Urban Forest Restoration Ecology:

Factors influencing native tree regeneration and practitioner decision-making processes

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ABSTRACT

Restoration ecology is a new and growing scientific field in the natural sciences. Typically restoration efforts have been focused in remote and rural landscapes. However, recently there has been a rise in the number of projects aimed at restoring native forests in urban environments. The long-term success of these projects depends on the sustainability of restoration plantings and their capacity to self-regenerate over time. Our scientific understanding of how to effectively restore functional, sustainable urban forests is increasing, yet we know little about mechanisms driving juvenile native tree regeneration and recruitment in these highly altered environments. Furthermore, ecological knowledge alone is insufficient to guide restoration outcomes in cities. Urban forests are socio-ecological systems and project outcomes are also influenced by the values and knowledge of restoration practitioners. In this thesis, I explore these ecological and social drivers of long-term urban forest restoration success in New Zealand.

In chapter 2 I investigate how drivers of juvenile native tree recruitment vary according to plant species successional status and growth stage. The long-term existence of restored forests depends on their capacity to follow the steps of ecological forest succession into maturity. It is crucial this includes recruitment of native tree seedlings into the sapling stage, especially for middle and late successional species. However, it is unclear whether the drivers that promote juvenile recruitment are generally applicable, or if they vary according to plant successional status and growth stage. Using a forest planting chronosequence approach and negative binomial generalised linear models, I investigated what drivers promote recruitment processes of early and mid-late successional native woody juveniles in 79 restored urban forests across nine New Zealand cities. I found that mid-late successional trees respond to different drivers than early successional species and the relative importance of particular drivers varies according to the trees growth stage. The effects of canopy cover appear generalizable across successional status but not growth stage, while the opposite is true for the effects of microclimate. Older forests host greater seedling abundance, and larger forest patch size is important for mid-late successional species. These results indicate that to

promote urban forest successional progress and hence recruitment of native tree juveniles, management approaches should vary depending on restored forest age and site conditions.

The third chapter explores how restoration practitioners in Aotearoa (New Zealand) make decisions in their efforts to re-establish native urban forests and why there is a gap between science-based best-practice restoration and on-the-ground implementation. The science-practice gap is well-documented in the applied sciences, but little is known about how it manifests in the urban forest restoration context where there are multiple objectives and many diverse stakeholders involved. To remedy this knowledge gap, we administered an online survey to practitioners involved in urban forest restoration. We found there is a tenuous link between scientific knowledge and urban restoration practice due to breakdowns in knowledge transfer and barriers to implementation. When restoring, practitioners tend to prioritise planting or weed control over other vital elements such as project planning and quantifiable monitoring. Objectives are commonly broad, vague, and focused on restoration of simple structural ecosystem components but not important functional attributes. Results show that practitioners value interactions with ecology experts and fellow practitioners equally to traditional forms of science communication (e.g. journal articles) as sources of restoration knowledge. This chapter suggests that prioritising interactive, interpersonal science communication and encouraging collaboration between scientists and practitioners would help strengthen knowledge transfer. Additionally, providing practitioners with time-saving resources, adequate funding, and guidance to navigate socio-ecological constraints that arise in urban projects will improve restoration outcomes.

This thesis broadens our understanding of social and ecological drivers of urban forest restoration success and highlights opportunities for improving the efficacy of urban restoration efforts. This research allows us to develop restoration ecology theory and refine best-practice methods for restoration of native urban ecosystems in Aotearoa.

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Clockwise from top left: Waitakere Ranges, Auckland; Dolbel Reserve, Napier; Pukekura Park, New Plymouth.

"This is a recreation reserve, with houses all around, and we want local people to enjoy it for its forested-ness, but also to enjoy walking through it, looking across it from their properties, and appreciating the fullest possible ecosystem benefits an urban forest can bring. We want to extend the ecosystem restoration into the neighbourhood…and as a group we want to learn and enrich our own lives through caring for our urban forest."

– Survey respondent (Chapter 3)

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1 CHAPTER 1

INTRODUCTION

1.1 Research Topic

This thesis presents research about urban forest restoration with specific focus on social and ecological drivers of urban forest restoration success.

The United Nations has declared 2021-2030 the decade of ecosystem restoration with the hope of reversing widespread ecosystem degradation and combating the global climate and extinction crisis (UN Water, 2019). Typically, restoration efforts are located in more remote or rural conservation areas, isolated from the most intense anthropogenic pressures (Miller & Hobbs, 2002). However, it is increasingly recognised that urban environments should be restored to encompass more native greenspace (Gaston, Ávila-Jiménez, & Edmondson, 2013; Sanderson, Walston, & Robinson, 2018) and can provide unique opportunities for studying ecosystem restoration (Barot *et al.*, 2019; Piana, Aronson, Pickett, & Handel, 2019). Compared to more remote areas, there is an increased capacity for maintenance and monitoring of restoration projects in cities due to a large volunteer workforce, making the implementation of restoration initiatives easier and more efficient (Clarkson & Kirby, 2016; Soanes & Lentini, 2019). Furthermore, such urban restoration projects increase the provision of ecosystem services (e.g. carbon sequestration, reductions in air and water pollutants, reduced urban heat island effects, amenity values, improved public health) that save money and benefit urban residents in a multitude of ways (Endreny, 2018; Hartig & Kahn, 2016; Nowak & Crane, 2002). The benefits of these ecosystem services make restoration initiatives attractive to local authorities like city councils (McDonnell & MacGregor-Fors, 2016; Oldfield, Warren, Felson, & Bradford, 2013; Pataki *et al.*, 2011). However, cities also present a unique set of challenges to restoration success. These include: soil compaction and modification (Gregory, Dukes, Jones, & Miller, 2006; Jim, 1998; Scharenbroch, Lloyd, & Johnson-Maynard, 2005), low native propagule pressure (Overdyck & Clarkson, 2012; Suding, Gross, & Houseman, 2004), habitat fragmentation and loss of connectivity (Lindig-Cisneros & Zedler, 2000), local extirpation of agents of seed dispersal or pollinators (Rader, Bartomeus, Tylianakis, & Laliberté, 2014), high exotic propagule pressure (Aikio, Duncan, & Hulme, 2012; Sullivan, Meurk, Whaley, & Simcock, 2009), urban heat island effects (Kalnay & Cai, 2003; Oke, Crowther, McNaughton, Monteith, & Gardiner, 1989), and the need for sustained support from a large range of stakeholders (Fox & Cundill, 2018; Wallace & Clarkson, 2019).

To restore urban forests from scratch that are functional and sustainable, projects must be guided by ecological theory (MacMahon, 1998; Matzek, Covino, Funk, & Saunders, 2014; Seavy & Howell, 2010; Wallace & Clarkson, 2019). However, even with inputs from ecological theory, reconstructing a forest from scratch in a completely degraded area (as defined by Stanturf, Palik, & Dumroese, 2014) is a difficult task (Waldron & Xi, 2013). Ecological restoration aims to recreate complex ecosystems over relatively short time spans that in natural conditions would develop over centuries (Hilderbrand, Watts, & Randle, 2005). The complexity of natural ecosystems means that restoration must rely on simplified conceptual models, and as such, the outcomes of restoration projects are variable with many falling short of expectations (Hilderbrand *et al*., 2005; Suding, 2011). In particular, recruitment failure can arrest successional development and negatively affect the resilience of restored forests (Acácio, Holmgren, Jansen, & Schrotter, 2007; Ettinger, Lee, & Montgomery, 2017).

Previous research has described the developmental trajectories of restored urban forests over time and factors that may impede or promote regeneration. For example, Wallace, Laughlin, & Clarkson, (2017) found that as initial plantings of early successional tree species age, canopy closure triggers the stabilization of understory microclimate and reduces light availability, shading out herbaceous weeds and creating conditions suitable for native seedling establishment approximately 20 years after initial planting (Wallace, Laughlin, & Clarkson, 2017). Other similar studies have found that a lack of seed rain and non-native dominated seed banks limit regeneration of native species in urban forests (Overdyck & Clarkson, 2012), while canopy composition can influence the establishment and survival of species in the understory (Laughlin & Clarkson, 2018). Management practices also play an important role. Johnson & Handel (2019) found that sustained management interventions (i.e. mechanical weed removal, herbicide application, watering) improve urban restoration outcomes and shift community composition towards increased indigenous plant dominance.

Generally we know the speed at which reconstructed forests grow and that they can regenerate themselves somewhat, but we don't understand exact mechanisms driving juvenile tree regeneration or the transition between growth stages from germinated seedlings to saplings (Piana *et al.*, 2019). This thesis takes this next step by investigating whether drivers of native tree seedling regeneration vary according to a species' successional status and seedling growth stage. Filling this knowledge gap is important, both for advancing our theoretical understanding of how juvenile tree regeneration fits into successional dynamics in restored urban forests, and also to provide best-practice recommendations tailored to each stage of the forest restoration process.

This thesis also addresses a related topic regarding transfer of research knowledge to restoration practitioners. To ensure that ecological research findings translate into improved restoration outcomes, it is crucial that information is transferred to practitioners and implemented (Hulme, 2014; Pullin & Knight, 2001). There are few studies that document the practices and experiences of urban forest restoration practitioners or investigate the links between science and practice in an urban forest restoration context. Understanding practitioner experiences and use of scientific information is an important first step towards diminishing the gap between ecological knowledge regarding best-practice restoration, and on-the-ground implementation (Hulme, 2014; Knight *et al.*, 2008).

1.2 Background

Urban landscapes are expanding, with over 50% of the world's population now living in cities (McDonnell & Hahs, 2015; Sasaki, Ishii, & Morimoto, 2018). Urban environments are characterized by frequent disturbances, habitat fragmentation, extensive areas of impervious surfaces, high concentrations of pollutants, and urban heat island effects (Aronson *et al.*, 2016; Ignatieva, Stewart, & Meurk, 2011; McKinney, 2008). As a result, urban areas typically have greatly reduced indigenous vegetation cover and are often described as sites of bioitic homogenization with low biodiversity values (Barot *et al.*, 2019; Lepczyk *et al.*, 2018; McKinney, 2008; Miller, 2005). Cities have also been described as the focal points of an 'extinction of experience' with an increasing number of urban residents disconnected from nature (Miller, 2005; Soga & Gaston, 2016).

In response to these dual problems of biodiversity loss and an increasing disconnect between people and nature, urban forest restoration projects have gained momentum, increasing in frequency throughout New Zealand and around the world (Clarkson & Kirby, 2016; Oldfield *et al*., 2015; Standish, Hobbs, & Miller, 2013). These restored urban forests provide habitat for native fauna (Alvey, 2006; Sandström, Angelstam, & Mikusiński, 2006), provide beneficial ecosystem services (Elmqvist *et al*., 2015; Endreny, 2018), are important sites for the conservation of endangered plant species (Soanes & Lentini, 2019), and perhaps most importantly, increase the visibility of nature in cities, providing opportunities for urban residents to reconnect with and appreciate nature without having to travel long distances (Miller, 2005; Standish *et al*., 2013). There is a growing body of scientific literature associated with urban ecological restoration, but as a relatively new field, many knowledge gaps still remain regarding the best methods for efficient, cost-effective and successful long-term restoration outcomes (Oldfield *et al*., 2013).

One measure of restoration success is the sustainability of plantings and their capacity to self-regenerate over time. Without regeneration and recruitment of native tree seedlings, forests will be limited to a single generation (Oldfield *et al*., 2013). Chapter two of my thesis uses a chronosequence approach, whereby space is substituted for time, meaning I have collected data from urban forests reconstructed from scratch at different points spanning the last 60 years. This experimental framework allowed me to investigate what abiotic and biotic factors constrain or promote native tree regeneration and recruitment in these forests as they develop through the crucial first stages of establishment and succession. I studied relationships between microclimate and plant community composition and structure in 79 restored urban forests of varying ages across nine New Zealand cities. Using statistical modelling approaches, I investigate forest age, forest patch size, herbaceous ground cover, forest canopy openness, and microclimate under the forest canopy in relation to the abundance of both early and mid-late successional native woody seedlings across three stages of growth (height tiers). The results of this study enable me to make management recommendations regarding best practice methods for urban forest restoration that promote native tree regeneration in the early stages of forest development, and recruitment of saplings in the latter stages. It is particularly important that we understand the drivers of recruitment of mid-late successional saplings as these long-lived tree species are crucial for forest ecosystem resilience and signal an important stage in forest development dynamics (Laughlin & Clarkson, 2018; Oliver & Larson, 1990).

Improving our theoretical understanding of what constitutes best-practice ecological restoration is vital, but this knowledge alone is insufficient for improving restoration outcomes (Higgs, 2005). To be useful, knowledge must be effectively transferred to practitioners and implemented. Chapter three of my thesis uses a practitioner survey approach to investigate how city councils and community groups involved in urban forest restoration make decisions in their efforts to re-establish native forests, how they access scientific knowledge, and how the human dimensions of these projects influence restoration outcomes. This survey incorporated open-ended and fixed-answer questions designed to explore how practitioners design plantings, choose plant species, what resources they utilize during the planning phase, what obstacles they face in achieving restoration objectives, and what resources they would like to have available to them. The information gained from this survey allowed me to identify weak links between science and practice as well as opportunities to strengthen these links.

This thesis contributes to a growing body of literature that provides guidance on how to effectively restore urban forests. Urban restoration actions take place within socio-ecological systems and therefore focusing solely on understanding ecological dynamics can result in crucial determinants of restoration success being overlooked (Crandall *et al*., 2018; Fernández-Manjarrés, Roturier, & Bilhaut, 2018). My work avoids this shortcoming by combining an ecological and social science approach. It advances scientific knowledge through the investigation of factors promoting early and mid-late successional native seedling regeneration and recruitment, and provides an overview of the tenuous link between research and implementation in urban forest restoration. Together, the results from these two inter-related pieces of research will help to maximise the efficiency, sustainability and overall success of forest restoration projects in our urban landscapes.

1.3 Research questions

This thesis investigates the following questions:

1) What factors constrain or promote native juvenile tree regeneration and recruitment in restored urban forests?
2) Do these factors vary by tree species successional status and juvenile growth stage?
3) To what extent is urban forest restoration practice informed by restoration ecology research?
4) How can we strengthen the information transfer link between restoration researchers and practitioners?

1.4 Thesis objectives

This thesis aims to:

1. Investigate what factors constrain or promote regeneration and recruitment processes for native woody species in different successional categories in restored urban forests of varying ages.
2. Investigate how restoration practitioners make decisions regarding urban forest restoration with the aim to diminish the gap between theoretical best-practice restoration and on-the-ground implementation.

1.5 Thesis overview

This thesis comprises four chapters, one of which has been submitted for publication.

- Chapter 1: provides the introduction and background for the entire thesis
- Chapter 2: is a data chapter investigating drivers of early successional and mid-late successional native seedling regeneration, establishment and recruitment
- Chapter 3: is a data chapter that has been submitted to the journal *Urban forestry and urban greening* under the authorship of Sarah Busbridge, Bruce D. Clarkson and K. J.

Wallace. It explores practitioner decision-making processes and the links between science and practice in urban forest ecological restoration.

- Chapter 4: synthesises the results and highlights implications for practical application.

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2 CHAPTER 2

DRIVERS OF NATIVE TREE RECRUITMENT IN RESTORED FORESTS DIFFER BY PLANT SPECIES SUCCESSIONAL STATUS AND JUVENILE GROWTH STAGE

2.1 Abstract

Projects aimed at restoring forests from scratch in urban landscapes using native species are increasing in frequency throughout New Zealand. While the establishment of early-successional species to form an initial canopy is often relatively successful, the long-term existence of these forests depends on their capacity to regenerate and persist. This entails recruitment of tree seedlings into the sapling stage, especially for middle and late successional species. Here, I investigated drivers that promote recruitment processes for species in different successional categories (early vs. mid-late) in 79 restored urban forests spanning 4 to 58 years since initial reconstruction from scratch across nine New Zealand cities. Negative binomial generalised linear models were used to determine relationships between abundance of early and mid-late successional native woody juveniles across three growth stages and forest age, patch size, herbaceous ground cover, canopy openness, and site microclimate characteristics such as mean air and soil temperatures. As planted forests age, canopy openness decreases and the understory microclimate cools and stabilises. This cooler microclimate promotes increased abundance of only the mid-late juvenile trees, while canopy closure promotes an increase of both early and mid-late native tree seedling germination and recruitment. The strength of this effect decreases as seedlings grow taller, and is no longer important for juveniles of either successional status once they reach the sapling stage. These findings indicate that seedlings and saplings may have different light requirements. Herbaceous cover plays a role during early growth as it is inversely related to short seedling abundance, but once seedlings reach sapling height there is no longer a relationship. Increased age of the restored forest is the most important predictor of greater short seedling

abundance, although larger forest patch size is similarly important for just the mid-late successional woody species. These results indicate that to promote urban forest successional progress and hence recruitment of native tree juveniles, management approaches should vary depending on the age, site conditions, and developmental stage of a restored forest.

2.2 Introduction

Urban forest restoration projects vary in scale and the scope of what they aim to achieve (Clarkson & Kirby, 2016; Gobster, 2010; Standish *et al*., 2013). Yet one common goal is that plantings are sustainable and self-regenerate over time to produce long term forest ecosystem benefits. To provide practitioners with reliable guidelines on how restore functional, self-perpetuating forests that follow a natural successional trajectory to maturity, it is important to understand the precise drivers of regeneration and recruitment processes and how best to manage them (Piana, Aronson, Pickett, & Handel, 2019; Wallace & Clarkson, 2019).

Urban environments present unique challenges to functional restoration success as ecological dynamics in cities are often severely altered and differ considerably to those found in non-urban areas (Grimm *et al*., 2008), where the majority of restoration ecology research has taken place. Restoration practitioners working in cities must contend with challenges such as soil compaction and modification (Gregory *et al*., 2006; Jim, 1998; Scharenbroch *et al*., 2005), low native propagule pressure (Overdyck & Clarkson, 2012; Sullivan *et al*., 2009), local extirpation of seed dispersal agents or pollinators (Rader *et al*., 2014), high non-native invasive plant species propagule pressure (Aikio *et al*., 2012; Meurk & Hall, 2006; Overdyck & Clarkson, 2012; Sullivan *et al*., 2009), urban heat island effects (Kalnay & Cai, 2003; Oke *et al*., 1989), and frequent disturbances (McKinney, 2008).

Models of forest succession provide the conceptual basis for planning ecological restoration (Pickett *et al*., 2001; Young, Petersen, & Clary, 2005). Forest succession describes temporal patterns of change in plant community composition and structure in response to disturbance (McCook, 1994). These changes in vegetation modify the abiotic environment of a site, thereby facilitating or inhibiting the colonisation or survival of juvenile trees based on their niche requirements (McCook, 1994). In a simplified model of secondary succession, fast growing, light demanding pioneer species are the first to colonise and establish, creating conditions suitable for the establishment of more long-lived, shade-tolerant later successional species (Crawley, 1997; Huston & Smith, 1987). Ecological restoration aims to mimic this process through planting of early successional native tree seedlings and subsequent management interventions (Johnson & Handel, 2016). However, the successional models used to inform restoration have been developed from data in non-urban, largely intact forest systems, and must be modified to be applicable to reconstruction of urban forests (Johnson & Handel, 2016).

Urban restoration ecology is a relatively new discipline and there is a need for more research to inform urban forest successional models. In particular, there are still a number of knowledge gaps regarding best practice methods for restoring urban forests that are self-sustaining with adequate native regeneration (Oldfield *et al*., 2013). At present, projects are relatively successful in achieving the establishment of the initial canopy but few have demonstrated subsequent recruitment of mid-late successional long-lived tree species into the canopy. These mid-late successional species are essential for delivering ecosystem services and providing faunal habitat and forest resilience (Oliver & Larson, 1990; Suganuma, Assis, & Durigan, 2014).

Previous studies in urban environments have shown that as initial plantings age, canopy closure triggers the stabilization of understory microclimate and causes reductions in light availability, which shades out herbaceous weeds and creates conditions suitable for native seedling germination (Wallace, Laughlin, & Clarkson, 2017). Additional constraints to native regeneration include non-native plant propagule pressure and native seed dispersal limitations. Even under a native canopy, seed banks may be dominated by invasive non-native species and the fragmented nature of urban forests mean there are often few nearby native seed sources (Overdyck & Clarkson, 2012). As natural colonization by native species is highest at sites located less than 100m from existing native vegetation (Sullivan *et al*., 2009), this can limit the abundance and species richness of regenerating seedling populations. Where dispersal limitations are present, enrichment planting of mid-late successional species is often undertaken to further successional processes. Like early successional species, the survival of these enrichment plants is subject to environmental conditions, but also determined by forest age, extent of canopy cover and canopy composition (Laughlin & Clarkson, 2018).

These prior studies provide the basis for current best-practice management recommendations (i.e. planting densely to fast-track the establishment of an initial canopy, considering local dispersal constraints, undertaking enrichment planting). However, many urban forest restoration projects are still largely trial and error initiatives (Robinson & Handel, 1993) without plans or clear management for facilitating later stages of succession due to lack of empirical knowledge. My work aims to address this gap by ascertaining the drivers that promote native juvenile tree recruitment and determining whether they are universally consistent, or if they vary by plant successional status and growth stage.

In non-urban forests, ecological research has shown that adult tree niches are often broader than the juvenile niche (Grubb, 1977). In other words, adult trees can often persist where juveniles of the same species may not, and therefore the presence of many species are limited by mortality during the germination and establishment growth stages (Grubb, 1977; Oldfield *et al*., 2013; Young *et al*., 2005). This change in niche requirements across growth stages is referred to as an "ontogenetic niche shift" and is relatively understudied in plant ecology (Young *et al*., 2005). In the context of restoration, ontogenetic niche shifts can arrest successional development without management interventions such as enrichment plantings (Young *et al*., 2005). For example, Paterno *et al*. (2016) showed that the facilitative effects of a canopy on mid-late successional seedlings can become inhibitory once seedlings reach a certain size. Other studies investigating drivers of seedling regeneration have shown microclimate is a crucial driver at seedling germination and establishment stages (McLaren & McDonald, 2003), while herbaceous vegetation cover constrains young seedling numbers (Kuijper *et al*., 2010). These studies have all taken place in non-urban forests, however, and there is a need for more research on drivers of seedling establishment and recruitment specific to restored urban forests, which exist in highly altered contexts.

Here I address this knowledge gap by investigating drivers that promote regeneration and recruitment processes for woody species in two different successional categories in 79 restored urban forests spanning 4 to 58 years in age across nine New Zealand cities. Specifically, my research question was: Do restored urban forest attributes (i.e. age, patch size, herbaceous ground cover, canopy openness, and understory microclimate) affect juvenile tree regeneration differently across a) juvenile growth stage (i.e. height) and b) successional status? I hypothesized that: 1) early successional species would be more tolerant of microclimate fluctuations regardless of their growth stage, and they would appear earlier in a planted forest's successional trajectory, and 2) small mid-late successional seedlings would require the cooler microclimate associated with a closed canopy, while the taller saplings would be more tolerant of microclimate fluctuations but instead require more light (i.e. greater canopy openness). The results of this study allow us to make management recommendations regarding best practice methods for urban forest restoration that promote native tree regeneration in the early stages of forest development, and recruitment of saplings during later stages.

2.3 Methods

Study sites

My study took place across the North Island and South Island of New Zealand. Historically, 75% of New Zealand's main islands were covered in continuous temperate rainforest (Nicholls, 1980). Due to widespread clearing for agriculture and silviculture, only 23% of the total area of New Zealand now remains in native forest (Star, 2002).

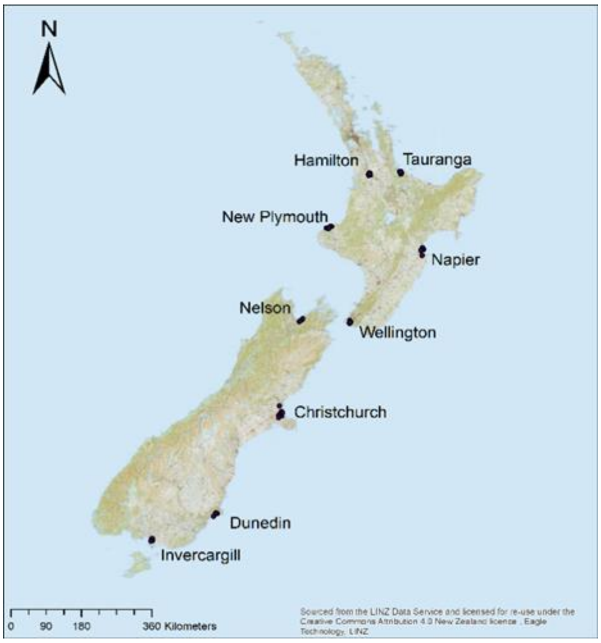

Figure 2.1: Locations of 9 research cities in New Zealand, each of which had 9 research sites except Dunedin and Invercargill, which had 8 each.

Table 2.1. Annual climate values for the nine study cities.

City	Mean air temperature (°C)	Mean min temp (°C)	Mean annual solar radiation ($MJ/m^2/day$)	Mean Vapour pressure deficit (kPa)	Mean annual water deficit (mm)
Hamilton	13.97	4.40	14.80	0.36	46.93
Tauranga	14.08	4.59	15.13	0.42	24.15
New Plymouth	13.43	5.61	14.81	0.33	9.39
Napier	13.43	3.44	14.73	0.52	165.72
Wellington	12.10	4.04	14.03	0.34	62.79
Nelson	11.81	1.71	15.14	0.44	81.50
Christchurch	11.57	1.06	13.83	0.47	202.35
Dunedin	9.78	2.00	12.38	0.37	47.80
Invercargill	9.94	1.19	12.36	0.33	17.49

Data were collected from restored forest patches reconstructed from scratch in five North Island cities (Hamilton, Tauranga, New Plymouth, Napier and Wellington) and four South Island cities (Nelson, Christchurch, Dunedin, Invercargill), spanning a latitudinal range of climate (-46.45137 to -37.67842) (Fig. 2.1; Table 2.1). Seventy-nine urban restoration sites were selected for this study to form a chronosequence spanning 4 years to 58 years since initial planting. Chronosequences use space as a substitute for time in order to study the temporal dynamics of systems and uncover potential causal links between response and predictor variables (Dornelas *et al.*, 2013).

In each city, nine 20 x 10 m (200 m^2) permanent plots were established (except for Dunedin and Invercargill, which only had eight plots) in forests planted using native species. Plots were located within city limits in developed, built up urban and peri-urban areas, were planted from scratch as a single initial cohort, and did not contain streams or seepages. The area of the forest patches the plots occurred within ranged between 0.1 ha to 77.3 ha, with a mean size of 8.01 ha. Forest patch size was measured such that it included any remnant forest if present adjacent to stands of restored forest. Plot edges were typically >1m from the forest patch edge, but this was not possible in a minority of cases due to the small size of some restored forest patches. The site name, location, age and patch size of study plots are listed in Appendix 2.1, Table 2.3.

Data collection

Vegetation Survey

Within each plot a vegetation survey was completed using National Vegetation Survey (NVS) protocols (New Zealand National Vegetation Survey Databank). Trees and shrubs were both included in the survey, and hereafter are both referred to as trees or juvenile trees for simplicity. Each adult tree (diameter at breast height DBH ≥ 2.5cm) within the plot was identified to species level, classified as native or non-native using NVS standard classification (New Zealand National Vegetation Survey Databank), and the diameter at breast height (DBH) of each stem (≥2.5 cm) was recorded. These data were then summed to find total basal area of the plot. Saplings, (trees >1.35m high and <2.5cm DBH) within the plot were classified as native or non-native, and tallied by species. Both 'germinated' and 'recruited' growth stages of seedlings (<1.35m tall) within 10 circular subplots (each with 1.5m radius) were identified to species level, and tallied by height tier: >15cm, 16-45cm, 46-105cm, 106cm-135cm. Seedling tallies were then added together across the 10 circular subplots (totalling 70.7m^2) and scaled up to stems per 200m^2 (the area of each plot). All native tree species recorded were assigned a successional status following de Silva, (2019) and Wallace *et al*., (2017). If species were not present in either of these sources, successional status was assigned using information available in published literature, planting guides, websites and databases. Using Microsoft Access (version 15.0, Microsoft, Washington, U.S.), seedling tally data was separated into categories based on seedling growth stage (germinated seedlings: 0-15cm, recruited seedlings: 16-135cm, saplings: 136cm+) and successional status (early, or mid-late). These two successional status groups were used for this study because many middle and late successional species appear to overlap somewhat in establishment timing during first decades of forest development and sample sizes of species in the mid-late categories were small. All plant species identified in study plots are listed in Appendix 2.2, Table 2.4 along with their successional status, and the growth stages that were found present. Percent cover of herbaceous ground cover was visually estimated within each plot.

Abiotic data

Canopy openness was measured at the centre and four corners of each plot at breast height (~1.3m from the ground) at three occasions over a 12 month period (at 0, 6, and 12 months) using a convex spherical densiometer (Convex model A; Forestry Suppliers, Jackson, Mississippi, USA). These 15 values per plot were averaged to represent a single plot-level percentage canopy openness value (used as a proxy for light level near the forest floor). Air temperature (^{o}C) was measured every four hours for 12 months using HOBO data loggers (model MX2301A; Onset, Cape Cod, Massachusetts, USA) attached 1m above ground level to the tree trunk closest to the centre of each plot. HOBOs were suspended inside radiation shields to ensure direct solar radiation did not confound general ambient temperature measurements. Soil temperature (^{o}C) was measured every four hours for a three month period (24 March 2018 – 24 June 2018) at the centre and a single corner of each plot using thermochron iButtons buried at 10cm depth (iButton dataloggers model DS1921G-F5; Maxim Integrated, San Jose, California, USA). This timeframe occurred during New Zealand's autumn and winter, but temperature varies widely at any time of year due to the oceanic island climate. For both air and soil temperature we computed the mean, variance, and maximum values for the recorded range of temperatures at each plot.

Statistical analyses

Statistical analyses were carried out in R Studio version 3.4.3 (R Core Team 2018).

While 79 plots were measured, analysis was only undertaken on 75 plots. Four restored forest plots (one from Hamilton, one from Christchurch, one from Invercargill, one from Dunedin) were excluded from the final analysis as they were adjacent to large patches of mature forest, and therefore subject to more intense seed rain than all other restored forests plots, or had a monospecific canopy planted and so were flooded with propagules of a single species.

First, to investigate important ecosystem attributes in forest successional development, bivariate plots were inspected and fitted with linear regression models to log transformed data for the relationships between forest age and both basal area and canopy openness. The same procedure was used to investigate relationships between canopy openness with herbaceous ground cover, air temperature variance, and maximum air temperature. I evaluated candidate models based on significance (α =0.05) and R^2 values. Second, relationships between predictor variables (forest age, canopy openness, herbaceous ground cover, mean air and soil temperature) and juvenile tree abundance were evaluated. A generalized linear model (GLM) with a Poisson error distribution was trialled to begin with but was unsuitable as the data were overdispersed (residual deviance was more than twice that of the residual degrees of freedom) and included zeros. This lead me to instead use a

GLM with a negative binomial distribution error, which is commonly used to model overdispersed count data that includes many zeros (Ver Hoef & Boveng, 2007; Warton, 2005). Negative binomial GLMs were used to test the effects of all the predictor variables on native woody juvenile tree regeneration abundance. Forest age and patch size were also modelled together using a multiple regression to look for interactions and understand how much of an influence each predictor variable had on regeneration abundance across the two different successional status categories and three different growth stages. Resulting models were evaluated based on significance (α=0.05). All negative binomial models were fitted using the glm.nb function in the 'MASS' package in R (Venables and Ripley 2002). Separate models were fitted for each of the two successional status categories (early and mid-late) and for each of the three growth stages (height tiers) respectively. Bivariate linear regression model fit outputs were plotted using the ggplot() function in the 'ggplot2' package in R (Wickham, 2006), and multiple regression model fit outputs were plotted using the wireframe() function in the 'Lattice' package in R (Sarkar, 2008).

2.4 Results

Important attributes in forest ecosystem development

As forests developed (age in years since initial planting increased), basal area increased from $0.004m^2$ per $200m^2$ (the plot area) to $1.84m^2$ per $200m^2$ (Fig. 2.2A) and average canopy openness decreased from 88.82% to 5.89% (Fig. 2.2B). Forests with low canopy openness hosted lower herbaceous ground cover (a drop from 96.88% to 0%; Fig. 2.2C), less fluctuation of air temperature (43.6 ^{o}C swings in open canopies, became 17.5 ^{o}C swings in closed canopies; Fig. 2.2D), and lower maximum air temperatures (34.7 ^{o}C maximum in open canopies, became 22.86 ^{o}C under mostly closed canopies; Fig. 2.2E).

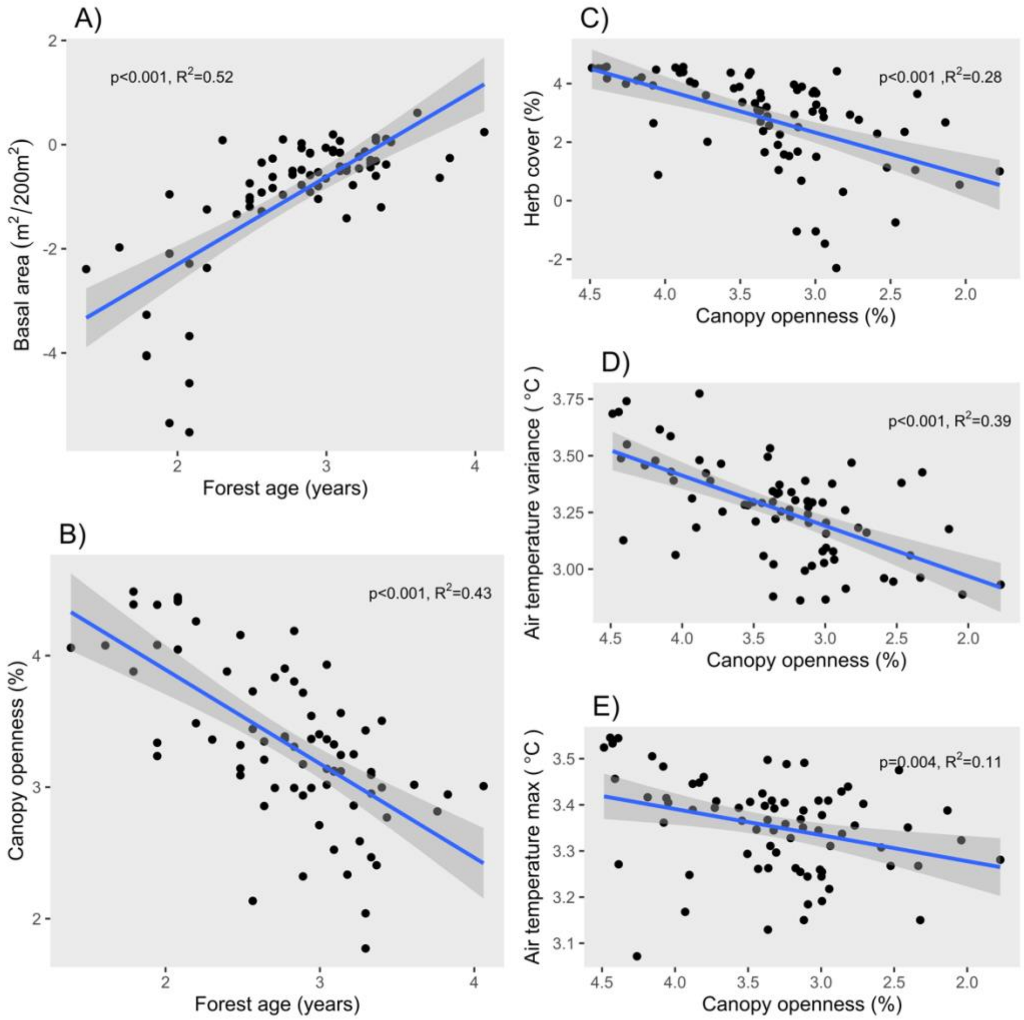

Figure 2.2 Changes in ecosystem attributes driving native tree recruitment during forest development after initial restoration planting.

Drivers of juvenile tree recruitment

Juvenile trees respond to the degree of canopy openness and forest age (Fig. 2.3 & 2.4). However, the shape of the response curve changes depending on both the juvenile's growth stage and successional status. Early successional juveniles grow more abundantly in older forests (Fig. 2.3A, 2.3C, 2.3E) and under less open canopies (generally characteristic of older forests) (Fig. 2.4A, 2.4C). In comparison, mid-late successional juvenile abundance has no relationship with forest age (Fig. 2.3B, 2.3D, 2.3F), but a similar relationship to canopy openness as early successional species with more small juveniles found under less open canopies (Fig. 2.4B, 2.4D). Although the abundance of both successional stages of ‘germinated’ and ‘recruited’ seedlings increased as canopy openness decreased, ‘saplings’ had no relationship with canopy openness regardless of successional status (Fig. 2.4E, Fig. 2.4F).

The abundance of ‘germinated’ and ‘recruited’ early successional trees appears positively correlated with forest age (Fig. 2.3) and negatively correlated with declines in canopy openness (Fig. 2.4), while abundance of ‘germinated’ and ‘recruited’ mid-late successional species is correlated with only declining canopy openness and has no relationship with forest age (Fig. 2.4, Fig. 2.3).

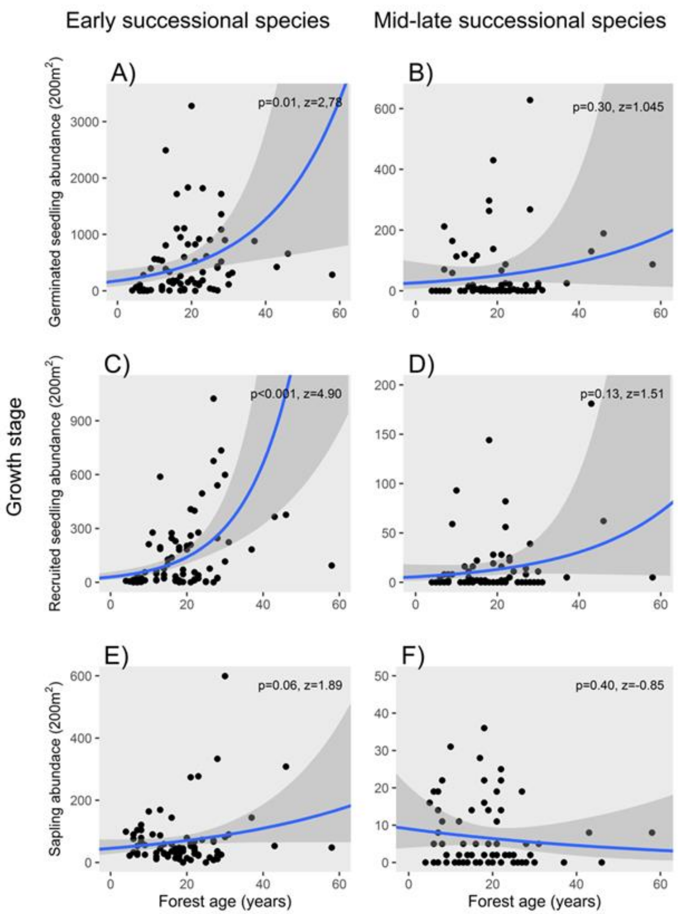

Figure 2.3 Abundance of regenerating native juvenile trees of early successional species (left) and mid-late successional species (right) per plot (200m^2) as related to forest age in planted urban forests (n=75, df=73).

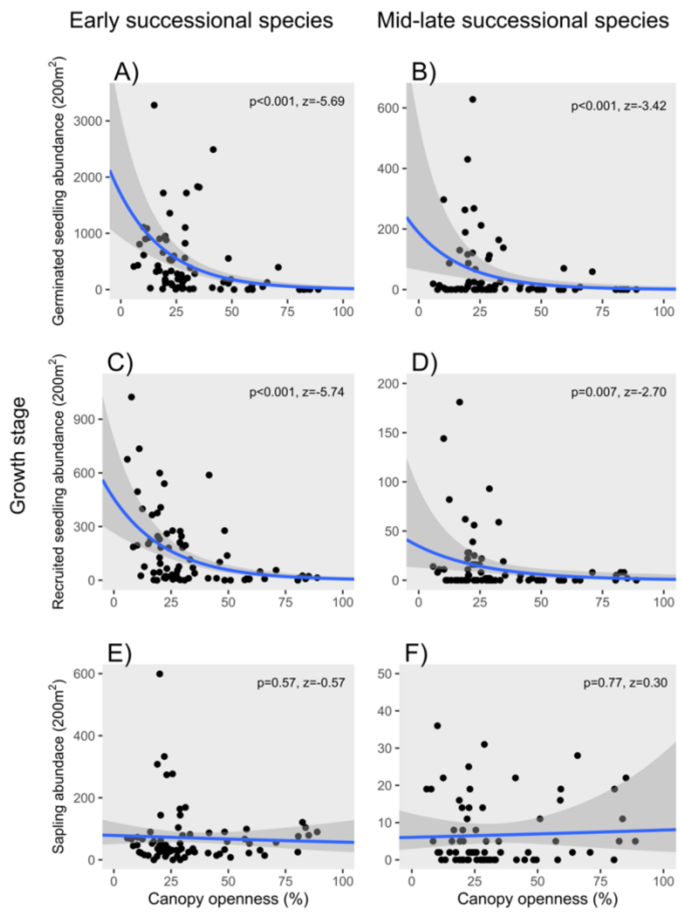

Figure 2.4 Abundance of regenerating native juvenile trees of early successional species (left) and mid-late successional species (right) per plot (200m^2) as related to canopy openness in planted urban forests (n=75, df=73).

The relationship between juvenile tree abundance and percent herbaceous ground cover is similar for both successional classes. There are significantly fewer 'germinated' and 'recruited' seedlings where herbaceous groundcover is high, but there is no relationship once juveniles reach the 'saplings' growth stage (Fig. 2.5).

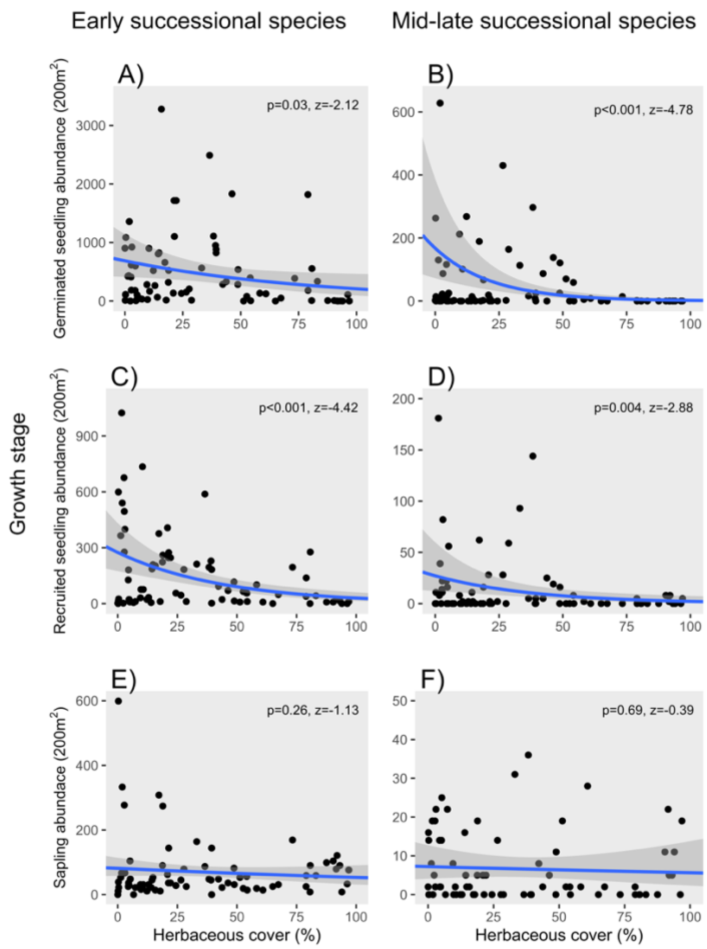

Figure 2.5 Abundance of regenerating native juvenile trees of early successional species (left column) and mid-late successional species (right column) per plot (200m²) as related to herbaceous ground cover (%) (n=75, df=73).

The abundance of mid-late successional 'germinated' and 'recruited' seedlings is tied to air and soil temperature, with higher numbers of seedlings present at cooler air temperatures (Fig. 2.6, Fig. 2.7B & D). 'Sapling' abundance is also related to air and soil temperature, but in opposite directions depending on the species successional status (Fig. 2.6 & 2.7). By the crucial sapling stage, when trees are truly on their way to the canopy, warmer air and soil temperatures appear to have a positive effect on early successional species (Fig. 2.6E & 2.7E), and a negative effect on mid-late successional species (Fig. 2.6F & 2.7F).

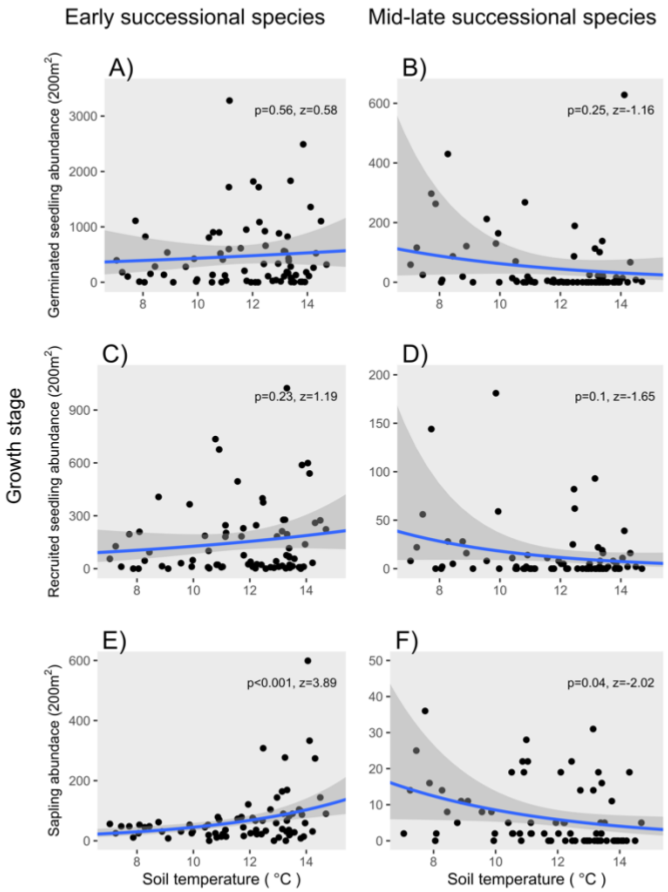

Figure 2.6 Abundance of regenerating native juvenile trees of early successional species (left column) and mid-late successional species (right column) per plot (200m²) as related to mean plot soil temperature at 10cm depth in restored urban forests (n=75, df=73).

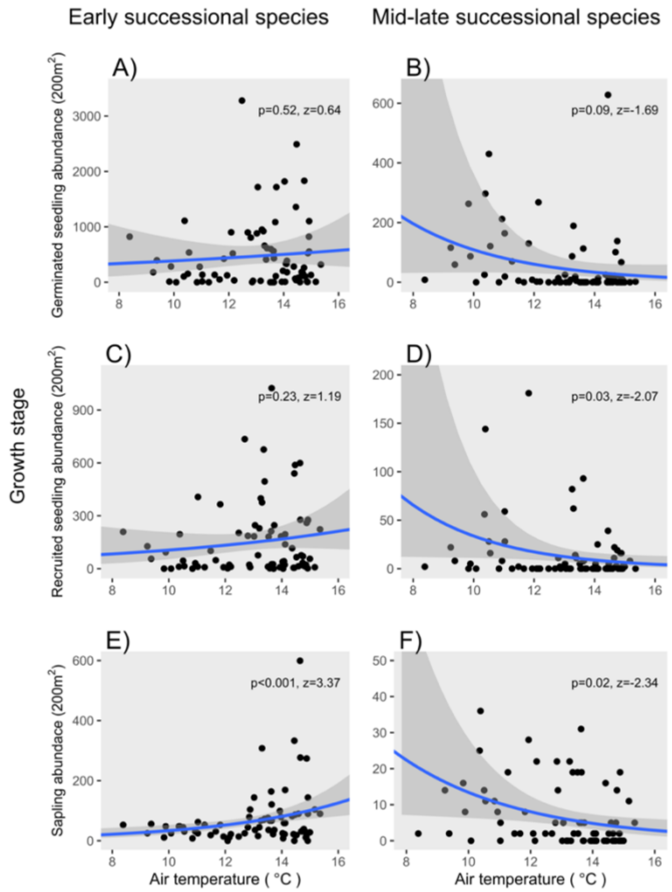

Figure 2.7. Abundance of regenerating native juvenile trees of early successional species (left column) and mid-late successional species (right column) per plot (200m²) as related to mean plot air temperature in restored urban forests (n=75, df=73).

When investigating drivers of the juvenile tree abundances, the relative impacts of forest age and forest patch size were looked at together within each successional category (Table 2.2.; Fig 2.8). Results show that forest age is an important driver of ‘germinated’ and ‘recruited’ seedling abundance in both successional status categories (Table 2.2). In comparison, forest patch size is irrelevant for early successional species abundance, but has an important effect on mid-late juvenile abundance, particularly at the ‘recruited’ seedling growth stage. Neither age nor patch size is an important driver of sapling abundance for both successional status categories.

Table 2.2. Multiple regression results for abundance of regenerating native juvenile trees of different successional statuses as related to the predictor variables of forest age and patch size.

Successional Status	Growth Stage	Forest Age		Forest Patch size		Interaction	
		Z value	Pr(>|z|)	Z value	Pr(>|z|)	Z value	Pr(>|z|)
Early	A) Germinated (<15cm)	2.376	0.02*	0.72	0.47	-0.58	0.56
	C) Recruited (16-135cm)	3.73	<0.001***	0.16	0.87	-0.57	0.57
	E) Saplings (>135cm)	1.07	0.29	-0.64	0.51	0.38	0.71
Mid-late	B) Germinated (<15cm)	1.88	0.06 .	1.71	0.08 .	-0.92	0.36
	D) Recruited (16-135cm)	3.02	0.003**	3.38	<0.001***	-1.58	0.11
	F) Saplings (>135cm)	-1.19	0.24	-0.62	0.53	0.99	0.32

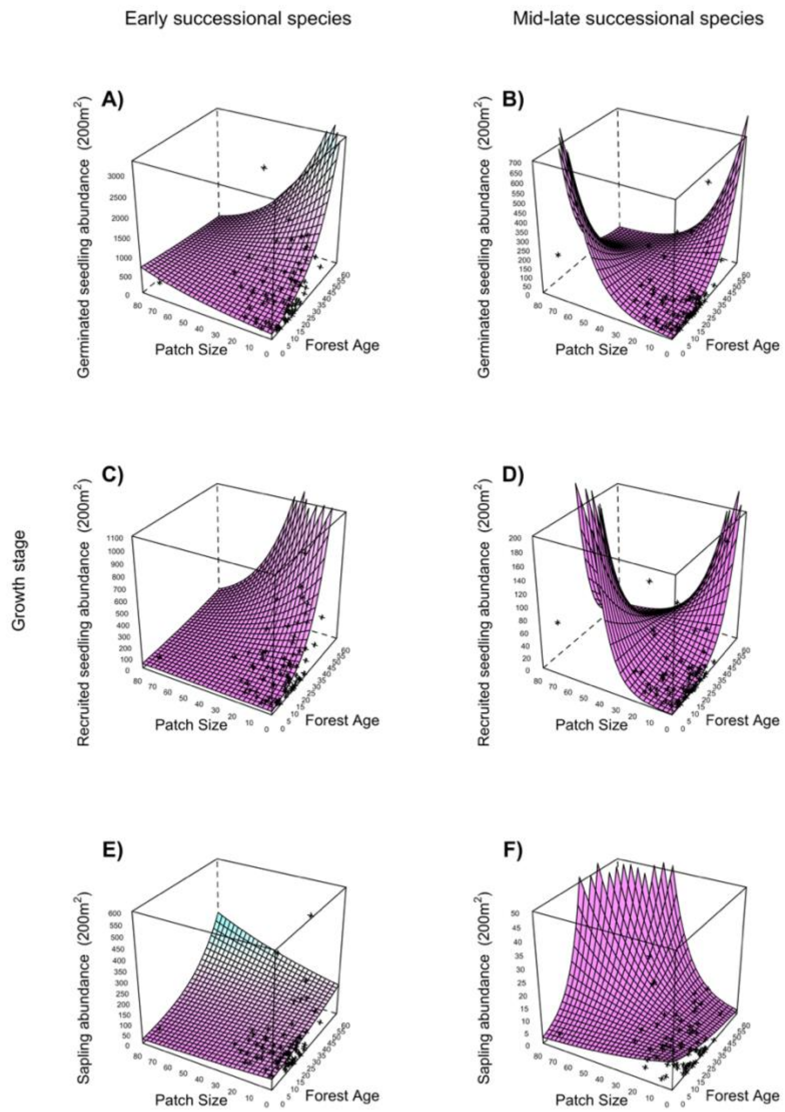

Figure 2.8. Juvenile tree abundance as predicted by restored forest patch size and forest age (time since initial planting).

2.5 Discussion

Important attributes in forest ecosystem development

Restored forest planting age and canopy closure are key developments that create conditions suitable for native seedling regeneration (Bertacchi *et al*., 2016; Gerhardt, 1996; Lebrija-Trejos *et al*. 2010; McLaren & McDonald, 2003; Wallace *et al*., 2017). As forests age, increases in tree basal area cause decreases in canopy openness which is related to reduced herbaceous ground cover and air temperature fluctuations, and a reduction on average in maximum understory air temperatures. Previous research indicates that time since restoration is a key predictor of restoration success (Crouzeilles *et al*., 2016) and understory sapling density (Suganuma & Durigan, 2015). My results expand upon this by showing that although forest age is of particular importance for regeneration of early successional species, abundance of the smaller mid-late successional species is actually more strongly related to canopy cover. Since mid-late successional species tend to be slower growing and prefer established forest conditions (Cornelissen, Castro-Diez, & Carnelli, 1998; Lebrija-Trejos *et al*., 2010), it's possible that the relatively short 58-year forest age span in this study may not have fully captured the relationship between forest age and abundance of mid-late successional juvenile trees. However, it is most likely that canopy cover is a stronger predictor of younger, sensitive mid-late successional juvenile abundance because it has a direct influence on the environmental conditions of restored forests (i.e. microclimate, light availability) (Aussenac, 2000; Vieira & Scariot, 2006; Wallace *et al*., 2017).

Nevertheless, even if forests have not yet developed conditions suitable for small mid-late successional seedling regeneration, it may be possible to introduce mid-late tree species that were germinated elsewhere earlier than they would arrive naturally. Tolerance of juvenile trees to environmental stresses increases as they become older (Niinemets, 2010), implying that young restored forests that are unsuitable for small seedling survival, can be suitable for sapling survival. My results suggest that although the smallest mid-late successional species may not be spontaneously germinating in the environmental conditions of young restored forests, the very same species introduced as saplings could survive. This reinforces the need for management interventions in the form of enrichment planting to fast track development to a late-successional forest state, as conditions suitable for natural germination and recruitment of later-successional seedlings are not guaranteed to develop with time.

The relationship found between forest age and juvenile tree abundance exemplifies the expected patterns of the classic competitive self-thinning process found in forest generational turnover (Bormann & Likens, 1979; Franklin et al., 2002). However, the novel aspect of this work goes beyond known competitive interactions to identify the abiotic drivers of juvenile tree recruitment. Specifically, how abiotic drivers differ by juvenile growth stage and successional status is revealed.

Drivers of juvenile tree recruitment

Light

Seedlings and saplings have different light requirements in restored urban forests. While a generally closed canopy has a positive effect on small seedlings in both successional states, this effect breaks down once seedlings mature into saplings. Studies in non-urban forest ecosystems have also shown that canopy presence has a facilitative effect, improving survival and establishment of small seedlings, but this effect transitions to neutral or negative as seedlings grow towards the canopy (Miriti, 2006; Paterno *et al*., 2016; Urza, Weisberg, Chambers, & Sullivan, 2019). This is attributed to ontogenetic changes in niche requirements as trees increase in size, in combination with shading and increased competition for resources (Messier *et al*., 1999; Paterno *et al*., 2016). Small seedlings are often more shade tolerant than saplings as they have a higher ratio of photosynthetic to non-photosynthetic biomass and thus lower maintenance costs (Kneeshaw, Kobe, Coates, & Messier, 2006; Kunstler, Coomes, & Canham, 2009; Waring, 1987). In comparison, larger plants have greater light requirements, but are more tolerant of environmental stresses such as large swings in temperature due to their larger carbon reserves (Grubb, 1977; Niinemets, 2010). For these latter growth stages, environmental buffering benefits provided by the canopy become negligible in the trade off for greater light access.

This has important implications for the management of urban restoration plantings. High density planting to fast track canopy closure of early successional plantings is considered best practice in urban forest restoration due to high risk of invasion by herbaceous weeds when light availability is initially high (McAlpine, Lamoureaux, & Westbrooke, 2015; Standish, Robertson, & Williams, 2001). This practice is vital to reduce weed control costs while creating conditions conducive to seedling regeneration as quickly as possible. However, recent studies have also shown that restored forests often have an excess of canopy trees that are all the same age due to these high initial planting densities and being planted in a single initial cohort (Oliveira, Oliveira, Suganuma, & Durigan, 2019; Sasaki *et al*., 2018). While important for establishment, this planting design does not foster natural forest patch dynamics, which include periodic canopy tree death and light gap formation (Brokaw & Busing, 2000; Muscolo, Bagnato, Sidari, & Mercurio, 2014). The spatial result of forest patch dynamics is multiple generations of trees growing in close proximity and many-layered height tiers (Muscolo *et al*., 2014; Schliemann & Bockheim, 2011). Therefore, best practice for initial canopy establishment can potentially later constrain or slow recruitment of saplings due to low light availability, and limit the value of enrichment planting without selective thinning of the overstory to mimic natural patch dynamics (Oliveira *et al*., 2019; Piana *et al*., 2019). My results are consistent with prior work (e.g. Paterno *et al*. 2016) exemplifying that low canopy openness no longer has a positive effect beyond the seedling stage.

Once reconstructed forests host an understory of established native juvenile trees, practitioners should evaluate whether light is sufficient for continued growth, and consider strategic canopy thinning through removal of branches if not. Small canopy gaps create favourable environments for the growth of juvenile trees by increasing light availability without increasing exposure to microclimatic extremes (Lusk & Laughlin, 2017). Yet, natural light-gap dynamics can take a long time to re-establish (Suganuma & Durigan, 2015). Strategic canopy thinning five or six decades after initial plantings will aid successional progression if combined with enrichment planting, although trade-offs will need to be carefully considered. If sites have a high risk of invasion by herbaceous weeds (common in urban spaces) or are heavily used and at risk from anthropogenic disturbances such as trampling, dense stands are likely to be more protected (Lehvävirta & Rita, 2002). Furthermore, to encourage recruitment of mid-late successional species it is important that thinning is conservative and only small gaps are created to ensure that cooler understory temperatures are maintained (Schliemann & Bockheim, 2011), and light demanding species with higher growth rates are not given a competitive advantage (Kneeshaw & Bergeron, 1998).

Herbaceous ground cover

Consistent with other studies (e.g. Kuijper *et al*., 2010; McAlpine *et al*., 2015; Standish *et al*., 2001; Vandenberghe, Freléchoux, Gadallah, & Buttler, 2006), results showed that herbaceous ground cover has a negative effect on seedlings at the smallest growth stages. Abundance of small seedlings is signficantly lower amongst high herbaceous ground cover, possibly due to competitive pressure for resources. However, once seedlings overtop herbaceous weeds, they are no longer subject to their competition pressure. Canopy formation is also an important influence as it causes fundamental changes in the understory environment and therefore what species can thrive there. For example, a shift towards less light can cause senescence of light-demanding herbaceous weeds in addition to meeting niche requirements of native tree species (McAlpine *et al*., 2015; Standish *et al*., 2001). These results indicate that if strategic canopy thinning is undertaken to create light gaps, practitioners should wait until spontaneously regenerating trees or planted enrichment species have reached sapling height (~135cm) to safeguard them against the detrimental effects of competition with herbaceous weed cover.

Temperature

Temperature drives abundance of juvenile trees depending on their successional status. Small, early successional seedlings regenerate across a range of soil and air temperatures, while mid-late successional seedlings are more abundant in cooler, stable microclimates, regardless of growth stage. This is likely due to physiological differences between the two groups. Early successional species typically have smaller leaves, faster growth rates, and are adapted to tolerate more difficult hot and dry environmental conditions (Cornelissen, 1999; Lebrija-Trejos *et al*. 2010). In comparison, mid-late successional species typically have traits related to the acquisition and conservation of resources (Lebrija-Trejos *et al*., 2010). For example, they tend to have larger leaves which enhance net carbon gain by intercepting more light in shaded conditions (Cornelissen, 1999; Lebrija-Trejos *et al*., 2010). However, large leaf size also increases the likelihood these species will sustain large water losses from transpiration and experience embolism under high temperatures if adequate water is not available (Duan *et al*., 2018).

Urban forest microclimate will not only be affected by canopy cover, but also by forest patch size, patch shape (extent of edges), and urban heat island impacts (Davies-Colley, Payne, & Van Elswijk, 2000; Oke et al., 1989). My results indicate that these dynamics will affect the regeneration and establishment of mid-late successional species. This has important implications for ecological restoration practice in and outside cities. Because of the urban heat island effect, city conditions today provide a window into conditions in rural environments under future global climate change scenarios. Future research exploring the interactions of regional climate (i.e. water deficit scales) and forest understory microclimate would provide further valuable insights, as drought is also known to have a strong negative effect on seedling recruitment and survival (Anderson-Teixeira *et al*., 2013)

Forest age and forest patch size

Multiple regression allowed me to disentangle the relative impacts of forest age and forest patch size and find they differ by successional status of the juvenile trees. When modelled with patch size, forest age is an important driver of small seedling abundance regardless of successional status, while patch size is important only for the smaller growth stages of mid-late successional species. This indicates that conditions become more suitable for the germination and establishment of both early and mid-late successional seedlings as forests age, although mid-late successional species are likely to be more abundant in large, older forests than in small patches.

This could be due to dispersal constraints or environmental stressors innate to small urban forest patches limiting the colonisation and survival of mid-late successional species. Previous research has shown that birds preferentially visit larger forest patches (Cole, Holl, & Zahawi, 2010; Fink, Lindell, Morrison, Zahawi, & Holl, 2009). Mid-late successional species are more likely to have large seeds that are bird dispersed and so small forest patch size can reduce the chance that seeds will arrive (Kelly *et al*., 2010; Wotton & Kelly, 2011). Small forest patches also have a high edge to interior ratio. As a result, they are subject to greater edge effects such as increased light availability and greater temperature fluctuations (Davies-Colley et al., 2000). As previously noted, these conditions can be detrimental to mid-late successional species that prefer cooler temperatures, and can lead to increased competition through growth of light-demanding herbaceous ground cover. The negative impact of edge effects in small forest patches are likely to be amplified by urban heat island effects that further increase drought and temperature stress (Shochat, Warren, Faeth, McIntyre, & Hope, 2006), creating inhospitable conditions for mid-late successional species in early growth stages.

Neither age nor patch size had a significant effect on saplings, regardless of successional status. This could because many sites contain saplings that have been planted during initial restoration efforts, which may be obscuring the effects of age or patch size on spontaneously grown sapling abundance. Nevertheless, regardless of source, the presence of both early and mid-late successional saplings in restored forests reveals what conditions are suitable for their continued survival. Even if seeds arrive but cannot successfully germinate or establish, recruitment limitations can likely be bypassed by enrichment planting of saplings of a sufficient height (Young *et al*., 2005). My results show that even small forest patches are worthy of enriching with mid-late successional tree species once the forest reaches a certain age (~10 years), as sapling-aged plants can persist there.

2.6 Conclusion

These results have important implications for the management of restored urban forests.

First, to promote natural (i.e. spontaneous) regeneration, management approaches should vary depending on the age of a restored forest. Early in forest development, practitioners should prioritise canopy closure and removal of herbaceous weed species to encourage regeneration and recruitment of small native tree seedlings. Later in forest development when saplings are evident in the understory, canopy thinning should be undertaken to create small light gaps and recruit these mid-late successional saplings to the canopy. Second, management actions should be tailored to the forest size. In small restored forest patches affected by limited seed dispersal and edge effects, enrichment planting of mid-late successional species is absolutely vital to ensure forest successional progression. Restoration practitioners should invest in large mid-late successional seedlings (>100cm) to maximise their resilience and chances of survival in urban environments.

In summary, this study shows that drivers of native tree seedling recruitment differ according to species successional status and the growth stage of seedlings in restored urban forests. The effects of canopy openness appear to be generalizable across successional status, while the effects of microclimate and patch size vary. Once juvenile trees reach the sapling growth stage an ontogenetic shift occurs and the positive effects of full canopy cover on seedling survival becomes neutral. At this point, saplings no longer require canopy cover for survival and instead likely require more light for growth to recruit into the canopy. For establishment of early successional species forest age is the most important factor, but mid-late successional species require the cooler microclimates associated with low canopy openness and larger patch size. Small forest patches are less likely to host natural regeneration of mid-late successional tree species because of sensitivity at the seedling stages, but still provide conditions suitable for the persistence of more resilient saplings if they are introduced through enrichment planting.

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2.8 Appendix 2.1

Table 2.3. Restored urban forest site name, location, age and patch size.

Site	City	Age (years)	Size (ha)
Tauhara	Hamilton	30	0.83
Minogue	Hamilton	38	2.44
Waiwhakareke old	Hamilton	13	1.32
Waiwhakareke young	Hamilton	6	7.95
Hamilton lake	Hamilton	19	3.2
Brymer	Hamilton	22	1.1
Featherstone	Hamilton	17	0.1
Tills	Hamilton	23	0.4
Avalon	Hamilton	12	0.46
McCardle's bush	Tauranga	31	9.86
Carmichael playground	Tauranga	11	1.36
Ohauiti old	Tauranga	17	0.1
Bethlehem	Tauranga	21	2.82
Ohauiti intermediate	Tauranga	14	0.32
Johnson reserve	Tauranga	22	14.9
Ohauiti young	Tauranga	5	4.37
Millbrook	Tauranga	16	4.75
Challenge reserve	Tauranga	14	2.13
Waipu lagoon	New Plymouth	30	1.49
Salaman reserve	New Plymouth	27	4.26
Airport	New Plymouth	28	1.31
Te henui	New Plymouth	13	12
Pukekura park	New Plymouth	10	18.8
Herekawe coastal	New Plymouth	4	8.9
Peringa park	New Plymouth	12	1.19
Herekawe inland	New Plymouth	16	2.71
Huatoki restored	New Plymouth	46	30
Halliwell reserve	Napier	6	1.37
Harakeke reserve	Napier	7	0.37
Otatara park	Napier	25	0.65

Dolbel young	Napier	8	7.2
Westshore reserve	Napier	19	0.6
Friends bush	Napier	37	0.34
Karamu stream	Napier	20	0.1
Dolbel kauri walkway	Napier	23	11
Dolbel Colenso block	Napier	23	0.28
Mt albert	Wellington	26	7.34
Alexandra road	Wellington	18	18.5
Manawa kariori north	Wellington	27	2.64
Manawa kariori south	Wellington	27	2.64
Tawatawa reserve	Wellington	24	2.99
Izard park	Wellington	22	3.32
Owen street	Wellington	8	14.9
Telford terrace	Wellington	14	8.77
Old chest hospital	Wellington	8	3.19
Whakatu drive	Nelson	16	1.9
Newman grove	Nelson	28	0.11
Murphy reserve young	Nelson	8	1.65
Murphy reserve old	Nelson	17	1.65
Pipers reserve	Nelson	6	13.2
Waste station	Nelson	28	0.22
Whitehead park	Nelson	20	20.3
Titoki	Nelson	13	31.8
Bobs track	Nelson	29	5
Matawai	Christchurch	43	0.44
Riccarton bush	Christchurch	39	9.65
Marshland road	Christchurch	12	0.49
Aynsley terrace	Christchurch	28	0.43
Radcliffe road	Christchurch	7	1.7
Styx living lab	Christchurch	15	0.27
Travis wetland	Christchurch	18	0.27
Wigram east	Christchurch	25	0.81
Halswell quarry	Christchurch	17	0.43

Upper leith walkway	Dunedin	21	18.8
Signal hill	Dunedin	29	148
Prospect park	Dunedin	21	18.8
Frasers gully	Dunedin	15	0.98
Craigieburn young	Dunedin	7	18
Craigieburn intermediate	Dunedin	18	18
Craigieburn old	Dunedin	58	18
Island park	Dunedin	9	77.3
Estuary walkway	Invercargill	21	0.99
Waihopai river	Invercargill	11	0.1
Rance covenant old	Invercargill	22	27
Thomsons bush exterior	Invercargill	7	20.9
Thomsons bush interior	Invercargill	9	20.9
Bushy point young	Invercargill	12	27
Kew bush	Invercargill	19	3.79
Rance covenant young	Invercargill	18	27

2.9 Appendix 2.2

Table 2.4 All plant species identified in this study, along with successional status and growth stage.

Scientific name	Successional Status	Growth stage
Agathis australis	Late	Recruited, sapling, adult
Alectryon excelsus	Mid	Germinated, recruited, sapling, adult
Aristotelia serrata	Early	Germinated, recruited, sapling, adult
Beilschmiedia tawa	Late	Germinated, recruited, sapling
Brachyglottis repanda	Early	Germinated, recruited sapling, adult
Carpodetus serratus	Early	Germinated, recruited, sapling, adult
Coprosma areolata	Mid	Germinated, recruited, sapling
Coprosma crassifolia	Early	Germinated, recruited, sapling
Coprosma grandifolia	Mid	Germinated, recruited, sapling, adult
Coprosma linariifolia	Mid	Germinated, recruited, adult
Coprosma lucida	Early	Germinated, recruited, sapling, adult
Coprosma obconica	Early	Germinated, recruited, sapling
Coprosma propinqua	Early	Germinated, sapling

Coprosma propinqua x robusta	Early	Germinated, recruited, sapling, adult
Coprosma repens	Early	Germinated, recruited, sapling, adult
Coprosma rhamnoides	Early	Germinated, recruited, sapling
Coprosma rigida	Early	Germinated, recruited, sapling, adult
Coprosma robusta	Early	Germinated, recruited, sapling, adult
Coprosma rotundifolia	Mid	Germinated, recruited, sapling, adult
Coprosma spathulata	Late	Germinated, recruited, sapling
Coprosma species	Early	Germinated, recruited
Coprosma tenuifolia	Late	Germinated, recruited, sapling, adult
Coprosma virescens	Early	Germinated, recruited, adult
Cordyline australis	Early	Germinated, recruited, sapling, adult
Cordyline banksii	Early	Adult
Corokia buddleioides	Early	Germinated, recruited, sapling, adult
Corynocarpus laevigatus	Early	Germinated, sapling, adult
Cyathea dealbata	Early	Recruited, sapling
Cyathea medullaris	Early	Sapling, adult
Dacrycarpus dacrydioides	Early	Germinated, recruited, sapling, adult
Dacrydium cupressinum	Late	Adult
Dicksonia squarrosa	Early	Sapling, adult
Dodonaea viscosa	Early	Germinated, recruited, sapling, adult
Dysoxylum spectabile	Late	Germinated, recruited, sapling, adult
Elaeocarpus dentatus	Late	Recruited, sapling
Elaeocarpus hookerianus	Late	Germinated, recruited, sapling, adult
Entelea arborescens	Mid	Adult
Fuchsia excorticata	Mid	Germinated, recruited, sapling, adult
Fuscospora cliffortioides	Mid	Adult
Fuscospora fusca	Mid	Sapling, adult
Geniostoma ligustrifolium	Late	Germinated, recruited, sapling, adult
Griselinia littoralis	Mid	Germinated, recruited, sapling, adult
Hedycarya arborea	Mid	Germinated, recruited, sapling, adult
Hoheria angustifolia	Early	Germinated, recruited, sapling, adult
Hoheria glabrata	Early	Germinated, recruited
Hoheria populnea	Early	Germinated, recruited, sapling, adult
Hoheria sexstylosa	Early	Germinated, recruited, sapling, adult
Knightia excelsa	Mid	Germinated, recruited, sapling, adult
Kunzea robusta	Early	Germinated, recruited, sapling, adult

Laurelia novae-zelandiae	Late	Germinated, recruited, sapling, adult
Leptospermum scoparium	Early	Germinated, recruited, sapling, adult
Litsea calicaris	Late	Germinated, recruited
Lophomyrtus obcordata	Early	Germinated, recruited, sapling, adult
Melicope simplex	Mid	Germinated, recruited, sapling,
Melicope ternata	Mid	Germinated, recruited, sapling, adult
Melicytus lanceolatus	Mid	Germinated, adult
Melicytus micranthus	Early	Germinated, sapling
Melicytus ramiflorus	Early	Germinated, recruited, sapling, adult
Metrosideros excelsa	Early	Recruited, sapling, adult
Metrosideros robusta	Late	Recruited, sapling, adult
Myoporum laetum	Early	Germinated, recruited, sapling, adult
Myrsine australis	Early	Germinated, recruited, sapling, adult
Myrsine divaricata	Early	Germinated, recruited, sapling
Nestegis cunninghamii	Late	Adult
Olearia arborescens	Early	Recruited, sapling, adult
Olearia avicenniifolia	Early	Germinated, recruited, sapling
Olearia lineata	Early	Sapling, adult
Olearia odorata	Early	Germinated, recruited, sapling,
Olearia paniculata	Early	Germinated, recruited, sapling, adult
Olearia solandri	Early	Germinated, recruited, sapling, adult
Olearia traversiorum	Early	Sapling, adult
Pennantia corymbosa	Early	Germinated, recruited, sapling,
Piper excelsum	Early	Germinated, recruited, sapling, adult
Pittosporum colensoi	Mid	Adult
Pittosporum crassifolium	Early	Germinated, recruited, sapling, adult
Pittosporum eugenioides	Early	Germinated, recruited, sapling, adult
Pittosporum ralphii	Early	Germinated, recruited, sapling, adult
Pittosporum species	Early	Recruited, sapling,
Pittosporum tenuifolium	Early	Germinated, recruited, sapling, adult
Pittosprum hybrid	Early	Sapling, adult
Plagianthus divaricatus	Early	Sapling,
Plagianthus regius	Early	Germinated, recruited, sapling, adult
Podocarpus laetus	Mid	Adult
Podocarpus totara	Mid	Germinated, recruited, sapling, adult
Prumnopitys ferruginea	Late	Germinated, recruited, sapling, adult

Prumnopitys taxifolia	Mid	Germinated, recruited, sapling, adult
Pseudopanax arboreus	Early	Germinated, recruited, sapling, adult
Pseudopanax colensoi var. colensoi	Mid	Germinated, recruited, sapling
pseudopanax colensoi var. ternatus	Early	Germinated, recruited, sapling, adult
Pseudopanax crassifolius	Mid	Germinated, recruited, sapling, adult
Pseudopanax crassifolius x lessonii	Early	Germinated, recruited, sapling,
Pseudopanax ferox	Mid	Sapling, adult
Pseudopanax hybrid	Early	Germinated, recruited, sapling, adult
Pseudopanax laetus	Early	Germinated, recruited, sapling, adult
Pseudopanax lessonii	Early	Recruited, sapling, adult
Pseudowintera colorata	Early	Adult
Rhopalostylis sapida	Mid	Germinated, recruited, sapling,
Schefflera digitata	Early	Adult
Sophora chathamica	Early	Germinated, sapling, adult
Sophora microphylla	Early	Germinated, recruited, sapling, adult
Sophora molloyi	Early	Recruited, sapling, adult
Sophora tetraptera	Early	Germinated, recruited, sapling, adult
Streblus heterophyllus	Mid	Germinated, recruited, sapling, adult
Urtica ferox	Early	Germinated, recruited,
Veronica parviflora	Early	Sapling, adult
Veronica salicifolia	Early	Adult
Veronica speciosa	Early	Sapling,
Veronica stricta	Early	Sapling, adult
Veronica strictissima	Early	Adult
Vitex lucens	Mid	Recruited, sapling, adult
Weinmannia racemosa	Late	Recruited, sapling

3 CHAPTER 3

A TENUOUS LINK: INFORMATION TRANSFER BETWEEN ECOLOGICAL RESEARCH AND RESTORATION PRACTICE

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3.1 Abstract

Despite a growing theoretical understanding of restoration ecology, this scientific knowledge is not typically well linked to restoration practitioner decision-making. Restoration projects are increasing worldwide due to the myriad ecological, social, cultural and economic benefits restored ecosystems provide, but if not linked to ecological theory these projects may not achieve long-term goals such as restoration of ecosystem structure and function. This limited transfer of knowledge is detrimental to public projects in particular because such ventures require sustained support by a high turnover of many stakeholders. We conducted a New Zealand-wide survey of urban forest restoration practitioners using a combination of open-ended and fixed-answer questions to better understand drivers behind their planning, implementation, and management of restoration projects. We chose urban forest restoration in public spaces as a model system because these practitioners are faced with extraordinary social pressures to restore under severely degraded ecological conditions, and therefore require reliable, efficient methods for success. Our goals were to 1) understand trends in practitioner decision making 2) identify weak links in knowledge transfer between restoration ecology research and ecological restoration practice, and 3) suggest targeted methods to strengthen information transfer between researchers and practitioners. Our survey identified a tenuous link between current researcher knowledge regarding best-practice restoration and practitioner actions. While best-practice recommends setting of specific, measurable objectives to restore ecosystems, practitioners tend to have broad, vague objectives and focus on restoration of simple ecosystem properties such as a canopy of early successional plant species. Practitioners also prioritise management actions like planting or weed control over project planning and quantifiable monitoring, despite projects requiring these for long-term success. Results indicate practitioners source knowledge equally through interpersonal interactions (e.g. with ecologists and fellow practitioners) and traditional forms of best-practice communication (e.g. books and scientific articles). This suggests that prioritising interactive, interpersonal modes of science communication and encouraging collaboration between scientists and practitioners could help strengthen knowledge transfer. Additionally, providing practitioners with time-saving resources (e.g. restoration planning and monitoring templates), adequate funding, and guidance to navigate socio-ecological constraints that arise in urban projects will improve restoration outcomes.

Keywords: decision-making, practitioner, research-implementation gap, restoration ecology, restoration practice, survey, urban ecology, urban restoration.

3.2 Introduction

The discipline of restoration ecology has matured into a mainstream science with a rapidly growing associated body of peer-reviewed literature (Wortley, Hero, & Howes, 2013). Discoveries reported in this literature provide key recommendations for how to cost-effectively restore resilient, self-sustaining ecosystems (MacMahon, 1998; Seavy & Howell, 2010; Wallace & Clarkson, 2019). However, this knowledge is not always implemented in restoration projects because it is typically available only through publications in costly, restricted-access journals (Anderson, 2014; Pullin & Knight, 2005; Sunderland, Sunderland-Groves, Shanley, & Campbell, 2009). Journal publications are of low value to many practitioners due to the use of esoteric terminology, management recommendations that are un-tested in the field, or recommendations that are difficult to extrapolate to different restoration environments (Anderson, 2014; Hulme, 2014). Furthermore, practitioners typically have limited time to read, translate and apply research findings, and so tend to rely on their own personal experience as management guidance instead (Anderson, 2014; Hulme, 2014; Pullin & Knight, 2005). These dynamics result in a separation between restoration ecology researchers and ecological restoration practitioners and therefore a disconnect between scientific knowledge and practitioner implementation (Anonymous, 2007; Burke & Mitchell, 2007).

The urban research-implementation gap

This research-implementation gap has been well documented in the applied sciences (Hulme, 2014), including restoration ecology (Allen, Covington, & Falk, 1997; Bernhardt *et al*., 2007; Cabin, Clewell, Ingram, McDonald, & Temperton, 2010) and the closely related field of conservation management (Anonymous, 2007; Knight *et al*., 2008; Lauber, Stedman, Decker, & Knuth, 2011; Sutherland, Pullin, Dolman, & Knight, 2004). Little is known about how it manifests in the context of urban forest restoration, but it is likely also hindered by break downs, weak links and barriers (Fig. 3.1). Urban restoration projects have recently gained momentum in cities worldwide in recognition of the myriad benefits they provide (Clarkson & Kirby, 2016; Oldfield *et al*., 2015; Soga & Gaston, 2016; Standish *et al*., 2013). These include: increased native biodiversity (Alvey, 2006; Sandström *et al*., 2006), provision of ecosystem services (Endreny, 2018; Nowak & Crane, 2002), improved human health and wellbeing (Kardan *et al*., 2015; Takano, Nakamura, & Watanabe, 2002), and increased opportunities for urban residents to reconnect with nature (Miller, 2005; Rosa, Profice, & Collado, 2018). As a relatively new field, urban restoration is an opportunity to implement restoration projects that provide these numerous benefits. There is increasing capacity to achieve this if using a scientifically underpinned approach, yet successful implementation remains challenging and often requires more than scientific knowledge alone (Higgs, 2005).

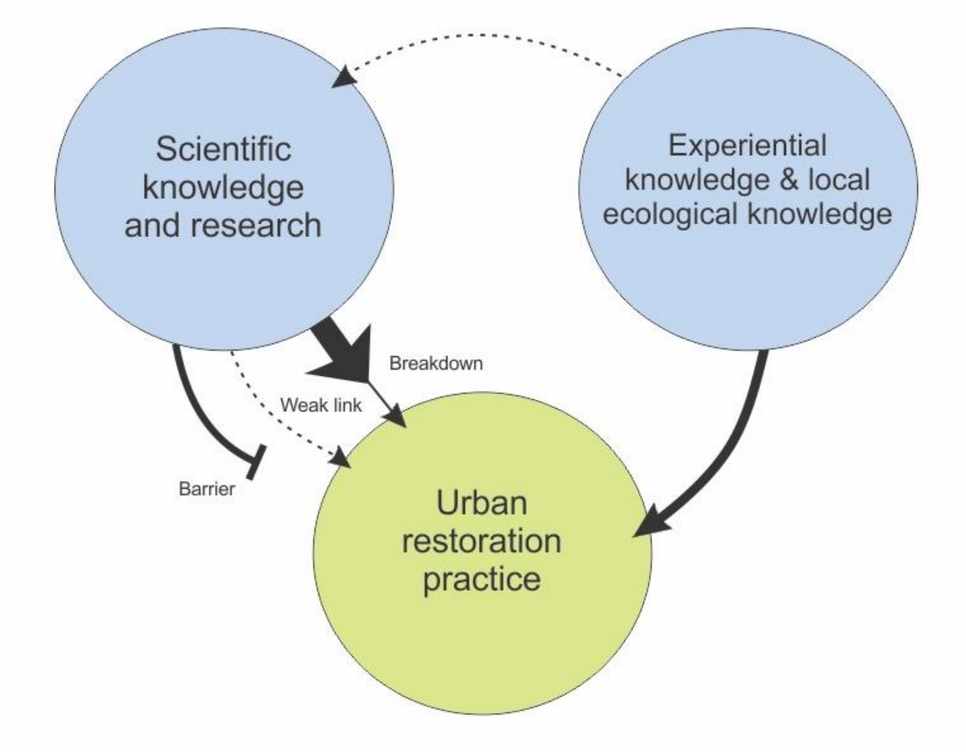

Figure 3.1. Two bodies of knowledge are important for successful urban restoration practice: Scientific knowledge and local experiential knowledge.

Urban forests are socio-ecological systems

Practitioner perspectives, values and knowledge determine the development of restoration plantings. Practitioners must navigate decisions regarding restoration objectives and which methods to use in their unique projects (Burke & Mitchell, 2007; Hertog & Turnhout, 2018). Tailoring management recommendations to a specific project is challenging even when aware of current research and best-practice management. Further, even highly relevant research findings may not address the range of complex external factors and constraints practitioners face when making decisions (Anderson, 2014). Restoration projects typically include multiple and sometimes conflicting objectives (Hagger, Dwyer, & Wilson, 2017), and the urban context presents a unique set of altered ecological dynamics such as soil compaction and modification, low native propagule pressure, local extirpation of agents of seed dispersal or pollinators, and high exotic propagule pressure (Standish *et al*., 2013; Sullivan *et al*., 2009). Urban projects also entail unique social constraints such as a human activities, stakeholder preferences, and governance systems that determine what is feasible in cities (Aronson *et al*., 2016; Burke &

Mitchell, 2007; McPhearson *et al*., 2016; Reid, 2018; Stanturf *et al*., 2014). Design of urban ecological restoration projects must simultaneously address these social and ecological elements in order to realise long-term success (Grimm *et al*., 2008).

Research objectives

Understanding how urban restoration practitioners use science to inform restoration is an important first step towards strengthening the science-practice link (Hulme, 2014). Despite increases in focus and funding for urban forest restoration, there is little information available about how practitioners incorporate science into their motivations, knowledge, and decision-making processes (but see Jay & Stolte, 2011). Here, we aimed to address this knowledge gap by investigating how restoration practitioners in New Zealand make decisions in their efforts to re-establish native urban forests. To achieve this, we distributed an online survey consisting of a combination of quantitative and qualitative questions designed to examine practitioner motivations, knowledge, and decision-making processes regarding management and monitoring of their forest restoration projects. Our main objectives were to: 1) understand trends in practitioner decision making 2) identify weak links in knowledge transfer between restoration ecology research and ecological restoration practice, and 3) suggest targeted methods to strengthen the information transfer link between restoration researchers and practitioners. We summarise our findings in five sections entitled: 1) *What's the goal? Urban forest restoration objectives*, 2) *What's most important? Urban forest restoration priorities*, 3) *How to manage? Restoration site ongoing care*, 4) *Did it work? Urban forest restoration monitoring* and 5) *What caused problems? Setbacks faced by practitioners.*

3.3 Methods

Survey participant selection and data collection

We used an online survey created using SurveyMonkey® to collect data from urban forest restoration practitioners throughout New Zealand. Surveys provide a useful method for identifying what practices exist and exploring people's perceptions, values, attitudes, and levels of knowledge (Crandall *et al*., 2018). An invitation to complete the survey was emailed to 114 individuals we had previously worked with. This included regional and city council staff, independent consultants, and restoration group volunteers involved in urban forest restoration. The survey invitation was sent on 12 November 2018 and was open to responses until 10 December 2018. Two reminder emails were sent 10 and 20 days after the inital email invitation, respectively. Individuals were asked to forward the email to others they knew were involved in urban forest restoration and an invitation to complete the survey was also publicised through the host research program (People, Cities and Nature) Facebook page. As it was not possible to get a fully representative sample of all urban forest restoration practitioners, results are only intended to give an indicative overview of urban restoration practice. Context-based studies, such as this which focuses on only urban forest restoration practitioners, are valuable for producing practice orientated knowledge regarding what is happening in a given setting, and ensuring complexities or contradictions are not lost in generalisations (Hodgetts & Stolte, 2012).

Survey design

The survey consisted of a combination of 18 questions which were quantitative (fixed answer) or qualitative (free-text) (Appendix 3.1). The survey was designed to be thorough without high completion time (~15-20 minutes) so as to avoid participant fatigue and diminishing levels of detail in responses (Braun & Clarke, 2013). The majority of questions were free-text such as 'What is your restoration project objective(s)?', however four questions were closed-ended with two of these (Q. 5 & Q. 7) using Likert scales - a commonly used tool for measuring attitudes in social science research (Allen & Seaman, 2007; Croasmun & Ostrom, 2011). Likert scale questions required the survey participant to rate importance according to a five-tiered categorical scale. An assymetrical Likert scale was chosen to provide a neutral response option, and reduce the likelihood of response bias (Croasmun & Ostrom, 2011). Categories were: 'Not at all important', 'Of little importance', 'Moderately important', 'Important', and 'Very important'. A 'N/A' category was also available if participants did not consider the question relevant. Fixed-answer questions were appended with requests such as: 'Please specify any other important considerations not listed', and 'Please specify names or details of resources used where possible', to ensure information not listed as response options in the survey design was also gathered. The survey preface included a question asking recipients to confirm that their restoration project was located within city limits. Our final question – 'Is there anything else you would like to add?' ensured participants could provide any additional information they considered important.

The survey was pilot-tested by one young and one elderly person who were involved in restoration projects on a professional and avocational basis, respectively. Based on feedback we refined the survey to ensure questions and terminology were clear to both demographics. Names identifying individuals, groups and locations have been omitted from quotes to maintain participant confidentiality.

Survey analysis

We summarised data from fixed answer questions by frequency with results presented as proportions of the total responses recieved. We treated the Likert-scale responses as continuous data and used descriptive statistics to calculate weighted means and percentages of responses (Sullivan & Artino 2013), allowing a comprehensive overview of these results. For qualitative data, we used thematic analysis to identify patterns of meaning across open-ended survey responses (Braun & Clarke, 2006). This approach offers an accessible, flexible method for analysing qualitative data while still allowing for a complex and detailed understanding (Braun & Clarke, 2006). Survey data was analysed using NVivo (NVivo 12, www.qsrinternational.com/ products_nvivo.aspx). This software enables passages of text identified as relating to a theme or category to be manually coded or tagged and indexed into "nodes" (Bazeley & Jackson, 2013; Silver, 2014). Coding and analysis involved muliple stages of coding and re-coding the data into nodes based on themes and sub-themes identified in the literature (deductive) as well as in the survey responses (inductive) (Swain, 2018).

The survey responses for restoration project objectives were classified according to a slightly modified version of the categories used by Hallett *et al*. (2013) (Appendix 3.2). These categories were based on the attributes of restored ecosystems and relate to ecosystem form, function and stability as outlined in the Society for Ecological Restoration Primer on Ecological Restoration (2004), as well as additional social goals and related attributes as defined by Hallet *et al* (2013). We use the term 'ecosystem resilience' rather than 'ecosystem stability' which was used by Hallet *et al*., as ecosystems are dynamic and rarely stable. The social goal, 'economic benefits' used by Hallet *et al*. was changed to 'provision of ecosystem services' as this more accurately reflected our data, and the social goal 'cultural values' was split into 'societal values' and 'Indigenous cultural values or use' to distinguish between these two important aspects of culture. If responder's objectives were too broad to assign to any of these categories (i.e. 'restoration of urban forest reserves') they were excluded from categorisation. Nodes were also created for some questions on the basis of specific categories. For example, for questions such as 'What maintenance has occurred during the course of restoration activities?' (Q13) responses were coded into several categories that were determined by responses (e.g. non-native plant control, enrichment planting, pruning, mulching, irrigation). These categories were then summarised by frequency and the proportion of participants that answered the question who undertook that specific activity was calculated. Blank or incomplete responses to survey questions were excluded from analysis. Sample size varies slightly between questions because not all participants answered all questions.

Our motivation for this research was to understand the link between ecological theory and practioner implementation in order to improve outcomes of urban forest restoration projects. Therefore our interpretation of survey responses and selection of themes and patterns are of interest are framed in this context.

3.4 Results and Discussion

Survey response rates and demographics

Of the 114 practitioners we contacted, 67 completed the survey (59% response rate). Two participants' answers were omitted before analysis as their work was not urban or involved roadside specimen trees rather than forest patch ecosystem restoration. Remaining participants (n=65) consisted of 23 people involved with restoration as part of their employment on a professional basis (35%), and 42 were involved on an avocational basis as volunteers (65%). Of these participants, 55% identified as male, 45% identified as female (n=65). Not all participants provided further demographic information, but the majority of those who did were over 60 years of age (58%, n=58), and almost all (98%) identified as New Zealand European/Pākehā (n=57). Responses were from participants in 10 New Zealand cities (Wellington, Hamilton, Auckland, Tauranga, Napier, Chirstchurch, Invercargill, Nelson, Dunedin, New Plymouth) with the highest number of responses in Wellington (21.5%), Hamilton (16.9%) and Auckland (15.4%).

We summarise our findings in five sections that follow.

1) What's the goal? Urban forest restoration objectives

Consistent with other studies (e.g. Galbraith, Bollard-Breen, & Towns, 2016; Jones & Kirk, 2018) we found that project objectives are rarely well defined (e.g. "restore the reserves native flora", "improve bio-diversity and the occurrence of native forested areas within our Urban areas"). This is despite research indicating that clearly defined objectives are a key component of successful restoration (Hallett *et al*., 2013; Palmer *et al*., 2005). It has been hypothesized that such trends may reflect avocational perspectives of restoration as opposed to that of professionals (Galbraith *et al*., 2016; Weng, 2015). However, here we found professional practitioners also describe poorly defined objectives, while at least one clearly defined objective was described by an avocational practitioner.

The majority of participants reported objectives related to ecosystem form (Fig. 2), with attributes related to indigenous species most frequently reported (e.g. "to clear the area of invasive weeds and plants, [and] to plant natives over the cleared area"). These results are similar to those found by Hallett *et al*., (2013). Interestingly, a minority of participants described objectives characteristic of short-term restoration milestones such as achieving the "establishment of primary successional species." or "canopy closure". Although these are essential first steps, expanding objectives to include establishment of late-successional species (Laughlin & Clarkson, 2018) and restoration of ecological function (Wallace, Laughlin, Clarkson, & Schipper, 2018; Wright *et al*. 2009) is crucial for long-term forest development and urban restoration success.

Only a minority of participants reported objectives related to ecosystem function (Fig. 3.2). These most commonly included goals relating to restoring ecosystem functional groups (e.g. through the provision of habitat). These results contrast with those of Hallett *et al*., (2013) who found most projects had objectives related to ecosystem function. This may reflect a difference between urban and non-urban restoration objectives. As many urban forest restoration projects are relatively small-scale and multi-purpose, restoring ecosystem function may be considered infeasible, impracticable, or low priority given the many other urban-specific restoration objectives.

Practitioners rarely cited ecosystem resilience as an objective (Fig. 3.2) despite a key definition of restoration success being sustainable forests that persist (Reid, 2018; Society for

Ecological Restoration International Science: Policy Working Group, 2004). Only three practitioners stated a goal for their restored ecosystems to be "self-sustaining", and no practitioners mentioned objectives related to ecosystem resilience to stress events, despite weather-related events being cited as a common setback (see Section 5). Ensuring climate change resilience in restored forests is an urgent challenge (Choi *et al*., 2008; Stanturf *et al*., 2014), but our results show this is yet to be incorporated into urban project objectives. Remedying this is vital, as urban forests are subject to urban heat island effects and thus are even more vulnerable to the extremes of climate change (Oke *et al*., 1989).

Socially-focused objectives were more common than objectives related to ecosystem function and resilience, although still only stated by a minority, lower than found by other studies (Fig. 3.2; e.g. Hallett *et al*., 2013; Peters, Hamilton, & Eames, 2015). This may be detrimental to urban restoration projects as inclusion of social goals can increase community support for projects and thus their long-term sustainability (Fox & Cundill, 2018; Hallett *et al*., 2013; Higgs, 2005; Wallace & Clarkson, 2019). Generally, however, our results indicate human-use values are important in decision-making even if not an explicit objective. Of participants who described social objectives, societal values such as improving amenity or recreational values of the site were most common (20%). This was followed by a minority (11%) focused on the provision of ecosystem services for humans (e.g. listed as firebreak management, flood mitigation, or improved water quality). Restoration objectives focused on indigenous cultural values were absent, although one practitioner expressed a desire to ".see more information and involvement regarding the cultural aspects of [their restoration site]". This failure to incorporate Māori cultural aspirations into restoration projects has been previously noted as a detrimental feature of community group restoration in New Zealand (Wehi & Lord, 2017).

A number of project objectives included a combination of social and ecological attributes. For example, one participant describes their project objective:

> *To restore the health of an urban stream, bring back at least some of the native forest species that would have once grown at our site, encourage birdlife - and almost as important, bring people together on the project and encourage more people to enjoy the park.*

This is positive as it illustrates that projects are capturing opportunities to obtain multiple benefits (Mansourian, Stanturf, Derkyi, & Engel, 2017).

In summary, project objectives described by practitioners are not clearly defined and reflect a tendency to focus on short-term, achievable goals related to ecosystem form as opposed to more long-term, challenging (but essential) commitments. This is understandable given practitioners desire to see tangible short-term results (Cabin, 2007). However, it is important to incorporate objectives related to ecosystem processes, function and persistence, and social values in addition to restoring native vegetation cover. Not doing so may compromise the longevity of plantings and their ability to adapt to altered climate (Stanturf *et al*., 2014), which in turn will compromise amenity values and provision of ecosystem services.

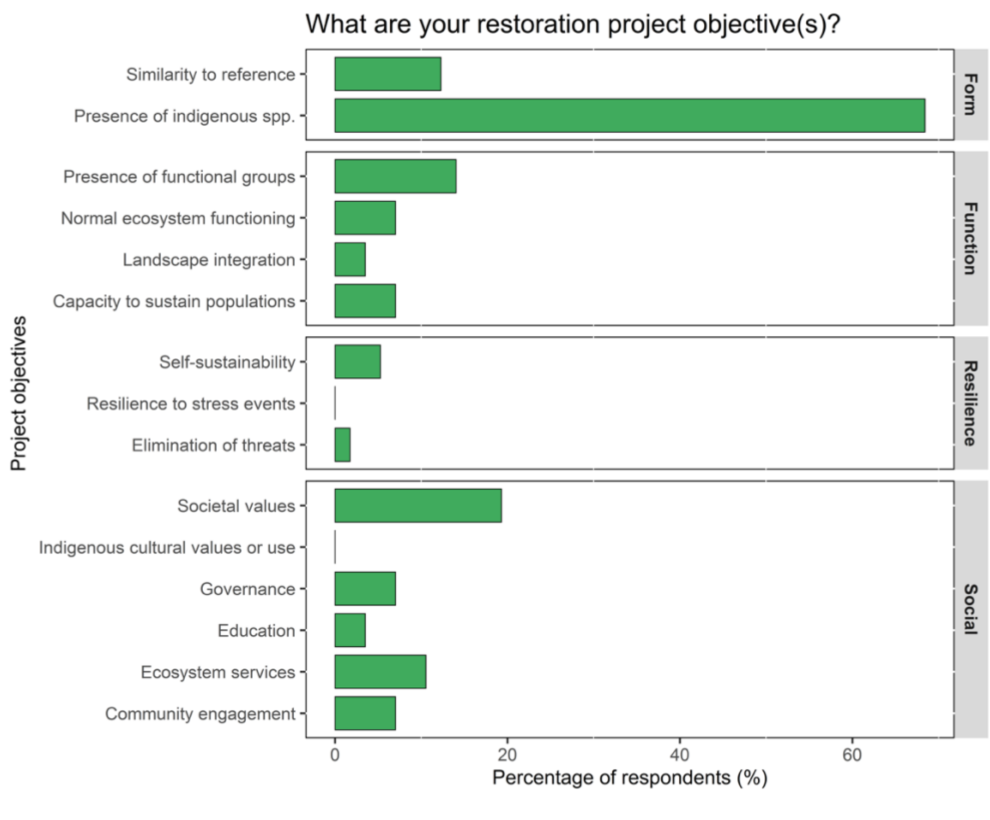

Figure 3.2. Description of project objectives from urban forest restoration practitioners (n=57).

2) What's most important? Urban forest restoration priorities

Restoration practitioners balance a range of considerations when managing projects. Our survey revealed that practitioners' highest priorities were native species planting and native wildlife habitat provision, followed by habitat or species conservation (Fig. 3.3). These aligned with the most common restoration objectives found in section one of this study. Interestingly, practitioners also rate restoring ecosystem processes and function as an important consideration here, despite only a minority including this in their project objectives (section 1; Fig. 3.2). Practitioners are least concerned about ‘restoring to a historical state or pre-human condition’, ‘erosion mitigation’, and ‘climate change reduction through carbon off-setting’ (Fig. 3.3). However, these considerations are still on average rated as important by practitioners.

In free-text sections, practitioners listed additional considerations such as: reducing the costs and difficulties of site management or maintenance, plant availability, weed eradication, amenity values, volunteer interests, and practical considerations regarding what is achievable. A number of participants noted that stakeholder concerns and involvement are important to take into account. For example, one participant notes:

> *“Once [sic] important consideration was to make the restoration accessible and responsive to the local community's needs, interests and concerns: so keeping in mind things like maintaining viewshafts for reserve neighbours, being able to reassure locals that increasing the number of people actively involved in caring for the reserve would be a positive experience (e.g. more passive surveillance of the area would tend to decrease chances of burglaries) rather than negative (i.e. responding to some concerns about attracting strangers into the neighbourhood).”*

Other functions of the restoration site are also an important consideration. One practitioner notes: “Overall the site had to work first and foremost as a stormwater basin, so design of urban forest could not impede on this function.”

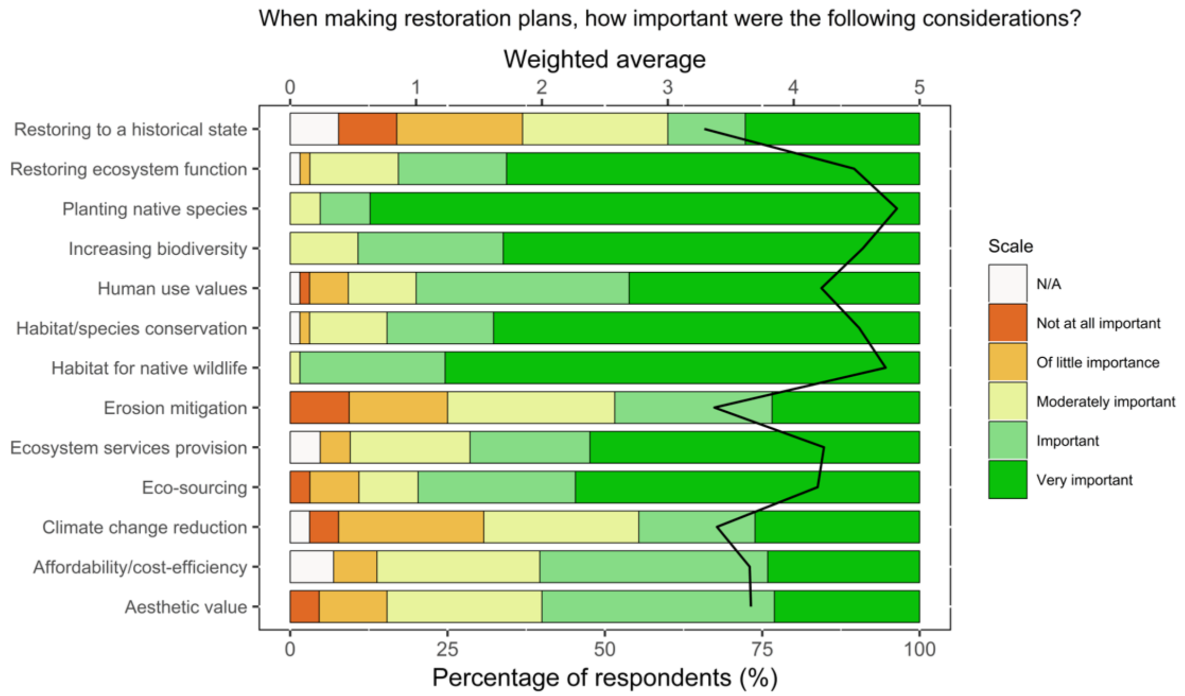

Figure 3.3 Restoration practitioners' (n=64) importance ratings out of 13 considerations (i.e. priorities) when designing restoration plantings.

In regards to selecting what species to plant, restoration practitioners are most concerned with whether plants are native and the ecological suitability of plants for a site, followed by the hardiness or likelihood of survival of the plant species (Fig. 3.4). The lowest rated concerns are personal or public plant preferences, the cost of plants, and their aesthetic value. Further considerations added by practitioners in free-text included: growth rate (so as to suppress weeds), the availability of eco-sourced plants, whether species had historically existed at the restoration site, how locally distinctive the species was, and the ease of management (e.g. *Cordyline australis* (G. Forst.) an indigenous monocot tree commonly known as tī kouka, was mentioned as being undesirable due to its prolific leaf litter). As with planning plantings, participants mentioned that retaining support of local residents was an important consideration when choosing planting species. For example, one practitioner commented:

> *We avoided planting tall tree species in particular areas, to prevent loss of view shafts for surrounding properties. This was a deliberate act, to keep reserve neighbours on-side with the project, and increase likelihood of more active support from neighbours as the project grows.*

Some practitioners note that plant choices are not made by them but rather "governed by what the council has grown and supplied". Specific to urban restoration, one participant also notes the importance of considering the suitability of plants for the public setting of the planting:

> *In some places I avoid some species that council contractors won't recognise as a planting (e.g. native grasses and carexes*[sic]*) or that volunteers won't be able to distinguish from similar weeds (e.g. I don't plant toetoe if there is pampas). Working next to rail, I avoid trip hazards, and need to ensure the size will be right. Working next to footpaths…it's important to ensure it won't become a hazard. I have used nettles in an area we didn't want people going.*

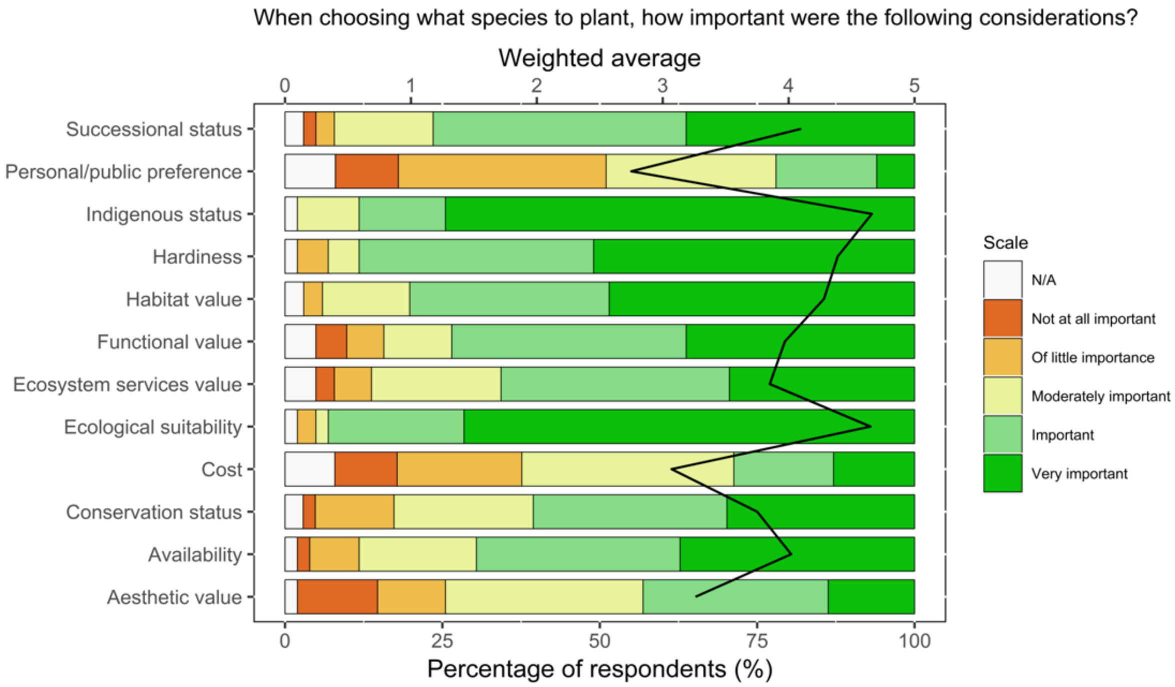

Figure 3.4.Restoration practitioners' (n=63) importance ratings for 12 considerations (i.e. priorities) when choosing what species to plant in an urban forest restoration project. .

Sources of restoration knowledge

Survey participants were also asked what resources they utilized when planning restoration projects. Consistent with other studies (e.g. Bernhardt *et al*., 2007; Seavy & Howell, 2010), we found that practitioners collectively use a wide variety of resources. Both professional and volunteer practitioners used an average of five different types of resources to plan plantings (n=62).

Discussions with co-workers or other volunteers were the primary resource used by both professional and volunteer practitioners, followed by discussions with plant or ecology experts (Fig. 3.5), both results in alignment with other research (Seavy & Howell, 2010; Sutherland *et al*., 2004). Unfortunately, there is survey evidence suggesting practitioners perceive interactions with experts (including through field trips and workshops) as rare opportunities (Seavy & Howell, 2010). This indicates that increasing opportunities for interpersonal dialogue between scientists and practitioners will help strengthen the science-practice link, albeit dependent on scientists having the science communication skills to translate research findings into on-the-ground actions for specific sites (Anderson, 2014). Although under a third of survey participants had attended workshops (Fig. 3.5), a different survey of urban forest restoration workshop attendees (n=81), found that 88.9% of those attending workshops stated they were either 'very likely' or 'likely' to apply information learned at the workshop in their restoration work (People, Cities and Nature, unpublished data.). Similarly, a study by Davis *et al*., (2013) found many fire science users valued workshops and interactive science delivery methods that include 'face time' between scientists and practitioners.

Other resources differ in their use by professional and avocational practitioners. Almost half of professional practitioners (47.8%) accessed information in academic or scientific publications, while only 26.2% of avocational practitioners reported accessing scientific publications (Fig. 3.5). These results fit within the contradictory findings of other research, some of which show practitioners rarely access primary scientific literature (e.g. Bernhardt *et al*., 2007; Sutherland *et al*., 2004), while other times it is an important information source (e.g. Seavy & Howell, 2010; White, Lindberg, Davis, & Spies, 2019). It is likely that the accessibility and perceived importance of scientific publications are determined by practitioners' experiences and their restoration project context. In particular, the lower number of avocational practitioners using scientific publications may reflect the cost barrier to accessing most academic journals. In comparison, the opposite trend was true for hardcopy books, pamphlets, and reports, which are more accessible and were more popular with avocational practitioners (64.3% vs. 47.8%).

Avocational practitioners are more likely to plan plantings using knowledge sourced from discussions with co-workers or volunteers (85.7% avocational vs. 60.9% professional), or with friends and family (31% vs. 4.3%), while professional practitioners were more likely to use reference forest sites as guidelines for restoration (43.5 professional vs 31% avocational). Interestingly, discussions with avocational practitioners (through community groups) are an important resource used by 43.5% of professional practitioners, as well as avocational practitioners themselves (38.1%), indicating there is a significant amount of knowledge sharing between and among these two groups. This represents a 'web' or 'network' mode of knowledge transfer (Davis *et al*., 2013) between practitioners. These multiple transfers of knowledge are important as they strengthen the likelihood scientific knowledge will be disseminated amongst practitioners if it is introduced to the network. Web-based tools are one of the least-used resources for practitioners from both groups (Fig. 3.5), perhaps reflecting the older demographic represented by survey participants. However, Seavy & Howell, (2010) found that web-based tools were not perceived as important or widely available by restoration practitioners and land managers, which may also explain our results. This is unfortunate as most cutting-edge research is available online only, or may experience a substantial lag time (years) before being put into hardcopy forms or becoming dispersed via face-to-face communication.

Practitioners also specified using resources not listed in the survey categories. These included: the practitioner's own experience and knowledge, council management plans, local seminars, habitat specialists, landscape architects, historical data, local nurseries, and societies or groups including Botanical societies, QEII trust (land-covenanting institution), or local Forest and Bird branches (an ecological advocacy organisation). Reliance on a practitioner's personal experience for project management is consistent with other studies (e.g. Bernhardt *et al*., 2007; Sutherland *et al*., 2004) and highlights how the weak link between restoration scientists and practitioners is disadvantageous to both parties. Not only are practitioners disconnected from up to date research, scientists are also disconnected from the valuable practical ecological knowledge and strategic knowledge held by practitioners (Hulme, 2014; Sutherland *et al*., 2004).

In one instance a practitioner noted that advice sought had been rejected, stating: "The planting densities advocated by advisors are impractical due to cost, size of area, labour required, dry summers etc." This implies that even when practitioners access expert knowledge, recommendations may not be implemented due to a range of practical constraints.

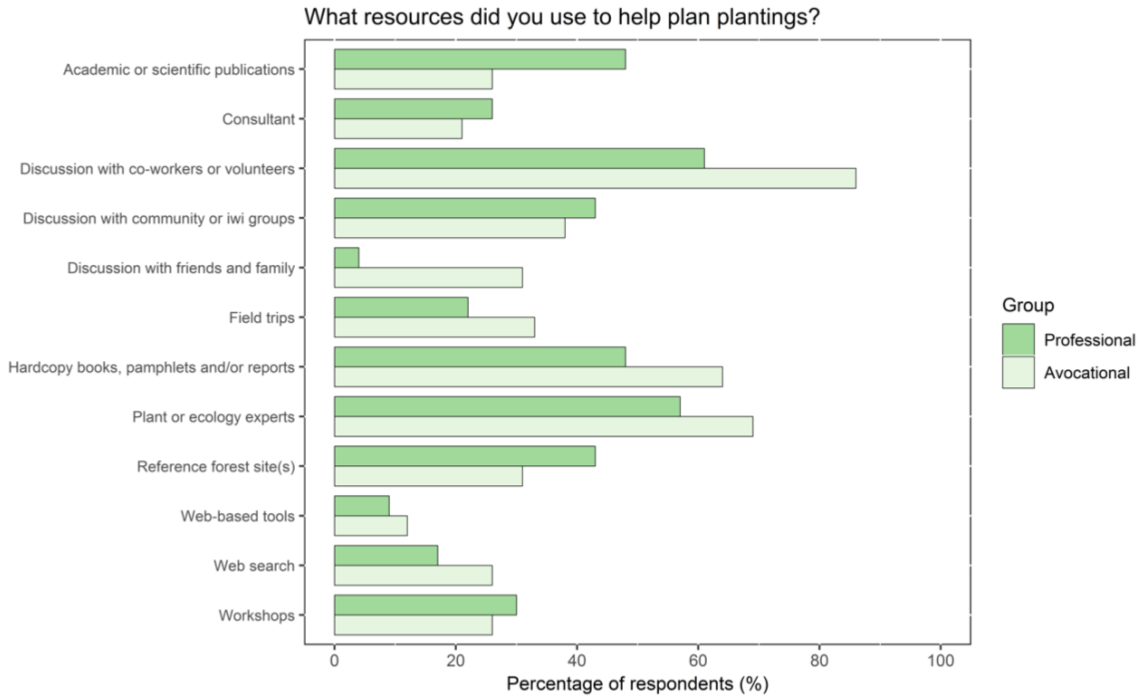

Figure 3.5 Resources used by avocational (n=42) and professional (n=23) urban forest restoration practitioners when planning restoration plantings.

In conclusion, practitioners rate many considerations as important but top priorities typically relate to increasing the presence of indigenous species (Fig. 3.2). When planning how to achieve objectives, practitioners utilise different resources, varying slightly between professional and volunteer groups, but interactions with co-workers and experts are most frequently used, eclipsing scientific publications. This indicates an opportunity to improve transfer of scientific knowledge by prioritising interactive, interpersonal modes of communication and encouraging collaborative partnerships between scientists and practitioners.

3) How to manage? Restoration ongoing care

Non-native plant control is the most common site maintenance activity (76%). Interestingly, attitudes towards non-native plants range widely, from dislike to one respondent emphasizing the importance of grass cover, stating, "A living 'mulch' of grass around the trees helps keep moisture in the soil during summer." However, this reasoning is not scientifically supported and can also increase the mortality of desirable regenerating seedlings (Anton, Hartley, & Wittmer, 2015). Another noted that, "weeds provide shelter from wind and sun" to establishing restoration plants. There is some support for this, as established non-native tree species can act as a nurse crop to stabilise understory microclimate and reduce herbaceous groundcover (Stanturf *et al*., 2019). However, if these non-native species are deciduous they can also alter the light and nutrient availability and microclimate of a forest by shedding their leaves. This annual leaf drop favours establishment of non-native herbaceous understorey species and decreases chances of native seedlings spontaneously regenerating (Cornwall *et al*. 2008; Wallace, Laughlin, & Clarkson, 2017).

Releasing plants (removing weeds surrounding the base of plants) is the second most common site maintenance activity (31%), while less-used site maintenance includes: mulching (13%), watering (13%), pruning (10%), replacing dead plants (5%), fertiliser application (3%), and installing plant protectors (3%). Installing drainage systems, undertaking enrichment planting, and "maintaining light gaps for podocarps" are also mentioned by one participant each. Some maintenance that occurred was to ensure human use values of the site are maintained e.g. pruning next to paths, or controlling weeds to "achieve the amenity outcome desired".

When asked how site maintenance changed over time, two perspectives were expressed. Several practitioners said that weed control practices are "ongoing", "continuous" or "endless" - a perspective also reported elsewhere (e.g. Jay & Stolte, 2011). However, other practitioners stated maintenance lessened after plant establishment and forest canopy development. Current best practice recommends dense initial restoration plantings to achieve rapid canopy closure, reduce light availability and shade out herbaceous weeds, thereby reducing the length of time that intensive weed control is required and hastening native tree regeneration (Wallace *et al*., 2017). The two practitioner perspectives may reflect a difference between practitioners who use scientific underpinnings to inform their management and those who do not.

Sourcing plants and collecting seeds

Plant nurseries are the most common source of restoration plants (56.5%). About half (48.4%) of participants also collected and propagated seed locally, and 20% of these from the restoration site itself. A quarter of practitioners (26%) use plants provided by a city or regional council and one practitioner used plants provided by a trust.

Survey responses showed that practitioners consider eco-sourcing a top priority (55.6% of rating as 'very important' and 25.4% rating as 'important'; Fig. 3.3). This practice involves sourcing plant propagation material from individuals in naturally occurring forest patches within the local area of the restoration project to ensure ecological suitability, increase establishment success, and reduce outbreeding depression (Breed, Stead, Ottewell, Gardner, & Lowe, 2013). In total, 80.6% of practitioners surveyed either obtained plants from nurseries that claim to eco-source or use their own locally sourced seeds. This widespread adoption of eco-sourcing practices is consistent with another recent practitioner survey (Cooper, Catterall, & Bundock, 2018). Eco-sourcing has been included in general restoration guidelines since the 1990s (Cooper *et al*., 2018) and provides an example of successful science communication. However, it also highlights difficulties in science communication when scientific findings are constantly refined or change with context. For example, strict eco-sourcing practices can lead to poor restoration outcomes when seed is collected from highly fragmented or bottlenecked populations such as are typical in urban landscapes. In this context, eco-sourcing can actually lead to reduced intraspecific genetic diversity and decreases resilience of a population to changing climatic conditions or other stressors (Breed *et al*., 2013; Broadhurst *et al*., 2008; Cooper, Catterall, & Bundock, 2018; Hufford & Mazer, 2003; Prober *et al*., 2015). It is now proposed that using diverse, 'ecologically appropriate' plant material from a broader landscape will increase adaptive capacity and this is more important that using only locally adapted plant material, particularly in degraded environments (Jones, 2014; Prober *et al*., 2015) like urban forests. That the majority of urban practitioners are prioritising eco-sourcing seeds from an area that is often smaller than the local ecological district indicates a substantial lag time between refinement of best-practice methods for urban contexts and implementation. This lag time would be reduced by strengthening the link between ecological research and restoration practice.

4) Did it work? Urban forest restoration monitoring

Monitoring deficiencies are a widespread issue in restoration projects (Rey-Benayas, Newton, Diaz, & Bullock, 2009; Wortley *et al*., 2013) and urban forest restoration is no different. Less than a quarter of practitioners conduct baseline (i.e. pre-restoration) data collection (23.7%; Fig. 6), which they defined loosely and included taking photo points (8.5%) or recording pre-existing vegetation (7%). Two practitioners reported baseline data was collected for certain projects only (Fig. 3.6; 3.4%) and four reported that baseline data collection was undertaken, but by an external agency such as a city council or university (Fig. 3.6; 6.8%).

Post-restoration monitoring was more common (59.6%; n=57; Fig. 3.6). Two practitioners said they monitor only a subset of their projects, (3.5%: Fig. 3.6) and four practitioners reported that monitoring is sometimes done by an external agency such as a land-covenanting institution, a city council, or a university (7%: Fig. 3.6). However, when practitioners were asked about methods, it became apparent that monitoring consists of mainly informal, visual inspections (56%, n=39). Participants who undertook monitoring described practices like, “Very simple looking at sites afterwards”, “Checking to see if plants had survived”, and checking on sites to “make sure plants are not smothered by weeds”. These results echo similar research that has found social-qualitative measures such as visual appearance are commonly used to assess project success (Bernhardt *et al*., 2007), and comprehensive ecological monitoring of restoration progress is largely non-existent (Galbraith *et al*., 2016; Hagger *et al*., 2017).

Other post-restoration monitoring (n=39) included plant survival (36%), plant growth or plant height (26%), canopy cover (13%), the impacts of pest species (e.g. rabbit browse) or plants displaying disease symptoms (8%), diversity in the restored forest (5%), regeneration (5%), and soil retention (2%). However, it was unclear whether these practices occurred in a structured, quantifiable form, or are more informal and observational. For example, “growth and survival rates” could refer to the collection of numerical data, or it could simply refer to informal visual assessments. More formal practices employed by practitioners included photo points (13%) and plot-based methods to collect numerical data (10%). These proportions are substantially lower than those reported by Peters *et al*. (2016) (54% and 45% respectively). However Peters *et al*. was not focused solely on urban areas. As urban restoration projects are typically relatively small-scale, this may contribute towards low monitoring rates.

No practitioners mentioned monitoring of socioeconomic indicators despite these being a key element of successful restoration (Wortley *et al.*, 2013) and human use values being rated an important consideration in restoration objectives in survey responses (Fig. 3.3).

The general lack of restoration project monitoring and interpretation signifies another weak link between ecological research and restoration practice. Pre-restoration and post-restoration monitoring are crucial components of restoration projects. Monitoring enables us to determine progress by ensuring restoration activities are successful in producing desired outcomes (Wortley *et al.*, 2013), and allows for better communication of project status to stakeholders and funding agencies (Hagger *et al.*, 2017; Chris Jones & Kirk, 2018; Stanturf *et al.*, 2014; Westgate, Likens, & Lindenmayer, 2013). Furthermore, it enables restoration techniques to be improved through adaptive management (Mansourian *et al.*, 2017). Without monitoring, practitioners tend to overestimate project success (Bernhardt *et al.*, 2007), which may result in long-term failures (Stanturf *et al.*, 2014), and could also dissuade practitioners from searching for up-to-date best-practice information to improve restoration outcomes.

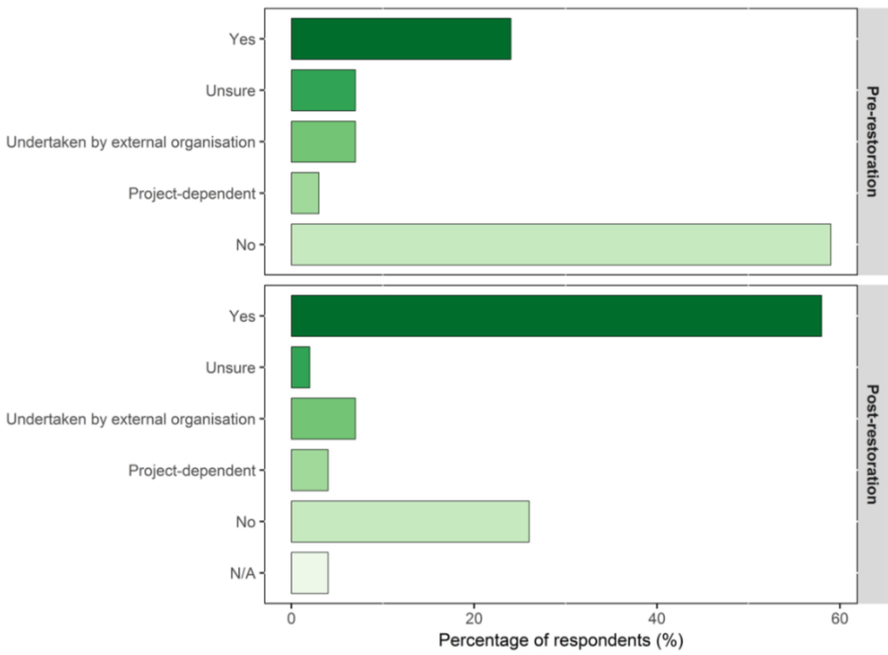

Figure 3.6 Monitoring of urban restoration projects. Percentage of practitioners who reported undertaking pre-restoration monitoring (i.e. baseline data collection) (n=59) and post-restoration monitoring (n=57) is shown on the x-axis.

When participants were asked to explain absence of monitoring, resource availability was the most common reason (lack of time, funding, labour). However, many responses reflected a view that monitoring is unimportant, laborious, or superfluous. For example, responses to this question included: “cannot see the point in doing it”, and, “a lot of the formal monitoring I have seen is a waste of effort”. Practitioners also indicated they consider other activities of higher priority; for example, “[monitoring] was not as important as getting the right plants in the ground”. Some practitioners reported a combination of these views, e.g. “identifying plant species and counting the number within a given square was considered but was felt to be a very time consuming…and did not add anything to the restoration project”, and, “formal monitoring is costly and for this work not really needed. We are going for gross change that is highly visible.” Other studies also report criticisms of monitoring as too costly and labour intensive (Bernhardt *et al*., 2007; Clewell & Rieger, 1997; Peters *et al*., 2016; Wurtzebach, Schultz, Waltz, Esch, & Wasserman, 2019) or dispensible (Jones & McNamara, 2014). The view that monitoring is unnecessary may be partially explained by a lack of clearly, initially defined project objectives. These provide the basis for identifying what to monitor (Block, Franklin, Ward, Ganey, & White, 2001; Jones & Kirk, 2018) and the relationship between poorly defined objectives and a lack of monitoring has emerged elsewhere (Bernhardt *et al*., 2007; Galbraith *et al*., 2016; Jones & Kirk, 2018). Other reasons for not carrying out monitoring cited by practitioners included organizational issues, or a view that restoration had been successful so monitoring was not required.

In summary, thorough monitoring is not a priority for most urban forest restoration practitioners. As a result management decisions often take place without the evidence-based knowledge derived from monitoring (Cook, Hockings, & Carter, 2010). Practitioners who do not monitor view it as time-consuming and unimportant in comparison to other restoration activities. Encouragement of monitoring practices will require effective communication to practitioners about the multiple benefits of forming a clear restoration plan including objectives and well designed monitoring and interpretation of monitoring results. Such plans can be simple but are still valuable, and evidence shows that with appropriate training, volunteer practitioners are capable of using empirical methods to monitor restoration plantings (Galbraith *et al*., 2016; Peters *et al*., 2016). Monitoring toolkits for New Zealand projects exist but are not yet widely used by practitioners (Peters *et al*., 2016). Although monitoring requires sustained costs, these are small relative to the value of the forests it protects (Lovett *et al*., 2007). Practitioners must be adequately resourced to carry out long-term monitoring themselves or enabled to engage in collaborative involvement (as is is already happening some places, Fig. 3.6), which can have a significant positive effect on monitoring activities (Peters *et al*., 2015).

5) What caused problems? Setbacks faced by practitioners

Environmental events

When practitioners were asked about setbacks in achieving restoration objectives, 40% (n=60) said they experienced setbacks due to weather-related factors (e.g. drought or frost), stochastic events such as slips or tree-falls, or other abiotic factors such as salt spray or inhospitable soil type. Practitioners note that setbacks could sometimes have been avoided or resulted from a trial and error approach, leading to adaptation of practices. Environmental conditions such as droughts are likely to increase due to climate change. Therefore, ensuring restored ecosystems have adaptive potential by including plants with traits for coping with climate change is crucial for long-term sustainability (Choi *et al*., 2008; Cooper *et al*., 2018; Laughlin, 2014). Our results indicate these considerations are not incorporated into urban restoration practitioners decision-making, and more guidance on designing plantings to increase resilience in the face of environmental pressures is paramount.

Lack of resources

As with monitoring (section 4), a lack of resources was reported as an obstacle to achieving restoration objectives (18.3% of respondents), specifically a shortage of workers and finances: "We always have to compromise [on] how many plants we can plant because the budget is limited". This is a common constraint in restoration work and widens the gap between research and implementation (Cabin, 2007). Restoration is a long-term process but sustained funding is rare, particularly for monitoring, even though monitoring can ultimately increase cost-effectiveness and success by enabling adaptive managment (Stanturf *et al*., 2019). It is a wicked problem that without long-term funding, restoration outcomes will be compromised, yet if projects are not 'successful' in the short term, it is difficult to sustain funding flow. There continues to be a need for more long-term investment in restoration projects.

Invasive non-native species

Invasive non-native species are sometimes the cause of setbacks, such as herbivory by non-native vertebrates (i.e. rabbits) (13.3% of practitioners), and non-native plant invasion (15%). However, non-native plant control sometimes results in non-target effects (e.g. spray drift resulting in native plant elimination) and can therefore be an issue itself (11.7%). Urban areas typically have high non-native propagule pressure so it is difficult to eliminate this setback entirely (Overdyck & Clarkson, 2012; Sullivan *et al*., 2009). However, problems caused by invasive non-native species can be minimised by the use of best-practice methods, including mammalian pest control, planting densely to fast-track canopy closure and shade out herbaceous weeds, and minimising spray drift by spraying on calm days or using plastic shields around desirable plants.

Stakeholder unity

Conflict amongst stakeholders such as councils, landowners, contractors, or the public is reported to cause some urban forest restoration project setbacks (11.7%). This includes issues such as local authorities "changing objectives" and "conflicting aims", and problems with neighbouring landowners: "people cut down and poison the trees we plant in the coastal reserve as they are worried they will block their views". This highlights the importance of maintaining stakeholder interest and unity - a complex task in populated urban contexts. Some practitioners indicated setbacks are related to poor communication between community restoration groups and city council employees or other involved organisations. One practitioner describes the challenge this poses:

> *We have struggled to get clarity on support (and permissions) for doing restoration work on a council managed reserve. On several occasions we have found marker tape, tracking tunnels and bait stations in or near the reserve, without any signage or indication of who else is working in the area. So, a lack of coordination between local groups (us), [non-goverment organisation], Council, and biosecurity contractors has caused a few issues. We have struggled at times to make sense of what development is happening around the reserve and on what timeframes, which does impact on our restoration planning and may impact on our activities.*

Setbacks are also caused in minority of case by plant theft, site vandalism, and disagreements or disorganisation within restoration groups. For example, as one practitioner states, “There are many compromises with volunteers who have set ideas”.

These reports of stakeholder disharmony suggest more advice should be made available to pracititoners for navigating socio-political challenges, considering the important role stakeholder agreement plays in determining restoration project success (Fox & Cundill, 2018; Wallace & Clarkson, 2019). Our survey indicates that practitioners understand the important role of stakeholder buy-in, but they lack guidance on how to promote this without sacrificing other objectives such as ecological values. As one practitioner notes:

> *Much of the advice and technical information seems to be designed from a purist technical standpoint. This is not so helpful from an urban forest restoration perspective, because the forest remnant is itself highly modified, there are many people/groups to consider – not just one owner.*

To address this, practitioners could be equipped through multidimensional training and/or a link to social science experts, increasing their capacity to successfully navigate the social, economic, and management dimensions of urban restoration (Meli, Schweizer, Brancalion, Murcia, & Guariguata, 2019; Nelson, Schoennagel, & Gregory, 2008). Additionally, providing more guidance on design of plantings that both maintain ecological integrity and support human use-values would help support stakeholder unity. Finally, ensuring stakeholder engagement and participatory processes are incorporated into restoration projects from the outset will enhance communication and ensure values of the many are embedded in the restoration process (Druschke & Hychka, 2015; Fox & Cundill, 2018; Guerrero *et al*., 2017). Such an approach will have the additional benefit increasing the probability that community members will play an advocacy role for the restoration project (Fox & Cundill, 2018).

Support requested

When asked what additional support or information practitioners would like to have for their restoration work, 35% said no additional support is needed (n=49). This was due to a view that restoration projects were successful (e.g. “Nothing at this stage, we are doing well”), or because they had adequate support (e.g. “We are already pretty well supported by [the Council] and our own wider group of supporters”). Restoration community groups in New

Zealand typically receive support from local councils or the Department of Conservation (a government agency) (Peters *et al*., 2015).

In spite of this, most practitioners (65%) still said they need additional support. Of these, almost half would like more restoration information or training (46.9%), specifically: standards for monitoring and reporting the results, development of a restoration plan, successional (i.e. enrichment) planting, the latest science and innovations, and best-practice methods in general. Improving access to restoration information can be done by synthesizing research findings into an easily accessible synopsis written in plain language for use by practitioners (Anderson, 2014). Such documents have been ranked by practitioners as the most important source of information for restoration decision-making (Seavy & Howell, 2010), and can increase the likelihood practitioners will implement effective management interventions (Walsh, Dicks, & Sutherland, 2015). Another document for support of practitioners is a simple restoration plan template. Support documents could be disseminated via many avenues (e.g. workshops, soical media, websites), with regular updates with scientific advacements. However, producing documents such as research synopses is a complex task that requires financial support for the work involved (White *et al*., 2019).

Other support desired by practitioners includes more resources (e.g. financial assistance or equipment, 34% and more labour, 31%). One practitioner mentioned that they need “more experienced volunteers” and younger volunteers to take over the organisation of the project. A minority of practitioners expressed a desire for support to improve community engagement, networking, or improved communication and coordination between stakeholders (9%). For example, one participant commented, “The biggest problem is conflicts with other people on the site. There needs to be coordination of all the public land agencies in each area”. This reiterates the need for improved stakeholder engagement and a more coordinated approach to urban restoration work.

3.5 Conclusion

The tenuous link between ecological research and restoration practice

Our findings indicate that practitioner objectives, priorities, and management practices often do not reflect current scientific knowledge regarding best-practice restoration. Broad, vague project objectives, a focus on ecosystem components rather than ecosystem function and resilience, a tendency to prioritise implementation over project planning and evaluation, and ineffective knowledge transfer between researchers and practitioners all contribute towards this disconnect between ecological research and restoration practice.

Even if equipped with up-to-date knowledge regarding best practice restoration, implementation can be particularly difficult in urban settings where there are many diverse stakeholders involved and multiple objectives. Limited time, labour, funding and sometimes technical expertise further restrict what is possible for practitioners. However, the commitment of these under-resourced practitioners, their broad range of restoration priorities, and the range of knowledge resources they utilise shows a remarkable desire that urban forest restoration projects have successful outcomes.

Removing barriers and strengthening the research-implementation link

Restoration outcomes can be improved by ensuring practitioners have the capacity to implement best-practice restoration techniques dynamically in complex urban environments. To achieve this, we must make sure practitioners are aware of current best-practice restoration, that they understand why it is important, and they know how to apply it in their projects. Publications in academic journals will not suffice, instead, a two-way communication channel must exist. Practitioners must be able to ask site-specific questions and voice conerns about practical constraints. This two-way channel consists of interactive, interpersonal communication and collaborative partnerships between scientists, trained science communicators, and practitioners. This format will encourage dialogue and knowledge transfer to the benefit of all parties. Barriers to restoration implementation must also simultaneously be removed where possible through increased funding for urban restoration efforts, supportive policies, and increased training opportunities for practitioners. Finally, urban restoration practitioners require assistance to translate broad restoration visions into clearly defined objectives and measurable success criteria.

This study is the first to specifically target New Zealand urban forest restoration practitioners, and as such provides an important overview of the perspectives and decision-making processes of these practitioners and the range of socio-ecological constraints that contribute towards the research-implementation gap in cities. We have highlighted opportunities for improving the efficacy of urban restoration efforts that will also likely be applicable in other restoration contexts globally.

Conflict of Interest

Nothing to declare.

Authors contributions

S.B. and K.W. conceived the ideas and designed the survey. S.B. designed the methodology and analysed the data. S.B. led the writing of the manuscript. All authors contributed to the drafts and gave final approval for publication.

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Ethics approval

Approval for research using human subjects was granted by the University of Waikato Faculty of Science and Engineering Human Research Ethics Sub-Committee (Approval number: FSEN_2018_16). The survey was prefaced with a written description of the project and all participants were aware of the nature of the research. All gave their informed consent to participate in the research with the knowledge that a manuscript was to be published based on the results.

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3.7 Appendix 3.1

Busbridge, Clarkson and Wallace: Urban forest restoration practitioner survey questions:

Please enter your contact details (these will be kept confidential)

a. Name:
b. Company/Institution:
c. City/Town:
d. Email address:

1. My responses are regarding my work in URBAN (within city limits) forest restoration (Y/N)

2. Why did you decide to get involved with urban forest restoration efforts?

3. What is your restoration project objective(s)?

4. How did you select your restoration site?

5. When making restoration plans, how important were the following considerations? **(Please enter a value between 1 and 5, with 1 being not at all important and 5 being very important)**

Consideration	Rating (1-5, or NA)
Aesthetic value	
Planting native species	
Provision of ecosystem services	
Providing habitat for native wildlife	
Greenspace benefits/human use values	
Affordability/cost-efficiency	
Using eco-sourced plants	
Habitat or species conservation	
Restoring ecosystem processes and function	
Increasing biodiversity	
Climate change reduction through carbon offsetting	
Erosion mitigation	
Restoring to a historical state or pre-human condition	

Please specify any other important considerations not listed:

6. What resources did you use to help plan plantings (e.g. planting density, planting methods)?

Resource	Used? (Y/N)
Hardcopy books, pamphlets and/or reports	
Academic and scientific publications	
Reference forest site(s)	
Web search	
Web-based tools	
Consultant	
Plant or ecology experts	
Workshops	
Field trips	
Discussion with co-workers or other restoration volunteers	
Discussion with friends and family	
Discussion with community or iwi groups	

Please specify names or details of resources used where possible:

7. When choosing what species to plant, how important were the following considerations? **(Please enter a value between 1 and 5, with 1 being not at all important and 5 being very important)**

Consideration	Rating (1-5, or NA)
Availability	
Cost	
Hardiness/likelihood of survival	
Ecological suitability	
Aesthetic value	
Functional value	
Habitat value (e.g. provision of food or other resources for wildlife)	
Indigenous status (native or endemic to NZ)	
Successional status	
Personal/public preference	
Conservation status	
Ecosystem services value	

Please specify any other important considerations not listed:

8. What resources did you use when choosing what species to plant?

Resource	Used? (Y/N)
Hardcopy books, pamphlets and/or reports	
Academic and scientific publications	
Reference forest site(s)	
Web search	
Web-based tools	
Consultant	
Plant or ecology experts	
Workshops	
Field trips	
Discussion with co-workers or other restoration volunteers	
Discussion with friends and family	
Discussion with community or iwi groups	

Please specify names or details of resources used where possible:

9. Where were plants used in the restoration planting sourced from, or seeds collected from?

10. Monitoring and evaluation:
    a) Did you undertake any pre-restoration monitoring? (baseline data collection prior to restoration)
    b) Did you undertake any post-restoration monitoring?
    c) If you did undertake monitoring, what were you measuring and how?
    d) If no monitoring was undertaken, why not?

11. Have there been any setbacks or obstacles to the achievement of your restoration objectives, or compromises you have had to make? If so, what were they?

12. What site preparation, if any (e.g. weeding, pruning, spraying, watering etc.) did you undertake prior to commencement of the restoration work?

13. What maintenance has occurred during the course of restoration activities, and has it changed over time? If so, how?

14. What additional support and/or information would you like to have available for your restoration work?

15. How has the restoration work been funded (labour, plants, materials, herbicides, etc.)?

16. Is there any policy you are aware of mandating whether you should do restoration in your city?

17. What are your primary responsibilities in relation to the restoration planting?
18. Is there anything else you would like to add?

3.8 Appendix 3.2

Attributes of restored ecosystems as outlined in the Society for Ecological Restoration Primer on Ecological Restoration (2004), as well as additional social goals and related attributes as defined by Hallet *et al* (2013).

Category	Abbreviated attribute/goal	Full attribute/goal definition	Example survey response for this attribute definition
Form	Similarity to reference conditions	The restored ecosystem contains a characteristic assemblage of the species that would occur in a reference ecosystem and that provide appropriate community structure	"To restore the native flora of [the site] back to, as near as is practical, pre European status…"
Form	Presence of indigenous species	The restored ecosystem consists of indigenous species to the greatest practicable extent (includes removal of non-native species)	"To clear the area of invasive weeds and plants, to plant natives over the cleared area"
Function	Presence of functional groups	The functional groups necessary for the continued development and/or persistence of the restored ecosystem are either represented, or have the potential to colonize (includes habitat provisioning)	"…recruit seedlings [and] receive seedlings via avian dispersal"
Function	Capacity of the physical environment to sustain populations	The physical environment of the restored ecosystem is capable of sustaining populations of the species necessary for its continued stability or development along the desired trajectory	"Freshwater catchments - improve habitat for eels, freshwater fish and crustacea…"
Function	Normal functioning	The restored ecosystem apparently functions normally for its ecological stage of development, and signs of dysfunction are absent	"Restore health and ecology to a coastal forest"

Function	Landscape integration	The restored ecosystem is suitably integrated into a larger ecological matrix or landscape, with which it interacts through abiotic and biotic flows and exchanges	"Improving the quality and connectivity of the scattered remnant patches of indigenous vegetation along the [site] corridor…"
Resilience	Elimination of threats	Potential threats to the health and integrity of the restored ecosystem from the surrounding landscape have been eliminated or reduced as much as possible	NA
Resilience	Resilience to stress events	The restored ecosystem is sufficiently resilient to endure the normal periodic stress events in the local environment	NA
Resilience	Self-sustainability	The restored ecosystem has the potential to persist indefinitely under existing environmental conditions without human intervention. Aspects of its biodiversity, structure, and functioning may change as part of normal ecosystem development and may fluctuate in response to normal periodic stress and occasional disturbance events. As in any intact ecosystem, the species composition and other attributes of a restored ecosystem may evolve as environmental conditions change	"To create a self-sustaining habitat sanctuary that represents the original diversity of the [site]."
Social	Community engagement	The restoration builds support and connections among the local community	".[to] bring people together on the project and encourage more people to enjoy the park…"
Social	Societal values	Societal human-use values are promoted through the restoration (e.g. provides recreational opportunities, enhances landscape aesthetics)	"To provide the public with a natural habitat for them to appreciate and enjoy."

Social	Māori cultural values or use	Māori cultural values are promoted or enhanced through the restoration (e.g. Restores culturally important species, allows for retention traditional practices such as customary harvest)	NA
Social	Ecosystem services	Ecosystem services are provided or enhanced through ecosystem restoration	"To mitigate flood effects in the catchment…"
Social	Educational outreach	Educational opportunities are incorporated in restoration planning	"To rebuild the forest community in our local reserve to conserve indigenous biodiversity…and as a local teaching/learning resource"
Social	Governance	Institutions with governance capacity either fund, mandate or maintain the restoration effort (e.g. complies with legal mandates, partners non-profit organizations with federal agencies)	"…to create urban forest site as per [City Council] Urban Forest Strategy"

4 CHAPTER 4

THESIS SYNTHESIS

4.1 Discussion

Restoring forests is a difficult task that involves complex interactions and feedback loops (Waldron & Xi, 2013). It is important that projects are guided by ecological theory to maximise likelihood of success and ensure that limited resources are put to use effectively (Wallace & Clarkson, 2019). However, much of the research on forest restoration has taken place in forests outside cities. Urban forest ecosystems have altered dynamics as they are subject to a different array of stressors, and involve a unique social element due to proximity with humans (Aronson *et al.*, 2016; McPhearson *et al.*, 2016). Although urban-specific research and knowledge of how to restore urban forest ecosystems is growing, a number of knowledge gaps remain. Remedying these is important as space and budgets for promoting biodiversity in cities is limited and thus ensuring such projects are successful and sustainable in the long term is crucial.

This thesis broadens understanding of social and ecological drivers of urban forest restoration success and highlights opportunities for improving the efficacy of urban restoration efforts. It investigates what factors constrain or promote regeneration and recruitment processes for native tree species across a chronosequence of restored forests in New Zealand cities. The research presented here also explores how restoration practitioners in New Zealand make decisions in their efforts to re-establish native urban forests, and identifies factors contributing towards the gap between research and implementation. Together these two chapters allow me to make recommendations on how to both improve restoration practice, and increase the likelihood such recommendations will be implemented.

Chapter 2 presents novel findings on how ecological drivers of native tree recruitment from germination to the sapling stage vary by plant species successional status in restored urban forests. Using a chronosequence approach and statistical modelling techniques, I showed that canopy closure is an important development that promotes germination and recruitment of young trees of all successional stages (early, mid, late). However, I identified an important ontogenetic shift that occurs during juvenile growth whereby the positive effects of canopy closure become neutral once seedlings grow into saplings. In contrast, the key driver of herbaceous ground cover has a negative effect on the smallest seedlings, but no effect on saplings which are released from competition pressures due to their height. The cooler, more stable temperatures provided by the canopy benefit mid-late successional species, but not early successional species. My work shows that while forest age is an important driver of seedling abundance of all successional statuses, patch size is also key for mid-late successional species to thrive. These results demonstrate that although conditions suitable for mid-late successional seedling germination and recruitment are not guaranteed to develop in small restored forest patches, these patches are nevertheless suitable for the survival of the larger mid-late successional saplings because they are less sensitive to environmental stresses. This affirms the value of restoring even small urban forests because with interventions such as enrichment planting of mid-late successional saplings, they too can contribute to a city-wide network of established forest habitat.

Chapter 3 addresses the lack of information available about how practitioners incorporate restoration science into their motivations, knowledge, and decision-making processes. A survey approach of urban restoration practitioners found that there is often a tenuous link between ecological science and urban restoration practice due to breakdowns in knowledge transfer and barriers to implementation. Survey responses showed project objectives are often broad, vague and reflect a focus on revegetation or establishing an initial canopy rather than restoring ecosystem processes or function. Other issues included a prioritization of implementation activities over project planning and evaluation, insufficient resources to support best practice, and social constraints that restrict what is possible for practitioners to achieve. We also found that practitioners aren't always aware of up-to-date research. Scientists typically rely on journal publications to communicate research findings, but our results showed that practitioners actually relied more on discussions with other practitioners and plant or ecology experts as sources of restoration knowledge.

In conclusion, this thesis addresses knowledge gaps regarding drivers of native tree regeneration, and how the science-practice gap manifests in an urban restoration context. Chapter 2 illustrates that mid-late successional tree juveniles do not respond to the same drivers as early successional species, and that time since restoration does not guarantee that conditions will become suitable for these species to colonise spontaneously. Chapter 3 is one of the first studies to specifically target urban forest restoration practitioners and provides an important overview of their perspectives and decision-making processes, as well as the range of socio-ecological constraints that contribute towards the research-implementation gap in cities. It demonstrates that there is often a tenuous link between ecological research and urban restoration practice. Although Chapter 2 has important best-practice management applications for ensuring restored forests follow the steps of succession into maturity, Chapter 3 highlights that there are likely to be roadblocks to implementing these as the practitioner objectives are typically more focused on short term goals.

This thesis highlights how to effectively overcome factors that can impede the long-term success of restored urban forests. This research advances our understanding of both ecological theory and the factors that contribute towards science-practice gaps in urban forest restoration. Furthermore, many discoveries reported in this thesis are applicable in other restoration contexts globally, in and outside of cities. Together, the results from these two inter-related pieces of research will help to maximise the efficiency, sustainability and overall success of forest restoration projects in our urban landscapes, ensuring that cities maximise the ecosystem service benefits of urban forests, and that urban dwellers can enjoy Aotearoa's native forest in their backyard.

This research has the following implications for best-practice in urban forest restoration management:

- When establishing newly-planted urban forests, practitioners should prioritise canopy closure and reduction of herbaceous weed species as early as possible to encourage regeneration of small seedlings of all successional statuses

- Dense initial planting is an effective strategy to help fast track canopy closure, but to later recruit late-successional saplings to the canopy, small light gaps may need to be created 20-30 years later.

- Enrichment planting of mid-late successional species should be an important part of all urban restoration projects, but is absolutley vital in small restored forest patches to ensure successional progression.

- Juvenile trees used in enrichment planting should be ≥100cm to maximise their chances of survival.

- Restoration project monitoring should include measures of regeneration processes as indicators of success.

- All restoration projects should include formal restoration plans with clearly defined objectives that extend beyond establishing an initial canopy.

- Ongoing monitoring is crucial to ensure projects are progressing towards objectives, and to inform adaptive management.

- Stakeholder engagement from the outset is important to avoid setbacks and secure long-term project support.

This research has the following implications for improving knowledge transfer and closing the science-practice gap in urban ecological restoration:

- Publishing research in academic journals is unlikely to result in improved restoration outcomes by itself. More interactive, interpersonal forms of science communication where practitioners can discuss project-specific concerns should be considered.

- Practitioners would benefit from useful time-saving resources such as regularly updated, accessible research syntheses as well as restoration plan and monitoring templates.

- More collaborative partnerships between scientists, science communicators and practitioners would be beneficial for all parties involved to further ecological theory and improve restoration management.

- Urban restoration guidelines should cover all aspects of restoration including social, cultural, economic and management dimensions.

- Barriers to implementing best-practice must be removed where possible through increased funding for urban restoration efforts, supportive policies, and increased training opportunities.

Recommendations for further research:

- In relation to Chapter 2, further research should be done to identify optimal light requirements of mid-successional saplings for accelerating their growth. This way management recommendations can be made to specify the precise degree of canopy thinning or canopy gap creation that will promote recruitment of mid-late successional saplings into the canopy.

- An evaluation of the relative importance of dispersal constraints, and role of below ground resources (water and nutrients) in the same study plots would help provide an more comprehensive overview of factors constraining regeneration.

- In relation to Chapter 2, further research is required to investigate how factors that constrain or promote seedling regeneration, establishment and recruitment vary with regional climate in New Zealand. For example, is the positive effect of shading weaker in cities that have more moderate climates, and high annual precipitation? Such research would aide in the development of management recommendations tailored to each city.

- In relation to Chapter 3, further research may undertake more in-depth interviews of urban practitioners to acquire a more detailed understanding of how they access and use scientific knowledge and their decision-making processes.

4.2 References

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