

Assessing the Ecological Impact of Urban Greening: A Case Study of Roadside Planting in Wellington, New Zealand

By

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ABSTRACT

Roadside reserves in Wellington, New Zealand have been the target of a government-led, community-implemented urban greening initiative for the past 25 years. Prior studies of urban greening have shown numerous benefits to neighbourhoods and communities through increased engagement and stewardship, yet there remains a need for research into the ecological effects these programmes have on individual urban landscapes. This research conducted site surveys to determine the variation in ecological functioning and biodiversity within 36 reserves involved in the Wellington Free Plants Programme (FPP). These measures were compared to historical planting data for each site retrieved from council records. Candidate models were constructed based on novel and classical ecological theory, which sought to explain observed variation between physical and ecological measures across study sites and the relationship between these variables and biodiversity. Sites were small with an area ranging from 5.9m^2 to 246.5m^2 (mean = $37.8 \pm 49.5\text{m}^2$), and biodiversity levels (assessed using a Shannon-Weiner Index) ranged from 0.1 to 2.9 (mean = 2.1 ± 0.7). The top performing candidate models to predict biodiversity included area, shape, and seedbank density. An examination of the effect of varying urban greening efforts across these sites utilised a multivariate analysis which included measures of ecological functioning, biodiversity, the number of years a site had been planted, and the

number of individual plants provided over those years. A significant negative relationship was found between site disturbance and the number of planting years ($F_{33,1} = 4.092$, $p = .051$) while a somewhat significant positive relationship was found between biodiversity and the number of individual plants provided ($F_{33,1} = 3.536$, $p = .069$). These results indicate that current urban greening efforts contribute to the ecological health of roadside reserves and that the patterns and processes governing the biological composition of these reserves may be partially explained with traditional ecological theory.

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Chapter 1

INTRODUCTION

The field of urban ecology has undergone a paradigm shift in recent years, as cities are increasingly seen as a part of ecosystems instead of as altered landscapes in which ecosystems persist (Alberti et al. 2003). In addition to naturally occurring greenspace, built areas within cities may contribute to, and themselves constitute, broadly defined habitat. Moving towards a consideration of the ecology of cities as opposed to the ecology in cities is forming a cornerstone of modern urban ecology (McDonnell et al. 2009). This dichotomy consists of two distinct ecological approaches: “ecology of cities” is a more inclusive approach which seeks to characterise individual cities as functioning ecosystems, while “ecology in cities” targets all non-human biological components (Grimm et al. 2000; Niemela et al. 2009; Wu 2008)

Ecology of cities builds upon ecology in cities, and has moved from a nested subset into a distinct field of ecological enquiry (Niemela et al. 2009). Niemela et al. (2009) attribute much of this progression to the abandonment of traditional ecological equilibrium theory in which ecosystems are assumed to be the healthiest when in balance. The view of urban systems as heavily disturbed is “...giving way to dynamism and the recognition that systems are often not in

equilibrium.” (Niemela et al. 2009, p.12). The multi-disciplinary framework required for a socio-inclusive “ecology of cities” approach is likely to be more resource intensive and may explain why progress has been comparatively slow (McDonnell et al. 2009).

Although urban areas vary in terms of size and intensification, studies have shown consistency between cities in regards to carbon sequestration and the availability of limiting nutrients, indicative of successful ecological functioning (Nowak & Crane 2002; Pickett et al. 2011). Still, it is believed that these ecosystems cannot fully replicate the level of functionality found in their ‘natural’ counterparts (Kowarik 2011). While there has been a marked expansion in urban field research over the last decade, there remains a great need for research surrounding species interactions, functional dynamics, and the ecological processes that contribute to urban environments (Heneghan et al. 2009).

In cities, high levels of environmental disturbance combine with high availability of resources (either by conventional proximity to humans or imported for use), which influence species assemblage (Turner 1989; McKinney 2006). This often results in an abundance of urban-adapted generalists that can displace local biota with more specific resource requirements (Shochat et al. 2006). New Zealand’s urban centres are distinct from their counterparts in Europe, North America, and Australia in the disproportionate level of exotic species that dominate their ecosystems (Freeman & Buck 2003). Widespread urbanisation and the threat this poses to locally-adapted native species, makes research on urban

ecosystems in a New Zealand context particularly urgent. While the novelty of urban environments makes the application of classic ecological theory difficult, it does, however, allow for the development and testing of novel ecological functions in a unique 'socio-ecological' system. (Grimm et al. 2008; Gaston et al. 2010).

Biotic and abiotic processes in urban environments differ from environments with less direct anthropogenic influence, and as such, restoration efforts are being increasingly viewed with humans as the main source of process manipulation (Burke & Mitchell 2007). For instance, human activities can lead to increased temperatures, which may alter season patterns and effect species migrations and breeding (Shochat et al. 2006). It is difficult to minimise human influence on an urban environment, however some cities have made attempts to channel involvement towards positive ecological outcomes. For instance, the Wellington City Council operates a Free Plants Programme (FPP), providing native plants to residents for planting on roadside sites. Such initiatives may be the key to enhancing urban habitat and resource availability, which has been shown to increase arthropod communities, plant productivity, and contribute significantly to overall ecological health (Muller et al. 2014).

Roadside sites are a common characteristic of modern urbanisation and represent an interesting junction between the processes of natural succession and artificially high levels of disturbance. Often found on public lands, these sites are removed from the usual difficulties that surround experimental designs in urban

ecology. Parks, gardens, and reserves represent moderate levels of urbanisation and, holding with the intermediate-disturbance hypothesis (Connell 1978) and resource availability, should exhibit relatively high indices of species richness. These sites are often the focus of local restoration projects and, despite their size, can have a significant impact on the city's ecosystem: small restoration sites (<500m²) have already been shown to act as valuable seed sources in New Zealand's most urbanised areas (Sullivan et al. 2009). This is especially important as recent studies have shown local seed banks and seed source availability to be limitation factors in urban succession (Albrecht et al. 2011; Overdyck & Clarkson 2012). Additionally, Viles and Rosier (2001) found that roadside greenways represent an opportunity to increase connectivity, biodiversity, and overall ecological health in New Zealand, by targeting disused land and mitigating the localised and detrimental ecological effects of passing vehicles (e.g. pollution and invasion vectors).

As such, the importance of quantifying the ecological functioning of these roadside reserves and their contribution to the urban landscape should not be understated. Whether the functioning of roadside reserves is a reflection of a larger ecological apparatus or a component thereof should be a question posed by ecologists, city planners, and stakeholders alike.

Urban ecology research has typically been observational in nature, due in large part to difficulties associated with property rights, human interference and most importantly, the high variability in spatial landscapes (Gaston 2010).

However, community-based restoration efforts like the Wellington City Council's FPP, which seeks to turn urban wastelands (disused sites with spontaneous vegetation) into purposefully planted roadside reserves, provide a unique opportunity to put existing observational methodologies into an experimental design.

Local-scale factors (both ecological and managerial) are suggested to play a more significant role predicting and enhancing biodiversity in urban environments than those of a larger landscape-scale (Schwartz et al. 2013). Examples of commonly studied ecological factors include decomposition rates, disturbance, and seedbank characteristics. Relative measures of these site characteristics can provide a better understanding of the long-term, ecological benefits associated with small-scale urban restoration projects. When considered in terms of volunteer community actions, this type of analysis may help to determine whether any ecological value is contributed by programmes with known social benefits. This will help inform future implementation and management of community-based restoration strategies.

Growing pressure placed on global ecology as a product of anthropogenic expansion will be best served by a compendium of "international, comparative work on...management systems of urban green space" (Niemela 2014, p. 299), including an assessment and monitoring scheme.

This study seeks to examine sites with similar land-use qualities (i.e. road reserves) in regards to ecological variation as well as whether any of this variation

can be explained as a function of treatment over time. Ecological studies in urban environments often focus on one of several themes, including changing land-use analysis, 'natural' plot comparisons, and numerous gradient analyses (Gaston 2010; McDonnell et al. 2009; Pickett et al. 2011). Combining aspects of what has been traditionally considered separate avenues of ecological study will hopefully provide valuable insight into the interplay between ecological health and government and community involvement and the relationship between these elements as a function of time.

Urban ecology research often focuses on broad aggregations of variables (i.e. patch distance from industrial/commercial centres and land-conversion factors like impermeable surface cover and vegetation loss (McDonnell & Hahs 2013). Though there is value to broad-scale pattern analysis, there is a growing need to investigate the underlying mechanisms and processes within established urban matrix frameworks. This has been done with varying degrees of success in recent years (Bonnington et al. 2013; Williams & Winfree 2013), yet the variation in urban intensity and landscape across the world's cities calls for the wider application of city-specific research to characterise novel ecological processes (Ramalho & Hobbs 2012). Moving from large-scale analysis to a finer, habitat-level scale is difficult as patterns break down and individual patch dynamics become more prominent drivers of ecosystem composition and function (Kowarik et al. 2011). Therefore, urban ecosystems are better defined as complex mosaics which may require an examination of the smaller, individual patches that comprise the

urban landscape. By using a localised approach, it is possible to shine light on underlying ecological processes that may have widespread application.

Cities often exhibit higher levels of species richness than neighbouring, rural areas, as well as supporting rare and endangered species (Alvey 2006). For example, Christchurch, New Zealand is a city surrounded by pastoral lands, and, as such, maintains a higher plant diversity than its regional surroundings (Stewart et al. 2004). The relationship between increased diversity and ecological stability has been debated for decades and, at its best possibly, can be considered an oversimplification of system complexities (McNaughton 1977; Tilman 1996).

In 2014, Bonthoux et al. found that urban wasteland sites in Western and Central Europe supported greater species richness and diversity than similarly sized, intentional green spaces, leading to the recommendation that future management plans prioritise the protection and inclusion of wasteland in conservation efforts. The Wellington region FPP has been working within a similar framework over recent years by providing locally-sourced plants to residents interested in creating novel green space along local public wastelands (roadside reserves). The expressed goals of the programme are to enhance ecosystem services and functions by promoting clean water, regulating runoff, storing carbon, increasing native plant populations, and providing diverse habitat for native birds, lizards, and insects, yet most of these lack evidentiary support and have been poorly explored in urban settings (Pataki et al. 2011).

While variation in species diversity across the urban gradient has been the focus of numerous studies there is significantly less literature devoted to how patterns change as a function of time (Luck & Smallbone 2010). This is unsurprising given the added logistic and resource requirements of long-term ecological monitoring. Programmes like the FPP provide an excellent framework for post hoc investigations; changes in programme goals and implementation, such as the removal of most exotic species from plant offerings, can provide an excellent opportunity for broad, temporal analysis. There is substantial research addressing the human benefits to community-based urban greening efforts ranging from psychological and social (Sommer et al. 1994; Armstrong 2000) to reductions in neighbourhood crime rates (Westphal 2003). The ecological benefits, however, resist broad theoretical approaches due to localised system complexity and need to be examined on an individual initiative and city basis (Hostetler et al. 2011).

This research recognises the need for an evaluation of small-scale urban ecological systems, with a focus on comparative functioning and the associated restoration value of community-based efforts, the long-term environmental outcomes of which have been poorly studied (Peters et al. 2015). It is expected that the study sites are likely to be representative of heavy fragmentation due to their proximity to highly urbanised landforms (i.e. roads), and as such, may exhibit a reduction in their functional potential. A healthy ecosystem is dependent upon not only internal species interactions, but numerous abiotic factors and ecological functions which are inextricably linked to species diversity (Meyer 1997). The

relationship between biodiversity and ecological functioning in small, urban greenspace is still poorly understood and requires greater research attention (Schwartz et al. 2013).

Although the active planting of native species may result in an increase in species-specific connectivity, the additional stressors associated with active human involvement (e.g. disturbance, exotic introductions, etc.) may not warrant these types of actions. Indigeneity of plantings (at least at the individual level) within the FPP planted sites is positively correlated with survival rates (Berentson 2013), but the degree to which the forced assemblage of native plant species contributes to overall biodiversity and ecosystem function is not known. Similarly, community group access and habitat manipulation at these sites may lead to an increase in measurable disturbance, or altered decay rates and soil seed bank availability. Through an examination of these measures this thesis seeks to explore the relationships between ecological functioning and biodiversity within small-scale urban bush sites and any benefits conferred through increased community planting efforts.

Chapter 2

THE RELATIONSHIP BETWEEN PHYSICAL FACTORS, ECOLOGICAL FUNCTIONS AND BIODIVERSITY IN SMALL URBAN PATCHES.

INTRODUCTION

The vegetative structure of urban greenspaces, like all ecosystems, is governed by the complex interactions of a variety of processes (McDonnell et al. 2009). In natural environments, the interplay between disturbance, decomposition, and biodiversity are well documented across both habitat and landscape scales (Attiwill 1994; Hooper et al. 2005). In urban environments, habitat patterns across landscapes (i.e. rural-urban gradient analyses) are similarly well documented, but small-scale processes are less well-known (McDonnell & Hahs 2008; Ramahlo & Hobbs 2012).

Ecosystem health is reflected in the consistency of processes and functions which are considered normal given the size, shape and historical context of a given site (Tzoulas et al. 2007). These processes and functions can only be judged for efficiency and normalcy given locally specific baselines to determine relevant changes as sites are exposed to the outside processes of surrounding landscape. Using decomposition, disturbance, and soils seed banks to characterise ecological function allows for research into the current and projected ecological health of

urban landscapes. When combined with site-specific biodiversity, these measures can help inform the dynamics that are fundamental to understanding an urban ecological system.

Biodiversity becomes an increasingly important factor in ecosystem function as the urbanisation of an area increases (Alberti 2008). Studies examining green space within the urban matrix have shown that diversity is often greater in areas where vegetation more closely approximates pre-urbanisation land cover (Koh & Sodhi 2004). Decomposability in an urban environment is widely variable and largely governed by the local physical and biological environment (Carreiro et al. 1999). At smaller scales, decay rates are determined by a combination of the chemical composition of leaf litter, biodiversity, and microclimates, contrasting with larger scales where climate has been found to be the largest predictor variable (Turner 1989). Disturbance in urban areas is well researched and presents in many forms, yet is commonly observed through the introduction of biogeographic barriers (i.e. roads, buildings, parking lots) which can increase habitat fragmentation and alter landscape connectivity (Alberti 2008).

Urban soils are generally characterised as of a poor quality, with reduced nutrient availability, high pH and high heavy metal pollution (Craul 1992). However, Pickett et al. (2008) found evidence in two major North American cities that urban soils may vary significantly in quality across the urban landscape. The highly variable nature of soils across the urban landscape necessitates an understanding of specific seed traits and abundance which have been shown to be

critical for successful recruitment and seedling establishment in urban forest restoration (Overdyck & Clarkson 2012).

Although site area has long been a more widely considered predictor of ecological organisation and function than that of site perimeter, research has become more inclusive of the latter (Horak 2016). Both area and perimeter are included in the preliminary analysis of this chapter and any statistical difficulties that may arise as a result of modelling collinearity will be addressed and discussed.

Socio-economic factors have been shown to be positively correlated with increased species richness in plants (Hope et al. 2003; Martin et al. 2004). To incorporate a socio-economic component this research utilised deprivation information available from the New Zealand Deprivation Score, an index categorising mesh block areas based on multiple socio-economic variables (Atkinson et al. 2014).

The aim of this chapter is to assess variation in the physical characters and ecological functioning of 36 roadside reserves that have been planted as part of the FPP and determine what relationship if any, this variation has with biodiversity. The primary questions being explored are as follows:

- 1) How do small-scale urban bush sites function in comparison to each other?*
- 2) Does variation in ecological functioning effect local biodiversity?*

It is expected that site-specific biodiversity will correlate with higher rates of decomposition as increased biodiversity can lead to greater variation in the

chemical composition of detritus and leaf litter. This is expected to result in higher decomposition efficiency, which will be largely regulated by local biotic factors (abiotic factors are not expected to vary significantly between sites). Biodiversity across sites is expected to decrease as a function of disturbance. Although increased site disturbance may limit recruitment in certain species and decrease the potential biomass through seedling establishment reduction (e.g. trampling), it is likely any observed relationship will have a weak correlation to both biodiversity and decomposition across all sites.

METHODS

Site Selection

Sites were selected from archived planting data provided by the Wellington City Council as part of the FPP. 150 randomly selected sites were culled down to 36 based on the following criteria: site is found on public land, site represents reasonably safe working conditions, there exists a strong likelihood that the site selected has been planted in conjunction with the FPP (archival records were limited to the address of recipient and brief descriptions of proposed planting site). Site locations can be found in Appendix I.

Site Characters

Study methods for the 3 measures of ecological function investigated in this thesis are expanded upon below. The full list of site characters included in analysis can be found in Table 1.

Decomposition analysis utilised Lipton® pyramid tea bags to determine decay rates, providing standardised bag weight, composition and mesh size. 6 tea bags were buried 8 cm deep in a random array at each of the sites and retrieved 90 days later. Bags were cleaned of loose debris and roots, dried at a constant temperature of 20°C for 48 hours, and weighed. This value was compared to an average pre-burial weight (1.94 g) determined through a random sampling of 15 bags given similar drying treatment. Due to logistic difficulties in the retrieval of all 6 bags from each site (bag locations were discreetly, but visibly marked and there were multiple incidents of missing bags on subsequent site visits) the original experimental design of repeated measures over time was abandoned for a single survey of any remaining bags. While this does not allow for insight into changing decomposition rates over time as a function of available temperature and rainfall data, it will provide a comparative measure of decay for analysis. However, due to the varying number of bags retrieved per site (1-6) it was decided to only include the bag with the lowest weight (highest decomposition) in analysis.

Seed bank character was intended to be assessed with emergence and extractive measures to provide both density and viability estimations (Abella et al. 2013). 3 soil core samples (3cm x 0-6cm) were taken from each site, washed using a 150um mesh sleeve, and hand sorted using US Standard #35 sieve (500um). Due

to the high frequency of low and zero values recorded, the final analysis will consist solely of the core exhibiting the highest density of seeds. As all core samples were randomly placed, the seed density value can be interpreted as an optimal representation of recruitment potential. Seeds were then sprouted over 45 days. However, due to low a low incidence of emergence within the prescribed timeframe and logistical constraints preventing the extension of said timeframe this aspect of the seedbank measure was not included in the final analysis.

Site biodiversity was assessed visually with vegetative counts and was largely limited to woody species. Where sites were thoroughly accessible and relatively small ($<10\text{m}^3$) full vegetative counts were conducted. At larger sites ($>10\text{m}^3$) or those sites where through access was impossible (dense or hazardous conditions) transects were established with 1m quadrats placed every 5m. Species level identification along with number of individuals present was recorded.

Disturbance is widely regarded as a significant driver of ecological assemblage and functioning (Attiwill 1994). The difficulty in qualifying and quantifying disturbance in ecological research is apparent in both temporal and logistic approaches. Due to the inaccessible and steep nature of many of my study sites it was decided to limit disturbance measures to erosion, slips, and other visibly evident disturbances along the edges of the study sites. It was hypothesized that much of the internal site disturbance found would be a result of intentional and unintentional human disturbance as a result of the planting scheme to which these sites are or have been subjected. It is hoped that much of this variation will be explained through differences in planting intensity across

sites, the examination of which will form the bulk of the analysis in Chapter 3.

Disturbance was determined through a visual assessment of site edges (due to site proximity to public access and roadways, as well as steep slopes, the majority of expected disturbance was assumed to be in relation to vegetative trimming/edging, mowing, and bank slippage). Disturbance was measured as a percentage of the total accessible site edge.

All field work was conducted October-January 2017.

Statistical Methods

A model selection approach was chosen due to the observational nature of much of the collected data and the assumed complexity of the ecological systems in question. This approach enables multiple hypothesis testing and is ideal for *post hoc* analysis when conducting research within the confines of non-experimentally designed scenarios (Johnson & Omland 2004). Candidate models were developed based on the proposed explanatory power of variables and variable interactions (Table 2). These models were compared utilising an information-theoretic approach where ranking was establishing based on Akaike Information Criterion (AIC). The small sample size of the study was corrected for using second order AICs and ultimately weighted to establish a given model being the best fit within the candidate set (Burham & Anderson 2002).

Table 1

Site characters surveyed

Variable(s)	Category	Measure	Method
Area; Perimeter	Physical	m ² ; m	Site boundaries mapped with GPS area and perimeter calculated in GIS
Slope	Physical	degrees	Measured using clinometer
Aspect	Physical	degrees from north	Measured using compass
Shape	Physical	ratio	perimeter/area
Deprivation Index	Socioeconomic	rank	Retrieved from existing data *
Disturbance	Ecological	%	Explained in text
Decomposition	Ecological	g	Explained in text
Seedbank Density	Ecological	highest sampled value	Explained in text
Biodiversity	Ecological	Shannon-Weiner Index	Explained in text

* Atkinson et al. 2014

Table 2

Candidate models for factors affecting site biodiversity

Variable(s)	Reference(s)
Seedbank	Albrecht et al. 2011; Overdyck & Clarkson 2012
Decomposition	Sayer 2006; Hattenschwiler et al. 2005
Disturbance	Attiwill 1994
Area	Cornelis & Hermy 2004
Slope	Chapin et al. 2011
Shape	Helzer & Jelinski 1999
Degrees from North	Chapin et al. 2011
Deprivation Index	Iverson & Cook 2000
Area x Shape	
Perimeter + Slope	
Degrees from North + Slope	Chapin et al. 2011
Seedbank x Perimeter	
Seedbank + Disturbance	Eriksson & Eriksson 1997
Seedbank + Decomposition	
+ Disturbance	

RESULTS

Descriptive Statistics

Physical factors measured at each site varies widely (Table 3). The surveyed area of sites showed 63.9% were considered small in size ($>25\text{m}^2$) while only 2 sites were moderate-to-large ($>150\text{m}^2$). The median slope for surveyed sites was 19.2 degrees and considered moderate while 27.8% of surveyed sites had a slope that was considered steep ($>25\%$). Site shape (perimeter-area ratio) ranged from 0.4 to 4.8 with a median ratio of 1.9. 75% of sites were found to have a ratio of <2.5 , with 44.4% of those sites exhibiting a ratio of between 1.5 and 2 (Figure 1). Sites with a higher ratio value were found to be less regular in shape than sites with a lower ratio value.

The lone social factor included in this study, deprivation index, was found to vary widely across sites, ranging from a score of 1 to 9 (out of a possible 10) with a median score of 4. Although 27.8% of sites had a middle score of 5 indicating neither association with affluence or poverty, 55.6% of sites were scored >5 indicating that these sites were located within relatively affluent neighbourhoods (Figure 1). All physical/social factors were assessed for collinearity with a significant relationship found between area and perimeter ($p<0.01$) (Table 4).

Table 3

Summary statistics for physical/social factors surveyed

	Area	Perimeter	Slope	Degrees from North	Deprivation Index	Shape
Mean	37.8	56.4	20.4	172.7	4.2	2.2
Standard Error	8.3	6.7	2.2	18.0	0.3	0.2
Median	20.5	42.1	19.2	151.5	4.0	1.9
Standard Deviation	49.5	39.9	13.2	107.7	2.0	1.0
Skewness	3.1	2.6	0.9	0.2	0.7	1.0
Range	240.6	210.6	53.6	331.0	8.0	4.4
Minimum	5.9	9.8	2.1	21.0	1.0	0.4
Maximum	246.5	220.4	55.7	35.0	9.0	4.8

Table 4

Correlation matrix for physical/social factors surveyed

	Area	Perimeter	Slope	Degrees from North	Deprivation Index
Area	-	0.824*	0.036	0.075	0.245
Perimeter	0.000	-	0.025	0.137	0.111
Slope	0.835	0.885	-	0.055	0.265
Degrees from North	0.664	0.426	0.749	-	-0.200
Deprivation Index	0.150	0.519	0.118	0.243	-

Note – correlation coefficients are presented in the top tier while p-values are presented in the bottom tier.

*Correlation is significant at the 0.01 level (2-tailed).

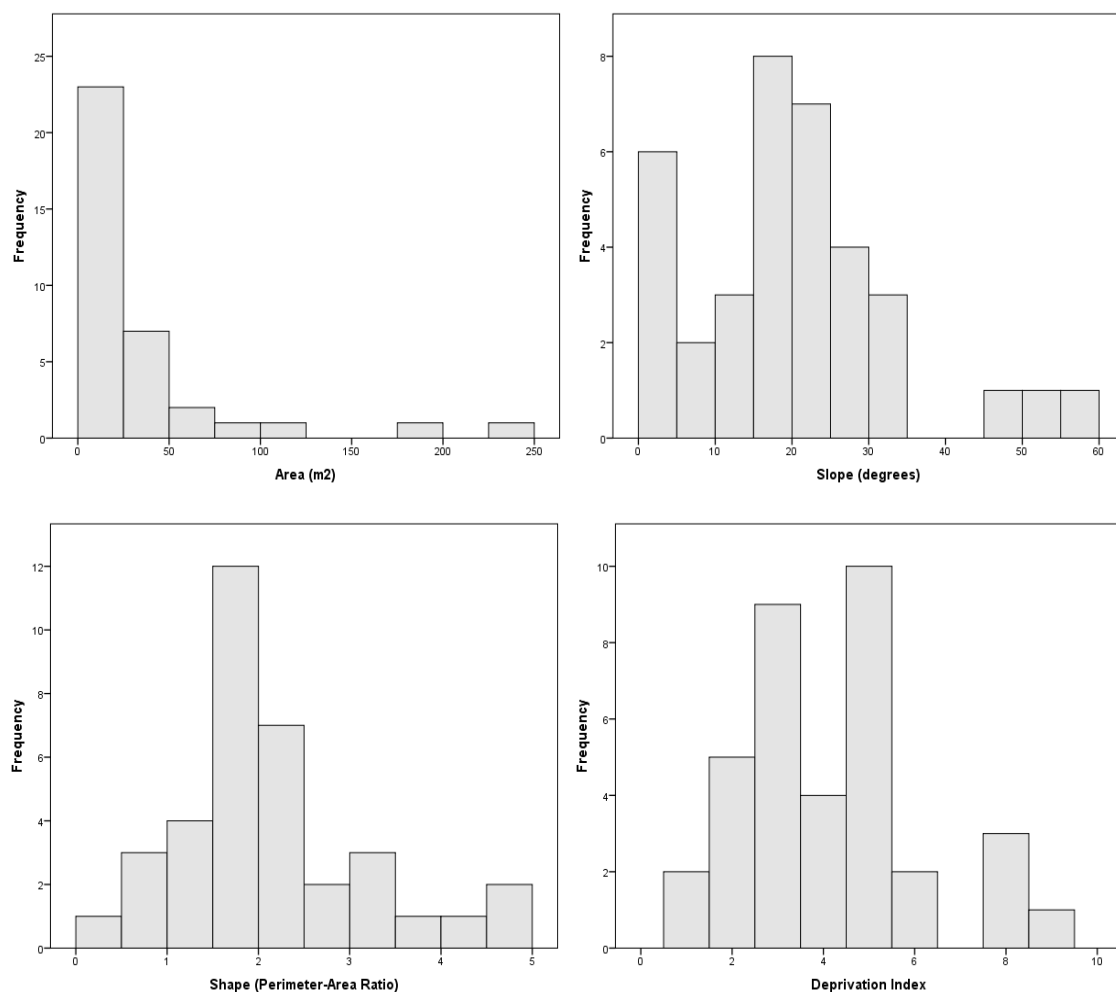


Figure 1. Frequency distribution of area, slope, shape, and deprivation index at surveyed sites.

Ecological factors measured at each site varied widely (Table 5). The variation found in decomposition was especially pronounced with final bag weights ranging from 0.2g to 1.9g with 63.9% of sites presenting a final bag weight between 1.0g and 1.5g. Median disturbance across sites was 20% with 72.2% of sites exhibiting low levels of disturbance (<30%). Only 1 site was considered heavily disturbed (>80%). Biodiversity levels were generally high with 66.6% of

sites maintaining an index score of 2.0 or higher, 37.5% of which were between 2.5 and 3.0. Seedbank densities were considerably lower than expected with seed numbers ranging between 3 and 34 individual seeds. 27.7% of cores sampled were found to have more than 20 seeds per 42.41 cm³ (Figure 2). These variables were assessed for collinearity with no significant relationships found (Table 6).

Table 5

Summary statistics for ecological factors surveyed

	Seedbank Density (#)	Decomposition (g)	Disturbance (%)	Diversity Index
Mean	16.9	1.3	23.5	2.1
Standard Error	1.4	0.1	2.7	0.1
Median	14.5	1.3	20.0	2.1
Standard Deviation	8.4	0.3	16.1	0.7
Skewness	0.7	-1.3	1.8	-1.4
Range	31.0	1.7	72.5	2.8
Minimum	3.0	0.2	7.5	0.1
Maximum	34.0	1.9	80.0	2.9

Table 6

Correlation matrix for ecological factors surveyed

	Seedbank	Decomposition	Disturbance
Seedbank	-	-0.174	-0.122
Decomposition	0.311	-	0.030
Disturbance	0.478	0.860	-

Note – correlation coefficients are presented in the top tier while p-values are presented in the bottom tier.

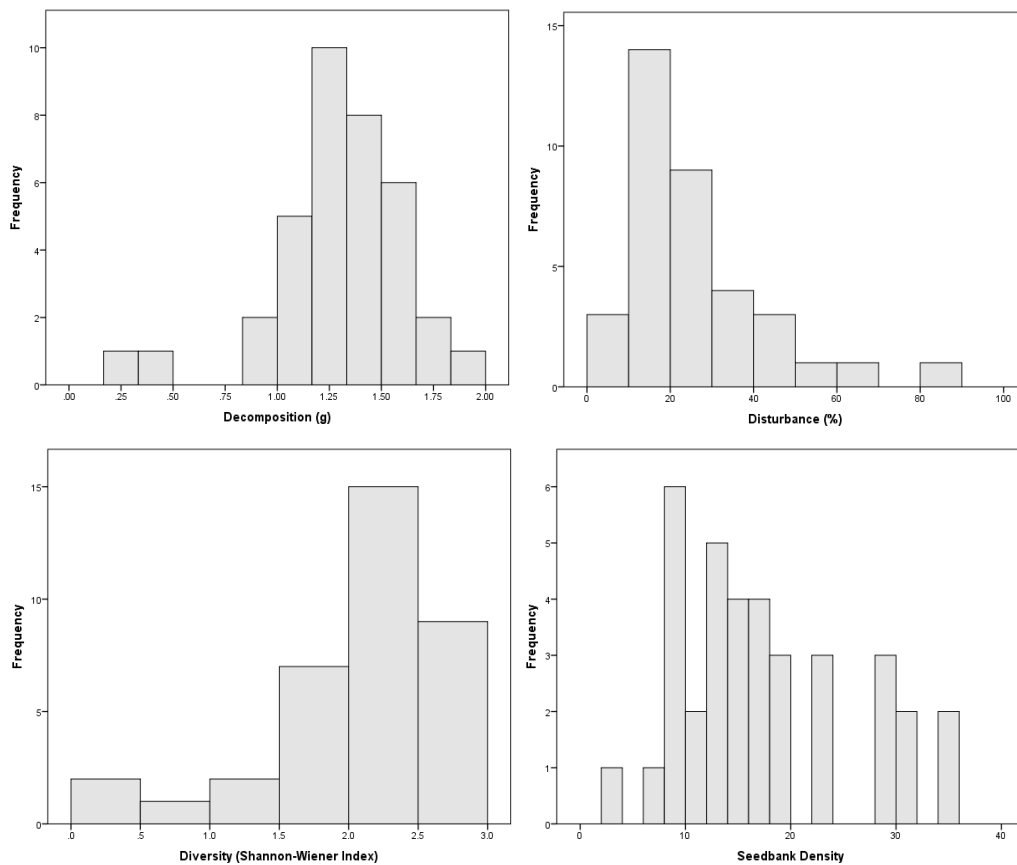


Figure 2. Frequency distributions for decomposition (final bag weight), disturbance, diversity index, and seedbank density at surveyed sites.

Model Selection

Although site perimeter was included in preliminary analysis, initial correlation assessment suggested that measures of perimeter and area were indeed linked. To avoid any statistical bias in the modelling associated with multicollinearity (Graham 2003), perimeter was not included in final model selection.

Fixed-effects models are presented in descending order according to their AIC weights (w_i) in Table 7. A 95% confidence set includes models 1 – 12 (sum of AIC weights = 0.9677). This set excludes Model 13 (Degrees from North + Slope) and Model 14 (Seedbank + Decomposition+ Disturbance), although as all models were within 10 Δ AIC of each other none can be omitted as lacking any explanatory power.

Table 7

Explanatory models describing site biodiversity

	Model	k	AIC	AICc (n=36)	Δ AIC	w_i
1	Area	3	-37.103	-36.353	0	0.2774
2	Site Shape	3	-35.574	-34.824	1.529	0.1291
3	Seedbank	3	-35.345	-34.595	1.758	0.1152
4	Deprivation Index	3	-34.910	-34.160	2.193	0.0927
5	Degrees from North	3	-34.578	-33.828	2.525	0.0785
6	Seedbank * Perimeter	5	-35.028	-33.028	3.325	0.0526
7	Disturbance	3	-33.348	-32.598	3.755	0.0424
8	Slope	3	-33.341	-32.591	3.762	0.0423
9	Decomposition	3	-33.339	-32.589	3.764	0.0422
10	Seedbank + Disturbance	4	-33.583	-32.293	4.060	0.0364
11	Perimeter + Slope	4	-33.357	-32.067	4.286	0.0325
12	Shape * Area	5	-33.640	-31.640	4.713	0.0263
13	Degrees from North + Slope	4	-32.625	-31.335	5.018	0.0226
14	Seedbank + Decomposition + Disturbance	5	-31.663	-29.663	6.690	0.0098

Note – Models are ranked by AIC weights (w_i)

Three single-factor models performed best based on AIC analysis of the candidate set. Although Area was ranked the highest ($AICc = -37.103$, $w_i = 0.2774$), Seedbank and Shape are within 2 ΔAIC and are therefore considered indistinguishable within the modelling confines (Burnham & Anderson 2002). Area was shown to have a weak positive relationship with site biodiversity ($R^2 = 0.080$). Although the majority of sites were small in size ($>50 \text{ m}^2$) these sites represented the full range of diversity levels found ($0.092 - 2.88$) (Figure 3). Both sites with an area over 150 m^2 were found to have a diversity index over 2.5. The shape ratio of sites was found to have a weak negative relationship with biodiversity ($R^2 = 0.061$). No sites with a perimeter-area ratio greater than 3.5 were found to have a diversity index over the mean for all sites (2.056) (Figure 4). Seedbank was found to have a weak positive relationship with site diversity ($R^2 = 0.055$) (Figure 5).

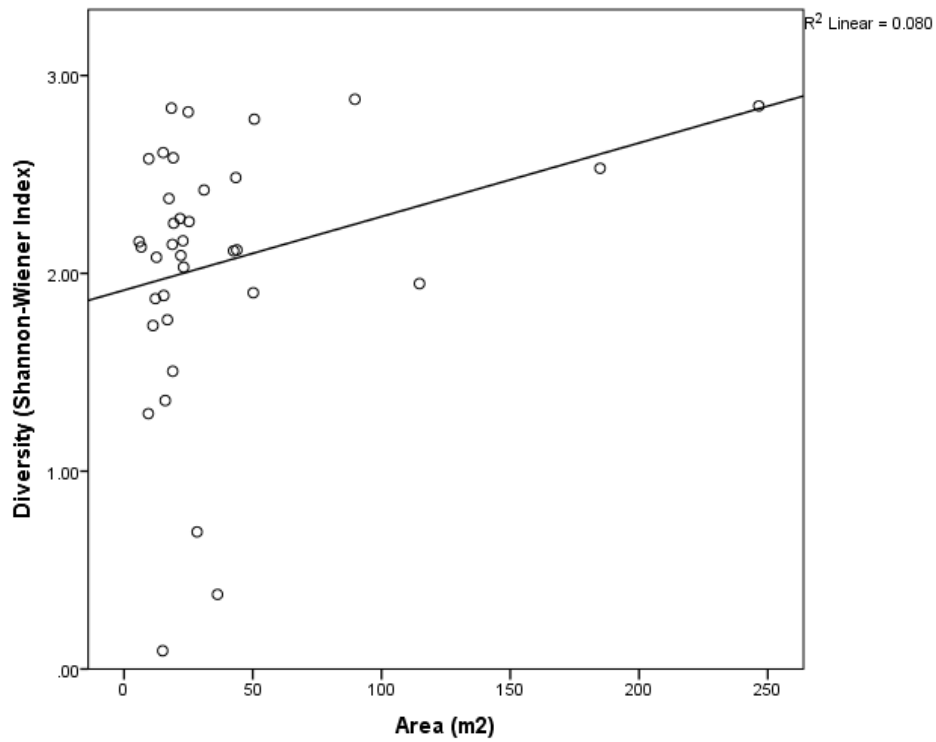


Figure 3. Site diversity as a function of area within each of the 36 sites surveyed.

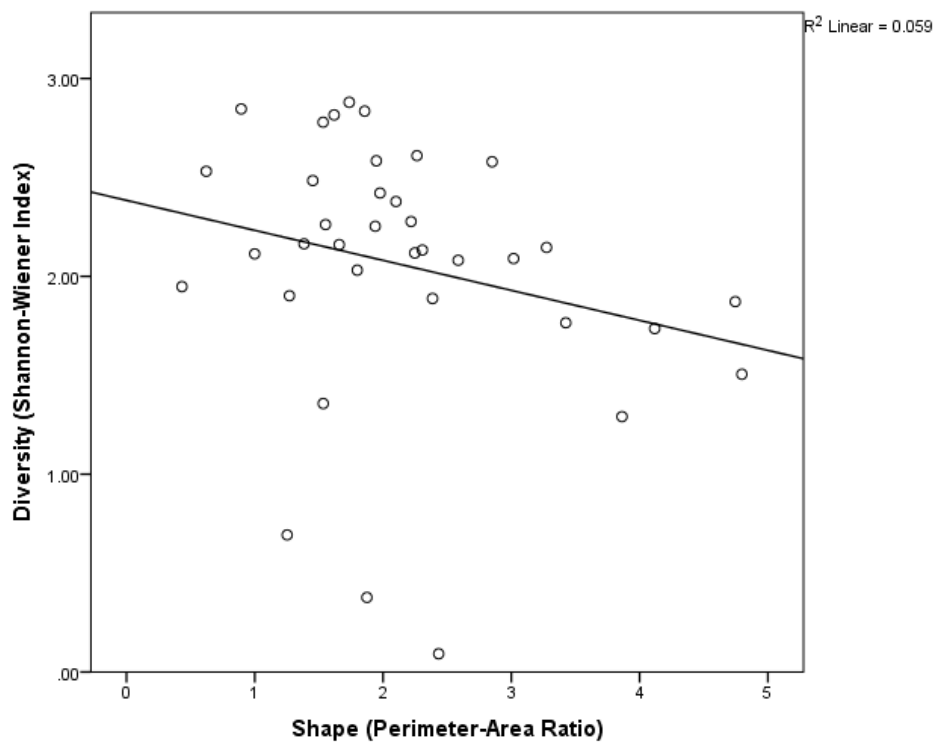


Figure 4. Site diversity as a function of site shape within each of the 36 sites surveyed.

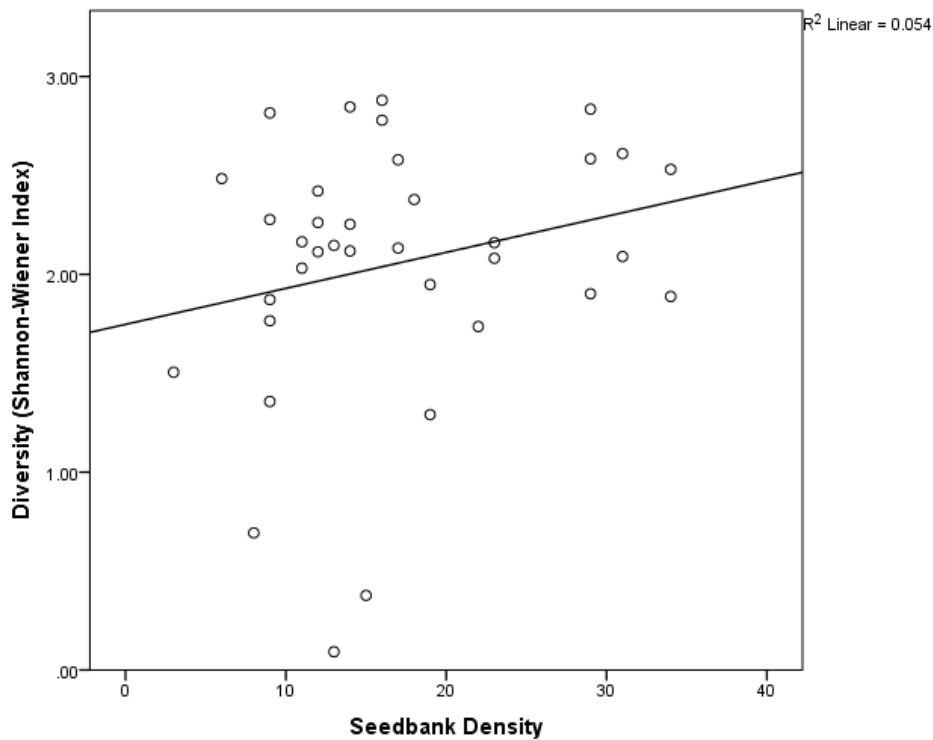


Figure 5. Site diversity as a function of seedbank density (# seeds/126.63 cm³) at each of the 36 sites surveyed.

DISCUSSION

Ecological Factors

Seedbank density was the only ecological factor found to play a significant role in predicting biodiversity within sites. This relationship is unsurprising as an increase in species types would also indicate an increase in seed types, some of which may be more readily identifiable. The existing floral composition of sites would be expected to be reflected in the seedbank, especially given the low

disturbance levels found across the majority of sites. The addition of an emergence measure in future studies would aid in seed identification and help in establishing a direct relationship between seedbank density, composition, observed biodiversity. However, these results highlight the importance of including seedbank analysis in long term monitoring and its potential as a biodiversity indicator (Duelli & Obrist 2003).

Model 14, the combined effect of disturbance, decomposition, and seedbank on biodiversity, was not contained within the 95% confidence interval set. Although this is contrary to initial hypotheses it is not unsurprising given the scale of the study sites. Small, fragmented land should theoretically be governed by edge effects and localised disturbance, as has been previously found in New Zealand podocarp-broadleaf forests (Young & Mitchell 1994). The consistency of the findings of this study with research conducted in less human-dominated environments would indicate that there is potential for a careful and deliberate application of traditional ecological theory to novel, urban ecosystems.

Physical Factors

Similarly, the finding that area and shape were among the best performing models explaining biodiversity conforms with existing ecological theory based on work done with larger areas (Schwartz et al. 2013; Young & Mitchell 1994). A negative relationship was found between biodiversity and increases in the perimeter-area ratio. As sites with a larger ratio have an increased perimeter for a comparatively small area, this would indicate an increase in the deleterious species

effects associated with edge exposure. However, the expression of this negative relationship may also be a result of several indirect physical factors. Site accessibility or visual exposure could theoretically lead to an increase in human involvement. It was thought that increases in exposed edges, especially those surrounded by roads and foot path, would yield higher levels of disturbance. However, disturbance levels were found to be generally low and modelling suggests a poor explanatory power of the variable. This may be indicative of a more complex relationship that was not uncovered in the course of this research.

Additionally, there may be individual species characteristics governing establishment and recruitment within these sites. Native species have the potential to perform more favourably than non-natives within a purposely planted regiment (Berentson 2013), but as this study did not make distinctions between species status it is possible that the high vector pressure represented by roadways is actually a driving force in site biodiversity.

Although small sites ($< 50 \text{ m}^2$) were found to have varying amount of biodiversity, the only three sites with a diversity index of less than 1.00 were all classified as small. Similarly, the only two sites with areas greater than 150 m^2 both had diversity indexes over 2.00. Again, this may be indicative of the complexities associated with ecological systems whereby any number of factors may be responsible for the observed characters present at sites. While smaller sites were found to be less consistently diverse than their larger counterparts, they may still represent value for the conservation of habitats and species with a

naturally-occurring isolated distribution (Schwarz 1999). The effects of increases in an perimeter-area ratio is well documented, yet changes in our understanding of the supposed permanence of such fragmented landscapes has become a cornerstone of restoration ecology and is crucial to our re-imagining of the ecological potential of the urban landscape (Young 2000).

Socioeconomic Factors

Deprivation index was not found to be a significant predictor of site biodiversity. The scope of this research did not extend to population measures as explicit models which has been shown to have a negative correlation with biodiversity potential as well as ecosystem services (Tratalos et al. 2007). While previous research has shown that increases in residential vegetation can exhibit a strong relationship with localised socio-economics (Martin et al. 2004), no such pattern was detected during the course of this study. It was surprising that the deprivation index, which varied significantly across sites, was not correlated to biodiversity. It is possible that the topographical nature and comprehensive planning of the Wellington region (i.e. Outer Green Belt), designed for city-wide access to greenspace (Beatley 2016), reduces the inequity of experience often associated with socioeconomic gaps. Similarly, although socio-economic boundaries in New Zealand cities have been found to be significantly related to accessible local biodiversity, much of this disparity is attributed to mature, established gardens on private property (Hand et al. 2016), and thereby removed from the scope of this study.

Further Research

Ultimately, the restoration value of small urban road reserves will be largely governed by local priorities. Small urban patches have the same associated risks that patchiness has long represented in non-urban landscapes, acting as ecological sinks which negatively affect population growth rates (Pulliam 1988). Although urban patches may support higher proportions of exotic species than rural counterparts, levels of biodiversity (both native and exotic) can be expected to drop precipitously over time as succession events move away from pioneer species (Angold et al. 2006). Small patches within an urban landscape can increase select species diversity but proximity to larger sources help eliminate the sink potential of small, isolated urban areas increasing overall diversity within the urban matrix (Loss et al. 2009). Examining small habitats within the context of the larger urban matrix may help inform observed levels of biodiversity as well as overall ecological functioning.

The ramifications this has for urban greening initiatives is currently under researched and, as such, long-term monitoring programmes are being developed and implemented (Pickett et al. 2008). To fully understand ecological functioning and interaction dynamics within restoration sites multiple measurements are required over significant periods of time (Ruiz-Jean & Aide 2005). The obvious drawback to explicit long term monitoring schemes lies in increased cost and resource investment which might explain the prevalence of one-off experimental designs.

The longevity of urban patches is largely determined through city planning and development as well as community engagement. Although the rationales associated with urban greenspace preservation and restoration is well established (Clewel & Aronson 2006; Peters et al. 2015; Svendsen & Campbell 2008), further research is needed to quantify the ecological value of urban greening efforts. This is explored in the following chapter.

Chapter 3

THE ECOLOGICAL EFFECTS OF COMMUNITY PLANTING OVER A 25 YEAR PERIOD IN WELLINGTON, NEW ZEALAND

INTRODUCTION

Urban greenspaces are frequently targeted by community restoration groups in a variety of ways (e.g. native planting, weeding, public policy initiatives, increased aesthetics), and are thus considered to be major drivers of urban vegetation (Conway et al. 2011). Recent trends have shown an increase in the level of community-based contributions to restoration and conservation efforts, largely attributable to expanding educational outreach and economic considerations (Bramston et al. 2011). Additionally, volunteer community restoration efforts are often designed with social, ecological, and economic considerations (Clewett & Aronson 2013). Engaging the public can also ease many of the complexities and political controversies associated with ecological restoration efforts, leading to enduring and self-sustaining, citizen science-based projects (Buzier et al. 2012; Theobald et al. 2015).

Small-scale restoration efforts in particular offer not only the most accessible avenue for volunteer involvement, but may also provide urban ecologists a unique opportunity to evaluate the socio-ecological benefits of such engagement in an experimental context. Sites which are manipulated through

planting efforts are likely to be the target of future community attention which may consist of continued planting, weeding, or recreation (Tzoulas et al. 2007; Krasny et al. 2014). These activities may contribute significantly to levels of biodiversity, both directly, as would be seen with purposeful native planting, and indirectly, through an increase in localised disturbance and altered decay rates. Community planting programmes therefore provide a unique opportunity for research of both ecological effect as well as an opportunity to examine social impact on the community.

Urban greening efforts targeting comprehensive green networks on city scales have been implemented in a number of European and America cities, and, in recent years have begun to take shape throughout New Zealand (Viles & Rosier 2001). The Wellington City Council's Free Plants Programme (FPP) has been operating for 25 years providing individual plants to local residents to be planted on public land, most often small roadside reserves. These reserves border public and private land and represent an excellent post hoc opportunity to examine the ecological effects of an urban greening scheme.

In a review of 301 articles presenting restoration efforts and success indices, Wortley et al. (2013) found that 98.5% of studies indicated an ecological focus, with only 2% of those also containing an examination of related social aspects or impacts. A similarly sized study examined the self-proclaimed objectives of community restoration groups and found that 72.9% of groups prioritise social objectives, the most common being 'community-building' and 'education and awareness' (Peters et al. 2015). The stark differences between

these findings illustrate the need for comprehensive research into the socio-ecological effects of community-based restoration activities and an examination of the measurable ecological impacts of volunteer involvement. This will help to ensure that both ecological health and continued support to/from volunteer organisation continues unabated (Peters et al. 2015).

Successful landscape connectivity within an urban system requires regionally specific ecological analysis (Viles & Rosier 2001). The establishment of individual urban greening initiatives across a spectrum of social and ecological conditions will facilitate the development of broader theory, and in turn, inform future management and development decisions. There is considerable evidence supporting the ecological value that unmanaged land represents in the urban landscape: allowing urban areas to vegetate spontaneously without human influence can support habitat provisioning for plant diversity (Robinson & Lundholm 2012) and arthropod assemblages (Small et al. 2003; Gödde et al. 1995).

As restoration efforts involve considerable time and resources the impact they have must be carefully evaluated. Local diversity and ecological functioning measures are among the most commonly assessed metrics in determining the success of ecological restoration projects (Wortley et al. 2013). These endeavours can have demonstrable effects on ecosystem functioning including the support of local biodiversity (Cooper et al. 2007). This chapter examines planting regime variation within FPP sites and the relationship between these variables and ecosystem function and biodiversity.

The primary question being explored is as follows:

How do planting efforts, defined as the number of years a given site has been planted and the number of individual plants provided to that site, over a 25 year period effect ecological functioning and biodiversity within small roadside reserves?

It was hypothesised that planting intensity (combined variation in number of planting years and planted individuals) is unlikely to affect ecological function including biodiversity. However, it is possible that the repeated addition of large numbers of individual plants may increase local biodiversity, especially on a short term basis. As a result of this, recently planted sites may boast temporarily high diversity levels and this may lead to significant correlations between planting intensity and one or more dependant variables, the most likely of which would be biodiversity.

METHODS

The aim of this chapter was to assess whether there is a relationship between the planting history of sites and measures of ecological function (as outlined in the previous chapter) and biodiversity. The primary question being explored is as follows:

Does planting intensity (planting years and number of planted individuals) within the Wellington City Council's Free-Plants Program (FPP) contribute to local biodiversity and ecosystem function?

Data Collection

Planting intensity was assessed through two distinct measures: the number of years a given site had been subjected to planting through the FPP (planting years) and the cumulative number of individual plants provided (planted individuals) (Table 8). Determining both measures presented similar challenges in that the available database information was organised not in accordance with planting location, but rather residential data regarding the individual requesting plants. A thorough database investigation and repeated site visits indicated that any given site may have been subjected to planting by multiple members of the neighbourhood or community, the involvement of whom would not be directly linked to the selected site (original site selection was largely informed through the specificity of targeted site descriptions). To compensate for this planting data for any given site was expanded to include recorded incidences of planting which exhibited a strong likelihood that the selected site was the targeted destination. Properties were often listed as programme recipients in multiple years and any recurring property with conflicting targeted site locations was not included in the cumulative numbers used in the final analysis. No property was assumed to have been involved in the planting of a site if located more than 100m from the site, unless explicitly indicated in the database.

Table 8

Historical site variables surveyed

Variable	Category	Methods
Planting Years	Historical Context	Retrieved from existing database sources*
Planted Individuals	Historical Context	Retrieved from existing database sources*

*Data provided by Wellington City Council (Berhampore Nursery records for years 1990-2015)

Statistical Methods

A MANOVA was used to test the relationship between the number of years a site had been planted and the total number of individuals provided with measures of decomposition, disturbance, seedbank density and biodiversity. Predictor variables were tested separately as a combined analysis failed Levene's Test. Univariate testing was conducted to determine any significance of between-subjects effects (Bolker et al. 2008).

RESULTS***Descriptive Statistics***

Historical context variables measured at each site varied widely (Table 9). The number of planted individuals at each site ranged from 10 – 601 with a mean of 89.4. 34 out of 35 surveyed sites received less than 200 individual plants over

the 25 year data period with 37.1% of sites receiving less than 50 individual plants (Figure 6). The number of years a site was planted ranged from 1 – 6 years with a mean of 2.5. 31.4% of surveyed sites were determined to have been planted once, while a single site was found to have been planted on more than 5 separate years (Figure 7). Both maximum values for planting years and planted individuals were captured in the same site. A significant correlation was found between planting years and planted individuals (Pearson $r = 0.819$, $p < 0.01$).

Planting years and planted individuals frequencies can be found in Figure 6 and Figure 7.

Table 9

Summary statistics for historical site variables surveyed

	Planting Years	Planted Individuals
Mean	2.5	89.4
Standard Error	0.3	17.3
Median	2.0	58.0
Standard Deviation	1.5	102.3
Skewness	0.9	3.9
Range	5.0	591.0
Minimum	1.0	10.0
Maximum	6.0	601.0

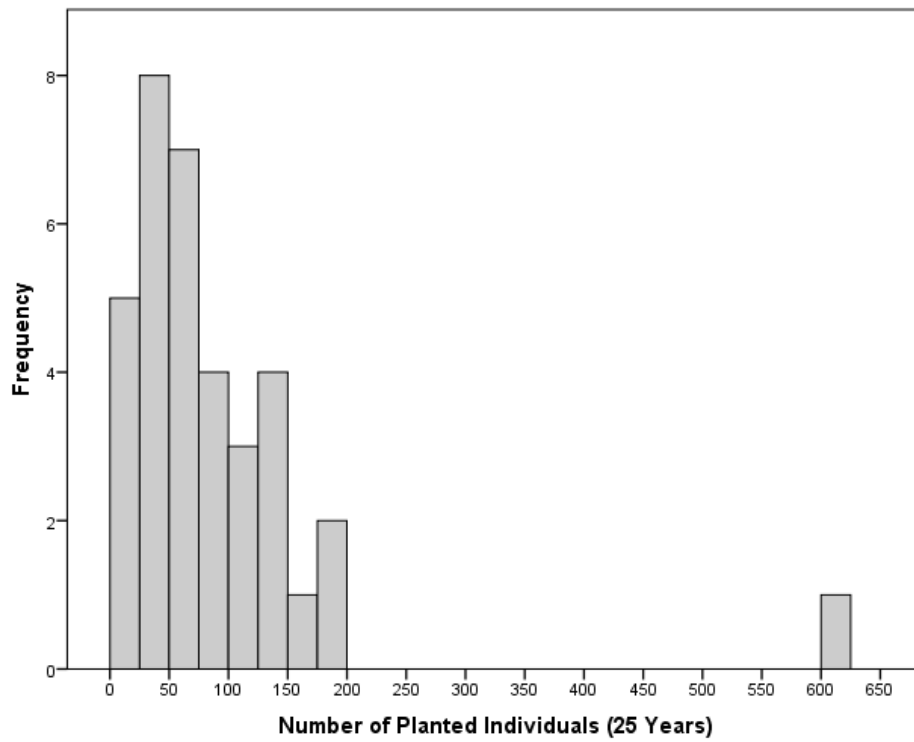


Figure 6. Frequency distribution for the number of planted individuals

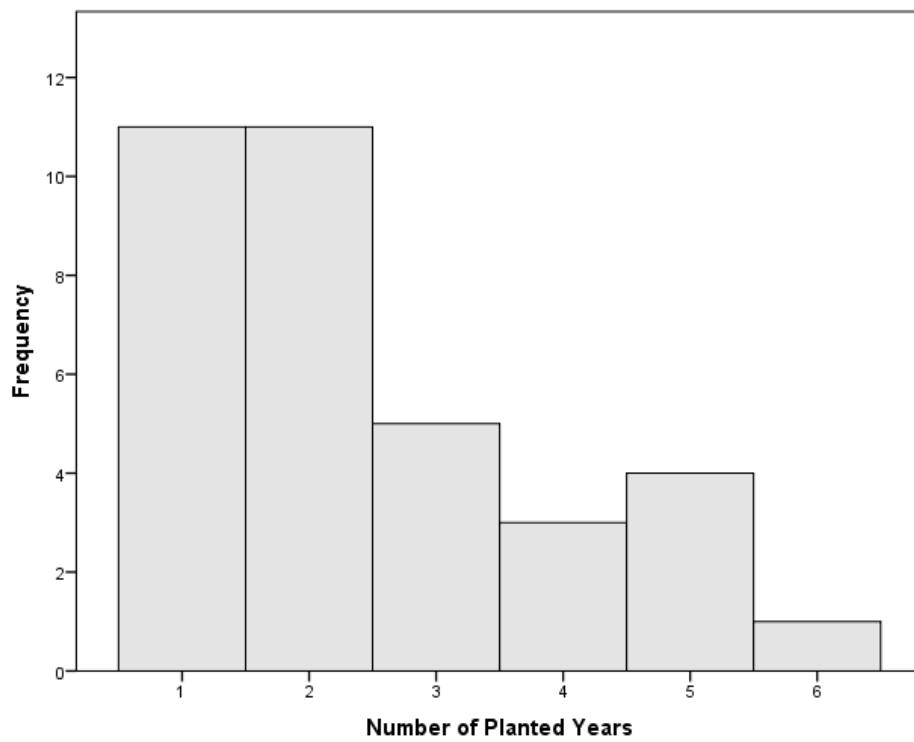


Figure 7. Frequency distribution for the number of planting years

Planted Individuals

The number of planted individuals did not explain combined measures of biodiversity and ecosystem function (Pillai's Trace = 0.201, $F_{30,4} = 1.881$, $p = .140$). Univariate testing found a somewhat significant positive relationship between the number of planted individuals and biodiversity ($F_{33,1} = 3.536$, $p = .069$). No significant relationship was found between the number of planted individuals and disturbance ($F_{33,1} = 1.535$, $p = .224$). No significant relationship was found between the number of individuals planted and seedbank density ($F_{33,1} = 0.045$, $p = .833$). No significant relationship was found between the number of planted individuals and decomposition ($F_{33,1} = 1.699$, $p = .201$).

Although not statistically significant, a slight negative relationship was described for both seedbank density and disturbance as a function of an increase in planted individuals, while a slight positive relationship was described for decomposition as a function of the same measure (Figure 8).

A potential outlier was identified and can be seen in Figure 8. Multivariate testing was re-run without this site and no significance was found (Pillai's Trace = 0.194, $F_{29,4} = 1.473$, $p = .236$).

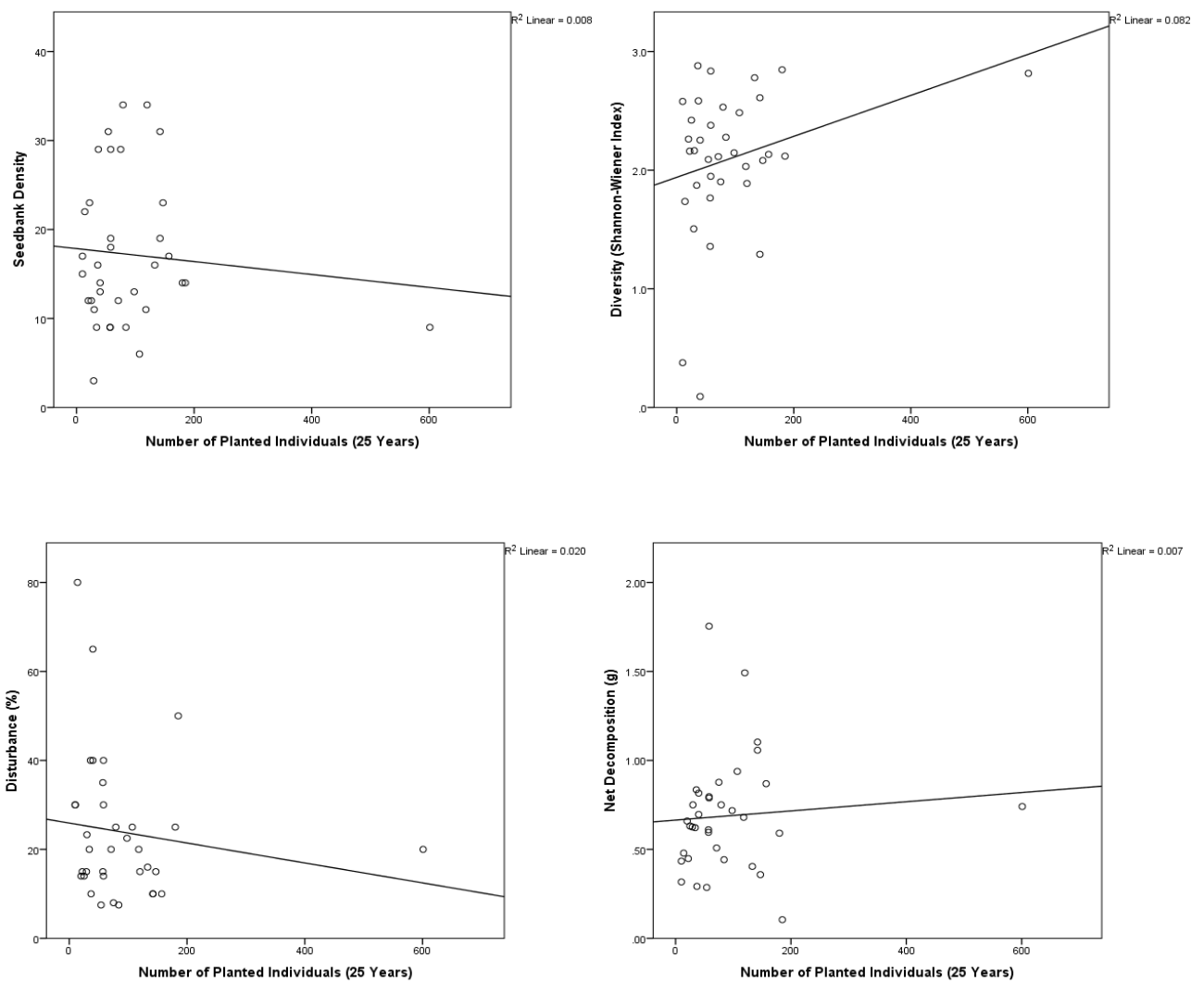


Figure 8. Four ecological factors as a function of the number of individuals planted at each study site over a 25 year period.

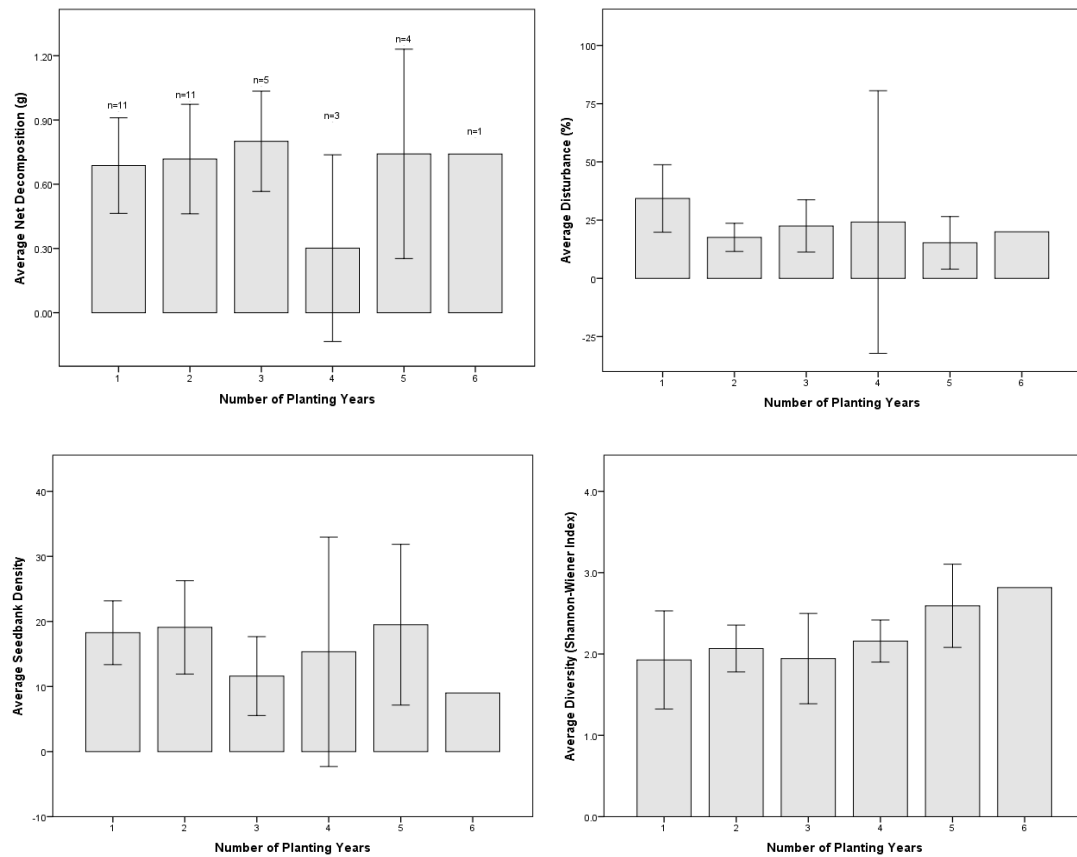
Planting Years

The number of planting years come close to explaining combined measures of biodiversity and ecosystem function (Pillai's Trace = .245, $F_{30,4} = 2.429$, $p = .069$).

Univariate testing found a near significant negative relationship between the

number of planting years and disturbance ($F_{33,1} = 4.092$, $p = .051$). No significant relationship was found for the number of planting years and decomposition ($F_{33,1} = 0.034$, $p = .885$). No significant relationship was found for the number of planting years and seedbank density ($F_{33,1} = 0.790$, $p = .384$). No significant relationship was found between number of planting years and biodiversity ($F_{33,1} = 1.654$, $p = .207$) (Figure 9).

Multivariate testing was again re-run without the site identified as a potential outlier and no significance was found (Pillai's Trace = 0.222, $F_{29,4} = 2.063$, $p = .112$).



Error bars represent 95% CI

Figure 9. The average measure of four ecological factors as a function of the number of years a given site was planted.

DISCUSSION

It is not surprising that the number of planting years and planted individuals was found to be strongly correlated; fidelity to any particular site would likely result in long-time residents repeatedly requesting plants for a specific location. Accepting this, the most salient question becomes the extent to which repeated greening efforts affect these sites. Although planting intensity

(combined measure of planting years and planted individuals) was not found to have a significant effect on ecological functions and biodiversity, there is weak support for a relationship between these measures and planting years.

Planting Years

Interestingly, only the disturbance measure showed a significant relationship to planting years. As the number of years a site was planted increased reduced levels of disturbance were observed. Repeated planting can be considered a site commitment on the behalf of residents, which may explain this result. Prolonged greening efforts can impart significant feelings of stewardship towards residents (Armstrong 2000), and as such, residents may be more vigilant of site degradation, augmenting planting efforts beyond those associated with the FPP. Similarly, continued planting efforts may be indicative of increased residence time or neighbourhood age which have also been linked to increases in biodiversity and greenspace (Loss et al. 2009). Although sites planted in conjunction with the FPP are removed from public works lists of mowing and weeding targets, there was evidence of clearing activity found in the course of this research which may have contributed to measured disturbance.

Planted Individuals

The only somewhat significant relationship found for planted individuals was site biodiversity. As no such relationship was described for planting years, these results support the initial hypothesis that increasing the number of plants

provided to site would increase the measure of site biodiversity, especially on an immediate basis. Combining the lapsed time between the most recent planting events with repeated biodiversity measures would provide insight into ongoing mortality rates associated with this urban greening initiative. Previous research has shown that survival rates of planted seedlings in the Wellington region is influenced by species-specific and human factors (Berentson 2013), and may be improved through a concerted focus on providing larger, more robust seedlings (Anton et al. 2015).

The FPP has transitioned to providing almost entirely native plants in recent years, maintaining high stocks in conjunction with the Berhampore Nursery. In general, participation in the FPP is largely correlated with an existing knowledge of horticulture, which likely extends to the local suitability of various plants (Berentson 2013). This is promising as recent research has shown that urban greening initiatives are often limited by residents unaware of native plant availability and a nursery trade with limited supply (Torres-Camacho et al. 2017).

Future Research

Community-based urban greening efforts often lack a goal-driven operating framework and may benefit from an adaptive co-management approach (Olsson & Folke 2004). Although urban greening initiatives are often community-led and in response to a catalytic event (i.e. neighbourhood degradation or natural disaster/disturbance) (Krasny et al. 2014), the involvement of scientific agencies

and local governing bodies can result in longer-lasting, wider-impacting restoration efforts (Svendsen & Campbell 2008; Ostrom & Cox 2010).

This research has shown preliminary evidence that the level of planting intensity within small roadside reserves have not measurably affected the ecological function and biodiversity of the study sites. There has been, however, no current effort to ascertain any influence upon the larger urban landscape as a result of local planting efforts. Similar programmes have prioritised large areas to act as ecological 'hubs', facilitating a source dynamic thereby aiding in biological diversification and species movement across the urban gradient (Walmsley 2006). While there is undeniable value in engaging with a resident's desire to beautify their local surrounds and increasing feelings of stewardship with nature (Andersson et al. 2014; Chapin et al. 2010), the extent to which this aides in the creation of sustained refuges for priority taxa is currently unquantified. Future research should focus on the extent to which the FPP supports increased landscape connectivity between planting sites.

As no attempt was made to ascertain site age as a product of land development, it is possible that some variation measured within sites is also correlated with non-FPP related succession dynamics. The age of urban patches has been shown to positively affect diversity levels of arthropods (McIntyre 2000), as has the age of neighbourhoods been positively correlated with diversity levels of plants (Martin et al. 2004). Age dynamics associated with small-scale urban restoration warrants further study in the Wellington region.

Expected increases in city size and density will place additional stresses and value on urban ecosystem function and services, underpinning the need for sustainable socio-ecological frameworks (Chapin et al. 2010; Savard et al. 2000). Compounded with current research gaps associated with community planting programmes in countries other than the USA (Guitart et al. 2012), there is a growing need for comprehensive research with an early emphasis on a local and city-level scale.

CONCLUSIONS

Modelling conducted over the course of this research indicate that the size and shape of roadside reserves are the most important predictors of biodiversity. Holding with traditional ecological theory, the area and shape of reserves describe potential recruitment limitations associated with edge effects and habitat fragmentation. This functional similarity between small urban patches and larger ones outside of the urban landscape suggests the need for additional research into the effect of scale when examining urban habitat. The novelty of urban ecosystems in the field of ecology is predicated on an assumption of dissimilarity, the extent of which will only become better understood. Additionally, further research into seedbank analysis as a biodiversity indicator for urban habitat is recommended, specifically within the context of a long-term monitoring framework.

Increases in planting years were associated with a reduction in site disturbance and increases in planted individuals were associated with higher biodiversity. These findings support the FPP goals and will hopefully aid in future management decisions. Sites targeted for future urban greening may benefit from efforts which utilise specific restoration goals. As the edges of roadside reserves

are generally paved roads or footpaths a concerted focus towards planting along these edges may provide site buffering and the promotion of biodiversity.

Combining environmental, social and historical factors may present a more comprehensive understanding of the factors affecting diversity within and across an urban landscape. Understanding the complexities governing urban ecosystems is critical for sustainable greenspace development. Key issues include individual patch dynamics and position within urban landscape, biological indicators, and the informed development of a comprehensive, locally specific, socio-ecological framework.

The value of urban greenspace is subjective; ecologists, local managers, community groups and stakeholders may possess varying metrics for successful urban greening. While the definition of success may differ, they are not necessarily mutually exclusive. The establishment and maintenance of urban biodiversity and robust ecological functioning requires a “participatory management” approach, incorporating socio-ecological dynamics that can result in “different and potentially complementing profiles” within the urban landscape (Andersson et al. 2014, p.449).

The adaptation of traditional ecology theory to novel urban ecosystems is as complex as it is critical: urban landscapes are complex mosaics in which humans and our constructs act as both backdrops and drivers of ecological function and habitat dynamics. Any application of an “ecology of cities” methodology to urban restoration must begin with the acceptance that humans influence the

connectivity and organisation of habitat patches as well as the assemblage of species within these patches (Niemelä et al. 2009). As biotic succession is both directly, and indirectly, influenced by human activity (Bradshaw 2003), a successful framework for the future of greenspace research, planning, and management requires such considerations. As research has shown that biodiversity gains associated with ecological restoration initiatives can be significantly compromised when site selection is not based on biodiversity objectives (Mason et al. 2012), the promotion of healthy ecological functioning and biodiversity within urban greenspace will require ongoing community engagement, biological monitoring, and explicit restoration goals.

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APPENDIX I:

Location of study sites across Wellington

