TWENTY YEARS OF PROVIDING FREE PLANTS IN AN URBAN NEW ZEALAND SETTING; WHAT AFFECTS COMMUNITY PARTICIPATION AND PLANTING SUCCESS?

BY

PAUL BERENTSON

A thesis submitted to the Victoria University of Wellington in fulfilment

of the requirements for the degree of Master of Science

Victoria University of Wellington

2013

ABSTRACT

An urban greening programme in Wellington, New Zealand providing free plants to city residents was evaluated with the following objectives:

1. To assess the levels of plant survival after five, ten, and fifteen years and determine factors contributing to observed survival;
2. To investigate factors influencing participation in the programme;
3. To quantify the some of the socioeconomic factors relating to programme participants and environmental factors relating to sites.

Data were collected from a combination of council records, site surveys and postal questionnaire surveys. The study found that plant survival was generally poor, but was mainly influenced by indigeneity of the plants. Contrary to many theories of exotic invasiveness, New Zealand native plants were 4.3 times more likely to survive than exotic plants. Site based effects were not found to influence survival significantly; nor were specific plant traits, or year of planting. A small sample of these sites was matched to questionnaire responses and it was found that length of residence by programme participants increased the performance of the best model indigeneity, indicating that increasing length of residence was a predictor of better survival of plantings. The questionnaire respondents included both those who had participated in the programme and those who had not. The sample population, however, was quite distinct from the general population of the region, being older, wealthier, having higher levels of education, and twice as likely to own their own home. As suggested by previous research looking at the effects of socioeconomic factors on urban forestry or urban greening participation was shown in this study to be mainly affected by the age of the respondent, which increased the odds of participation by 200% between the youngest and oldest age groups. This socioeconomic model was improved when two factors were included: the number of trees outside their property, and, horticultural knowledge of the participant. This indicates that participants might be more motivated by personal interest in horticulture, than in improving environmental conditions.

ACKNOWLEDGEMENTS

This research was made possible by the provision of access to plant records by the Wellington City Council, who also provided funding to facilitate research activities. The Wellington Zoo Trust also provided funding through their Conservation Fellowship. I would like to thank both these funders for their generous support.

A big thankyou goes to all the Wellington residents who completed my questionnaire and shared their experiences of urban planting in our wild little city.

I was lucky enough to have two supervisors assisting me throughout this thesis, and I sincerely thank Dr Heiko Wittmer and Dr Wayne Linklater for their enthusiasm and ongoing commitment to this research.

And most importantly of course, a huge thankyou to my family Jess, Ngaio (and now one more) Florence. Your support and sacrifices and constant encouragement got me through.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

INTRODUCTION

Urban Greening is a concept increasingly recognised as an important aspect of modern city design (Lane & Raab, 2002; Tidball & Krasny, 2007; Wellington City Council, 1998). The initiative of planting trees on otherwise disused land has many potential benefits, ranging from the aesthetic to the functional. Functional benefits might be physical such as erosion control, improvement of air quality and provision of wind shelter (Cameron et al., 2012; Summit & Sommer, 1998); ecological by increasing resources and connectivity (Nowak & Dwyer, 2007); social such as reducing stress (Ward Thompson et al., 2012) or empowerment (Westphal, 2003); or economic by increasing land values (Wachter & Wong, 2008). Taking planting beyond the traditional parks and gardens however, introduces some greater challenges to plant establishment and growth and both environmental and human influences become important. This study evaluates a programme encouraging planting of urban and suburban sites in Wellington with the aim of indentifying factors that affect planting success and public planting participation.

Background

The Wellington City Council (WCC) has since 1990 provided free native plants once a year for residents to plant on public land such as road reserve or other reserve, either for aesthetic or ecological restoration purposes. The programme is referred to in this thesis as the Free Plants Programme (FPP). The motivation for assessing this urban restoration programme in particular comes from the fact that records have been kept describing location of sites and plants provided. This presents a rare opportunity to compare field (Figure 1) and questionnaire surveys with historic planting data in an urban setting.

Road reserves are the strip of land between a developed road and the boundary of the adjacent property. In general a road reserve is intended to allow for future widening of roads, should it be required, and in Wellington it usually varies in characteristic according to the topography of the landscape. In particular, the steep terrain of much of the city's suburban areas requires roads to often be terraced into a hillside, therefore the width and slope of the road reserve is often governed by the slope of the hill, resulting in a range of widths, aspects, slopes, and degree of soil modification by excavation. Given this variation it is expected that site characteristics may affect the outcomes of plantings. Most road reserves in Wellington are relatively unmanaged, having occasional mowing or weed control, by cutting or herbicide spraying. Apart from designated road reserves, other potential planting sites are public reserves such as the "green-belt" of undeveloped land designated by Council plans (Wellington City Council, 2004a).

The intention of the FPP is to improve Wellington's environment by "making Wellington a better place to live" with the specific aim being to "increase our native plant populations; and provide habitats for our native animals and reduce the weed problem" (Wellington City Council, 2010). Recently, the WCC has developed an approach grounded in restoration ecology and further states that native eco-sourced plants are used: "to keep the distinctiveness of Wellington's local flora; to avoid the risk of planting species that could become invasive; and, as local plants are better suited to Wellington's conditions they are easier to grow". To date, the success or otherwise of the programme has not been evaluated. As such it is unknown whether the plants have been successfully established, whether the scheme has had any effect on the Wellington environment, or met the programme objectives.

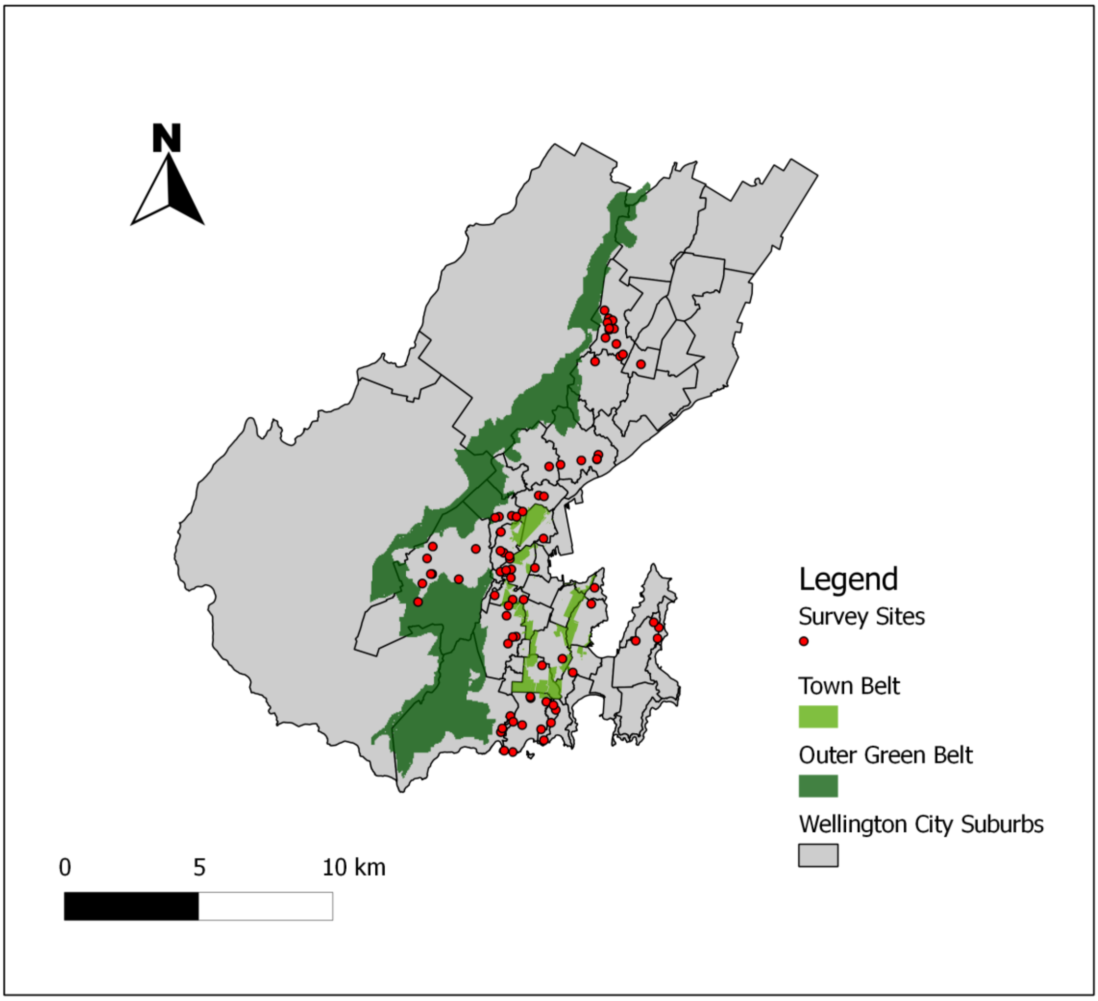


Figure 1: Sites surveyed for this study showing Wellington City suburbs and green belts (Wellington City Council).

Urban Greening

The ecology of cities includes some unique challenges and opportunities for restoration and conservation efforts. Challenges can result from the level of human disturbance to the environment, while opportunities may exist due to the accessibility of sites to active management and proximity to distinctive source populations of ecologically managed reserves.

Ecological restoration projects aim to restore a level of functionality to an ecosystem that has been degraded, damaged, or destroyed (Clewell, Aronson, & Winterhalder, 2004). However, the FPP's stated objectives correspond only very loosely to these recommendations. Given the scale and age of most of the FPP sites, they cannot be expected to function "normally" or as a "wild" ecosystem or match a particular reference site. Therefore, the scheme cannot be considered a formal restoration project, or evaluated as such. Instead the FPP better fits the definition of a "Community Greening" project, which integrates natural, human, social, financial, and physical capital (Tidball & Krasny, 2007).

Urban greening schemes such as the FPP are not uncommon internationally; some focus on obvious targets for restoration such as planting banks along the Mississippi river in urban areas (Lane & Raab, 2002), while others focus on smaller more diverse urban sites and may emphasise a community element, to which some of their success has been attributed. For example Summit and Sommer (1998) observed a San Francisco community planting scheme and recommended that programmes be easy to engage with, highlight personal benefits to participation and involve community groups. The opposite is also true in that urban planting schemes should provide community benefits in themselves, and increased satisfaction with their environment (Sommer, Learey, Summit, & Tirrell, 1994). Golf courses have provided some comparatively large areas where biodiversity has been encouraged through active management (e.g. Burgin & Wotherspoon, 2008; Terman, 1997). Private gardens have also been focussed on as contributing to landscape level biodiversity (Gaston, Smith, Thompson, & Warren, 2005; Thompson et al., 2009), and programmes providing plants to residents have been shown to increase resident knowledge of native species and weeds (Marynissen & Campbell, 2006).

It has been recognised that ecological gains such as increased biodiversity and environmental benefits such as improved air quality and reduced erosion can be achieved if certain challenges that exist in urban settings can be overcome. The challenges to establishing ecologically functioning communities in urban environments are due to the relatively high level of human disturbance. Such disturbance results in fragmented habitats and small patch size, modifies the physical environments and introduces biotic pressures from exotic species. These environmental conditions often require special management to overcome them. For example McPherson et al. (1997) highlighted benefits from trees in Chicago such as improved air quality and energy conservation, but emphasize the need to match specific plants to site specific conditions to ensure gains are realised. Furthermore, ecological management techniques suitable for managing at the landscape scale of wilderness sites may need to be adapted to suit local urban sites that do not coincide with traditional ecological temporal or spatial scales (Borgström, Elmqvist, Angelstam, & Alfsen-Norodom, 2006). Therefore, while there are many small sites in New Zealand urban and rural landscapes that could, with intensive management be restored to a state similar to their "natural" or pre-human state, achieving this will not always be practicable.

The smaller scale of urban sites can allow for intensity of management that would not be feasible on larger sites. Successful examples such as the Karori Wildlife Sanctuary (Zealandia) in Wellington used existing geographical features for the construction of their predator proof fence, and subsequently have achieved success in establishing some ecosystem functioning in terms of vegetation dynamics (Blick, Bartholomew, Burrell, & Burns, 2008). The effort and expense in this type of activity is however, very large.

Smaller sites such as those found in normal many urban and suburban landscapes are less likely to have the resources, or scale to achieve any level of ecosystem functioning. While planting of these sites using an ecological restoration approach may provide aesthetic and cultural benefits as well as a level of plant structure, actual gains for biodiversity and conservation are less obvious. In particular, small scale restoration sites may have limited potential for supporting viable resident populations of flora and fauna, and this is especially likely to be true of small urban sites that can be fragmented by intense human activity. The difficulty in maintaining populations in small sites is recognised in the theory of island biogeography (MacArthur & Wilson, 1967), where the stability of a population in any location is influenced by the size of the habitat or site, and the distance to source populations. For example, it has been estimated that the minimum forest area needed to support a diverse range of New Zealand forest birds is in the tens of km^2 (East & Williams, 1984). Although island biogeography theory must be used cautiously and account for autecological idiosyncrasies of individual species (Simberloff & Abele, 1976), from a conservation perspective, in principle it may be assumed that increasing tendency towards local extinction due to small populations associated with small sites can be offset to an extent by recruitment from source populations.

In urban areas, small scale restoration sites may also suffer from reduced immigration required to maintain high levels of native biodiversity (Bastin & Thomas, 1999). The nature of these effects in the urban environment depends upon the spatial scale of the site and degree of human alterations, changing the mix of species for example from favouring native to favouring synanthropic introduced species (Marzluff, 2005) or specific trophic groups (Gibb, 2002).

Area and isolation effects are, however, influenced or mitigated to an extent by the degree of alteration to the matrix (Kupfer et al., 2006), that is, the patchwork of habitat between the major population habitats in a landscape. The benefit of these smaller sites may therefore be in the manner in which they add to the ecological functioning of the landscape on a larger scale. For example, restoring a patchwork of sites may increase matrix permeability - the degree to which an organism is able to disperse through the landscape in general between habitat patches - and such sites may also act as “stepping stones” along corridors between larger habitat areas (Kupfer, Malanson, & Franklin, 2006). Beyond being a simple corridor between habitats, the sites may also provide additional resources including food, perching trees for birds, and seed sources that can be visited on a short term basis. Regardless of the scale of the site, it will provide sufficient viable habitat for at least some resident smaller plants and organisms.

The establishment of plants is the key goal in starting this matrix permeability improvement (Meurk & Swaffield, 2000). Generally, in human-disturbed habitats, plant species composition is likely to be influenced by environmental conditions (e.g. soil, light, competition); the pool of species able to colonize from neighbouring sites; the barriers preventing pollen/seed flow among sites; and the frequency and intensity of human-induced disturbance (Muratet, Machon, Jiguet, Moret, & Porcher, 2007). Once the original plants are established it is hoped that they start a succession process either by acting as nursery plants that facilitate the establishment of target species (Francisco M. Padilla & Pugnaire, 2006), or if they are the target species themselves and act as a source of propagules (Clewell et al., 2004).

The establishment of New Zealand native plants in urban areas has been shown to be affected by several factors including, for example, density of planting and distance from other native vegetation (Sullivan, Williams, Timmins, & Smale, 2009). These factors, if shown to be important for the establishment of plants, can inform management decisions and result in greater planting success and greater cost efficiencies. Back-yard habitat has been examined within the context of wider town landscape connectivity, and has been found to provide sufficient linkages to maintain viable metapopulations (Rudd, Vala, & Schaefer, 2002).

Urban planting occurs in environments that are often very different from wild ecosystems. The influence of several factors are key to successful plantings in the town environment, which are examined in this study in the categories of biotic, abiotic, and human factors.

Biotic effects on plantings

A key determinant of planting success is the suitability of specific plants to the environment they are transplanted to. There are several biotic factors relevant to this study that are recognised as influencing this success. The strategies that plants have evolved to enable them to survive in a particular environment rely on their being able to access sufficient resources, and therefore differing resource availability provides conditions suiting different plant groups (Vallet, Daniel, Beaujouan, Rozé, & Pavoine, 2010). These strategies are often categorised according to the Competitor, Stress Tolerator or Ruderal – the C-S-R Triangle theory (Grime, 1977). In this theory plants have evolved traits to exploit a certain environment characterised by a combination of disturbance regime, and biotic or abiotic conditions. Each species will likely be most successful on very particular combination. In general, *competitors* are most successful in environments with low intensity stress and disturbance, by out competing other plants for resources. Under stresses such as climatic extremes, or poor nutrients, *stress tolerators* are likely the only plants to thrive. Likewise, in environments of high intensity disturbance, *ruderal* species will be the only ones to thrive often due to their characteristic short life-cycles and large, persistent seed production. Urban sites, however, may not be typical sites for ruderal plants as human caused disturbance regimes would tend to vary in intensity and often be present for short short periods during development. The typical disturbance-dominated environment would need ongoing disturbance to maintain a ruderal dominated flora, for example flood-prone or fire-prone areas. The types of disturbance are also unlikely to be consistent in their effects on the landscape, for example human caused disturbance that may only be temporary states that either result in increased heterogeneity by fragmenting the landscape, or decreased local heterogeneity by, for example, removing the topsoil layer and creating homogenous pioneer communities (Rebele, 1994). Predicting the response of plants to these variable conditions is, therefore, not easy to generalise.

Human impacts greatly vary, particularly if the wider city and suburban region is considered. Hill *et al.*(2002) reviewed several scales of human impacts on urban environments such as Hemeroby – a bio-indicator scale that compares urban floras across varying scales of human activity, and while it was not attempted in this study to quantify the impact of human disturbances on the sites studied, it is important to bear in mind the biotic effects that these scales represent. Increasing human disturbance is associated with increasing annuality of plants (the proportion of plants that are annuals) and xenicity (exotic richness). Hill *et al.*(2002) revealed that urban plants are often those that favour irregularly disturbed sites, rather than the classic ruderal species that are specialists of highly disturbed environments. Urban centres do not have defined boundaries and it is well recognised that the rural to urban gradient produces species compositional changes as human influences become more intense, favouring exotic species (e.g. McKinney, 2002). Increasing proportion of exotics has also been found to be negatively correlated with distance to dwellings on urban wasteland sites (Muratet et al., 2007). The wide variation in type and intensity of human impacts can be expected to produce a wide variety of ecological responses.

Species that thrive on urban sites are often considered to be invasive, however, many of the factors that make a species invasive are to do with their efficient dispersal and fast life-histories. These factors are less relevant to plants that are planted by humans, having already overcome the biological requirement to reproduce and establish from propagules. Instead, the plants could be viewed as principally in competition for resources. Therefore invasiveness must be considered as a combination of species trait and habitat, for example, Thompson *et al*. (1995) found that in cool damp environments, competitiveness is the most important factor in invasiveness.

It appears from the literature that there are many theories regarding the response of plants in urban environments, however it is far from easy to generalise what that response may be across species or sites. Urban sites are hugely variable and potentially unpredictable environments.

Indigineity

Many restoration programmes recommend the use of native plants (e.g. Middleton, Bever, & Schultz, 2010; Miyawaki, 1999). This is expected to provide a variety of benefits including supporting local fauna, being evolutionarily adapted to local conditions, and resistant to local pests and diseases (Dorner, 2002), avoiding introduction of potentially weedy species, and, particularly if using eco-sourced plants, supporting genetic diversity of local flora. Net primary productivity (NPP) is often higher in native dominated sites (Eviner & Chapin III, 2003).

General species traits

Eviner & Chapin III (2003) list the following as potential traits that affect NPP: the growth rate, allocation, phenology, nutrient use efficiency, resource requirements, traits that influence access to resource pools, and traits that influence conditions that limit growth. Species competitiveness can be affected by growth form, for example, height and biomass are key traits that increase a species' competiveness and show greater ability to compete at less frequently disturbed sites (Dietz, Steinlein, & Ullmann, 1998).

Invasiveness

Exotics are often assumed to be more invasive than natives in any given environment. Hierro, Maron & Callaway (2005) reviewed the theories explaining invasiveness and categorised a plants potential advantage in a novel environment as the resulting from one or more of the following conditions: 1) plants in novel environments are free from their natural enemies; 2) plants in novel environments evolve rapidly selecting for invasive traits (evolution of invasiveness); 3) they have weapons such as growth inhibiting chemicals (allelochemicals) that are new and more effective in a novel plant community (novel weapons); 4) they are better adapted to the intensity of anthropogenic disturbance regimes; 5) they present more intense propagule pressure, 6) they fill an empty niche in the community, and, 7) they exploit a relatively poor diversity in the novel community (species richness). In the context of a planting programme, the degree to which these factors are significant will depend on the specific characteristics of the programme, including site type, species planted and human management actions. Later successional processes are likely to be affected more by these processes than the initial plantings, which have overcome some barriers to establishment such as dispersal and germination. The degree to which weediness of exotics affects planting success is one of the questions this study hopes to contribute.

Abiotic factors affecting plantings

If plants have evolved traits to suit certain environments, we should be able to make some assumptions regarding which species will thrive in an urban-suburban environment if we can characterise that environment; however, this is not straightforward. On one hand, the environmental factors that influenced the evolution of the species traits such as invasiveness or ruderality, are often considered to be characteristic of human influenced landscapes, for example urban - suburban environments are often characterised by disturbance events and increased nutrients (Vallet et al., 2010). This means they are often characterised in terms of the plants that favour them, for example, Ruderal from C-S-R triangle theory (see above). Additionally, the increased nutrients resulting from moderate levels of urban development may mean that on a rural to urban gradient Net Primary Productivity (NPP) is highest in suburban areas (Shochat, Warren, Faeth, McIntyre, & Hope, 2006). On the other hand, as discussed above the boundaries of urban zones are not clearly defined and a city contains a gradient of environmental types.

Other effects of urban environments are less likely to favour a particular species trait, but are nonetheless likely to have an effect on growth. An effect that is often observed in studies of the urban environment is the "heat island" (e.g. McDonnell et al., 1997), where temperatures in urban areas are higher than the surrounding countryside due to the heat retaining materials used in urban areas, and the excess heat generation by human activity. This is generally considered to have favourable impacts on flora in temperate zones (Shochat et al., 2006).

Propagule pressure (the quantity and frequency of propagules released into an environment) of exotics is often an influence in urban environments due to surrounding land use, which can result in garden escapees (Borgmann & Rodewald, 2005; Vidra & Shear, 2008). This may contribute to the observation of species richness often being high in urban habitat patches (Niemelä, 1999) compared with rural areas. Modified urban soils may also affect planting by shifting soil quality, competitive regimes, seedling establishment, and disturbance patterns (Pavao-Zuckerman, 2008).

Regional Characteristics of the Present Study Environment

The areas included in the study are for the most part relatively small suburban Wellington urban locales (Figure 1), many of which are bounded by the Wellington Town Belt – a green-belt of relative wilderness including native and exotic trees, and a variety of successional phases from regenerating scrub through to established timber species and native lowland forest. The Wellington greenbelt was created in 1841, and is to be managed in a way that protects and enhances natural, landscape, cultural and historic values (Wellington City Council, 1998). Along with parks and nature reserves, the greenbelt is the principle area of biodiversity in the city, providing habitat for a range of flora and fauna. The greenbelt could thus benefit from smaller FPP planting projects by providing improved matrix.

The original soils in the area consist primarily of Porirua silt loams which have poor physical properties but, apart from a lack of phosphorus, have a good nutrient content (Wellington City Council, 2004b). However, on most urban sites the soils have been modified significantly, either as a direct result of landscape modification to enable construction of roads and buildings, or through incidental damage due to erosion, pollution and past or present land use.

Wellington receives slightly higher average daily temperature and slightly less rainfall than the national average (Maclean, 2009) at 1270mm per annum. The region tends to get low rainfall in the summer and high rainfall in the winter. High rainfall at any time of the year can cause flooding along streams and rivers (Greater Wellington Regional Council, 2012) and the strong winds caused by funnelling between the Tararuas and the South Island result in a climate favouring a hardy and salt tolerant plants in all but the most sheltered valleys (Gabites, 1993).

Human factors affecting plantings

In any planting or restoration programme, the plants and physical site characteristics are only parts of the mix of factors contributing to the planting conditions. The other major factor influencing the plants and sites is the human input, before, during and after planting. These factors range from decisions about when, what and where to plant; what alterations to the site need to be made before planting; what methods are used to plant the seedlings; and, what regime of care is needed once the plants are in the ground. All these decisions then need to be implemented, possibly on an ongoing basis. In a voluntary programme such as the FPP, the human input is relatively unstructured compared to an organised restoration programme. Plants are selected by experienced nursery staff, however, once plants are in the possession of the participant, it is up to individuals to use suitable methods for planting and efforts in post-planting care. This is expected to lead to a wide variety of planting techniques being used in the programme and presents an opportunity to investigate whether human influences can be shown to affect planting success over and above biotic and abiotic conditions. Human influences of on urban planting schemes are reasonably well studied. Studies focus on both the direct effect of human activities in care of plants, and participation in planting programmes generally.

Human Actions

Human influences directly affect plant survival in two ways: through intentional care of plants and through accidental or deliberate damage to plants.

Socioeconomic factors have been shown to be associated with damage and care of establishing plantings in urban areas. Lower socioeconomic areas in some cities have been shown experience greater tree damage through vandalism (Nowak, McBride, & Beatty, 1990). Other damage resulting from human action is usually related to vehicle use or foot traffic (Nowak, Kuroda, & Crane, 2004). However, in the circumstances of the road reserves that feature in this study, mowing or herbicide spraying by residents or local government may cause unintentional damage.

Participation

Participation in urban planting has been shown to be influenced by a variety of socioeconomic factors such as age (Straka, Marsinko, & Childers, 2005), income (Greene, Millward, & Ceh, 2011), or employment status (Zhang, Hussain, West, & Letson, 2007). Explanation for these influences range from concern for the environment and protection of trees (Jones, Davis, & Bradford, 2012), to personal benefit through increased satisfaction with trees planted (Sommer et al., 1994) or actual financial gain through increase in real-estate value (Wachter & Wong, 2008). Another relationship observed between home-ownership and planting is that tree planting tends to be greatest early in a resident's tenure in a home (Summit & McPherson, 1998).

Participation is also affected by physical environmental variables such as percent of forested land (Wall, Straka, & Miller, 2006), and age of housing stock

(Greene et al., 2011), although it may be argued that these are also indirect socioeconomic effect, where established suburbs, closer to parks have more expensive housing and therefore have associated higher socioeconomic factors. Summit & Sommer (1998) note that while many environmental programme guidelines focus on environmental protection as motivating participation, a stronger motivating force may be personal advantage.

Biotic, abiotic and human influences on urban plants are all well studied individually, however, it is less common to be able to study them in combination. In this study, data on these three factors are combined in an analysis of planting success.

AIMS

Biotic and Abiotic Influences on Planting Success

The aim of this study was firstly to find if any factors have influenced the successful establishment of plants provided as part of the FPP. Several site and species characteristics were measured and modelled to examine their effect on the survival of plants at selected 76 randomly selected sites. This will give an indication of whether site factors or species factors are more important in successful establishment, and will inform management decisions around types of plants provided for particular sites.

Human Influences on Planting Success

The second aim was to examine the effect that the actions of the FPP participants had on planting success. Household postal-questionnaire surveys were used to collect data on factors that directly and indirectly might affect plantings.

Influences on Participation

The third aim was to examine the factors that influenced participation in planting public spaces. A participating population and a non-participating population were surveyed and compared to find which factors may account for the decision to participate.

Thesis Structure

This thesis includes analyses based around the three aims above. The results and discussion are presented under these headings.

METHODS

Two distinct data sets were collected: 1) data were collected about sites and the species recorded in WCC records as being planted at these sites by conducting site surveys, and database and literature searches; 2) a postal questionnaire was conducted to collect data on human actions and socioeconomic factors relating to households at addresses of FPP participants. These two data-sets are referred to in this thesis as *Site Survey* data, and *Questionnaire* data respectively. The data-sets overlap to the extent that 25 of the Questionnaire responses were from the same addresses that were responsible for planting of sites that were site-surveyed. This provides the opportunity to examine the second aim above – Human influences on planting success. Data are therefore discussed in the context of being from either the Questionnaire or Site Survey data-sets. Analyses of the two data-sets are discussed in the context of the three aims: Biotic/Abiotic Influences on planting Success; Human Influences on Planting Success; and, Influences on Participation.

Data Collection

Site Surveys

The research is based both on data collected as part of the management of the planting programme, and on data collected in the course of this research. The approach is to examine the effect of certain treatments (site or species factors) on a response variable (plant survival). As this is a post hoc analysis of a programme that was not originally set up as an experiment, treatments were not strictly randomly allocated, with allocation being influenced by availability of plants at time of distribution, changes in planting policy and site information available to the nursery (N. OliverSmith, Personal communication, 4 February 2011), however, the 76 sites surveyed were chosen as a random sample from three of the 21 years the programme had run (1990 – 2011). The years 1995, 2000, and 2005 were chosen, being five, ten and 15 years after the start of the programme, to ensure that conditions in a particular year did not unduly bias results.

Data were collated from existing records which were collected by the City Council, and new data were compiled by field survey and derived from Geographic Information System (GIS) calculations on existing data layers. Records of plantings from three years were transcribed from paper to electronic database. The information consists of: species planted; street address of participant in planting, including in some cases a brief amount of information on location of planting site; year of planting. The species provided by the WCC for planting by community groups in the years surveyed are listed in Appendix 3. A random sample of 76 addresses was taken from the three years and these addresses were Geo-referenced and mapped in a GIS, then maps were produced as guides to site visits.

The motivation for assessing this urban restoration programme in particular comes from the fact that records have been kept describing location of sites and plants provided. While the existing data presented a rare opportunity to compare field and questionnaire surveys with historic planting data in an urban setting, it also, presented some challenges in analysing the historic data. This was principally due to the low level of detail in the records kept, and the difficulty in identifying the actual site and individual plants planted. Council records consisted primarily of the address and name of the person receiving the plants (the participant) and the species provided. Occasionally, a brief description of the intended site was included, which was used to find the site.

Apart from difficulties in pinpointing specific sites, the programme presented some methodological challenges. As plants were not individually recorded or marked, it could not be established with certainty during site surveys that those individuals being recorded were the same plants initially supplied by the programme. The uncertainty was mitigated by analysing success on the basis of presence or absence of a planted species at the site. The rationale for this was that if one or more individuals of a species recorded as having been planted at a site had established there, it is likely that that species could be successful it that combination of environment and planting method used at the site. This dependent variable is named in analyses *Species Success*, and refers to the success of that particular species at a particular site.

Site visits were conducted during the period between December 2010 and July 2011. Each of the randomly selected addresses was visited and its suitability for inclusion in field sampling was evaluated based on whether the actual planting site could be identified. This was based on information provided in council records, or if lacking, identification of the plants provided by council. If a planting site could not be identified at an address, a new site was chosen, being the next closest from the same year.

Those planting sites that could be identified were surveyed, collecting general site characteristics, and a count of existing examples of plants provided as part of the FPP. The type of data collected during site visits is summarised in Table 1. Additional environmental and geographic data were compiled from various sources, and are outlined in Table 2.

Table 1

Factors Measured by Site Surveys

Factor	Category	Levels / measures	Method
Species survival	Response	Number of plants	Count of each species of plant provided by programme at each site; converted to presence/absence for analysis
Site type	Site: physical	Garden, mown grass, untended road reserve, bush	Visual assessment of site primary function or management regime
Aspect	Site: physical	Degrees from north	Measured using compass, then converted to degrees from north
Slope	Site: physical	Degrees	Measured using clinometer
Land-use bordering	Site: social	Reserve, road, residential, footpath, garden, stream	Visual assessment of types of land-use bordering the site on up to four sides
Canopy closure	Site: physical	Percent	Measured using spherical densitometer, averaged over 4 readings from cardinal points. Sites smaller than 10m across the longest dimension had one point at centre of site measured. Larger sites had measurements every 10 m along a transect running the longest path across the site.

Table 2

Additional Site and Species Factors Compiled From Existing Data

Factor	Category	Levels / Measures	Method
Area / perimeter	Site: physical	m2 / m	Site boundaries were plotted as a vector layer over ortho-photos, then area and perimeter of sites derived in GIS
Elevation	Site: physical	m	Point sample from centre point of each site from the 5m digital elevation model (DEM) provided by Wellington City Council
Topographic exposure	Site: physical	Topex score	The Topex score was calculated at 5m resolution and point sampled for the centre point of each site from the DEM. See Appendix 2 for procedure.
NZ native	Species	Native, exotic	Information gathered from existing databases
Drought tolerant	Species	Tolerant/not tolerant	Existing databases/various sources*
Shade tolerant	Species	Tolerant/not	Existing databases/various sources*
Exposure tolerant	Species	Tolerant/not	Existing databases/various sources*
Early successional	Species	Early/late	Existing databases/various sources*
Form	Species	Tree/shrub/clump	Existing databases/various sources*

Note: * (Greater Wellington Regional Council, 2010; Landcare Research, n.d., 2011)

Questionnaire

Questionnaire design

Data were collected from residents in the areas planted using a postal questionnaire. The questionnaire was designed to improve our understanding of the motivations to participate in the FPP on public land, the human factors influencing the success of plantings, and gauge people's opinions of what contributed to successful plantings.

A target of approximately 200 respondents was sought. Based on a predicted response rate of approximately 25%-30% for self-administered postal surveys (Fox, Crask, & Kim, 1988), 750 questionnaires were sent out. Questionnaires were sent to all addresses of participants in the Free Plants Programme from the same three years included in site surveys. Following data entry of legible, legitimate addresses, the address points were mapped and it was visually determined that a reasonably even spread of points across the survey area were present. The numbers of addresses from the three years 1995, 2000 and 2005 were 373, 164 and 213 respectively, giving a total of 750 households invited to participate in the survey. The numbers of respondents in each of the three years were 115, 54 and 59 respectively, giving a total of 229 and a response rate of 30.5%.

The questionnaires were addressed to the "householder" with the expectation that a proportion of the participants of the FPP would no longer be resident at the address, but that the new resident would be invited to respond to the survey. In this way the questionnaire gathered responses from the two separate populations: those who participated in the FPP, and those who did not. It was expected that, because the latter population were resident at prior addresses of previous FPP participants, this would control to an extent for some of the social and neighbourhood factors expected to influence participation in the public planting activity. For instance, Greene, Millward and Ceh (2011) found that participation in urban forestry programmes varied regionally but was based on factors such as income and house age. By surveying addresses whose past residents were known to have participated in the FPP, these factors could be assumed to be have less influence, and therefore the other influences on participation would be more apparent.

The survey was designed to meet the following requirements suggested in literature as facilitating good response rates and analysis of the data: attract the interest of participants, be sufficiently brief to avoid response fatigue, consist of easily understandable plain English questions, provide easily interpretable unambiguous answers, elicit the information required for analysis, and allow for ease of comparison of this data with other data such as census data (Dillman, 2007; Fox et al., 1988). For the purpose of comparison to NZ Census data the questionnaire distribution area was confined to the Statistics New Zealand boundary for Wellington City. Salience to the target audience, in particular was considered important in the response rate of this population due to the nature of the population selection (Dillman & Carley-Baxter, 2000), which was targeted to specific group of Wellington City residents. It was assumed that the salience of the questionnaire to the targeted households would be high because members of these households had planted a public area and would therefore consider completing a questionnaire in order to convey their opinion of urban planting generally and the FPP programme. Other motivating factors may have been the assumed interest in biodiversity and the environment of the target population, and a willingness to engage in "public good" activities.

The survey was kept short, and it was estimated the 3 page survey should take most respondents no longer than ten minutes to complete. The survey was piloted on a trial population of twenty participants who provided feedback as to the navigability and intelligibility of questions and instructions. This feedback was considered in the final draft of the survey.

Survey questions were a mix of open-ended and closed-ended questions according to the type of data sought and the purpose of the question. The open-ended format was used for more general questions designed to motivate the respondent's interest and gauge general opinions on urban planting or environmental engagement. Some used a partially closed, unordered format (e.g. Question 8.3). This format was used to ensure the correct type of response was elicited in terms of type of activity, while avoiding the problem of respondents being unable to answer because the appropriate category was not provided. Closed-ended questions were used where more specific answers were sought regarding planting techniques, environmental awareness and socioeconomic data. These included a Likert scale (e.g. Question 1), or an unordered list (e.g. Question 16). Open-ended questions and the open part of partially closed questions require further extraction of themes before categorisation and analysis can be performed, however they do not restrict the possible answer.

The questionnaire collected data from respondents in the following four areas: descriptive site data, such as the distance from the householder's address and the level of planting already present in the neighbourhood; planting methods such as watering or weeding regime; social context as in who was involved in the planting; and socio-economic data of the household involved. The Questionnaire is provided in Appendix One.

Mail-out organisation

The information held by WCC on plantings was used to identify households for questionnaire delivery. All addresses recorded as having plants supplied by the FPP in the same years selected for the site surveys were entered into a database. These records are held on paper, which is generally a typed or handwritten summary of addresses with some names included; this meant that an element of human error was introduced to the data before this research began. In some cases addresses did not exist, and some addresses were excluded on the basis of uncertainty as to their legitimacy. It is also possible that the record of species provided were not accurate, and it is feasible that different plants than the ones recorded were delivered. Some addresses were duplicated between two or more years of the FPP, indicating that plants had been supplied on more than one occasion to these addresses, most likely to the same household, or to a subsequent resident if tenure had changed. Duplicate addresses and those outside the site survey area, as defined by the Statistics New Zealand data for Wellington City were removed. The addresses were then geocoded using the Google geocoding API through the GPSVisualizer.com Batch Geocoder (Schneider, 2012). Any addresses that were not able to be confirmed were removed, resulting in 750 addresses remaining to include in the survey.

Ethics approval was gained from the Victoria University Human Ethics Committee under the condition that all data were confidential. Due to the spatial nature of the data and their relevance to physical addresses, the data were collected with reference to street addresses. All data were, however, entered and stored with reference to a numeric code, with the corresponding street addresses stored separately. This meant the collection was effectively anonymous as no names were required to be included on the questionnaires.

The survey was conducted in three waves of postal mail-outs. Firstly, an initial letter was sent inviting the householder to participate in the research and providing notification that a questionnaire would be received in the mail approximately one week later. Secondly, the questionnaire was mailed with an information letter, instructions and a freepost pre-addressed return envelope. The third mail-out followed one week after the questionnaire with a reminder letter to all addresses thanking the householders for their interest in the research and providing a reminder that the surveys were requested to be completed and returned within 6 weeks. Five householders responded to the email address provided on this letter to ask for mislaid surveys to be replaced. These replacements were sent to participants electronically and returned either using the freepost number on the participants own envelope, or returned electronically.

Data entry

All returned questionnaires were entered into a spreadsheet by the researcher. The spreadsheet was formatted to allow only valid answers to categorical factor questions via drop-down lists, and text strings for free text answers. All answers were recorded against a numeric code, separate from the address list, as per the ethics approval description.

Statistical Methods

For the three analyses Model selection is used to explore the data. This approach has the following benefits over traditional null hypothesis testing: the analysis is not restricted to testing a single model against an arbitrary probability threshold; relative support for competing hypotheses can be quantified; and, where appropriate, model averaging can be used to draw inference from more than one model (Johnson & Omland, 2004).

An information-theoretic approach was used for model comparison as outlined by Burnham and Anderson (2002). Models were compared by ranking with Akaike Information Criterion (AIC) scores. Second order AICs were calculated to correct for small sample size (AICc). This method compares models based on the degree to which each minimises information loss or the Kublick-Liebler distance (Wagenmakers & Farrell, 2004), therefore it identifies the best model given the data and set of models, but does not indicate absolute fit of each model. Models within two AIC of each other (Δ AIC < 2) are considered to be indistinguishable, while a model Δ AIC > 12 of the top model is considered to be implausible (Burnham & Anderson, 2002). AIC weights (w_i) were calculated and used to infer the probability of a model being the best in the candidate set. A 95% confidence set of models was created by summing w_i of the top models so that $\sum w_i > 95$.

Logistic regression models were developed to assess the relationship between various biotic and site traits, or human actions and the presence of individuals of the planted species; and social factors that influenced participation in public planting. Survival of plants was measured as the presence during the site survey of a particular species that was recorded as having been planted at that site. ‘Present’ indicates at least one member of a planted species was found during the survey, ‘absent’ indicates no members of the species were found at the site. Presence/absence was coded 1/0 and used as the dichotomous response variable in the regressions. Participation in public planting is also a dichotomous variable, where 1 indicates the questionnaire respondent was involved in planting outside their own property, or 0 indicating they had never planted outside their current property.

Biotic and abiotic Influences on Planting Success

As the surveys were conducted at a range of different sites, and were planted on three separate years, an analysis of the effect that these variables had on model performance was undertaken. A global model was created that included all factors from the models in Table 1 and Table 2 as fixed-effects and random-effects of the Site or Year were added either individually or together. The four models were compared using Akaike Information Criteria (Zuur, Ieno, Walker, Saveliev, & Smith, 2009), to find if model performance was improved by accounting for these effects (Table 10). Models were compared in the R statistical analysis programme using the *glm* function to calculate AIC corrected for small samples (AICc) for fixed-effects; and, the *lmer* function for mixed-effects logistic regressions. The initial comparison of models used a global model of fixed effects and the two potential random effects were varied (Table 10
Global *models for planting success based on biotic and abiotic traits*), therefore, these mixed-effects models were fitted in the lmer function using restricted maximum likelihood (REML) rather than maximum likelihood, which would be used if the fixed effects were varied (Pinheiro & Bates, 1998). Further analyses used fixed effects only models, which were compared in the *glm* function and fitted using the iteratively reweighted least squares method.

Several predictive models were proposed based on the combinations of species traits or site traits listed in Table 1 and Table 2, or a combination of both, where literature supported. The number of observations was 291; that is, 291 different species-site combinations. The number of groups or sites was 76. These models and their rationales are listed in Table 3: Species Traits, Table 4: Site Traits, and, Table 5: Interactions.

Table 3

Candidate Models for Species Traits Affecting Presence of Planted Species

Candidate models	Rationale	Reference*
Indigeneity	Native species are genetically more suited to local conditions than exotic spp. because they have evolved here	20, 22, 27
Drought tolerant	A principal cause of seedling mortality is lack of water. Certain species can withstand this better due to evolved traits	5, 15, 19, 21
Shade tolerant	Shade tolerance plants have a competitive advantage in low light conditions, either by faster growth, or by comparative fitness/survival over non-shade tolerant plants	12, 19, 29
Early successional	Early successional or pioneer species are better suited to poor soil and exposed sites	2, 8, 23, 25, 26
Growth form	Plant assembly rules theorise that plant community makeup is driven by resource use, with certain guilds of plants using particular resources at a site. One method to characterise a guild is in terms of growth form. Therefore it is hypothesised that species of similar growth form use similar resources (e.g. canopy layer/light, root depth/nutrients) and variability of resource availability between sites will result in competitive advantage to certain guilds of plants at each site.	7, 8, 13, 25

***Note*: see * references below**

Table 4

Candidate Models for Site Traits Affecting Presence of Planted Species

Candidate models	Rationale	Reference*
Garden Site-type (evidence of soil improvement/ cultivation/ weeding/)	Urban sites are often characterised by poor soils and invasive species. Sites that were essentially maintained as residential gardens had mitigated these factors to the highest degree.	1, 28
Land use bordering Site(reserve, road, residential, footpath, garden, stream)	Follow-up care of seedlings is important for survival. Sites bordering residential properties are likely to be more accessible and therefore may receive more frequent follow-up care.	1, 4, 14
Canopy closure	Canopy closure affects the levels of light reaching seedlings, and will favour shade tolerant species	23
Area and perimeter	The greater the area of a site, the smaller the edge effects on vegetation growth. Edge effects may be positive (e.g. availability of resources such as light) or negative (e.g. increased damage by wind, people etc)	23, 24
Topex (exposure) Aspect (degrees from north) + slope Elevation (temperature effect)	Wind damage, desiccation and temperature affect seedling survival, and are higher at exposed sites (e.g. north/south aspect, low topex) and more elevated sites.	6, 8, 14, 16, 18, 24

***Note*: see * references below**

Table 5

Candidate Models for Species Trait / Site Interactions Affecting Presence of Planted Species

Candidate models	Rationale	Reference*
Topex × Exposure tolerant	Effects of exposed site mitigated by exposure tolerance of plant.	8, 18
Topex × Growth form	Effects of exposed site mitigated by lower growth form.	18
Canopy cover × Shade tolerance	Shade tolerant plants do better under more canopy cover.	3, 19, 29
Site type garden × Late successional	Later successional species that are not drought or poor soil tolerant may survive better in better soil of garden sites.	2, 8, 23
Canopy cover × Early successional	Canopy cover may be a proxy measure for vegetation density, with early successional plants surviving better with less cover.	2, 8, 23

***Note*: see * references below**

1 Austin, 2002
2 Bellingham et al., 2011
3 Berkowitz, Canham, & Kelly, 1995
4 Borgmann & Rodewald, 2005
5 Brodribb & Hill, 1998
6 Chapman, 2000
7 Cole & Spildie, 2006
8 Criddle, Smith, & Hansen, 1997
9 Simberloff & Dayan, 1991
11 Davis & Meurk, 2001
12 Grubb, 1998
13 Lee, McGlone, & Wright, 2005
14 Lugo-Pérez & Sabat-Guérnica, 2010
15 Marañón et al., 2004
16 Melo-Abreu et al., 2010
18 Mikita & Klimánek, 2010
19 Niinemets & Valladares, 2006
20 O'Brien & Krauss, 2008
21 Padilla & Pugnaire, 2007
22 Pokorny et al., 2005
23 Pratt, 1999
24 Puigdefábregas *et al*., 1999
25 Raman, Mudappa, & Kapoor, 2009
26 Smaill, Ledgard, Langer, & Henley, 2011
27 Stewart, Ignatieva, Meurk, & Earl, 2004
28 Sullivan, Meurk, Whaley, & Simcock, 2009
29 Valladares & Niinemets, 2008

Human Influences on Planting Success

Questionnaire results were analysed in three ways: 1) a comment analysis and summary statistics from the questionnaire; 2) A fixed-effects logistic regression analysis to similar to the biotic and abiotic trait analysis for those planting sites that matched respondent addresses, with the addition of questionnaire explanatory variables; and 3) a fixed-effects logistic regression of questionnaire variables affecting respondents participation in public planting.

Models were again compared using an Information Theoretic approach by comparing the AIC Values. It is worth noting again that AIC penalises models for the number of parameters included (2k) therefore there is a trade-off between fit and complexity. Increasingly complex models must increase model performance accordingly.

Descriptive Statistics

Survey population

Socioeconomic data from the Questionnaire were compared to data from the 2006 New Zealand Census of Populations and Dwellings (Census) was accessed online from Statistics New Zealand describing the Wellington City population.

The median for age and income were calculated according to the formula in Figure 2 below (StatSoft Inc, 2012).

$$\text{Median} = L + I * \frac{N/2 - F}{f}$$

where:

- L = lower limit of the interval containing the median
- I = width of the interval containing the median
- N = total number of respondents
- F = cumulative frequency corresponding to the lower limit
- f = number of cases in the interval containing the median

Figure 2: Formula to calculate median from interval data

Themes from questionnaire

Responses from the two questions regarding respondents' opinions on factors that had contributed to, or hindered plant survival were analysed for key themes. This was performed using a word counter to quantify the most common words, which were then were grouped into common areas. A graphical analysis of the frequency of these themes was undertaken.

Human influences on plant survival

Those responses that matched addresses physically surveyed in the Site Surveys were included in a further regression analysis using only those sites, and including some additional explanatory variables from the Questionnaire. The response variable was the same binary variable as in the Site Survey analysis: presence/absence of species provided at each site.

Addresses of Questionnaire responses were matched to addresses of Site Surveys, which resulted in 25 matches. Five of these matching survey responses were found to be from respondents who did not indicate they had participated in planting outside of their property. This is likely to be because the tenure had changed and they were not yet resident at the address when the planting occurred. These addresses were removed from the analysis. The twenty Site Survey – Questionnaire response matches included multiple species records per site, which totalled 85 species survival data points.

The n = 85 dataset is small for the analysis, however the chance to investigate the human influences on plant survival in a post hoc analysis is rare, and it was considered this offset some concerns regarding sample size. Nevertheless, preliminary nature of the analysis is acknowledged and conclusions are drawn carefully. Furthermore, models are limited to the inclusion of single additional variables from the questionnaire, to avoid over-fitting models to small samples (Anderson, 2008).

The best performing model from the Site Survey analysis (*Indigineity*) was used as a base model. The following factors from the questionnaire were added to the base model, creating the candidate set for this analysis to assess if they improve model performance for this subset of sites: *Time resident at address, Distance from address to site, Horticultural experience, Group involvement* and post-planting care factors *Weeding, Watering, Fertilizer* and *Mulch*. The response variable was the same binary variable as for the original analysis: *Species Success*. The Candidate models for this analysis are listed in Table 6.

Table 6

Candidate models for surveyed sites matching questionnaire responses

Candidate model	Rationale	Reference
Indigineity	Best performing model from Survey analysis	
Indigineity + Time resident at address	Time at residence could affect survival due to the residents greatest energy in tree planting being early in residents tenure.	(Summit & McPherson, 1998)
Indigineity + Distance from address to site	Personal interest is important predictor of valuing trees- closer to house more value gained.	(Summit & Sommer, 1998)
Indigineity + Horticultural experience	Horticultural experience may provide skills and knowledge to provide correct conditions for plant survival. Tree knowledge and landscape experience shown to increase degree to which trees are valued.	(Jones et al., 2012)
Indigineity + Group involvement	Group involvement has benefits for restoration programmes and tree survival	(Austin, 2002; Curtis & De Lacy, 1994; Sklar & Ames, 1985)
Indigineity Watering Fertilizer Mulch	Weeding, Watering, Fertilizer application and Mulching have been shown to increase survival of transplanted tree seedlings in natural environments.	(Austin, 2002; Dostalek, Weber, Matula, & Frantik, 2007; Dostálek et al., 2005; Lai & Wong, 2005; Navarro Cerrillo, Fragueiro, Ceaceros, del Campo, & de Prado, 2005)

Table 7

Candidate models for questionnaire factors influenced involvement in planting outside their property

Candidate model	Rationale	Reference
Socioeconomic + Environmental Affinity	Increased awareness of an environmental programme willingness to engage in one	(Straka et al., 2005; Zhang et al., 2007)
Socioeconomic + Time at Address	Most planting effort early in tenure. Age of house shown to affect tree planting rate.	(Greene et al., 2011; Summit & McPherson, 1998)
Socioeconomic + Horticultural Experience	Participants motivated by enjoyment of and confidence in their ability to successfully plant trees	(Jones et al., 2012; Moskell, Allred, & Ferenz, 2010)
Socioeconomic + Number of Trees Outside	Personal advantage – environmental improvement, either biodiversity or aesthetic Increasing real-estate value	(Summit & Sommer, 1998; Wachter & Wong, 2008)

***Note*: Models include the best performing model from the analysis of socioeconomic factors influencing participation.**

Influences on Participation

All 229 questionnaire responses were included in an analysis of factors influencing whether the respondent was involved in planting outside their own property. The response variable for this analysis was also binary: whether or not the respondent indicated that they had planted any plants anywhere outside their own property. Logistic regressions were performed on this data using the candidate models.

Several studies have found socioeconomic variables affect participation in planting programmes. The following questionnaire variables have been found to influence participation in activities such as urban tree planting, volunteerism and support for tree or general environmental protection: age (Straka et al., 2005; Zhang et al., 2007); education (Wall et al., 2006); income (Greene et al., 2011; Wall et al., 2006; Zhang et al., 2007); and, employment status (Wall et al., 2006; Zhang et al., 2007). These variables were checked for collinearity, and un-correlated variables were used in models to establish as base model representing the overall effect of socio-economic factors on planting participation.

The best performing model was then used as a base model to investigate the influence of other questionnaire variables on participation in the FPP, while accounting for socioeconomic factors. These candidate models are listed in Table 7**.**

The variable Environmental Affinity is a compound variable calculated from responses to two questions on the questionnaire. The first: *what are the three greatest threats to New Zealand biodiversity?,* was scored 0, 1 or 2 according to how many of the 2 threats from the New Zealand Biodiversity Strategy categories – habitat loss and pest plants /animals. The second variable was respondents' reported frequency of visits to green-spaces, which were interval categories, converted to scores (see Appendix One - Wellington Road Reserve Planting Survey). The scores from both questions were averaged and converted to a variable between zero and one.

RESULTS

Descriptive Statistics

Site Surveys

The number of individual plants planted at each site varied between four and 152, while the number of different species at each site varied between one and 13. The number of individuals present at each site varied between one and 99.

The physical characteristics of site varied widely (Table 8). Topographic exposure (topex) scores show that most sites were moderately exposed, with 18 very exposed sites (topex score <20) (Figure 3). The median elevation was 122 meters above sea level, and the range of sites was 2.8 meters (sea level) to 253 meters (Figure 4). Most sites were moderately sloping (5 – 15 degrees). Two sites were quite steep (> 25 degrees) (Figure 5). The majority of sites were small ($< 50\ m^2$), though three were comparatively large $> 300\ m^2$ (Figure 6). These variables were checked for collinearity; no significant correlations were found (Table 9).

Species success was higher for native plants. The proportion of successful native plantings (measured as presence of at least one individual of a planted species) was 64 percent compared with 32 percent for exotic plantings (Figure 7).

Growth form showed slightly higher species success for trees at 60 percent, compared with 57 percent for clump forming species, and 53 percent for shrubs (Figure 8).

Table 8

Summary statistics for Area, Slope, Elevation and Topex for sites surveyed

	Area (m2)	Slope (degrees)	Elevation (m)	Topex score
Mean	60.3	11.5	113.7	38.5
Standard Error	11.8	0.8	6.6	2.9
Median	26.5	9.7	122.2	35.3
Standard Deviation	103.0	6.6	57.4	25.0
Skewness	3.9	1.0	-0.2	0.5
Range	591.9	31.0	250.5	112.0
Minimum	1.8	2.8	2.8	-12.4
Maximum	593.7	33.7	253.3	99.5

Table 9

Correlation matrix for site variables

	Elevation	Topex	Area	Slope
Elevation	—	-0.177	0.030	-0.168
Topex	0.132	—	0.116	-0.028
Area	0.800	0.327	—	0.115
Slope	0.153	0.812	0.327	—

Note**: correlation coefficients are shown in the top section of the matrix; p-values are shown in the bottom section.**

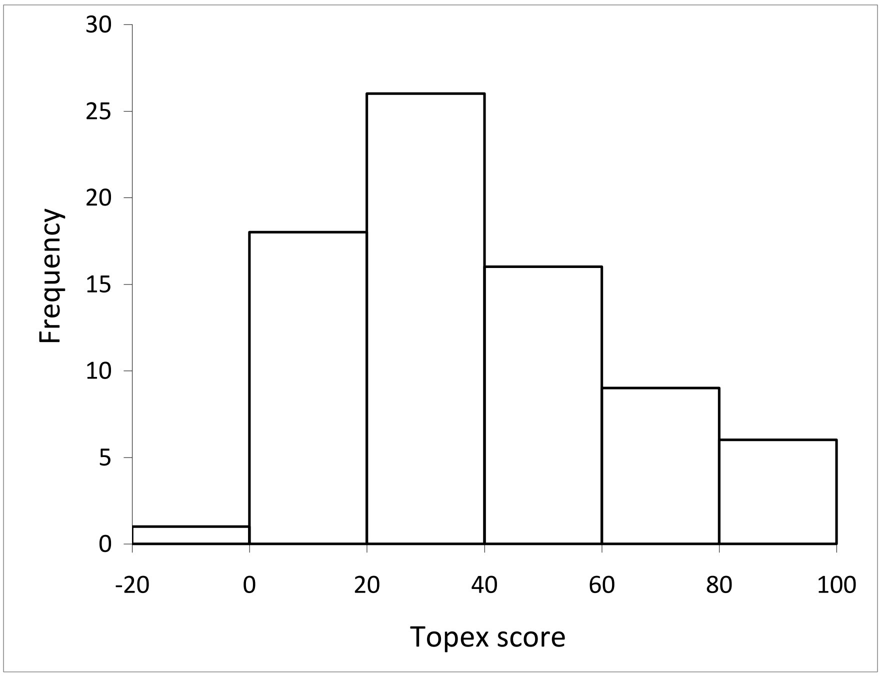


Figure 3: Frequency distribution for Topographic Exposure (Topex) score for surveyed sites. Higher Topex score indicates increasing protection from wind provided by the landscape topography in all directions.

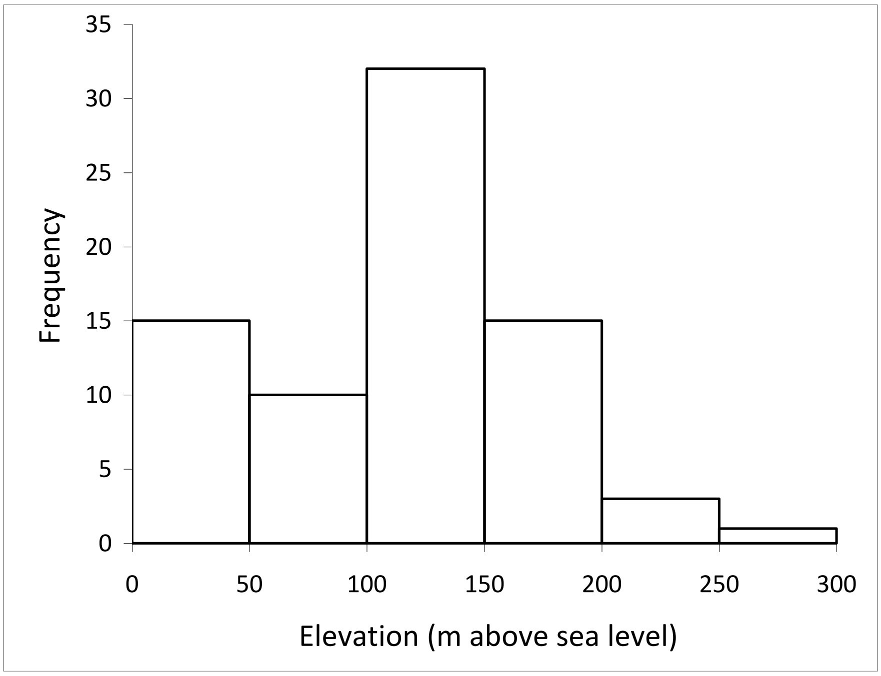


Figure 4: Frequency distribution for Elevation of surveyed sites.

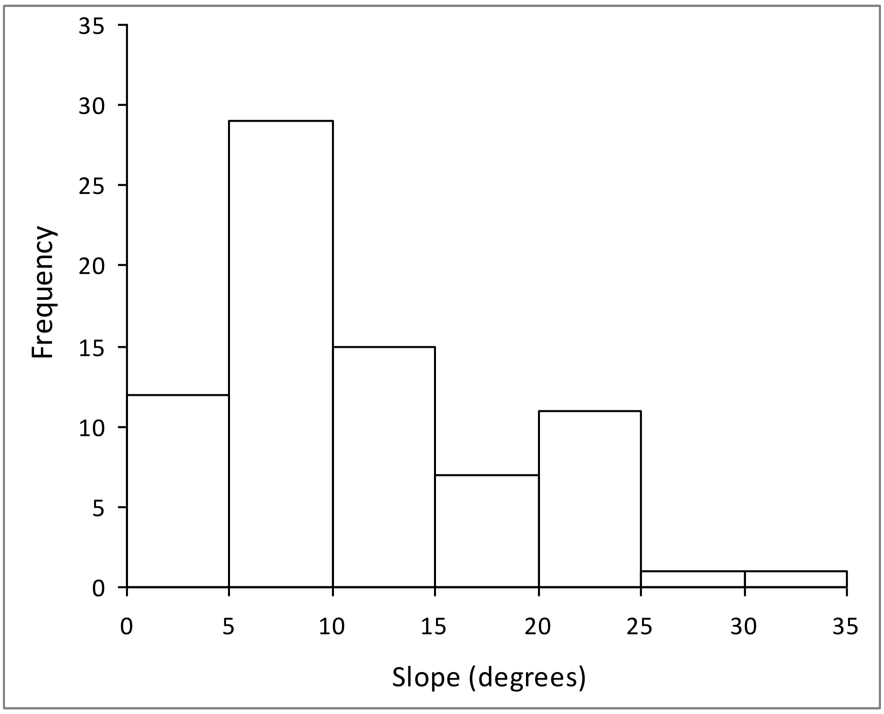


Figure 5: Frequency distribution for Slope of sites surveyed.

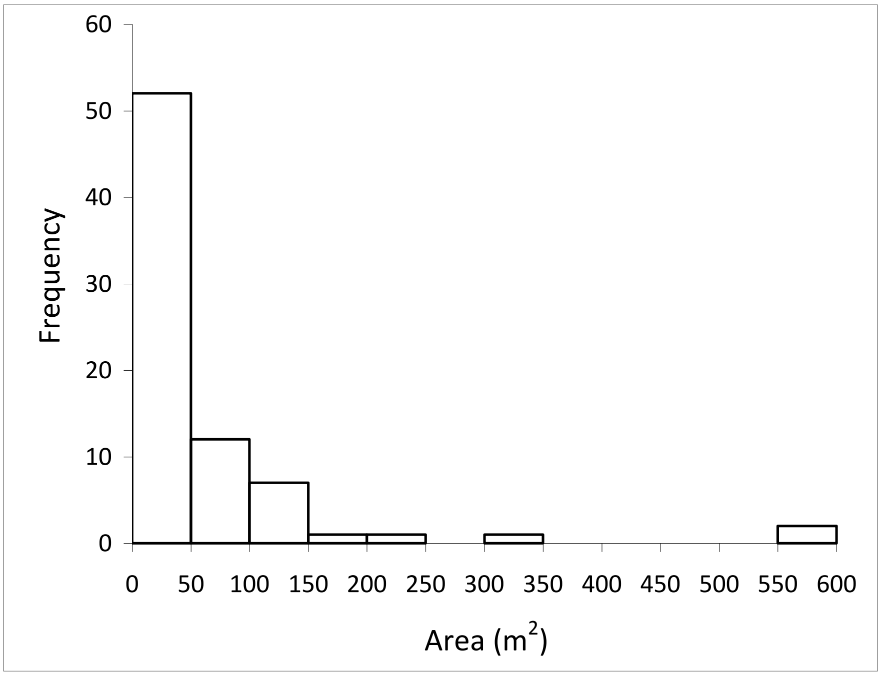


Figure 6: Frequency distribution for Area of sites surveyed.

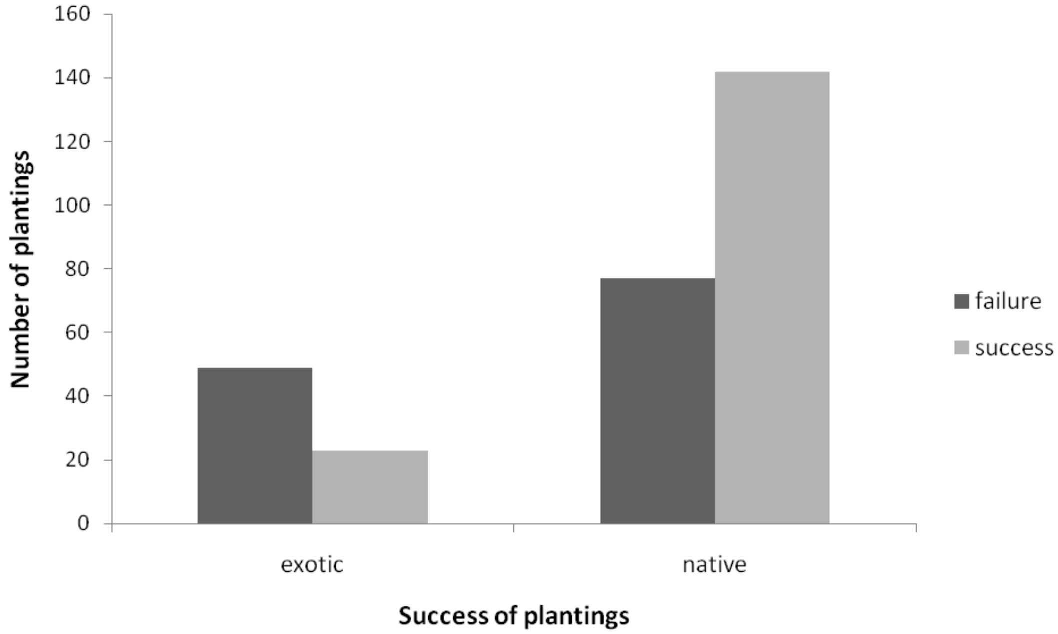


Figure 7: Frequency of species success for natives and exotics at all sites. Successful plantings are presence of each planted species at each site.

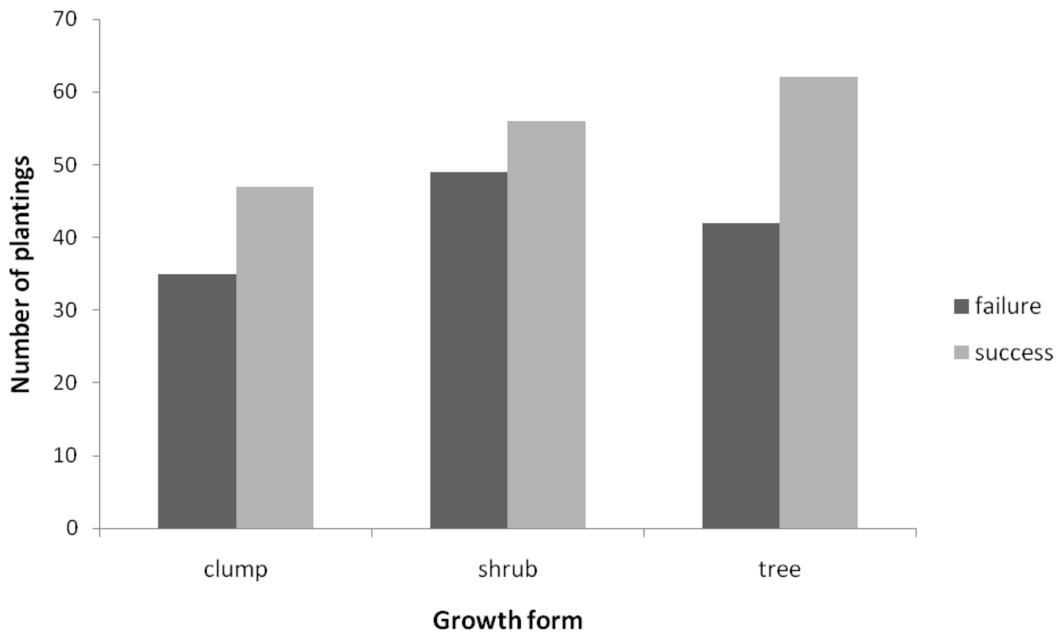


Figure 8: Frequency of species success for three growth forms: shrub, clump and tree. Successful plantings are presence of each planted species at each site.

Questionnaire

Population

The questionnaire was sent to 750 households, and 229 responses were received. The socioeconomic data from the sample population was compared with the same data from the 2006 Census.

The median age of respondents was 54.3 years, whereas the median age of the population in Wellington City is 33.1. Eighty-five percent of respondents had a post school qualification compared with 55.5% of people aged 15 years and over in Wellington City. The median household income of respondents approached $100,000 compared with $74,200 in Wellington City. Ninety-six percent of respondents owned their own home, compared with 47.1% home owners in Wellington as a whole.

Sixty-six percent of respondents indicated they would be prepared to be contacted again with a survey related to biodiversity. The number of households that report having planted land outside their own property was 152. Of these, 100 recall having received plants from WCC FPP.

Themes

Factors that contributed to the survival of plants reported by those respondents who had planted outside their own properties fell into six categories: aftercare (including watering, weeding and trimming), application of fertiliser or compost, mulching, selection of the right species for the environment, wind protection – either already existing or provided specifically for the planting, and all other factors. The frequency of each response is plotted in Figure 9.

More than half of the respondents indicated that they had provided some sort of care of the plants after planting day, and that this had contributed to the survival of some of the plants. The next most common response was suitability of species to the site conditions, or general perceived hardiness of the plant. Application of fertiliser and compost, or mulch both made up less than 10%, and wind protection just over.

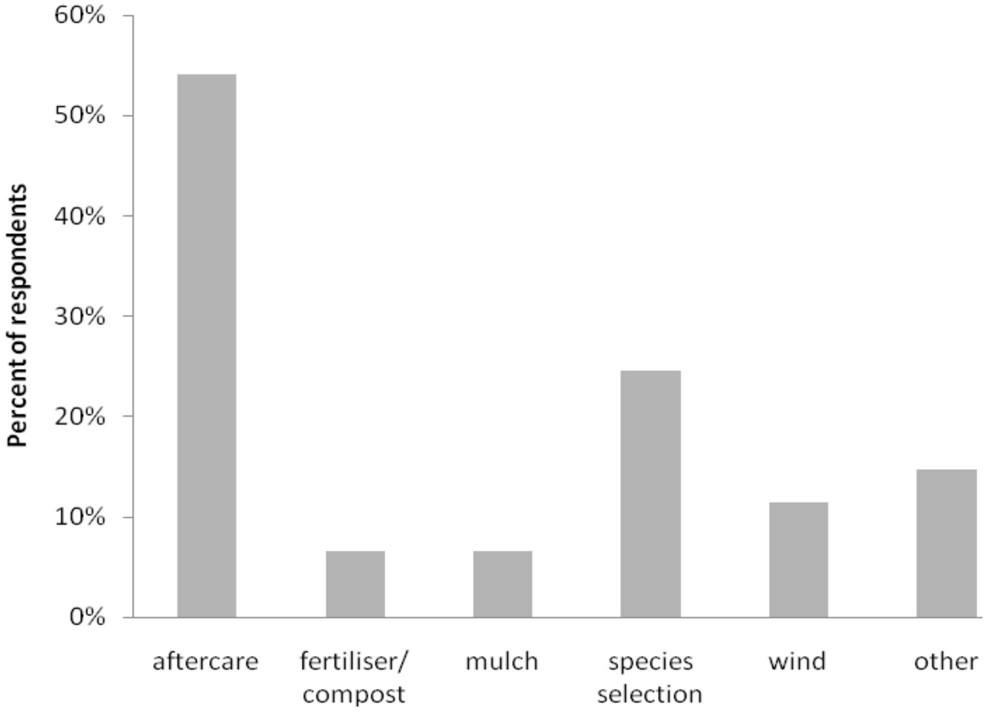


Figure 9: Factors reported by respondents as having contributed to plant survival. Respondents could indicate more than one factor.

Factors that hindered to the survival of plants, as reported by those respondents who had planted outside their own properties are shown in Figure 10; these fell into six categories. Damage - intentional or otherwise, and wind were the two most common responses, with approximately one third of respondents each. Poor site conditions such as soil or aspect was next most common, at 22%. Lack of water and the impact of weeds were slightly less frequent responses. Lack of care generally and species selection did not feature frequently, with only 6% and 3 % respectively.

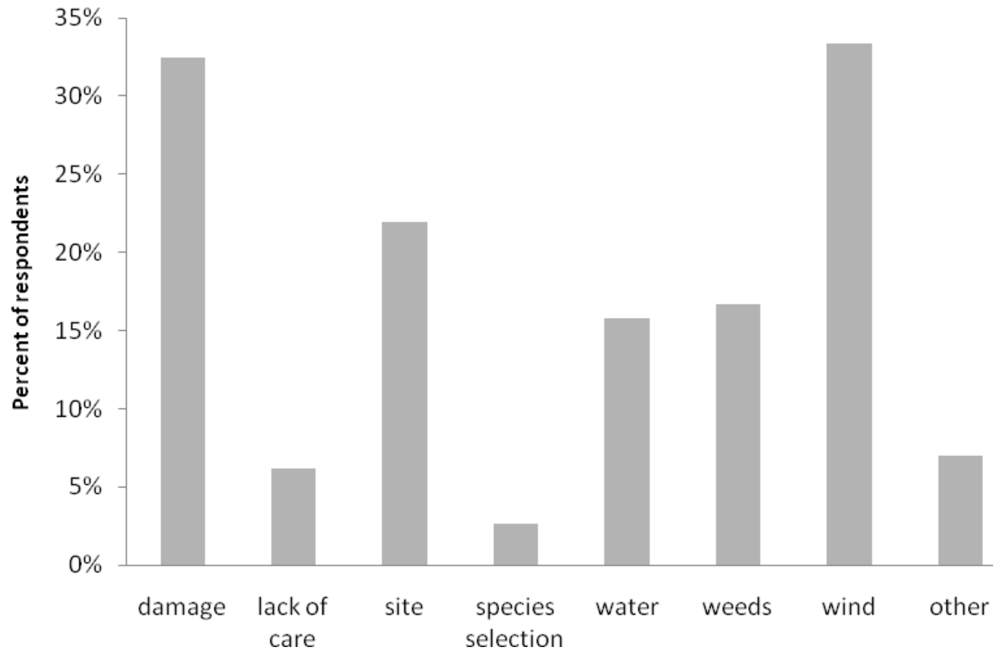


Figure 10: Factors reported by respondents as having hindered plant survival

Sixty-six percent of respondents indicated that they had obtained plants from the Council as part of the FPP. Only 18% indicated that they obtained plants solely through the FPP, meaning that 82% sourced some or all plants themselves (Figure 11).

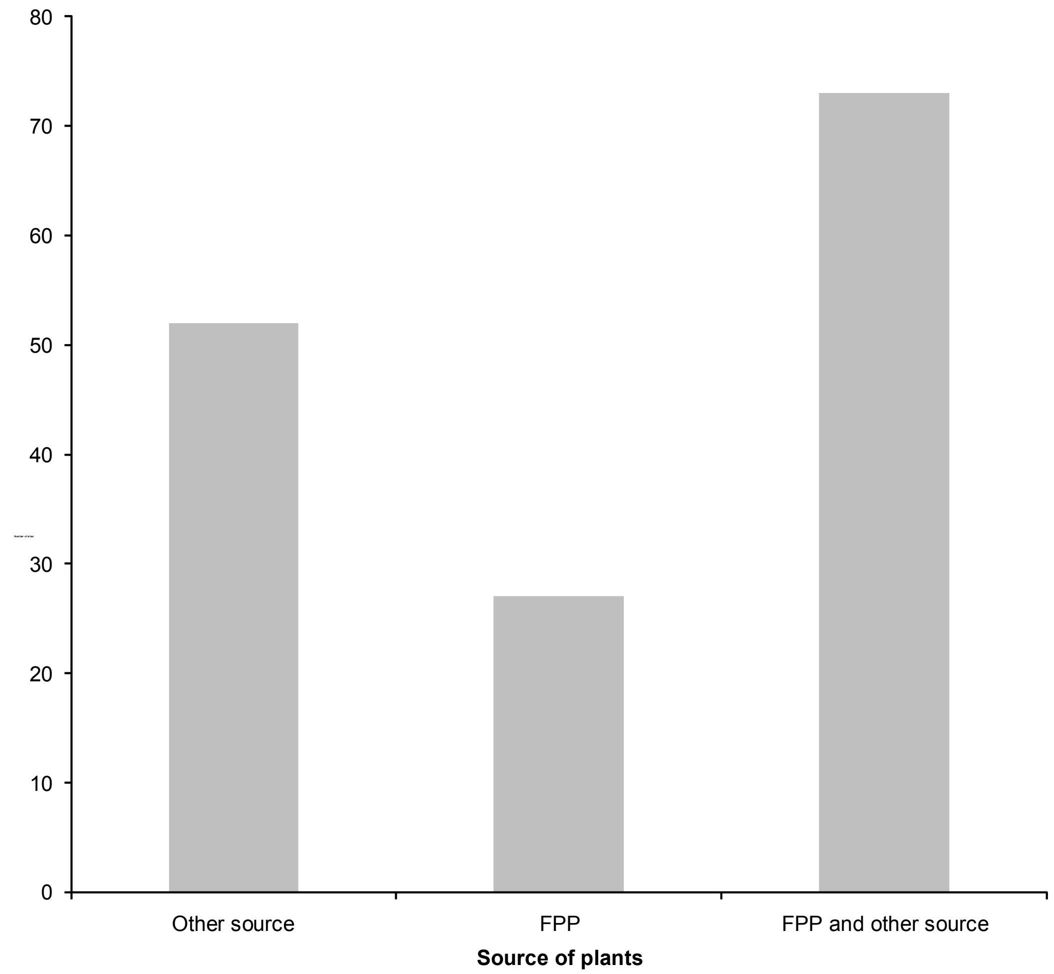


Figure 11: Frequency of questionnaire responses of from participants of planting outside their own property reporting source of plants used at each site, either Free Plants Programme (FPP), Other Source (such as retail or self propagated), or both. The total number of respondents was 152.

Model Selection

Biotic and abiotic factors affecting planting success

The global model with Year and Site random-effects mixed models, and the fixed-effects global model are listed in Table 10. All models tested are mixed effects (random and fixed) or fixed-effects logistic models. The random intercept assumes there may be heterogeneity between sites or between years and represents the combined effect of all omitted site-specific or year-specific covariates that cause some plants to have performed better than others. Modelling the *Site* and *Year* variables as random-effects limits how much variation is attributed to them. The fixed effect global model performed better than the mixed effects models (Δ AIC =3.5 & 7.8 for year and site respectively), indicating that there is unlikely to be significant site- or year-specific factors that were not measured in the surveys. These results show that the fixed-effects included in the global model are sufficient to capture variation between sites and years. The random-effects were not included in further analyses.

Fixed-effects models are listed in rank order with their AIC weights (w_i) in Table 11. Interaction terms, where included are recorded as factor1 × factor2.

Table 10

Global models for planting success based on biotic and abiotic traits

Global Model	k	AICc (n = 291)	Δ AIC
Fixed	25	380.8	0
Year as random-effect	26	384.3	3.5
Site as random-effect	26	388.6	7.8
Year and Site as random-effects	27	391.7	10.9

***Note*: Models are ranked by AIC value. Random-effects of year planted and variation between sites are included singly, and in combination.**

Table 11

Models describing planting success based on biotic and abiotic traits

	model	k	AIC	AICc (n = 291)	Δ AIC	wi
1	Indigineity	2	378.2	378.3	0.0	0.815
2	Indigineity + Growth Form	3	381.6	381.7	3.5	0.145
3	Canopy closure + Shade tolerance + Canopy closure × Shade tolerance	4	385.8	386.0	7.7	0.017
4	Aspect North + Slope + Topex + Elevation + Aspect North × Slope	6	386.7	387.0	8.7	0.010
5	Canopy closure + Early successional + Canopy closure × Early successional	4	387.4	387.6	9.3	0.008
6	Topex + Exposure tolerant + Topex × Exposure tolerant	4	389.6	389.7	11.4	0.003
7	Aspect North + Slope + Topex + Elevation + Aspect North × Slope	5	390.8	391.1	12.8	0.001
8	Topex + Drought tolerant + Topex × Drought tolerant	4	393.6	393.7	15.5	0.000
9	Garden + Early successional + Garden × Early successional	4	393.6	393.8	15.5	0.000
10	Topex	2	395.2	395.3	17.0	0.000
11	Topex + Growth Form + Topex × Growth Form	4	395.4	395.5	17.2	0.000
12	Topex + Growth Form	3	398.8	398.9	20.7	0.000
13	Area + Perimeter + Area × Perimeter,	4	401.9	402.0	23.8	0.000
14	Garden + Residential border	3	403.2	403.3	25.0	0.000
15	Area + Perimeter	3	404.0	404.0	25.8	0.000

Note: Models are ranked by Akaike Information Criteria

One of the two single-factor models, *Indigeneity*, performed the best amongst the candidate set. The AICc = 377.3, w_i = 0.8283, meaning that this model had 82% chance of being the best model in the set. This model (1), *Survival ~ Indigeneity* gives the odds of survival for native species as 4.35 that of exotic species (e^1.4712) (z = 4.709, p = 2.48e-06).

A 95% confidence set of models includes the top two Models 1 and 2, with the sum of AIC weights = 0.9682.

Model 2 includes *Growth form* as well as Indigineity (AICc = 380.8, w_i = 0.1399), however this factor was not significant at either level (Tree: z = -0.734, p = 0.4627, Shrub: z = -0.363, p = 0.7163). Comparing these two top model weights, the evidence ratio for model one is 0.828/0.140 = 5.92 times more likely than model 2 to best describe the data.

Other models tested were much less likely to be the best of the set to describe the data, with weights of 0.016 and less. Any of the models that included physical site characteristics were implausible (i.e. *Aspect north*, *Slope*, *Topex*, *Elevation*, *Garden*, *Area*, *Perimeter*, and *Residential border*). The species traits *Shade tolerant*, *Early successional*, *Drought tolerant* and *Exposure tolerant* were also only found in implausible models.

Human influences on planting success

Table 12 shows the Akaike Information Criterion (AICc) Weights for the twenty sites that were physically surveyed for species survival, and which had questionnaire responses. The best model from analysis of biotic and abiotic variables, which contained the single factor *Indigineity* (Model 3) was improved by the addition of the questionnaire variable *Time at address* (Model 1), and *Mulching* as follow-up care (Model 2).

Using AIC weights to compare the models indicates relative support for each model in the set, the best model in this analysis, Model 1 (w_i = 0.6128), is 4.9 times more likely to be the best model in the set than Model 2 (w_i = 0.1256) (0.6128/0.1256), and 9.9 times more likely than Model 3 w_i = 0.0621.

If a 95% confidence set of models was to be generated, models 1 to 7 need to be included to cumulatively sum AIC weights to >0.95. In order to use information contained in these models, multi-model inference would be required (Burnham & Anderson, 2002). However, if Model 3 (the base model) is compared to lower ranked models 4 to 7, although Δ *AIC* <2, it is noted that these models are the hierarchically more complex versions, that is the *Indigeneity* Model with one extra parameter added. As AIC penalises the number of parameters in a model (K) by adding 2 for each extra (AIC = -2L.L. + 2K) it can be concluded that the additional parameters in models 4 to 7 do not reduce deviance compared to the their base Indigeneity model, so are therefore uninformative parameters and should be excluded from further analysis (Arnold, 2010). None of the additional parameters in these models were statistically significant. Using the same criteria for the *Mulch* model, although the performance is slightly improved, the Δ *AIC* <2 again, and is therefore substantially equivalent to the simpler Indigineity model, and should also be excluded.

Applying these rules of parsimony and model fit, it can be concluded that inference should be drawn from Model 1 only, with the caveat that it has 61% support, but nonetheless is the best given the data and model set. Model 1 gives the odds of survival of natives as 3.7 (e^1.307) times that of exotics (i.e. a 270% increase), if *Time at Address* is held constant at any level. *Time at Address* gives a 9% (e^0.087) increase in odds of survival for each additional year a participant was resident at their address, when *Indigineity* is held constant at either level (native or exotic). In the sample population, the association of longer length of residence with increased success was more apparent for native plants then for exotics (Figure 12).

Table 12

Models of planting success for Indigineity plus Questionnaire variables.

	Model	k	AICc (n = 85)	Δ AIC	wi
1	Indigineity + time at address	3	105.8	0.0	0.613
2	Indigineity + FolMulch	3	109.0	3.2	0.126
3	Indigineity	2	110.4	4.6	0.062
4	Indigineity + distance from address to site	3	110.4	4.6	0.061
5	Indigineity + Group involvement	3	111.5	5.7	0.035
6	Indigineity + FolWatering	3	111.7	5.8	0.033
7	Indigineity + horticultural experience	3	112.1	6.2	0.027
8	Indigineity + FolWeeding	3	112.5	6.7	0.022
9	Indigineity + FolFertilizer	3	112.5	6.7	0.021

***Note*: The best performing model from analysis of planting success in site surveys, *Indigineity*, was used as the base model, with Questionnaire variables added individually. The models are ranked by AIC.**

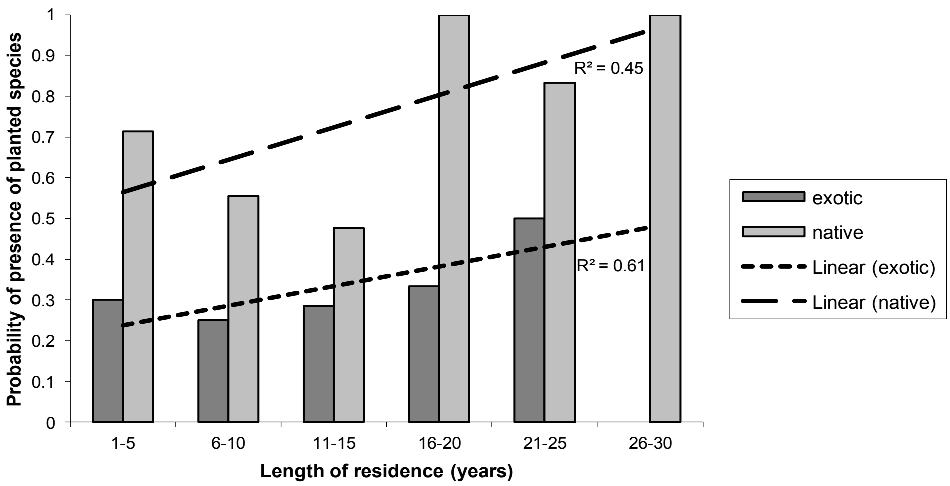


Figure 12: Probability of presence of a native or exotic planted species during site surveys for length of residence groups of the participant planting the plant.

Planting participation

The aim of the first analysis of participation in planting outside the respondent's property was to find which socioeconomic factors played a role in determining participation rates in planting outside of questionnaire respondents' properties. The dependent variable used was a binary variable for each questionnaire respondent named *PlantedOut,* and coded 1 indicating the respondent had planted anywhere outside their property, or 0 indicating they had not. Socioeconomic data were collected on *Age, Household Income, Education* and *Employment status*. *Employment status* was re-coded into a dichotomous variable *Working* indicating whether the respondent was working or not. Collinearity was examined using a matrix of Spearman's Rank (r_{pb}) correlations for ordinal-ordinal relations, and Point Biserial (ρ) correlations for dichotomous-ordinal relations as appropriate. Income showed multicollinearity with all three other variables: Age, Education and Working, and was removed from further analysis (ρ = -0.231, p = 0.016; ρ = 0.09, p = 0.039; r_{pb} = 0.156, p = 0.019 respectively). Working was also strongly correlated with age, and was also removed from further analysis (r_{pb} = -0.436, p = <0.001).

The remaining socioeconomic factors *Age* and *Education* were combined into the three possible regression models to predict participation using the *PlantedOut* dependant variable. The results of logistic regressions are listed in Table 13, ranked by AIC weight. Models 1 and 2 performed similarly (w_i = 0.494 & 0.451 respectively). Model 2 is, however, a hierarchically complex version of Model 1, and is less than 2 AIC difference so is therefore assumed that *Education* is an uninformative parameter in Model 2. Comparing AIC weights of Models 1 and 3, Model 1 is nine times more likely than model 3 to be the best model in the set

(0.4943/0.0548).This analysis shows that Age is the most important socioeconomic factor measured influencing participation in planting outside of a property.

Table 13

Socioeconomic models for participation in planting public spaces

	Model	k	AICc (n = 228)	Δ AIC	wi
1	Age	2	289.9	0.0	0.494
2	Age + Education	3	290.1	0.2	0.451
3	Education	2	294.3	4.4	0.055

***Note*: Models are ranked by AIC.**

The parameter estimate for *Age* from Model one shows that as a participant's age moves from one age bracket to the next older bracket, the odds of participating in planting outside their own property increases by 54% (odds = e^0.4343 = 1.544). This means that between the youngest age bracket (18 – 25 years) and the oldest (Over 70 years) we would expect an increase in the odds of participation of approximately 200% (i.e. a difference of four age brackets at 54% each). In other words, the ratio of participants to non-participants in the Over 70 Years age bracket is expected to be twice that of the ratio of participants to non-participants in the 18-25 age bracket. The proportion of respondents who indicated they participated was highest for the 51-70 years age group, with 70% of respondents in this age group participating (Figure 13). This was also the most numerous group of respondents and made up 47% of all respondents.

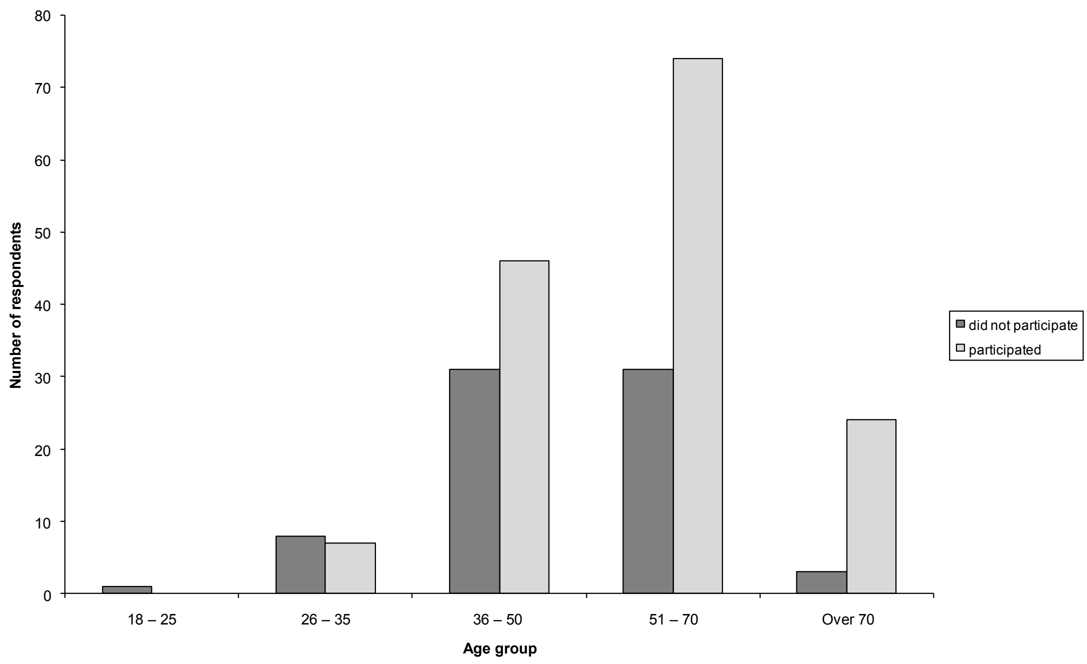


Figure 13: Numbers of respondents from each age group, having participated in planting outside their own properties or not.

The aim of the second analysis of participation was to find if any of the other factors measured in the questionnaire could improve the performance of the socioeconomic model from the first analysis. Model 1 was used as a base model to account for socioeconomic factors influencing planting behaviour, while investigating the influence of other hypothesised influences measured in the questionnaire. The four factors *Horticultural Knowledge*, number of trees outside the respondent's residence (*Trees Outside*), *Time at Address* and *Environmental Affinity* were individually added to the *Planted Outside ~ Age* model to create 4 new models. The models are shown in Table 14, ranked in order of AIC weight. Models 4 and 5, *Age* + *Trees Outside* and *Age* + *Horticultural Knowledge* performed similarly and together make up a 97% confidence set. These models were both a substantial improvement over the base model of *Age* alone, being 24 times as likely (*Trees Outside*) and 21 times as likely (*Hort knowledge*) to be the best models in the set, given the data. In the sample population, the association between number of trees outside a property and participation in planting was most marked at tree levels of 6 – 10 and above (Figure 14).

As there is considerable model selection uncertainty, drawing inference from either model becomes complex, however the parameter estimates from the two top models indicate that age, horticultural knowledge and the number of trees outside a property, are all positively associated with the resident's participation in planting outside their own property. The degree to which the odds of their participation in the planting increases is in the range of 30% - 45% for each unit increase of the variable, as measured by the questionnaire.

Table 14

Models explaining participation in planting public spaces..

	Model	k	AICc (n = 228)	Δ AIC	wi
4	Age + Trees Outside	3	283.6	0.0	0.515
5	Age + Hort Knowledge	3	283.8	0.3	0.454
1	Age	2	289.9	6.3	0.022
6	Age + Time at Address	3	291.6	8.1	0.009
7	Age + Environmental affinity	3	291.7	8.1	0.009

***Note*: Models use the best performing socioeconomic model (shown in bold) from Table 13 as a base model; individual questionnaire variables are added. Models are ranked by model weight**

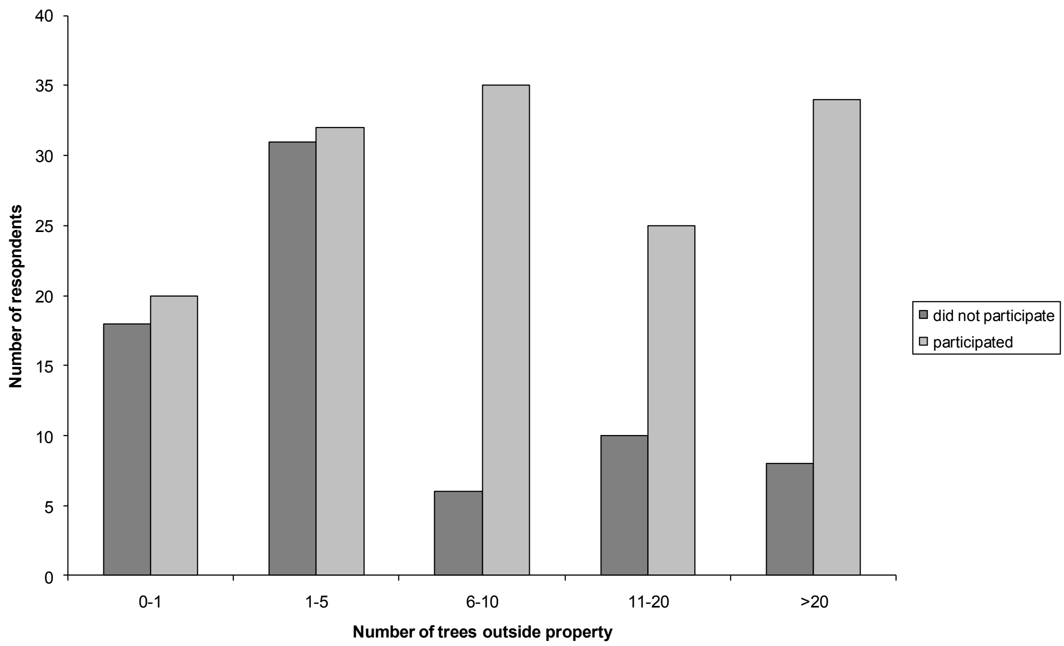


Figure 14: Number of respondents from each range of numbers of trees outside their property, having participated in planting outside their own properties.

DISCUSSION

Biotic and abiotic factors affecting planting success

The results of this analysis have clearly indicated one model that best describes the factor influencing species success. The model predicting New Zealand native species to have 4.4 times the chance of survival over non-natives is the simplest and has a high likelihood (82%) of being better than any other model tested. This provides clear and easily interpreted evidence in support of focussing resources on providing native plants to Wellington residents. This supports findings of other research indicating that native plants perform better in plantings, possibly due to their evolutionary adaptation to local conditions (e.g. O'Brien & Krauss, 2008; Pokorny et al., 2005; Stewart et al., 2004).

Between-site variation was not shown to be important in the models used to describe the data collected in this study, either as a random effect or a fixed effect. This indicates that it is not likely there were site factors that were not measured in this study and would have improved any of the models' fit, and that those that were measured did not explain species success as well as Indigeneity. This is surprising given the amount of variation in sites for variables often assumed to affect growth such as aspect (Lugo-Pérez & Sabat-Guérnica, 2010) and exposure (Mikita & Klimánek, 2010). It is possible that this result has been influenced by the dependent variable used. The methodological issues encountered in retrospectively assessing planted sites with virtually no record keeping meant that the study was based cautiously on the presence or absence of any individual member of a planted species. In cases where the environment is harsh due to wind exposure or poor light conditions, plants may be just "holding on", and the effect of their failure to thrive is not captured by the statistic. This statistic also does not capture the degree of success that species may have had, so a species with high survival rates is ranked the same as a species with a single surviving individual. Whether or not a species is thriving however, this result does indicate that in this study any particular site-species combination has a higher chance of being a possible success if the species in question is native.

Of the six species factors included in the models, the simplest model, containing 'Indigeneity' on its own, proved to be the best. This has clear implications for management for programmes of this type. The regression for this model shows that when native species are provided to residents, they have 4.35 times higher likelihoods of success than non-native plants. While the results show the average effect of all native species, and it cannot be assumed that all native species will outperform exotics, for a programme such as the FPP, where the objective is to generally improve the quantity and quality of urban plants, the cost effectiveness of establishing natives is clear.

Where appropriate, use of non-native plants will still be advantageous to meet specific needs, which may be either aesthetic or functional, for example the use of winter flowering tree species to provide a more consistent supply of food to nectar feeders, or fast growing exotics for erosion control. However, there is a far wider range of exotic plants than native plants to choose from, and these results indicate that greater care is perhaps required in selecting which exotic species are likely to be easily established. Additionally, some of the exotic species are no longer considered appropriate for general planting due to their tendency towards weediness. Native plants have the advantage of being adapted to local conditions, as well as free from problematic tendencies of some non-natives such as potential weediness. Native plants can, however, be considered problematic if they have a tendency towards weediness outside their locally-native regions. For example the locally non-native *Pittosporum crassifolium* (karo) and *Hoheria populnea* (houhere, lacebark) have been reported as becoming invasive in the Wellington region (Perrie, 2013), as have the widely planted pohutukawa (*Metrosideros excelsa*) (Greater Wellington Regional Council, 2010). Therefore some consideration of potential weediness needs to be considered even with non-locally native plants.

This highlights another surprising indication from the analysis: that native plants outperformed exotics, which are traditionally viewed as potentially invasive. Hierro *et al*. (2005) argue that invasiveness should be viewed from a biogeographic viewpoint and therefore must include reference to the plant's performance in its native environment, where it may display similar impacts. The authors consider that many assumptions about the impacts of species in recipient than in native communities are mainly anecdotal. For those species that are expected to be invasive in New Zealand, they may thus also display aggressive growth in their own country, and vice versa, New Zealand natives are just as likely to display aggressive invasive characteristics, therefore inferring invasiveness on the basis of xenicity is problematical. In another meta-analysis of invasives, Daehler (2003) found that in 94% of analyses, native performance was superior to the invasives in at least some growing conditions.

Of the seven theories of invasive traits reviewed by Hierro *et al.* (2005), only three seem relevant to planting success: Natural enemies (Exotics are released from natural enemies that control their population growth), Disturbance (Exotics are adapted to disturbance type and intensity that are novel to natives), and empty niche (Exotics utilize resources unused by the locals). The other theories rely on factors that have been mitigated due to the fact that the plants have already been established by human action. Such factors are: reproductive or dispersal success (Propagule pressure), rapid local evolution (Evolution of invasiveness), or direct competition with neighbouring native plants (Novel weapons, species richness). These three potentially relevant theories still presuppose that the plant is in competition with natives, however this is not necessarily the case in the sites included in this study. The mix of species in the plantings are inconsistent, as are the levels of weeding; both these factors influence the level and type of competition experienced by the exotics.

Other factors included in this study may also still be of value to consider when evaluating the appropriateness of a plant – site combination. Survival over a wide variety of conditions for a wide variety of plants was retrospectively assessed. Further experimental research could identify whether specific known plant - environment associations are important in urban settings, or whether the effect of anthropogenic influences is greater than in nature. For example, while topographic exposure did not have an overall effect, it would be useful to know whether sensitive plants are more successful in particularly sheltered sites.

Topographic exposure, or the degree to which a site is protected from wind would intuitively be an important factor in plant establishment, and has previously been modelled to predict vegetation zones when combined with other climatic data on wind direction and speed (Mikita & Klimánek, 2010). Other studies however, have concluded that topographic exposure on its own may be too simple a measure to describe the complexity of airflow induced by complex terrain (Mitchell, Hailemariam, & Kulis, 2001; Perry & Wilson, 2010). Similarly, aspect, or degrees from north alone may have been insufficient to capture the effect of wind or sun exposure without microclimate data such as terrain ventilation or hill shading.

Tolerance of plants to abiotic site conditions such as shade, exposure or drought may have been overwhelmed by the influence of anthropogenic effects such as aftercare provided by the participants of the programme.

Site size and the interaction of area and perimeter were hypothesised to affect plants due to their influence on edge effects. Habitat edges affect microclimate (e.g. available light, temperature, air movement), ecological flows such as seed rain and herbivore movement, and flows of resources (Ries, Fletcher, Battin, & Sisk, 2004). A larger site will have a smaller proportion of its area exposed to edge effects, while a higher area to perimeter ratio results in a less elongated shape, with lower edge effects relative to the same sized site with lower ratio. The low rank of these factors and models is probably due to the small size of the sites, meaning the entire site is influenced by edge effects. For example, Young & Mitchell (1994) found that in New Zealand Podocarp forest, fragments less than 0.9 ha were governed in their entirety by edge effects. As the largest site in this study was 593 m^2 (0.06 ha), edge effects would be likely to govern all the sites to the same extent.

Much of the literature on planting success in a restoration context focuses on human actions in terms of methods of planting and after care (e.g. Dostalek et al., 2007; Lai & Wong, 2005; Navarro Cerrillo et al., 2005). These are usually actions designed to mitigate the effects of poor site, inhospitable environment, or lack of resource. While it was not found in this study that any of the abiotic factors measured explained species success as precisely as indigeneity, it would be interesting to know whether the actions themselves had any effect.

Human influences on planting success

Questionnaire respondents were in several ways atypical of Wellington City residents. The sample was based on a population that was older, wealthier, had higher levels of education and was twice as likely to own their own home as the general Wellington population. As the questionnaire was not delivered to a random sample of the population, but rather a subset pre-selected on their already having participated in the FPP programme, this is unsurprising. These figures provide an interesting starting point to investigate whether these factors had any effect on participation in the public-site planting. These will be discussed later in the section.

The purpose of the analysis of questionnaire results was also to discover if any human influences had an effect on species success, measured during the site surveys. Human influences could either be deliberate actions such as plant care or site preparation, or they could be indirect influences such as socioeconomic factors. The results show that each of these may have a small effect; however the base model of Indigeneity again showed the strongest effect, tripling the chance of survival over exotic plants. Although the small sample size means that these results must be interpreted cautiously, it does lend weight to the substantial evidence reported in literature regarding the influence of human actions such as shelter and mulch on plant survival.

The 9% increase in odds of any species being successful for each year the participant was resident at their address may show the effect that those who stay longer at a property will invest more resources in improving their local environment. Given that the maximum length of residence reported in the questionnaire was 50 years, this effect could be significant. It may also be an effect of familiarity with local environmental conditions and reflect the experience gained through horticultural trial and error. That is, participants who have spent longer at an address have learnt what is required for plant survival there, such as wind shelter or water.

A third explanation for this effect could be related to the personal advantage motivation included in Summit and Sommer's (1998) analysis of urban tree planting behaviour. If a participant is motivated by increasing utility such as shade or shelter, or increasing property value (Wachter & Wong, 2008), then the length of a residence may determine the degree of benefit they might expect to gain from that planting as it becomes increasingly established.

The observed effect of mulching is also expected, and has been shown to improve seedling survival in several studies (e.g. Athy, Keiffer, & Stevens, 2006; Devine, Harrington, & Leonard, 2007; Dostalek et al., 2007). Mulching has the effect of increasing soil moisture retention and stabilising soil temperature, thereby reducing water stress on the plant and encouraging soil microbial activity. This is especially important for young seedlings with developing roots. Weeds are also suppressed, which benefits the seedlings by reducing competition.

The response rate of 30.5% was reasonable for this type of survey (Fox et al., 1988), and coupled with the two thirds proportion of those respondents who indicated that they would be willing to be contacted in future for other biodiversity research, could be interpreted as a population who are engaged with the programme, or urban greening in general. The predicted saliency of the questionnaire to the surveyed population appears to be confirmed by this result.

Those respondents who planted outside their own properties were asked if they thought any factors had helped or hindered the success of the plantings. The most commonly reported factor helping the survival of plants was basic weeding and watering. This is not surprising as lack of water and competition from invasive species are commonly reported as placing stress on plants and adversely affecting survival (Jacobs, Ross-Davis, & Davis, 2004; Jusaitis, 2005; Yelenik & Levine, 2010). It is not clear, however, how much of this result might be respondents' expectations of what influences success, rather than objective observations of plant survival at their sites. This would need to be investigated using experimental procedures to separate whether the weeding and watering provided a significant advantage over the natural rainfall and comparative growth rates in the local environment.

Twenty-five percent of respondents reported that they thought the species of the plant that they planted had an effect on whether or not the plant survived. Though it is not clear from the answers whether they thought that native species were more likely to survive, given the performance of indigeneity in the overall species survival analysis previously in this thesis, this adds some additional weight to the importance of type of plant.

When asked what had hindered plant survival, the results were not exactly the reverse of the factors that helped survival. For instance, weeds and lack of water were only fourth and fifth most commonly mentioned. Instead, damage either physical, or by the effects of wind were the two top responses. These are potentially catastrophic events that the respondents had little control over, such as vandalism or accidental destruction by mowing or vehicles. Severe wind events are also common in the study region, and there may be little that participants may have been able to do to mitigate such events. Species selection was the least commonly reported factor hindering survival.

Together, the results of helping and hindering factors tend to paint a picture of human action in plant management being significant in the absence of catastrophic environmental conditions. Human efforts may be overwhelmed, however, by severe environmental conditions when they occur. Once again, further empirical studies would be needed to investigate these effects more thoroughly.

Other potential human influences on seedling survival had little evidence supporting them in this study, though due to the small sample size, the interpretation of this should be limited. The literature, and the results of the participant opinion themes, suggests that there is good reason to assume that such fundamental plant care factors such as watering and weeding will enhance survival of seedlings, however further studies would be needed to confirm these effects for the specific circumstances of this study, which are particular to the region and the organisation of the programme. For example, a hypothesis could be posited that the Wellington's rainfall is high enough that watering of plants is not significant for their survival, as is the case for plants adapted to certain arid environments (e.g. Castell & Terradas, 1994).

The other factors that were hypothesised might indirectly influence species survival, but which were not found to be significant, showed fairly skewed distributions; 85% of respondents had not planted the site as part of a community/neighbourhood group, and 85% of respondent's planting sites were directly adjacent to their properties.

Once again however, the scale of the effect of indigeneity was the by far the biggest, with an expected 270% increase in odds of success of natives over exotics, for any given length of residence. This adds weight to the evidence from the Site Survey analysis, that indigeneity is the strongest influence on survival of the factors measured in this study.

Participation in planting

Socioeconomic influences

Of interest to programme managers is motivating participation. Goals such as habitat creation rely on a certain level of participation to create a useful number of sites, and total area, to have an influence on the matrix at a landscape level (Jules & Shahani, 2009; e.g. Kupfer et al., 2006; Rudd et al., 2002). To achieve a greater level of participation in programmes encouraging planting of public space, it is useful to understand what influences people to participate in such activities. Various studies have examined the role of socioeconomic factors in participation in urban forestry programmes and have found that factors such as age (Zhang et al., 2007), education (Wall et al., 2006) and income (Greene et al., 2011) played a role in participation, or respondents' indicated likelihood of participation.

In this study the questionnaire data were used to examine what influenced participation in planting outside the respondents own property. Of the 229 questionnaire respondents 152, or 66% indicated that they had planted outside their own property.

Socioeconomic factors were assumed to play a role in this, and it was found that the age of the respondent most accurately predicted whether or not they had been involved in planting land outside their own property. The best model showed that the odds of planting outside increased by 55% for each increase of the five age brackets. This supports other findings (e.g. Coulthard, Walker, & Morgan, 2000; Kirsch & Van Der Zanden, 2002) that older citizens are more involved in gardening, and more engaged in civic activities. Other socioeconomic factors that may be associated with participation were excluded due to their correlation with age, but whether or not one or the other has more of a causative effect on the response variable is not assumed.

For example, it could be argued that whether a respondent was working or not has a more causative effect than simply their age on their propensity to engage in community activity such as this; however, as age is correlated with the other socioeconomic variables, we can assume that any improvement in the age model indicates a real effect.

Other influences on participation

For the circumstances of the FPP investigated in this study, it was hypothesised that factors other than socioeconomic might also influence participation. Using the base model of Age identified above as representing the influence of the measured socioeconomic factors, it was found that model performance was improved by the addition of two factors: number of trees outside the respondent's property; and, their self-reported horticultural knowledge. Unsurprisingly, increasing horticultural knowledge positively affected the odds of participating in planting outside. The self-reported nature of the variable may indicate confidence in the respondent's ability and / or interest in tree planting generally. Involvement in horticulture as a hobby may also result in increased opportunity to find out about the FPP, and prior knowledge such as this has been shown to be associated with willingness to participate in a programme (Straka et al., 2005).

Less expected was the result that greater numbers of trees on the street outside respondent's properties was positively associated with participation. The environment that respondents live in is expected to influence their decision to participate in environmental improvement projects such as the FPP, and the number of trees outside a homeowner's property was thought likely to be an important environmental factor due to their visibility to the resident and potential effects on the property, such as providing shelter or shade. It was expected that participation might be motivated by the attempt to mitigate a lack of trees on a street by planting. There are however, several possible reasons that could be hypothesised for the opposing result. One explanation might be that those streets that do not already have mature trees on them are less likely to have room for any planting plots. Respondents with fewer trees outside their properties may be as likely to desire to plant outside, but are constrained by the lack of space, or other factors preventing trees being planted. This is perhaps an aspect of the local Wellington geography, where there is a high degree of variation in the type and size of road reserve between streets, which is usually constrained to a large degree by topography. As Wellington is built on a series of fault lines that have resulted in steep valleys and ridges with a degree of slope instability (Begg & Johnston, 2000), many residential roads are terraced into hillsides, leaving little width for traditional road reserve areas.

A second possible explanation for the effect of trees outside the property is selection bias, where homeowners with an affinity for trees and tree-planting may be likely to move to neighbourhoods that have established trees. Their interest in trees also influences them to become involved in tree planting programmes.

It must also be acknowledged that some of the results may be influenced by the study design. Respondents were asked: “How many trees are there on the street directly outside your property?” It was intended that this quantify the “greenness” of the street prior to planting, however, it may have been interpreted as including some plants provided as part of the FPP, in which case the dependent and independent variables are confounded. This effect would be mitigated to a large extent by the fact that the question asks specifically for “trees” rather than plants, and the majority of plants provided by the programme were shrubs or small plants, and that there did not appear from the responses or questionnaire pilot to be confusion about how this could be answered. Given the substantial uncertainty involved in interpreting these results, further conclusions should be drawn with caution.

The model that included Environmental affinity did not perform as well. Although Length of residence at the address has been shown in this study to influence species success, it has not been shown here to influence participation.

Considering the evidence from the two models where odds of participating in planting public spaces increases both with number of trees on street and horticultural knowledge, a hypothesis could be formed that the motivation to plant is more influenced by a personal interest in horticulture, rather than a desire to improve the environment by establishing more plants. This could be an important consideration in determining how to motivate people to participate in programmes such as the subject of this study. Programme managers could potentially expect good uptake with residents who have an established knowledge and enthusiasm for horticulture. Increasing awareness of a programme amongst this group could be the most efficient way to increase participation. Conversely, if managers wish to broaden the scope of those who are involved, marketing the programmes to demographics outside the typical wealthier, older, horticulturally-knowledgeable resident could open the benefits of planting to a much wider base. Given the age of participation in horticulture programmes has been shown to be decreasing (Kirsch & Van Der Zanden, 2002), and the social benefits to urban greening (Nowak & Dwyer, 2007), this could have positive outcomes beyond the scope of biological restoration aims.

CONCLUSIONS

Models developed in this study to describe successful establishment of species in an urban planting scheme of public places in Wellington showed that the most important factor measured in this study is indigeneity. Based on their presence at sites up to 15 years after planting, native species substantially outperformed exotics. Other species traits and site attributes measured in this study were not useful in predicting presence of a species planted, nor were the random-effects of year planted or site. This simple result provides an easy base of support for current practices in many planting schemes, where native plants are favoured.

There was some degree of evidence that the length of residence by participants in planting public land increased the success of the planting, over and above the effect of indigeneity. It is proposed here that this could be influenced by local knowledge of environment conditions, or increased motivation to put resources into maintaining a site. Further research could identify if this is an important factor.

Participation in public planting activity was shown to be mainly affected by age and horticultural knowledge. This provides an interesting basis for further research into what barriers there are to participating in such activities, and if urban biodiversity managers are looking for ways to increase participation, this could provide an interesting starting point.

These findings show that there can be value in assessing the outcomes of community planting programmes from a scientific framework. In future, programmes such as the FPP could improve the knowledge gained from them by viewing the plantings as a “designed experiment” as described by Felson and Picket (2005), where ecologists enter into partnerships with community groups, planners and landscape architects. This is expected to have the outcomes of allowing research in multiple locations, while providing aesthetic and functional benefits to the community.

These findings should help managers to best allocate resources to achieve the outcomes of community biodiversity projects.

REFERENCES

Anderson, D. R. (2008). Model Based Inference in the Life Sciences: A Primer on Evidence. *Biometrics*, *64*(4), 1309–1309. doi:10.1111/j.1541-0420.2008.01138_13.x

Arnold, T. W. (2010). Uninformative parameters and model selection using Akaike's Information Criterion. *Journal of Wildlife Management*, *74*(6), 1175–1178. doi:10.2193/2009-367

Athy, E. R., Keiffer, C. H., & Stevens, M. H. (2006). Effects of mulch on seedlings and soil on a closed landfill. *Restoration Ecology*, *14*(2), 233–241. doi:10.1111/j.1526-100X.2006.00125.x

Austin, M. (2002). Partnership opportunities in neighborhood tree planting initiatives: building from local knowledge. *Journal of Arboriculture*, *28*(4), 178–186.

Begg, J. G., & Johnston, M. R. (2000). *Geology of the Wellington area*. Lower Hutt: Institute of Geological & Nuclear Sciences Ltd.

Bellingham, P. J., Peltzer, D. A., Walker, L. R., Peter, J., Duane, A., & Lawrence, R. (2011). Contrasting impacts of a native and an invasive exotic shrub on flood-plain succession. *Biological Invasions*, *16*(1), 135–142.

Berkowitz, A. R., Canham, C. D., & Kelly, V. R. (1995). Competition vs. Facilitation of Tree Seedling Growth and Survival in Early Successional Communities. *Ecology*, *76*(4), 1156. doi:10.2307/1940923

Blick, R., Bartholomew, R., Burrell, T., & Burns, K. C. (2008). Successional dynamics after pest eradication in the Karori Wildlife Sanctuary. *New Zealand Natural Sciences*, *33*, 3–14.

Borgmann, K. L., & Rodewald, A. D. (2005). Forest Restoration in Urbanizing Landscapes: Interactions Between Land Uses and Exotic Shrubs. *Restoration Ecology*, *13*(2), 334–340. doi:10.1111/j.1526-100X.2005.00042.x

Borgström, S., Elmqvist, T., Angelstam, P., & Alfsen-Norodom, C. (2006). Scale mismatches in management of urban landscapes. *Ecology and Society*, *11*(2), 1–30.

Brodribb, T., & Hill, R. S. (1998). The photosynthetic drought physiology of a diverse group of southern hemisphere conifer species is correlated with minimum seasonal rainfall. *Functional Ecology*, *12*(3), 465–471. doi:10.1046/j.1365-2435.1998.00213.x

Burgin, S., & Wotherspoon, D. (2008). The potential for golf courses to support restoration of biodiversity for BioBanking offsets. *Urban Ecosystems*, *12*(2), 145–155. doi:10.1007/s11252-008-0076-5

Burnham, K., & Anderson, D. (2002). *Model selection and multimodel inference: a practical information-theoretic approach* (2nd ed.). Springer-Verlag, New York.

Cameron, R. W. F., Blanuša, T., Taylor, J. E., Salisbury, A., Halstead, A. J., Henricot, B., & Thompson, K. (2012). The domestic garden – Its contribution to urban green infrastructure. *Urban Forestry & Urban Greening*, *11*, 129–137. doi:10.1016/j.ufug.2012.01.002

Castell, C., & Terradas, J. (1994). Effects of water and nutrient availability on water relations, gas exchange and growth rate of mature plants and resprouts of Arbutus unedo L. *Annals of Botany*, *73*, 595–602.

Clewell, A., Aronson, J., & Winterhalder, K. (2004). *The SER International primer on ecological restoration*. *Ecological Restoration* (Vol. 2, pp. 206–207). Tuson: Society for Ecological Restoration International Science & Policy Working Group.

Cole, D., & Spildie, D. (2006). Restoration of plant cover in subalpine forests disturbed by camping: success of transplanting. *Natural Areas Journal*, *26*(2), 168–178. doi:10.3375/0885-8608

Coulthard, M., Walker, A., & Morgan, A. (2000). *People's perceptions of their neighbourhood and community involvement: Results from the social capital module of the General Household Survey 2000*. London: Her Majesty's Stationery Office (HMSO).

Curtis, A., & De Lacy, T. (1994). Landcare in Australia: Does it Make a Difference? *Journal of Environmental Management*, *46*, 119–137.

Daehler, C. C. (2003). Performance comparisons of co-occurring native and alien invasive plants: implications for conservation and restoration. *Annual Review of Ecology, Evolution, and Systematics*, *34*(1), 183–211. doi:10.1146/annurev.ecolsys.34.011802.132403

Davis, M., & Meurk, C. (2001). *Protecting and restoring our natural heritage: A practical guide*. Wellington, NZ: Department of Conservation.

Devine, W. D., Harrington, C. a., & Leonard, L. P. (2007). Post-planting treatments increase growth of Oregon white oak (Quercus garryana Dougl. ex Hook.) seedlings. *Restoration Ecology*, *15*(2), 212–222. doi:10.1111/j.1526-100X.2007.00205.x

Dietz, H., Steinlein, T., & Ullmann, I. (1998). The role of growth form and correlated traits in competitive ranking of six perennial ruderal plant species grown in unbalanced mixtures. *Acta Oecologica*, *19*(1), 25–36. doi:S1146609X98800052

Dillman, D. A. (2007). *Mail and internet surveys: The tailored design method* (2nd ed., Vol. 2006, p. 542). John Wiley & Sons. doi:10.2501/S0021849907070237

Dillman, D. A., & Carley-Baxter, L. R. (2000). Structural determinants of mail survey response rates over a 12 year period, 1988-1999. In *Proceedings of the Survey Research Methods Section, American Statistical Association* (pp. 394–399).

Dorner, J. (2002). *An introduction to using native plants in restoration projects. globalrestorationnetwork.org* (pp. 1–66). Plant Conservation Alliance, U.S. Environmental Protection Agency.

Dostalek, J., Weber, M., Matula, S., & Frantik, T. (2007). Forest stand restoration in the agricultural landscape: The effect of different methods of planting establishment. *Ecological Engineering*, *29*(1), 77–86. doi:10.1016/j.ecoleng.2006.07.016

Dostálek, J., Weber, M., Matula, S., Petruš, J., Koželuhová, K., Frantík, T., & Možný, M. (2005). Planting of Woody Species in an Agricultural Landscape: a Case Study in the Valová River Floodplain. *Acta Pruhoniciana*, (80), 2–65.

East, R., & Williams, G. (1984). Island biogeography and the conservation of New Zealand's indigenous forest-dwelling avifauna. *New Zealand Journal of Ecology*, (7), 27–35.

Eviner, V. T., & Chapin III, F. S. (2003). Functional matrix: A conceptual framework for predicting multiple plant effects on ecosystem processes. *Annual Review of Ecology, Evolution, and Systematics*, *34*(1), 455–485. doi:10.1146/annurev.ecolsys.34.011802.132342

Fox, R. J., Crask, M. R., & Kim, J. (1988). Mail Survey Response Rate: A Meta-Analysis of Selected Techniques for Inducing Response. *Public Opinion Quarterly*, *52*(4), 467–491.

Gaston, K. J., Smith, R. M., Thompson, K., & Warren, P. H. (2005). Urban domestic gardens (II): experimental tests of methods for increasing biodiversity. *Biodiversity and Conservation*, *14*(2), 395–413. doi:10.1007/s10531-004-6066-x

Gibb, H. (2002). Habitat fragmentation in an urban environment: large and small fragments support different arthropod assemblages. *Biological Conservation*, *106*(1), 91–100. doi:10.1016/S0006-3207(01)00232-4

Greater Wellington Regional Council. (2010). *Wellington Regional Native Plant Guide*. *Native Plants*. Wellington, NZ.

Greater Wellington Regional Council. (2012). Wind and Storms. Retrieved June 01, 2012, from http://www.gw.govt.nz/Wellington-region-weather-and-climate/

Greene, C. S., Millward, A. a., & Ceh, B. (2011). Who is likely to plant a tree? The use of public socio-demographic data to characterize client participants in a private urban forestation program. *Urban Forestry & Urban Greening*, *10*(1), 29–38. doi:10.1016/j.ufug.2010.11.004

Grime, J. (1977). Evidence for the existence of three primary strategies in plants and its relevance to ecological and evolutionary theory. *American Naturalist*, *111*(982), 1169–1194.

Grubb, P. J. (1998). A reassessment of the strategies of plants which cope with shortages of resources. *Perspectives in Plant Ecology and Systematics*, (1), 3–31.

Hierro, J. L., Maron, J. L., & Callaway, R. M. (2005). A biogeographical approach to plant invasions: the importance of studying exotics in their introduced and native range. *Journal of Ecology*, (93), 5–15. doi:10.1111/j.1365-2745.2004.00953.x

Hill, M. O., Roy, D. B., & Thompson, K. (2002). Hemeroby, urbanity and ruderality: bioindicators of disturbance and human impact. *Journal of Applied Ecology*, *39*(5), 708–720. doi:10.1046/j.1365-2664.2002.00746.x

Jacobs, D. F., Ross-Davis, A. L., & Davis, A. S. (2004). Establishment success of conservation tree plantations in relation to silvicultural practices in Indiana , USA. *New Forests*, (28), 23–36.

Johnson, J. B., & Omland, K. S. (2004). Model selection in ecology and evolution. *Trends in Ecology & Evolution*, *19*(2), 101–8. doi:10.1016/j.tree.2003.10.013

Jones, R. E., Davis, K. L., & Bradford, J. (2012). The value of trees: factors influencing homeowner support for protecting local urban trees. *Environment and Behavior*, (1), 1–27. doi:10.1177/0013916512439409

Jules, E. S., & Shahani, P. (2009). A broader ecological context to habitat fragmentation: Why matrix habitat is more important than we thought. *Journal of Vegetation Science*, *14*(3), 459–464. doi:10.1111/j.1654-1103.2003.tb02172.x

Jusaitis, M. (2005). Translocation trials confirm specific factors affecting the establishment of three endangered plant species. *Ecological Management and Restoration*, *6*(1), 61–67. doi:10.1111/j.1442-8903.2005.00220.x

Kirsch, E., & Van Der Zanden, A. (2002). Demographics and volunteer experiences of Oregon Master Gardeners. *HortTechnology*, (12), 505–508.

Kupfer, J., Malanson, G. P., & Franklin, S. B. (2006). Not seeing the ocean for the islands: the mediating influence of matrix-based processes on forest fragmentation effects. *Global Ecology and Biogeography*, (15), 8 – 20. doi:10.1111/j.1466-822x.2006.00204.x

Lai, P. C. C., & Wong, B. S. F. (2005). Effects of tree guards and weed mats on the establishment of native tree seedlings: implications for forest restoration in Hong Kong, China. *Restoration Ecology*, *13*(1), 138–143. doi:10.1111/j.1526-100X.2005.00016.x

Landcare Research. (n.d.). Ecological Traits of New Zealand Flora. *Ecotraits Database*. Retrieved from http://ecotraits.landcareresearch.co.nz/

Landcare Research. (2011). The Green Toolbox Database. Retrieved from http://www.landcareresearch.co.nz/resources/tools/green-toolbox

Lane, C., & Raab, S. (2002). Great River Greening: a case study in urban woodland restoration. *Ecological Restoration*, *20*(4), 243–251.

Lee, W., McGlone, M., & Wright, E. (2005). Biodiversity inventory and monitoring: a review of national and international systems and a proposed framework for future biodiversity monitoring by the Department of Conservation. *Landcare Research Contract Report*. doi:10.1.1.111.326

Lugo-Pérez, J., & Sabat-Guérnica, A. M. (2010). Structure and composition of woody plants in urban forest remnants with different adjacent land-use and slope aspect. *Urban Ecosystems*, *14*(1), 45–58. doi:10.1007/s11252-010-0139-2

MacArthur, R. H., & Wilson, E. O. (1967). *The Theory of Island Biogeography*. Princeton, N.J.: Princeton University Press.

Maclean, C. (2009). "Wellington region - Climate", updated 3-Mar-09. *Te Ara - the Encyclopedia of New Zealand*. Retrieved from http://www.teara.govt.nz/en/wellington-region/3

Marañón, T., Castro, J., Zamora, R., Villar, R., Zavala, M. A., Quero, J. L., … Mendoza, I. (2004). Regeneration of tree species and restoration under contrasted Mediterranean habitats: field and glasshouse experiments. *International Journal of Ecology and Environmental Sciences*, *30*(3), 187–196.

Marynissen, P. R., Campbell, D. M., Preston, C., Watts, J. H., & Crossman, N. D. (2006). Rural lands incentives program and gardens for wildlife. In *15th Australian Weeds Conference, Papers and Proceedings, Adelaide, South Australia, 24-28 September 2006: Managing weeds in a changing climate.* (pp. 462-465). Weed Management Society of South Australia.

Marzluff, J. M. (2005). Island biogeography for an urbanizing world: how extinction and colonization may determine biological diversity in human-dominated landscapes. *Urban Ecosystems*, *8*(2), 157–177. doi:10.1007/s11252-005-4378-6

McKinney, M. (2002). Urbanization, biodiversity, and conservation. *BioScience*, *52*(10), 883–890. doi:10.1641/0006-3568

McPherson, E., Nowak, D., & Heisler, G. (1997). Quantifying urban forest structure, function, and value: the Chicago Urban Forest Climate Project. *Urban Ecosystems*, *1*, 49–61.

McDonnell, M. J., Pickett, S. t. A., Groffman, P., Bohlen, P., Pouyat, R. V., Zipperer, W. C., … Medley, K. (1997). Ecosystem processes along an urban-to-rural gradient. *Urban Ecosystems*, *1*, 21–36.

Meurk, C. D., & Swaffield, S. (2000). A landscape ecological framework for indigenous regeneration in rural New Zealand-Aotearoa. *Landscape and Urban Planning*, *50*(1-3), 129–144. doi:10.1016/S0169-2046(00)00085-2

Middleton, E. L., Bever, J. D., & Schultz, P. a. (2010). The effect of restoration methods on the quality of the restoration and resistance to invasion by exotics. *Restoration Ecology*, *18*(2), 181–187. doi:10.1111/j.1526-100X.2008.00501.x

Mikita, T. O., & Klimánek, M. A. (2010). Topographic exposure and its practical applications. *Journal of Landscape Ecology*, *3*(1), 42–51.

Mitchell, S. J., Hailemariam, T., & Kulis, Y. (2001). Empirical modeling of cutblock edge windthrow risk on Vancouver Island, Canada, using stand level information. *Forest Ecology and Management*, *154*(1-2), 117–130. doi:10.1016/S0378-1127(00)00620-4

Miyawaki, A. (1999). Creative ecology: restoration of native forests by native trees. *Plant Biotechnology*, *16*(1), 15–20.

Moskell, C., Allred, S. B., & Ferenz, G. (2010). Examining motivations and recruitment strategies for urban forestry volunteers. *Cities and the Environment*, *3*(1), 1–28.

Muratet, A., Machon, N., Jiguet, F., Moret, J., & Porcher, E. (2007). The role of urban structures in the distribution of wasteland flora in the Greater Paris Area, France. *Ecosystems*, *10*(4), 661–671. doi:10.1007/s10021-007-9047-6

Navarro Cerrillo, R. M., Fragueiro, B., Ceaceros, C., del Campo, A., & de Prado, R. (2005). Establishment of Quercus ilex L. subsp. ballota [Desf.] Samp. using different weed control strategies in southern Spain. *Ecological Engineering*, *25*(4), 332–342. doi:10.1016/j.ecoleng.2005.06.002

Niemelä, J. (1999). Ecology and urban planning. *Biodiversity and Conservation*, *8*, 119–131.

Niinemets, Ü., & Valladares, F. (2006). Tolerance to shade, drought, and waterlogging of temperate northern hemisphere trees and shrubs. *Ecological Monographs*, *76*(4), 521–547. doi:10.1890/0012-9615(2006)076[0521:TTSDAW]2.0.CO;2

Nowak, D., & Dwyer, J. F. (2007). Understanding the benefits and costs of urban forest ecosystems. In J. E. Kuser (Ed.), *Urban and community forestry in the northeast* (pp. 25–46). Springer Netherlands.

Nowak, D., Kuroda, M., & Crane, D. (2004). Tree mortality rates and tree population projections in Baltimore, Maryland, USA. *Urban Forestry & Urban Greening*, *2*, 139–147.

Nowak, D., McBride, J., & Beatty, R. (1990). Newly planted street tree growth and mortality. *Journal of Arboriculture*, *16*(5), 124–129.

O'Brien, E. K., & Krauss, S. L. (2008). Testing the Home-Site Advantage in Forest Trees on Disturbed and Undisturbed Sites. *Restoration Ecology*, *18*(3), 359–372. doi:10.1111/j.1526-100X.2008.00453.x

Padilla, F. M., & Pugnaire, F. I. (2007). Rooting depth and soil moisture control Mediterranean woody seedling survival during drought. *Functional Ecology*, *21*(3), 489–495. doi:10.1111/j.1365-2435.2007.01267.x

Padilla, Francisco M., & Pugnaire, F. I. (2006). The role of nurse plants in the restoration of degraded environments. *Frontiers in Ecology and the Environment*, *4*(4), 196–202. doi:10.1890/1540-9295(2006)004[0196:TRONPI]2.0.CO;2

Pavao-Zuckerman, M. a. (2008). The nature of urban soils and their role in ecological restoration in cities. *Restoration Ecology*, *16*(4), 642–649. doi:10.1111/j.1526-100X.2008.00486.x

Perrie, L. (2013). Plant Conservation Conference and weedy native plants. *Te Papa blog*. Retrieved from http://blog.tepapa.govt.nz/2013/05/29/plant-conservation-conference-and-weedy-native-plants/

Perry, T. E., & Wilson, J. S. (2010). Evaluating the vulnerability of Maine forests to wind damage. In J. S. Pye, John M.; Rauscher, H. Michael; Sands, Yasmeen; Lee, Danny C.; Beatty (Ed.), *Advances in threat assessment and their application to forest and rangeland management. Gen. Tech. Rep. PNW-GTR-802. Portland, OR* (pp. 237–246). U.S. Department of Agriculture, Forest Service.

Pinheiro, J., & Bates, D. (1998). *lme and nlme: Mixed-Effects Methods and Classes for S and S-PLUS. October* (3rd ed.). Madison: Bell Labs, Lucent Technologies and University of Wisconsin.

Pokorny, M. L., Sheley, R. L., Zabinski, C. a., Engel, R. E., Svejcar, T. J., & Borkowski, J. J. (2005). Plant functional group diversity as a mechanism for invasion resistance. *Restoration Ecology*, *13*(3), 448–459. doi:10.1111/j.1526-100X.2005.00056.x

Pratt, C. (1999). *Factors affecting the establishment, growth and survival of native woody plant communities on the Canterbury Plain, New Zealand (*Master's thesis, Lincoln University, New Zealand). Retrieved from http://hdl.handle.net/10182/1113

Raman, T. R. S., Mudappa, D., & Kapoor, V. (2009). Restoring rainforest fragments: survival of mixed-native species seedlings under contrasting site conditions in the Western Ghats, India. *Restoration Ecology*, *17*(1), 137–147. doi:10.1111/j.1526-100X.2008.00367.x

Rebele, F. (1994). Urban ecology and special features of urban ecosystems. *Global Ecology and Biogeography Letters*, *4*(6), 173–187.

Ries, L., Fletcher, R. J., Battin, J., & Sisk, T. D. (2004). Ecological responses to habitat edges: Mechanisms, Models, and Variability Explained. *Annual Review of Ecology, Evolution, and Systematics*, *35*(1), 491–522. doi:10.1146/annurev.ecolsys.35.112202.130148

Rudd, H., Vala, J., & Schaefer, V. (2002). Importance of backyard habitat in a comprehensive biodiversity conservation strategy: a connectivity analysis of urban green spaces. *Restoration Ecology*, *10*(2), 368–375. doi:10.1046/j.1526-100X.2002.02041.x

Schneider, A. (2012). gpsvisualizer.com. Retrieved from http://www.gpsvisualizer.com

Shochat, E., Warren, P. S., Faeth, S. H., McIntyre, N. E., & Hope, D. (2006). From patterns to emerging processes in mechanistic urban ecology. *Trends in Ecology & Evolution*, *21*(4), 186–91. doi:10.1016/j.tree.2005.11.019

Simberloff, D., & Abele, L. (1976). Island biogeography theory and conservation practice. *Science*, *191*(4224), 285–286.

Simberloff, D., & Dayan, T. (1991). The guild concept and the structure of ecological communities. *Annual Review of Ecology and Systematics*, *22*, 115–43.

Sklar, F., & Ames, R. G. (1985). Staying Alive: street tree survival in the inner-city. *Journal of Urban Affairs*, *7*(1), 55–66.

Smaill, S., Ledgard, N., Langer, E., & Henley, D. (2011). Establishing native plants in a weedy riparian environment. *New Zealand Journal of Marine and Freshwater Research*, *45*(3), 357–367. doi:10.1080/00288330.2011.589456

Sommer, R., Learey, F., Summit, J., & Tirrell, M. (1994). The social benefits of resident involvement in tree planting. *Journal of Arboriculture*, *20*(3), 170–175.

StatSoft Inc. (2012). *Electronic Statistics Textbook*. Tulsa, OK: StatSoft.

Stewart, G. H., Ignatieva, M. E., Meurk, C. D., & Earl, R. D. (2004). The re-emergence of indigenous forest in an urban environment, Christchurch, New Zealand. *Urban Forestry & Urban Greening*, *2*(3), 149–158.

Straka, T. J., Marsinko, A. P., & Childers, C. J. (2005). Individual characteristics affecting participation in urban and community forestry programs in South Carolina, US. *Journal of Arboriculture*, *31*(3), 131–137.

Sullivan, J., & Meurk, C. (2009). Restoring native ecosystems in urban Auckland: urban soils, isolation, and weeds as impediments to forest establishment. *New Zealand Journal of Ecology*, *33*, 60–71.

Sullivan, J., Williams, P. A., Timmins, S. M., & Smale, M. C. (2009). Distribution and spread of environmental weeds along New Zealand roadsides. *New Zealand Journal of Ecology*, *33*(2), 190–204.

Summit, J., & McPherson, E. (1998). Residential tree planting and care: a study of attitudes and behavior in Sacramento, California. *Journal of Arboriculture*, *24*, 89–97.

Summit, J., & Sommer, R. (1998). Urban tree-planting programs — A model for encouraging environmentally protective behavior. *Atmospheric Environment*, *32*(1), 1–5. doi:10.1016/S1352-2310(97)00175-1

Terman, M. R. (1997). Natural links: naturalistic golf courses as wildlife habitat. *Landscape and Urban Planning*, *38*(3-4), 183–197.

Thompson, K., Austin, K. C., Smith, R. M., Warren, P. H., Angold, P. G., & Gaston, K. J. (2009). Urban domestic gardens (I): Putting small-scale plant diversity in context. *Journal of Vegetation Science*, *14*(1), 71–78. doi:10.1111/j.1654-1103.2003.tb02129.x

Thompson, K., Hodgson, J. G., & Rich, T. C. G. (1995). Native and alien invasive plants: more of the same? *Ecography*, *18*(4), 390–402. doi:10.1111/j.1600-0587.1995.tb00142.x

Tidball, K. G., & Krasny, M. (2007). From Risk to Resilience: What Role for Community Greening and Civic Ecology in Cities? In A. E. J. Wals (Ed.), *Social Learning Towards a more Sustainable World* (pp. 149–164). Wagengingen, The Netherlands: Wagengingen Academic Press.

Valladares, F., & Niinemets, Ü. (2008). Shade tolerance, a key plant feature of complex nature and consequences. *Annual Review of Ecology, Evolution, and Systematics*, *39*(1), 237–257. doi:10.1146/annurev.ecolsys.39.110707.173506

Vallet, J., Daniel, H., Beaujouan, V., Rozé, F., & Pavoine, S. (2010). Using biological traits to assess how urbanization filters plant species of small woodlands. *Applied Vegetation Science*, *13*(4), 412–424. doi:10.1111/j.1654-109X.2010.01087.x

Vidra, R. L., & Shear, T. H. (2008). Thinking locally for urban forest restoration: a simple method links exotic species invasion to local landscape structure. *Restoration Ecology*, *16*(2), 217–220. doi:10.1111/j.1526-100X.2008.00387.x

Wachter, S., & Wong, G. (2008). What is a tree worth? green-city strategies, signaling and housing prices. *Real Estate Economics*, 213–239. doi:10.1111/j.1540-6229.2008.00212.x

Wagenmakers, E.-J., & Farrell, S. (2004). AIC model selection using Akaike weights. *Psychonomic Bulletin & Review*, *11*(1), 192–6.

Wall, B., Straka, T., & Miller, S. (2006). An econometric study of the factors influencing participation in urban and community forestry programs in the United States. *Arboriculture & Urban Forestry*, *32*(5), 221–228.

Ward Thompson, C., Roe, J., Aspinall, P., Mitchell, R., Clow, A., & Miller, D. (2012). More green space is linked to less stress in deprived communities: Evidence from salivary cortisol patterns. *Landscape and Urban Planning*, *105*(3), 221–229. doi:10.1016/j.landurbplan.2011.12.015

Wellington City Council. (1998). Open space strategy for Wellington. Retrieved from http://wellington.govt.nz/~/media/your-council/plans-policies-and-bylaws/plans-and-policies/a-to-z/capitalspaces/files/capitalspacespolicy.pdf

Wellington City Council. (2004a). Wellington's Outer Green Belt Management Plan. Retrieved from http://wellington.govt.nz/your-council/plans-policies-and-bylaws/plans-policies/a-z-index/outer-green-belt-management-plan

Wellington City Council. (2004b). Town Belt Management Plan. Retrieved from http://wellington.govt.nz/your-council/plans-policies-and-bylaws/plans-policies/a-z-index/town-belt-management-plan

Westphal, L. (2003). Urban greening and social benefits: a study of empowerment outcomes. *Journal of Arboriculture*, *29*(3), 137–147.

Yelenik, S. G., & Levine, J. M. (2010). Processes limiting native shrub recovery in exotic grasslands after non-native herbivore removal. *Restoration Ecology*, *18*(2), 418–425. doi:10.1111/j.1526-100X.2009.00598.x

Young, A., & Mitchell, N. (1994). Microclimate and vegetation edge effects in a fragmented podocarp-broadleaf forest in New Zealand. *Biological Conservation*, *67*(1), 63–72. doi:10.1016/0006-3207(94)90010-8

Zhang, Y., Hussain, A., West, J. D., & Letson, N. (2007). Public attitudes toward urban trees and supporting urban tree programs. *Environment and Behavior*, *39*(6), 797–814. doi:10.1177/0013916506292326

Zuur, A., Ieno, I., Walker, N., Saveliev, A., & Smith, G. (2009). *Mixed effects models and extensions in ecology with R*. New York: Springer.

APPENDICES

APPENDIX ONE - WELLINGTON ROAD RESERVE PLANTING SURVEY

Please answer this survey to the best of your memory. Tick ☑ the appropriate box or write your answer in the space provided or use additional paper if required. Feel free to discuss the answers with others in your household. This will take less than 10 minutes of your time to complete.
We would appreciate it if you could complete and return the survey by 31 December 2011.

Please start here

1. How often did you visit a green space (park, reserve or public gardens) in the last year?

- ☐ Not at all
- ☐ Once or twice a year
- ☐ About once a month
- ☐ Two to three times a month
- ☐ Two or more times a week

2. What in your opinion are the three greatest threats to native plants and animals in New Zealand?

i. ....

ii. ....

iii. ....

3. How experienced do you consider yourself to be in gardening / horticulture?

Inexperienced **1 2 3 4 5** Experienced

4. How many trees are there a) on your property **0 1-5 6-10 11-20 >20**

b) on the street directly outside your property? **0 1-5 6-10 11-20 >20**

5. What trees would you like to see planted on your street? (*Please list names or use descriptions if you prefer*)

____

6. What do you like or dislike about the plants planted on your street?

____

____

7. Please answer the following questions about planting on and off your property by circling the appropriate answer or writing in the space provided.

	a) On your property?	**b) Outside your property (e.g. on road reserve)?**
How many trees or shrubs have you or a member of your household planted?	0 1-5 6-10 11-20 >20	0 1-5 6-10 11-20 >20
Please list the 3 things that you planted most of.	1. .... 2. .... 3.	1. .... 2. .... 3. ....
Approximately how much money have you spent on plants at your current address?	....	....

If you have planted outside of your property (e.g. on road reserve) please go to question 8; if you have never planted outside of your property, go to question 9 (page 3).

8. Please complete the following table about planting you have participated in *outside* of your own property (for example on public/council land such as road reserve). Please estimate to the best of your memory. If you have planted more than one site outside your property, please describe the one you have spent the most time on.

What is the size of site planted? (approx. length and breadth in meters)	................metres *by* ....metres
Where was the site in relation to your property?	☐ Directly adjacent *or* Distance .... (metres from my property)
What was the dominant ground cover before planting / preparation?	☐ Grass ☐ Scrub ☐ Bare earth ☐ Other ....
Where did you obtain the plants? Tick as many boxes as applicable.	☐ Wellington City Council Free Plants Programm ☐ Retail outlet ☐ Propagated myself ☐ Other ....
What year and season was the site planted?	
Who was involved?	☐ Just me ☐ My household ☐ Friends/family ☐ Neighbours ☐ Community group ☐ Other ....
Were any of the following procedures undertaken to prepare the site before planting?	*please describe* ☐ Weed removal .... ☐ Fertilizer .... ☐ Other ....
Has any follow-up care of the plants been carried out by yourself or others?	*please describe* ☐ Weed removal .... ☐ Fertilizer .... ☐ Mulch/weed mat .... ☐ Wind protection .... ☐ Water .... ☐ Other ....
Do you think anything has helped the survival of the plants? Please describe.	
Do you think anything has hindered the survival of the plants? Please describe.	

9. Are you: ☐ Female

☐ Male

10. How long have you lived at your current address?

....

11. Do you

☐ Own your own home
☐ Rent

12. Which age group you belong to?

☐ 18 – 25
☐ 26 – 35
☐ 36 – 50
☐ 51 – 70
☐ Over 70

13. What is your highest educational level?

☐ No formal qualifications
☐ High school qualification
☐ Vocational or Trade qualification
☐ Bachelor degree
☐ Postgraduate degree

14. Which describes your annual *household income* before tax?

☐ Less than $10,000
☐ $10,001 - 30,000
☐ $30,001 - 50,000
☐ $50,001 - 70,000
☐ $70,001 – 100,000
☐ $100,000 or more
☐ Don't know

15. Which ethnic group/s do you belong to?

☐ New Zealand European
☐ Maori
☐ Pacific Islander
☐ Asian
☐ Other. Please specify ....

16. What is your employment status?

☐ Retired
☐ Employed full-time
☐ Employed part-time
☐ Student
☐ Beneficiary/Unwaged

17. Would you be happy to be contacted to for further research into urban biodiversity?

☐ Yes ☐ No

Preferred method of contact: ☐ Email ☐ Mail ☐ Phone

Please provide contact details: .... …………………………………

Thank you very much for your help.
Please return by 31st December 2011.

Return of the survey implies you have consented to participating in this study. You will receive a letter reminding you to return the survey; if you have already returned the survey, please ignore this letter.
Surveys can be returned in the freepost envelope provided, or send to:
Freepost number 3589, School of Biological Sciences, Victoria University of Wellington, PO Box 600, Wellington 6140

APPENDIX TWO – GIS PROCEDURE FOR CALCULATING TOPEX

Topographic exposure (Topex) was originally defined as the sum of the angles to the hozrizon for the eight main compass directions. A refinement to this measure has been suggested by limiting the distance to a stated radius (topex-to-distance), which removes the effect of distant hills that are unlikely to affect wind speed greatly, and accounts for plateaus, which would otherwise be calculated similarly to hilltops[1]. For this was calculated according to the two kilometre distance limited procedure[2]. The procedure has been adapted for the Grass GIS software mapcalc function[3]. The greatest angle to any point at 100 meter intervals up to two kilometers along each of the eight main compass directions was calculated for each cell at a five meter resolution.

The angle at each direction was calculated with the formula below (example for westerly compass direction) for each cell on eight separate layers at 5 meter resolution (one for each compass direction), which were then added together to create a single layer with the summed angles:

```
max(atan(((dem[0,-20] - dem)) / (100)),atan(((dem[0,-40] - dem)) / (200)), …
atan(((dem[0,-400] - dem)) / (2000)))
```

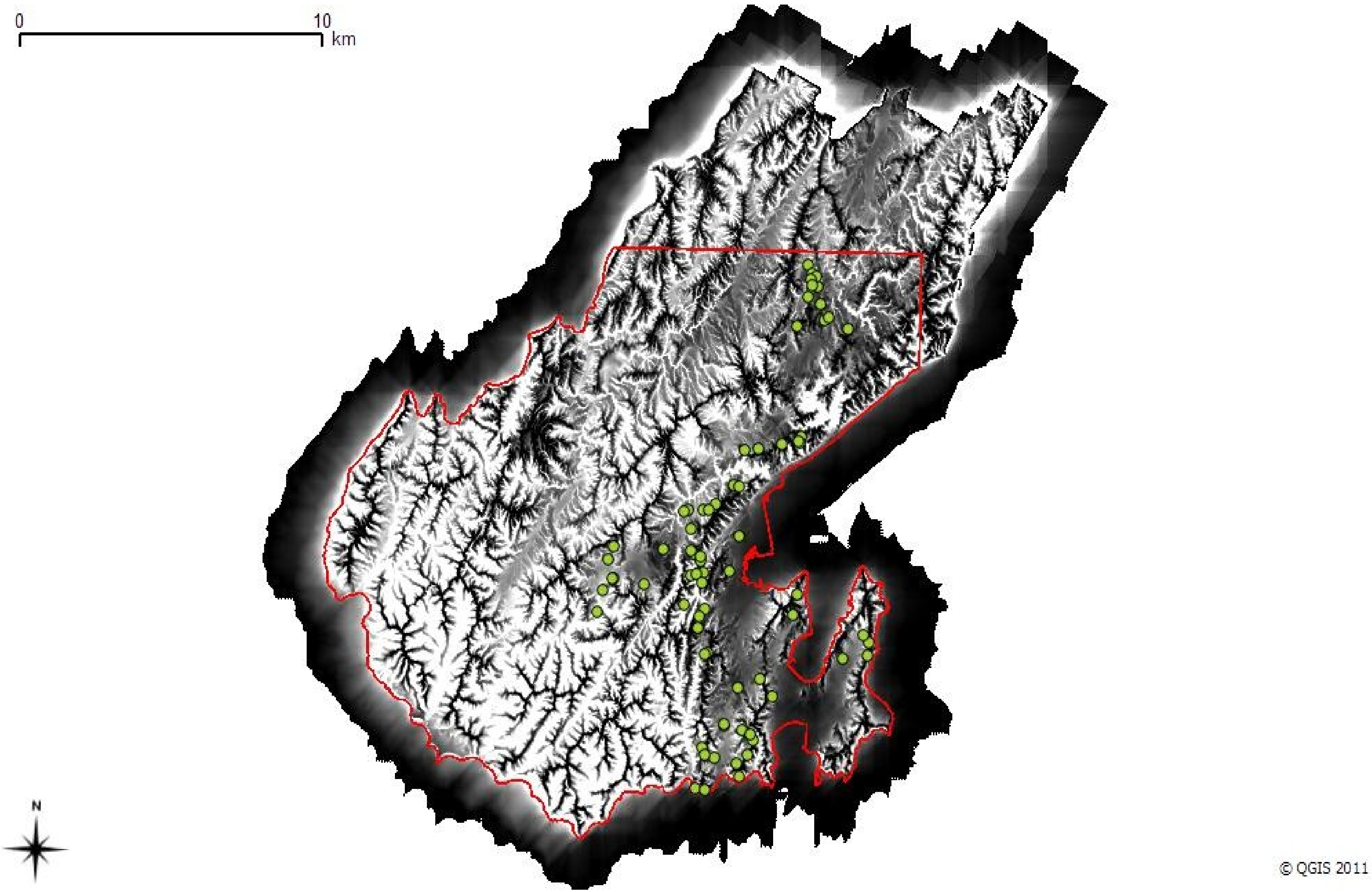
The resulting layer is displayed below, including the sites that were point sampled to obtain the topex score. Darker shading represents higher Topex score and indicates greater protection from wind.

---

[1] C. P. Quine and. M. S. White; 1998. The potential of distance-limited topex in the prediction of site windiness. *Forestry*, Vol 71, No. 4

[2] S.J. Mitchell, N. Lanquaye-Opoku, H. Modzelewski, Y. Shen, R. Stull, P. Jackson, B. Murphy, J.-C. Ruel; 2008. Comparison of wind speeds obtained using numerical weather prediction models and topographic exposure indices for predicting windthrow in mountainous terrain. *Forest Ecology and Management* 254, 193–204

[3] Popkin, J; 2011. Calculating Topographic Exposure With Grass http://jamiepopkin.blogspot.co.nz/2011/01/calculating-togographic-exposure-with.html


0
10
km
N
© QGIS 2011

APPENDIX 3 – SPECIES LIST

Species provided by Wellington City Council in any of the years surveyed.

Species	*Native (n)/ exotic (e)*
Acacia sp.	e
Agapanthus praecox	e
Aloe striata	e
Aristotelia serrata	n
Arthropodium cirratum	n
Asplenium sp.	n
Astelia fragrans	n
Austroderia fulvida syn. *Cortaderia fulvida*	n
Azalea sp.	e
Banksia integrifolia	e
Brachiglottis sp.	n
Brachyglottis monroi	n
Calistermon sp.	e
Camelia sp.	e
Carex comans	n
Carex testacea	n
Chionochloa	n
Choisya sp.	e
Cistus "Pink Ice"	e
Cistus sp.	e
Clianthus puniceus	n
Coprosma acerosa	n
Coprosma hybrid	n
Coprosma kirkii	n
Coprosma propinqua	n
Coprosma prostrata	n
Coprosma repens	n
Coprosma robusta	n
Coprosma talbrockiei	n
Coprosma williamsii	n
Cordyline australis	n
Corokia "frosted chocolate"	n
Corokia macrocarpa	n
Corokia virgata	n
Correa sp.	e
Crassula "flame"	n
Dodonaea viscosa	n
Euphorbia glauca	n
Euryops sp.	e
Fatsia japonica	e
Gazania sp	e
Genista sp.	e
Griselinia littoralis	n
Hebe sp.	n
Hedera sp.	e
Helleborus sp.	e
Hosta sp.	e
Hydrangea sp.	e
Iberis semperivens	e
Kunzea ericoides	n
Lagunaria sp.	e
Lavandula sp.	e
Leptospermum scoparum	n
Macropiper excelsum	n
Melicytus crassifolius	n
Meryta sinclairii	n
Mesembryanthemum sp.	e
Myosotidium hortensia	n
Myrsine australis	n
Olearia angustifolia	n
Olearia cheesemanii	n
Olearia paniculata	n
Ozothamnus leptophyllus	n
Penstemon sp.	e
Phormium purpurea	n
Phormium tenax	n
Pittosporum crassifolium	n
Pittosporum eugenioides	n
Pittosporum tenufolium	n
Poa cita	n
Podalyria sp.	e
Polygala sp.	e
Pseudopanax arboreus	n
Pseudopanax lessonii	n
Psoralea sp.	e
Rosmarinus officinalis	e
Sedum palmeri	e
Sophora sp.	n
Sutherlandia sp.	e
Teucrium sp.	e
Viburnum tinus	e