Effects of the Design and Management of Urban Reserves on Native Bird Communities in Auckland City, Aotearoa/New Zealand

A thesis presented in partial fulfilment of the requirements for the degree of

Master of Science

In

Conservation Biology

At Massey University, Albany, New Zealand

Lydia Clare Tyrrell

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Abstract

As the human population of Auckland City continues to increase, intense development pressure is causing an increase in housing density and widespread biodiversity loss, greatly increasing the importance of urban forest reserves as tree cover on private land is rapidly lost. The effectiveness of urban biodiversity conservation measures needs to be researched and understood if successful mitigation action is to conserve avian diversity in densifying cities. Little is known about the abundance and diversity of birds in Auckland's urban reserves, and to date there have been no published studies on the breeding of native birds in the city. More knowledge is needed about the ability of reserves to sustain resilient urban bird populations, and about which habitat and reserve design factors are most important. In this study, 28 reserves on the Auckland Isthmus divided into four categories of shape and size, were surveyed for avian abundance and diversity. A breeding survey was also conducted to record presence of breeding success for tui (*Prosthemadera novaeseelandiae*), piwakawaka/New Zealand fantail (*Rhipidura fuliginosa*) and riroriro/grey warbler (*Gerygone igata*). Native bird diversity was found to differ significantly between reserve size categories, with smaller reserves showing less diversity. Strong evidence was found of greater native bird diversity at sites with higher vegetation quality and greater vegetated area of adjacent landcover, indicating that the negative impact of smaller area could be offset by these factors. Reserve shape was not found to affect native bird communities, with small narrow reserves showing greatest native bird abundance and large narrow reserves showing the greatest native bird diversity. Successful breeding of the three study species was observed in reserves of all shape and size categories.

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Figure 1.1- Riroriro/grey warbler (*Gerygone igata*) fledgeling photographed during breeding survey

Nature and cities

As human populations have converged into burgeoning cities across the globe in recent history, the inclusion of natural habitats in increasingly urbanised landscapes has rarely been prioritised. Natural habitats are fragmented and destroyed to make way for human-made structures, and enormous worldwide urban growth has come at the cost of nature. The growth of cities causes large-scale environmental changes, from natural or agricultural landcover to anthropogenic landcover such as buildings and paved surfaces. Such major changes in landscape causes loss, alteration, and fragmentation of wildlife habitats, driving local and regional extinctions (Chace and Walsh, 2006, Clergeau et. al., 2006). Urbanisation is a rapidly growing phenomenon throughout the world, with the proportion of humans living in cities expected to rise from the current 55% to 68%, and global urban landcover projected to reach 1.7 × 10^6 km^2 by the year 2050 (United Nations, 2018, Zhou et. al., 2019). Modern city planners have begun to consider the importance of protecting urban biodiversity in some countries, but the perception of a required trade-off between land-use for human needs and biodiversity conservation can be a major constraint on investment in urban habitat creation. Scientific understanding of urban ecosystems has gained interest and progressed rapidly in recent years, but there is still much yet to be learned. Urban bird populations are often studied because birds are easily monitored and are commonly used as biodiversity indicators, so a large and growing body of knowledge regarding urban birds exists from cities around the world.

Researchers in urban and landscape ecology agree almost unanimously that larger habitat patches are more advantageous for the conservation of biodiversity, but we have only just begun to explore this assumption in relation to the effects of the heterogeneity of the urban matrix (the non-habitat area of a landscape) on populations existing in fragmented habitat in varying urban densities. The advantages of different reserve network design priorities for the purpose of biodiversity conservation have been debated in the literature for many decades, with the most studied comparison often referred to as “single large or several small” (or SLOSS), which explores whether better conservation outcomes can be achieved by retaining one large habitat patch or dividing available habitat area into many smaller patches (Patterson and Atmar, 1986, Ovaskainen, 2002).

When considering the potential advantages of a network of connected smaller habitat fragments, metapopulation theory (detailed below on page 4) is extremely applicable to urban bird populations, which usually reside in habitat fragments within the urban matrix that birds can fly between. Field-based studies have shown that native bird species can utilise residential gardens as habitat, which would suggest that habitats, although altered and disrupted, are not actually isolated in medium-density urban residential areas with large, maturely vegetated gardens; and such an urban matrix could hardly be considered inhospitable (van Heezik et. al., 2013). Metapopulation modelling, which explores the potential viability of regional populations existing as a set of sub-populations residing in connected habitat fragments, fails to account for variability in the matrix conditions, but shows that the reduction of migration costs is one of the main advantages of conserving fewer large habitat patches rather than many small patches, an advantage that is irrelevant in contexts where habitat patches are well-connected, or where the surrounding matrix may actually enable movement between habitat fragments and also provide resources (Schippers et. al., 2009). The optimal configuration of natural habitats in an urban landscape is immensely important to understand, not only for the conservation of biodiversity, but also for human well-being and connection with nature. Planning and management of urban green space should consider both social and ecological advantages while striving to optimize integration of and synergy between human and natural systems.

Urban reserves

Urban reserves (frequently referred to as greenspaces) may include grassed areas, forest, riparian zones, and wetlands, and are frequently designed to meet not only environmental but also (often predominantly) social outcomes, such as facilitating active transport, providing spaces for communities to socialise, and for sporting and other recreational activities (Shannahan et. al., 2015). The social benefits of interaction with nature are well-proven and wide-ranging, from psychological benefits such as the reduction of stress, depression and anxiety, to physical benefits such as reduction in obesity rates and improved immune system functioning (Hartig and Staats, 2006, Nutsford, et. al., 2013, Rook, 2013, Dadvand et. al., 2014, Carrus et. al. 2015, Wheeler et. al., 2015). Positive attitudes towards urban green spaces are often conditional upon perceived standard of maintenance and perceived utility to visitors (Bonnes et. al., 2011), however, the social benefits derived from ecosystem services provided by urban forest are less commonly acknowledged, and, although they have rarely been comprehensively analysed in the past, are recently becoming more well-understood (Dobbs et. al., 2011, Klimas et. al., 2016, Lafortezza et. al., 2018). As the recognition of ecosystem services such as carbon-sequestration, micro-climate regulation, stormwater attenuation and pollution reduction increases, investment in the creation of more biodiverse urban greenspace for human benefit can easily be validated (Klimas et. al., 2016, Lafortezza et. al., 2018), however, land is in premium demand in most cities, and this is a major limitation on expanding urban reserves. For this reason, urban reserves tend to be small, a factor which restricts their conservation value, especially for species which are vulnerable to disturbance.

The effect of habitat fragment size is often studied in relation to bird population density and diversity. In the urban landscape, the size of reserves has been shown to have a primary role in positively influencing bird species richness and the number of bird species nesting, thus making it a prime contributing factor of urban avian diversity (Chang and Lee, 2016, Leveau et. al., 2019). Even though many cities, especially throughout the developing world, contain only small habitat patches, the majority of studies exploring bird populations in urban habitat patches focus on larger patches (>2ha), which may be because small patches are of less conservation interest as they consist mostly or entirely of edge habitat (Collinge, 1996, Carbo-Ramirez and Zuria, 2011). Modelling has shown that patch area alone is insufficient to explain the persistence of animal populations in fragmented habitat patches (Aurambout et. al., 2005). The implausibility of expanding the area of habitat patches in growing and densifying cities necessitates more detailed and thorough understanding of how networks of existing small patches can be enhanced to maximise the conservation value that they are able provide.

There is a scarcity of studies in the literature which explore the effect of habitat fragment shape in urban landscapes (see review by Ignatieva et. al. 2011), so shape was selected along with size as a perimeter by which to categorise the reserves in this study. The effect of shape on habitat fragments has most commonly been explored in relation to ecosystem disturbance caused by edge effects such as increased exposure to the conditions (pollution, noise etc.) of the matrix. Fragments with less-round shapes have a greater proportion of area impacted by edge effects, and so species which are sensitive to edge effects are negatively impacted by a high edge-to-core ratio. Urban reserves, however, have a high level of disturbance due to human activity, and may therefore be fundamentally unsuitable for species which are most highly sensitive to disturbance (Rodriguez-Prieto et. al., 2014). Improved understanding of the effect of reserve shape could be gained by directly comparing bird populations found in reserves of different shapes, a comparison which does not appear to have been recorded in the literature so far.

Despite the severe negative impacts on bird diversity caused by the human-modified urban landscape, habitat factors have been found to be more important than landscape factors in determining bird species richness within urban reserves (Clergeau et. al., 2001), so it is essential to scrutinize factors such as vegetation composition and structure when comparing different reserve designs. Plant species richness is one such factor which is important to consider in regard to avian conservation, as native bird species richness has been found to be positively correlate with native plant biomass (Day, 1995). Herbaceous stratum height and the presence of tree cavities are further examples of vegetation factors that have been found to influence bird species richness in small urban habitat patches (Carbo-Ramirez and Zuria, 2011, Strohbach et. al., 2013, Matsuba et. al, 2016). The landcover of the area adjacent to reserves has also been found to influence bird communities in small urban reserves, with an increased proportion of vegetation cover associated with greater species richness (Mason et. al., 2007, Carbo-Ramirez and Zuria, 2011).

Although it Is generally accepted that larger habitat patches are better for retaining bird species diversity, in densely urbanised and growing cities where retrospective creation of such habit is not possible, the effective management of a network of smaller habitat patches and linear habitats connected throughout a city may be the most practical and effective focus for conservation planning and resources. If urban native bird species are managed as metapopulations this could provide an effective strategy for preserving as much as possible of the biodiversity endemic to the region that can tolerate the constant human disturbance ubiquitous throughout the urban landscape. There is emerging theory and evidence that a metapopulation existing in a heterogenous connected network of different sizes and shapes of habitat patches may perform equally to or better than a landscape with fewer larger, less connected patches (Schippers et. al., 2009). Increasing the body of knowledge regarding native bird populations in urban reserves of varying shapes and sizes will help to inform the development of the most optimal reserve design and management strategies.

Metapopulations

Metapopulation biology focusses on the dynamics of migration in fragmented habitats and the regional persistence of species with unstable local populations (Hanski, 1998). In this theoretical framework, a network of habitat patches inhabited by asynchronistically fluctuating local populations are demographically connected by migration and colonisation dynamics, which can allow for the replacement of vulnerable populations in small patches that may become extinct periodically, and would be otherwise unviable (Hanski and Gilpin, 1991). Because birds are mobile, their presence in urban reserves is not necessarily an indication of a sustainable population: instead, reserves may be acting as population sinks, with abundance maintained through immigration from source populations elsewhere (van Heezik et. al. (2008(a)). Patch occupancy may fluctuate, with factors such as temporal variation in resource abundance causing some patches to be unoccupied at times. Although it has been established that some small urban reserves have no value as habitat for bird breeding, they may provide resources valuable to the persistence of the regional population at certain times (Donnelly and Marzuff, 2004).

The equivalent of demographic stochasticity can be explained in metapopulation theory as extinction-colonisation stochasticity, whereby the balance between extinction of local populations and colonisation of unoccupied habitat patches must be above replacement to ensure long-term persistence. Each population within the metapopulation must produce at least one new population in its lifetime for the metapopulation to persist (Hanski, 1998). When a metapopulation contains a larger source population, or a higher proportion of patches in the network are sources of colonists, this can provide a constant source of new colonists which the causes the extinction rate of nearby small populations to decrease, so the metapopulation becomes more stable and resilient (Hanski, 1998). To understand its long-term viability of a metapopulation, it is therefore necessary to know what proportion of habitat patches occupied by a metapopulation are source populations.

Modelling has shown that metapopulation functioning benefits from a heterogeneous variety of well-connected sizes and shape of habitat patches (Schippers et. al., 2009). Many smaller habitat patches spread throughout a landscape reduces the metapopulation’s vulnerability to regional stochasticity (such as drought or disease) by reducing the chance that all populations will be affected at once (Hanski, 1998, Schippers et. al., 2009). The advantage of fewer large patches becomes greater when the matrix is inhospitable and connectivity is poor, reducing colonisation rates and raising migration costs, especially for species for which dispersal ability is weak (Schippers et. al., 2009). Maintaining functioning metapopulations within a connected network of smaller habitat patches is attractive from a human perspective due to requiring smaller pockets of land and distributing the social benefits of nature more widely and accessibly throughout a city, and is a much more feasible approach than retrospectively allocating large areas of land for reserves in already built-up areas. Greater understanding of the metapopulation viability of urban native bird populations in fragmented habitats is crucial to inform planning and management decisions intended to support biodiversity in cities, not just to conserve native birds and the ecosystems they are a part of, but also to benefit humans.

Greenways

As new approaches to environmental planning and management of urban areas emerge, it is imperative to understand the value, or lack thereof, that different types of greenspace can provide to biodiversity conservation in urban areas. The term greenways refers to long narrow reserves which are often created in areas that are of low value for other uses, such as disused railway lines or riparian corridors, and are usually designed to facilitate active transport and recreation (Horte and Eisenman, 2020). This makes them a socially desirable, low-cost, and convenient type of urban reserve to create, which helps to explain their growing popularity, but if their value to biodiversity conservation is less than that of other reserve types there is a danger that the proliferation of greenways could lead to a false sense of urban ecological protection or enhancement. Greenway designs frequently emphasise the social and economic (rather than the ecological) advantages of this reserve type, and they have most commonly been studied regarding their social benefits and their ability to connect urban residents with nature, with relatively few studies addressing their value to biodiversity (Horte and Eisenman, 2020).

Long, narrow habitats are usually considered as connecting elements between habitats rather than as valuable habitat per se, and studies have shown that greenways can significantly increase connectivity in a landscape (Ignatieva et. al. 2011, Carlier et.al., 2019, Horte and Eisenman, 2020). Greenways can aid connectivity by providing dispersal corridors and by permeating the urban environment, thus enabling the both the ease of movement between habitat patches and the interception of migrating individuals in the matrix which are then channelled into habitat patches (Angold et. al., 2006). Continuous canopy cover throughout the greenway is a key factor to ensure effective connectivity (Carlier et.al., 2019). If the ecological connectivity of greenways is prioritised in their design, they can facilitate metapopulation functioning, which is dependent on connections between habitat fragments, especially when the matrix is inhospitable, such as in densely urbanised areas.

Greenways can also provide habitat for urban birds, but the value of that habitat to species which are most negatively impacted by urbanisation is dependent on a range of factors (Mason et. al., 2007). Although multi-use reserves can have negative impacts on some species due to factors such as frequent human disturbance, it has been proposed that smaller reserves in urban environments are unlikely to be capable of providing habitat for highly sensitive species (Rodriguez-Prieto et. al., 2014). Greenway width significantly impacts urban-sensitive and urban-avoider species of birds, which are not found in greenways less than 50m in width, and some of which are not found in greenways less than 300m wide (Mason et. al., 2007). Bird species that are well-adapted to urban environments can make use of greenways with varying types of vegetation structure, however, they may have less dependence on corridors due to their ability to utilise the matrix as secondary habitat (Matsuba et. al., 2016). Urban-avoider bird species (sensitive to urban disturbances) are more likely to be dependent on corridors and are unlikely to benefit from greenways with sparse understory vegetation, regardless of their width (Matsuba et. al., 2016). The proportion of a greenway that is human-managed (e.g. paths and mowed grass) has been found to be the most important factor for predicting species-richness of urban-sensitive species such as forest-interior and insectivorous birds (Mason et. al., 2007). Adjacent landcover is another significant factor, with urban-sensitive species showing lower richness and abundance where adjacent landcover has a greater proportion of impervious surface and bare-earth (Mason et. al., 2007).

Reserves which are long and narrow in shape have a longer edge in relation to their core area, and this can increase the likelihood of invasion by problematic species such as nest-predators, which are found in greater abundance closer to forest edges (Keyser, 2002.). It has been established that the narrower a greenway is, the greater the abundance of mammalian nest-predators, indicating that such reserve shapes are likely to incur higher costs to maintain breeding bird populations within them as predator populations need to be controlled with greater effort (Sinclair et. al., 2005). Without sufficient predator control measures there is a risk that long narrow reserves could become population sinks due to high mortality while breeding and high nest failure.

Although it has been established that the width and vegetation of an urban greenway will largely determine its conservation value, comparisons between greenways and other habitat shapes are conspicuously absent from the literature as of yet. The ecological value of long narrow urban reserves has most commonly been studied in relation to connectivity and bird habitat provision, and this current study aims to add to the body of knowledge by assessing whether narrow reserves are of similar value for the conservation of urban birds when compared to round reserve shapes.

Urban native birds

Studies have shown that increased urbanization tends to lead to a decrease in avian species richness and an increase in avian abundance and biomass and select for omnivorous and granivorous species (Chace and Walsh, 2006, Ortega-Álvarez and MacGregor-Fors, 2009). Exotic species that are successful urban exploiters often replace native species that cannot tolerate or exploit the urban environment (Dar and Reshi, 2014, Xu et. al., 2018). It is therefore of great importance to the conservation of urban biodiversity to maintain viable regional populations of those native species which can adapt to the challenges of life in the city. Native bird populations are vital for maintaining ecosystem functioning through the provision of ecosystem services such as native plant pollination and seed dispersal (Kelly et. al., 2010). Urban birds are also important to people, and bird species richness has been found to positively relate to neighbourhood well-being (Luck et. al., 2011). A New Zealand study found that planting native tree species specifically to attract native birds to urban gardens is a common activity undertaken by more than half of the survey participants, indicating the value that people place on connecting with nature through native birds (Charles and Linklater, 2015).

Urbanisation causes immense changes in landcover as natural landcover is replaced by built and paved landcover, massively reducing, degrading and fragmenting the available habitat area for native wildlife. Landcover is the principal factor affecting bird species density (number of species per square kilometre) across cities worldwide (Aronson et. al., 2014). The level of densification can be understood as the proportion of land covered by human-modified (i.e. built and paved) surfaces, and increased urban density has been demonstrated to negatively impact native bird richness (Paz Silva et. al, 2015). The proportion of vegetated landcover has the inverse effect, showing positive effects on urban native bird populations (Paz Silva et. al, 2015, van Heezik and Adams, 2016). While native vegetation in particular is usually found to be positively associated with native bird species richness (Day, 1995, Donnelly and Marzluff, 2004, Chace and Walsh, 2006, van Heezik et. al., 2008(b)), there is also evidence of exotic trees supporting native birds in the New Zealand urban context (Gray and van Heezik, 2016).

Although still considered to be part of the urban matrix rather than valuable habitat, residential urban areas are more hospitable to native bird species when compared to densely urbanised areas which have a greater proportion of built and paved landcover. A survey of birds at commercial, industrial and residential sites Dunedin, New Zealand, found that exotic bird species were dominant in areas that were commercial, industrial, or had structurally simple gardens, but bush-dependant natives made up 24-32% of species in residential areas with larger, more structurally complex gardens, and 50-75% of species in bush fragments (van Heezik et. al., 2008(b)). Spill-over effects of habitat fragments have also been shown in urban bird studies, for example another Dunedin study showed that built areas which border on undeveloped areas have higher native bird density than other built areas that do not (van Heezik and Adams, 2016). These studies imply that in situations where habitat fragments are surrounded by a residential urban matrix, native birds from a source population can utilise and disperse through well-vegetated private gardens and are not isolated from populations in other nearby habitat patches. In contrast, where the surrounding matrix is highly urbanised, urban-sensitive bird species are less likely to cross edges (Hodgson et. al., 2007), meaning that connecting corridors between habitat fragments become much more crucial.

Native birds in urban areas are also affected on a species-specific basis by factors associated with their ability to adapt to the urban environment. Omnivorous and granivorous species are better able to utilise urban food sources and are less likely to be constrained by lack of food, while species capable of nesting on human-built structures are less likely to be constrained by lack of nesting sites (Chace and Walsh, 2006). Different species of bird also vary in vulnerability to predation, and the density of predator species has been found to be higher in urban habitats (Sorace, 2002). The biodiversity outcomes of a connected network of smaller reserves may require some trade-offs: while there are native bird species that can be found in reserves with high levels of disturbance or fragmentation, many others are not so tolerant to disturbance, and small reserves may be insufficient to support the persistence of some native bird species because of the different minimum patch size thresholds which have been identified for different species (Donnelly and Marzuff, 2004). Urban greenspaces should be planned and managed towards meeting the needs of the greatest number of native bird species possible, although some species which are most highly vulnerable to disturbance may not be able to adapt to urban habitats regardless of conservation efforts.

Reserves in Auckland

The city of Auckland has undergone and continues to undergo rapid population growth, resulting in an increase in housing density and widespread biodiversity loss (Haines, 2011; Curran-Cournane et. al., 2014; Wyse et. al., 2015). The human population of Auckland is projected to grow from 1.7 million to 2.5 million by 2040, creating immense development pressure in the region (Curran-Cournane et. al., 2014). As has been the case in many cities in around the world, Auckland's growth has historically been largely accommodated by outward expansion of residential areas, serviced mainly by motorway construction. This urban sprawl has led to concerns over environmental sustainability and inadequate or failing infrastructure, and regional policy strategy documents have been advocating strongly for urban containment and urban intensification since 1999 (Dupuis and Dixon, 2002). The urban densification process poses a threat to urban greenspace, and the provision of urban greenspace in a compact city has been described as a major challenge, with loss of private greenspace rarely offset by provision of more public greenspace (Haaland and Bosch, 2015). The increased area covered by impervious surfaces that is associated with increased urban density causes a decrease in species diversity (Dallimer et. al., 2012). Rapid urban growth is a significant factor in species extinctions as it impacts heavily on ecological processes and commonly leads to native species being replaced by exotic species (Goddard et. al., 2010).

While some suburbs contain forest reserves where biodiversity will continue to be protected as the city densifies, many parts of Auckland have only recreational green spaces providing facilities such as playing fields and playgrounds with very little biodiversity value. Tree cover (including both native and non-native trees) in Auckland City is just 6%, 63.2% of which is on private land, and only approximately 15% of that is protected, meaning that more than half of the meagre remaining tree cover is vulnerable to clearance as the city densifies (Wyse et. al., 2015). A recent survey of bird species diversity and abundance (which included passing sea birds) on nine of the volcanic reserves of the Auckland Isthmus found 38 species of bird present, 18 of which were native (Landers et. al., 2019). This study compared relative abundance of species recorded but did not analyse differences in bird communities and diversity between sites. A subsequent study of urban bird communities in Auckland city found that bird diversity decreased with increased urban density, and that within habitat fragments greater native bird richness and abundance was associated with less noise and a less densely urbanised surrounding matrix (Heggie-Gracie et. al., 2020). There seems to be a paucity of research in the literature to date regarding the avian diversity of New Zealand’s largest city, and the factors which influence it.

Increasing the area of large urban reserves in a growing and densifying city such as Auckland, with high demand and prices for land, is largely unfeasible, particularly when housing shortages exemplify an apparent direct competition between human needs and biodiversity conservation. An alternative to continuously providing quixotic recommendations for an increase in large urban reserves is to explore more practical opportunities for enhancing urban reserve networks in ways that meet social and infrastructural goals as well as biodiversity objectives. Connectivity between habitat patches will become increasingly important as the urban density of Auckland city increases. Low lying tracts of land that are natural drainage areas and flood periodically are unsuitable for built infrastructure, and therefore considered to be of low value to humans, but can potentially provide valuable connectivity and habitat for urban biodiversity, while simultaneously meeting human goals such as flood mitigation, carbon sequestration, and the provision of walkways and cycleways. If the creation of more greenways connecting small habitat patches all over the city were to be considered as a solution for maintaining metapopulations of native birds throughout the city, evidence that small and narrow reserves are capable of supporting native bird breeding would be vital to justify this approach. This study aims to contribute towards filling a current knowledge gap by comparing the habitat value of different reserve shapes and sizes for maintaining metapopulations of urban birds.

Research Aims

- To explore the effect of reserve size and shape on avian diversity, abundance, and native bird breeding in urban reserves on the Auckland Isthmus
- To glean insight into what habitat factors may be most important for diversity and breeding of urban native birds
- To elucidate whether a network of small and narrow reserves could provide value as habitat for birds as well as connecting larger habitat patches

Chapter 2: Survey of study sites and associated factors

Image 2.1- Excellent field assistant recording measurements during vegetation survey in a reserve with mature forest, Dingle Dell, Auckland, New Zealand

Introduction

In order to compare bird populations in urban reserves of different shape and size, it is necessary to ensure that the different size/shape reserve categories do not significantly differ in factors which are likely to be important in influencing urban bird communities. The selected factors to measure for this purpose were initially: groundcover, understory complexity, trunk circumference, tree density, tree species diversity, reserve landcover, (these vegetation factors were later combined into a composite vegetation quality score), adjacent landcover and distance to closest forest patch.

It was essential to conduct a detailed survey of the vegetation in each study site because bird communities can be positively affected by increased vegetation complexity in their habitats, and significant variation in vegetation quality between site categories would confound the comparison of reserve shape and size. Vegetation factors such as the proportion of tree cover, vegetation diversity, the secession stage of the forest and tree density have been shown to affect the avian species richness and abundance of a habitat patch. (Dobbs et. al., 2011, Chang and Lee, 2016). Urban-sensitive bird species have been found to have lower species richness and abundance in urban reserves with less understory density, so this was also considered to be a meaningful factor to explore (Matsuba et. al., 2016). A vegetation quality score (VQS) was created to combine the most important vegetation factors and test whether the different shape/size reserve categories are of comparable quality. Tree species diversity is often positively correlated with reserve size, and positively influences bird species richness (Donnelly an Marzuff, 2004, Chang and Lee, 2016), so this was considered to be an influential factor to test independently, as well as being included in the VQS. The landcover of reserves was also tested independently as well as being included in the overall VQS because the proportion of reserves covered in pavement and mowed grass has previously been found to be the most important factor influencing abundance and richness of urban-sensitive bird species in narrow urban reserves (Mason et. al.,2007).

In a fragmented landscape such as the urban environment, regional populations may be more likely to function as metapopulations, and small reserves may function as important habitats that support such metapopulations. For metapopulations to persist in a landscape of fragmented habitats, connectivity between patches is necessary (Hanski and Gilpin, 1991). The distance to the closest forest patch was measured for each site because habitat patch connectivity has been found to have a significant influence on bird species richness in urban reserves, and connectivity combined with patch area has been shown to influence overall bird abundance (Shanahan et. al., 2011). Adjacent landcover was an important related factor to measure because the density of built areas in the surrounding matrix of reserves may also affect the patch connectivity, with the effect varied for different species, for example a study from New South Wales in Australia found that nectivorous bird species were more likely to move between patches over a matrix of high-density housing, while insectivorous species tended to cross the through areas of low density housing (Hodgson et. al., 2007). Furthermore, the landcover of the area adjacent to narrow urban reserves has been shown to influence the richness and abundance of urban-sensitive bird species (Mason et. al., 2007).

Two habitat factors which were deemed to be particularly important to avian breeding activity as well as avian diversity were predator control and the presence of freshwater. Predation is a major cause of nest failure and juvenile mortality in urban habitats (van Heezik et. al., 2008(a)). Decreased distance to freshwater has previously been found to be associated with an increase in bird species richness in urban greenspaces (Zivanovic and Luck, 2016). We also considered a source of freshwater to be a factor that could influence whether birds establish breeding territories in a habitat patch or not because time and energy must be prioritised towards feeding the incubating parent and the offspring during the breeding period, and flying some distance to access freshwater detracts from this prime objective.

For my study covered in this thesis, I have selected 28 urban reserves of different area and shape which are situated on the Auckland Isthmus. Care has been taken to choose reserves with similar vegetation conditions in order to avoid confounding vegetation factors when exploring the size and shape effect. It is also important to ensure that there is no significant difference in both the connectivity and the adjacent landcover of the reserve shape/size categories if bird populations are to be accurately compared. In this chapter we analyse the plant and landcover parameters of these reserves in order to determine whether such confounding factors are present.

Research Questions

1. Do tree cover, tree diversity and vegetation quality score differ between the different shape and size categories?
2. Does the adjacent landcover score differ between the different shape and size categories?
3. Does the distance to closest forest patch differ between different shape and size categories?
4. Does the presence of predator control measures and freshwater differ between the different shape and size categories?

Methodology and Results

Study sites

The vegetation, landcover, bird populations and the breeding of three native species were surveyed in 28 reserves on the Auckland Isthmus (see **figure 2.2** below). The reserves surveyed for this study have areas ranging between 5,593 m² and 186,529 m², are of varying shapes, and all located within a matrix predominantly consisting of medium-density housing. All of the study sites are multi-use reserves with walkways running through them and varying amounts of grassed area.

Figure 2.2- Map of Auckland Isthmus with study sites outlined in yellow (from Google Earth Pro)

Vegetation Survey

We used footpaths in each reserve as sampling transects when possible (see **figure 2.3,** page 19). In reserves where paths were not present in a large portion of the reserve, we set up transects. The vegetation was sampled every 20 m along the path or transect using Point Centre Quarter (PCQ) survey method (see Bryant et. al., 2004). At each 20m point, a 2m line was measured perpendicular to the path/transect, and a sample point marked, on alternating sides where possible. Where this was not possible due to physical obstructions such as water or very steep banks, the survey point was marked on the same side as the previous point. At each sample point a cross was marked out on the ground, and in each quarter the tree closest to the centre but within 10 m of it which had a trunk circumference greater than 15 cm was sampled. If there was no tree with a circumference greater than 15 cm within 10 m of the point, then it was recorded as “no tree”. The parameters measure included the species, the distance from the centre point to the tree and the trunk circumference at breast height. Each plant under 2 m tall that was touching the line between the centre point and the tree sampled in each quarter was identified and categorized into height categories of 0-50 cm, 50cm-100 cm, and 100-200 cm, to give an indication of the understory complexity. A score for groundcover: 0 = nothing, 1 = grass, 2 = thick cover of leaf litter/seedlings/weeds was given for each line.

Figure 2.3: Dingle Dell Reserve, Auckland, New Zealand, showing transect line in red (from Google Earth Pro)

Shape/size reserve categories

Google Earth Pro was used to measure the area, perimeter, length, and width of each site. The length was recorded as the longest straight line between two edges of the reserve and the width was measured as a line perpendicular to the length at the widest point of the reserve.

The width divided by the length of the 28 study sites was used for a measurement of reserve shape. The 14 sites above the median value for length/width were classified as round (or R) and the sites below the median classified as narrow (or N). Within the narrow category, the 7 sites with an area above the median were categorised as narrow large (NL), and the sites with an area below the median as narrow small (NS). The same was then done for the round category, which was dived into round large (RL) and round small (RS) categories.

We chose to use width/length as a determinant for shape rather than an area/perimeter ratio due to the often indented edges of reserves that were basically round in shape skewing this data in a way that was not practically useful when comparing round to narrow reserve shapes.

Permanova was used to test the differences of habitat features between the treatment (NL, NS, RL, RS). The distance to the nearest forest is log (V+1) transformed before the data was normalised. The Euclidian distance is used to build the resemblance table. Unrestricted permutation of raw data was performed for the single factor permanova.

Table 2.1- *PERMANOVA* result

PERMANOVA table of results

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Sh	3	20.538	6.8459	1.1615	0.315	999
Res	24	141.46	5.8943			
Total	27	162				

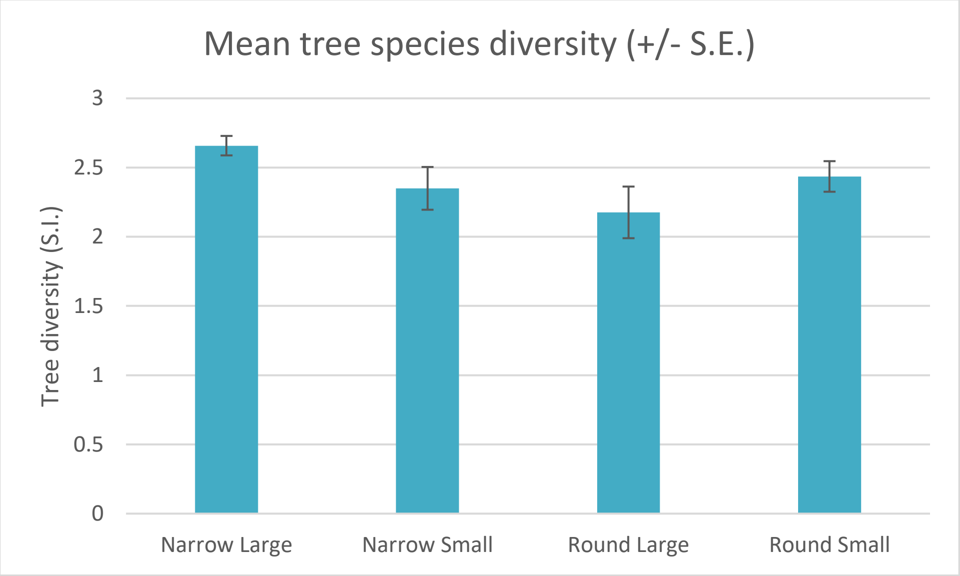
There is no evidence the parameters investigated differ between the reserve shape and size categories. Further exploration of each of the measured factors follows below.

Tree species diversity

Shannon index was used to measure the diversity of plants using the equation:

$$H = \sum_{i=1}^{S} - (P_i * \ln P_i)$$

Figure 2.4- Mean tree species diversity for reserve shape/size categories (+/- S.E.)



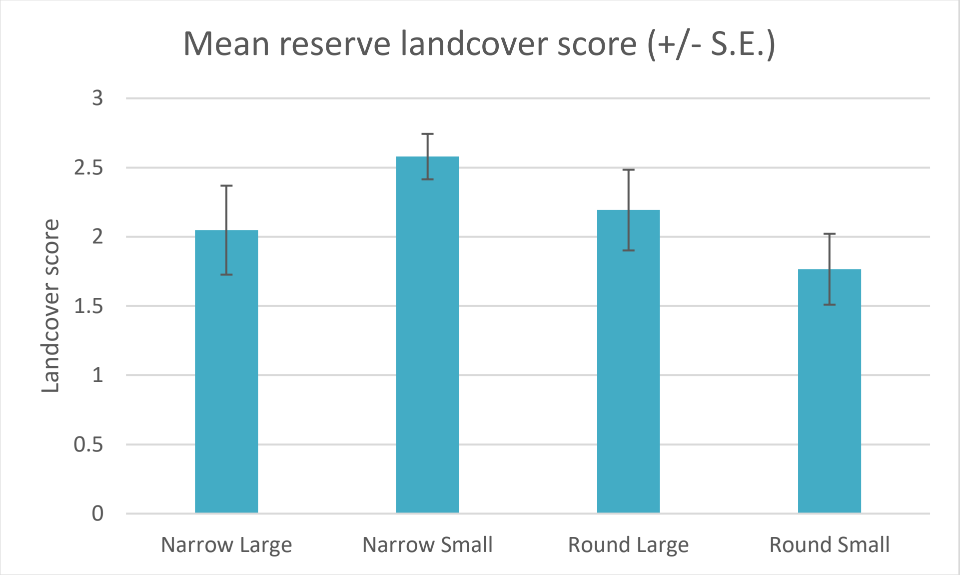
There was no significant difference (ANOVA, *p*= 0.124) in mean tree species diversity between the reserve categories, and a Levene's test showed no evidence of problems with variance assumption (*p*= 0.131).

Reserve landcover score (RLS)

For each site, areas of water (measuring patches larger than 100 m2) were measured and added together. The same was then done for areas of tree covered area, grass covered area and built-up area. Paths were measured as part of the built-up area if they were more than 2 m in width. A reserve landcover score (RLS) was generated with the intention of representing the habitat value of each landcover type, the assumption being that tree cover is the most valuable, grass and water have some value, and built areas have a negative habitat value. The proportion of the total reserve area covered by each landcover type was calculated and then the following equation was calculated whereby: T= proportion of tree cover, G= proportion of grass, W= proportion of water and B= proportion of built area:

RLS= (2 x T) + G + W + (-1 x B)

Figure 2.5- Mean reserve landcover score for reserve shape/size categories (+/- S.E.)



Analysis showed no significant difference (ANOVA, *p*= 0.207) in LS between the reserve categories and no evidence of problems with variance assumption (Levene-test, *p*= 0.157).

Vegetation Data Analysis

Groundcover scores for each sample point were added together and then averaged to give a groundcover score for the whole site. The average trunk circumference for each site were calculated as an indication of the maturity of the forest, and the mean distance between the tree and centre point at each site was used as a measure of tree density.

To calculate an understory complexity score (UCS) for each site, we first assigned a understory score (US) of 0-3 depending on the presence or absence of tree and shrub seedlings, climbers, ferns and grasses in each height category a score of 0-3: 0 for nothing in any category, 1 for presence in 0-0.5 m, a 2 for presence in 0.5-1 m, and a 3 for presence in 1-2 m. A Shannon index (H) was then calculated for the plant diversity in each of the height categories and the understory complexity score was calculated as:

UCS = US + H

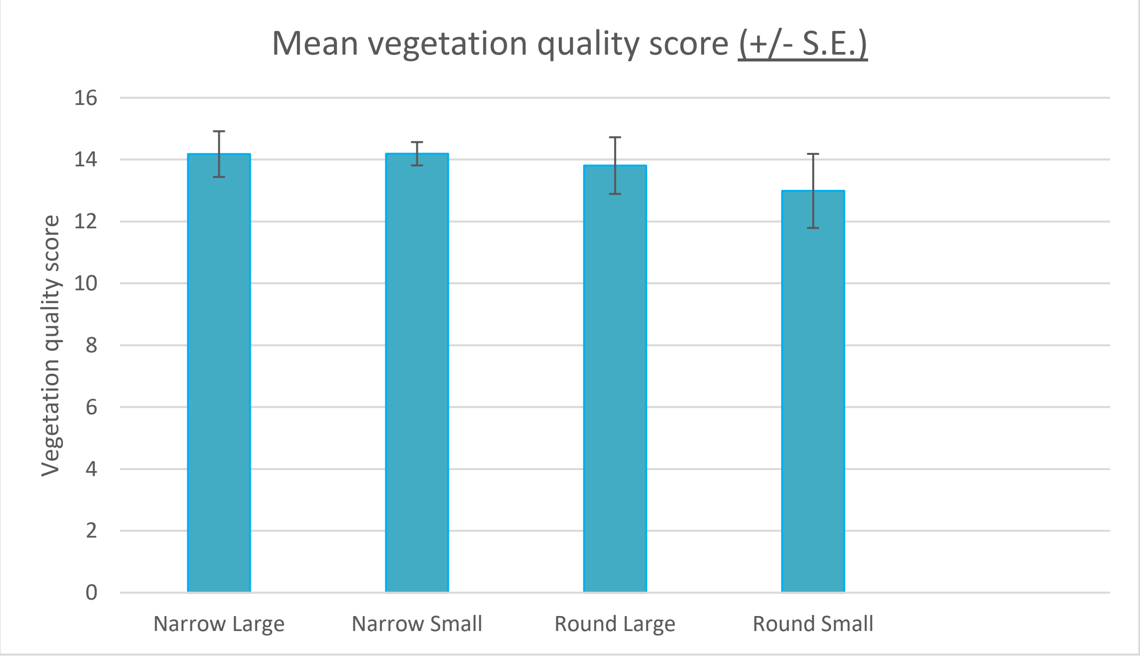
Vegetation quality score

The trunk circumference and the tree density were found to be co-related variables, so trunk circumference was selected as the indicator of forest maturity. Understory complexity score and groundcover score were also found to be co-related variables, and understory complexity score was selected to use for further analysis because it is known to be a factor which can influence bird diversity.

The mean trunk circumference for each site was given a ranked score between 1 and 5. A score of 1 was given to sites with an average trunk circumference of 50-100 cm, a 2 was given for 101-150 cm, a 3 for 151-200 cm, a 4 for 201-250 cm and a 5 for 251 cm and above. The sum of landcover score (LS), Shannon diversity index for tree species (H), understory complexity score (UCS) and trunk circumference score (TCS) was used as a vegetation quality score (VQS). Although tree species diversity and landcover score were included in the vegetation quality score, they were also tested as separate variables because they have been considered to be important variables effecting bird populations in previous studies, and also they were not strongly related to any other variables.

VQS = LS + H + UCS +TCS

Figure 2.6- Mean vegetation quality score for reserve shape/size categories (+/- S.E.)



Statistical analysis found no significant difference (ANOVA, *p*= 0.734) in VQS between treatments and no evidence of problems with variance assumption (Levene-test, *p*= 0.499).

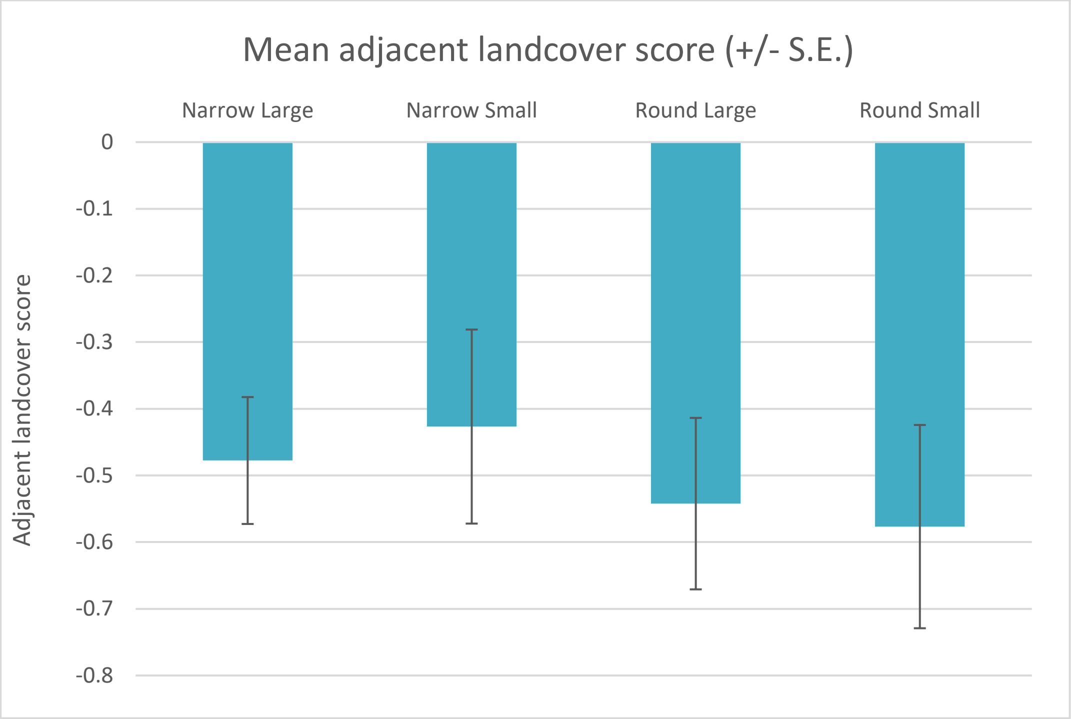
Adjacent landcover score

The adjacent land surrounding each reserve within a 200 m radius around the reserve was measured using google earth pro (see **figure 2.4**), and the area of water, tree, grass and built up areas greater than 1000m2 were measured and added together for each landcover category. The same method used to generate landcover score was then applied to generate an adjacent landcover score (ALS) score: T= proportion of tree cover, G= proportion of grass, W= proportion of water and B= proportion of built area:

ALS= (T x 2) + G + W + (B x -1)

Figure 2.7: Roy Clements Treeway, Auckland, NZ, showing adjacent landcover area within blue line (from Google Earth Pro)

Figure 2.8- Mean adjacent landcover score for reserve shape/size categories (+/- S.E.)

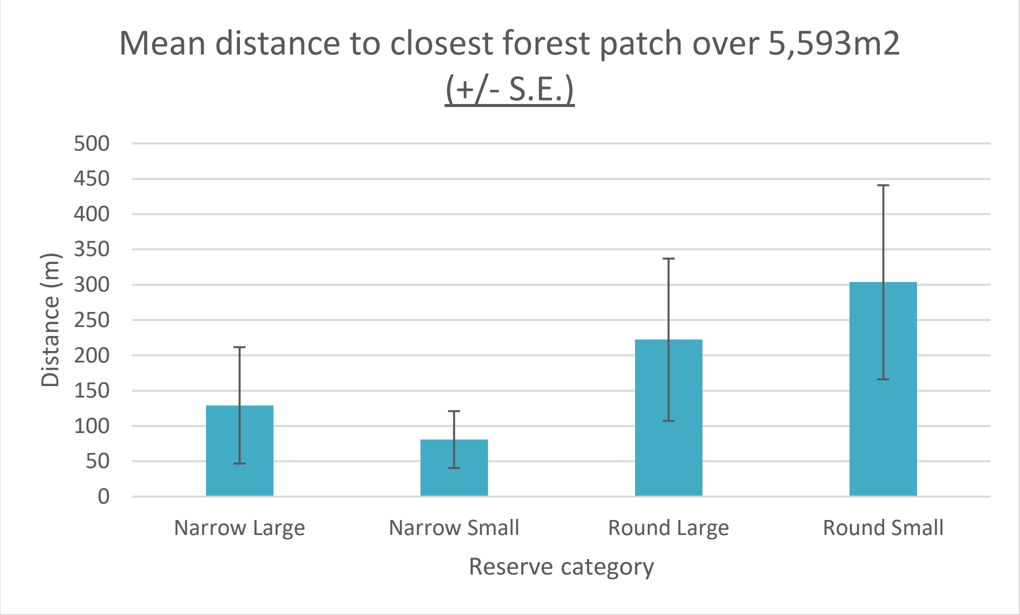


There is no significant difference (ANOVA, *p*= 0.207) in mean adjacent landcover score between the different treatments and no evidence of problems with variance assumption (levene-test, *p*= 0.865).

Distance to closest forest patch

Distance to closest forest patch greater than 5,593 m^2 (the area of the smallest study site) was measured using google earth pro, and if the closest patch was connected to the study site by continuous vegetation cover the distance was recorded as 0. An arbitrary cut-off for habitat patch area was chosen because different species of birds have different territory sizes, and a minimum habitat patch size for bird species found in Auckland is currently unknown.

Figure 2.9- Mean distance to closest forest patch for reserve shape/size categories (+/- S.E.)

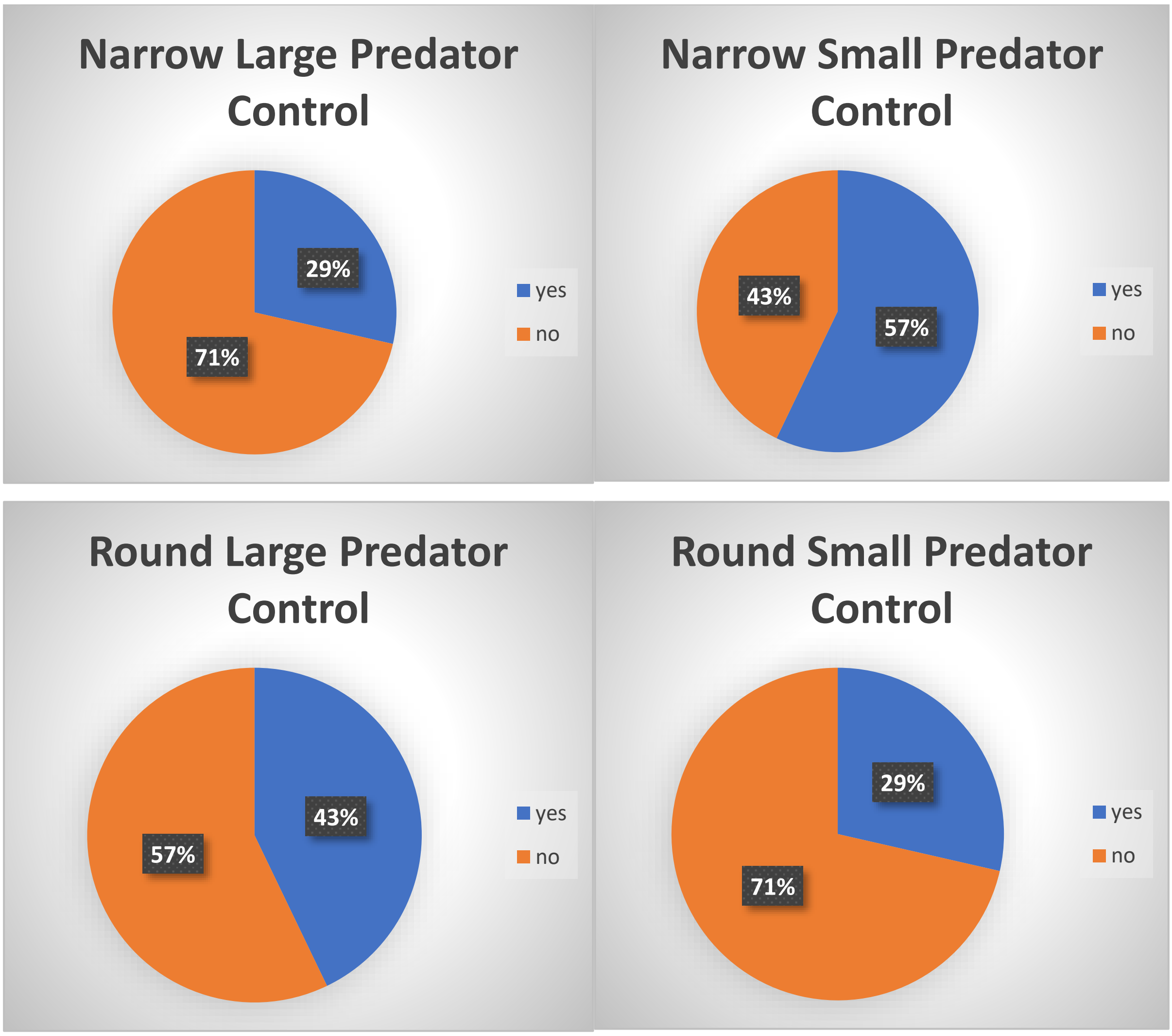


There was no evidence of a significant difference in distance to closest forest patch between the treatments. Kruskal-Wallis Rank Sum Test: Chi-square value = 1.8902, df=3; *p*= 0.596.

Factors which could affect bird breeding

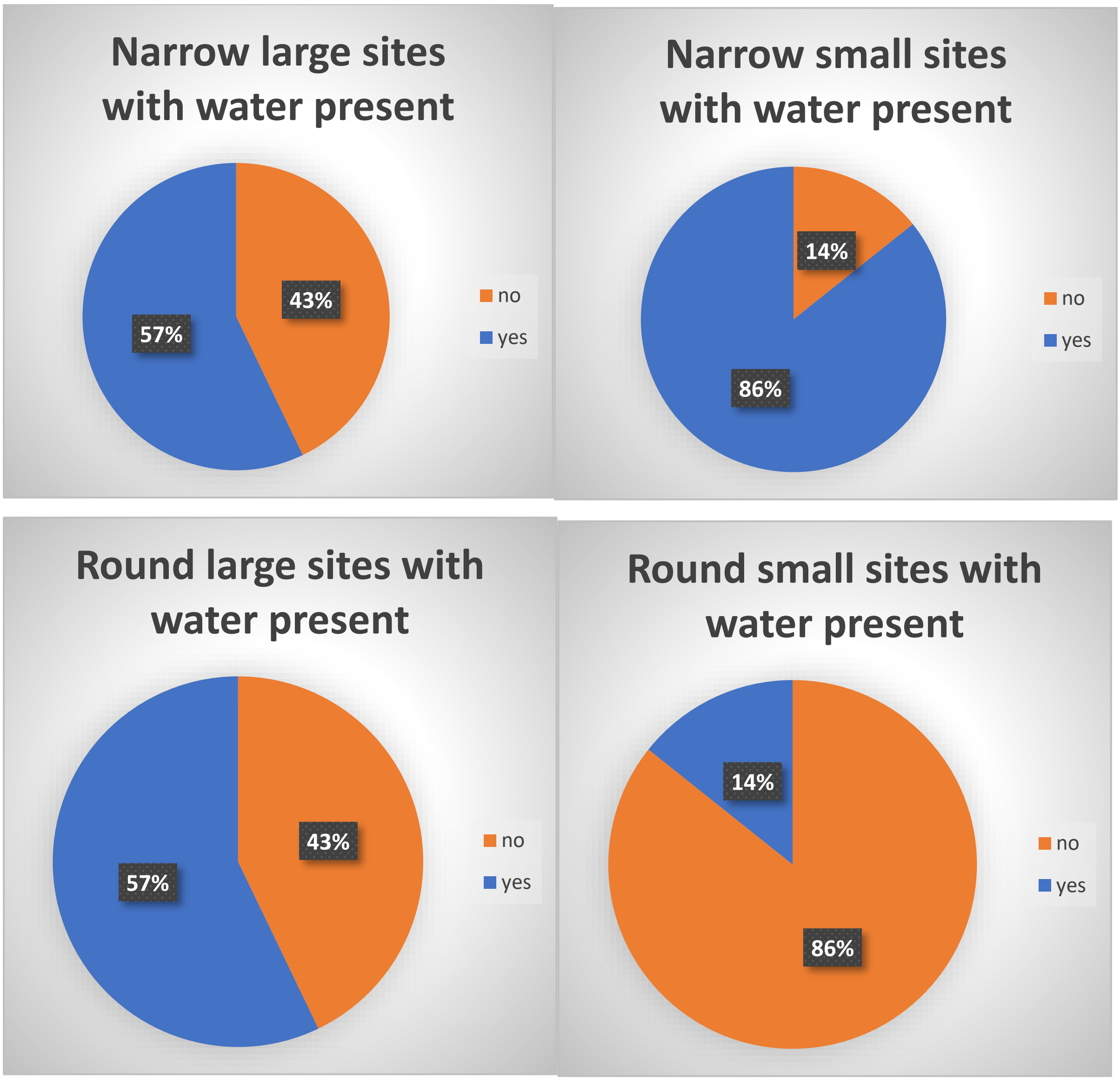
The presence of predator control measures and a source of freshwater were recorded during surveys of each site as these factors may influence the suitability of habitat for successful bird breeding. Auckland Council was contacted to confirm the presence of predator control operations at the sites where they had been observed, and the lack of predator control at sites where none had been seen. The proportion of sites in each shape/size category where predator control was seen are indicated in the chart below (**figure 2.10**, page 27) as the "yes" proportion, and the proportion of sites where predator control was not seen were marked as "no". The same applies to the chart (**figure 2.11**, page 28) displaying the proportion of sites in each shape/size category where a freshwater source was present.

Figure 2.10- Comparison of the proportion of each reserve category with predator control



The number of reserves in the narrow small category with predator control (yes) was highest at 57.14% and was double the proportion of both narrow large and round small categories which had 28.57%. Statistical analysis was unfortunately not possible for this data because of the small data set and the binary nature of the data.

Figure 2.11- Comparison of the proportion of each reserve category with freshwater present



There was a large difference in the proportion of sites with freshwater present (yes) between the size/shape categories and statistical analysis found weak evidence of an association between water presence and categories (Fisher’s exact test, *p*= 0.071). The narrow small category had 85.71% with freshwater present, four times the round small category which had only 14.3% with freshwater present.

Discussion

The initial permanova test showed no statistical differences between reserve shape and size categories for the factors tested, and then further analysis confirmed this for each of the separate factors tested. Because there was no statistically significant difference found between the shape/size categories for any of the variables considered as important influences of bird diversity and abundance, the bird populations of the different reserve categories can be compared without confounding factors influencing the difference in bird data between categories.

A review of previous studies has found that larger habitat patches have greater plant diversity, which is thought to be the reason that they are able to support greater bird species diversity (Donnelly and Marzuff, 2004), however, our reserve shape/size categories showed no significant difference in tree species diversity. The landcover of reserves, another factor which has been shown to be important in influencing bird diversity (Mason et. al.,2007), was also found not to be significantly different between the different shape/size reserve categories. The VQS was used to test the combined factors of vegetation diversity, coverage, complexity, and maturity, as a composite variable of vegetation factors known to influence bird diversity, and no significant difference was found between the different reserve categories. The landcover of the area adjacent to urban reserves has previously been found to influence bird diversity (Mason et. al., 2007), and when the adjacent landcover of the four reserve categories were compared no significant difference was found. Habitat patch connectivity has been shown to significantly influence bird species richness in urban reserves (Shanahan et. al., 2011), and when the average distance to the closest forest patch for each reserve category was compared, no significant difference was found.

Of the two factors which we considered likely to affect bird breeding in particular, larger differences between categories were seen, and, although the binary data was difficult to analyse with a small sample size (28 sites) and predator control was not able to be tested for significance, presence of water showed a weak difference. These two factors must be scrutinized when comparing the different reserve categories in analysing the data from the bird breeding survey.

Accuracy of the adjacent landcover score could have been greater if the different landcover types had been measured at a finer scale as was done for the landcover measurements within sites, but time constraints did not allow for this. Accuracy of the comparison of connectivity measurements for the sites in this study could have been improved if a minimum habitat patch size for urban birds species in Auckland was known, as this could have been used as a minimum patch size rather than the area of the smallest study site, which was the arbitrary value selected.

Conclusion

There was no significant difference detected between the shape/size reserve categories for any of the variables known to affect bird diversity and abundance that were analysed. No significant difference was found between categories for tree diversity, reserve landcover, vegetation quality score, adjacent landcover score or distance to closest forest patch. This means that the effect of the shape and size of reserves on urban bird populations can be tested independently of potentially confounding variables.

Chapter 3: Effect of urban reserve shape and size on bird abundance and diversity

Figure 3.1- Silver-eye (*Zosterops lateralis*) juveniles photographed during a bird count

Introduction

In the urban environment, increased anthropogenic disturbances and the introduction of exotic species cause changes to animal community assemblages. Some species (often referred to as urban exploiters or synanthropic species) can adapt to human activity and disturbance and are abundant in urban areas; however many other species (urban avoiders or non-synanthropic species) are less tolerant of anthropogenic disturbance and their populations decrease in urban areas or become locally extinct (Dar and Reshi, 2014, Xu et. al., 2018). In avian communities, urbanisation tends to select for omnivorous and granivorous species that can utilise urban resources, i.e. eating human food waste and nesting on human-made structures (Carbo-Ramirez and Zuria, 2011). The exclusion of species that are vulnerable to anthropogenic disturbance or which occupy more specialized niches leads to community assemblages becoming more structurally simple (Chace and Walsh, 2006, Clergeau et. al., 2006). The fitness benefit of being able to use urban resources results in successful urban exploiters such as the globally ubiquitous rock pigeon (*Columba livia)* and house sparrow (*Passer domesticus)* dominating urban environments and replacing non-synanthropic native species, causing an increase in avian biomass and a decrease in both functional and species diversity, leading to the global biotic homogenisation of urban environments (Chace and Walsh, 2006, Clergeau et. al., 2006, Aronson et. al., 2014, Dar and Reshi, 2014, Paz Silva et. al, 2015). Although a few species can sustain their populations in urban areas that lack natural habitats, most urban avian species require natural habitats such as urban reserves and green spaces in order to persist in an urban environment. Therefore, urban biodiversity loss can be mitigated by the preservation or creation of natural habitats within the urban matrix. Urban reserves, or urban green spaces, are created and maintained for social reasons such as aesthetic enhancement of the urban environment, for infrastructural purposes such as transport and drainage management, and for ecological purposes such as providing habitat for native birds. Factors that influence the success of urban reserves in preserving or enhancing biodiversity in cities can be grouped into habitat factors and landscape factors.

Habitat factors

Adjacent landcover

The landcover of the matrix in which urban reserves exist is an extremely important landscape factor, and at a closer scale the qualities of the area directly surrounding a reserve is a highly influential habitat factor, as a high proportion of natural landcover in the adjacent area can increase the effective habitat size of the reserve, decrease edge effect, and enhance connectivity. The proportion of land covered by buildings in areas adjacent to small urban habitat patches can have a negative impact on avian species richness (Mason et. al., 2007, Carbo-Ramirez and Zuria, 2011, Nielson et. al., 2014, Heggie-Gracie et. al., 2020). Private gardens in residential urban areas may provide important food resources and nesting sites for native birds and increase greenness of the surrounding matrix of the reserve (Gray and van Heezik, 2016). A reduction in contrast between the habitat and the surrounding matrix can greatly reduce edge effect, and so is particularly important to consider when investigating the effects of habitat shape (Ewers and Didham, 2006). Connectivity is also affected by the quality of the matrix because an inhospitable matrix can prevent or inhibit the movement of birds across habitat edges (Hodgson et. al., 2007). Adjacent landcover of reserves is an important variable to explore in relation to native bird diversity because it has been established that this is a significant factor influencing native bird diversity in the urban New Zealand context (van Heezik and Adams, 2016, Heggie-Gracie et. al., 2020).

Vegetation

Bird communities are well known to be influenced by vegetation structure and composition (see review by Chace and Walsh, 2006). The proportion of urban reserves covered by vegetation is an important factor in predicting the avian diversity found within them, but the quality of vegetation matters as well as the quantity (Rodrigues et. al., 2018). Higher vegetation complexity provides superior foraging and nesting opportunities for birds and has a positive influence on the diversity of forest birds in urban habitat patches (Kang et. al., 2015). Measuring vegetation complexity requires examining the structure, maturity and diversity of vegetated areas, factors which have also been explored separately in the urban context. Increased vertical vegetation structure has been shown to support increased species richness and diversity of urban birds, and herbaceous stratum height is particularly important for ground-foraging and shrub-nesting species (Donnelly and Marzluff, 2004, Carbo-Ramirez and Zuria, 2011). The maturity of trees in urban reserves is also an important component of vegetation complexity to consider, as a greater number of woody trees in a reserve significantly increases bird species richness, and the presence of trees with cavities has also been shown to be beneficial to bird species richness in urban green space (Strobach et. al., 2013, Aida et. al, 2016). Urban native bird species richness increases in relation to an increase in plant species diversity, and positively responds to the preservation of native vegetation characteristics in particular (Donnelly and Marzluff, 2004, Chace and Walsh, 2006). In the New Zealand urban context, increased vegetation complexity has been shown to positively influence native bird diversity, and an increase in proportion of native vegetation cover significantly increases native bird diversity and also increases the proportion of native to exotic birds (van Heezik et. al., 2008(b), Heggie-Gracie et. al., 2020).

Water

Although many bird species are able to fly beyond reserve perimeters to seek water, this ability can be reduced according to the weaker flying ability of some species and the level of aversion to crossing a potentially inhospitable urban matrix if building density is high. A water source within the reserve eliminates these potential issues and can provide greater food foraging opportunities for some species as well. This advantage has been confirmed in a study conducted in Sydney, Australia, which found that bird species richness in urban greenspaces increases with decreased distance to fresh water (Zivanovic and Luck, 2016).

Landscape factors

Landcover

Across cities worldwide, landcover has been found to be the most important factor affecting bird species density (number of species per square kilometre) (Aronson et. al., 2014). Landcover has a major effect on species richness and is heterogenous to varying degrees in different cities which have differing levels of urban densification. Urban densification causes a greater proportion of land to be covered by buildings and impervious surfaces which reduces the availability of suitable habitat for non-synathropic species, and many studies have shown that this negatively affects total avian species richness (Chace and Walsh, 2006, Ortega-Alvarez and MacGregor-Fors, 2009, Fontana et. al., 2011, Aronson et. al., 2014). Native bird richness has also been found to respond negatively to impervious surface cover and increased building density (Paz Silva et. al, 2015). In the New Zealand context, increased building density has been shown to have negative impacts on native bird communities and to have neutral or positive effects on exotic species (van Heezik and Adams, 2016, Heggie-Gracie et. al., 2020).

The landcover of suburban areas in cities generally consist of a heterogenous mix of buildings, impervious (paved) surfaces, private gardens and public green space. Public greenspace can include areas of widely differing vegetative diversity, from grassed areas such as sports fields to reserves containing natural habitats such as forest patches and wetlands. Private gardens in cities can support plant and bird diversity to varying degrees, as clearly illustrated in a study of gardens in Dunedin, New Zealand, which showed that the size of the vegetated area of properties was most important factor in explaining plant and bird diversity found within them (van Heezik et. al., 2013). Many studies in different parts of the world have shown that urban avian species richness is positively influenced by increased area of vegetation cover (Fontana et. al., 2011, Sandstrom et. al., 2006, van Heezik et. al. 2013, Paz Silva et. al, 2015, Heggie-Gracie et. al., 2020). Native bird richness has also been shown to respond positively to increased vegetation cover (Paz Silva et. al, 2015, Heggie-Gracie et. al., 2020). At a landscape scale, it can generally be concluded that the relative proportion of built and paved area in relation to vegetated area will strongly influence both the overall and native bird species richness of a city.

Connectivity

Habitat patches that are preserved or created within the urban matrix are isolated from one another to varying degrees, which can inhibit dispersal by disabling animal movements, thus reducing gene flows. The isolation of a small population can lead to inbreeding and population decline. Small habitat patches are often unable to sustain populations in the long-term without connections to other habitat patches that allow a network of small populations to function as a metapopulation, allowing geneflow between patches and population recovery after local extinctions, thus maintaining viable populations in fragmented landscape (Hanski and Gilpin, 1991). The connectivity of urban reserves has been widely recognized as being of vital importance to the persistence of metapopulations within a highly modified matrix (see review by Ignatieva et. al. 2011). While the positive effect of connectivity is ecologically complex because different species are differently affected by isolation, the most important functions of connectivity at the landscape level are dispersal and colonization between patches, and a significant reduction in ecosystem services for biodiversity has been found to be caused by the destruction of connectivity (Ng et. al., 2013).

Connectivity between habitat patches can be provided through continuous corridors of vegetation, often referred to as greenways, or with small pockets of habitat acting as steppingstones between larger patches. Whether it is more useful to enhance connectivity through the provision of linking corridors or steppingstones between larger habitat patches may depend on the species which conservation efforts are aimed at preserving (Angold et. al, 2006, Ignatieva et al., 2011, Qi et. al., 2017). Species that can fly, including most bird species, may be able to utilise steppingstones to move between habitat patches, while mammals generally make better use of corridors (Angold et. al, 2006). It has been established that avian diversity in urban reserves is positively influenced by greater connectivity (i.e. Shannahan et. al., 2011, Kang et. al., 2015), but connectivity has been found to be less important than park size in influencing number of species at the community level (Chang and Lee, 2016). Habitat patches functioning as steppingstones within an urban matrix can benefit avian diversity by enhancing connectivity for some bird species, although the building density of the matrix will affect which species of bird are likely to cross it (Hogson et. al., 2007). Corridors of forest linking habitat patches can provide the extra benefits of extending habitat for birds, eliminating the need to cross over urban matrix to move between habitats, and enhancing ecological functioning (Horak, 2016). It has also been proposed that for some species, corridors could benefit dispersal by intercepting organisms dispersing through the matrix and channelling them into habitat patches (Angold et. al., 2006, Horak, 2016). Corridors can potentially have the detrimental effect of increasing mammalian predator presence, a consequence which is likely to be particularly harmful in the New Zealand context, where bird species have evolved in the absence of, and so are highly vulnerable to, introduced mammalian predators.

Shape

The impact of habitat fragment shape on any given species depends on how the species responds to various edge effects, as habitat fragments which are more long and narrow have a greater edge (or perimeter) length in relation to area, and are therefore more exposed to edge effects. When two adjacent ecosystems are separated by an abrupt transition, the effects of the interaction between the two systems are known as edge effects, and can cause changes in the biotic and abiotic conditions of an ecosystem which affect the organisms within it (Murcia, 1995). In urban environments, habitat fragments are surrounded by a highly contrasting environment of mostly built and human-modified areas (the urban matrix), and the transition between habitat fragments and the surrounding urban matrix can be abrupt, causing significant edge effects. There are three categories into which edge effects can be grouped: 1.) abiotic effects changing the physical environment, for example increased exposure to weather, 2.) direct biological effects caused by these physical changes, and 3.) indirect biological effects whereby direct biological effects lead to changes in species interactions (Murcia, 1995).

Human-caused disturbance creates major abiotic and biotic edge effects in urban habitat fragments and has been shown to be an important factor negatively influencing avian abundance and species richness in small urban forest patches (Kang, et. al, 2015). Human activity such as the creation of noise can interfere with bird behaviour, and urban reserves with minimal core area are unlikely to provide habitat suitable for species that are significantly impacted by anthropogenic disturbances (Carbo-Ramirez and Zuria, 2011). Management practises in urban reserves can also negatively impact bird species, and reserves with more core area may be less affected by practises such as the pruning of vegetation. Despite the negative impacts of the disturbance associated with edge effect, there are many species which benefit from habitat edges, as patch margins can benefit dispersing organisms, and can in some cases have more species rich and heterogeneous vegetation structure than core areas (Gonzalez et. al., 2010, Horak, 2016).

A second major edge effect in urban habitat fragments is the biotic effect of increased invasibility, and the indirect biotic effects this causes. Habitat fragments that have more edge area and less core area can favour species which are well adapted to disturbance such as invasive exotic species, which can outcompete or predate native species. Invasive species populations such as *Rattus rattus* can build up in a favourable matrix (i.e. with an abundance of resources such as human food waste) and spill over into a habitat fragment, and longer edges provide greater opportunity for colonization. The breeding of urban birds can be severely impacted by increased invasion by nest-predator species, which are found in greater abundance closer to forest edges in urban areas (Keyser, 2002.). An increase in predator populations can reduce viability of urban bird populations by increasing mortality by predation and decrease breeding success, meaning that predator control measures may be more necessary to maintain diverse bird populations in reserves which are less-round in shape.

Reserves with a greater core-to-edge ratio can provide more suitable habitat for species that are sensitive to disturbance and can provide a for a greater range of more specialised niches, which can then facilitate greater species diversity, for example, shade-tolerant species of vegetation (and therefore the fauna that depend on them) may be absent from reserves with a high proportion of edge effected area (Gonzalez et. al., 2010). A New Zealand study looking at forest patches in a mostly agricultural matrix found that the increased edge effect of irregularly shaped habitat fragments reduced populations of core-dwelling species by 10-100%, depending on the level of sensitivity to edge effects of the particular species (Ewers and Didham, 2007). This effect was found to be greater in smaller fragments than in large ones. A major constraint of this study is that it treated edge effect as homogenous and did not account for the strong impact that varying degrees of contrast between the habitat and matrix can have on edge effect (Ewers and Didham, 2007). In an urban setting, the level of contrast between habitat patch and matrix is heterogenous, and highly variable depending on the level of urban densification, so edge effects can vary significantly in this context. This study includes surrounding matrix quality as a factor in exploring the effects of reserve shape on urban bird communities, as surrounding matrix has been identified as an important factor determining edge effects on urban birds (Hodgson et. al., 2007), but has not yet been studied specifically in relation to urban habitat shape.

Studies focussing on habitat fragment shape are few in the literature, and have most frequently been conducted within an agricultural matrix and may not be applicable in urban areas where human activity is so prevalent and ubiquitous that species which are highly sensitive to disturbance are unlikely to persist regardless of core-area size (Rodriguez-Prieto et. al., 2014). The provision of habitat in urban areas is constrained by land availability, and large round reserves may be impossible to create retrospectively, but although narrow reserves along waterways or transport routes may be much more practically possible to create, planners have tended to view narrow reserves have as corridors rather than habitat provision in the urban context (Ignatieva et. al. 2011). It is generally accepted that the increased edge effect of less-round urban reserves may negatively impact bird diversity to some degree, however, this impact could be less significant in a residential urban matrix which includes private gardens, with less contrast between habitat and matrix. Theoretical modelling comparing the ecological advantages of different designs of reserve networks at a landscape scale suggest that a mixture of habitat patch sizes and shapes could provide the most positive outcomes, and linear habitat elements may be valuable both as corridors and as habitat (Schippers et. al., 2009). Studies of urban greenways (long narrow reserves) have found that width is a determinant of both predator density and habitat quality, with wider greenways having lower nest-predator densities and higher abundance and richness of urban-sensitive species (Sinclair et. al, 2005, Mason et. al, 2007). It is important to further explore the possibilities of creating narrow reserves to enhance urban biodiversity in existing cities due to the greater likelihood of actually being able to implement the creation of such reserves in highly urbanised areas with scarce land availability.

Size

It has been well established that urban reserve size is positively correlated with bird species richness, an effect which has been recorded consistently in cities throughout the world (Carbo-Ramirez and Zuria, 2011, Strobach et. al., 2013, Nielson et. al., 2014, Kang et. al., 2015, Aida et. al, 2016, Chang and Lee, 2016, Leveau et. al., 2019). The increase in avian diversity that comes with an increase in reserve size has been attributed to the greater plant diversity found in larger reserves providing a greater variety of resources for a more diverse range of bird species (Donnelly an Marzuff, 2004). Some studies have placed more emphasis on a strongly related factor, attributing the increased habitat heterogeneity found in larger reserves, of which vegetation diversity is a crucial element, to be the most important aspect influencing the ability of larger reserves to support greater bird species diversity (Nielson et. al., 2013, Chang and Lee, 2016). Larger reserves are generally able to incorporate a greater range of abiotic and landscape features and therefore can provide more various microhabitats and resources for a greater range of specialised species (Nielson et. al., 2013, Chang and Lee, 2016). This increase in habitat for specialised species means that as well as species diversity, increased patch size can also positively influence diversity of functional groups in urban forest birds (Kang, et. al, 2015).

A study of small green sites in Boston, America, found that patch size was the most important factor affecting bird species richness and concluded that small greening projects should focus on connecting or increasing larger patches (Strobach et. al., 2013). Although the authors believe that allocating larger areas in fast-developing cities is practically impossible, this conclusion is still echoed in a study of small greenspaces throughout Pachuca, Mexico, which also identified herbaceous stratum height and landcover of the adjacent area as other important factors influencing bird species richness in small urban habitat patches (Carbo-Ramirez and Zuria, 2011).

Patches of habitat in urban areas are often small because of competing land use requirements and it is important to know more about what other landscape and habitat features influence avian diversity and abundance in order to improve urban biodiversity. Although studies of both larger and smaller urban habitat patches continuously recommend increasing fragment size, area alone has been shown to be insufficient to estimate the persistence of populations in habitat fragments (Aurambout et. al., 2005).

The majority of studies exploring bird populations in urban habitat patches focus on larger patches (>2ha), perhaps because small patches are made up mostly or entirely of edge habitat, however there are many cities throughout the developing world (where the majority of future urban growth is predicted) in which only small habitat fragments exist (Collinge, 1996, Carbo-Ramirez and Zuria, 2011). Therefore, it is important to understand the role of small urban greenspaces, especially the increasingly popular design of small narrow riparian greenspace, in sustaining avian metapopulations and diversity.

This study compares categories of reserve shape and size while attempting to control for variation in factors such as vegetation quality and surrounding matrix by using reserves with an even range of these factors in each shape/size category. In this study we investigate the effect of a range of habitat and landscape factors on avian diversity and abundance using 28 urban reserves divided into 4 shape and size categories situated in Auckland City, New Zealand.

Research questions

1. Do vegetation quality, adjacent landcover and habitat connectivity significantly affect the native bird diversity of urban reserves?

2. Does the shape and size of reserves affect diversity and abundance of bird species in general?

3. Does the native bird diversity differ between urban reserves of different shape and size?

Methodology

Bird Counts

The same paths and/or transects used for the vegetation survey were also used for bird counts (see page 18 for details). I walked at a steady slow pace along transects and recorded birds seen and heard. Observed birds were categorised into 1.) seen within 10m on either side of the transect, 2.) seen outside of 10m but within the reserve area, or 3.) Birds heard but not seen (which may or may not have been within the reserve area). Birds that had been heard and then were subsequently seen were transferred to be recorded as seen and removed from the heard category. Flying birds were counted if they were in front of me and going from left to right or in front to behind, but not counted if flying from right to left or coming from behind me. Bird counts were conducted between the hours of 1200 and 1600 on days with favourable weather conditions, with no rain and wind less than 25km/hour (as stated on metservice weather website).

Bird Data Analysis

A Shannon index was used to calculate bird species diversity for each site in each season, autumn, winter and spring (see chapter 2 page 21 for equation). Shannon index was also calculated for native bird diversity at each site in each season. The length of transects used for bird counts and nest searches was measured and multiplied by 20m (10m search area on either side of transect) to give a transect area for each site. The total number of birds seen within 10m was divided by the transect area of each site to give a bird abundance score for each site in each season. The total number of native birds seen within 10m was divided by the transect area of each site in each season to give a native bird abundance score for each season.

Statistical analysis

Multidimensional scaling (MDS) and principle component ordination (PCO) were used to explore the influence of reserve shape and size on bird communities, using bird survey data. Brey-Curtis distance of untransformed abundant data was used to generate the resemblance matrix. Permutational Anova (permanova) test were used to detect the statistical significance. Site 28 is an outlier and has been excluded from analysis. All mutivariant analyses were performed using software Primer V7 (Anderson & Gorley, 2008).

ANOVA tests were used to analyse the effect of reserve shape and size on bird diversity, bid abundance, native bird diversity, and native bird abundance, and Levene's test was used to assess problems with assumptions of variance for these ANNOVA tests.

Results

Figure 3.2- Community composition PCO plot

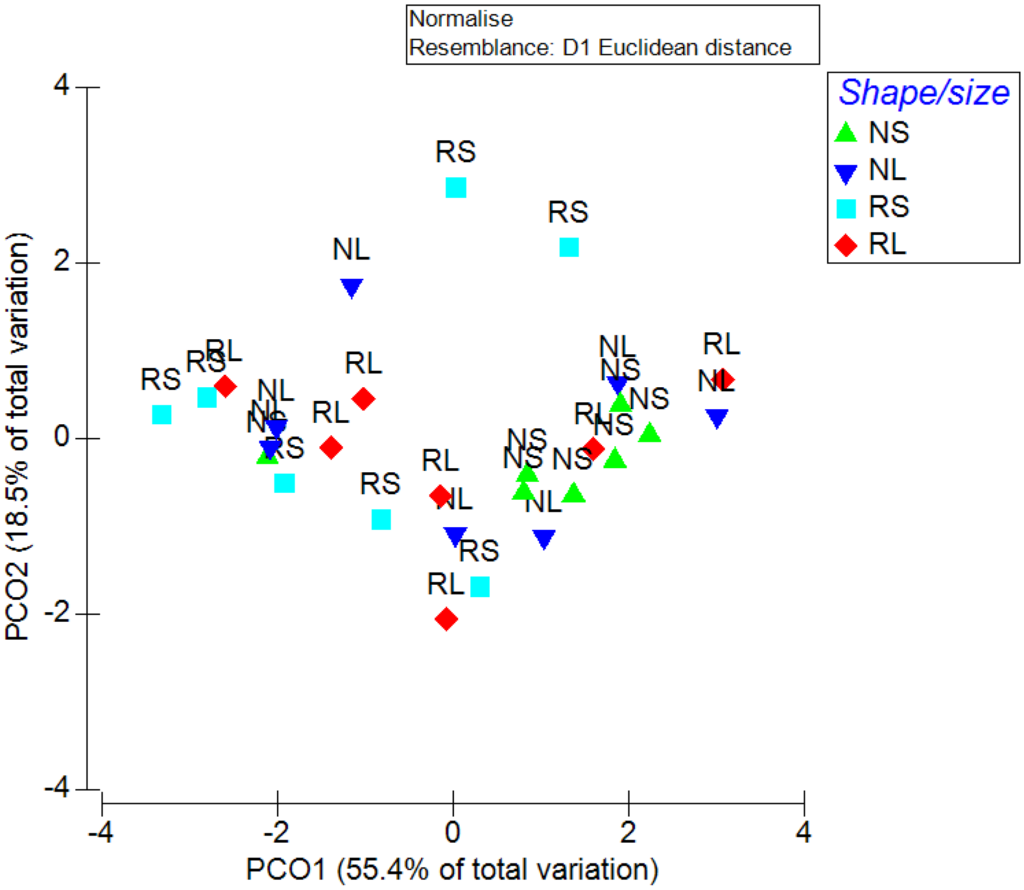
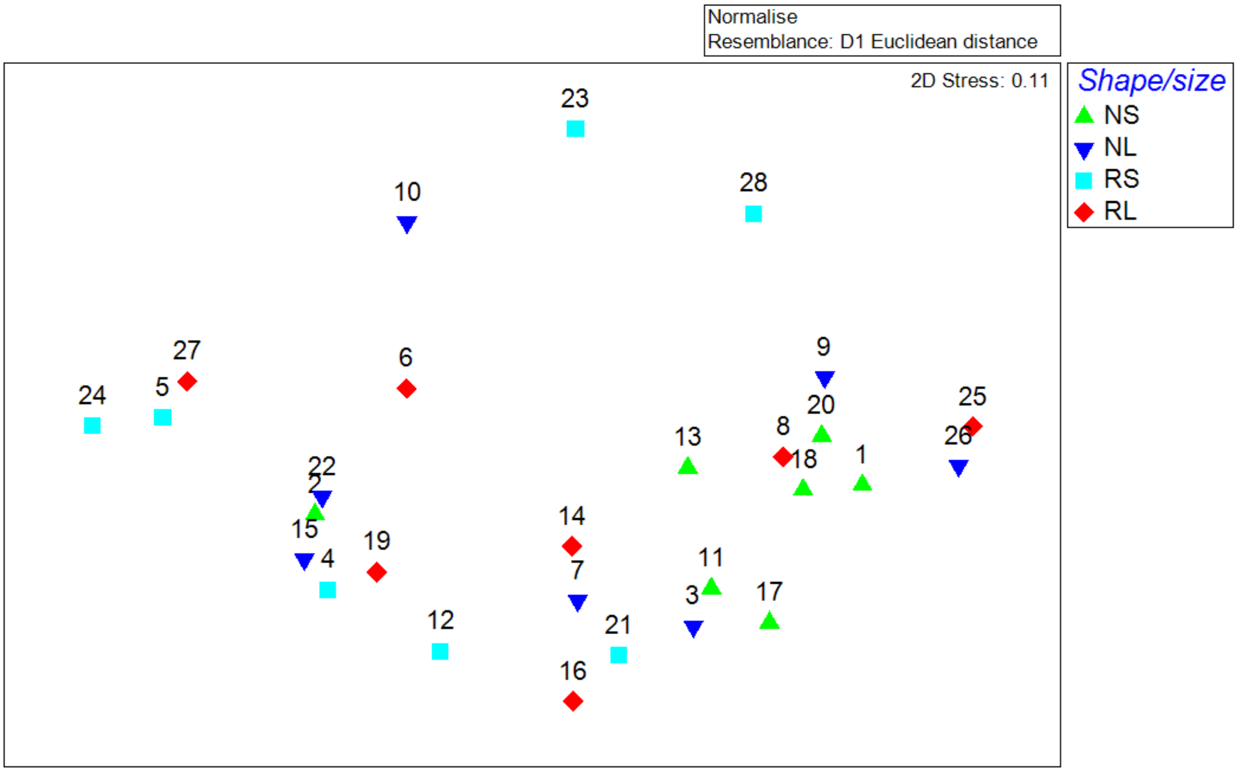


Figure 3.3- Community composition MDS plot



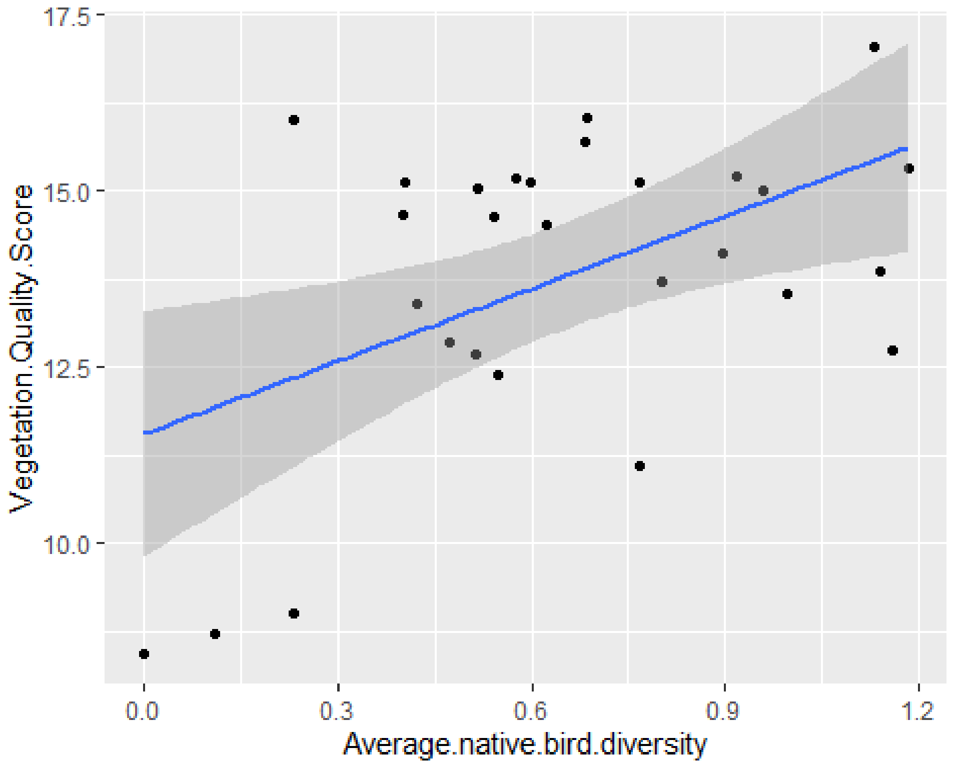
Both PCO (figure 3.2) and MDS (figure 3.3) plots on bird survey data (species and abundance) indicate dissimilarities in avian community structure between small sites, especially round small sites, and large sites. Permanova test analysis confirmed the significant effect of shape and size on bird community composition (*p*= 0.007).

Habitat factors effect on native birds

There was no evidence of an effect of vegetation quality score on native bird abundance (t-test [of slope *Beta*= 0], *p*= 0.414), although residual plot suggests some evidence of non-linearity and changing variance.

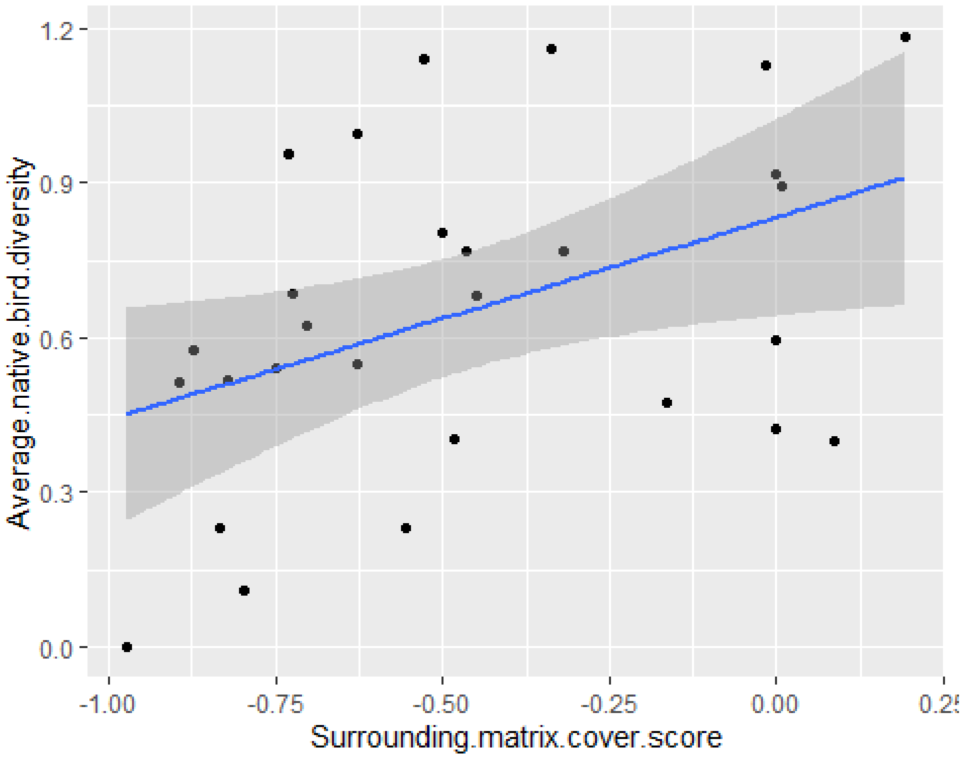
There was also no evidence of an effect (t-test [of slope *Beta*= 0], *p*= 0.346) of adjacent landcover score on native bird abundance.

Figure 3.4- Mean native bird diversity in relation to VQS



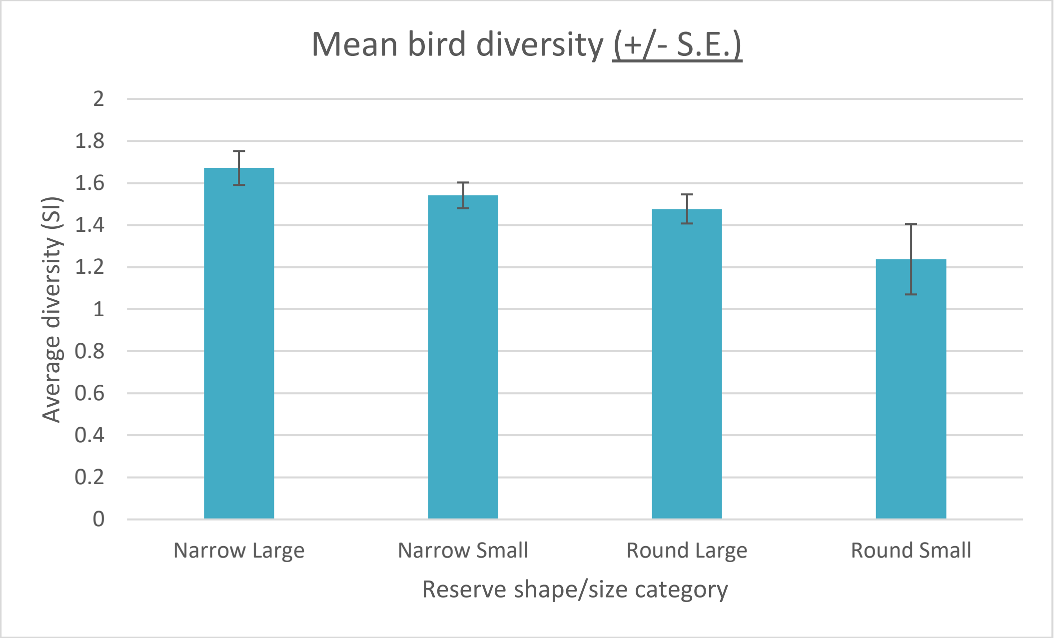
Testing the effect of vegetation quality score on native bird diversity showed strong evidence of an effect (t-test [of slope Beta=0], *p*= 0.007). Greater vegetation quality was associated with greater native bird diversity. Residual plot suggests some evidence of non-linearity.

Figure 3.5- Mean native bird diversity in relation to adjacent landcover



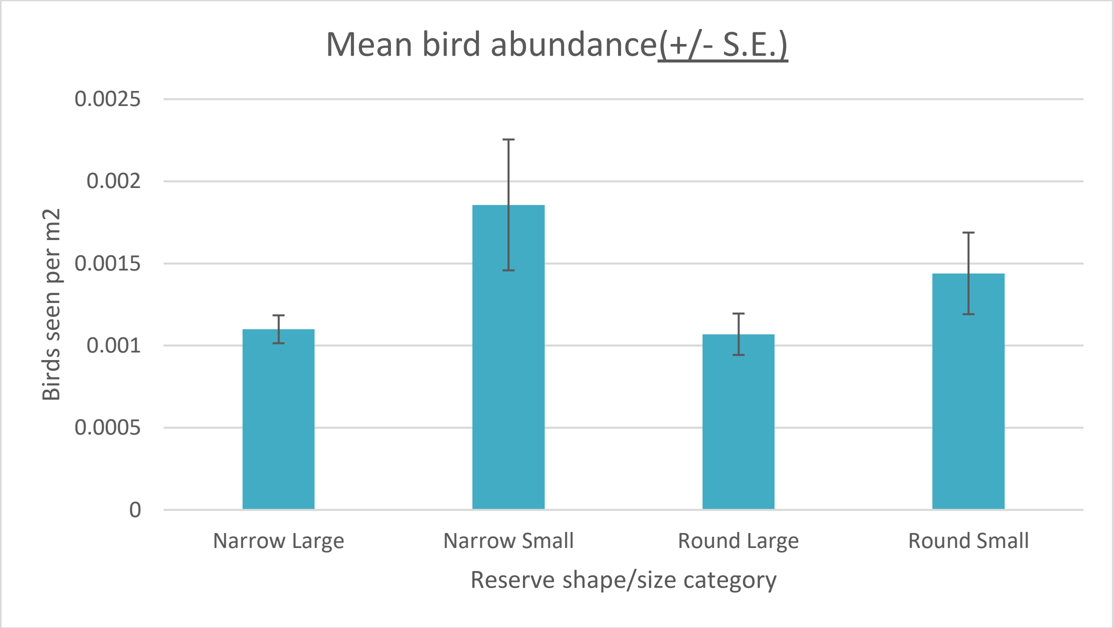
The adjacent landcover score (labeled as surrounding matrix cover score in the above graph) also had a significant effect on native bird diversity (t-test [of slope Beta=0], *p*= 0.023). Increased quality of adjacent landcover was associated with an increase in native bird diversity.

Figure 3.6-Mean bird diversity (Shannon Index) for reserve shape/size categories (+/- S.E.)



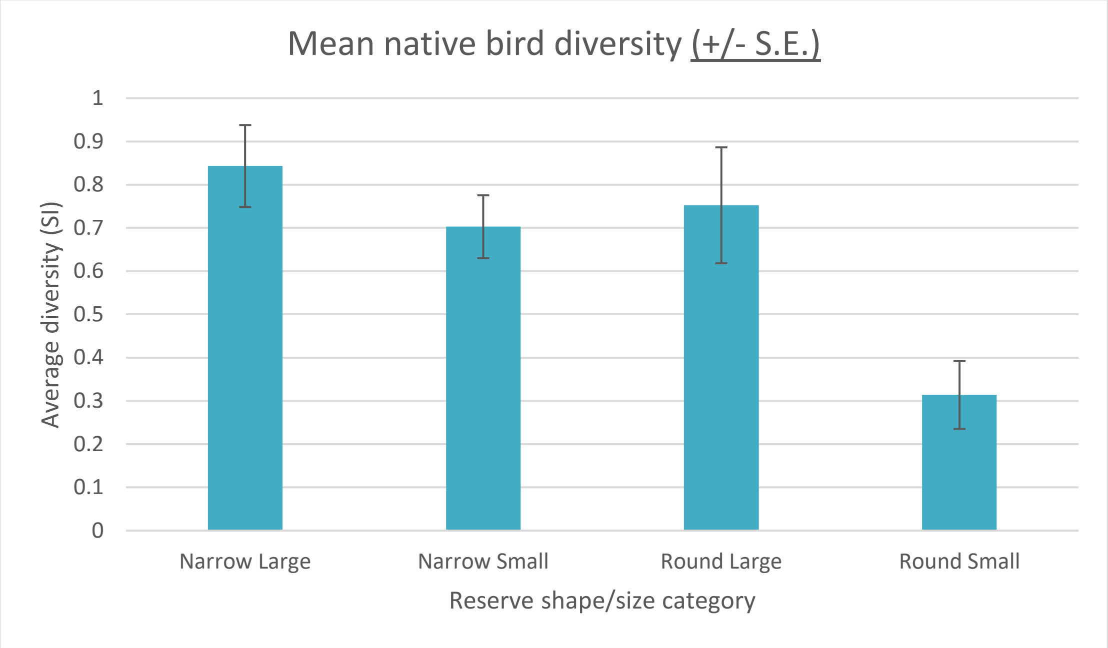
The shape and size category had a significant effect on bird diversity (ANOVA, *p*= 0.047). The difference appears to be between narrow large and round small reserve categories, but there is weak evidence of problems with variance assumption (Levene-test, *p*= 0.063). When the outlier with very low diversity was removed from the small round category, the test returned a non-significant result (ANOVA, *p*= 0.123) with no problem with assumption of variance (Levene-test, *p*= 0.258).

Figure 3.7- Mean bird abundance for different reserve shape/size categories (+/- S.E.)



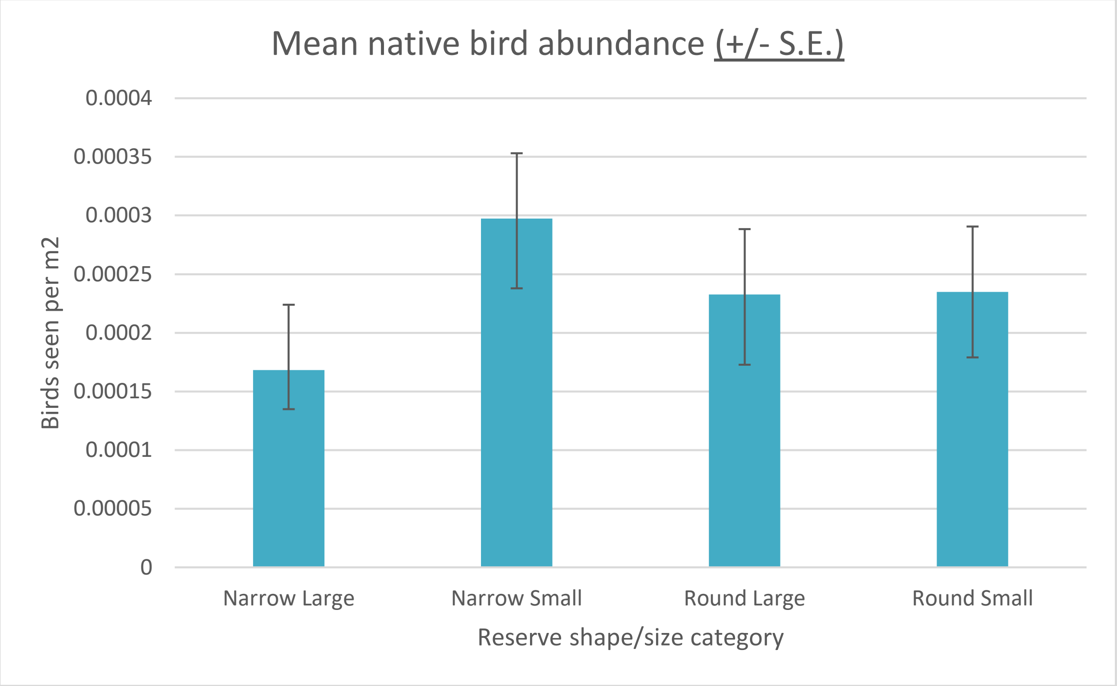
There was no statistically significant (ANOVA, *p*= 0.112) difference in mean bird abundance between different reserve shape/size categories and no evidence of problems with variance assumption (levene-test, *p*= 0.232).

Figure 3.8- Mean native bird diversity (Shannon Index) for different reserve shape/size categories (+/- S.E.)



Strong evidence of differences in mean native bird diversity between shape and size categories was found (ANOVA, *p*= 0.004), with the round small category showing lower native bird diversity than the other categories. There was no evidence of problems with variance assumption (levene-test, *p*= 0.374).

Figure 3.9- Mean native bird abundance for different reserve shape/size categories (+/- S.E.)



Analyzing the mean native bird abundance of each reserve category showed no significant difference (ANOVA, *p*= 0.418) and no evidence of problems with variance assumption (levene-test, *p*= 0.714).

Discussion

We found strong evidence that vegetation quality score (VQS) positively effects native bird diversity, which is consistent with previous studies (Donnelly and Marzluff, 2004, Chace and Walsh, 2006, van Heezik et. al., 2008(b)). Native bird diversity increases with an increase in vegetation quality score, indicating that improving the vegetation cover, density and maturity of urban reserves can positively influence their capacity to support native bird populations, regardless of the reserve size or shape. Although we incorporated forest maturity into the dataset by measuring the average trunk circumference as a component of VQS, the presence of tree cavities is another important aspect of forest maturity that has been shown to be related to urban bird diversity and was not recorded in this study (Strobach et. al., 2013, Aida et. al, 2016). The vertical vegetation structure of habitat patches is also a vegetation factor that has previously been shown to significantly influence urban bird diversity and that we did not explore (Donnelly and Marzluff, 2004, Carbo-Ramirez and Zuria, 2011). Although data was collected on the understory complexity, which was measured for 3 different height categories, this data was incorporated into the VQS rather than being explored as a separate variable, and data on tree height was not collected. Tree species data was collected to determine the consistency of tree diversity between reserve categories, and this data in combination with the understory vegetation species data could have been used (if time constraints allowed) to explore the proportion of native plant species compared to exotics, which has previously been shown to influence native bird diversity (van Heezik et. al., 2008(b)).

A greater vegetated proportion of the adjacent landcover has been shown to have a positive effect on urban bird species richness in previous studies (Mason et. al., 2007, Carbo-Ramirez and Zuria, 2011, Nielson et. al., 2014, Heggie-Gracie et. al., 2020). Our study also confirms this result- the adjacent landcover score had a significant positive effect on native bird diversity ($p=0.023$). During the data collection, native birds of several species were observed flying in and out of private gardens adjacent to the reserve, suggesting that a matrix including private gardens can effectively extend the available habitat area of the reserve, in accordance with previous studies which have shown the value of private gardens in supporting native bird populations (van Heezik et. al, 2013, van Heezik and Adams, 2016).

Native bird diversity was the most important aspect of this study in terms of the conservation of urban biodiversity, and we found strong evidence (p=0.004) of a difference between reserve shape and size categories, with the round small category showing lower native bird diversity than the other categories (see **figure 3.8** page). The lower native bird diversity found in round small reserves is consistent with the literature which unanimously shows that larger reserves have greater avian diversity (Donnelly an Marzuff, 2004). What was not expected however, was finding that narrow small reserves did not have lower native bird diversity than narrow large or round large reserves, confirming previous findings that reserve size alone cannot explain urban bird diversity (Aurambout et. al., 2005). This finding is inconsistent with other studies on small urban reserves that found patch size to be the most important factor in predicting bird diversity (Carbo-Ramirez and Zuria, 2011, Strobach et. al., 2013, Kang, et. al, 2015). Round small sites were also found to differ from other reserve categories in regards to avian community structure. Shape and size were found to have a significant effect on bird community composition (p=0.007), with bird species and abundance data indicating dissimilarities in avian community structure between small sites, especially round small sites, and large sites.

Finding that the narrow reserves have greater species richness and native bird diversity than rounder reserves is very unexpected. Some factors which may have contributed to this finding are that round small reserves (which showed the lowest native bird diversity) also had the lowest freshwater presence (only 14.3% of sites) and lowest predator control (28.57% of sites), while narrow small sites had the highest presence of freshwater (85.71% of sites) and the highest predator control (57.14% of sites). It is possible that the greater presence of freshwater, which has been previously shown to increase bird diversity (Zivanovic and Luck, 2016), made the narrow small habitats more attractive to some species of bird, and that predator control measures offset the disadvantage of greater edge effect leading to higher abundance of predators which has been shown in previous studies (Keyser, 2002, Sinclair, 2005). These findings provide new evidence that narrow urban reserves, even when small in area, are capable of providing habitat for native bird populations, and presence of freshwater and predator control are likely to be either contributing or determining factors to their ability to do so. Further research directly comparing reserves of similar shape and area with and without predator control and freshwater is necessary to determine just how crucial these factors are.

The similarity between the native bird diversity in round large and narrow large sites adds new evidence to the findings of previous studies looking at diversity of vegetation which found that when the positive effect of increased area is removed, testing habitat patch shape independently has shown no significant effect on vegetation diversity of forest patches (Hill and Curran, 2001, Gonzalez et. al., 2010). All of the study sites had walking paths throughout the reserve area, and with pedestrians and dogs (often not on leashes) regularly seen using them it can be confirmed that there is no pristine core area in any of the study sites that was completely free from human disturbance. This may explain why reserve narrowness does not have a detrimental effect on bird diversity in an urban setting, as the increased edge effect of less-round habitat shapes is largely irrelevant in habitats where the entire habitat area is affected by anthropogenic disturbance, as has been previously proposed (Rodriguez-Prieto et. al., 2014). Interestingly, the narrow large category and the round large category have an identical proportion of sites with freshwater in them (both 57%), so this is a consistent factor when comparing these categories. Likewise, the presence of predator control measures was not higher in narrow large (29%) reserves than in round large reserves (43%), and yet narrow large sites still have a slightly higher bird diversity, both for native species and all species, than round large ones, further adding weight to the findings of this study that reserve shape does not explain urban native bird diversity.

It is likely that the significantly lower native bird diversity found in round small reserves could be mitigated by improving the vegetation quality and/or the adjacent landcover of these reserves, as we found that both of these factors had significant positive effects on native bird diversity. Adjacent landcover could potentially be improved by educating the public on the benefits of nature and encouraging tree planting in private gardens with policies such as rates reductions for properties with trees. Previous studies in urban New Zealand have also found a significant effect of vegetation quantity and quality, so we recommend investing in improving the vegetation of small reserves as a sound strategy to improve native bird diversity in urban areas, which can help to offset the effect of vegetation loss occurring on private land due to densification (van Heezik et. al., 2008, Heggie-Gracie et. al., 2020). Further research on these very small (> 47,557 m^2) reserves could help to establish thresholds for minimum vegetation quality by comparing native bird communities in a larger sample size of reserves with similar area but significantly different vegetation quality.

Conclusion

The shape of urban reserves did not significantly affect bird diversity or native bird diversity, and this is new evidence that does not appear to already exist in the literature. The size of reserve did not have a significant effect on native bird diversity, although round small sites did show lower native and total bird diversity than all the other categories, indicating that habitat factors such as presence of predator control and freshwater may be especially crucial for bird populations in smaller reserves. In concurrence with previous studies, vegetation quality and the landcover of the surrounding matrix were shown to have a significant effect on native bird diversity. The findings of this study provide evidence that urban reserves of varying size and shape can support diverse native bird populations, and although large round areas of habitat may not be practically possible to create of maintain in many cities, other shapes and sizes of reserve are still able to enhance urban biodiversity.

This study attempts to control for variation in factors such as connectivity and vegetation complexity by using reserves with an even range of these factors in each size/shape category. Further studies on this topic should compare a larger number of sites of similar size and shape but with different categories of vegetation complexity, connectivity and surrounding matrix to find out more about how these factors interact with the effects of reserve size and shape to influence urban bird diversity. Future studies should also incorporate tree height, number of tree cavities, predator control measures and distance to a freshwater source as measured variables which could potentially influence avian diversity.

Chapter 4: Effects of reserve shape, size and habitat features on the breeding of tui (*Prosthemadera novaeseelandiae)*, piwakawaka (*Rhipidura fuliginosa)* and riroriro *(Gerygone igata)* in urban reserves on the Auckland Isthmus

Figure 4.1- Active piwakawaka/fantail (*Rhipidura fuliginosa)* nest recorded during breeding survey

Introduction

With the expansion of global urbanisation, the conservation of nature and biodiversity in urban areas is becoming increasingly important, not only for maintaining regional biodiversity (Padilla and Rodewald, 2015), but also for the wellbeing of people (Carrus et. al, 2015). Natural habitats in urban areas are often severely degraded due to habitat fragmentation, anthropogenic disturbance and pollution. Many native species that are sensitive to disturbance become locally extinct in the urban environment (Donnelly and Marzuff, 2004) resulting in altered community structures with greater abundance of human affiliated species (Clergeau et. al., 2006, Dar and Reshi, 2014, Xu et. al., 2018). If native bird species that are capable of adapting to some level of urbanisation are to persist in cities long-term, they must be able to successfully reproduce in the generally degraded habitats of the urban environment. A lack of breeding success in urban areas can jeopardize the regional population viability of native bird species (Padilla and Rodewald, 2015). The breeding success of native birds in urban areas can be severely negatively impacted by habitat loss and fragmentation, insufficient resources, predation by invasive species, and many other kinds of anthropogenic disturbance such as air, water and noise pollution (Chace and Walsh, 2006, van Heezik et. al. (2008(a)). Improving the planning and management of urban greenspace at both a habitat and landscape scale could help to mitigate these negative impacts, with the aim of eventually achieving sustainable and resilient regional populations of native birds throughout urban areas. Because natural habitats in cities are fragmented but not necessarily isolated, the application of metapopulation theory to the regional management of urban habitats at a landscape scale is highly relevant and has the potential to improve urban conservation outcomes.

As outlined in the general discussion (see chapter 1), a metapopulation refers to a network of habitat patches inhabited by asynchronistically fluctuating sub-populations of a species. These patches are not completely isolated and the migration of individuals between patches can occur, enabling the genetic diversity of the regional population to be maintained through the movement of individuals between sub-populations. Environmental and demographic stochasticity pose extinction risk to the sub-population in each habitat patch, but the immigration of individuals from other patches can allow sub-populations to recover from stochastic events, or allow habitat patches where sub-populations have become extinct to be recolonised (Hanski, 1998). Such a metapopulation dynamic can greatly reduce the chance of both local and regional species extinction. If the connectivity between habitat patches can allow a species to be managed regionally as a metapopulation, this approach could increase the viability of populations of native species in urban environments and improve urban biodiversity.

Understanding the breeding activity of native birds in habitat patches is extremely important when considering these patches as part of a network inhabited by sub-populations functioning as a metapopulation. Species can migrate to and be present in habitat patches that are unfavourable for various reasons, and these patches may be acting as population sinks due to a lack of breeding success, high mortality, or both, and have the potential to negatively impact on the viability of the metapopulation as a whole (Schippers et. al., 2009, van Heezik et. al. (2008(a)). Attempting to increase connecting links to such patches may result in an increase in the number of colonists that die or fail to breed in these population sinks. To assess the value of a discrete patch of habitat to a metapopulation it is essential to know whether it can support the successful breeding of a sub-population. Information regarding the breeding of native species in urban reserves is important for conservation management of urban habitats, and if a species is to be managed as a metapopulation then the factors which contribute to the viability of sub-populations need to be more thoroughly understood. For fragmented habitats, the size and shape of a fragment can affect the availability of resources and the magnitude of various edge effects, such as predation or noise pollution, and therefore determine the quality of the fragments as breeding habitats. Habitat factors such as vegetation quality, adjacent landcover, and freshwater availability are also likely to affect the breeding success of native bird species in urban habitats, although the impact of some of these factors have not yet been quantified in published studies so far.

Reserve area and shape

Many studies from different parts of the world have illustrated the importance of habitat patch area to urban bird populations (Donnelly and Marzuff, 2004, Carbo-Ramirez and Zuria, 2011, Strobach et. al., 2013, Nielson et. al., 2014, Huang et. al., 2015, Kang et. al., 2015, Aida et. al, 2016, Chang and Lee, 2016, Leveau et. al., 2019). As well as increasing bird species richness, increased reserve size has also been found to be positively correlated with an increased number of nesting species (Huang et. al., 2015, Chang and Lee, 2016). A study comparing bird breeding in small, medium and large reserves in urban (meaning densely urbanised in this instance), suburban and exurban (rural/suburban combination) areas found that large urban and medium suburban parks had an increased density of nests compared to the other reserve types, and medium suburban parks also had greater nest success (Donnelly and Marzuff, 2004). The higher density of nests in large and medium urban reserves indicates that such reserves are population sources, while small urban reserves with a conspicuous absence of nests are likely to be population sinks, demonstrating the vital importance of connectivity for maintaining urban bird communities in smaller reserves (Donnelly and Marzuff, 2004).

The total area of suitable habitat available to a metapopulation is of primary importance to its long-term viability, but the most ecologically advantageous configuration of this habitat area is a subject of ongoing debate (Ovaskainen, 2002). The long-term viability and resilience of a metapopulation may require some larger patches to be included in the habitat patch network to provide a consistent source of dispersing colonists to bolster sub-populations in small patches. Populations in larger patches are expected to be larger, and therefore are more resistant to demographic stochasticity and less likely to go extinct (Hanski, 1998). Larger patches may also be vital to support viable populations of species with weak dispersal ability, or which require core-area habitat. Because the effects of patch size and shape impact different species to differing degrees, it is important to build a body of knowledge exploring these effects to increase our understanding of which habitat types are likely to support breeding populations of which species, and which habitat factors are of the greatest importance to the conservation of urban native bird species. The effect of habitat shape on urban native bird breeding appears not to have been investigated in published studies to date, so this current study could begin to address a crucial knowledge gap.

Modelling has suggested that a mixture of large, small, and linear connecting habitat elements within a landscape can be more ecologically advantageous than the configuration of the same habitat area into one or a few large patches (Schippers et. al., 2009). If a metapopulation of native birds is to persist in a city long term, the proportion of source populations within the metapopulation must outweigh the negative impact of sink populations, so investigating the effect of reserve shape and size on native bird breeding is essential to understanding whether a reserve network is likely to contain a great enough proportion of source populations to support a viable metapopulation long-term. Small or linear habitat patches may be the most practical types of habitat patches to create in a highly urbanised landscape, but they should be able to support some level of reproduction in order to be considered as part of the network of valuable habitat, rather than functioning only as connecting elements (Schippers et. al., 2009). Previous studies of urban bird communities investigating reserve size or shape have found that reserve area is the most significant factor influencing bird breeding, and that long narrow reserves (greenways) are capable of providing bird breeding habitat of varying quality depending on their width (Donnelly and Marzuff, 2004, Sinclair et. al, 2005, Mason et. al, 2007, Huang et. al., 2015). This study attempts to add to the current body of knowledge by directly comparing the ability of reserves of smaller and larger sizes and of narrower and rounder shapes to provide suitable breeding habitat for three species of native birds in an urban environment.

Habitat features

As well as variation in territory sizes, different species of native birds have differing habitat requirements such as needing a certain level of vegetation complexity for foraging or nesting. Understanding the potential limiting factors that could impact the breeding of each species is important when assessing the ability of urban reserves to provide suitable breeding habitat. Vegetation diversity has been shown to positively influence bird breeding in urban reserves, which is likely to be due to the associated increase in invertebrate diversity as well as availability of food sources such as seeds and nectar (Huang et. al., 2015). A New Zealand study found that vegetation structure and composition have a greater influence on native bird diversity in urban areas than general bird diversity, indicating that vegetative habitat characteristics are particularly important to native birds (van Heezik et. al., 2008(b)). As well as controlling for vegetation quality when comparing the effect of reserve size and shape, this study also explores the effect of vegetation quality on the breeding of each of the three native species studied.

The quality of the adjacent landcover is another important habitat feature to consider because a suburban matrix with greater vegetation cover than densely urbanised matrix may provide resources for some species of native bird (Gray and van Heezik, 2016). A study from Beijing, China, found that greenness of the surrounding area of parks was a significant positive factor for bird breeding (Huang et. al., 2015). Connectivity is an associated factor which may indirectly influence breeding as migration between smaller habitat patches is likely to be important for juvenile dispersal and mate-finding. This study controls for adjacent landcover quality when comparing reserve shape/size categories, and also explores the effect of adjacent landcover on the breeding of the three native species studied.

Another example of an important factor influencing the suitability of habitat patches for bird breeding is the distance that birds need to fly to access freshwater, as this becomes especially important in the breeding season when time and energy are maximised towards providing food for offspring or the incubating parent. Although a Canadian study including freshwater streams as a habitat factor found that some species of bird can only be found in habitats which had freshwater (Melles et. al., 2003), there appear to be few studies exploring this factor in the literature, and there may not be any to date which relate it to native bird breeding.

Predator control

Predation is a principal cause of nest failure for urban birds (Donnelly and Marzuff, 2004). New Zealand has especially vulnerable endemic species because of the historical lack of mammalian predators, which has allowed native birds to evolve without behavioural strategies to avoid predation by mammals. In New Zealand, as in other countries, the birds that adapt to urban areas are comprised of a mixture of both introduced and native species, but in New Zealand even those native species which are resilient to anthropogenic disturbance and able to utilise urban resources are still particularly vulnerable to predation by introduced mammals, and predation has been shown to be a major cause of both nest failure and juvenile mortality for native birds in urban reserves (van Heezik et. al., 2008(a)). Although the predation of all three species of native bird selected for this study by introduced mammals has been recorded in the literature, more knowledge about differing responses of the breeding activity of each species to predator control measures is needed to inform minimum predator control management requirements to supporting breeding populations (Gillies and Clout, 2003, van Heezik et. al., 2008(a), Ruffell and Didham, 2017). Although it can generally be said that predator control measures are important for the breeding and survival of native bird species in New Zealand, such measures have been found to have less positive effect when applied with less frequency and also when applied to areas with less forest cover, the latter indicating that efforts to control predators are likely to be differentially advantageous in different types of reserve (Ruffell and Didham, 2017). The effective allocation of management efforts to control predators and maximise breeding success can be improved through greater understanding of which habitats are likely to be especially valuable for breeding, and which species are particularly vulnerable to nest and juvenile predation. Predator control measures are likely to be more necessary in reserves of narrow shape with greater edge effect, which can be expected to have a higher abundance of nest predators (Sinclair et. al., 2005). This study includes the presence of predator control measures as a factor to analyse in relation to the breeding of three urban-adapted native species in order to explore the importance of this management tool in ensuring the long term persistence of these species in urban reserves.

Study species

The species selected for this study were three of the most commonly observed native species in urban reserves on the Auckland Isthmus: tui (*Prosthemadera novaeseelandiae*), piwakawaka (*Rhipidura fuliginosa*) (also known as New Zealand fantail) and riroriro (*Gerygone igata*) (also known as grey warbler*)*. At least one of these three species was seen or heard at every reserve surveyed. All three species are still widespread throughout New Zealand and appear to have adapted to anthropogenic disturbance and mammalian predators more successfully than most other native species (Miskelly, 2018). All three species are known to be able to cross a suburban matrix such as the one in this study area (Hodgson et.al, 2007), but also all three are either sedentary or, in the case of tui, show high site-fidelity, returning to the same breeding sites each year. It could be considered surprising that these species are commonly observed in many urban areas of New Zealand because all three are partially (tui) or fully insectivorous (piwakawaka and riroriro), and it has often been documented in the literature that insectivorous species decline with increased urbanization (Chace and Walsh, 2006).

Tui are medium-sized, mainly nectivorous birds of the Meliphagidae family, which can travel long distances (up to 30km) in search of food, regularly returning to the same seasonal nectar sources and breeding sites with high site-fidelity (Bergquist, 1985). In the breeding season their diet shifts to include arthropods as well as nectar due to increased protein requirements, they become territorial, and their range is restricted to around 0.5km in diameter, with related individuals nesting in adjacent territories and moving off together in family groups to other foraging sites after fledging (Bergquist, 1985). Tui have shown a significant, long-term, positive response to the removal of mammals (O'Donnell and Hoare, 2012, Miskelly, 2018). One of the very few published studies documenting avian diversity in Auckland city found that tui are one of the three native species with the highest relative abundance on the Auckland Isthmus, a result that was similar in the above chapter (see chapter 3 results) surveying reserves in the same area (Landers et. al, 2019).

Riroriro are an endemic passerine, one of the two lightest native bird species found in New Zealand, and inhabit all types of native forest throughout the country (Gill, 1982). Adults are sedentary, and in the breeding season they become territorial and form monogamous pairs to build enclosed nests that are often parasitized by the shining cuckoo (*Chrysococcyx lucidus),* which migrates to New Zealand over summer. Towards the end of the breeding season grey warblers expand their territories to overlapping home ranges through which they move in small foraging groups of 3-6 that do not include recent progeny, and may associate with groups of other small native insectivores, including fantails (Gill, 1982). Juvenile riroriro have been found to be highly mobile, and it is thought that most disperse over distances up to 2.9km (Gill, 1982). In urban areas, riroriro have been recorded in bush fragments and larger private gardens with greater vegetation complexity but are virtually absent in highly urbanised areas (van Heezik et. al. (2008(a)).

Piwakawaka are another small native bird that have been recorded in urban bush fragments as well as in residential areas with more complex and mature vegetation (van Heezik et. al. (2008(a)). Piwakawaka are weak fliers, meaning that the cost of migration is high, and connectivity is especially important. Their breeding seems to be highly sensitive to stochasticity, with a study of fantail breeding in Dunedin City recording 60-70% nest survival in a good year, and only 5-7% in a bad year (van Heezik et. al. (2008(a)). The same study also found that predation by rats (*Rattus rattus*), possums (*Trichosurus vulpecula*) and possibly mice (*Mus musculus*) was a major cause of nest failure, and that under predation pressure piwakawaka prefer to nest in trees of 6-7m in height which are largely unavailable in urban gardens, and so rely on reserves with larger trees (van Heezik et. al. (2008(a)).

Riroriro and piwakawaka have both shown significant increases in abundance over a ten year period of predator control (O'Donnell and Hoare, 2012), but have also shown significant declines in abundance after the removal of mammalian predators and reintroduction of missing native bird species, indicating that these species are less susceptible to predation than other native species, and can be outcompeted by other natives when mammalian predators are removed (Miskelly, 2018). Both piwakawaka and riroriro are also known to inhabit and penetrate habitat edges and are able to disperse across residential areas (van Heezik et. al. (2008(a)), and so could be considered successful urban adapters in low to medium density urban areas.

This study compares presence of breeding, siting of juveniles, and spring abundance for three species of native birds in reserves of different shape and size and explores habitat factors which may contribute to suitability of breeding habitat. Higher vegetation quality score, adjacent landcover score and the presence of predator control are expected to positively influence the breeding activity of all three species. Freshwater presence is hypothesised to be of particular importance to piwakawaka/fantails, which are not strong flyers.

Research Questions

1. Does the shape and size of urban reserves influence the breeding of the three study species?
2. Does the shape and size of urban reserves influence the spring abundance of each study species?
3. Does predator control show an obvious effect on the presence and success of breeding for the three study species?
4. Does water availability, vegetation quality, adjacent landcover, tree species diversity and connectivity influence breeding of the three study species?

Methodology

Study sites

The same 28 reserves used for the bird diversity survey and which were analysed and categorised into four shape/size categories in the above chapters were also surveyed for bird breeding activity. Refer to chapter 2 methodology (page 18) for further details regarding these sites.

Breeding Activity Survey

During the summer months of December, January and February, each of the 28 study sites was surveyed for signs indicating the presence of breeding tui (*Prosthemadera novaeseelandiae*), piwakawaka/New Zealand fantail (*Rhipidura fuliginosa*) and riroriro/grey warbler (*Gerygone igata*). The same paths or lines (where paths were not present in a large portion of the reserve) through each reserve that were used for the vegetation survey and bird counts were walked at a steady slow pace, with visual and auditory searching for any nests in surrounding vegetation. A similar amount of search time was spent on reserves of similar size, ranging from 2 hours for large sites, to 30 minutes for the smallest sites. Any nests of any bird species that were seen were recorded. If any of the three study species of bird were seen or heard they were observed until it could be determined whether they were engaging in breeding behaviour. If seen collecting invertebrates to feed young they were followed until the nest or juveniles were found and recorded. The following and observation of study species birds was included in the survey time. Nest surveys were conducted between the hours of 12pm and 4pm on days with no rain and wind less than 25km/hour (as stated on Metservice weather website). Nests found were photographed for to confirm which species they belonged to, and the species of tree in which the nest was found was recorded, as well as the estimated height, vertical index and horizontal index of the nest. Juveniles which were observed during the subsequent bird counts were also added to the juveniles seen data.

Breeding data analysis

Potential confounding factors related to the breeding of native birds that were explored for this study were predator control, presence of a freshwater source, vegetation quality score, adjacent landcover score and distance o closest forest patch. The four categories for reserve shape and size do not show any significant difference in the measured habitat factors (see chapter 2), however it may be useful to compare the different detection of breeding for these three native species in relation to the habitat features to inform future studies in this area.

Binary logistic regression was used to test the effect of habitat features on the presence of breeding activities of three native bird species, tui, piwakawaka and riroriro. For each species, we included variables of habitat size and shape category, distance to the closest forest patch over 5000m2, vegetation quality score, presence of predator control, surrounding matrix score and presence of water as covariant in the model. The test was conducted in SPSS (version 24).

Vegetation quality score was calculated using the following equation: VQS = LS + H + UCS +TCS (see chapter 2 page 23 for further details).

Results

Evidence of breeding (siting of active nest or parents with juveniles) for at least one the three study species was found at 89.29% of the reserves surveyed, and juveniles were sited at 71.43% of sites.

Table 4.1- Measurements and tree species for nests found during breeding survey

Site name	Species	Nest tree	Height (m)	VI (m)	HI (m)	Nest notes
Dingle Dell	piwakawaka	kawakawa	1.8	1.8/2	0/1	not active
Francis Street Reserve	piwakawaka	karaka	3			active- incubation parent and mate nearby
Ayr Reserve	piwakawaka	mahoe	2.2	2.2/6.5	0/0.8	looks very fresh and new, and a pair are hanging around
Kepa Bush Reserve	piwakawaka	mahoe	3	3/6.0	1.5/0.8	not active but at least 5 piwakawaka around here
Watling Reserve	tui	puriri	9	9/10.0	1/1.5	not active
Owairaka	tui	pohutakawa	12	12/17.0	1.5/2	not active
Dingle Dell	tui	kanuka	10	10/14.0	4/1.5	active- can hear chick and see parents flying in to feed
Dingle Dell	tui	pohutakawa	14	14/15.0	3/1.5	not active
Dingle Dell	tui	totara	16	16/20.0	6/2.0	not active
Francis Street Reserve	tui	totara	12			not active
Nirvana park	tui	pohutakawa	10	10/11.0	05./1	not active
Olympic Park	tui	kanuka	5	5.5.5	0.5/1	not active, close to seen juvenile
Miranda Reserve	tui	kanuka	7	7/9.0	2/1.8	not active
Gribblehurst	tui	totara	8	8/12.0	6/1.5	not active
Jagger's Bush	tui	Ligustrum lucidum	18	18/19	0/2.0	not active

Wellpark Reserve	tui	matai	7			not active
Fowlds Park	tui	pohutakawa	8	8/12.0	4/0.5	active- one chick, parents feeding
Kepa Bush Reserve	tui	kanuka	9	9/10.0	2.5/1.5	active, 1 chick being fed
Kepa Bush Reserve	tui	kanuka	14	14/15.0	4/1.5	not active
Kepa Bush Reserve	tui	kanuka	13	13/15	4/1.0	not active
Mount Saint John	tui	pohutakawa	16	16/18	5/1.5	not active
Mount Saint John	tui	pohutakawa	15	15/17	4/1.5	not active
Wellpark Reserve	tui	kanuka	13			not active
Potters Park	tui	pohutakawa	12	12/14.0		not active

Active and inactive nests of tui and piwakawaka were found, but only one active (seen during an autumn bird count and so not recorded on the above table 4.1) and no inactive nests were found for riroriro, which is probably due to field researcher inexperience as well as their nests being quite well-hidden. The nests recorded for piwakawaka were found between 1.8 and 3m from the ground in kawakawa (*Macropiper excelsum*), karaka (*Corynocarpus laevigata*) and mahoe trees (*Melicytus ramiflorus*). Tui nests were seen at estimated heights of 5-18m in a variety (6 species) of trees, but the majority (70%) were seen in kanuka (*Kunzea ericoidesvar. Ericoides*) (35%) and pohutakawa (*Metrosideros excelsa*) (35%) trees.

Shape and size reserve categories

Table 4.2- Proportion of each reserve type with evidence of breeding found

Species	Narrow large	Narrow small	Round large	Round small
Tui breeding	57.14%	57.14%	85.71%	71.43%
Piwakawaka breeding	57.14%	42.86%	28.57%	42.86%
Riroriro breeding	14.29%	28.57%	28.57%	14.29%

Table 4.3- Proportion of each reserve type with juveniles seen

Species	Narrow large	Narrow small	Round large	Round small
Tui juvenile seen	57.14%	28.57%	42.86%	28.57%
Piwakawaka juvenile seen	57.14%	14.29%	14.29%	42.86%
Riroriro juvenile seen	14.29%	28.57%	14.29%	14.29%

Evidence of breeding was found at a higher proportion of round large sites for tui and riroriro, but round large reserves were the lowest reserve category for piwakawaka breeding. Piwakawaka had the highest proportion of sites where breeding was found and where juveniles were seen in the narrow large category, even though the narrow large sites had the lowest spring abundance for piwakawaka. Spring abundance for tui was highest at round large and round small sites, while evidence of breeding was highest at round large also, closely followed by round small; but it was narrow large sites which showed the largest proportion with juveniles tui seen. Riroriro had the greatest number of round large sites with evidence of breeding, but the highest rate of juveniles seen at narrow small, which also showed much higher riroriro spring abundance than the other reserve categories.

Figure 4.2- Tui mean spring abundance for each shape/size category (+/-S.E.)

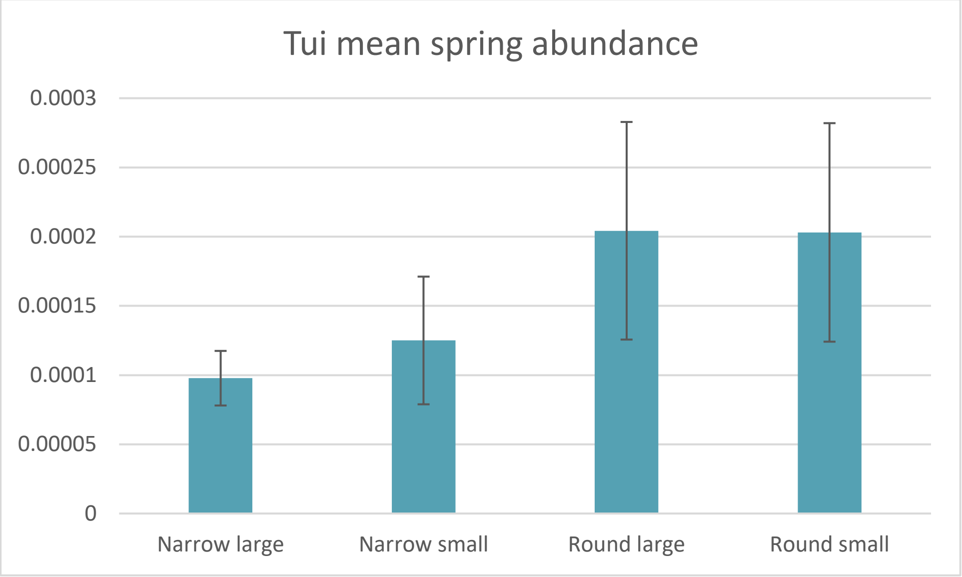


Figure 4.3- Piwakawaka mean spring abundance for each shape/size category (+/-S.E.)

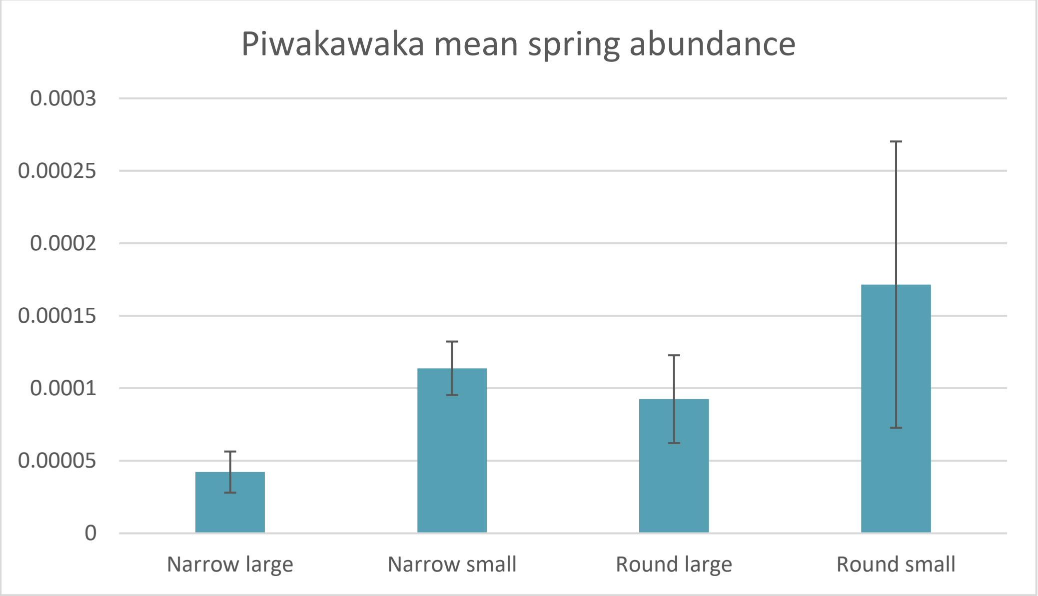
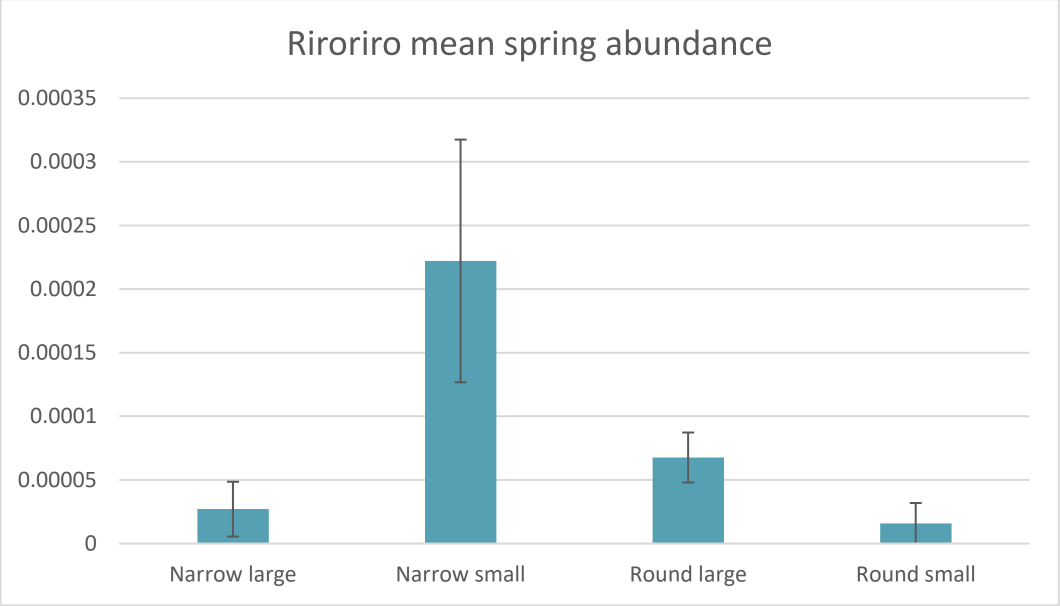


Figure 4.4-Riroriro mean spring abundance for each shape/size category (+/-S.E.)



Habitat features

The binary logistic regression tests showed no statistical significance of all variables tested influencing whether these native species breed at a site, however, there is a strong trend showing that vegetation quality influence whether tui and riroriro breed at a site. There is also a trend showing predator control affecting the breeding of piwakawaka, and predator control, connectivity and adjacent landcover affecting the breeding of Riroriro.

Table 4.4 Binary logistic regression test result- Tui

Variable	B	S.E.	Wald	df	Sig.	Exp(B)	95% C.I.for EXP(B) Lower	95% C.I.for EXP(B) Upper
Distance (m) to closest forest patch(over 5,593m2)	-.002	.003	.620	1	.431	.998	.992	1.003
predator control	.387	1.171	.109	1	.741	1.473	.148	14.620
Vegetation Quality Score	-.623	.375	2.764	1	.096	.536	.257	1.118
SHAPE	.221	.476	.216	1	.642	1.248	.491	3.172
Adjacent landcover	.870	1.598	.296	1	.586	2.387	.104	54.687
presence of water	-1.186	1.164	1.038	1	.308	.306	.031	2.991
Constant	10.469	5.739	3.327	1	.068	35220.988		

Table 4.5 Binary logistic regression test result- Piwakawaka

							95% C.I.for EXP(B)	
Variable	B	S.E.	Wald	df	Sig.	Exp(B)	Lower	Upper
predator control	1.685	1.075	2.458	1	.117	5.395	.656	44.363
Distance (m) to closest forest patch(over 5,593m2)	-.003	.003	.964	1	.326	.997	.992	1.003
Vegetation Quality Score	.046	.302	.023	1	.880	1.047	.579	1.892
SHAPE	-.092	.449	.042	1	.837	.912	.378	2.198
Adjacent landcover	-.191	1.470	.017	1	.896	.826	.046	14.726
presence of water	.029	1.060	.001	1	.978	1.029	.129	8.211
Constant	-1.128	4.400	.066	1	.798	.324		

Table 4.6 Binary logistic regression test result- Riroriro

							95% C.I.for EXP(B)	
Variable	B	S.E.	Wald	df	Sig.	Exp(B)	Lower	Upper
predator control	1.886	1.553	1.475	1	.225	6.594	.314	138.429
Distance (m) to closest forest patch(over 5,593m2)	.005	.003	2.649	1	.104	1.005	.999	1.012
Vegetation Quality Score	.049	.298	.027	1	.869	1.050	.586	1.885
SHAPE	-.436	.543	.645	1	.422	.646	.223	1.874
Adjacent landcover	2.615	1.961	1.778	1	.182	13.673	.293	638.861
presence of water	-.947	1.364	.482	1	.488	.388	.027	5.623
Constant	-.867	4.835	.032	1	.858	.420		

Table 4.7-Tui breeding

Parameter Estimates

Parameter	B	Std. Error	95% Wald Confidence Interval Lower	Upper	Hypothesis Test Wald Chi-Square	df	Sig.
(Intercept)	-11.148	5.4181	-21.767	-.529	4.234	1	.040
Distance (m) to closest forest patch(over 5,593m2)	.003	.0027	-.002	.008	1.488	1	.223
Vegetation Quality Score	.682	.3479	5.421E-5	1.364	3.842	1	.050
predator control	.384	.9722	-1.521	2.290	.156	1	.693
(Scale)	1[a]						

Table 4.8- Piwakawaka breeding

Parameter Estimates

Parameter	B	Std. Error	95% Wald Confidence Interval: Lower	95% Wald Confidence Interval: Upper	Hypothesis Test: Wald Chi-Square	Hypothesis Test: df	Hypothesis Test: Sig.
(Intercept)	1.743	4.1748	-6.439	9.926	.174	1	.676
Distance (m) to closest forest patch(over 5,593m2)	.003	.0027	-.002	.008	1.105	1	.293
Vegetation Quality Score	-.081	.2862	-.642	.480	.081	1	.776
predator control	-1.770	.9264	-3.586	.045	3.652	1	.056
(Scale)	1[a]						

Dependent Variable: Piwakawaka breed

Model: (Intercept), Distance (m) to closest forest patch(over 5,593m2), Vegetation Quality Score, predator control

a. Fixed at the displayed value.

Table 4.9- Riroriro breeding

Parameter Estimates

Parameter	B	Std. Error	95% Wald Confidence Interval Lower	95% Wald Confidence Interval Upper	Hypothesis Test Wald Chi-Square	Hypothesis Test df	Hypothesis Test Sig.
(Intercept)	2.801	4.0254	-5.088	10.691	.484	1	.487
Distance (m) to closest forest patch(over 5,593m2)	-.002	.0022	-.007	.002	1.233	1	.267
Vegetation Quality Score	-.067	.2659	-.588	.455	.063	1	.802
predator control	-.664	1.0112	-2.646	1.318	.431	1	.511
(Scale)	1[a]						

Dependent Variable: Riroriro breed

Model: (Intercept), Distance (m) to closest forest patch (over 5,593m2), Vegetation Quality Score, predator control

a. Fixed at the displayed value.

To further explore these results, habitat factors have been displayed below with comparisons made between sites where there was evidence of breeding or juveniles were found, and sites where there was not.

Table 4.10- Comparison of breeding found at sites with and without predator control

Species	Breeding in sites with predator control	Breeding in sites with no predator control
Tui	63.64%	70.59%
Piwakawaka	72.73%	23.53%
Riroriro	27.27%	23.53%

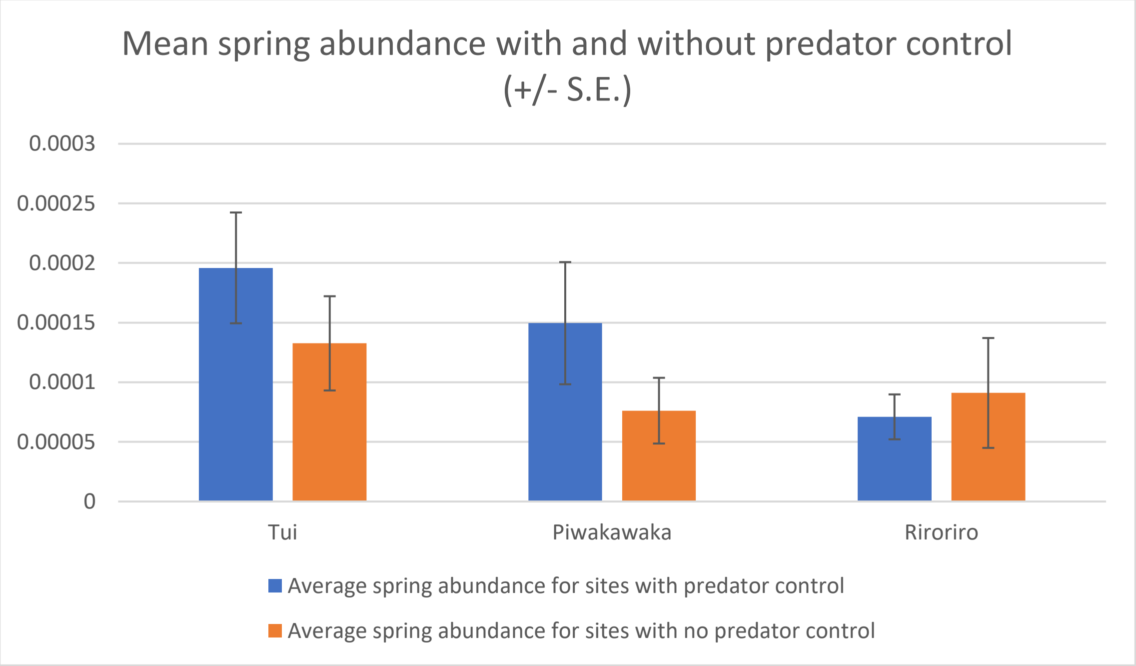
There was little difference between reserves with or without predator control for tui and riroriro breeding, but piwakawaka had a much higher rate of breeding seen at reserves with predator control.

Table 4.11- Comparison of juveniles seen at sites with and without predator control

Species	Juveniles seen in sites with predator control	Juveniles seen in sites with no predator control
Tui	36.36%	41.18%
Piwakawaka	54.55%	17.65%
Riroriro	27.27%	11.76%

Juvenile piwakawaka, tui and riroriro were seen at a greater proportion of sites which had predator control.

Figure 4.5- Mean spring abundance for sites with and without predator control



The mean spring abundance was higher for tui and piwakawaka at sites with pest control, but this was not the case for riroriro.

Table 4.12- Comparison of breeding found at sites with and without freshwater

Species	Breeding in sites with water	Breeding in sites with no water
Tui	66.67%	69.23%
Piwakawaka	60%	23.08%
Riroriro	26.67%	23.08%

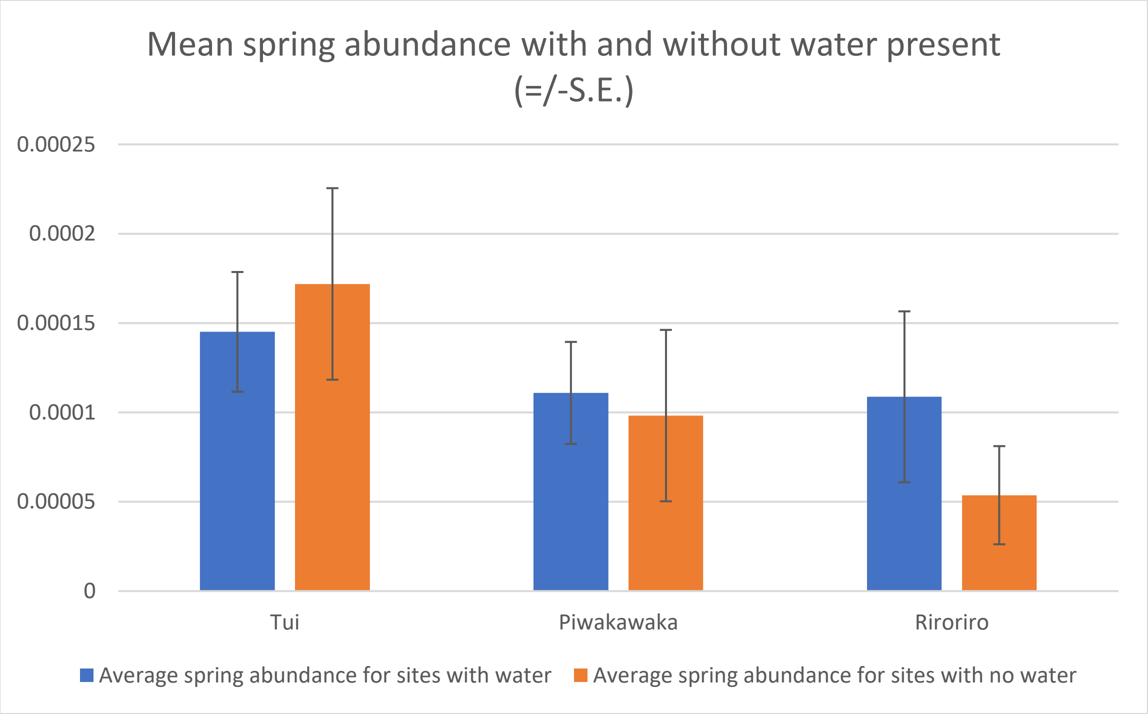
Although there appeared to be little influence of the presence or absence of freshwater for tui and riroriro breeding, piwakawaka were found to be breeding at a much higher rate at reserves with a source of freshwater.

Table 4.13- Comparison of juveniles seen at sites with and without freshwater

Species	Juveniles seen in sites with water	Juveniles seen in sites with no water
Tui	33.33%	46.15%
Piwakawaka	40%	23.08%
Riroriro	20%	15.38%

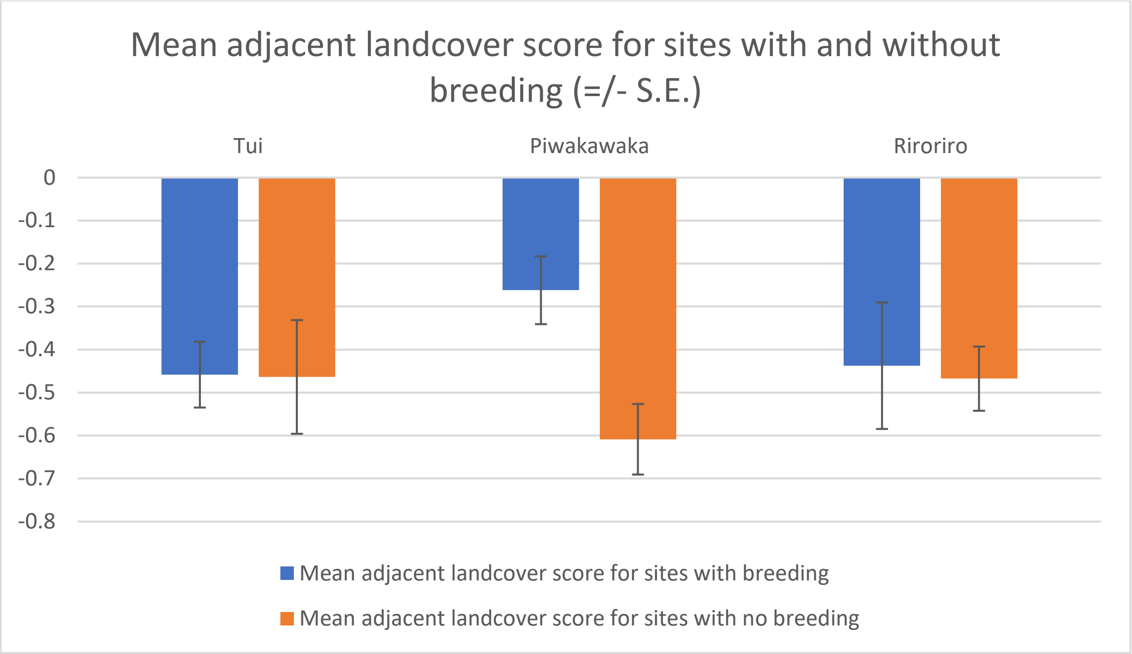
Piwakawaka juveniles were seen at almost twice the proportion of reserves with a source of freshwater that those without.

Figure 4.6- Mean spring abundance for sites with and without freshwater source (=/- S.E.)



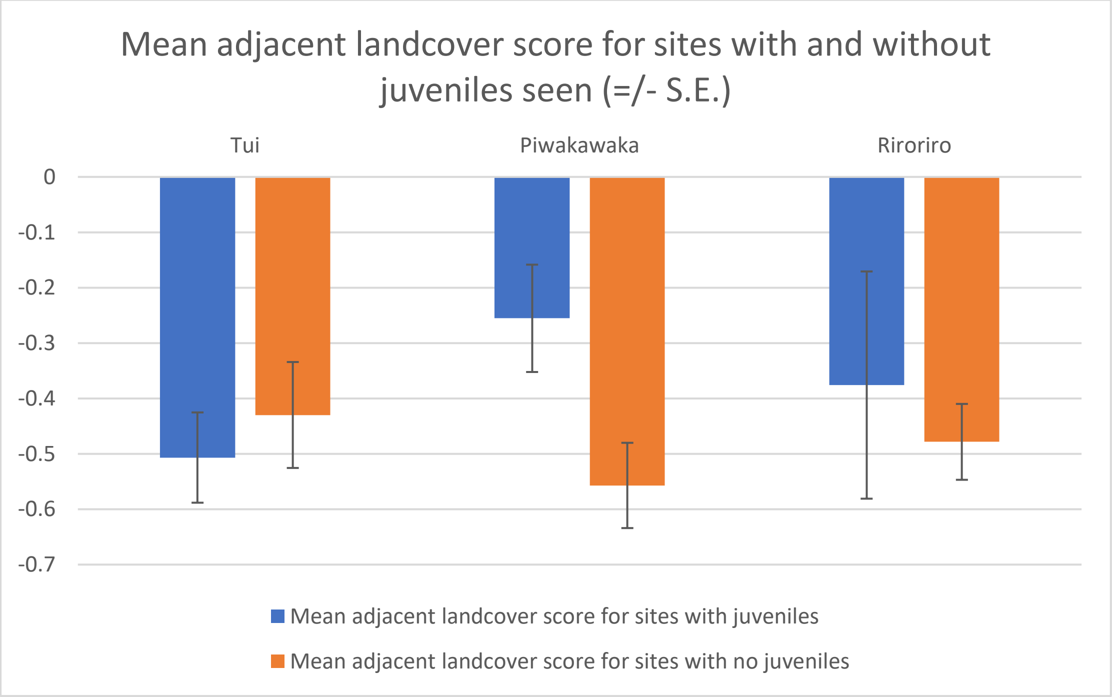
The mean spring abundance for piwakawaka and riroriro was higher at sites with a source of freshwater, but this was not the case for tui.

Figure 4.7- Mean adjacent landcover score for sites with and without evidence of breeding (=/- S.E.)



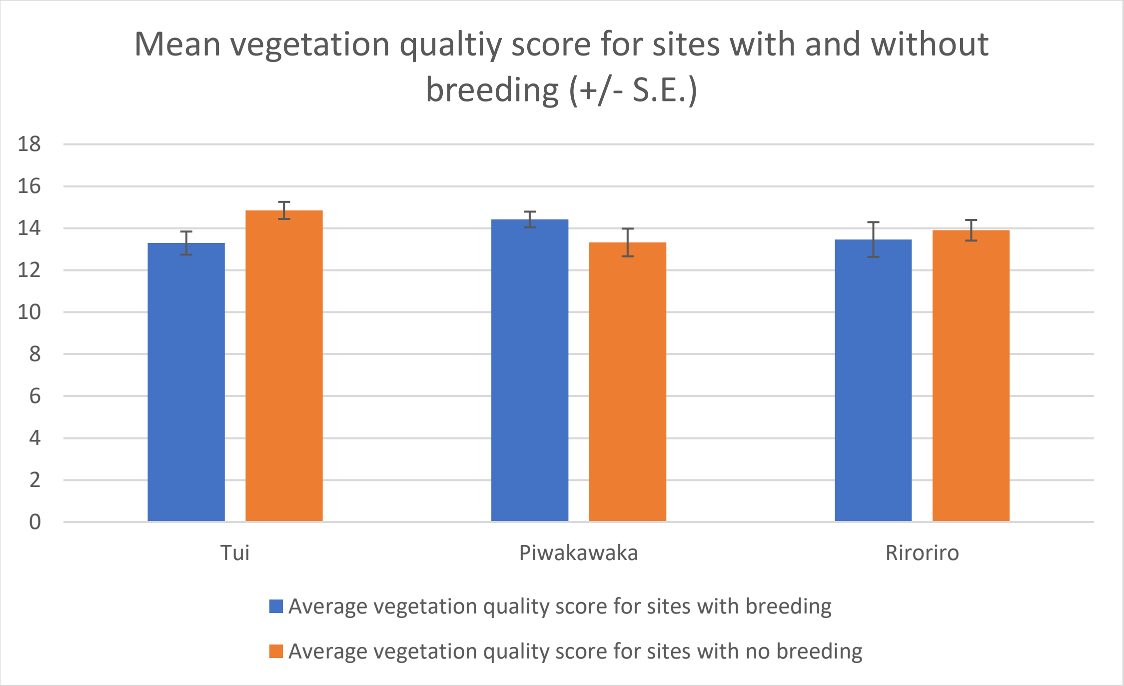
The mean adjacent landcover score was higher for reserves where evidence of breeding was found for all three species, with the greatest difference being for piwakawaka.

Figure 4.8- Mean adjacent landcover score for sites with and without juveniles seen (=/- S.E.)



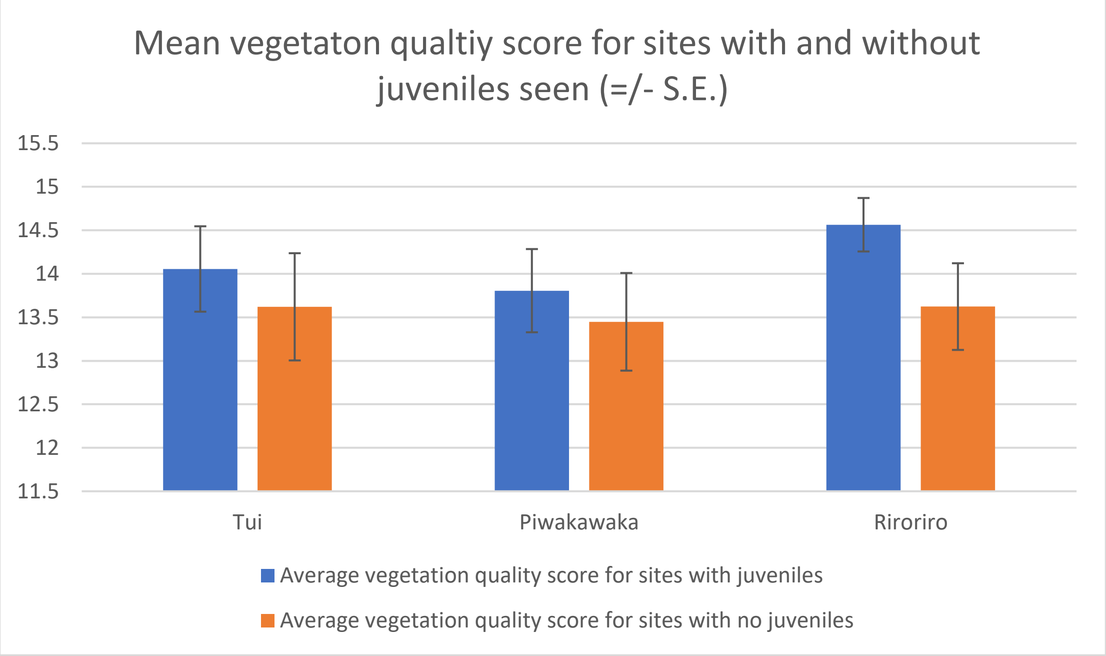
The mean adjacent landcover score was higher for the reserves where juvenile piwakawaka and riroriro were seen, but not for tui.

Figure 4.9- Mean vegetation quality score for sites with and without evidence of breeding (+/- S.E.)



The reserves where piwakawaka were found to be breeding had a slightly higher mean vegetation quality score, but this was not the case for tui and riroriro.

Figure 4.10- Mean vegetation quality score for sites with and without juveniles seen (+/- S.E.)



The mean vegetation quality score was higher for the sites where juveniles were seen for all three species.

Discussion

Reserve shape and size

Tui had the highest spring abundance at round large sites and the greatest proportion of round large sites had evidence of breeding, as might well be expected for a species with larger territories (Bergquist, 1985). Evidence of tui breeding was observed at a higher proportion of round small and narrow small sites, but juveniles were only seen at just over a quarter of both small reserve categories, which could be an indication of nest failure, or because smaller sites provide limited resources and tui fledgelings may have quickly dispersed from these reserves. Tui need larger breeding territories of around 0.5km in diameter and tend to nest near related individuals (Bergquist, 1985), so the round large sites could potentially be preferential because they provide sufficient area to allow family groups to establish breeding territories near one another. Interestingly, although a greater proportion of round large sites had evidence of tui breeding activity, it was the narrow large category which showed the highest proportion of sites with juveniles seen, and this was also the only category where the proportion of sites with evidence of breeding seen was the same as the proportion with juveniles seen. This result could indicate high breeding success for tui at narrow large sites and may be consistent with a previous urban bird breeding study which found medium suburban reserves to have the highest breeding success of the reserve sizes surveyed (Donnelly and Marzuff, 2004).

Piwakawaka had the highest spring abundance at round small sites, but they did not have the highest level of evidence of breeding or juveniles found at these sites. This finding could indicate that factors other than reserve shape and size were more important for the breeding of piwakawaka, which would not be surprising given that they require smaller territories and are edge-adapted. Evidence of piwakawaka breeding and juveniles seen were highest at narrow large sites, showing a good rate of successful breeding, despite a low proportion of narrow large reserves having predator control. Once again, this finding could support previous findings of medium-sized suburban reserves having the highest breeding success (as found by Donnelly and Marzuff, 2004).

Riroriro showed the highest level of evidence of breeding at round large and narrow small sites, which both had higher spring abundance than the other two categories, with narrow small sites in particular showing much higher spring abundance than other reserve categories. The narrow small category also had the highest proportion of sites where riroriro juveniles were seen. Because only one nest was found for riroriro the search effort cannot be exactly quantified, and all other breeding observations made were of parents feeding fledged juveniles and are evidence of successful breeding rather than including breeding attempts. The data for juveniles seen includes sightings of juveniles during subsequent bird count observations as well as during the nest search. Furthermore, several shining cuckoo juveniles were observed (often being fed by riroriro) during the bird counts and breeding survey, which were representative of riroriro breeding effort which was not accounted for in this study.

It is possible that the area or width of some of the smaller sites are insufficient to provide for an optimal tui breeding territory, but the breeding of piwakawaka, which have smaller territories, was not lower for small narrow sites than for any other categories except narrow large sites, indicating that further studies exploring the minimum habitat area and width for the breeding territories of different native species are needed to better inform reserve planning

Contrary to other studies (e.g. Donnelly and Marzuff, 2004), small urban reserves were found to have some value as breeding habitats for native birds. Narrow small sites were not the lowest category for spring abundance, detecting the presence of breeding, or sighting juveniles for any of the three study species. This indicates that it is possible for smaller linear elements to be managed as valuable breeding habitat as well as increasing connectivity, catching dispersers migrating through the matrix, and reducing the cost and increasing the likelihood of birds migrating between habitat patches. Small reserves of either shape category did not appear to be acting as population sinks due to area, and providing that the observed breeding can be managed in a way that supports the highest possible nest success and juvenile survivorship, which is likely to require some level of predator control, these reserves could potentially act as population sources to varying degrees depending on their vegetation quality and adjacent landcover. The difference in level of breeding activity for the different species in each reserve type could be an indication of the effect of territory size. There are likely to be different minimum reserve areas and widths required for each of the three different species to establish breeding territories.

The inclusion of larger habitat patches within the network is important to provide the greatest sources of colonists and because larger populations are more resilient to some kinds of stochasticity, however, this study contributes practical evidence to the theoretical suggestion that remaining larger fragments in the urban landscape should be connected by linear habitat elements where possible, and smaller stepping stones where it is not (Schippers et. al., 2009). This study also provides further evidence that linear habitat elements can be valuable as habitats, and not just as connecting corridors as they have often been viewed (Ignatieva et. al. 2011).

Habitat features

There was a strong trend seen in the binary logistic regression showing that vegetation quality affects tui breeding. Tui had higher spring abundance at sites with predator control, but sites with predator control did not show a greater proportion of sites where evidence of breeding or juveniles were seen. Presence of a freshwater source was not associated with an increase spring abundance, evidence of breeding, or juveniles seen for tui, which is not surprising as they are strong fliers capable of accessing water some distance away. Shape and size of reserves appear to be more important for tui due to their larger territory sizes and family group proximity preference, and some habitat features appear to be less important because of their strong flying ability.

Although statistically significant results for this were not able to be obtained in this study, an indication that piwakawaka are sensitive to habitat features was apparent in that they showed greater abundance at sites with predator control and a source of freshwater, and also showed much higher evidence of breeding and juveniles seen at sites with predator control than sites without predator control. Sites with a source of freshwater showed higher evidence of breeding and showed higher rate of sites with juveniles seen. This is likely to be due to advantages of both proximity to drinking water, and also freshwater invertebrates, which piwakawaka feed on. Piwakawaka also showed more evidence of breeding at sites with higher average vegetation quality, and higher rates of juveniles seen. Sites where breeding of piwakawaka was observed also showed a much greater mean adjacent landcover quality than sites without evidence of piwakawaka breeding. These findings suggest that piwakawaka could be a useful indicator species in urban reserves in that although they are well adapted to edge habitat and disturbance, they are also sensitive to differences in habitat quality factors.

Spring abundance for riroriro was not higher at sites with predator control but was twice as high at sites with freshwater than without. Sightings of riroriro juveniles was higher in sites with predator control and in sites with a source of freshwater. Sites with riroriro juveniles seen had a higher vegetation quality score than sites without, and the binary logistic regression showed a strong trend of vegetation quality affecting riroriro breeding. Riroriro breeding also showed a positive response to adjacent landcover quality, with sites where juveniles were seen having a higher average landcover quality than sites where none were seen. It is probable that observer bias (the lack of ability to spot riroriro nests by visual search in the absence of activity) may have influenced the results for riroriro but results of juvenile sightings indicate that their breeding is influenced by variation in habitat features. Their response in this study to adjacent landcover quality may be in accordance with the observations of previous studies that found riroriro move freely in and out of habitat fragments and suburban gardens, and it is possible that they could be less dependent on large forest fragments than other native species (van Heezik et. al. (2008(a)).

Predator control was confirmed, as expected, to be important to the breeding success of all three urban native bird species studied. Future studies should seek to gather more information on the effect of the frequency and extent of predator control measures to provide specific guidelines for the management requirements of urban native bird breeding. Although we did not find lower rates of successful breeding in narrow reserves during this study, predator control is likely to be especially important in narrow reserves due to their longer perimeter and greater invasibility, so if this reserve type is to become prevalent in urban conservation then the ongoing monitoring of nest predation will be crucial to ensure that these habitat patches are not acting as population sinks. Our study found some indications of vegetation quality influencing the breeding of native bird species, and all 3 species had a higher mean vegetation quality at sites where juveniles were seen. This factor should be explored further in future studies on urban native bird breeding, and our observations of particular tree species being predominantly selected for nesting (see **Table 4.1**, page) could suggest that these tree species are particularly important to include in planting plans for urban reserves to ensure optimal native bird breeding habitat.

Conclusion

This study shows a previously unreported finding- that the shape of urban reserves does not influence their ability to provide breeding habitat for native birds. Narrow large reserves were the category with the most juveniles seen for both tui and piwakawaka, showing that the potential increase in edge effect caused by less round habitat shapes is not an important factor for the breeding of urban tolerant native bird species in a medium-density urban environment. Reserve size appeared only to affect bird species with larger breeding territories, with tui showing greater breeding success at large round and narrow reserves. Piwakawaka bred most successfully at narrow large and round small, and riroriro at narrow small sites (where they were also seen in much higher abundance).

Reproduction should be possible in smaller linear habitat patches for them to be considered part of a metapopulation’s habitat network (Schippers et. al., 2009) and this current study shows that such reserves can indeed provide breeding habitat. Future studies assessing urban avian metapopulation viability should look at the value of small linear habitat elements such as street trees and motorway plantings, which are often considered part of the matrix, in terms of connectivity, resource provision, and mortality risk. Understanding the habitat area, width, and quality thresholds at which linear patches become valuable as habitat as well as connectors could enable this habitat type to be effectively enhanced throughout cities, contributing to the sustainability of metapopulations. A set of area and width thresholds should be developed for a range of native indicator species that have a wide variety of habitat and territory requirements. Piwakawaka and tui appear to be a useful combination of indicator species for native bird biodiversity in urban reserves in New Zealand as although both species are able to adapt to the urban environment, they are of different guilds and have differing habitat requirements.

It is important to better understand which habitat patches are likely to be valuable as population sources and which are likely to be less valuable, so that a balance can be maintained in the landscape to support a viable metapopulation. Rather than focussing, as the literature overwhelmingly insists, only on increasing the area of existing habitat patches, instead prioritising the creation of narrow reserves permeating the urban matrix could provide more breeding habitat for native birds as well as connecting small, medium and larger habitat patches throughout a city. The addition of linear habitat patches to existing landscapes containing a mixture of medium and small patches could have a threefold advantageous effect on the overall viability of a regional metapopulation: 1.) increasing connectivity (and decreasing migration costs), 2.) increasing total habitat area, and 3.) increasing the number of populations in the regional metapopulation, all of which factors are major influences on metapopulation viability. Although it has been previously reported that the presence of bird species which are sensitive to urban disturbance require linear habitats to be of certain minimum widths (Mason et. al, 2007), this current study suggests that it is also important to establish minimum widths for breeding territories as well.

Climate change will increase extreme weather events and species ranges may shift, so linear habitat elements and more small patches are all the more important to increase resilience to stochastic events and facilitate species movement. To better inform planning and management decisions regarding urban reserves, area and width thresholds should be established of what can be considered 1.) a large patch that can support valuable source populations, 2.) a small patch or narrow corridor which may provide foraging opportunities and connectivity but is not valued as breeding habitat, and 3.) a medium patch or corridor of sufficient width that can provide breeding habitat as well as resources and connectivity. Thresholds for native indicator species that require more area, such as tui, and that are sensitive to habitat features, such as piwakawaka, should be investigated and developed into guidelines for urban reserve planning and management. More evidence regarding the breeding success of native birds in urban reserves is desperately needed in a world where increasing urbanisation is threatening both the conservation of species and the connection between humans and the natural environment.

Chapter 5: General discussion

Figure 5.1- Active tui (*Prosthemadera novaeseelandiae)* nest recorded during breeding survey

Research summary

Our results confirmed that the size of reserves has a significant effect on native bird diversity, with larger reserves having higher diversity in terms of species richness and abundance. There is also an indication that reserve size has a significant effect on overall bird diversity and may positively affect the breeding of the native bird species studied. Such trends have also been found by previous studies of urban bird populations (Donnelly and Marzuff, 2004, Carbo-Ramirez and Zuria, 2011, Chang and Lee, 2016, Leveau et. al., 2019).

Reserve shape was a particularly important factor to explore due to the scarcity of studies addressing the effect of urban habitat shape and the recent proliferation of greenways, urban greenspaces of a narrow or linear design (Ignatieva et. al. 2011, Horte and Eisenman, 2020). Our study did not find an effect of reserve shape on bird diversity, native bird diversity or breeding of the three native species studied. In fact, narrow large reserves were found to have the highest mean bird diversity and the highest mean native bird diversity of all four reserve categories. This is likely to be because, as previous studies have concluded, human disturbance is ubiquitous in urban reserves, negating the impact of edge effects (Rodriguez-Prieto et. al., 2014). The narrow reserve categories, when compared to the round categories, showed higher rates of juvenile birds seen for all of the 3 study species, thus providing evidence of breeding success at narrow reserves and therefore that these habitats can contribute greatly to urban bird diversity.

As has been previously stated in the literature, the persistence of populations in fragmented habitat patches cannot be explained by patch area alone (Aurambout et. al., 2005). Various aspects of the habitat features studied were found to have significant effects on avian diversity, native bird diversity, and breeding. That vegetation diversity, structure and composition are an important influence on bird communities has been well-established in the literature, and many studies have shown that increased vegetation quality has a positive effect on avian diversity in urban areas (Donnelly and Marzluff, 2004, Chace and Walsh, 2006, Carbo-Ramirez and Zuria, 2011, Strobach et. al., 2013, Kang et. al., 2015, Aida et. al, 2016, Rodrigues et. al., 2018). This effect has been shown to be of particular importance to native birds in the New Zealand urban context (van Heezik et. al., 2008(b)), so we were interested in testing the effect of our vegetation quality score, which incorporated diversity, structure and maturity, on native bird diversity and found that our results affirmed that the effect of vegetation quality on native bird diversity was indeed significant. This result highlights the importance of assessing and improving the vegetation quality of existing reserves by increasing the proportion of tree cover, the plant diversity, and the structural complexity. There was also a trend found for vegetation quality affecting whether tui breed at a site or not, indicating that further studies comparing native bird breeding at sites with varying vegetation quality are warranted.

The landcover adjacent to urban reserves has previously been found to significantly influence bird diversity, with a greater proportion of vegetated landcover associated with greater bird diversity and a greater proportion of paved and built landcover having a negative effect, (Mason et. al., 2007, Carbo-Ramirez and Zuria, 2011, Nielson et. al., 2014, Heggie-Gracie et. al., 2020). In New Zealand adjacent landcover has been shown to be a significant factor influencing native bird diversity in urban reserves (van Heezik and Adams, 2016, Heggie-Gracie et. al., 2020), and our study also confirms that a greater proportion of vegetation of the landcover adjacent to reserves has a significant positive effect on native bird diversity.

Piwakawaka showed a trend toward more evidence of breeding at sites with predator control. This was not surprising as predation is known to be a major cause of nest failure and juvenile mortality in urban reserves (Donnelly and Marzuff, 2004, van Heezik et. al., 2008(a)). It has been previously recorded that rats (*Rattus rattus*), possums (*Trichosurus* *vulpecula*) and possibly mice (*Mus musculus*) predate piwakawaka nests in urban areas, and that nest survival can be as low as 5-7% (van Heezik et. al. (2008(a)). Our finding highlights the importance of ongoing predator control measures in urban reserves, particularly during the breeding season.

The effect of the presence of a freshwater source, or lack thereof, on bird breeding had not been previously studied (to the best of my knowledge). We recorded higher spring abundance, evidence of presence of breeding activity and juveniles seen in sites containing a source of freshwater for the two smaller species studied, piwakawaka and riroriro. However, the data set was too small for meaningful statistical testing. Further research on this topic could be valuable, as a previous study has found that in urban greenspaces bird species richness increases with decreased distance to fresh water (Zivanovic and Luck, 2016). If it can be confirmed that the presence of freshwater has a significant positive effect on native bird communities, the creation and maintenance of riparian urban reserves could be justifiably prioritised above other reserve types.

Improving urban habitats for native bird populations

Increasing the number of habitat patches available to a metapopulation tends to increase their patch occupancy and resilience to stochasticity, consequently reducing the chance of metapopulation extinction (Hanski, 1998). Therefore, any opportunity to increase the number of valuable habitat patches in a city should be taken. Connectivity is also a crucial factor to consider because metapopulations with greater connectivity between patches have higher recolonization rates and lower mortality-during-migration rates, which can be a very high cost for species which are weak dispersers or are crossing an inhospitable matrix with high mortality risk (Schippers et. al., 2009).

It has been pointed out by the authors of previous studies investigating avian species richness in urban reserves that although increasing the size of reserves would be of great benefit to the avian diversity found within them, in densifying cities it is practically impossible to do so due to the high demand for land (Carbo-Ramirez and Zuria, 2011). Narrow habitat patches are much easier to acquire land for as they can be multi-purpose, and biodiversity objectives can be combined with infrastructural necessities, which is why greenways have become increasingly popular reserve types to create in urban areas (Horte and Eisenman, 2020). A network of narrow reserves (biodiverse greenways) could be used to achieve multiple outcomes such as improving flood resilience and transport options, which would help to justify the acquisition of land for increasing biodiversity in nature-poor urban areas.

The narrow sites in this study were situated in areas with a high level of tree cover in the residential surrounding matrix, meaning that the provision of connectivity was less important, however in a densely urbanised and nature-poor urban setting such reserves could be vital to ensure connectivity. This study shows that narrow reserves can provide valuable habitat for a diverse urban native bird community and also for the breeding of the native species studied. I recommend that linear habitats with sufficient area, width and vegetation quality are created throughout cities at every possible opportunity to enhance urban avian biodiversity. Because Auckland, like many cities around the world, is becoming more densely urbanised, connecting corridors are also becoming more important. This study begins to provide practical evidence to support the previously modelled theory (Schippers et. al., 2009) that a network of small and large patches connected by linear habitat elements can support a viable regional population of urban native birds and could provide a solution to the problem of biodiversity loss in densifying cities.

There was an indication of an effect of reserve size implied particularly for the breeding of tui, which require larger habitat size territories than the other two species studied, piwakawaka and riroriro, and had higher rates of breeding success (juveniles seen) detected at round large and narrow large sites. This illustrates the necessity of a variety of habitat patch sizes in a network to provide for the requirements of different species, and it should be cautioned that the creation of new linear habitat elements in an urban landscape cannot justify the sacrifice of larger patches. Furthermore, the creation and maintenance of narrow reserves should take account of recommendations from previous studies on greenways which show that the vegetation complexity should be maximised, the proportion of managed (e.g. paved, mowed and sprayed) area should be minimised, paths should be near reserve edges wherever possible, and the width of greenways should be as great as possible, to provide the best conservation outcomes (Mason, 2007, Matsuba et. al., 2016). Our finding that greater vegetation quality has a significant positive effect on avian diversity in urban reserves also confirms that vegetation diversity and maturity should be prioritised to support the greatest avian diversity. The development of a set of minimum standards for urban reserve design to achieve conservation outcomes for native urban birds could be extremely advantageous to ensure that urban greenspaces created for social and infrastructural outcomes such as transport routes can also provide habitat valuable to native birds.

Adjacent landcover is another factor which we found to have a significant effect on native bird diversity, and which previous studies have also identified as significantly influencing urban bird diversity (Mason et. al., 2007, Heggie-Gracie et. al., 2020). Although adjacent landcover is not within the control of the greenspace management, reserves with highly urbanised adjacent landcover should be considered a greater priority for the improvement of vegetation quality than reserves with a greener adjacent landcover as they are more likely to be vitally important for the provision of habitat and connectivity in such areas.

Predator control is particularly important in reserves with a larger perimeter to core ratio (Sinclair et. al., 2005), and our finding that piwakawaka had higher rates of breeding and juveniles seen at reserves with predator control confirms that ongoing investment in predator control methods will be highly important for maintaining native bird diversity in urban reserves, and also implies that predator control throughout the breeding season may be necessary to ensure that reserves do not become population sinks.

The presence of a freshwater source is a factor affecting urban avian conservation which should be explored more extensively, as it has been previously found to positively influence bird diversity and may be of particular importance for some species of bird which are not strong fliers (Melles et. al., 2003). Although our study did not obtain sufficient data to test for statistical significance, we did observe that riroriro and piwakawaka had higher average abundance and a higher rate of juveniles seen at reserves with freshwater, and we also noted that the reserve shape/size category with the lowest native bird diversity was the round small category, which had (weakly significantly) a lower proportion of reserves containing freshwater. These results warrant further investigation and suggest that reserves created around a water source such as wetlands or riparian reserves along urban creeks could be particularly beneficial to urban avian conservation.

Social benefits of enhanced reserve networks

There are many powerful reasons to enhance urban reserve networks for the benefit of humans. Natural settings have been found to be capable of promoting psychological well-being), for example decreased counts of anxiety/mood disorder treatments have been found to be associated with decreased distance to urban green space and increased proportion of green space within the neighbourhood (Hartig and Staats, 2006, Nutsford, et. al., 2013). Stress reduction and attention restoration are examples of benefits derived from natural settings for which much empirical evidence has been found (Carrus et. al. 2015). Evidence has also been found which indicates that physical health can be increased with increased access to greenspace, for example increased greenness of surrounding environment has been found to be associated with decreased prevalence of obesity and sedentary behaviour in children (Dadvand et. al., 2014). The health and well-being benefits derived from greenspace have been shown to be greater when the greenspace is more biodiverse (Carrus et. al., 2015, Wheeler et. al., 2015). The physical benefits derived from greenspace are likely to be greater with increased biodiversity because immunological health and regulation are positively impacted by exposure to organisms present in biodiverse areas (Rook, 2013). This large and growing body of knowledge means that increasing the prevalence and the vegetation quality of greenways and other types of urban reserves can be an easily justified investment in human well-being.

A further social advantage of creating greenways or narrow reserves throughout a city is their ability to improve access and proximity to nature for the great many people living in urban areas with very little tree cover. The benefits of access to and interaction with nature are not equally distributed, with many studies in cities around the world, including Auckland, NZ, documenting findings such as more tree cover, greater tree protection, and higher species diversity in areas of socio-economic advantage (Shanahan et. al, 2014, Wyse et. al., 2015, Leong et. al. 2018). Surveys have also found that people are more likely to frequently use greenspace when the time taken to reach the greenspace is shorter (Dallimer et. al., 2014). Ensuring that people in cities have access to nature is vital for securing ongoing support and funding for nature conservation because direct contact with nature fosters environmental awareness, pro-environmental attitudes, and willingness to conserve nature (Zhang et. al. 2014, Charles and Linklater, 2015). Cities are home to the largest and densest human populations and have the potential to be the greatest interface between humans and nature. Without access to nature, more than half the world's people may not care about or support the preservation or restoration of biodiversity.

Increasing focus on the infrastructural and economic benefits of urban forest can help to rationalize greater investment in improving natural habitats in a city. Quantifying and valuing the ecosystem services provided to society by urban green spaces is necessary to fully take account of the benefits they contribute, and to accurately inform planning decisions involving their creation and maintenance (Klimas et. al., 2016). Ecosystem services provided by urban forest include micro-climate regulation, the reduction of noise and air pollution, stormwater attenuation and carbon sequestration (Schwendenmann and Mitchell, 2014, Klimas et. al., 2016). If greenways were created as active transport routes and by daylighting and restoring streams and wetlands with native vegetation throughout a city, as well as the aforementioned ecosystem service benefits, this also could result in substantially improved access to nature for the inhabitants of the city, and potentially provide connecting corridors, a greater number of habitat patches, and greater overall habitat area for urban native birds as well.

Future research

To obtain greater understanding the influence of reserve shape on urban avian diversity, it is vital to gain more knowledge of the impact of reserve width. An American study established that urban reserves which were less than 50m in width were not utilised by urban-sensitive bird species, and some urban-sensitive species were not found in reserves with a width less than 300m (Mason et. al., 2007). If future studies can determine a minimum reserve width for supporting native avian diversity for New Zealand species this could help to avoid resources and land being allocated to the creation of greenways and linear reserves which have social value, but little or no biodiversity value. Comparing the native bird diversity and breeding in narrow urban reserves of different widths is necessary to establish a width threshold for native birds.

We observed a trend for vegetation quality score affecting whether tui breeding was found at a site or not, which could possibly be because of food resources supplied by the nectar of some trees. Although tui also include invertebrates in their diets in the breeding season, phenology studies of sites where tui breed and do not breed could confirm whether the presence of particular species of trees which provide nectar resources is positively associated with tui breeding. This information could be extremely useful in conjunction with reserve size thresholds, as reserve managers could ensure that the correct tree species are available for the provision of tui food in reserves which are of adequate size for their family-group adjacent breeding territories.

Although we obtained tree species data in our vegetation survey, we did not have the capacity to analyse differences between sites in the proportion of native and exotic trees, or to explore the effect of such differences on native bird diversity and breeding. Because previous studies have already shown that urban native bird communities are positively influenced by the presence of native vegetation it was not considered to be a research priority for the current study, but future studies assessing the effects of vegetation quality on urban avifauna could explore the importance of this factor further to hep provide guidance for the planting of urban greenspaces (Day, 1995, Donnelly and Marzluff, 2004, Chace and Walsh, 2006, van Heezik et. al., 2008(b)).

Our finding that both the vegetation quality and the adjacent landcover of reserves had significant effects on native bird diversity confirms that improving the vegetation of existing urban reserves and the areas adjacent to them should be a conservation priority. The relative importance of vegetation quality in reserves situated within a matrix of various urban densities could be a valuable factor to explore to aid the most effective allocation of effort and resources to improve urban reserves. It is likely that the provision of both habitat and connectivity are much more important to maintain the presence of native birds in areas with a highly urbanised matrix, and quantifying this facet could provide persuasive evidence to prioritise the improvement of reserves in the most densely urbanised areas.

We recorded higher presence of breeding activity for piwakawaka and riroriro at sites with a source of freshwater but did not obtain enough comparable data to statistically analyse successfully. It was also thought possible, but unable to be confirmed, that a lower proportion of round small sites containing freshwater could be a contributing factor to the lower native bird diversity found in these sites. A comparison of a larger set of data on native bird breeding and diversity data collected at reserves with and without freshwater, with similar size, shape, and vegetation quality, could potentially provide more definitive results regarding the importance of freshwater to native bird breeding and diversity in the urban context. Tree heights and presence of tree cavities are vegetation factors which were not measured in this study but have been previously shown to influence bird diversity. Future studies comparing native bird diversity at different types of urban reserve should include these vegetation factors if possible.

An important limitation of metapopulation modelling is that it struggles to incorporate the effect of the quality of the matrix on population dynamics, which is a hugely significant weakness considering that mortality during migration can be extremely costly to populations existing within an inhospitable matrix such as a densely urbanised landscape, and has been considered the main disadvantage of the several small habitat patch network in comparison to single large patches (Hanski, 1998, Schippers et. al., 2009). Future studies on bird populations in urban reserves should incorporate data on the landcover of the matrix when analysing connectivity.

During data collection for this study, observing the neighbourhoods surrounding the study sites appeared to suggest lower tree cover in the lower socio-economic neighbourhoods, and the round small and round large reserves in these neighbourhoods appeared to be those with less tree cover. Although it was beyond the scope of this study, the link between socio-economic factors and surrounding matrix cover could be an interesting factor to explore for future studies. The literature shows that poor neighbourhoods tend to have less tree cover, and reserves in these neighbourhoods also tend to have less tree cover (Leong et. al.,2018). Narrow reserves created around daylighted streams or natural drainage areas, whether large or small, are a potential exception to this trend as they tend to be planted with a high coverage of vegetation to reduce water run-off and improve drainage flows.

Because connectivity is particularly important for avian diversity in small habitat fragments (i.e. Shannahan et. al., 2011, Kang et. al., 2015), exploring this factor more thoroughly in future studies is recommended. The connectivity measurement in this study (distance to closest forest patch) was not able to be applied meaningfully due to potential difference in landscape cover affecting the importance of the distance to closest forest patch, i.e. if the matrix between patches provides resources and is effectively just an extension of the habitat area then the distance to another habitat patch becomes largely irrelevant. Also, without dispersal data it is not possible to know whether or not small habitat patches are able to act as steppingstones for various species. Studies assessing the level of gene flow between populations inhabiting reserves throughout the city could determine the extent to which populations in habitat patches are demographically isolated, and also the extent to which regional populations are functioning as metapopulations.

Conclusion

Reserves which are narrow in shape can provide valuable habitat for native birds in residential urban areas. Size of reserves is an important factor to consider in reserve planning as reserves below a certain area may not be capable of providing useful habitat for a diversity of native bird species, although improving the vegetation quality of small reserves can increase their value to urban avian diversity. Planting and vegetation management plans should be developed for existing urban greenspaces which currently have low tree cover, plant diversity, and vegetation complexity in order to increase the diversity of native birds that can be supported within them. Reserve area and width thresholds should be developed, as well a minimum vegetation quality threshold, to ensure that the proportion of reserves acting as populations sinks for native bird species do not jeopardize the persistence of regional metapopulations. A city-wide network of narrow reserves created around daylighted streams and transport routes could not only provide multiple positive social and infrastructural outcomes, but also provide suitable habitat for diverse populations of urban adapted native birds, provided they have high quality vegetation. Managing regional populations of native birds inhabiting a network of smaller reserves throughout a city as connected metapopulations could be an effective strategy to ensure long-term resilience.

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