The Urban Environment, Children and Biophilia:

Do children use and prefer biodiverse urban habitats?

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Abstract

Today nearly half of the world's children will grow up in the highly modified and artificial urban landscape (UNICEF, 2012). These children will experience an environment largely alien to the more natural one we have evolved to be in, leaving them to grow up in a state of 'biological poverty' (Kellert and Wilson, 1993; Turner et al., 2004). For these children urban green spaces provide a residual link to the natural world, preserving opportunities to experience and connect to biodiversity. However, the distribution and quality of urban green space across cities is patchy and increasing parental safety concerns may restrict access to even nearby natural areas. As such many urban children may be in danger of growing up in isolation from nature and the developmental and wellbeing benefits it provides (Louv, 2008).

In this thesis I aim to assess children's habitat use in urban areas to explore whether they are able to access and use urban green spaces. I predict that children will preferentially seek out and use the biodiverse areas available to them, as would be expected by the Biophilia Hypothesis, proposed by EO Wilson as the innate affiliation towards the natural world (Wilson, 1984). To explore children's habitat use patterns, I used data collected from interviews with 126 children in two urban centres of New Zealand. I applied approaches developed from wildlife research to estimate children's home ranges and habitat use. I estimated the biodiversity present within each child's neighbourhood and home range area to assess how much biodiversity is available and used by each child in their day-to-day movements. Further, I applied resource selection analysis to gain quantitative estimates of children's habitat preferences.

Overall, I found that the biodiversity available to children in their neighbourhoods was varied but generally high, with some form of green space located close to all children. I found home range size was a key determinant of how much biodiversity a child had access to. However I also found evidence for a continued decline in home range size, with a median home range size of less than three hectares. Further, over a quarter of children had restricted ranges which prevented them from accessing any biodiverse green habitats, indicating that declining home range sizes could be facilitating a disconnection to nature.

Yet, for the majority of children who did have access to nature in their neighbourhood, they did not show any preference for these more biodiverse habitats available. Instead, children spent most of their time outdoors either in their garden or on residential streets. Further resource selection analyses identified gardens, streets and both paved and green sports fields as being the most preferred habitat types. These suggest children's habitat use is motivated by selecting sites that are close to their home and which support play, rather than biodiversity values (Moore, 1986; Jansson and Persson, 2010).

The lack of biophilic behaviour in children despite available and accessible biodiversity suggests that children may be spending more time indoors with electronic media (Pergams and Zaradic, 2006). Through a combination of lack of availability, declining independence and the allure of modern technology, children are growing up isolated from nature and ignorant of the benefits it affords (Louv, 2008). This trend is concerning as it means future generations will grow up with little knowledge of or empathy for the environment (Pyle, 1978; Miller, 2005). It is therefore important to improve children's connection to nature by integrating biodiversity into the urban environment, and children's lives, to a greater degree. I explore the possible roles of urban planning, school education, and parent's in supporting this.

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Chapter 1: General Introduction

1.1 Humans and the Urban Environment

The human race is well on its way to becoming a truly urban species, but we are only beginning to understand how this shift in our choice of habitat affects our behaviour and well-being. Since the 1800s we have undergone rapid urbanisation from a predominantly rural past to become a majority urban population by 2010 (UN, 2012). This pace of change is not expected to slow; by 2030 the UN projects the global urban population will climb to 60% and already levels of urbanisation in developed nations surpasses 80% (UN, 2012). Urban areas are usually recognised as areas with a high density human population and/or the domination of artificial land cover of built structures (Farinha-Marques et al., 2011). Urban areas can range from mega-cities of more than 10 million people to small towns and villages of more than 1000 people (Takano et al., 2002; Statistics New Zealand, 2013a). As with any other species that are subject to drastic changes in environment, our increasing urbanism will likely impact and alter many aspects of lifestyle and well-being. Today's urbanites dwell in a near exclusively human constructed environment that is far removed from the natural systems we once interacted with on a daily basis and evolved in association with (Grinde and Patil, 2009). This increasing detachment from natural environments and species has been found to be linked to many changes to our activities, health and welfare (Bratman et al., 2012).

Urbanisation involves the clearance of ecosystems and vegetation and replacement with layers of impervious surfaces and buildings. Most of the remaining open ground is then reconfigured into urban green spaces with only highly constrained and artificial forms of nature permitted. Not surprisingly therefore, urbanisation often necessitates a dramatic loss of species, making it a central threat to global biodiversity and a key contributor to the extinction of species (Grimm et al., 2008). As such, urban areas often have low naturalness and animal species richness, meaning the majority of humans now live in a state of "biological poverty" (Grimm et al., 2008; Turner et al., 2004).The biodiversity of cities is often far removed from its original state, with the loss of many native species unable to adapt to change in habitat (Sol et al., 2014) and an immigration of exotic species, often introduced by city residents (Dunn and Henneghan, 2011). These include urban generalist species which have colonised many cities, leading to the homogenisation of biota across cities (McKinney, 2002(Schwartz et al., 2006). To accommodate growing human populations, urban areas are predicted to expand to cover 6% of the Earth's land surface by 2030, a three-fold increase from levels in 2000 (Seto et al., 2011). Despite this small total area, urban metropolises exert a disproportionate drain on environmental resources at local and global scales, with ecological footprints up to triple their actual land area (Wackernagel et al., 2006).

Despite the drastic modifications of natural landscapes necessitated by urbanisation, urban areas are by no means empty of biodiversity. A great diversity of green spaces can be encapsulated within urban areas, ranging from fragments of remnant ecosystems to ornamental parks and gardens through to abandoned brownfield sites (Recher and Serventy, 1991; Livingston et al., 2003; Rink and Herbst, 2011). These fragmented patches of green can add up to a significant proportion of an urban area; on average around 19% in western cities (Fuller and Gaston, 2009). Additionally, the influence of humans through introducing and maintaining diverse green spaces helps boost species richness above its normal capacity. This is clearly visible in desert cities such as Phoenix, Arizona, where humans foster much higher species richness then is supported naturally in surrounding desert (Hope et al., 2003). Even in less extreme environments several studies have found cities have higher total plant species richness compared to neighbouring natural areas (Luck, 2007; Grimm et al., 2008). However, this may partially be an artefact that many cities were preferentially developed in biodiversity hotspots (Kuhn et al., 2004). As such, cities offer a unique assemblage of retained native and rare species mixed with a diverse set of introduced species (Kuhn et al., 2004; Angold et al., 2006). While historically ecologists disdained the nature of cities, urban areas are now being recognised as dynamic and rich habitats in their own right which play critical roles in the sustainability of the city and well-being of its residents (Miller and Hobbs, 2002; Dearborn and Kark, 2009).

The conservation of green space in cities is being increasingly emphasised in urban management due to the diverse values these spaces hold. As the loss of biodiversity often results in the loss of ecosystem services, these green patches can help to restore these systems. For instance, urban green spaces can help to reduce the heat-island effect of cities (Wong and Yu, 2005), limit levels of atmospheric pollution (Nowak et al., 2006), and restore carbon and hydrological cycles (Denardo et al., 2005; Pickett et al., 2008). Even small features can have considerable benefit, a single street tree can improve air quality and temperature, reduce noise from the street and provide aesthetic and well-being values (Bolund and Hunhammar, 1999). In particular the importance of nature in the city for human well-being has recently been emphasized (Bratman et al., 2012). However even these remaining urban green areas in cities may be under threat, as evidence points to trends of densification in urban areas through the paving over and re-development of green spaces (Pauleit et al., 2005). Whether such densification of cities, as opposed to allowing low density but sprawling cities, provides a more sustainable future for the inevitable growth of cities is under debate (Lin and Fuller, 2013). A key component of this argument is role of biodiversity in providing well-being benefits to urban residents through connections to nature.

1.2 Nature Connection and Well-being

The value of nature in supporting peoples' health and well-being has been appreciated informally across cultures for many centuries (Bratman et al., 2012). More recently the link between nature and health has seen an explosion of attention. This was largely catalysed by a landmark study in (1984) by Roger Ulrich which confirmed this traditional knowledge (Ulrich, 1984). Ulrich showed that patients recovered faster and reported less pain when recovering from gall bladder surgery, simply if their window had a natural view of trees, compared to the recovery of patients who looked towards a brick wall. Similar effects have been found in relation to mental health, where nature can help to reduce stress levels (Korpela et al., 2002), improve attention (Berman et al., 2008) and support higher cognitive functioning (Tennessen and Cimprich, 1995). These benefits can have immediate effect but also can accumulate over a life-time for those living in greener environments, adding up to greater longevity (Takano et al., 2002). For example, Alcock et al. (2013) tracked participant's mental health over 5 years and found those who moved to greener areas had significantly better mental health, whereas those who moved to less green locations showed a decline in mental health. Other studies have shown green spaces support better physical health by encouraging greater amounts of physical activity (Babey et al., 2008; Coombes et al., 2010). For children especially, nature plays a critical role in their healthy development and well-being (Kellert, 2005). Children share similar benefits as adults from nature, including lower stress (Wells and Evans, 2003), reduced symptoms of ADHD (Kuo and Faber Taylor, 2004), lower obesity levels (Bell et al., 2008) and improved cognitive functioning (Wells, 2000). More specifically, natural spaces also help support children's learning and development by stimulating more complex and enriching play (Fjørtoft and Sageie, 2000;Samborski, 2010).

Despite the multitude of benefits of nature, there is increasing evidence of a lack of connection to nature in urban societies (Nabhan and St Antoine 1993; Miller 2005). The removal of nature from cities coupled with urbanites' hectic schedules leaves little opportunity to make use of and appreciate nearby nature. As such, an estimated 90% of our lives are now spent indoors (Evans and McCoy, 1998). If green space is less available in urban areas then it is less likely to be used (Coombes et al., 2010). Even when nature is nearby, people will bypass it in favour of taking time-saving routes (Nisbet and Zelenski, 2011). These declines in time spent in nature are particularly severe in children. Natural England (2005) have reported a decrease in time spent in woodlands from 40 to 10% of play time compared to previous generations. In the USA, children aged 6 to 9 on average spend less than 30 minutes a day outdoors (Hofferth, 2009). It appears that time previously spent outdoors is being replaced with increased screen time (Hofferth, 2009). In teenagers, Pergams and Zaradic (2006) found that decreases in national park visits, which have declined for the first time in 50 years, were associated with increases in the use of electronic entertainment. In children, this disconnection with nature and the resultant detrimental effects to their development and health has been termed the Nature Deficit Disorder (Louv, 2008). Overall, this widening division between humans and nature has been described as the third major crisis facing the human race alongside climate change and biodiversity loss (Sampson, 2013).

Concern over the disconnection from nature extends beyond the detrimental effects on individual's wellbeing as it is feared it might translate to a decline in conservation ethos. In 1978 Pyle characterised this concept of decline in time and connection to nature as the "extinction of experience" (Pyle, 1978). This is a process whereby, through continual decreased experience of nature, individuals have a decreased knowledge and empathy for it. This unfamiliarity with natural ecosystems is already thought to be a major reason for the perceived lack of concern for today's environmental issues (Miller, 2005). A higher number of experiences in nature has been linked to both greater emotional connectivity and preference for nature (Bixler et al., 2002), and the expression of positive environmental behaviours in adulthood (Wells and Lekies 2006). Extinction of experience results in children being unable to identify local plants and animals, name endangered species, or recognise the link between the resources they consume and the species and environmental systems that produce them (Verboom and Kralingen, 2004; Bebbington, 2005; Miller, 2005). Instead, Balmford et al. (2002) showed children today have a better knowledge of different pokemon than of common wildlife species. This growing lack of contact with nature in children is proposed to create "environmental generational amnesia", whereby increasingly degraded environments become the norm, and later generations will become steadily less aware of nature's functions and value (Kahn et al., 2009).

1.3 Biophilia in Urban Children

This reduced connection to nature might be considered unexpected under EO Wilson's theory of Biophilia. This theory proposes that humans have an innate affiliation towards natural things, and this might even be an evolutionary inherited trait (Wilson and Kellert 1993). More biodiverse and green environments would likely have supported greater survival in early humans and so today we show innate preferences for features of green and plentiful landscapes (Kaplan and Kaplan, 1989). Research has repeatedly shown that people across cultures prefer views of natural landscapes to built ones, and prefer more green urban areas in comparison to less green (Kaplan and Kaplan 1989; Ulrich 1993; Hartig and Staats, 2005). In field-based experiments people show greater preference for greener landscapes (Lindemann-Matthies and Bose, 2007; Lindemann-Matthies et al., 2010) and the more species-rich and complex the habitats the greater the benefits to well-being (Kaplan 2001; Fuller et al., 2007). This preference for nature is also visible outside of experiments, in the biased site selection of homes and businesses in more natural areas (Wolf, 2005) and higher property values in greener areas (Luttik, 2000).

Children are also expected to express a biophilic response. Children demonstrate preferences for more species-rich areas and choose natural spaces as their favourite places, especially for play (Moore, 1986; Sobel, 1993; Maxey, 1999). If the biophilia hypothesis is true, then the current disconnections with nature would be unexpected as children and adults should seek out and make use of urban green areas. Yet, a disconnection to nature in children is reported (Louv, 2008) and this is of concern due to childhood being the critical time to establish an enduring connection to nature (Kahn, 2002). The decline in nature connection has been particularly linked to the urban environment (Miller, 2005), though there are many possible factors which could be involved. Three main potential factors which could be altering children's connections to nature are discussed below.

i) Availability of biodiversity

This first factor assesses whether urban areas support biodiverse habitats within children's neighbourhoods and places they regularly visit. While biodiversity can be high in cities, these patches of public green space might not be present within a child's neighbourhood, but instead clustered in other areas. For instance, lower socio-economic regions of cities often have lower quantity and quality of biodiversity (Whitford et al., 2001; Hope et al., 2003; Mathieu et al., 2007). The green areas present in lower-socioeconomic areas were also found to be less diverse and structurally complex compared to others. Many green spaces, particular parks and playgrounds designed for children are highly artificial and sterile interpretations of nature (Moore and Young, 1978; Herrington and Studtmann, 1998). As such they may be inadequate as spaces to attract and engage children and to allow connections to nature (Nabhan and Trimble, 1995; Jansson and Persson, 2010). Therefore, the impoverished nature of urban areas may be triggering a decline in connection to nature.

ii) Accessibility of biodiversity

If urban areas do contain biodiverse green spaces, these may not be at a scale that is accessible to a given child due to physical barriers and parental restrictions. Much of a cities' biodiversity can be locked up in private property and while many cities have standards for minimum distances to green spaces, these are often not achieved (Barbosa et al., 2007). For instance, in Manchester, while overall green space provision is high, accessibility for individuals is below recommended standards (Kaźmierczak et al., 2010). Previously accessible green spaces have now been found to lie outside of children's independent ranges due to increases in parental restrictions (Veitch et al., 2008). The reduction of children's home range size over the past few decades has been well documented, with decreases found in licences to walk to school or cycle on the road without an adult (Hillman, 1993). This decline has been linked to increased traffic levels and parental safety concerns (Karsten, 2005).

As such, the number of possible green spaces a child can visit become increasingly small, leaving them with little choice as to the type or quality of biodiversity they can interact with.

iii) Biophilia is not present in urban children

This factor addresses specifically the hypothesis of biophilia against its antithesis of videophilia; the attraction to indoor, sedentary activities associated with modern technology (Pergams and Zaradic, 2006). Studies have shown that children highly value and enjoy natural sites, with such sites even described as exerting a 'pull' on children to extend their range and explore new areas (Payne and Jones, 1977). However today, perhaps as a function of the urban environment, or the variety of today's technology, children now seek out the nearest computer screen rather time outdoors (Pergams and Zaradic, 2006). Further some studies have shown that a biophilic response to nature is not ubiquitous (Dallimer et al., 2012; Qiu et al., 2013). Others have found that rural children, who are assumed to have greater access to nearby biodiversity, do not use it any more than urban children, indicating perhaps biophilia is not present (Clements, 2004).

This project seeks to explore these questions for children in New Zealand by utilizing methods developed from wildlife research. Wildlife ecologists have long sought to understand an animal's use of different habitats, gaining information on which habitats are important to a given species and hence which need to be preserved. Specifically, this project will use resource selection analyses which measure the selection of different habitats by an animal in relation to that habitats availability in the environment (Manly et al., 2002). An animal's habitat selection is a behavioural response to environmental elements which support its survival (Boyce and McDonald, 1999). As such, if we assume biophilia is a genetic trait which encourages preference to biodiverse habitats then we can explore for selection of biodiversity in humans as other animals select for resources such as shelter and food. By applying this to children's use of the urban environment we can explore whether natural habitats are valued and used by children in their routine activities.

1.4 Thesis Structure

The main aims of this thesis are to assess the habitat use of children growing up in New Zealand and investigate possible factors which either restrict or support access as to, well as preference for more biodiverse habitats. New Zealand has a rich and unique ecology, but it is also one of the world's most urbanised nations (UN, 2012). This study will combine quantitative measures of biodiversity with children's self-reported use of their environment to assess children's habitat preferences. This combined methodology reflects the current focus on integrative approach between social and ecological fields which is essential within an urban context (McIntyre et al. 2000; Alberti et al., 2003).

Chapter 2: Urban Biodiversity

This chapter describes the development of a new methodology to predict biodiversity across an urban landscape. It first defines biodiversity above simple metrics of greenness or species richness and also takes into account how a child might perceive and value biodiversity. This method was utilised to measure biodiversity across urban environments and used to create a predictive model which was used in the following chapters to compare biodiversity across urban areas.

Chapter 3: Children's home ranges and biodiversity

This chapter seeks to explore the question of how much biodiversity is present in a child's neighbourhood and home range. Information from interviews with children will be used to estimate children's home ranges and the accessibility of different habitats in their neighbourhood. Habitat maps incorporating the biodiversity values derived in chapter 2 will be used to compare available and accessible biodiversity among the children and the effects of demographic factors.

Chapter 4: Testing Biophilia: Resource Selection Analysis

This chapter seeks to ask the question of whether urban children in New Zealand cities show a biophilic response to their environment. Under this hypothesis it is expected that children will seek out the most biodiverse habitats available to them. This chapter will describe the selection preferences of different habitats by children using resource selection analysis. It will also explore any variation in these responses with demographic factors and environmental factors such as the availability of biodiversity.

Chapter 5: General Discussion

This chapter will summarise the main results and conclusions of the previous three chapters. Interpretation of results in relation to the key aims of the study; whether children are preferentially seeking out and using biodiversity, or whether the accessibility or availability of biodiversity could be preventing this, is assessed. Implications of these findings for the conservation of biodiversity within cities, the importance of accessibility in children's green spaces and urban planning are reviewed. Key areas for future research to further explore these findings are discussed.

Chapter 2: Measurement and prediction of biodiversity values across urban habitats

2.1 Introduction

2.1.1 Urban Habitats and Biodiversity

Urban biodiversity is simply the diversity within and between species which is present within urban areas or those found along its fringes (Muller et al., 2010). How urban areas themselves are defined however can vary widely, though in general they describe areas with large human population densities and a high cover of impervious surfaces (Farinha-Marques et al., 2011). The principal difference between biodiversity in a city and of other areas is the dominance of human influence on its form, composition and processes. Urban biodiversity is critical to supporting valuable ecosystem functions and providing benefits to city residents (Dearborn and Kark, 2009). As such, urban biodiversity is inextricably linked to people and so must be defined in context with their values and use of urban areas. Especially as it is this biodiversity which most people will interact with on a daily basis. Therefore it is this nearby nature that will be the main influence on how people view and value urban biodiversity, and so affect the urban planning of green spaces and management of biodiversity (Miller and Hobbs, 2002; Alberti et al., 2003).

Urban ecologists have had to diverge from methodologies applied in more natural systems for the classification of habitats and biodiversity, which are driven more by anthropogenic processes and require integration of both ecological and social definitions (McIntyre et al., 2000). Biodiversity is often assessed at the habitat level, where habitat units are usually defined based on topographic and vegetation characteristics. However for urban areas the function or land-use of an area is usually more important (Cadenasso et al., 2007). At its most coarse scale, the urban environment can be defined into four major categories: grey (built), green (vegetated), blue (water) or brown (wasteland) sites (Farinha-Marques et al., 2011). Finer-scales of classification identify areas such as commercial or residential sites, and may take into account the building density of such areas (Cadenasso et al., 2007). Often these habitats are described within the framework of an urban gradient, which describes a common pattern in city layouts, where a densely built-up central city area falls out to less dense and greener areas towards the city limits (McDonnell and Hahs, 2008). Composition and overall richness of biodiversity have been found to vary significantly across this urban gradient (McDonnell and Hahs, 2008). Often there is a peak of diversity in peri-urban habitats, where greater coverage of vegetation and high habitat heterogeneity supports diverse plant and avian assemblages (McKinney, 2008; Kowarik, 2011). However, no unified classification system for urban areas exists and so classifications often vary between urban studies depending on the topic and scale of focus.

Biodiversity has also been represented in a diverse number of ways. At one end of the scale, biodiversity can be defined in its most complete form, that of the UN's (1992) definition as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems." However, a complete census of all these aspects of biodiversity is near impossible and so studies of biodiversity rely upon specific measures related to a few of these aspects which can act as indicators for overall biodiversity. Here it is useful to consider Noss's (1990) three categories of biodiversity indicators, which largely mirror that of the more encompassing UN definition. Noss divided biodiversity into compositional, structural and functional biodiversity. Such definitions are useful in that they capture the many different aspects of biodiversity which together help support a healthy sustainable ecosystems. They also provide many different measures of biodiversity to be assessed, providing a more complete picture of the state and change of biodiversity in the dynamic urban environment.

The most common aspects of biodiversity investigated are species richness and vegetation complexity. The most common taxa surveyed are plants, birds and invertebrates, which can act as surrogate species and indicators of diversity in other species (Oliver and Beattie, 1996; Blair, 1999). Studies tend to show that urban areas can support high richness of species, though this varies greatly between different habitat types and is composed of a mix of native and introduced species (Dunn and Heneghan, 2001; Stewart et al., 2009). Particularly biodiverse habitats in urban areas include wasteland sites, which can be havens for rare species (Strauss and Biedermann, 2006), and managed green spaces like gardens and parklands can support rich species assemblages and high levels of green cover (Livingston et al., 2003; Goddard et al., 2010).

Within species richness, the composition of species present in a location can reveal much about the ecological integrity and quality of a habitat (Stewart et al., 2009; Chong et al., 2014). In particular, the presence of native species is of particular interest in urban areas, as native species can often be lost through urbanisation and the associated influx of alien species (McKinney, 2006; Sol et al., 2014). Native species can provide biodiversity values through providing greater ecosystem services then exotic species (Isaacs et al., 2008), and provide a greater sense of place for urban residents (Horwitz et al., 2001). However, children in particular have poor species identification skills, and often show greater knowledge (Balmford et al., 2002; Bebbington, 2005) and preference (Bizerril, 2004; Nates et al., 2010; Ballouard et al., 2011) for charismatic exotics over native species. While the overall more natural habitats are more preferred over urban landscapes (Hur et al. 2010;Voigt et al., 2014) discussed further below), the presence of native species over exotics species has not yet, to my knowledge, investigated in relation to human well-being and nature connection. As such, while native species are of high ecological value, exotic species likely provide many of the same opportunities as natives to support children's connection to nature, and therefore are treated equally in this study.

The structural complexity of vegetation within an environment has also been found to be a useful measure of biodiversity as it correlates highly with compositional richness. There is a strong relationship between more complex vegetation structures and greater diversity of avian (Clergeau et al., 2001; Sandström et al., 2006) and invertebrate communities (Smith et al., 2006). Habitat structure is a combination of the composition, abundance, and vertical and horizontal arrangement of vegetation (Byrne, 2007). More complex structures are thought to increase the number of micro-habitats available to be exploited, which supports greater numbers of species (Krebs, 2001). Studies which incorporate measures of habitat structure in their assessments therefore allow fine-scale exploration of factors which can encourage biodiversity in urban areas.

However, this approach requires intensive and time-consuming methods, which mean they are often limited to particular taxonomic groups or within specific habitat types only. In attempts to characterise biodiversity in a more rapid system, others have used more general indicators for biodiversity which can also be applied across different habitat types. Known as "rapid biodiversity assessments", these rely on indicators that if applied equally across sites can provide a ranking of biodiversity value. For instance, due to the strong link between structural richness and species richness, many have used structural richness as a more easily assessed measure of biodiversity (Hermy and Cornelis, 2000; Young and Jarvis, 2001; Tzoulas and James, 2010).

At the other end of the spectrum, biodiversity can be estimated using more general and coarse approaches. Growing interest in the importance of green spaces for the well-being of the urban human population has led to increasing number of studies which have used general and qualitative measures of biodiversity are used to assess this role of green space sites (Ulrich, 1984; Wells and Evans, 2003). However these ignore the diversity of urban systems which cannot be captured by simply identifying sites as natural or not, or using subjective scales of greeness. Within the ecological field, coarse measures of biodiversity have been used to infer biodiversity across large areas, such as using proportional cover by vegetation as a biodiversity indicator (Hur et al., 2010). The disadvantage of these methods is they do not take into account the wide variation within and between urban habitats, which can be dependent on factors such as vegetation type and amount, density of buildings and socio-economic factors (Cadenasso et al., 2007). Advances in the remote sensing field have allowed more fine-scale mapping of vegetation types across urban areas, allowing relatively quick assessment of the composition of an urban area (Mathieu et al., 2007). The diversity of these habitats and relative amount of natural and greener land-covers has been used to infer biodiversity levels (Honnay et al., 2003). Yet these methods lack a connection between what is mapped from aerial photos and what is present on the ground.

More recently, studies have adopted a multi-scale and multi-dimensional approach to take into account the variability present within and between urban habitats. For instance, Bino et al. (2008) combined fine and coarse scale analyses to show landscape green cover is an effective predictor of avian species richness. Angold et al. (2006) assessed biodiversity across an urban area using site-based surveys and landscape-scale assessments to infer patterns in biodiversity across taxonomic groups. Others have begun to combine both ecological and social values of green spaces. In 2011, Luck et al. used a combined approach of field surveys of avian diversity and vegetation structure combined with aerial imagery of green cover and density. They further combined these variables with more social well-being values to assess whether these ecological characteristics were correlated to well-being, indicating functional values for these green spaces. Such methods allow for a more comprehensive understanding of the diverse roles of green spaces in urban environments, and improvements in imaging and GIS systems support greater integration between scales of assessment.

2.1.2 Prediction of the biodiversity value of urban habitats for children

This study is part of a larger project exploring children's use and connection to nature in an urban environment. As such, the measures of biodiversity used must reflect children's perceptions and values of nature. First of all this means the attributes and functions relevant to children must to be used to describe and identify urban areas, such as whether sites are public, private or encourage play and sports. Additionally we must define biodiversity in a way that is relevant to how children might detect and perceive biodiversity in their day-to-day lives. Here I apply a holistic approach to defining biodiversity which is based on Noss's (1990) three indicators of biodiversity: compositional richness, structural complexity within habitats and the functional value of the habitat. All three of these could also affect how children view and respond to urban environments.

As expected under the biophilia hypothesis, children show preference for more species-rich environments (Benkowitz and Kohler, 2010; Samborski, 2010). However, the biodiversity that a person may perceive on a walk in the park can be different to that classically used to assess species richness in urban habitats. For instance, a lay-person may not perceive the diversity of plant species or bird songs, and may not recognise at all the more cryptic species which require effort and time capture and identify. Studies have shown that people can accurately perceive different levels of species richness, though often underestimate high richness areas and underestimate low richness sites (Fuller et al., 2007; Qiu et al., 2013). As such, the use of morphospecies as a more coarse measure of biodiversity acts well as a more realistic indicator of detectable biodiversity. Morphospecies, or recognisable taxonomic units, is a method to estimate species richness by estimating the number of species based on easily identifiable morphological differences that can be recognised by a non-specialist (Oliver and Beattie, 1996). While this method reduces the accuracy of these assessments, it does act as a reliable of indicator of true species richness (Oliver and Beattie, 1996; Pik et al., 1999).

Estimating functional values of biodiversity means identifying measures which reflect aspects that might encourage children to interact and connect with nearby nature. This is thought to be valuable, as spending time in green space has been shown to have many benefits to children's health, well-being and knowledge of the natural world (Kahn and Kellert, 2002). In contrast to preferences by adults (Qiu et al. 2013), children may prefer more messy and wild environments, providing them with more freedom to explore and make use of the environment itself (Kellert 1997; Pyle 2002). The coverage of vegetation, or greenness, of an area has been found to be a useful predictor of biodiversity (Whitford et al., 2001; Bino et al., 2008). Additionally, the level of greenness and structural richness of an area may be used as a proxy by humans to infer species richness levels (Fuller et al., 2007; Qiu et al. 2013). More natural environments are shown to support more native fauna (Daniels and Kirkpatrick, 2006; Parsons et al., 2006; Turner, 2006) and low management regimes, such as for vacant land, can allow unique assemblages to develop (Zerbe et al., 2003; Rink and Herbst, 2011). However intensive management of green spaces can also support high species richness, but this is composed to a higher degree by invasive species (Livingston et al., 2003).

Here I apply an approach which will allow for landscape-level prediction of biodiversity, while additionally being specific to the structural richness of each habitat unit, therefore taking account of the heterogeneity within and between urban habitats. Biodiversity was assessed in a range of urban habitats which integrated species richness, structural and perceptible values of biodiversity, measured in ways that a child could perceive biodiversity. Biodiversity values were measured in the field and the best predictors of biodiversity used to construct a predictive model to be applied to this landscape approach. This methodology could therefore be used to assess values of biodiversity related to children across the urban environment.

2.2 Methods

2.2.1 Study Area

I collected field samples of biodiversity across three New Zealand cities; Auckland, Wellington and Dunedin. Both Auckland and Wellington are situated in New Zealand's North Island and are two of the country's most populace urban settlements with populations of around 1,400,000 and 200,000 respectively (Statistics New Zealand, 2013b). Dunedin resides in the southern half of the South Island and has a population of around 120,000 (Statistics New Zealand, 2013b). Auckland and Wellington both enjoy sub-tropical climates while Dunedin resides in a temperate zone. However, despite climatic differences, Dunedin and Wellington have been found to have more similar ecological environments than to Auckland (Clarkson et al., 2007). Dunedin and Wellington are both smaller cities with a central area ringed by hills and a substantial green belt. In contrast, Auckland is a much larger city with a more sprawling urban form and low plains geography. While variations in biodiversity could be expected due to variations in biogeographic factors, their biological composition is more likely to converge due to the strong human influences on urban environments. For instance, Loram et al. (2008) assessed flora diversity across five cities in Scotland, Northern Ireland, Wales and England and found similar levels of species richness, diversity and composition.

2.2.2 Biodiversity sampling design

I defined habitats here as a continuous land-cover type which are classified by appearance and land-use. Thirteen distinct habitat types were classified for urban areas. Descriptions, sample sizes and field values for each habitat surveyed are provided in Appendix Table 2A.1. I selected sample sites within each city as diverse but representative examples of that habitat type. Efforts were taken to sample habitats across the urban gradient and in different neighbourhoods. A minimum of 10 samples for each habitat were collected in total, with time constraints restricting further data collection. Gardens were only sampled within Dunedin as access had to be organised in advance. Participating gardens were largely collected by advertising in the University of Otago Zoology Department. Few lower quality gardens were volunteered however, which means a smaller sample size for this group. Agricultural habitats from Wellington were not surveyed as no accessible examples were found to be present within the study area. Habitat surveys were run prior to the interviews with children on where they spent time outdoors, however, surveys were carried out many of the neighbourhoods of the schools and so some habitats surveyed were directly used by children. I carried out the majority of surveys in Dunedin between April and June 2013, during the austral autumn season. Further samples were collected in Wellington in June and Auckland in August of 2013. Surveys were only run on fine weather days without rain between 8am and 6pm.

Within each sample habitat site individual “features” were identified, which represent artificial and natural structures in an environment which add structural complexity (Young and Jarvis, 2001). Here I defined 10 main habitat elements based on previous work by Hermy and Cornelis (2000), Young and Jarvis (2001) and Tzoulas and James (2010). The addition of each one of these features would add structural heterogeneity to a site, thereby cumulatively adding new niches for species. These features are listed in Appendix Table 2A.2 with mean biodiversity value and their contribution value; calculated as the average proportional value of that feature in comparison to the mean of the habitat of which is resides within. Features had to cover an area of greater than $1m^2$ to be recorded as being present within a habitat site. Within a single habitat site one sample was collected for each feature present at that site. This allowed for assessment of biodiversity for both the habitat and for each feature present within it

Sampling design involved measuring biodiversity metrics at each feature present in an individual habitat site. This allowed two scales of sampling to take place at the same point; one at the fine-scale level of the feature and the other at a larger scale to characterise biodiversity in the habitat surrounding the feature. For this reason sampling points could not be selected randomly and instead were selected as representative parts of that feature for that habitat, such as in the middle of lawns or by the most common tree species present. Each sample plot was centred in the middle of the target feature. Two areas were delimited for each sample plot; a 1m diameter feature plot and a 5m habitat plot. The feature plot aimed to identify very fine-scale variations in biodiversity within a habitat, which could be important as children may favour spending time with particular features of habitats. The habitat survey plot was sized to get a more general measure of the biodiversity of the habitat site. At each sample measures of the three indicators of biodiversity: i) compositional, ii) structural and iii) functional, hereafter termed wildness, were measured. The specific methodology for each biodiversity metric, described in more detail below, is provided in Table 2.1.

i) Compositional:

Surveys of compositional biodiversity were recorded within a 5m diameter plots. Within a 5 minute count, the number of unique invertebrate and bird species recognised were recorded. During the 5 minute count the observer remained in the centre of the plot but could pivot at will. Additionally, the immediate area around the observer of the target feature could be searched by, for instance, looking into the bark of trees, through the leaves of a hedge, or under leaves on the ground. This aimed to better capture the value of that feature to a child who may be inquisitively exploring their environment.

ii) Structural Complexity:

The structural complexity of the site was assessed by measuring the number of strata occupied by vegetation and the number of plant growth forms present in the sample area (Hercock, 1997). Plant growth forms distinguish different plant forms by size and appearance. Here I defined ten discrete plant growth forms which cover the most common types of plants in urban areas and are listed in Appendix Table 2A.3.

iii) Wildness:

The wildness of a site is an indicator of how natural and undisturbed a site appears, which has value to urban children in being a more aesthetically pleasing and engaging environment for play (Pyle, 2002). This included a measure of the green cover of the 5m diameter plots, including measures for both floor and canopy cover of vegetation. The level of management of a site was assessed on a scale of 0 to 4; 0 representing a heavily controlled environment and 4, an area which was largely self-determined. Indicators used to infer management level included mown lawns, trimmed vegetation, cleaned artificial structures, as well as a lack of weeds and litter. The naturalness of a site was also scored from 0 to 4, with the degree of alteration from its natural state used as a measure of naturalness (Aplet et al., 2000; Kaźmierczak et al., 2010). Under this definition highly landscaped and degraded sites were seen as unnatural while natural areas included native woodlands, grasslands and beaches.

Table 2.1. Scoring metrics which estimate a BioScore for a single sampling point. Each BioScore is the sum of equally weighted biodiversity indicators, which are each the aggregate of specific biodiversity metrics.

Biodiversity Indicator	**Biodiversity Metric**	**Description and method**
Compositional Richness	Bird species richness (BSR)	Number of bird species sighted or heard within 5 minute count
	Invertebrate species richness (ISR)	Number of invertebrate morphospecies sighted, within 5m buffer during 5 minute count
	Plant species richness (PSR)	Number of plant morphospecies sighted within the 5m buffer
Structural Complexity	Strata number (S)	Number of strata occupied by vegetation, 1 point for each strata occupied by vegetation, strata levels were: <1, 1-2, 2-5, 5-12 and > 12m
	Number plant growth forms (PGF)	Count of plant growth forms in 5m buffer out of 10 possible (listed in Appendix Table 2.3)
Wildness	Management (M)	How controlled the sample area is by current human influences. Scored from 0-4, where 0 represents a completely controlled environment and 4 indicates an environment under very little human influence
	Naturalness (N)	How altered the site has been in comparison to a native state. Scored from 0 to 4, where 0 represents a completely altered and now artificial environment, while 4 represents a near pristine environment
	Greeness (G)	Percentage coverage of vegetation of the floor and canopy (each out of 100%) within 5m buffer. Re-scaled to range between 0 and 4

The biodiversity metric scores were summed together to obtain an aggregate score for each of the compositional, structural and wildness indicators. These were then added together to obtain a single feature's biodiversity value, termed "BioScore", as in Eq. 1:

$$\text{Eq. 1: } BioScoreFeature = \left(s_1(BSR + ISR + PSR) + s_2(S + PGF) + s_3(M + N + G)\right)$$

where s denotes scaling of the indicator scores so that each is equally weighted with a maximum score of 10. Scores were scaled by taking the maximum recorded score of any individual sample and scaling this to equal 10. I calculated a single habitat's BioScore by averaging the BioScores of all features present within that site. As sampling effort varied between sites based on the number of features in a site, taking the average across features removed any effect from this variation in survey effort. I then used these average BioScores for each habitat site to explore the relationship between habitat characteristics and biodiversity value.

To explore whether biodiversity values varied between different features I also calculated BioScores for each feature type. In addition to this I calculated a 'contribution score' which represented the average amount of biodiversity that each feature type contributed to the mean BioScore of the habitat it was within. For example, within a single habitat site, a bareground feature may have a BioScore of 4 while a group of trees may be 16, while the average BioScore of the entire habitat site is 13. Therefore, the tree group contributes a lot more to the biodiversity of the site in comparison to the bareground.

2.2.3 Data Analysis

The effects of habitat type and other factors on the BioScore were explored using linear regression models in R (R Core Team, 2013). BioScores were normally distributed within each factorial group using histograms and Shapiro-Wilk tests of normality. I checked for collinearity between pairs of predictor variables and found none were correlated above a 0.7 threshold. Model residuals were not found to be spatially autocorrelated using Moran's tests with the package spdep (Bivand et al., 2014), with all Moran's values less than 0.03 (p=0.75) (following Bivand et al., 2008). Models were assessed based an information theoretic approach, which selects the most parsimonious model with the greatest predictive ability (Burnham and Anderson, 2002). Models were created based on *a priori* hypotheses on which variables would act as possible predictors of the BioScore, as well as factors which could have altered the survey results. Variables assessed in models were habitat type, feature richness (as both a linear and quadratic effect), city and survey date and time. Models were assessed using AICc values, Akaike's weight (*w*) and R^2 values, which allow ranking of models and measures of overall model fit. AICc values were used in place of AIC values as these take into account small sample sizes (Burnham and Anderson, 2002).

The effect of cities was difficult to identify due to few samples from both Auckland and Wellington (n= 20 for both). Values for Wellington and Auckland did not fall outside of the range of values from Dunedin and so were included in the same overall dataset.

2.3 Results

A total of 151 sites were surveyed across the 13 different habitat types. The resultant BioScore showed an intuitive gradient from high values for more green and natural habitat types, such as woodland and vacant lots, to low values in more artificial and paving-dominated landscapes (Figure 2.1). Overall, woodland had the highest BioScore while Open Public Areas (OPA) had the lowest. Within green urban areas there is a gradient from the higher scores of more natural and structurally diverse habitats down to those more managed and homogenous habitats, such as recreational green. Garden type 1 and 2 scored the highest for compositional score and structural scores, due to their high diversity of plant species. Within the three separate garden categories there was a clear trend of higher BioScores for garden type 1 decreasing to garden type 3, with less green coverage and lower animal and plant species richness.

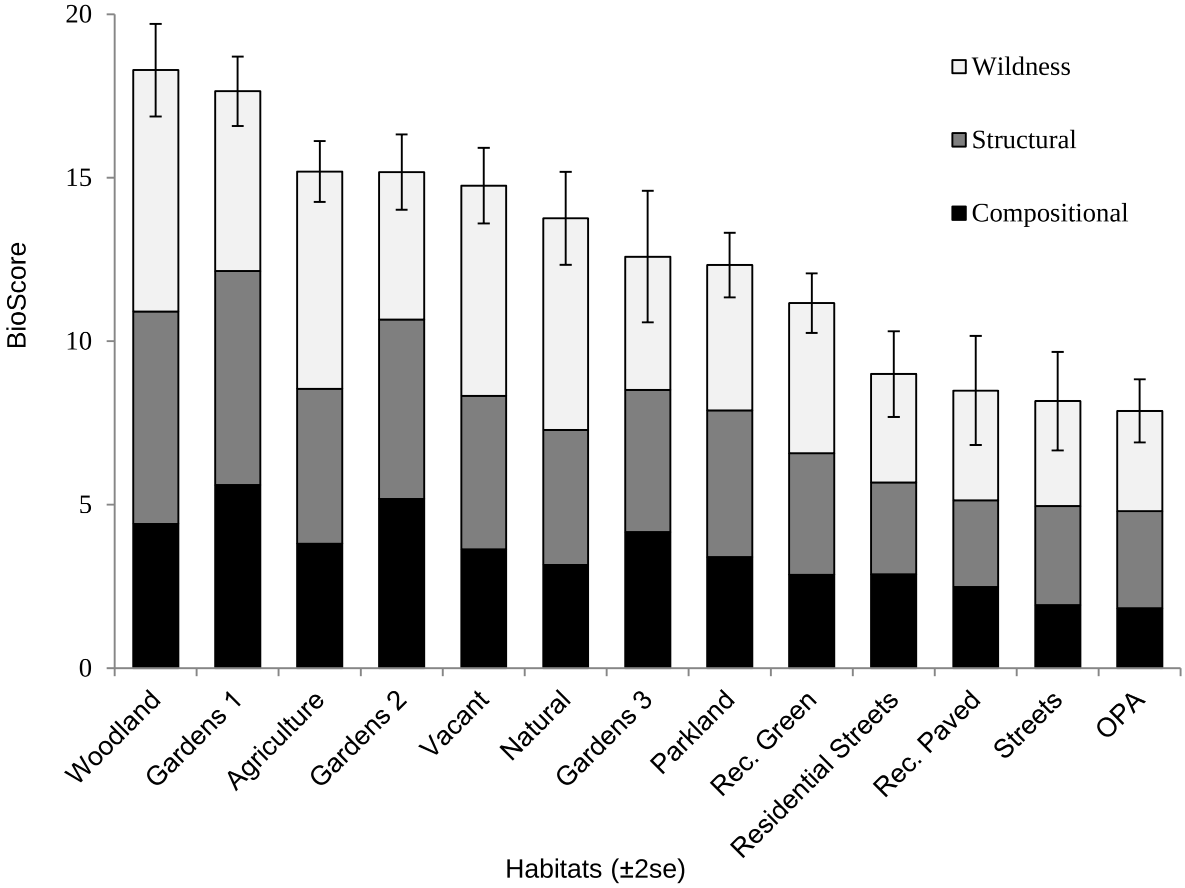


Figure 2.1. Combination of biodiversity metrics within the average BioScores for each habitat type. Values are shown for the three indicator values of the BioScore: compositional (species richness), structural (vegetation complexity) and wildness (greenness, naturalness and management of site). Standard error bars are provided for the total BioScore value of each habitat. Habitat type had a significant effect on BioScore (Table 2.2), with an R^2 value of 0.7.

BioScore also showed a significant positive relationship with feature richness (Figure 2.1). Although this relationship peaked at around 6-7 features per site, which may indicate some redundancy in the number of features in a site. However, it could also be due to the sampling design, which took the average BioScore across features, which may decrease the overall BioScore if low-quality features, such as paved ground, are present. As each feature is likely to add some new species assemblage, if biodiversity was measured cumulatively this relationship would likely not be present. However, counts of species richness here did not take account of differences in species composition between different features. Despite this, model ranking indicated a linear form for the effect of feature richness of BioScore provided a better model fit then a quadratic relationship, which was not included in any of the top four models (Table 2.2).

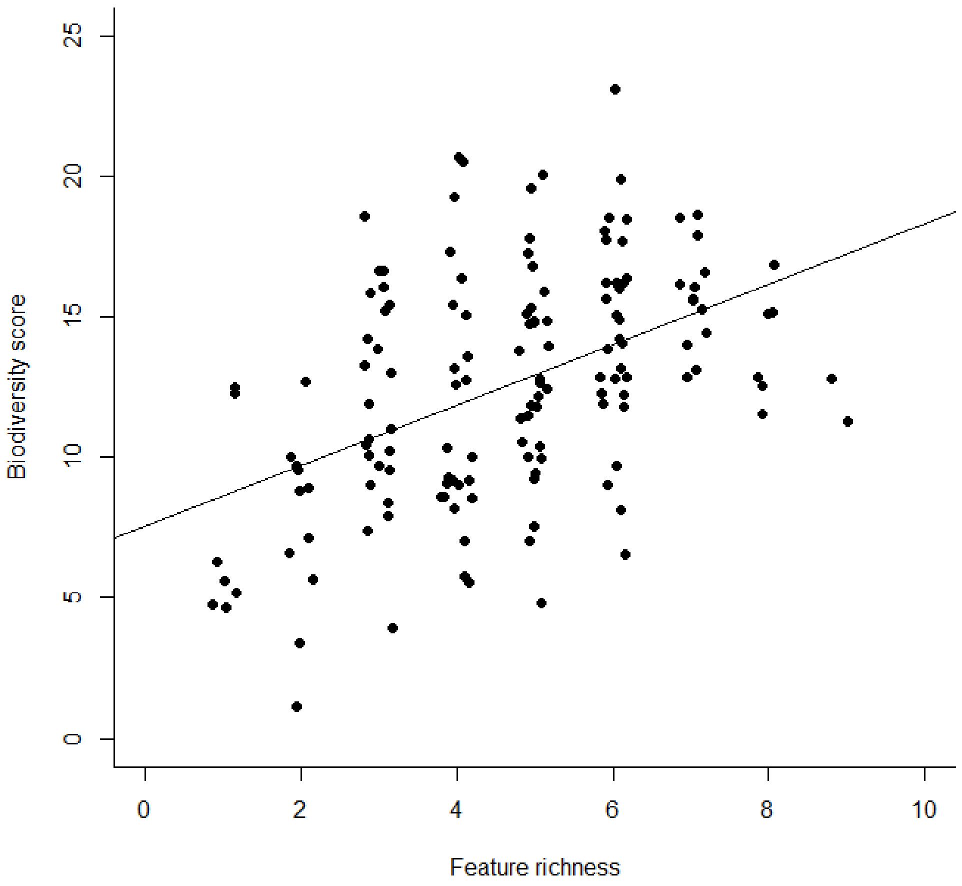


Figure 2.2. Relationship between BioScore and feature richness. The regression line estimates the effect of features richness across all habitats, with a slope of 1.08 and adjusted R^2 value of 0.22.

BioScores were also assessed for each individual feature type present across all habitat types, these values are provided in Appendix Table 2A.2. Of the near 700 features surveyed, the most common features across urban habitats were paved (116) and grass (115). The natural features tended to have higher biodiversity scores then artificial features of paving and fences, with the exception of natural bare ground, which also scored low. In contrast the most valuable features across habitats in terms of biodiversity were tree group, tall vegetation and tree line, largely due to their addition of structural diversity while allowing additional vegetation to grow in an understory, increasing compositional richness.

Linear models were used to identify the best model for predicting biodiversity across urban landscapes; a comparison of the best performing models is provided in Table 2.2. The two most important parameters for predicting BioScores were habitat type and feature richness, while city effects and date and time of survey were found to be insignificant. Model 1, where the effect of feature richness is constant across each habitat type, is ranked as the best model based on AICc values and model weight. The adjusted R^2 was however slightly higher for model 2 which incorporated an interaction between feature richness and habitat type. Within each habitat type feature richness also showed a positive trend with BioScore, apart from Garden Type 1. As discussed earlier, this may be due to a bias in sampling design rather than a true trend for this specific habitat type. As such, Model 1 was selected over Model 2 as a more reliable and parsimonious predictor of biodiversity for all habitat types. The results of this top model are shown in Table 2.3

Table 2.2. Comparison of best models for BioScore prediction, with models ranked by ΔAICc (AICc difference), which is used to estimate Akaike weights (*w)*, representing the likelihood of that model being most parsimonious out of the candidate set. F-values indicate the strength of effect of each parameter in the model *w* (Akaike weight) and adjusted R^2 values. FR = Feature Richness

Model	Equation	ΔAICc	Parameter significance	df	F-value	w_i	Adj. R^2
1	Habitat	0	<0.001	12	40.32	0.8	0.79
	+ FR		<0.001	1	37.26		
2	Habitat	4.31	<0.001	12	44.15	0.09	0.81
	+ FR		<0.001	1	40.8		
	+ Habitat*FR		0.017	12	2.08		
3	Habitat	4.6	<0.001	12	39.98	0.08	0.79
	+ FR		<0.001	1	36.95		
	+ City		0.817	2	0.44		
4	Habitat	13.65	<0.001	12	44.21	0.00	0.81
	+FR		<0.001	1	40.86		
	+City		0.8	2	0.49		
	+Date		0.72	1	1.44		
	+Time		0.26	1	0.38		
	+Habitat*FR		0.02	12	2.2		

Table 2.3: Summary of results from the best performing model (Model 1) with estimate and 95% confidence intervals (CI) for each habitat group and feature richness.

Parameter	Estimate	Lower 95% CI	Upper 95% CI
Agriculture	13.4	11.25	14.66
Gardens 1	13.49	9.49	12.85
Gardens 2	12.39	8.07	11.72
Gardens 3	10.96	8.89	12.08
Natural	11.51	2.98	6.33
OPA	5.87	6.88	10.16
Parkland	9.51	6.26	9.50
Rec. Green	8.99	5.18	8.49
Rec. Paved	8.09	5.25	8.54
Streets	8.11	6.50	9.80
Residential Streets	9.30	11.21	14.24
Vacant	13.98	13.50	16.55
Woodland	16.14	12.80	13.28
Feature Richness	0.9	12.25	12.25

2.4 Discussion

The biodiversity of different urban spaces was found to be dependent upon the habitat type and feature richness of that site. Therefore, the type and quality of urban habitats will affect how much biodiversity a child will perceive and be able to interact with in their neighbourhood. The most valuable habitats in this regard were woodland, gardens type 1 and 2, agriculture and vacant lots. These all had high values for green cover, species richness and structural complexity. In contrast, the lowest scoring habitats were open public space (OPA), recreational paved and streets, which had low values for amount of green cover, species richness across all three taxa groups and higher levels of management. This may make these habitats less aesthetically pleasing and less supportive as an engaging environment for urban children in supporting connections to nature (Pyle, 2002).

While only the number of features was included in this model, there were differences in the biodiversity value measured for each feature type. Vegetation and tree features scored highly by adding greater green cover and structural heterogeneity to a site. These features are also more likely to support more diverse assemblages of animal species (Krebs, 2001). In contrast, artificial or plain natural features, such as bare ground and lawns, provided little in terms of biodiversity. However, these are the features that are most common across urban landscapes and may be the ones most used by people in their day-to-day activities, as paths for walking, or sports fields for play.

More rigorous measurement of biodiversity, i.e. the use of true species types, would undoubtedly show a much stronger variation between different habitat types. Here, particularly for invertebrate species, only species which were easily distinguished as being unique were counted, thus leading to much lower estimates of species richness then have been recorded in urban habitats (Smith et al., 2006). Additionally I did not compare the relative numbers of native to exotic species, which would have provided information on the quality of the habitats and how well habitats may have supported children's understanding of local flora and fauna(Lindemann-Matthies, 2005). However this study used measures of biodiversity that a child would be able to perceive in an urban area, which meant a large proportion of the variation between habitats was not recognised. Studies have found lay-people's estimates of species richness are often underestimated, particularly for more natural habitats (Lindemann-Matthies et al., 2010) Cryptic biodiversity may be easily overlooked by children, leading to an under-appreciation of local flora and fauna within urban habitats. Previous studies have shown people's responses to green environments are more related to the biodiversity they perceive to be present, as opposed to what is actually there (Fuller et al., 2007). It is hypothesised that people use cues to assess the quality of a habitat, such as number of diverse plant forms, or how aesthetically pleasing these are arranged and managed (Fuller et al., 2007; Qiu et al., 2013). As such, these cues should be incorporated by both the wildness and structural complexity indicators of biodiversity.

While a limited sample size for Wellington and Auckland prevented a more robust assessment of between-city variation, the effect of cities appears to be limited. As urban areas are highly modified landscapes, site characteristics appear to have much more impact on site biodiversity than the larger scale impact of different ecoregions of the city itself (Loram et al., 2008). Further, it would be expected that the relative rankings of the habitats would vary similarly between cities, particularly if differences in feature richness can account for some of the differences between cities.

The best model for predicting biodiversity estimated that BioScores increase uniformly with feature richness across all habitat types. The strong performance of Model 2, which allowed the effect of feature richness to vary for each habitat type, indicates there is likely variation between the different habitat types. This individual relationship between habitats and features may be better estimated with a larger sample size, which may improve the fit of this model and accuracy of predictions. The method of calculating BioScores used here could be creating a bias by underestimating scores for habitats with a high number of features, by less biodiverse features ‘pulling down’ the whole score. This model also assumes that each feature is equally valuable in supporting biodiversity in a habitat. However, contribution values of the different features to the habitats they are within shows clear differences between complex natural features, compared to plain natural or artificial features. Greater survey effort could identify these more fine-scale trends between habitats and features to more accurately predict biodiversity then was possible here.

Further improvements could include accounting for varying proportional coverage of features in a habitat, in addition to just whether they are present or absent. In this way, although habitats may contain a very small amount of a strongly positive or negative contributing biodiversity feature, their input to the BioScore would be equal, despite their proportion most likely have a significant influence on the biodiversity of that site. An improved method could weight the contribution of different features by their proportional composition within the site. For instance, Tzoulas and James (2010) scaled their estimates of biodiversity by the proportional cover of different features.

Other factors which were not taken into account here, but could influence biodiversity in urban areas, include the age of a habitat site and its spatial context in terms of connectivity to other green spaces (Kuhn et al., 2004; Cook et al., 2011). The size of an area has also been shown to have a strong association with biodiversity, with larger sites often holding a greater number of features and species (Cadenasso et al., 2007; Daniels and Kirkpatrick, 2006; Cornelis and Hermy, 2004; Loram et al., 2008). Here, the size of a habitat could be taken into account when using this model for prediction of biodiversity across an urban landscape by multiplying the site’s BioScore by its area. This method was applied in chapter 3 to predict and compare biodiversity across particular urban areas.

Despite these limitations, this method provides a unique approach to modelling biodiversity which takes account of children's perception and values of biodiversity. Field-based measures were used to identify key predictors of biodiversity which can be applied across diverse urban landscapes instead of relying upon more general indicators of biodiversity such as the proportion of green cover. It builds upon previous work by incorporating the heterogeneity of structural richness within and between habitat types to better predict biodiversity (Cadenasso et al., 2007, Young and Jarvis, 2001). This therefore represents an efficient, comprehensive method of predicting values of biodiversity across urban habitats at a fine-scale. It is particularly in novel in assessing biodiversity metrics applicable to children. Often, assessments of biodiversity across cities do not account for the factors which most lay-people would notice and appreciate in the environment. It is critical to recognise these human values of biodiversity to better identify which sites and features of sites are connected to greater preference and use of green spaces.

Chapter 3: Estimating biodiversity within children's neighbourhoods and home ranges

3.1 Introduction

The rapid urbanisation of the human population has been blamed for causing a growing disconnection from the natural world (Maller et al., 2009), which has been linked to negative effects on our individual, societal and environmental well-being (Orr, 1994; Bratman et al., 2012). Urbanisation results in the replacement of green vegetation with built structures and impervious surfaces, transforming natural landscapes into areas of biological poverty (Turner et al., 2004; Aronson et al., 2014). Over the past 200 years of urbanisation, the importance of green space in urban planning paradigms has varied, leading to patchy provisions of nature across urban landscapes (McDonnell and Hahs 2008(Watson, 2009). However, today more than ever the benefits of urban green spaces are being recognised, leading to assessments of the quantity, quality and accessibility of biodiversity present in urban areas (Kuhn et al., 2004; Loram et al., 2008; Barbosa et al., 2007; Kaźmierczak et al., 2010).

Nature in a city can be rich, but distributed patchily across neighbourhoods, leading to inequalities in accessibility for urban residents (Whitford et al., 2001; Pauleit et al., 2005; Sliuzas and Kuffer, 2008). This presence of biodiversity in an urban area, hereafter described as "available" biodiversity, can vary between cities depending on their age and urban form. (Fuller and Gaston, 2009) found that green space in European cities could vary from 2% up to 46% of their total area. However, across a single cityscape the provision of biodiversity can be biased towards the more affluent regions (Whitford et al., 2001; Pauleit et al., 2005). Further, it is important to also take into account the accessibility of green spaces when considering how these spaces relate to urban residents. Accessibility of green spaces refers to whether areas are open to the public, whether they are within walking or cycling distance for that person, and whether they are perceived as safe to visit (Harrison et al., 1995). For instance, a large proportion of green space could be locked up in private property; Mathieu et al. (2007) found private gardens composed roughly 35% of total urban area in Dunedin, New Zealand.

When available biodiversity is present within public spaces, access can still vary depending on the potential visitor's demographics and mobility. For example, compared to adults, children often have lower accessibility to green spaces, due to restrictions on their freedom to travel independently, being either driven or accompanied by an adult to sites (Veitch et al., 2008; Freeman and Quigg, 2009).

Recommendations as to accessible green spaces should be within urban areas vary, with minimum distances of 900m (around 15 minutes walking), recommended by the European Environment Agency, compared to 300m (5 minutes walking), advocated by English Nature (2005). The former has often been achieved across European cities (Barbosa et al., 2007), whereas the latter was met for only 36.5% of households in Sheffield, UK (Barbosa et al., 2007). As such, while availability of green space can might be high, its accessibility at an individual level can still be poor (Kaźmierczak et al., 2010). For children, higher availability of nearby green space has been linked to benefits for both mental and physical health (Coombes et al., 2010; Wells and Evans, 2003). Urban residents' use of these sites however has been found to decline with increasing distance from home (Ellaway et al., 2005; Coombes et al,. 2010), and with decreasing greenness of the site (Ellaway et al., 2005).

While studies have shown urban areas can be surprisingly biodiverse (Kuhn et al., 2004), this biodiversity may not be present at a scale that is both available and accessible to urban children. Children often have confined independent ranges, curbed by urban barriers such as major roadways and parental restrictions on independent movement (Carver et al., 2008; Villanueva et al., 2012). These factors have caused with a significant decline in children's scale of movement over the past few generations (Hillman, 1993; Veitch et al., 2008). This decline has been more pronounced in urban areas (Kyttä, 1997; van der Spek and Noyon, 1997; O'Brien et al., 2000). Parent's concerns for safety are thought to be a key driver of children's declining independence, particularly over the dangers of traffic; now a dominating presence in today's cities (Karsten, 2005; Timperio et al., 2004). As such, many neighbourhood green spaces are now unreachable for local children despite being accessible when first designed (Veitch et al., 2008). This lack of both available and accessible biodiversity in urban areas has been proposed to be facilitating the development of a Nature Deficit Disorder in children (Louv, 2008), whereby an inability to interact with nature has detrimental effects on children's health and knowledge of the natural world.

The lack of opportunities for children to interact with nature is in part due to the declining ability of children to explore and interact with nature in their neighbourhood on their own (Freeman, 1995; Pyle, 2002). Independent use of space supports the development of a diversity of skills in children (Wells, 2000; Tranter and Pawson, 2001), and greener, more natural environments are thought to facilitate this development and learning (Fjørtoft and Sageie, 2000; Samborski, 2010). Such spaces within children's neighbourhoods are thought to encourage greater exploration and range expansion, with these natural areas exerting a 'pull' on children (Payne and Jones, 1977). Previous work exploring children's ranges and movement have assessed independent movement by measuring the maximum distance children travel from home on their own (Hart, 1979; Freeman and Quigg, 2009), with recent results showing most children do not travel further than 1 km from home (Veitch et al., 2008). Other studies have used qualitative measures of independence, such as the number of "licenses" or allowances, allocated to children from their parents, to use different spaces or travel to certain places alone (Kyttä, 2004); that have also been shown to be in decline (Hillman, 1993). The average child aged 9 today are now limited to the range of a 7-year old in the 1970s (Hillman, 1993). Typically these studies find girls do not travel as far from home as boys (Tranter and Pawson, 2001; Spilsbury et al., 2009), who have more license to explore on their own (Coates and Bussard, 1974; Hart, 1979).

The degree to which biodiversity is both available and accessible to urban children is critical to understanding the role of the urban environment in fuelling any growing disconnection to nature in children (Louv, 2008). For instance, a disconnection could be occurring because the biodiversity present in urban areas is simply too artificial or sterile to foster an appreciation of nature (Nabhan and Trimble, 1994; Pyle, 2002). Alternatively rich biodiversity could be present, but inaccessible for children due to the impassibility of urban environments coupled with parent's concerns for safety. It is clear that children's use of biodiverse areas needs to be evaluated against the availability and accessibility of green areas in the urban domain. The application of concepts and methodologies commonly used to determine home range size and habitat-use by wildlife can be applied to children to answer these questions.

The exploration of animals' home ranges has been a major topic in wildlife studies (Powell and Mitchell, 2012). In this study a child's home range is defined as an area which encapsulates children's most used spaces, and these spaces, which are spaces where the child has free use of. Free-use refers to movement and use of space that the child can make without being accompanied by an adult. This area is estimated using Minimum Convex Polygons (MCPs) which create a polygon drawn around a focal subject's location points. MCPs were one of the earliest methods to estimate home ranges, and they remain one of the most commonly used due to their broad applicability and simplicity (Burgman and Fox, 2003). As children will be identifying the spaces they use most frequently, density-methods to identify these areas were not required (Boyle and Lourenço, 2009). Further MCPs provide more conservative estimates of home range area then density methods (Boyle and Lourenço, 2009). As the urban environment is largely dominated by private property, I further classify home ranges into available, which include all areas within the MCP boundary, and accessible, which only includes those areas which the child has access to.

Often animal home ranges are estimated by attached tracking methods, such as Global Positioning System (GPS) collars, to determine an animal's location and movements at periodic intervals. Here, in this study, children's locations will be determined during a one-on-one interview, during which the children use an aerial photograph of their neighbourhood to place dots on the specific places they spend the most time when outdoors. This method varies from approaches in the human geography field, as children were asked to place dots in relation to where they spent the most time outdoors, thereby assessing the most commonly used areas, as opposed to the greatest distance travelled from home. It emphasises those areas which are most frequently used, and ignores one-off long distance travels. This is more compatible to the wildlife field's definitions of home range, that relates to the most valuable habitats used by animals and excludes any "occasional sallies" that may sporadically take place (Burt, 1943; Powell and Mitchell, 2012).

In this chapter I assess how much biodiversity is available and accessible at both a neighbourhood and home range scale for children in two New Zealand cities. This information will be valuable in addressing the question of whether children in urban areas are growing up in isolation from nature. This evaluation was part of a larger project, in which children were interviewed about, *inter alia*, where they spent the most time. I used this information to estimate children's home ranges and identify accessible and inaccessible areas. Biodiversity of urban areas was estimated based on predictions of the "BioScores" of 13 different habitat types, described in chapter 2. I applied the BioScores predictions to habitats present in a hand-drawn habitat map of children's neighbourhoods that incorporate information from aerial imagery, ground-truthed visits to sites and information from children's interviews. I then compared the relative biodiversity content of children's home ranges and neighbourhoods against socio-demographic factors.

3.2 Methods

3.2.1 Study Area

Children's home ranges were assessed in two major urban centres in New Zealand; Auckland and Wellington. In each city three schools were selected with similar urban environments; suburban areas dominated by residential suburbs. Schools in each city represented low (1-4), medium (5-7) and high (8-10) decile rankings which reflect the socioeconomic status of students attending the school. These rankings are applied to schools by the Ministry of Education, with lower deciles corresponding to schools the larger proportions of students from lower socioeconomic communities (Ministry of Eduction, 2009). Schools were selected to be located within areas of similar availability of public green space so that children's response to green space could be assessed across socioeconomic gradient. Both the children's interviews and ground-truthing of children's home ranges and neighbourhoods were run concurrently in Wellington during June 2013 and in Auckland during August 2013.

3.2.2 Children's interviews

In each school, a year 5-6 class with children aged 9-11 was selected. Approximately 20 children were interviewed from each class leading to sample sizes of 61 and 65 for Wellington and Auckland respectively. Interviews were conducted as part of a larger study on children's neighbourhoods and time spent outdoors. The study was approved by the University of Otago Ethics Committee (#13/119). I used information gathered in these interviews on the sites children spent the most time to estimate home range size. After the child had identified key sites in their neighbourhood and those they visited when outdoors, they were asked to place over 40 dots in these areas. They were asked to place more dots in places they spend more time and fewer dots in places they spend less time. Children were asked to tell the interviewer about where they were placing the dots, and to be as specific as possible with where they put their dots, avoiding placing them on buildings and in places where they must be accompanied by an adult. After the interview the dots were reviewed in relation to the information in the interview and dots were removed if they fell outside of the area the child stated they could go by themselves or if the child was identifying an indoor space. If dots had been misplaced into neighbouring areas by accident, these dots were moved to the nearest edge of accessible habitat.

3.2.3 Drawing home ranges

Home ranges for each child were created by treating the placed dots as analogues of GPS locations that would have been obtained from a tracked animal. Home ranges were drawn using 100% Minimum Convex Polygons using Hawth's Analysis Tools extension (Beyer, 2004) within ArcGIS (v10.1, Esri 2012). I used a minimum sample size of 30 dots per child, as this is a minimum sample size recommended in wildlife studies (Millspaugh and Marzluff, 2001; Baasch et al., 2010). Children who did not reach this number were removed from the analysis, resulting in seven children being removed, and a final sample size of 118. Figure 3.1 exhibits the home ranges of the local children of the three schools in Auckland, while Figure 3.2 shows a single child's location points and the MCP home range estimated around them. For comparison, the home ranges for Wellington are provided in Appendix Figure 3A.1. I calculated the maximum distance from home children usually travelled as the Euclidean distance (straight line) of the furthest location point from home.

3.2.4 Defining buffers

The neighbourhood buffer was defined as a 500m radius circle around the child's home in order characterise the habitat availability within the nearby-neighbourhood (an example is shown in Figure 3.2a). I used the distance of 500m because this was the median maximum distance from home that children travelled during a pilot experiment with four children in Dunedin. I used this standard measure to compare differences in available biodiversity between the neighbourhoods of each child. This buffer enclosed, for most children, an area of just over 78 hectares. However a few of the children's neighbourhoods included habitat areas that did not fit within the urban classification defined in chapter 2, and could not be attributed a biodiversity value. These areas, such as marine open water and active mines, were therefore left blank on the habitat map.

3.2.5 Defining accessibility

Accessible areas are defined here as public spaces that the child is allowed to visit independently and any privately owned areas that the child indicated they were allowed to visit, such as a friend's garden. Accessibility was defined for each child and this information was added into the habitat map. At first all green spaces (excluding gardens) were assumed to be accessible unless proven otherwise, and then accessibility was either supported or removed after on visiting the site. This allowed two forms of home range to be assessed; an available home range and an accessible home range. The available home range included all habitats present within the home range boundary; whereas the accessible home range only those the child had access to. Figure 4.2 illustrates the difference between a) available habitats and b) accessible habitats within a child's home range.

2.3.6 Urban habitats and biodiversity mapping

Based on the aerial imagery and results of ground-truthing, habitat maps for both scales of children's home range and neighbourhoods were created in ArcGIS. Property boundaries were based on information from New Zealand primary parcels, sourced from Land Information New Zealand (LINZ, 2012). The habitat map for the main study area in Wellington and Auckland are provided in Appendix Figures 3A.2 and 3A.3. Each site, a discrete area of a given habitat type, was identified as a particular habitat type and the number of features in each site were predicted at first using the aerial map and then ground-truthed in person. The relative biodiversity value of that site was calculated using the BioScores defined and calculated in Chapter 2. The BioScore of a given site was calculated based on its habitat type and feature richness. I investigated the relative habitat composition of neighbourhood buffers and home ranges by calculating the average proportional area of each habitat with these areas and I also calculated the number of children who had access to each of these habitats at these two scales. It is important to note however that the proportional area values do not take into account the home ranges or buffers where these habitats were not present, therefore only indicating the proportional area when these habitats are present and not the area of that habitat than an average child has access to.

I calculated an overall BioScore for each child's neighbourhood buffer, available home range and accessible home range. I used two methods to estimate these scores, the first is described as an area score, where each site's biodiversity score was multiplied by that site's area. These were summed across all sites present within a home range or buffer to generate the overall score. Alternatively, I also calculated a proportional score, which multiplied each site's BioScore by its percentage composition of that child's buffer or home range area. The area score provides an indicator of the total biodiversity present in children's home ranges and buffer areas, based on the assumption that larger habitats would contain more biodiversity (Daniels and Kirkpatrick, 2006; Cornelis and Hermy, 2004; Moroney and Jones, 2006). In comparison, proportional scores assessed the relative quality of biodiversity within children's urban areas by removing the effect of different home range sizes. Neighbourhood buffers were only assessed using the proportional score to allow comparability as some children's buffers contained areas that were not mapped as they lay outside the classification system and so received no biodiversity value.

3.2.7 Habitat map ground-truthing and error rate

As many sites as possible within the home range and buffer of each child were visited to correct for differences from what was visible from the aerial imagery. I prioritised sites for ground-truthing as 1) sites the child has visited 2) sites the child has mentioned, 3) areas where accessibility needs to be established and 4) other green areas that are more likely to have cryptic features. Ground-truthing of sites involved inspecting sites from viewpoints and either walking across the particular habitat or around the boundary, until the majority habitat was viewed. Any elements of a habitat which appeared from the map to be likely to hold cryptic features or have recently been altered were specifically assessed as well. Errors in the aerial map, due to changes since the image was taken or the presence of cryptic features hidden by others, were recorded as either a change to the habitat type, or the number of features present. Habitat type changes included a complete change of habitat, such as building of a house on a vacant lot, or a major change in boundaries of the habitat. A change in feature richness was recorded if cryptic features were identified, or if features present in the aerial map had been removed e.g. if trees had been cut down. I used these categories to assess the accuracy of drawing maps for unknown areas that could not be visited due to time constraints. The percentage of ground-truthed habitats that had errors recorded was calculated and the percentage of habitats out of the total that were not ground-truthed was also recorded.

As private gardens often took up the majority area of each neighbourhood, only those identified by children as accessible locations were checked. However, these could be viewed only from the outside and so feature richness was estimated based on this information and from the aerial imagery. To account for the misidentification of feature richness without ground-truthing, I applied an error rate to these gardens. During the surveys of biodiversity, described in chapter 2, before visiting gardens I predicted the number of features in the garden using aerial imagery and then checked in person during the survey. For the 28 gardens visited, on average I underestimated feature richness by 0.6 features per garden. This value was then added to the estimated feature richness for children's accessible gardens. Gardens that were not accessed by children in the study could not be feasibly ground-truthed in person due to time constraints. However, any major change in the type of garden, such as redevelopment from low density to high density housing, was recorded if seen. Classification of these gardens was made using the aerial imagery into one of the three garden types (defined in chapter 2). Average feature richness for each garden type was then calculated from the estimated feature richness of the gardens accessed by children in Auckland and Wellington, as well as those gardens in Dunedin assessed in the biodiversity survey. The estimated feature richness was 6.9, 5.8 and 5.3 for garden types 1, 2 and 3, respectively.

3.2.8 Data Analysis

The dependent variables of biodiversity in the buffer, available and accessible home range, and home range size were assessed individually using the statistical programme R (R Core Team, 2013). The relative effects socio-demographic factors were assessed using linear mixed models using the package lme4 (Bates et al. 2013). Scores were non-normally distributed but were transformed for analysis using box-cox transformations, via the package MASS (Venables and Ripley, 2002). I formed candidate model sets based on *a priori* hypotheses of possible interactions from scientific literature and after exploratory analysis (Zuur et al., 2010; Grueber et al., 2011). Continuous variables were standardized prior to model fitting (Schielzeth, 2010). Models were ranked using AIC values, Akaike's weights (Burnham and Anderson, 2002) and R^2 values using the MuMin package (Barton, 2013), which estimate the amount of variance explained by both the fixed and random effects in each model (Nakagawa and Schielzeth, 2013). The assumptions of normality of residuals and homogeneity of variance were confirmed using qq plots and residual versus fitted plots (Zuur et al., 2009).

Linear mixed effect models were used to take account of spatial correlation between children living in the same neighbourhood (Millar and Anderson, 2004). Neighbourhoods were identified as clusters of children living near each other within a similar urban environment, i.e. with a similar urban layout and amount of green cover. As the schools were zoned, most children lived close to their corresponding school, though some lived in the neighbourhood of other schools and others lived in completely separate neighbourhoods. In total seventeen different neighbourhoods were identified, of which three contained only a single child. City was considered in the models as a fixed effect rather than a random effect as it contained too few levels to be modelled correctly (Bolker et al., 2009).

Variation in BioScores within buffer and home range areas were assessed against a set of factors including city, deprivation index, ethnicity and gender. Deprivation was assessed using the 10 point scale of socioeconomic index for New Zealand (Salmond et al., 2007). This was attributed to each child based on the index of the area their home was located within. Eight different ethnicities of children were identified in the study, with Pākehā (European descent) and Pasifika (Pacific Islander descent) ethnicities making up the vast majority of children in the study, composing 39% and 29% of the sample respectively.

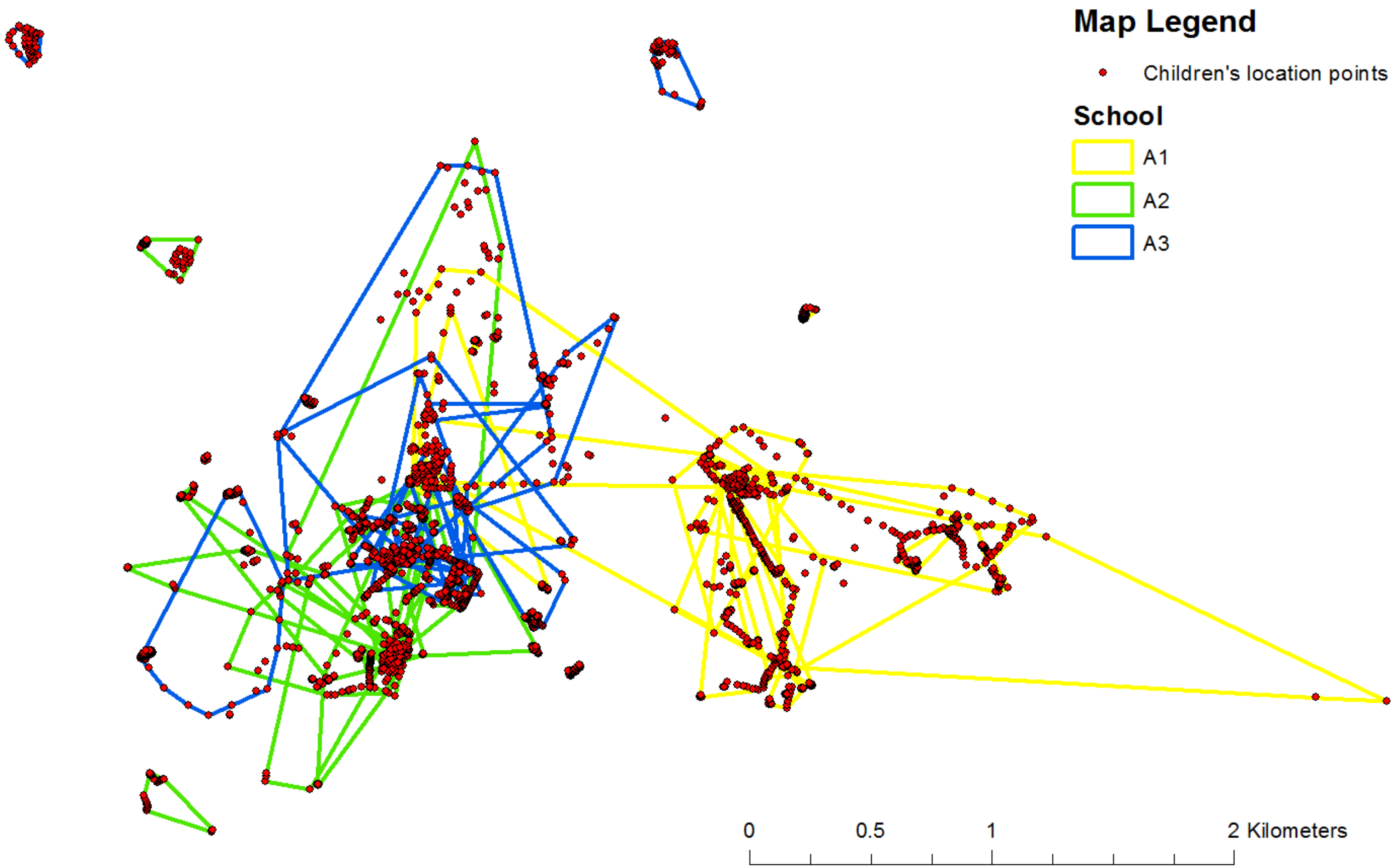
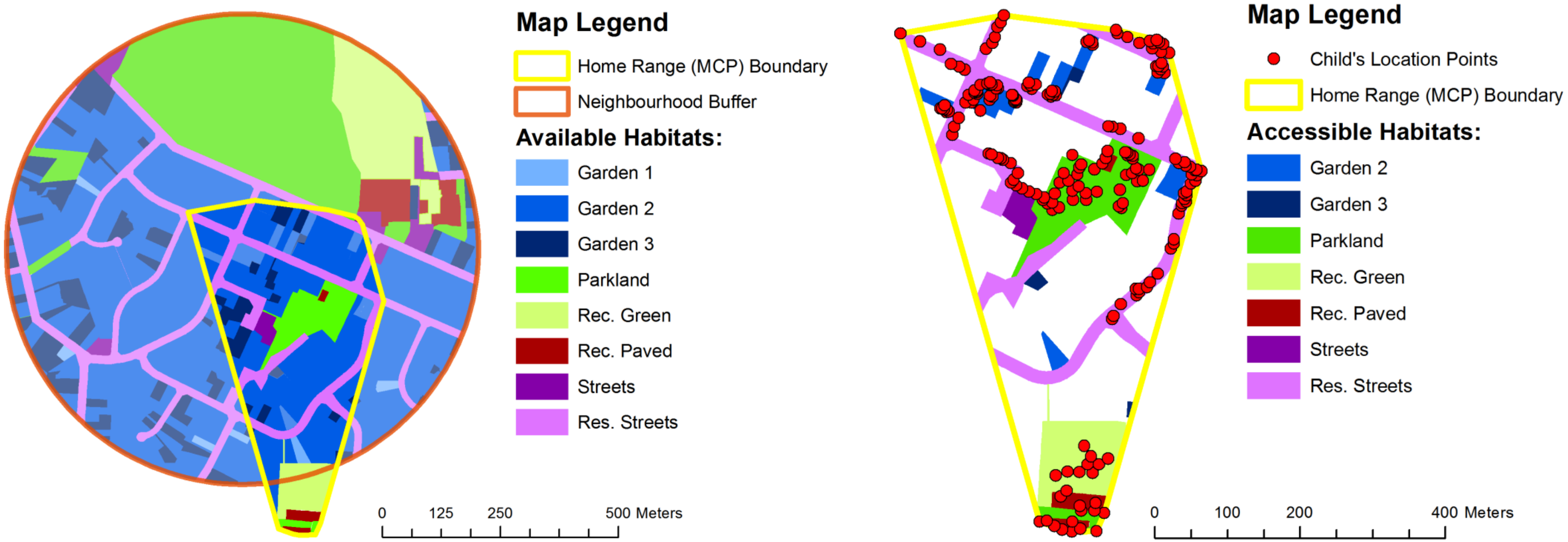


Figure 3.1: Home ranges of children in Auckland grouped by their school. School A1 is the high decile (i.e. low deprivation index local area), School A2 is the medium decile, and School A3 is the low decile. Children’s location points are points were placed onto an aerial map by children in relation to the areas where they spend most time outside. Minimum Convex Polygons (MCPs) were drawn around these points to characterise the home range area.



a)

b)

Figure 3.2. Example of a single child's a) neighbourhood buffer and home range, with the habitats available within both, and b) used location points and accessible habitats within the home range. The location points were placed by the child using aerial imagery of their neighbourhood and indicate the spaces where they spend the most time outdoors. Home ranges were drawn using Minimum Convex Polygons (MCP) around these location points. Accessible habitats are those which are publicly accessible or others that the child has indicated they have access to. Gardens were categorised at three levels (chapter 2), with garden type 1 being the most biodiverse to garden type 3 the least.

3.3 Results

3.3.1 Habitat map error rate

Overall, rates of error in misclassification of habitat type or misidentification of the number of features present within habitats were low (Table 3.1). This indicates the use of the aerial map was accurate in allowing fine-scale identification of habitats and features and that we can be confident in the estimation of habitat characteristics that were unable to be ground-truthed in this study. Overall around one fifth of the total area was not ground-truthed due to lack of accessibility and time constraints.

Table 3.1. Error rate for ground-truthing habitat maps in Auckland and Wellington for all habitat groups apart from gardens. From the habitats ground-truthed the percentage of errors between aerial imagery and actual composition on the ground was recorded as either changes to habitat type or the number of features in that habitat.

City	Number of sites	Percentage of sites ground-truthed	Feature list error rate (% of sites)	Habitat type error rate (% of sites)
Auckland	1883	77.5	4.4	2.9
Wellington	2215	79.5	8.0	1.0
Total	4098	78.6	6.3	1.7

3.3.2 Buffer Biodiversity

Biodiversity in each child's buffer was assessed using the proportional biodiversity score; calculated as the BioScore site multiplied by that site's percentage composition of the total buffer area. One main outlier was present in the data with a much lower score compared to others; this child lived in the city centre of Wellington, which is highly urbanised with little green space, in contrast to all other children who lived in peri-urban suburbs. This outlier was removed as, although it likely accurately reflects the lower biodiversity in the city centre, it skewed the data from other children, who lived at a different point along the urban gradient. With this outlier removed we can be more confident in the interpretations of the results for the remaining children.

On average, biodiversity in children's nearby neighbourhoods was similar between cities (Auckland median = 1333.36, Wellington median =1325.41). However there was a much greater range of values in Wellington than in Auckland, likely due to the greater spatial segregation between the neighbourhoods of interest (see Appendix Figure 3A.1). In contrast, in Auckland all three schools were located close together despite still covering a gradient of socio-economic classes (Figure 3.1).

Both cities showed a decline in buffer biodiversity with increasing deprivation (Figure 3.3a), although this trend was stronger in Wellington than Auckland (Figure 3.3 b, c). Figure 3.3 b) and c) show the three schools of each city, sorted by decreasing decile rank which corresponds to increasing deprivation of the local area. This shows the greater difference between schools and their local area in Wellington in comparison to Auckland, where the differences between schools are not as strong. The high decile school in Wellington (W1 in Figure 3.3 c) had overall the highest biodiversity scores, which were supported by a large proportion of surrounding woodland. In comparison the low decile school (W3) was in a much higher density urban area, though still was surrounded by Wellington's green belt (see Appendix Figure 3A.2). Linear mixed models identified deprivation alone as being the most likely model based on Akaike's weights. Deprivation index of the child's home location explained 35% of the variation between biodiversity scores in the buffer based on the marginal R^2 value. The results of this top model are shown in Table 3.4, while comparison with the closest ranking models out of the candidate set is provided in Appendix Table 3A.4.

Comparisons of the relative availability and accessibility of habitats within children's neighbourhood buffers is provided in Table 3.2.The habitat with the greatest area within children's neighbourhoods was inaccessible gardens, on average taking up more than half of the buffer area. Streets composed the next largest area, with both residential and other streets composing over 20% of the buffer area. Woodland was the most available green habitat in terms of area (17%), followed by parkland (7%). The least common habitats were gardens which the child has access to (<1%), recreationally paved habitats and vacant land (both <2%). Overall green spaces took up a third of the buffer area. In terms of accessibility, measured as the number of children with that habitat present within their buffer, gardens and residential streets were present in all children's buffers. The most accessible form of green space was parkland, with 100% of children having access to a park within 500m of their home. The majority of children also had access to woodland, vacant land and recreational green habitats.

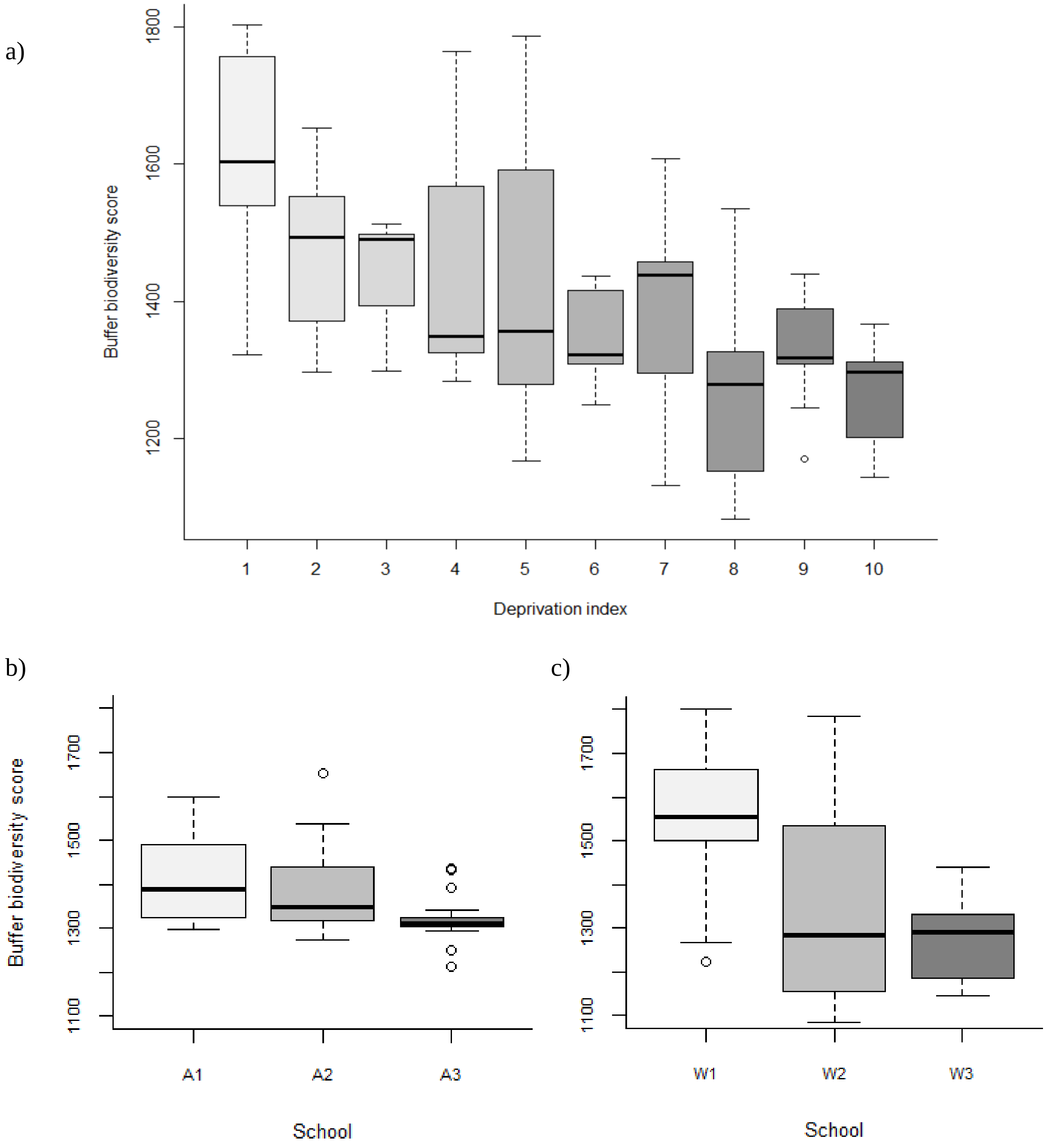


Figure 3.3. Bar and whisker plot of biodiversity scores of children's neighbourhood (500m radius buffer around home). The median biodiversity scores are shown by the bold horizontal bar, boxes define the interquartile range of values, whiskers include values 1.5 times the interquartile range and more extreme points are shown as dots. Figure a) shows an overall decline in biodiversity score with increasing deprivation index. Plots b) and c) compare the differences between b) Auckland and c) Wellington, where children are grouped by their school which fell into a high (1), medium (2), and high (3) decile rankings, with higher decile rankings here corresponding to higher levels of deprivation.

3.3.2 Home range size

Home range shapes and sizes varied across children with home range areas recorded from a few square metres up to 118 hectares. However, the median home range size for children in this study was just over 6 hectares (Table 3.2). Further, as approximately half of the total home range area was identified as being inaccessible to children, this resulted in an actual estimate of the area that children could utilise of 2.73 hectares. Boys tended to have larger total home ranges than girls and also travel further from home. The mixed model analysis also identified that gender had a significant effect on home range size ($F = 5.87$, $df = 2$). However, both the effect size and the R^2 values for this model were very small, and overall gender did not improve model fit over the null random model (delta AIC < 1; see Table 3.4).

Table 3.2. Median values with interquartile range for children's movement and biodiversity metrics for all children and for each gender. The effect of accessibility on estimates of home range size and biodiversity within home ranges is shown.

Group	Maximum distance from home (m)	Home range total area (ha)	Home range accessible area (ha)	Available home range BioScore	Accessible home range BioScore
All	501.66 (229.21-809.76)	6.16 (1.59-19.27)	2.73 (0.78-7.52)	67.5 (20.97-262.14)	27.58 (9.01-94.18)
Boys	571.41 (346.7-875.9)	7.24 (2.15-23.9)	3.62 (1.08-12.02)	87.1 (25.89-298.43)	79.5 (13.14-146.34)
Girls	344.29 (151.78-642.38)	2.89 (1.3-13.77)	1.56 (0.48-4.66)	37 (17.01-197.67)	78.57 (5.63-19.35)

Table 3.3. The availability of each habitat measured as their proportional area within children's neighbourhood buffer and accessible home range (MCP), as well as their accessibility, measured as the percentage of children who had that habitat type present in either the buffer or home range. Note, sample size for neighbourhood buffers was 126 and 118 for home ranges and proportional areas values do not include the home ranges were these habitats were not present. Estimates of percentage area are shown with ± 2 standard error.

	Neighbourhood Buffer		**Accessible home range (MCP)**	
Habitat	Percentage of buffer area (±2 s.e.)	Percentage of buffers with habitat present	Percentage of MCP Area (±2 s.e.)	Percentage of children with access
Agricultural	1.9 (1.46)	19.8	8 (14.79)	1.6
All Gardens	0.8 (1.09)	100	34.7 (5.93)	92.9
Inaccessible Gardens	52.6 (6.37)	-	-	-
Natural	2.6 (0.8)	49.2	10.7 (5.7)	11.1
OPA	2.9 (1.2)	12.7	3.4 (3.71)	3.2
Parkland	7 (1.08)	100	16.4 (3.29)	69.8
Rec. Green	3.1 (0.6)	74.6	21.4 (5.73)	74.6
Rec. Paved	1.5 (0.19)	84.1	13.5 (3.58)	84.1
Street	9.7	92.9	12.2 (3.13)	62.7
Res. Street	11.1 (0.62)	100	30.3 (3.42)	91.3
Vacant	1.6 (0.28)	91.3	5.3 (1.35)	50
Woodland	16.7 (3.12)	82.5	27.8 (6.89)	39.7

3.3.3 Available MCP Biodiversity

The biodiversity score of the available home range area was highly correlated with home range size, with a Spearman's correlation rho = 0.97. Home range size could not be used as a predictor in models as it was implicitly included in the biodiversity scores, by area being multiplied by each habitat biodiversity score, under the assumption based on previous work that biodiversity richness would increase with area (Cornelis and Hermy, 2004; Daniels and Kirkpatrick, 2006). Here the main effect on biodiversity in the home range was gender; with boys having more biodiverse home ranges than girls (Table 3.2). This may be due to the fact that boys tended to have larger home ranges than girls, but could also reflect differences in habitat selection for more or less biodiverse areas. However alternative models failed to improve model fit over the null, with all models providing very poor explanatory power (R^2 values less than 0.1).

Exploration of the overall quality of available home ranges, based on the proportional composition of habitats as part of the home range as opposed to their total area, identified buffer biodiversity and ethnicity to be the two most important factors. The top-ranked model selected by AIC weights suggested more biodiverse buffers supported more biodiverse MCPs, and variation in overall estimates for each ethnicity, with notably Middle Eastern, Māori and Indian having on average lower levels of biodiversity in their available home range compared to other ethnicities. Deprivation was included in the second-ranked model, indicating there may be some effect of deprivation but it is weaker at this smaller scale. These values are listed in Table 3.4 with the other top-ranked models for response variables, while a comparison of candidate models is provided in Appendix Table 3A.3.

3.3.4 Accessible MCP Biodiversity

Accessible habitats within children's home ranges comprised on average 52% of the total home range area. Home range size was again highly correlated with biodiversity, with more biodiversity present within boys accessible home ranges compared to girls. This pattern was however reversed when examining proportional accessible biodiversity, with girls having higher biodiversity values, although the strength of this relationship was very small ($R^2 = 0.06$). However, for proportional accessible biodiversity, linear mixed model analysis indicated the main effects were buffer biodiversity and ethnicity. Buffer Biodiversity had a positive effect on proportional accessible biodiversity, while Middle Eastern and Māori ethnicities had the lowest average levels of biodiversity in the accessible home range (Table 3.4). Gender was also identified as being important, as was included in second-ranked model which had a high Akaike's weight.

I also assessed the relative availability and accessibility of habitats within children's accessible home ranges (values in Table 3.3). Here, in contrast to at the scale of the buffer, accessible gardens composed the largest proportion of accessible home ranges, composing over a third of the accessible area. While the proportional area of green space within the accessible home range increased in comparison to the buffer, the proportion of children with this habitat present within their home range declined, especially for woodland, vacant land and natural habitats.

Table 3.4. Summary of top ranked models as selected by Akaike's weights (*w*) for of the main response variable assessed here. Each of these models included a random effect to take account of spatial correlation within neighbourhoods, the variance explained by this effect is indicated by V_N. Conditional R^2 (c) values indicates the amount of variation explained by both the fixed parameters and random effect (neighbourhood) of each model, while marginal R^2 (m) reflects the variance explained by the fixed effects only (Nakagawa and Schielzeth, 2013). The back-transformed effect size, 95% confidence intervals (CI), and parameters significance are shown for each model. Model sets for each response are supplied in the appendix (Tables 3A.4-9).

Response	Model Summary		Fixed effects summary				
			Fixed effects	Estimate	95% CI	t-value	df
Buffer biodiversity	*w*	0.98	Intercept	1542.13	1459.8 - 635.5	23.93	0.03
	R^2(c)	0.55	Deprivation	-38.05	-31.77 - 42.4	6.91	0.03
	R^2(m)	0.35					
	V_N	<0.01					
Home range size	*w*	0.59	Intercept	3.67	1.02 - 1.06	19.43	115.97
	R^2(c)	0.05	(Female)	4.88	0.57- 0.84	2.5	115.97
	R^2(m)	0.05	Male				
	V_N	0					
Home range available biodiversity (area)	*w*	0.62	Intercept	45.95	25.8 - 77.75	21.01	115.98
	R^2(c)	0.04	(Female)	0.28	0.05 - 0.52	2.38	115.98
	R^2(m)	0.04	Male				
	V_N	<0.01					
Home range available biodiversity (proportional)	*w*	0.76	(Intercept)	13.55	12.26 - 14.79	14.77	86.7
	R^2(c)	0.47	African	-0.02	-0.02 - 3.44	-0.03	105.73
	R^2(m)	0.34	East Asian	-0.65	-5.21 - 3.22	-0.64	108.57
	V_N	101.8	Indian	-0.72	-4.9- 2.57	-0.82	106.57
			Māori	-2.55	-8.04 - -1.00	-2.12	103.78
			Middle Eastern	-0.17	-5.75-5.26	-0.12	81.97
			Other	-0.71	-4.39 - 2.00	-1.0	108.97
			Pacifika	-0.01	-3.22 - 3.19	-0.02	108.97
			Pākehā	1.02	2.01 - 3.9	5.51	108.27
			Buffer biodiversity				108.6
Home range accessible biodiversity (area)	*w*	0.67	Intercept	17.67	10.16 - 29.6	28.41	115.97
	R^2(c)	0.06	(Female)	26.33	3.4 - 92.4	2.73	115.97
	R^2(m)	0.06	Male				
	V_N	0					
Home range accessible biodiversity (proportional)	*w*	0.61	(Intercept)	12.68	11.28 - 14.04	16.67	0.01
	R^2(c)	0.38	African	-0.05	-1.64 - 1.77	20.76	0.01
	R^2(m)	0.36	East Asian	-0.37	-2.39 - 2.0	12.83	0.01
	V_N	0.30	Indian	-1.05	-2.85 - 0.85	15.03	0.01
			Māori	-3.1	-5.3 - -0.71	7.42	0.01
			Middle Eastern	-0.25	-3.0 - 2.98	8.24	0.01
			Other	-0.79	-2.18 - 0.72	29.72	0.01
			Pacifika	-0.03	-2.5 - 1.5	35.02	0.01
			Pākehā	1.21	0.79 - 1.67	5.73	0.86
			Buffer biodiversity				

3.4 Discussion

The approach used here is novel, combining wildlife methodologies with children's interviews to assess children's use of urban space. The use of minimum convex polygons to estimate the size and shape of children's home ranges provides one of the few estimates of the size of these ranges as opposed to just maximum distances travelled (Spilsbury et al., 2009; Villanueva et al., 2012). Further the use of dots reflecting children's most frequently used spaces identified the most important areas within children's home ranges and individual important features such as particular trees. Gathering spatial-use information via interviews was useful in collecting additional information normally unachievable in wildlife studies. Here the use of dots placed by children themselves provided a relatively easy method of gathering information of space use and avoided many issues associated with the use of GPS tags. For instance, the dots reflected space use over a much longer period of time than can be often collected using GPS devices. Additionally, which spaces were and were not accessible to each child could be explicitly identified. This issue of accessibility of different habitats to wild animals is often a major assumption of habitat-use studies (Beyer et al., 2010). Here it was found approximately half of children's home ranges contained inaccessible habitat, severely reducing the true availability of different habitats and of overall biodiversity. Assessments of the provision of green space within cities that do not take into account whether these sites are accessible, could therefore overestimate biodiversity richness of urban areas and opportunities for children to connect with nature.

The use of wildlife methods did however provide some challenges. For instance, minimum convex polygons at times calculated unrealistic home ranges, by either excluding areas it was known the child must have used, such as streets to get to a park, or by including areas of little importance to the child, such as inaccessible gardens they could not use. An improved approach could integrate our knowledge of how children use space (i.e. that they must use particular streets to get to a park) to draw home ranges that include these network paths (Villanueva et al., 2012). This method would be valuable in more accurately assessing accessibility of green space by measuring the distances children must travel as opposed to straight-line distances across inaccessible habitats.

Urban children had a median home range size of about six hectares and a maximum distance from home estimated at just under 500m. However, as much of the home range area was inaccessible, this meant the actual area a child could exploit was 2.73 hectares. This is much smaller than median home ranges from other recent studies assessing home range area. Villanueva et al. (2012) assessed children's 'activity areas' using MCPs and identified a median size of around 36 hectares. However, this area included all areas children travelled and was not limited to only those places they could go independently without an adult. Spilsbury et al. (2009) also used interviews with children to assess independent, 'alone', home range and found a more similar estimate of 10.4 hectares. The smaller estimate of six hectares I have estimated is likely due to the fact that children were asked to only identify their most used areas and not the maximum distance they were allowed to go as Spilsbury et al. (2009) did. As such the approach I have used focuses on the most frequently used spaces and therefore identifies a much smaller area where children spend most of their time. This shows that, excluding occasional long trips, children spend most of their time in a very confined area, which could be due to increasing parental concerns for safety as well as increasingly sedentary lifestyles facilitated by today's level of technology (Valentine and McKendrck, 1997; Karsten, 2005; Pergams and Zaradic, 2006).

Children growing up in the urban areas of Auckland and Wellington are exposed to different amounts of available biodiversity at both neighbourhood and home range scales. Within children's buffer areas, the main effects on the availability of biodiversity was deprivation level, with lower socio-economic areas supporting lower levels of biodiversity, as has been found in previous work (Whitford et al. 2001; Pauleit et al., 2005; Sliuzas and Kuffer, 2008). However, while more biodiverse buffers supported more biodiverse accessible home ranges, the effect of deprivation did not transfer to this finer-scale. This may indicate that the effect of deprivation may act primarily on the inaccessible portions of biodiversity in neighbourhoods, such as private gardens, which have been shown to be more biodiverse and larger in size in more affluent neighbourhoods (Pauleit et al., 2005; Cook et al., 2011). In this study, it was noted that the more biodiverse gardens were more prevalent in higher socioeconomic areas, whereas the least biodiverse garden type was most common in more deprived areas.

As such, while biodiversity in private property may vary by socioeconomic status, the provision of accessible biodiversity was more equally allocated across neighbourhoods. In the urban neighbourhoods assessed here, all children had access to a park within their buffer area and a high proportion also had access to other green areas, indicating an equal provision across socioeconomic gradients. This therefore meets with recommendations set by the European Environmental Agency. Previous work on accessibility of green spaces has shown higher accessibility in lower socioeconomic areas, although the quality of these sites was found to be lower (Ellaway et al., 2005; Barbosa et al., 2007). Children growing up in less deprived areas are more likely to have immediate access to greater biodiversity in their garden, but biodiversity in public spaces overall was found here to be equally accessible.

The amount of biodiversity that was available and accessible was highly linked to home range size. Larger home ranges, by necessity of how biodiversity values were predicted, had higher levels of biodiversity within them, and also had access to a greater diversity of habitat types. During children's interviews, eight children indicated they were not allowed to leave their garden unaccompanied and the vast majority of children reported some form of restriction on movement. While the biodiversity in private gardens can be rich, this cannot substitute for larger, more natural and unmanaged habitat types which allow more free interactions with nature (Pyle, 2002). Here the only identified demographic effect on home range size was gender, which is a common pattern that has been found in earlier studies (O'Brien et al., 2000). However, the difference between boys and girls' home ranges was found here to be very small, indicating a narrowing of this gender gap, which has been recognized in more recent studies of children's geographies (Valentine, 1997; Matthews, 2001). The very small home range sizes reported here, common across socio-demographic groups, fits in with the current trend of declining independence of urban children (O'Brien, 2000). This decline in children's free mobility could lead to fewer opportunities to connect with nature, which itself could have negative consequences for children's well-being and development of a diversity of skills (Pyle, 2002; Wells and Evans, 2003; Louv, 2008).

The relative value of biodiversity within home ranges (proportional biodiversity), showed biodiversity in the available home range area was affected by both biodiversity in the buffer and the child's ethnicity. As would be expected, biodiversity in the available home range was supported by more biodiverse buffers. However, as buffer biodiversity did not have any effect on home range size itself, there does not appear to be any presence of biodiversity acting as a 'pull' on children to travel further (Payne and Jones, 1977). Both available and accessible biodiversity in the home range did vary for different ethnic groups, with Middle Eastern, Indian and Māori groups encountering lower levels of biodiversity compared to other ethnicities. These groups did not show any dissimilarity to other ethnic groups in terms of home range size, indicating these differences in biodiversity may be due to which habitats were selected as part of the home range. Previous research has shown ethnic minorities spend less time outdoors (Hofferth, 2009).

Overall, these results indicate that the amount of biodiversity available and accessible to children in urban areas is more likely to be determined by individual and social, rather than demographic, factors (Hofferth and Sandberg, 2001). In particular home range size showed no relationship to ethnicity or level of deprivation. Instead it is likely that both children's preferences and parents' influence on home range selection itself determine the amount of biodiversity encountered in their home range. Such factors could include parent's safety concerns affecting their child's mobility, such as whether they let their children walk alone to school. Other factors could relate to children's own decisions and motivations for using outdoor space, which could be affected by how many friends children have in the local area and what activities they enjoy outdoors (Jansson and Persson, 2010). These factors could be in part mediated by cultural norms of different ethnicities affecting habitat selection preferences. While it might be expected that most children would seek out the most biodiverse areas available in their neighbourhood (Wilson and Kellert, 1993), the variation in biodiversity within the home ranges between genders and ethnic groups suggests there may be some differences in selection preferences. These possibilities can be assessed in the following chapter assessing habitat-use and selection preferences of children.

Chapter 4: Testing Biophilia: Children's use and selection of urban habitats

4.1 Introduction

The biophilia hypothesis, made famous by EO Wilson in 1984, proposes that humans have an innate, inheritable trait to affiliate towards the natural world (Wilson, 1984). Since 1984, support for this theory has grown as humans have been shown to prefer green and natural landscapes over urban and built landscapes (Kaplan and Kaplan, 1989; Kaplan, 2001; Lindemann-Matthies et al., 2010). This response is explained as an evolutionary-adapted behaviour, as more biodiverse areas would have been signals to our early ancestors of more productive environments and preferential sites for settlement (Kaplan and Kaplan, 1989; Kellert and Wilson, 1993). As such, a preference for more biodiverse environments would be positively selected for, reflected in human habitat use. The expression of this trait can still be seen in the preferential founding of cities in biodiversity hotspots (Kuhn et al., 2004), as well as the beneficial effects green areas have been shown to provide to human well-being (Grinde and Patil, 2009; Bratman et al., 2012). Biophilia has also been shown to extend to children, with natural places often named as their favourite outdoor places (Korpela et al., 2002; Moore, 1986). Natural areas are thought be more attractive sites for children's play, supporting more engaging and creative play experiences (Fjørtoft and Sageie, 2000; Samborski, 2010). There is a positive association between children's use of green spaces and their greenness (Loukaitou-sideris, 2003; Ellaway et al., 2005; Castonguay and Jutras, 2009), and additionally, natural spaces are considered by children to be one of the most valuable aspects of their neighbourhood (Chawla and Malone, 2003; Elsley, 2004).

Yet the expression of biophilia may be under threat, especially in the many children who are now growing up in urban areas. Urbanisation is associated with declines in the availability of green space, with natural areas being built over or replaced with artificial green environments such as gardens and parks (Pauliet et al., 2005). Children are now faced with a landscape that is not designed to support their mobility, but rather that of traffic (Freeman, 1995; O'Brien et al., 2000; Karsten and Van Vliet, 2006) that creates many barriers to their independent movement (Carver et al., 2008). The domination of cars and fears for children's safety has led to declines in children's mobility, with children increasingly spending time close to home, in their gardens, and indoors (Freeman and Quigg, 2009; Spilsbury et al., 2009). In tandem with this, children are spending increased time with electronic media, leading to less active use of the outdoors and natural spaces (Pergams and Zaradic, 2006, Natural England 2009). These factors are thought to be preventing children from connecting to nature and fostering the growth of a biophilia response (Orr, 1994; Miller, 2005). This stifling of biophilia is concerning, as lack of connection to nature during childhood could mean future generations will have little interest in, and not value nature (Miller, 2005; Wells and Lekies, 2006). Wray-Lake et al. (2010) already report a decline in environmental concern in adolescents since the 1990s.

However, opportunities to connect with nature have not completely disappeared from urban areas, which can support more diverse assemblages of habitats and species than natural areas (Kuhn et al., 2004). Therefore urbanisation does not necessitate a decline in nature connection (Nairn et al., 2003), especially if children show innate biophilic behaviour. Valuable green areas in cities include remnant natural areas such as native woodland (Recher and Serventy, 1991; Soga et al., 2014), managed green areas such as parks with high species richness, and abandoned, wild areas which can support unique and rare species (Rink and Herbst, 2011). Biodiversity can exist in the most urbanised features, such as on built structures (Lundholm, 2010). These diverse sites provide an array of biodiversity experiences that can be more varied than within a single natural system. If urban children have an innate biophilic response to their environment, we would expect them to seek out these areas preferentially, even if they only comprise a small proportion of the total area.

In this chapter I aim to discover the degree to which urban children express biophilia by assessing their use and selection of habitats within their neighbourhood. Information on children's habitat use will be useful in identifying which habitats are important to children, as well as estimating how much biodiversity they are likely to encounter in their routine activities. Additionally, I will use resource selection analysis (RSA) to quantify children's preference for particular habitat types. RSA allows inferences on preference to be made as it estimates the probability of use of a habitat in relation to its availability (Manly et al., 2002). RSA has been applied to humans before, for instance to assess the habitat-use of hunters (Stedman et al., 2004), however it has never before been applied to the study of children's habitat use. The results of this more quantitative analysis would therefore provide new insights into children's habitat preferences and enable comparison with the qualitative methods often used to date.

RSA is widely used in wildlife studies to model how a particular animal interacts with its environment, what environmental features drive its use of space, and to infer which habitats are most important to that animal. If an animal uses a given resource to a greater degree than its proportional availability, that resource is said to be preferred (Johnson, 1980). Habitat-use by children can be assessed under the same concept, with higher proportional use of habitats indicating greater preference by children. However, in contrast to wildlife, a child's home provides the majority of resources animals would normally need to utilise their home range for, such as shelter and food. For children instead the main motivation to travel outdoors, excluding mandatory trips such as to school, appears to be play. Play is the main motivation named by children for their reason to visit particular outdoor areas (Korpela et al., 2002; Min and Lee, 2006; Kalvaitis and Monhardt, 2012). Other factors that could affect how they use space could include social factors, such as travelling to meet with friends (Jansson and Persson, 2010, Rehrer et al., 2011). Children may also show avoidance of particular habitats, such as those they are not allowed to visit alone, or that they feel are unsafe (O'Brien et al. 2000). Examples of this include major roadways that children are not allowed to cross (Timperio et al., 2004) or parks they view as being unsafe (Castonguay and Jutras, 2009). Children have also been shown to avoid industrial areas, cemeteries and the grounds of schools other than their own (Rehrer et al., 2011). Biodiversity is therefore just one possible factor in many that could be influencing children's selection of urban habitats.

In this chapter I will quantify children's habitat-use in two urban centres of New Zealand. I will assess whether children are showing a biophilic response by comparing their frequency of use of habitats, and by assessing biodiversity value of the most frequently used areas, and additionally by applying RSA to measure children's habitat preferences. I will also investigate the effects of demographic and environmental parameters on children's selection and use of biodiversity. This information could help identify the relative roles of urbanisation, declining children's independence and decline in the biophilia response play in the loss of children's connection to nature.

4.2 Methods

4.2.1 Children's habitat-use and home ranges

I explored children's use of urban habitats and estimated their home range size using information on where they spent the most time outdoors, gathered during one-on-one interviews. During these interviews children placed dots on an aerial map in relation to where they spent the most time outdoors, placing more dots in places they spent more time, and less dots in places they spent less time. Children were asked to place at least 30 dots on the map, as this is a minimum number recommended for home range and resource selection analyses (Millspaugh and Marzluff, 2001; Baasch et al., 2010). I used these dots to define children's home range areas as the area most frequently used, and estimated their size using 100% Minimum Convex Polygons (MCPs; described in more detail in chapter 3). Out of the 126 children interviewed, home ranges were drawn for 118 who placed more than 30 dots on the map. A habitat map was drawn for each child's home range and surrounding neighbourhood that classified landcovers into 13 habitat types, which were defined in chapter 2. The map was drawn based on aerial photographs and ground-truthed information in ArcGIS (v10.2, Esri, 2013). The accessibility of habitats was identified as sites either being publicly accessible, or private areas to which the child indicated during the interview they had access to. I explored which habitats were most used by children by calculating the proportion of dots within each habitat type.

I compared the use of habitats against demographic factors including city, deprivation level, gender and the biodiversity score of the child’s home range. Deprivation level was estimated for each child by the deprivation score of the meshblock which their home was located (Salmond et al., 2007) and was divided into low, medium and high deprivation groups for comparison. Biodiversity scores were calculated to estimate aspects of biodiversity children would relate to (chapter 2) and were calculated across children’s home ranges and neighbourhoods in chapter 3. The biodiversity score of children’s home range was categorised into equally sized high, medium and low groups.

I then estimated how much biodiversity children would be exposed to when spending time in each of these habitats by predicting the biodiversity across children’s habitat maps using the BioScores calculated in chapter 2. Each child’s use of biodiversity was then calculated by assigning each dot the BioScore of the habitat it was placed within. For each child I averaged their dot’s BioScores to produce an estimation of how much biodiversity each child is exposed to in their day-to-day activities. Further, I calculated a proportional-use score to compare children’s average use of biodiversity against the average biodiversity score of their home range. This average biodiversity score of the home range represents how much biodiversity a child would be encountering if they were using different habitats in their home range in direct relation to their availability.

I used linear mixed models to explore the possible effects of demographic factors on the use of biodiversity by children. Mixed effect models were used to control for pseudoreplication of children living within the same neighbourhood (Miller and Anderson, 2004). Models were ranked using Akaike’s weights based on AIC values (Burnham and Anderson, 2002). Marginal and conditional R^2 values were estimated to compare the model fit for the fixed effects only and fit for both the fixed and random effects respectively (Nakagawa and Schielzeth, 2013). Degrees of freedom for the mixed models were estimated via the package lmerTest (Kuznetsova et al., 2014). Models were checked that they met the assumptions of normality and homogeneity of variance in the model residuals (following Zuur et al., 2009).

4.2.2 Defining Resource Selection Analysis: Used vs. Available Design

I applied a Used-Available design for the resource selection analysis (Manly et al., 2002), which compares data on the habitats children used to a measure of their availability, measured specifically to each child. I defined within the available habitat which areas are accessible to children, so I specifically compared the use of habitats to their accessibility, following the same format as a Used-Available design. Here the use of habitats was estimated using the location points placed by children as indicators of where they spend the most time outdoors.

The accessibility of urban habitats was calculated by generating randomly placed points within each child's accessible home range using the 'generate random points' tool within ArcGIS. Recommendations on the number of available points to use can vary (Manly et al., 2002), although what is most important is that the available points accurately characterise the accessible area (Buskirk and Millspaugh, 2006). I used a 1:4 ratio of used-to-accessible points as, in an earlier pilot study, this ratio was found to successfully identify all habitats present in four children's home ranges. In the pilot children on average placed 50 dots, which meant 200 random points would be generated for each child in the main study. However, in practise, 300 dots were estimated to allow for some to be removed if they fell in areas that were found to be inaccessible during ground-truthing. The median number of dots placed by children in the main study was 52, thereby maintaining, for most children, a 1:4 ratio of used-to-accessible.

For each used and accessible point I calculated the nearest Euclidean distance to each accessible habitat type as well as distance to the child's home. This distance approach provides continuous measures of use of habitats in comparison to compositional methods (Conner et al., 2003), thereby reducing the number of zero-values that can cause problems during statistical analysis (Zuur et al., 2010). Further, it allows the incorporation of responses of habitats that are not directly used (Conner et al. 2003). This would allow preference for biodiverse areas that aren't directly used to be identified (English Nature, 2002; Conner et al., 2005; Hedblom et al., 2014); for instance, children have shown preference for play areas that are surrounded by native habitats (Jansson and Persson, 2010). Distances to each habitat from each used and accessible point were calculated using the *Near* function in ArcGIS.

4.2.3 Scale and design of analysis

Habitat selection by animals can vary depending on the scale of analysis and multi-scale assessment of habitat preference is recommended to capture a more complete picture of an animal's habitat preferences (Boyce, 2006; Meyer and Thuiller, 2006). Here I analysed resource selection of children at two scales (sensu Johnson, 1980); third-order selection of patches within the home range; and second-order selection of a home range within a wider area.

At the broader scale of selection, the second-order, the selection of each child's home range was compared against an area based on each child's maximum distance travelled from home (MD Buffer).

I used an accessible area defined for each child, rather than the study area landscape typically used (Meyer and Thuiller, 2006), as children are bounded in their selection of home range by their home location. The MD Buffer was defined as a buffer around a child's home which was then altered depending on any specific information the child had given on restrictions on their movement to ensure only accessible areas were included. For instance, if a child indicated they were not allowed to pass a certain road, then the MD Buffer was altered to exclude this area. Availability of habitats within the MD Buffer was characterised in the same format as for children's home ranges, with 300 random points generated in accessible habitats. Eight children had MD Buffers that did not extend beyond their home range area. These children were not allowed to venture past their garden unaccompanied, making their home range and accessible area the garden area only. Therefore, these children cannot show selection of their home range, and were removed from the analysis. Figure 4.1 illustrates these two different scales of analysis, with one child's MD Buffer and home range shown.

In addition to the two orders of selection, I applied two different approaches at the third order of selection. These two approaches (summarised in Table 4.1) altered how the availability of habitats was measured and were devised to adapt to the very high heterogeneity of accessible habitats between children's home ranges. The first approach is termed 'strict' as it follows the normal approach to RSA analysis, where I only measured distances from used and random locations to accessible habitats present within that child's home range. The typical response to cope with high heterogeneity in habitat availability between individuals in RSA is to merge habitat groups together (Aebischer et al., 1993). I followed this method here by reducing the original 13 habitat types down to four; garden habitats, biodiverse habitats, play habitats and street habitats (listed Table 4.2). However, even with this reduction in habitat types, only 30 children had access to the full set of habitat groups within their home range. Therefore, I split the children into different datasets depending on which set of habitats were available to them, leading to three sub-groups of children for independent analysis. Therefore this strict response assesses the response of children to those accessible habitats present within their home range only.

The above strict design approach meant a reduction in detail with the amalgamation of habitats types, potentially averaging out any differences in responses to specific habitats by children. Despite this reduction in habitat types, the number of children for three of the four sub-groups remained below 50% of the total number of children. To allow for detailed habitat responses to be assessed and inference across the total sample of children to be made, I allowed distances to be measured to those accessible habitats present outside the home range area. This approach, which I termed 'adapted', allowed distances to be measured to accessible habitats that were present outside of the child's home range. This therefore allowed all children to have a value for all habitat types and therefore be assessed as one group. It also took account of the response of children to nearby habitats that they do not specifically use (Conner et al., 2003), and acknowledged the physical inaccessibility of habitats as these will all have similarly larger values compared to others for both used and accessible points. As such, this approach falls as an intermediate of selection between the second and third orders, where accessibility is defined by random points within the home range, but the selection of each point is assessed with habitats outside this area. This approach could cause spurious results to occur, as habitats could be shown to be selected without ever being used if children spent time near to habitats. This could reflect a real preference to be near particular habitats; studies have shown people can gain benefits from simply viewing nature (Ulrich, 1984; Hedblom et al. 2013), although it may also identify selection by children when they may not be specifically choosing to be close to a habitat. I compared the results using the same method I used with the strict method to assess if the adapted method could be used as an indicator for selection at the strict level with the larger and more detailed dataset.

Table 4.1. The format of the resource selection analysis, where two different orders of scale (Johnson 1980), and two different approaches to measuring habitat availability, were applied to overcome differences in habitat availability between children.

Order	**Approach**	**Description**
3: Selection of patches within home range	Strict	Distances measured only to those accessible habitats present within the child's home range. Models were run for sub-groups of children based on availability to different habitats
	Adapted	Distances measured to accessible habitats both inside and outside the child's home range.
2: Selection of home range within MD Buffer	Strict	Distances measured habitats inside the Order 2 Area (MD Buffer) Models were run for sub-groups of children based on availability to different habitats

Table 4.2. List of the individual habitats grouped together into three broad habitat groups. Further, within each habitat group, the rarer habitats were combined into larger sub-groups. The number of children (out of 118) with access to each habitat group at each order of selection is shown.

Group	**Habitats**	**Number of children with access to habitats at:**	
		Third order	**Second order**
Biodiverse (B)	1. Woodland 2. Natural group (agricultural, vacant land and natural areas)	30	80
Play (P)	1. Parkland 2. Recreational group (recreational paved and recreational green)	69	99
Street (S)	1. Streets group (residential streets, streets and open public paved areas (OPA))	107	115
Gardens (G)	1. Gardens (Garden 1, Garden 2 and Garden 3)	117	118

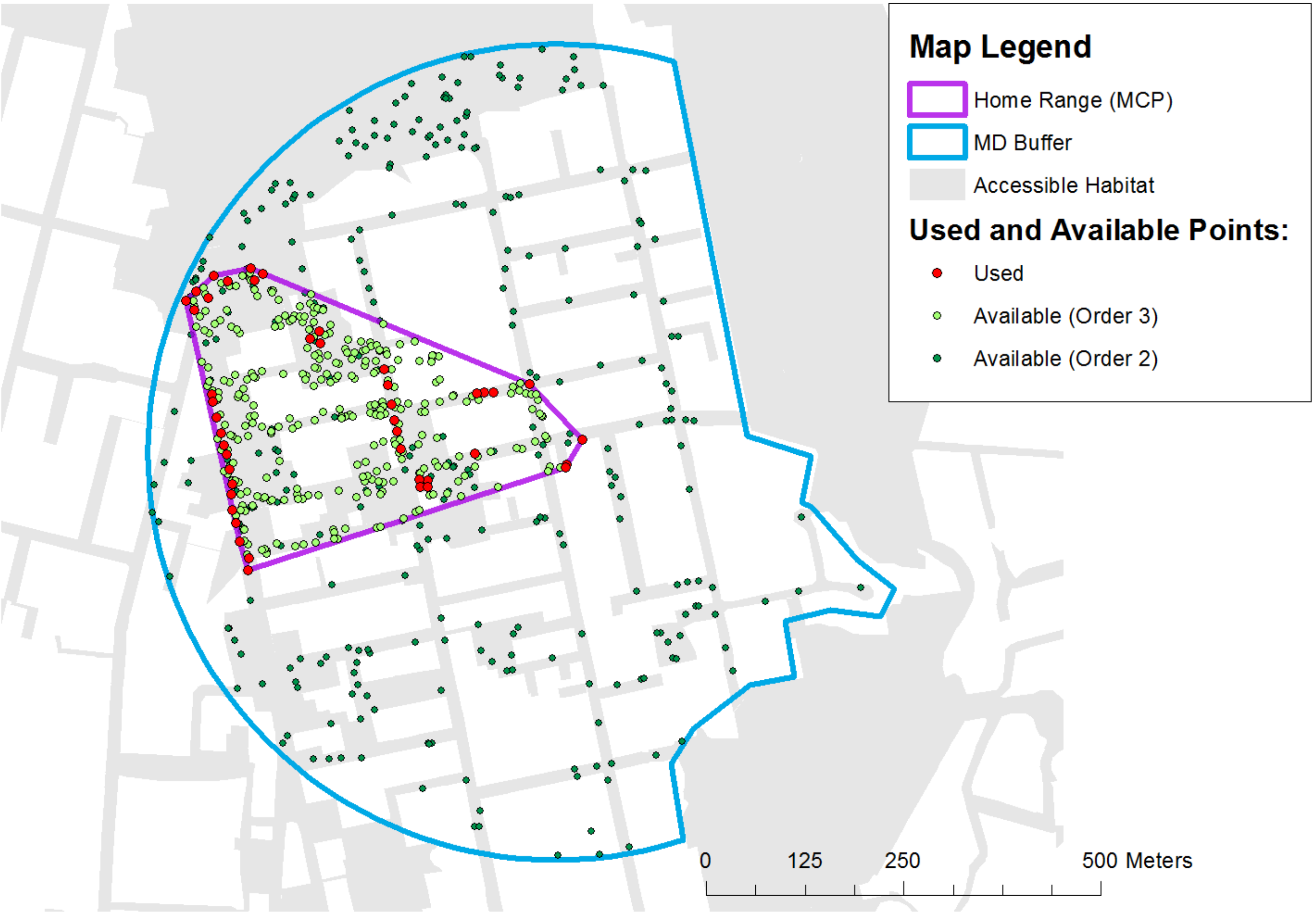


Figure 4.1. Illustration of the scales of assessment used here to assess children's habitat selection preferences. The child's home range provides the boundary for the third order and describes the area the child uses most frequently, calculated as a minimum convex polygon around the child's used (red dot) locations. The MD (maximum distance from home buffer) provides the boundary for order 2, and represents the area accessible to each child, based on the maximum distance travelled at home and altered to exclude the areas the child has identified as being inaccessible. In this example the MD buffer has been amended to exclude woodland habitat on right edge of the buffer which the child was not allowed to visit alone. Within these two areas Available points were randomly drawn to characterise the available area, with one set being applied to the third order (light green) and the other to second order (dark green).

4.2.4 RSA statistical analysis

I used General Linear Mixed Models (GLMMs) for the resource selection analysis, with the response (point used by child or an available coded as 1 or 0, respectively) as the dependent variable in a logistic regression framework (Manly et al., 2002). The logistic regression analysis takes the form of:

$$\hat{w}(x) = \exp(\beta_0 + \beta_1 x_1 + \ldots . \beta_n x_n)$$

where the regression coefficients (β) are estimated for each habitat covariate (x) in the model for each value, 1 to n. I used a random effect with this model to account for replication within each child (Gillies et al., 2006) GLMMs are also useful in this context at they can account for unbalanced samples between individuals (Gillies et al., 2006). Negative coefficients estimated from the model indicate a positive preference by children, as it indicates that a decrease in distance to a habitat is associated with an increased probability of use.

I created candidate model sets based on *a priori* hypothesis of how children might respond based on previous studies (Anderson and Burnham, 2002). These hypotheses were divided into four groups based on different habitat groups that children might find important; habitats close to home, natural habitats (e.g. woodland), play habitats (sports fields and parks) and street habitats (described in Table 4.3). These four groups of hypotheses represented four different possible motivations for children using outdoor space. The first, “close to home”, predicts that children, will prefer to stay close to their home, which is suggested from previous research (Min and Lee, 2006; Veitch et al., 2008; Jansson and Persson, 2010). The biophilia hypothesis is included in the second group of hypotheses that test the assumption that children seek out and use the most biodiverse spaces available to them. Here biodiverse spaces were broken into five different models to represent the different values of biodiversity calculated in Chapter 2, such as the wildest habitats or the most structurally diverse. These different facets of biodiversity have been suggested to affect children’s value of natural spaces (Pyle, 2002; Benkowitz and Kohler, 2010; Samborski, 2010). Third, the play hypothesis assume children use outdoor space mainly for play, and most of their time outdoors will be in the habitats designed for play such as recreational paved, recreational green and parks (Jansson and Persson, 2010). Lastly, the street model assumes that children will show strongest selection for streets as they travel between different habitats, as well as use street areas for running errands, such as to the local diary, and to meet friends and for play (Tranter and Pawson, 2001; Karsten, 2005). In addition global and null models were run for comparison with the *a priori* hypotheses.

These model sets were applied to each scale and design of analysis and top models were selected based on Akaike’s weights calculated from AIC scores (Burnham and Anderson, 2002). Analysis was performed in R using the Lme4 package (R Core Team 2012; Bates et al., 2014). The models were fitted with a binomial family and logit link function. Collinearity between variables was tested using

Spearman rank correlations. A correlation threshold of r = 0.6 (Hosmer and Lemeshow, 2005) was applied: only gardens and distance to home variables were correlated, and so were not included in the same model. Possible effects of demographic fixed factors were investigated by using these factors to subset the data, and then compare model selection between these groups.

Table 4.3. List of hypotheses used in the resource selection analysis. BioScores used to test different responses to different aspects of biodiversity were calculated in chapter 2.

ID	Model Group
Full (all accessible habitat types)	
1a	Gardens + Full
1b	Home + Full
Close to home	
2a	Gardens only
2b	Home only
Biodiverse habitats	
4a	Highest BioScores: Woodland, gardens, vacant land
4b	Highest species richness: Woodland, garden
4c	Highest structural richness: Woodland, gardens, parkland
4d	Highest naturalness: Woodland, vacant, natural areas
4e	Highest greenness: Woodland, vacant, parkland
Play habitats	
5a	Formal play areas: Rec. paved, rec. green, parks, gardens
5b	Play areas without gardens: Rec. Paved, rec. Green, parks.
5c	Sports sites only: Rec. Paved, rec. green
Street habitats	
6a	Streets, residential streets, OPA

4.3 Results

4.3.1 Children’s use of biodiversity

Linear mixed model analysis revealed the more biodiverse accessible home ranges supported higher use of biodiversity in children’s day-to-day movements (Table 4.4). However, this response would be shown if children were simply using biodiversity at an equal rate to its availability. The proportional use of biodiversity score, assessed whether children’s use of biodiversity was greater, or less than expected given the estimate of the null use of their home range. This showed the majority of children (73%) used more biodiversity than would be expected under a null-use of their home range. Analysis of this response showed little effect of demographic factors, with none of the candidate models improving significantly from the null (Table 4.5). Only gender had a weak effect, with boys showing greater proportional use of biodiversity compared to girls.

Table 4.4. Top five candidate models ranked by Akaike's weights examining children's use of biodiversity within their accessible home range. Model summary values for each model list the Akaike weight (*w*) of the model, the conditional and marginal R^2 values, and the variance in the model explained by the random neighbourhood effect (V_N) included in all models. Fixed effect summary reports the variables included in each model, their coefficients, t-values, standard error and degrees of freedom.

Model Rank	**Model Summary**		**Fixed effects summary**				
			Fixed effects	**Coefficient estimate**	**Standard error**	**t-value**	**df**
1	*w*	0.99	Intercept	12.47	0.2	61.4	13.34
	R^2(c)	0.49	Home range biodiversity	1.4	0.15	9.4	108.12
	R^2(m)	0.45					
	V_N	0.2					
2	*w*	0	Intercept (African)	12.27	0.56	21.87	0.01
	R^2(c)	0.48	+ East Asian	-0.49	0.69	-0.7	0.01
	R^2(m)	0.46	+ Indian	-0.43	0.91	-0.5	0.01
	V_N.	0.11	+ Maori	0.97	0.79	-0.5	0.01
			+ Middle east	0.64	1.08	1.23	0.01
			+ Other	1.18	1.24	0.59	0.01
			+ Pacifica	0.52	0.62	0.95	0.01
			+ Pakeha	0.1	0.62	0.83	0.01
			+ Buffer biodiversity	-0.12	0.19	-0.64	0.01
			+Home range biodiversity	1.55	0.56	0.17	0.01
3	*w*	0	Intercept	12.65	0.41	30.51	44.32
	R^2(c)	0.51	Home range biodiversity +	1.6	0.33	4.8	113.97
	R^2(m)	0.46	Deprivation +	-0.04	0.06	-0.68	69.84
	V_N.	0.26	Home Range Bio. * Dep.	-0.04	0.06	-0.7	0.06
4	*w*	0	Intercept (Female)	12.12	0.27	45.53	30.76
	R^2(c)	0.51	Home range biodiversity +	1.48	0.15	9.74	109.94
	R^2(m)	0.46	(Male)	0.6	0.29	2.05	112.45
	V_N	0.21					
5	*w*	0	Intercept	12.9	0.44	29.1	49.02
	R^2(c)	0.45	Home range biodiversity +	-1.47	0.18	8.38	109.46
	R^2(m)	0.44	Deprivation +	-0.08	0.44	-1.13	70.89
	V_N	0.27	Buffer biodiversity	-0.21	0.21	-1.01	113.92

Table 4.5. Top five candidate models ranked by Akaike's weights examining children's proportional use of biodiversity within their accessible home range. Model summary values for each model list the Akaike weight (w) of the model, the conditional and marginal R^2 values, and the variance in the model explained by the random neighbourhood effect (V_N) included in all models. Fixed effect summary reports the variables included in each model, their coefficients, t-values, standard error and degrees of freedom.

Model Rank	**Model Summary**		**Fixed effects summary**				
			Fixed effects	**Coefficient estimate**	**Coefficient standard error**	**t-value**	**df**
1	w	0.77	Intercept (Female)+	1.04	0.02	49.19	26.85
	R^2(m)	0.08	Male	0.07	0.02	3.28	114.16
	R^2(c)	0.17					
	V_N	<0.01					
2	w	0.21	Intercept	1.087	0.002	63.94	13.31
	R^2(m)	0					
	R^2(c)	0.09					
	V_N	<0.01					
3	w	0	Intercept (Female)+	1.04	0.02	48.35	0.03
	R^2(m)	0.08	Home range biodiversity+	0.001	0.012	0.15	0.01
	R^2(c)	0.17	Male	0.07	0.02	3.24	0.01
	V_N	<0.01					
4	w	0	Intercept (Female)+	1.06	0.03	32.85	30.68
	R^2(m)	0.08	Male +	1.14	0.04	32.-3	37.59
	R^2(c)	0.21	Deprivation	-0.004	0.004	-0.89	56.31
	V_N	<0.01					
5	w	0	Intercept	1.11	0.07	17.04	94.81
	R^2(m)	<0.01	Home range biodiversity	-0.002	0.005	-0.42	109.09
	R^2(c)	0.09					
	V_N	<0.01					

4.3.2. Children's habitat use

Across all children in the study, the most used habitat types were gardens (40% of used locations) and residential streets (15%). Around 10% of time was spent on playfields and less than 5% in woodland, vacant or natural habitats combined. Children in Auckland and Wellington differed in their use of habitats, which may be due to varying availability of these habitats in children's home ranges (Figure 4.2). In Auckland, children showed greater use of gardens and recreational green spaces (i.e. sports fields) than children in Wellington: both of these were more available in Auckland. In contrast, in Wellington children used residential streets and recreational paved (i.e. paved courts) to a greater degree.

There was little difference between the genders, with boys showing only slightly higher use of habitats outside of gardens than girls, who instead used gardens and residential streets most intensively (Figure 4.3). The availability of habitats did not vary significantly between low, medium and high deprivation groups, with only woodland showing a higher availability for medium and low deprivation children compared to high deprivation children (Figure 4.4). Deprivation appeared to have more influence on children's use of habitats, with lower deprivation children showing higher use of residential streets but less frequent use of recreational paved and parkland in comparison to children from more deprived areas.

I also assessed whether greater provision of more biodiverse habitats within children's home ranges altered their use, by comparing children with low, medium and high biodiversity scoring home ranges. As would be expected, children with more biodiverse home ranges had greater proportions of natural habitats, with the most significant change in the availability of woodland habitats (Figure 4.5). Children with more biodiverse home ranges tended to show greater use of the biodiverse habitats they had access to, although the difference between groups was small. The major difference between the groups was the proportional availability and use of gardens, with children with low biodiversity home ranges having much greater availability and use of garden habitat. This is likely due to the fact that children with low biodiversity scores tended to have smaller home ranges, and therefore, gardens comprised the majority of space.

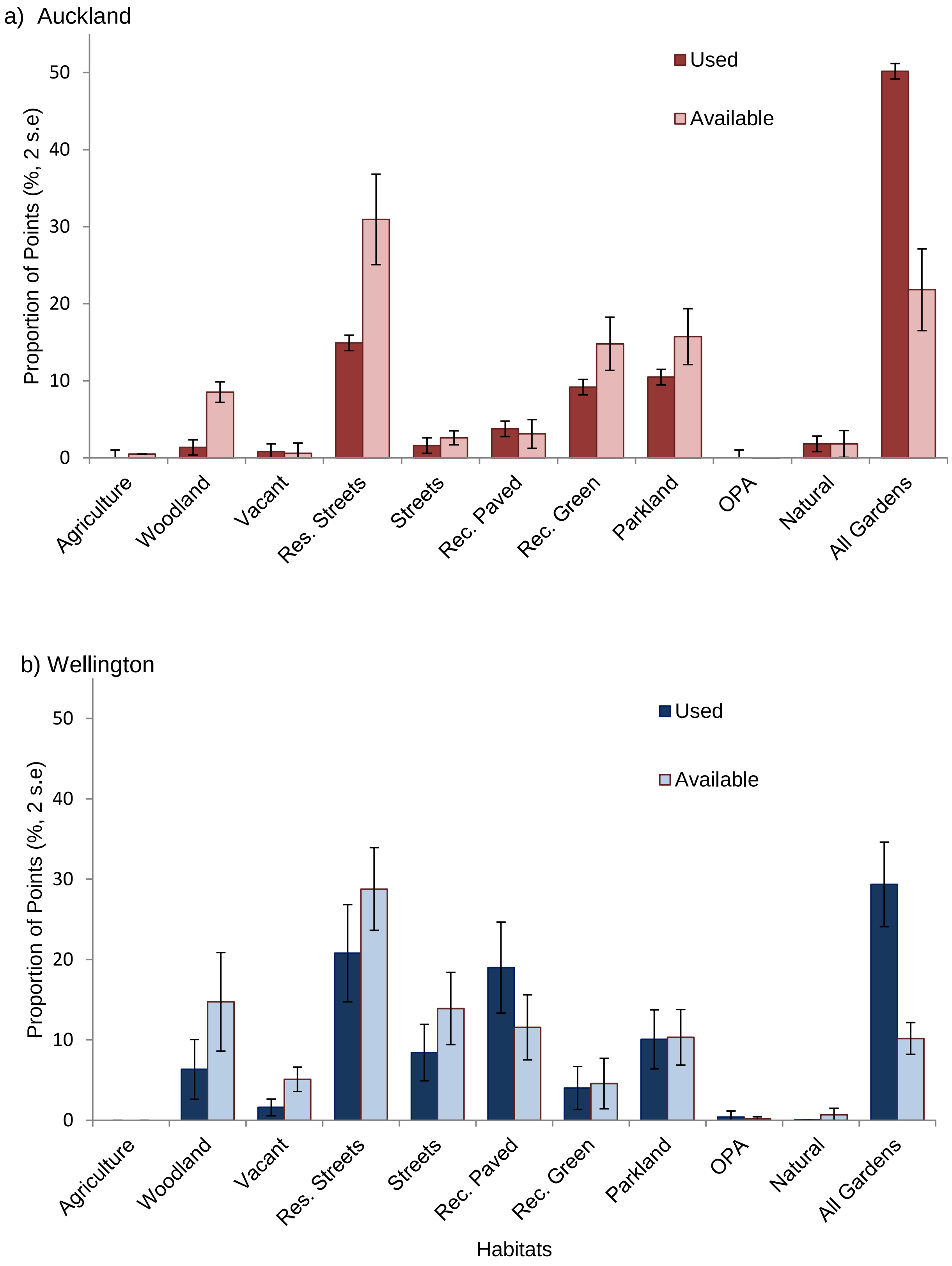
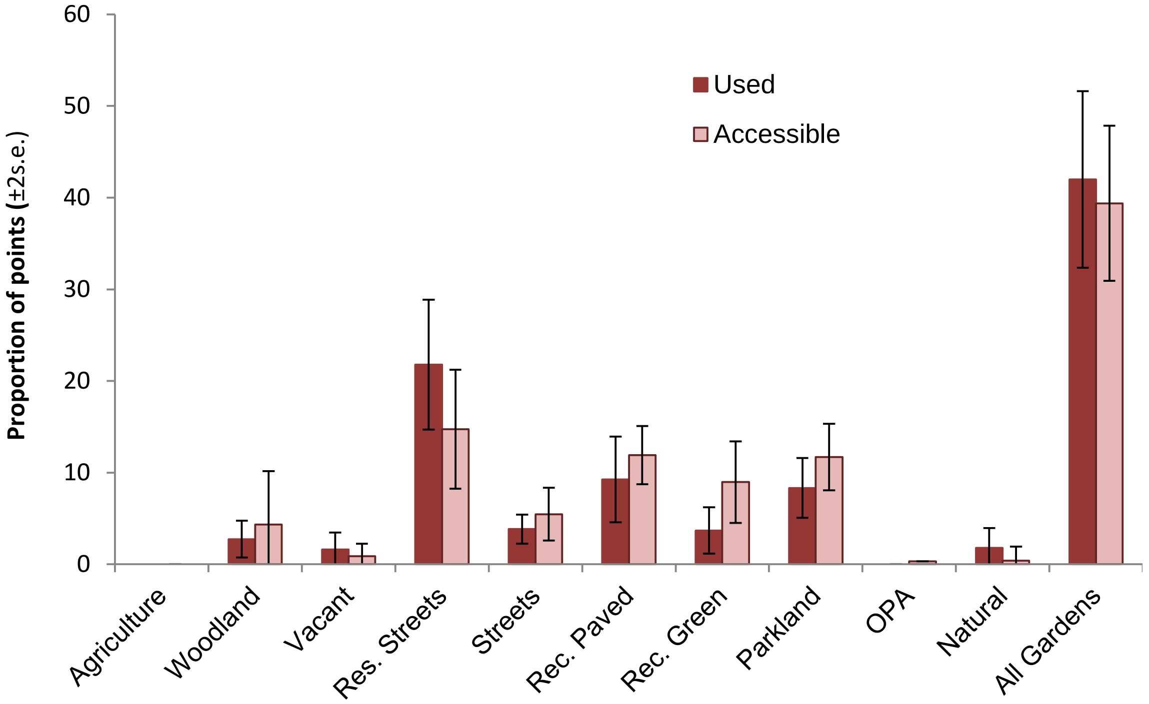


Figure 4.2. Comparison of average habitat availability and use of habitats for children in a) Auckland, and b) Wellington. Used points were children's location placed on aerial map in the places they spend the most time; available proportions were taken from randomly generated points within each child's home range.

a) Girls



b) Boys

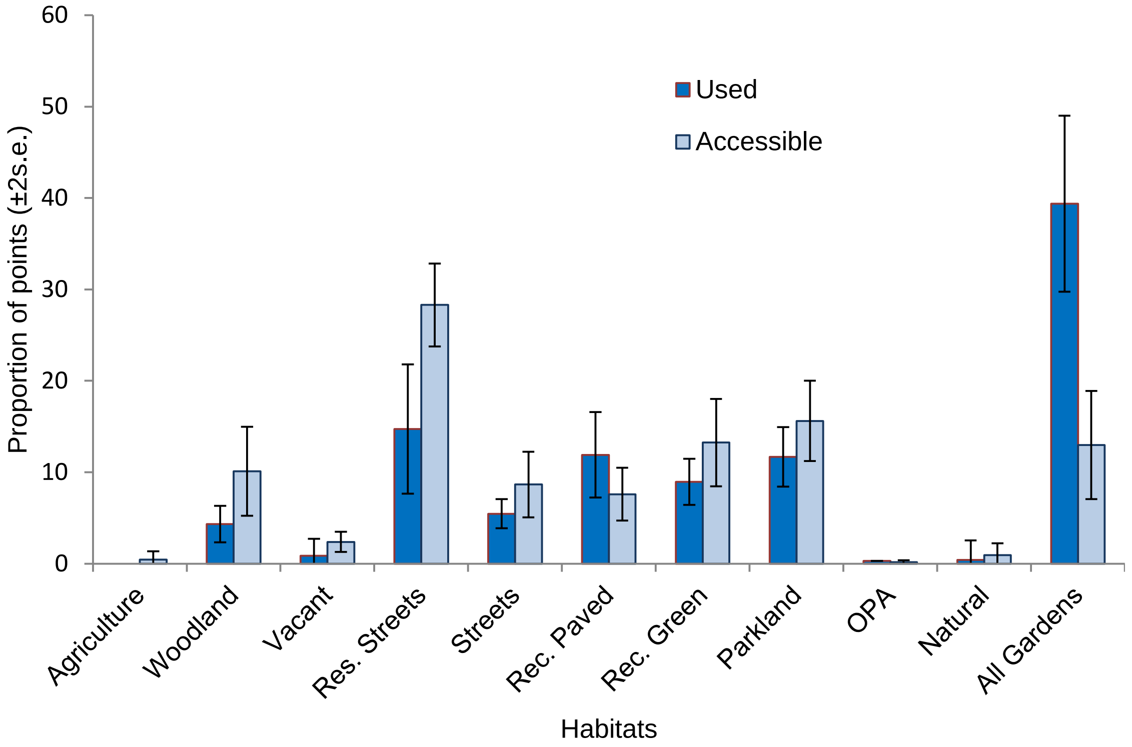


Figure 4.3. Comparison of average habitat availability and use of habitats for a) girls, and b) boys. Used points were children’s location placed on aerial map in the places they spend the most time; available proportions were taken from randomly generated points within each child’s home range.

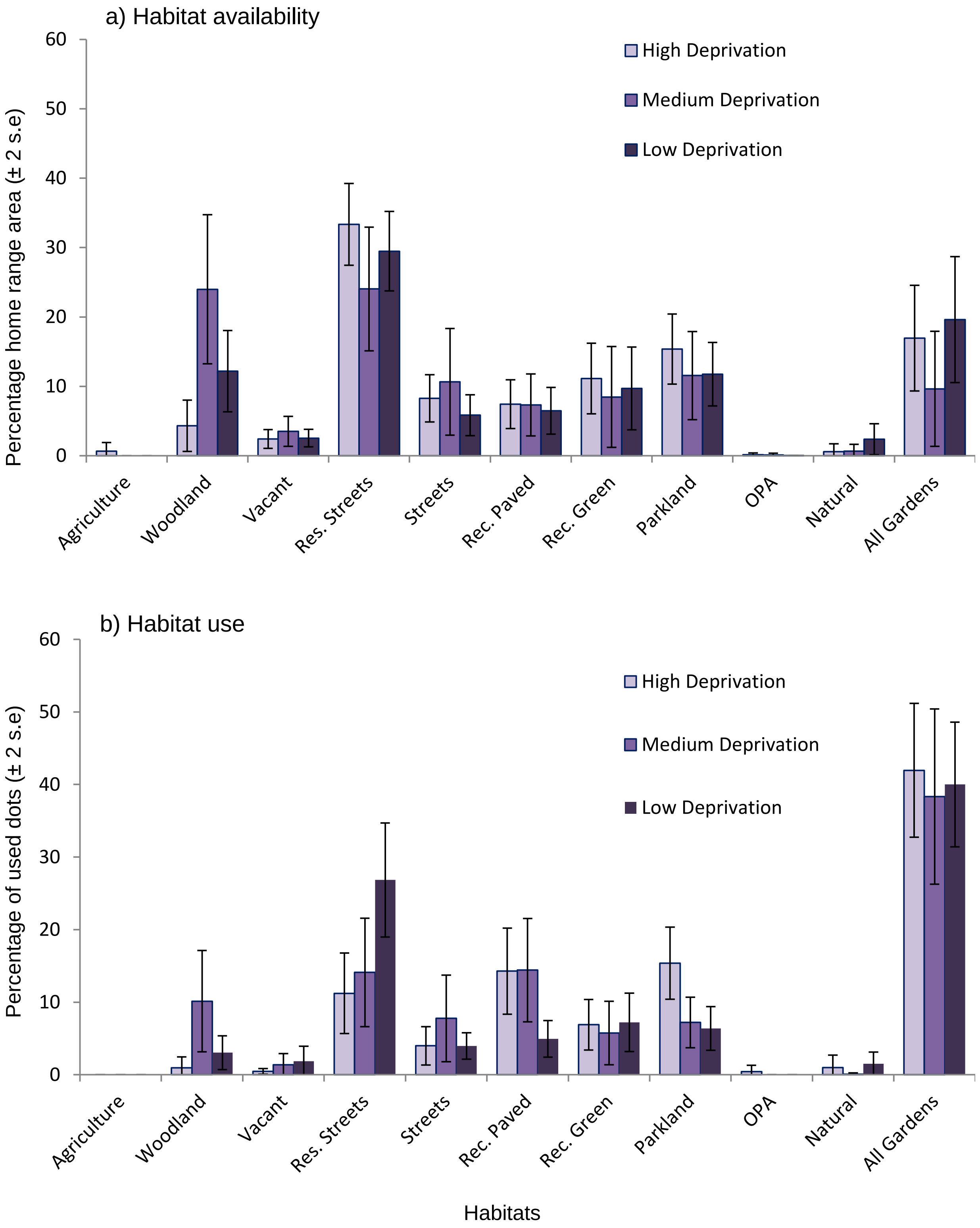


Figure 4.4. Comparison of a) habitat availability and b) habitat use by children living in low, medium and high deprivation areas. Used points were children’s location placed on aerial map in the places they spend the most time; available proportions were taken from randomly generated points within each child’s home range.

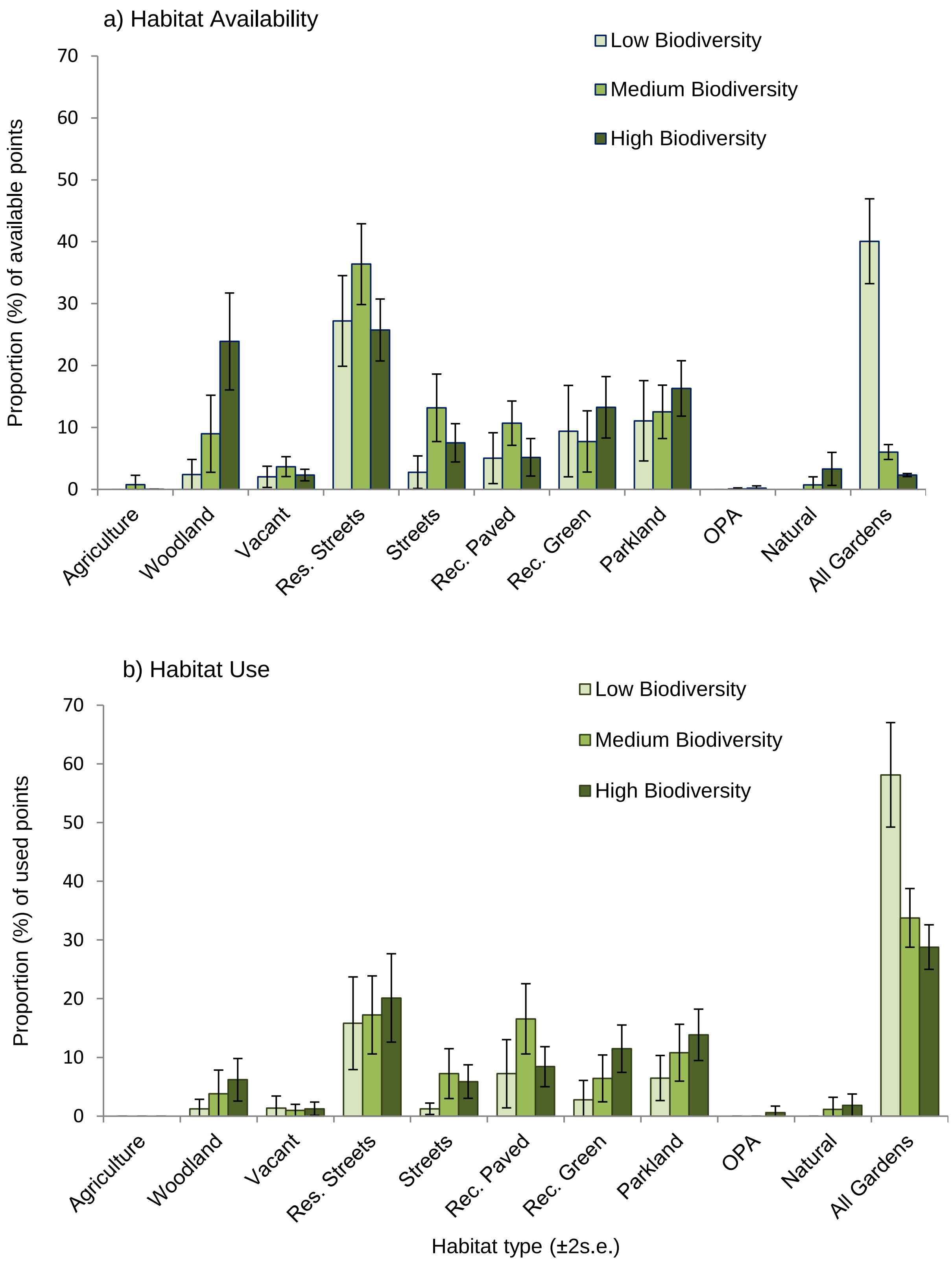


Figure 4.5. Comparison of the average proportion points within each habitat type for (a) the randomly generated available points within children's accessible home range, and (b) children's used location points. Children are grouped by the biodiversity score of their home range into high, medium and low groups (Biodiversity scores calculated in chapter 3; and habitat types were defined in chapter 2).

4.3.3 Children's habitat preferences

i) Third Order - Strict

At the third order, I quantified children's habitat selection within their home range using both strict and adapted approaches. In the strict approach, only a quarter (n= 30) of all children had access to the full set of grouped habitats within their home range. Comparison between the different groups of children based on accessibility to different habitats showed a high degree of similarity in their habitat preferences. Across all groups, children most strongly preferred to be close to home, and most strongly avoided parkland, woodland and street habitats (Figure 4.6a). Each group was found to be best explained by the same top model (1b), which composed the full set of habitats except for gardens which was replaced with home (see Table 4.6 for a summary of top models, and Appendix Tables 4A.1 – 4A.3 for candidate model results). However all three top-models performed poorly in explaining the total variance within the data, with marginal R^2 values around 0.1. Note that negative regression coefficients indicate a positive selection response, as it denotes that decreasing distance to a particular habitat is associated with an increased probability of use by children.

ii) Third Order - Adapted

The adapted approach, which was applied to third order selection, identified the same top models (1b) as the strict approach (see Appendix Table 4A.4 for candidate models). A comparison of selection coefficient values to those from the third order strict approach, the adapted approach yielded similar responses, appearing to provide an approximate average of selection of habitats (see Figure 4.7 for a comparison against both third and second orders). The exception was increased avoidance of parkland and streets, and a decrease in the negative selection for woodland, in the adapted versus strict approach. The greater avoidance of parkland and streets may reflect the greater availability of these habitats which the adapted approach takes account of, and therefore its lack of use by exclusion from the home range is more strongly expressed. The more positive selection of woodland could be interpreted as children preferring to be close to, but not within, woodland habitats. The adapted approach did not improve the predictive power of the resource selection models at this order, with marginal R^2 values lower than those for the strict approach, which is likely due to the greater variability in children's responses brought from the larger sample size assessed at the adapted approach (Table 4.6).

As responses from the adapted approach did not deviate significantly from responses from the strict approach for most habitats, I ran further analyses using the adapted approach to model the responses of categorical factors on children's selection preferences. Assessment of selection for each specific habitat type showed children had the strongest positive selection to be close to home, and responded negatively to residential streets and parkland. I found there was significant variation between the two cities, which were earlier found to vary in the availability of habitat types. Children in Auckland showed strong selection to be close to gardens, which tended to comprise a greater portion of their home range in comparison to children in Wellington. For Wellington children, distance to home was found to be the more important predictor (Figure 4.8(a)). Auckland children also showed strong negative selection of residential streets and parkland habitats, which were the second most abundant habitat type in Auckland. Wellington however showed stronger negative selection of streets and woodland, both of which were more available in these neighbourhoods.

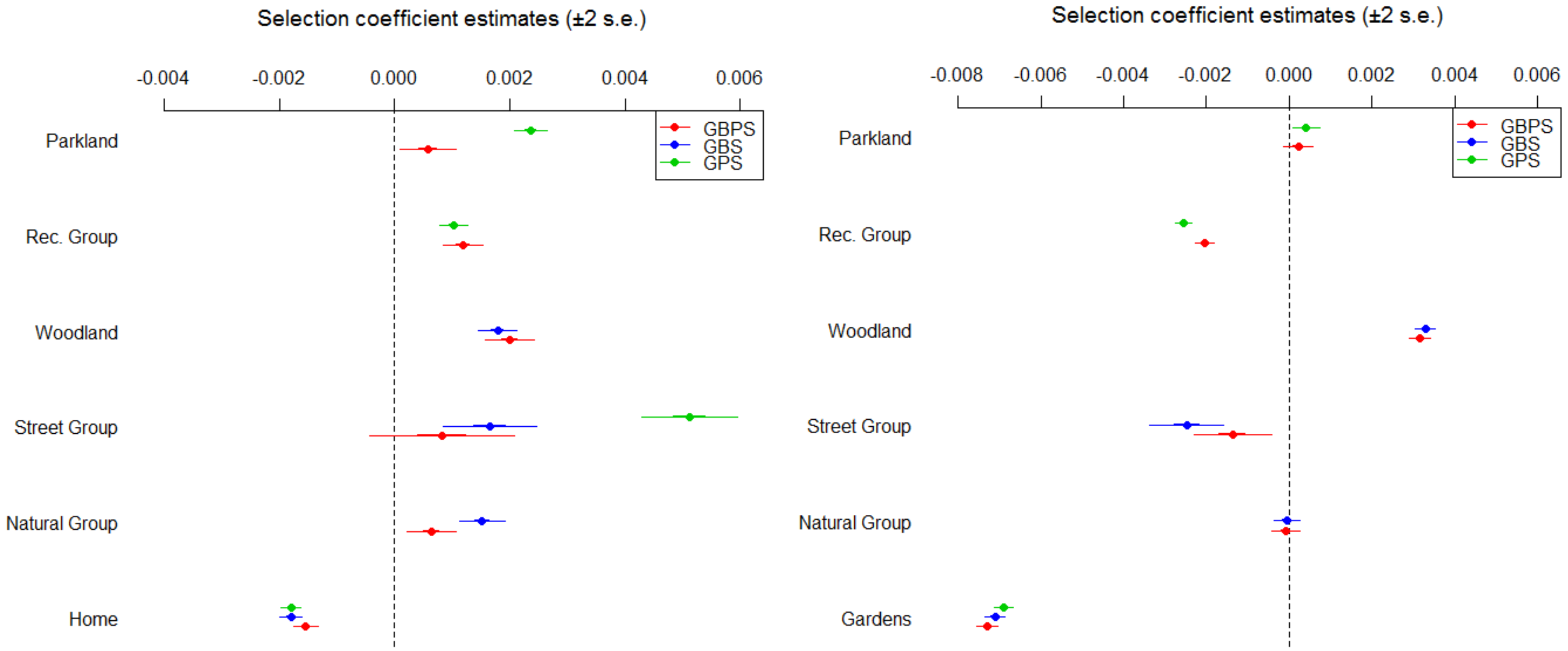
A comparison of genders at the adapted third order (Figure 4.9(a)), reveals that distance to home was found to be a better predictor for girls' habitat selection than distance to gardens, which was for boys the better predictor. Boys showed selection for streets and woodland, while girls showed selection for recreational green. Although there was clear variability between cities and genders, taking this information into account by modelling them separately did not improve the explanatory power of the models, with R^2 values remaining low (Appendix Table 4A.4). Only boys showed an increase over the full adapted model, with a higher marginal R^2 value indicating that boys had more similar selection preferences in comparison to girls.

iii) Second Order - Strict

At the second order of selection assessing children's selection of their home range within the MD buffer, there was significant change in how children responded to accessible habitats. While at the third order selection the top models included selection for home, at the second order this switched to selection for gardens and sports grounds. Table 4.6 provides summary of top models, candidate models provided in Appendix Tables 4A8-4A10. Across the different accessibility groups, children showed very similar responses to environment (Figure 4.6). Woodland was identified was being more strongly avoided than at the third order, likely due to its higher availability at this broader scale of assessment making its comparative lack of use more visible. The predictability of models showed improvement from the third order, with the fixed effects accounting for around half of the variation in the data.

At the second order, I also investigated if there was any variation between the two cities and the two genders, within each of the GBPS, GBS and GPS groups. I found an improvement in predictability only in modelling the Auckland children independently (Appendix Table 4A.11). This indicates more similar selection patterns by children in Auckland compared to Wellington, which is likely due to the fact these children were responding to a shared environment to a greater degree than the more spatially segregated children in Wellington children. Comparison between the two cities showed Wellington children had a much greater preference for sports grounds and streets in comparison to Auckland children (Figure 4.8 (b)). Auckland children show strong avoidance of streets but more positive selection for woodland. This shows that children in Wellington are driving the selection for sports grounds and streets in the combined dataset seen in Figure 4.5.

I also found there was variation in habitat selection between genders at the second order (Figure 4.9 (b)). For most of the accessible groups the same top models were selected as at the third order: boys showed stronger selection for gardens while girls showed greater selection to be close to home (apart from girls within the GBS group). However, in contrast to the third order, where boys showed greater preference for street and recreational habitats, at the second order girls showed much stronger selection for both these habitat groups. However both genders showed increased negative preference for woodland habitat compared to their response at the third order.



a) Third order (strict)

b) Second order (strict)

Figure 4.6. Children's habitat preferences estimated by the top-ranked model's selection coefficients, where negative values indicate preference for that habitat type. Bars represent responses for different datasets where children were grouped based on their accessibility to different habitat types. For example, GBPS refers to children who have access to the 4 habitat groups of Gardens, Biodiverse habitats, Play habitats and Street habitats. Coefficients values (± 2 standard error) are shown for children's (a) third-order selection of habitats within the home range and (b), second-order selection of the home range itself.

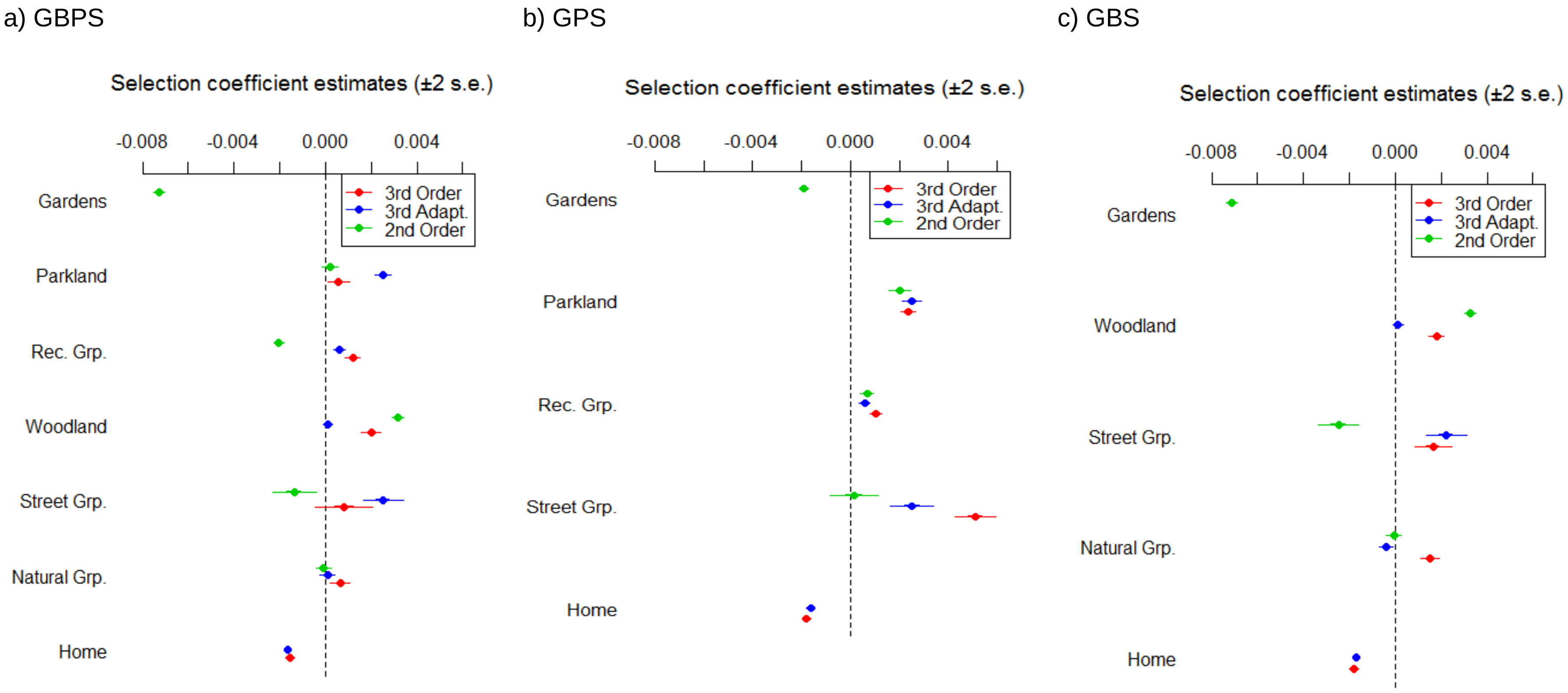
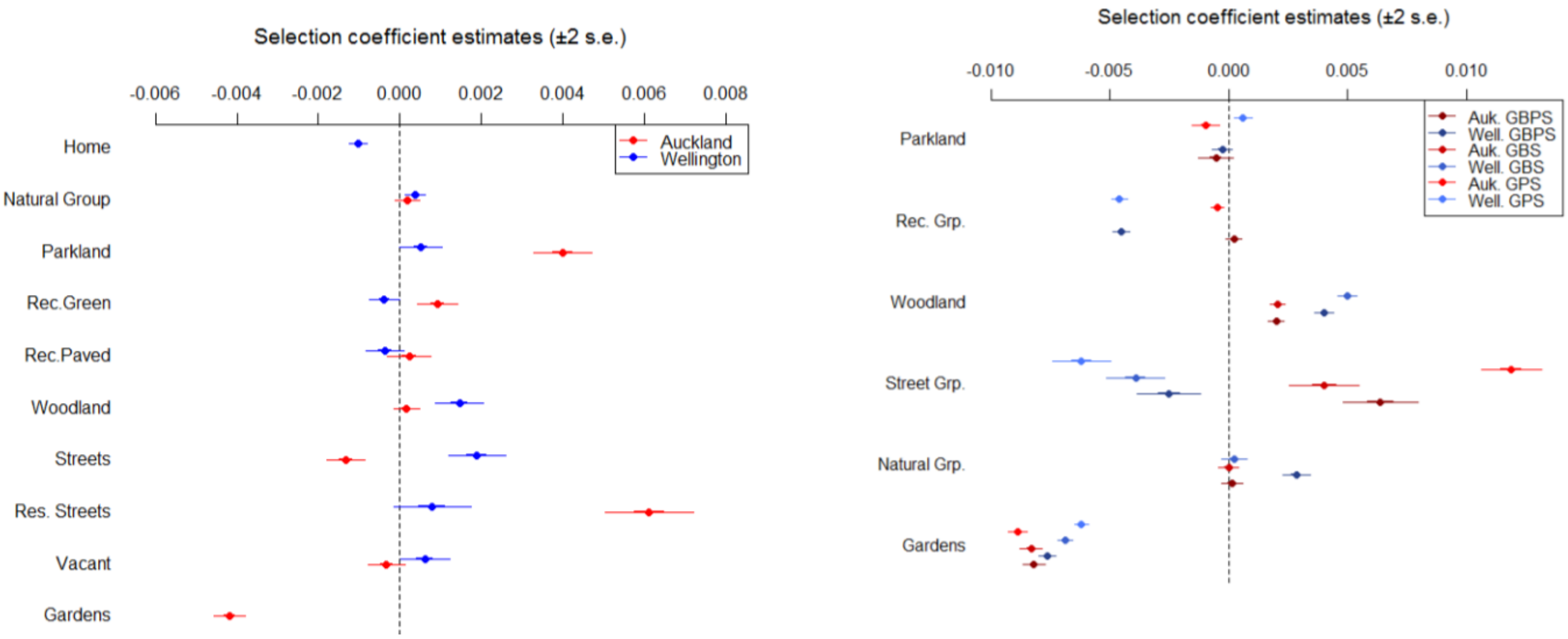


Figure 4.7. Comparison of selection coefficients for the top-ranked models. At the third order (strict and adapted approaches) and second order. Negative coefficients indicate selection for (i.e. to be closer to) a given habitat. Top ranked models for third order was 1b, while at second order was 1a. Orders were compared using the subsets of children who had access to these habitats for the strict third and second order, into a) GBPS, b) GPS, and c) GBS. Sample sizes varied with 3rd order on average containing between 30 and 70 children, the adapted third order allowed comparison of all 118 children, and the second order considered between 80 and 110 children, depending on the sub-group.



a) Third order

b) Second order

Figure 4.8. Comparison between children's selection preferences in Auckland and Wellington at a) the adapted third order (selection of space within the home range), and b) the second order (selection of the home range). The adapted approach allowed distances to be taken to all main habitat types. The second order some habitats were grouped together to prevent a decline in sample size and models were run for groups of children defined by their accessibility to different habitats. Negative coefficients signify children are preferentially using that habitat type in relation to its availability.

a) Third order

b) Second order

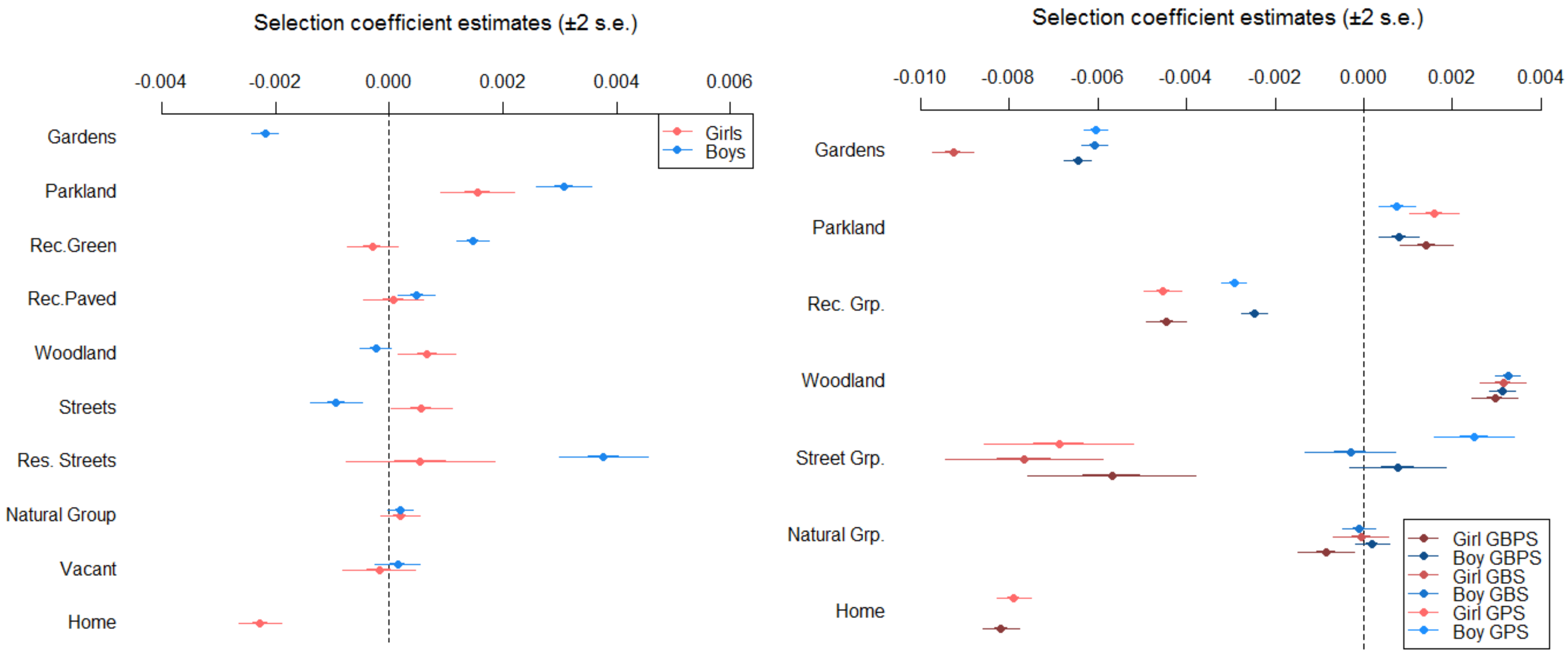


Figure 4.9 Comparison between girls' and boys' selection preferences at a) the adapted third order (selection of space within the home range), and b) the second order (selection of the home range). The adapted approach allowed distances to be taken to all main habitat types. For the second order some habitats were grouped together to prevent a decline in sample size and models were run for groups of children defined by their accessibility to different habitats. Negative coefficients signify children are preferentially using that habitat type in relation to its availability.

Table 4.6. Summary of top-ranked model performance and coefficient values for each overall model of each order, approach and dataset. Blank spaces indicate that habitat was not available to children within that dataset while '-' indicates the habitat is available but not included in the top model. Full lists of candidate models for each of these groups can be found in the appendix.

	Order:	**Third Order**				**Second Order**		
	Approach:	**Strict**			**Adapted**	**Strict**		
	Dataset:	**GBPS**	**GPS**	**GBS**	**Full**	**GBPS**	**GPS**	**GBS**
Model summary	Best model	1b	1b	1b	1b	1a	5a	1a
	Akaike’s weight (*w*)	1	1	1	0.92	1	0.73	1
	Sample size (n)	30	69	40	118	80	100	84
	R^2(m)	0.12	0.11	0.12	0.09	0.52	0.52	0.5
	R^2(c)	0.2	0.2	0.25	0.21	0.73	0.67	0.7
Habitat coefficient estimates (± 1 s.e.)	Gardens	-	-	-	-	-72.9E^{-04} (1.33E^{-04})	-68.9E^{-04} (1.21E^{-04})	-71.1E^{-04} (1.26 E^{-04})
	Natural Group	6.4E^{-04} (2.2E^{-04})		15.2 E^{-04} (2.0 E^{-04})	0.98E^{-04} (1.709E^{-04})	-0.81E^{-04} (1.68E^{-04})		-0.57E^{-04} (1.64 E^{-04})
	Street Group	8.2E^{-04} (6.44E^{-04})	51.3 E^{-04} (4.2 E^{-04})	16.6 E^{-04} (4.1 E^{-04})	25.3E^{-04} (4.507E^{-04})	-13.7E^{-04} (4.82E^{-04})	-	-4.8E^{-04} (4.79 E^{-04})
	Woodland	20.01E^{-04} (2.22E^{-04})		17.88 E^{-04} (1.68 E^{-04})	0.96E^{-04} (1.087E^{-04})	31.5E^{-04} (1.30E^{-04})		32.8E^{-04} (1.26 E^{-04})
	Sports grounds	0.87 E^{-04} (1.76E^{-04})	10.3 E^{-04} (1.2E^{-04})		6.26E^{-04} (1.29E^{-04})	-20.3E^{-04} (1.13E^{-04})	-25.5E^{-04} (1.08E^{-04})	
	Parkland	5.8 E^{-04} (2.46E^{-04})	23.7E^{-04} (1.5E^{-04})		25.1E^{-04} (1.92E^{-04})	2.20E^{-04} (1.79E^{-04})	4.13E^{-04} (1.72E^{-04})	
	Home	-15.4E^{-04} (1.06E^{-04})	-18.0E^{-04} (0.9E^{-04})	-18.0 E^{-04} (1.0 E^{-04})	-16.5E^{-04} (0.79E^{-05})	-	-	-
Candidate models in Appendix Table:		4A.1	4A.2	4A.3	4A.4	4A.8	4A.9	4A.10

4.4 Discussion

In this chapter I have compared three different measures to quantify how much biodiversity children are using in their daily routines and to determine whether they exhibit a biophilic response to their environment. At first, the comparison of children's use of biodiversity suggested that children were behaving in accord with the biophilia hypothesis. In terms of the amount of biodiversity encountered by children, I found the majority showed a greater proportional use of biodiversity in relation to the amount accessible to them in their home range. Additionally the existence of biophilia in children was suggested by the trend of children with more biodiverse home ranges showing greater use of biodiverse habitats. This trend would be expected if greater exposure to biodiversity helps create connections and foster a biophilic response (Benkowitz and Kohler, 2010; Samborski, 2010). However, even children with greater amounts of biodiverse habitats available to them, spent little time in the more natural habitats, such as woodland. In contrast children appear to spend most of their time, excluding the garden, in low biodiversity habitats, such as streets and paved sports courts. The high use of biodiversity in relation to their home range is likely due to the large amount of time spent in gardens, which are relatively biodiverse compared to the rest of the home range area. As such this suggests that, despite children stating natural spaces are they favourite (Moore, 1986; Korpela, 2002), in reality they spend little time in these areas.

The results from children's use of biodiversity and habitats provided mixed results in terms of whether they were expressing biophilia in their choice of habitats. The application of resource selection functions allowed children's use of habitat to be put in context with its availability, thereby providing a measure of children's preference of urban habitats. The resource selection analysis, across the different scales and different levels of habitat accessibility, were largely in agreement and suggest that children do not behave as expected under the biophilia hypothesis. Instead, the strongest preference that children demonstrated was to be close to home, or to their own and friend's gardens. In most cases, children at best showed no positive or negative response to biodiverse habitats, but many children showed avoidance of biodiverse habitats. In particular, woodland, despite being the highest ranked habitat in terms of biodiversity, was often found to be the habitat most avoided by children.

Deprivation level proved to have little effect on the availability of habitats within children's home range area, which is expected as schools were selected to be in areas with similar amounts of biodiversity. Deprivation still had some effect of the use of habitats. Children in lower deprivation neighbourhoods showed higher use of residential streets, which may due to these areas perceived as safer for children to roam on their own (Weir et al., 2006). In contrast, medium and high deprivation children showed greater use for recreational paved and high deprivation also showed high use of parkland, in comparison to low deprivation children. As deprivation did not significantly affect home range size (chapter 3), it appears higher deprivation children are not limited by range, but instead do not spend time on the street. Alternatively, it may be that low deprivation children do not spend as proportionally as much time in recreational paved and parkland habitats. As these areas are accessible, their lack of use could be due to restrictions on children's free time, as opposed to physical restrictions on movement (Hofferth, 2009). Children from higher socioeconomic classes have been found to typically engage in more organised activities (Sener et al., 2008), which could limit the available time to make use of nearby green spaces.

Children in both Auckland and Wellington showed similar responses to biodiverse habitats, although they did show more varied behaviour in terms of the less biodiverse habitats. Differences in both children's use and preference for different habitats appeared to be linked to varying availability between the two cities. Children in Wellington generally had a greater accessibility to woodland, non-residential streets and recreational paved habitats within their home range boundaries in comparison to Auckland. In contrast, children in Auckland had a greater accessibility to gardens, parkland and recreational green. Typically the greater availability of habitats encouraged greater use by children, yet the resource selection analysis indicated these habitats were avoided. For example, children in Wellington had proportionally more access and time spent in woodland than Auckland children, yet at both orders of selection woodland was avoided by Wellington children more than Auckland. This is likely because the higher availability of these habitats causes the comparative lack of use to become more significant. In other cases though, the difference in selection could be due to differences in the habitats between cities. One of the main divergences in children's responses between the cities was in their preference for streets, with Auckland children responding negatively while Wellington children responded favourably. This difference is likely due to the fact that much of the street habitat in Wellington was the city centre, while in Auckland this constituted a motorway and industrial warehouses, which are unlikely to be spaces which encourage use by children (Rehrer et al., 2011).

Consistent differences were also present in the habitat selection patterns between boys and girls. Typically, distance to home was a better predictor of girls' habitat use than distance to gardens, which may represent greater influence of parental safety concerns leading them to remain close to their home (Veitch et al., 2008; Spilsbury, 2009). Unexpectedly, girls showed stronger preference for recreational habitats, including play fields and courts, in comparison to boys. Boys however, spent proportionally more time in these habitats than girls. The lower preference shown by boys may be due to the fact that boys tended to be more diverse in the habitats they visited, while girls tended to spend the majority of time in either gardens, streets and recreational paved. With more diverse habitat use, boys would have had more access to different habitats, but used each one individually to a lesser degree, therefore resulting in a lower indication of preference.

Habitat preference was also found to alter depending on the order of selection. At the third order, selection of habitats within the home range was much more variable between the children who had different accessibility of habitat groups (GBPS, GBS and GPS). At the third order, no specific habitat type proved to be strongly affecting children habitat use, with children only showing a selection for locations closer to home. In contrast, selection of home range within children's accessible area (the MD buffer) showed much clearer patterns of selection between the three accessibility groups. The more distinct patterns of children's home range selection, as opposed to habitats within the home range, resulted in greater predictability at the second order in comparison to the third. This suggests that children in other New Zealand neighbourhoods and cities may show similar behaviour in terms of home range selection. However, it is evident that the availability of habitats affects selection at this order, with Auckland having greater explanatory power than Wellington, likely due to the greater overlap in children's neighbourhoods in comparison to Wellington.

At the second order, children exhibited strong preference for gardens, but also showed preference for street and recreational habitats. The switch from a stronger selection for gardens at the second order, in comparison to home at the third order, reflects the fact that children's home ranges were not centred around the child's home. Instead home ranges protruded out in a specific direction from the child's home (Villanueva et al., 2012), often making the home one of the corners of the home range polygon. The selection for gardens (their own and friend's) and recreational habitats at the second order, suggests that it is the presence of these habitats that children extend their home ranges to include. These habitats may therefore be the most important for children to have access to within their home range, even though their everyday use is not high in comparison to other habitats as indicated by a lack of selection at the third order. The strong selection for sports grounds is likely driven by these areas being places where children can go to play, a central motivation in children's use of outdoor space (Korpela et al., 2002; Min and Lee, 2006; Kalvaitis and Monhardt, 2012).

The strong preference to be close to gardens and home supports the theory that declining children's mobility is negatively altering children's use and connection to biodiversity. However, it does not appear in this case that the cause of the decline in home range size is due to parental restrictions. Habitat availability for the strict approaches considered only habitats that children had identified as being accessible to them. Therefore the measures of children's habitat preference show that despite biodiverse areas being accessible, they are not preferred habitats. Parents could be influencing their children's use of space in others ways instead of spatial boundaries. Increasing safety concerns can transfer to children, making them afraid to travel far and visit new places on their own (O'Brien et al., 2000). Additionally, replacement of free-time with organised activities may mean that children simply do not have time to explore the wilder areas of their neighbourhood (Hofferth, 2009). The increase in structured activities would also explain the high selection for sports grounds where these activities take place.

While parental restrictions on children’s movements may be affecting the accessibility of nearby nature, as evidenced by some children reporting they were not allowed to leave their garden unaccompanied, it is clear that children’s home ranges are declining in size and this is reducing the amount of accessible biodiversity in their home range. Only a quarter of children in this study had access to the four different habitat groups; biodiverse, play, streets garden habitats, within their home range. Here I found that children with less biodiverse home ranges, which were linked to home range size in Chapter 3, tended to spend more time in gardens. Gardens were the most preferred habitat type and represent a valuable combination of both being safe and close to home, encouraging play as well as generally containing high levels of biodiversity. As such, if children’s home range sizes continue to shrink, gardens may be the main source of biodiversity that children interact with on a daily basis. While gardens can be biodiverse, they are often highly artificial and managed habitats, and do not support the same engaging play and connection to nature afforded by more natural sites (Pyle, 2002).

Chapter 5: General Discussion

In this study I have applied a variety of methods to assess the availability, accessibility, use and preference of biodiversity by children in urban neighbourhoods. This information has implications for our understanding of how children use the urban realm and whether they are showing a biophilia response by preferentially using biodiverse habitats accessible to them. In this chapter, I summarise the main conclusions of this work and place these in the context of current discussions of urban design paradigms and the changing nature of children's outdoor activities. I will review the main techniques I have applied and possible avenues for future research.

5.1 Review of findings

5.1.1 Biodiversity in urban habitats

In Chapter 2 I defined biodiversity as a combination of species richness, structural diversity and wildness components. I selected and measured these specific aspects of biodiversity to produce an estimate of biodiversity which would reflect how children may perceive and value biodiversity. Surveys of biodiversity across the thirteen habitats types showed biodiversity could be predicted by both habitat type and the feature richness of that site. Feature richness is the number of features (out of a possible ten) that a single habitat site held, and provides a measure of structural diversity across that site. Greater structural diversity of habitats has been linked to higher levels of biodiversity and has also been used as an indicator of biodiversity itself (Hercock, 1997; Young and Jarvis, 2001; Cornelis and Hermy, 2004; Tzoulas and James, 2010).

Overall, the most biodiverse urban habitat was woodland, which had high scores in all three aspects of biodiversity; species richness, structural complexity and wildness. The least valuable habitats in terms of biodiversity values were streets and open public areas (OPAs), which, when they contained natural features, were usually highly landscaped with low species richness and proportions of green cover. Gardens proved to be highly biodiverse habitats, especially garden type 1, which was defined as containing a large proportion of mature vegetation, and consequently scored highly for amount of green cover and structural diversity. In some cases the highly managed landscapes had higher scores than natural areas, such as beaches and grasslands. Yet these natural areas were one of the few habitats, including woodland and vacant land, which scored highly in terms of wildness.

I used these BioScores to predict the biodiversity present across children's neighbourhoods by assessing the biodiversity present within a 500m radius buffer around the child's home. Even though these schools were selected to fall within neighbourhoods containing similar levels of biodiverse habitats nearby, the level of deprivation still negatively affected biodiversity within this area. However across neighbourhoods there was a high availability of parks, sport fields and woodland within the buffer, and in total accessible green space occupied over a quarter of the total buffer area. As the availability of accessible green spaces was similar across the socioeconomic gradient, it appears the deprivation is affecting the quality of biodiversity in private gardens, which are included in the assessment of biodiversity within the neighbourhood. Private gardens were not included in the estimation of biodiversity within children's accessible home ranges, and at this scale deprivation did not have a significant effect on accessible biodiversity. At the neighbourhood scale, typically lower deprivation areas had a greater proportion of the more biodiverse garden type 1, while higher deprivation areas tended to have smaller gardens with a greater amount of impervious surfaces (garden type 3). Children in lower socioeconomic neighbourhoods therefore have equal access to public green spaces, but less biodiversity immediately available to interact with in their own gardens. This is significant as gardens were found to be the most used habitat in the urban environment, and as such, children from different backgrounds may be encountering different levels of biodiversity despite showing similar habitat preferences.

5.1.2 Children's home ranges

A key concern in children's geography has been the decline in children's home range size over the past few decades (O'Brien et al., 2000). Here I recorded smaller home range areas than have been recently reported (Spilsbury et al., 2009; Villanueva et al., 2012). This is likely due to the fact that home ranges were defined as children's most frequently used area, as opposed to including rare long distance movements, and I also only included the area of accessible habitats within the home range boundary. Children's median home range size was found to be less than three hectares and the median maximum distance travelled from home was 498m. Boys tended to have larger home ranges than girls and also travelled further from home. This indicates that the gender gap still exists in children's home range size, likely due to continued greater restrictions on movement for girls in comparison to boys (O'Brien et al., 2000). Larger home ranges are important in supporting children's connection to nature, as larger home ranges enclosed a greater total amount of biodiversity and a greater diversity of habitat types.

5.1.3 Children's use and preference for urban habitats

Children spent the greatest proportion of their time in gardens (40%), followed by residential streets (18%), recreational paved (11%) and parks (10%). The least used habitats were OPA, vacant land and agricultural habitats (all less than 1%). Children were found to use proportionally more biodiversity in relation to the amount available within their home range which is likely due to high use of gardens. By comparing children's use of habitats in relation to their accessibility using resource selection analyses, I found children most preferred to be either close to home or close to their own or other accessible gardens. At the level of selection of habitats within the home range, children did not show strong selection for any habitat outside of gardens, but did display strong negative responses towards streets, parkland and recreational green. In contrast, at the second order of selection, children showed much more predictable patterns in the selection of their home range, displaying preference for garden, street and recreational habitats (recreational green and paved). This change in preference from the third to second order indicates that children base their home range area around these key habitats of gardens, streets and recreational habitats. However, while these areas are included in their home range, they are not used to a great extent, and instead children spend the majority of their time in the garden, resulting in the strong preference to be close to home in the third order analysis.

Children's preferences for different habitats varied between gender and between the two cities. Auckland and Wellington had different availabilities of habitats, and typically the greater availability of habitats supported greater use by children. However, the resource selection analysis showed that overall girls showed stronger preference to be close to home than boys, which corresponds to the smaller home range sizes recorded. Boys showed more diverse use of habitats than girls who tended to spend most of their time in gardens or on residential streets. Boys showed less strong selection, either for or against habitats, in comparison to girls. Instead boys tended to visit a greater diversity of habitats, but did not show particular favouritism for one particular type, which meant that the resource selection analysis often estimated this as negative selection (as availability of the habitats was high in the home range but its use was low). This indicates the importance of considering the selection of habitats in relation to their use. Individually both metrics make identification of which habitats are important to children difficult, as high use could just be due to high availability, while high selection could just be due to low availability. As such it is important to qualify one against the other to make inferences on children's habitat use patterns.

5.2 Conclusions on Children, Urban Biodiversity and Biophilia

In chapter 1, I introduced three hypotheses of possible reasons for the decline in nature connection in urban children; i) lack of biodiversity within urban areas, ii) lack of accessibility of available biodiverse areas, and iii) lack of a biophilic response in children, as evidenced by children showing a preference to be in natural areas. In this section I will review the evidence gathered across the past three chapters for each hypothesis.

5.2.1 Availability of biodiversity

Urbanisation has been associated with low levels of biodiversity through the removal of native green space and replacement with built structures largely inhospitable for wildlife (Aronson et al., 2014). Cities are also highly managed and artificial environments, with few native species and instead domination by common generalist species (McKinney, 2006; Sol et al., 2014). As the human population becomes increasingly urban, a decline in connection to nature has been reported (Nabhan and St. Antoine, 1993; Louv, 2008), and these two trends are thought to be linked (McKinney, 2005). I assessed the amount of biodiversity available to children by characterizing a buffer around the child's home to measure nearby nature to each child. Within this area I found all children had access to some form of green space, in addition to their gardens, within the buffer limits. Therefore all children, had access to a green space, although the relative quality of these sites could vary. Additionally, the availability of the more natural spaces was lower in comparison to the more managed green spaces. However, I found that on average over a quarter of children's buffers were composed of green space, indicating that in these urban neighbourhoods a high availability of biodiverse habitats within the urban environment could be found. As such, within the urban environment biodiverse habitats are present and available for children to exploit to forge a connection to nature.

5.2.2 Accessibility of biodiversity

The majority of interviewed children indicated they had limits set on how far they could go from home. This could mean that despite the presence of biodiverse habitats within the urban domain, children may not have the freedom to visit it on their own. In this study I found smaller home range areas than previously reported (Spilsbury, 2009; Villanueva et al., 2012), and these smaller home ranges contained less total biodiversity and fewer habitat types. Shrinking of home range size could be preventing children from accessing the most biodiverse areas in their neighbourhood. Indeed, a small number of children indicated they were not allowed outside of their garden without supervision. However, even those children who had access to biodiverse habitats within their accessible area (MD Buffer; 72% of the sample) showed no preference for the more biodiverse habitats accessible to them. At this second order children showed preference for private gardens, streets and sports grounds, but not the wilder and more natural sites such as woodland. This indicates that accessibility, for the majority of children, is not restricting their use of biodiverse habitats. Instead children are choosing to spend time in low-to-medium areas of biodiversity, which tended to be those which support structured play activities.

However, in addition to physical boundaries creating an area in which they are allowed to roam, children's use of the urban environment could also be constrained temporally. The amount of children's free time has been found to be in decline, providing less free-time for children to venture from their home and explore wilder areas of their neighbourhood. On average children aged 9-12 in

2003 spent around twenty minutes a day outdoors after school, which has declined from forty minutes since 1997 (Hofferth, 2009). Instead children's days are increasingly filled with sedentary indoor activities, and any outdoor time is spent in structured sports activities (Hofferth, 2009). This would explain children's preference to be close to home as well as sports fields where these organized activities take place (Hofferth and Sandberg, 2001). While in lower socio-economic areas the availability of biodiversity may be lower, children from wealthier backgrounds tend to have busier schedules (Sener et al., 2008), and so these children as well may be limited in opportunities to connect to nature.

5.2.3 Biophilic response in children

The evidence indicates that these children, who had both available and accessible biodiversity present, did not seek out and preferentially use these biodiverse habitats accessible to them. Both children's habitat use and resource selection analyses suggest low preference for these habitats. Additionally, there was only a small increase in the use of habitats with increasing availability in the home range. Figure 5.1. summarises this information; it compares the proportion of habitats accessible within the area available to children – the neighbourhood buffer, the area accessible to them – their home range, and what they use – the used location points. The figure shows that, for children who have access to these habitats only show greater selection in comparison to their availability for gardens and recreational paved, as identified by the selection at the second order. The graph illustrates three main responses children had to different habitat types. Habitats which show a continual decline from buffer to use indicate habitats of least preference by children, namely woodland and streets. Habitats which show a peak within the home range indicate these habitats are often visited, but not used intensively. Those habitats which show an increase from buffer to use identify those habitats most important to children; their garden and recreational paved. This largely mirrors the selection preferences identified for children in Chapter 4, which additionally showed selection of streets: this likely reflects a preference to be close to residential streets.

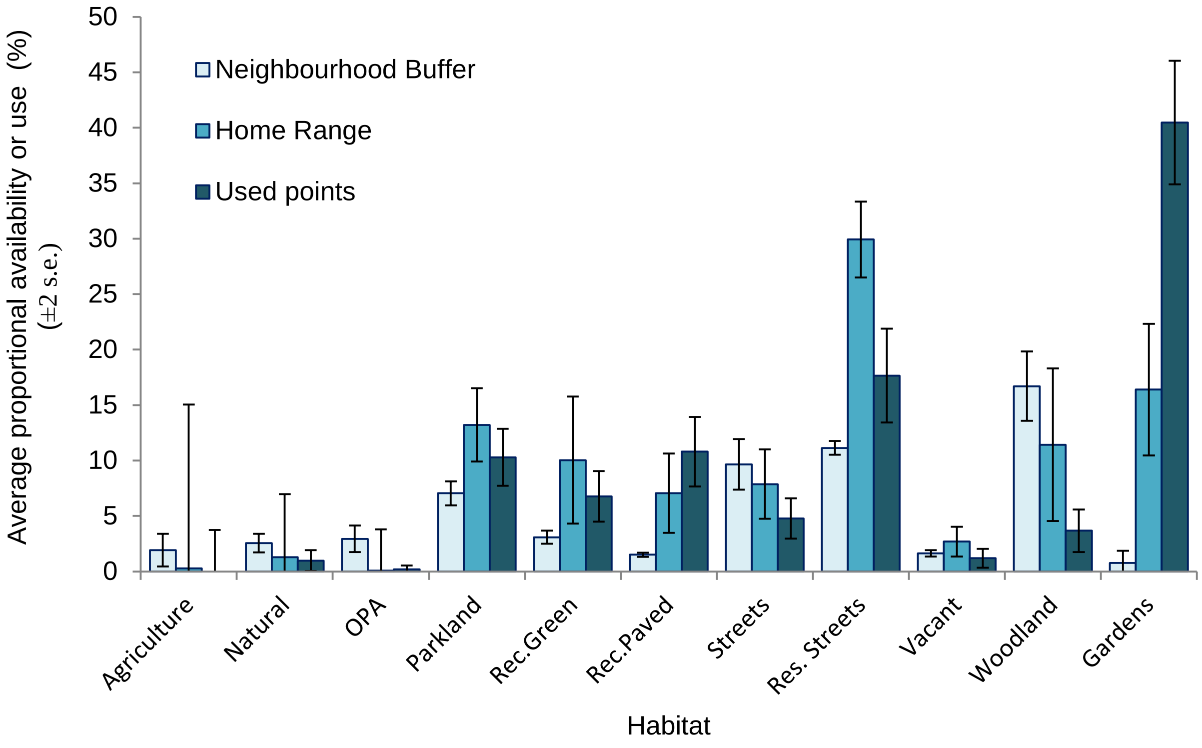


Figure 5.1. Proportional availability of and use of habitats measures as the percentage composition of buffer and accessible home range areas, and the percentage of dots placed in each habitat. Note, the average values for home range area and proportion of used dots only account for the children with access to those habitats. 67% of children had access to all habitats within their accessible area (MD Buffer).

Instead of biophilia, children may be more motivated by its antithesis; videophilia (Pergams and Zaradic, 2006). Videophilia is the attraction of modern electronic media which has been suggested to be distracting children from the less immediate rewards of spending time in nature (Pergams and Zaradic, 2006). Videophilia is associated with greater proportions of time spent indoors and reduction in physically active outdoor activities. This would explain the strong preference to be close to home identified here.

This finding therefore demonstrates that these urban children are not exhibiting a biophilic response to their environment, and that the simple presence of nearby nature is not enough to encourage its use by children. Biophilia is a trait the expression of which must be triggered by spending time in biodiverse areas and fostering a connection to nature (Nabhan and St Antoine, 1993; Orr, 1994). The prevention of an innate biophilia response in today's generation of children could be being facilitated by the urban environment. If urbanization is associated with declining home range sizes, this means that children are never afforded the opportunities to connect with nature and therefore never develop an appreciation for it. Additionally, the biodiversity accessible to children may not be of a form which will encourage their use and interaction with nature. Some urban green areas can be highly managed, and children can be discouraged from playing in natural areas for fears of disturbing the flora and fauna. Children have complained about the sterile nature and 'cookie-cutter' design of many parks designed for their use (Maxey, 1999). Instead, what children may require is greater access, freedom and encouragement to explore unmanaged, wild areas, which are thought to better support children's connection to nature (Pyle 2002; Samborski, 2010).

5.3 Implications for urban design and children's independence

The absence of biophilia in children is concerning as future generations may grow up with a poor knowledge and awareness of the natural world (Pyle 1978; Miller, 2005). Establishing a connection to nature as a child has been found to be an important predictor of conservation ethos in adulthood (Wells and Lekies, 2006) and already adolescents are showing lower levels of environmental concern in comparison to previous generations (Wray-Lake et al., 2010). It is therefore important to support children's connection to nature to foster biophilia, and ensure future generations are environmentally literate and aware of its value for individual, community and environmental well-being (Orr, 1994; Miller, 2005). In the next section I outline some possible strategies for supporting urban children's connection to nature.

5.3.1 Urban Design

Although biodiverse areas were not highly prioritized sites for use by children, their conservation in the urban landscape is critical to maintain the availability of these areas for children. In this study, all children had at least one form of green space available, but the availability of more natural spaces was lower. The availability of green space in the cities I examined was high across the socioeconomic gradient as schools were selected to be similar provisions of green space. However this may not be the same for other children as many studies have found unequal provision of green space across socioeconomic gradients (Whitford et al., 2001; Pauleit et al., 2005). Natural and woodland habitats should be prioritized for conservation as they are valuable in supporting native biodiversity in the surrounding neighbourhood (Soga et al. 2014); as well as providing engaging experiences when children do visit these areas (Pyle, 2002). Additionally new wild areas can be established through the restoration of abandoned urban plots. These restorations should focus less on making sites aesthetically pleasing, which adults may consider to be more important (Gobster et al., 2007), and instead allow spontaneous and unmanaged regeneration, supporting both biodiversity and children's play activities (Pyle 2002; Rink and Herbst, 2011). Restoration of wild areas has been shown to improve community understanding of ecological values and increase the frequency of use of these areas (Ward Thompson et al., 2013).

As well as preserving natural habitats children currently do not spend time in, improving the biodiversity value of habitats children already spend time in could support their connection to nature. Improving the biodiversity and greenness across all urban habitats has been suggested to provide ecosystem services within cities (Bolund and Hunhammar, 1999). If this biodiversity was made accessible to children this would add an additional service of affording children opportunities to connect to nature (Chiesura, 2004). One key area where improving biodiversity could be focused is school grounds. Children's time spent outdoors in their school's paved and green areas represents on the main times children are provided with unstructured time outdoors. Increasing the biodiversity present in school grounds can be coupled with education on biodiversity (Tranter and Malone, 2004). Greening of school grounds should prioritise native species, which tend to be rarer in urban environments and which children know the least about (Ballouard et al., 2011; Dunn and Henneghan, 2011). More biodiverse school yards have been shown to support more complex and enriching play (Fjørtoft and Sageie, 2000), as well as improve children's knowledge of species and preference for more species-rich habitats and native species (Lindemann-Matthies, 2005; Benkowitz and Kohler, 2010; Castonguay and Jutras, 2009; Samborski, 2010). If children garner a connection to nature within the relatively safe school environment, they may become more inclined to explore the biodiverse areas outside of school as well.

Children spent most of their time outside in their gardens, and for many children who had small home ranges gardens provided the greatest source of biodiverse habitat available to them. With continued decline in children's home range size gardens represent a last line of defence in connection to nature. In terms of children's values, gardens provide suitable areas for connection to nature, as they are safe, support play and can be structurally diverse and species-rich. Gardens can provide ecosystem services to individual's houses and the nearby neighbourhood (Cameron et al., 2012), as well as support connections to nature in adults and the children who visit them (Freeman et al., 2012). However gardens are becoming a rare commodity. As cities grow, pressure on space is leading to greater infill of garden areas with built structures and impervious surfaces (Pauleit et al. 2005; Perry and Nawaz, 2008). The value of gardens in terms of providing opportunities for children to connect to nature has important implications for the current debate on how future city growth should be managed (Lin and Fuller, 2013). Currently, as a response to the negative impacts from the past trend of urban sprawl, new city designs tend to follow a more compact city form (Neuman, 2005). This often means the loss of gardens in favour of more dense housing and provision of public green space instead, which could confer overall benefits to urban biodiversity (Sushinsky et al., 2013). However, if the quality and quantity of public green space provided is not high, and if children's mobility continues to be restricted, particularly as has been reported for children living in high-density housing (O'Brien et al., 2000) that future high-density urban growth could further reduce children's connection to nature.

5.3.2 Children's independence

It is clear that parental influences are playing a part in how children use their neighbourhood's public space (Hofferth and Sandberg, 2001). Parents could support children's connection to nature by supporting greater use of the neighbourhood where safe, and allowing children more unrestricted free time to explore their neighbourhood. Children have been found to engage in more enriching play when not under supervision (Castonguay and Jutras, 2009). Parents have begun increasingly limiting which activities children can take part in when in natural environments. For instance, one UK survey reported half of children were not allowed to climb a tree without adult supervision (Playday Survey, 2008), removing the opportunities for children to physically interact with nature. Encouraging children to walk to school, instead of being driven, would also increase children's time spent outdoors after school (Wen et al., 2009). Additionally, a lack of biophilia in children may be due to lack of encouragement by their parents, who are children's main source of information on biodiversity outside of school (Benkowitz and Kohler, 2010).

5.4 Review of methods and future research

5.4.1 Biodiversity Scores

In this study I created a BioScore system which aimed to measure aspects of biodiversity that would be important for children. This included surveys for 'detectable' biodiversity using morphospecies, which provide an indicator of species richness but often underestimates true richness than more intensive methods (Oliver and Beattie, 1996). This generalisation was useful here as it more likely captured how children would view biodiversity, who may fail to notice many cryptic species. I also considered biodiversity in terms of the structural richness within sites which has could be a surrogate indicator of biodiversity to humans (Barbosa et al., 2007) and also has been linked to more engaging play habitats for children (Fjørtoft and Sageie, 2000, Samborski 2010). The resulting biodiversity scores ranked urban habitats as would be expected, but the differences between habitats were relatively small.

This is in part due to the use of morphospecies to measure species richness, which reduced the variation between urban habitats by only identifying the most visible species, and suggests that children may not recognise the variation in richness between natural and more urban habitats. However, part of this may also be due to the survey method I applied. Within each surveyed site, samples of habitat features were averaged to create that habitat's biodiversity score, thereby providing an estimate of the average amount of biodiversity within that habitat in any given 5m^2 radius sample plot. What this method lacked was an account of the relative proportions of the different features in each habitat (such as applied in Tzoulas and James, 2010). Taking the proportional contribution of features to a habitat site would provide a more accurate estimate of the average level of biodiversity within a given 5m^2 plot in that site. This could be achieved by weighting feature's BioScores by their proportional area.

5.4.2 Application of wildlife habitat use methods to children

Studies of animal habitat use assume animals use their environment in a way that maximizes their fitness (Boyce and McDonald, 1999). This concept can be extended to children who, instead of venturing outdoors in search of food and shelter, are motivated to venture outdoors by errands, structured activities, and to meet and play with friends. Unlike tracking studies of wildlife however, I could make use of information not normally available from children's interviews, and gather information more quickly representing movements over a greater time-scale than possible using wildlife methods which rely on tracking devices. Of most significance was information obtained from children on the accessibility of habitats, which drastically reduced the amount of area and biodiversity actually accessible to children.

The main drawback of this method of assessing children's spatial use based on interviews was its subjectivity. It relied on accurate responses by children on how they perceived where they spent the most time outdoors. Children have been shown to overestimate their home range size in comparison to that reported by their parents (Spilsbury, 2005). During interviews children could be over-estimating the time they spend in different habitat, although similar proportions of time spent in different habitats were found in previous research (Cunningham et al., 1994). Children's subjectivity in where they perceive they spend the most time however may result in them also placing dots in areas they feel are important. Further research could additionally use more temporally explicit approaches, such as GPS data to get an accurate picture of how accurate children's perceptions are of their time outdoors, and how much time they tend to spend in different habitats (Fjørtoft et al., 2009; Oliver et al., 2014). These two methods, interview dot-mapping and tracking methods combined would allow both temporal and spatial specificity, yet reflect children's habitat selection at both a short and long-timescales.

The application of wildlife methods at times provided unrealistic measures of children's home range size. Minimum convex polygons are a commonly used estimator of home range size, but can be vulnerable to creating unrealistic forms. For some children it was clear the home range was excluding areas children used, such as streets they must take to reach a particular used destination. An improved method would be a network approach (Villanueva et al., 2012), which uses a network map of a neighbourhood's streets to include the areas children must travel to access different portions of their home range.

In this study I have assessed children's preference of habitats using resource selection analysis, and evaluated evidence for biophilia through the relative selection of habitats of different biodiversity content. This could be expanded to produce a more complete picture of what drives children's habitat use by incorporating the children's values of outdoor place with different habitats. For instance, habitats could be attributed values for how safe they feel, how good they are for play and exploration, and biodiversity could be included as just one of these possible motivations for use of space. This would also allow assessment of how biodiversity varies in relation to these other motivations. Here, distance to different urban habitats explained around half of the variation in children's selection of their home range. Remaining variance in a child's spatial use is likely dependent on both their own personal and their families' preferences for outdoor activities, social connections and views of neighbourhood safety (Spilsbury, 2005; Veitch et al., 2007; Hofferth, 2009). Therefore, inclusion of these motivations would better account for individual variation in why children use space, and better identify those spaces that are most important. In this way we could identify which urban green spaces contain high values for children and biodiversity.

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Appendix

Table 2A.1 Description of the 13 urban habitat types with mean BioScore, mean feature richness (number of structural features within each site) and total sample size.

Habitat	Description	Mean BioScore (s.e)	Mean feature richness (s.e.)	Sample size
Agriculture 	Horticultural or pastoral land-use area	15.19 (0.46)	4.4 (0.37)	10
Garden 1 	Garden rich areas (>1/3 of lot size is garden) rich in tree and scrub vegetation elements	17.64 (0.53)	6.7 (0.29)	10
Garden 2 	Garden rich areas (<1/3 of lot size is garden) poor in tree and scrub vegetation	15.17 (0.57)	6.55 (0.16)	11
Garden 3 	Garden poor areas (<1/3 of lot size as garden), poor in tree and scrub vegetation	12.59 (1.01)	5.14 (0.51)	7

Natural 	Non-woodland area largely retained its native state, e.g. beaches	12.00 (0.71)	5.58 (0.38)	12
Open Public Area (OPA) 	Largely paved area open for public use	6.90 (0.48)	4.67 (0.28)	12
Parkland 	Area dominated by lawns with flowerbeds and planted trees.	11.02 (0.45)	6.31 (0.36)	13
Recreational Green 	Area of managed grassland designed for sports activities	11.11 (0.46)	5 (0.51)	13
Recreational Paved 	Paved area designed for sports or play, e.g. school playgrounds, tennis courts	7.95 (0.83)	3.17 (0.57)	12

Street 	A street or road in a non-residential area and associated landscaped vegetation	8.48 (0.76)	3.42 (0.45)	12
Residential Streets 	Street in a residential area and associated landscaped vegetation	9.11 (0.65)	2.46 (0.35)	13
Vacant lots and fringe vegetation 	Dominantly vegetated areas with unmanaged or un-landscaped vegetation, such as abandoned sites and road verges	14.98 (0.58)	3.46 (0.37)	14
Woodland 	Area dominated by groups of trees, including native bush remnants to forests for logging	18.31 (0.71)	4.5 (0.29)	12

Table 2A.2. Description of the ten features which add structural diversity to habitat sites. Average BioScore is calculated for that feature's immediate 1m^2 area, while mean contribution value represents how much the 1m feature contributes to the mean BioScore of the habitat site it is within. Average BioScore and contribution value are shown ± 2 standard error.

Feature	**Definition**	**Mean BioScore (±s.e.)**	**Mean contribution value (±s.e.)**	**Sample size**
Bare ground	Area of naturally bare-ground such as beaches or rocks but not trodden ground	7.39 (0.59)	0.56 (0.04)	6
Fence	Permanent fence or wall over 50cm in height	7.29 (0.38)	0.59 (0.03)	108
Grass	Grasses under 1m in height	9.18 (0.18)	0.73 (0.02)	115
Low vegetation	Herbaceous and woody plants under 1m in height.	9.85 (0.45)	0.76 (0.01)	81
Paved	Area covered with impervious or other artificial surface like gravel	4.41 (0.92)	0.38 (0.01)	116
Tree Group	Group of more than 3 trees with connected canopies	19.66 (0.92)	1.28 (0.06)	60
Tree Line	More than 3 trees in a line with connected canopies	19.46 (0.98)	1.45 (0.07)	53
Single Tree	3 or less trees with or without connected canopies	19.52 (0.71)	1.59 (0.06)	81
Tall Vegetation	Herbaceous, grass or woody (non-trees) vegetation over 1m in height	17.15 (0.67)	1.12 (0.04)	49
Water	Permanent natural or artificial body of water	12.01 (1.22)	0.77 (0.06)	30

Table 2A.3. Definitions of plant growth form used as a measure of habitat structural complexity (d.b.h = trunk diameter at breast height).

Plant Growth Form	Description
Large Tree	Woody plant with <3 main trunks > 20cm d.b.h.
Small Tree	Woody plant with <3 main trunks < 20cm d.b.h.
Tall Shrub	Woody plant with >3 main trunks >1m tall
Small Shrub	Woody plant with >3 main trunks <1m tall
Tall Grasses and Ferns	Grasses over 1m in height, ferns and flax plants
Herbaceous plants	Green-stemmed plants excluding grasses
Grasses	Grass and grass-like species <1m in height
Lichens and Mosses	Lichens, mosses and clubmosses
Aquatic	Freshwater and marine vegetation
Climbers	Woody or green stemmed plants which have grown on other structures

Figure 3A.1 Home ranges for children, estimated using Minimum Convex Polygons (MCP) of three schools in Wellington.

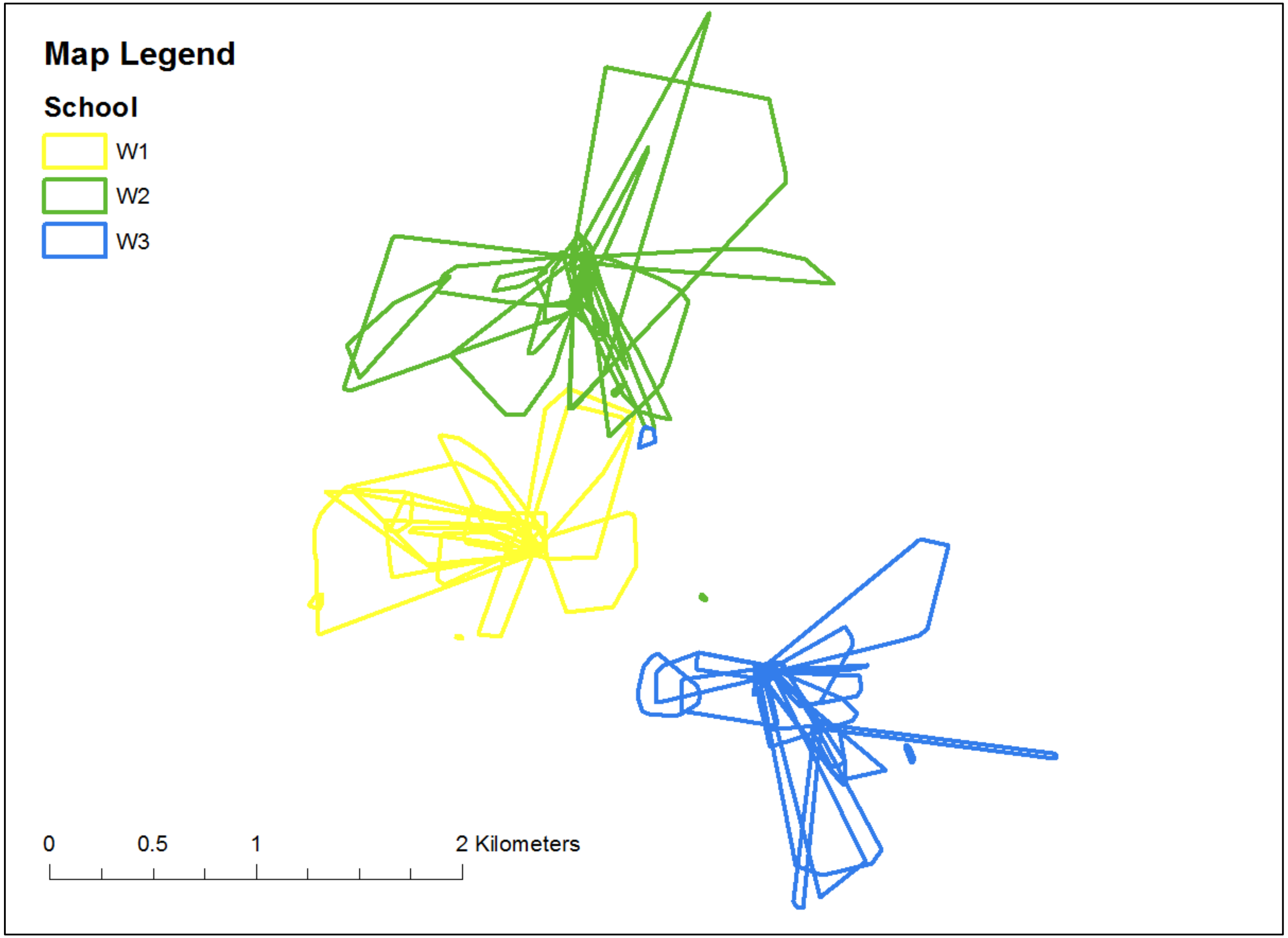

Figure 3A.2. Map of children's buffer areas for Wellington, with all available habitats shown. Neighbourhood buffers of three children living outside the main study area are not shown.

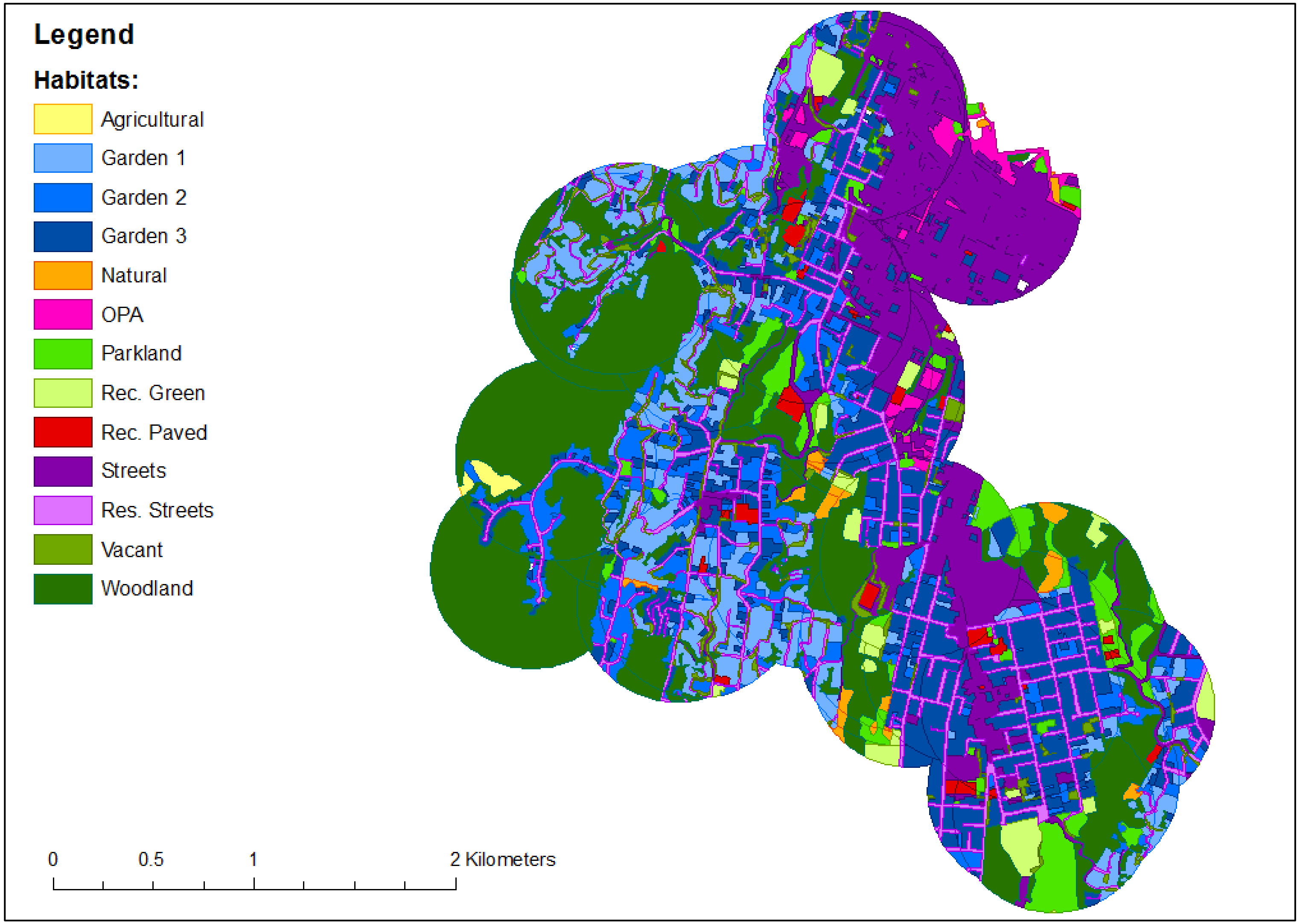

Figure 3A.3. Habitat map of the central Auckland neighbourhood buffers. Neighbourhood buffers of seven children who lived outside of this main area are not shown.

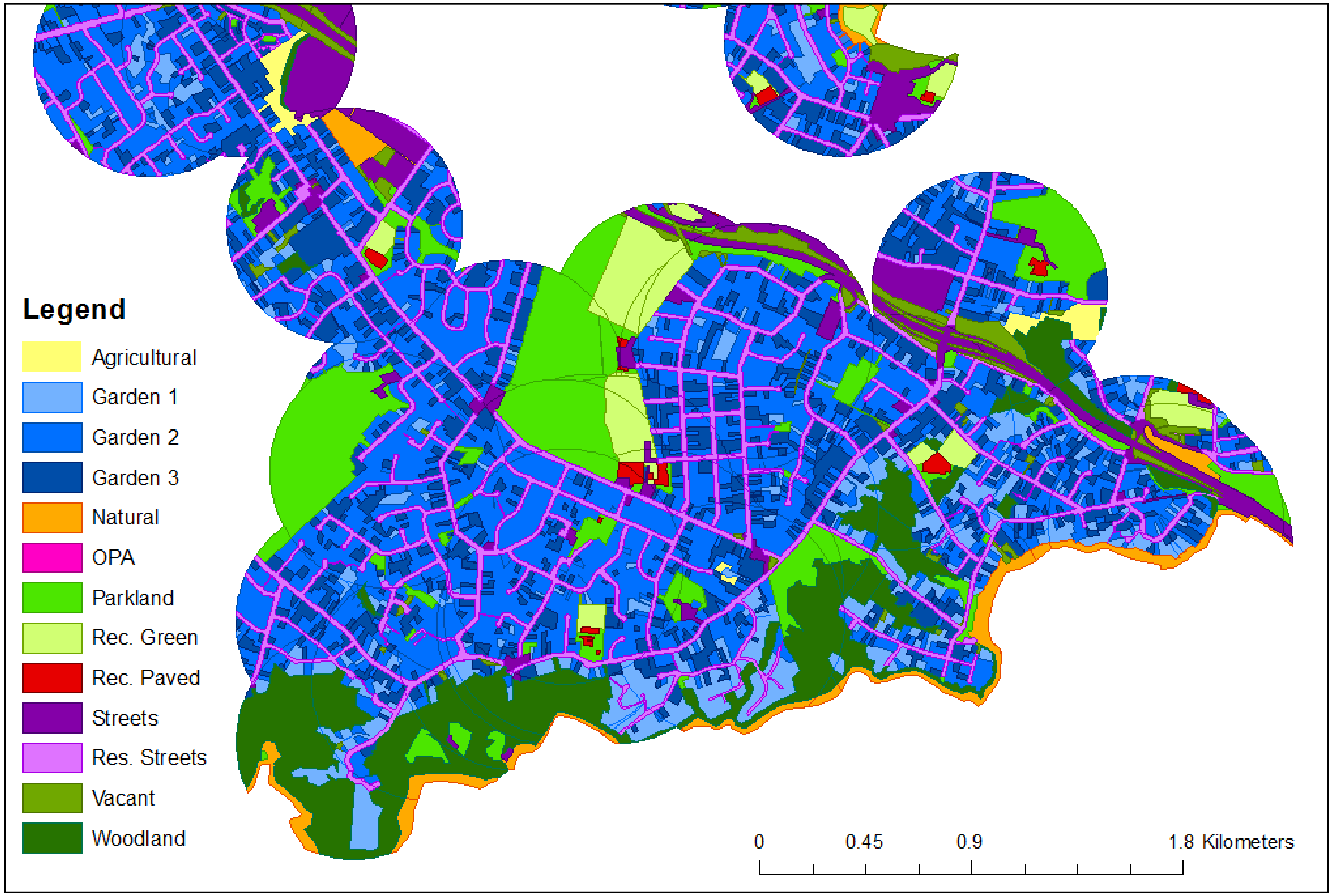

Table 3A.4. Candidate models testing effects on biodiversity in the buffer area. All models were fitted with neighbourhood as a random effect. Listed are values for AIC, *w* (Akaike's weight) and both marginal and conditional R^2 values.

Top buffer biodiversity models

Model	Equation	AIC	ΔAIC	*w*	R^2(m)	R^2(c)
B2	Deprivation	-2717.97	0	0.98	0.35	0.55
B0	1	-2708.72	8.25	0.02	0	0.42
B3	City*Deprivation	-2666.15	50.82	0	0.38	0.38
B4	Ethnicity*Deprivation	-2341.76	375.21	0	0.44	0.56
B5	Deprivation*City + Deprivation*Ethnicity	- 2292.94	424.03	0	0.48	0.6

Table 3A.5. Candidate model set for assessing children's home range size. All models were fitted with neighbourhood as a random effect. Listed are values for AIC, *w* (Akaike's weight) and both marginal and conditional R^2 values.

Home range size models

Model	Equation	AIC	ΔAIC	*w*	R^2(m)	R^2(c)
H2	Gender	172.38	0	0.59	0.048	0.05
H0	1	173.13	0.75	0.4	0	0
H6	Deprivation	181.64	9.26	0	<0.01	0
H3	Gender*Deprivation	185.91	13.53	0	<0.01	0.07
H4	Gender + Buffer biodiversity	188.14	15.76	0	<0.01	0.05
H7	Buffer biodiversity	189.5	17.12	0	<0.01	0.05
H5	Ethnicity	190.59	18.21	0	<0.01	0.05
H8	Buffer biodiversity + Deprivation*City + Gender	207.36	34.98	0	0.07	0.12
H9	Deprivation * Buffer biodiversity	216.00	43.72	0	0.08	0.08
H1	Deprivation*Ethnicity + Gender*Deprivation + Buffer biodiversity + Age + City	252.32	79.94	0	0.19	0.19

Table 3A.6. Ranking of the five candidate models assessed by Akaike's weights. All models were fitted with neighbourhood as a random effect. Listed are values for AIC, *w* (Akaike's weight) and both marginal and conditional R^2 values.

Available MCP biodiversity (area)						
Model	Equation	AIC	ΔAIC	*w*	R^2(m)	R^2(c)
V2	Gender	240.96	0	0.64	0.05	0.05
V0	1	242.09	1.13	0.36	0	0
V3	Gender*Deprivation	253.48	12.52	0	0.06	0.06
V4	Ethnicity	255.28	14.32	0	0.05	0.05
V5	Buffer biodiversity	256.88	15.92	0	0.09	0.09
V1	Buffer biodiversity + Ethnicity*Dep + City*Dep + Gender	308.42	67.46	0	0.15	0.17

Table 3A.7. Candidate models assessing proportional home range biodiversity models ranked by Akaike's weights. All models were fitted with neighbourhood as a random effect. Listed are values for AIC, *w* (Akaike's weight) and both marginal and conditional R^2 values.

Available MCP biodiversity (proportional) models						
Model	Equation	AIC	ΔAIC	*w*	R^2(m)	R^2(c)
VP6	Buffer Biodiversity + Ethnicity	769.46	0	0.76	0.3	0.48
VP1	Buffer Biodiversity + Deprivation +Ethnicity	771.77	2.31	0.24	0.31	0.48
VP8	Ethnicity	795.1	25.64	0	0.08	0.4
VP5	Buffer biodiversity	800.69	31.23	0	0.25	0.42
VP7	Buffer Biodiversity + Deprivation	803.08	33.62	0	0.26	0.42
VP4	Deprivation*Gender + Buffer biodiversity	804.57	35.11	0	0.26	0.42
VP3a	Deprivation*Gender	815.16	45.7	0	0.11	0.32
VP2	Gender	818.28	48.82	0	0.01	0.32
VP0	1	820.6	51.14	0	0	0.34

Table 3A.8. List of six candidate models considered in the assessment of biodiversity in children's accessible home ranges. All models were fitted with neighbourhood as a random effect. Listed are values for AIC, *w* (Akaike's weight) and both marginal and conditional R^2 values.

Accessible MCP biodiversity (area) models

Model	Equation	AIC	ΔAIC	*w*	R^2(m)	R^2(c)
C2	Gender	106.91	0	0.7	0.06	0.06
C0	1	108.62	1.71	0.3	0	0
C3	Gender*Deprivation	122.59	15.68	0	0.07	0.07
C5	Buffer biodiversity	125.05	18.14	0	0	0
C6	Ethnicity	130.47	23.56	0	0.05	0.05
C1	Ethnicity*Gender	139.33	32.42	0	0.02	0.15

Table 3A.9. Candidate models set for assessing factors explaining biodiversity in children's accessible home ranges. All models were fitted with neighbourhood as a random effect. Listed are values for AIC, *w* (Akaike's weight) and both marginal and conditional R^2 values.

Accessible MCP biodiversity (proportional)

Model	Equation	AIC	ΔAIC	*w*	R^2(m)	R^2(c)
CP10	Buffer biodiversity + Ethnicity	292.61	0	0.54	0.35	0.37
CP5	Buffer biodiversity	294.1	1.49	0.25	0.28	0.34
CP8	Ethnicity	295	2.39	0.16	0.38	0.4
CP9	Buffer biodiversity + Gender	297.49	4.88	0.05	0.28	0.34
CP0	1	309.19	16.58	0	0	0.24
CP2	Gender	310.3	17.69	0	0.02	0.22
CP4	Buffer biodiversity + Deprivation * Gender	310.8	18.19	0	0.28	0.33
CP7	Ethnicity	314.04	21.43	0	0.1	0.25
CP3	Deprivation*Gender	318.16	25.55	0	0.1	0.2
CP6	Buffer biodiversity * Deprivation	318.48	25.87	0	0.27	0.34

Table A.1 Candidate model rankings for resource selection analysis third order selection using the strict approach with the GBPS dataset (children who had access to gardens, biodiverse, play and street habitats, n = 30).

Third Order (strict) – GBPS dataset				
Model	Equation	AIC	ΔAIC	*w*
1b	Natural Grp + Street Grp + Woodland + SpG + Parks + Home	9669	0	1
1a	Gardens + Natural Grp.+ Streets + Woodland + SpG + Parks	9781	112	0
4c	Gardens +Woodland + Parks	9827	158	0
4b	Gardens + Woodland	9850	181	0
4a	Gardens + Woodland + Natural Grp.	9852	183	0
5a	Gardens + SpG +Parks	9904	235	0
2b	Home	9909	240	0
6b	Gardens + Streets + SpG	10002	333	0
4e	Natural Grp. + Woodland + Parks	10011	342	0
2c	Gardens	10020	351	0
5b	SpG + Parks	10037	368	0
5c	SpG	10128	459	0
6a	Street Grp.	10186	517	0
Global	Gardens + Natural Grp. + Street Grp + Woodland + SpG + Parks + Home	9664	*-4.6*	
Null	1	10194	*525*	

Table 4A.2. Candidate model rankings for resource selection analysis third order selection using the strict approach with the GBS dataset (children who had access to gardens, biodiverse, play and street habitats, n = 40).

Third Order (strict) – GBS dataset				
Model	Equation	AIC	ΔAIC	*w*
1b	Natural Grp + Street Grp + Woodland + SpG + Parks + Home	12573	0	1
1a	Gardens + Natural Grp.+ Streets + Woodland + SpG + Parks	12716	143	0
4a	Gardens + Woodland + Natural Grp.	12726	153	0
4b	Gardens + Woodland	12743	170	0
2c	Gardens	12905	332	0
6b	Gardens + Streets + SpG	12960	387	0
2b	Home	13244	671	0
6a	Street Grp.	13246	673	0
Global	Gardens + Natural Grp. + Street Grp + Woodland + SpG + Parks + Home	12574	1	
Null	1	13244	671	

Table 4A.3. Candidate model rankings for resource selection analysis third order selection using the strict approach with the GPS dataset (children who had access to gardens, biodiverse, play and street habitats, n = 69).

	Third Order (strict) – GPS dataset			
Model	Equation	AIC	ΔAIC	*w*
1b	Natural Grp + Street Grp + Woodland + SpG + Parks + Home	21845	0	1
1a	Gardens + Natural Grp.+ Streets + Woodland + SpG + Parks	21962	117	0
5a	Gardens + SpG +Parks	22080	235	0
6b	Gardens + Streets + SpG	22232	387	0
5b	SpG + Parks	22339	494	0
2b	Home	22348	503	0
2c	Gardens	22441	596	0
5c	SpG	22589	744	0
6a	Street Grp.	22825	980	0
Global	Gardens + Natural Grp. + Street Grp + Woodland + SpG + Parks + Home	21841	-4	
Null	1	22854	1009	

Table 4A.4. Candidate model rankings for resource selection analysis third order selection using the strict approach with the full dataset (distances taken to habitats outside of home range so all 118 children included).

Model	Equation	AIC	ΔAIC	*w*
1b	Home +Natural + Vacant + Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks	37198	0	0.92
1a	Gardens + Natural + Vacant + Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks	37203	5	0.08
5a	Gardens + Rec. Paved +Rec. Green + Parks	37273	75	0
4c	Gardens +Woodland + Parks	37337	139	0
2b	Home	37506	308	0
2c	Gardens	37510	312	0
4b	Gardens + Woodland	37512	314	0
4a	Gardens + Woodland + Vacant	37513	315	0
5b	Rec. Paved +Rec. Green + Parks	37770	572	0
4e	Vacant + Woodland + Parks	37857	659	0
5c	Rec. Paved +Rec. Green	37901	703	0
6a	Street + Res. Street	37996	798	0
4d	Woodland + Vacant + Natural	37198	821	0
Global	Gardens + Home +Natural + Vacant + Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks	37164	-34	-34
Null	1	38063	865	865

Table 4A.5. Model summary for third order (adapted) selection between genders and between cities for each dataset. Top models as selected by Akaike's weight are shown, with the number of children included in each model. Marginal and conditional R^2 values provide a measure of total variance explained by the fixed effects, and the fixed and random effects, respectively.

Category	Top Model	Akaike weight	Sample size	R^2(m)	R^2(c)
Female	1b	1	49	0.07	0.19
Male	1a	1	69	0.13	0.25
Auckland	1a	1	63	0.1	0.22
Wellington	1b	1	55	0.04	0.13

Table 4A.6. Candidate models for third order adapted approach comparing children in Auckland and Wellington. Models are sorted by Model ID and the top-ranked model for each city is shown in bold. Global and null models are provided below for comparison.

Model ID	Equation	Auckland				Wellington			
		AIC	ΔAIC	*w*	rank	AIC	ΔAIC	*w*	*rank*
1a	**Gardens + Natural + Vacant + Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks**	**18903**	**0**	**1**	**1**	**18009**	**24**	**0**	2
1b	**Natural + Vacant + Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks + Home**	**19089**	**186**	**0**	**4**	**17985**	**0**	**1**	**1**
2b	Home	19365	462	0	8	18055	70	0	5
2c	Gardens	19225	322	0	7	18073	88	0	9
4a	Gardens + Woodland + Natural + Vacant	19210	307	0	5	18050	65	0	3
4b	Gardens + Woodland	19220	317	0	6	18059	74	0	7
4c	Gardens +Woodland + Parks	19045	142	0	3	18054	69	0	4
4d	Woodland + Vacant + Natural	19843	940	0	12	18146	161	0	11
4e	Natural + Vacant. + Woodland + Parks	19491	588	0	10	18174	189	0	13
5a	Gardens +Rec. Paved +Rec. Green +Parks	19036	133	0	2	18064	79	0	8
5b	Rec. Paved +Rec. Green + Parks	19417	514	0	9	18055	70	0	6
5c	+ Rec. Paved +Rec. Green	19678	775	0	11	18168	183	0	12
6a	Res. Street + Street	19856	953	0	13	18128	143	0	10
Global	Gardens + Home +Natural + Vacant + Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks	18878	-25			17986	1		
Null	1	19887	984			18175	190		

Table 4A.7 Candidate models for third order adapted approach comparing genders. Models are sorted by Model ID and top-ranked models for each gender are shown in bold. Global and null models are shown below the candidate models for comparison.

Model ID	Equation	Girls				Boys			
		AIC	ΔAIC	*w*	rank	AIC	ΔAIC	*w*	rank
1a	**Gardens + Natural + Vacant.+ Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks**	**15793**	**62**	**0**	**4**	**21342**	**0**	**1**	**1**
1b	**Natural + Vacant + Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks + Home**	**15731**	**0**	**1**	**1**	**21392**	**50**	**0**	**2**
2b	Home	15748	17	0	2	21733	391	0	8
2c	Gardens	15815	84	0	8	21694	352	0	6
4a	Gardens + Woodland + Natural + Vacant	15812	81	0	7	21696	354	0	7
4b	Gardens + Woodland	15810	79	0	6	21694	352	0	5
4c	Gardens +Woodland + Parks	15794	63	0	3	21517	175	0	4
4d	Woodland + Vacant + Natural	15978	247	0	13	22013	671	0	12
4e	Vacant + Woodland + Parks	15964	233	0	12	21849	507	0	10
5a	Gardens + S Rec. Paved +Rec. Green +Parks	15795	64	0	5	21431	89	0	3
5b	Rec. Paved +Rec. Green + Parks	15920	189	0	9	21754	412	0	9
5c	Rec. Paved +Rec. Green	19678	775	0	10	21884	542	0	11
6a	Res. Street + Street	15960	229	0	11	22030	688	0	13
Global	Gardens + Home +Natural + Vacant + Res. Street + Street + Woodland + Rec. Paved +Rec. Green + Parks	15738	7			21326	-16		
Null	1	15981	250			22085	743		

Table 4A.8. Candidate model rankings for resource selection analysis second order selection using the strict approach with the GBPS dataset (children who had access to gardens, biodiverse, play and street habitats, n = 80).

Second order - GBPS dataset				
Model	Equation	AIC	ΔAIC	*w*
1a	Gardens + Natural Grp.+ Streets + Woodland + SpG + Parks	20924	0	1
1b	Natural Grp + Street Grp + Woodland + SpG + Parks + Home	21230	306	0
4c	Gardens +Woodland + Parks	21293	369	0
4a	Gardens + Woodland + Natural Grp.	21308	384	0
4b	Gardens + Woodland	21308	384	0
6b	Gardens + Streets + SpG	21617	693	0
5a	Gardens + SpG +Parks	21651	727	0
2c	Gardens	22154	1230	0
2b	Home	22716	1792	0
4e	Natural Grp. + Woodland + Parks	26473	5549	0
5b	SpG + Parks	26473	5549	0
5c	SpG	26480	5556	0
6a	Street Grp.	26638	5714	0
Global	Gardens + Natural Grp. + Street Grp + Woodland + SpG + Parks + Home	20237	-687	
Null	1	26850	6026	

Table 4A.9.Candidate model rankings for resource selection analysis second order selection using the strict approach with the GBS dataset (children who had access to gardens, biodiverse, play and street habitats, n = 84).

Second Order – GBS dataset				
Model	Equation	AIC	ΔAIC	*w*
1a	Gardens + Natural Grp.+ Streets + Woodland + SpG + Parks	21975	0	1
4a	Gardens + Woodland + Natural Grp.	21990	15	0
4b	Gardens + Woodland	22514	539	0
1b	Natural Grp + Street Grp + Woodland + SpG + Parks + Home	22873	898	0
2c	Gardens	23444	1469	0
2b	Home	27327	5352	0
4e	Woodland + Natural Grp.	27399	5424	0
6a	Street Grp.	21975	0	0
Global	Gardens + Natural Grp. + Street Grp + Woodland + SpG + Parks + Home	21460	-515	
Null	1	27761	5786	

Table 4A.10. Candidate model rankings for resource selection analysis second order selection using the strict approach with the GPS dataset (children who had access to gardens, biodiverse, play and street habitats, n = 100).

Second order – GPS dataset				
Model	Equation	AIC	ΔAIC	*w*
5a	Gardens + SpG +Parks	27412	0	0.73
1a	Gardens + Natural Grp.+ Streets + Woodland + SpG + Parks	27414	2	0.27
6b	Gardens + Streets + SpG	27419	7	0
2c	Gardens	28063	651	0
1b	Natural Grp + Street Grp + Woodland + SpG + Parks + Home	28098	686	0
2b	Home	29102	1690	0
5b	SpG + Parks	33360	5948	0
5c	SpG	33360	5948	0
6a	Street Grp.	33760	6348	0
Global	Gardens + Natural Grp. + Street Grp + Woodland + SpG + Parks + Home	34000	-602	
Null	1	26810	6588	

Table 4A.11. Model summary for second order selection between genders and between cities for each dataset. Top models as selected by Akaike's weight are shown, with the number of children included in each model. Marginal and conditional R^2 values provide a measure of total variance explained by the fixed effects, and the fixed and random effects, respectively.

Category	**Dataset**	**Top Model**	**Akaike weight**	**Sample size**	**R^2(m)**	**R^2(c)**
Female	GBPS	1b	1	31	0.47	0.75
	GBS	1a	1	35	0.48	0.68
	GPS	1b	1	37	0.45	0.75
Male	GBPS	1a	1	49	0.53	0.74
	GBS	1a	1	49	0.5	0.7
	GPS	1a	1	63	0.53	0.66
Auckland	GBPS	1a	1	31	0.62	0.79
	GBS	1a	1	34	0.63	0.78
	GPS	1a	1	47	0.64	0.77
Wellington	GBPS	1a	1	49	0.56	0.71
	GBS	1a	1	50	0.48	0.63
	GPS	1a	1	53	0.52	0.66