HELSINGIN YLIOPISTO
HELSINGFORS UNIVERSITET
UNIVERSITY OF HELSINKI

Relationships Between Tree Cover, Built Cover, and Bird Diversity in an Urbanizing Landscape: A Case Study of Central Viikki, Helsinki

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Sini Ruohomäki
Supervisor: Ian MacGregor-Fors

HELSINGIN YLIOPISTO

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Tiivistelmä - Referat – Abstract

Kaupungistumisen aiheuttamat maankäytön muutokset ovat yksi globaalin luontokadon merkittävistä syistä, kun luonnon ekosysteemit muutetaan rakennetuksi ympäristöksi. Kaupunkirakenteen tiivistämisessä pyritään keskittämään maankäytön muutokset jo valmiiksi rakennetuille alueille, mutta se voi silti aiheuttaa luonnon monimuotoisuuden kannalta haitallista elinympäristöjen häviämistä ja pirstoutumista. Kaupunkeja on kuitenkin mahdollista rakentaa nykyistä luontoystävällisemmin, ja sitä varten tarvitaan tietoa kaupungistumisen vaikutuksista luonnon monimuotoisuuteen. Linnut ovat monipuolinen, laajasti tutkittu ja elinympäristön muutoksille herkkä lajiryhmä, mikä tekee niistä käyttökelpoisia kaupunkiluonnon monimuotoisuudesta kertovia bioindikaattoreita.

Tässä tutkimuksessa selvitän kaupungin tiivistymisen mahdollisia vaikutuksia paikalliseen linnustoon Viikissä, Helsingissä. Viikki kuuluu ekologisesti arvokkaimpiin alueisiin Helsingissä, ja se tunnetaan erityisesti monipuolisesta linnustostaan. Tutkin, miten kaksi maankäytön mittaria — puiden latvuspeittävyys ja rakennettu pinta-ala — ovat yhteydessä lintujen lajirunsauteen ja lukumääriin. Päiväaktiivisiin lintuihin keskittyvä kartoitus tehtiin kesällä 2024.

Tulosten mukaan puiden latvuspeittävyydellä on vahva yhteys lintujen lajirunsauteen. Puiden latvuspeittävyydelle tunnistettiin kynnysarvo 14 %, eli kun latvuspeittävyys yhden hehtaarin koealalla oli alle 14 %, lintujen lajirunsaus oli sillä huomattavasti vähäisempi, ja vastaavasti laajempi latvuspeittävyys oli yhteydessä korkeampaan lajirunsauteen. Rakennetun pinta-alan yhteys lajirunsauteen oli myös merkitsevä, mutta heikompi kuin latvuspeittävyyden. Kummallakaan maankäytön muuttujalla ei ollut merkittävää yhteyttä lintujen lukumäärään. Keski-Viikin kaupungistumisen skenaariomallinnus osoitti, että latvuspeittävyyden ja rakennetun pinta-alan todennäköiset yhdistelmät johtavat todennäköisesti pääosin lintujen lajirunsauden vähenemiseen. Tulokset korostavat puiden latvuspeittävyyden ylläpitämisen ja kasvattamisen merkitystä kaupunkiympäristössä lintujen, erityisesti metsäisten lajien, kannalta. Tulosten pohjalta väitän, että lintujen ja muun luonnon monimuotoisuus olisi syytä ottaa huomioon jo kaupunkisuunnittelun aikaisessa vaiheessa, jotta voidaan suunnitella ja rakentaa luontoystävällisempiä kaupunkeja.

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Tiivistelmä - Referat - Abstract

Urbanization causes major land-use changes, contributing to global biodiversity loss as natural ecosystems are transformed into built environment. While urban densification aims to concentrate development within already built areas, it can still result in habitat loss and fragmentation, posing challenges for urban biodiversity. Understanding how urbanization affects biodiversity is needed for informing more biodiversity-friendly urban planning. Birds—being diverse, well-studied, and responsive to habitat changes—serve as useful bioindicators of urban biodiversity.

This study explores how future urban development in Viikki, Helsinki, may affect local bird communities. Viikki is one of Helsinki's most ecologically valuable areas, particularly known for its rich birdlife. I investigated the relationships between two land cover metrics—tree cover and built cover—and bird species richness and abundance. Field surveys of diurnal landbird communities were conducted during summer 2024.

The results show a strong positive relationship between tree cover and bird species richness. A threshold of 14% tree cover was identified, meaning that 1-hectare sites with less than 14% tree cover hosted fewer bird species, while sites with higher tree cover were associated with notably higher species richness. Built cover showed a weaker negative association with species richness. Neither of the land cover variables showed a significant relationship with bird abundance. Scenario modelling of urban development in Central Viikki suggested that most of the plausible variations in tree and built cover would lead to declines in bird species richness. These findings highlight the importance of maintaining tree cover in urban landscapes, particularly for woodland bird species. I argue that biodiversity considerations should be integrated early in urban planning processes to promote more biodiversity-friendly cities.

Keywords
Urban biodiversity, urbanization, urban planning, birds

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1 Introduction

1.1 Urbanization

Urbanization is a global megatrend transforming the world. Land-use change, including expansion of urban land-use, is one of the main drivers of global environmental change and biodiversity loss (IPBES, 2019). Cities are made up of artificial infrastructure, buildings and various types of altered environments, which eliminates the local natural ecosystem (Guneralp & Seto, 2013). Urbanization tends to be a more lasting or even irreversible type of habitat loss than, for example, land-use for agricultural or forestry purposes (McKinney, 2002). Urban land-use is intensive and diverse, as accommodating the needs of the urban people requires different residential, industrial and commercial, recreational and social facilities (Pauleit & Breuste, 2011). Urban systems differ from non-urban systems in that they have continuous massive flow of inputs, such as energy and materials, and outputs, like waste and heat, which alters the ecosystem ecology (Adler & Tanner, 2013). Rapid habitat modification in urban areas challenges organisms with different types of disturbance and conditions than in non-urban systems (Adler & Tanner, 2013). One major form of ecological alteration in urban systems is soil sealing, meaning that built surfaces such as roads, buildings, and pavements cover the ground replacing vegetation and often preventing natural ecosystem processes (Pauleit & Breuste, 2011). Ecosystem processes are closely tied to urban land use and cover, meaning that urban planning and land-use regulations can significantly impact ecological dynamics and biodiversity in cities (Pauleit & Breuste, 2011). While urban development poses a major threat to wildlife populations, its ecological impacts can be mitigated through biodiversity-friendly planning (Ikin et al., 2010; Plummer et al., 2020).

Urban development can take place in different ways and forms depending on planning (or lack of planning). For instance, urban expansion (growth or sprawl) refers to a phenomenon of the city expanding to the surrounding, often rural or natural land, which leads to increase in urban land-use (Bhatta, 2010). Urban expansion is typically characterized by low density building, long distances, and uncoordinated planning taking large amount of space (Bhatta, 2010; Zhang et al., 2023) It can even lead to informal conversion of land to urban use, occupation of hazardous sites and lack of community infrastructure and services (Angel, 2023). Urban expansion is associated with adverse socio-economic effects like social inequality and segregation as well as high costs for public services (Zhang et al., 2023). Lack of proper planning can also lead to the destruction of valuable ecological assets, such as wetlands or other high-quality green spaces (Angel, 2023). Other ecological consequences include higher energy consumption, increases in urban heat islands, and air pollution (Zhang et al., 2023). Due to the various negative effects, urban expansion is often described as an undesirable land-use pattern (Ewing, 2008).

In contrast, urban densification is a form of urban development that many cities actively pursue by promoting infill development and consolidation, to avoid the negative consequences of urban expansion (Teller, 2021). Infill development refers to development of already built-up areas with existing essential infrastructure (Bhatta, 2010). This kind of compact city approach is linked to limited loss of natural areas as well as many other benefits such as efficient transportation systems, walkability, reduced energy use, and enhanced social and economic opportunities (Bibri et al., 2020). Infill development and compact city planning is promoted by the European Union (EU) as EU aims for reaching a target of “no-net-land take by 2050” as part of the Soil Strategy 2030 (European Commission, 2021). No-net-take target is justified by following arguments: land conversion to artificial surfaces leads to less resilient ecosystems, decreased potential for carbon storage and biodiversity maintenance, increased surface run-off during floods and increased effects of heatwaves in cities as well as diminished ecological land functions and direct loss of natural areas (European Commission, 2021).

Despite the benefits and political steer towards densification, researchers have pointed out some of its risks and drawbacks. Densification tends to lead to a reduction in urban green space within cities (Pauleit & Breuste, 2011; Berghauser Pont et al., 2021, Haaland & Konijnendijk van den Bosch, 2015), while many urban environmental problems—such as urban heat island effect, increased flood risk, and air and noise pollution—are related to lack of green areas (Berghauser Pont et al., 2021). Additionally, urban areas often overlap with naturally biodiversity-rich areas, such as coastal zones and riverine areas, which may exacerbate the negative effects of habitat loss and fragmentation resulting from land-use change (Ricketts & Imhof, 2003; Colding, 2007; Seto et al., 2012). As urban areas continue to grow, integrating biodiversity considerations into spatial planning can help avoid unintended consequences for biodiversity and ecosystem functioning. According to the globally adopted Kunming-Montreal Global Biodiversity Framework, human-induced species extinction should be halted by 2050 (Convention on Biological Diversity, 2022). At the EU level, the Biodiversity Strategy for 2030 sets a goal for Europe's biodiversity being on a path to recovery by 2030 (European Commission: Directorate-General for Environment, 2021). Research and knowledge of urban biodiversity can be used to improve practices related to urban planning and management so that they would meet the aims to conserve biodiversity without compromising the associated costs, uncertainties, trade-offs and risks (Ossola et al., 2018).

1.2 Birds in urban environments

Urban environments may be hostile and challenging for organisms, yet numerous species—from beneath the surface to above the skyline—live in these human-dominated landscapes (Knapp et al., 2021). Research and recognition of urban biodiversity has grown since 1990s with the focus not only on which species are found in cities but also on how they are able to persist, adapt and become established and how they are linked to ecosystem functioning (Rega-Brodsky et al., 2022). Birds are among the most studied taxonomic groups in urban settings (McKinney, 2008; Rega-Brodsky et al., 2022). Birds are widely recognized as useful indicators of overall biodiversity—also known as bioindicators— (Lepczyk et al., 2008; Evans et al. 2009; Suarez-Rubio & Thomlinson, 2010; MacGregor-Fors et al., 2013; BirdLife International, 2022; European Environment Agency, 2024). They represent a diverse species group with essential ecological roles from predators to seed dispersers and scavengers (BirdLife International, 2022). They are much studied, well-known and diverse taxa, which can form complex communities in urban areas and respond quickly to habitat changes (MacGregor-Fors et al., 2013). Birds are also relatively easy to survey (Plummer et al. 2020). These qualities make them a useful choice of taxa to inform biodiversity-friendly urban planning (Plummer et al., 2020; BirdLife International, 2022).

Concerningly, bird populations are declining significantly both globally (BirdLife International, 2022), at the EU level (EEA, 2024) and locally in Finland (BirdLife Finland, 2023). Bird population numbers are one indicator of the health of the environment and can be used for assessing the success of EU's goal to halt biodiversity loss and put it on the path of recovery by 2030 (EEA, 2024). Beyond their ecological role, birds also contribute to human well-being, as they are among the most visible urban wildlife and widely appreciated by people (Clucas & Marzluff, 2011). Birdwatching is a popular activity with various positive health and well-being effects (Peterson et al., 2024) and exposure to birdsong soundscape improves mental well-being (Stobbe et al., 2022). According to Cox et al. (2017) both vegetation cover and bird abundance are associated with better mental well-being for urban residents. However, bird-human interactions are more complex and can also include conflicts and disservices such as aggression towards people or pets and disturbing noises (Pejchar et al., 2025). Still, given the ecological importance, sensitivity to environmental change, and mostly positive influence on human well-being, birds serve as valuable indicators for biodiversity conservation and the creation of sustainable and healthy cities.

Urban environments form a complex mosaic of green, aquatic and built habitats (Adler & Tanner, 2013) each supporting diverse bird communities. Green spaces are diverse patches which vary in size, use, management, and disturbance as well as in history and purpose (Hostetler et al., 2011). Green spaces, such as woodlands, parks, cemeteries, and street trees, provide nesting sites and food resources for many bird species, including songbirds (*Passeriformes*) and woodpeckers (*Picidae*) (Hostetler et al., 2011). Aquatic habitats—rivers, ponds, wetlands, and canals—support waterbirds such as ducks and geese (*Anseriformes*). Built areas, including buildings, roads, and paved surfaces, are habitats with little natural vegetation but can provide habitats for species like rock doves (*Columba livia*) and house sparrows *(Passer domesticus).* Urban brownfields and landfills attract scavengers, such as gulls (*Larus* spp.) and crows (*Corvus* spp.), taking advantage of human-generated waste (Hostetler et al., 2011). The availability, size, quality, and connectivity of these habitat types play a critical role in shaping bird diversity and population dynamics across the urban landscape (Lepczyk et al., 2017).

Urban green habitats are often highly fragmented, breaking continuous green areas into smaller, isolated patches (Marzluff & Ewing, 2008). These habitats are often described as habitat islands due to their isolation inside dense built matrix (Fernández-Juricic & Jokimäki, 2001; Oliver et al., 2011). Drawing from Island Biogeography Theory (MacArthur & Wilson, 1967) fragmentation poses a major threat to biodiversity as smaller patches tend to host fewer species and are more exposed to edge effects, such as increased disturbance, making these small, isolated subpopulations more vulnerable to local extinctions (Haddad et al., 2015). In such fragmented landscapes, species often depend on access to larger nearby habitat areas to survive (Adler & Tanner, 2013). However, the ability to move between areas in fragment landscape and utilize resources from different habitats depends on species mobility and adaptability, like capability to fly and tolerate urban disturbance (Schneider et al., 2015). Fragmentation leads to smaller patch sizes and reduced connectivity, both of which can negatively affect urban biodiversity. Plummer et al., (2020) found that contiguous green areas are more suitable for accommodating breeding birds than numerous smaller habitat patches. MacGregor-Fors et al., (2013) recommend that green areas, such as parks, should ideally range between 20 and 100 hectares depending on the city's size and geographic location, while parks smaller than 1 hectare should be avoided if the aim is to increase biodiversity. Lepczyk et al. (2017) and Fernández-Juricic and Jokimäki (2001) suggest that a patch size of approximately 10 to 35 hectares is sufficient to support most urban-dwelling bird species. Evans et al. (2009) argue that patch size influences bird assemblages more than connectivity. However, MacGregor-Fors et al. (2013) emphasize that interconnectivity of green areas can help urban-sensitive species to live in cities and argue that green corridors should be established especially along main waterways and boulevards, and they should be as natural as possible and at least width of 100 m. Beninde et al. (2015) conclude that for maintaining overall biodiversity, large green spaces of at least 50 hectares are essential to prevent rapid species loss. Moreover, they highlight that the best strategy for supporting urban biodiversity is to maintain a network of sufficiently large habitat patches connected by ecological corridors.

1.3 Quantifying urbanization

When studying how urbanization impacts biodiversity, it is important to understand what is considered "urban" and how it is quantified to make comparisons between studies and cities possible (McDonnell & Hahs, 2008; Moll et al., 2019; MacGregor-Fors, 2011; Rega-Brodsky et al., 2022) Urban areas are generally defined as regions with a higher density of human-made structures and population compared to surrounding areas (Adams & Lindsey, 2011). Since urban environments vary widely in size, structure, and habitat composition, a common approach is the use of urbanization gradients, as proposed by McDonnell and Pickett (1990). Use of urbanization gradients enables analysis of the factors and responses to environmental changes resulting from urbanization (McDonnell & Hahs, 2008). The measures used for urbanization gradient can be broard or specific (McDonnell & Hahs, 2008) and are typically based on land use or land-cover, while there is also a call for demographic metrics (McDonnell & Hahs, 2008; Moll et al. 2019).

One common broad measure for urbanization gradient is the proportion of built surfaces (McDonnell & Hahs, 2008). For example, McKinney (2002) simplifies urbanization gradient into four categories by impervious surface cover: rural (no or only little impervious), urban fringe (<20% impervious), suburbia (20%-50% impervious), urban core (> 50% impervious). A more recent categorization by MacGregor-Fors (2011) divides urban areas into (1) sparsely developed (0–33% built cover), (2) moderately developed (34–66% built cover), and (3) highly developed (67–100% built cover). Alternatively, MacGregor-Fors et al. (2011) consider areas urban if they have more than 50% built cover at the landscape scale and a minimum population density of 1,000 people/km^2, integrating also demographic metrics. As demonstrated, the definition of "urban" varies, which can make it challenging to compare and generalize findings across urban ecological studies, as MacGregor-Fors (2011) argues. Nevertheless, urbanization gradients remain an intuitive and relatively simple tool for understanding urbanization's ecological impacts (Beninde et al., 2015).

1.4 Urban biodiversity patterns

For most taxa, including birds, diversity is likely to be lower in cities and especially low in the urban core compared to the surrounding areas (McKinney, 2008; Aronson et al., 2014). The Intermediate Disturbance Hypothesis (IDH), originally introduced by Connell (1978), suggests that biodiversity is highest at intermediate levels of disturbance. This theory has been applied in urban ecology as urban environments can be seen as highly disturbed environments due to human activities. Some studies (Jokimäki & Suhonen 1993; Blair, 1996; Lepczyk et al., 2008; Guetté et al. 2017) have shown that bird species richness—the number of different species present—tends to peak in areas with moderate levels of human development compared to more natural or heavily urbanized areas. However, most studies have not found evidence of this pattern for birds (Marzluff et al., 2001; Thompson et al., 2022) and show instead that bird diversity declines with increasing urbanization (Chase & Walsh 2006; Lepczyk et al. 2008, Chamberlain et al. 2017; MacGregor-Fors & Schondube, 2012; Escobar-Ibáñez et al. 2020). Accordingly, Fox (2013) argues against the validity of the IDH in urban settings. Plummer et al. (2020) and Evans et al. (2015) emphasize that species' responses to urbanization are highly variable, making broad generalizations difficult.

Another generalized pattern is that bird abundance—the number of individual birds—tends to be higher in highly urbanized areas or city centers (Donnelly & Marzluff, 2004; Chase & Walsh 2006; Ortega-Álvarez & MacGregor-Fors, 2009; Kurucz et al., 2021). More specifically, bird species richness and abundance generally decline along the urban gradient, with only few species thriving in highly urbanized environments (MacGregor-Fors et al., 2013). As a result, city centers often have a high abundance of a few dominant, generalist species, while overall species richness remains low and specialist species are largely absent (Clergeau et al., 2006; Kurucz et al., 2021). Moreover, bird communities in highly developed urban areas tend to resemble one another, supporting a similar set of urban-tolerant species —particularly omnivores, granivores, and cavity-nesters (Chace & Walsh, 2006).

Cities present both opportunities and challenges for birds, and different species respond differently to urban conditions. Cities can offer increased resource availability (Evans et al., 2015), higher food abundance (Shochat, 2004), supplemental feeding by humans (Clucas & Marzluff, 2011), and greater availability of nesting sites (Kurucz et al., 2021). On the other hand, window collisions—which kill billions of birds annually (Klem, 2009)— and predation by free-roaming cats (van Heezik et al., 2010) are significant risks. Understanding why some species persist or even thrive in urban environments, while others disappear requires looking beyond habitat availability to broader ecological and evolutionary patterns. Species may succeed in cities by being preadapted, adjusting to novel conditions, or evolving over time (Adler & Tanner, 2013). Birds have been initially categorized by Blair (1996) into three groups based on their response to urbanization: urban avoiders, urban adapters, and urban exploiters. This framework was later refined by Fischer (2015), who introduced the three following categories: urban avoiders, urban utilizers, and urban dwellers. Urban avoiders are species that prefer natural areas and are rarely found in urban environments, except in large, undisturbed natural spaces embedded within cities. Urban utilizers are species that take advantage of urban resources but do not primarily breed in cities, while urban dwellers are species that successfully establish and persist in urban areas. The key difference between urban utilizers and urban dwellers lies in population dynamics: while utilizers rely on surrounding natural habitats for survival, dwellers are self-sustaining within urban environments (Fischer et al., 2015). These categories reflect species-specific habitat preferences and adaptability to urbanization. Consequently, habitat changes that favor urban dwellers tend to negatively impact urban avoiders, and vice versa (MacGregor-Fors et al., 2013).

As urban areas expand and natural habitats decline, certain species are becoming more prevalent in cities, while others struggle to persist. Well-known examples of urban-dwelling species include for example the house sparrow (*Passer domesticus*), feral pigeon (*Columba livia domestica*), and European starling (*Sturnus vulgaris*) (Aronson et al., 2014). These species are not only highly related to urbanized sites but also cosmopolitan, occurring in over 80% of cities included in a global analysis by Aronson et al. (2014). Not all species are as flexible, and according to Marjakangas et al. (2024), approximately 22% of bird species can tolerate highly modified, human-dominated environments. However, they emphasize that these human tolerance indices may be uncertain due to extinction debt, where species in degraded habitats persist temporarily but may go extinct if they fail to adapt to increasing human pressures. Overall, urbanization negatively impacts bird populations worldwide, as concluded by Aronson et al. (2014). Therefore, conservation efforts, including biodiversity-friendly urban planning, are needed to protect biodiversity.

1.5 Tree cover and built cover as indicators of urban ecological sustainability

Biodiversity loss is among the most pressing sustainability challenges cities are facing, and there is a growing need for diverse sustainability solutions and reliable methods to assess their effectiveness (European Commission, 2021) Indicators are widely used tools for evaluating and monitoring sustainability efforts (Singh et al., 2012; United Nations General Assembly, 2017; European Commission, 2021). Indicator refers to an ‘‘operational representation of an attribute (quality, characteristic, property) of a system’’ (Gallopin, 1997). In practice, sustainability indicators are used as ways to summarize, simplify, and quantify the complex dynamics in our environment into manageable and usable information (Singh et al., 2012). Indicators are valuable beyond scientific research by providing practical tools to assess the effects of various urban policies, designs, infrastructures, and sustainability interventions for city planners, managers, and policymakers (Kenney et al., 2017; European Commission, 2021). As Kenney et al. (2016) explain, indicators serve as reference tools that help assess status, rates of change, and long-term trends using measured or modeled data. According to Hiremath et al. (2013) sustainability indicators should be policy relevant, scientifically founded, readily implantable, and useful for planning purposes. Essential characteristics of effective indicators are that they should be systematically updated and provide a comparative baseline for measuring change (Kenney et al., 2016). Land cover being an important factor affecting ecosystem processes and biodiversity, land cover metrics — such as tree cover and built cover — can be used as easily measurable urban ecological sustainability indicators (Pauleit & Breuste, 2011; Hautamäki & Laita, 2024).

Tree cover refers to the proportion of land area covered by tree canopy, as viewed from above (Hautamäki & Laita, 2024). Tree cover, as a part of overall vegetation cover, is a key factor influencing bird species richness and abundance by providing food resources, nesting sites, and shelter (Radford et al., 2005; Trollope et al., 2009; MacGregor-Fors et al., 2013; Villaseñor et al., 2021). Trees enhance habitat complexity, which is generally linked to higher biodiversity (Evans et al., 2009). However, Ferenc et al. (2014) emphasize that tree cover is especially important for woodland birds. Villaseñor et al. (2021) found that greater tree cover at the landscape level increases both species diversity and abundance, while Radford et al. (2005) demonstrated that bird species richness—particularly among woodland-dependent species—declines sharply when tree cover falls below 10% at the landscape scale. Trollope et al. (2009) suggest that tree cover strongly correlates with the occurrence of urban-sensitive species, while Villaseñor et al. (2021) emphasize its role in supporting diverse bird groups through critical food provisioning. Species-specific responses to tree cover seem to vary; woodpeckers, woodland birds, and hole-nesting species generally benefit from higher tree density, while many urban generalists may not (Sandström et al., 2006). According to Tremblay and St Clair (2011) urban sensitive bird species require canopy cover from 20% to 40% for movement between sites, while more urban tolerant species may need as little as 2% to 4% to do so. Additionally, the height and age of trees contribute to habitat quality, as taller, older trees provide more nesting cavities, food, and structural complexity (MacGregor-Fors et al., 2013; Jokimäki et al., 2014). Beyond birds, urban trees offer numerous ecosystem services, including climate change mitigation and adaptation, aesthetic and cultural benefits, and contributions to human health and well-being (Salmond et al., 2016; Hautamäki & Laita, 2024). Recognizing these benefits, the EU Nature Restoration Law mandates no net loss of urban tree cover and promotes a threshold of 10% tree cover at the city level by 2030, with an increasing trend thereafter (Regulation (EU) 2024/1991). Konijnendijk et al. (2022) recommend 30% tree cover as a target for cities, yet most urban areas remain well below this level (Croeser et al., 2024).

Built cover (also referred to as urban land cover or artificial land cover) describes the proportion of land occupied by built structures, including buildings, roads, and other infrastructure, within a given area (Pauleit & Breuste, 2011). It focuses on impervious surfaces, such as asphalt and concrete, which cover substantial portions of urban environments. These sealed surfaces disrupt natural ecosystem processes, including the flow of water, energy, nutrients, and air, contributing to urban problems such as heat islands, increased flood risk, and pollution (Pauleit & Breuste, 2011). Built cover typically increases with urban development; for example, Angel et al. (2005) estimated that in high-income countries, each new city inhabitant requires approximately 350 m^2 of built space. Built cover is often considered the counterpart of vegetation cover, which is fundamental to biodiversity because plants serve as primary producers supporting various taxa. However, vegetation can coexist with built structures through green solutions like street trees, green roofs, vertical gardens, and planter boxes. On the other hand, agricultural fields are not classified as built cover, though they may also lack vegetation at certain times (Brunbjerg et al., 2018). In large-scale urban studies, vegetation cover is frequently identified as a key driver of species richness across various taxonomic groups (Beninde et al., 2015). Globally, Aronson et al. (2014) found vegetation cover to be the strongest explanatory variable for bird and plant species density. The conversion of vegetated areas into sealed built surfaces thus has inevitable consequences for biodiversity. Built cover is often used as a simple proxy for urbanization intensity (McDonnell et al., 2008). In addition to surface cover, the three-dimensional structure of the urban form—including building height, density, and layout—also influences habitat availability and microclimates. However, studies investigating the effects of built cover on bird diversity have produced somewhat mixed results. For example, study by García-Arroyo et al. (2025) from Lahti, shows bird species richness decreasing especially in sites with most built cover and distinct bird community compositions at 50% built cover threshold. Andersson and Colding (2014), on the other hand, found minimal differences in bird communities across Swedish urban areas with similar tree cover, suggesting that urban form alone may not strongly shape species richness. In contrast, Evans et al. (2009) reported declines in both species richness and individual bird densities with increasing building density in British cities. Their study also showed that bird communities respond positively to the structural complexity and species richness of woody vegetation but negatively to human disturbance. Similarly, MacGregor-Fors (2011) highlights the significant role of human activity—including noise and pedestrian movement—in shaping urban biodiversity. Together, these findings suggest that built cover alone doesn’t fully explain biodiversity patterns. Instead, the interaction between urban form, green infrastructure, and human disturbance seem to play a critical role in shaping bird communities.

1.6 The Central Viikki urban development case

Helsinki, the capital and largest city of Finland, has experienced significant population growth over recent decades. The Helsinki City Strategy, “A Place for Growth” (2021–2025), outlines the city’s development priorities, including nature objectives (City of Helsinki, 2021). The strategy emphasizes urban growth and housing development, particularly through renovation and infill construction within the existing built environment, with much of this development concentrated along rail networks. The strategy acknowledges the need to balance urban growth with environmental values:

A growing city requires a reconciliation between the values associated with compact living and those of the surrounding natural environment. (City of Helsinki, 2021).

Central Viikki is one of the areas designated for future infill development in Helsinki. The new light rail route, “Jokeri,” which connects Eastern Helsinki and Espoo, began service in October 2023 (Raidejokeri, n.d.). This light rail passes through Central Viikki, making the area a target for further urban development and densification. The City of Helsinki (n.d.) aims to add 7000 new residents, expand services, and create a more vibrant urban environment in the area. The Central Viikki zoning framework (kaavarunko) (City of Helsinki (n.d.) serves as the key planning document guiding this development. A zoning framework provides a strategic vision for future land use and urban structure and functions as an intermediate plan between the city-wide master plan (yleiskaava) and detailed local plans (asemakaava). Although the framework itself is not legally binding or subject to appeal, it plays a critical role in shaping how development proceeds and helps integrate ecological considerations early in the planning process (City of Helsinki, n.d.). In this study, the zoning framework serves as the central planning material through which potential development scenarios and their ecological implications are analyzed.

Viikki is an ecologically important area as together with Vanhankaupunginlahti bay, it forms the largest nature conservation area in Helsinki and one of the most significant urban nature sites in Europe (Helsinki City Planning Department, 2016). The Vanhankaupunginlahti bird wetland Natura 2000 site (316 ha) is protected under the EU Habitats and Birds Directives (SCI & SPA) and the Nature Conservation Act. The site is also safeguarded under the EU Water Framework Directive, with bird populations as the main conservation focus (Helsinki City Planning Department, 2015). The diverse habitats of Vanhankaupunginlahti, including wetlands, shoreline forests, meadows, and fields, have historically supported a rich breeding bird community. Long-term bird monitoring since the 1940s indicates population fluctuations, with declines between 1940–1970 followed by a recovery in the 1990s and 2000s, driven by water quality and land-use changes (Helsinki City Planning Department, 2015).

According to the Natura Assessment for the New Helsinki Master Plan (Helsinki City Planning Department, 2015), the population within 1 km of the Natura 2000 site is expected to grow by 23,000 residents by 2049. While the development is primarily infill construction, it is expected to impact farmland, open landscapes, and migratory waterbirds, according to the assessment. However, the Natura 2000 area itself will not be directly affected, and the remaining fields should still support migratory birds and species reliant on open landscapes, according to the assessment (Helsinki City Planning Department, 2015). While increased recreational use may put pressure on Natura-area edge habitats, the ecological integrity of Vanhankaupunginlahti and Viikki's wetlands is expected to remain functionally intact, ensuring compliance with conservation laws. The key concern remains the long-term connectivity between Viikki's fields and wetlands, which is critical for several bird species (Helsinki City Planning Department, 2015). While Central Viikki is not part of the nature conservation areas, which are mainly located along the southern coastline, parts of Central Viikki meet the Important Bird and Biodiversity Area (IBA) criteria (BirdLife Finland, 2024). This ecological significance of Viikki highlights the importance of understanding how densification may impact local biodiversity and how urban planning could mitigate the potential negative consequences.

Sustainability efforts are often evaluated at the city level, which is reasonable from a governance perspective and valuable from ecological perspective, but it risks overlooking important local-scale biodiversity patterns. Gupta et al. (2012) emphasize the need for neighborhood-level analysis, as green space availability and quality can vary significantly across urban gradients. For example, in Helsinki city-level tree cover averages 32%, but the proportion varies widely across neighborhoods—from less than 5% to over 65% (Kinnunen, 2023). Since urban planning decisions—and possible biodiversity loss—are typically implemented at the neighbourhood level through zoning, it is essential to assess sustainability indicators and integrate biodiversity considerations at this local scale early in the planning process, rather than addressing them reactively at later stages (Hohti et al., 2022). Zoning plays a key role in shaping urban development and determining how sustainability goals translate into practical outcomes. Careful assessment of ecological values before development is also key part of ecological compensation, a tool likely to become more common in the future (Hohti et al., 2022). Ecological compensation involves offsetting unavoidable habitat loss by improving or restoring nature elsewhere (Ministry of the Environment, 2024). However, such measures can only be effective if guided by ecological data from the start. Although numerous studies have examined the relationship between birds and land cover metrics, it is important to investigate these dynamics in cities located in boreal regions. Seasonal variation in boreal climate can influence species behavior and community composition in ways that differ from patterns observed in temperate or tropical cities (MacGregor-Fors, 2022). Moreover, medium-sized boreal cities remain relatively understudied compared to larger cities in temperate regions (García-Arroyo et al., 2025). As Viikki undergoes urban densification, balancing urban development needs with ecological sustainability is important.

2 Aims of the study

The overarching aim of this study is to explore how tree cover and built cover shape bird diversity at the neighborhood scale in Viikki, Helsinki, and how this knowledge can support biodiversity considerations in urban development planning. Specifically, I examine how the two urban land cover metrics are related to bird species richness and abundance. Additionally, I explore the bird community composition in relation to land-use type. Lastly, I aim to compare different plausible urbanization scenarios and their potential effects on the bird community. The study focuses on the diurnal landbird communities of Central Viikki, where most future urban development is planned, while also including data from the nearby southern area to capture variation across the local landscape.

I aim to answer the following specific research questions:

a) *How are bird species richness and abundance related to tree cover and built cover in Central Viikki?*
b) *How might urban development in Central Viikki affect local bird communities under different development scenarios?*

Based on previous research (see Section 1.5), I expect that areas with higher tree cover will support higher bird species richness, whereas areas with higher built cover will be associated with lower species richness. In contrast, I anticipate bird abundance to be higher in sites with greater built cover, reflecting the dominance of urban-adapted species in more urbanized environments (see Section 1.4). I also expect that different bird groups, such as urban dwellers and urban avoiders, respond differently to urban densification.

3 Methods

3.1 Study area

This study was conducted in Viikki (60° 13′ 30.18″ N, 25° 1′ 1.52″ E), a district in Helsinki. Helsinki is located in southern Finland and lies within the hemiboreal vegetation zone. Situated approximately eight kilometers northeast of the city center, Viikki is geographically central and known for its unique combination of cultural-historical agricultural landscapes and natural areas (City of Helsinki, 2022). Viikki is recognized as an ecologically significant area for birdlife (Helsinki City Planning Department, 2016). Parts of the area, including zones covered by the current zoning framework, qualify as Important Bird and Biodiversity Areas (IBA) according to the criteria set by BirdLife International. IBA sites are globally acknowledged for their importance in conserving bird populations and biodiversity. Viikki qualifies for this status primarily as a breeding and staging area for wetland bird species, based on assessments using the year 2000 as a reference point (BirdLife Finland, 2024).

For the purposes of this study, the Viikki area was divided into two zones: a northern area, defined by the boundary of the zoning framework plan (outlined in black in Figure 1), and a southern area, located outside the zoning framework area. The northern area corresponds to the area covered by the current zoning framework plan and represents the core area of interest for future development. However, as this area includes relatively few natural sites, a southern area was added to include more natural habitats, such as forest patches. This expanded the range of land cover types—particularly increasing variation in tree cover—and allowed for a more robust assessment of how habitat characteristics influence bird diversity across Viikki.

The northern Viikki zone is characterized by low-density residential development and the University of Helsinki campus. The area features a mosaic of built-up land, fields, and various types of urban green space. The southern Viikki zone consists mainly of open agricultural fields, forest patches, an arboretum, and wetland areas that are integral to Helsinki's urban green network (City of Helsinki, 2022). This zone includes the Viikki-Vanhankaupunginlahti Nature Reserve, the largest in Helsinki, which plays a vital role in supporting nesting and migratory bird populations (City of Helsinki, 2023). The area also hosts the Viikki Research Farm, managed by the University of Helsinki, which is used primarily for agricultural and forestry science research (University of Helsinki, 2024).

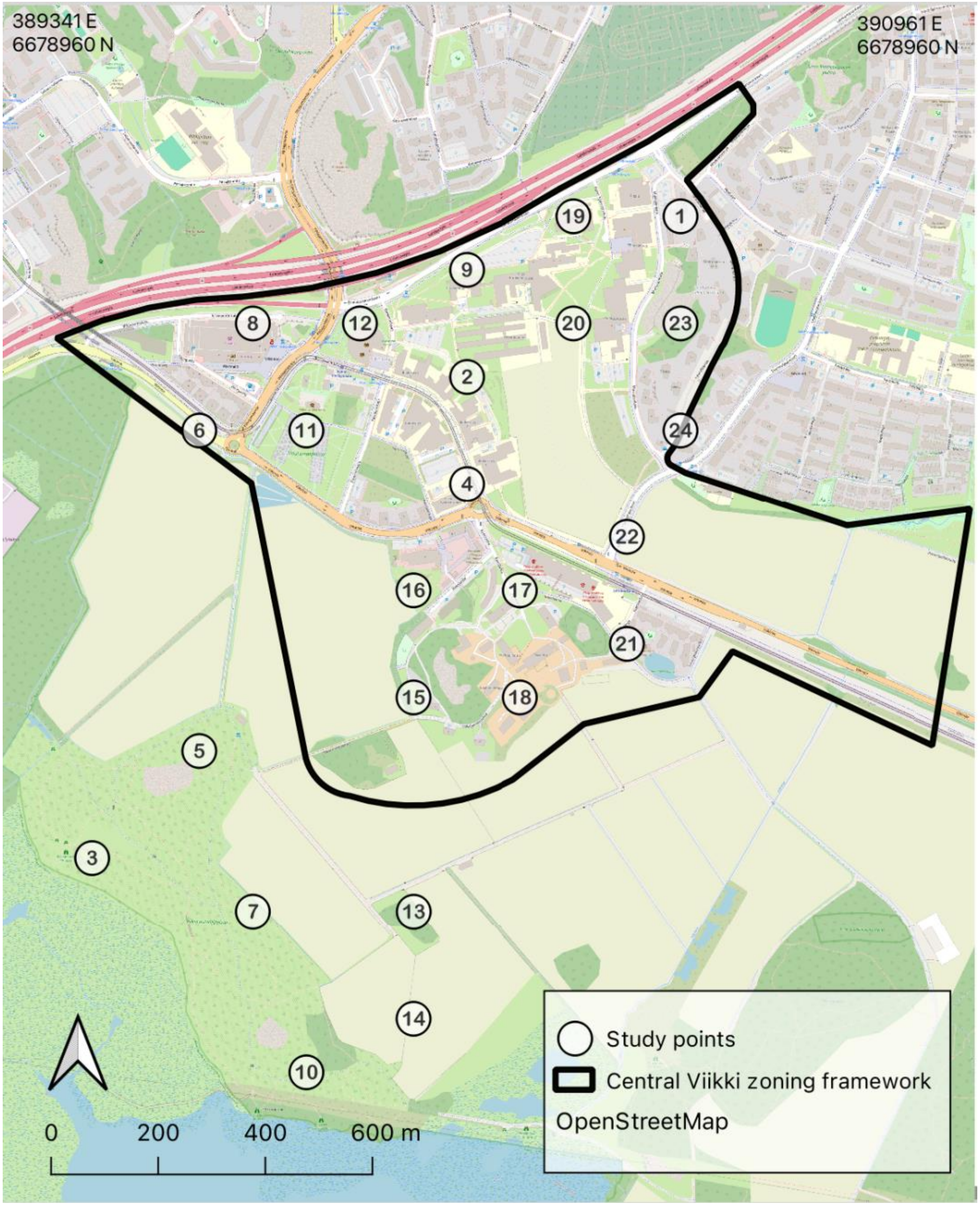

Figure 1. Map of the study area Viikki, Helsinki. The map shows 24 study points and the Central Viikki zoning framework. The base map is from OpenStreetMap. Coordinate

system: ETRS89 / ETRS-TM35FIN (EPSG:3067), coordinates shown in meters (Easting, Northing).

3.2 Bird survey

This study employed a standardized point count method to survey land birds, following bird monitoring guidelines (Hildén et al., 1991; Gregory et al., 2007). The aim was to assess both species richness, defined as the number of unique bird species observed per site, and abundance, defined as the total number of individual birds recorded at each site. Abundance was calculated as the cumulative number of individuals observed across three visits to each site. As repeated sightings of the same individuals may have occurred, the abundance measure reflects general bird activity and detectability, rather than absolute population size.

The survey was conducted across 24 study sites. Sites were selected using a regular sampling method (Gregory et al., 2007), designed to provide even coverage of the study area. The area was divided into one-hectare grid cells with study points located at the center of each square. To avoid double-counting, study points were placed a minimum of 200 meters apart. Due to the urban nature of the area, some sites were excluded because they were inaccessible (e.g., private property or traffic infrastructure). The sites covered different land-use types including residential, recreational, commercial, agricultural, and different types of greenspaces. Of the final 24 study points, 18 were located within the northern zoning framework area and 6 in the southern area.

Bird surveys were conducted over a 10-day period between July 15 and July 24 in 2024. Each site was visited three times by the author. Surveys were carried out in the early morning, beginning at sunrise (approximately 4:00 a.m.) and ending by 7:00 a.m., when bird activity is typically highest and human disturbance is minimal. All surveys were conducted during suitable weather conditions, avoiding heavy rain or strong winds. Birds seen or heard within the site boundaries were recorded. To maximize species detection while avoiding double-counting individuals, each site was surveyed for a total of 10 minutes using two consecutive 5-minute point counts, following MacGregor-Fors (2022). Birds flying high overhead were excluded, as the focus was on individuals actively using the site for foraging, nesting, or perching.

3.3 Tree and built cover quantification

The tree and built cover of the study area were quantified by using QGIS software (QGIS.org, 2024). Polygons representing tree and built cover were manually digitized from high-resolution Google Maps satellite imagery (Google, 2024), where trees and built structures were clearly distinguishable. After digitizing, the polygons were intersected with grid squares using the intersection tool in QGIS. The surface area of each polygon within each grid square was calculated using QGIS's built-in area calculation tool, with results presented in both square meters and as a percentage of the grid cell. This method allowed for precise estimation of built and tree cover within the study area.

In this study, built cover refers to infrastructure that directly covers the land surface, including buildings, roads, parking lots, and other artificial structures composed of materials such as asphalt or concrete. Gravel roads were included as built cover, as they are constructed surfaces that inhibit vegetation growth, relevant to biodiversity-related considerations in this study. Tree cover represents the coverage provided by tree foliage from an aerial perspective. This was estimated based on satellite imagery. Thus, the height of trees included in the calculation was not exact, but as a rule of thumb the height limit of more than two meters were used. Previous field observations and familiarity with the landscape helped in distinguishing tree cover in complex spots.

3.4 Land-use categorization

Each bird observation site was assigned to one of four habitat categories: urban, park, forest, or agricultural. The classification was based on a combination of field observations at site, aerial imagery and zoning material, focusing on dominant land use and landscape structure. While the categorization is somewhat subjective, it aimed to reflect ecologically meaningful differences between sites relevant to bird communities.

3.5 Scenario modelling

To explore how different urban development choices might affect bird diversity in Central-Viikki in the future decades, I created four hypothetical land-cover scenarios varying in tree cover and built cover: *Forest City, Moderate Urbanisation, Threshold,* *High Urbanization and Concrete Jungle.* Each scenario was defined by percentage values for tree cover and built cover. *High Urbanization, Threshold,* and *Moderate Urbanization* are based on plausible planning outcomes (based on Helsingin kaupunki (2023), Helsingin kaupunkiympäristö (2023) and Architect Johanna Mutanen personal communication, September 19, 2024). *Concrete Jungle* and *Forest City* are based on theoretical extremes.

Table 1. Hypothetical urban development scenarios for the Central Viikki

Scenario Name	**Built Cover (%)**	**Tree Cover (%)**	**Description**
Concrete Jungle	100	0	Extreme densification; no remaining green cover
High Urbanization	70	5	Highly urbanized environment with minimal vegetation
Threshold	60	14	Between high and moderate urbanization with 14% tree cover
Moderate Urbanization	50	30	Urban development with increase in tree cover
Forest City	0	100	Hypothetical green scenarios with no built infrastructure
Current Situation	36	11	Represents present-day land-use conditions in Central Viikki

3.6 Statistical analysis

I performed various statistical analyses to understand the effects of tree and built cover on bird communities in the Viikki area. I started with simple models to visualize the bird data and to assess the relationships and continued with more complex models to assess the possible changes in bird populations in different urbanization scenarios. Species richness and total bird abundance were first summarized per study site. Descriptive statistics were calculated, and boxplots were used to visualize differences across landscape. All statistical analyses were conducted in R (Version 2024.09.1+394). To assess the effects of land cover on bird communities, linear models (LM) and generalized linear models (GLM) were used to test the relationships between tree cover, built cover, and the response variables: species richness and abundance. Model assumptions were tested and met.

To explore differences in bird community composition between sites, I applied non-metric multidimensional scaling (NMDS) using the 'metaMSD' function of the 'vegan' R-package (Oksanen, 2025). NMDS is an ordination technique that arranges sites in a low-dimensional space based on their species composition, preserving the rank order of dissimilarities between sites (Oksanen, 2025). Bray–Curtis dissimilarity was used as the distance measure. The resulting ordination was visualized with 'ggplot2' and 'ggrepel', producing two complementary NMDS plots: (1) site scores with land use type coloring and site numbers, (2) species scores with species names with font size scaled to species abundance. These visualizations were used to explore patterns of bird community composition in relation to land use.

Additionally, a classification and regression tree (CART) analysis (Breiman et al., 1984) was used to identify potential threshold effects in species richness. CART is a machine learning algorithm that is increasingly applied in ecological modelling (Debeljak and Džeroski, 2011). It is a statistical method that systematically divides the data into groups based on the explanatory variables that best account for variation in the response variable. The method evaluates how well these divisions explain the outcome using deviance as a measure of variability (Crawley, 2013). Each division is based on the rank order of one predictor variable at a time, allowing the method to handle datasets with many explanatory variables, including both continuous and categorical types, as well as potential collinearity or interactions between them (Jackson and Bartolome, 2002). Compared to simpler models, CART analysis can accommodate complex, non-linear, and indirect relationships among variables, and for these reasons, CART has become a widely used tool in ecological research for identifying meaningful thresholds and patterns (Debeljak and Džeroski, 2011). In this study, regression trees were built using the 'rpart' package in R (R Development Core Team; Therneau et al., 2013).

Finally, I estimated species richness under five urban development scenarios using a Generalized Linear Model (GLM) the 'predict()' function in R. For each scenario, I specified realistic values of tree cover and built cover based on observed ranges in the dataset and potential planning outcomes (see Table 1.) While the full model—including tree cover, built cover, location, and all interactions—provided a better statistical fit (AIC = 130.86) than the simpler model (AIC = 138.25), the simpler model including only tree and built cover (rich ~ tree + built) (see Appendix 1.) was used for scenario-based predictions. This model was easier to interpret and more suitable for visualizing how changes in tree and built cover influence predicted richness. The scenarios represented different combinations of tree and built cover (see Table 1), and predicted values were visualized as bar plots and compared to the modelled current situation.

4 Results

4.1 Bird species richness, abundance, and community composition

A total of 48 bird species were recorded during the survey (see Appendix 2. for a full list of species). The most frequently observed species across the study sites were Eurasian Blackbird (*Turdus merula*), detected at 21 sites, White Wagtail (*Motacilla alba*) at 18 sites, and Great Tit (*Parus major*) at 16 sites. In terms of total individual counts, the most abundant species were the Common Starling (*Sturnus vulgaris*) with 220 individuals and the House Sparrow (*Passer domesticus*) with 219 individuals, followed by White Wagtail (98), Great Tit (92), and Blue Tit (*Cyanistes caeruleus*) (82 individuals). Fourteen species were recorded at only a single site each. Four of the observed species are classified as endangered according to the 2019 Red List of Finnish Species (Hyvärinen et al. 2019): house sparrow (*Passer domesticus*), barn swallow (*Hirundo rustica*), European greenfinch (*Chloris chloris*), and white-backed woodpecker (*Dendrocopos leucotos*).

Overall, species richness tended to be higher at the southern sites compared to the northern ones, as shown in Figure 2. Out of all species, 30 species were recorded in the northern 18 sites and 39 species in the six southern sites. The highest number of bird species recorded at a single site was 19 species**,** observed at both site 5, which is characterized by a wooded and rocky hill adjacent to open fields, and site 13, a cow pasture with scattered trees in an otherwise open landscape. The lowest numbers of species were recorded at site 14, an open field in the southern part of Viikki with three species, and at site 8, located next to the Prisma shopping center, with four species.

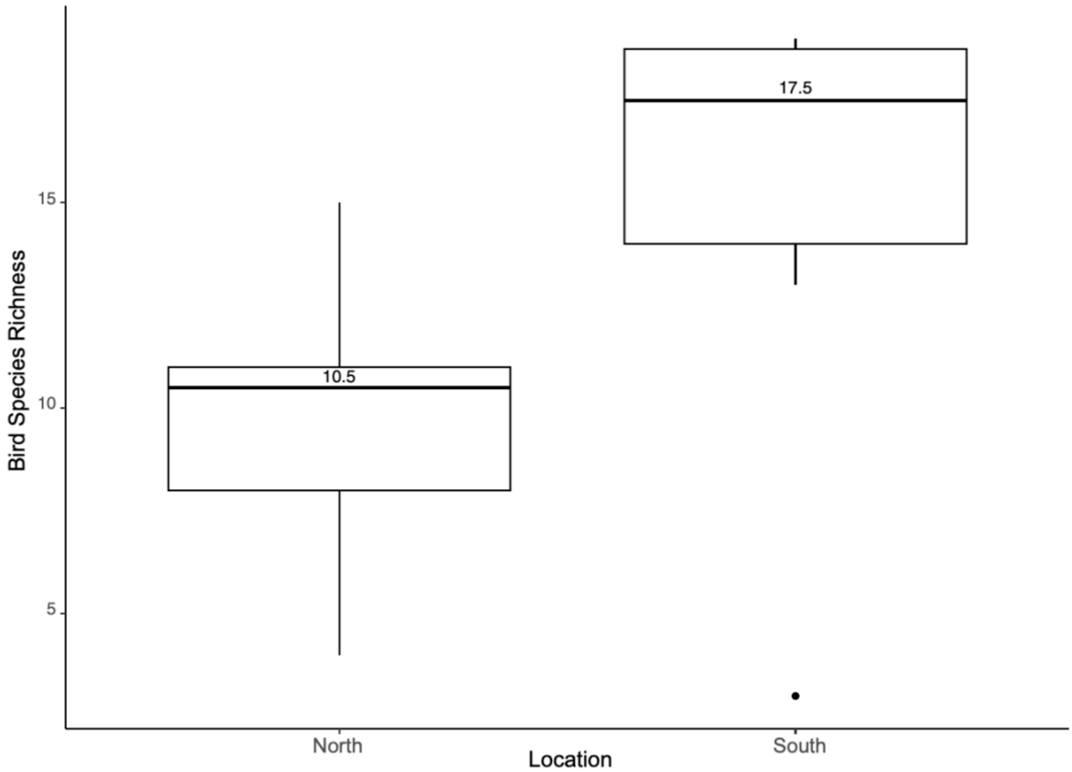

Figure 2. Box-and-whisker plot illustrating bird species richness in the northern (North) and southern (South) areas of Viikki. The southern area exhibits a higher median species richness (17.5) with less variability, while the northern area shows a lower median (10.5) species richness and a wider range of values.

To illustrate variation in bird communities across land-use types, Table 2 summarizes species richness, abundance, and the most abundant species within four habitat categories. The number of survey sites varied by habitat type: 13 urban, 5 forest, 3 agricultural, and 3 park sites. For each habitat category, the five to six most abundant species are listed. Forest sites had the highest mean species richness (14 species), while urban sites had the lowest (9 species). Agricultural sites showed the highest mean bird abundance (95 individuals per site), whereas urban sites had the lowest (38 individuals per site) mean bird abundance.

Table 2. Summary of bird species richness, abundance, and the most abundant species across habitat categories in Viikki. The complete species list with abundance values across habitat categories is available in Appendix 2**.**

Habitat category	Total number of species observed	Mean richness per site	Mean abundance per site	Most abundant species per habitat category
Urban (13 sites)	26	9	38	*Passer domesticus* *Sturnus vulgaris* *Motacilla alba* *Parus major* *Cyanistes caeruleus*
Park (3 sites)	19	13	57	*Sturnus vulgaris* *Motacilla alba* *Passer domesticus* *Parus major* *Corvus cornix*
Agricultural (3 sites)	24	11	95	*Passer domesticus* *Sturnus vulgaris* *Corvus corax* *Motacilla alba* *Hirundo rustica*
Forest (5 sites)	31	14	54	*Cyanistes caeruleus* *Parus major* *Turdus merula* *Erithacus rubecula* *Fringilla coelebs* *Spinus spinus*

Non-metric multidimensional scaling (NMDS) in Figure 3 illustrates patterns of bird community composition across the survey sites. Sites with similar species compositions are positioned closer together, while sites with more distinct communities are placed further apart. Forest sites tended to cluster more closely, indicating similar species assemblages, whereas urban sites appeared more dispersed, suggesting higher variability in community composition. Agricultural sites appeared relatively distinct in species composition, with one site (14) especially separated from others. Park sites showed intermediate patterns, overlapping with other sites and reflecting mixed species compositions. In NMDS in Figure 4, both survey sites and species scores are displayed in the same ordination space. Species scores indicate the position of species in the ordination space based on their occurrences and abundances across sites. This NMDS plot visualizes species abundance using label size: species that are more abundant across all surveyed sites are displayed with larger font sizes. Species positioned closer to particular sites indicate stronger associations with those sites, while species further apart reflect greater dissimilarity. Species that cluster tightly in the bottom-left area of the plot have been grouped and labeled as **Clustered forest species* to improve readability.

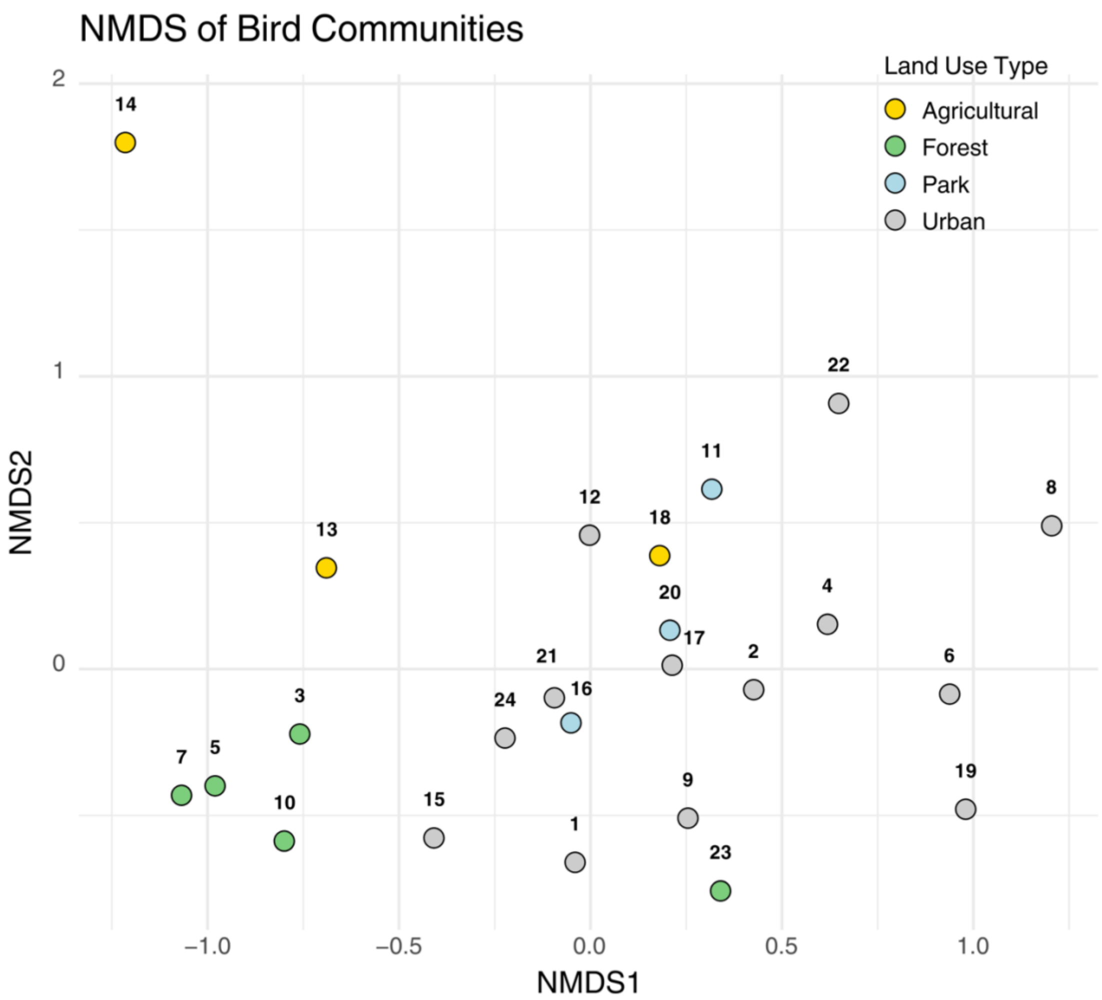

Figure 3. Non-metric multidimensional scaling (NMDS) plot showing variation in bird community composition across the survey sites. Sites are labelled by site number and colored by land-use type.

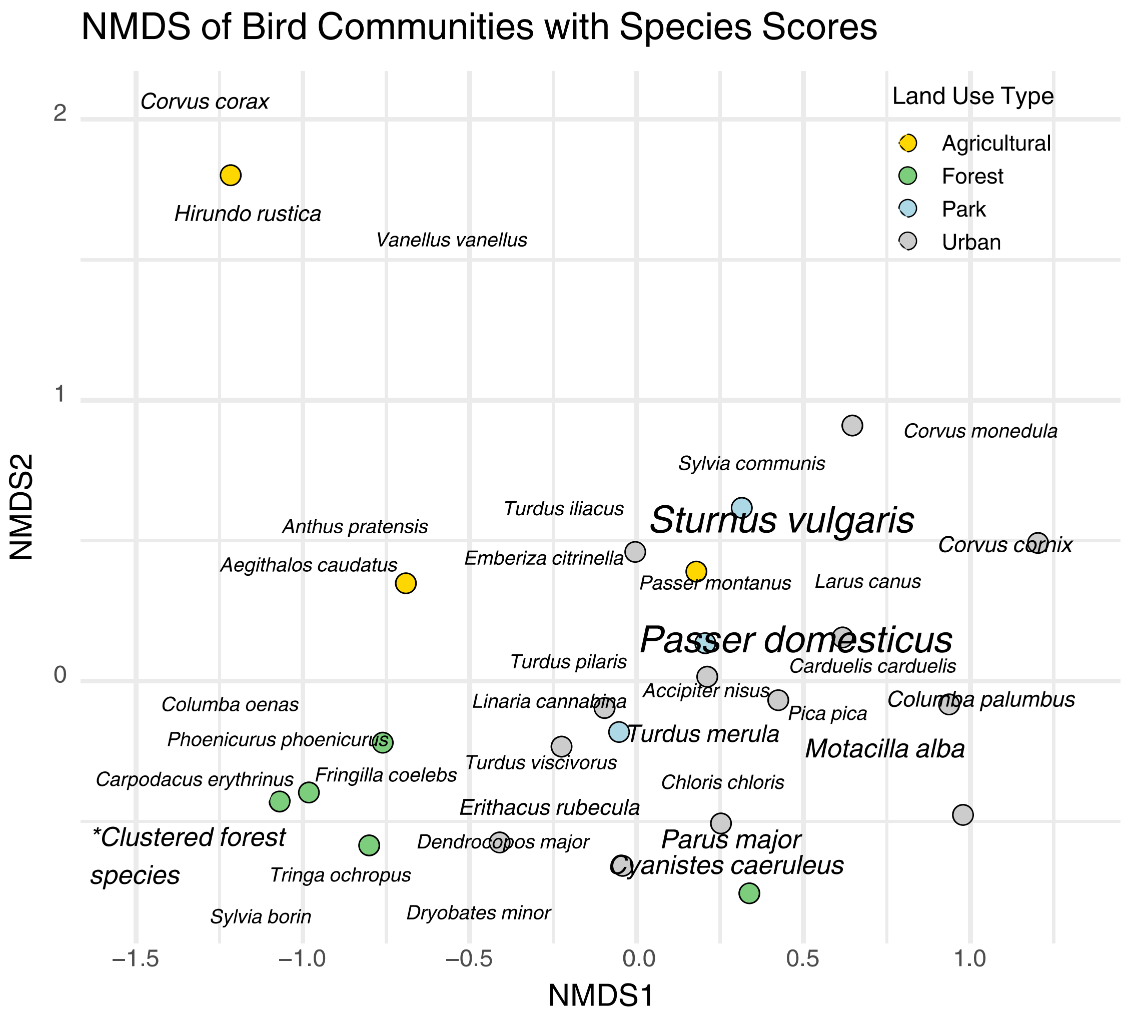

Figure 4. Non-metric multidimensional scaling (NMDS) of bird communities with species scores. In this NMDS plot, font size corresponds to the relative abundance of each bird species across all surveyed sites—larger labels indicate more abundant species. *Clustered forest species include: *Turdus philomelos, Phylloscopus trochilus, Garrulus glandarius, Periparus ater, Troglodytes troglodytes, Certhia familiaris, Luscinia Luscinia, Muscicapa striata, Sylvia articpia, Dendrocopos leucotos*

4.2 Tree and built cover in relation to bird richness and abundance

This section examines the relationships between land cover variables (tree cover and built cover) and bird species richness and abundance. Figure 5 illustrates the relationship between observed bird species richness and tree cover. As shown in Table 3, a linear model showed a significant positive relationship between tree cover and bird species richness ($\beta = 0.0010$, SE = 0.0003, $t = 3.39$, $p = 0.003$), explaining 34.3% of the variation in species richness ($R^2 = 0.343$). However, there is considerable spread in the data. Most observations are concentrated on the left side of the x-axis, indicating that a significant number of sites have tree cover below 2500 square meters. A noticeable gap exists between around 3000 and 7000 square meters of tree cover, with few observations in that range. Figure 6 illustrates the relationship between observed bird species richness and built cover. As shown in Table 4, a linear model showed a significant negative relationship between built cover and bird species richness (β = –0.00077, SE = 0.00033, t = –2.31, p = 0.031), explaining approximately 19.5% of the variation in species richness ($R^2 = 0.195$). Neither tree cover (β = –0.00051, p = 0.84) nor built cover (β = –0.00022 , p = 0.93) showed a significant relationship with bird abundance.

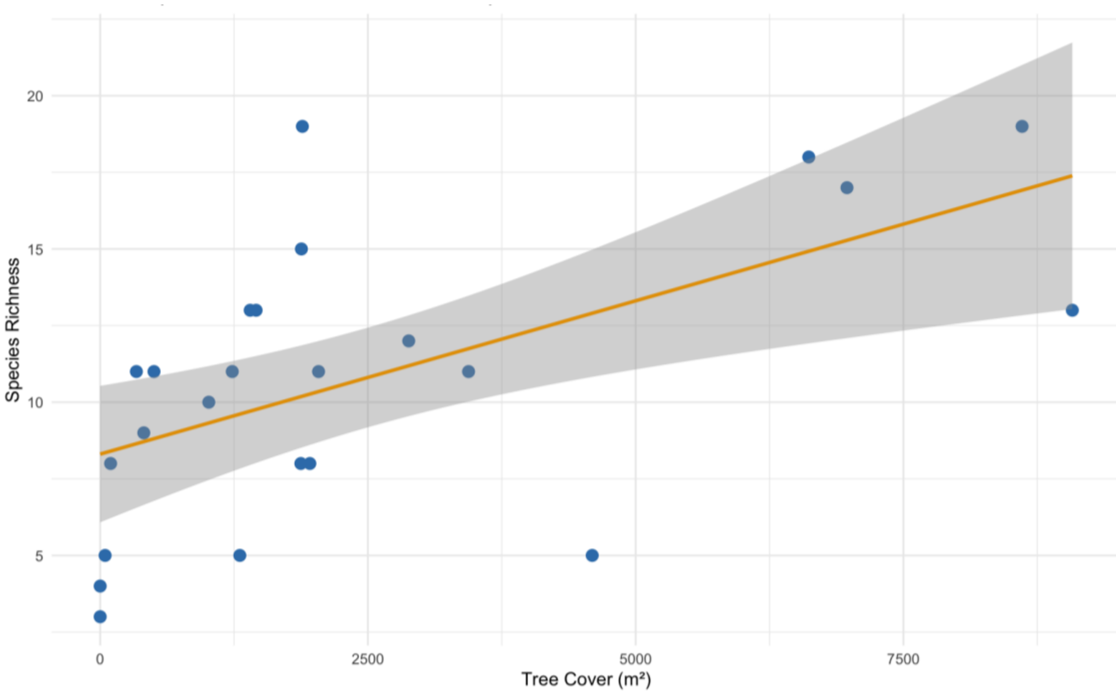

Figure 5. Relationship between tree cover and bird species richness across 24 survey sites. Points represent individual sites, and the solid line shows the fitted linear regression with 95% confidence interval (shaded). The model indicates a significant positive relationship ($R^2 = 0.34$, p = 0.003, n = 24).

Table 3. Summary of the linear model (LM) for the relationship between tree cover (m^2) and bird species richness.

Variable	Estimate	Std. Error	t-value	*P*
(Intercept)	8.31	1.07	7.74	< 0.001
Tree cover	0.0010	0.0003	3.39	**0.003**

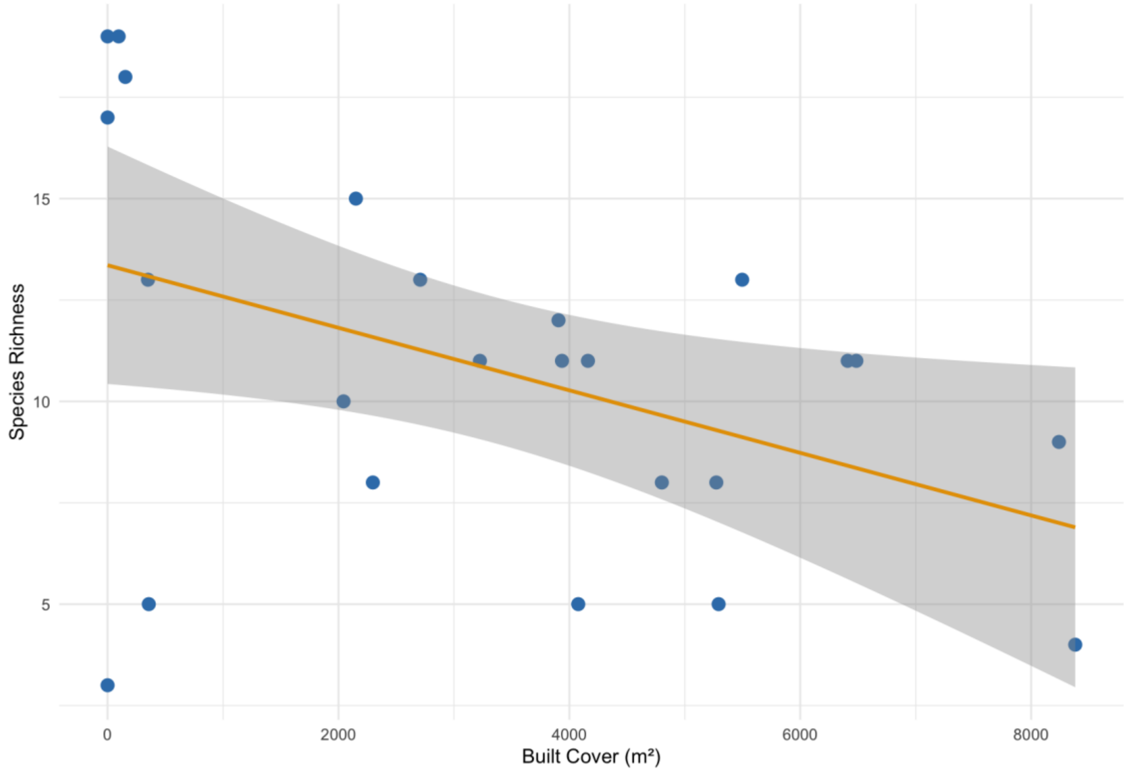

Figure 6. Relationship between built cover and bird species richness across 24 survey sites. Points represent individual sites, and the solid line shows the fitted linear regression with 95% confidence interval (shaded). The model indicates a significant negative relationship ($R^2 = 0.195$, $p = 0.031$, $n = 24$).

Table 4. Summary of the linear model (LM) for the relationship between built cover (m^2) and bird species richness.

Variable	Estimate	Std. Error	t-value	*P*
(Intercept)	13.36	1.41	9.46	< 0.001
Built cover	-0.00077	0.00033	-2.31	**0.031**

To further examine the relationships between tree cover, built cover, location (North and South), and bird species richness, I constructed a generalized linear model (GLM) with a Poisson distribution and a log-link function. The full model included all main effects and interaction terms (richness ~ tree × built × location). As shown in Table 5, tree cover had a highly significant positive relationship with species richness ($p < 0.0002$), while built cover and location alone did not have a significant relationship. However, the three-way interaction between tree cover, built cover, and location was also significant ($p < 0.0005$).

Table 5. Summary of the results of the generalized linear model (GLM) testing the relationship between tree cover, built cover, location, and their interactions, and bird species richness.

Variable	df	Resid. Deviance	*P*
Tree cover	1	33.73	**0.0002**
Built cover	1	33.13	0.4404
Location	1	32.60	0.4638
Tree cover × Built cover	1	31.83	0.3824
Tree cover × Location	1	29.16	0.1018
Built cover × Location	1	27.98	0.2790
Tree cover × Built cover × Location	1	15.74	**0.0005**

To explore potential threshold effects of tree cover on bird species richness, a regression tree model was constructed. The model identified a threshold at 1352 m² of tree cover (approximately 14% of a hectare), above which sites supported notably higher average species richness, as shown in Figure 7.

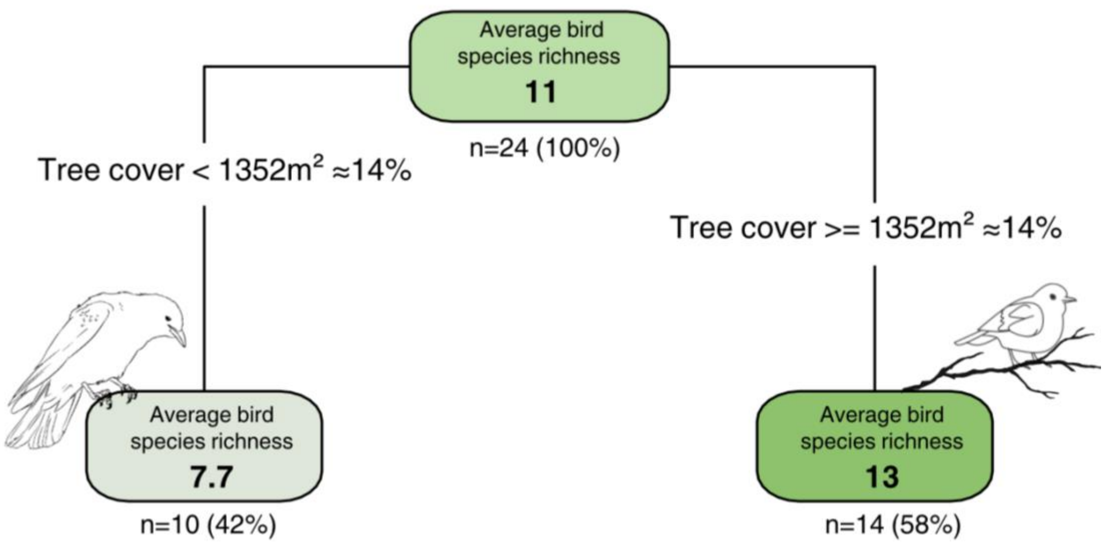

Figure 7. Regression tree showing the effect of tree cover on bird species richness. The CART model splits at a threshold of 1352 m² of tree cover (approximately 14% of a hectare study site)**,** with sites exceeding this value supporting a higher average species richness (13 species) compared to sites with less tree cover (7.7 species). The number of sites and their proportion within each group are displayed under the boxes. Illustration created by the author using Canva.

4.3 Predicted richness under urban development scenarios

Bird species richness was estimated under five hypothetical urban development scenarios for the Central Viikki zoning framework area (see Table 1) and compared to the current situation using a generalized linear model (GLM) with tree cover and built cover as explanatory variables (see Appendix 1). As shown in Figure 9, the current landscape, with 36% built cover and 11% tree cover, yielded a modelled species richness of 9.5. The "Concrete Jungle" scenario (100% built, 0% tree cover) produced the lowest richness estimate (7.5), corresponding to a 21.1% decline. In contrast, the "Forest City" scenario (0% built, 100% tree cover) yielded the highest predicted richness (18.8), a 97.9% increase compared to current conditions. Scenarios with intermediate land cover values, such as "Threshold" (60% built, 14% tree) and "Moderate Urbanization" (50% built, 30% tree), showed modest changes in species richness (9.1 and 10.4, respectively).

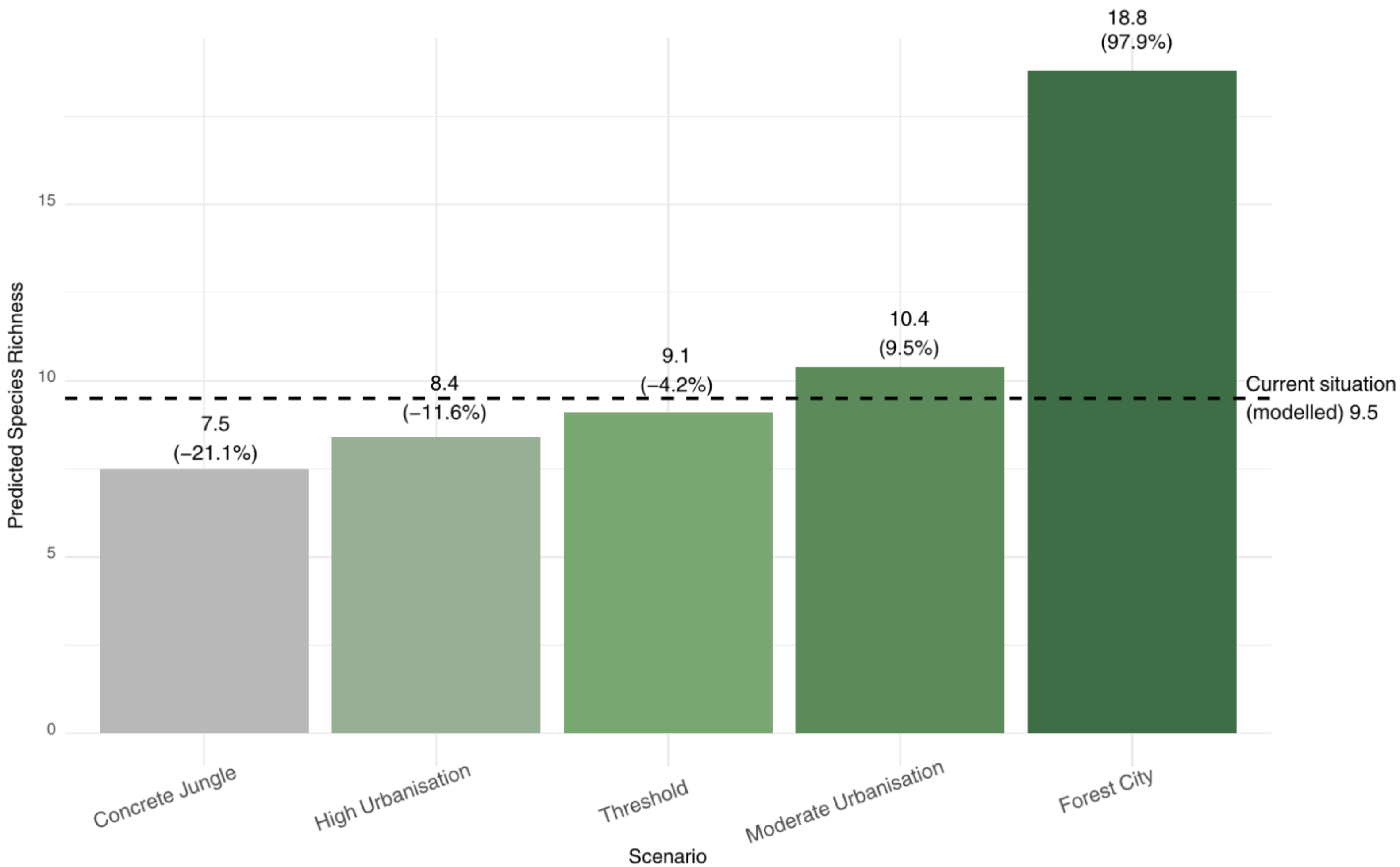

Figure 8. Predicted bird species richness for the Central Viikki zoning framework area under five urban development scenarios based on tree and built cover, using a generalized linear model (GLM) with Poisson distribution. Bars show model-predicted species richness per hectare. Numbers above bars indicate the predicted value and relative change (%) compared to the current situation. Model-predicted species richness under current land-cover situation is marked with a dashed line.

5 Discussion

This thesis aimed to assess bird diversity in Viikki in relation to tree and built cover and to evaluate how future urban development might affect local bird communities. The findings contribute to the broader understanding of biodiversity patterns in urban environments and offer practical insights for integrating ecological considerations into urban planning.

The findings, including surveys from a narrow window of time, confirm that Viikki supports a diverse bird community, with 48 species recorded, which accounts for nearly one-third of the breeding bird species in Helsinki (Helsingin kaupunki, 2019). While the most frequently observed species were common generalists such as Eurasian Blackbird (*Turdus merula*), White wagtail (*Motacilla alba*) and Great tit (*Parus major*), the total also included four species classified as endangered in Finland (Hyvärinen et al., 2019). However, as noted by BirdLife Finland (2023), focusing solely on red-listed species may overlook broader patterns of biodiversity loss, as many common species are also in decline.

I compared the Central Viikki area, with urban development plans, with sites from the southern more natural forest and agricultural areas. The comparisons (see Figure 2) showed that southern sites hosted much more species per site and in total than the northern Central Viikki. Habitat-based comparison (see Table 2) showed higher average and total species richness in forest sites. Agricultural sites had the highest overall bird abundance and a distinct species composition, including species such as ravens and barn swallows. These sites are located in the immediate vicinity of the nature conservation zone and biodiversity-rich wetlands in Viikki, which likely contributes to their observed species diversity. These patterns highlight the ecological value of Viikki's heterogeneous landscape, which comprises forest patches, fields and a farm, parks, and built-up areas. In contrast, urban sites recorded lowest at both species richness and abundance. With the planned addition of 7000 residents and significant increases in built infrastructure in Central Viikki, the landscape will likely transition from moderately to highly urbanized following the categorization by MacGregor-Fors

(2011). This development underscores the importance of incorporating biodiversity considerations into urban planning.

The statistical analyses reinforce the significance of tree cover as a key factor influencing bird species richness. The positive relationship between tree cover and species richness aligns with my hypothesis and previous studies (Radford et al., 2005; Trollope et al., 2009; MacGregor-Fors et al., 2013; Villaseñor et al., 2021). Non-metric multidimensional scaling (NMDS) was used to explore variation in bird community composition between the survey sites (Figures 3 and 4). The NMDS ordination of site scores revealed that forest sites tended to cluster more closely together, indicating relatively similar species assemblages dominated by woodland-associated species, such as woodpeckers, and hole-nesting species, which often rely on higher tree densities and mature trees (Sandström et al., 2006; Ferenc et al., 2014). MacGregor-Fors et al. (2013) and Jokimäki et al. (2014) also highlight the importance of tree age and structure. During the digitization process, it was noted that mature trees contributed disproportionately to overall canopy area. This emphasizes the need to prioritize the preservation of existing large trees, as newly planted trees may take decades to deliver comparable ecological benefits.

A regression tree analysis (see Figure 7) identified a clear threshold of 1352 m^2 (~14% of a one-hectare study site). Above this threshold, species richness increased notably and correspondingly decreased with lower tree cover. This finding is in line with earlier research, such as Radford et al. (2005), who identified a 10% tree cover threshold for bird species richness, and Konijnendijk (2022), who proposes a 30% tree cover target at the neighborhood scale. These results suggest that a minimum of 14% tree cover may be necessary to support local bird diversity In Central Viikki, while aiming for 30% could yield even greater ecological benefits. At present, tree cover in Central Viikki is approximately 11.5%, suggesting that increasing tree planting could help meet these thresholds and support higher species richness. Additionally, mainting and increasing also other stuctural vegetation, such as shrubs and meadows, is important for many species (e.g. Evans et al. 2009, Beninde et al., 2015).

Built cover showed a statistically significant negative correlation with species richness as I hypothesized, but the relationship was weaker than that of tree cover. The highest richness was observed at sites with no built cover, yet not all non-urban sites were species-rich, and some urban sites had relatively high species richness emphasizing that factors beyond land cover—such as habitat structure, food availability, and green space size (Evans et al., 2015; Plummer et al., 2020)—also influence bird communities. These findings support broader urban ecological theory suggesting that urbanization often reduces species richness (McKinney, 2002, 2008; Lepczyk et al., 2008), though not all species respond equally. The relatively weak relationship between built cover and species richness observed in this study reflects the mixed evidence reported in previous research. While Evans et al. (2009) found a decline in both species richness and abundance with increasing building density, Andersson and Colding (2014) observed minimal differences in bird communities across Swedish urban areas with similar tree cover, suggesting that vegetation structure may outweigh the influence of built form alone. This underlines the importance of focusing not solely on built density but also on the quality and composition of green infrastructure within urban environments.

Contrary to what I hypothesized, bird abundance did not correlate with tree or built cover. This was an unexpected result as many studies suggest that more urbanized areas tend to host higher bird abundance (Donnelly & Marzluff, 2004; Ortega-Álvarez & MacGregor-Fors, 2009; Kurucz et al., 2021). In this study, abundance was calculated as the cumulative number of individuals observed across three visits per site. This approach may reflect general bird activity rather than consistent site-level abundance. This study was conducted at a local scale within a moderately urbanized area, with few highly urbanized sites and a notable absence of common urban species such as feral pigeons (*Columbia livia domestica*), which may also help explain why this pattern was not observed. Nonetheless, the NMDS of study sites and species scores (Figure 4), illustrated that only a few species—notably common starling (*Sturnus vulgaris*), house sparrow (*Passer domesticus*) and white wagtail (*Motacilla alba*)—occurred frequently and in high numbers especially in the more urban sites, which is consistent with patterns observed in other studies of urban bird communities dominated by few generalist species (Clergeau et al., 2006; Kurucz et al., 2021 MacGregor-Fors et al., 2013).

Scenario modelling further illustrated how land-use changes might affect bird diversity in Central Viikki (Figure 8). Predictions based on a simplified GLM of tree and built cover (see Appendix 1.) indicated that increasing built cover generally leads to a decline in species richness, while increasing tree cover has a mitigating or positive effect. For example, a hypothetical scenario “Concrete jungle” with 100% built cover and no trees was associated with a predicted richness reduction of over 21%, while a more realistic “High Urbanization” scenario (70% built cover, 5% tree cover) still resulted in a predicted decline of nearly 12%. On the contrary, “Moderate urbanization” scenario (50% built, 30% tree cover) with higher built cover than the current one but also higher levels of tree cover showed potential for even slightly enhancing species richness. These findings underline the importance of maintaining tree cover in urban development to support biodiversity.

Changes in species composition are also expected as urbanization progresses and species observed in Viikki reflect varying levels of urban tolerance. Following the categorization by Fischer et al. (2015), generalists such as House sparrow and Common starling can be considered urban dwellers. These species are well-adapted to human-dominated environments (Aronson et al. 2014) and are expected to remain common under moderate or even high urban development. In contrast, many species like Coal tit (*Periparus ater*), Eurasian Treecreeper (*Certhia familiaris)*, and White-backed woodpecker (*Dendrocopos leucotos*), which were only observed in forest-type sites, likely represent urban avoiders that depend on more continuous, undisturbed habitats. There were notable number of forest dwelling species as illustrated in Figure 4. However, most of the larger forested areas lie outside the current Central Viikki zoning framework, meaning that direct impacts from development are most likely limited. However, indirect effects, such as increased recreational pressure, edge effects, and reduced connectivity, can still threaten these species. Maintaining or expanding tree cover and taking care of connectivity between green areas may help mitigate these impacts, as also suggested by MacGregor-Fors et al. (2013).

It is important to note that this study focused primarily on land bird species, as the surveyed sites did not include significant water bodies. Consequently, waterbirds and wetland-associated species—groups for which Viikki is particularly important—were not represented in the data. Future urban planning and biodiversity strategies for Viikki should also consider the needs of these species. Furthermore, the observed patterns mainly reflect species associated with terrestrial habitats, and the desirability of promoting more woodland bird species in urban areas ultimately depends on broader conservation and planning goals. Notably, woodland species are often favoured by people due to their aesthetic qualities while urban species tend to be more prone to conflicts with humans (Pejchar et al. 2025).

While this study focused on tree and built cover as key variables, they explained the variation in species richness only to a limited extent. For example, patch size and connectivity are also critical for supporting urban bird communities (Plummer et al., 2020; MacGregor-Fors et al., 2013). Several studies recommend minimum patch sizes ranging from 10 to 100 hectares to maintain breeding bird populations (Lepczyk et al., 2017; Fernández-Juricic & Jokimäki, 2001; MacGregor-Fors et al., 2013). While tree cover and built cover can serve as useful indicators of ecological sustainability due their easy usability and association with biodiversity, it is important to consider also other variables when assessing sustainability and ecological effects of urban planning.

While the study offers important insights into urban bird diversity, several limitations should be considered when interpreting the results. The results are closely tied to the specific landscape context of Viikki, which may limit their generalizability to other urban areas. Other important environmental factors—such as green space connectivity, vegetation structure, food availability, and human disturbance—were not included in the models. Moreover, the GLM predictions should be interpreted only as indicative trends, not precise forecasts. The bird survey was conducted in a single season during July, potentially leading to under-detection of early breeders and excluding migratory species that utilize agricultural fields as stopover sites (Ellermaa, 2018). Methodological limitations include the relatively small sample size, possible observer bias (Gregory et al. 2007), and manual digitization of land cover data, which may introduce some subjectivity.

The findings of this study suggest several important directions for future research. Longitudinal studies monitoring bird communities in Central Viikki would provide deeper insights into how urban development impacts bird diversity and overall biodiversity over time. The results of this study could serve as comparative baseline for future studies. Studies spanning the full timeline of development—including the pre-construction, construction, and post-construction phases—would allow researchers to capture both immediate disturbances and longer-term community responses to habitat changes. Surveys could be conducted across multiple seasons to include also migration and wintering birds. Broadening the focus beyond birds to include other taxa, such as insects, amphibians, or mammals, would offer a more comprehensive understanding of urban biodiversity dynamics and how different species groups respond to urbanization pressures.

While this study confirmed the importance of overall tree cover, more detailed investigations could clarify which tree features, such as species or age, most effectively support diverse bird communities and other taxa. In addition, examining how three-dimensional urban form, including building height, façade design, and the incorporation of bird-friendly architectural features, influences bird movement and collision risks would provide valuable insights for urban biodiversity management. More generally, future studies should extend beyond traditional green spaces, such as parks and forests, to investigate biodiversity within built-up urban environments, which are expanding rapidly in many cities. Furthermore, it would be important to investigate how urban greenery and bird diversity contribute to broader ecosystem services, including human well-being, and how such wider benefits can strengthen the case for biodiversity-friendly urban planning. An important question for future work is not only how much biodiversity urban areas can support, but also what types of biodiversity are prioritized and why. Clarifying these goals could help guide planning and conservation efforts more effectively.

Finally, integrating ecological research with urban planning tools, such as zoning data, spatial modelling, and scenario-based predictions, would strengthen the ability to anticipate biodiversity outcomes under different development strategies and make informed decisions. Birds are valuable indicators of biodiversity and monitoring their responses to land-use change can inform sustainable urban planning strategies. By building on these directions, future research can help bridge the gap between urban ecological knowledge and practical land-use planning, ensuring that biodiversity considerations are embedded early in the design and development of cities.

6 Conclusions

This study examined how tree cover and built cover are related to bird diversity in the Central Viikki area and explored how future urban development might affect local bird communities. The results showed that tree cover had a strong positive relationship with bird species richness. Tree cover exceeding 14% per site was associated with substantially higher richness, while sites below this threshold showed a notable decline. Built cover was negatively associated with species richness, but the relationship was weaker than that of tree cover. No clear relationship was found between land cover and bird abundance.

Scenario-based predictions indicated that urban densification in Central Viikki—with increasing built cover—would likely reduce bird species richness unless sufficient tree cover is maintained. Maintaining and increasing tree cover may mitigate losses in bird diversity even in densely built urban settings. Species composition is also expected to shift under urban development, with generalist urban-dweller species becoming more dominant, while forest-associated urban-avoiders may decline. If the goal is to promote diverse bird populations in Central Viikki, planning should prioritize the retention of mature trees, planting new trees and the integration of vegetative cover within the built environment. Aiming for a minimum threshold of 14% tree cover may serve as a practical benchmark for biodiversity-friendly urban development.

Future research could build on these findings by conducting studies across different phases of urban development—before, during, and after construction—to monitor changes in bird communities over time. Expanding the scope to other taxonomic groups, such as insects, amphibians, or mammals, would also provide a more comprehensive picture of urban biodiversity dynamics. Integrating ecological research with urban planning tools, such as spatial models and scenario-based predictions, would help anticipate biodiversity outcomes under different development strategies and support evidence-based urban planning.

7 Ethical considerations

All bird surveys were conducted with care to minimize disturbance to birds and their habitats, particularly during the sensitive breeding season. Observations were carried out quietly and at a distance, avoiding any disruption to nesting or feeding behavior. The study involved no capturing, handling, or marking of birds. The methods followed standard guidelines for bird surveys (e.g. Hildén et al. 1991, Gregory et al. 2007).

This study was conducted in accordance with the guidelines for the responsible conduct of research issued by the Finnish National Board on Research Integrity (TENK).

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Appendices

APPENDIX 1. Results of the Generalized Linear Model testing the relationship of tree cover and built cover on bird species richness.

Variable	df	Resid. Deviance	*P*
Tree cover	1	33.728	**0.0002**
Built cover	1	33.133	0.4404

APPENDIX 2. Full list of observed bird species and their abundances across different land-use types.

Species name	**Habitat category** Agricultural	Forest	Park	Urban	Total
Accipiter nisus				1	1
Aegithalos caudatus	3				3
Anthus pratensis	2				2
Carduelis carduelis	1	1	8	6	16
Carpodacus erythrinus		1			1
Certhia familiaris		4			4
Chloris chloris	1	4	3	11	19
Columba oenas	1	1			2
Columba palumbus	14		3	13	30
Corvus corax	35	4			39
Corvus cornix	1		10	27	38
Corvus monedula				13	13
Cyanistes caeruleus		38	7	37	82
Dendrocopos leucotos		1			1
Dendrocopos major	2	6		3	11
Dryobates minor				1	1
Emberiza citrinella	7				7
Erithacus rubecula	1	17	2	11	31
Fringilla coelebs	1	11	2	5	19
Garrulus glandarius		1			1
Hirundo rustica	27				27
Larus canus			6	7	13
Linaria cannabina	2	1	3	5	11
Luscinia luscinia		1			1
Motacilla alba	28	5	25	40	98
Muscicapa striata		8			8
Parus major	6	33	14	39	92
Passer domesticus	79		15	125	219
Passer montanus	1	1		19	21
Periparus ater		2			2
Phoenicurus phoenicurus		1			1
Phylloscopus collybita		6			6
Phylloscopus trochilus	1	9			10
Pica pica			8	6	14
Regulus regulus		5		1	6
Spinus spinus		11			11
Sturnus vulgaris	65		48	107	220
Sylvia atricapilla	1	8	1		10

Sylvia borin		2			2
Sylvia communis			2		2
Tringa ochropus		1		1	2
Troglodytes troglodytes		3			3
Turdus iliacus	1	1	2	1	5
Turdus merula	3	22	8	32	65
Turdus philomelos		8			8
Turdus pilaris			5	12	17
Turdus viscivorus				1	1
Vanellus vanellus	1			3	4
Total	**284**	**217**	**172**	**527**	**1200**