# Patterns of tree diversity, composition, and structure across public and private land use types

Emma Sandhu Bacon

A Thesis in The Department of Biology

Presented in Partial Fulfillment of the Requirements
For the Degree of Master of Science (Biology) at
Concordia University
Montreal, Quebec, Canada

December 2025

CONCORDIA UNIVERSITY

**School of Graduate Studies**

This is to certify that the thesis prepared

By: Emma Sandhu Bacon

Entitled: Patterns of tree diversity, composition, and structure across public and private land use types

and submitted in partial fulfillment of the requirements for the degree of

**Master of Science (Biology)**

complies with the regulations of the University and meets the accepted standards with respect to originality and quality

Signed by the final Examining Committee:

______________________________________ Chair
*Dr. Emma Despland*

______________________________________ Examiner
*Dr. Marie-Julie Favé*

______________________________________ Examiner
*Dr. Emma Despland*

______________________________________ Examiner
*Dr. Danijela Puric-Mladenovic*

______________________________________ Supervisor
*Dr. Alain Paquette*

______________________________________ Supervisor
*Dr. Carly Ziter*

Approved by ______________________________________
Dr. Robert Weladji, Graduate Program Director

______________________________________
Dr. Pascale Sicotte, Dean of Faculty

# ABSTRACT


Patterns of tree diversity, composition, and structure across public and private land use types

Emma Sandhu Bacon

As cities expand tree planting to promote benefits to residents, they require strong baseline knowledge on urban forest conditions. Historically, public tree databases have been used to assess urban forest composition, structure and diversity, yet recent studies suggest trees on private land can differ in important ways from public trees, influencing urban forest benefits and resilience. We used a city-wide network of plots spanning multiple urban gradients to assess six private and public land use types (residential, public right-of-way, institutional, commercial, parks (excluding woodlands), vacant lots) and asked how urban forest composition and structure differed. Our study presents one of the most complete urban forest inventories and analyses for a large North American city, capturing the diversity, structure, and composition of trees across multiple land use types. We found strong differences in tree abundance, size, and diversity among land use types, reflecting contrasting management practices and planting histories. Hedges emerged as a dominant feature across all land use types – but most prominently on residential land – contributing a substantial proportion of trees, particularly *Thuja occidentalis*. Residential land and public rights-of-way supported the greatest tree abundance and species richness, while parks contained larger trees and greater basal area. Despite overlapping species, dominance patterns differed across land use types. Our work emphasizes that public green spaces are insufficient to represent citywide urban forest composition and structure, and private green spaces must be considered to better understand the urban forest. It also reveals that overall, the urban forest is much more diverse than previously thought.

## Acknowledgments

I would like to thank my committee members Dr. Emma Despland and Dr. Danijela Puric-Mladenovic for their thoughtful comments and feedback, as well as Elyssa Cameron for field advice that greatly improved my field season. I would also like to thank Dr. Bella Richmond, Dr. El-Amine Mimouni, and Dr. Eric Pedersen for thoughtful advice on statistical modelling, and Aaron Baxter for teaching me everything I know about geospatial software and creating beautiful maps. Thank you to Dr. Marine Fernandez for her invaluable guidance during data collection, and for conducting the initial tree inventory in the summer of 2022. I am also grateful to Sarah Tardif, Dr. Laura Schillé, Dr. Rita Sousa-Silva, Dr. Maria Faticov, and Dr. Marine Fernandez for their foundational work in the initial development of the Montreal Urban Observatory. I would like to thank my dedicated tree research assistant Raphaël Bouchard-Piché, as well as lab members Aliaksei Fiodarau, Lauren Bianco, Michael Paulauskas, Kayleigh Hutt-Taylor, Mackenzie Guest, Émily Ducharme, and Bella Richmond for supporting me in collecting tree data, and Daniel Lesieur for guiding me on keeping my large dataset well-organized. I am grateful to all my labmates in the ZULE lab and PaqLab for their guidance and encouragement throughout my thesis. Their thoughtful feedback on my writing, presentations, and research helped shape my work and made my MSc experience unforgettable.

I would like to deeply thank my co-supervisors, Dr. Carly Ziter and Dr. Alain Paquette, for their ongoing support and mentorship. Your guidance was invaluable not only in shaping my research but also in supporting me professionally and personally, helping me grow as a researcher and as a person. You both go above and beyond for your students, creating an incredibly supportive and inspiring environment.

Finally, I would like to thank my incredible support system outside of academia — my parents, Jyoti Sandhu and Shawn Webb, Léo and Tamsin Bacon; my siblings, Ash Bacon, Léo

Bacon, Vica Webb, and Imogen Bacon; my close friends in Montreal and afar; and my partner, Emile Diedericks. Your encouragement, patience, and unwavering support over the past two years have meant the world to me and helped me get through this journey. Thank you.

# Contribution Statement

The Montreal Urban Observatory – the study system for this thesis – was developed prior to the start of my graduate research, initially to sample pollens, then biodiversity in general. Its creation involved initial contributions from Dr. Rita Sousa-Silva, Daniel Lesieur, Mélanie Desrochers, Hendrick Paquette-Ambroise, Elyssa Cameron, followed with Sarah Tardif, Dr. Laura Schillé, Dr. Maria Faticov, Dr. Marine Fernandez, under the direction of Dr. Alain Paquette, Dr. Carly Ziter, and Dr. Isabelle Laforest-Lapointe, who were all involved in the design of the observatory framework and the process of site selection for the plots.

Under the supervision of Dr. Marine Fernandez, an initial tree inventory was conducted in summer 2022, surveying trees within a 100 m buffer surrounding each plot center. For this thesis, I re-surveyed all trees included in that original dataset and expanded the sampling area to a 200-m buffer around each plot center. I led the literature review and identification of clear research questions and hypotheses, adaptation of sampling methods to those new questions, data cleaning, analysis, interpretation, and writing of the thesis. Dr. Carly Ziter and Dr. Alain Paquette provided feedback throughout the research process.

# Table of Contents:

# List of Figures

**Figure 6.** Principal Coordinates Analysis (PCoA) ordination of tree species composition across six different land use types. Plot shows distances between 101 subplots based on 307 tree species, calculated using Bray-Curtis dissimilarities. Points represent subplots and are colored by land use type. The dispersion of species composition within each land use type is illustrated with 90% confidence ellipses around the centroid of each group. Axes show the first two PCoA axes, with the percentage of variance explained indicated in parentheses.

**Figure A2.1.** Principal Coordinates Analysis (PCoA) ordination of tree species composition across six different land use types. Plot shows distances between 101 subplots based on 307 tree species, calculated using Jaccard dissimilarities. Points represent subplots and are colored by land use type. The dispersion of species composition within each land use type is illustrated with 90% confidence ellipses around the centroid of each group. Axes show the first two PCoA axes, with the percentage of variance explained indicated in parentheses.

**Figure A2.2.** Average total area and plantable area across six land use types. (A) Total area (hectares), (B) Plantable area (hectares). Grey data points represent raw data values of subplots. Coloured points represent the mean value for each land use type and error bars show ±1 standard error of the mean. Each panel displays a different metric, with y-axes scaled independently.

# List of Tables

**Table A3.4.** Fixed and random effects results for mixed model analysis (zero-one inflated beta GLMM) predicting Hurlbert's PIE evenness at the subplot level (n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

**Table A3.5.** Fixed and random effects results for a linear mixed model analysis predicting log-transformed basal area (standardized by total subplot area, n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

**Table A3.6.** Fixed and random effects results for a linear mixed model analysis predicting log-transformed basal area (standardized by plantable area) at the subplot level (n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

## Land Acknowledgement

This research was conducted on the Island of Montreal, which is an ancestral and unceded territory, a place known as Tiohtiá:ke in Kanien'kéha and Mooniyang in Anishinaabemowin. The Kanien'kehà:ka Nation is recognized as the custodians of these lands and waters, although it is historically known as a gathering place for many Indigenous Peoples, including, but not limited to, the Kanien'kehà:ka of the Haudenosaunee Confederacy, Huron/Wendat, Abenaki, and Anishinaabeg. Today, it is home to a diverse population of Indigenous and other peoples.

For more information, please consult Concordia's Indigenous educational resources for faculty and students:

https://www.concordia.ca/library/guides/indigenous-fac-res.html

# Introduction

Urban green spaces are essential for both human wellbeing and the ecological functions of cities (Jabbar et al., 2022; United Nations, 2019). Trees are an important component of urban green spaces which maintain important functions and ecosystem services. Collectively, all trees in our cities make up the urban forest, spanning diverse social and ecological contexts across public and private land. For example, the urban forest regulates air temperature and improves human wellbeing (Berglihn and Gómez-Baggethun, 2021; Bowler et al., 2010; Cadotte et al., 2011; McPherson et al., 2011; Nowak et al., 2013). Given their critical role in the ecological, social, and economic functioning of cities, urban forests have become a growing focus of research, planning, and policy. However, not all urban forests are created equally – the diversity, composition, and structure of trees can vary widely across and within different cities (Anderson et al., 2021; Jenerette et al., 2016; Landry et al., 2020; Mejía et al., 2024), influencing the benefits they provide and the resilience of the urban forest to stress and disturbance.

The capacity of the urban forest to continue providing multiple services relates to its resilience (Pataki et al., 2013; Yachi and Loreau, 1999). Within the context of urban forests, resilience can have diverse interpretations, including ecological, social, and institutional aspects (Roman et al., 2021). Ecological resilience refers to the capacity of the urban forest to mitigate stressors and changes, and recover from disturbances (Huff et al., 2020). Tree functional traits and other characteristics influence the resilience of the urban forest and the quality and quantity of ecosystem services it provides (Nitoslawski et al., 2016). Different tree species have unique life-history strategies, which allow them to thrive under diverse environmental conditions (Paquette et al., 2021). Functionally diverse urban forests are less vulnerable to threats such as urban stressors, disease, and exotic competitors because there is an increased chance of some species surviving or adapting to variable conditions and stressors, thus maintaining the benefits that the forest provides (Cadotte et al., 2011; Isbell et al., 2015). Such functional variation depends on the composition of species within the urban forest.

However, recent research shows that worldwide, urban forests have historically had low taxonomic diversity, dominated by only a few species (Paquette et al., 2021). For example, an analysis of diversity in 32 Canadian street tree populations found that four species – Norway maple (*Acer platanoides*), green ash (*Fraxinus pennsylvanica*), American elm (*Ulmus americana*), and honey locust (*Gleditsia triacanthos*) make up more than one third (34%) of all street trees (Martin et al., 2025). This pattern reflects both the practical constraints of diversifying the urban forest (i.e. tolerance to urban conditions, nursery stock, maintenance requirements) and aesthetic preferences that have historically favoured monocultures or low-diversity plantings (Hilbert et al., 2023; Plant and Kendal, 2019).

Alongside their global counterparts, many Canadian municipalities are committing to planting more trees, which can contribute to a range of environmental and human health benefits that enhance overall urban wellbeing (Wolf et al., 2020). However, these urban forest strategies sometimes lack attention to forest diversity (Ordóñez Barona et al., 2024). To make informed decisions on urban forest management, we must carry out studies that assess multiple dimensions of the urban forest in parallel. For example, a study of Barcelona's urban forest found that both tree species richness and abundance were positively related to life expectancy (Padullés Cubino and Retana, 2023), highlighting the importance of not only the number of trees in cities, but tree diversity as well. A large study of Canadian cities shows that tree diversity can also contribute positively to mental health (Buxton et al., 2024).

Beyond species richness and tree abundance, other forest metrics have important implications for urban forest management and ecosystem services. Species richness alone may overestimate urban forest diversity because all species are weighed equally, whereas rare species may contribute little to overall urban forest function (Kendal et al., 2014; Morgenroth et al., 2016). Metrics that incorporate evenness provide a more meaningful measure of community structure by reflecting the balance among species. While tree abundance indicates the number of individual trees, metrics related to structure, such as tree size, density, and basal area, offer further insight into the ecological capacity of the urban forest, as does tree spatial distribution. For example, large trees contribute disproportionately to ecosystem services such as cooling, shading, and carbon storage (Alonzo et al., 2025; Wujeska-Klause and Pfautsch, 2020; Zhang et al., 2020), whereas the spatial density and distribution across the city influences the uniformity of these benefits (Rendon et al., 2024; Yin et al., 2024). Considering diversity, composition, and structure therefore provides a more in depth understanding of the urban forest's capacity to provide benefits and helps identify where management efforts can most effectively enhance ecosystem services (Paquette et al., 2021; Sousa-Silva et al., 2021).

There is surmounting evidence that tree diversity, composition, and structure is not evenly allocated throughout the urban forest (Anderson et al., 2021; Avolio et al., 2015; Fan et al., 2019; Roman et al., 2018). Urban tree diversity and abundance is influenced by many factors such as land management type, municipal policies, socioeconomic status, and population density (Berland et al., 2011; Roman et al., 2018). Many studies have focused on patterns of tree abundance and diversity by documenting patterns along urbanization gradients and across different socioeconomic groups. For example, urban canopy cover and tree species richness are often positively correlated with wealth (Avolio et al., 2015; Gerrish and Watkins, 2018). Other studies have found significant race-based inequity in access to the urban forest, with predominantly white neighbourhoods having more tree canopy and higher tree species diversity (Locke et al., 2021; Watkins and Gerrish, 2018). Social vulnerability and urban forest diversity are intertwined, suggesting that residents in low-income neighbourhoods may not have equal access to the ecosystem services that trees provide (Landry et al., 2020). Despite this within-city heterogeneity, urban forest research and consequently many urban forest management strategies are often based on city-wide trends (Conway and Vander Vecht, 2015; Sousa-Silva et al., 2023a). Moreover, city-wide assessments typically consider only public trees, capturing only part of the urban forest (Keller and Konijnendijk, 2012; Ma et al., 2021).

Trees found on private land are thought to account for upwards of half of all urban trees and play a critical role in shaping the urban forest (Avolio et al., 2015; Shakeel and Conway, 2014). For example, in our study area of Montreal, private trees are often estimated at 50% of all trees (Larouche et al., 2021). However, private trees are commonly excluded from municipal or openly accessible inventories (Keller and Konijnendijk, 2012), largely because municipalities inventory only the trees they are responsible for. Private trees are also more difficult and time consuming for researchers and practitioners to access in the field (Dyson et al., 2019). As a result, studies of urban forest benefits often include only street trees or park trees (Larouche et al., 2021), overlooking the contribution of private trees. Recent efforts have begun to include private properties, particularly residential yards, in urban forest assessments. Municipalities such as Halifax, NS and Cambridge, ON have included private trees as part of their urban forest inventory, but these assessments have been either done at a small scale or using canopy mapping (City of Cambridge, 2015; Duinker and Foster, 2017). Local initiatives such as the Neighbourwoods© program (Kenney and Puric-Mladenovic 1995) similarly have monitored urban tree structure and function across land uses in Southern Ontario. New research is testing remote sensing using aerial imagery and/or airborne LiDAR to map both public and private urban forest (Fleming et al., 2025; Ucar et al., 2018), but most studies find it is less reliable for identifying species or capturing fine-scale diversity metrics (Chen et al., 2024). Even within studies considering private trees, the emphasis is often placed on residential land – rarely accounting for other non-public spaces such as schools, hospitals, or commercial sites (Dyson, 2019). Consequently, a large portion of the urban forest's diversity and structural variation remains poorly characterized.

Urban green spaces are shaped not only by public policies but also by the actions and preferences of private landowners, including residents, businesses, and institutions. Green spaces on private land often have different species and management practices than public green spaces, reflecting local priorities (Pearce et al., 2015), aesthetic preferences (Conway, 2016; García, 2025; Tyrväinen et al., 2003), and functional needs (Drew-Smythe et al., 2023). For example, trees present on public land are typically valued for their stress resilience and compact growth, while individual residents may select trees for aesthetic or cultural purposes such as flowering and colour (Avolio et al., 2015; Hutt-Taylor and Ziter, 2022; Pataki et al., 2013). The availability of species to individual managing bodies also varies across different land use types, and different actors use different criteria to select the species and number of trees for their green space (Conway and Vander Vecht, 2015).

As a result, privately owned and governed green spaces can hold unique species and differ in forest diversity, composition, and structure compared to publicly managed green spaces. For example, residential yards are found to contain higher canopy than other land uses (Ossola et al., 2019) and harbor high biodiversity, providing habitat for wildlife (Davies et al., 2009; Fornoff et al., 2017). Schools, churches, hospitals, and commercial areas can serve as additional sources of tree diversity and abundance, supporting ecosystem services and connectivity across the city. Studies have shown that these non-residential private sites contain species not found elsewhere and contribute to heterogeneity in urban forests (Chambers-Ostler et al., 2024; Hutt-Taylor and Ziter, 2022; Sousa-Silva et al., 2023b). Even within the umbrella of private land, different stakeholders may have differ needs or preferences that impact the urban forest. For a more holistic understanding of the urban forest, it is important to consider not only the contrast in tree diversity, composition, and structure between public and private land but also to examine the different land use types that make up these broad categories.

Urban forest composition is highly context-dependent and can vary substantially between cities due to differences in climate, planning history, cultural preferences, and land-use patterns (Jenerette et al., 2016; Mejía et al., 2024; Ordóñez-Barona, 2017; Roman et al., 2018; Yang et al., 2015). Because these local factors shape the urban forest uniquely, generalizations from other cities may be unreliable for guiding management. To make informed urban forest management decisions, it is therefore important to collect fine-scale data on tree species composition, structure, and distribution within the city of interest. In Montreal, Canada, some previous research has examined urban forest characteristics, but mainly at the neighbourhood scale. A study conducted in the residential neighbourhood of Notre-Dame-De-Grâce found that private trees contributed an additional 53 species not found in the public inventory (32% percent of species surveyed) yet tended to be smaller than trees on public land (Hutt-Taylor and Ziter, 2022). However, another study located in downtown Montreal found that public trees had higher diversity and abundance than private trees, with small trees dominating public and private green spaces (Sousa-Silva et al., 2023b). These two studies may differ in their findings due to the dominance of different land use types present in their respective study areas. In residential areas, where there is more private land, private trees may contribute more to the urban forest compared to a city's downtown, where there is higher population density, limited space for trees, and the public right of way makes up a higher proportion of plantable area. Overall, these examples highlight how the drivers of tree diversity and structure differ across the urban landscape even within the same city. To better understand the contributions of different land use types to the urban forest at the city scale, we need to capture the range of factors that may be driving differences in the distribution and diversity of trees across the urban landscape.

To expand on work investigating the contribution of trees on private land to the urban forest, we created a tree inventory of the Island of Montreal spanning varying gradients of vegetation cover, population density, and mean household income. Specifically, we asked how multiple metrics of urban forest diversity, composition, and structure vary across six land use types: commercial, institutional, parks (not including woodlands), residential, public right-of-way, and vacant lots. This offers an initial assessment for stakeholders to understand the contributions of different land use types to the urban forest and to guide targeted management.

Since diverse green spaces are managed with different goals in mind, we expected tree diversity, composition, and structure to differ among land uses. However, there is limited previous work on how less studied private land use types, such as institutional and commercial land, differ from other land use types at a city-wide scale. For example, we may expect that institutional green spaces have a species composition similar to residential yards, since they are both private land use types and they may have similar preferences for trees. However, we could also expect that institutions are more similar to parks in tree size and density, since they both contain large grassy areas with trees chosen for ease of maintenance and canopy. While we approached this study as exploratory and could not generate strong predictions for every land use type, we did have reasonable expectations for some. For instance, we expected that residential green spaces would have higher species richness and lower species evenness than other land use types at local scales due to the prevalence of rare species that result from individual preferences in land management (Hutt-Taylor and Ziter, 2022). Similarly, we expected that species composition would differ between residential land and the public right-of-way to the different priorities in tree selection from city managers and homeowners (Conway and Vander

Vecht, 2015). We also expected the public right-of-way and parks to have higher evenness and lower species richness since public trees are often dominated by only a few species specifically chosen for their low maintenance and high resilience (Liu and Slik, 2022).  Generally, we expected residential and commercial land to have more small trees than public land due to preferences for smaller ornamental trees and the presence of privacy hedges, which are typically composed of small trees (Dilley and Wolf, 2013; Hutt-Taylor and Ziter, 2022).

## Methods

*Study Area*

We conducted our study on the island of Montreal. The island covers an area of 500 $km^2$ and has a population of two million inhabitants (Statistics Canada, 2022). It is composed of 16 independent municipalities, including the city of Montreal, Canada’s second largest city.

Within the boundaries of the island of Montreal, we used 22 plots for our study that belong to the Montreal Urban Observatory (Paquette et al., 2025), a network of 25 plots established in 2021 to study biodiversity, ecosystem functions, and human health (Fig. 1). We excluded three plots from our analyses due to the presence of dense forested areas (i.e., urban woodlands), a land use type not present in other plots and difficult to sample using consistent methodology. The Montreal Urban Observatory spans a variety of urban settings, ranging from suburban areas to the downtown core. These plots were selected to represent a diversity of urban landscapes across three different gradients: vegetation density, population density, and mean household income (Paquette et al., 2025). Plots were not chosen to be proportionally representative of the land use and sociodemographic context of the Island of Montreal but rather span the variation in hypothesized drivers of ecosystem function and biodiversity in cities. We defined plot area as a 200 m radius buffer around the plot center. This scale allowed us to sample green spaces within a variety of land-use types across a range of different urban landscapes, which facilitates extrapolating our results to other urban contexts.

*Tree Inventory*

We conducted a tree inventory in summer 2023 and 2024 within a 200 m buffer (area = 125,663 $m^2$ or 12.6 ha) surrounding each plot center. We defined a tree as a woody perennial individual 2 m or more in height. In cases where multiple stems occurred in close proximity, we considered stems as belonging to the same tree if they were visibly connected at or near the base. When it was not possible to determine a whether two stems were part of the same individual, we recorded stems as separate individuals. Each tree within the buffer (total of 28,339 trees) was geo-located and identified at the species level when possible. In cases where we could not confidently identify the species, we identified the tree to genus (n = 911 trees, 3.2%). We also measured diameter at breast height (DBH) for each tree when possible; otherwise, DBH was estimated (n = 1 656, 5.8%). We used Montreal's public tree database to locate all publicly managed trees (Ville de Montréal, 2023) and remeasured and identified these trees to ensure their accuracy and update DBH measurements. All trees not inventoried by the city but located inside the public domain were also included. To ensure consistency across all land use types, we removed cultivar classification from the public tree data when present and classified trees at the species level. For neighbourhoods where there was no public tree inventory available, and to locate private trees, we used satellite imagery of the plots to identify the location of trees (Google Maps, 2025). To access private trees, we gained permission from landowners to include their trees in our study or estimated species and DBH from a distance (e.g., when inaccessible, permission to access land not obtained, but visible from the street).

Many trees, especially on private land, were part of hedges. We considered a hedge to be a continuous, closely planted row of trees of the same species. To identify the number of individuals in one hedge, we counted the number of individuals within a one metre section of the hedge (in an area representative of the entire hedge, typically within the middle of the hedge) and then estimated the length of the entire hedge in the field or by using satellite data. We then calculated the estimated number of individuals in the hedge by multiplying the length by the number of individuals within the one metre section. In total, plots contained 419 hedges ranging from 4 metres to 100 metres in length and ranging from 3 to 250 individuals (n = 11 366 total hedge stems).

*Land Use Types*

To determine land use types within plots, we used a land-use classification polygon layer (Communauté métropolitaine de Montréal, 2024) containing 11 classes of land use. To ensure sufficient sample sizes and facilitate comparison across land use types, land use classes with similar characteristics were aggregated into broader land use types. We consolidated the land-use categories into 6 land use types containing trees or other vegetation: commercial, institutional, park, public right-of-way, residential, and vacant lot (Table 1). For example, offices, industrial land, and commercial land were aggregated into a single 'Commercial' category. Each plot was subdivided into multiple subplots according to the land use types present. A subplot was defined as a unique land use type within a plot (e.g. Plot 01A_Residential, Plot01A_Commercial), with all polygons in a single plot belonging to the same land use type (e.g. residential) being merged to form a single subplot polygon, allowing ecological metrics to be calculated at the land use level within each plot (Fig. 2).

Both residential land and the public right-of-way subplots were present in all 22 plots, while commercial land (n = 15), institutional land (n = 18), parks (n = 14), and vacant lots (n = 10) were only present in a subset of plots. Subplots which contained no trees accounted for approximately 4% of subplots (n = 4, 2 in commercial, 1 in institutional, 1 in parks). In most cases, these areas were not truly plantable (i.e., contained mostly built area) and we removed these subplots to improve consistency across land use types. This resulted in 101 subplots for analysis, allowing us to show differences in tree diversity and abundance in areas where trees are present.

*Plantable Area*

Trees can only grow within the area of our land-use types that do not contain built infrastructure. For example, only a proportion of residential land is considered green space, where trees can grow. We consider the areas that can theoretically support green infrastructure as “plantable area”. To ensure only plantable area within subplots was considered, we created a built area polygon layer using building assessment layer from the City of Montreal (Données Québec, 2023) and OpenStreetMap (OpenStreetMap contributors, 2025) in QGIS (QGIS Development Team, 2025). To calculate total built area at the subplot-level, we merged building data, road data, waterways, parking lots, and railways into single polygons for each land use type in each plot. Certain OpenStreetMap layers were provided as lines and required buffering to approximate area. Buffers were applied based on average feature widths to exclude areas where tree planting is unlikely. Railways were buffered to 4 m total width by using satellite imagery to estimate the area covered by the railway. Road lanes were buffered to 3 m, based on average road width in Canada (City of Toronto, Transportation Services, 2017). We visually estimated the width of individual waterways since there were only two present across plots. While this approach may over- or underestimate individual features, since roads can vary in size, it provides a consistent estimate of built area across subplots. To ensure the accuracy of our built area layer, we manually drew polygons over built infrastructure that was not included within the existing layers using satellite imagery (e.g. tennis courts, cemented vacant lots, playground structures) (Google Maps, 2025). Smaller structures such as pools, sheds, and privately owned sidewalks were not included in our built area layer due to the lack of existing layers and the disproportionate amount of time required for manual digitization relative to their limited contribution to total built surface area. However, this exclusion may have greater influence on results for residential land, where such infrastructure is more common. As our analysis focuses on relative rather than absolute built area, this level of uncertainty was considered acceptable. Using our built area layer and the land-use classification layer, we removed built area from subplots to create a polygon of plantable area to use for our analyses (Fig. 2).

*Species Classification*

Due to visibility issues when collecting tree data from a distance, we were unable to identify some trees to the species level (n = 911, 3.2%). To include these trees in diversity analyses, we assigned species based on the most abundant species within that genus in the dataset. For example, in the *Acer* genus, *Acer platanoides* was the most common tree species across all plots. We assigned trees in the *Acer* genus that were not identified to the species level as *Acer platanoides*. This approach assumes that unidentified trees most likely belong to the most common species across our dataset rather than a rare or less common species, allowing us to estimate species data for analysis in a conservative way. In cases where species identification was unreliable (e.g., some individuals belonging to the *Malus* genus, all individuals in the *Crataegus* genus) individuals were assigned to a general "spp." category, preventing artificial inflation of species richness.

Many trees on both private and public land were hybrids (n = 366 individuals, 1.3%, belonging to 78 different hybrid species). Some hybrids are common and naturally occurring, while others are cultivars and exclusively planted. For taxa with multiple hybrids, we evaluated on a case-by-case basis whether to retain the hybrid as a separate species. Hybrids that were similar to one of their parent species or were cultivars with little ecological or morphological differentiation were reassigned to the parent species group (Table A1). Hybrids with unique morphological traits that were visually identifiable in the field, have distinct reproductive lineages, are naturally hybridizing, or have known ecological significance were retained as separate species. For example, *Acer x freemanii,* which is widely planted in urban settings due to its retention of specific traits from both parent species, was retained as a separate species, while *Ulmus* x 'Homestead', which is an elm hybrid that is a cross of an *Ulmus pumila* and an *Ulmus pumila* hybrid, was reassigned to its parent species *Ulmus pumila.* This approach ensures that the dataset still captures the presence of unique hybrid taxa that are more likely to influence community composition, while minimizing inflation of species counts due to cultivars and hybrids.

*Tree Abundance, Diversity, and Size Structure*

We calculated three types of taxonomic diversity for each land use type for each subplot: species richness, effective number of species based on Shannon entropy ($q = 1$), and Hurlbert's probability of interspecific encounter (PIE). Both species richness and the effective number of species were calculated under the Hill number framework (Hill, 1973). Hill (1973) introduced a family of diversity indices, referred to as Hill numbers, which convert classical diversity measures to express diversity in terms of species equivalents, or effective number of species. For a given diversity metric, such as Shannon diversity index (1948) or Simpson diversity index (1949), the effective number of species is the number of equally common species that would produce the same diversity value. According to the framework, species richness ($q = 0$) counts species equally, regardless of their relative abundance. When $q = 1$, the Hill number is the exponential of Shannon entropy, which quantifies diversity by weighing each species' frequency, allowing it to focus on common species while being less sensitive to rare species, which are usually not ecologically meaningful. This value corresponds to the number of equally common species that would give the same Shannon diversity, making $q = 1$ (hereafter referred to as the effective number of species) useful for comparing directly to species richness as they are expressed in the same unit. Hurlbert's probability of interspecific encounter (PIE) is a measure of evenness ranging from 0 to 1 that estimates the probability that two individuals selected at random within a sample will belong to different species (Hurlbert, 1971). PIE values closer to 1 indicate higher evenness, while values closer to 0 indicate low evenness within a sample.

We calculated tree abundance as the number of trees within each subplot. For multi-stemmed trees, we consolidated all stem measurements into one individual DBH as per Magarik et al. (2020). Due to lack of access to certain trees, we estimated DBH for 1,656 trees in the dataset, (5.8%). In most cases, we visually estimated DBH in the field (n = 1,259, 4.4%). Throughout the field season, we regularly calibrated our visual estimates through randomly selecting individual trees to both estimate and measure to increase the accuracy of our visual estimation. However, we were unable to visually estimate 404 trees (1.4%) due to obstructed view of their trunks. To include these individuals in size analyses, we estimated DBH using species-specific averages. We calculated these averages separately for each species within each land use type, rather than across all plots (see appendix I). This approach accounts for potential differences in tree size associated with land use type.

We calculated basal area for each tree as the cross-sectional area at breast height. To obtain subplot-level basal area($m^2$/ha), we summed tree basal area and standardized by the total subplot area. We also calculated basal area based on the plantable area within each subplot to accurately reflect subplot area that has the capacity to support tree growth. To examine urban forest structure, i.e., the size of trees across different land use types, we plotted size distribution curves using DBH measurements.

*Statistical Analysis*

*Rarefaction*

Due to the heterogenous nature of urban landscapes, our plots did not contain equal amounts of each land use type. In some cases, entire land use types were absent from a plot, leading to variation in both the number of subplots sampled and their relative size. Because species richness typically increases with the number of individuals encountered, it is strongly dependent on the size of the sampling area (Chiu, 2023; Colwell et al., 2012). Traditionally, ecologists use rarefaction to down-sample larger sites until they are the same size as the smallest sample, to allow for comparison of equally sized sites (Chao and Jost, 2012). However, these sites will have different degrees of completeness, representing different proportions of the true assemblage depending on the site's diversity (Chao et al., 2014; Chao and Jost, 2012). Standardizing samples by coverage rather than size or individual abundance addresses the bias that can occur with size-based rarefaction. We decided to rarify all diversity metrics (species richness, effective number of species, and PIE evenness) using coverage-based rarefaction and extrapolation based on sample completeness using the R package *iNEXT*, which implements Hill numbers (Hsieh et al., 2025). For each subplot, abundance data were assembled into species-by-abundance vectors and analyzed with *iNEXT* to calculate Hill numbers for species richness and effective number of species. Diversity estimates were standardized to 95% sample completeness. Estimates were obtained through bootstrap resampling (200 replicates) to generate 95% confidence intervals for each subplot. Because PIE evenness is not implemented in the *iNEXT* framework, we calculated it separately by first determining the rarefaction sample size corresponding to 95% coverage for each subplot (using the *estimateD* function in *iNEXT*). We then rarefied each subplot's species count to the standardized sample size and computed PIE evenness from the rarefied counts using the *mobr* package in R (McGlinn et al., 2024). This provided a single standardized estimate of PIE evenness per subplot; however, unlike Hill numbers, it was calculated as a point estimate without bootstrapped confidence intervals.

*Linear Mixed Models*

To evaluate the effects of land use type on diversity metrics (i.e., tree abundance, species richness, effective number of species, and PIE evenness), we used mixed-effects models using R packages *lme4* (Bates et al., 2025) and *glmmTMB* (Brooks et al., 2025) with diversity metric as the response variable, and land use type as the independent variable. *Tree Density:* Because subplots varied substantially in size, results of comparing absolute tree counts could be driven by area and not ecological patterns. To account for this, we modeled tree abundance using a generalized linear mixed model (GLMM), including the log-transformed area of each subplot as an offset, thus effectively modeling tree density (Warton, 2022). We used a negative binomial distribution to account for overdispersion in the data. *Species Richness and Effective Number of Species:* Coverage-based rarified values for species richness and effective number of species were modeled with GLMMs with a Gamma distribution (with a log-link) to account for continuous positive skewed data. *Evenness:* Our study contained 4 subplots that contained only one species, which means that the probability of drawing two different species is zero, leading to a PIE evenness value of zero. To account for zeros in the data, we modeled evenness using a zero-inflated beta regression mixed-effects model. *Basal Area:* We modeled basal area per subplot area and basal area per plantable area using a linear mixed-effects model with a log-transformed response to account for the positive skew of basal area values.

For all models, plot was included as a random effect to account for non-independence among subplots within plots. All model residuals were visually inspected for normality using diagnostic plots. We then evaluated model significance using the *Anova* function in the *car* package in R (Fox et al., 2024), generating p-values using Type II Wald chi-square tests. Finally, to explore differences between individual land use types, we calculated Tukey-adjusted pairwise comparisons using the *emmeans* package, which provides estimated marginal means and contrasts on the model response scale (Lenth et al., 2025).

*Species Ordination*

To visualize differences in species composition between land use types, we used the *ape* package in R to perform a principal coordinates analysis (PCoA) (Paradis et al., 2024). PCoA is an ordination method that summarizes patterns of dissimilarity in species composition among sites into a few axes, making complex multivariate data easier to visualize. We compiled species data in a subplot-by-species matrix and calculated Bray-Curtis dissimilarity between subplots to capture differences in species composition while accounting for relative abundances. The first two principal coordinates were extracted and visualized, with variance explained by each axis reported. To identify which species were most strongly associated with variation in community composition, we used the *envfit* function in the *vegan* package (Oksanen et al., 2025) to fit species vectors onto the ordination, and significance was assessed with permutation tests. We considered species with significant correlations ($p < 0.001$) to contribute meaningfully to differences across land use types. For interpretation, we associated each species with the land use type whose centroid in ordination space aligned most closely with its fitted vector. To assess whether species composition differed among land use types, we calculated multivariate dispersion using the *betadisper* function in vegan (Oksanen et al., 2025). We further tested differences in dispersion by performing an ANOVA. Our ANOVA results revealed significant differences in dispersion among land use types, indicating that variability in species composition was not homogenous across groups. Given the significant differences in dispersion observed, we refrained from conducting PERMANOVA, because it assumes homogeneity of dispersions. Instead, we focused on pairwise comparisons of beta dispersion using Tukey's HSD test to identify which land use types differed from each other in compositional variability. To complement this abundance-based analysis, we also calculated Jaccard dissimilarities to focus on presence/absence patterns, highlighting whether differences were driven by species occurrence rather than abundance. We compared results using both Bray-Curtis and Jaccard dissimilarities and found similar patterns, so we decided to perform our analysis using Bray-Curtis distances (see appendix II; Fig. A2.1). All statistical analyses were done in R 4.4.2 (R Core Team, 2024).

# Results

*Urban Forest Inventory Summary*

In total, our inventory resulted in a dataset of 28,339 trees across 22 plots. We identified 101 genera and 307 species (Table 2). Raw tree abundance (without accounting for area) was highest on residential land, followed by the public right-of-way, and lowest in vacant lots (which contained less than 1% of the plantable area across all subplots). Trees belonging to hedges accounted for approximately 40% of all trees and were present in all land use types, most prominently on residential land, where they accounted for almost 50% of trees. Raw (unrarefied) species richness was highest on residential land, followed by the public right-of-way. Species richness was lowest on commercial land and in vacant lots with intermediate numbers on institutional land and in parks.

The five most abundant tree species across the entire inventory were Eastern White-Cedar (*Thuja occidentalis*) (35%), Norway Maple (*Acer platanoides)* (7.2%), Common Lilac (*Syringa vulgaris*) (4.6%), Manitoba Maple (*Acer negundo*) (3.9%), and Silver Maple (*Acer saccharinum*) (3.8%) (Table 2). In total, these five species represent 54 % of trees across the entire inventory. Private land use types (i.e., commercial, institutional, residential, vacant lots) contributed an additional 92 species not found on public land (i.e., the public right of way, parks). Specifically, residential land alone contributed an additional 69 species not found in any other land use type.

Both the total area and the plantable area within our plots varied across land use types. Residential land represented 46% of the total area and 55% of plantable area across plots, while the public right-of-way represented 30% of the total area and 22% of plantable area. Vacant lots covered the lowest area (0.66%) and plantable area (0.84%). At the plot level, residential subplots were larger (5.7 ha) and contained more plantable area (3.3 ha) on average when compared to other land use types (Fig. A2.2). Overall, public land represented 39% of our overall study area and 31% of plantable area, while private land represented 63% of our study area and 69% of plantable area.

*Tree DBH and Size Distribution*

Tree DBH ranged from 0.5 to 175 cm, with a median value of 8 cm. Median DBH differed across land use types (Table 2). The highest median DBH was 26 cm in parks, with all other land use types having similar and substantially lower median DBH (7 – 10 cm).

Tree size distribution also varied across land use types. Tree size was right-skewed across all land use types, with a predominance of small trees under 25 cm DBH (Fig. 3). Residential land had the highest density of small trees and the strongest skew, with an especially high density of stems < 10 cm. The public right-of-way also showed a right-skewed distribution but contained proportionally more medium and large trees than other land use types. When hedges were removed, the residential and public right-of-way peaks were substantially reduced yet still show a right-skewed distribution. Comparatively, parks and vacant lots exhibited more even distributions across DBH classes.

*Tree Density*

Tree density differed significantly among land use types ($\chi^2_{(df = 5)} = 36.8$, $p < 0.001$) (Fig. 4A) (Table A3.1). According to model estimates, mean tree densities per subplot ranged from 174 trees ha$^{-1}$ [95% CI: 126-241] on institutional land to 562 trees ha$^{-1}$ [95% CI: 419-754] in the public right-of-way. Residential areas and vacant lots had comparable tree density (279 trees

$ha^{-1}$ and 275 trees $ha^{-1}$, respectively), while parks and commercial land had tree densities of 204 trees $ha^{-1}$ and 212 trees $ha^{-1}$. Pairwise comparisons show that the public right-of-way supported higher tree densities than all other land use types ($p \leq 0.006$), including institutional, ($p < 0.0001$), parks ($p = 0.0003$), commercial land ($p = 0.0004$), and residential land ($p = 0.0056$), except vacant lots, where the difference was marginal ($p = 0.089$). No other pairwise comparisons were statistically significant (all $p > 0.22$), indicating broadly similar densities in commercial, institutional, park, residential, and vacant lot land uses.

*Species Richness*

Species richness (coverage-based rarefied to 95% completeness) also differed significantly among land use types ($\chi^2_{(df = 5)} = 172.6$, $p < 0.001$) (Fig. 4B) (Table A3.2). Mean richness per subplot ranged from 7 species [95% CI: 4-10] in vacant lots to 59 species [95% CI: 44 -79] on residential land. The public right-of-way had similar species richness to residential land, with a mean of 54 species [95% CI: 41-72]. Pairwise comparisons showed that both residential land and the public right-of-way had substantially higher richness than all land use types (all $p < 0.001$) but did not significantly differ from each other ($p = 0.998$). Institutional land and parks also supported more species than commercial land and vacant lots ($p \leq 0.001$) but did not differ from each other ($p = 0.978$), while commercial land and vacant lots had similarly low levels of species richness ($p = 0.997$).

*Effective Number of Species*

The effective number of species (or ENS) also differed significantly among land use types ($\chi^2_{(df = 5)} = 68.2$, $p < 0.001$) (Fig. 4C) (Table A3.3). Mean estimates of ENS per subplot ranged from 5 species [95% CI: 3-6] in vacant lots to 17 species [95% CI: 13-23] in the public right-of-way. Pairwise comparisons showed that institutional land, parks, the public right-of-way, and residential areas all supported significantly higher ENS than commercial land ($p \leq 0.0039$), while vacant lots did not differ from commercial areas ($p = 0.99$). Institutional, public right-of-way, and residential land had higher ENS than vacant lots ($p \leq 0.0002$), while parks also had higher ENS than vacant lots ($p = 0.04$). There were no significant differences in ENS among institutional, parks, public, right-of-way, and residential land ($p \geq 0.134$).

*Evenness*

Land use types also significantly affected evenness ($\chi^2_{(df = 5)} = 21.6$, $p < 0.001$) (Fig. 4D) (Table A3.4). Mean Hurlbert's PIE values at the subplot level ranged from 0.66 [95% CI: 0.57-0.74] in commercial areas to 0.85 [95% CI: 0.79-0.89] in the public right-of-way. Institutional land (0.84 [95% CI: 0.78-0.89]) and parks (0.80 [95% CI: 0.72-0.86]) also had relatively high evenness, while residential areas (0.78 [95% CI: 0.72-0.84]) and vacant lots (0.75 [95% CI: 0.63-0.84]) had intermediate evenness. Pairwise comparisons show that the public right-of-way and institutional land had significantly higher evenness than commercial land ($p = 0.0006$ and 0.0026, respectively). No other pairwise differences among land use types were statistically significant (all $p \geq 0.095$).

*Basal Area*

According to our mixed model, basal area (standardized by total subplot area) differed significantly among land use types ($\chi^2_{(df = 5)} = 57.9$, $p < 0.001$) (Fig. 4E) (Table A3.5). Back-transformed means ranged from 0.9 $m^2$/ha [95% CI: 0.6–1.5] in commercial areas to 7.9 $m^2$/ha

[95% CI: 4.9–12.7] in parks. Public rights-of-way also had the second highest basal area (5.6 m²/ha [95% CI: 3.8–8.2]), while institutional land (2.9 m²/ha [95% CI: 1.9–4.4]), residential areas (3 m²/ha [95% CI: 2–4.3]), and vacant lots (3 m²/ha [95% CI: 1.7–5.3]) had intermediate values. Pairwise comparisons showed that commercial land had significantly lower basal area than all other land use types (all $p \leq 0.012$). Parks also supported significantly higher basal area than institutional land ($p = 0.018$) and residential areas ($p = 0.014$). No other pairwise differences were statistically significant (all $p \geq 0.089$).

When we look at basal area standardized by plantable area, basal area also differed significantly among land use type ($\chi^2_{(df = 5)} = 50.6$, $p < 0.001$) (Fig. 4F) (Table A3.6). Basal area ranged from 3.9 m²/ha on commercial land [95% CI: 2.5–6.2] to 20.7 m²/ha in the public right-of-way 95% CI: 14.3–30.1], with parks also supporting relatively high densities (11.9 m²/ha [95% CI: 7.5–19.]). Pairwise comparisons revealed that commercial land had significantly lower basal area than parks ($p = 0.009$) and the public right-of-way ($p < 0.001$). The public right-of-way contained significantly higher basal area than residential areas ($p < 0.001$), institutional land ($p = 0.001$), and vacant lots ($p < 0.001$). No other pairwise comparisons were statistically significant (all $p \geq 0.095$).

*Tree Species Composition*

While public land use types accounted for only 70% of all species, private land use types included 86% of all species in the total inventory. *Thuja occidentalis* was the most abundant species on both private and public land accounting for 42.4% and 20% of tree abundance respectively (Fig. 5). Of the ten most abundant species found in private land use types, only *Thuja occidentalis* accounts for more than 10% of trees, with other species such as *Syringa vulgaris* (6%) and *Acer platanoides* (5.7%) representing less of the overall composition (Fig.

5A). This highlights the dominance of *Thuja occidentalis* on the urban landscape. In public land use types, *Thuja occidentalis* is less dominant, with the second and third most abundant species, *Acer platanoides* and *Acer saccharinum,* representing 10.6% and 7% of trees (Fig. 5B). Despite the dominance of *Thuja occidentalis* based on number of stems, larger species such as *Acer saccharinum* and *Acer platanoides* make up the bulk of urban forest relative importance (species basal area / total basal area) on both public and private land (Fig. 5C and 5D).

When looking at individual land use types, residential land accounted for the highest proportion of all species (81.1%), followed by the public right of way (67.1%), institutional land (34.5%), and parks (32.2%). Commercial land and vacant lots both respectively accounted for 12.4% of all species. *Thuja occidentalis* was the most common species in all land use types, ranging from 21.8% to 45.2% of all trees, except for parks., which were dominated by *Acer saccharinum* and *Acer platanoides* (Table 2). Parks overall displayed less dominance from a few select species, with no species accounting for more than 12.9% of the total tree abundance.

Points closer together in the PCoA indicate more similar species assemblages, while greater distances indicate increased dissimilarity among sites (Fig. 6). The first two PCoA axes explained 18.8% (Axis 1) and 10.2% (Axis 2) of variation, capturing 29% of overall composition differences. Residential land and the public right-of-way formed distinct clusters in ordination space. Institutional, commercial, vacant lots and park subplots showed greater overlap in centroid positions, suggesting similarities in community composition among these land use types. These visual patterns were supported by our quantitative analysis of multivariate dispersion, which showed significant differences in variability among land use types (ANOVA: $F_{(df = 5)} = 10.3$, $p < 0.001$). The public right-of-way (average distance to median = 0.43) and residential land (0.42) were significantly less variable than institutional land (0.61), parks (0.61), commercial land (0.59), and vacant lots (0.58) . Pairwise comparisons revealed that residential land and public right-of-way subplots were significantly more homogenous compared to institutional, park, vacant lot, and commercial land (p values ranging from 0.00005 – 0.02 across all pairwise comparisons), but did not differ from one another ($p = 0.999$). Institutional, park, vacant lot, and commercial land use types did not significantly differ from one another. Together, these results suggest that residential land and the public right-of-way are both more uniform in their species composition across plots, and distinct from other land use types, while other land use types of have overlapping and more variable species composition. At the species level, 71 species showed significant associations with a particular land use type ($p < 0.001$), with four most strongly associated with the public right-of-way and the remaining 67 associated with residential land (Table A2).

# Discussion

This study explored how patterns of urban forest diversity, composition, and structure differ among public and private land uses within the urban landscape. Land use types differed in tree diversity, composition, and structure, matching our expectations that the unique preferences and management goals of different stakeholders influences the urban forest on a fine scale. Specifically, we found that species present on both public and private land differed in dominance and composition, meaning that current metrics considering only public land are not fully capturing the urban forest. This study represents one of the most comprehensive analyses of an urban forest inventory conducted for a large North American city. By incorporating plantable area and tree size into our work, we show not only which land use types contain the most trees or the most species, but rather how much each of these spaces are contributing to the overall urban forest. Finally, our findings demonstrate the importance of assessing a variety of diversity metrics to understand urban forests and the potential of different land use types for future planting initiatives. While species richness and tree density identified where trees and species are most abundant, measures of evenness, composition, and size structure revealed differences essential for guiding planting initiatives that enhance resilience and long-term forest benefits.

Across the entire tree inventory, *Thuja occidentalis* was the most abundant tree species, comprising 35% of all trees (Table 2). Most of these belonged to hedges, which were most abundant on residential land, where they serve as privacy barriers or yard boundaries (Blanusa et al., 2019). Consistent with previous work in Montreal showing high abundance of *Thuja occidentalis* and other conifers on private land (Hutt-Taylor and Ziter, 2022), we also found this species dominating commercial, institutional, residential, and vacant lot land uses to varying degrees. However, our inventory also recorded a disproportionately high number of *Thuja occidentalis* on public land, where it was the most abundant species in the public right-of-way. In contrast, Montreal's municipal inventory reports *Thuja occidentalis* as only 0.54% of public trees (Ville de Montréal, 2023). This discrepancy likely reflects methodological differences. Our field inventory included all woody stems over 2 m within our study area, whereas municipal inventories typically focus on larger individual, managed street or park trees and may exclude hedges that do not meet criteria for their tree inventory. Some of the hedges on public land may also result from residents planting trees along the public right-of-way at the edges of their properties. Hedges are a common feature in urban landscapes and have the potential to provide important ecosystem services but are currently ignored by many urban planning policies (Blanusa et al., 2019; Gonçalves et al., 2021). Conifers are often thought to be rare or underrepresented in urban forests (Paquette et al., 2021; Wood and Dupras, 2021), likely because most studies focus on street trees, where conifers are not ideal due to their low branches (Kuntz et al., 2006). However, our results reveal a strong dominance of *Thuja occidentalis* in hedges, suggesting that hedges may represent an overlooked source of structural and functional diversity in cities. Future research should evaluate the contribution of hedges to urban forest benefits and determine whether this is proportional to their abundance.

Although *Thuja occidentalis* dominated in stem count abundance across public and private land uses, large-stature species such as *Acer saccharinum* (Silver maple) and *Acer platanoides* (Norway maple) accounted for most relative importance (Fig. 5). These species likely contribute more to canopy cover and ecosystem services due to their greater size (Nowak and Aevermann, 2019; Suchocka et al., 2023). This pattern suggests that while *Thuja occidentalis* is abundant in the urban forest, other species may be more important and provide more benefits due to their relative basal area. Tree size distributions across all land uses were right-skewed, dominated by smaller trees under 25 cm DBH, particularly on residential land, which showed the highest density of small stems (<10 cm) (Fig. 3), similar to Sousa-Silva et al. (2023). We initially hypothesized that this skew was driven by the high number of hedges, which are generally composed of small individuals. However, the trend persisted even after hedge trees were removed, indicating that the predominance of small trees is not simply an artifact of the high number of hedges in our plots. Unlike Hutt-Taylor and Ziter (2022), who found that public land use types tend to contain more large trees than small, our results show that Montreal's urban forest is dominated by smaller trees across land use types. This difference may arise partly from recent planting initiatives in the city, such as the "Un arbre pour mon quartier" (English: *A tree for my neighbourhood*) program by la Société de verdissement du Montréal métropolitain (Soverdi), which helps provide trees to residents and institutions (Soverdi, 2025), or because our study includes very small trees (<5 cm DBH), which were not incorporated in their analysis. We may also be seeing smaller trees overall than in Hutt-Taylor and Ziter (2022) because they sampled an older neighborhood dominated by large, established street and park trees. However, the Sousa-Silva et al. (2023) plot was located in downtown Montreal, where there are fewer large trees overall due to newer plantings and less available space, which is why we may see smaller trees in their study. Our broader, city-wide sampling captures trends across diverse neighborhoods, reflecting that as a whole, Montreal is dominated by smaller trees. In their research, Pearce, Kirkpatrick, and Davison (2013) warn that the shift toward smaller tree species on private land threatens the long-term functionality of the urban forest, as large trees contribute disproportionately to canopy cover, cooling, carbon storage, and habitat provision, and their decline will make it increasingly difficult to maintain urban forest benefits. In our data, it is not possible to determine whether the prevalence of small trees reflects the dominance of small-statured species or simply younger individuals that have yet to reach maturity. Future research should therefore track changes in urban forest size distributions over time to better understand these dynamics and how they may influence management priorities and canopy cover goals.

We found that total tree abundance was highest on residential land (Fig. 4A), yet this pattern largely reflects the amount of residential area within our plots rather than higher tree density. When standardized by area, residential land supported moderate tree densities compared to other land use types. The public right-of-way, by contrast, exhibited the highest tree density. Unlike other land use types, the public right-of-way has relatively little plantable area beyond narrow boulevards and medians, but trees are planted in the available space at regular intervals. This approach results in a higher number of trees per hectare of plantable area. These results are further supported by our basal area results. When standardized by total subplot area, parks supported the highest basal area, likely reflecting the presence of mature, large-stature trees (Fig. 4E). In contrast, when basal area was standardized by plantable area, the public right-of-way showed the highest basal area, suggesting that although there is less basal area on the public right-of-way overall, trees in these areas are densely packed within the available plantable area (Fig. 4F). Similarly to Anderson et al. (2023), who showed that street trees enhance local biodiversity and biomass in denser neighborhoods, our results indicate that public rights-of-way make a disproportionate contribution to urban forest structure relative to their limited plantable area, underscoring their importance in maintaining canopy and increasing tree biomass in cities. Although significant funding exists to expand tree planting on public land, future initiatives must also consider where additional trees can realistically be planted and whether the public right-of-way remains the most effective space for expansion, given that it already sustains the highest tree density and basal area among land use types. By comparing basal area at these two different scales, we thus see how using subplot area estimates reflect the contribution of each land use type to overall canopy structure and urban forest composition, while plantable area estimates capture how efficiently space is used for trees.

Residential areas contain large amounts of plantable area, but much of this space serves multiple functions beyond tree planting, such as space for other vegetation for gardening or leisure. The number of trees in an individual residential yard depends on the individual homeowners' choices, maintenance capacity, and aesthetic preferences, which can widely differ (Conway, 2016). As a result, even though residential land encompasses a large share of the urban landscape, tree density remains moderate because not all plantable surfaces are intended for tree planting. Basal area at the subplot and plantable area scale was also low in residential spaces, indicating that trees may be smaller due to preferences for small-stature species (Hutt-Taylor and Ziter, 2022). For example, ornamental species such as common lilac (*Syringa vulgaris*) were common on private land but rarer as a public tree, and such species tend to have low relative importance and contribute less to overall forest structure yet are easier to maintain than large trees. Even if residents are interested in planting more trees in their yards, lack of information, maintenance costs and time constraints can dissuade them from adding more tree density (Almas and Conway, 2018; Riedman et al., 2022). To effectively increase the number of trees on residential land, programs must therefore align with residents' goals for their yards and provide ongoing support for tree care, not just initial planting. Urban forest initiatives should also recognize that not all residents will wish to plant or maintain additional trees, as property goals and preferences vary widely.

Institutional land had the lowest tree density, with all other land use types also possessing moderate tree densities. Different land use types have different built infrastructure and purposes that influence their potential for trees and the spatial needs required of their vegetation (Narváez Vallejo et al., 2024). Recognizing how the functional role of each land use type limits tree density can help urban forest managers tailor greening strategies to each land use type. The City of Montreal, supported by almost $49 million in federal funding as part of Canada's 2 Billion Trees program, has announced its goal of plant over 300,000 trees on the city's public lands and additional programs to plant 200,000 trees on private properties across the city (Natural Resources Canada, 2025). To properly implement these programs, we must have strong knowledge on which land use types have the capacity to support more trees. Institutional lands represent key opportunities for increasing tree density by partnering with schools, hospitals, and other organizations to integrate additional plantings within available open space. Similar approaches could also be applied to commercial land and vacant lots, which currently have relatively low tree density and basal area, to expand overall tree cover and biomass across the urban forest. However, we stress also the importance of placing that addition canopy where it will benefit the most people, especially those more likely to develop health conditions related to environmental exposure (Li et al., 2025; Sousa-Silva et al., 2023a). From this perspective, institutional land (or commercial land, vacant lots) may not necessarily be good candidates.

While unrarefied species richness was highest in residential areas, our inventory showed that when rarefied to account for sampling effort, the public right-of-way and residential land had substantially higher species richness than all other land use types (Fig. 4B). This finding is supported by previous work in Montreal (Hutt-Taylor and Ziter, 2022; Sousa-Silva et al., 2023b), but historically, studies in other cities find that tree diversity is lower on the public right-of-way compared to residential land (Avolio et al., 2018; Nitoslawski et al., 2016). In the past, urban foresters would plant rows of monoculture along streets, which reduced diversity in these spaces (Hilbert et al., 2023). However, similar richness between residential and public land may also reflect spatial turnover across neighbourhoods. Even if individual streets are planted as monocultures, such as *Acer platanoides* in one neighbourhood and *Tilia cordata* in another, diversity appears high when viewed across the entire city. This pattern also may partly reflect limitations in the land use data used to classify land use types. Residents often plant trees along property edges that border the public right-of-way, creating an overlapping area between private and public spaces which are difficult to determine as public or private trees when dealing with large datasets. These factors could inflate apparent richness on both land types and blur distinctions in species diversity.

Lower species richness on institutional, park, and commercial land likely reflects design constraints and management priorities that favor uniform plantings, open recreation areas, or ornamental landscaping over species variety. Vacant lots had the lowest species richness, likely because these minimally managed areas are colonized by a few opportunistic species that come to dominate, limiting overall diversity. These differences emphasize that each land use type fulfills distinct social and functional roles, which must be considered when developing strategies to enhance overall urban forest biodiversity. For example, institutional and commercial lands could increase diversity by planting a mix of medium- and large-stature species that provide shade and habitat, while also incorporating small flowering or fruiting species to improve aesthetic and ecological value. Vacant lots could be seeded with native shrubs and small trees to create pockets of habitat and increase structural complexity. Even small interventions along property edges or medians could enhance connectivity and species richness. By targeting these specific land use types and working with the relevant stakeholders - such as municipal managers, private property owners, and community groups - urban forest planners can strategically increase both species richness and functional diversity across the city.

Contrary to our expectations, we did not find substantial differences in effective number of species and evenness between public and private land use types. Previous research suggests that residential areas and other private land often harbor unique species due to individual planting preferences and an increased presence of introduced and exotic species (Avolio et al., 2018; Bourne and Conway, 2014; Chambers-Ostler et al., 2024). Previous work has found that effective number of species is higher on public land than private land (Sousa-Silva et al., 2023b). However, this finding may be because their study took place in a downtown area, where public trees are relatively more abundant compared to other parts of the city. Additionally, differences in the amount of each land use type across the study area were not accounted for. In our study, the public right-of-way, institutional land, parks, and residential areas all supported relatively high effective number of species compared to commercial areas and vacant lots (Fig. 4C), suggesting that land use types with more active management and planting initiatives may have more balanced abundances. Although residential land and the public right-of-way did not differ significantly in species richness or effective number of species, residential areas exhibited slightly higher species richness, while the public right-of-way had a higher effective number of species. The public right-of-way may have higher effective number due to fewer rare or ornamental species compared to private yards, since residents have authority over which species they plant in their yards, leading to more unique and one-off species occurring on residential land.

While the evenness of urban tree communities is often assumed to be low due to the dominance of a few species (Hilbert et al., 2023), we observed relatively high evenness across most land use types (Fig. 4D), consistent with previous work in Montreal (Hutt-Taylor and Ziter, 2022). Specifically, the public right-of-way and institutional land had the highest evenness, while parks, residential land, and vacant lots showed intermediate values, and commercial land had the lowest evenness. These patterns likely reflect a combination of planting practices, management objectives, and spatial constraints. For example, commercial spaces often have limited planting area and prioritize a few ornamental or low-maintenance species, which can reduce evenness. Parks, despite having higher effective number of species, were not necessarily more even because a few dominant species, such as maples, still make up most of the individuals, reducing evenness among species. These results suggest that the prevalence of rare species in all land use types prevents strong differences in evenness.

While the public right-of-way and residential subplots supported comparable species richness, their tree communities were distinct in species composition (Fig. 6). Because city managers and homeowners prioritize different traits and functions in tree selection (Conway and Vander Vecht, 2015), we expected residential and public land uses to support distinct species compositions. Public rights-of-way are generally planted by municipalities with a limited set of species selected for stress tolerance, low maintenance, and uniform growth (Conway and Vander Vecht, 2015; Martin and Conway, 2025), resulting in relatively consistent assemblages across sites, as we found in our results. While research shows that residents are influenced by individual preferences (Conway, 2016; Su et al., 2022) and we would expect that to lead to differences in composition among yards, we found that residential land was similarly homogenous to the public right-of-way across our plots. This is surprising, because previous research in Montreal (Hutt-Taylor and Ziter, 2022; Sousa-Silva et al., 2023b) found that the public right-of-way differed from residential land. One possible explanation for this difference is that our study captured city-wide patterns across diverse neighbourhoods, while previous work focused on a single neighbourhood where social, economic, and environmental conditions were more consistent. The broader spatial extent of our sampling may have averaged out localized differences between residential land and the public right-of-way, leading to greater similarity. We also found that residential land and the public right-of-way significantly differed in composition from other land use types. Parks, institutional grounds, commercial areas, and vacant lots showed a greater overlap in ordination space, suggesting that there are similarities in community composition across these land use types. This may be because these land uses are shaped by a combination of functional goals and landscape designs across the city, resulting in overlapping species assemblages across sites.

When looking at which species significantly influenced composition across land use types, only residential land and the public right-of-way stood out, highlighting their distinct roles in structuring urban tree communities. Residential land had the largest number of associated species, which may reflect the combined effects of homeowner planting preferences and diverse landscaping practices. The public right-of-way, by contrast, had only a few associated species. For example, Eastern white-cedar (*Thuja occidentalis*), Norway spruce (*Picea abies*), and common pear (*Pyrus communis*) were correlated with residential land, while honey locust (*Gleditsia triacanthos*), Kentucky coffeetree (*Gymnocladus dioicus*) and gingko (*Gingko biloba*) were associated with the public right-of-way. This pattern is consistent with expectations, as conifers – typically bottom-heavy – and fruit trees are not common on public land, while these latter species are top-heavy classic street trees that are commonly planted on the public right-of-way but occur less frequently on other land use types.

Across all land use types, we observed differences in composition and structure metrics both within and among plots. For example, parks across different neighbourhoods (i.e., among plots) varied in tree species composition and size distribution, with some containing predominantly maple trees, while others included more diverse mixes of native and ornamental species – but these differences among parks were smaller than those observed between parks and other land use types, such as residential yards or commercial lots. Similarly, residential subplots differed in tree abundance and species richness depending on homeowner preferences, yard size, and management practices, yet overall residential land consistently supported higher richness than commercial or vacant lots. These patterns highlight that while neighbourhood-level dynamics such as local planting initiatives, socio-economic factors, or municipal management practices can influence the urban forest within an area, land use type remains a stronger predictor of diversity, composition, and structure across the city. Understanding both plot-level and land use type-level variation is therefore critical for urban forest planning, as it allows managers to target interventions that account for both local context and broader land use patterns.

We hypothesized that land use types would differ in their tree diversity, composition, and structure, but it was unclear whether all private land use types would exhibit similar patterns, or whether some private types might be more similar to public land use types than others. Our results demonstrate that we need to look specifically at different land use types rather than lump them into private vs. public to fully characterize the urban forest. For example, institutional land exhibited composition metrics more similar to parks, aligning with a public land use type, but structure metrics were more similar residential land, reflecting characteristics of private land. Similarly, commercial land displayed low species richness and basal area akin to vacant lots, yet their tree density was similar to both parks and institutional land, possessing characteristics of private and public land uses. These findings indicate that we cannot simply group land use types into private versus public, because doing so would obscure important differences in diversity, composition, and structure that are relevant for urban forest management. Instead, it is necessary to evaluate each land use type individually, across the city, to accurately capture patterns in diversity, composition, and structure and to guide targeted management strategies.

## Conclusion

This study represents one of the most comprehensive urban forest inventories and analyses conducted for a large North American city, capturing both public and private trees across multiple land use types. Although other initiatives such as i-Tree Eco offer important urban forest monitoring frameworks across North America, they are typically applied to smaller datasets and often emphasize woodlots or dense forest stands. By contrast, our inventory covers a larger spatial extent and purposefully incorporates all local land use types—including those that are typically under-sampled—providing a more contextually complete picture of the urban forest. Given the limited number of comparable large-scale inventories, there remains a need for more comprehensive assessments to understand urban forest diversity, composition, and structural trends nationwide.

We aimed to explore the similarities and differences in the urban forest among land use types under different management. We found that both public and private land use types possess meaningful differences in tree diversity, composition, and structure across the city of Montreal. Understanding differences in tree diversity and structure within different land use types can allow for better management of the urban forest by allowing both municipal and private stakeholders to target land uses that have the potential to improve urban forest diversity and resilience, and consequently people's wellbeing. Our findings reveal several actionable insights. For example, institutional land currently has low tree density and diversity based on the available plantable area, suggesting these spaces are underutilized opportunities for expanding urban forest cover. Furthermore, hedges, particularly those that include coniferous species such as *Thuja occidentalis*, contribute substantially to urban forest structure and should be considered in planting and management strategies. Private land use types represent a large proportion of plantable land across our study sites, demonstrating the potential of private land to further planting initiatives across the city. As cities dedicate themselves to planting more trees, finding ways to assess private trees on a city-wide scale will allow for a better understanding of the urban forest and drive initiatives that focus on private land to increase urban forest diversity and resilience. Our work also highlights the need for focusing on urban forest metrics past species richness and abundance, as tree dominance, size structure, and species composition can vary greatly between land use types and provide insights on the current state of the urban forest.

Beyond these specific findings, this research establishes a critical foundation for future work on Montreal's urban forest, providing the baseline data and analytical framework needed to explore how urban forests interact with human health, wildlife habitat, other trophic levels, and the city's social and cultural dimensions. Our tree inventory set the stage for long-term monitoring and interdisciplinary research linking urban forests to ecosystem services, equity, and quality of life in Montreal. Our work not only advances our understanding of the current urban forest but also lays the groundwork for future research that will guide the sustainable planning and management of greener, healthier cities in Canada.

# References


Almas, A., Conway, T., 2018. Resident Attitudes and Actions Toward Native Tree Species: A Case Study of Residents in Four Southern Ontario Municipalities. Arboric. Urban For. AUF 44, 101–115. https://doi.org/10.48044/jauf.2018.009

Alonzo, M., Ibsen, P.C., Locke, D.H., 2025. Urban Trees and Cooling: A Review of the Recent Literature (2018 to 2024). Arboric. Urban For. AUF 51, 420–444. https://doi.org/10.48044/jauf.2025.023

Anderson, E.C., Avolio, M.L., Sonti, N.F., LaDeau, S.L., 2021. More than green: Tree structure and biodiversity patterns differ across canopy change regimes in Baltimore's urban forest. Urban For. Urban Green. 65, 127365. https://doi.org/10.1016/j.ufug.2021.127365

Anderson, E.C., Locke, D.H., Pickett, S.T.A., LaDeau, S.L., 2023. Just street trees? Street trees increase local biodiversity and biomass in higher income, denser neighborhoods. Ecosphere 14, e4389. https://doi.org/10.1002/ecs2.4389

Avolio, M., Pataki, D., Gillespie, T., Jenerette, G., McCarthy, H., Pincetl, S., Weller-Clarke, L., 2015. Tree diversity in southern California's urban forest: the interacting roles of social and environmental variables. Front. Ecol. Evol. 3. https://doi.org/10.3389/fevo.2015.00073

Avolio, M.L., Pataki, D.E., Trammell, T.L.E., Endter-Wada, J., 2018. Biodiverse cities: the nursery industry, homeowners, and neighborhood differences drive urban tree composition. Ecol. Monogr. 88, 259–276. https://doi.org/10.1002/ecm.1290

Bates, D., Maechler, M., Bolker, B., Walker, S., Christensen, R.H.B., Singmann, H., Dai, B., Scheipl, F., Grothendieck, G., Green, P., Fox, J., Bauer, A., Krivitsky P.N., Tanaka, E., Jagan, M., Boylan, R.D., 2025. lme4: Linear Mixed-Effects Models using "Eigen" and S4. https://doi.org/10.32614/CRAN.package.lme4

Berglihn, E.C., Gómez-Baggethun, E., 2021. Ecosystem services from urban forests: The case of Oslomarka, Norway. Ecosyst. Serv. 51, 101358. https://doi.org/10.1016/j.ecoser.2021.101358

Berland, A., Shuman, B., Manson, S.M., 2011. Simulated Importance of Dispersal, Disturbance, and Landscape History in Long-Term Ecosystem Change in the Big Woods of Minnesota. Ecosystems 14, 398–414. https:/doi.org/10.1007/s10021-011-9418-x

Blanusa, T., Garratt, M., Cathcart-James, M., Hunt, L., Cameron, R.W.F., 2019. Urban hedges: A review of plant species and cultivars for ecosystem service delivery in north-west Europe. Urban For. Urban Green. 44, 126391. https://doi.org/10.1016/j.ufug.2019.126391

Bourne, K.S., Conway, T.M., 2014. The influence of land use type and municipal context on urban tree species diversity. Urban Ecosyst. 17, 329–348. https://doi.org/10.1007/s11252-013-0317-0

Bowler, D.E., Buyung-Ali, L., Knight, T.M., Pullin, A.S., 2010. Urban greening to cool towns and cities: A systematic review of the empirical evidence. Landsc. Urban Plan. 97, 147–155. https://doi.org/10.1016/j.landurbplan.2010.05.006

Brooks, M., Bolker, B., Kristensen, K., Maechler, M., Magnusson, A., McGillycuddy, M., Skaug, H., Nielsen, A., Berg, C., Bentham, K. van, Sadat, N., Lüdecke, D., Lenth, R., O'Brien, J., Geyer, C.J., Jagan, M., Wiernik, B., Stouffer, D.B., Agronah, M., Akdur, H.T.K., Bové, D.S., Krieger, N., 2025. glmmTMB: Generalized Linear Mixed Models using Template Model Builder. https://doi.org/10.32614/CRAN.package.glmmTMB

Buxton, R.T., Hudgins, E.J., Lavigne, E., Villeneuve, P.J., Prince, S.A., Pearson, A.L., Halsall, T., Robichaud, C., Bennett, J.R., 2024. Mental health is positively associated with biodiversity in Canadian cities. Commun. Earth Environ. 5, 310. https://doi.org/10.1038/s43247-024-01482-9

Cadotte, M.W., Carscadden, K., Mirotchnick, N., 2011. Beyond species: functional diversity and the maintenance of ecological processes and services. J. Appl. Ecol. 48, 1079–1087. https://doi.org/10.1111/j.1365-2664.2011.02048.x

City of Cambridge, 2015. Urban Forest Plan 2015-2034. City of Cambridge, Ontario. https://www.cambridge.ca/en/learn-about/resources/Accesible-PDFs/Cambridge-Urban-Forest-Plan-2015-2034.pdf

Chambers-Ostler, A., Walker, H., Doick, K.J., 2024. The role of the private tree in bringing diversity and resilience to the urban forest. Urban For. Urban Green. 91, 127973. https://doi.org/10.1016/j.ufug.2023.127973

Chao, A., Gotelli, N.J., Hsieh, T.C., Sander, E.L., Ma, K.H., Colwell, R.K., Ellison, A.M., 2014. Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. Ecol. Monogr. 84, 45–67. https://doi.org/10.1890/13-0133.1

Chao, A., Jost, L., 2012. Coverage-based rarefaction and extrapolation: standardizing samples by completeness rather than size. Ecology 93, 2533–2547. https://doi.org/10.1890/11-1952.1

Chen, J., Liang, X., Liu, Z., Gong, W., Chen, Y., Hyyppä, J., Kukko, A., Wang, Y., 2024. Tree species recognition from close-range sensing: A review. Remote Sens. Environ. 313, 114337. https://doi.org/10.1016/j.rse.2024.114337

Chiu, C.-H., 2023. Sample coverage estimation, rarefaction, and extrapolation based on sample-based abundance data. Ecology 104, e4099. https://doi.org/10.1002/ecy.4099

City of Toronto, Transportation Services, 2017. Road Engineering Design Guidelines: Lane Widths Guideline. Road Eng. Des. Guidel. Version 2.0.

Colwell, R.K., Chao, A., Gotelli, N.J., Lin, S.-Y., Mao, C.X., Chazdon, R.L., Longino, J.T., 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. J. Plant Ecol. 5, 3–21. https://doi.org/10.1093/jpe/rtr044

Communauté métropolitaine de Montréal, 2022. Utilisation du sol - données géoréferencées. Données fournis par la Communauté métropolitaine de Montréal. https://observatoire.cmm.qc.ca/produits/donnees-georeferencees/#utilisation_du_sol

Conway, T.M., 2016. Tending their urban forest: Residents' motivations for tree planting and removal. Urban For. Urban Green. 17, 23–32. https://doi.org/10.1016/j.ufug.2016.03.008

Conway, T.M., Vander Vecht, J., 2015. Growing a diverse urban forest: Species selection decisions by practitioners planting and supplying trees. Landsc. Urban Plan. 138, 1–10. https://doi.org/10.1016/j.landurbplan.2015.01.007

Davies, Z.G., Fuller, R.A., Loram, A., Irvine, K.N., Sims, V., Gaston, K.J., 2009. A national scale inventory of resource provision for biodiversity within domestic gardens. Biol. Conserv. 142, 761–771. https://doi.org/10.1016/j.biocon.2008.12.016

Dilley, J., Wolf, K.L., 2013. Homeowner Interactions with Residential Trees in Urban Areas. Arboric. Urban For. AUF 39, 267–277. https://doi.org/10.48044/jauf.2013.034

Données Québec, 2023. Bâtiments 2D [Jeu de données]. Données Qué. https://www.donneesquebec.ca/recherche/dataset/vmtl-batiment-2d

Drew-Smythe, J.J., Davila, Y.C., McLean, C.M., Hingee, M.C., Murray, M.L., Webb, J.K., Krix, D.W., Murray, B.R., 2023. Community perceptions of ecosystem services and disservices linked to urban tree plantings. Urban For. Urban Green. 82, 127870. https://doi.org/10.1016/j.ufug.2023.127870

Duinker, P., Foster, D., 2017. The HRM Urban Forest in 2016: An i-Tree Eco Assessment for Halifax Regional Municipality. Dalhousie University, Halifax, NS. https://www.itreetools.org/documents/319/FosterDuinker_2017_iTreeEcoForHalifax_Feb2017.pdf

Dyson, K., Ziter, C., Fuentes, T.L., Patterson, M.S., 2019. Conducting urban ecology research on private property: advice for new urban ecologists. J. Urban Ecol. 5, juz001. https://doi.org/10.1093/jue/juz001

Dyson, K., 2019. Vegetation communities on commercial developments are heterogenous and determined by development and landscaping decisions, not socioeconomics. Urban For. Urban Green. 14, e0222069. https://doi.org/10.1371/journal.pone.0222069

Fan, C., Johnston, M., Darling, L., Scott, L., Liao, F.H., 2019. Land use and socio-economic determinants of urban forest structure and diversity. Landsc. Urban Plan. 181, 10–21. https://doi.org/10.1016/j.landurbplan.2018.09.012

Fleming, A., Conway, T.M., Sleightholm, P., McKay, J., 2025. Aerial Imagery as a Tool for Monitoring Urban Tree Retention: Applications, Strengths and Challenges for Backyard Tree Planting Programs. Arboric. Urban For. AUF. https://doi.org/10.48044/jauf.2025.022

Fornoff, F., Klein, A., Hartig, F., Benadi, G., Venjakob, C., Schaefer, H.M., Ebeling, A., 2017. Functional flower traits and their diversity drive pollinator visitation. Oikos 126, 1020–1030. https://doi.org/10.1111/oik.03869

Fox, J., Weisberg, S., Price, B., Adler, D., Bates, D., Baud-Bovy, G., Bolker, B., Ellison, S., Firth, D., Friendly, M., Gorjanc, G., Graves, S., Heiberger, R., Krivitsky, P., Laboissiere, R., Maechler, M., Monette, G., Murdoch, D., Nilsson, H., Ogle, D., Ripley, B., Short, T., Venables, W., Walker, S., Winsemius, D., Zeileis, A., R-Core, 2024. car: Companion to Applied Regression. https://doi.org/10.32614/CRAN.package.car

García, I., 2025. Residential Green Infrastructure: Unpacking Motivations and Obstacles to Single-Family-Home Tree Planting in Diverse, Low-Income Urban Neighborhoods. Sustainability 17, 7412. https://doi.org/10.3390/su17167412

Gerrish, E., Watkins, S.L., 2018. The relationship between urban forests and income: A meta-analysis. Landsc. Urban Plan. 170, 293–308. https://doi.org/10.1016/j.landurbplan.2017.09.005

Gonçalves, S.F., Lourenço, A.C. de P., Bueno Filho, J.S. de S., de Toledo, M.C.B., 2021. Characteristics of residential backyards that contribute to conservation and diversity of urban birds: A case study in a Southeastern Brazilian city. Urban For. Urban Green. 61, 127095. https://doi.org/10.1016/j.ufug.2021.127095

Google Maps, 2025. Google Satellite [basemap]. Google Maps. https://www.google.com/maps/@45.4459392,-73.6329728,12z?entry=ttu&g_ep=EgoyMDI1MTAyOS4yIKXMDSoASAFQAw%3D%3D

Hilbert, D.R., Koeser, A.K., Andreu, M.G., Hansen, G., Roman, L.A., Thetford, M., Thompson, G.L., 2023. Conceptualizing the human drivers of low tree diversity in planted urban landscapes. Ambio 52, 1532–1542. https://doi.org/10.1007/s13280-023-01876-7

Hill, M.O., 1973. Diversity and Evenness: A Unifying Notation and Its Consequences. Ecology 54, 427–432. https://doi.org/10.2307/1934352

Hsieh, T.C., Ma, K.H., Chao, A., 2025. iNEXT: Interpolation and Extrapolation for Species Diversity. https://doi.org/10.32614/CRAN.package.iNEXT

Huff, E.S., Johnson, M.L., Roman, L.A., Sonti, N.F., Pregitzer, C.C., Campbell, L.K., McMillen, H., 2020. A Literature Review of Resilience in Urban Forestry. Arboric. Urban For. AUF 46, 185–196. https://doi.org/10.48044/jauf.2020.014

Hurlbert, S.H., 1971. The Nonconcept of Species Diversity: A Critique and Alternative Parameters. Ecology 52, 577–586. https://doi.org/10.2307/1934145

Hutt-Taylor, K., Ziter, C.D., 2022. Private trees contribute uniquely to urban forest diversity, structure and service-based traits. Urban For. Urban Green. 78, 127760. https://doi.org/10.1016/j.ufug.2022.127760

Isbell, F., Craven, D., Connolly, J., Loreau, M., Schmid, B., Beierkuhnlein, C., Bezemer, T.M., Bonin, C., Bruelheide, H., de Luca, E., Ebeling, A., Griffin, J.N., Guo, Q., Hautier, Y., Hector, A., Jentsch, A., Kreyling, J., Lanta, V., Manning, P., Meyer, S.T., Mori, A.S., Naeem, S., Niklaus, P.A., Polley, H.W., Reich, P.B., Roscher, C., Seabloom, E.W., Smith, M.D., Thakur, M.P., Tilman, D., Tracy, B.F., van der Putten, W.H., van Ruijven, J., Weigelt, A., Weisser, W.W., Wilsey, B., Eisenhauer, N., 2015. Biodiversity increases the resistance of ecosystem productivity to climate extremes. Nature 526, 574–577. https://doi.org/10.1038/nature15374

Jabbar, M., Yusoff, M.M., Shafie, A., 2022. Assessing the role of urban green spaces for human well-being: a systematic review. GeoJournal 87, 4405–4423. https://doi.org/10.1007/s10708-021-10474-7

Jenerette, G.D., Clarke, L.W., Avolio, M.L., Pataki, D.E., Gillespie, T.W., Pincetl, S., Nowak, D.J., Hutyra, L.R., McHale, M., McFadden, J.P., Alonzo, M., 2016. Climate tolerances and trait choices shape continental patterns of urban tree biodiversity. Glob. Ecol. Biogeogr. 25, 1367–1376. https://doi.org/10.1111/geb.12499

Keller, J., Konijnendijk, C., 2012. Keller and Konijnendijk: A Comparative Analysis of Municipal Urban Tree Inventories Short Communication: A Comparative Analysis of Municipal Urban Tree Inventories of Selected Major Cities in North America and Europe. Arboric. Urban For. 38, 24–30. https://doi.org/10.48044/jauf.2012.005

Kendal, D., Dobbs, C., Lohr, V.I., 2014. Global patterns of diversity in the urban forest: Is there evidence to support the 10/20/30 rule? Urban For. Urban Green. 13, 411–417. https://doi.org/10.1016/j.ufug.2014.04.004

Kenney, W. A., Puric-Mladenovic, D.. 1995. Neighbourwoods© (neighbourwoods.org).

Kuntz, P., Environmental Design and Management Ltd., The Bristol Group, 2006. St. John’s Urban Forest Management Master Plan (Master Plan Report). City of St. John’s. https://www.stjohns.ca/en/building-development/resources/Planning-Reports/St.John'-Urban-Forest-Master-Plan.pdf

Landry, F., Dupras, J., Messier, C., 2020. Convergence of urban forest and socio-economic indicators of resilience: A study of environmental inequality in four major cities in eastern Canada. Landsc. Urban Plan. 202, 103856. https://doi.org/10.1016/j.landurbplan.2020.103856

Larouche, J., Rioux, D., Bardekjian, A.C., Gélinas, N., 2021. Urban forestry research needs identified by Canadian municipalities. For. Chron. 97, 158–167. https://doi.org/10.5558/tfc2021-017

Lenth, R.V., Piaskowski, J., Banfai, B., Bolker, B., Buerkner, P., Giné-Vázquez, I., Hervé, M., Jung, M., Love, J., Miguez, F., Riebl, H., Singmann, H., 2025. emmeans: Estimated Marginal Means, aka Least-Squares Means. https://doi.org/10.32614/CRAN.package.emmeans

Li, L., Kross, A., Ziter, C.D., Eicker, U., 2025. Analyzing spatial patterns of urban green infrastructure for urban cooling and social equity. Urban For. Urban Green. 112, 128983. https://doi.org/10.1016/j.ufug.2025.128983

Liu, J., Slik, F., 2022. Are street trees friendly to biodiversity? Landsc. Urban Plan. 218, 104304. https://doi.org/10.1016/j.landurbplan.2021.104304

Locke, D.H., Hall, B., Grove, J.M., Pickett, S.T.A., Ogden, L.A., Aoki, C., Boone, C.G., O'Neil-Dunne, J.P.M., 2021. Residential housing segregation and urban tree canopy in 37 US Cities. Npj Urban Sustain. 1, 1–9. https://doi.org/10.1038/s42949-021-00022-0

Ma, B., Hauer, R.J., Östberg, J., Koeser, A.K., Wei, H., Xu, C., 2021. A global basis of urban tree inventories: What comes first the inventory or the program. Urban For. Urban Green. 60, 127087. https://doi.org/10.1016/j.ufug.2021.127087

Magarik, Y.A.S., Roman, L.A., Henning, J.G., 2020. How should we measure the DBH of multi-stemmed urban trees? Urban For. Urban Green. 47, 126481. https://doi.org/10.1016/j.ufug.2019.126481

Martin, A.J.F., Conway, T.M., 2025. Historical drivers of street tree species selection: A comparative archival study of Canadian prairie cities. Trees For. People 22, 101044. https://doi.org/10.1016/j.tfp.2025.101044

Martin, A.J.F., Olson, L.G., Bacon, E.S., Almas, A.D., Ziter, C.D., Conway, T.M., 2025. A biogeographical analysis of taxonomic diversity and native species dominance in 32 Canadian street tree populations. Urban Ecosyst. 28, 158. https://doi.org/10.1007/s11252-025-01764-w

McGlinn, D., Xiao, X., McGill, B., May, F., Engel, T., Oliver, C., Blowes, S., Knight, T., Purschke, O., Gotelli, N., Chase, J., 2024. mobr: Measurement of Biodiversity. https://doi.org/10.32614/CRAN.package.mobr

McPherson, E.G., Simpson, J.R., Xiao, Q., Wu, C., 2011. Million trees Los Angeles canopy cover and benefit assessment. Landsc. Urban Plan. 99, 40–50. https://doi.org/10.1016/j.landurbplan.2010.08.011

Mejía, G.A., Groffman, P.M., Avolio, M.L., Bratt, A.R., Engebretson, J.M., Grijseels, N., Hall, S.J., Hobbie, S.E., Lerman, S.B., Litvak, E., Locke, D.H., Narango, D.L., Padullés Cubino, J., Pataki, D.E., Trammell, T.L., 2024. How do urban trees vary across the US? It depends on where and how you look. Front. Ecol. Environ. 22, e2777. https://doi.org/10.1002/fee.2777

Morgenroth, J., Östberg, J., Konijnendijk van den Bosch, C., Nielsen, A.B., Hauer, R., Sjöman, H., Chen, W., Jansson, M., 2016. Urban tree diversity—Taking stock and looking ahead. Urban For. Urban Green. 15, 1–5. https://doi.org/10.1016/j.ufug.2015.11.003

Narváez Vallejo, A., Schwarz-v.Raumer, H.-G., Eisenberg, B., 2024. The interplay of land-use and land-ownership as a key for urban greening management. Urban For. Urban Green. 99, 128442. https://doi.org/10.1016/j.ufug.2024.128442

Natural Resources Canada, 2025. Federal Investment Contributes to the Planting of 500,000 Trees in Montreal. https://www.canada.ca/en/natural-resources-canada/news/2025/03/federal-investment-contributes-to-the-planting-of-500000-trees-in-montreal.html.

Nitoslawski, S.A., Duinker, P.N., Bush, P.G., 2016. A review of drivers of tree diversity in suburban areas: Research needs for North American cities. Environ. Rev. 24, 471–483. https://doi.org/10.1139/er-2016-0027

Nowak, D.J., Aevermann, T., 2019. Tree compensation rates: Compensating for the loss of future tree values. Urban For. Urban Green. 41, 93–103. https://doi.org/10.1016/j.ufug.2019.03.014

Nowak, D.J., Greenfield, E.J., Hoehn, R.E., Lapoint, E., 2013. Carbon storage and sequestration by trees in urban and community areas of the United States. Environ. Pollut. 178, 229–236. https://doi.org/10.1016/j.envpol.2013.03.019

Oksanen, J., Simpson, G.L., Blanchet, F.G., Kindt, R., Legendre, P., Minchin, P.R., O'Hara, R.B., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H., Barbour, M., Bedward, M., Bolker, B., Borcard, D., Borman, T., Carvalho, G., Chirico, M., Caceres, M.D., Durand, S., Evangelista, H.B.A., FitzJohn, R., Friendly, M., Furneaux, B., Hannigan, G., Hill, M.O., Lahti, L., Martino, C., McGlinn, D., Ouellette, M.-H., Cunha, E.R., Smith, T., Stier, A., Braak, C.J.F.T., Weedon, J., 2025. vegan: Community Ecology Package. https://doi.org/10.32614/CRAN.package.vegan

OpenStreetMap contributors, 2025. OpenStreetMap [Building Footprints Montreal] [WWW Document]. OpenStreetMap Found. Available as open data under the Open Data Commons Open Database License (ODbL) at openstreetmap.org.

Ordóñez Barona, C., St Denis, A., Jung, J., Bassett, C.G., Delagrange, S., Duinker, P., Conway, T., 2024. A content analysis of urban forest management plans in Canada: Changes in social-ecological objectives over time. Landsc. Urban Plan. 251, 105154. https://doi.org/10.1016/j.landurbplan.2024.105154

Ordóñez-Barona, C., 2017. How different ethno-cultural groups value urban forests and its implications for managing urban nature in a multicultural landscape: A systematic review of the literature. Urban For. Urban Green., Special feature:TURFGRASS 26, 65–77. https://doi.org/10.1016/j.ufug.2017.06.006

Ossola, A., Locke, D., Lin, B., Minor, E., 2019. Yards increase forest connectivity in urban landscapes. Landsc. Ecol. 34, 2935–2948. https://doi.org/10.1007/s10980-019-00923-7

Padullés Cubino, J., Retana, J., 2023. Socioeconomics explain tree diversity, abundance, and composition in the compact city of Barcelona, Spain. Landsc. Urban Plan. 236, 104778. https://doi.org/10.1016/j.landurbplan.2023.104778

Paquette, A., Sousa-Silva, R., Faticov, M., Schillé, L., Bacon, E., Cameron, E., Fraysse, J., Gagnon Koudji, E., Poirier, S., Rondeau-Leclaire, J., Tardif, S., Handa, T., Laforest-Lapointe, I., Puric-Mladenovic, D., Ziter, C., 2025, in-review. Montreal Urban Observatory: research platform to monitor urban forest ecosystems for global change adaptation and health. Ecosphere.

Paquette, A., Sousa-Silva, R., Maure, F., Cameron, E., Belluau, M., Messier, C., 2021. Praise for diversity: A functional approach to reduce risks in urban forests. Urban For. Urban Green. 62, 127157. https://doi.org/10.1016/j.ufug.2021.127157

Paradis, E., Blomberg, S., Bolker [aut, B., cph, Brown, J., Claramunt, S., Claude, J., Cuong, H.S., Desper, R., Didier, G., Durand, B., Dutheil, J., Ewing, R.J., Gascuel, O., Guillerme, T., Heibl, C., Ives, A., Jones, B., Krah [aut, F., cph, Lawson, D., Lefort, V., Legendre, P., Lemon, J., Louvel, G., Marotta, F., Marcon [aut, E., cph, McCloskey, R., Nylander, J., Opgen-Rhein, R., Popescu, A.-A., Royer-Carenzi, M., Schliep, K., Strimmer, K., Vienne, D. de, 2024. ape: Analyses of Phylogenetics and Evolution. https://doi.org/10.32614/CRAN.package.ape

Pataki, D.E., McCarthy, H.R., Gillespie, T., Jenerette, G.D., Pincetl, S., 2013. A trait-based ecology of the Los Angeles urban forest. Ecosphere 4, 1–20. https://doi.org/10.1890/ES13-00017.1

Pearce, L.M., Davison, A., Kirkpatrick, J.B., 2015. Personal encounters with trees: The lived significance of the private urban forest. Urban For. Urban Green. 14, 1–7. https://doi.org/10.1016/j.ufug.2014.11.003

Pearce, L.M., Kirkpatrick, J.B., Davison, A., 2013. Using Size Class Distributions of Species to Deduce the Dynamics of the Private Urban Forest. Arboric. Urban For. AUF 39, 74–84. https://doi.org/10.48044/jauf.2013.011

Plant, L., Kendal, D., 2019. Toward Urban Forest Diversity: Resident Tolerance for Mixtures of Tree Species Within Streets. Arboric. Urban For. AUF 45, 41–53. https://doi.org/10.48044/jauf.2019.004

QGIS Development Team, 2024. QGIS Geographic Information System (Version 3.38). Open Source Geospatial Foundation Project. https:/qgis.org

Rendon, P., Love, N., Pawlak, C., Yost, J., Ritter, M., Doremus, J., 2024. Street tree diversity and urban heat. Urban For. Urban Green. 91, 128180. https://doi.org/10.1016/j.ufug.2023.128180

Riedman, E., Roman, L.A., Pearsall, H., Maslin, M., Ifill, T., Dentice, D., 2022. Why don't people plant trees? Uncovering barriers to participation in urban tree planting initiatives. Urban For. Urban Green. 73, 127597. https://doi.org/10.1016/j.ufug.2022.127597

Roman, L.A., Conway, T.M., Eisenman, T.S., Koeser, A.K., Ordóñez Barona, C., Locke, D.H., Jenerette, G.D., Östberg, J., Vogt, J., 2021. Beyond 'trees are good': Disservices, management costs, and tradeoffs in urban forestry. Ambio 50, 615–630. https://doi.org/10.1007/s13280-020-01396-8

Roman, L.A., Pearsall, H., Eisenman, T.S., Conway, T.M., Fahey, R.T., Landry, S., Vogt, J., van Doorn, N.S., Grove, J.M., Locke, D.H., Bardekjian, A.C., Battles, J.J., Cadenasso, M.L., van den Bosch, C.C.K., Avolio, M., Berland, A., Jenerette, G.D., Mincey, S.K., Pataki, D.E., Staudhammer, C., 2018. Human and biophysical legacies shape contemporary urban forests: A literature synthesis. Urban For. Urban Green. 31, 157–168. https://doi.org/10.1016/j.ufug.2018.03.004

R Core Team, 2024. R: A Language and Environment for Statistical Computing (Version 4.4.2.). R Foundation for Statistical Computing, Vienna, Austria. https:/www.R-project.org/

Shakeel, T., Conway, T.M., 2014. Individual households and their trees: Fine-scale characteristics shaping urban forests. Urban For. Urban Green. 13, 136–144. https://doi.org/10.1016/j.ufug.2013.11.004

Shannon, C.E., 1948. A mathematical theory of communication. Bell Syst. Tech. J. 27, 379–423. https://doi.org/10.1002/j.1538-7305.1948.tb01338.x

Simpson, E.H., 1949. Measurement of Diversity. Nature 163, 688–688. https://doi.org/10.1038/163688a0

Sousa-Silva, R., Cameron, E., Paquette, A., 2021. Prioritizing Street Tree Planting Locations to Increase Benefits for All Citizens: Experience From Joliette, Canada. Front. Ecol. Evol. 9. https://doi.org/10.3389/fevo.2021.716611

Sousa-Silva, R., Duflos, M., Ordóñez Barona, C., Paquette, A., 2023a. Keys to better planning and integrating urban tree planting initiatives. Landsc. Urban Plan. 231, 104649. https://doi.org/10.1016/j.landurbplan.2022.104649

Sousa-Silva, R., Lambry, T., Cameron, E., Belluau, M., Paquette, A., 2023b. Urban forests – Different ownership translates to greater diversity of trees. Urban For. Urban Green. 88, 128084. https://doi.org/10.1016/j.ufug.2023.128084

Soverdi, 2025. La Société de Verdissement du Montréal Métropolitain: Un arbre pour mon quartier. Un Arbre Pour Mon Quart. https://unarbrepourmonquartier.org/a-propos/fonctionnement/.

Statistics Canada, 2022. Focus on Geography Series, 2021 Census - Montréal (Census division). https://www12.statcan.gc.ca/census-recensement/2021/as-sa/fogs-spg/page.cfm?lang=e&topic=1&dguid=2021A00032466.

Su, K., Ordóñez, C., Regier, K., Conway, T.M., 2022. Values and beliefs about urban forests from diverse urban contexts and populations in the Greater Toronto area. Urban For. Urban Green. 72, 127589. https://doi.org/10.1016/j.ufug.2022.127589

Suchocka, M., Heciak, J., Błaszczyk, M., Adamczyk, J., Gaworski, M., Gawłowska, A., Mojski, J., Kalaji, H.M., Kais, K., Kosno-Jończy, J., Heciak, M.W., 2023. Comparison of Ecosystem Services and Replacement Value calculations performed for urban trees. Ecosyst. Serv. 63, 101553. https://doi.org/10.1016/j.ecoser.2023.101553

Tyrväinen, L., Silvennoinen, H., Kolehmainen, O., 2003. Ecological and aesthetic values in urban forest management. Urban For. Urban Green. 1, 135–149. https://doi.org/10.1078/1618-8667-00014

Ucar, Z., Bettinger, P., Merry, K., Akbulut, R., Siry, J., 2018. Estimation of urban woody vegetation cover using multispectral imagery and LiDAR. Urban For. Urban Green., Wild urban ecosystems: challenges and opportunities for urban development 29, 248–260. https://doi.org/10.1016/j.ufug.2017.12.001

United Nations, 2019. World urbanization prospects: the 2018 revision. Department of Economic and Social Affairs, Population Division. https://population.un.org/wup/assets/WUP2018-Report.pdf

Ville de Montréal, 2023. Arbres publics sur le territoire de la Ville [CSV dataset]. Données Montr. https://donnees.montreal.ca/dataset/arbres

Warton, D.I., 2022. Analysing Discrete Data: The Generalised Linear Model, in: Warton, D.I. (Ed.), Eco-Stats: Data Analysis in Ecology: From t-Tests to Multivariate Abundances. Springer International Publishing, Cham, pp. 231–263. https://doi.org/10.1007/978-3-030-88443-7_10

Watkins, S.L., Gerrish, E., 2018. The relationship between urban forests and race: A meta-analysis. J. Environ. Manage. 209, 152–168. https://doi.org/10.1016/j.jenvman.2017.12.021

Wolf, K.L., Lam, S.T., McKeen, J.K., Richardson, G.R.A., van den Bosch, M., Bardekjian, A.C., 2020. Urban Trees and Human Health: A Scoping Review. Int. J. Environ. Res. Public. Health 17, 4371. https://doi.org/10.3390/ijerph17124371

Wood, S.L.R., Dupras, J., 2021. Increasing functional diversity of the urban canopy for climate resilience: Potential tradeoffs with ecosystem services? Urban For. Urban Green. 58, 126972. https://doi.org/10.1016/j.ufug.2020.126972

Wujeska-Klause, A., Pfautsch, S., 2020. The Best Urban Trees for Daytime Cooling Leave Nights Slightly Warmer. Forests 11, 945. https://doi.org/10.3390/f11090945

Yachi, S., Loreau, M., 1999. Biodiversity and ecosystem productivity in a fluctuating environment: The insurance hypothesis. Proc. Natl. Acad. Sci. 96, 1463–1468. https://doi.org/10.1073/pnas.96.4.1463

Yang, J., La Sorte, F.A., Pyšek, P., Yan, P., Nowak, D., McBride, J., 2015. The compositional similarity of urban forests among the world's cities is scale dependent. Glob. Ecol. Biogeogr. 24, 1413–1423. https://doi.org/10.1111/geb.12376

Yin, Y., Li, S., Xing, X., Zhou, X., Kang, Y., Hu, Q., Li, Y., 2024. Cooling Benefits of Urban Tree Canopy: A Systematic Review. Sustainability 16, 4955. https://doi.org/10.3390/su16124955

Zhang, J., Gou, Z., Zhang, F., Shutter, L., 2020. A study of tree crown characteristics and their cooling effects in a subtropical city of Australia. Ecol. Eng. 158, 106027. https://doi.org/10.1016/j.ecoleng.2020.106027

## Figures and Tables

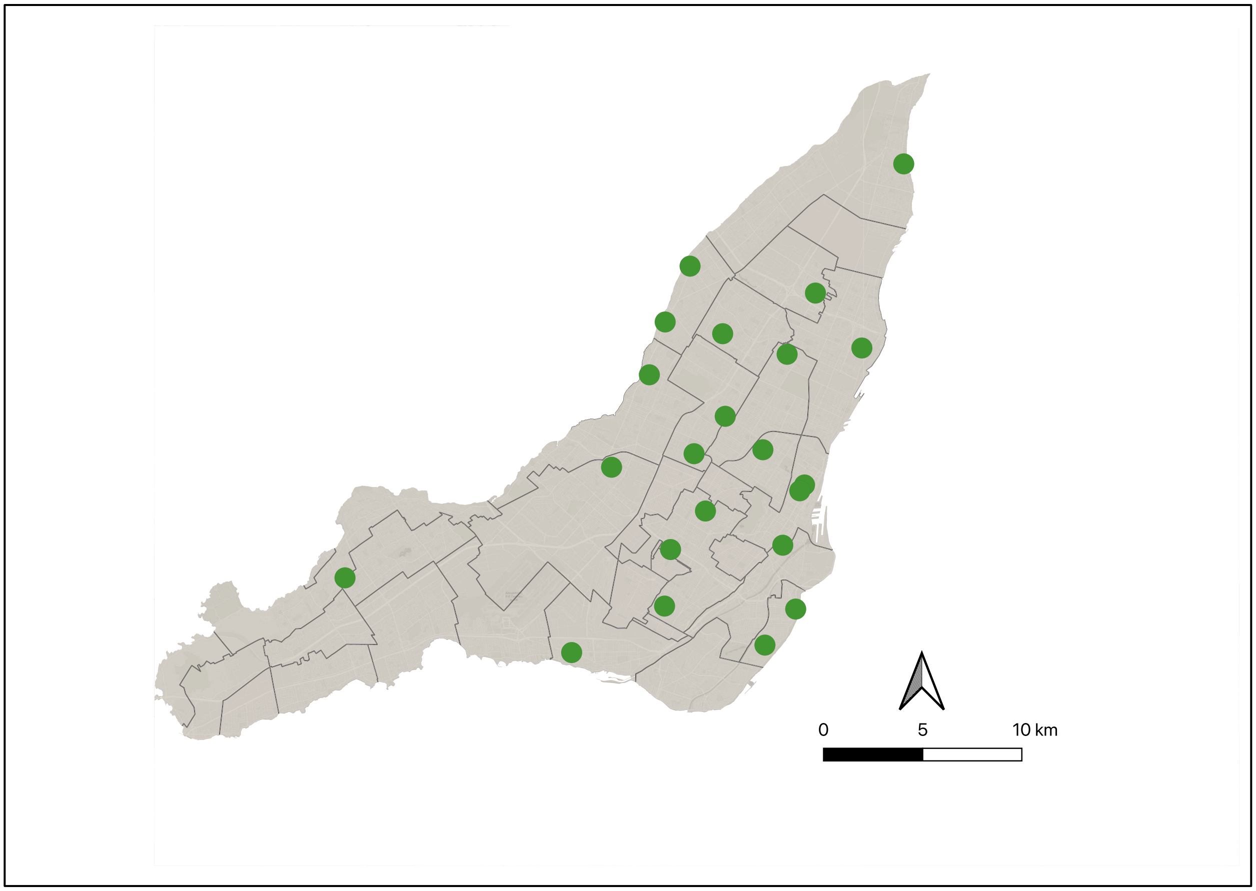


**Figure 1.** Visual representation of study sites for tree inventory across the island of Montreal, Quebec, part of the Montreal Urban Observatory. Green dots represent plot location and are not representative of size. The black lines delineate borough and municipality administration limits for the agglomeration of Montreal. Background map: Grey basemap, XYZ Tiles, © OpenStreetMap contributors. Map created using QGIS 3.28.

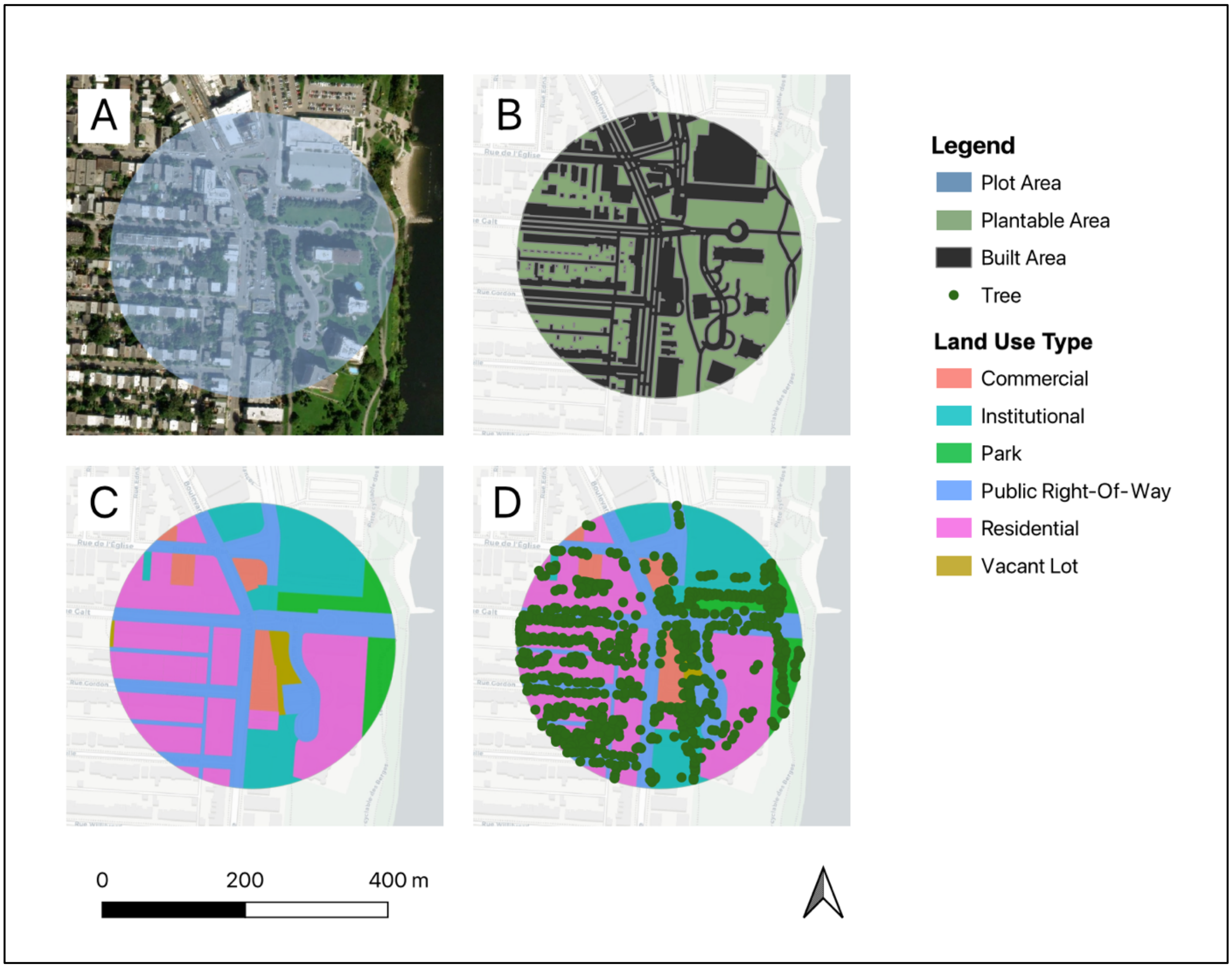


**Figure 2.** Visual representation of plot (16A) located in the Verdun borough of Montreal, Quebec. (A) Satellite imagery showing the plot boundary and buffer. (B) Built area (dark grey) and plantable area (green). (C) Land use types within the plot. (D) Land use types with overlaid tree polygons. Background maps: © OpenStreetMap contributors. Map created using QGIS 3.28.

**Table 1.** Description of public and private land use type classifications and site characteristics. Samples refer to the number of plots which contain that land use type out of a total 22 plot study area. Area and plantable area are presented as the sum across all plots.

| **Land use type** | **Description** | **Samples (N)** | **Area (ha)** | **Plantable area (ha)** |
|---|---|---|---|---|
| Commercial | Offices, shopping complexes, commercial businesses, and industrial sites | 15 | 13.4 | 2.4 |
| Institutional | School grounds, hospitals, and places of worship | 18 | 30.4 | 15.5 |
| Park | Publicly managed parks | 14 | 18.3 | 12.9 |
| Public Right-Of-Way | Land allotment for street trees, medians | 22 | 81.5 | 28.4 |
| Residential | Low density (1 dwelling), medium density (2-4 dwellings), and high density (5+ dwellings) residential homes, including apartment complexes | 22 | 125.3 | 72.2 |
| Vacant Lot | An empty or unused area of land not visibly managed by landowners | 10 | 1.6 | 1.1 |

**Table 2.** Summary tree inventory by land use type in study area. Shown are the total number of trees, number of genera, number of species, percentage of trees identified as hedges, and the median diameter at breast height (DBH) (cm) for each land use type.

| Land use type | No. of trees | No. of genera | No. of species | % hedges | Median DBH (cm) | Dominant species (desc.) |
|---|---|---|---|---|---|---|
| **Commercial** | 327 | 23 | 38 | 29.9 | 9.0 | *Thuja occidentalis* (26.3%)<br>*Ulmus pumila* (15.9%)<br>*Acer platanoides* (11.2%) |
| **Institutional** | 1,892 | 50 | 106 | 24.3 | 8.4 | *Thuja occidentalis* (21.8%)<br>*Acer platanoides* (7.8%)<br>*Acer saccharinum* (7.1%) |
| **Park** | 1,385 | 52 | 99 | 6.28 | 25.6 | *Acer saccharinum* (13%)<br>*Acer platanoides* (10.3%)<br>*Thuja occidentalis* (7.8%) |
| **Public Right-Of-Way** | 7,425 | 77 | 206 | 31.29 | 10.0 | *Thuja occidentalis* (22.3%)<br>*Acer platanoides* (10.6%)<br>*Gleditsia triacanthos* (6.5%) |
| **Residential** | 17,041 | 94 | 249 | 48.9 | 7.21 | *Thuja occidentalis* (45.2%)<br>*Syringa vulgaris* (6.6%)<br>*Acer platanoides* (5.3%) |
| **Vacant Lot** | 269 | 26 | 38 | 26.8 | 8.30 | *Thuja occidentalis* (29%)<br>*Acer negundo* (10.4%)<br>*Acer platanoides* (9.7%) |

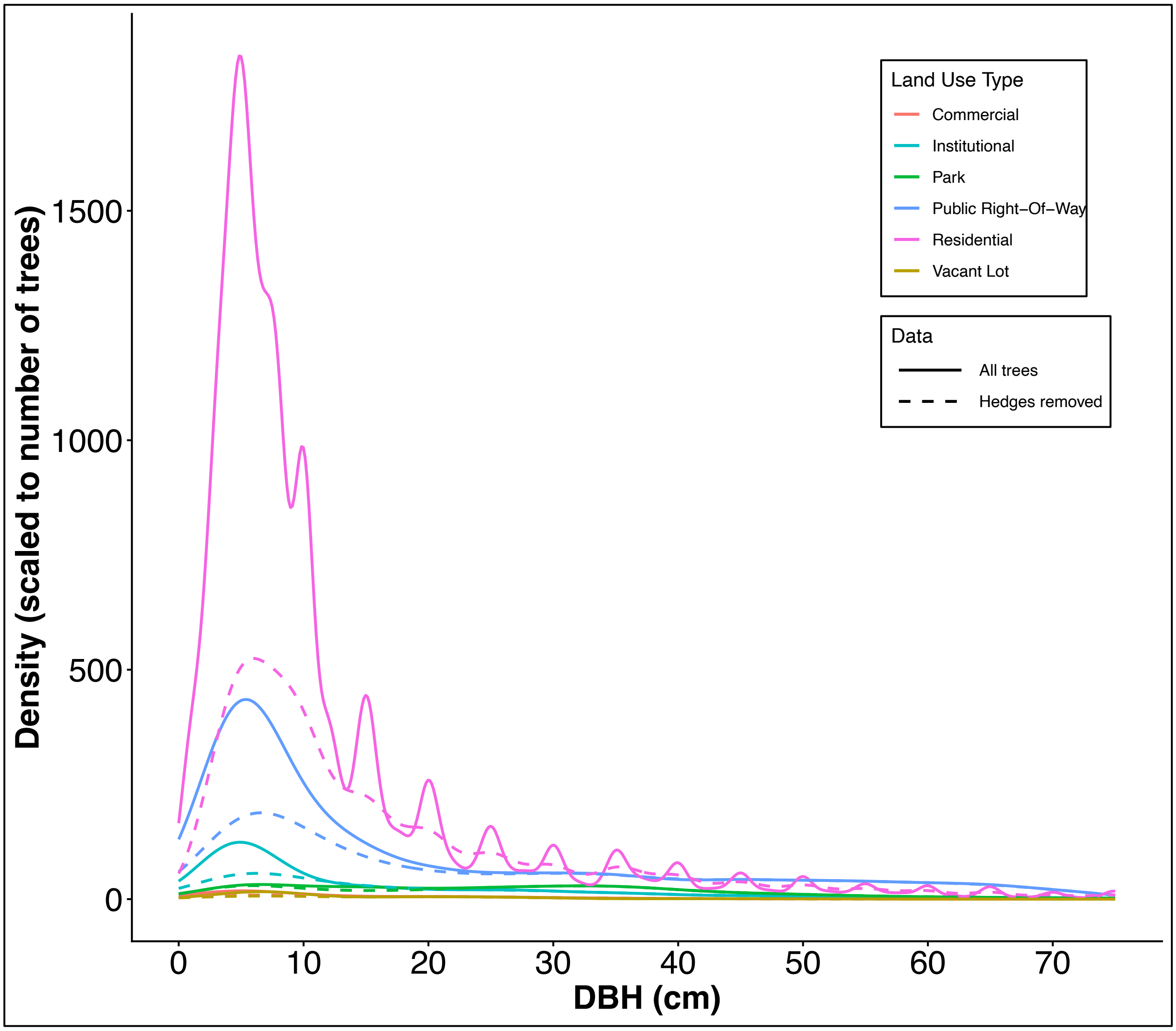


**Figure 3.** Scaled kernel density plot comparing the distribution of DBH (cm) in six land use types. Solid lines represent show all trees and dashed lines show distribution with hedges removed. Curves coloured by land use type. The y-axis represents smoothed tree counts, scaled to the number of trees in each land use type (density x number of trees). DBH is shown up to 75 cm for visualization purposes; 409 trees with larger DBH values are not shown. Tree size distribution in residential land showed pronounced peaks at whole-number DBH values (e.g., 10, 15, 25). This pattern is likely results from a higher number of estimated DBH in residential areas, in part due to the high proportion of hedges.

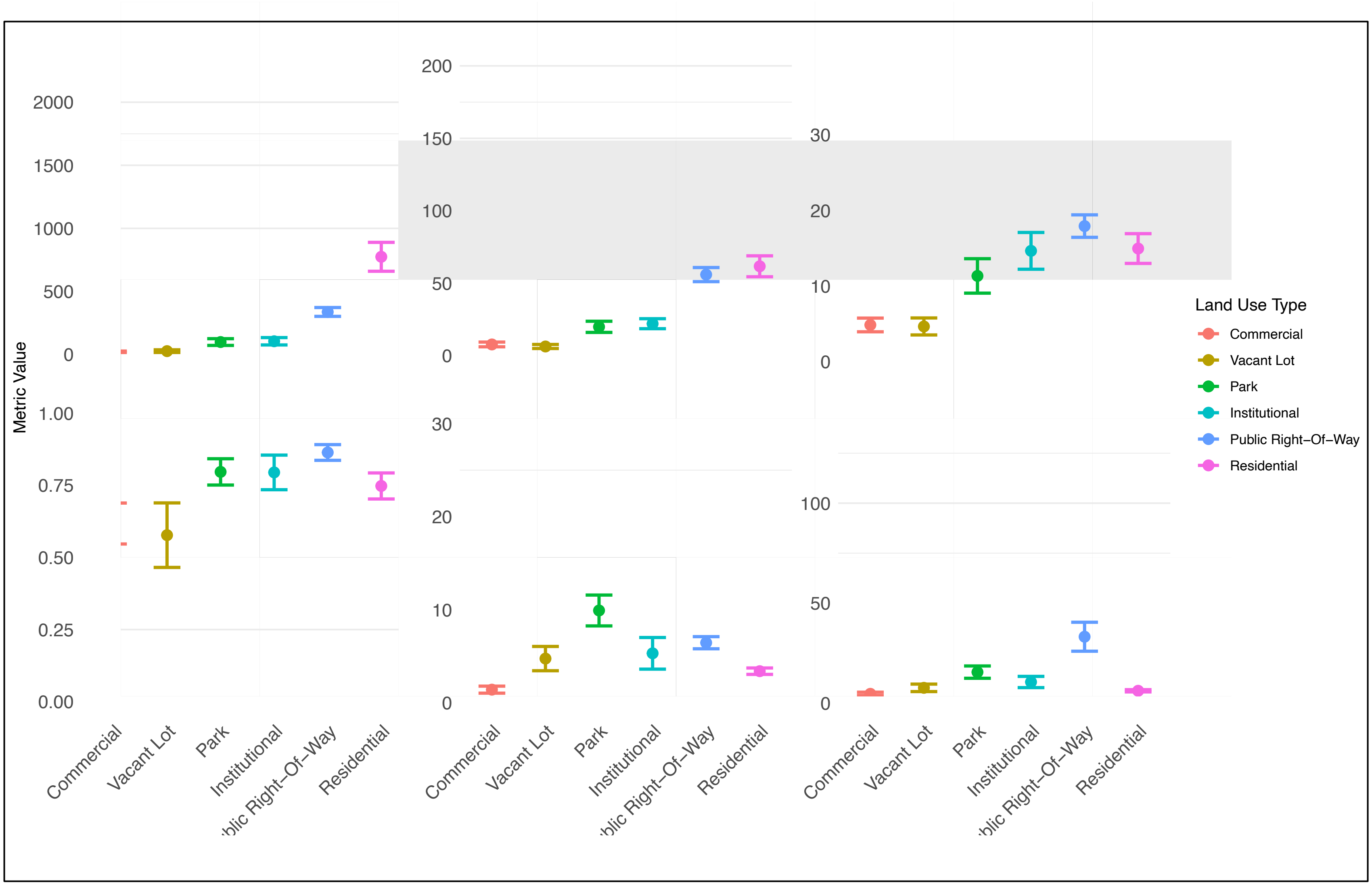


**Figure 4.** Variation in tree diversity, structure, and size in study area across six land use types. (A) Tree abundance, (B) Species richness, (C) Effective number of species, (D) Hurlbert's PIE evenness, (E) Basal area, standardized by subplot area, and (F) Basal area, standardized by plantable area. Grey data points represent raw data values of subplots. Coloured points represent the mean value for each land use type and error bars show ±1 standard error of the mean. Each panel displays a different metric, with y-axes scaled independently. Panels B–D present values derived from coverage-based rarefaction, while other panels show unrarefied values.

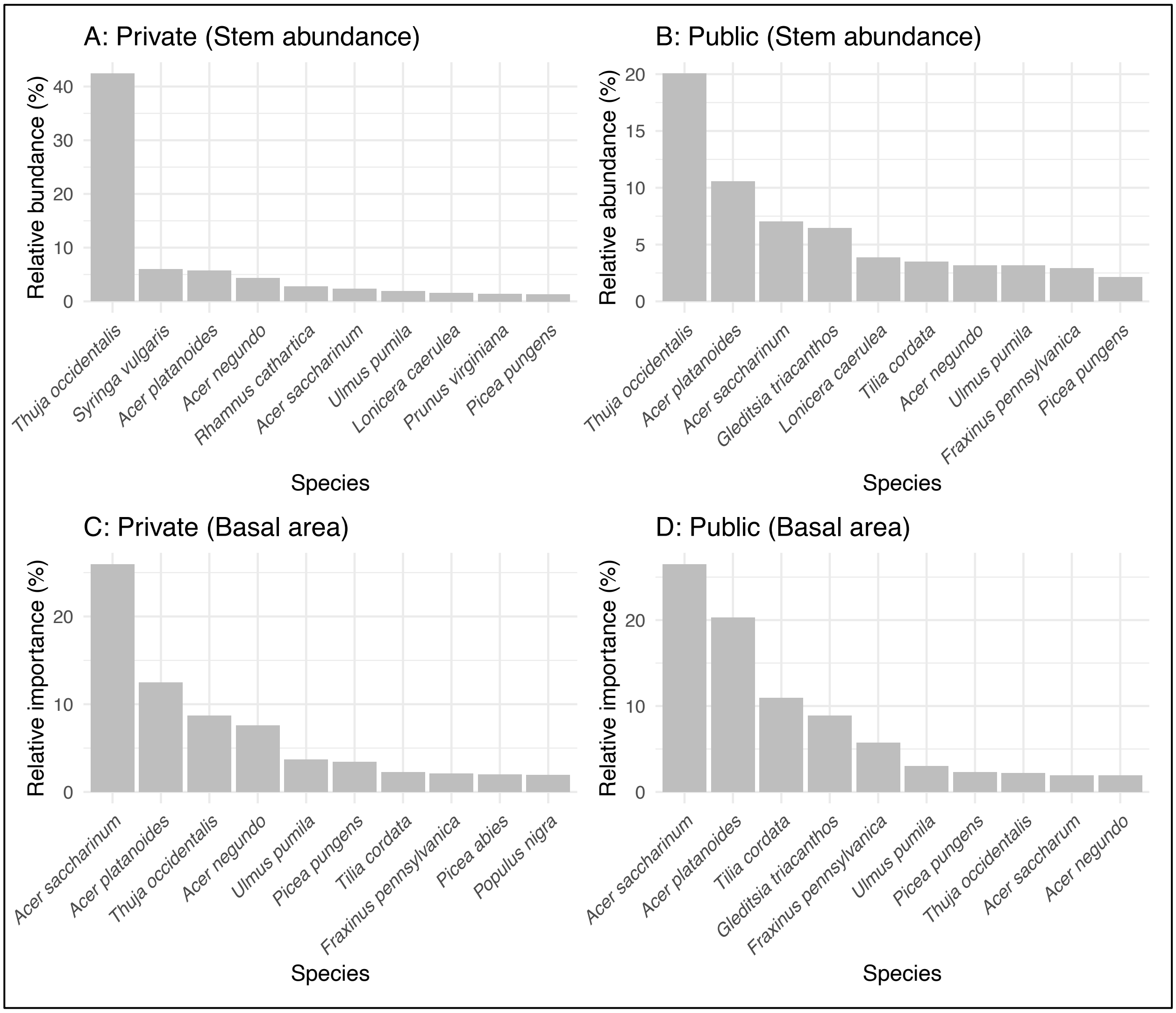


**Figure 5.** Top ten most common tree species on private (commercial, institutional, residential, vacant lots) and public (parks, public right-of-way) land use types based on relative abundance (A, B) and relative importance (C, D). Bars represent the proportion of individuals (abundance) or basal area contributed by each species. Species names are shown in italics and ordered by decreasing percent contribution within each panel.

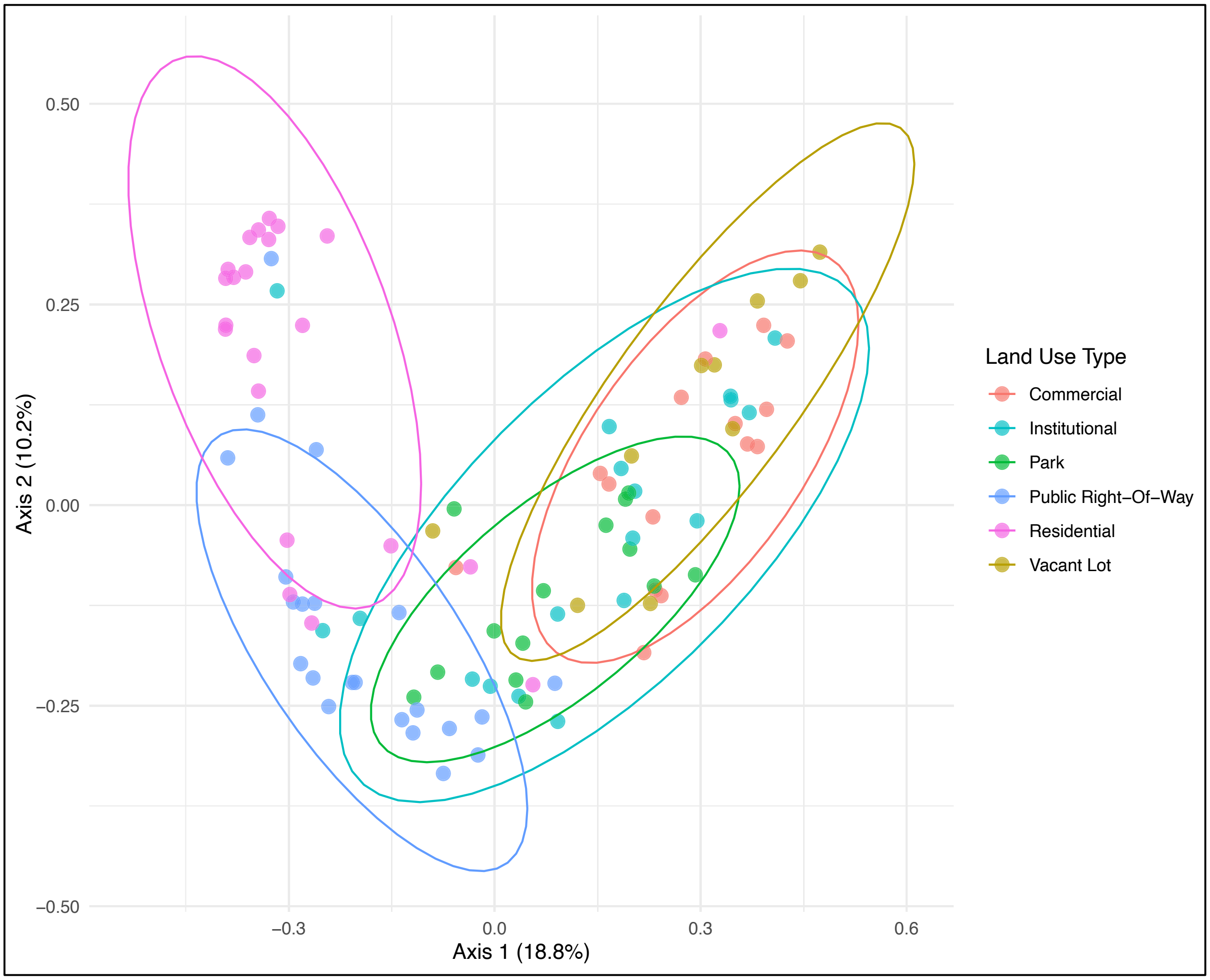


**Figure 6.** Principal Coordinates Analysis (PCoA) ordination of tree species composition across six different land use types. Plot shows distances between 101 subplots based on 307 tree species, calculated using Bray-Curtis dissimilarities. Points represent subplots and are colored by land use type. The dispersion of species composition within each land use type is illustrated with 90% confidence ellipses around the centroid of each group. Axes show the first two PCoA axes, with the percentage of variance explained indicated in parentheses.

# Appendix I: Species classification and DBH estimation

**Table A1.** List of hybrid species present in sampling and their reclassification based on parental lineages. Hybrids that were closely similar to one of their parent species or were cultivars with little ecological or morphological differentiation were reassigned to the parent species group. Hybrids with unique morphological traits that were visually identifiable in the field, have distinct reproductive lineages, are naturally occurring, or have known ecological significance were retained as separate species. To reclassify hybrids, we used *Plants of the World Online, The Kew Data Portal*, from the Royal Botanic Gardens (https://powo.science.kew.org). If species information was not available on the Kew Data Portal, we used local sources, such as *Arbres Hydro-Québec* (https://arbres.hydroquebec.com) or *Oregon State University Landscape Plants* (https://landscapeplants.oregonstate.edu). For some *Malus* species, hybrid formulas were not disclosed due to patent rights. In these cases, hybrids were reclassified as *Malus spp.*

| **Hybrid species** | **Parent species (hybrid formula)** | **Reclassification** | **Number of individuals** |
|---|---|---|---|
| *Acer* x *freemanii* | *A. rubrum* x *A. saccharinum* | *Acer* x *freemanii* | 155 |
| *Aesculus* x *hybrida* | *A. flava* x *A. pavia* | *Aesculus* x *hybrida* | 2 |
| *Amelanchier* x *grandiflora* | *A. arbor* x *A. laevis* | *Amelanchier* x *grandiflora* | 11 |
| *Crateagus* x mordenensis | *C. laevigata* x *C. succlenta* | *Crataegus* spp. | 4 |
| *Magnolia* x *brooklynensis* | *M. acuminata* x *M. liliiflora* | *Magnolia* x *brooklynensis* | 1 |
| *Malus* x 'Dolgo' | N/A | *Malus* spp. | 2 |
| *Malus* x *floribunda* | *M. baccata* x *M. toringo* | *Malus* x *floribunda* | 6 |
| *Malus* x 'Golden Raindrops' | N/A | *Malus* spp. | 1 |
| *Malus* x 'Harvest Gold' | N/A | *Malus* spp. | 2 |
| *Malus* x 'Indian Magic' | N/A | *Malus* spp. | 2 |
| *Malus* x 'Makamik' | N/A | *Malus* spp. | 2 |
| *Malus* x 'Prairifire' | N/A | *Malus* spp. | 2 |
| *Malus* x 'Radiant' | N/A | *Malus* spp. | 2 |
| *Malus* x 'Red Splendor'' | N/A | *Malus* spp. | 1 |
| *Malus* x 'Centurion' | N/A | *Malus* spp. | 1 |
| *Philadelphus* x *virginalis* | *P. coronarius* x *P. microphyllus* x *P. pubescens* | *Philadelphus* x *virginalis* | 1 |

| | | | |
|---|---|---|---|
| *Platanus* x *acerfolia* | *P. occidentalis* x *P. orientalis* | *Platanus* x *acerfolia* | 2 |
| *Prunus* x *cistena* | *P. cerasifera* x *P. pumila* | *Prunus* x *cistena* | 1 |
| *Quercus* x *warei* | *Q. bicolor* x *Q. robur* | *Quercus* x *warei* | 3 |
| *Syringa* x *persica* | *S. laciniata* x *S. afghanica* | *Syringa* x *persica* | 1 |
| *Tilia* x *europaea* | *T. cordata* x *T. platyphyllos* | *Tilia* x *europaea* | 7 |
| *Tilia* x *flavescens* | *T. americana* x *T. cordata* | *Tilia* x *flavescens* | 1 |
| *Ulmus* x 'Cathedral' | *U. davidiana var. japonica* x *U. pumila* | *Ulmus davidiana* | 2 |
| *Ulmus* x 'Frontier' | *U. minor* x *U. parvifolia* | *Ulmus* x 'Frontier' | 3 |
| *Ulmus* x 'Homestead' | *U. pumila* x *(U. pumila* x *U. minor)* | *Ulmus pumila* | 47 |
| *Ulmus* x 'Patriots' | *U. davidiana var. japonica* x *(U. hollandica* x *(U. minor* x *U. pumila))* | *Ulmus* x *hollandica* | 4 |
| *Ulmus* x 'Sapporo Autmn Gold' | *U. davidiana var. japonica* x *U. pumila* | *Ulmus pumila* | 2 |
| *Ulmus* x 'Morton' | *U. davidiana var. japonica* x *U. davidiana var. japonica* | *Ulmus* x 'Morton' | 71 |
| *Ulmus* x 'New Horizon' | *U. davidiana var. japonica* x *U. pumila* | *Ulmus* x 'New Horizon' | 7 |
| *Ulmus* x *hollandica* | *U. glabra* x *U. minor* | *Ulmus* x *hollandica* | 17 |

**Tree DBH estimation**

Due to lack of access to certain trees, we estimated DBH for 1,663 trees in the dataset, (5.8%). In most cases, we visually estimated DBH in the field (n = 1,259, 4.4%). However, 404 trees (1.4%) were unable to be visually estimated due to obstructed view of their trunks. To include these individuals in size analyses, we estimated DBH using species-specific averages. These averages were calculated separately for each species within each land use type, rather than across all plots. We applied this approach when at least five individuals of the same species were present within a given land use type. When fewer than five individuals of a species occurred within a land use type, species averages were instead calculated using all DBH measurements for that species across all plots. For species with fewer than five individuals in total across all plots, DBH was estimated as the mean of the available measurements, regardless of sample size. In certain cases, this procedure was further adapted to site-specific conditions. For example, in Plot 08A, a 200 m stretch of fenced railway contained 158 trees, many of which could not be measured directly due to restricted view. To address this, we conducted visual DBH estimates for trees within three accessible 10–20 m sections located at the beginning, middle, and end of the stretch. These values were then used to calculate species-specific averages applied to the unmeasured trees within the railway section.

# Appendix II: Supplementary data visualization and analysis

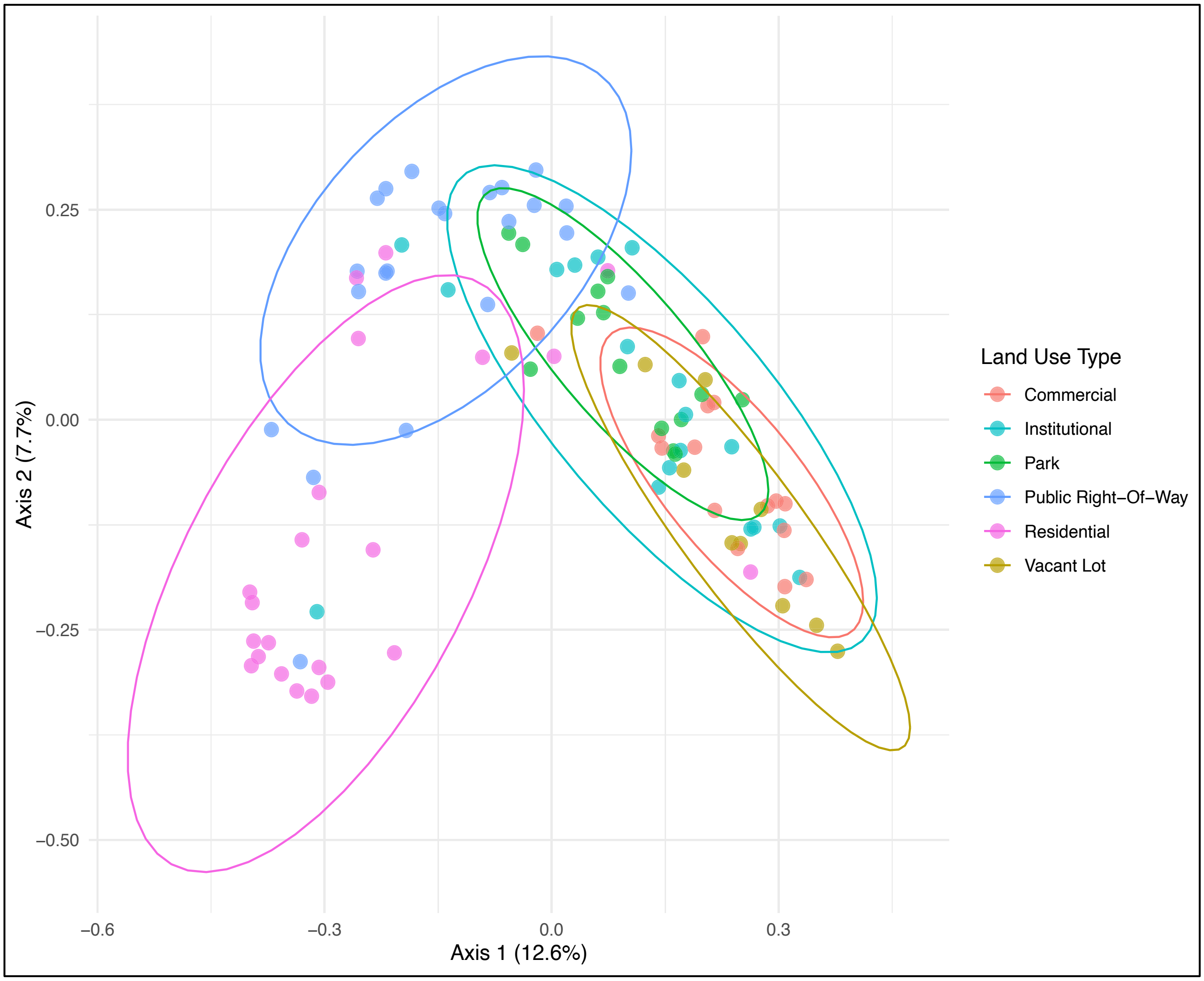


**Figure A2.1.** Principal Coordinates Analysis (PCoA) ordination of tree species composition across six different land use types. Plot shows distances between 101 subplots based on 307 tree species, calculated using Jaccard dissimilarities. Points represent subplots and are colored by land use type. The dispersion of species composition within each land use type is illustrated with 90% confidence ellipses around the centroid of each group. Axes show the first two PCoA axes, with the percentage of variance explained indicated in parentheses.

**PCoA ordination: Jaccard**

We performed PCoA ordinations using both Bray-Curtis and Jaccard dissimilarities to examine patterns of species composition among the 101 subplots. Bray-Curtis distances incorporate species abundances, while Jaccard distances are presence-absence based, focusing only on species occurrence. The Bray-Curtis ordination explained 29% of the variation on the first two axes and showed clear separation among land use types. The Jaccard ordination produced similar patterns of positioning among land use types, although the first two axes explained slightly less variation (20.3%) than Bray-Curtis. Visually, the positions occupied by the different land use types in ordination space in the Jaccard ordination were similar to Bray-Curtis. Dispersions based on Jaccard distances were slightly higher overall, with Public Right-of-Way (0.528) and Residential (0.508) sites again showing lower dispersion compared to Commercial (0.628) or Park (0.640) sites. Post-hoc Tukey comparisons largely mirrored the patterns observed with Bray-Curtis distances, with Public Right-of-Way and Residential consistently showing smaller distances to centroid than other land use types. Overall, both dissimilarity measures produced comparable ordination structures, supporting our decision to report results using Bray-Curtis distances in the main analyses, as it incorporates species abundances and provides a more nuanced representation of species composition than Jaccard distances.

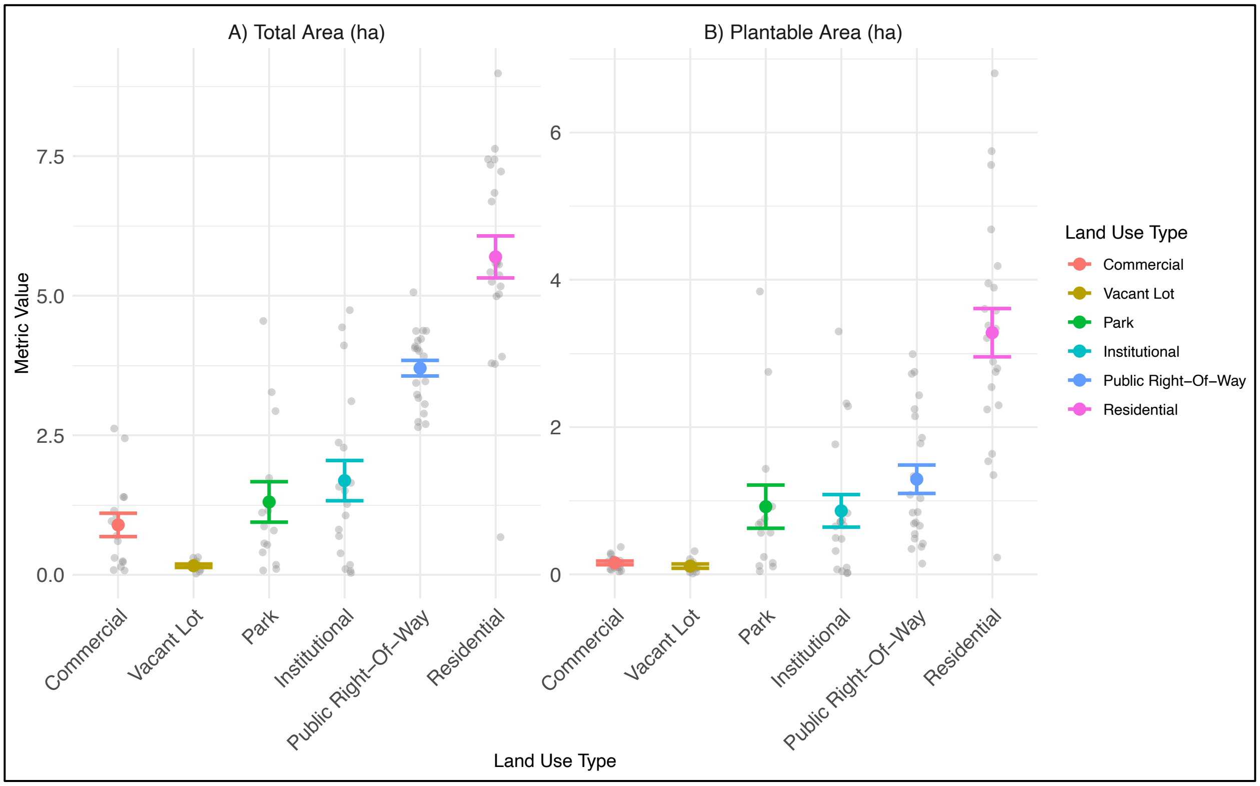


**Figure A2.2.** Average total area and plantable area across six land use types. (A) Total area (hectares), (B) Plantable area (hectares). Grey data points represent raw data values of subplots. Coloured points represent the mean value for each land use type and error bars show ±1 standard error of the mean. Each panel displays a different metric, with y-axes scaled independently.

**Table A2.** Species significantly associated with land use types. Species abundance patterns were correlated with the first two axes of a PCoA of community composition, and each species was assigned to the land use type whose centroid in ordination space was most aligned with its vector. The table lists only species with highly significant correlations ($p < 0.001$), the land use type with which each species is most strongly associated, the strength of the correlation (r), and the total number of individuals recorded across the entire inventory.

| Species | Correlation (r) | Land use type | Total individuals |
|---|---|---|---|
| *Thuja occidentalis* | 0.58 | Residential | 10046 |
| *Picea abies* | 0.48 | Residential | 228 |
| *Pyrus communis* | 0.45 | Residential | 203 |
| *Syringa reticulata* | 0.44 | Residential | 377 |
| *Juniperus communis* | 0.42 | Residential | 101 |
| *Malus spp.* | 0.39 | Residential | 294 |
| *Acer platanoides* | 0.38 | Residential | 2049 |
| *Prunus virginiana* | 0.37 | Residential | 360 |
| *Abies balsamea* | 0.37 | Residential | 37 |
| *Syringa vulgaris* | 0.36 | Residential | 1306 |
| *Magnolia kobus* | 0.35 | Residential | 38 |
| *Hydrangea paniculata* | 0.35 | Residential | 92 |
| *Prunus cerasus* | 0.34 | Residential | 60 |
| *Malus hupehensis* | 0.33 | Residential | 101 |
| *Acer saccharinum* | 0.33 | Residential | 1073 |
| *Betula pendula* | 0.31 | Residential | 22 |
| *Cotinus coggygria* | 0.31 | Residential | 68 |
| *Gleditsia triacanthos* | 0.31 | Public Right-Of-Way | 725 |
| *Sorbus aucuparia* | 0.31 | Residential | 72 |
| *Syringa pubescens* | 0.30 | Residential | 78 |
| *Taxus canadensis* | 0.30 | Residential | 137 |
| *Taxus cuspidata* | 0.30 | Residential | 23 |
| *Malus sylvestris* | 0.30 | Residential | 77 |
| *Ginkgo biloba* | 0.30 | Public Right-Of-Way | 169 |
| *Acer palmatum* | 0.29 | Residential | 72 |
| *Pinus strobus* | 0.29 | Residential | 71 |
| *Salix integra* | 0.28 | Residential | 40 |
| *Magnolia liliiflora* | 0.28 | Residential | 14 |
| *Morus alba* | 0.27 | Residential | 79 |
| *Picea glauca* | 0.26 | Residential | 144 |

| | | | |
|---|---|---|---|
| *Philadelphus coronarius* | 0.26 | Residential | 24 |
| *Acer negundo* | 0.25 | Residential | 1123 |
| *Fraxinus americana* | 0.25 | Residential | 136 |
| *Cornus alba* | 0.24 | Residential | 11 |
| *Juniperus scopulorum* | 0.24 | Residential | 70 |
| *Picea pungens* | 0.23 | Residential | 448 |
| *Morus nigra* | 0.23 | Residential | 12 |
| *Ulmus pumila* | 0.23 | Residential | 654 |
| *Ulmus americana* | 0.23 | Residential | 288 |
| *Malus baccata* | 0.22 | Residential | 27 |
| *Tilia cordata* | 0.22 | Public Right-Of-Way | 437 |
| *Prunus avium* | 0.21 | Residential | 74 |
| *Picea rubens* | 0.20 | Residential | 24 |
| *Acer freemanii* | 0.20 | Residential | 155 |
| *Prunus persica* | 0.19 | Residential | 13 |
| *Ulmus* x *Morton* | 0.19 | Public Right-Of-Way | 71 |
| *Quercus macrocarpa* | 0.19 | Residential | 80 |
| *Prunus cerasifera* | 0.19 | Residential | 25 |
| *Morus rubra* | 0.18 | Residential | 35 |
| *Pinus banksiana* | 0.17 | Residential | 10 |
| *Betula papyrifera* | 0.16 | Residential | 35 |
| *Fraxinus pennsylvanica* | 0.16 | Residential | 466 |
| *Gymnocladus dioicus* | 0.16 | Public Right-Of-Way | 163 |
| *Callitropsis nootkatensis* | 0.16 | Residential | 9 |
| *Acer tataricum* | 0.15 | Residential | 78 |
| *Syringa josikaea* | 0.15 | Residential | 8 |
| *Amelanchier canadensis* | 0.15 | Residential | 108 |
| *Weigela florida* | 0.15 | Residential | 14 |
| *Celtis occidentalis* | 0.15 | Residential | 305 |
| *Rhamnus cathartica* | 0.15 | Residential | 654 |
| *Lonicera tatarica* | 0.15 | Residential | 64 |
| *Ulmus glabra* | 0.15 | Residential | 52 |
| *Aesculus hippocastanum* | 0.14 | Residential | 47 |
| *Sambucus canadensis* | 0.13 | Residential | 18 |
| *Rosa gallica* | 0.13 | Residential | 17 |
| *Prunus domestica* | 0.13 | Residential | 164 |
| *Salix babylonica* | 0.12 | Residential | 17 |

| | | | |
|---|---|---|---|
| *Viburnum trilobum* | 0.12 | Residential | 8 |
| *Picea laxa* | 0.11 | Residential | 19 |
| *Rosa canina* | 0.10 | Residential | 41 |
| *Forsythia viridissima* | 0.09 | Residential | 27 |

# Appendix III: Mixed model analysis

**Table A3.1.** Fixed and random effects results for mixed model analysis (negative binomial distribution with log link and log(ha) offset) predicting tree density (trees $m^2$/ha) at the subplot level (n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

| Predictor | Estimate | Std Error | z value | P value |
|---|---|---|---|---|
| **Commercial** | 5.085 | 0.184 | 27.71 | **<0.0001** |
| **Institutional** | -0.198 | 0.241 | -0.82 | 0.412 |
| **Park** | -0.040 | 0.257 | -0.16 | 0.876 |
| **Public Right-Of-Way** | 0.975 | 0.231 | 4.21 | **<0.0001** |
| **Residential** | 0.274 | 0.227 | 1.21 | 0.227 |
| **Vacant Lot** | 0.261 | 0.288 | 0.91 | 0.365 |
| | **Random effect (SD)** | | | |
| | 0.151 | | | |

**Table A3.2.** Fixed and random effects results for mixed model analysis (Gamma distribution with log link) predicting species richness (coverage-based rarefied values) at the subplot level (n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

| Predictor | Estimate | Std Error | z value | P value |
|---|---|---|---|---|
| **Commercial** | 2.00 | 0.178 | 11.23 | **<0.0001** |
| **Institutional** | 1.093 | 0.223 | 4.90 | **<0.0001** |
| **Park** | 0.924 | 0.236 | 3.91 | **0.0001** |
| **Public Right-Of-Way** | 1.990 | 0.212 | 9.37 | **<0.0001** |
| **Residential** | 2.075 | 0.212 | 9.81 | **<0.0001** |
| **Vacant Lot** | -0.127 | 0.259 | -0.49 | 0.625 |
| | **Random effect (SD)** | | | |
| | 0.223 | | | |

**Table A3.3.** Fixed and random effects results for mixed model analysis (Gamma distribution with log link) predicting effective species diversity (q = 1, ENS) at the subplot level (n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

| **Predictor** | **Estimate** | **Std Error** | **z value** | **P value** |
|---|---|---|---|---|
| **Commercial** | 1.50 | 0.179 | 8.42 | **<0.0001** |
| **Institutional** | 1.13 | 0.212 | 5.32 | **<0.0001** |
| **Park** | 0.828 | 0.228 | 3.63 | **<0.001** |
| **Public Right-Of-Way** | 1.34 | 0.202 | 6.63 | **<0.0001** |
| **Residential** | 1.15 | 0.202 | 5.68 | **<0.0001** |
| **Vacant Lot** | 0.063 | 0.250 | 0.25 | 0.803 |
| **Random effect (SD)** | | | | |
| | 0.276 | | | |

**Table A3.4.** Fixed and random effects results for mixed model analysis (zero-one inflated beta GLMM) predicting Hurlbert's PIE evenness at the subplot level (n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

| Predictor | Estimate | Std Error | z value | P value |
|---|---|---|---|---|
| **Commercial** | 0.675 | 0.196 | 3.45 | **<0.001** |
| **Institutional** | 0.998 | 0.267 | 3.74 | **<0.001** |
| **Park** | 0.712 | 0.273 | 2.61 | **0.009** |
| **Public Right-Of-Way** | 1.029 | 0.252 | 4.09 | **<0.001** |
| **Residential** | 0.608 | 0.245 | 2.48 | **0.013** |
| **Vacant Lot** | 0.411 | 0.316 | 1.30 | 0.192 |
| | **Random effect (SD)** | | | |
| | 0.353 | | | |

**Table A3.5.** Fixed and random effects results for a linear mixed model analysis predicting log-transformed basal area (standardized by total subplot area, n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

| Predictor | Estimate | Std Error | t value | P value |
|---|---|---|---|---|
| **Commercial** | -0.089 | 0.231 | -0.39 | 0.701 |
| **Institutional** | 1.147 | 0.296 | 3.88 | **<0.001** |
| **Park** | 2.156 | 0.317 | 6.80 | **<0.001** |
| **Public Right-Of-Way** | 1.807 | 0.284 | 6.36 | **<0.001** |
| **Residential** | 1.170 | 0.284 | 4.12 | **<0.001** |
| **Vacant Lot** | 1.192 | 0.347 | 3.43 | **<0.001** |
| | **Random effect (SD)** | | | |
| | 0.329 | | | |

**Table A3.6.** Fixed and random effects results for a linear mixed model analysis predicting log-transformed basal area (standardized by plantable area) at the subplot level (n = 101) for six land use types: commercial, institutional, park, public right-of-way, residential, and vacant lot.

| Predictor | Estimate | Std Error | t value | P value |
|---|---|---|---|---|
| **Commercial** | 1.37 | 0.226 | 6.06 | **<0.001** |
| **Institutional** | 0.491 | 0.294 | 1.67 | 0.099 |
| **Park** | 1.11 | 0.315 | 3.53 | **<0.001** |
| **Public Right-Of-Way** | 1.66 | 0.282 | 5.89 | **<0.001** |
| **Residential** | 0.339 | 0.282 | 1.20 | 0.232 |
| **Vacant Lot** | 0.4231 | 0.345 | 0.67 | 0.504 |
| | **Random effect (SD)** | | | |
| | 0.261 | | | |