Taxonomic Diversity, Pest Vulnerability, and Carbon Storage of the Urban Forest in Winnipeg, Manitoba, Canada

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A thesis submitted to the Faculty of Graduate Studies

in partial fulfillment of the requirements for a

Master of Science in Bioscience, Technology, and Public Policy.

March 2024

Portions of this thesis are repeated for clarity to facilitate future publication of single chapters.

Abstract

Canadian prairie cities face a number of challenges when managing urban forests, one of which is reduced tree diversity due to more severe climate constraints to tree survival. This thesis reports on diversity and carbon storage for the city of Winnipeg, Manitoba, Canada. Approximately 24,500 trees were surveyed and measured across 77 Winnipeg neighborhoods, including trees on private lots, which had not been previously reported for the city. Using these data, I evaluated tree species diversity measures for city neighborhoods and compared diversity measures between trees on public and private property. Private properties exhibited higher tree diversity and better health status across all metrics. I also adapted the Pest Vulnerability Matrix (Laćan & McBride, 2008) to environmental conditions found in the city of Winnipeg to identify pests with the most potential to impact city forests and neighborhoods as well as areas most at risk of new pest invasion. Exploring carbon storage in the city, I used methods developed by Wayson et al. (2015) to create prediction intervals (a measure of reliability for the prediction of an observation) around biomass equations used by city foresters. I then estimated carbon storage in residential areas across the city. I found 58% of carbon stored in trees surveyed was in American elm (*Ulmus americana*), and no other tree species in the survey had an equivalent amount of stored carbon (based on mean DBH). This research incorporates the first large scale private tree inventory within Winnipeg, providing a more comprehensive assessment of tree species diversity and carbon storage values across the city. This study will allow urban forest managers to have a clearer understanding of the existing tree inventory and implications for future urban forest management activities to protect and increase the city's urban forest resource.

Acknowledgements

Thanks to my committee, Richard Westwood, Andrew Park, and Martha Barwinsky for their continued guidance and feedback. Thank you to external examiner Kamni Gill for your thoughtful suggestions through the defense and final revisions.

Thanks to the Westwood Lab, including Matthew Russell, Connor Watson-Savage, Justis Henault, and Martine Balcaen for support completing this thesis. And to Vanessa Schulz and Symantha Strempler, whose good spirits on long field days were invaluable to the data collection process.

Thanks also to Kerienne LaFrance, for their time and help gathering the data for this project, and to the City of Winnipeg Urban Forestry Branch as a whole for keeping this urban forest healthy and standing for generations to come.

Treaty 1 territory is the ancestral territory of Anishinaabe, Cree, Dakota, Dené, and Ojibway-Cree peoples, and the traditional homeland of the Métis Nation. As a settler and researcher of biodiversity on Treaty 1 territory, I have benefited from a long tradition of land stewardship and knowledge exchange by past and current Indigenous residents of these lands, and I extend my truest gratitude for these efforts, which continue even in the face of ongoing colonial violence. I acknowledge the importance of solidarity with and commitment to the Land Back movement in Treaty 1 and beyond.

Finally, for being there for me through late nights, early mornings, and all the times in between on this thesis and beyond, thanks to my partner Kaitlyn Duthie-Kannikkatt.

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List of Abbreviated Terms

1. DBH - Diameter Breast Height
2. DED - Dutch Elm Disease
3. EAB - Emerald Ash Borer
4. LIDAR - Light Detection and Ranging
5. PVM - Pest Vulnerability Matrix
6. UFMP - Urban Forest Management Plan
7. USDA - United States Department of Agriculture
8. UTM - Urban Tree Database

Chapter 1

Introduction

Urban forests provide environmental, health, and aesthetic benefits to the world's citizens. Urban forest managers face many challenges when it comes to managing these forests, including ensuring citizen safety, care for urban aesthetics, and maintaining the integrity and health of the forest (Rines, 2007). A changing climate also brings increased threats to trees from drought, flooding, and increased pest activity (Ordóñez & Duinker, 2014). Creating a climate- and pest-resilient urban forest requires careful planning and a detailed understanding of the complement of tree species that will best thrive in the future.

More recently, city managers have begun to look to these urban trees to help balance city carbon budgets. Forests are one of the few carbon sinks found in urban areas (Zhao et al., 2023) and as such it is necessary to be able to accurately survey and measure urban forests to determine their carbon storage and sequestration capacity.

Winnipeg, Manitoba is home to an urban forest of over 3 million trees, including the largest remaining urban elm (*Ulmus americana* L.) canopy in North America (City of Winnipeg 2021). The city has a long history of community activism devoted to these trees, and the City of Winnipeg recently adopted its first comprehensive urban forest strategy (City of Winnipeg, 2023). Winnipeg's urban forest is under threat from insect pests, diseases, and the impacts of climate change, and forest managers will need to prioritize forest protection and enhancement methodologies to avoid significant future losses to the forest canopy.

This thesis describes research conducted from 2002 to 2020 in Winnipeg's urban forest to examine diversity and carbon storage. Data on taxonomic and size diversity of trees on public and private property across the city was collected by student researchers including myself, over this timespan. I then analyzed these data to explore the following research questions:

1) How diverse is Winnipeg's urban forest in terms of tree species?

    i) How does tree diversity vary by neighborhood?

    ii) Do private and public tree holdings differ in diversity, and if so,

    iii) How do private holdings contribute to tree diversity in Winnipeg?

2) What neighborhoods are most susceptible to current and future pest invasions? Is there a relationship between pest vulnerability and species diversity?

3) Can I estimate carbon storage in Winnipeg?

   i) What changes in carbon storage might we expect given the city of Winnipeg's plan for 1:1 replacement of lost trees?

   ii) Can I use published regression parameters and simulation techniques to derive measures of statistical variability in in carbon storage in Winnipeg's urban forest?

   iii) What is the estimated mass of carbon stored in surveyed neighbourhoods in Winnipeg's urban forest, and what are the boundaries around these estimates?

The thesis contains a literature review (Chapter 2), which includes a discussion of diversity measurements and benefits, as well as diversity management in Canada's urban forests. The literature review also examines biomass measurement and carbon capture and provides an overview of both USDA and i-Tree biomass calculations, both of which were used later in the thesis. Chapter 3 describes data collection methods for the thesis and addresses research questions 1 and 2. In this chapter I report diversity measures like richness, evenness, and dominance, as well as pest vulnerability outcomes determined using the Pest Vulnerability Matrix (Laćan & McBride, 2008). Chapter 4 addresses research question 3, providing prediction intervals around i-Tree biomass and other biomass equations and comparing carbon storage across tree species in Winnipeg's urban forest. Chapter 5 includes a discussion of results from chapters 4 and 5, and Chapter 6 provides conclusions for the thesis.

Chapter 2

Literature Review

This literature review consists of two components:

1. An exploration of biodiversity in urban forests, including measurement, sources of biodiversity, and managing for pests by diversifying forests and
2. An examination of the urban forest carbon capture literature and a review of factors that affect carbon capture and methods for carbon measurement.

2.1 Biodiversity and the Urban Forest

2.1.1 What is an urban forest?

McPherson et al. (1997) describe urban forests as follows: "Urban forests are small pockets of green in a gray landscape. They are ribbons of life meandering through a largely artificial environment. They are enclaves of serenity and biological diversity tucked within suburban development and busy streets."

Urban forests are often considered the sum of all woody and associated vegetation in and around dense human settlements, ranging from small communities in rural settings to metropolitan regions (Miller 1997). In residential areas, this can include trees managed by municipalities, trees planted and/or tended on private properties, and trees that have grown without human support or inputs. Many municipalities also include large swaths of unmanaged trees that resemble natural forests when inventorying total urban forest holdings (Sanders 1984); in Winnipeg, this includes areas like Assiniboine Forest, a small aspen-oak dominated woodland. A full description of Winnipeg's urban forest can be found in Section 2.1.8.

2.1.2 Measuring Biodiversity

2.1.2.1 Taxonomic, functional, and phylogenetic biodiversity

Biodiversity, or the measure of the variety of life (Magurran, 2004), is a fundamental biological characteristic of any ecosystem, including urban forests. Biodiversity measures the variety and abundance of species in an area (Magurran, 2004), but the metrics used to measure that biological variety will depend on the desired research outcomes. Researchers most commonly reference taxonomic biodiversity, which measures variety of species or other taxonomic rank (Humphries et al., 1995). This thesis reports taxonomic diversity, which has a number of benefits. Taxonomic diversity is the most common diversity measure and can allow researchers to compare data across studies; however, the proliferation of different taxonomic measures of diversity can also make comparison difficult. Using taxonomic diversity also allows researchers to identify samples by taxonomic rank, often an easier method than DNA analysis or defining functional or biological traits. This measurement of biodiversity also has readily identifiable indices and is useful where spatial units are comparable. In the case of this thesis, relatively comparable urban forest neighborhoods (similar climate, soil conditions, access to sunlight, etc.) made taxonomic diversity metrics a viable and efficient option for the research conducted.

Taxonomic biodiversity can fail to capture differences in ecosystem services provided by the populations in question. For this reason, researchers interested in ecosystem service delivery will often choose to use functional diversity measures instead. Functional diversity measures variety in functional traits, or traits that describe a species' use of resources (Bagousse-Pinguet et al., 2019). Taxonomic diversity can also fail to capture diversity because species-level distinctions are subject to human biases like visual distinctiveness or economic value (Miller et al., 2018).

Because of these limitations, some researchers argue that phylogenetic diversity, or the measure of branch lengths in a phylogenetic tree, is a more comprehensive measure of diversity (Miller et al., 2018). Generally branch lengths are measured in terms of nucleotide substitutions compared to the parent branch, with longer branches indicating greater genetic divergence (McLennan, 2010). Taxonomic diversity measures met the requirements of this study from an urban forest planning perspective, and measuring genetic and functional diversity was not feasible. Thus, I report taxonomic diversity in the following chapters.

2.1.2.2 Species richness and evenness

Taxonomic diversity is usually described in terms of richness and sometimes in terms of evenness. Richness provides a count of species (or other taxonomic rank); species richness by count is the simplest and most universal measure of biodiversity (Humphries et al., 1995). Evenness is a biodiversity measure that quantifies the proportional representation of each species at a location (Magurran, 2004). Heterogeneity indices combine both approaches in a single index

(Heip et al., 1998). Heterogeneity indices can be parametric, ie., based on observations following a particular abundance model. For instance, Log Series α is appropriate for cases where many species have only one observed individual (Fisher et al., 1943; Magurran, 2004). More commonly reported in the scientific literature are non-parametric heterogeneity indices, which are not based on a particular abundance model. The most enduring non-parametric index is Shannon's Diversity Index. However, this index is widely criticized for confounding richness and evenness, and errors may arise if some species are not sampled within a community (Magurran, 2004). Non-parametric indices used in the following chapters include Simpson's 1/D, Simpson's evenness measure, and the Berger-Parker *d* index. Simpson's 1/D accounts for richness and evenness, and it is less affected by sample size and less sensitive to rare species than similar indices (Magurran, 2004; Simpson, 1949). Simpson's evenness index was chosen in this study as a complementary metric to Simpson's 1/D; Simpson's evenness index reports only the evenness component of the more widely used D metric. The Berger-Parker index is a simple statistic that reports dominance of the most represented species, an important consideration for foresters looking to diversify urban forest holdings. Equations for all metrics used can be found in Appendix C of Chapter 2.

2.1.2.3 Alpha, beta, and gamma diversity

Finally, the diversity within and between study sites can be described using alpha (α), beta (β), and gamma (γ) diversity. Alpha diversity describes biodiversity at a single study site, while beta biodiversity describes the differences in diversity across a range of sites (Magurran, 2004). Gamma diversity is a measure of diversity over an entire study area or larger landscape (Jost, 2007). Beta diversity is sometimes described in terms of the relationship between alpha and gamma diversity: most simply, Whitaker (1960) described beta diversity as $\beta = \gamma/\alpha$, while Lande (1996) proposed $\beta = \gamma - \alpha$. Other researchers have worked to separate beta diversity from alpha and gamma diversity, including popular methods based on average dissimilarity between pairs of plots (Izsak & Price, 2001).

2.1.3 Benefits of diversity

Increased resilience to environmental disturbance, including pest resilience can be linked to increased species diversity (Laćan & McBride, 2008). Many pests found in Winnipeg, such as the emerald ash borer (*Agrilus planipennis* Obenberger, hereafter EAB) and Dutch elm disease

(*Ophiostoma ulmi* (Buisman) and *Ophiostoma novo-ulmi* (Brasier), hereafter DED), target specific tree taxa. Higher tree species diversity implies that individuals of pest-specific host trees will be scattered more widely. I would therefore expect disease transmission and pest dispersal among susceptible trees to be slowed, and that fewer trees or tree species would be lost to any given pest or pathogen (Poland & McCullough, 2006; Nitoslawski et al., 2016).

There is also speculation that increased urban tree diversity will reduce forest loss caused by climate change. Ordóñez and Duinker (2014) describe a two-pronged approach to assessing climate change vulnerability in urban forests. Their approach targets the ecological and social components, or "clusters," of climate resilience. Increased diversity operates in the ecological cluster to maximize species-level adaptive capacity and is influenced by city policy and budgets of the social cluster (Ordóñez & Duinker, 2014). The presence of non-native tree species may be important in achieving this resilience, since many exotic species may be better suited to the changing conditions than their native counterparts (Woodall et al., 2010; Almas & Conway, 2016). It is worth noting that while many cities account for forest services like carbon storage and sequestration in their climate planning, few account for the potential impact of climate change on the urban forest health (Brandt et al., 2016).

Tree diversity also affects the provision of ecosystem services by the urban forest (Alvey, 2006). For instance, pollutant filtration and release varies by tree species (Grote et al., 2016), as does capacity for storm water retention and reduction of urban heat island effects (Davenport et al., 2016). Increased biodiversity has also been shown to increase citizens' reported well-being in urban environments (Carrus et al., 2015), and diverse urban forests can provide residents with fuel and fruit, or provisioning services. By identifying and planting trees that yield specified combinations of beneficial ecosystem services, urban forest managers can optimize diversity to serve the needs of their communities.

2.1.4 Sources of Diversity in Urban Forests

Williams et al. (2009) described a model for understanding sources of plant diversity in urban settings. Their model identifies four filters, which serve to add, subtract, and preserve species over time. These filters are listed below:

1. Habitat Transformation: When an area is initially settled, species present are incorporated into the urban forest or deleted from the landscape.
2. Fragmentation: Over time, areas of contiguous vegetation are fragmented. Some native species are lost, and non-native species are introduced.
3. Urban Environment: Only those species able to cope with high stress environmental conditions of the urban forest (salt, root crowding, poor soil, etc.) persist.
4. Human preference: Species are removed and planted based on human preference over time.

Wealth inequality and property value are correlated with tree diversity. Research has shown that there is tree canopy cover is generally lower in low-income areas of North American cities (Hope et al., 2008; Landry & Chakraborty, 2009; Jesdale et al., 2013; Schwarz et al., 2015). Hope et al. (2008) called this the “luxury effect,” and observed that urban greenspace and human access to resources were highly correlated. Kinzig et al. (2005) found greater biodiversity in higher-income neighborhoods in Phoenix, Arizona. They suggested these residents might have more resources available for purchasing plants, as well as more interest in public beautification. Other explanations for the luxury effect include neighborhood choice, the idea being that high-diversity areas attract people and drive up property values in these parts of cities, and environmental context (eg. in drier climates, wealthy neighborhoods have more diverse, healthy vegetation because residents can afford irrigation) (Leong et al., 2018). Interest in the luxury effect is rising as climate change exacerbates urban heat islands, putting the most vulnerable communities at risk due to lack of green infrastructure (Jesdale et al., 2013).

Konijnendijk (2021) developed a set of guidelines to gauge cities’ success in urban greening and resident access to trees and green space, the 3-30-300 Rule. This rule suggests that all homes should have three trees “of a decent size”, that all neighborhoods should have at least 30% canopy cover, and that all residents live within 300 meters of a greenspace (Konijnendijk, 2021). The application of this rule allows researchers and planners to manage for equity and greenspace access, and to compare performance within and across cities. Ling (2021) looked at this rule as applied to Toronto and Mississauga, and found that of 180 homes surveyed, only 12% met the 3-30-300 rule.

In Winnipeg, winter climate limits tree diversity. Winnipeg is in USDA plant hardiness zone 3. Between 1975 and 2020, Winnipeg experienced an average minimum winter temperature of -33°C (Prairie Climate Centre, 2020). See Table 2.1 for historical minimum temperatures and number of very cold (less than -30°C) temperature days in Winnipeg from 1950 to 2020. In their study of 20 U.S. and Canadian cities, Jenerette et al. (2016) showed that minimum winter temperature is a better predictor of tree diversity than summer maximum temperature, annual precipitation, or socio-cultural factors. City foresters have admitted to being hesitant to plant some species that would otherwise be considered hardy in Winnipeg due to worries about low temperatures (eg., most *Acer* species) (City of Winnipeg, 2023).

Table 2.1 Coldest minimum temperature (°C) and number of days below -30°C in Winnipeg, Manitoba (Prairie Climate Centre, 2020)

Year	1950	1960	1970	1980	1990	2000	2010	2020
Coldest Minimum Temp (°C)	-35.4	-37.3	-36.8	-37.3	-35.8	-34.5	-35.3	-33.8
No. Days Below -30°C	12.2	15.8	13.1	15.2	11.8	8.8	8	7.4

2.1.5 Measuring tree diversity in urban forests

In urban forests, tree diversity is usually quantified in terms of species richness and evenness, and in terms of age and size (Nitoslawski et al., 2016). Species richness is the measure of total species in an area, and in urban forestry, is often gauged using the “10-20-30 Rule.” Developed by geneticist Frank Santamour, this rule suggests that city foresters should aim for no more than 10% representation of any given tree species, 20% any genus, and 30% any family (Santamour, 1999). It does not appear there was a specific rationale for these proportions beyond his personal experience as a forest researcher; these benchmarks were suggested to prevent catastrophic loss of urban forests from lethal pest invasions which target specific tree species.

The 10-20-30 rule is not accepted by all researchers. Some pests are generalist feeders that feed on a variety tree species, like spring cankerworm (*Paleacrita vernata* Peck) and fall cankerworm (*Alsophia pometaria* L.) (LaFrance & Westwood, 2006). Others target trees that are weak or diseased, like the shot hole borer (*Scolytus rugulosus* Miller) that may cost the state of California an estimated $616 million in lost urban trees over the next 10 years (McPherson et al., 2017). In such cases keeping trees healthy is of more concern than maintaining diversity. On the other hand, some cities already surpass Santamour's proportions (eg. the most abundant tree species makes up no more than 5% of the forest, no genus more than 10%, etc.), but could still benefit from more ambitious targets with greater levels of tree diversity (Kendal et al., 2014).

Age structures in urban forests can be considered in a similar manner to the 10-20-30 rule. An ideal age class composition for the urban forest is described by Richards (1983) as 40% juvenile, 30% semi-mature, 20% mature and 10% senescent trees. This concept was suggested to provide a solid base of healthy, mature trees, with enough young trees to replace them over time (McPherson & Kotow, 2013). To quantify adherence to this rule, McPherson and Kotow (2013) calculated the absolute difference between the four ideal values and the percentages observed at their research locations, then summed the three values for an indication of percent deviation from the ideal distribution. This metric was then used to give a "grade" across several metrics. Of the 29 forests graded, 13 received their lowest grades in age structure, largely because juvenile trees were underrepresented (McPherson & Kotow, 2013).

Many biodiversity measures besides species richness are used to measure forest resilience, including species evenness and dominance. See section 2.1.1.2. for discussion of these measures and their uses.

2.1.6 Non-native species

Although urban forests are increasingly managed to conserve native species (Sjöman et al., 2016), the benefits of using non-native trees are also well established (Chalker-Scott, 2015). Many argue that non-native plantings risk introducing invasive species and diminishing the "superiority" of native species, based on the idea that native trees are best adapted to the environment in which they evolved (Sjöman et al., 2016). However, the urban environment often does not match its natural surroundings or has lost its natural infrastructure. Urban trees must cope with additional stressors like poor soils, pollutants, salt, and lack of root space. In interviews with 16 Canadian urban foresters, the most common reason exotic tree species were planted over native counterparts was their resilience to urban conditions (Almas & Conway, 2016). Non-native trees may also provide habitat for wildlife and pollinators, and have shown few adverse effects on urban insect, bird, and non-woody plant populations (Chalker-Scott, 2015). It is worth noting that much of the literature investigating and promoting the use of native trees comes from regions of the world with higher native tree diversity than Winnipeg, like central Europe, California and New England (Aronson et al., 2017; Kendle & Rose, 2000; Pawlak et al., 2023).

Non-native tree species serve to enhance diversity in urban forests, and the presence of these non-native trees does not necessarily undermine the benefits of a diverse urban forest. In a review paper of North American urban forests, Chalker-Scott (2015) found that both provisioning and habitat services for wildlife are preserved in urban forests with high proportions of non-native trees.

2.1.7 Tree diversity in Winnipeg's Urban Forest

The City of Winnipeg manages over 300,000 trees, and City foresters face a number of challenges when managing for a diverse urban forest. Foremost is the inheritance of historic planting regimes, which resulted in large monocultures of American elm and ash (*Fraxinus* sp.). This trend was prominent in North American urban forests for many decades leading into the 1970s, as foresters sought out “ideal” street trees to plant en masse and Winnipeg followed suit.

Limited availability of local bulk nursery stock can limit the numbers of trees available for planting. Many species that might be good candidates for boulevard planting are not currently grown in the quantities required locally. Moreover, many trees that are used for private property plantings are not well-suited for boulevard use. The foremost example of this is spruce (*Picea* spp.) plantings. Spruce was the third most numerous genus found on private property in the surveys conducted for this thesis, but spruce are not planted on boulevards because they can block line of sight in the right-of-way areas on streets due to their crown composition (City of Winnipeg, 2009).

Finally, budget constraints directly impact tree diversity. Funds sometimes need to be redirected by the City of Winnipeg Urban Forestry Branch to keep up with tree removals as a result of invasive pests rather than increasing levels of tree replacement and diversity that matches removal rates. The need for tree removal resources also impacts other management activities and may lead to the city falling behind in other core maintenance practices, such as pruning (City of Winnipeg, 2023).

American elm and ash are the species most vulnerable to lethal pests in the Winnipeg, comprising 50% of the street and park tree inventory in the city (City of Winnipeg, 2023). These trees are particularly vulnerable to pests such as EAB and DED. See Table 2.2 for the ten most abundant tree species by count on growing on street right-of-ways and in parks. The City of Winnipeg (2009) has Tree Planting Guidelines and Acceptable Tree Species policies (City of Winnipeg, 2017) to guide tree planting and tree replacement efforts and improve resilience to pest outbreaks.

2.1.8 Managing for Diversity in Winnipeg's Urban Forest Strategy

The City of Winnipeg adopted the Winnipeg Comprehensive Urban Forest Strategy in December 2023. The Strategy explicitly calls for the City of Winnipeg reach tree species targets aligned with Santamour's 10-20-30 Rule, recommending a diversity target of no more than 10 percent of any single species and 20 percent of any genus in the City's street and park tree inventory (City of Winnipeg, 2023). Recommended trees for new and replacement planting include but are not limited to: Manitoba maple (*Acer negundo* L.), silver maple (*Acer saccharinum* L.), northern hackberry (*Celtis occidentalis* L.), bur oak (*Quercus macrocarpa* Michx.), linden (*Tilia* spp), and Japanese tree lilac (*Syringa reticulata* Blume). In addition to proposing growing contracts with local nurseries to improve access to desired species, the Strategy suggests establishing a civic nursery to source uncommon species and trial new varieties for climate suitability, disease resistance, and salt tolerance (City of Winnipeg, 2023).

Table 2.2 The 10 most numerous tree taxa by count on street right-of-ways and park land in Winnipeg, MB, as reported in Draft Winnipeg Urban Forest Strategy (City of Winnipeg, 2023).

Public Property Species	**Common Name**	**Percent Representation**
Fraxinus pennsylvanica	Green ash	29%
Ulmus americana	American elm	17.5%
Tilia spp	Linden	9%
Ulmus pumila	Siberian elm	6%
Quercus macrocarpa	Bur oak	6%
Acer negundo	Manitoba maple	5%
Picea pungens	Blue spruce	3%
Fraxinus nigra	Black ash	3%
Picea glauca	White spruce	3%
Populus spp	Poplar, Cottonwood	3%

2.1.9 Diversity Surveys in Other Urban Forests in the Prairie Provinces

2.1.9.1 Saskatoon

The City of Saskatoon has a smaller urban forest than Winnipeg, with approximately 100,000 city-managed trees (City of Saskatoon, 2016), compared to 300,000 managed in Winnipeg (City of Winnipeg 2023). While Winnipeg has been managing DED for over 40 years, Saskatoon had its first confirmed case in 2015 (Barwinsky, 2016). The city is also managing for cottony ash psyllid (*Psyllopsis discrepans* Flor), allocating over one million dollars in funding to ash removal (Tank et al., 2017). The city also committed $50,000 to the development of an urban forestry management plan (UFMP) in 2018 (City of Saskatoon, 2018).

Saskatoon has used Tree Plotter software to inventory their urban forest (Tree Plotter, 2018). This software allows users to survey and map trees and provides an interactive map that can be made available to the public. It is compatible with i-Tree records (i-Tree is a widely used carbon and diversity accounting software developed by the U.S. Forest Service; see section 2.2.1 below), but also has its own add-on modules with similar capabilities, such as ecosystem service valuation and offline recording capabilities (Tree Plotter, 2018). Ordóñez and Duinker (2015) also conducted a climate change vulnerability assessment of Saskatoon's urban forest in workshop formats using qualitative data solicited from local experts in urban forestry and/or climate change. They found that increased snow in shoulder seasons, higher wind speeds, and drought were expected to have the most serious effects on the forest (Ordóñez & Duinker, 2015). Adaptive solutions suggested by participants included improved drainage in parks, preferential planting of conifers, and the development of a climate plan for the urban forest (Ordóñez & Duinker, 2015).

2.1.9.2 Edmonton

The city of Edmonton has over one million trees under city management (City of Edmonton, 2019c). Neither DED nor EAB are present in the city. One serious pest is European elm scale (*Eriococcus spurius* Modeer), a sap-sucking insect that causes elms to lose their leaves and is lethal if the infestation is heavy (City of Edmonton, 2019b). A cosmetic pest of concern is the ash leaf cone roller (*Caloptilia fraxinella* Ely), which produces larvae that feed on and nest in ash leaves, causing unsightly curling. The city is not currently applying pesticides for ash leaf cone roller, as a high proportion of the population is destroyed yearly by parasitic wasps (City of Edmonton, 2019b). Their ten-year UFMP was implemented in 2012. It calls for 20% canopy cover over the city and specific protections for native species and natural areas, among other initiatives (City of Edmonton, 2012).

In 2009, Edmonton used i-Tree Streets and i-Tree Eco to inventory their urban forest (City of Edmonton, 2012). i-Tree Streets showed high proportions of elm and ash, while i-Tree Eco, which accounts for more cover in natural areas, showed a largely aspen and poplar dominated canopy (City of Edmonton, 2012). Currently, the city uses OpenTreeMap ® to inventory their trees. This software allows them to catalogue trees as well as value ecosystem services and model planting scenarios (opentreemap, 2019). City residents are encouraged to edit the city's public tree map to include newly planted trees and uncatalogued trees on private property (City of Edmonton, 2019d). The city also maintains a separate, searchable database for fruit trees on public property (City of Edmonton, 2019a).

2.1.10 Pest Management and the Pest Vulnerability Matrix

One important reason cities manage for diversity and climate resilience is to reduce outbreaks of lethal pests. Pest outbreaks in urban forests tend to be more severe than those in natural forests (Meineke et al., 2013; Raupp et al., 2010). This occurs for a variety of reasons, including the lack of parasitoids in urban settings (Bennett & Gratton, 2012; Burkman & Gardiner, 2014), the higher temperatures in urban areas (Dale & Frank, 2014; Meineke et al., 2013), and the susceptibility of already stressed urban trees to pest outbreak (Flückiger & Braun, 1999). Unfortunately, pest outbreaks in urban forests are likely to increase with increasing temperatures (Meineke et al., 2013). Tubby and Webber (2010) suggest that climate change will exacerbate pest outbreaks in cities by:

- inducing physiological changes in host trees that make it easier for pests to access trees,
- promoting development and survival of pests,
- impacting pest predators and competitors, and
- increasing fit of local climate to non-native pests

A recent study of Canadian Urban Forestry Management Plans reported that more than 60% of cities developed their UFMP in response to pest and disease outbreaks (Ordóñez & Duinker, 2013). In this section, I review Winnipeg's major pests and describe the Pest Vulnerability Matrix, a tool for assessing pest vulnerability across a city.

2.1.10.1 Pests of Winnipeg's Urban Forest

The two most serious pests in Winnipeg's urban forest are emerald ash borer (EAB) and Dutch Elm Disease (DED). Emerald ash borer is an introduced phloem-feeding boring beetle that kills trees in the genus *Fraxinus*. The beetle's larvae feed under the bark and eventually girdle the tree, resulting in the tree's death in 1–3 years. EAB feeds on and is lethal to all species of North American ash (Poland & McCullough, 2006). The pest was first found in Winnipeg in November 2017 and was expected to kill most of the ash population of >356,000 trees over the next ten years (Barwinsky, 2016). To date (2024) the infestation has been confined to a very small area of the city (MacDonald et al., 2022) and has not spread to adjacent areas yet.

DED is caused by a fungal pathogen that infects elm xylem tissue. The xylem becomes lethally blocked with mycelia and the pathogen's ability to overcome the tree's defenses increases over time (Strobel & Lanier, 1981). In North America, DED is spread by the elm bark beetles *Scolytus multistriatus* Marsham and *Hylurgopinus rufipes* Eichhoff which move between healthy and infected trees to lay eggs and construct brood galleries under the bark (Wood, 1982; Russell 2021). DED is widespread, causing a high degree of mortality to North American and European elm species, although many Asian species, including Siberian elm (*Ulmus pumila* L.), are less affected (Strobel & Lanier, 1981). Winnipeg has the largest remaining urban elm canopy in North America (>230,000 trees), and the city has been managing DED for over 40 years (M. Barwinsky, personal communication, November 6, 2018).

Other insect pests are found sporadically in Winnipeg's urban forests. They include spruce budworm (*Choristoneura freemani* Freeman), fall and spring cankerworm (*Alsophila pometaria* and *Paleacrita vernata*) and forest tent caterpillar (*Malacosoma disstria* Hubner). These are not directly lethal to trees, although defoliation can be unsightly and repeated infestation can leave trees susceptible to more serious infections or pest outbreaks (City of Winnipeg, 2019). One of the most serious of these defoliating pests in recent years has been elm spanworm (*Ennomos subsignaria* Hubner), which the City manages by spraying affected trees with the biological insecticide *Bacillus thuringiensis* var. *kurstaki* (City of Winnipeg, 2020a). The City of Winnipeg also manages for black knot fungus (*Dibotryon morbosum* Schwein) by removing infected Schubert chokecherry trees (*Prunus virginiana* 'Schubert'). This pathogen attacks *Prunus* species. producing large, black galls on stems of infected trees (Winnipeg Public Works Department, 2009). Oak decline, a stress-induced condition prevalent in the City's Bur oak (*Quercus macrocarpa* Michx.), is also being monitored. Oak decline develops over time due to environmentally-related stress and dieback can result in subsequent infection by the two lined chestnut borer (*Agrilus bilineatus* Weber) (Winnipeg Public Works Department, 2012).

2.1.10.2 Pest Vulnerability Matrix

The Pest Vulnerability Matrix (PVM) is an Excel-based tool developed by Igor Laćan and Joe McBride (University of California, Berkeley) to determine the most serious pests and most susceptible tree species over a given area within urban forests (Laćan & McBride, 2008). Users input tree species by percent representation and pests that are present or are expected to be present in a forest. Pests are pre-classified by the software into low, moderate, and high severity classes based on lethality, and are assigned to tree host species present in the city. The program calculates the number of pests affecting any given species as well as the percent of trees species affected by a given pest and organizes the information into an accessible visual matrix for candidate forests (Laćan & McBride, 2008).

Though useful for visualization of common tree species and pests, PVM is not a comprehensive pest management planner, and Lacan and McBride (2008) recommend using only the 20 most numerous tree species for maximum utility. PVM was developed for users in California but has been adapted in multiple contexts to design planting regimes and identify vulnerable neighborhoods, including use in Toronto, Ontario (Vecht & Conway, 2015).

To quantify the output of the PVM, McPherson and Kotow (2013) calculated a pest score for each species in a given area by weighting and summing the severity classes of potential pests reported by the PVM (3,5, and 7 for low, moderate, and high severity pests). The area was then assigned an overall pest vulnerability score, the sum of each species' pest scores multiplied by their percent representation (McPherson & Kotow, 2013) with the results used by urban forest managers to prioritize integrated pest management activities over large urban landscapes.

2.2 Carbon Capture in Urban Forests

The carbon captured by urban trees could contribute to the mitigation of global climate change. Carbon capture consists of the carbon that is already contained in a given carbon reservoir (storage) and the net amount of carbon removed from the atmosphere to the reservoir over a given time period (sequestration) (Nowak et al., 2013). Pasher et al. (2014) report that in Canada alone, urban forests store an estimated 34 million tonnes of carbon and sequester 2.5 million tonnes of carbon yearly. Forests can be either carbon source or sink; yearly growth sequesters carbon and the energy savings associated with the cooling effects of urban trees reduce emissions, while tree death and subsequent decay release carbon, and act as a carbon source if the trees are not replaced (Nowak, 1993). It is possible for carbon balance to be positive or negative, depending on these conditions in a given year.

Based on the 2023 Comprehensive Urban Forest Strategy (City of Winnipeg, 2023), Winnipeg's urban forest sequesters 39,000 tonnes of carbon annually and stores an estimated 509 thousand tonnes of carbon, valued at $39 million (i-Tree estimates carbon value at $51.23 USD/tonne (Nowak 2020)). This includes 172 thousand tonnes stored in the City's American elms. The City of Winnipeg used i-Tree Eco to determine the carbon numbers in this report (City of Winnipeg, 2023).

2.2.1 Factors that Affect Carbon Capture

Factors that affect growth rate also affect carbon sequestration in trees. Nowak (1993) reported variation in growth rates based on tree species and age. Older trees tend to sequester less carbon yearly than their younger counterparts. Competition, use of root space, and allocation of resources to reproduction also play a role in carbon uptake (Thomas, 2011). Climate, soil type, and access to sunlight also affect growth (Boukili et al., 2017).

In cities, trees face unique environmental conditions that can negatively affect growth, including soil compaction, scarce rooting space, root damage and/or removal for construction, impermeable pavement surfaces, and pruning practices for safety and utility that do not always promote tree health (Mullaney et al., 2015). On the other hand, urban trees often face less competition from other trees than their natural counterparts, and they may have more access to nutrients and water than trees found outside cities in the same regions (McHale et al., 2009).

2.2.2 Measuring Carbon in Trees

Researchers began to recognize the potential of forests as carbon sinks during the 1970s (Dyson, 1977). Since then, biologists have been using forestry calculations developed for measuring timber harvests, including those for volume, weight, density, and growth, to estimate carbon storage and sequestration in trees.

2.2.3 Methods

2.2.3.1 Direct measurements

Direct measurement of carbon stocks is usually done destructively. Trees are felled and subdivided for measurement. Roots can also be excavated. Tree volume can be measured by displacement in water, called xylometry, and although this is most accurate, it is difficult to do and thus not common (Kershaw et al., 2016). More frequently, volume is measured by measuring sections of a tree as distinct, geometric solids like cones, cylinders, and neiloids; some examples include the Newton, Smalian, and Huber formulas (Kershaw et al., 2016). More recently, studies in urban settings have used LIDAR to estimate tree volumes, which allows measurement without felling the tree (Lefsky & McHale, 2008; McHale et al., 2009).

Ideally direct measurements of biomass would be taken to measure biomass. Green weight includes the weight of water in plant tissues, while dry biomass measurements are taken after the wood has been dried in an oven (Kershaw et al., 2016). To determine carbon content (in % or g C g^{-1} dry weight) from a dried sample, researchers use elemental analysis or mass spectral analysis (Kershaw et al., 2016).

Assessing carbon sequestration requires measuring growth over time. Researchers can use dendrochronological methods to age trees, and this is usually how predictive equations for age based on size are developed (Kershaw et al., 2016). However, urban forestry professionals often use other means to determine age in trees. These can include using city planting records, examining aerial photography, and interviewing residents and tree protection associations (Łukaszkiewicz & Kosmala, 2008; McPherson & Peper, 2012).

2.2.3.2 Developing allometric equations

Tree allometric equations describe relationships between some fundamental tree measurement (eg. DBH) and another property of the tree that is more difficult to determine directly (eg. biomass) (Parresol, 1999). From direct measurements, foresters have developed allometric equations for volume, weight, biomass, carbon storage, carbon sequestration, and more. Volumetric equations predict volume based on DBH or height and DBH, generally by species (Nowak, 2020). This is often accomplished by transforming the data using a natural log function, and then completing a linear, or less often, a weighted least squares regression (McHale et al., 2009). This has been shown to underestimate values in large trees, so sometimes a correction factor is added for this as well (Parresol, 1999). This can be multiplied by a dry-weight density factor to record biomass; the Global Wood Density database provides species-specific values widely used in urban forestry (Nowak, 2020). Often a below-ground volume adjustment is made at this stage to account for root biomass (McPherson et al., 2016). Some equations allow the user to calculate biomass directly from DBH or DBH and height. From biomass, percent carbon is calculated, usually at or near half the biomass value (Thomas & Martin, 2012). There is variation in percent carbon depending on species, ranging from 43% to 55% in boreal species (Thomas & Martin, 2012).

2.2.3.3 Scaling Up

Once carbon storage equations have been developed, researchers work to hone and refine them for accuracy (Nowak, 2020). Sometimes multiple equations are combined. For example, Nowak et al. (2020) combined carbon equations that covered a range of diameters for a given species, so that a single equation works for a larger size range for that species. Many equations can also be combined into a user-friendly database that performs calculations across multiple species. Sections 2.4 and 2.5 will discuss two such collections: i-Tree and the United States Department of Agriculture Urban Forest Database.

2.2.4 Growth and storage in urban trees

When researchers first began measuring trees in urban areas, they used estimates based on direct measurements from non-urban forests. These were readily available, but many researchers have since raised concerns over their ability to accurately describe urban trees. Rhoades and Stipes (2006) observed that trees in urban settings, being grown in more open environments than non-urban trees, develop a stronger taper. In some cases, urban trees grow larger than their non-urban counterparts due to increased access to water and nutrients and decreased competition for light (Rhoades & Stipes, 2006). However, some literature shows that urban trees are smaller due to stressful growing conditions (Close et al., 1996; McHale et al., 2009). McHale et al. (2009) found that biomass calculations developed from urban trees differed significantly from other allometric equations in the literature, including those used by i-Tree's precursor, STRATUM. Depending on which of 11 species was examined, these authors found that equations from literature overestimated mature tree biomass for urban trees, and that they were accurate in medium sized trees. These equations could also underestimate biomass in small trees or underestimate biomass for all tree sizes (McHale et al., 2009).

In recent years, efforts have been made to account for the differences in growth produced by different allometric equations. The simplest measures involve using a common adjustment factor for urban trees. For example, i-Tree reduces biomass estimates by 20% for open-grown trees (see section 2.2.3 for more information). Some researchers have developed urban-forest-specific growth equations (McPherson et al., 2016; Monteiro et al., 2016; Troxel et al., 2013), such as those in USDA's Urban Tree Database (McPherson et al., 2016), discussed below.

2.2.5 USDA Urban Tree Database

The United States Department of Agriculture has developed an Urban Tree Database (UTD). The UTD is available online, and includes the raw data used to derive allometric equations (McPherson et al., 2016). The database includes 171 species-specific equations to derive volume, woody and foliar biomass, and a variety of growth characteristics. There are also general-use equations based on tree form for species not listed in the database. The UTD was developed over 20 years using 14,487 trees from 6 geographical regions of the United States (McPherson et al., 2016). It can be viewed as a successor to i-Tree Streets, using more sophisticated statistical analysis to build growth equations. The biomass equations reported in i-Tree Streets were developed only using best fit to linear, exponential, and logarithmic functions, while UTD equations were based on best fit to 12 equation forms over 7 parameters. UTD users do not submit their data to the USDA; it is an open-source catalogue of equations, including a user guide with step-by-step instructions for calculation (McPherson et al., 2016). Users only need DBH to calculate carbon storage in some species, while they need DBH and height for others.

The UTD user guide does not give specific instructions for calculating carbon sequestration, although McPherson et al. (2016) do state that the database can be used to do so. Some equations provided calculate dry weight biomass from DBH and height, while some calculate biomass from DBH alone (McPherson et al., 2016). Equations were developed by species and region, testing 12 equation forms across 7 parameters (McPherson et al., 2016). Users must account for belowground biomass by multiplying the dry weight biomass values by 1.28. To calculate carbon storage from the UTD, authors suggest multiplying biomass by a factor of 0.5, and to calculate carbon dioxide stored, to multiply by a factor of 3.67 (McPherson et al., 2016). The database does include species-specific equations for predicting age using DBH by region (McPherson et al. 2016), so it may follow that the user could project tree size into the future or past using these equations to measure sequestration against current carbon storage.

2.2.6 i-Tree Eco

i-Tree is an accessible toolset that allows urban foresters to measure a variety of traits using free software developed by the USDA (i-Tree Tools, 2012). Of the many i-Tree tools available, the City of Winnipeg has used i-Tree Eco software to examine a variety of factors, including tree density, compensatory value, runoff avoided, pollution removal, and carbon storage, in Winnipeg's urban forest (City of Winnipeg 2018). I-Tree Eco allows users to input a sample of city trees with measurements as well as a number of local variables, including weather, pollution levels, and regional geography (Maco & Nowak, 2011). While UTD outputs are largely based on DBH, i-Tree Eco uses a number of tree and site measurements to determine outputs (see Table 2.3 for a complete breakdown of inputs and outputs). The program can return a range of data, including species structure and distribution, value estimates, and carbon storage and sequestration data (i-Tree Tools, 2012). Although it was first derived for use in U.S. cities, the makers of the software now claim to accommodate both Canadian and international cities (Maco & Nowak, 2011). I-Tree Eco has also been used to quantify forest structure in Edmonton (City of Edmonton, 2019d) Toronto (Toronto Parks, Forestry, and Recreation, 2008), and Halifax, as well as a number of smaller Canadian municipalities (Foster & Duinker, 2017).

Table 2.3 Inputs and outputs of i-Tree Eco, directly as published in Nowak (2020). D= directly used; I= indirectly used; C= conditionally used.

	DERIVED VARIABLES		ECOSYSTEM SERVICES										
DIRECT MEASURES	Leaf Area	Leaf Biomass	Carbon Storage	Gross Carbon Sequestration	Net Carbon Sequestration	Energy Effects	Air Pollution Removal	Avoided Runoff	Transpiration	VOC Emissions	Compensatory Value	Wildlife Suitability	UV Effects
Species	D	D	D	D	D	D	I	I	I	D	D		
Diameter at breast height (d.b.h.)			D	D	D						D	D	
Total height	D	D	C	C	C	D	I	I	I	I		D	
Crown base height	D	D	C				I	I	I	I			
Crown width	D	D	C				I	I	I	I			
Crown light exposure			C	D	D								
Percent crown missing	D	D	C	C	C	D	I	I	I	I			
Crown health (condition/ dieback)				D	D						D	D	
Field land use				D							D	D	
Distance to building						D							
Direction to building						D							
Percent tree cover						D	D	D				D	D
Percent shrub cover							D					D	
Percent building cover						D							
Ground cover composition							I					D	

Most if not all equations used in i-Tree Eco are derived from non-urban-forest growth equations (Nowak, 2020). Biomass values of all trees that are reported as having 4-5 faces open to sunlight are multiplied by 0.8, as observed by Nowak (1994) in Chicago street trees. Nowak (2020) commit to seeking out more urban-forest specific equations in the next i-Tree Eco update, including integration of data from GlobAllomeTree (http://www.globallometree.org/accounts/login/), a free, international web platform for sharing tree allometric equations. Carbon storage in individual trees is capped at 7500 kg in i-Tree to prevent overestimation of storage in very large trees. Like the UTD, i-Tree Eco uses a factor of 0.5 to convert biomass to carbon storage. When a biomass equation for a species is not in the i-

Tree database, the average value of the other members of the genus or that of the closest phylogenetically-related tree is used.

2.2.7 Species-Specific Carbon Capture

In the following sections, I summarize the carbon capture literature for the three most common trees in Winnipeg's residential urban forest. All biomass equations referenced are available in Appendix F.

2.2.7.1 Ulmus americana

American elm is the most common street tree in Winnipeg's urban forest. It is the largest elm species that grows in Canada, with a graceful, vase like form that made it popular as a shade tree across North America in the first half of the 20th century (Bey, 1990). In some habitats, American elm can reach 35 meters in height, and can grow as large as 175 cm in DBH (Farrar, 1995).

American elm biomass equations in i-Tree, as reported by Nowak (2002, 2020), were derived from equations developed using destructive sampling and hand measurement in Minnesota, Wisconsin, and Illinois forests (Hahn, 1984). I-Tree uses a wood density value of 0.46 tonnes/m^3 for American elm, from the Global Wood Density Database.

The UTD biomass equations for American elm were derived from Lefsky and McHale (2008), who used LIDAR technology to quantify tree biomass in urban street trees in Fort Collins, Colorado. The USDA also uses a wood density value of 0.46 g/cm^3. See Figure 2.1 for a comparison of i-Tree and USDA values for American elm.

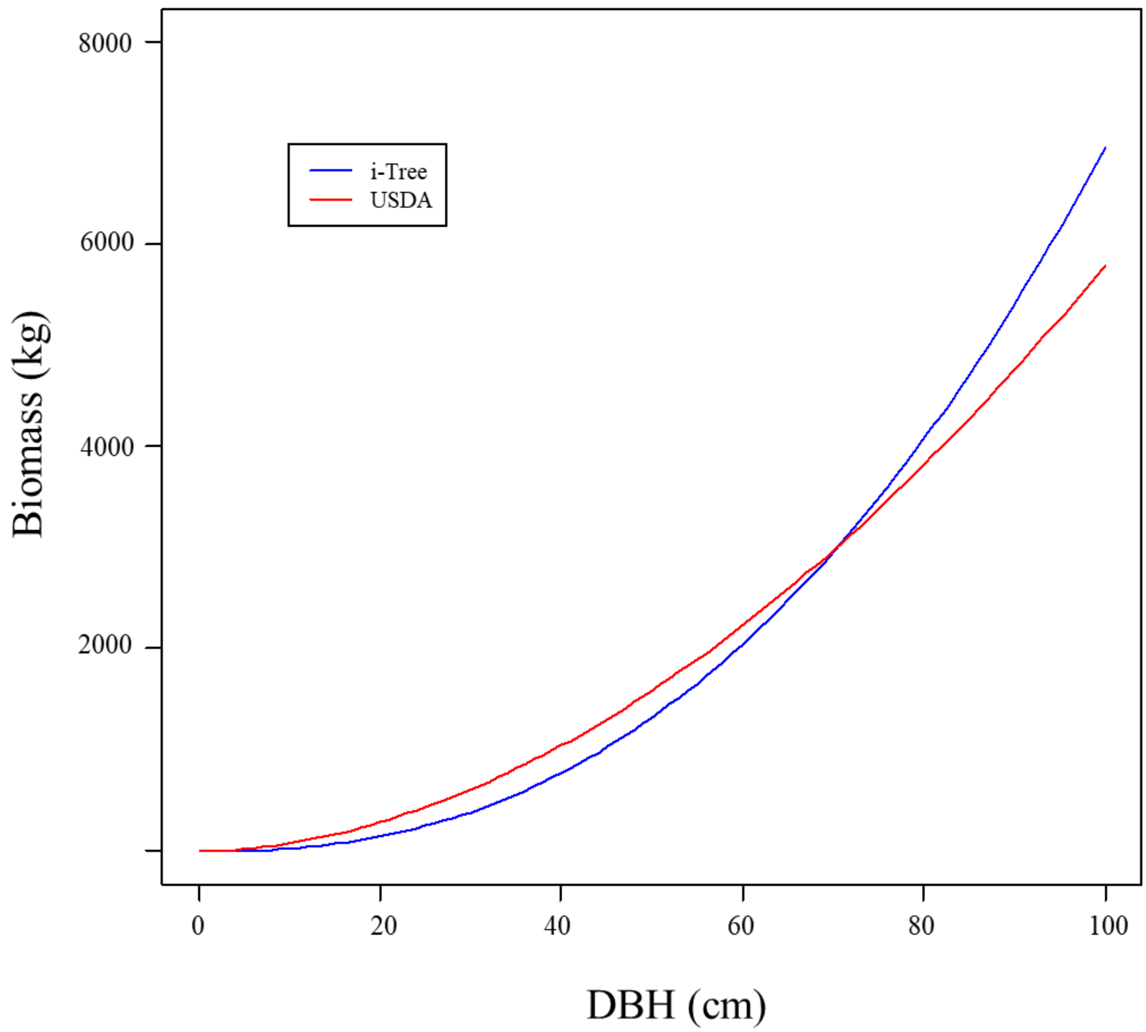

Figure 2.1 American elm biomass curves for USDA and i-Tree equations.

2.2.7.2 Fraxinus spp

The second most common trees in the inventory in the current study included ash species. Green ash (*Fraxinus pennsylvanica* Marsh.), black ash (*Fraxinus nigra* Marsh.), and Manchurian ash (*Fraxinus mandshurica* Rupr.) were all identified in the present study, but the majority of observations were made to genus rank only (60% of all ash observations). The City of Winnipeg public tree database indicates that 88% of ash trees on public property are green ash, 7% black ash, and 4% Manchurian ash.

Green ash was widely planted in Winnipeg during the 1960's in anticipation of the arrival of DED (City of Winnipeg, 2023). Ash does not have a consistent growth form. Green ash can grow well in urban environments, although it may grow slowly in degraded soils (Kennedy, 1990). In Canada, green ash can grow to 25 meters, and may reach a DBH of 60 cm (Farrar, 1995).

i-Tree uses green ash biomass equations derived from those developed by Schlaegel (1984) using destructive sampling and hand measurement on green ash trees from the Mississippi Delta (Nowak, 2020; Nowak, 1994). These require the input of both DBH and height (Nowak, 2020). I-Tree uses a wood density value of 0.53 tonnes/m^3 for green ash, from the Global Wood Density Database.

Like American elm, the UTD biomass equations for green ash were derived from Lefsky and McHale (2008), who used LIDAR technology to quantify tree biomass in urban street trees in Fort Collins, Colorado. The UTD also uses a wood density value of .053 tonnes/m^3. See Figure 2.2 for a comparison of i-Tree and UTD values. Because i-Tree's biomass equation requires height, the i-Tree height regression for ash was used to estimate height used in biomass equation for this figure.

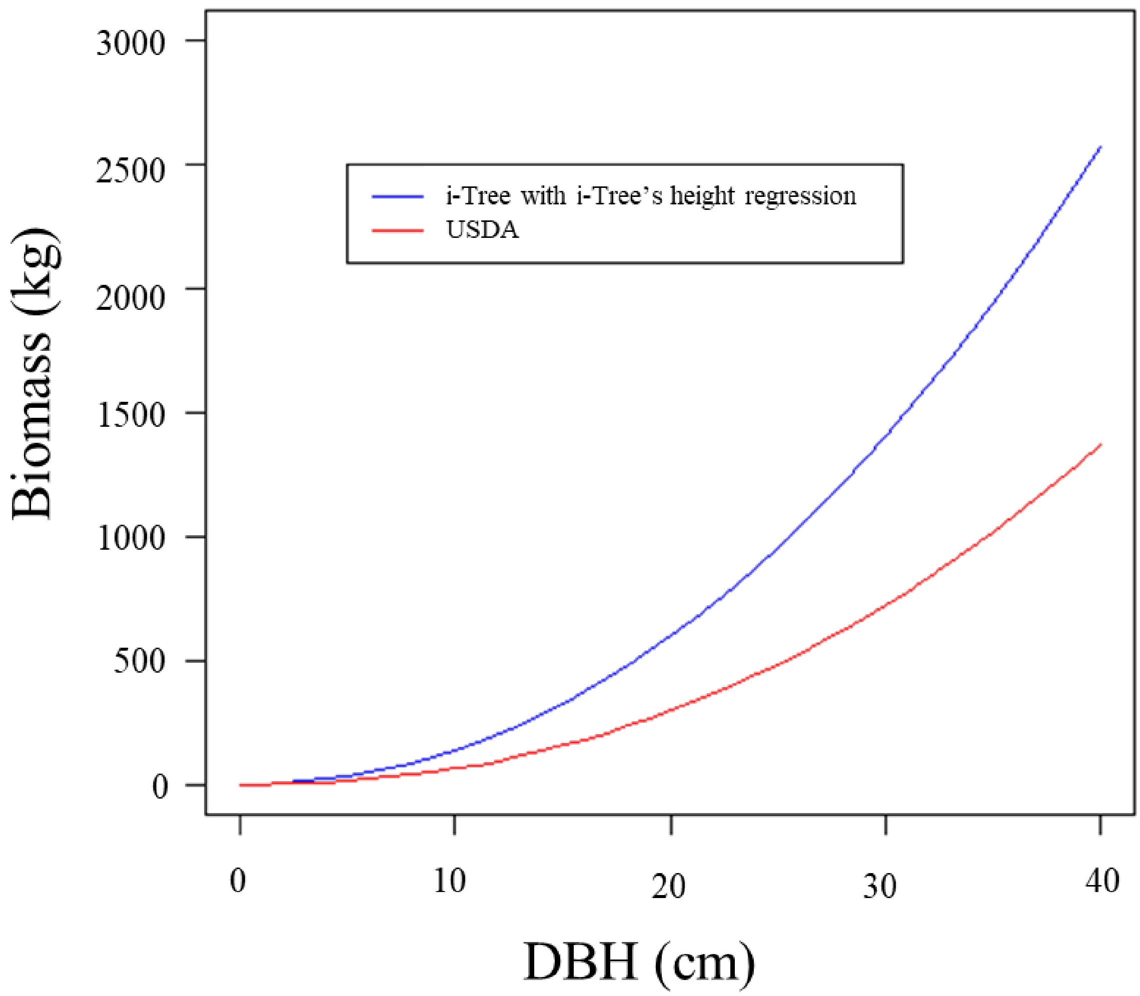

Figure 2.2 Green ash biomass curves for USDA and i-Tree equations, using i-Tree's height regression.

2.2.7.3 Picea spp

Spruce trees are common yard trees in Winnipeg. Due to their dense, low foliage, spruce trees are not permitted as boulevard trees on most city streets because they block traffic sightlines. White spruce (*Picea glauca* Moench), Colorado spruce (*Picea pungens* Engelm.), Norway spruce (*Picea abies* L.), and black spruce are found in the city, although in this study all spruce species were grouped into a *Picea spp* taxon for ease of identification at a distance. Of the spruces in the City of Winnipeg public tree database, 56.7% were Colorado spruce, 43.0% were white spruce, and the remaining 0.3% were Norway spruce and black spruce (*Picea mariana* Mill.).

White spruce can grow to 25 meters in the wild, with a DBH of 60 cm. Colorado spruce are generally a bit larger than white spruce, and in urban settings white spruce grow to 30 meters and can reach 90 cm in DBH (Farrar, 1995).

For spruce, i-Tree biomass equations are derived from studies using destructive sampling and hand measurement of spruce species in the northeast United States (Nowak, 2020; Nowak, 1994; Tritton, 1982). A general spruce biomass equation is available, as well as one for white spruce. i-Tree uses a wood density value of 0.36 tonnes/m^3 for Colorado spruce and 0.33 tonnes/m^3 for white spruce, both from the Global Wood Density Database.

The UTD biomass equations for spruce trees were derived from McHale et al. (2009), who used LIDAR technology to quantify tree biomass in urban street trees in Fort Collins, Colorado. The UTD also uses a wood density value of 0.36 tonnes/m^3 for Colorado spruce; no white spruce were sampled. See Figure 2.3 for a comparison of i-Tree and UTD values.

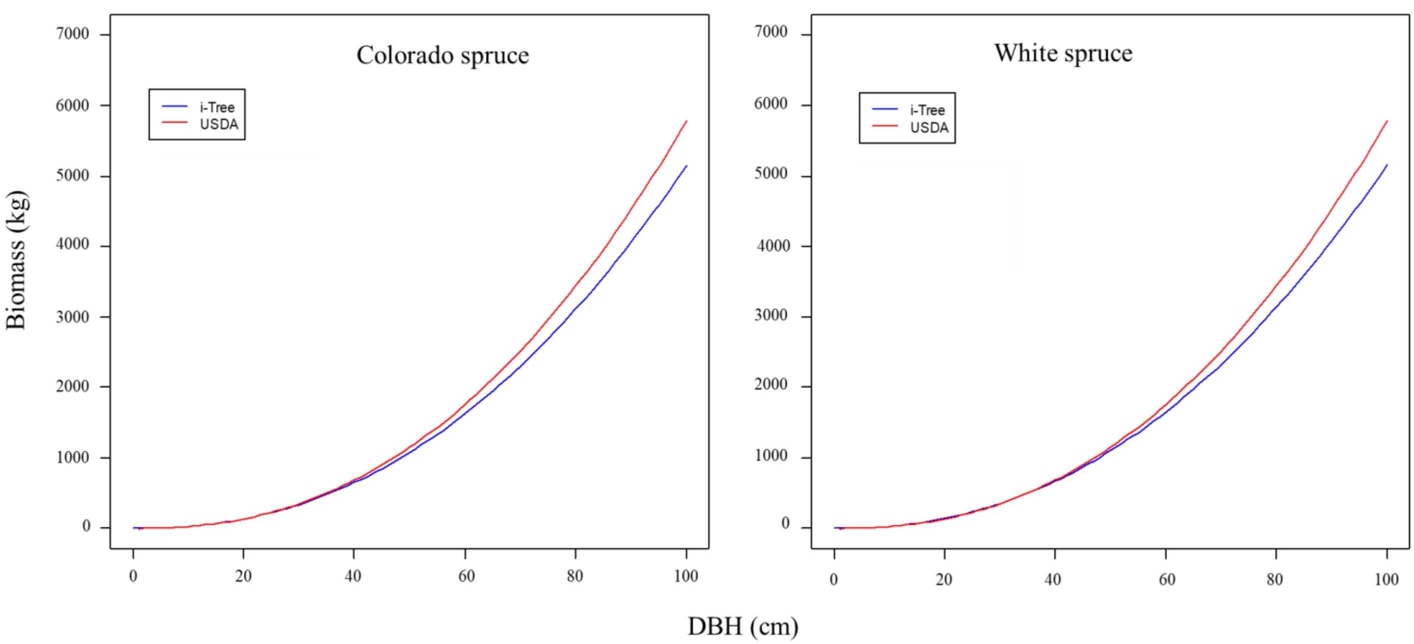

Figure 2.2 Colorado spruce (left) and white spruce (right) biomass curves for USDA and i-Tree equations.

2.2.8 Carbon Management in Winnipeg's Urban Forest

In September 2018, The City of Winnipeg committed to the *Winnipeg Climate Action Plan*, which calls for city-wide reduction in greenhouse gas emissions by 20% by 2030 relative to 2011 levels, and 80% by 2050 (City of Winnipeg, 2018). This includes a stated goal (7.2) to "Increase and Preserve Tree Canopy" (City of Winnipeg, 2018). See Appendix C for specific content of this plan.

The City of Winnipeg *State of the Urban Forest Report* (City of Winnipeg, 2021) summarizes carbon estimates based on i-Tree findings across the City. Carbon storage across the city was estimated at 509,348 t and valued at $39.2 million CAD. This total carbon storage includes 98,500 t in public trees.

Although it does not contain specific carbon targets, the 2023 Winnipeg Urban Forest Strategy (City of Winnipeg, 2023) outlines specific actions to impact tree size and number. In Policy 6, the strategy describes actions to "promote carbon sinks," calling for a 1:1 replacement of trees removed from boulevards and parks and planting 760,000 new trees by 2065. Moreover, the strategy proposes a target of no more than two percent annual loss of elms to disease (City of Winnipeg, 2023). If met, this target would allow the city to replace larger trees slowly over time. The strategy also calls for the City to establish new standards that will improve health and longevity of newly planted trees, increasing the number of trees that survive to maturity for maximum carbon capture over their lifespan (City of Winnipeg, 2023).

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Chapter 3

Tree Diversity and Pest Vulnerability

3.1 Introduction

City foresters are increasingly managing for diversity in urban forests across North America (Morgenroth et al., 2016). This approach contrasts with historical attempts to plant “the perfect street tree,” which led to the planting of American Elm (*Ulmus americana* L.) monocultures in many North American cities and towns (Haugen, 1998). These monocultures were later decimated by the introduction and spread of Dutch elm disease (DED) (*Ophiostoma ulmi* Buisman) (Rosen, 2015). Winnipeg, Manitoba epitomizes this trend, with historical planting regimens of American elm and later, green ash (*Fraxinus pennsylvanica* Marsh), being planted in large numbers. These species now dominate older neighborhoods (City of Winnipeg, 2023). More recently, through its comprehensive urban forest strategy, the City of Winnipeg has developed a planting plan that aims to diversify the urban forest. The plan has a stated goal to ensure that no more than 10 percent of any species and 20 percent of any genus should appear in the tree inventory (City of Winnipeg, 2023).

In northern forests, tree species diversification efforts are limited by climatic conditions. Winnipeg is one of the few major cities in USDA hardiness Zone 3b in North America, where trees experience significant climate stressors compared to other Canadian cities (Table 3.1). The City of Winnipeg Urban Forestry Branch has only 39 tree species on the approved boulevard tree planting list, as Winnipeg city foresters have chosen to avoid some tree species due to concerns about survival over harsh winters (City of Winnipeg, 2023).

Table 3.1 USDA Hardiness Zones for major Canadian cities.

City	USDA Hardiness Zone
Winnipeg	3b
Calgary	4a
Edmonton	3b
Halifax	6a
Montreal	4b
Ottawa	4b
Regina	3a
Toronto	6a
Vancouver	8b

Urban foresters cite numerous benefits of tree species diversification:, including resilience to climate change, maintenance of aesthetic values, and perhaps foremost, pest control and reduction of pest impacts (see section 2.1.2). Monoculture plantings leave large portions of the urban forest vulnerable to severe pest impacts, particularly from species-specific pests (City of Winnipeg, 2023). In Winnipeg, DED and emerging pests such as emerald ash borer (*Agrilus planipennis* Fairmaire), and cottony ash psyllid (*Psyllopsis discrepans* Flor) have the potential to affect over 50% of the street and park trees (City of Winnipeg, 2023).

Another challenge in diversifying urban forests lies in maintaining an accurate tree inventory upon which to base management decisions. The City of Winnipeg has a complete inventory of street trees and trees in public parks (City of Winnipeg, 2023), but an inventory of trees on private property has been lacking. City foresters often base their choice of trees on practicalities like cost, availability of stock in bulk, suitability to the boulevard environment, or ease of pruning and cleanup (eg. minimal flowers and/or fruit). However, homeowner purchasing and planting decisions may be guided by different considerations. In a study of homeowners in Mississauga, Ontario, Conway (2016) found that homeowners were most likely to select trees for aesthetic reasons, for screening property boundaries, and for shade. Thus, the mix of trees on private property is likely to differ from the mix used in boulevard plantings and therefore may serve as a significant source of diversity in the urban forest.

In this chapter, I explore patterns of tree diversity in Winnipeg, and seek to answer the following research questions:

1) How diverse is Winnipeg's urban forest in terms of tree species?
    i) How does tree diversity vary by neighborhood?
    ii) Do private and public tree holdings differ, and if so,
    iii) How do private holdings contribute to tree diversity in Winnipeg?
2) What neighborhoods are most susceptible to current and future pest invasions? Is there a relationship between pest vulnerability and species diversity?

3.2 Methods

3.1.1 Data Collection

Tree surveys were completed on public boulevards and private front yards in Winnipeg neighbourhoods between the years of 2002 and 2019. One third of the data were collected by the author (2019) with the remaining data collected by urban-forestry-trained undergraduate forest ecology research assistants studying at the University of Winnipeg. During the first 10 years of the survey (2002 to 2012), city neighbourhoods were chosen ad hoc to represent three broad age classes of trees based on neighbourhood age (newer neighbourhoods built after 1980, middle aged neighbourhoods built between 1940 and 1980 and older neighbourhoods built prior to 1940). The term "neighborhood" is used to refer to the City of Winnipeg's neighborhood classification utilized for zoning purposes and electoral districts (City of Winnipeg, 2020) (Figure 3.1). After 2014, neighbourhoods were chosen to provide more replicates of the three broad age classes and to fill in gaps in areas of the city not well surveyed previously. Of Winnipeg's 197 residential neighborhoods, parts of 99 neighborhoods were surveyed (see Appendix A for neighborhoods surveyed, including statistical summaries). The survey unit was considered to be one city block. Blocks within each neighborhood were chosen randomly from street maps in the first 10 years of the study. For the remaining years of the study, ArcMap software was used to choose blocks at a density of 4 blocks/km^2, the average density for the first ten years of the study.

Survey crews identified and measured all boulevard trees and trees in the front yards of residential properties on city streets within sampled neighbourhood blocks. Backyards and back lanes (although only older neighborhoods had back lanes) were not included in the survey due to accessibility and privacy limitations. Parks, schools, riverbanks, and other open public spaces were also excluded from the sampling. Only trees were included in this study, as shrubs and annual vegetation were difficult to identify at a distance and have no standardized unit of measurement, such as DBH (diameter at breast height).

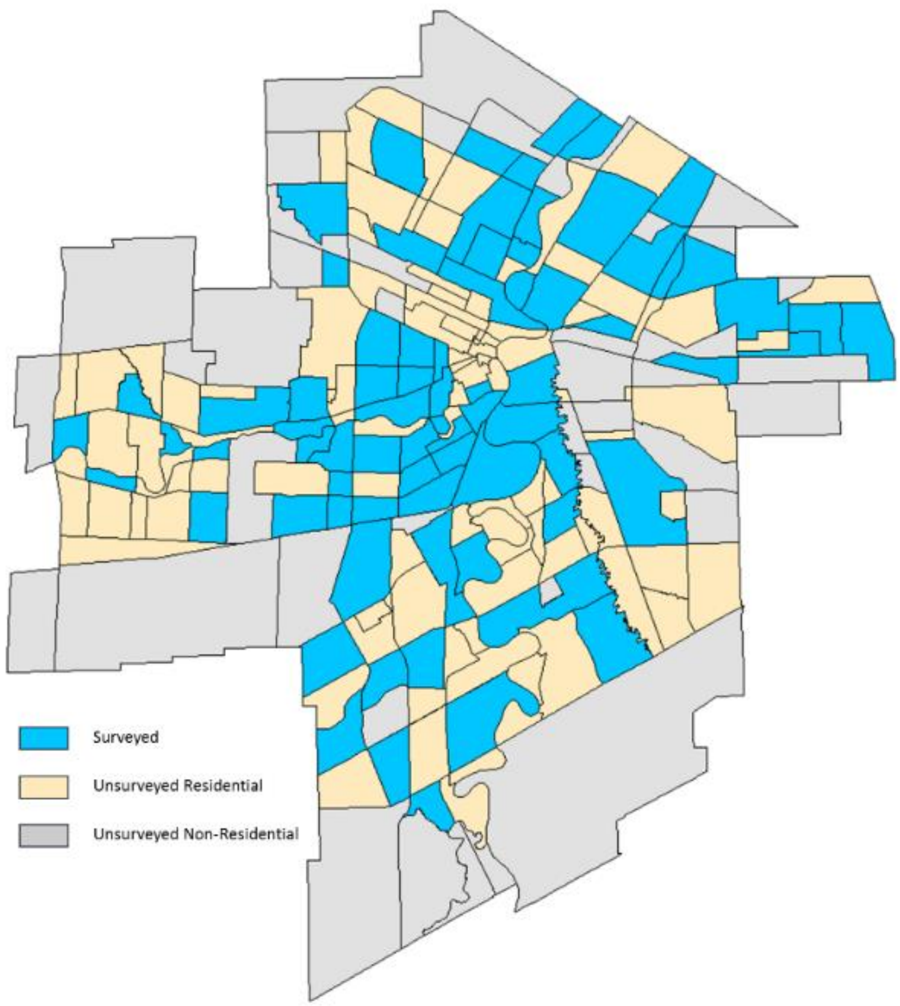


Figure 3.1 Neighborhoods zoning units surveyed in Winnipeg, MB

Surveyors recorded the following data from individual trees: DBH, height to the top of the crown, crown closure, species or genus of trees on public and private property (genus if species could not be determined due to the distance from adjacent public property), crown dieback (scale of 1-4, 1: no dieback, 2: 25% dieback, 3: 50% dieback, and 4: >75% dieback), tree location (private or public property), and the closest street address. For each block, data collection included average spacing between boulevard trees, a count of missing trees (spaces between boulevard plantings where there was evidence a tree was removed), and a count of infill trees (trees that were replanted after the removal of an existing boulevard tree and were significantly younger than remaining boulevard trees). Each property was matched to a city database listing its real estate value, area of the building footprint, property frontage length, area of the entire property, zoning designation, and year built. See Table 3.2 for a summary of variables and instruments used to measure variables. See Appendix I for photographs of surveys being undertaken.

Table 3.2 A summary of variables recorded during street surveys.

Variable	Tools Used	Description	Unit	Measured by Tree or Block
Diameter Breast Height	DBH tape (Forestry Suppliers Inc), estimation when trees not accessible	Diameter of tree 1.3 meters from the ground	cm (nearest 1 cm)	Tree
Height	clinometer (Suunto PM-5), estimation when trees not accessible	Measurement of tree height	m (nearest 1 m)	Tree
Species	---	Species of tree, or genus when trees not directly accessible	---	Tree
Dieback	---	A measure of dieback from 1-4; 1: no dieback, 2: 25% dieback, 3: 50% dieback, and 4: 75%-100% dieback	---	Tree
Spacing	tape measure	Average spacing between trees	m (nearest 1 m)	Block
Missing trees	---	Count of missing trees on boulevards	---	Block
Infill trees	---	Count of infill trees on boulevard	---	Block

3.1.2 Analysis

3.1.2.1 Tree Diversity

In some neighbourhoods, rarefaction analyses (Figure 3.2) showed that initial surveys had likely not captured the full proportion of the tree diversity of a neighbourhood. Rarefaction allows users to view species accumulation based on random sampling to build a rarefaction curve; the portion of the curve before an asymptote is reached generally indicates that the population is under sampled (Gart et al., 1982, Oksanen et al., 2019). Some adjacent neighbourhoods were therefore amalgamated (where homes were approximately the same age,

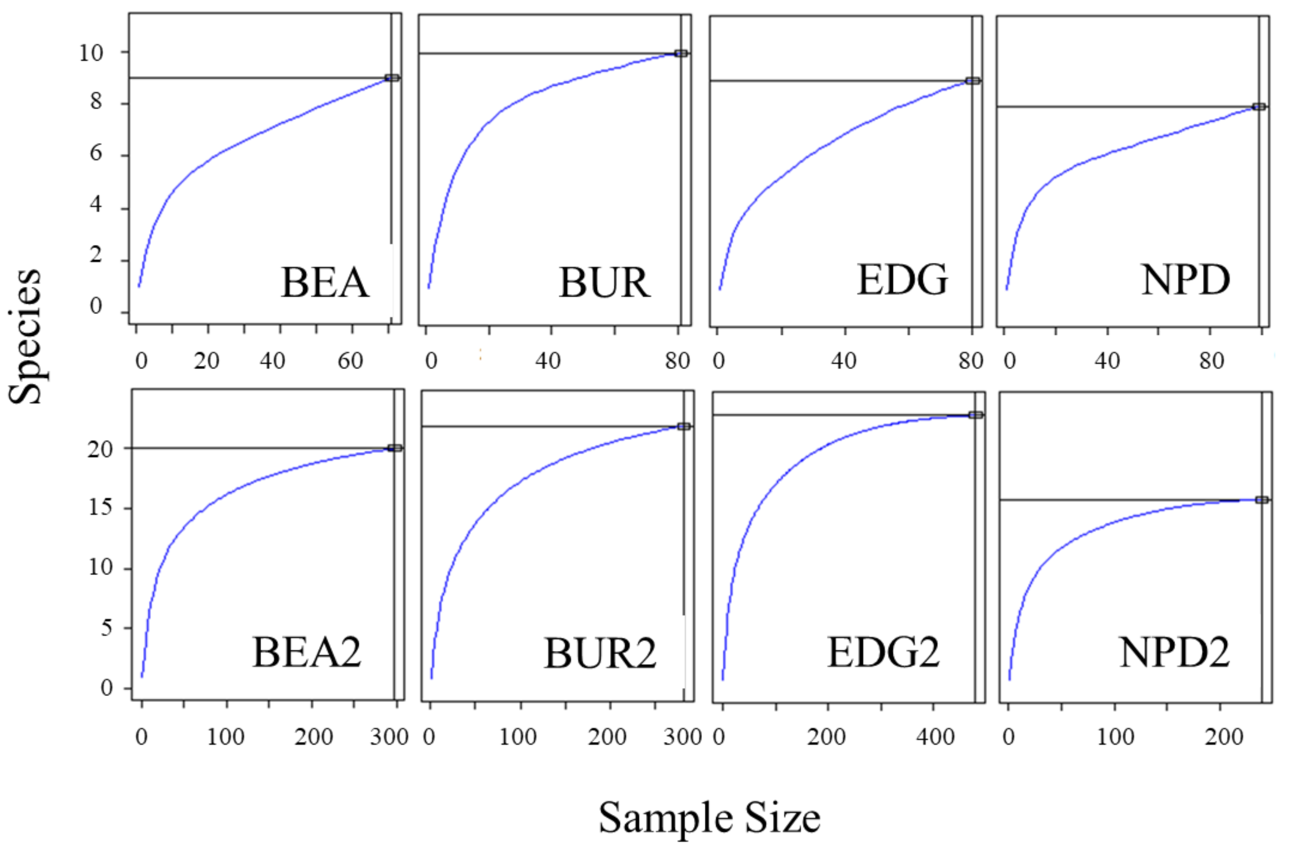


Figure 3.2 Example of rarefaction curves for tree diversity for four neighborhoods. The top row shows curves before amalgamation, the bottom row is after amalgamation. Neighourhoods: BEA = Beaumont, BEA2 = Pembina Strip, Beaumont, and Maybank, BUR = Burrows Central, BUR2 = Burrows Central and Shaughnessy Park, EDG = Edgeland, EDG2 = Edgeland and Sir John Franklin, NPD = North Point Douglas, NPD 2 = North Point Douglas and William Whyte.

value and with comparable lot sizes) to obtain larger samples for species diversity assessment. Thirty-two of the original 99 neighborhoods were amalgamated with adjacent neighborhoods. Wherever possible, undersampled neighborhoods were preferentially amalgamated, rather than joining undersampled and adequately sampled neighborhoods, for a final total of 75 amalgamated neighborhoods. This procedure also provided a more realistic representation of neighbourhoods based on physical similarities in tree composition, and a helpful reduction for urban forest managers, especially in terms of eliminating very small neighbourhoods. For a full list of amalgamated neighborhoods, see Appendix A.

After amalgamation a series of diversity and health indicators were calculated. See Appendix C for a list of indicators, formulas, and sample calculations. Diversity indices for each amalgamated neighbourhood (hereafter denoted as neighbourhood) included species richness, Simpson's 1/D Reciprocal Index, Simpson's Evenness Index, and Berger-Parker Dominance. Species richness, a count of species over a given area, is a simple and universal measure of diversity (Magurran, 2004). Simpson's 1/D Reciprocal Index (Simpson, 1949) reports the reciprocal of the probability of drawing two individuals of the same species at random from a population. The reciprocal is used to make the index more intuitive, with lower values indicating lower diversity. Simpson's 1/D accounts for richness and evenness, and it is less affected by sample size and less sensitive to rare species than similar indices (Simpson, 1949; Magurran, 2004). Simpson's Evenness Index was chosen in this study as a complementary metric to Simpson's 1/D; Simpson's evenness index reports only the evenness component of the more widely used 1/D metric by dividing 1/D by the sample size (n).

All equations are listed in Appendix C, Table C1. . Simpson's 1/D accounts for richness and evenness, and it is less affected by sample size and less sensitive to rare species than similar indices (Magurran, 2004; Simpson, 1949). Simpson's evenness index was chosen in this study as a complementary metric to Simpson's 1/D; Simpson's evenness index reports only the evenness component of the more widely used D metric.

Tree diversity in urban forestry is often gauged using the "10-20-30 Rule." Developed by geneticist Frank Santamour, this rule suggests that urban foresters should aim for a tree inventory containing no more than 10% of any given tree species, 20% any genus, and 30% any family (Santamour, 1999). For more background on the 10-20-30 Rule, see section 2.1.4. I report percent values for species, genus, and family for each neighborhood for comparison to this target.

Species richness and 10-20-30 values were calculated using species as listed in Appendix B, Table B1. All other diversity indicators were calculated using tree genera from Appendix B, Table B1, with American elm and Siberian elm calculated separately. For a summary of all indicators by neighborhood, see Appendix C, Table C2.

The final indicator, Berger-Parker Dominance, reports the proportion of the most populous species as a decimal ranging from 0-1. It is considered to a be an accurate measure of the numerical importance of the most abundant species in a sample (Magurran, 2004). In this thesis, a high Berger-Parker Dominance value indicates a neighborhood was heavily planted with one species (likely influenced by municipal planting), and being closer to a monoculture, is potentially more prone to severe pest infestation and large potential tree losses to the area.

Health was measured by assessing tree dieback. Mean dieback value was used to compare tree health across neighborhoods. Dieback was assessed as the approximate percentage of the crown that contained dead or dying branches.

Once all indicators were compiled, I used Wilcoxon's Matched-pairs Signed Rank Tests to compare public and private trees in each neighborhood by indicator. This test allows users to test if two matched groups (in this case, neighborhoods with trees on private and public property) are statistically different and can be used even when the data are not normally distributed. I also plotted relationships between number of trees on a property and various home characteristics, testing the relationships using Kendall's Tau statistic (a ranked correlation coefficient). Due to high heteroscedacity, relationships could not be determined by regression, however preliminary results of these plots are shown in Appendix A.

3.1.2.2 Pest Vulnerability

The Pest Vulnerability Matrix (PVM) was first developed by Laćan and McBride (2008) for use across several municipalities in central California (see Literature Review for further details). The PVM allows users to assess tree taxa present across columns and pests (including bacterial, fungal, and insect pests) present along rows in a particular sample tree population. Each pest is assigned a severity impact on each host tree species: low, moderate, or severe impact as represented by yellow, orange, and red coloring respectively. In Laćan and McBride's PVM, there is also crosshatching for cultivars that may show resistance to certain pests, but because the inventory compiled for my study did not include cultivar identification, this component was removed.

Several other modification to the PVM analysis were made. With the expert advice of a Winnipeg urban forester and review of the literature, I also removed pests not present in Winnipeg and added those not included in the original PVM matrix (Laćan & McBride, 2008), as well as removing tree species not found in my inventory or the City of Winnipeg inventory. Then I added pest species that the City of Winnipeg has on their urban forest pest watchlist that are likely to appear in the near future (City of Winnipeg, 2023). See Appendix D for a full list of changes to Laćan and McBride's original PVM.

I then used the PVM to calculate measures of vulnerability using many of the same calculations found in McPherson and Kotow (2013). I began by calculating the number of pests targeting each taxa in the PVM. I then multiplied these counts by 1 (low severity), 2 (moderate severity), or 3 (high severity) and summed these values for a *taxon pest score*. For instance, *Juniperus* species in Winnipeg would receive a pest severity score of 10 (8 low severity pests X 1 + 1 moderate severity pest X 2). These weights are an imperfect method to describe pest impact and must be determined by the practitioner using expert knowledge applicable to the region being studied to best reflect management realities. In Winnipeg, some pests that are still rare or not observed have the potential to be exceedingly impactful (e.g emerald ash borer, spongy moth (*Lymantria dispar* L.), Asian longhorned beetle (*Anoplophora glabripennis* Motschulsky). Though McPherson and Kotow (2013) used weights of 1, 3, and 5, I reduced the weights to reduce outsize effects of more severe pests on taxon pest scores. As categorizations of severity are subjective, this allowed inclusion of impacts of "less severe" pests that are nonetheless present in Winnipeg's forest in the calculations. See Table D3 in Appendix D for all taxon pest scores.

I then multiplied the calculated taxon pest scores by percent abundance of trees in each neighborhood. From this, I was able to identify and rank the most threatened tree species in each neighborhood. I report the number of times each taxon was ranked first, second, or third in a neighborhood ("No. 1 risk, No. 2 risk, No. 3 risk"), as reported in McPherson and Kotow (2013).

Subsequently the total score for each taxon was multiplied by the percent taxon abundance in each neighborhood to produce a matrix of pest scores for each taxon in the neighborhood. I then summed each neighborhood's scores for a *neighborhood pest score*. McPherson and Kotow (2013) use an importance value which incorporates abundance and tree size instead of using only abundance to make this calculation. I decided not to use size so that smaller trees on private properties were not undervalued. See Table 4 in Appendix D for all neighborhood pest scores.

To determine which pests have the capacity to affect the largest areas of forest in Winnipeg, I calculated the percent of each neighborhood forest potentially at risk for infestation by each severe pest.

I also wanted to compare neighborhood pest vulnerability to other diversity indicators discussed in the previous section. To do so, I calculated Kendall's Tau for each major diversity indicator. Kendall's Tau Statistic allows comparison of ranked lists; Tau values range from -1 to 1, with values of 1 indicating ranked lists are the same and values of -1 indicated ranked lists are reversed.

Because Santamour's 10-20-30 Rule is used as a measure of resilience, I also wanted to explore the relationship between the neighborhood pest scores and the 10-20-30 rule. Using Excel's Solver package (Mason, 2013), which allows users to define constraints on a system to determine optimum outputs, I was able to explore tree complements in a hypothetical Winnipeg urban forest to maximize and minimize neighborhood pest scores while creating "idealized urban forests" that adhered to the 10-20-30 rule. I first constrained the hypothetical urban forest to a 10-20-30 makeup and required that no taxon fall below 1% abundance. I then used the LP Simplex method (Mason, 2013) for linear models to compute the maximum and minimum pest scores possible under these constraints.

I also tried to explore the relationship between neighborhood age and pest vulnerability, but found no meaningful correlations were observed; this work is not reported in this thesis.

3.3 Results

3.1.3 Survey Summary Data

In total, 24,549 trees were measured across 99 neighborhoods. After the rarefaction analysis and amalgamation of specific neighbourhoods, 75 neighbourhoods were included in the analysis (hereafter referred to simply as neighborhoods). There were 39 tree species within 26 genera and 15 families found in the survey (see Appendix B). Thirty-two percent of observations were trees were on public property, 67% on private property, and 1% were addresses with neither trees nor shrubs. Across all neighbourhoods 69% of trees were given a dieback rating of one, 24% a rating of two, 5% a rating of three, and 1% a rating of four.

See Figures 3.3 and 3.4 for a summary of height and diameter ranges by genus. The most numerous tree species in the survey were American elm (*Ulmus americana*, 23%), ash (*Fraxinus* species,15%), eastern white cedar (*Thuja occidentalis*, 14%), spruce (*Picea* species, 13%), and maples (*Acer* species, 8%).

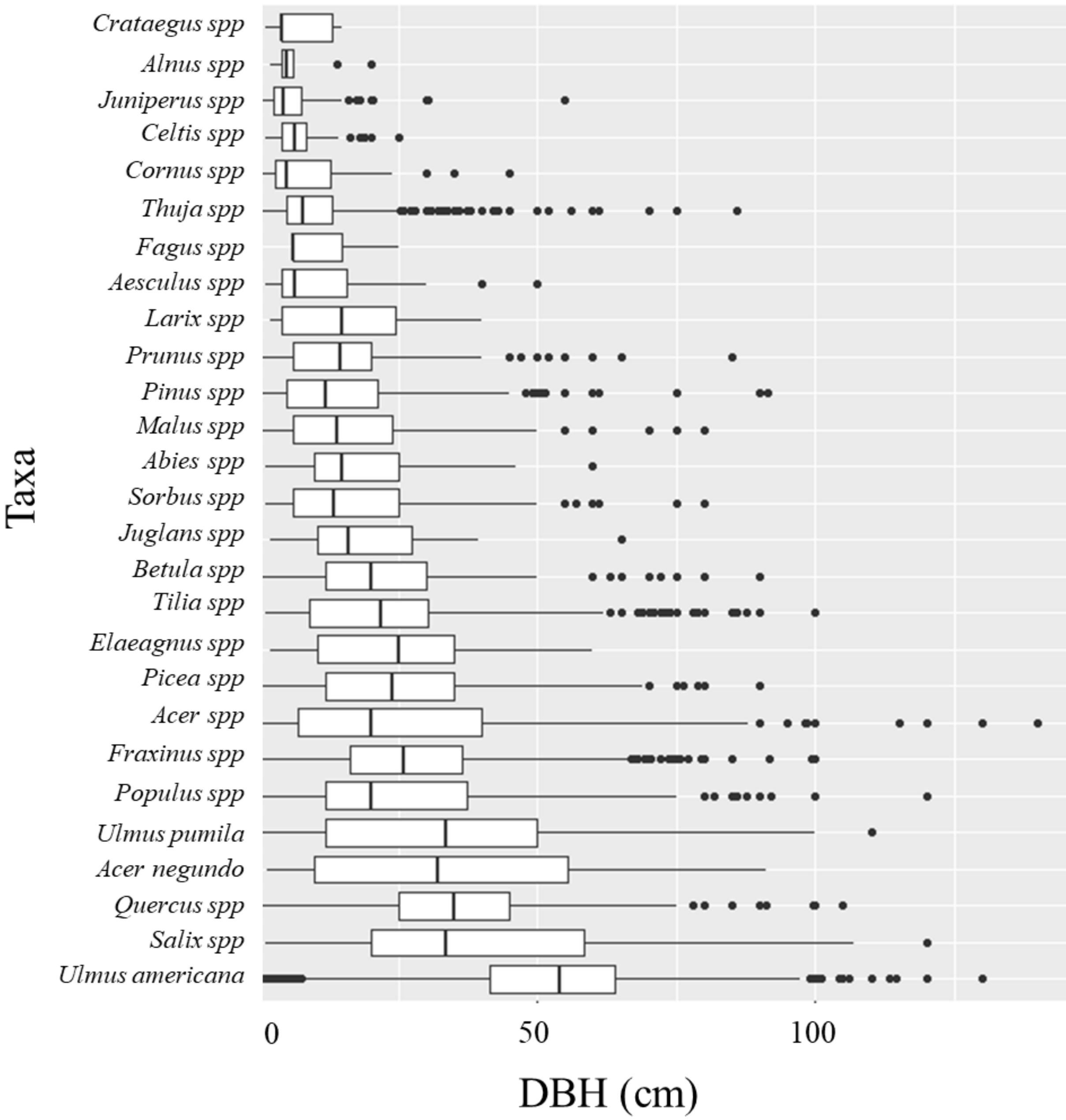


Figure 3.3. DBH ranges for tree genera recorded in the survey. American elm, Siberian elm, and Manitoba maple are broken out further. Boxplot shows central line at median, first and third quartiles bounded within the white box, and outliers as points.

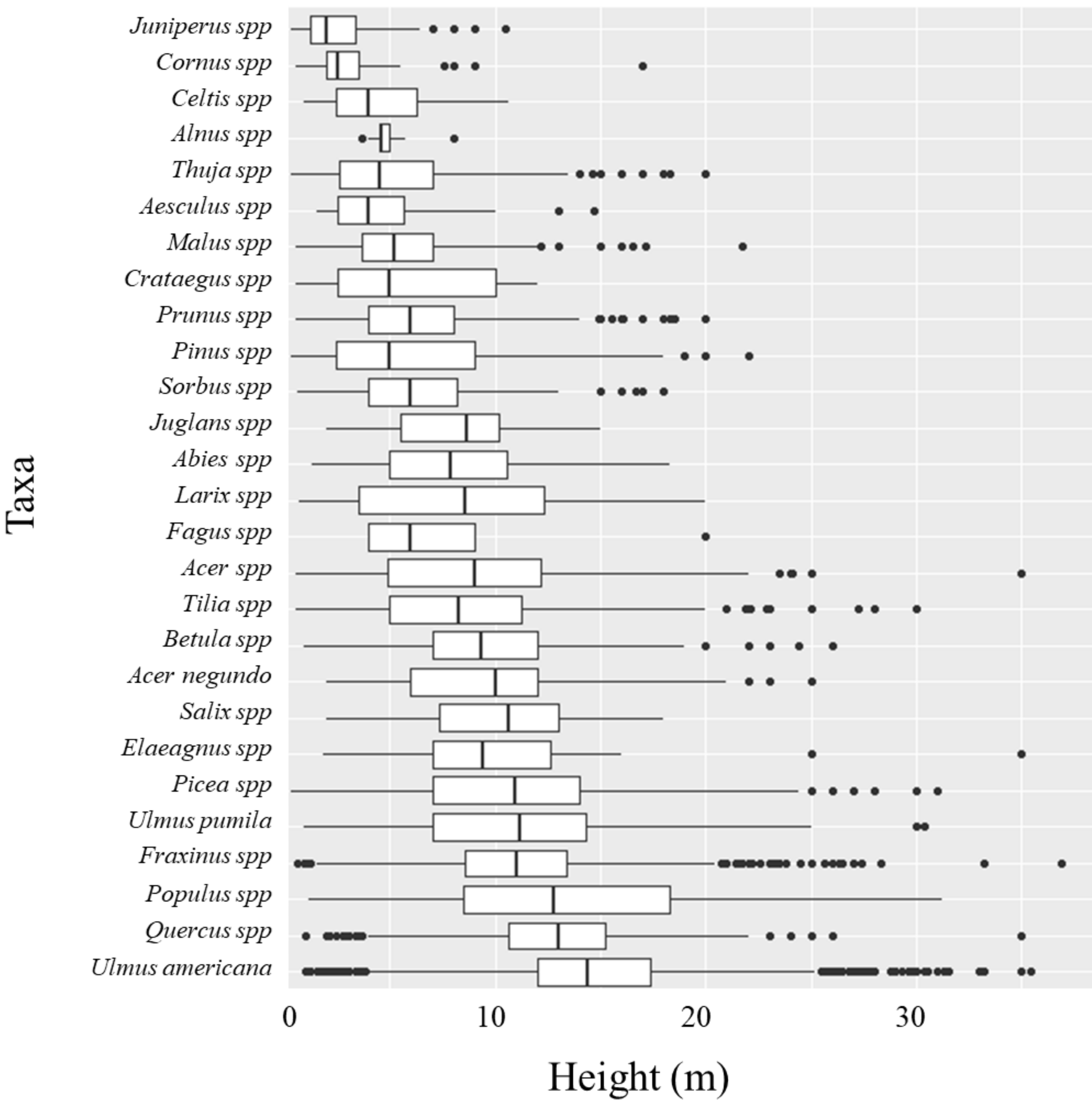


Figure 3.4. Height ranges for tree genera recorded in the survey. American elm, Siberian elm, and Manitoba maple are broken out further. Boxplot shows central line at median, first and third quartiles bounded within the white box, and outliers as points.

3.1.3.1 Diversity and Health Indicators

All diversity and health indicators are summarized in Appendix C.

Species Richness

Species richness within city neighborhoods varied from 6 to 25 species (mean =15 ± 4 species). The most widely distributed tree species were spruce species (75 neighborhoods), American linden (74 neighborhoods), cedar species (71 neighborhoods), apple species (70 neighborhoods), and Manitoba maple (69 neighborhoods).).

Simpson's Reciprocal Index of Diversity (1/D) and Evenness

Simpson's 1/D averaged 5.76 for all neighbourhoods, with neighborhoods ranging from values of 2.24 to 10.93. Simpson's evenness averaged 0.03 for all neighbourhoods, with neighborhoods ranging from values of <0.01 to 0.15.

Berger-Parker Dominance

Berger-Parker dominance averaged 0.23 for all neighbourhoods in the city, with American elm being the most common species. Dominance values in neighbourhoods varied from a low of 0.13 in one neighbourhood where tree species were quite evenly represented to 0.64 in one neighbourhood where American elm was the dominant species, making up 64% of the neighborhood. The most dominant trees by neighborhood were American elm (29% of neighborhoods), Ash species (28%), Eastern white cedar (16%), Spruce species (12%), and Linden species (5%).

10-20-30 Values

Of all neighborhoods surveyed, none met Santamour's benchmark of 10% or less for the maximum value for a single species, and only 6 had 20% or less as the maximum value for a single genus. However, 31 of the 75 neighborhoods surveyed had 30% or less as the maximum value for a single family. (See Figure 3.5)

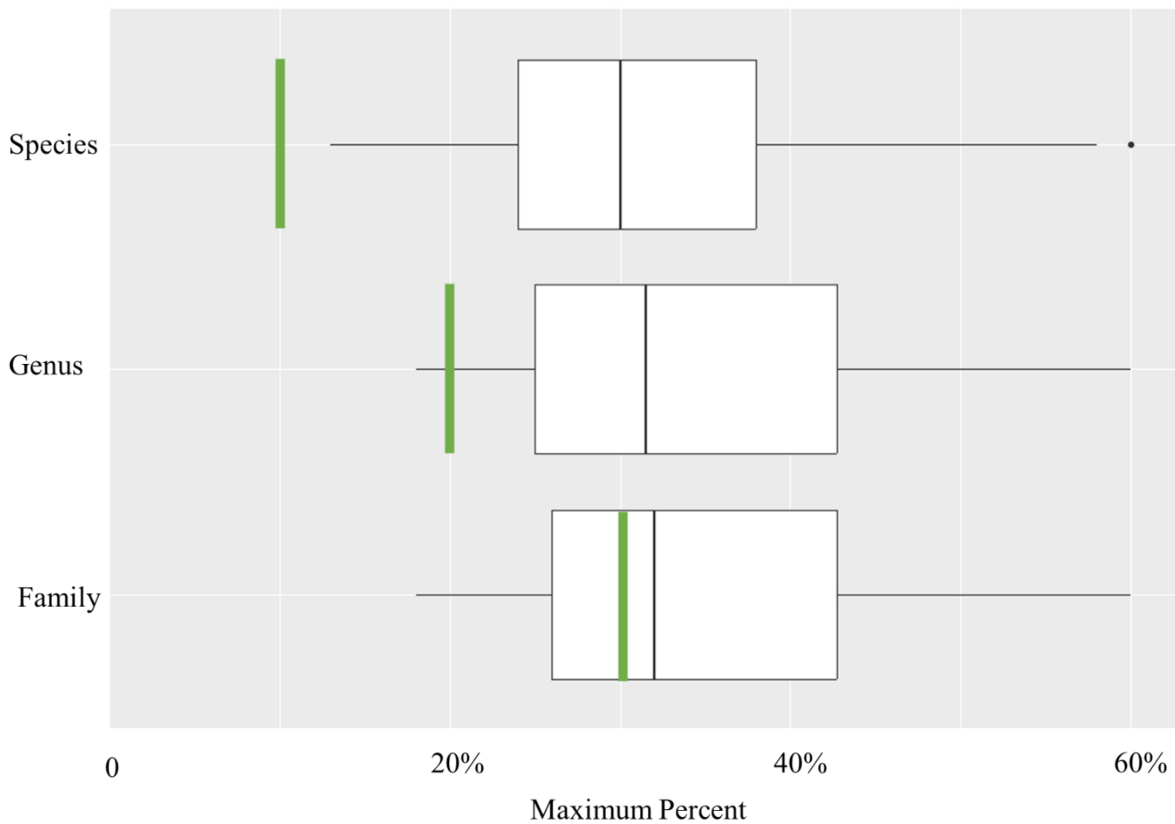


Figure 3.5 Percent range for maximum species, genus, and family values across surveyed neighborhoods. Boxplot shows central line at median, first and third quartiles bounded within the white box, and outliers as points. Green lines are shown at Santamour's target values.

Tree Dieback

Across all neighbourhoods, 69% of trees were ranked as category 1, 24% in category 2, 5% in category 3 and 2% in category 4. When converted to dieback percentage, mean dieback for all neighborhoods was 8.7%, with values ranging from mean dieback of 0.7% to 23.5%.

3.1.4 Comparing Public and Private Forest

Of trees surveyed, 14,628 trees were on private property and 9,921 on public property (largely boulevards, or the public greenspace along roadways). An additional 446 addresses were surveyed that had no trees on either public or private property. Of the addresses surveyed, 67%

had at least one private tree, 28% had no private trees but at least one public tree, and 5% had no trees.

3.1.4.1 Diversity Indicators

Wilcoxon signed rank analysis indicated that private holdings were more diverse across all measures (Table 3.3). See also heat maps for visual reference, Figures 3.6-3.10.

Table 3.3 *V* and *p* values for Wilcoxon Signed Rank Tests comparing public and private tree holdings in surveyed neighborhoods.

Indicator	**Wilcoxon's V**	**p**
Simpson's 1/D	20	<.001
Simpson's Evenness	634	.007
Berger-Parker Dominance	2711	<.001
Mean Dieback	2582	<.001

Figure 3.6 Species richness on public and private property in Winnipeg.

Figure 3.7 Simpson 1/D on public and private property in Winnipeg.

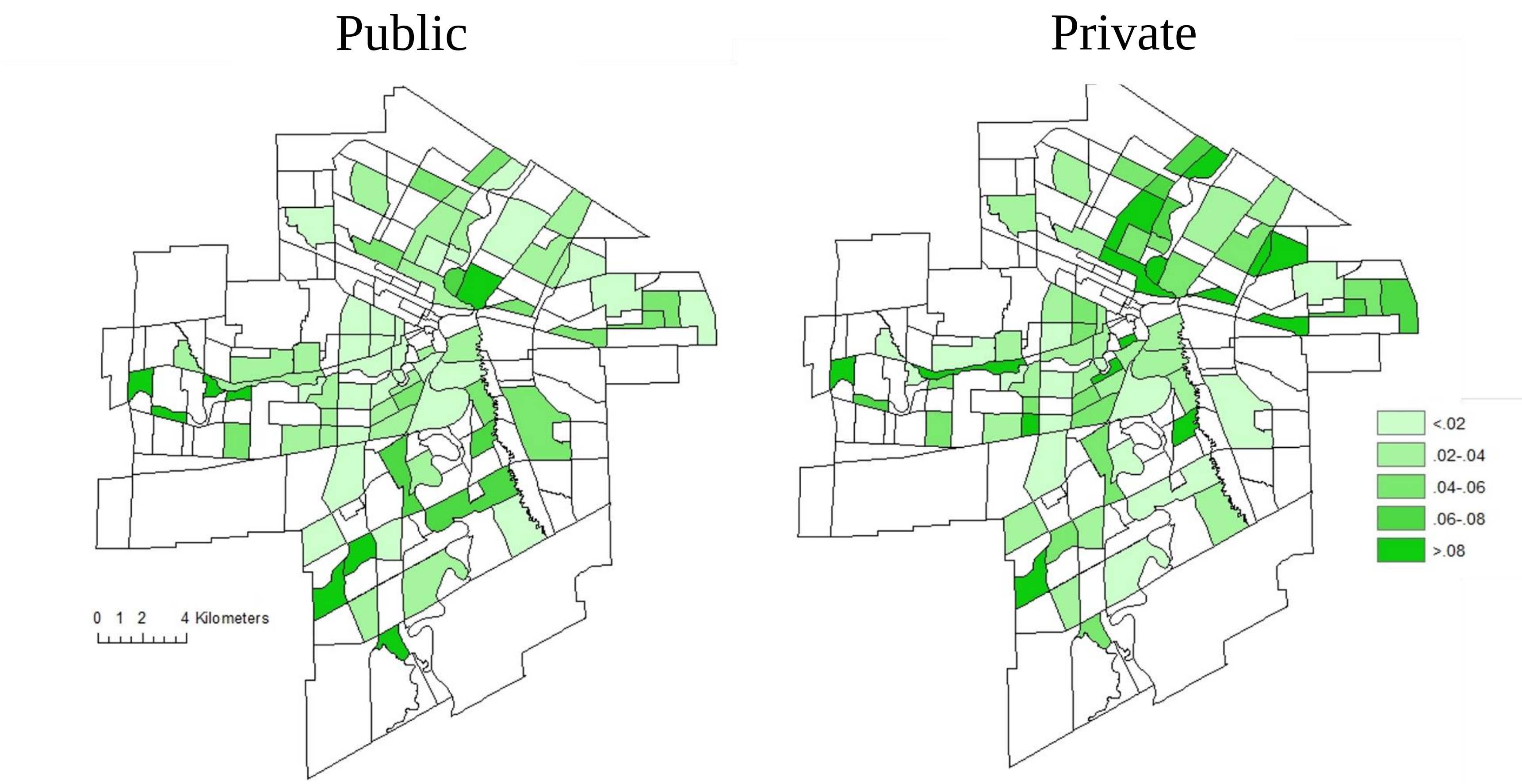


Figure 3.8 Simpson's measure of evenness on public and private property in Winnipeg.

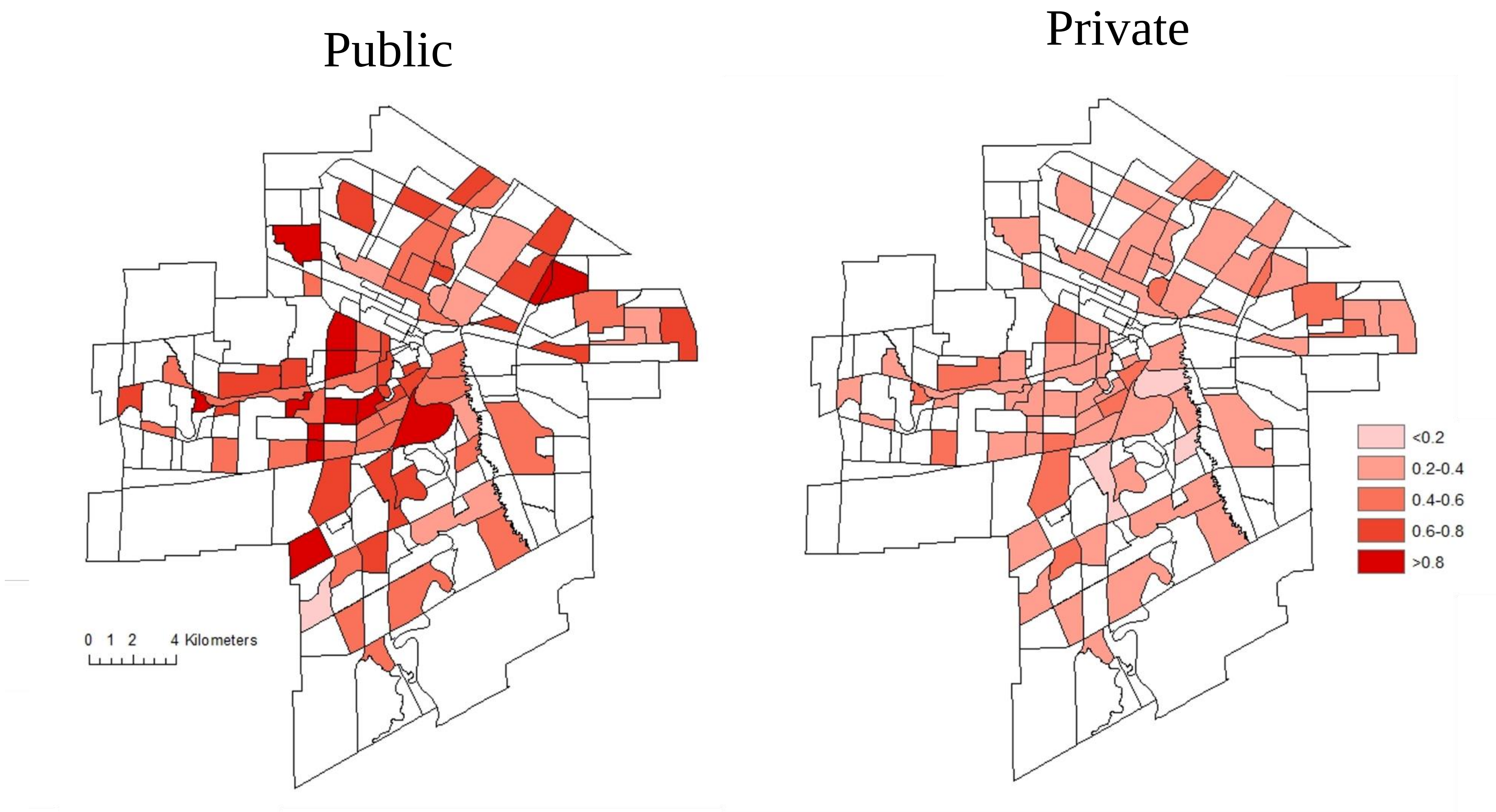


Figure 3.9 Berger-Parker dominance on public and private property in Winnipeg.

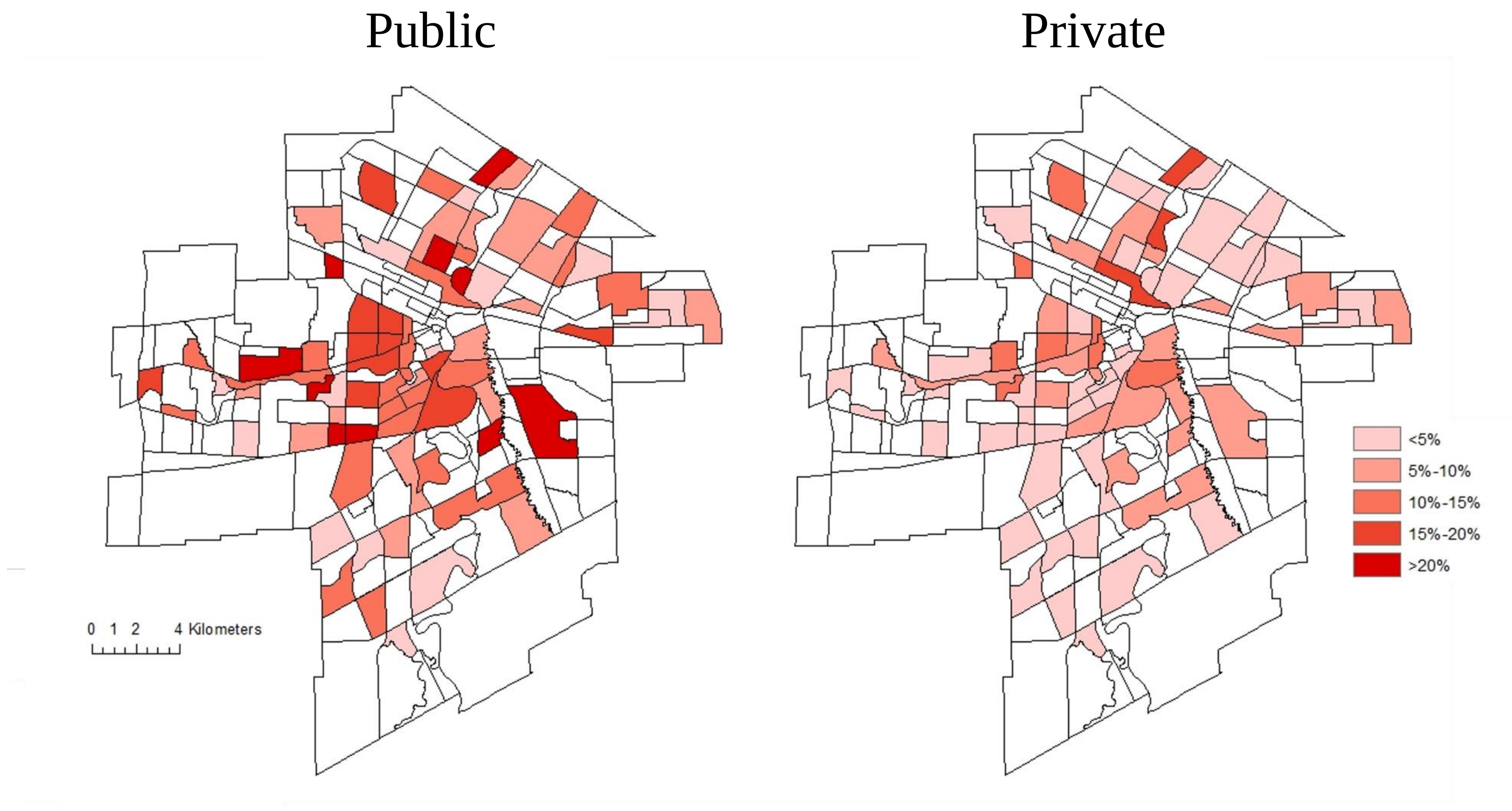


Figure 3.10 Mean dieback on public and private property in Winnipeg.

3.1.4.2 Private Trees and Home Characteristics

For this analysis, only single- and two-family residences were examined. Although regression was not possible due to the heteroscedasticity of the dataset, relationships between home characteristics and number of trees were plotted for visual assessment (see Figure 3.11 and 3.12). Homes with higher assessed values, more land, and more living area were all correlated with higher tree counts using Kendall's Tau statistic (see Table 3.4).

Table 3.4 Correlation coefficients for Kendall's Tau statistic used to test the relationship between home characteristics and number of trees.

Home Characteristic	Correlation Coefficient
Value ($CAD)	0.138
Land Area (m^2)	0.233
Living Area (m^2)	0.100

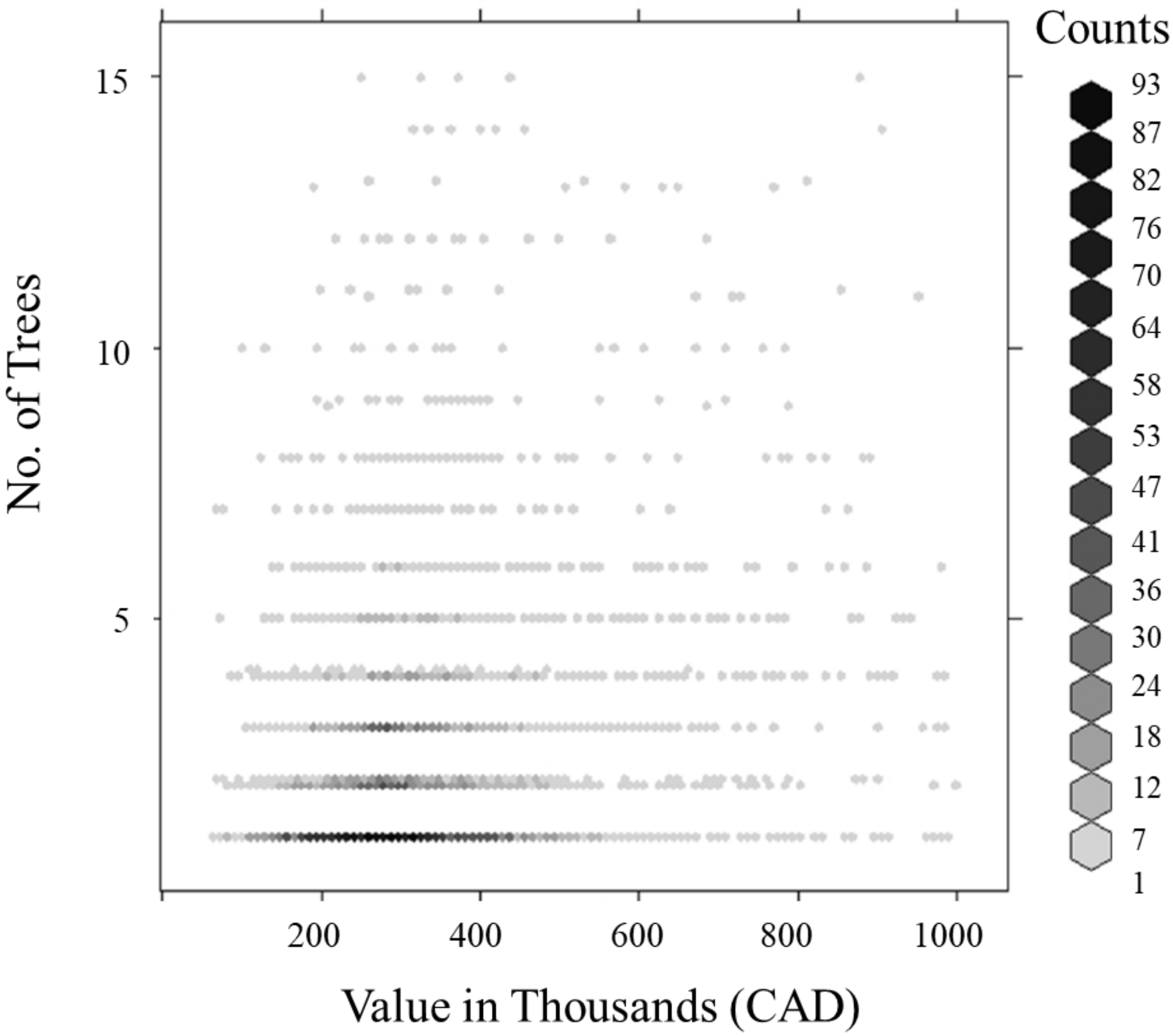


Figure 3.11 Number of trees as a function of value (CAD). Counts on right reflect values of overlapping points, combined for ease of viewing.

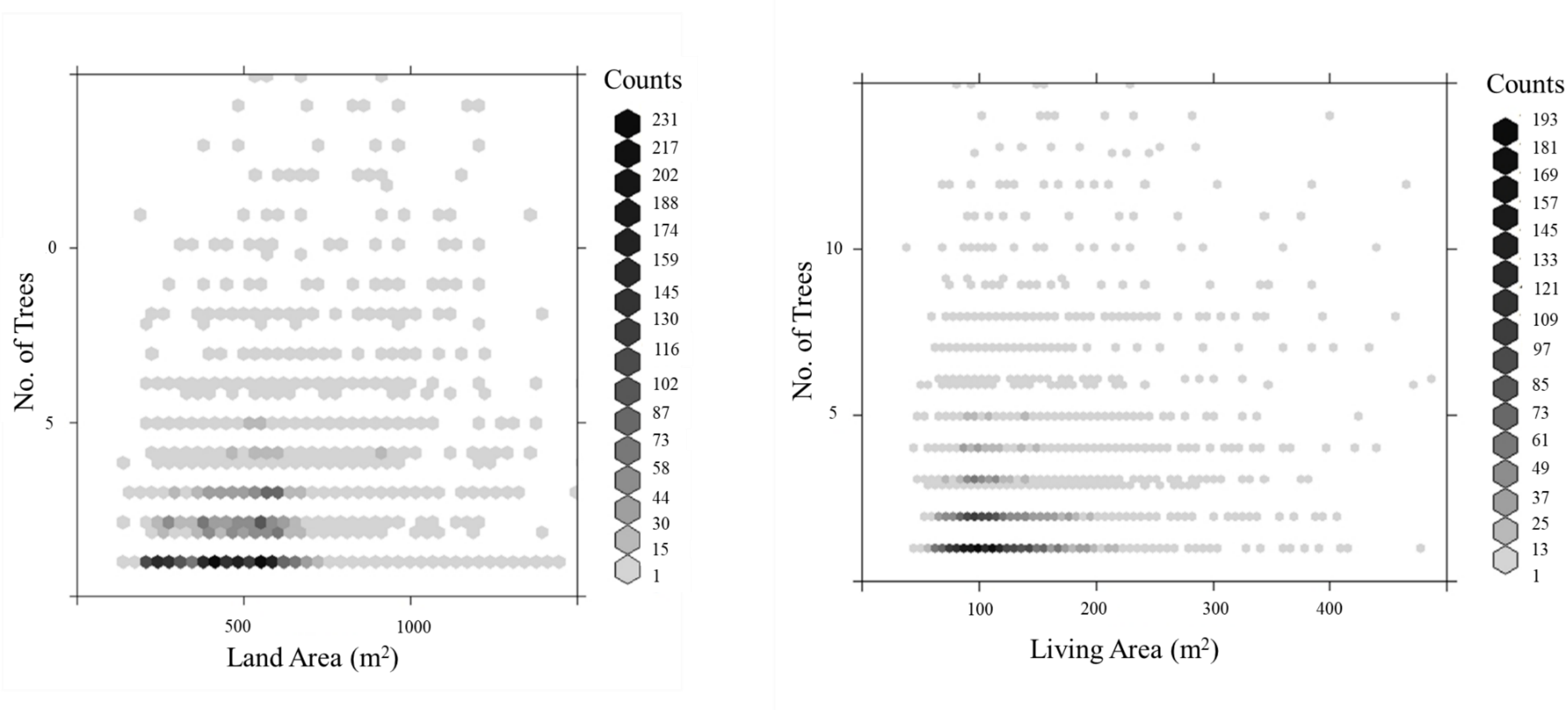


Figure 3.12 Number of trees as a function of land area and living area. Counts on right reflect values of overlapping points, combined for ease of viewing.

3.1.5 Pest Vulnerability

Table 3.5 shows the species that are at the highest risk across Winnipeg's neighborhoods, as determined by ranked taxon pest scores. Ash was the most threatened taxon in Winnipeg and the only taxa which was threatened in 10 or more neighbourhoods.

Table 3.5 Number of neighborhoods for which each taxon was rated as the first, second, or third highest risk. "Other *Acer spp*" refers to any *Acer* observations that were not recorded as *Acer negundo*.

Taxon	No. 1 risk	No. 2 risk	No. 3 risk	Total Score
Fraxinus spp	35	14	8	57
Ulmus americana	25	12	6	43
Other *Acer spp*	3	15	24	42
Picea spp	1	16	17	34
Thuja spp	2	7	6	15
Quercus spp	5	2	4	11
Tilia spp	1	2	4	7
Acer negundo	1	2	0	3
Prunus spp	1	2	0	3
Ulmus pumila	0	2	2	4
Malus spp	0	0	3	3
Populus spp	0	1	1	2
Salix spp	1	0	0	1

When the number of neighbourhoods were analyzed to determine which pests would be most widespread in the city, Asian Longhorned Beetle had the potential to have the highest impact on city neighborhoods (Table 3.6). It should be noted that this pest has not yet been found in Winnipeg; its closest current location is in southern Minnesota.

Table 3.6 Number of neighborhoods in which potential future and current pests and disease are (or have the potential to be) the most wide spread. Percentages are based on the 70th, 80th, and 90th percentiles of percent urban forest affected. Only pests deemed to have "severe" impacts were ranked.

**pest is rare or not yet observed in the City of Winnipeg*

Pest Common Name	Pest Latin Name	24-33%	34-46%	>46%	Total
Asian longhorned beetle*	*Anoplophora glabripennis*	8	18	46	72
Armillaria root rot*	*Armillaria spp*	24	5	1	30
Dutch elm disease	*Ophiostoma ulmi*	7	11	4	22
Elm bark beetles	*Hylurgopinus rufipes*	7	11	4	22
Anthracnose	*Apiognomia spp*	9	8	4	21
Cottony ash psyllid*	*Psyllopsis discrepans*	9	8	4	21
Emerald ash borer*	*Agrillus planipennis*	9	8	4	21
Two lined chestnut borer	*Agrilus bilineatus*	0	0	1	1
Bronze birch borer	*Agrilus anxius*	0	0	0	0

Table 3.7 shows how ranked lists of indicators compare to a ranked list of neighborhood pest scores using Kendall's Tau Statistic. The most closely aligned indicator was Berger-Parker Dominance.

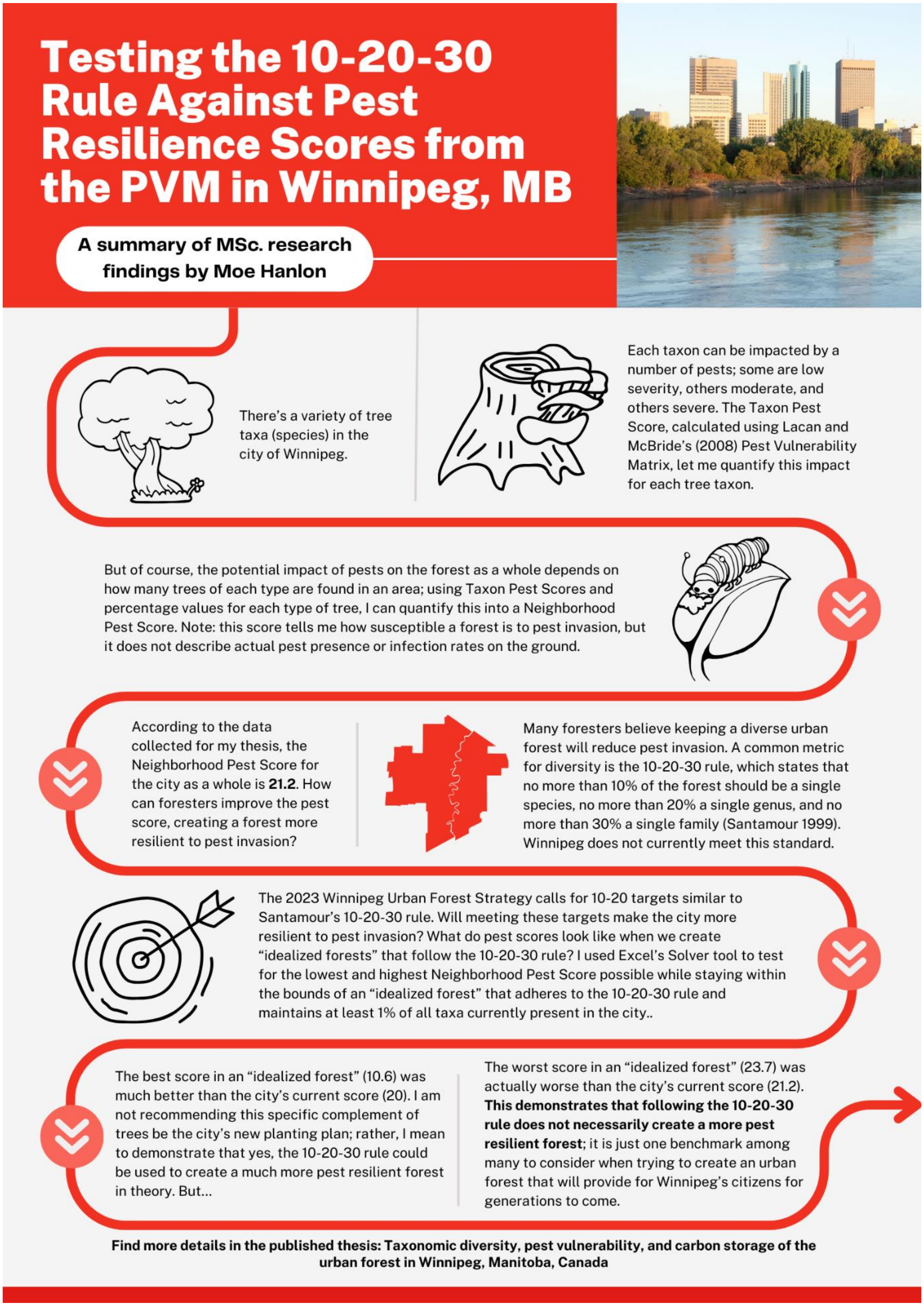
Table 3.7 Tau values for neighborhood pest score rankings vs. diversity indicator rankings. Tau values range from -1 to 1, with values of 1 indicating ranked lists are the same and values of -1 indicated ranked lists are reversed

Indicator	Tau	p
Species Richness	0.1513	0.0641
Berger-Parker Dominance	0.3248	<.001
Simpson's 1/D	0.1654	0.0358
10-20-30 Adherence	0.3038	0.001
Health	0.2614	<.001

Tree complements for the citywide dataset as well as the idealize urban forests under 10-20-30 rule constraints are shown in Table 3.8. The pest score for the entire city was 21.2; the minimum value possible under 10-20-30 constraints is nearly half the actual city's pest score (10.6), but the maximum possible pest score is higher than the city's current performance (23.7). I was also able to create a poster for outreach explaining some results from the PVM section of this thesis, Figure 3.13 below.

Table 3.8 Maximum and minimum neighborhood pest scores as well as tree complements for idealized forests under 10-20-30 conditions.

	City of Winnipeg	10-20-30 idealized forest with minimum pest score	10-20-30 idealized forest with maximum pest score
Juniper (*Juniperus* spp.)	1%	10%	1%
Cedar (*Thuja* spp.)	13%	10%	1%
Pine (*Pinus* spp.)	1%	1%	1%
Larch (*Larix* spp.)	0%	10%	1%
Fir (*Abies* spp.)	0%	10%	1%
Spruce (*Picea* spp.)	13%	1%	1%
Box elder (*Acer negundo*)	6%	1%	1%
Other maples (*Acer* spp.)	7%	1%	10%
Ohio Buckeye (*Aesculus glabra*)	0%	10%	1%
Ash (*Fraxinus* spp.)	14%	1%	10%
Dogwood (*Cornus* spp.)	0%	1%	1%
Catalpa (*Catalpa* spp.)	0%	10%	1%
Walnut (*Juglans* spp.)	0%	1%	1%
Mountain ash (*Sorbus* spp.)	1%	2%	1%
Oak (*Quercus* spp.)	5%	1%	10%
Linden (*Tilia* spp.)	6%	10%	1%
Birch (*Betula* spp.)	2%	1%	10%
Alder (*Alnus* spp.)	0%	1%	2%
Willow (*Salix* spp.)	1%	1%	10%
Poplar, Cottonwood, Aspen (*Populus* spp.)	1%	1%	10%
American Elm (*Ulmus americana*)	20%	1%	10%
Siberian Elm (*Ulmus pumila*)	3%	1%	1%
Hackberry (*Celtis* spp.)	0%	1%	1%
Apple, Crabapple (*Malus* spp.)	3%	1%	10%
Cherry and Plum (*Prunus* spp.)	4%	1%	1%
Hawthorn (*Crataegus* spp.)	0%	1%	1%
Russian Olive (*Elaeagnus* spp.)	0%	10%	1%



375 Figure 3.13. A poster to condense and simplify findings from Section 3.1.5.

3.4 References Chapter 3

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Chapter 4

Estimating carbon storage in Winnipeg's urban forest

4.1 Introduction

Cities across the world are turning to their urban forests as a way to help balance municipal greenhouse gas budgets (Zhao et al., 2023). As such, it is increasingly important that urban forest managers be able to provide estimates of carbon storage and sequestration by urban trees. However, reliable estimates of urban tree growth and carbon storage remain elusive due to variability in species-specific growth patterns, environmental and climatic conditions, and differences between patterns of growth in urban and non-urban trees.

As discussed in Chapter 1, any variable that affects growth also affects carbon sequestration in trees, and environmental conditions have a pronounced impact on tree growth. All trees experience differential growth based on access to sunlight, water, soil nutrients, and rooting space, among other factors (Boukili et al., 2017). In urban areas, trees are often subject to nutrient-deficient, compacted soils, poor drainage, increased temperatures, and low air quality, all of which can negatively impact growth (Mullaney et al., 2015). However, some studies show that urban trees grow larger than their natural counterparts, likely due to wide spacing between trees and enhanced access to sunlight (McHale et al., 2009). Moreover, urban trees are often selected for desirable traits like canopy architecture, color, flowering, etc. (see Figure 4.1), which may lead to significant divergence from the growth patterns of the same species in a wild population (Santamour, 1999).

Figure 4.1 Left: *Thuja* species have been bred for a variety of architectures including squat, bush-like forms (front center) and columnar forms (against home). Right: Columnar *Populus* species are frequently used as hedges in the city of Winnipeg (Photo credit: Google Maps Streetview).

The City of Winnipeg Urban Forest Strategy (City of Winnipeg, 2023) sets targets to reduce tree loss and plan for tree replacement. It also includes key actions for increasing carbon storage and sequestration in the urban forest. The Strategy calls for shortening the pruning cycle to seven years for street trees, improving health outcomes, and increasing tree longevity. The strategy also includes disease tracking and management measures, including prioritizing rapid removal of diseased American elms that are considered “brood” trees, a measure intended to prolong the lifespan of the city’s elm canopy and spread out tree replacements over a longer period of time (City of Winnipeg, 2023). In an effort to increase the extent of the tree canopy, the plan also calls for planting 17,000 new trees per year, as well as replacing lost trees at a 1-to-1 ratio (City of Winnipeg, 2023).

The City of Winnipeg has used i-Tree Eco software (one of several i-Tree tools available) to estimate carbon storage in Winnipeg’s urban forest (City of Winnipeg, 2023). i-Tree is an accessible toolset that allows urban foresters to measure a variety of traits using free software developed by the United States Department of Agriculture (USDA) Forest Service (i-Tree Tools, 2017). Using i-Tree Eco, the City of Winnipeg estimated current carbon storage in Winnipeg's boulevard trees at 509,000 tonnes (t).

i-Tree Eco allows users to input a sample of city trees with diameter and height measurements, as well as many local variables, including weather, pollution levels, and land use (Maco & Nowak, 2011). The program can return a range of results, including species composition and distribution, the monetary value of trees, and carbon storage and sequestration data (i-Tree Tools, 2017). Although it was first derived for use in U.S. cities, the architects of the software claim that it can now accommodate both Canadian and international cities (Maco & Nowak, 2011). In addition to Winnipeg, i-Tree Eco has been used to quantify urban forest structure in Edmonton (City of Edmonton, 2019) Toronto (Toronto Parks, Forestry, and Recreation, 2008), and Halifax, as well as some smaller Canadian municipalities (Foster & Duinker, 2017).

Given the variability in tree growth and subsequent carbon storage, it is imperative that cities be able to both estimate carbon storage in trees and calculate some measure of statistical confidence in the accuracy of those measurements. Allometric studies that drive most estimates of carbon storage are often published without accompanying estimates of statistical variability, such as standard errors or confidence intervals. This challenge is compounded when programs like i-Tree Eco combine the predictive equations from these studies into new equations for the program, and reports of confidence published in the original studies are often lost when equations are combined; for example, i-Tree Eco describes a method by which a main equation is sometimes supplemented with another for very old trees, and the whole equation is smoothed for easy use (Nowak 2020).

In this chapter, I sought to address the following research objectives:

1) Explore changes in carbon storage given the city of Winnipeg's plan for 1:1 replacement of lost trees.
2) Use published regression parameters and simulation techniques to derive measures of statistical variability in in carbon storage in Winnipeg's urban forest; and
3) Estimate the mass of carbon stored in surveyed neighbourhoods in Winnipeg's urban forest and describe the boundaries around these estimates.

4.2 Methods

The survey data used for exploring tree diversity was also used to explore carbon storage in Winnipeg's urban forest in this chapter. To do so, I used height and DBH data from individual trees. Trees on public boulevards were measured using a DBH tape for diameter and a clinometer for height. Height and DBH measurements for private trees were estimated based on adjacent public tree measurements. According to the research coordinator for the first fifteen years of the survey, estimates for private trees were assessed to be within 10 to 15% of actual values.

I began by finding biomass equations reported in Understanding i-Tree: A Summary of Programs and Methods (Nowak 2020) and calculating biomass and carbon values for tree species surveyed using these equations. These i-Tree equations can be found in Appendix F, Table F1. For species for which no biomass equations were available, I followed the protocol set out in Nowak (2020), calculating the mean of all values for biomass in a corresponding genus or the next closest taxon (Nowak 2020). These values can be found in Appendix F, Table F3. It is worth noting that I did not use reported i-Tree Eco carbon storage or calculate an approximation thereof. i-Tree Eco, as discussed in section 2.2.6, uses a variety of measurements like crown light exposure to adjust the baseline carbon values derived from biomass equations. I did not have these measurements for many trees in the survey, and so chose to focus on values derived from the i-Tree biomass equations, which require only DBH and, sometimes, height.

These i-Tree carbon calculations allowed me to explore my first research question, comparing storage of each species of tree at its average DBH to the carbon storage of surveyed elm and ash at their average DBH values.

I then moved on to my second research question, using methodology described by Wayson (2015) to create pseudo-data for these i-Tree biomass equations (with some supplements, see Appendix F, Table F2) which do not have original datasets easily available. With these pseudo-data, I was able to create prediction intervals to describe uncertainty in the original equation.

I then used the Wayson et al. (2015) biomass and carbon calculations and the City of Winnipeg Open Data Portal to estimate potential carbon limits by neighborhood and map carbon storage for my final question. I used predicted values from Wayson et al. for the tree species surveyed and used i-Tree biomass calculations for carbon values for the remaining species.

4.2.1 Carbon Replacement Values

I began by comparing average carbon storage (as determined using i-Tree equations) between tree species to determine the Carbon Replacement Value for Winnipeg's two most threatened species, American elm and green ash. Carbon Replacement Value was defined as the number of trees of a given species (at mean surveyed DBH for that species) needed to equal the carbon storage of an American elm or green ash at the average surveyed DBH for American elm or green ash, respectively (eq. 1). For this calculation, i-Tree biomass equations were used to align with carbon calculations used by the City of Winnipeg.

$$(eq. 1)\ \text{Elm Carbon Replacement Value}_{\text{Species X}} = \frac{\text{Carbon Storage Elm}_{\text{Average DBH Elm}}}{\text{Carbon Storage Species X}_{\text{Average DBH Species X}}}$$

Though it is easy to see any given tree that is lost and removed should be replaced by one newly planted tree, the replacement value can be used to give city foresters an idea of the number of trees needed to replace the carbon stored in Winnipeg's lost American elms and green ash, which are some of the city's largest and most numerous trees.

4.2.2 Developing Prediction Intervals

Because i-Tree carbon values are reported by the City of Winnipeg Urban Forestry Branch, I wanted to develop a method to describe potential error in the original biomass equations used by i-Tree. With assistance from Dr. Andrew Park, I was able to develop prediction intervals for biomass and carbon storage in individual trees based on methods used by Wayson et al (2015). Wayson et al. (2015) calculated the expected variability in regression equation parameters (e.g. standard errors of estimate) using coefficients of variation (r^2) and sample size, checking against the original dataset to verify their method. These authors developed a method to generate pseudo-data to emulate the error structure of the original allometric biomass equations. I followed Wayson et al.'s (2015) procedure, with some methodological adjustments described below (Figure 4.2), to estimate biomass and carbon storage in Winnipeg's dominant urban trees.

To determine which dominant tree species to use for the Wayson et al. (2015) procedure, I estimated carbon storage using i-Tree biomass equations for all trees surveyed (see Appendix F). I chose to analyze the eight tree taxa that each accounted for ≥1.5% of Winnipeg's urban-forest carbon storage in my survey using i-Tree biomass equations. Taken together, these eight taxa accounted for 94% of the total carbon in trees surveyed (see Table 4.1). I also included Eastern White Cedar (*Thuja occidentalis* L.), which was highly represented by stem count, though it only made up 0.6% of the i-Tree carbon storage estimate for the city.

Table 4.1 Dominant trees based on percent carbon in trees surveyed for this study, as calculated in using i-Tree biomass equations (Nowak 2020).

Common Name	**Latin Name**	**Percent Carbon of Trees Surveyed**
American Elm	*Ulmus americana*	57.6%
Ash	*Fraxinus* spp	8.8%
Oak	*Quercus* spp	6.7%
Spruce	*Picea* spp	6.4%
Boxelder	*Acer negundo*	6.0%
Siberian Elm	*Ulmus pumila*	3.1%
Linden	*Tilia* spp	2.5%
Birch	*Betula* spp	1.5%
Willow	*Salix* spp	1.5%
Eastern White Cedar	*Thuja* spp	0.6%

The completion of this analysis required a minimum set of parameters from the original allometric equations from which to derive the pseudo-data. The Wayson et al. (2015) method required r^2, original sample size, and the range of diameters at breast height (dbh) that were represented in the original data.

Correspondence with i-Tree developer David Nowak indicated that i-Tree did not have error estimates for the allometric equations it currently uses. Nor did they have access to the original datasets from which i-Tree biomass equations were derived. Most tree species in i-Tree have biomass equations derived from several previous studies. As such, I chose to use r^2 values from the original studies. In the event that several estimates were published, I used the mean of the published r^2 values. I also used the combined DBH range of these original equations. For example, if one study had a DBH range of 3–60 cm for a particular species, and another had a range of 45–120 cm, I would use a DBH range of 3–120 cm for that species.

Where possible, the forms of species-specific equations can be found in the i-Tree manual (Nowak 2020, Appendix F). Because some i-Tree equations used height in addition to DBH, I was unable to use the Wayson et al. (2015) method to estimate error for these equations. If i-Tree lacked a biomass equation for any species or the equation included height, I used equations from the literature referenced by i-Tree, or if none of those equations were usable, then I used USDA equations from the urban tree database (McPherson et al. 2016). See Appendix F for all equations used in this paper.

I first generated 10,000 random points from a uniform distribution within the dbh range of the published dataset in question. I then made 996 sets of "fuzzed" data, creating biomass values estimates that were larger or smaller than the values calculated from the original equation parameters. This was done by varying a set of numbers from 1 to 200 in increments of 0.2 (996 values), then creating a matrix where each of these values was multiplied by the 10,000 random dbh values within the range of the original dataset and a value from a 10,000 number random uniform distribution (0 ±1). The values in this matrix of "fuzz factors," scaled for different levels of variability, were then added to the 10,000 random biomass values, creating 996 "fuzzed" pseudo-datasets.

For each pseudo-dataset, I then calculated the r^2 value using the sum of squared residuals (eq. 2-4) as described in Wayson et. al (2015):

(eq. 2) $$R^2 = 1 - \frac{\mathrm{SS}_{\mathrm{residuals}}}{\mathrm{SS}_{\mathrm{total}}}$$

where:

(eq. 3) $$\mathrm{SS}_{\mathrm{residuals}} = \sum_i (\mathrm{biomass}_{\mathrm{pseudo}_i} - \mathrm{biomass}_{\mathrm{orig}_i})^2$$

and:

(eq. 4) $$\mathrm{SS}_{\mathrm{total}} = \sum_i (\mathrm{biomass}_{\mathrm{pseudo}_i} - \overline{\mathrm{biomass}_{\mathrm{pseudo}}})^2$$

I then queried the resulting collection of r^2 values to find the pseudo-dataset whose r^2 most closely matched that of the original (published) biomass equation. Wayson et al (2015) did this to identify the pseudo-data that came closest to replicating the variance structure of the original data on which the allometric equations were based.

From the best pseudo-dataset, I repeatedly drew samples of 100 diameters and biomass, and compiled 1000 pseudo-datasets using a uniform distribution. For each of these pseudo-datasets, I calculated regression coefficients using the form of the original i-Tree/other biomass equations. These analyses were done using the pred.fit procedure from the R investr package.

From here, I calculated standard errors for each DBH value along the predictive equations (eq.5), as described in Crawley (2013):

(eq. 5) $$se_{\hat{y}} = \sqrt{s^2 \left[\frac{1}{n} + \frac{(x-\bar{x})^2}{SSX}\right]}\,.$$

This allowed us to create 95%, 80%, 20%, and 5% prediction intervals by multiplying the standard error values and the corresponding t-value. Prediction intervals tell the observer how likely it is that a single new observation will fall within the interval given the specific values of the independent variables. I chose to calculate prediction intervals because they are more conservative than confidence intervals and allowed me to put intervals around the biomass of a single tree with a specific DBH value, rather than the mean biomass of a group of trees with a particular biomass (the outcome of calculating confidence intervals). Finally, I used the average values of the equation parameters and prediction intervals to predict biomass values and associated errors associated with the actual species-specific data sets available for Winnipeg's urban forest. See Figure 4.2 for a summary of the methods used based on Wayson et al. (2015).

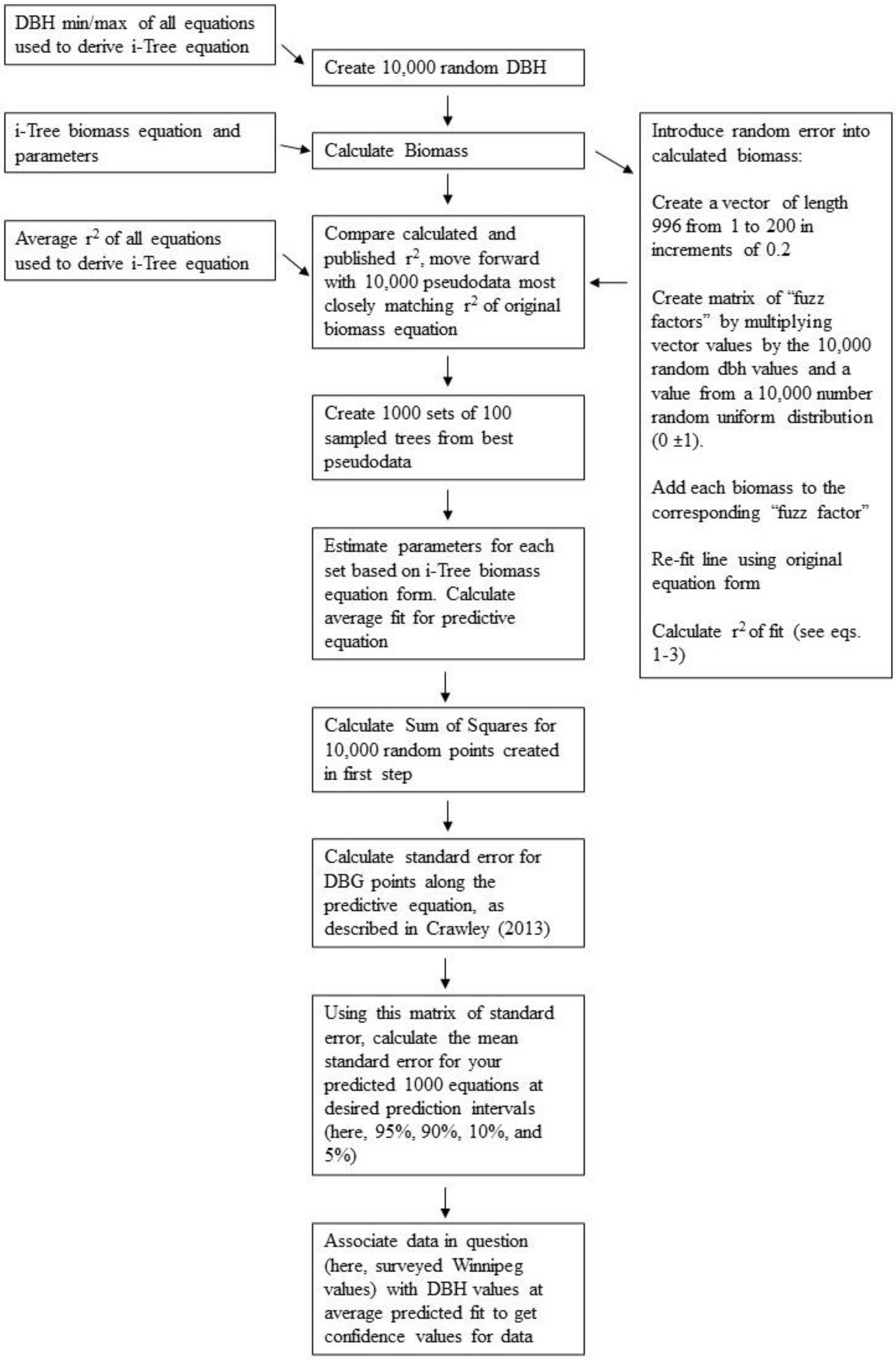


Figure 4.2 A summary of methods for predicting biomass prediction intervals, based on Wayson et al. (2015).

4.2.3 Neighborhood Carbon Storage

I also wanted to provide a carbon storage estimate together with prediction values across the surveyed residential portions of the city. To do this, I standardized our predicted average carbon storage values and carbon storage values at each prediction interval by frontage width (kg of carbon/ m frontage) for each neighborhood using the City of Winnipeg tax parcel database (City of Winnipeg 2022). I also included the i-Tree biomass values for all trees species not calculated in the Wayson et al. (2015) methods. I chose frontage meter because standardizing by area was not possible for front yards only, and property frontage was easily accessible in the City of Winnipeg database. I defined “residential” neighborhoods as those containing at least 100 residential addresses (see Appendix E for residential zoning codes included in this analysis). I then used total residential frontage per neighborhood to calculate total carbon storage at the mean and each prediction interval for all neighborhoods surveyed.

4.3 Results

Carbon values as calculated using i-Tree biomass equations (Nowak 2020) for individual trees in the survey are shown in Figure 4.3, and in aggregate in Figure 4.4. Results showed that American elm stores far more carbon than any other Winnipeg tree species, both per tree and across all surveyed trees.

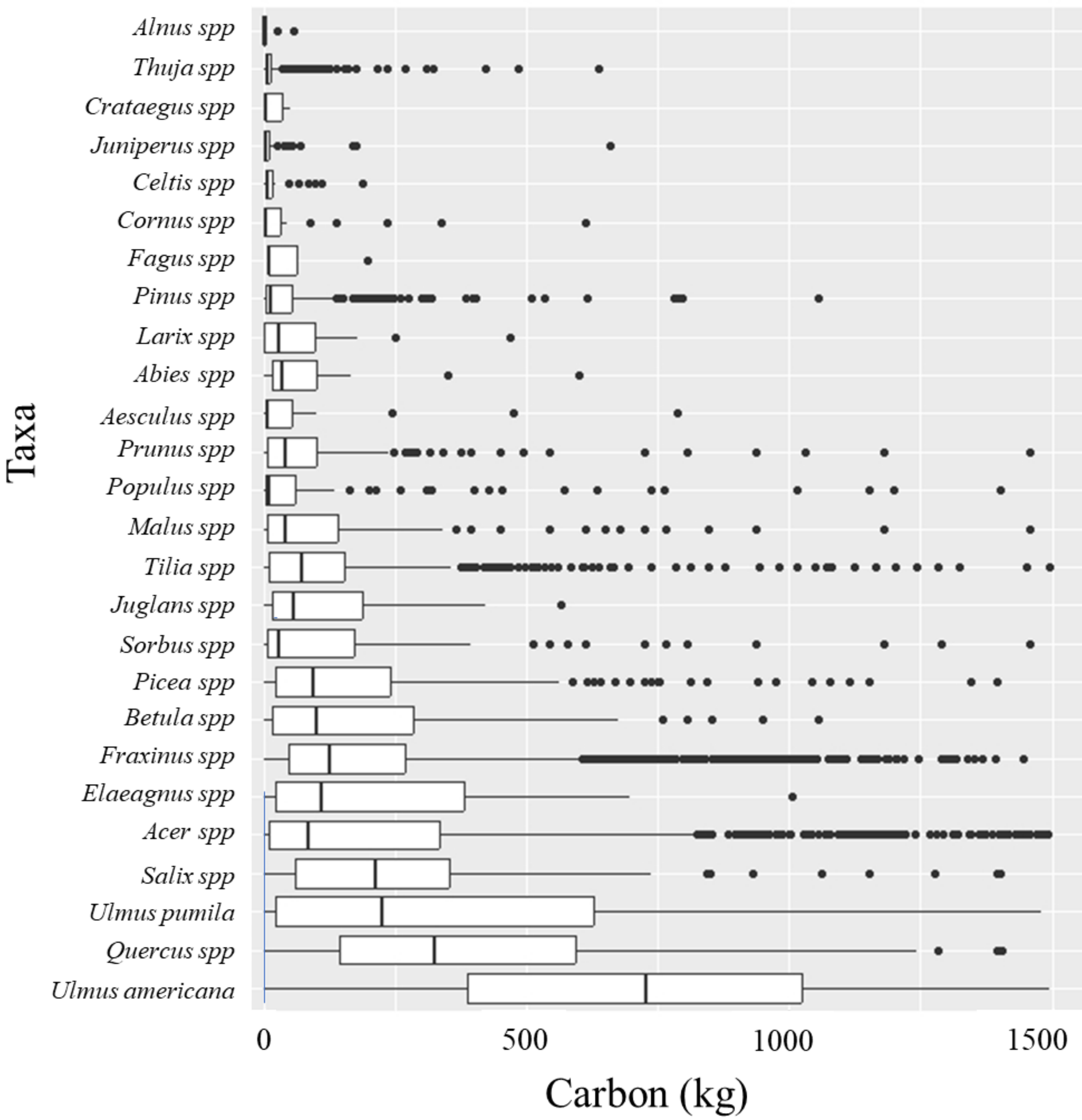


Figure 4.3 Estimated carbon storage (kg) distribution of individual trees as calculated using i-Tree biomass equations for all surveyed species.

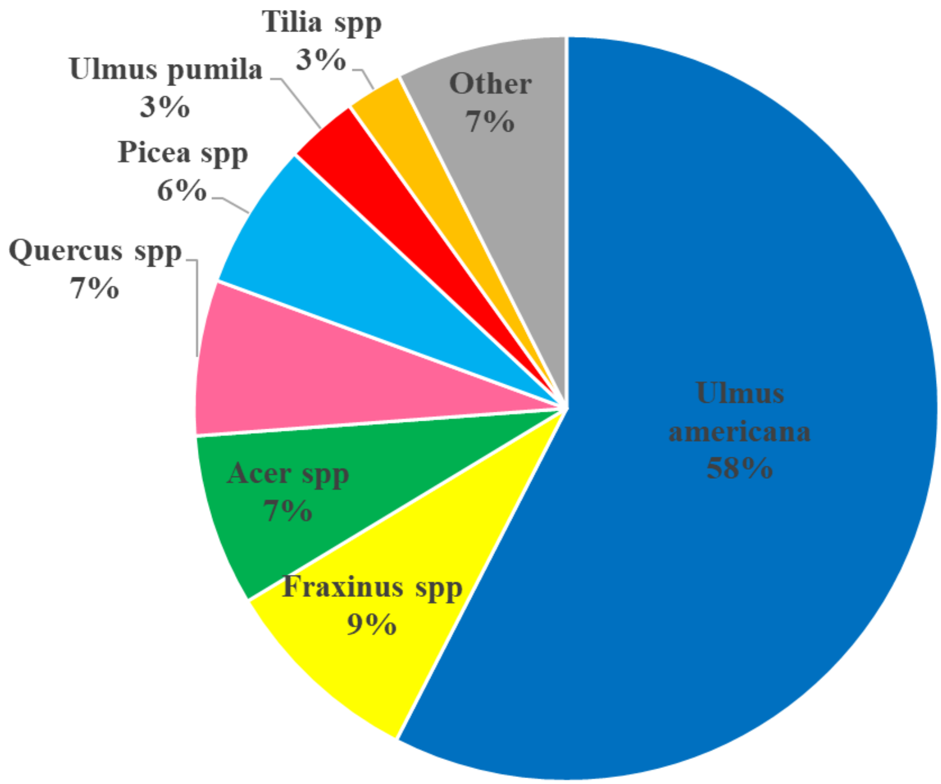


Figure 4.4 Total carbon storage per species in survey, as calculated using i-Tree biomass equations.

4.3.1 Carbon Replacement Values

Table 4.3 shows the Carbon Replacement Value of American elm and green ash for tree species observed in this study. No species has greater carbon storage capacity than American elm, and all but 7 species have only a quarter the biomass of the average Winnipeg elm.

Table 4.3. The DBH, estimated carbon storage, and replacement value for American elm and green ash for the average size of Winnipeg tree species. Species marked with a plus sign are those under imminent threat. Species with an asterisk are rarely observed at full size due to their recent introduction to Winnipeg's city forest, so Carbon Replacement Values for these species are likely much lower.

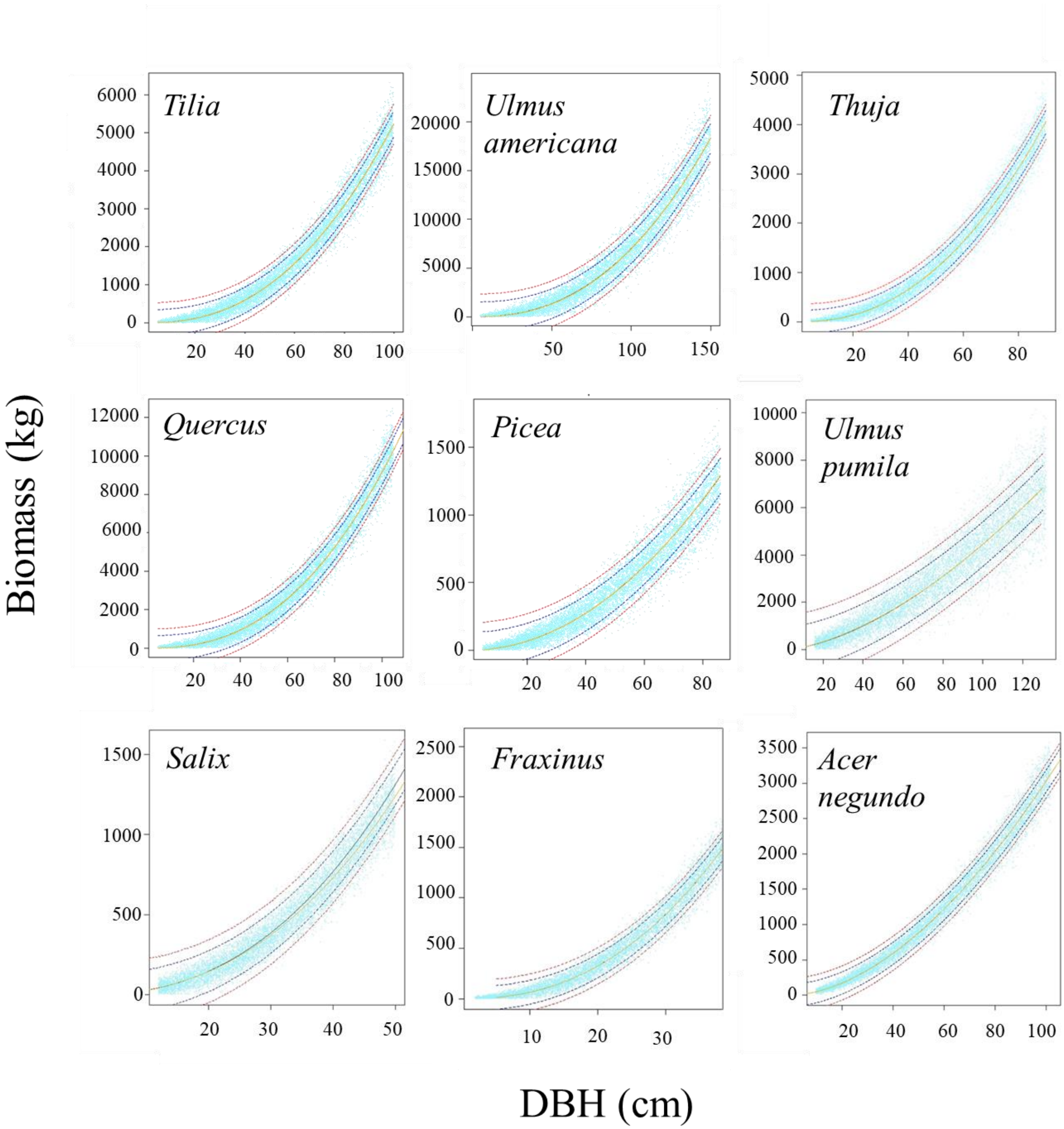
Species	Average DBH (cm)	Carbon (kg)	Elm Carbon Replacement Value	Ash Carbon Replacement Value
Ulmus americana[+]	51.9	1444.0	1.0	0.2
Salix spp	39.9	846.6	1.7	0.3
Quercus macrocarpa	35.4	690.5	2.1	0.4
*Celtis occidentalis**	37.5	499.3	2.9	0.5
Ulmus pumila	32.9	485.1	3.0	0.5
Acer saccharinum	33.2	473.1	3.1	0.5
Acer negundo	27.0	379.6	3.8	0.7
Pinus strobus	36.3	363.8	4.0	0.7
*Aesculus glabra**	37.5	362.8	4.0	0.7
Elaeagnus angustifolia	23.4	319.5	4.5	0.8
Betula papyrifera	23.4	299.9	4.8	0.8
Fraxinus pennsylvanica[+]	27.9	248.7	5.8	1.0
Acer rubrum	21.8	231.9	6.2	1.1
Juglans spp	20.5	231.2	6.2	1.1
Picea spp	24.7	219.8	6.6	1.1
Pinus resinosa	24.0	209.8	6.9	1.2
Tilia americana	22.6	156.9	9.2	1.6
Sorbus americana	17.9	154.0	9.4	1.6
Pinus banksiana	20.1	130.9	11.0	1.9
Acer saccharum	15.5	128.7	11.2	1.9
Prunus virginiana	16.4	123.0	11.7	2.0
Malus spp	16.4	122.8	11.8	2.0

4.3.2 Developing Prediction Intervals

I was able to construct prediction intervals for nine species in the study. Figure 4.5 shows these intervals and Table 4.2 reports the biomass values for the prediction intervals at the mean values of each species. Eastern white cedar has the widest prediction interval at the mean dbh, followed by American linden, while American elm, Bur oak, and spruce, which have similar widths at the same point.

Table 4.2. Carbon values as estimated using Wayson et al. (2015) method for nine Winnipeg tree species for which prediction intervals were calculated.

Species	Neighborhoods Represented	Trees Surveyed	Total Carbon Surveyed (t)	Mean Carbon/tree Surveyed (kg)	Carbon SD Surveyed (kg)
Ulmus americana	69	5602	5030	898	820
Fraxinus spp	75	3585	1785	498	595
Quercus spp	41	1169	584	500	661
Picea spp	75	3195	547	171	221
Acer negundo	69	1377	290	211	278
Ulmus pumila	59	631	280	445	417
Tilia spp	74	1486	211	142	241
Salix spp	33	157	106	676	961
Thuja spp	71	3325	49	15	46



655 **Figure 4.5** 5%-95% and 20%-80% prediction intervals (shown as red and blue dashed lines) around predicted average biomass for nine tree species. Pseudo-data points are shown in teal.

4.3.3 Neighborhood Carbon Storage

Figure 4.6 shows a histogram of estimated neighborhood carbon storage per meter of frontage. Carbon was estimated as 50% of the dry biomass, as indicated in Nowak (2020).

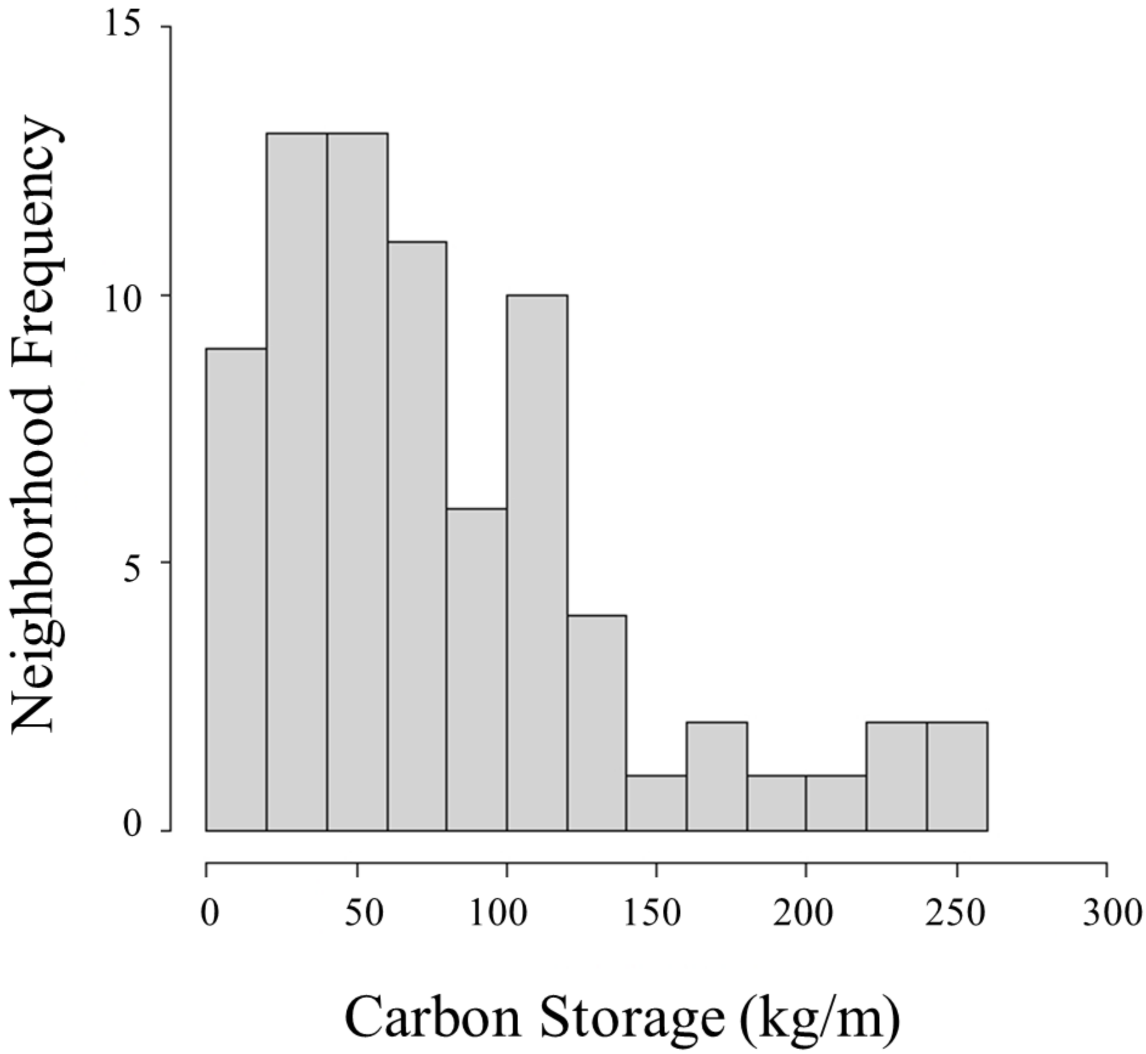


Figure 4.6 Frequency of neighborhoods with estimated standardized carbon storage (kg/frontage meter) as determined by mean values derived from the method of Wayson et al. (2015).

A heat map of carbon across surveyed neighborhoods is shown in Figure 4.7. A trend towards high carbon storage per meter frontage in the city center is evident. Total carbon storage in trees in surveyed residential areas in the city (in front yards only) came to an estimated 120,000 t, with a 95% CI of 55,800 - 225,300 t, and an 80% CI of 70,500 - 188,500 t.

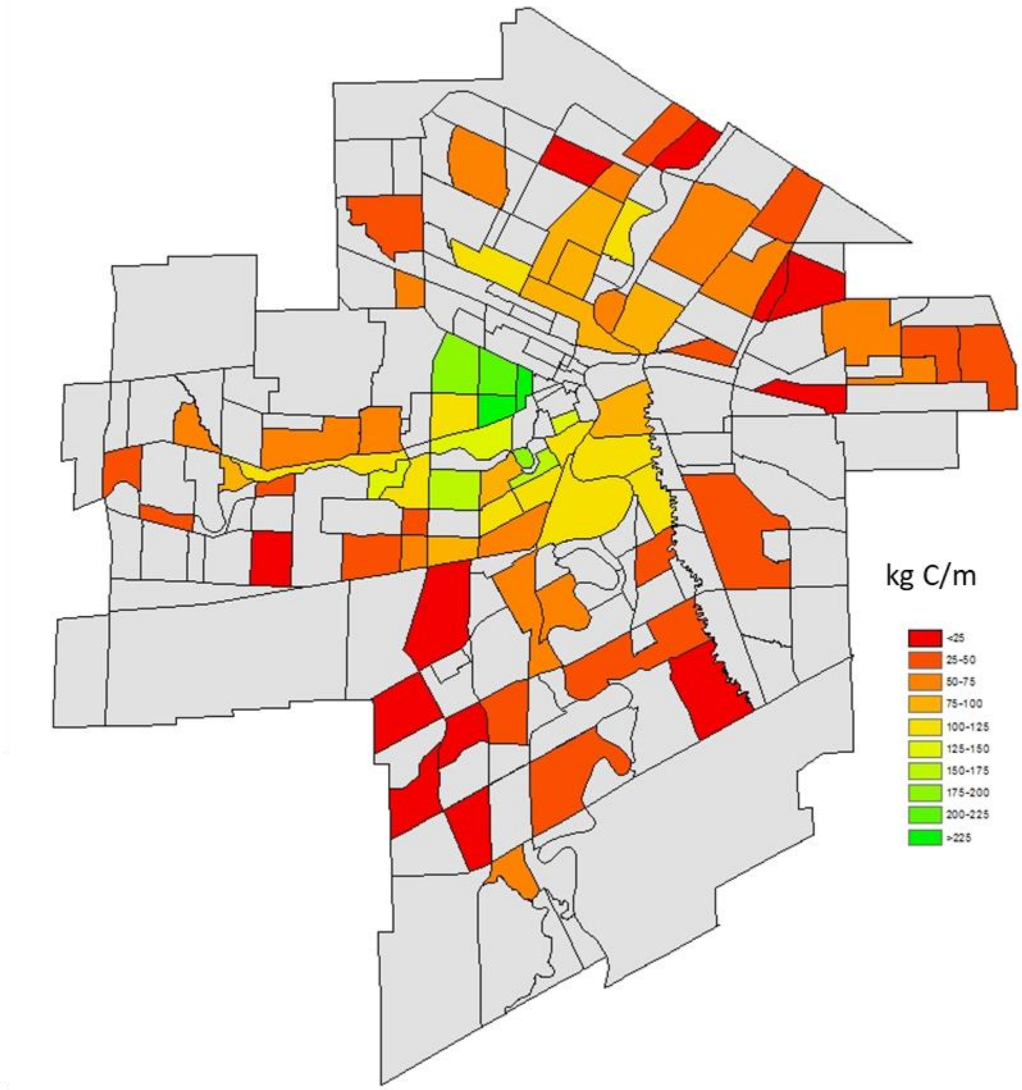


Figure 4.7 A heat map of standardized carbon values (kg C/residential frontage m) across surveyed neighborhoods in Winnipeg.

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Chapter 5

General Discussion

5.1 Diversity and Pest Vulnerability

Tree diversity in the City of Winnipeg has traditionally remained low due to climatic conditions in this northern city, the historical planting of monocultures, and the limitations of local nursery stock, among other factors (Rosen, 2015; City of Winnipeg, 2023). Results showed fairly low tree diversity across Winnipeg's residential component of its urban forest. No neighborhoods surveyed met the standards set by Santamour's (1999) 10-20-30 rule or the 10-20 target set by the City of Winnipeg in its 2023 Urban Forestry Strategy. Only 38 tree species were observed in the study, and 68% of neighborhoods had fewer than 19 species of trees.

The relatively low species richness in many neighborhoods means there is room for diversification, even given the few tree species options available for planting. The effects of climate change are already evident in the increasing minimum temperatures of the region's winters and decreased duration of extreme cold weather events (Prairie Climate Centre, 2023). Although local impacts of climate change on Winnipeg's urban forests remain to be investigated, increased temperatures may mean less cold-tolerant, exotic tree species not currently grown in the city may be become viable planting options over time. The ability to withstand freeze-thaw cycles may become one of the more important factors when choosing tree species to plant in the future. As discussed in section 2.1.5, non-native species can enhance diversity and help urban foresters maintain the ecosystem services provided by urban forests.

Winnipeg neighborhoods exhibit variation in diversity across many measures. Standard deviations for diversity metrics across neighborhoods were high relative to metric means. Furthermore, ranked lists indicate that when taken in aggregate, some neighborhoods outperform others across all metrics. By taking many measures of diversity into account, city foresters may target neighborhoods most in need of attention and identify neighborhoods that appear diverse by simple measures like species richness but suffer from poor tree health or a highly species dominant planting regime. Below, I identify some neighborhood "types" that exhibit poor performance across several diversity measures and suggest measures that may improve diversity:

5.1.1 American elm-dominated canopies

Some of the neighborhoods with the lowest diversity and evenness scores were dominated by American elms, These neighborhoods often had high mean dieback scores as well, likely from observation of DED-infected trees when the area was surveyed. While monitoring and removing DED-infected trees to retain the mature canopy as long as possible, urban forest managers can also prioritize diversity among new replacement trees.

Slow-growing, long-lived trees may be the best option for any residential canopy. These trees will grow large crowns over time, contributing to the City's canopy cover and carbon storage goals while cooling and beautifying their neighborhoods. In Winnipeg, successful species might include bur oak, silver maple, basswood, Ohio buckeye, and hackberry, among others. Managers may be hesitant to use species with faster growth rates (e.g., *Salix*, *Populus*, or Manitoba maple) due to structural and safety concerns and the trees' shorter lifespans, but finding some locations for these species could supplement the rapid loss of mature canopy over the short term. Neighborhoods surveyed that fit this type of canopy include Sargent Park, Broadway-Assiniboine, Luxton, and North River Heights.

5.1.2 Ash-dominated canopies

Ash-dominated canopies have low diversity and high evenness. Notably, in some of these neighborhoods American elm makes up a large proportion of the remaining canopy. In St Matthews, for example, 45% of trees sampled were ash species and an additional 28% were American elm. Because EAB and cottony ash psyllid are less prevalent in the city than DED, dieback values tended to be lower. at the time of surveys. Because widescale ash removal and replacement efforts have not been necessary to date, species richness in ash-doiminated neighborhoods was often lower than in elm-dominated neighborhoods, where infill trees were often novel species.

Again, prioritizing EAB and cottony ash psyllid detection in these neighborhoods is key to slowing canopy loss. Replacement trees should be varied and, where possible, resistant to both future pest invasions. These may be good neighborhoods to test boulevard planting of tree species that are less common in the city, as the timeline for canopy turnover will likely be longer. The slow spread of EAB thus far in Winnipeg indicates that boulevard ash trees may not see rapid removal over time, so incentivizing private tree planting in these neighborhoods may also increase diversity before major ash losses are seen. Neighborhoods surveyed that fit this canopy type include Eaglemere, Grassie, St. Matthews, Daniel Mcintyre, and Inkster-Faraday.

5.1.3 Neighborhoods with low species richness

Some neighborhoods had mid-level rankings when compared to other neighborhoods for evenness, diversity, and health measures but very low species richness nonetheless (<10 species). Even if they are not dominated by a single species or heavily populated with American elm or ash, it is still worth prioritizing diversification efforts in these neighborhoods. Though these forests are less susceptible to current pests, future pest invasions could affect large proportions of the canopy here. Because private property owners can plant a wider variety of species than is possible on boulevards, these neighborhoods are also good candidates for outreach efforts to private homeowners to encourage and incentivize new tree plantings. This might include a variety of conifers, fruit trees, and nut trees that the City of Winnipeg does not commonly plant on boulevards but which are popular with homeowners. Neighborhoods surveyed that fit this type include Earl Grey, Rivergrove, McMillan, and Riverbend.

Because the Winnipeg forestry department has limited resources, it may not be possible to track and manage for every diversity metric discussed in this thesis. It is vital that neighborhood diversity targets account for both richness and evenness; Simpson's 1/D is a measure that does this, encapsulating both facets of diversity. However, its meaning is not easily communicated to the public, and management based on this measure might be hard to plan for (eg. calculating how many new species it would take to raise a 1/D value by 0.2). That said, the City has already set a 10-20 diversity target, managing for evenness, so setting targets for species richness would be a simple and effective addition. The City could choose a richness target based on, for example, average richness in Winnipeg neighborhoods, focussing efforts on improving areas with low richness. On the other hand, the City might choose instead to increase richness in every neighborhood, aiming to add 1-4 new species (preferably from a different Genus, or even Family) to each neighborhood that are not currently found (or very rarely found) on public or private property in that neighborhood.

Though it is beyond the scope of this thesis to provide detailed analysis of the subject, it is worth discussing the relationship between diversity in the urban forest and social equity in Winnipeg. As discussed in section 2.1.2, diversity can impact ecosystem services in urban forests, including stormwater retention, filtration of air pollutants, and cooling. Furthermore, if managing for diversity is managing for a resilient forest, then it is worth managing for diversity in all neighborhoods, especially those where most residents live below the poverty line. Canopy cover is one of the primary tools for managing the urban heat island effect, and the Winnipeg Urban Forest Strategy explicitly describes the potential negative impacts of longer heat waves and low canopy cover on health outcomes in urban residents (City of Winnipeg 2023).

The luxury effect (section 2.1.3), in which areas with lower property values and lower average incomes have less trees and urban greenspaces, has been reported in the city's Urban Forest Strategy (City of Winnipeg 2023). Evidence shows residents in high poverty areas of the city have access to fewer trees per person, lower average canopy cover, and lower tree diversity than city residents as a whole, and consequently are exposed to higher than average temperatures in their neighborhoods (City of Winnipeg 2023). If the luxury effect is less evident in Winnipeg than other cities, it is because of a legacy of boulevard tree planting and care that has left Winnipeg's urban core with a mature elm canopy. As these old trees are removed with age and the impacts of DED, it is vital that the City replace these trees with enough saplings to maintain and improve canopy cover and diversity, with an aim of tracking and reducing the disparity between low and high-income areas over time.

These suggestions are based on the data collected for this project, which were taken from samples of blocks in study neighborhoods and include residential boulevard trees and front-yard private trees. That said, these recommendations have been formulated with data that included detailed inventories of trees on private property, which the City of Winnipeg has not previously been able to use. City of Winnipeg efforts at diversification are ongoing and based in long term planning and knowledge of Winnipeg's urban forest; these suggestions are meant to augment and inform other planning processes.

Trees found on private property are vital to the City's tree diversity. Sixty percent of trees surveyed were on private property, and many tree species are rare on boulevards but common on private property (eg. Spruce, white cedar). The data shows that street trees have higher dieback values than trees on private property and have poorer diversity across all measures than private trees when looked at as a whole. The Winnipeg Urban Forest Strategy includes commitments to incentivizing private tree planting and developing a recommended species list for homeowners (City of Winnipeg, 2023), with an emphasis on the city's low-income neighborhoods. Because of the wider range of tree choices for small-scale buyers and the relatively large private land area in residential areas, this focus on diversifying private tree plantings could greatly impact diversity outcomes in years to come. The analysis shows that there is room for growth for tree plantings on private property, with the majority of addresses surveyed having only one tree on private property (in the front yard), regardless of home characteristics like value, land area, and living area. Although I was unable to survey the back yards of private residences due to privacy concerns there may also be significant additional opportunities for further increases in trees on private property in Winnipeg.

The Winnipeg urban forest PVM developed in this paper is intended to be used by urban forest managers as a tool to visualize and compare pest impacts across the city's neighborhoods. While poor PVM scores were correlated with low Berger-Parker and 10-20-30 adherence scores, our analysis showed that there are potential city species proportions in Winnipeg that would have a worse overall pest score than the actual score with current species proportions, even while adhering to Santamour's 10-20-30 rule. Diversity and pest management are related, but managing for diversity alone will not inherently mean less pest impacts on the city. The PVM shows that the particular impacts of current and future pests, as well as planting based on host tree species, must be taken into account for best management of pests and preservation of Winnipeg's urban forest for the future.

When I used the PVM to calculate the potential impact of pests on city forests, the top two pests are scarce or not yet present in the city: Asian longhorned beetle and *Armillaria* root rot. These pests have a wide range of potential hosts and their impacts are typically severe to lethal. The potential impacts of these particular pests could mean that some neighborhoods, though ostensibly diverse by the metrics used in this report, were identified as highly susceptible to pest invasion using the PVM (Rockwood, Sturgeon Creek, Crescent Park, Margaret Park, and Springfield North).These neighborhoods may not be the first priority for managers as they deal with neighborhoods impacted by current pests like DED and EAB, but diversifying plantings to slow the spread of Asian longhorned beetle and *Armillaria* root rot could be highly beneficial to these neighborhoods in the long run. Tree taxa that are not susceptible to either pest include linden, hawthorn, Russian olive, and catalpa.

5.2 Estimating Carbon Storage

Accurate carbon accounting remains a challenge for urban forestry professionals. I-Tree is one of the most widely used urban forestry surveying tools in North America, but the methods used in developing the program's biomass equations are opaque for potential users. Moreover, the datasets used to create the original equations on which i-Tree biomass equations were built are not publicly available. I was able to create prediction intervals for some species, some of which were 2-3 times the estimated value within the 80% confidence value, showing that we can only give carbon estimates within very broad boundaries. Our work does not address variability in carbon storage that arises from differences between the climate, soil, stand density, and other growing conditions at the site of the source data (usually a forest stand) vs. the city survey site. The best methods for carbon estimation come from weighing oven-dried segments of a sacrificial harvest (see section 1.2.3), which was beyond the scope of this project but might be the best option for city foresters and university researchers in slow growing northern urban forests to compare biomass values given by programs like i-Tree to actual biomass values.

The regional distribution of carbon storage in Winnipeg (Figure 4.7) is probably a consequence of the city's development pattern. The oldest central neighborhoods are populated with American elms in the public boulevards, and secondary development included monocrop ash plantings, which are not as large as American elm at maturity. Finally, the newest residential neighborhoods built on the outskirts of the city have very young trees. As DED and EAB continue to necessitate the removal of a component of the city's mature canopy, a wider variety of tree species will be planted across neighborhoods on public boulevards. It remains to be seen how homeowners and property managers will respond to tree loss on private lots.

It also worth emphasizing that our methods only describe carbon storage in the front yards and boulevards of surveyed neighborhoods; backyards, parks, and forested areas are not included in this report. Backyards might hold larger trees than front yards, and the species might differ significantly from front yards, as backyard trees may be more likely to result from untended sprouts than intentional front yard planting. Trees also may grow larger in parks than boulevards due to better space for rooting and less competition for sunlight. Further research would be needed to speak with certainty of these areas.

Because I did not look at sequestration or replacement rates, I cannot say whether Winnipeg's urban forest serves as a net carbon source or sink. Few tree species in the survey grow to the size of mature elms, so it is possible that the city will see a decrease in carbon storage over the course of the elm die-off due to DED if trees are replaced at a 1:1 ratio as called for in the 2023 urban forestry strategy (City of Winnipeg, 2023). However, the plan also calls for planting 17,000 new trees a year, with 60% of those planted by the City being "large" trees (trees with large DBH at maturity).

Total carbon balance will depend on whether annual tree loss is brought to a manageable level. It also depends on the species and number of trees chosen for replacement and new plantings, as well as growth rates of remaining old trees and newly planted trees and disposal methods for removed trees; if the City can develop partnerships with more businesses looking to harvest and utilize lumber from felled trees, the carbon storage of those trees can be extended past the "lifetime" of the tree. It is likely that short-term storage will be reduced due to the ongoing loss of large trees and the relatively small amount of carbon sequestered during the early growth of newly planted trees. Over the long term, maintaining or increasing current levels of carbon storage will depend on the number of trees planted, survival rates, and whether new urban tree species can achieve the same size as current American elm and ash canopies.

Moreover, the carbon storage reported here does not account for greenhouse gas emissions emitted in the production and distribution of young trees for city planting. Petri et al. (2016) showed that the carbon costs of nursery production could mean that a tree is not carbon neutral for almost three decades, although many trees in Winnipeg are far older than this and have continued to grow and sequester carbon at large DBHs. Although boulevard trees are often planted at large DBH to prevent vandalism, smaller trees are an option for homeowners and may provide cost savings as an added benefit. Some smaller, less energy intensive options could include bare root stock, transplanting trees at a small DBH, and growing trees from seed or cuttings at home.

Because of the extreme level of uncertainty in carbon calculations, it is worth discussing whether tracking carbon and managing for maximum carbon storage and sequestration is a worthwhile endeavor. While it is unclear what the beneficial outcomes of tracking carbon in urban forests are beyond carbon accounting at a very localized level, managing for canopy cover has a variety of well-documented benefits: lower temperatures, lower cooling bills, higher home values and better health outcomes (City of Winnipeg, 2023; Nowak, 2020). If the City is already managing for increased canopy cover, it is likely that priority will fall on tall, long-lived trees with large crowns, ie., trees with high carbon storage over time. With minimal effort, the City could pursue increased canopy cover while also tracking simpler size-related metrics with less uncertainty than carbon storage, like total tree basal area.

5.3 Limitations

Because this project was conducted over many years by student researchers, limitations exist in the dataset and methodologies used. Some limitations were due to inaccessibility of trees on private property. Only front yards were surveyed, so the results here do not apply to backyards in Winnipeg. This implies that at the very least neighborhood species richness and carbon storage are higher on private property than reported here, although to what extent I cannot say with the data collected. There is also likely a large bias in the dataset in terms of measurements, as measurements for trees on private property were always estimated based on measurements on nearby trees on boulevards to prevent those doing surveys from trespassing on private yards.

Trees measurements were not adjusted for growth over time, and potential die-off over time was not accounted for in my dataset. Growth rates for Winnipeg trees are not available, and urban tree growth rates are extremely regional (McPherson et al., 2016). Die-off rates have been reported for elm, but rates for other species do not exist, and I did not want to bias the data by only adjusting for die-off in one species. These limitations could mean that carbon values are slightly under- or overreported.

Finally, although our method for estimating carbon confidence values is statistically robust, the best way to determine biomass and carbon storage is always by direct measurement, which was beyond the scope of this project.

5.4 References

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Chapter 6

Conclusions

Winnipeg's urban forest is subject to a variety of pressures, from the extreme and prolonged cold events of a northern prairie climate to the introduction and spread of potentially deadly pests in an urban forest with a legacy of monoculture tree planting. Maintaining and increasing forest diversity under these conditions is a challenge that requires knowledge of the present diversity measures in the city and the potential pest threats to this diversity. These pressures also affect tree growth and survival, key components to increasing carbon storage in Winnipeg's forest. This thesis seeks to provide a baseline summary of tree diversity on both public and private property in the city's residential neighborhoods, as well as a tool for assessing pest threats to city neighborhoods based on their diversity. It also describes carbon storage across the surveyed portions of the city and provides confidence values for that carbon storage based on i-Tree biomass values currently in use by the City of Winnipeg for its forest carbon accounting.

In terms of tree diversity, results showed low diversity across a number of measures. That said, trees on private property were highly diverse compared to their boulevard counterparts, and some neighborhoods far outperformed others in terms of tree diversity. This means that City of Winnipeg urban foresters can bolster diversity not only with City of Winnipeg tree plantings, but by fostering public-private partnerships, as called for in the Urban Forest Strategy. This information could also allow the City of Winnipeg to target neighborhoods most in need of diversification, especially if used in concert with the PVM.

The Pest Vulnerability Matrix (PVM) developed for this paper can be used as a preliminary guidance tool for the City of Winnipeg to examine how insect pests may affect trees in city neighborhoods based on the makeup of that neighborhood's tree population. It also allowed me to show that while diversity and pest resilience are related, a diverse forest in terms of the City of Winnipeg's own 10-20 target (no more than 10% of a given species in an area, and no more than 20% of a given genus) may be more subject to pest invasion than a less diverse forest, depending on the makeup of the trees.

Finally, this thesis explores carbon storage in the city. Starting with an exploration of i-Tree methodology, I was then able to develop confidence intervals for i-Tree biomass equations, the software used by the City of Winnipeg in its Urban Forest Strategy. I was also able to estimate carbon storage in neighborhoods surveyed across the city and describe “replacement values” for species in the city inventory that may be used to replace ash and American elm trees as they are removed.

This work includes the largest and most thorough survey of private trees in the city of Winnipeg. As such, the conclusions drawn here are notable, especially those pertaining to the contribution of private trees to city diversity and pest resilience. In addition, it includes the first modification of the PVM for a northern climate, highlighting the potential impacts of pests on a city with few options for tree diversification. It also provides a method, though imperfect, for any planners using i-Tree to create prediction intervals around carbon estimates being used for forest planning and municipal carbon accounting.

Future research on the topics explored in this thesis may take many forms. Monitoring of trees planted and lost across the city, especially on private property, will help the City of Winnipeg see trends in taxonomic and size diversity over time. This is especially important with the advent of the Urban Forest Strategy, which includes plans to augment tree planting and replace elms lost to DED on public and private property. It would also be worthwhile to test the predictions of pest vulnerability made by the PVM, comparing PVM rankings to actual infestation and removal rates. Finally, establishing Winnipeg-specific biomass equations for major species based on actual harvested city trees would be a critical endeavor, allowing City of Winnipeg urban foresters to compare i-Tree predictions to the actual numbers and allowing for improved, regional carbon predictions.

As the pressures of climate change and pest invasion intensify over the coming years, urban municipalities will need to ensure the long-term sustainability of their tree canopies. But these challenges are not a death sentence for urban forest ecosystems. Intentional action towards diversification, including establishing partnerships with private landholders, can help support resilient neighbourhood forests. Careful monitoring and efficient removals can help to mitigate the impacts of pest infestations. Carbon storage can be maximized as the city navigates replacing ash and American elm trees with new species. These strategies, taken together, will allow cities like Winnipeg to make significant strides towards cultivating a healthy and diverse urban forest that can be enjoyed for generations to come.

Appendix A Characteristics of surveyed homes by neighborhood, as reported by the City of Winnipeg tax parcel database in 2022

Neighborhood	Average Home Age	Average Property Value	Average Land Area (m^3) per Home	Average Living Area (m^3) per Home	Average Living Area/Land Area	Per Capita Median Income
Alpine Place and St. George	64	$288,724	475	101	0.21	$ 32,603
Armstrong Point	95	$639,086	1498	288	0.19	$ 59,421
Birchwood and Bruce Park	90	$260,761	378	119	0.31	$ 38,437
Bridgwater Forest	7	$621,443	529	207	0.39	$ 37,144
Bridgwater Trails and Bridgwater Centre	4	$1,200,451	1319	234	0.18	$ 34,275
Brockville and Linden Woods	35	$485,436	664	178	0.27	$ 45,034
Burrows Central and Shaughnessy Park	63	$215,000	218	69	0.32	$ 28,574

Canterbury Park	31	$303,852	441	120	0.27	$	43,332
Chalmers	87	$184,617	365	86	0.23	$	28,865
Colony and West Broadway	110	$214,712	336	161	0.48	$	22,642
Crescent Park	67	$348,766	655	118	0.18	$	36,847
Crescentwood	99	$552,797	712	231	0.32	$	54,426
Dakota Crossing	24	$388,732	488	145	0.30	$	43,493
Daniel Mcintyre	104	$157,143	275	108	0.39	$	25,069
Eaglemere	19	$390,298	510	147	0.29	$	37,076
Earl Grey	103	$262,635	282	117	0.42	$	35,068
East Elmwood	76	$199,009	374	81	0.22	$	32,295
Ebby-Wentworth and Grant Park	67	$254,137	448	94	0.21	$	29,242
Edgeland and Sir John Franklin	84	$290,536	419	101	0.24	$	39,045
Elmhurst	32	$453,718	655	170	0.26	$	47,278

Fort Richmond	45	$395,847	637	154	0.24	$	26,012
Glendale	43	$288,889	592	111	0.19	$	37,588
Glenelm	93	$209,544	327	93	0.29	$	36,481
Glenwood	92	$244,945	384	89	0.23	$	39,851
Grassie	25	$359,503	637	133	0.21	$	37,904
Inkster-Faraday and Jefferson	72	$250,974	521	105	0.20	$	31,570
J. B. Mitchell	63	$325,078	451	100	0.22	$	38,794
Kern Park and Melrose	78	$199,964	373	86	0.23	$	39,768
Kildare-Redonda	56	$264,872	541	99	0.18	$	35,747
King Edward	72	$218,175	377	87	0.23	$	36,405
Lord Roberts and Riverview	87	$304,868	499	114	0.23	$	38,424
Luxton	102	$193,031	342	111	0.32	$	34,505
Margaret Park	49	$276,068	476	113	0.24	$	32,918
Mathers	63	$372,290	566	126	0.22	$	34,787

Mcmillan	113	$361,500	516	187	0.36	$	34,699
Meadows and Radisson	56	$252,563	426	89	0.21	$	40,585
Minto	97	$217,399	296	101	0.34	$	35,617
Mission Gardens	15	$320,167	465	97	0.21	$	37,376
North Point Douglas and William Whyte	103	$126,172	330	98	0.30	$	21,694
North River Heights	92	$398,725	466	152	0.33	$	54,828
Norwood West and Norwood East	95	$313,757	460	127	0.28	$	40,445
Old Tuxedo	68	$993,602	1202	282	0.23	$	60,179
Pembina Strip, Beaumont, and Maybank	69	$283,729	554	96	0.17	$	30,896
Riverbend	46	$284,040	625	96	0.15	$	42,586
Rivergrove	30	$405,172	602	151	0.25	$	43,747
River-Osborne	124	$303,078	490	305	0.62	$	29,867
Rockwood	71	$272,927	451	102	0.23	$	33,062

Rossmere-A and Rossmere-B	64	$263,876	520	97	0.19	$	35,144
Sargent Park	84	$234,351	343	96	0.28	$	32,676
Seven Oaks	75	$247,602	488	105	0.21	$	36,020
Silver Heights and Deer Lodge	71	$286,963	556	117	0.21	$	40,867
South River Heights	65	$385,117	540	134	0.25	$	47,117
South Tuxedo	37	$697,806	1009	226	0.22	$	57,073
Southboine	49	$625,468	1517	215	0.14	$	35,930
Spence	107	$167,074	325	133	0.41	$	19,886
Springfield North	35	$345,496	552	136	0.25	$	44,579
St. John's	101	$172,023	366	110	0.30	$	24,751
St. Matthews	106	$162,473	293	121	0.42	$	24,185
Sturgeon Creek	59	$279,964	583	102	0.18	$	68,331
Templeton-Sinclair	39	$336,059	644	140	0.22	$	34,790
The Maples	43	$321,611	585	126	0.21	$	28,994

Tissot and Central St. Boniface	94	$203,880	409	84	0.21	$ 30,245
Tyndall Park	43	$281,015	466	100	0.21	$ 32,694
Valley Gardens and Springfield South	41	$304,359	543	109	0.20	$ 33,442
Vialoux	44	$525,934	1064	172	0.16	$ 37,605
Vista, Meadowood, and Minnetonka	54	$346,284	890	129	0.15	$ 39,769
Waverley Heights	44	$355,808	613	117	0.19	$ 33,026
Whyte Ridge	32	$442,197	667	153	0.23	$ 47,020
Windsor Park and Southdale	55	$318,415	636	116	0.18	$ 41,252
Wolseley	105	$290,465	312	131	0.42	$ 36,368
Woodhaven	68	$324,972	925	126	0.14	$ 46,810

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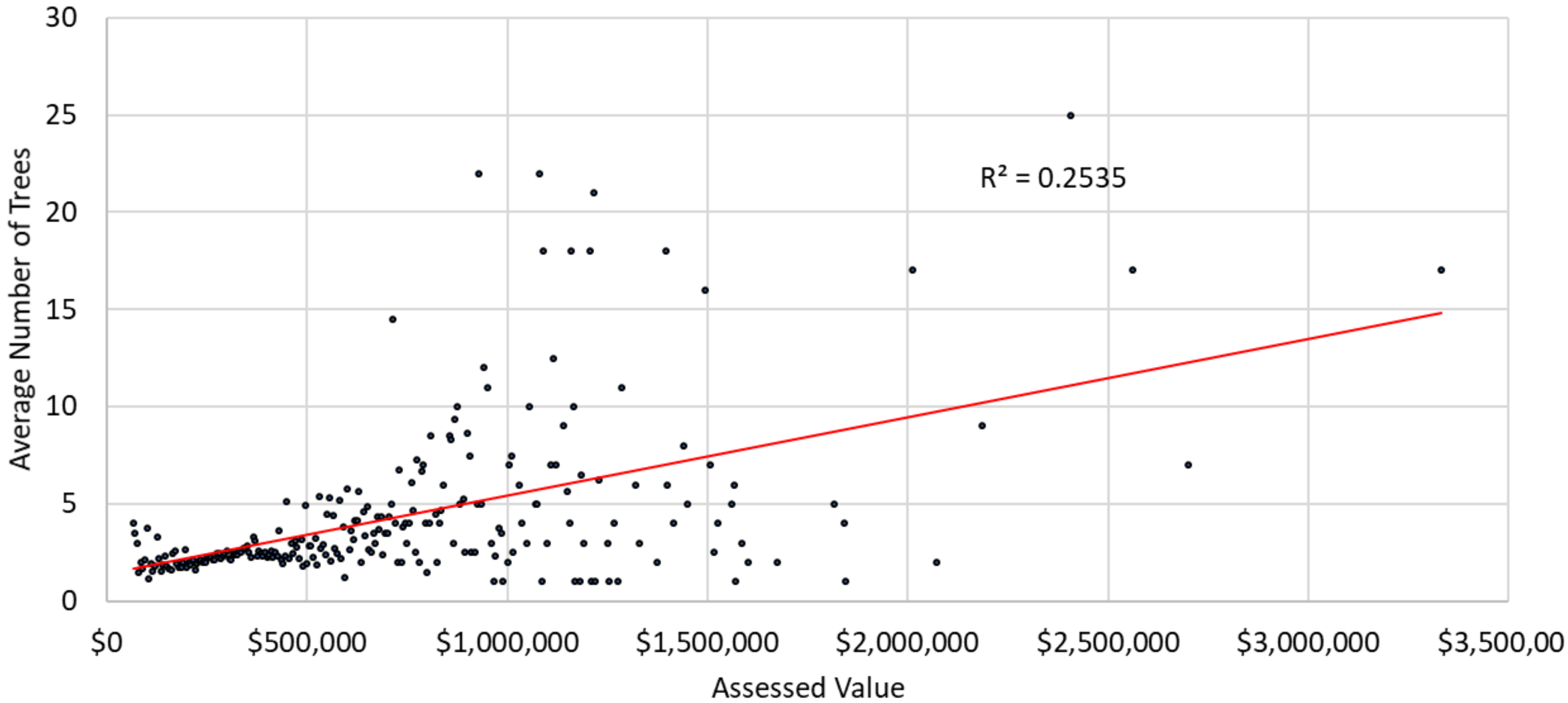

Figure A1. Average number of trees on private property as a function of assessed value in residential homes in survey.

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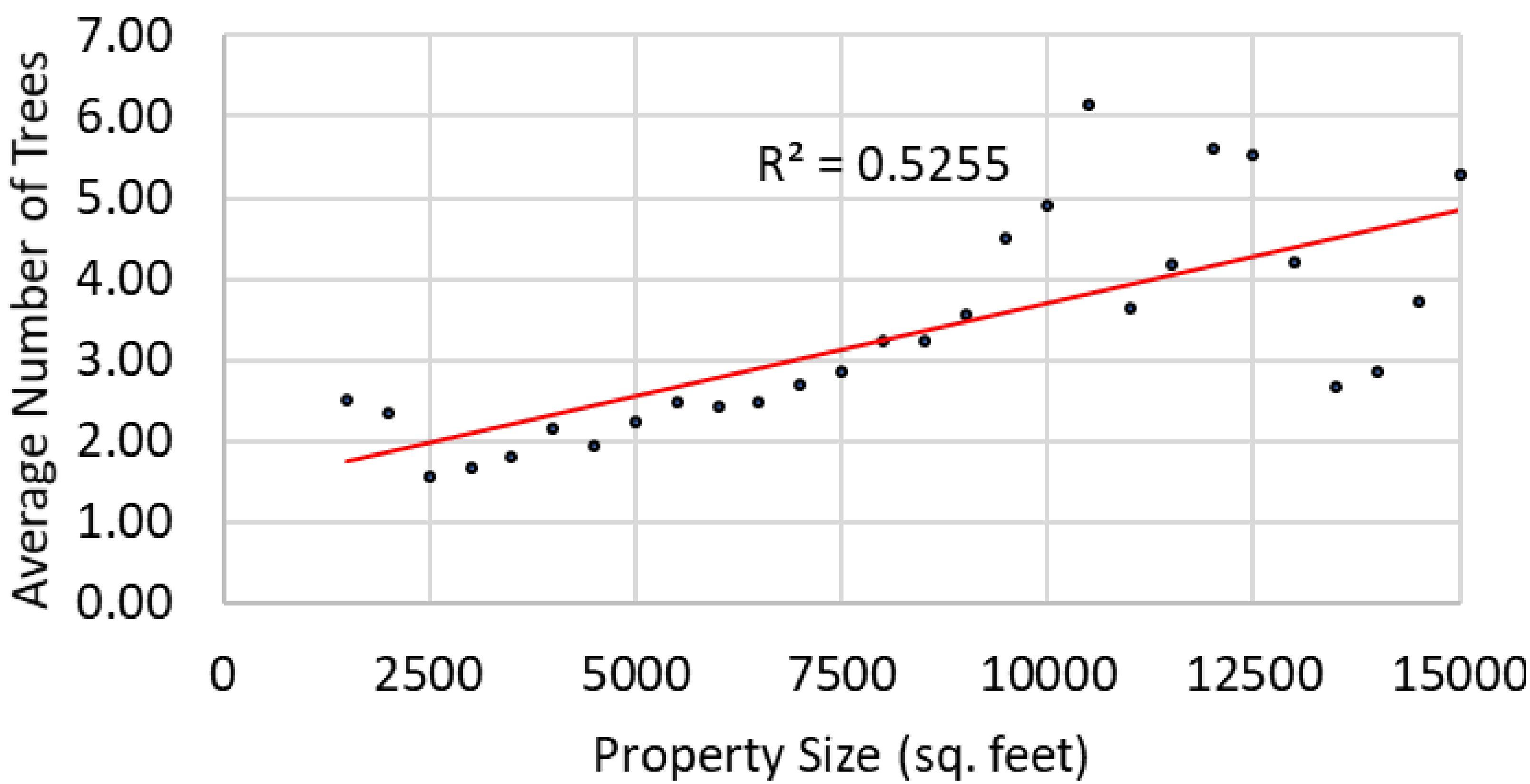

Figure A2. Average number of trees on private property as a function of property size in residential homes in survey (only properties under 15,000 square feet shown).

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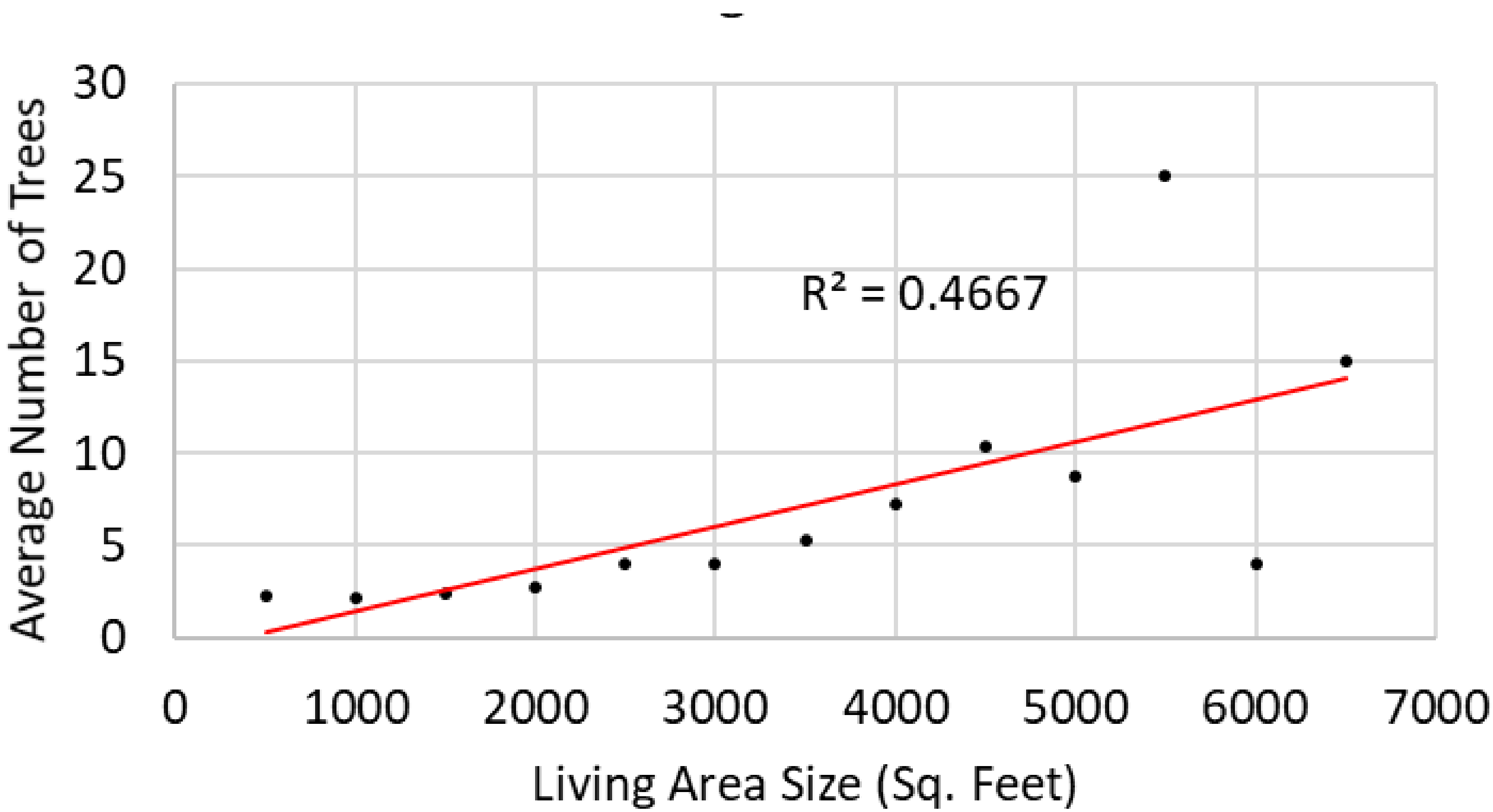
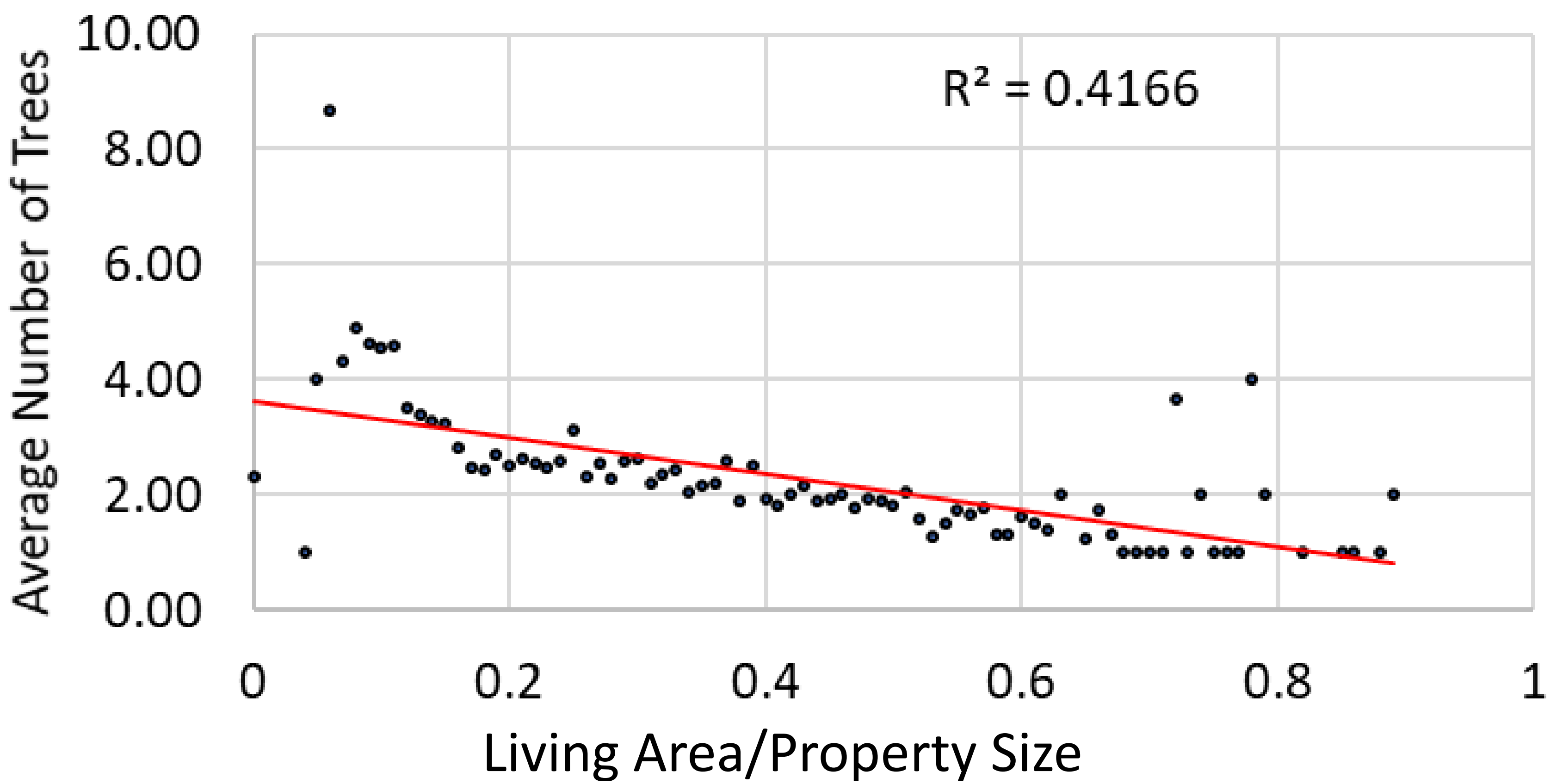


Figure A3. Average number of trees on private property as a function of living area in residential homes in survey.



20 Figure A4. Average number of trees on private property as a function of living area/property size in residential homes in survey.

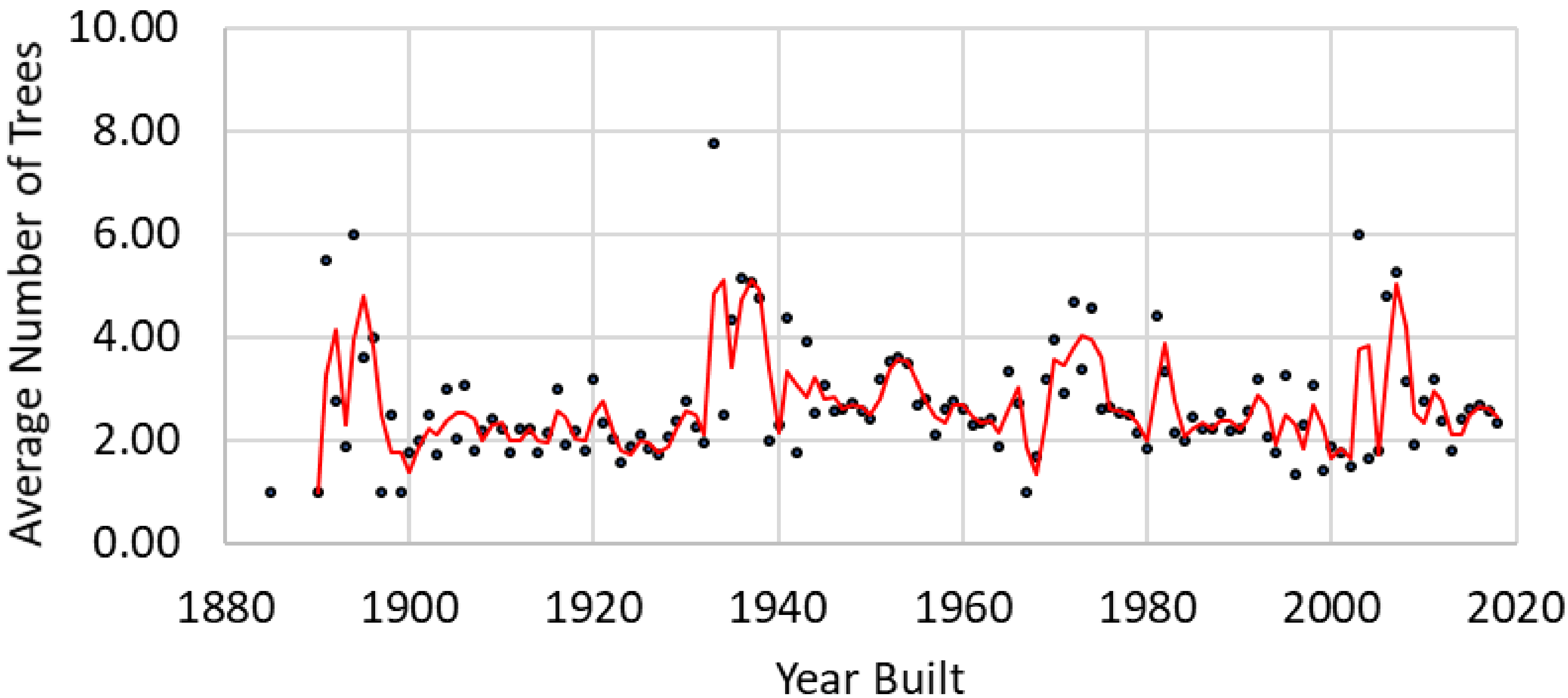


Figure A5. Average number of trees on private property as a function of year built in residential homes in survey.

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Appendix B Tree species, genera, and families observed in the study.

Manitoba native species based on *The Field Guide to the Trees of Manitoba* (Manitoba Sustainable Development 2019).

Common Name	Scientific Name	Genus	Family	Native to MB?
Balsalm Fir	*Abies balsamifera* Michx.	*Abies*	*Pinaceae*	Y
Amur Maple	*Acer ginnala* Maxim.	*Acer*	*Sapindaceae*	N
Boxelder	*Acer negundo* L.	*Acer*	*Sapindaceae*	N
Red Maple	*Acer rubrum* L.	*Acer*	*Sapindaceae*	N
Silver Maple	*Acer saccharinum* L.	*Acer*	*Sapindaceae*	N
Sugar Maple	*Acer saccharum* Marshall	*Acer*	*Sapindaceae*	N
Mountain Maple	*Acer spicatum* Lam.	*Acer*	*Sapindaceae*	N
Ohio Buckeye	*Aesculus glabra* Willd.	*Aesculus*	*Sapindaceae*	N
Alder	*Alnus spp*	*Alnus*	*Betulaceae*	N
Paper Birch	*Betula papyrifera* Marshall	*Betula*	*Betulaceae*	Y
Northern Catalpa	*Catalpa speciosa* Warder	*Catalpa*	*Bignoniaceae*	N
Common Hackberry	*Celtis occidentalis* L.	*Celtis*	*Cabbanaceae*	Y
Dogwood	*Cornus spp*	*Cornus*	*Cornaceae*	Some species

Hawthorn	*Crataegus spp*	*Crataegus*	*Rosaceae*	Some species
Russian Olive	*Elaeagnus angustifolia* L.	*Elaeagnus*	*Elaeagnaceae*	N
Beech	*Fagus* spp	*Fagus*	*Fagaceae*	N
Manchurian Ash	*Fraxinus mandschurica* Rupr.	*Fraxinus*	*Oleaceae*	N
Green Ash	*Fraxinus pennsylvanica* Marshall	*Fraxinus*	*Oleaceae*	Y
Walnut	*Juglans* spp	*Juglans*	*Juglandaceae*	N
Juniper	*Juniperus* spp	*Juniperus*	*Cupressaceae*	Some species
Larch	*Larix* spp	*Larix*	*Pinaceae*	Some species
Apple	*Malus* spp	*Malus*	*Rosaceae*	N
Spruce	*Picea* spp	*Picea*	*Pinaceae*	Some species
Jack Pine	*Pinus banksiana* Lamb.	*Pinus*	*Pinaceae*	Y
Red Pine	*Pinus resinosa* Sol.	*Pinus*	*Pinaceae*	Y
Eastern White Pine	*Pinus strobus* L.	*Pinus*	*Pinaceae*	Y
Scots Pine	*Pinus sylvestris* L.	*Pinus*	*Pinaceae*	N
Eastern Cottonwood	*Populus deltoides* Marshall	*Populus*	*Salicaceae*	Y
Trembling Aspen	*Populus tremuloides* Michx.	*Populus*	*Salicaceae*	Y
Amur Chokecherry	*Prunus maackii* Rupr.	*Prunus*	*Rosaceae*	N

Western Chokecherry	*Prunus virginiana* L.	*Prunus*	*Rosaceae*	Y
Bur Oak	*Quercus macrocarpa* Michx.	*Quercus*	*Fagaceae*	Y
Willow	*Salix spp*	*Salix*	*Salicaceae*	Some species
American Mountain Ash	*Sorbus americana* Marshall	*Sorbus*	*Rosaceae*	Y
Cedar	*Thuja spp*	*Thuja*	*Cupressaceae*	Some species
American Linden	*Tilia americana* L.	*Tilia*	*Malvaceae*	Y
Littleleaf Linden	*Tilia cordata* Mill.	*Tilia*	*Malvaceae*	N
American Elm	*Ulmus americana* L.	*Ulmus*	*Ulmaceae*	Y
Siberian Elm	*Ulmus pumila* L.	*Ulmus*	*Ulmaceae*	N

Appendix C Diversity Indicators

Table C1. Equations used to calculate diversity indicators.

Indicator	Equation	Variables
Simpson's 1/D	$1/\Sigma\left(\frac{n_{i[n_i-1]}}{\mathrm{N}[N-1]}\right)$	n is the number of individuals of the i^{th} species, and N is total number of individuals
Simpson's Evenness $E_{1/D}$	$\frac{(1/D)}{S}$	1/D is Simpson's 1/D, defined above, and S is the number of species in the sample
10-20-30 Divergence	(a-0.1)+(b-0.2)+(c-0.3)	*a* is the percent representation of the most populous species, *b* is the percent representation of the most populous genus, and *c* is the percent representation of the most populous family

Table C2. Diversity indicators for all surveyed trees by neighborhood.

Neighborhood	Species Richness	Berger-Parker Dominance	1/D	Simpson's Evenness	Mean Dieback
Alpine Place and St. George	19	0.21	9.78	0.07	13%
Armstrong Point	19	0.35	5.30	0.01	10%

Birchwood and Bruce Park	11	0.43	3.21	0.02	12%
Bridgwater Forest	15	0.46	4.12	0.04	2%
Bridgwater Trails and Bridgwater Centre	18	0.13	10.93	0.07	9%
Broadway-Assiniboine	7	0.60	2.58	0.03	2%
Brockville and Linden Woods	16	0.26	6.02	0.02	6%
Brooklands	15	0.37	5.18	0.04	14%
Burrows Central and Shaughnessy Park	18	0.25	7.06	0.03	7%
Canterbury Park	17	0.43	4.26	0.02	6%
Chalmers	16	0.20	8.51	0.04	2%
Colony and West Broadway	14	0.48	3.50	0.01	13%
Crescent Park	18	0.19	8.08	0.02	7%
Crescentwood	20	0.41	4.51	0.01	6%
Dakota Crossing	21	0.24	6.73	0.02	5%

Daniel Mcintyre	15	0.38	3.60	0.01	14%
Eaglemere	10	0.60	2.56	0.03	9%
Earl Grey	9	0.32	5.05	0.04	6%
East Elmwood	16	0.26	7.05	0.06	9%
Ebby-Wentworth and Grant Park	16	0.30	6.08	0.02	9%
Edgeland and Sir John Franklin	20	0.19	8.08	0.02	2%
Elmhurst	14	0.28	6.00	0.05	2%
Fort Richmond	20	0.26	7.04	0.02	2%
Glendale	15	0.26	8.87	0.16	4%
Glenelm	14	0.22	7.43	0.08	23%
Glenwood	18	0.24	7.60	0.02	8%
Grassie	14	0.53	3.23	0.02	2%
Inkster-Faraday and Jefferson	13	0.36	4.30	0.02	10%
J. B. Mitchell	17	0.25	6.91	0.03	3%
Kern Park and Melrose	16	0.21	6.82	0.05	4%
Kildare-Redonda	17	0.18	7.84	0.03	3%
King Edward	17	0.25	7.96	0.03	11%

Lord Roberts and Riverview	25	0.45	4.08	0.01	13%
Luxton	17	0.58	2.70	0.01	12%
Margaret Park	14	0.34	5.41	0.03	4%
Mathers	17	0.47	3.86	0.04	18%
Mcmillan	11	0.35	5.54	0.04	6%
Meadows and Radisson	16	0.27	5.86	0.01	9%
Minto	15	0.43	4.00	0.01	15%
Mission Gardens	9	0.47	3.56	0.06	13%
North Point Douglas and William Whyte	12	0.34	4.84	0.03	18%
North River Heights	26	0.51	3.42	0.00	14%
Norwood West and Norwood East	23	0.29	6.81	0.01	14%
Old Tuxedo	26	0.38	4.70	0.00	15%
Pembina Strip, Beaumont, and Maybank	19	0.26	7.21	0.04	6%

Richmond Lakes and Parc La Salle	20	0.21	9.45	0.04	4%
Riverbend	10	0.27	5.46	0.05	21%
Rivergrove	11	0.45	3.49	0.02	7%
River-Osborne	14	0.43	3.77	0.02	10%
Rockwood	20	0.31	6.41	0.02	8%
Rossmere-A and Rossmere-B	18	0.19	7.10	0.02	6%
Sargent Park	19	0.64	2.24	0.00	15%
Seven Oaks	16	0.34	6.00	0.03	13%
Silver Heights and Deer Lodge	22	0.35	5.71	0.01	12%
South Pointe	16	0.22	6.92	0.03	7%
South River Heights	22	0.29	6.19	0.01	8%
South Tuxedo	19	0.29	6.77	0.02	5%
Southboine	18	0.18	9.73	0.07	4%
Spence	17	0.44	3.71	0.01	12%
Springfield North	16	0.32	5.44	0.02	6%
St. John's	17	0.37	4.96	0.02	14%
St. Matthews	13	0.45	3.45	0.01	16%
Sturgeon Creek	26	0.21	7.83	0.01	9%

Templeton-Sinclair	17	0.24	6.39	0.04	6%
The Maples	20	0.31	5.57	0.02	14%
Tissot and Central St. Boniface	24	0.20	8.31	0.02	9%
Tyndall Park	17	0.42	3.96	0.01	4%
Valley Gardens and Springfield South	16	0.35	4.90	0.03	5%
Vialoux	17	0.27	7.21	0.05	2%
Vista, Meadowood, and Minnetonka	22	0.25	7.24	0.01	7%
Waverley Heights	18	0.26	6.06	0.02	6%
Whyte Ridge	19	0.30	5.23	0.02	3%
Windsor Park and Southdale	22	0.30	5.82	0.01	9%
Wolseley	23	0.32	5.36	0.01	15%
Woodhaven	17	0.56	2.94	0.01	1%
Winnipeg	39	0.23	8.01	0.00	10%

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Table C3. Highest percent values for species, genus, and family for Winnipeg neighborhoods. Santamour (1999) suggests no area have more than 10% of a single species, no more than 20% of a single genus, and no more than 30% of a single family.

Neighborhood	Species	Genus	Family
Alpine Place and St. George	21%	21%	21%
Armstrong Point	35%	36%	36%
Birchwood and Bruce Park	43%	43%	43%
Bridgwater Forest	46%	46%	46%
Bridgwater Trails and Bridgwater Centre	13%	22%	22%
Broadway-Assiniboine	60%	60%	60%
Brockville and Linden Woods	26%	26%	30%
Brooklands	37%	37%	37%
Burrows Central and Shaughnessy Park	25%	25%	27%
Canterbury Park	41%	43%	43%
Chalmers	20%	20%	20%
Colony and West Broadway	48%	51%	51%
Crescent Park	19%	19%	19%
Crescentwood	41%	42%	42%
Dakota Crossing	24%	24%	24%
Daniel Mcintyre	38%	38%	38%
Eaglemere	37%	60%	60%
Earl Grey	32%	32%	32%
East Elmwood	22%	29%	29%
Ebby-Wentworth and Grant Park	29%	30%	30%
Edgeland and Sir John Franklin	19%	19%	19%
Elmhurst	28%	28%	28%
Fort Richmond	26%	26%	26%
Glendale	26%	26%	28%
Glenelm	22%	22%	22%
Glenwood	24%	24%	24%
Grassie	52%	53%	53%

Inkster-Faraday and Jefferson	36%	36%	36%
J. B. Mitchell	25%	25%	25%
Kern Park and Melrose	21%	22%	22%
Kildare-Redonda	18%	18%	18%
King Edward	23%	25%	25%
Lord Roberts and Riverview	45%	45%	45%
Luxton	58%	60%	60%
Margaret Park	34%	34%	34%
Mathers	31%	47%	47%
Mcmillan	35%	35%	35%
Meadows and Radisson	23%	27%	27%
Minto	43%	44%	44%
Mission Gardens	40%	47%	47%
North Point Douglas and William Whyte	33%	34%	34%
North River Heights	51%	52%	52%
Norwood West and Norwood East	29%	38%	38%
Old Tuxedo	38%	39%	39%
Pembina Strip, Beaumont, and Maybank	25%	26%	26%
Richmond Lakes and Parc La Salle	21%	21%	21%
Riverbend	27%	27%	27%
Rivergrove	45%	45%	45%
River-Osborne	41%	44%	44%
Rockwood	31%	36%	36%
Rossmere-A and Rossmere-B	19%	26%	26%
Sargent Park	64%	65%	65%
Seven Oaks	34%	45%	45%
Silver Heights and Deer Lodge	35%	35%	36%
South Pointe	22%	22%	22%

South River Heights	29%	29%	29%
South Tuxedo	29%	29%	34%
Southboine	18%	19%	20%
Spence	44%	45%	45%
Springfield North	31%	32%	32%
St. John's	37%	39%	39%
St. Matthews	45%	45%	45%
Sturgeon Creek	21%	21%	24%
Templeton-Sinclair	24%	24%	25%
The Maples	31%	31%	31%
Tissot and Central St. Boniface	20%	20%	21%
Tyndall Park	42%	42%	42%
Valley Gardens and Springfield South	29%	35%	35%
Vialoux	27%	27%	30%
Vista, Meadowood, and Minnetonka	25%	25%	30%
Waverley Heights	26%	26%	29%
Whyte Ridge	30%	30%	30%
Windsor Park and Southdale	30%	30%	34%
Wolseley	32%	32%	32%
Woodhaven	56%	56%	56%
Winnipeg	23%	25%	25%

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Appendix D Pest Vulnerability Matrix

Table D1. Pests removed from Laćan and McBride's (2008) original Pest Vulnerability Matrix, developed in California, to better represent Winnipeg's urban forest

Pest	Reason for Removal
Abutilon mosaic virus: *Begomovirus* spp	No target species in Winnipeg forest
Ash dieback on Raywood ash and *Botryosphaeria* canker: *Botryosphaeria stevensii* Shoemaker	No target species in Winnipeg forest
Chestnut Blight: *Cryphonectria parasitica* Murrill	No target species in Winnipeg forest
Chinese Elm Anthracnose: *Stegophora ulmea* Fr.	No target species in Winnipeg forest
Dematophora root rot: *Rosellinia* spp	No record in Manitoba
Diamond scale: *Sphaerodothis neowashintoniae* Shear	No target species in Winnipeg forest
Drippy Oak and Drippy Nut Disease: *Brenneria quercina* Hildebrand	No target species in Winnipeg forest
Gray mold or *Botrytis* petal blight: *Botrytis* spp	Might affect dogwoods, but favors moister climates
Pitch Canker: *Fusarium circinatum* Nirenberg	No target species in Winnipeg forest
Sudden Crown Drop: *Thielaviopsis paradoxa* Dade	No target species in Winnipeg forest
Tar spot: *Rhytisma arbuti* Phillips	Maples can host, but no record in Manitoba
Greenhouse thrips: *Thysanoptera* spp	No target species in Winnipeg forest
Golden mealybug: *Nipaecoccus aurilanatus* Maskell	No target species in Winnipeg forest

Longtailed mealybug, Obscure mealybug: *Pseudococcus longispinus* Targioni Tozzetti, *Pseudococcus viburni* Signoret	No target species in Winnipeg forest
Pearleaf blister mite: *Phytoptus pyri* Pagenstecher	No target species in Winnipeg forest
Incense cedar scale: *Xylococculus macrocarpae* Coleman	Only occurs south of Oregon. No record in Manitoba.
Shothole borer: *Scolytus rugulosus* Müller	No record in Manitoba
Ground mealybug: *Rhizoecus falcifer* Kunckel d'Herculais	No target species in Winnipeg forest
Brown tea root disease: *Phellinus noxius* Corner	No target species in Winnipeg forest
Seiridium canker: *Seiridium cardinale* Wagener	No target species in Winnipeg forest
Cycas weevils: *Belidae*	No target species in Winnipeg forest
Ash dieback on Fraxinus excelsior: *Hymenoscyphus fraxineus* Baral	No target species in Winnipeg forest
Mediterranean pine engraver: *Orthotomicus erosus* Wollaston	No record in Manitoba
Goldspotted oak borer: *Agrilus auroguttatus* Schaeffer	Native to Mexico, recently arrived in California. No record in Manitoba.
Redhaired pine bark beetle: *Hylurgus ligniperda* F.	No record in Manitoba
Thousand Cankers Disease: *Geosmithia morbida* Kolařík	No record in Manitoba
Sooty Canker: *Hendersonula toruloides* Nattrass	No target species in Winnipeg forest
Myrtle rust: *Austropuccinia psidii* G. Winter	No target species in Winnipeg forest

Myoporum thrips: *Klambothrips myopori* Mound and Morris	No target species in Winnipeg forest
Tabebuia thrips: *Holopothrips tabebuia* Cabrera and Segarra	No target species in Winnipeg forest
Tipu psyllid: *Platycorypha nigrivirga* Burckhardt	No target species in Winnipeg forest

Table D2. Pests added to Lacan and McBride's (2008) original Pest Vulnerability Matrix, developed in California, to better represent Winnipeg's urban forest

Pest	Reason for Addition
Ash Flower Gall Mites: *Eriophyes fraxiniflora* Felt.	Present in Winnipeg
Two Lined Chestnut Borer: *Agrilus bilineatus* Weber	Present in Winnipeg
Black Knot: *Apiosporina morbosa* Shw.	Present in Winnipeg
Spruce Budworm: *Choristoneura fumiferana* Clemens	Present in Winnipeg
Spongy Moth: *Lymantria dispar* L.	Watch listed in Winnipeg
Cottony Ash Psyllid: *Psyllopsis discrepans* Flor	Watch listed in Winnipeg
Spring and Fall Cankerworm: *Paleacrita vernata* Peck, *Ahophila pometaria* L.	Present in Winnipeg
Forest Tent Caterpillar: *Malacosoma disstria* Hbn.	Present in Winnipeg
Elm Spanworm: *Ennomos subsignaria* (Hbn.)	Present in Winnipeg

Spiny Elm Caterpillar: *Nymphalis antiopa* L.	Present in Winnipeg

Table D3. Pest inventory used for the Winnipeg PVM showing number of hosts impacted by each pest by severity. An asterisk indicates pests not currently present in Winnipeg.

Pest	**Host Count by Impact**			**For more information about this pest:**
	Severe	Moderate	Low	
Anthracnose: *Apiognomonia; Cylindrosporium; Marssonia; Glomerella; Colletotrichum*	1	1	4	Douglas (2011)
***Armillaria* root rots**	5	11	0	Guillaumin, and Legrand (2013)
Dutch elm disease: *Ophiostoma ulmi* Buisman	1	0	0	Brasier (2000)
Cottony Ash Psyllid: *Psyllopsis discrepans* Flor	1	0	0	Hodkinson (2009)
Emerald Ash Borer: *Agrillus planipennis* Fairmaire	1	0	0	Herms and McCullough (2014)
Asian Longhorned Beetle*: *Anoplophora glabripennis* Motschulsky	11	0	0	Meng, Hoover, and Keena (2015)
Elm bark beetles: *Hylurgopinus rufipes* Eichh. *Seolytus multistriatus* Marsham,	1	0	0	Santini and Faccoli (2015)

Bronze Birch Borer: ***Agrilus anxius*** **Gory**	1	1	0	Muilenburg and Herms (2012)
Two Lined Chestnut Borer: ***Agrilus bilineatus*** **Weber**	1	0	0	Cote & Allen (1976); Muzika, Liebhold, and Twery (2000)
Bacterial leaf scorch: ***Xylella fastidiosa*** **Wells**	0	5	2	Sherald (2007)
Fireblight: ***Erwinia amylovora*** **Burrill**	0	3	0	Thomson (2000)
Hackberry dieback*: ***Mollicutes***	0	1	0	Poole et al. 2021
Cytospora	0	1	4	Kepley& Jacobi (2000)
Hypoxylon **and** ***Nectria*** **cankers**	0	0	10	Ostry (2013); Sakamoto et al. (2004)
Leaf spots and leaf blights; many species	0	0	6	Rai and Mamatha (2005)
Pinewood nematode: ***Bursaphelenchus xylophilus*** **Nickle**	0	2	0	Kim et al. (2020)
Powdery Mildews: ***Erysiphales***	0	1	15	Bert et al. (2016); Turechek et al. (2005)
Verticillium wilt: ***Verticillium albo-atrum*** **V.aa and** ***Verticillium dahliae*** **V.d**	0	9	0	Heimstra (1998); Keykhasaber et al. (2018)

Ash Yellows/Ash decline: *Candidatus* Phytoplasma fraxinii Griffiths	0	1	0	Bricker and Stutz (2004)
Spongy Moth*: *Lymantria dispar* L.	0	9	0	Davidson et al. (1999); Naidoo and Lechowicz (2001)
Spruce Budworm: *Choristoneura fumiferana* Clemens	0	1	0	Holsten (2011)
Black Knot: *Apiosporina morbosa* Shw.	0	1	0	Wilcox (1992)
Adelgids: *Adelges abietis* L., *Pineus strobi* Htg., *Adelges tsugae* Annand	0	1	3	Hain (1988); McClure (1996)
Waxy aphids: *Eriosoma, Stegophila, Prociphilus fraxinifolii* Riley, *Shivaphis celti* Das	0	1	4	Cranshaw (2011); Halbert and Choate (1999)
Other bark beetles: *Dendroctonus, Scolytus, Ips*	0	0	2	Six and Bracewell (2015); Smith and Hulcr (2025;
Alder flea beetle: *Macrohaltica ambiens* LeConte	0	1	0	Randall (2005)
Boxelder bugs: *Boisea trivittatus* Say	0	1	1	Peairs (2003)

Elm leaf beetle: *Pyrrhalta luteola* Muller	0	2	0	Miller and Ware (1992)
Clearwing borers: *Sesiidae*	0	2	4	Taft et al. (1991)
Spider mites: *Tetranychidae*	0	1	12	Pritchard and Baker (1952)
Diplodia tip blight: *Sphaeropsis sapinea* Dyko & B. Sutton	0	1	0	Koetter and Grabowski (2019)
Other native borers: *Lyctidae, Anobiidae, Cerambycidae*	0	4	10	Chiappini and Aldini (2011)
Sirococcus tip blight: *Sirococcus conigenus* Pers.	0	1	0	Kowalski (2010)
***Ceratocystis* canker**	0	1	0	Hinds (1972)
Ash Flower Gall Mites: *Eriophyes fraxiniflora* Felt.	0	0	1	Ascerno (1990)
Bacterial blight, bacterial canker, and bacterial blast: *Pseudomonas syringae* Van Hall and others	0	0	5	Bultreys and Kaluzna (2010); Burdekin (1972); Malvick (1988)
Crown gall: *Agrobacterium tumefaciens* Beijerinck and van Delden	0	0	4	Epstein et al. (2008); Garrett (1987)
***Fusarium* wilt and canker**	0	0	2	Okungbawa and Shittu (2012); Boyer (1961)

Heart rot and decay fungi: *Ganoderma* and others	0	0	1	Vasaitis (2013)
Root and crown rot: *Phytophthora* or *Pythium*	0	0	12	Ellis (2008)
Nematodes (Cyst, root knot, root lesion): *Meloidogyne, Xiphinema*	0	0	6	Elling (2013); Forge et al. (2021)
Scab: *Venturia*	0	0	4	Belete and Boyraz (2017);
Fungal shot hole: *Blumeriella jaapii* Rehm and others	0	0	1	Clement (2022); Park and Kim (2019)
Sudden oak death: *Phytophthora ramorum* Werres	0	0	2	Grunwald et al. (2019)
Twig blights: *Cryptocline, Discula, Kabatina, Phomopsis juniperovora* Hahn	0	0	3	Peterson (1982); Hecht-Poinar et al. (1989)
Spiny Elm Caterpillar: *Nymphalis antiopa* L.	0	0	6	Baker (2020)
Elm Spanworm: *Ennomos subsignaria* (Hbn.)	0	0	5	Fry et al. (2008)
Forest Tent Caterpillar: *Malacosoma disstria* Hbn.	0	0	21	Meeker, 2014
Spring and Fall Cankerworm: *Paleacrita*	0	0	5	LaFrance and Westwood (2006)

***vernata* Peck, *Ahophila pometaria* L.**				
Other defoliating *Lepidoptera* larvae	0	0	21	Mason (1987)
Lace bugs: *Corythucha*	0	0	5	Halbert and Meeker (2004
Armored scales: *Hemiberlesia lataniae* Signoret; *Diaspidiotus perniciosus* Comstock; *Lepidosaphes ulmi* L.; *Quadraspidiotus juglansregiae* Comstock and others	0	0	13	Miller and Davidson (2005)
Soft scales: *Coccus hesperidum* L., *Pulvinaria innumerabilis* Rathvon;; *Gossyparia spuria* Modeer; *Parthenolecanium fletcheri* Cockerell and others	0	0	9	Mahr (2020)
Blister gall mites: *Vasates laevigatae* (Hassan) and others	0	0	2	Davis and Beddes (2011)
Cottony alder psyllid: *Psylla alni L.*	0	0	1	Heslop-Harrison (1960)
***Cynipidae* gall wasps**	0	0	1	Egan et al. (2018)

Leafblotch miner and cypress tip miner: *Gracillariidae and Argyresthia cupressella* Walsingham	0	0	2	Auerbach and Alberts (1992); Antonelli and Foss (2003)
Gall midges: *Cecidomyiidae*	0	0	3	Stuart et al. (2012)
Whiteflies: *Aleyrodidae*	0	0	2	Pickett and Pitcairn (1999)
Shield bearers: *Coptodisca*	0	0	3	Brown (1990)
Other *Psylloidea*	0	0	1	Kabashima et al. (2014)
Silverspotted tiger moth and Tussock moths: *Halisidota argentata* Pack. and *Lymantriinae*	0	0	3	Duncan (1992); Cranshaw et al. (2009)
Willow gall sawflies: *Euura* and *Pontania*	0	0	1	Hjalten and Price (1996)
Eriophyid mites: *Aceria* and *Vasates*	0	0	3	Keifer (1982)
Juniper scale: *Carulaspis juniperi* Bouché	0	0	2	Dawasi and Addesso (2012)
Leaf beetles and flea beetles: *Altica, Chrysomela., Plagiodera*	0	0	2	Jolivet (2002); Parryl (1986)

Pine needle scale: *Chionaspis pinifoliae* Fitch	0	0	2	Tooker and Hanks (2000)
Conifer sawflies: *Megalodontoidea* and *Tenthredinoidea*	0	0	6	Haack and Mattson (1993)
Leafminers, including Apple-and-thorn skeletonizer: *Choreutis pariana* Clerk	0	0	4	Doganlar and Beirne (1981)
Mealybugs: *Pseudococcidae*	0	0	4	Flint (2016)
Western gall rust: *Endocronartium harknessii* Moore	0	0	1	Adams (1997)

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Table D4. Taxon pest scores for Winnipeg tree inventory.

Tree Taxa	Pest Score	Tree Taxa (cont.)	Pest Score
Juniper (*Juniperus* spp.)	10	Oak (*Quercus* spp.)	27
Cedar (*Thuja* spp.)	11	Linden (*Tilia* spp.)	10
Pine (*Pinus* spp.)	16	Birch (*Betula* spp.)	20
Larch (*Larix* spp.)	8	Alder (*Alnus* spp.)	19
Fir (*Abies* spp.)	7	Willow (*Salix* spp.)	27
Spruce (*Picea* spp.)	15	Poplar, Cottonwood, Aspen (*Populus* spp.)	30
Box elder (*Acer negundo*)	16	American Elm (*Ulmus americana*)	29
Other maples (*Acer* spp.)	29	Siberian Elm (*Ulmus pumila*)	16
Ohio Buckeye (*Aesculus glabra*)	8	Hackberry (*Celtis* spp.)	15
Ash (*Fraxinus* spp.)	30	Apple, Crabapple (*Malus* spp.)	20
Dogwood (*Cornus* spp.)	15	Cherry and Plum (*Prunus* spp.)	16
Catalpa (*Catalpa* spp.)	7	Hawthorn (*Crataegus* spp.)	12
Walnut (*Juglans* spp.)	16	Russian Olive (*Elaeagnus* spp.)	6
Mountain ash (*Sorbus* spp.)	11		

Table D5. Neighborhood pest scores for Winnipeg tree inventory

Neighborhood	Neighborhood Pest Score	Neighborhood (cont.)	Neighborhood Pest Score
Alpine Place and St. George	22	Minto	18
Armstrong Point	25	Mission Gardens	24
Birchwood and Bruce Park	27	North Point Douglas and William Whyte	27
Bridgwater Forest	16	North River Heights	20
Bridgwater Trails and Bridgwater Centre	22	Norwood West and Norwood East	23
Broadway-Assiniboine	25	Old Tuxedo	23
Brockville and Linden Woods	17	Pembina Strip, Beaumont, and Maybank	23
Brooklands	22	Richmond Lakes and Parc La Salle	18
Burrows Central and Shaughnessy Park	19	Riverbend	20
Canterbury Park	21	Rivergrove	17
Chalmers	21	River-Osborne	27
Colony and West Broadway	26	Rockwood	22
Crescent Park	22	Rossmere-A and Rossmere-B	19
Crescentwood	22	Sargent Park	24

Dakota Crossing	18	Seven Oaks	21
Daniel Mcintyre	27	Silver Heights and Deer Lodge	17
Eaglemere	23	South Pointe	19
Earl Grey	21	South River Heights	20
East Elmwood	21	South Tuxedo	18
Ebby-Wentworth and Grant Park	22	Southboine	21
Edgeland and Sir John Franklin	22	Spence	26
Elmhurst	19	Springfield North	21
Fort Richmond	17	St. John's	23
Glendale	20	St. Matthews	27
Glenelm	20	Sturgeon Creek	19
Glenwood	21	Templeton-Sinclair	18
Grassie	23	The Maples	20
Inkster-Faraday and Jefferson	25	Tissot and Central St. Boniface	20
J. B. Mitchell	21	Tyndall Park	22
Kern Park and Melrose	21	Valley Gardens and Springfield South	21
Kildare-Redonda	19	Vialoux	20
King Edward	21	Vista, Meadowood, and Minnetonka	21
Lord Roberts and Riverview	22	Waverley Heights	19

Luxton	26	Whyte Ridge	14
Margaret Park	22	Windsor Park and Southdale	17
Mathers	24	Wolseley	22
Mcmillan	23	Woodhaven	24
Meadows and Radisson	21	City of Winnipeg	21

Appendix E Zoning designations included for neighborhood carbon storage calculations

APARTMENTS	MULTIFAMILY CONVERSION
APARTMENTS MULTIPLE USE	MULTIPLE ATTACHED UNITS
COMMERCIAL ROW HOUSE	NURSING HOME
CONDO APARTMENT	RESIDENTIAL GROUP CARE
CONDO COMMERCIAL	RESIDENTIAL MULTIPLE USE
CONDO COMPLEX	RESIDENTIAL SECONDARY UNIT
CONDO INDUSTRIAL	ROOMING HOUSE
CONDO RESIDENTIAL	ROW HOUSING
CONDO VACANT*	SIDE BY SIDE
DETACHED SINGLE DWELLING	TRIPLEX
DUPLEX	VACANT RESIDENTIAL 1
MOBILE HOME	VACANT RESIDENTIAL 2

Appendix F i-Tree Biomass Equations

Table F1. i-Tree biomass equations used to estimate carbon, formatted for use in Microsoft Excel.

Species	Biomass Equation
Abies balsamifera	0.27965 * (DBH^2.04308)
Acer macrophyllum	EXP(-1.536895+(2.24355*(LN(DBH))+(0.0315/2)))
Acer rubrum	EXP(-1.84135+(2.36499*(LN(DBH))+(0.00913/2)))
Acer saccharinum	0.17789 * ((DBH^2*HEIGHT)^0.8467)
Acer saccharum	EXP(-1.46455+(2.3042*(LN(DBH))+(0.01354/2)))
Acer spicatum	EXP(-1.536895+(2.24355*(LN(DBH))+(0.0315/2)))
Alnus spp	(0.2896*DBH^2)-5.5963
Betula papyrifera	EXP(-2.41197+(2.56847*(LN(DBH))+(0.03474/2)))
Cornus spp	EXP(-1.98379+(2.38367*(LN(DBH))+(0.0381/2)))
Fagus grandifolia	EXP(-1.36313+(2.27798*(LN(DBH))+(0.0106/2)))
Fraxinus americana	EXP(-1.8446+(2.3762*(LN(DBH))+(0.05731/2)))
Fraxinus nigra	EXP(-1.905+(2.29776*(LN(DBH))+(0.08518/2)))
Fraxinus pennsylvanica	EXP(-2.19052+(0.8403*(LN(DBH^2*HEIGHT))+(0.13692/2)))
Juniperus spp	0.1632*DBH^2.2454
Picea abies	EXP(-1.77212+(2.25022*(LN(DBH))+(0.06069/2)))
Picea glauca	EXP(-1.73798+(2.22809*(LN(DBH))+(0.05189 /2)))
Picea rubens	EXP(-1.75175+(2.23587*(LN(DBH))+(0.05094/2)))
Picea spp	EXP(-1.87821+(2.25867*(LN(DBH))+(0.04823 /2)))
Pinus banksiana	EXP(-2.00907+(0.7914*(LN(DBH^2*HEIGHT))+(0.29405 /2)))

Pinus contorta	0.11886 *DBH^ 2.2333
Pinus echinata	0.01512 * (DBH^2*HEIGHT)^0.99415
Pinus elliottii	0.01865 * (DBH^2*HEIGHT)^0.97777
Pinus palustris	0.02455 * (DBH^2*HEIGHT)^0.95612
Pinus resinosa	EXP(-1.9363+(2.2825*(LN(DBH))+(0.0573/2)))
Pinus strobus	EXP(-2.82175+(2.42377*(LN(DBH))+(0.02545/2)))
Pinus resinosa	EXP(-1.9363+(2.2825*(LN(DBH))+(0.0573/2)))
Populus spp	EXP(-2.28909 +(2.44837*(LN(DBH))+(0.01442/2)))
Populus tremuloides	EXP(-2.51459+(2.4573*(LN(DBH))+(0.06754 /2)))
Prunus pensylvanica	EXP(-2.0349+(2.42467*(LN(DBH))+(0.05423/2)))
Prunus serotina	EXP(-2.00442+(2.44771*(LN(DBH))+(0.03475/2)))
Quercus macrocarpa	EXP(-2.38644 +(2.49236*(LN(DBH))+(0.06595/2)))
Thuja spp	EXP(-1.78066+(1.9944*(LN(DBH))+(0.09031/2)))
Tilia americana	EXP(-2.42943+(2.35806*(LN(DBH))+(0.25912/2)))
Ulmus americana	EXP(-2.22755+(2.39866*(LN(DBH))+(0.0602/2)))

Table F2. Equations and sources for species used to calculate predicted values where height was included in the i-Tree sources, rendering the i-Tree equation unusable in our confidence calculation methods.

Species	**Equation**	**Source**
Fraxinus spp	2.669*(((DBH1*0.393701)^2)^1.16332)	McPherson et al. (2016), citing Clark et al. (1985)
Acer negundo	(0.0019421 × DBH^1.785)*420	McPherson et al. (2016), citing Lefsky and McHale (2008)

Ulmus pumila	(0.0048879 * DBH^1.613)*540	McPherson et al. (2016), citing Zanne et al. 2009
Salix spp	0.17789 * ((DBH^2*HEIGHT)^0.8467)	McPherson (2016), citing McHale (2009) citing Standish et al. (1985)

Table F3. Trees observed in this study with no corresponding equation in i-Tree. Nowak et al. (2020) report that i-Tree calculates biomass for these trees by taking the mean of all values for biomass in a corresponding genus (or other closest taxa).

Observations	**i-Tree equations averaged**
Acer ginnala, Acer spicatum	*Acer macrophyllum, Acer rubrum, Acer saccharinum, Acer saccharum*
Aesculus glabra	*Acer macrophyllum, Acer rubrum, Acer saccharinum, Acer saccharum*
Celtis occidentalis	*Betula papyrifera, Fagus grandifolia*
Crataegus spp	*Prunus pensylvanica , Prunus serotina*
Elaeagnus angustifolia	*Betula papyrifera, Fagus grandifolia*
Fraxinus mandschurica	*Fraxinus americana , Fraxinus nigra, Fraxinus pennsylvanica*
Juglans spp	*Betula papyrifera, Fagus grandifolia*
Juniperus spp	*Juniperus virginiana*
Larix spp	*Picea glauca, Picea abies, Picea rubens, Pinus banksiana, Pinus contorta, Pinus echinata, Pinus elliottii, Pinus palustris, Pinus resinosa, Pinus strobus*
Malus spp	*Prunus pensylvanica , Prunus serotina*
Pinus spp, Pinus sylvestris	*Pinus banksiana, Pinus contorta, Pinus echinata, Pinus elliottii, Pinus palustris, Pinus resinosa, Pinus strobus*

Populus spp	*General populus equation*
Prunus spp	*Prunus pensylvanica , Prunus serotina*
Sorbus spp	*Prunus pensylvanica , Prunus serotina*
Tilia cordata	*Tilia americana*

Appendix G Carbon storage (T) for surveyed residential neighbourhoods

Table G1. Carbon storage (T) for surveyed residential neighborhoods, as reported for i-Tree and as estimated using methods from Wayson (2015).

Amalgamated Neighborhood	Estimated i-Tree Carbon (t)	Estimated Total Carbon (t): Predicted Average	Estimated Total Carbon (t): 95% confidence	Estimated Total Carbon (t): 80% confidence	Estimated Total Carbon (t): 20% confidence	Estimated Total Carbon (t): 5% confidence
Alpine Place and St. George	753	903	1496	1289	621	544
Armstrong Point	482	492	954	792	257	174
Birchwood and Bruce Park	1737	2047	4911	3910	989	673
Bridgwater Forest	13	16	211	143	7	6
Bridgwater Trails and Bridgwater Centre	11	18	533	353	5	5
Broadway-Assiniboine	362	359	808	651	173	115
Brockville and Linden Woods	587	752	1660	1342	411	364
Brooklands	1105	802	1179	1047	652	596
Burrows Central and Shaughnessy Park	2910	3268	4787	4256	2752	2612
Canterbury Park	376	920	1370	1212	781	728
Chalmers	2553	2728	4460	3854	1985	1743
Colony and West Broadway	2131	2299	4552	3764	1149	734
Crescent Park	956	992	2457	1945	418	312
Crescentwood	1088	1163	2722	2176	523	362
Dakota Crossing	547	881	1242	1116	724	670
Daniel Mcintyre	3594	4853	7596	6637	3382	2804
Eaglemere	59	119	200	171	79	62
Earl Grey	1235	1364	2791	2292	659	449

East Elmwood	699	679	1411	1155	316	209
Ebby-Wentworth and Grant Park	465	652	1328	1091	405	356
Edgeland and Sir John Franklin	1464	1939	3142	2721	1307	1094
Elmhurst	421	621	1131	952	453	415
Fort Richmond	1754	1841	3405	2858	1326	1149
Glendale	149	136	283	232	97	85
Glenelm	530	571	1090	908	286	169
Glenwood	2381	2341	3874	3338	1691	1460
Grassie	218	402	644	560	281	232
Inkster-Faraday and Jefferson	3830	4826	8955	7511	2679	2161
J. B. Mitchell	268	321	504	440	236	208
Kern Park and Melrose	982	1183	2025	1730	735	635
Kildare-Redonda	1068	1315	2380	2008	831	715
King Edward	1926	1630	2908	2461	1053	842
Lord Roberts and Riverview	4230	4334	9592	7753	1492	792
Luxton	2411	2498	4640	3891	1192	669
Margaret Park	508	545	1016	851	337	277
Mathers	266	455	642	577	361	321
Mcmillan	1093	1238	2599	2123	555	381
Meadows and Radisson	2765	3657	6254	5346	2278	1924
Minto	2187	2199	4675	3809	793	367
Mission Gardens	152	288	577	476	221	203
North Point Douglas and William Whyte	2300	2736	4918	4155	1618	1239
North River Heights	4285	4391	9337	7607	1624	811
Norwood West and Norwood East	3503	3520	7258	5950	2083	1610

Old Tuxedo	994	1019	2622	2061	298	190
Pembina Strip, Beaumont, and Maybank	1829	2102	2952	2655	1662	1499
Richmond Lakes and Parc La Salle	832	874	1268	1130	704	642
Riverbend	645	762	1588	1299	398	312
Rivergrove	89	114	253	204	50	38
River-Osborne	494	541	1248	1000	225	132
Rockwood	1601	1732	4096	3269	659	503
Rossmere-A and Rossmere-B	3847	4639	8679	7266	2645	2065
Sargent Park	4202	4293	9181	7471	1421	540
Seven Oaks	1783	1824	3694	3040	930	669
Silver Heights and Deer Lodge	3399	3470	6334	5333	1932	1347
South Pointe	50	66	612	421	19	13
South River Heights	921	1089	2083	1735	629	535
South Tuxedo	609	789	1430	1206	544	474
Southboine	211	231	629	489	90	68
Spence	1452	1635	3258	2690	811	544
Springfield North	630	1062	2033	1693	740	672
St. John's	1752	1948	4116	3358	862	574
St. Matthews	2359	3275	5190	4520	2292	1943
Sturgeon Creek	608	752	1056	950	625	581
Templeton-Sinclair	316	425	1003	801	252	214
The Maples	2191	2742	3542	3262	2366	2198
Tissot and Central St. Boniface	1504	1529	2243	1993	1176	1042
Tyndall Park	958	1453	2208	1944	1118	994
Valley Gardens and Springfield South	1327	2044	2885	2591	1579	1430

Vialoux	135	154	402	315	74	59
Vista, Meadowood, and Minnetonka	2806	3026	6709	5421	1711	1414
Waverley Heights	592	841	1396	1201	578	504
Whyte Ridge	496	519	1471	1138	301	288
Windsor Park and Southdale	2443	2775	4689	4019	1923	1683
Wolseley	3441	3498	6448	5416	1890	1256
Woodhaven	552	568	1464	1151	228	151

Appendix H City of Winnipeg Acceptable Tree Species for Planting

Table H1. Large Sized Trees (mature height 15 m or greater) - Deciduous

Note: Until further notice, Fraxinus (ash species and cultivars) will not be considered for planting due to the high risks associated with emerald ash borer.

Species/cultivar name	Common Name	Special note
Acer negundo 'Baron'	Baron Manitoba maple	Seed less cultivar of Manitoba maple
Acer saccharinum	silver maple and various cultivars	
Quercus macrocarpa	bur oak	
Populus x canadensis 'Prairie Sky'	Prairie Sky poplar	Parks and green spaces only; seedless
Populus deltoides	Cottonwood	Parks and green spaces only; seedless cultivars available
Salix pentandra	laurel leaf willow	Better suited to parks and green spaces
Tilia americana	basswood / American linden and various cultivars	
Ulmus americana	American elm	
Ulmus americana 'Brandon'	Brandon American elm	
Ulmus omericono 'Lewis and Clark'	Prairie Expedition elm	DED tolerant
Ulmus x 'Morton Glossy'	Triumph hybrid elm	DED tolerant
Ulmus pumila	Siberian elm	Limited use

Table H2. Large Sized Trees (mature height 15 m or greater) - Coniferous

Species/cultivar name	Common Name	Special note
Larix siberica	Siberian larch	Parks and green spaces only
Picea glauca ‘Densata’	Blackhills white spruce	Parks and green spaces only
Picea pungens	Colorado spruce — multiple cultivars	Parks and green spaces only
Pinus sylvestris	Scots pine	Parks and green spaces only

Table H3. Medium Sized Trees (mature height 9 m to 15 m) — Deciduous

Species/cultivar name	Common Name	Special note
Acer saccharum 'Jefselk'	Lord Selkirk sugar maple	Experimental
Aesculus glabra	Ohio buckeye	
Alnus hirsuta 'Harbin'	Prairie Horizon alder	Limited use for boulevards
Betulo popyrifero	Paper birch	Better suited to parks and green spaces
Betula papyrifera 'Varen'	Prairie Dream birch	Better suited to parks and green spaces
Celtis occidentalis 'Delta'	Delta hackberry	
ivlalus baccata	Siberian crabapple	
Juglans cinerea	Butternut	Better suited to parks and green spaces
Ulmus davidiana japonica 'Discovery'	Discovery elm	DED tolerant
Ulmus davidiana japonica 'Night Rider'	Night Rider elm	DED tolerant
Tilia cordata 'Golden Cascade'	Golden Cascade linden	
Tilia cordata 'Ronald'	Norlin littleleaf linden	
Tilia cordata 'Green Spire'	Green Spire littleleaf linden	
T. x flavescens 'Dropmore'	Dropmore linden	

T. x flavescens 'Glenleven'	Glenleven linden	
T. mongolica 'Harvest Gold'	Harvest Gold linden	

Table H4. Medium Sized Trees (mature height 9 m to 15 m) - Coniferous

Species/cultivar name	Common Name	Special note
Pinus cembra	Swiss stone pine	Parks and green spaces only; slow-growing

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Table H5. Small Sized Trees (mature height less than 9 m) - Deciduous

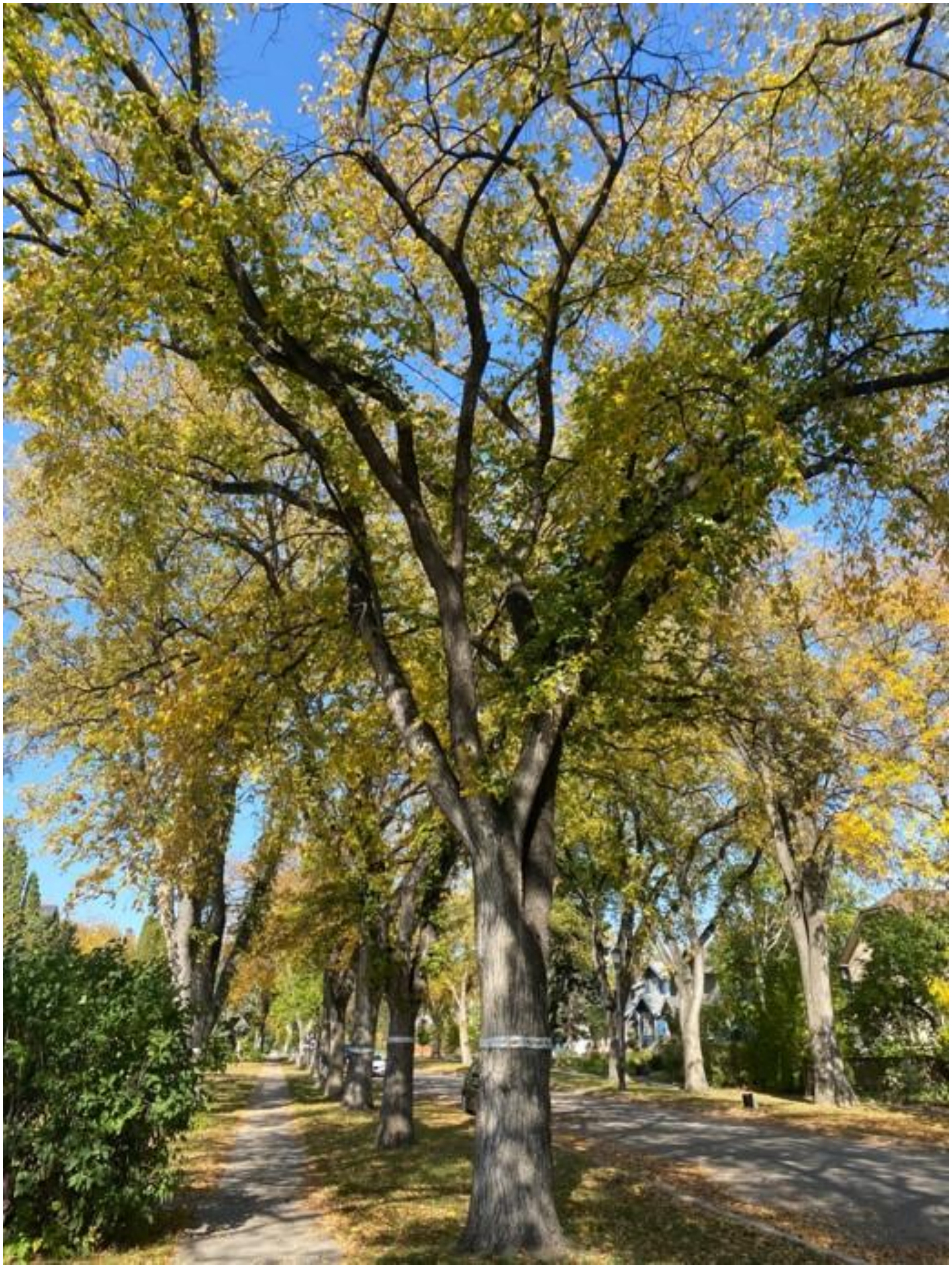
Species/cultivar name	Common Name	Special note
Acer ginnala	Amur maple	Tree form for boulevards
Acer ginnala 'Embers'	Embers Amur maple	Tree form for boulevards
Acer ginnala 'Ventura'	Ventura Amur maple	Tree form for boulevards
Acer tatarica 'GarAnn'	Hot Wings Tatarian maple	Tree form for boulevards
Crataegus x mordenensis 'Snowbird'	Snowbird hawthorn	Better suited to parks and green spaces
Crataegus x mordenensis ‘Toba’	Toba hawthorn	Better suited to parks and green spaces
Prunus maacki	Amur cherry	
Prunus maacki 'Goldrush'	Goldrush Amur cherry	
Prunus x‘Ming’	Ming Amur cherry	
Pyrus x ‘DurPSN303’	Navigator ornamental pear	

Sorbus aucuparia Rossica’	Russian mountainash	Limited use for boulevards
Sorbus decora	Showy mountainash	Limited use for boulevards
Syringa reticulata	Japanese tree lilac	
Syringa reticulata 'Ivory Pillar'	Ivory Pillar Japanese tree lilac	
Syringa reticulata 'Ivory Silk'	Ivory Silk Japanese tree lilac	
ivlalus x adstringens 'Durleo'	Gladiator rosybloom crabapple	Narrow columnar crown
ivlalus x adstringens 'Kelsey'	Kelsey rosybloom crabapple	
ivlalus x adstringens 'Pink Spires'	Pink Spires rosybloom crabapple	Narrow columnar crown
ivlalus x adstringens 'Selkirk'	Selkirk rosybloom crabapple	
ivlalus baccata 'Jeflite'	Starlite flowering crabapple	Very small fruit
ivlalus baccata 'Spring Snow'	Spring Snow flowering crabapple	No fruit

Table H6. Small Sized Trees (mature height less than 9 m) - Coniferous

Species/cultivar name	Common Name	Special note
Juniperus scopulorum ‘Medora’	Medora upright juniper	Parks and green spaces only
Thuja occidentalis and specific cultivars	Eastern white cedar pyramidal tree-form and pyramidal tree-form cultivars	Parks and green spaces only

Appendix I Photographs of tree surveys from University of Winnipeg course BIOL4475, Urban Forestry, courtesy of Dr. Richard Westwood



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