

MANAGING AND ENHANCING URBAN TREE DIVERSITY: A COMPARISON OF
SUBURBAN DEVELOPMENT IN TWO CANADIAN CITIES

by

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WHAT mystery pervades a well!
The water lives so far,
Like neighbor from another world
Residing in a jar.

The grass does not appear afraid;
I often wonder he
Can stand so close and look so bold
At what is dread to me.

Related somehow they may be,—
The sedge stands next the sea,
Where he is floorless, yet of fear
No evidence gives he.

But nature is a stranger yet;
The ones that cite her most
Have never passed her haunted house,
Nor simplified her ghost.

To pity those that know her not
Is helped by the regret
That those who know her, know her less
The nearer her they get.

— Emily Dickinson

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ABSTRACT

Given the widespread suburbanization of Canadian cities, it is crucial to explore the mechanisms by which subdivision development influences forest composition. This study assessed whether suburban neighbourhoods in two cities with contrasting pre-urbanized landscapes (woodlands and farm fields) exhibited differences in tree diversity losses or gains due to development. Trees were sampled in neighbourhoods representing two development decades and three land types: remnant woodland, streetscapes, and residential properties. Changes in species-selection decisions likely explain diversification in newer streetscapes in both cities. Older residential properties located adjacent to remnant forest stands were dominated by native trees, alluding to species dispersal across green spaces over time and the ecological importance of retaining woodland during development. Instead of simply aiming to maximize biodiversity in suburban areas, practitioners and stewards should acknowledge differences in forest composition based on land use evolution and tenure, and embrace flexibility and adaptation when shaping and managing tree-species diversity.

LIST OF ABBREVIATIONS USED

UFMP	URBAN FOREST MASTER PLAN
ON	ONTARIO
NS	NOVA SCOTIA
DBH	DIAMETER AT BREAST HEIGHT
HRM	HALIFAX REGIONAL MUNICIPALITY

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CHAPTER 1 INTRODUCTION

The world has become increasingly urbanized; over half of the global population lives in urban areas, which are expected to house two-thirds of the total population by 2050 (United Nations 2014). The tremendous current and projected growth of cities worldwide underscores the importance of sustainable urban management, in efforts to maximize the benefits of urban living while mitigating unwanted environmental, economic, and social consequences (Seto et al. 2011). The protection and creation of green infrastructure alongside the grey is understood to be crucial to the resilience of cities, and scholars, policy-makers, and planners alike have sought to develop and implement strategies meant to enhance services delivered by urban green spaces and associated vegetation, including trees (Hostetler et al., 2011; Tzoulas et al., 2007; Jim 2004).

Urban forests, which generally include any tree found within an urban area, provide numerous and multi-faceted benefits to urban dwellers, including (but certainly not limited to) reductions in energy and infrastructure maintenance costs, aesthetic enhancements, mitigation of the urban heat island effect, food sources, improvements in physical and psychological well-being, and recreational and learning opportunities (Duinker et al. 2015). Importantly, urban forests promote biodiversity in the city, arguably both an ecological good and a necessity for healthy, lasting green infrastructure. Species-rich forests play a valuable role in sustaining the many ecosystem services provided by trees, and have proven less vulnerable to environmental perturbations like the introduction of pests or diseases (Alvey 2006; Stone et al. 1996). Scholars have also noted that sufficient native tree-species representation within the urban forest can provide

habitat for native wildlife and help protect unique ecosystems by fostering the establishment of regionally appropriate species (Turner et al. 2005). Indeed, native trees have been shown to more effectively promote bird and insect diversity compared to non-native or naturalized species (Brändle & Brandl 2001; Ilkin et al. 2013; Barth et al. 2015). It can also be argued that we have a responsibility to protect the native diversity of local ecosystems (Kendle & Rose 2000). Biodiversity conservation is now recognized as a vital component of urban forest management, and has been incorporated in various ways in urban forest targets and policies (Ordóñez & Duinker 2013).

Cities themselves can be quite heterogeneous, as different regions within urban areas are created and developed to suit different needs, including institutional, commercial, and residential functions. Accordingly, tree canopy cover, structure, and species composition may vary according to land tenure and use (Bourne & Conway 2014; Dobbs et al. 2013). Indeed, it has been suggested that species richness might peak in moderately developed habitats in suburban and peri-urban areas, located between the urban core and the countryside (Bourne & Conway 2014; Alvey 2006). This trend could be due in part to landscape and habitat heterogeneity created from initial urban sprawl, when the presence of roads, green space, remaining natural habitat, and residential yards may encourage species diversity (McKinney, 2002).

In Canada, over 80% of the population is concentrated in cities, and population growth in suburban and peri-urban areas is far outpacing population growth occurring elsewhere. Suburbanization itself has wide-reaching impacts, such as natural habitat loss and fragmentation, pollution, and ecosystem degradation (Turner et al., 2005). The negative biological consequences of subdivision development demonstrate the

importance of considering how gains in ecological services could be acquired by promoting biodiversity in suburban areas. Furthermore, the majority of Canadian citizens experience and appreciate the natural world within urban environments, so it is imperative to create sustainable urban forests and maintain their positive benefits.

Relatively little research has been carried out in Canada on how suburbanization influences forest composition. No study has empirically examined how development patterns shape urban forests in Canadian cities. This study addresses these gaps in the literature, particularly given the need to quantify different potential gains or losses in biodiversity that arise from developing onto more natural areas, like woodlands or fields. One of two objectives for this thesis was to identify a broad range of drivers of tree-species composition in suburban neighbourhoods. The second objective was to quantitatively assess how development shapes tree-species composition in Canadian suburban neighbourhoods.

The thesis is divided into three papers. The first, Chapter 2, is a conceptual paper based on a review of the literature outlining factors that influence tree-species composition in suburban neighbourhoods. The list of factors is by no means exhaustive, but serves as a guideline and a reference for analyzing the results in subsequent chapters. Chapter 3 focuses on the effects of contrasting development patterns, namely the type of landscape present prior to urbanization, on tree diversity in subdivision developments in two Canadian cities: Halifax, Nova Scotia (developed onto woodland) and London, Ontario (developed onto farm fields). The third paper, Chapter 4, discusses the influence of land type on tree-species composition and species dominance within the same suburban neighbourhoods examined in Chapter 3. Finally, Chapter 5 summarizes the

contributions of this thesis to the growing body of urban forestry work, and emphasizes the importance of contemplating how species patterns may differ between spaces and over time. A research agenda is also outlined based on the findings of this work, and calls on scholars to explore how urban design and targeted education initiatives could promote appropriate forest diversity in suburban neighbourhoods.

CHAPTER 2 TREE DIVERSITY IN SUBURBAN AREAS

2.1 ABSTRACT

Tree diversity is crucial to the urban forest. Diverse urban forests provide habitat for a wider range of organisms, increase resilience to pests and disease, and, in the case where native tree species are well represented, contribute to the protection of regional ecosystems. Studies have shown that tree diversity can peak in the low- to mid-density neighbourhoods found in suburban and peri-urban areas, emphasizing the potential for biodiversity enhancement during and after subdivision development. Most studies quantifying tree-species composition in suburban areas focus on one or two major drivers of tree diversity, such as land use, socioeconomics and demographics, or the presence of natural features like parks or greenways. Furthermore, relatively little attention has been paid to the drivers of diversity for a variety of land types making up the entire urban forest, which represent differences in tree planting and establishment practices, ownership, and maintenance. This paper presents an overview of drivers of tree-species composition based on the literature, as well as factors that are not well studied but should be, because they play a role in determining the structure of the (sub)urban forest. These factors are examined in the context of four land types (street, residential property, park, remnant woodland) and are organized under the following major themes: biophysical characteristics, demographics, administration and neighbourhood management, professional cultures and paradigms, and community design. Based on what is known so far, a research agenda is also presented outlining major areas where research on urban tree diversity is lacking. The information presented in this paper can thus serve as a

guideline to inform urban forest management practices in order to strategically enhance tree diversity.

2.2 INTRODUCTION

Over the last twenty years, biodiversity conservation and improvement have gained traction among researchers and policy-makers, given increasingly rapid rates of biodiversity loss globally (Alvey 2006). Efforts to protect biodiversity have made headway in urban ecosystems, as evidenced by research and urban planning strategies (Savard et al. 2000; Dearborn & Kark 2009; Goddard et al. 2010; Marzok et al. 2014). Urban forests can contain a significant number of tree species native to a particular locale, and have been targeted for diversity enhancement in many cities (Cornelis & Hermy 2004; Alvey 2006; Ordóñez & Duinker 2013). Nonetheless, biotic homogenization has been identified as a challenge to biodiversity improvements in urban areas, as exotic and human-adapted species become more abundant in anthropogenic landscapes (Tait et al. 2005; McKinney 2008).

The purpose of this paper is to review the literature on factors influencing tree-species composition in urban areas and organize findings based on general themes and pertinence to suburban areas. An urban planner, an urban forester, and an arborist from the city of Halifax, Nova Scotia, were also consulted to identify and elucidate factors that are not well studied in the literature. For the purpose of this work, the definition of tree-species composition or diversity includes species richness and numbers (or proportions) of native and non-native species. The suburbs were chosen for study because research has shown that residential developments in particular have the potential to house a large number of native and non-native plant species on private properties, and could thus be

effectively targeted for biodiversity enhancement (McKinney 2002; Turner et al. 2005). Suburban areas can also exhibit greater landscape heterogeneity compared to dense urban cores and commercial or industrial strips (McKinney 2002), and could therefore be subjected to a wider range of factors influencing tree diversity, as well as opportunities for biodiversity gains, than urban centres. Finally, suburban areas are found around the world and, in many cases, represent the regions where the most urban growth is occurring (Gordon & Shirokoff 2014).

The following sections discuss the importance of urban forest diversity and characterize the “suburban forest” and neighbourhood. The rest of the paper explores the factors that influence the composition of the suburban forest; these are organized into five themes and examined according to their effect on each setting. Three important temporal distinctions were considered when identifying and describing drivers of tree diversity: 1) the factors at play when suburban developments are first created, 2) the factors that might influence diversity over time as suburban areas are initially settled, and 3) the factors that play a role when developments become part of the urban cityscape, and when mature trees are maintained and replaced.

2.3 DIVERSITY IN THE URBAN FOREST

Biodiversity loss has been identified as one of the most pressing ecological problems, and is predicted to have long-lasting effects globally. The implementation of international treaties, like the Convention on Biological Diversity (1992, 2011), as well as the passing of national and regional laws (e.g. Canadian Species at Risk Act 2003), reflects a growing interest in understanding the causes and consequences of declines in biodiversity, as well as a need to develop solutions (Turner et al. 2005; Cardinale et al.

2012). Among many repercussions, biodiversity loss can hamper ecosystem functioning and stability, decrease ecosystem resilience to environmental perturbations, and deplete outputs from provisioning and regulating ecosystem services (Cardinale et al. 2012). Research and policies aimed at preserving biodiversity often involve protecting large, untouched expanses of natural landscapes. However, improving diversity in urban environments, particularly if native species are prioritized, can have many benefits at local and global scales by slowing rates of biotic homogenization (Alvey 2006; McKinney 2008).

Consistent with these trends, diversity has become a focus for urban forest research and management. Many studies have explored the benefits of enhancing biodiversity in urban areas, and diversity metrics have been incorporated into core tenets and goals for urban forest management plans (Ordóñez & Duinker 2013). Indeed, more-diverse urban forests tend to have greater ecosystem productivity, as well as increased resilience to environmental changes related to climate, invasion events, and diseases (Alvey 2006; Ordóñez & Duinker 2012; Bourne & Conway 2014). It has been shown that the dominance of one species over others increases the vulnerability of the urban forest to pest and disease outbreaks, and increases mass mortality rates of affected trees (Alvey 2006; Lacan & McBride 2008). Species-rich forests also allow for the establishment of other organisms (birds, insects, mammals, microbes) that contribute to food webs and local ecosystem dynamics (Halifax Regional Municipality [HRM] 2013). A diverse assemblage of fruit-bearing trees can be a convenient and valuable food source for urban communities (Clark & Nicholas 2013). The maintenance of structural diversity is essential in avoiding even-aged conditions, when trees along one street or in one

neighbourhood reach the end of their life cycle at roughly the same time, or are ravaged by a species-specific pest (Steenberg et al. 2013). Diverse urban forests can also encourage positive conservation attitudes and educate residents about natural features and processes (Bourne & Conway 2014).

Similar arguments can be made for conserving native biodiversity in urban settings. Larger numbers of native tree species promote native bird, mammal, and insect diversity by providing suitable wildlife habitats within the city (Kendle & Rose 2000; Ilkin et al. 2013; Barth et al. 2015). From an ethical standpoint, one can argue for the importance of preserving the uniqueness of local and regional ecosystems, particularly if some native species are otherwise threatened or at risk (HRM 2013). Finally, the dominance of non-native and/or naturalized species over native species is a sign of decreasing ecological integrity due to the potential for invasion events and endemic species extirpation (Turner et al. 2005; Alvey 2006). It is thus worthwhile to examine the ecological, social, and economic relationships that can elucidate the *how* and the *why* underlying urban forest structure and diversity.

Although tree diversity can be defined in a number of ways, the most common diversity metrics used in the context of urban forests relate to age- and size-classes, to richness (the number of different tree species) and to evenness (the representation of a given species within the total number of individuals). Most papers quantifying tree-species composition in suburban areas focus on one or two major drivers of tree diversity, such as land type and use, education and income, and the presence of natural features like parks or greenways (Hope et al. 2003; Cornelis & Hermy 2004; Godefroid & Koedam 2007; Kendal et al. 2012a; Bourne & Conway 2014). Although some studies on plant

diversity have identified a wide variety of biophysical drivers of diversity in suburban areas (Čepelová & Münzbergová 2012), they fail to capture relevant socio-political and economic dimensions, including resource availability, administrative decisions, and management traditions. Other studies have examined both habitat-related and socioeconomic drivers of biodiversity in urban areas (McKinney 2008; Kowarik 2011), but do not refer explicitly to the urban forest, nor to varying temporal factors involved in suburban development. Furthermore, relatively little attention has been paid so far to differences in correlates of tree diversity for the land types making up the suburban forest, such as streets, residential properties, parks, and naturalized areas.

2.4 CHARACTERISING THE URBAN FOREST

There is no universal definition for “suburbia”, and the literature contains many conceptions of what constitutes the suburbs (Forsyth 2012). Different forms of subdivision development have been described based on street design and patterns, land use, buildings, and the presence of green and grey infrastructure (Wheeler 2015). Other factors, like commuting distance, access and location, modal transport, demographics, and culture have also been used in definitions of the term (Forsyth 2012). This paper characterizes the subdivision development fairly broadly for the purpose of envisioning its urban forest and the factors that might influence forest structure. It considers the suburbs, when initially developed, as the areas located between the urban core and the “countryside”, at the periphery of cities (Forsyth 2012; Turcotte 2008). These areas are typically residential, parceled into individually owned lots, which include detached or semi-detached dwellings and green space (e.g. lawn or garden). Various manifestations of

suburban development are pervasive worldwide and, in many urban areas, represent the primary land-use type (Wheeler 2015).

Any tree found within a city boundary, whether planted or naturally occurring, can be considered urban forest (Duinker et al. 2015). Urban forests are generally composed of either remnants of natural habitats or are intentionally planned and created for the purpose of increasing forest cover in urban areas (Turner et al. 2005; Ordóñez & Duinker 2014). In suburban areas, one might find trees planted along residential streets, in back or front yards on private residential property, and in public open spaces like parks. One might also find trees in areas that were either untouched during development or have naturalized since – these include patches of woodland and forest buffers running along property lines. Where suburban communities are gated, all trees found within the community, whether along streets or in open spaces, would be considered private. However, for the purpose of this paper, only neighbourhoods that are open to the public will be taken into account. Trees in suburban areas generally grow on one of the four land types that represent differences in development patterns, establishment practices, ownerships, and maintenance: street trees, property trees, park trees, and trees found in remnant stands (Table 1).

Studies have shown that tree-species richness is generally greatest in suburban and peri-urban areas compared to the countryside and the highly built-up urban core (McKinney 2008), although there are exceptions (Dobbs et al. 2013). Researchers have attributed this pattern to the heterogeneity and diversity of suburban landscapes, where multiple habitats (e.g. gardens, streetscapes, public green spaces, remnant woodland, agricultural land) occur in fairly close proximity (McKinney 2002; Hansen et al. 2005).

Additionally, private gardens reflect individual planting preferences; diverse ornamental species are planted for horticultural purposes, which often attract a wide range of organisms (Henderson et al. 1998; McKinney 2002; Turner et al. 2005). Evidently, suburban areas can exhibit greater landscape and species diversity compared to other urban areas despite the potential homogeneity of suburban housing design and development.

Suburban areas tend to develop at the periphery of cities, but often gradually become more central over time as cities grow (Turcotte 2008). As cities expand around and beyond suburban areas, landscape changes alter tree cover and tree-species composition. It is thus crucial to understand temporal context when determining drivers of urban diversity (Luck et al. 2009). As suburban developments age, new drivers of tree diversity may come into play. For example, the availability of nursery species at a given point in time influences the composition of planted trees during the early stages of development (Conway & Vander Vecht 2015), while elements of community design (e.g. the presence of remnant woodland adjacent to residential properties) can influence tree diversity as a neighbourhood ages and the canopy becomes more mature. In light of these considerations, phases of development should be taken into account when examining drivers of urban tree diversity: what drives tree diversity when a new neighbourhood is developed, when it is initially settled, and as it ages?

Suburban population growth is outpacing urban growth in many cities in North America, and in many regions of the world, suburban development accounts for the largest urban land area (Gordon & Shirokoff 2014; Wheeler 2015). Given the fact that suburban areas have the potential to include a range of landscapes and species,

identifying factors that shape the composition of trees found on various land types can provide insight into biodiversity enhancement in urban areas for planners and policy-makers (Dobbs et al. 2013). The following text presents an overview of ecological, socio-political, and economic factors that shape suburban forest composition.

Table 1 The land types on which trees are found in suburban areas.

Land Type	Ownership	Description
Street	Public	Along roads and boulevards, usually in a straight line; on tree lawns, medians; planted by developer (contractor) or municipality, and maintained by the municipality.
Property	Private	In front and back yards; can be planted in a row (hedges) or more randomly; either established naturally or planted; trees individually maintained by the property owner.
Park	Public	In parks and other open spaces otherwise dominated by lawn or impervious surfaces; planted and maintained by the city or municipality.
Remnant and/or regenerated	Private or public	In naturalized areas; includes forest buffers and patches in parks, between houses and/or residential developments; the trees are generally not individually maintained.

2.5 DRIVERS OF TREE-SPECIES COMPOSITION

2.5.1 Biophysical characteristics and natural features

The tree-species composition on all land types is influenced in part by ecological factors like climate, geology, weather, and topography. Trees located in areas with high impervious surface cover, like streets, are more likely to feel the effects of urban stresses in addition to the biophysical characteristics of a particular locale.

2.5.1.1 Geography and climate

Urban forest diversity can be influenced by the location of cities themselves. Kühn et al. (2004) found that cities in Germany were established in zones of naturally occurring high levels of native species richness, while Araujo (2003) demonstrated that a positive correlation exists between plant species richness and population density in Europe. Like non-urban vegetation communities, the species assemblage of trees that can thrive in cities depends on climate, temperature, and rainfall, which vary based on location (Kendal et al. 2012*b*; Kendal et al. 2014). Indeed, Dwyer et al. (2000) and Ramage et al. (2013) determined that the composition of urban trees was linked to native ranges of tree species in the surrounding biome, which is in turn determined by temperature and precipitation. The urban tree composition of a suburban neighbourhood therefore relies in part on where the urban area is located.

The type of soil, rock formation, topographic position, and other site-specific characteristics can also serve as a predictor for tree-species composition. Parent material types shape mineral composition, soil development, and soil quality (Keys et al. 2010). Slope gradient and elevation influence soil moisture and drainage, and determine growing conditions for tree species; for example, American beech (*F. grandifolia*) is more likely found on steeper slopes that are well drained, while red maple (*A. rubrum*) prefers flatter, wetter areas (Siegert et al. 2016). According to Kühn et al. (2004), the number of geological types found within a city positively correlated with native and overall plant species richness. The authors attributed this trend to other biophysical factors shaped by geology, like soil, relief, and natural habitats, a diversity of which promotes species richness. Although the drivers mentioned above do not explicitly relate to tree diversity in

suburban areas, they are important to consider because they influence urban tree diversity in general, including places at the periphery of cities.

2.5.1.2 Species traits and tolerances

Some cities, particularly coastal ones, are more likely to feel the effects of storms and hurricanes, which can result in tree mortality due to heavy precipitation and strong winds. Other cities located in very dry or very humid environments are more susceptible to drought or heavy rain. These factors inevitably dictate the types of trees that can thrive in urban and suburban environments; tree species traits related to growth rate, tolerances, and reproduction are key to determining the species composition of healthy trees in urban and suburban areas. Some tree species may be more suitable for street plantings due to higher tolerances for urban stresses like soil compaction, vehicular traffic and pollution, road maintenance, and construction (HRM 2013). As discussed previously, the success of planted and naturally established tree species can also depend on tolerances to site-specific characteristics like heat, sun exposure, soil type and moisture, and topography.

Extreme weather events are predicted to increase in frequency and intensity in the near future due to climate change, and urban planners and foresters need to consider which tree species should be prioritized over others (Roloff et al. 2009; Ordóñez & Duinker 2014). The range of native tree species is predicted to shift in many parts of the world due to climate change, which could also influence the distribution of these species in city environments (Kendal et al. 2012*b*). Although species traits are important considerations when planning future suburban forests, developers and planners may also be influenced (or constrained) by resource availability, planting traditions, and costs. These are discussed more thoroughly in the following sections.

2.5.1.3 Forest patches and habitat edges

On a neighbourhood level, the presence of natural features like forest remnants, patches, green corridors, and naturalized spaces can influence suburban tree composition. Forest stands located close to urban and suburban areas create transition zones, known as “forest edges”, along the perimeters of the contrasting habitat types (Godefroid & Koedam 2003*a*). Forest edges have the potential to house species that are generally not found in forest interiors, emphasizing the potential for these microhabitats to increase tree and plant species richness. Godefroid & Koedam (2003*a*) and Čepelová & Münzbergová (2012) discovered that overall species richness was greatest in these areas, and non-native species were also abundant in forest edges, due to landscape disturbances associated with the adjacent urban area. Areas that have undergone rapid and significant anthropogenic alterations are known to facilitate the invasion and propagation of non-native species (Byers 2002; Tait et al. 2005), which can in turn alter the species composition of remnant woodlands by promoting the establishment of exotic species in suburban areas.

Although edges along remnant woodland can serve as habitat for non-native species, studies have shown that preserving remnant patches can also conserve native species more effectively than anthropogenic landscapes like planned parks (Gong et al. 2013). Results like these indicate that “near-natural” habitats that have not been subjected to anthropogenic disturbances could mitigate the effects of biotic homogenization by preventing the establishment of non-native species (Gong et al. 2013; LaPaix and Freedman 2010). Kendal et al. (2012*b*) determined that spatial autocorrelation was an important predictor of native species distribution, which suggests that landscapes adjacent to forest patches containing native tree species, such as residential yards or

public green space, could also serve as important habitats for these native species. A positive relationship exists between native species richness and non-native species richness in forest patches in more-disturbed urban areas, indicating that anthropogenic changes resulting in landscape heterogeneity favour overall species richness (Zipperer, 2002; Godefroid & Koedam, 2003a; Gong et al. 2013). The size of forested patches also influences urban tree diversity; larger naturalized areas tend to hold more tree species than smaller ones (Hobbs 1988), and smaller forest patches may be more susceptible to anthropogenic disturbance and species invasions, potentially reducing overall species richness (Honnay et al. 1999). The size, shape, and land-use history of remnant and naturalized forest patches found in a suburban neighbourhood can therefore play a role in shaping its tree-species composition, mostly in remnant patches themselves and in contiguous areas.

2.5.2 Neighbourhood demographics and cultures

Socioeconomic and cultural drivers of suburban tree diversity manifest themselves mainly at the residential property level (Conway & Bourne 2013), although some studies have examined similar relationships pertaining to street and park trees (Pedlowski et al. 2002). Many of the trees found on residential properties are planted by home owners and reflect planting priorities, maintenance preferences, and environmental values – these factors can be shaped by demographic characteristics such as education, income, ethnicity, and gender. The tree-species composition on private properties can also be influenced by factors determined at the street or neighbourhood scale – these relate to fads and social norming.

2.5.2.1 Income and economic status

Household income seems to influence home-owner decisions regarding tree planting and removal on residential properties. Researchers have found that higher-income earners are more likely to plant trees to enhance the aesthetic beauty of their garden or home, and choose tree species according to fashion. They also value trees based on the ecosystem services that they provide, including food and habitat for wildlife (Kirkpatrick et al. 2012). According to Kinzig et al. (2005), groups with similar socioeconomic and cultural status tend to group together, suggesting that the median household income of a particular suburban neighbourhood could reveal trends related to planting preferences on private and public land. Some researchers suggest the existence of a “luxury effect”, whereby wealthier groups either favour diverse landscapes for settlement or create and maintain their own (Hope et al. 2003). Indeed, according to Hope et al. (2003) and Martin et al. (2004), perennial plant species richness is generally greatest in urban areas with higher family incomes and socioeconomic status. Pedlowski et al. (2002) also discovered that wealthier neighbourhoods contain more street and private property tree species than poorer neighbourhoods. In this case, the authors surmised that tree-species composition is influenced by home-owner planting and involvement, as well as municipal policies that may favour wealthier groups for public plantings (Pedlowski et al. 2002). Factors related to municipal decision-making and policy will be discussed further in section 4.3.

2.5.2.2 Property ownership

Little empirical research has been carried out regarding the influence of property ownership on tree-species composition. Kendal et al. (2012a) discovered that

neighbourhoods with more renters had a greater number of tree species on private properties. The authors attributed this pattern to the fact that different renters might plant different species in the same yard, contributing to overall species richness over time. However, studies have also shown that higher proportions of renters can also correlate with lower canopy cover in residential areas, due to lower income renters having fewer resources and less authority to plant and maintain trees on private property (Heynen et al. 2006; Landry & Chakraborty 2009). If canopy cover also correlates with tree species richness, then neighbourhoods with more renters may have lower overall species richness.

2.5.2.3 Education

The education level of home owners in suburban areas can also correlate with species richness on private property, although it is important to note that education often co-varies with income and socioeconomic status (Luck et al. 2009). According to Kirkpatrick et al. (2012), groups with both higher education and income levels were more likely to perceive trees as positive contributions to the urban landscape and understand the benefits of trees, which could influence tree species choices and shape tree species diversity on private properties (Luck et al. 2009; Meléndez-Ackerman et al. 2014).

2.5.2.4 Gender and age

Research has shown that along with income and education, gender and age may also shape tree species choices on private properties. Kirkpatrick et al. (2012) found that females were more likely to value trees for a wide range of reasons, including intrinsically and pragmatically, and be most knowledgeable about tree species. Home-owner age can also play a role; some studies have shown that older, retired people spend

more time tending to their garden, which could increase species richness depending on gardening priorities (Kendal et al. 2012a). However, homeowners who prefer to garden may wish to remove trees from their yard due to unwanted shade and maintenance costs.

2.5.2.5 Ethnicity and nationality

Demographics related to culture, nationality, and ethnicity can also correlate with neighbourhood tree-species composition by operating at the household level. Home owners with different cultural backgrounds can internalize contrasting perceptions of the urban forest, which are demonstrated through preferences related to tree planting and yard maintenance (Fraser & Kenney 2000). Research has shown that home owners of British origin gravitated towards shade and ornamental trees, Italian and Portuguese community members tended to favour fruit trees, and those of Chinese origin generally planted the fewest trees (Fraser & Kenney 2000). These results indicate what different cultural groups value in urban trees. Some plant trees for practical purposes like shade or food, while others choose species based on aesthetics and level of maintenance. The authors draw a connection between current urban landscaping practices and traditional land use and histories; people tend to value the natural features and processes associated with their cultural heritage. A suburban neighbourhood inhabited by a diverse group of people with distinct cultural values and traditions could thus create a more diverse landscape with high species richness, driven by “bottom-up” planting decisions (Kinzig et al. 2005). Conversely, a more homogenous community housing residents with similar cultural backgrounds may not exhibit the same level of species richness.

2.5.2.6 Fads and social norming

Although individual ethnic and cultural norms can significantly influence preferences for yard maintenance and tree planting, social rules operating at the neighbourhood level can also play a role in determining urban forest structure. It is well documented that home owners are either directly or indirectly influenced by the landscape, gardening, and planting practices of their neighbours (Nassauer et al. 2009, USA; Goddard et al. 2013). Researchers have proposed that replication or mimicry can occur at the street level, when particular landscape elements in front or back yards are “perceived and interpreted as a case or rule”, and subsequently are adopted by other residents (Julien & Zmyslony 2001, p.347). Landscape elements and maintenance activities that could influence the tree-species composition of the neighbourhood forest include gardening, lawn mowing, trampling, encroaching onto remnant woodland, planting location and, evidently, species choices (Hobbs 1998; McWilliam et al. 2014). Residents who initially settle into a new development may prefer garden beds and neat turf grass over stem density and tree-species richness, influencing future residents to plant fewer trees. Furthermore, residents who do plant trees may (unintentionally) encourage neighbourhood homogeneity related to planting location and tree species choice, potentially limiting overall tree-species richness (Jim 1993). Summit and McPherson (1998) found that home owners tended to plant trees within five years of residency, suggesting that in the case of a new subdivision, the tree-species composition of private properties in suburban neighbourhoods could be determined fairly soon after development and initial settlement.

Home-owner values about landscapes and tree species choices are also driven by dominant environmental paradigms and planting traditions. Kirkpatrick et al. (2013a) found that residents in Australian cities often made a conscious effort to remove non-native trees from their property, suggesting a backlash against outdated colonial landscape influences and a propensity towards intrinsically valuing “indigeneity” more than non-native tree species (Kirkpatrick et al. 2013a, p.175). Increasing pressure for urban dwellers to adopt more environmentally responsible behaviour may result in a newfound appreciation for nativeness in the urban landscape.

2.5.3 Administration and neighbourhood management

The drivers of urban forest structure related to administration and municipal management primarily influence trees planted on public land, namely streets and parks. However, the procurement of tree resources can play a large role in influencing tree-species composition on private properties. Residents often buy trees from local nurseries and garden centres, thus dictating the types of trees available for planting (Summit & McPherson 1998; Zipperer 2008; Conway & Vander Vecht 2015). Municipal factors affecting street and park trees also relate to the availability and cost of tree species (and cultivars) in nurseries and larger wholesalers, as well as urban forest management goals, policies, and operations.

2.5.3.1 Resource availability

The types of trees that residents choose to plant depend partly on cost and personal preferences, but also on the availability of species at local nurseries and retailers. In the city of Toronto, Canada, Conway & Vander Vecht (2015) found that ornamental tree species like the non-native Japanese maple (*A. palmatum*) are most

favoured by home owners buying trees from these stores. Customer demand thus influences tree species availability in nurseries a great deal; garden centres also identified the popularity of “container gardening” as a deterrent for stocking larger tree species (Conway & Vander Vecht 2015, p.6). Despite the buyers’ propensity towards ornamental species, half of the nurseries examined in this study stated that they were more likely to stock and recommend native tree species to customers, which also may be linked to store location and customer demand, as well as a shift in planting fashions towards prioritizing native species. On a neighbourhood level, the age of development may also correlate with the types of trees found on private properties due to nursery availability. Pincetl et al. (2013) determined that in Los Angeles, the number of tree species, particularly non-native ones, offered in local stores increased significantly from 1990-1989 to 1990-2011. This trend could be occurring due to changes in customer demand, suggesting that neighbourhoods established more recently may exhibit very different species composition patterns on residential properties compared to older neighbourhoods.

Although landscapers and contractors sometimes buy trees from local nurseries, planting projects occurring at the neighbourhood or municipal level for streets, parks, and other public areas often have greater access to regional wholesalers due to the sheer number of trees needed (Conway & Vander Vecht 2015). Disparities can exist between what urban foresters (or contractors) request for large-scale plantings and the availability of tree species in nurseries, leading to substitutions or sourcing of trees from another region (Sydnor et al. 2010; Conway & Vander Vecht, 2015). Depending on the planting priorities of the contractor or project, more ecologically favourable species (that are perhaps limited in quantity due to low demand) could be replaced by species that are

already common in the cityscape or are not as well adapted to the climate and physical environment (Conway & Vander Vecht 2015).

It is also important to recognize that landscape architects, developers, contractors, and urban foresters are themselves influenced by the traditions within their discipline, and subsequently may prioritize different factors when choosing tree species for planting, like soil condition, sun exposure, native status, aesthetics, species diversity, and space availability, among others (Conway & Vander Vecht 2015).

2.5.3.2 Urban forest targets and policies

Given the diverse benefits that urban forests provide, cities are recognizing the importance of engaging in strategic urban forest planning and management. One of the ways in which municipalities can do so is to create an urban forest management plan (UFMP), a document that presents a set of principles, guidelines, targets, and implementation strategies meant to promote a healthy and sustainable urban forest (HRM, 2013). In most cases, UFMPs create policies and rules to enhance urban tree diversity, often according to established standards (Santamour, 1990; HRM 2013; Ordóñez & Duinker 2013). These biodiversity guidelines, if implemented effectively and followed correctly by multiple actors (e.g. contractors, landscapers, foresters, community planting groups), play an important role in determining the species composition of trees planted on public land.

UFMP goals for urban forest diversity may vary between cities in terms of content and specificity. Many targets are inspired by the pervasive “10-20-30” rule, where no more than 10% of any species, 20% of any genus, and 30% of any family should be planted in a neighbourhood (Santamour 1990). Others are more ambitious, and

incorporate goals for native and heritage species representation (HRM 2013). Some municipalities will inevitably have a wider palette of native tree species to choose from for planting due to ecological factors, which could be reflected in diversity targets. However, in a review of fourteen Canadian UFMPs, Ordóñez & Duinker (2013) found that although all plans sought to promote “naturalness” in the urban forest, objectives were vague and there was seldom mention of actual numeric targets. Imprecise or ambiguous targets for tree species-selection may reflect a lack of knowledge about the composition of urban trees, and can result in disorganized and inefficient management practices.

The species richness and abundance of remnant trees in suburban areas may also be influenced by the lack of regulations surrounding suburban development, particularly if residents encroach into woodland around private properties by building fences and paths (McWilliam et al. 2014). For example, provisions for retaining forest buffers and riparian woodland in Halifax, Canada were introduced in municipal by-laws in 2006, but only apply to current projects and newer subdivision developments (HRM 2013).

Although laws were recently passed granting the municipality the right to protect remnant woodland in older suburban areas, implementation is difficult due to lack of resources. This illustrates how newer suburban developments may benefit from recent regulations aiming to protect native tree stands.

2.5.3.3 Urban forest management and operations

Targets and policies related to tree diversity should theoretically influence tree-species composition along streets and in public places, and the strategies used in the operationalization of UFMP goals should also be considered as important drivers of

urban forest structure. For one, the creation of a tree inventory can be useful for determining the “baseline” of urban forest composition. This information can subsequently be used to monitor urban forest composition and to develop specific strategies geared towards improving the representation of a particular species or enhancing tree diversity in certain neighbourhoods (Ordóñez & Duinker 2013). Inventories can also be particularly useful for determining plantable locations and suitable trees for a particular neighbourhood based on its current species composition (HRM 2013). However, inventories can be costly and time-consuming; software and modeling programs like i-Tree can also be used to describe and assess the state of urban forests.

Tree ownership and land tenure also influence the management of tree diversity. As mentioned previously, trees in suburban areas grow on multiple land types, reflecting differences in planting practices and ownership. Regulations and by-laws related to jurisdiction and oversight responsibilities of tree planting, removal, replacement, and maintenance vary across cities and countries, and can reflect potential differences in tree species assemblages (Jim & Liu 2001). Some municipalities have private tree protection by-laws, where the removal of large trees on private property is regulated and requires a permit (Toronto Municipal Code 2013).

In some cities, subdivision by-laws require a development officer’s approval for street tree species to be planted in new developments; however, it is difficult to determine whether these are enforced, and subsequently whether diversity targets are being met (J. Charles, personal communication, 2016). For cities that do not have established species diversity policies, tree-species selection and composition on public property generally

falls into the hands of arborists, developers, and landscape architects (J. Simmons, personal communication, 2016). In this case, the species composition of trees on public land will likely depend on the preferences and priorities of practitioners.

2.5.3.4 Neighbourhood management and engagement

Tree-species composition can be influenced by the constraints of neighbourhood management, particularly in colder and wetter climates where stormwater control, snow removal, and road salt applications for de-icing purposes are common considerations. Some tree species are more sensitive to the effects of de-icing salt (including reduction in photosynthesis and leaf necrosis) compared to others (Paludan-Müller et al. 2002). With this in mind, municipalities may constrain species selection for street trees based on salt tolerance. Snow removal techniques may also cause damage to street trees, which could alter species composition if trees require removal and replacement.

Resource availability is a major consideration in the operationalization of urban forest diversity objectives. Partnerships between municipalities and universities, industry, and community groups not only encourage resource- and knowledge sharing, but also foster public stewardship of the urban forest. UFMP goals for urban forest diversity that are defined and implemented at the neighbourhood level may also encourage citizen engagement, which can be important for achieving diversity targets. Neighbourhoods that do not hold many trees on municipal land would benefit from community-based strategies geared towards enhancing tree diversity on both public and private properties (HRM 2013). For example, initiatives encouraging residents and community groups to plant trees in the road verge or in public spaces can enhance neighbourhood tree-species diversity if implemented according to municipal guidelines and urban forest composition

targets. If neighbourhood diversity targets are set and citizens are educated about tree-species selection, it is more likely that appropriate species will be planted on public and private property, enhancing overall tree diversity.

2.5.4 Professional cultures and paradigms

Given the multidisciplinary nature of managing trees in the city, the field of urban forestry has piqued the interest of academics, scientists, policy-makers and practitioners. These groups reflect a multiplicity of values, priorities, influences, trends, and challenges that shape the conceptualization and management of urban forests. The definition of “urban forestry” itself is highly contextual and may depend on language, land-use history, and research traditions (Konijnendijk et al. 2006). Tree-species composition is therefore inevitably influenced by historical and contemporary trends as well as the cultures and priorities of the many professionals who engage in urban tree care and management.

2.5.4.1 Colonial history and influence

Cities located in countries with colonial histories may display European influences on tree-species composition on both public and private properties. In Canada, tree species in cities have traditionally been chosen from a pool of primarily European species, reflecting the country’s colonial history and landscape influences (Turner et al. 2005; Ordóñez & Duinker 2013). Some species have since naturalized and grow in parks and remnant woodland (Point Pleasant Park Plan 2008). In Melbourne, Australia, Dobbs et al. (2013) discovered that most non-native tree species planted in the city are endemic to Europe, and dominate residential properties and streets. Colonial settlers may also have influenced species choices for planting for pragmatic purposes. Large, native shade trees were often seen as a nuisance to builders and farmers, a perspective that some researchers

argue has pervaded Australian society to the point where native tree species are generally disliked (Kirkpatrick et al. 2013*a*). Christchurch, New Zealand, was designed as an “English garden city”, and non-native species have traditionally outnumbered native species in residential gardens (Stewart et al. 2004). These trends may be changing as some citizens begin to recognize the importance of protecting native species in urban areas (Kirkpatrick et al. 2013*a*). In Christchurch, urban dwellers are increasingly promoting the natural heritage of the area by planting native species on public and private residential properties and restoring woodland habitat.

2.5.4.2 Professional cultures and priorities

Many practitioners are involved in the research, planning, and management of urban forests. Differences in priorities and agendas can influence species selection and overall species richness in neighbourhoods, and can vary depending on where trees are being planted (Conway & Vander Vecht 2015). Kirkpatrick et al. (2013*b*) found that urban planners did not prioritize biodiversity conservation as much as arborists do when selecting street trees, indicating that planners may not consider street trees as an important contribution to wildlife habitat compared to trees on other land types. In comparison, municipal forestry staff in Toronto, Canada, prioritize native species and largely base their tree planting decisions on the species composition of nearby trees, looking to increase “both neighbourhood and district-wide diversity” (Conway & Vander Vecht 2015, p. 6).

When selecting a tree species to plant, landscape architects tend to consider available space, aesthetics, sun exposure, slope, and intended use above both species diversity and native status (Conway & Vander Vecht 2015). This result indicates that the

landscape architect, whose work is not necessarily tree-focused, may not consider the contribution that one or a few trees make to the overall diversity of the neighbourhood canopy.

2.5.4.3 “Naturalness” in the urban forest

The concept of naturalization has become an important component of urban forest management in recent years, as researchers and practitioners recognize the importance of preserving ecological integrity in urban ecosystems (Kendle & Rose 2000; Ordóñez & Duinker 2012; Toni & Duinker 2015). However, the nativeness agenda is still hotly debated as some researchers question the ecological benefits of prioritizing native tree-species for planting, and argue that the presence of non-native, non-invasive species is vital for urban biodiversity enhancement (Kendle & Rose 2000; Chalker-Scott 2015). Despite the lack of consensus, frameworks for naturalizing urban woodlands have been conceptualized and applied (Toni & Duinker 2015), and UFMPs have set guidelines and targets for native species plantings and species-at-risk protection (Ordóñez & Duinker 2013).

It has been shown that the conservation and enhancement of remnant woodland can protect native tree species (Ranta & Viljanen 2011), and encourage the dispersal and establishment of species in adjacent residential areas (Doody et al. 2010). The trend towards increasing the “naturalness” of the urban forest, and the incorporation of diversity targets in municipal planning, can increase native species representation in streets and other public spaces where planting occurs (Conway & Vander Vecht 2015). It is not clear whether diversity targets set out by municipalities regarding native species prioritization and species diversification will influence the planting preferences of

residents on private properties. Conway & Vander Vecht (2015) found that some retail garden centres only stocked native tree species, citing customer demand and the influence of native plantings in the public right-of-way. However, despite apparent resident support for native species, they may not know how to recognize a native tree or its ecological importance, emphasizing the importance of education and local community engagement (Doody et al. 2010).

Monocultures in street-tree plantings, which reflect distinctly “unnatural” tree establishment processes, are now widely recognized as detrimental to the resilience of the urban forest (Raupp et al. 2006). This is in part due to the mass tree mortality rates from diseases and insects that were experienced in North America in the last century (Poland & McCullough 2006; Ordóñez & Duinker 2013). These are continuing threats today. Dutch elm disease devastated urban forests throughout the United States and Canada, especially in cities where stately elms were predominantly planted in the streets for shade purposes (Steenberg et al. 2013). Ash trees were also commonly planted together alongside roads and in residential developments; these are also being decimated with the arrival of the emerald ash borer (Poland & McCullough 2006). As a result, urban foresters have diversified municipal street tree planting lists and have included more native species (Raupp et al. 2006). Although tree species diversification in city streets will likely occur in primarily newer developments, the tree-species richness in the streetscapes of older suburban neighbourhoods may increase when trees need to be removed and replaced due to decline and death.

2.5.4.4 Contemporary challenges

Climate change is predicted to directly affect trees in urban and suburban areas. The occurrence of temperature fluctuations, fires, extreme weather events, and species invasions threaten the stability and productivity of urban forests (Ordóñez & Duinker 2012; HRM 2013). Tree species that are adapted to warmer climates will likely tolerate northward climate shifts, and as such may become more common in urban environments that also experience the urban heat island effect (Leichenko & Solecki 2013). Climate change will therefore pose challenges for some non-adaptive native tree species (Ordóñez & Duinker 2014). Remnant woodlands adjacent to suburban neighbourhoods may be vulnerable to drought and fire (Leichenko & Solecki 2013). As mentioned above, a more homogenous urban forest dominated by few tree species will probably be less resilient to the effects of climate change and experience higher tree mortality rates, especially due to pests and diseases (Ordóñez & Duinker 2014).

It is to be expected that some tree species will fare better than others in the face of environmental change (Rostami 2011). Urban foresters can anticipate stressors related to climate change that trees will be exposed to (e.g. wind, drought) and choose species accordingly. Species with southern ranges but higher tolerances to stressors in northern environments (e.g. frost) may also be selected, thus shaping future tree-species composition (Ordóñez & Duinker 2014). Furthermore, foresters and managers can mitigate the potential climate impacts on tree diversity by protecting remnant forest patches, enhancing habitat connectivity in the urban forest, and increasing the population sizes of vulnerable tree species (Ordóñez & Duinker 2014).

2.5.5 Neighbourhood and community design

The development history and design of suburban communities can dictate planting spaces and opportunities, influencing the composition of trees on all land types. Factors relating to neighbourhood design include development history and traditions, park and green space creation, subdivision and housing arrangement, and land use.

2.5.5.1 Development patterns, history, and age

The ways in which suburban developments are created can play a role in how their urban forest is shaped (Hahs et al. 2009; Fahey et al. 2012). One factor that distinguishes development patterns is the landscape present immediately prior to urbanization, also known as the pre-urbanized landscape. It has been shown that urban tree cover, overall tree-species richness, and proportion of native species is greatest in urban areas developed onto naturally forested regions, compared to prairie land (Nowak et al. 1996; Fahey et al. 2012), suggesting that remnant and regenerating forest stands significantly influence urban forest composition. This theory is confirmed by the fact that occurrences of native tree species in the urbanized landscape have been shown to spatially correlate with previously forested areas housing these species (Fahey et al. 2012). In a similar vein, Hope et al. (2003) found that urban developments on previously farmed land housed many fewer woody plant species compared to areas that had not been cultivated. Residential development encroaching onto once forested farmland or prairie and desert land will likely require more reforestation than similar types of development encroaching onto woodland (Nowak et al. 1996; Heynen & Lindsey 2003). The tree-species composition in subdivision development that occurs in non-wooded areas may therefore be influenced more by the preferences of developers, planners, and residents.

As mentioned previously, the biophysical conditions of a site can dictate the types of tree species that are more likely to grow well (Kühn et al. 2004; HRM 2013).

Therefore development patterns and techniques that alter the ecological characteristics of an area, including soil, geology, and natural contours may also influence the composition of remnant areas as well as trees located on residential and public properties (Florgård 2000).

The age of a suburban neighbourhood may also explain its tree-species composition, particularly as landscapes mature and residents settle into their homes and plant (or remove) trees according to their own preferences (Hope et al. 2006). The date at which a subdivision was developed may also reflect a particular development pattern or trend that could influence the type and amount of green space it holds, as well as its urban forest structure. Hope et al. (2003) found that younger housing lots correlated with higher plant species richness, which could reflect changes in technology and preferences related to water use, landscaping, and environmental values. These results illustrate that planting fashions and changing values could explain differences in urban forest composition between neighbourhoods located in the same city, but developed at different times. Similarly, the species composition of trees in the public right-of-way reflects municipal diversity targets enacted at the time of neighbourhood development. Nitoslawski & Duinker (2016) found that streetscapes in newer subdivisions (<15 years) exhibit greater species richness, evenness, and proportions of native species compared to older subdivisions (>40 years), illustrating how neighbourhood age can reflect changes in policy and targets.

2.5.5.2 Design of parks and remnant areas

Research has shown that green spaces can help protect native tree species, and enhance overall species richness (Alvey 2006; LaPaix & Freedman 2010). Parks found in urban and suburban areas generally support both native and non-native tree species, which could explain why planned green spaces are typically species-rich (Gong et al. 2013; Nock et al. 2013). However, larger parks and remnant areas tend to exhibit higher species richness than smaller green spaces (Godefroid & Koedam 2003*a*, 2003*b*). The shape of a forest patch and habitat edges can also influence its tree-species composition, as non-native species are generally found around forest boundaries (Godefroid & Koedam 2003*a*; LaPaix & Freedman 2010; Pennington et al. 2010). Remnant woodland with a long perimeter and wide recreational trails could therefore increase species richness by promoting the establishment of exotic, opportunistic species, but possibly at the expense of native ones (LaPaix & Freedman 2010). The effectiveness of parks to protect native tree species may therefore depend on their ecological integrity; parks and green spaces with a high degree of hemeroby related to intensive landscaping and the presence of impervious surfaces typically do not reflect the natural history of a region, and may not encourage the establishment of native species that are vulnerable to urban stresses (LaPaix & Freedman 2010).

The presence of remnant woodland adjacent to residential neighbourhoods can present an opportunity for native species establishment on private properties, as species can disperse and become established in lawns and gardens (Doody et al. 2010). Given this phenomenon, it can also be argued that species in residential gardens may also disperse and become established in adjacent parks or remnant areas. Green space

connectivity within a neighbourhood can thus promote tree-species diversity by providing sufficient habitat for species to disperse and become established (Rudd et al. 2002).

Interestingly, research has also shown that the amount of public green space in a particular neighbourhood can positively correlate with the amount of green space on surrounding private residential properties (Troy et al. 2007). These trends indicate that native tree-species richness can be promoted on properties close to parks or remnant woodlands, depending on the planting and maintenance preferences of homeowners (Doody et al. 2010).

2.5.5.3 Urban morphology and land cover

Various forms of subdivision development exist and differ in terms of size, street and lot design, land use, building types, and the presence of green and grey infrastructure (Wheeler 2015). The spatial design of a suburban neighbourhood can influence urban forest structure and composition (Biggs et al. 2014). For starters, the total land area of a neighbourhood can correlate with the amount of land available for tree planting. One might expect that with fewer trees, one would also generally find fewer species. Higher housing and population density has been shown to positively correlate with the amount of impervious surfaces, and negatively correlate with abundance of trees as well as native species richness (Luck et al. 2009). These trends may be associated with the size of lots and gardens on residential properties as well as type of housing (Tratalos et al. 2007); studies have shown that larger gardens hold more species than smaller ones (Kendal et al. 2012a). Larger gardens generally positively correlate with vegetation cover and number of large tree species (Smith et al. 2005). Studies have shown that tree-species richness is high on residential land (Turner et al. 2005; Dobbs et al. 2013; Bourne & Conway 2014),

suggesting that a primarily residential neighbourhood with larger lots may house more tree species on private property.

Neighbourhood design can also determine the abundance and types of trees established on each land type. The presence of tree lawns, medians, or road verges along residential streets allows for street-tree planting by the municipality as well as neighbourhood residents who participate in “guerilla gardening” in front of their houses but in the public right-of-way. Depending on the development history, the majority of the canopy in some suburban developments is found on private properties and “pedestrian corridors” between residential lots (HRM 2013). These neighbourhoods may have lower tree-species richness simply due to the lack of street trees. Infrastructure may also restrict species selection in the public right-of-way. The presence of overhead power lines is an important consideration when planting along streets, as height restrictions may constrain species selection. Neighbourhoods with buried power lines may therefore exhibit greater species richness and size diversity as smaller and larger species could be planted without risk of damage from and to lines.

2.6 RESEARCH NEEDS

While this paper has presented a breadth of factors that could influence tree diversity in suburban areas, it is evident that some drivers are better studied than others. It would be worthwhile to carry out qualitative studies exploring the value of native tree species for residential home owners, particularly because it has been shown that native species can disperse from remnant woodland and establish on residential properties (Doody et al. 2010). Exploring the role of education and community engagement regarding native species protection and overall diversity enhancement can highlight important factors that shape how private gardens might contribute to urban biodiversity.

Given the lack of consensus on whether property ownership influences tree diversity, future studies examining urban forest values of renters and home owners could be carried out to identify barriers to biodiversity enhancement.

Few studies have examined how land use and design could influence tree-species selection and urban forest diversity. One can argue that the presence of road verges and tree lawns encourages street-tree planting, yet there is little to no empirical evidence about how the size, shape, and design of sidewalks and roads could shape tree diversity. It would also be worthwhile to test how neighbourhood design could increase green-space connectivity, and subsequently how tree diversity could be enhanced.

Finally, exploring how municipalities, developers, urban foresters, and other stakeholders communicate and cooperate during subdivision development can shed light on policies and compliance issues related to tree-species selection. Are tree-species lists and diversity targets being consulted appropriately? Are there consequences for non-compliance? Tree diversity is shaped by both policy and its implementation, therefore it is vital to consider the priorities of practitioners involved in the planning (and planting) process.

2.6 SUMMARY

Enhancing tree diversity should be a priority for urban forest managers, particularly given concerns about biodiversity loss as well as the many benefits and services that diverse urban forests provide. Despite the recognition that biodiversity management in the urban forest context is important, it is also fraught with uncertainty. Debate persists about the contributions of native and non-native species to tree diversity, and in many cases municipal diversity targets are lackluster and ill-defined. Disturbances

like pests, diseases, extreme weather events, and other climatic changes are likely to impinge on the success of trees in urban environments, especially species that are already vulnerable to urban stresses. These difficulties reinforce the utility of trial-and-error and adaptive management; not all trees nor plantable spaces should be considered equal. Instead of blindly aiming for *more* biodiversity, managers and practitioners might be better off envisioning the *right* kind of biodiversity.

In this paper, ecological, socioeconomic, cultural, and administrative drivers of tree diversity in suburban areas have been outlined. This list is neither exhaustive nor inflexible; other factors will likely be identified and studied as urban forest research continues, while some factors will be more relevant in certain contexts. The drivers described in this text should serve as a guideline that can be used to inform urban forest management practices and to develop strategic diversity targets. In doing so, it is worth considering how the urban forest itself is perceived to contribute to our cities. Are trees in the city prioritized during and after urban development? Do we consider forests to be an integral component of the urban landscape, retained and designed with purpose, or more of an afterthought dependent on available resources? Biodiversity enhancement and the maximization of its benefits are most effective when the green infrastructure takes precedence over the grey – when the city is built with the trees firmly in mind.

CHAPTER 3 THE INFLUENCE OF DEVELOPMENT PATTERNS

3.1 ABSTRACT

Is (sub)urban forest diversity shaped by previous land use? This study was designed to quantitatively assess the impacts of subdivision development on urban tree-species composition in two Canadian cities: Halifax, Nova Scotia, and London, Ontario. The main goal was to determine whether cities with contrasting pre-urbanized or pre-settlement landscapes – woodlands in Halifax and agricultural fields in London – also revealed differences in urban tree diversity losses and/or gains due to urbanization. In each city, four residential neighbourhoods representing two age categories, older and newer (40-50 years, <15 years), were examined and trees on three land types were sampled: public (street), private (residential), and remnant (woodland). All public street trees within the chosen neighbourhoods were inventoried and approximately 10% of the residential property lots were sampled randomly. Plots were examined in remnant forests in or near each city, representing the original forest habitats prior to agricultural and/or urban landscape transformations. Diameter at breast height, species richness and evenness, and proportions of native and non-native trees were measured. In both cities, streetscapes in newer neighbourhoods exhibit greater species richness and evenness, and are characterized by substantially more native trees. Despite this trend, developers and home owners continue to intensively plant non-native species on newer and smaller property lots. Older neighbourhoods in Halifax containing remnant forest stands hold the greatest number of native trees on private property, alluding to the importance of residual forest buffers and patches in promoting naturalness in the private urban forest. These results suggest that identifying and quantifying flows of species between green spaces

during and after development is valuable in order to effectively promote native species establishment and enhance overall urban forest diversity.

3.2 INTRODUCTION

Biodiversity conservation and enhancement are key considerations in urban forest research and policy, given rapid rates of urbanization worldwide and growing concerns about biodiversity declines (Alvey 2006). Many studies have explored the range of benefits and services that diverse urban forests provide, including greater ecosystem productivity, higher resilience to environmental perturbations, and increased wildlife habitat (Turner et al. 2005; Ilkin et al. 2013; Bourne & Conway 2014). Consequently, diversity targets have been incorporated into the core objectives of urban forest management plans (HRM 2013; Ordóñez & Duinker 2013).

Despite the recognition that diversity is crucial to urban forest health and city sustainability, urban foresters and scholars continue to disagree on the importance of native tree species contributions to overall forest diversity; some argue that native tree species promote the establishment of other native organisms and increase the ecological integrity of urban ecosystems (Alvey 2006; Ilkin et al. 2013; Barth et al. 2015) while others question the prioritization of nativeness over non-native, non-invasive species that are more resilient to urban stresses and are vital for urban biodiversity enhancement (Kendle & Rose 2000; Chalker-Scott 2015). Furthermore, biotic homogenization has been identified as a pressing challenge to biodiversity enhancement in cities, as widespread exotic species can outcompete and replace native ones in anthropogenic environments, potentially contributing to species extirpations and threatening local, unique ecosystems (Olden et al. 2004; Turner et al. 2005; McKinney 2006). These

arguments and concerns illustrate the importance of elucidating the factors shaping urban forest diversity as well as determining how diversity benefits can be strategically and effectively maximized.

Suburban development is of particular importance to urban biodiversity research, as studies have shown that both native- and non-native tree-species richness can peak in neighbourhoods developed at the periphery of cities, located between the countryside and the urban core (Alvey 2006; McKinney 2008). This trend has been attributed to the high level of landscape and habitat heterogeneity (e.g. private gardens, streetscapes, public green spaces including parks) that can occur in suburban and peri-urban areas (McKinney 2002; Deutschewitz et al. 2003; Hansen et al. 2005). Research has revealed that the type of land cover (e.g. fields, grassland, forests, roads, ruderal habitat) can dictate suburban plant diversity, as habitat edges often promote greater species richness as well as the establishment of rare species (Godefroid & Koedam 2003; Čepelová & Münzbergová 2012). Furthermore, relatively undisturbed forest patches have been shown to be more suitable for protecting native plant and tree species compared to areas that have undergone significant anthropogenic change (Gong et al. 2013). Diversity on residential properties is also influenced by home owner attitudes towards species selection, which often focus on aesthetics and promote exotic and ornamental species plantings (Turner et al. 2005; Bourne & Conway 2014). Patterns in landscape structure and species composition emphasize the need to consider the planning and design of subdivisions during and after development, especially if urban forest diversity is to be considered a priority.

The heterogeneity of suburban landscapes, including the multiple land types on which trees can grow, necessitates the involvement of various urban forest professionals and practitioners in urban forest management. Trees found in parks and along streets generally fall under the jurisdiction of the municipality, and are subject to policies and targets determined by urban foresters and planners pertaining to tree planting guidelines, species selection, and diversity goals (HRM 2013; Ordóñez & Duinker 2013). The land types found in suburban areas represent differences in tree establishment and management, reinforcing the importance of considering other factors such as policy and administration along with biophysical characteristics and home owner preferences when exploring how urban forest diversity is shaped.

Approximately two out of three Canadians live in suburban areas, which are currently experiencing much higher population growth rates compared to inner-city neighbourhoods located closer to city centres (Gordon & Shirokoff 2014). The extensive suburbanization occurring across the country highlights the need to consider how suburban development influences urban forest diversity and potentially provides opportunities for biodiversity enhancement in urban areas. Although the effects of urban development on urban forest biodiversity have been examined in a single Canadian city (Bourne & Conway 2014), researchers have yet to compare forest diversity losses and/or gains associated with development and previous land use in different cities. The goal of this study was thus to explore whether contrasting development patterns in two Canadian cities also determine differences in suburban tree diversity.

3.3 MATERIALS AND METHODS

3.3.1 Study sites

The two cities chosen for this study are Halifax, Nova Scotia (44.6478°N, 63.5714°W), and London, Ontario (42.9837°N, 81.2497°W). Although both cities are found in the eastern region of the country, they are situated approximately 1,500 km apart (Figure 1). Halifax is located in the Acadian forest region, a mixed broadleaf and temperate forest comprised of both broadleaf and coniferous species including red spruce (*Picea rubens*), yellow birch (*Betula alleghaniensis*), sugar maple (*Acer saccharum*), and balsam fir (*Abies balsamea*) (Farrar 1995; Loo & Ives 2003). Some relatively undisturbed hinterland forest stands are found in suburban and peri-urban neighbourhoods in Halifax [6]. London is located at the northern range of the Carolinian forest region, characterized by a wide variety of broadleaf tree species like hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), hickory (*Carya* spp.), and tulip tree (*Liriodendron tulipifera*). London is surrounded primarily by agricultural fields and some scattered remnant woodland corridors (City of London 2014).

In each city, four residential neighbourhoods representing two age categories were examined (Figures 2, 3 and Tables 2, 3). Neighbourhood selection was based on the pre-urbanized landscape, which was determined by consulting with urban foresters and planners as well as historical aerial imagery (Figures 4, 5). Other criteria for neighbourhood selection included size and general similarities in urban morphology related to street and sidewalk design, as well as presence of street trees and remnant woodland in Halifax. Approximately 10% of private property lots were randomly selected based on total residential property land area on ArcGIS© (version 10) using the

parcel data layers from each city's spatial database (Figure 6). The residents of selected properties were given a letter describing the study and requesting permission to access their property. If permission was not granted, the home immediately next to the original selected property was chosen. All streets within the boundaries of each neighbourhood were examined. In Halifax, four remnant stands located adjacent to each residential neighbourhood were sampled using five plots of 10 m x 10 m (0.20 ha total remnant forest sampled). In London, two remnant woodland sites located outside city limits were sampled using three plots of 20 m x 20 m (0.24 ha total remnant forest sampled).



Figure 1 Halifax, Nova Scotia is located in Atlantic Canada and London, Ontario is located in Central Canada.

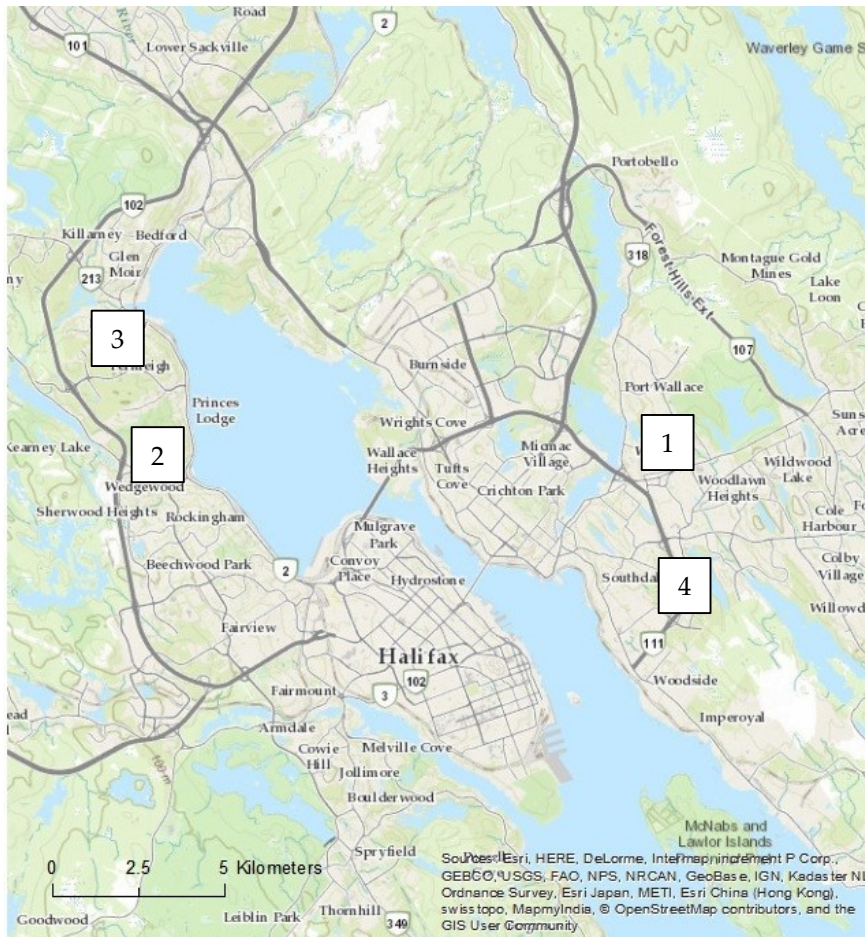


Figure 2 The neighbourhoods examined in Halifax: Tam O'Shanter (1), Birch Cove (2), Millview (3), and Russell Lake West (4).

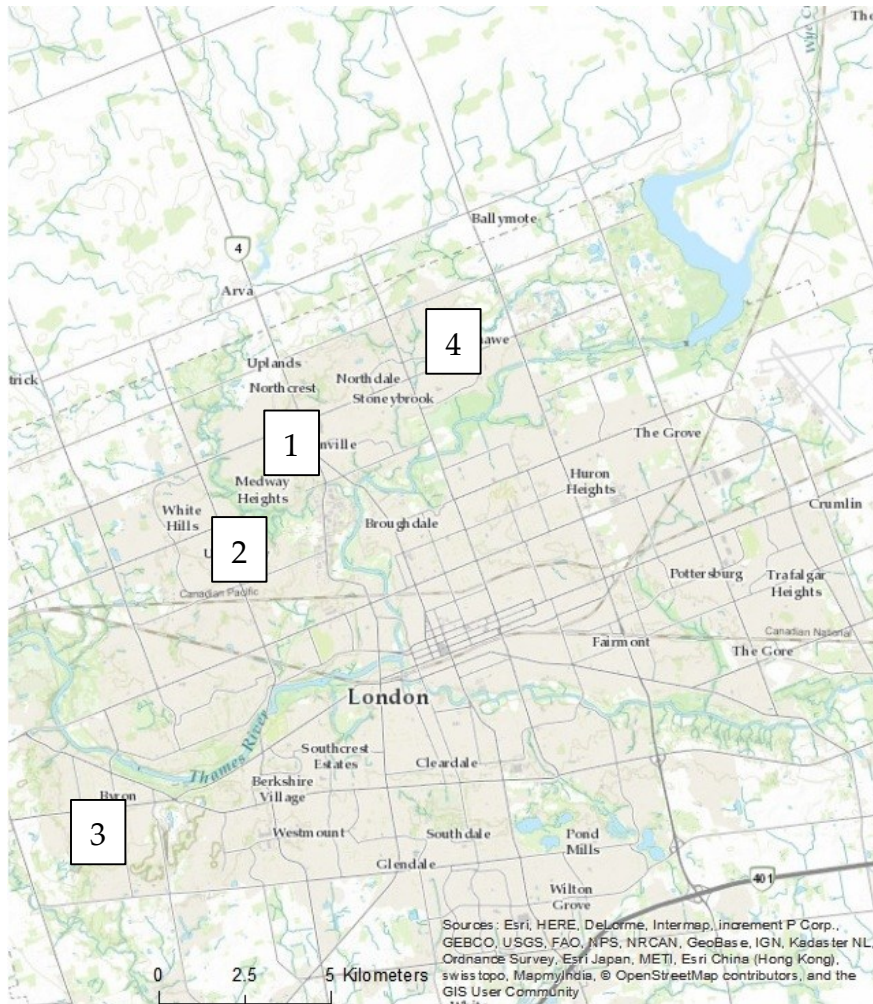


Figure 3 The neighbourhoods examined in London: Masonville (1), Sherwood Forest (2), Byron (3), and Stoney Creek (4).

Table 2 Neighbourhoods examined in Halifax.

Neighbourhood	Decade of development	Total number of lots	Number of selected lots	Average lot size \pm S.D. (m ²)
Tam O'Shanter	1960s-1970s	452	41	870 \pm 420
Birch Cove	1960s	250	26	890 \pm 280
Millview	2000s	478	49	830 \pm 300
Russell Lake West	2000s	300	30	710 \pm 320

Table 3 Neighbourhoods examined in London.

Neighbourhood	Decade of development	Total number of lots	Number of selected lots	Average lot size \pm S.D. (m ²)
Masonville	1960s-1970s	304	33	1270 \pm 300
Sherwood Forest	1960s	401	43	870 \pm 250
Byron	2000s	315	32	560 \pm 150
Stoney Creek	2000s	293	30	450 \pm 80



Figure 4 Aerial imagery showing the development of Stoney Creek (London, Ontario) from field to subdivision. Source: University of Western Ontario and Google Earth © Digital Globe.



Figure 5 Aerial imagery showing the development of Millview (Halifax, Nova Scotia) from woodland to subdivision. Source: Google Earth (c) Digital Globe.

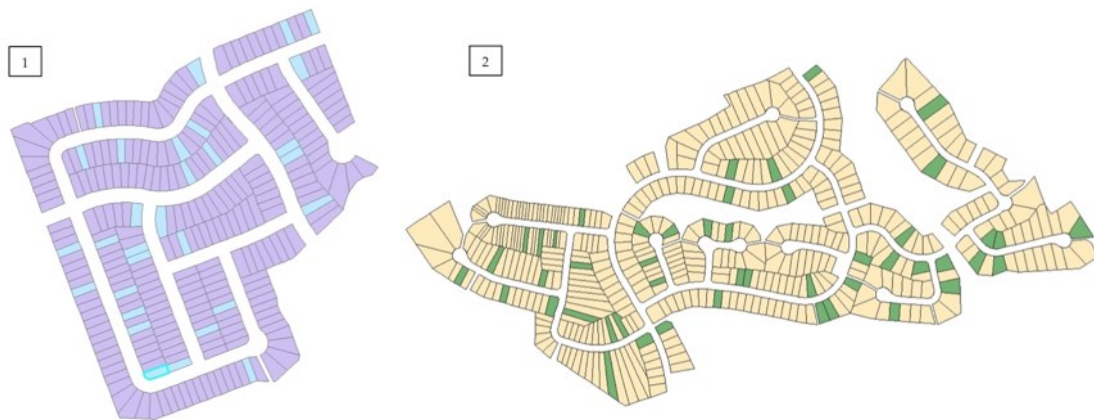


Figure 6 Neighbourhood extents and randomly selected private properties for Stoney Creek, London (1) and Millview Halifax (2).

3.3.2 Data collection

All sites in Halifax were visited between June and July 2015, while all sites in London were visited between June and August 2015. The street trees in each neighbourhood were censused. All trees within the boundaries of randomly selected

residential properties and within each plot in remnant woodland stands were measured. For each tree, the species was identified and diameter at breast height (DBH) was measured at 1.4 m from the base of the trunk using a diameter tape or calipers, depending on the size of the tree. Dead trees and shrubs (woody plants with height <5 m at maturity) were not measured.

Street trees were defined as trees either found on the road verge located between the street and the sidewalk, or, in the case where there was no sidewalk, located between the curb and private residential properties. Private property trees were defined as those found on landscaped and maintained areas of residential properties.

3.3.3 Data analysis

Tree species identified were denoted as “native” or “non-native” based on whether the natural range of the species is found in the province; London is found in Ontario, which has 85 native species, while Halifax is located in Nova Scotia with 42 native species (Farrar 1995; Kershaw 2001). Naturalized species that were introduced after European colonization, like Norway maple (*Acer platanoides*), were considered non-native.

For each land type and neighbourhood, species richness was calculated by summing all identified tree species. Proportions of native and non-native species were calculated by dividing the count of native/non-native trees by the total count of trees measured.

Two indices were used to calculate species diversity (Van Dyke 2008). The first is the Shannon-Weaver index, which takes into account both species richness and evenness:

$$H' = -\sum_i p_i \ln(p_i),$$

where p_i represents the count proportion of the i th species. Two-sample t -tests were carried out to determine whether results from this calculation varied significantly across neighbourhoods within and between cities.

The second diversity measure is the Simpson index of dominance, which describes the evenness of a community based on the probability that two randomly selected individuals belong to different species:

$$1 - D = \sum_i p_i^2,$$

where p_i represents the count proportion of the i th species.

Four separate chi-squared analyses were carried out on SPSS© (version 22) using 2 x 2 contingency tables to identify the factors that influence proportions of native and non-native trees on residential properties. For these calculations, tree data was combined from the two neighbourhoods of equal age class within the same city. Two variables were tested: the pre-urbanized landscape and the decade of neighbourhood development (or neighbourhood age). The tests compared counts of native and non-native trees in old neighbourhoods with different pre-urbanized landscapes, in new neighbourhoods with different pre-urbanized landscapes, in both old and new neighbourhoods in Halifax, and in both old and new neighbourhoods in London.

3.4 RESULTS

3.4.1 Site descriptions

The size of the neighbourhoods ranged from approximately 25 to 50 hectares. Newer neighbourhoods were generally comprised of smaller residential property lots compared to older neighbourhoods. Properties in all neighbourhoods examined contained detached or semi-detached homes surrounded by lawn. Many home owners chose to maintain garden beds or other landscaped areas to grow shrubs, hedges, herbaceous plants, and trees. Both newer and older neighbourhoods in Halifax retained some woodland area during development, including corridors located behind and between homes, and in some cases along pedestrian footpaths. Newer neighbourhoods in London were primarily surrounded by fields with some scattered forest corridors (Figure 7). Some areas of an older neighbourhood in London bordered a ravine with some woodland, while the other older neighbourhood housed a small patch of regenerated woodland.



(a) Russell Lake West, Halifax



(b) Byron, London

Figure 7 New subdivision developments in: (a) Halifax, where remnant woodlands are located adjacent; (b) London, where fields are located adjacent. Remnant stands in Halifax are generally found between property lines behind and beside housing developments as well as along pedestrian footpaths. In London, the scattered forest corridors that remain often exist further away from the development.

In Halifax, the remnant woodland sites were found adjacent to the subdivision, and were determined to be forested immediately prior to development, and remained so throughout the development process. Red maple (*Acer rubrum*) and red spruce dominated remnant sites in Halifax. Dominant trees were estimated to range between 50–100 years old. Other species sampled included yellow birch, striped maple (*Acer pensylvanicum*), and eastern hemlock (*Tsuga canadensis*). Two non-native species, Norway maple and European alder (*Alnus glutinosa*), were sampled in remnant plots located in close proximity to Birch Cove and Tam O’Shanter, respectively.

In London, remnant woodland was sampled at two sites located outside city limits. Dominant species included red maple, black cherry (*Prunus serotina*), and red oak (*Quercus rubra*); these trees were estimated to be between 100–200 years old. More uncommon species sampled included blue beech (*Carpinus caroliniana*), flowering dogwood (*Cornus florida*) and tulip tree.

3.4.2 Species diversity

3.4.2.1 General summary

In total, 82 tree species were identified in Halifax and 104 tree species were identified in London (Table 4). In both cities, the total species richness of private properties was greatest, followed by streetscapes and remnant woodland. Across all land types, London had a higher number of native species compared to Halifax. However, the selected neighbourhoods in Halifax housed a larger proportion of native species (33 native species sampled/42 total native species within the area of Nova Scotia = 79%) compared to London (52/85 = 61%).

Table 4 Combined species richness and raw counts of native and non-native species on each land type for both cities. Percentages represent proportion of native species sampled out of the total number of native species found in the province.

Land Type	Halifax			London		
	Native	Non-Native	Total	Native	Non-Native	Total
Remnant	16 (38%)	2	18	25 (29%)	0	25
Street	10 (24%)	23	33	21 (25%)	19	40
Private	28 (67%)	44	72	39 (46%)	45	84
Total number of species	33 (79%)	49	82	52 (61%)	52	104

3.4.2.2 Street trees

Similar patterns in street-tree composition were observed in Halifax and London (Figures 8, 9). Older neighbourhoods exhibited low species evenness and were dominated by non-native species, including Norway maple, littleleaf linden (*Tilia cordata*), crabapple (*Malus* spp.), and tree lilac (*Syringa reticulata*). In streetscapes, Norway maple accounted for 48% of all older trees in Halifax and 40% of all older trees in London, illustrating its popularity in street-tree planting in Canadian neighbourhoods developed 40-50 years ago.

The results from the Shannon-Weaver diversity index calculation, which takes into account species richness and evenness, indicated that the diversity of streetscapes in older neighbourhoods in both cities is significantly different from streetscape diversity in newer neighbourhoods (Table 5). A two-sample *t*-test was used to compare differences in the Shannon index calculations (e.g. diversity) of street trees between older (mean $H' = 1.30$, SE = ± 0.040) and newer (mean $H' = 2.56$, SE = ± 0.105) neighbourhoods in Halifax ($t = 11.17$, $p = 0.008$, d.f. = 2). The same test found a significant difference between the diversity of street trees in older neighbourhoods (mean $H' = 1.83$, SE = ± 0.040) and newer neighbourhoods (mean $H' = 2.58$, SE = ± 0.055) in London ($t = 10.95$, $p = 0.008$, d.f. = 2). Older streetscapes in London were slightly more diverse than older streetscapes

in Halifax ($t = 9.37, p = 0.011, d.f. = 2$), while a similar result was not found between newer streetscapes in both cities.

Street trees in newer neighbourhoods exhibited much greater species evenness, as illustrated by the more gradual slope in the species abundance distribution (Figure 3.9). The representation of Norway maple was also much smaller, accounting for approximately 11% of all newer street trees in Halifax and 3% of all newer street trees in London. Native stem count proportions were also higher in newer neighbourhoods compared to older ones, indicating that native species are more recently preferred for planting in city streets (Table 5).

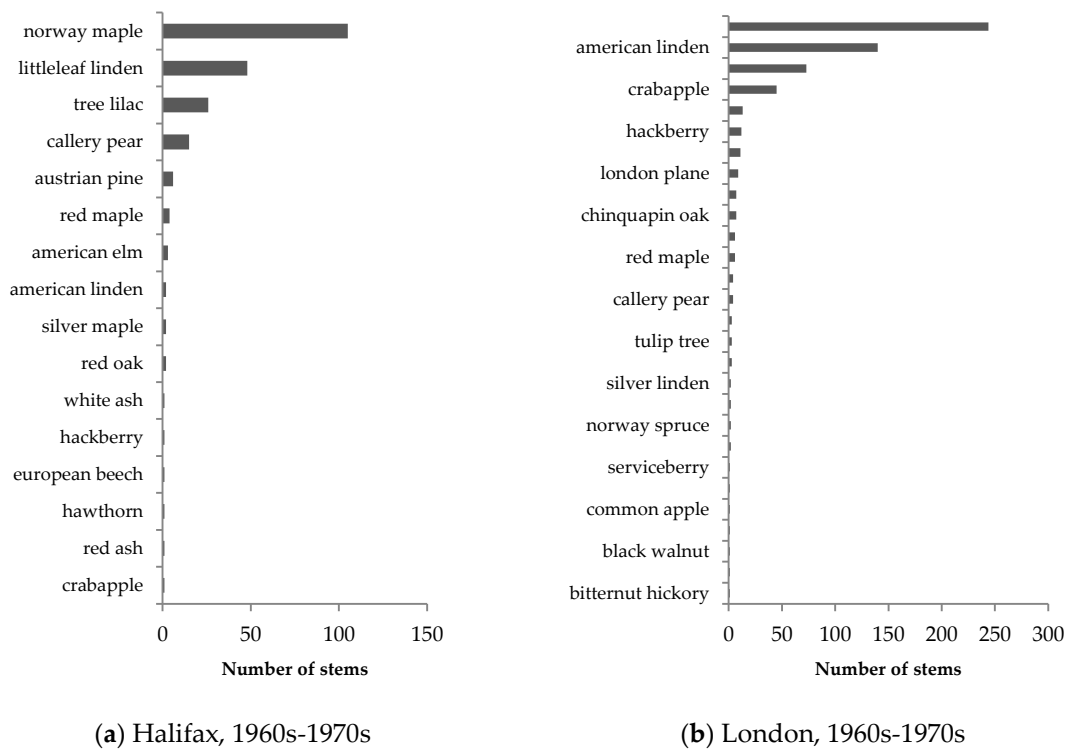
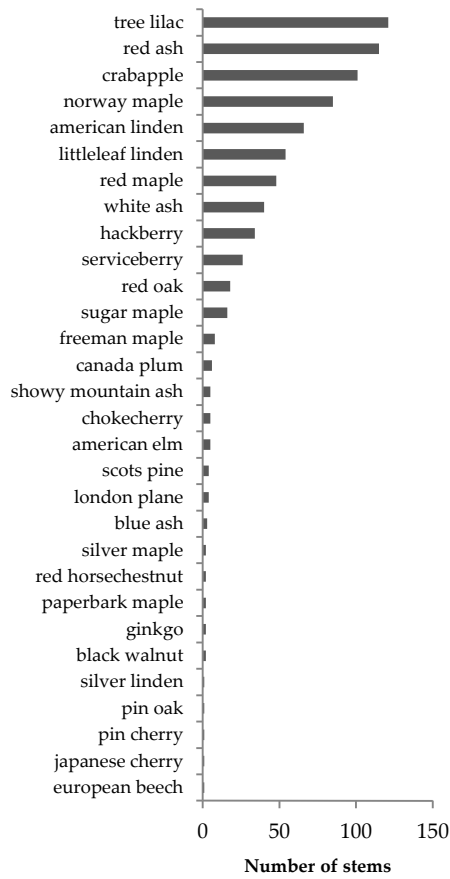
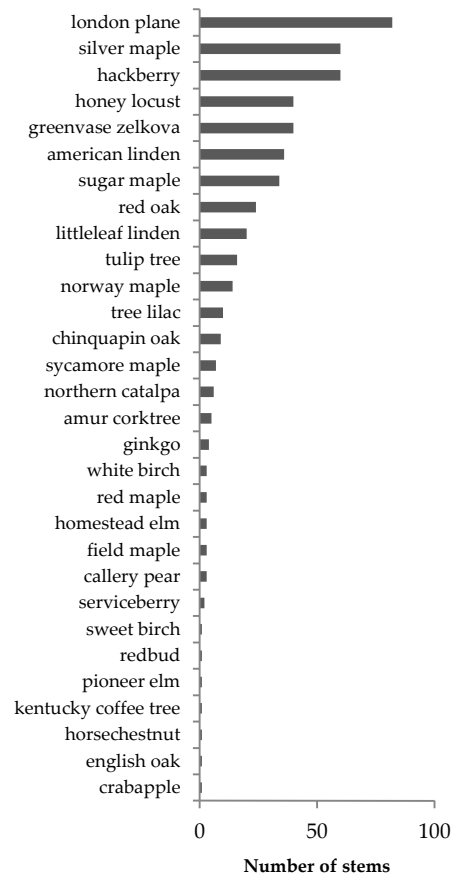


Figure 8 Combined relative species abundance of street trees in neighbourhoods developed 40-50 years ago: (a) Relative species abundance of street trees in two neighbourhoods in Halifax; (b) Relative species abundance of street trees in two neighbourhoods in London.



(a) Halifax, 2000s



(b) London, 2000s

Figure 9 Combined relative species abundance of street trees in neighbourhoods developed <15 years ago: (a) Relative species abundance of street trees in two neighbourhoods in Halifax; (b) Relative species abundance of street trees in two neighbourhoods in London.

Table 5 Diversity indices of street tree communities combined according to neighbourhood.

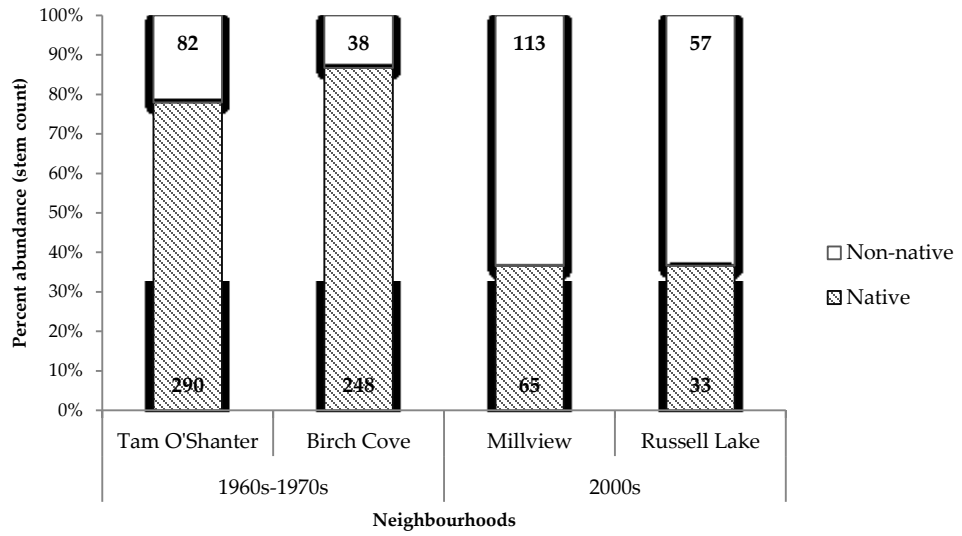
City	Decade of development	Neighbourhood	Shannon Index (H') ¹	Simpson Index ($1-D$) ²	Count proportion of native trees
Halifax	1960s-1970s	Tam O'Shanter	1.34	0.65	0.11
	1960s	Birch Cove	1.26	0.57	0.03
	2000s	Millview	2.45	0.90	0.36
	2000s	Russell Lake West	2.66	0.91	0.37
London	1960s-1970s	Masonville	1.79	0.68	0.26
	1960s	Sherwood Forest	1.87	0.77	0.45
	2000s	Byron	2.52	0.89	0.57
	2000s	Stoney Creek	2.63	0.91	0.62

¹ Higher number indicates a more diverse community; ² Higher number indicates a more even community.

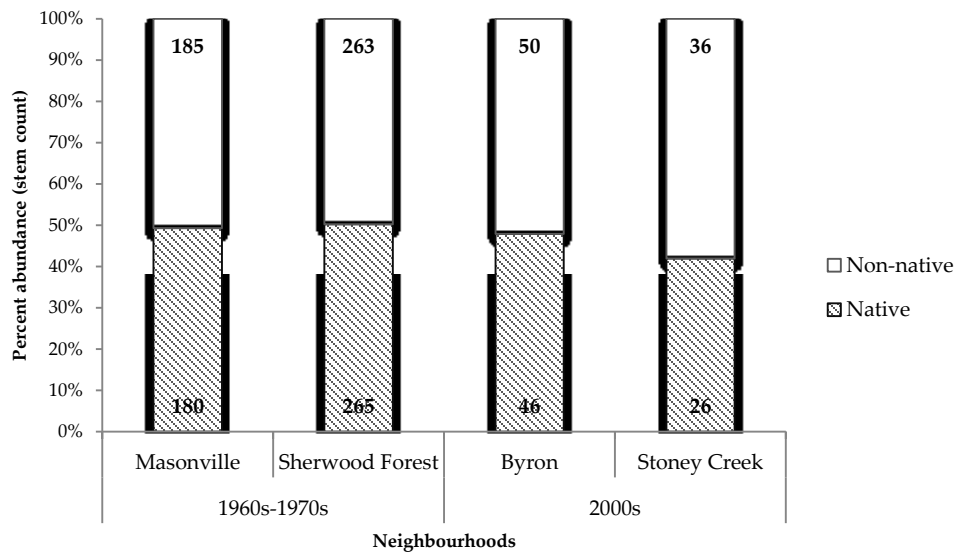
3.4.2.3 Residential property trees

Contrasting patterns in the composition and diversity of residential property trees were observed between the two cities. In Halifax, older neighbourhoods had a significantly greater proportion of native trees compared to newer neighbourhoods (Figure 10). Older neighbourhoods in Halifax also displayed higher proportions of native trees compared to older neighbourhoods in London. In London, there were no notable differences between proportions of native and non-native trees across old and new neighbourhoods.

No statistical difference was observed between the diversity of property trees in older and newer neighbourhoods within each city (Table 6). However, a statistical difference did exist ($t = 5.77$, $p = 0.029$, $d.f. = 2$) between older neighbourhoods in Halifax (mean $H' = 2.44$, $SE = \pm 0.155$) and older neighbourhoods in London (mean $H' = 3.46$, $SE = \pm 0.085$). A similar relationship was not observed between newer neighbourhoods in both cities.



(a) Halifax



(b) London

Figure 10 Counts and proportions of native and non-native trees for residential properties in each neighbourhood: (a) Halifax; (b) London.

Table 6 Diversity indices of residential property tree communities combined according to neighbourhood.

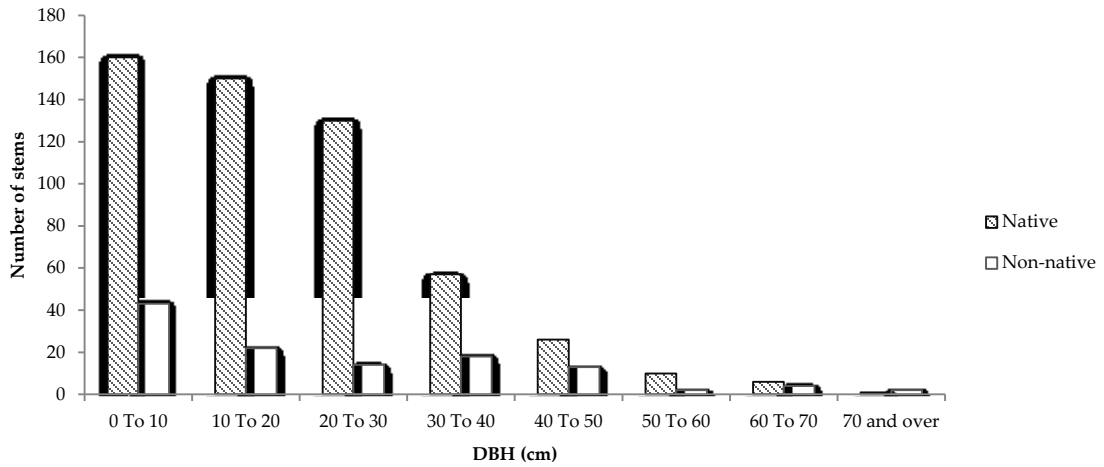
City	Decade of development	Neighbourhood	Shannon Index (H') ¹	Simpson Index ($1-D$) ²	Count proportion of native trees
Halifax	1960s-1970s	Tam O'Shanter	2.28	0.75	0.78
	1960s	Birch Cove	2.59	0.89	0.87
	2000s	Millview	3.09	0.91	0.37
	2000s	Russell Lake West	2.76	0.93	0.37
London	1960s-1970s	Masonville	3.37	0.95	0.50
	1960s	Sherwood Forest	3.54	0.95	0.50
	2000s	Byron	3.09	0.94	0.48
	2000s	Stoney Creek	3.14	0.95	0.42

¹ Higher number indicates a more diverse community; ² Higher number indicates a more even community.

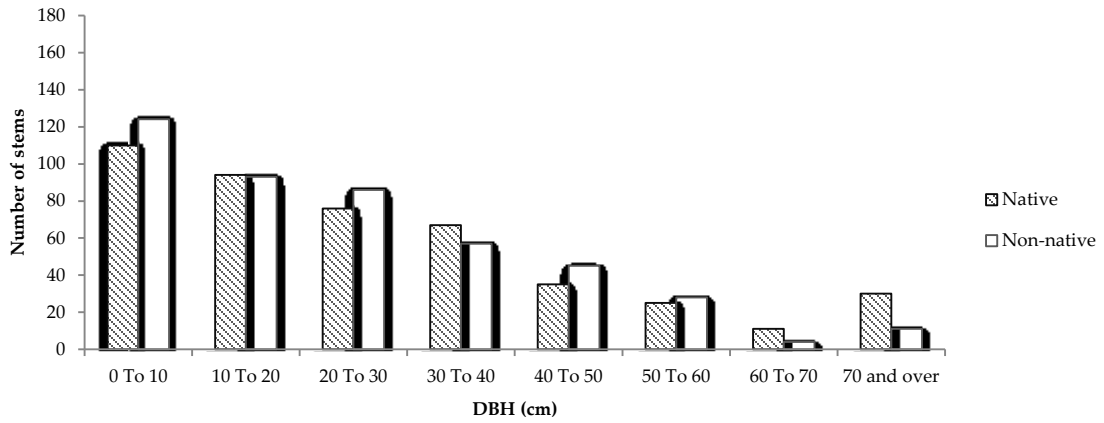
3.4.3 Size-class diversity

Size-class distributions were created to determine the composition of native and non-native trees according to DBH on residential properties, and to serve as a proxy for tree age. The size-class distribution of older neighbourhoods differed considerably between the two cities. In older neighbourhoods in Halifax, approximately 79% of small trees (DBH < 10cm) were native. In comparison, 47% of trees of the same size class in older neighbourhoods in London were native. Furthermore, the number of native and non-native stems remained relatively equal and consistent across size classes in London (Figure 11). Conversely, in Halifax almost 60% of all native trees had a DBH below 20 cm.

In newer neighbourhoods, the size-class distribution of native and non-native trees was similar across both cities (Figure 12). Some older native trees were left standing in developments in both cities, while non-native trees dominated newly planted stems.

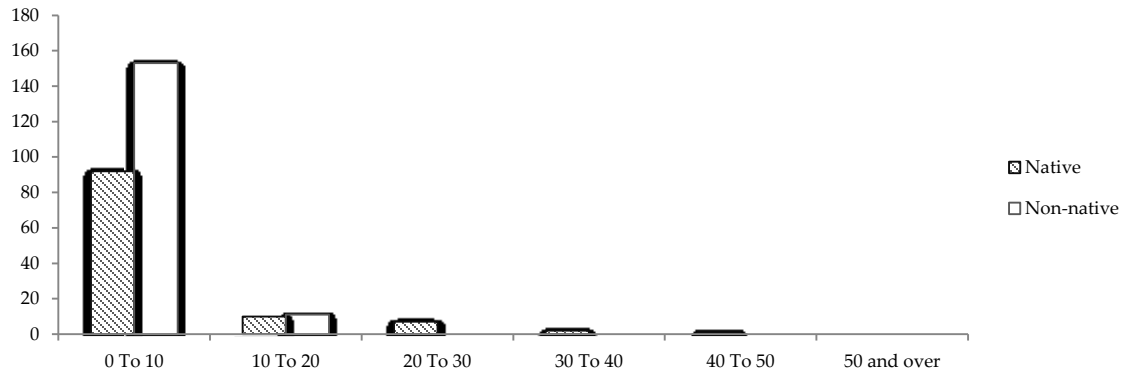


(a) Halifax, 1960s-1970s

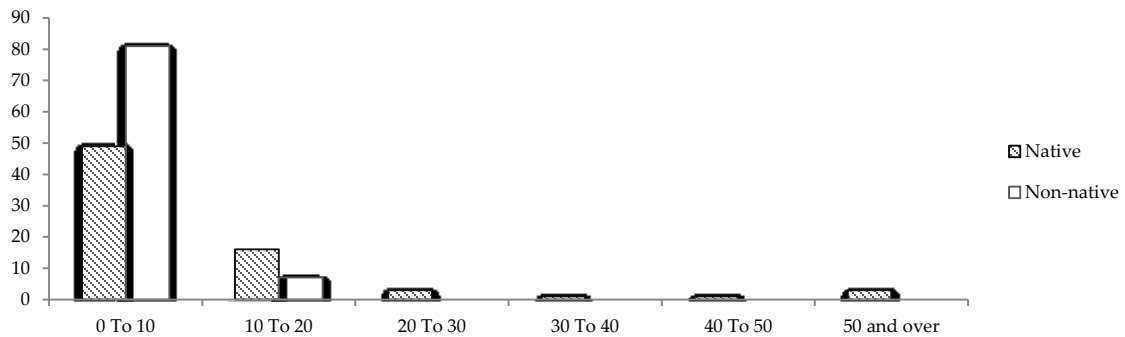


(b) London, 1960s-1970s

Figure 11 Size-class distribution of native and non-native trees on residential properties in neighbourhoods developed in the 1970s-1970s in: (a) Halifax; (b) London. Trees with a DBH=10 cm, 20 cm, etc. are included in the lower histogram bin.



(a) Halifax, 2000s



(a) London, 2000s

Figure 12 Size-class distribution of native and non-native trees on residential properties in neighbourhoods developed in the 2000s in: (a) Halifax; (b) London. Trees with a DBH=10 cm, 20 cm, etc. are included in the lower histogram bin.

3.4.4 Chi-squared analysis

Four contingency tables were used to analyze the relationship between two factors (the pre-urbanized landscape and age of development) and counts of native and non-native trees on residential properties (Tables A1-A4). A strong and statistically significant relationship was found between the type of pre-urbanized landscape and counts of native and non-native trees in neighbourhoods developed during the 1960s-1970s (Table 7). Halifax had a significantly greater number of native trees compared to non-native trees in older neighbourhoods, while London had virtually equivalent

numbers of native and non-native trees. The equivalent relationship in newer neighbourhoods was not found to be significant. Another important relationship existed in Halifax, but not in London, between decade of development and counts of native and non-native trees. Newer neighbourhoods in Halifax were dominated by non-native trees, while the opposite occurred in older neighbourhoods. In comparison, count proportions of native and non-native trees remained consistent across new and old neighbourhoods in London.

Table 7 Results and statistical significance of chi-squared tests.

Tests	df	X²	p-value
O ¹	1	166.42***	<0.005
N ²	1	3.36	0.067
H ³	1	180.85***	<0.005
L ⁴	1	0.98	0.323

¹ Relationship between the pre-urbanized landscape (Halifax and London) and counts of native and non-native trees in old neighbourhoods.

² Relationship between the pre-urbanized landscape (Halifax and London) and counts of native and non-native trees in new neighbourhoods.

³ Relationship between neighbourhood age and counts of native and non-native trees in Halifax.

⁴ Relationship between neighbourhood age and counts of native and non-native trees in London.

3.5 DISCUSSION

3.5.1 Development and tree-species composition

The various land types that are found in suburban areas reflect differences in tree establishment, planting practices, maintenance, and ownership, and are thus influenced by different drivers of tree-species composition. The sites sampled in this study ranged from near-natural remnant forests to the urbanized streetscape. In both cities, streets in older neighbourhoods were dominated by only a few species, including the ubiquitous Norway maple. In comparison, newer neighbourhoods had greater species richness and evenness as well as a better representation of native species, which have been shown to

support insect and bird diversity more effectively than non-native species (Burghardt et al. 2009; Ilkin et al. 2013).

Remnant stands represented the most “natural” forest habitat found in suburban areas, and contained few (if any) non-native trees. Nevertheless, plots in Halifax supported two non-native tree species, both of which are invasive. Given the close proximity of the remnant sites to the adjacent residential area, it is probable that these species were planted on private property some time after development occurred, and managed to disperse into the remnant stands.

Many residential properties were intensively landscaped, and housed common ornamental non-native species like Norway maple, blue spruce (*Picea pungens*), and Japanese maple (*Acer palmatum*). Properties in newer neighbourhoods had fewer trees, of which a majority were non-native species. Conversely, properties in older neighbourhoods in Halifax held a large amount of native trees and resembled the remnant woodland sites more closely than properties in newer developments in terms of species composition. The size-class distributions suggest that some trees were kept during development in neighbourhoods in both cities; it is likely that large (and old) non-native trees found in London were planted on farms or along fence rows. These patterns illustrate the importance of examining changes in species distributions across remnant forests and adjacent green spaces, particularly as some non-native species can threaten native habitats and potentially displace native species (Turner et al. 2005; McKinney 2006).

3.5.2 Policy and administration

Street trees are generally planted by a developer or contractor, and are maintained by the municipality. Despite the contrasting development patterns (e.g. the pre-urbanized landscape) of the selected neighbourhoods, the patterns exhibited in street-tree communities observed between the two cities are very similar. Given street-tree planting practices, it is certain that political and administrative factors play a large role in determining the types of trees found in city streetscapes.

The drastic difference between street-tree composition in newer versus older neighbourhoods in both cities could reflect changes in urban forest management and policy that have occurred in the last few decades. The infiltration of diseases and insects like Dutch elm disease and the emerald ash borer resulted in mass tree mortality rates in Canada in the last century, and has heightened awareness of the importance of diversifying planting stock (Poland & McCullough 2006; Raupp et al. 2006; Steenberg et al. 2013). Many urban forest management plans created more recently have developed targets to enhance native and overall species diversity, illustrating concerns about invasion events and urban forest canopy loss (Ordóñez & Duinker 2013). Higher evenness and native species representation in newer neighbourhoods in both cities may reflect a growing diversification trend among urban forest practitioners across Canada.

The differences observed in street-tree diversity between older and newer neighbourhoods could also be due in part to changes in nursery stock availability over time. Large-scale projects, like street-tree planting in new subdivision developments, often acquire trees from regional wholesalers instead of local nurseries (Conway & Vander Vecht 2015). At a given time, species selection may be limited due to location,

resources, or customer demand; some tree species requested by the developer or urban forester may be substituted or outsourced, potentially resulting in more unfavourable species choices and reducing diversity enhancements (Sydnor et al. 2010). However, some research in the United States has shown that nurseries are offering more species choices today compared to decades ago (Pincetl et al. 2013). More recent demand for a wider range of species may therefore be influencing nurseries and wholesalers to diversify their stocks, in turn promoting higher species richness and evenness as well as native species representation in newer subdivision developments.

Various practitioners are both directly and indirectly involved in the process of street-tree species selection and planting, including municipal officials, arborists, urban planners, foresters, developers, and contractors. These groups reflect differences in expertise, values, priorities, and professional paradigms (Kirkpatrick et al. 2013; Conway & Vander Vecht 2015); these in turn can dictate the species composition of street trees. For example, the popularity of tree lilac and London plane (*Platanus x acerifolia*) in newer neighbourhoods could be due to the preference of a particular urban forester or planner, based on factors such as tolerance to urban stresses, maintenance, growth rate, shade value, and cost.

3.5.3 The influence of remnant woodland

Although little research has been carried out on the subject, it has been shown that development patterns, including the landscape present prior to urbanization, can influence the composition of urban tree species. One study found that residential neighbourhoods developed from forested areas had higher tree species richness and a greater proportion of native species compared to regions developed from prairie land (Fahey et al. 2012).

Researchers have also suggested that the presence of remnant woodland located within or adjacent to residential areas can promote native species establishment and representation within the urban landscape (Doody et al. 2010; Ranta & Viljanen 2011).

The results of this study seem to illustrate these processes to some extent. The significant difference between both cities in native and non-native stem counts in older neighbourhoods suggests that native trees in Halifax may be dispersing from woodland areas retained during and after development onto adjacent residential properties. Furthermore, the much higher number of native trees found on older residential properties compared to newer ones in Halifax potentially demonstrates a lag effect; the tree-species composition of newer neighbourhoods may be influenced primarily by home owner planting preferences, while the species composition of older neighbourhoods may be shaped both by home owner planting decisions and the natural establishment of some trees over time as the development aged.

Assuming that stems with a DBH < 10 cm are younger than trees with a larger DBH, the size-class distributions for Halifax show that many native trees are younger, suggesting that these stems appeared on residential properties a while after development occurred. The results also indicate that this phenomenon was not observed to the same degree in London, which could be due to the lack of remnant woodland in and around the selected neighbourhoods. The size-class distributions of native and non-native trees in London are very consistent across new and old neighbourhoods, therefore it is more likely that the species composition on private properties is influenced primarily by the planting attitudes and preferences expressed by home owners. Furthermore, the strikingly similar size-class distributions of trees in newer neighbourhoods in Halifax and London

imply that the influence of remnant woodland is neither felt during nor immediately after development, but rather over time as trees are gradually given the chance to disperse into adjacent green space.

Despite the fact that private properties in Halifax had a better representation of native species, properties in London were generally more diverse, both in terms of species richness and evenness. This pattern could be attributed to differences in tree establishment; the tree-species composition of gardens in London mostly reflects individual planting preferences, and many home owners prefer non-native ornamental species for horticultural purposes, which contribute to overall species diversity (Henderson et al. 1998; Turner et al. 2005; McKinney 2006). In a similar vein, this trend could also explain why the diversity measures characterizing newer neighbourhoods in Halifax and London are more similar compared to the older neighbourhoods, where naturalization is likely occurring in Halifax, but not in London, due to native species dispersal.

Depending on diversity targets and priorities, neighbourhoods in cities like Halifax may benefit from retaining and protecting forest buffers and patches during and after development. Not only are remnant areas important for promoting native tree-species establishment, but other organisms may benefit from retaining woodland during development, illustrating the crucial role that urban forests play in biodiversity conservation (Barth et al. 2015).

3.5.4 Opportunities for diversity enhancement

In both cities, residential properties were found to be most diverse compared to other land types, confirming the results from other studies on land tenure and tree-species

composition (Turner et al. 2005; Dobbs et al. 2013; Bourne & Conway 2014). Home owner attitudes and preferences likely explain the diverse range of planting decisions; many social and economic factors taking place at the property- and neighbourhood-level can influence residential species diversity, such as income (Hope et al. 2003), education (Luck et al. 2009), ethnicity and nationality (Fraser & Kenney 2000), and planting fads (Nassauer et al. 2009). These patterns emphasize the potential for private yards and gardens to contribute to urban forest biodiversity (Doody et al. 2010; Goddard et al. 2010). It is thus important to consider how landscape conversion and development history could influence private property trees, particularly in the context of biodiversity management and enhancement.

In the context of native species diversity, it is worthwhile to explore home owner attitudes towards native species establishment, and whether residents knowingly allow native trees to establish on their property. Research has shown that although urban dwellers may in theory support the planting of native species, they might not be knowledgeable enough to identify native trees and recognize their ecological importance (Doody et al. 2010). Although results from this study have shown that native trees are probably dispersing onto private properties, the exact establishment mechanisms and potential decisions on the part of home owners have not been identified.

Subdivisions developed in cities like London, where most forested area has already been converted into agricultural fields, may benefit from targeted education initiatives and planting projects geared towards increasing native species representation and overall species richness, depending on the diversity goals of the municipality in question. Educating residents about the benefits of both native and non-native species

may result in more informed planting choices, and in turn could influence the availability of local nursery stock (Conway & Vander Vecht 2015). Education initiatives geared towards nursery workers and wholesalers could also influence resource availability and encourage the dissemination of information about tree diversity and its benefits to customers (Polakowski et al. 2011). London in particular has a relatively broad palette of native species to choose from, and encouraging a wider range of species selection on residential properties could give rise to diversity benefits at the city level.

Researchers have stressed the importance of increasing green space connectivity in urban and suburban areas for the purpose of increasing resilience to environmental change, providing wildlife habitat, and enhancing biodiversity (Rudd et al. 2002). Regardless of pre-urbanized landscape, suburban areas may effectively enhance diversity if the design of residential properties and neighbourhoods is considered and incorporated into biodiversity initiatives or urban forest management plans implemented at the city level (Goddard et al. 2010). It is worth noting that subdivision design should consider the possible negative impacts of residential encroachment onto remnant woodland, including loss and/or destruction of forest habitat, waste disposal and littering, and introductions of exotic species (McWilliam et al. 2015). Neighbourhood-level urban forest strategies could educate residents about remnant forest protection (particularly in riparian areas) and identify specific biodiversity issues and opportunities at the community level, encouraging partnerships between municipalities, planners, developers, researchers, and residents (Poland & McCullough 2006; Goddard et al. 2010). Citizen engagement can be particularly useful in fostering public stewardship of the urban forest and in achieving municipal tree diversity goals (HRM 2013).

3.5.5 Limitations and research needs

The advantages of this study design relate to the fact that a large number of trees on different land types were sampled, allowing for robust analyses and conclusions about species composition at the neighbourhood level. However, it is difficult to generalize the results of this study to other residential areas or other cities, as only four neighbourhoods in each city were examined. Furthermore, this study did not examine parks, a land type that has been extensively researched and plays an important role in urban forest diversity management (Godefroid & Koedam 2003; LaPaix & Freedman 2010; Pennington et al. 2010).

Characterizing urban forest diversity is rendered more complex with the introduction of different land types (e.g., streets, private properties, remnant stands) that each require particular sampling protocols in order to effectively capture trends related to land use. Given the sampling design, the areas investigated for each land type in each neighbourhood are necessarily different. Although species accumulation curves are often used to assess sampling saturation, urban species composition does not always reflect natural forest establishment and succession patterns that conform to traditional species-area relationships (Zhao et al. 2010*b*). Private property lots in particular do not represent plots sampled in natural settings (e.g. remnant woodland); for example, some homeowners plant a line of trees of the same species, while others choose to plant, say, eight trees representing eight species. The number of singletons is thus relatively high, which influences calculations for species curves. Therefore it may not be as relevant to plot traditional species curves for communities that were not naturally established.

Although this study focused on the pre-urbanized landscape as a major driver of urban forest diversity, there are many other biophysical, socioeconomic, and political factors that influence urban tree diversity. In residential areas, lot size, which was not controlled for in this case, has been shown to correlate with urban tree diversity (Tratalos et al. 2007; Kendal et al. 2012). However, the results from this study showed a negligible correlation between property lot size and both number of stems and species richness (Figures A1, A2). Exploring how other drivers pertaining to urban morphology and neighbourhood design could promote urban forest diversity and green space connectivity would also be an important endeavour, particularly as suburban areas continue to expand in Canadian metropolitan areas.

Future research should explore home owner decisions about species selection for planting and the natural establishment of native trees on residential properties from remnant woodland. It would be worthwhile to determine whether residents purposefully allow for native species establishment on private property, and what kind of motivations are involved. Furthermore, creating and implementing education and planting initiatives geared towards home owners and community groups could shed light on the types of strategies that are most effective for biodiversity promotion in residential areas.

3.6 SUMMARY

The results of this study emphasize the importance of considering both spatial and temporal contexts associated with urban forest creation and management. The land types found in suburban areas represent divergences in tree establishment, planting, and maintenance practices, and thus reflect various manifestations in tree-species composition. Urban forest policies and management traditions likely explain differences

observed in street-tree species composition between older and newer neighbourhoods. The pre-urbanized landscape, on the other hand, appears to be an important driver of urban forest composition on residential properties, as native species can naturally establish onto adjacent green spaces from forest buffers and patches, and potentially alter tree diversity over time.

Studies like these can encourage urban forest practitioners to reflect on the ways in which cities are developed, and how green infrastructure is created and molded based on the preferences and priorities of multiple professionals and urban dwellers. Envisioning how urban areas can be built to enhance urban forest diversity also allows for the opportunity to contemplate the kind of diversity that we would like to see in our cities. Although tree communities in London were generally more diverse, Halifax had greater proportions of native trees on residential properties. Is one case better than the other? What elements of urban forest diversity should be prioritized when developing and implementing management plans? These are crucial considerations, especially given growing concerns about biodiversity loss and environmental disturbances like pests, diseases, and climate change, which threaten the success of trees in urban landscapes. Given these difficulties, managers and practitioners should work towards achieving the *right* kind of biodiversity, instead of simply aiming for *more* biodiversity.

CHAPTER 4 LAND TYPES AND SPECIES DOMINANCE

4.1 ABSTRACT

The goal of this study was to explore where tree species are found in the suburban landscape using a neighbourhood-level approach and by assessing abundance (stem counts) and dominance (basal area). Four suburban neighbourhoods in two Canadian cities were chosen: Halifax, Nova Scotia and London, Ontario. In each city, trees were sampled on three land types that represented differences in tree establishment, planting practices, species selection, and maintenance: remnant woodlands, streets, and private residential properties. For each land type, diversity indices were calculated using relative stem counts and basal area, and proportional species representation was determined. The differences between diversity indices calculated using abundance and dominance were not found to be significant in woodland and street tree communities, but they were significant for trees growing on residential properties. Only a few native species were consistently represented in the majority of the canopy in remnant stands and private properties in Halifax, while London had a richer and more even distribution of species on properties. Newer neighbourhoods in both cities housed a greater number of small non-native stems, yet large native trees dominated in terms of basal area, emphasizing the importance of retaining trees during development to promote biodiversity in suburban neighbourhoods. The results illustrate that basal area is an important measure of tree-species dominance, and should be used in conjunction with stem counts when assessing tree diversity in an urban setting.

4.2 INTRODUCTION

Assessing the structure and tree-species composition of the urban forest is crucial to the understanding of its ecosystem functioning and resilience (Stone 1996; Alvey 2006). Planting for tree diversity decreases the risk of mass mortality due to pest and disease outbreaks, increases output of ecosystem services, and generates learning opportunities as well as positive conservation attitudes (Alvey 2006; Goddard et al. 2010). The presence of native tree species in particular can promote native insect and bird diversity as well as enhance the ecological integrity of urban ecosystems, illustrating the importance of determining how and why species patterns emerge (Burghardt et al. 2009; Ordóñez & Duinker 2012; Ilkin et al. 2013).

Urban areas incorporate different types of spaces on which trees are found. These are spaces that represent differences in land use, tree planting practices, species-selection decisions, and ownership (Kirkpatrick et al. 2011; Bourne & Conway 2014). Land types can be defined in various ways depending on the scale of analysis; some scholars may define land classes based on use, such as institutional, agricultural, commercial, and residential (Bourne & Conway 2014). For the purpose of this paper, the land types examined are found in suburban neighbourhoods (save for one exception in London) and are therefore defined by differences in land use and jurisdiction within residential areas. Exploring variations in land use and land tenure sheds light on how these differences determine urban forest composition and can inform management strategies for diversity enhancement (Alvey 2006). However, studies examining relationships between land type and urban species richness and composition have primarily been carried out at the county or municipal level using random or stratified sampling techniques (Iverson & Cook 2000;

Jim & Liu, 2001; Zhao et al., 2010a; Kirkpatrick et al., 2011; Kendal et al., 2012; Dobbs et al., 2013; Bourne & Conway, 2014). As such, these types of studies may fail to capture ecological and socio-political processes occurring at a neighbourhood scale related to development patterns, urban morphology, biophysical characteristics, and socio-demographics. Furthermore, determining patterns and trends in tree-species composition and structure at the neighbourhood level may foster citizen stewardship of the urban forest, which proves particularly useful if the private urban forest (comprised of trees located on private residential properties) plays a significant role in the provisioning of green infrastructure benefits. It has thus been suggested that conducting both neighbourhood-based and municipal-level urban forest assessments is ideal for effective stewardship and management of trees in the city (Steenberg et al. 2013).

A number of tools can be used to assess urban forest composition. Many scholars use species and individual abundance (e.g. stem count) measures to calculate diversity indices, like Shannon-Weaver, which incorporate species richness and evenness (Savard et al. 2000; Jim & Liu 2001; Dobbs et al. 2013; Bourne & Conway 2014). However, basal area, defined as the cross-sectional area of a tree stem measured at breast height, can also be used to calculate the same indices and can provide valuable information about the biodiversity provisioning services of a tree community as well as its vulnerability to environmental perturbations (Motz et al. 2010). Basal area correlates with tree size, leaf surface area, and biomass, and provides a good indication of the amount of canopy available for wildlife habitat (Bartelink 1997; Chen et al. 1997).

Furthermore, assessing tree composition on the sole basis of stem count does not reveal how much tree cover would be lost if a species- or genus-specific pest emerged.

To reduce the risk of mass mortality and the subsequent substantial loss of ecosystem services, a lack of overwhelming single-species dominance in the canopy is preferable. It has been suggested that native species should be prioritized in urban environments, as they are better adapted to the local ecosystem and are less likely to become invasive and out-compete other native species (Turner et al. 2005). Although some scholars disagree about the importance of nativeness in urban forests (Kendle & Rose 2000; Chalker-Scott 2015), it has been shown that native tree species might house a greater richness of insects and mites compared to more recently naturalized species (Brändle & Brandl 2001). Native trees promote native bird diversity more effectively than non-native trees, indicating the importance of having an adequate representation of nativeness in the canopy for biodiversity promotion (Ilkin et al. 2013; Barth et al. 2015).

The goal of this study was to assess the influence of land type on tree-species composition in suburban neighbourhoods, by considering basal area as well as the more conventional stem counts. Exploring forest diversity in suburban neighbourhoods in Canada is particularly worthwhile, given that two thirds of the country's population lives in suburban areas which continue to grow at rates greater than downtown centres and exurban regions (Gordon & Shirokoff 2014). The rapid suburbanization of Canadian metropolitan areas creates new challenges for natural resource management, emphasizing the necessity to retain, create, and protect sustainable and long-lasting urban forests.

4.3 MATERIALS AND METHODS

4.3.1 Study area and sites

Halifax, Nova Scotia and London, Ontario are located approximately 1,500 km apart, in two distinct forest regions. Halifax, a city primarily developing onto woodlands,

is located in the Acadian forest region, while London, developing onto farm fields, is located at the northern edge of the Carolinian forest region. In each city, four suburban residential neighbourhoods were chosen: two were developed 40-50 years ago, and two were developed less than 15 years ago.

For the purpose of this study, a suburban neighbourhood was defined as a relatively homogeneous group of residential detached or semi-detached dwellings surrounded by lawn or green space, developed at approximately the same time. Neighbourhoods in Halifax were delineated based on the urban forest neighbourhood classification system carried out for the Halifax Urban Forest Master Plan, which considered factors like development history and age, land use and zoning, and biophysical characteristics (Steenberg et al. 2013) (Figure 13). Neighbourhoods in London were delineated based on development age, urban morphology, and size (Figure 14). The neighbourhoods selected ranged from approximately 25 to 50 hectares.

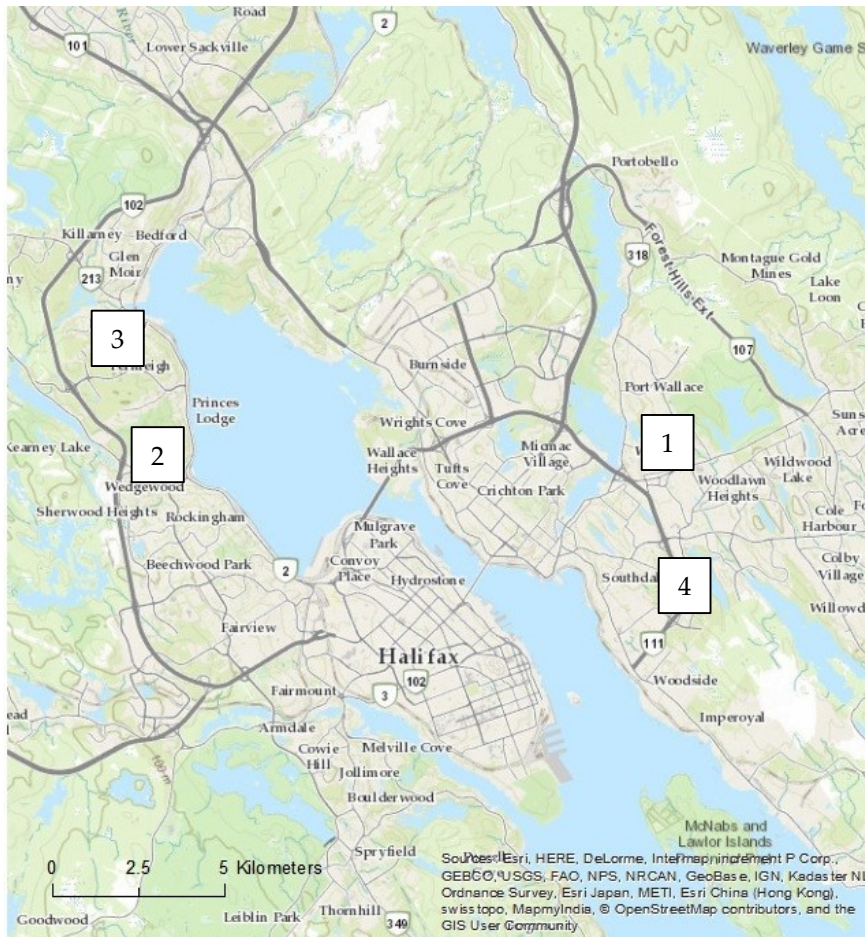


Figure 13 The neighbourhoods examined in Halifax: Tam O'Shanter (1), Birch Cove (2), Millview (3), and Russell Lake West (4).

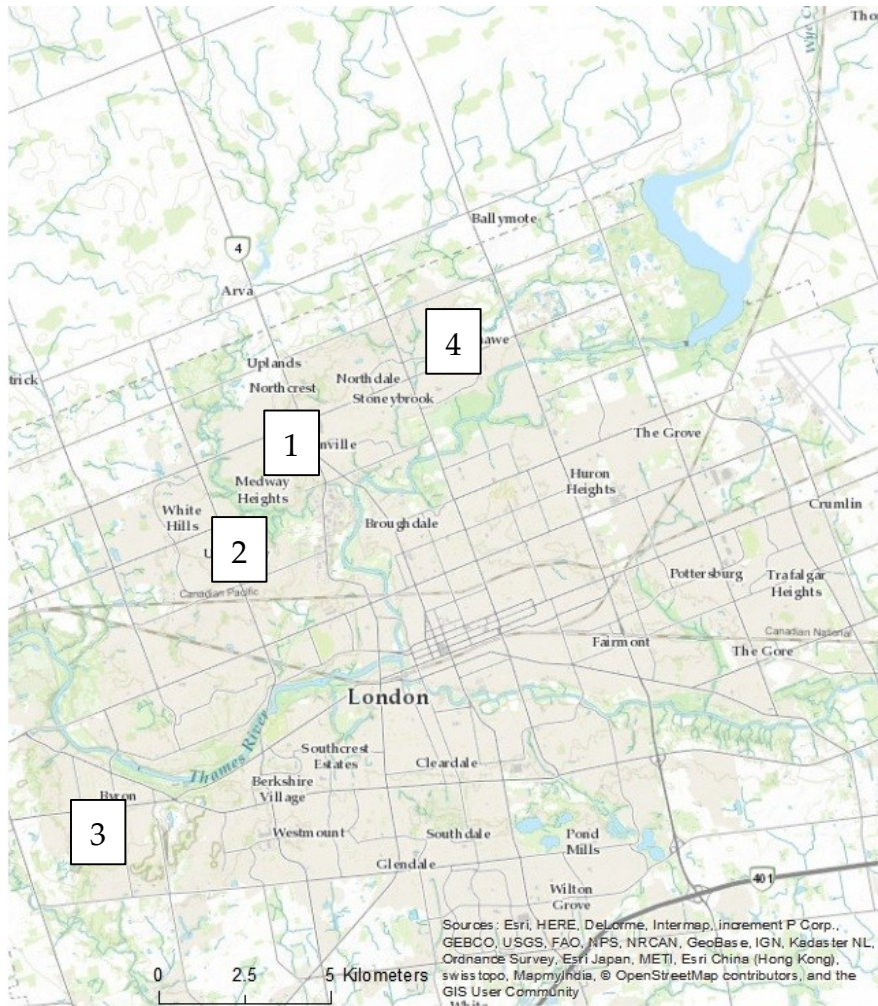


Figure 14 The neighbourhoods examined in London: Masonville (1), Sherwood Forest (2), Byron (3), and Stoney Creek (4).

4.3.2 Data collection

Three land types were examined in each city: remnant woodland, streets, and residential private properties. For the purpose of this paper, a tree “community” will refer to the trees found on a particular land type within a neighbourhood. Each tree sampled was identified at the species level and its stem diameter at breast height was measured. In Halifax, remnant stands were forested areas found adjacent to each selected residential development (Figure 15). At each site, five 10 m x 10 m plots were sampled (Figure A3). Given the lack of remnant stands within the city limits of London, two remnant sites were sampled outside the city in protected forest areas: Mosa Forest and Dorchester Swamp. At each site, three 20 m x 20 m plots were sampled (Figure A4). Smaller plots were measured in the Halifax remnant stands because of their small sizes and linear shapes.

Street trees were defined as those planted along streets, either between the curb and the sidewalk (in the “tree lawn”) or between the curb and residential front yards (Figure 16). All street trees within each selected neighbourhood were measured.

Residential property trees were defined as those growing on landscaped areas (e.g. lawn, flower beds) within the limits of residential property lots (Figure 17). Approximately 10% of the property lots within each neighbourhood were selected at random, and all trees within each lot were measured.



Figure 15 A remnant stand located adjacent to a suburban development.



Figure 16 Trees planted along a suburban street, in a tree lawn on the left and between the curb and residential front yards on the right.

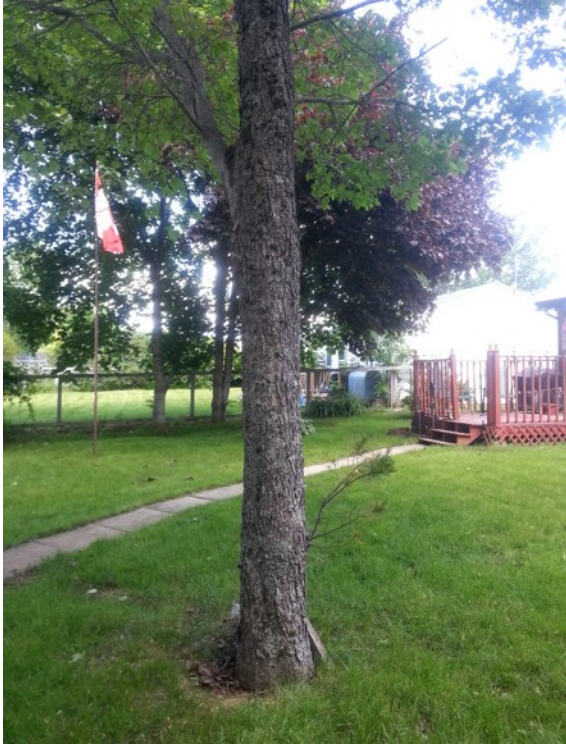


Figure 17 A tree growing in the back yard on a residential property.

4.3.2 Data analysis

For all species, relative abundance and relative dominance were measured on every land type and in each neighbourhood according to the following formulas

(McPherson & Rowntree 1989):

$$\text{Relative } \mathbf{abundance} = \frac{\# \text{ individuals of species } i}{\text{total \# of individuals}}$$

$$\text{Relative } \mathbf{dominance} = \frac{\text{basal area of species } i}{\text{total basal area}}$$

Matrices were created and colour-coded according to the values of relative abundance and dominance for each species (Table 8). These proportions were calculated for each species encountered. For the purpose of this analysis, only the largest

proportions up to a sum of 80% were included, to elucidate major patterns in species composition on each land type. Therefore some rarer species were left out of the analysis (Figures A5, A6). In each neighbourhood, the largest proportion (taking into account both abundance and dominance) was calculated first, and subsequently smaller proportions were added approximately 80% of the total was reached. This number was chosen to ensure that the majority of species contributing to the canopy were identified while eliminating the species that are more negligible and only account for 1% – 2% of the tree community.

Finally, the Shannon-Weaver Index (Van Dyke 2008; Motz et al. 2010) and native species proportions of each tree community were calculated using both relative abundance (stem count) and relative dominance (basal area). Two-sample *t*-tests were used to determine whether significant differences in compositional indices are observed when comparing the different measures.

Table 8 Colour code according to the relative abundance or dominance for each species.

Proportion of abundance or dominance	Colour
0 – 9.99%	
10 – 24.99%	
25 – 49.99%	
50 – %	

4.4 RESULTS

4.4.1 Remnant stands

Native species dominated remnant sites sampled in both cities, in terms of relative abundance and dominance. The sampling plots were also generally dominated by few species. Red maple (*Acer rubrum*) and red spruce (*Picea rubens*) were ubiquitous in all remnant stands sampled in Halifax, and combined they contributed between 40-60% of

the canopy in terms of abundance and 50-85% in terms of dominance. Other remnant species of note included American beech (*Fagus grandifolia*), and eastern hemlock (*Tsuga canadensis*) (Tables 9, 10).

Red maple also dominated stands sampled in London, contributing up to 65% of the basal area in one site (Tables 11, 12). Other large trees included black cherry (*Prunus serotina*), large-tooth aspen (*Populus grandidentata*), and white pine (*Pinus strobus*).

4.4.2 Street trees

In both cities, the dominance of one or a few species in the street was influenced by neighbourhood development age. Older neighbourhoods were much more likely to have only a couple of species account for 80% of the canopy. In Halifax, non-native species like littleleaf linden (*Tilia cordata*) and Norway maple (*Acer platanoides*) were most abundant and dominant in older neighbourhoods (Tables 9, 10). In London, American linden (*Tilia americana*) and Norway maple were street-tree species of choice (Tables 11, 12).

Newer neighbourhoods in Halifax had more-even representation of street trees, with red ash (*Fraxinus pennsylvanica*), crabapple (*Malus* spp.), tree lilac (*Syringa reticulata*), and Norway maple comprising approximately 50% of the canopy in terms of both abundance and dominance. In London, species like hackberry (*Celtis occidentalis*), silver maple (*Acer saccharinum*), London plane (*Platanus x acerifolia*), and sugar maple (*Acer saccharum*) were more commonly found.

4.4.3 Residential properties

Most species dominating private properties in older neighbourhoods in Halifax were those also found in remnant areas, including red maple, red spruce, and American

beech. Some exceptions include white birch (*Betula papyrifera*) and Norway maple. The latter was more popular in newer neighbourhoods, comprising approximately 20% of the canopy in terms of abundance. Although non-native species like blue spruce (*Picea pungens*) and Japanese maple (*Acer palmatum*) were more abundant in newer neighbourhoods, native species were far more dominant in terms of basal area.

In London, the most abundant property species in older neighbourhoods included a mix of both native and non-native species: white spruce (*Picea glauca*), eastern red cedar (*Juniperus virginiana*), blue spruce, and Norway maple. In newer neighbourhoods, a similar trend to Halifax was observed, where native species like hackberry, sugar maple, red oak (*Quercus rubra*) and ironwood (*Ostrya virginiana*) accounted for most of the total basal area.

Table 9 Relative abundance (based on stem count) of species in Halifax.

Native species	Tam O'Shanter			Birch Cove			Russell Lake West			Millview		
	Remnant	Street	Residential	Remnant	Street	Residential	Remnant	Street	Residential	Remnant	Street	Residential
alternate leaf dogwood						0.028						
american beech				0.203		0.115				0.075		
american elm												
balsam fir							0.153					
eastern hemlock				0.095								0.028
pin cherry						0.031						
red ash								0.131			0.156	
red maple	0.263		0.481	0.176		0.185	0.162	0.071	0.089	0.265	0.057	0.090
red oak	0.075		0.024									
red spruce	0.247		0.024	0.257		0.182	0.468		0.100	0.252		0.045
serviceberry	0.108					0.063						
showy mountain ash			0.035									
staghorn sumac			0.035									
striped maple											0.238	
trembling aspen			0.043									
white ash								0.079	0.111			0.022
white birch			0.032			0.154	0.117					
white spruce												0.084
yellow birch	0.065		0.024	0.081								
Non-native species												
american linden								0.071	0.022		0.092	
blue spruce									0.056			0.124
canada plum									0.033			
common apple									0.033			
crabapple								0.146			0.121	0.039
euopean alder	0.086											
hackberry											0.051	
japanese maple									0.044			0.079
littleleaf linden		0.333			0.181			0.086			0.061	0.039
norway maple			0.105		0.624	0.042		0.094	0.222		0.117	0.163
silver maple									0.056			
tree lilac		0.481						0.169	0.033		0.148	

Table 10 Relative dominance (based on basal area) of species in Halifax.

Native species	Tam O'Shanter			Birch Cove			Russell Lake			Millview		
	Remnant	Street	Residential	Remnant	Street	Residential	Remnant	Street	Residential	Remnant	Street	Residential
american beech				0.105		0.117				0.132		
eastern hemlock				0.191								
pin cherry						0.026						
red ash								0.169			0.237	
red maple	0.357		0.515	0.144		0.265	0.314	0.092	0.330	0.391		0.251
red oak	0.067		0.041						0.236			
red pine						0.025						
red spruce	0.320			0.365		0.134	0.548		0.186	0.225		0.175
showy mountain ash			0.025									
white ash						0.045		0.072				0.021
white birch						0.136						
white pine	0.053									0.070		
white spruce			0.027									0.024
yellow birch	0.051								0.073			
Non-native species												
american linden											0.157	
blue spruce												0.036
crabapple			0.028					0.166			0.112	
freeman maple												0.023
japanese maple												0.057
littleleaf linden		0.494			0.198			0.156			0.067	0.024
london plane												0.021
norway maple			0.166		0.680	0.064		0.072			0.125	0.147
tree lilac		0.362						0.128			0.107	

Table 11 Relative abundance (based on stem count) of species in London.

	Mosa	Dorchester	Masonville		Sherwood Forest	Byron		Stoney Creek		
Native species	Remnant	Remnant	Street	Residential	Street	Residential	Street	Residential	Street	Residential
american linden			0.101		0.348		0.070		0.078	0.048
bitternut hickory	0.040	0.057				0.023				
black cherry		0.156								
chinquapin oak									0.041	
eastern red cedar				0.079		0.119				
freeman maple				0.038		0.027		0.031		0.048
hackberry							0.103		0.146	0.032
honey locust			0.042	0.030		0.028	0.118		0.037	
ironwood	0.212							0.042		
large tooth aspen		0.047								
red maple	0.454	0.384								0.032
red oak	0.061							0.021	0.091	
red spruce								0.031		
redbud				0.022				0.042		0.048
silver maple							0.147		0.091	
sugar maple						0.072	0.059	0.156	0.082	0.048
tulip tree							0.033			
white birch								0.031		0.048
white cedar				0.041						
white oak	0.071									
white pine		0.047		0.025						
white spruce				0.117		0.044				0.048
witch hazel		0.081								
yellow birch		0.043								
Non-native species										
blue spruce				0.101		0.051		0.083		0.129
callery pear										0.065
columnar (european) hornbeam								0.031		
common apple						0.021				
common pear								0.021		
crabapple			0.056	0.049	0.091					
english oak								0.031		
european beech								0.073		0.065
european buckthorn						0.055				
greenvase zelkova							0.125			
japanese cherry										0.032
japanese maple				0.036		0.057		0.094		0.081
katsura tree										
littleleaf linden			0.098		0.141				0.068	
london plane							0.176		0.155	
norway maple			0.542	0.085	0.279	0.078		0.021	0.032	0.032
norway spruce				0.046		0.063		0.021		
sargent's cherry								0.073		
yellow cedar						0.023				

Table 12 Relative dominance (based on basal area) of species in London

	Mosa	Dorchester	Masonville		Sherwood Forest		Byron		Stoney Creek	
Native species	Remnant	Remnant	Street	Residential	Street	Residential	Street	Residential	Street	Residential
american linden			0.214		0.440	0.022			0.106	0.037
black cherry		0.346								
black walnut						0.051				
eastern red cedar				0.048		0.030				
flowering dogwood										
freeman maple				0.169		0.141				0.054
hackberry							0.089		0.123	0.420
honey locust				0.043		0.067	0.082			
ironwood								0.144		
large tooth aspen		0.124								
red maple	0.658	0.261				0.025				
red oak	0.157							0.219	0.127	
red pine				0.051						
shagbark hickory								0.136		
showy mountain ash										
silver maple				0.025		0.030	0.145			
sugar maple				0.047		0.096	0.043	0.083	0.123	0.223
white oak								0.289		
white pine		0.162		0.031						
white spruce				0.075		0.032				
Non-native species										
blue spruce				0.108		0.051				0.053
crabapple				0.030						
greenvase zelkova							0.248			
horse chestnut							0.047			
japanese cherry										0.030
littleleaf linden			0.140		0.174					
london plane						0.027	0.179		0.290	
norway maple			0.536	0.138	0.275	0.124				
norway spruce				0.039		0.096				
sycamore maple									0.057	

4.4.4 Diversity indices by abundance and dominance

No significant difference was observed in both street tree communities and remnant stands between indices calculated using abundance and those calculated using dominance (Table 13). However, a difference was found on residential properties, where the Shannon Abundance Index (mean $H' = 2.98$, SE = ± 0.15) was significantly higher than the Shannon Dominance Index (mean $H' = 2.41$, SE = ± 0.16) on all properties ($t = 2.65$, $p = 0.019$, d.f. = 14). Similarly, no differences in native proportions were found in streetscapes and remnant woodlands. On residential properties, native proportions were significantly higher when taking into account dominance (mean proportion = 0.73, SE = ± 0.07) compared to abundance (mean proportion = 0.54, SE = ± 0.05) on all properties ($t = 2.36$, $p = 0.033$, d.f. = 14). When examining older and newer neighbourhoods separately, no difference was observed on older properties, while a very significant difference between native abundance proportions (mean proportion = 0.41, SE = ± 0.03) and native dominance proportions (mean proportion = 0.78, SE = ± 0.07) was found on newer properties in both cities ($t = 4.66$, $p = 0.003$, d.f. = 6).

Table 13 Diversity indices of residential property tree communities according to neighbourhood.

City	Decade of development	Neighbourhood	Shannon Abundance Index (H')	Shannon Dominance Index (H')	Abundance proportion of native trees	Dominance proportion of native trees
Halifax	1960s-1970s	Tam O'Shanter	2.28	1.95	0.78	0.72
	1960s	Birch Cove	2.59	2.50	0.87	0.85
	2000s	Millview	3.09	2.58	0.37	0.56
	2000s	Russell Lake West	2.76	1.98	0.37	0.86
London	1960s-1970s	Masonville	3.37	3.12	0.50	0.57
	1960s	Sherwood Forest	3.54	2.93	0.50	0.60
	2000s	Byron	3.09	2.17	0.48	0.88
	2000s	Stoney Creek	3.14	2.06	0.42	0.80

4.5 DISCUSSION

4.5.1 Differences between land types within neighbourhoods

The matrices outlining species dominance based on land type illustrate the pervasiveness of some tree species in the suburban environment, as well as the ecological processes that contribute to species patterns. Native species found in remnant areas close to suburban developments in Halifax also tended to be found on private properties, which could be attributed to trees retained during development and/or species dispersal occurring from the remnants to the private properties (Doody et al. 2010; Ranta & Viljanen 2011). This trend was not observed in London, explained largely by the lack of remnant woodland located close to suburban neighbourhoods. Although red maple was both abundant and dominant in natural forest stands in London, it was not commonly planted in the streets or on private properties. Many species found only on private properties in both cities were ornamental trees, likely planted for horticultural and aesthetic purposes (Turner et al. 2005). These included Japanese maple, blue spruce,

European beech (*Fagus sylvatica*). Interestingly, eastern redbud (*Cercis canadensis*), a native species considered rare in Ontario, was planted on some residential properties.

While some tree species were found on only one land type, it is evident that some species flourish everywhere in urban environments. Norway maple was found in practically all street and property communities examined, which is to be expected given its invasiveness, tolerance to urban stresses, and popularity in street-tree planting in older neighbourhoods (Harrington et al. 2003). Norway maple can also thrive in more-natural conditions, and has been shown to erode native diversity by reducing plant species richness and density in its understory (Martin 1999). If planted in abundance along streets or on private properties in Halifax, Norway maple could invade and threaten the diversity of remnant stands located adjacent to subdivision developments.

4.5.2 Differences between abundance and dominance

Street tree communities exhibited the least variation between diversity indices calculated using abundance and dominance. This pattern is likely explained by the fact that street trees are generally planted around the same time, and therefore as the street trees age not all size classes are necessarily represented evenly. A lack of size or age diversity can pose a problem when all street trees reach the end of their life cycle at roughly the same time, when trees need to be removed and replaced (Steenberg et al. 2013). Considering basal area in diversity calculations may not provide additional useful information about tree-species composition, unless the tree species planted vary significantly in terms of growth rate. Although tree growth depends on various factors, including soil quality and nutrient availability, some tree species tend to grow faster than others; honey locust (*Gleditsia triacanthos*), callery pear (*Pyrus calleryana*), hackberry,

and some cultivars of Norway maple are fast-growing, while ginkgo (*Ginkgo biloba*), tree lilac, and some cultivars of littleleaf linden have slower growth rates (Bassuk et al. 2009).

In comparison, significant differences were observed between diversity indices calculated using abundance and dominance for residential property trees. Native proportions were much higher in newer neighbourhoods when basal area was taken into account, for two reasons. First, trees in newer neighbourhoods were planted recently, and are relatively small compared to many of the trees found on properties in older neighbourhoods. The small trees planted in newer neighbourhoods are not individually contributing as much to the canopy as would larger trees. Second, the majority of large trees found in newer neighbourhoods are native and were likely retained during development, when woodlands were encroached upon in Halifax and when fields with scattered trees were encroached upon in London. Despite the fact that non-native trees generally constituted the majority of stems in newer neighbourhoods (particularly in Halifax), a small number of large native trees dominated in terms of basal area. These included species like hackberry and sugar maple in London, and red maple and red oak in Halifax. The lack of evenness in basal area distribution also corroborates values found for the Shannon Dominance Index, which were significantly lower than values of the Shannon Abundance Index in newer neighbourhoods.

The trends point to the potential biodiversity benefits of preserving large remnant trees in residential neighbourhoods in both Halifax and London (Barth et al. 2015). Older and larger trees that provide more tree cover may contribute to biodiversity provisioning services more effectively than small, newly planted trees with less biomass and canopy cover (Ferenc et al. 2013). Furthermore, larger native trees may support other native

species of birds and insects, contributing to the persistence of local species and ecosystems (Ilkin et al. 2013).

4.5.3 Should nativeness be prioritized?

Although retaining large trees enhances biodiversity, it is essential to consider species richness and evenness when assessing urban forest resilience. Urban landscapes tend to promote the introduction of invasive species and alien pests, and it may not be beneficial to attempt to mimic natural settings in the urban forest (McKinney 2006; Aronson et al. 2015). The results of this study have shown that remnant woodlands tend to be dominated by few native species, which is reflected in species patterns of residential neighbourhoods in Halifax, some of which house an overwhelming abundance of red maple. Despite the high level of nativeness represented in the canopy, there is an elevated risk of mass tree mortality and tree cover loss if a species-specific threat emerges. Asian longhorned beetle, for example, is an invasive insect that has been spotted in Southern Ontario; it infests many native tree species, and has been shown to favour red maple in natural environments (Dodds & Orwig 2011).

Native biodiversity protection is crucial to sustainable urban forest management, but should not be prioritized to the extent where the risk of mass mortality and canopy loss is heightened due to the overwhelming dominance of one or two native tree species. Putting efforts towards increasing richness and evenness in city streets is an important step to mitigating the risk of canopy loss due to pests and diseases.

4.5.3 Limitations and research needs

It is worth mentioning that classification schemes of continuous data will inevitably pose problems related to the delineation of categories. In this case, the scheme

implies that a species making up 24% of the total (either in terms of abundance or dominance) is in a different class (illustrated by colour) compared to a species making up 26% of the total. To account for this difficulty, the number values of each proportion were added to the matrices, to provide more detailed information about the representation of each species in addition to elucidating general patterns from the coloured classification scheme.

Further research should be carried out on other land types found in suburban neighbourhoods, such as public parks. It would be worthwhile to determine whether comparing the abundance and dominance of species in parks would yield similar results to those found for residential properties. If so, municipalities should consider enhancing age diversity in addition to species diversity, to ensure that one or two species are not dominating the majority of the basal area and canopy.

4.6 SUMMARY

This study explored how tree species are spatially distributed in suburban landscapes, and how indices describing urban forest composition may provide complementary information when taking into account both abundance and dominance measures. The results illustrate that determining what constitutes “ideal” tree diversity is challenging; should native species representation or species richness and evenness be prioritized? Are the two mutually exclusive? Ultimately, some species will fare better on particular land types, and as evidenced by this study, some spaces are more promising for promoting naturalness in the suburban forest. If possible, developers should be encouraged to retain older, larger native trees during development in order to enhance the

canopy of newer neighbourhoods and mitigate losses in diversity that could occur in the process of landscape conversion.

CHAPTER 5 CONCLUSION

Urbanization creates human-dominated landscapes from natural ones, and in the process can fragment or destroy wildlife habitat, alter ecosystem dynamics, and degrade the ecological integrity of a region (McKinney 2006). Urban forests emerge from development by the retention of trees from woodlands or from purposeful planting along streets, in public green spaces, and on residential properties (Turner et al. 2005). Despite dramatic habitat replacement and landscape transformations incurred by development, biodiversity gains can be made in suburban residential areas, as evidenced by the abundance of both native and non-native tree species found in selected neighbourhoods. Species composition and diversity is influenced by a multitude of socio-demographic, political, economic, and ecological factors, taking place at the household, neighbourhood, and municipal levels. This work presents a compilation of knowledge about drivers of tree-species composition, and empirically explores how development patterns shape the suburban forest. By decoupling spatial and temporal contexts, patterns in tree composition at the neighbourhood level are more easily revealed and understood, and can thus effectively inform urban forest management strategies.

Many patterns in tree-species composition elucidated in this study were unique to the particular land types examined; remnant areas were dominated by native trees, streets housed a mix of native and non-native species tolerant to urban stresses, and private properties boasted a range of non-native ornamental species. This sampling design highlighted the potential for suburban neighbourhoods to express various manifestations of forest composition, to the point where more-urbanized settings like streetscapes and residential yards each exhibited greater species richness than more-natural forest stands.

In addition to the variety of tree species planted on residential properties, home owners in newer neighbourhoods also benefit immediately from the preservation of large native trees during development, as these trees contribute most to the canopy and subsequent ecosystem services.

Although it is useful to examine different land types separately for the purpose of capturing trends reflecting land use, tree establishment, and ownership, these spaces are in no way disconnected. Species can disperse between land types, potentially altering neighbourhood forest composition over time (Doody et al. 2010). An overwhelming abundance of native trees was found in older neighbourhoods in Halifax, which likely occurred due to the influence of native remnant stands retained during development. Furthermore, changes in the urban forest over time are not restricted to ecological processes; newer streetscapes exhibited far more species evenness and richness than older ones, reflecting an evolution in species-selection policies towards favouring both diversification and native species representation (Raupp et al. 2006). Studies have also suggested nursery stock availability could influence species-selection decisions (Pincetl et al. 2013).

This work focused only on the biophysical elements of tree diversity, and did not explore the attitudes and preferences of citizens and practitioners involved in shaping species composition on various land types. Various professionals have different skills and agendas, and may prioritize some factors over others when selecting and planting trees (Conway & Vander Vecht, 2015). Urban dwellers themselves experience the urban forest in a variety of ways, and are subjected to social, economic, and environmental influences when contributing to the private urban forest (Avolio et al. 2015; Pearce et al. 2015).

More research is needed in Canadian cities on how and why urban dwellers select tree species, including native and non-native trees. Using this information, scholars and practitioners can develop and test education initiatives focusing on the benefits of planting some trees over others. This would be especially beneficial in neighbourhoods located in close proximity to remnant areas, which could be threatened by the introduction of invasive species.

From a planning perspective, determining the extent to which urban morphology plays a role in forest composition is vital for designing neighbourhoods that integrate green infrastructure effectively to maximize its benefits. Studies have shown that some elements of urban design influence forest composition, like lot size, impervious surface cover, and housing density (Tratalos et al. 2007; Bigsby et al. 2014). Even transportation is relevant, as vehicle use can promote seed dispersal and potentially introduce undesirable species (Von Der Lippe & Kowarik 2007). Factors that are not as well researched include road and sidewalk size and arrangement, and the presence and size of the tree lawn. It is worthwhile to determine which elements of urban design are most relevant in shaping tree cover, canopy distribution, and species diversity in suburban neighbourhoods. In doing so, priorities for urban planners, foresters, and developers can be identified to inform plans for future subdivision developments where citizens will benefit from a healthy and long-lasting urban forest.

In conclusion, high population growth rates occurring in suburban areas in recent years emphasize the importance of determining how suburban development affects forest composition in Canadian cities. Enhancing urban forest diversity has numerous benefits, such as wildlife habitat promotion, reduction in ecosystem vulnerability, and increased

learning opportunities (Stone et al. 1996; Alvey 2006). Approximately 80% of Canadians live in metropolitan areas, and thus experience the natural world primarily within urban environments (Gordon & Shirokoff 2014). Urban dwellers value trees in the city for a variety of reasons, highlighting the importance of sustainable forest management in areas where people live, work, and play (Peckham et al. 2013).

The concept of tree diversity is recognized as essential to urban forest management, yet scholars disagree on management and enhancement priorities. Some believe in the importance of protecting native remnant stands for biodiversity promotion (LaPaix & Freedman 2010), while others welcome the introduction of non-native species for the purpose of diversification (Chalker-Scott 2015). Enhancing tree diversity in suburban areas requires a balance of both; it seems most appropriate to create and manage urban forests that celebrate unique regional ecosystems while also promoting the benefits of non-native, non-invasive species. Acknowledging and utilizing differences in forest composition between land types is useful for informing management and enhancement strategies. Instead of blindly aiming for more diversity on each type, practitioners and urban forest stewards should determine which spaces are more conducive to the success of particular species.

Given the multiplicity of contexts for the urban forest, it seems more appropriate to envision urban forest diversity and targets not as endpoints, but rather as an adaptive journey. Experimentation, trial-and-error, and mistakes are necessary steps for determining how trees fare in different places and over time. As environmental disturbances like invasive species, pests, and repercussions from climate change continue

to touch our cities, we must accept that flexibility and creativity are important qualities for urban forest management.

REFERENCES

- Alvey, A.A. 2006. Promoting and preserving biodiversity in the urban forest. *Urban For. Urban Green*. **5**(4):195-201. doi:10.1016/j.ufug.2006.09.003.
- Araujo, M.B. 2003. The coincidence of people and biodiversity in Europe. *Global Ecol. Biogeogr.* **12**(1):5-12. doi:10.1046/j.1466-822X.2003.00314.x.
- Aronson, M.F.J., Handel, S.N., La Puma, I.P., Clemants, S.E. Urbanization promotes non-native woody species and diverse plant assemblages in the New York metropolitan region. *Urban Ecosyst.* **18**(1): 31-45. doi:10.1007/s11252-014-0382z.
- Avolio, M.L., Pataki, D.E., Pincetl, S., Gillespie, T.W., Jenerette, G.D., McCarthy, H.R. 2015. Understanding preferences for tree attributes: the relative effects of socio economic and local environmental factors. *Urban Ecosyst.* **18**(1):73-86. doi:10.1007/s11252-014-0388-6.
- Barker, P.A. 1975. Ordinance control of street trees. *J. Arboric.* **1**(11):212-216. Available from <http://joa.isa-arbor.com/request.asp?JournalID=1&ArticleID=1356&Type=2> [accessed 8 March 2016].
- Bartelink, H.H. 1997. Allometric relationships for biomass and leaf area of beech (*Fagus sylvatica* L). *Ann. For. Sci.* **54**(1):39-50. <http://dx.doi.org/10.1051/forest:19970104>
- Barth, B.J., FitzGibbon, S.I., and Wilson, R.S. 2015. New urban developments that retain more remnant trees have greater bird diversity. *Landscape Urban Plan.* **136**(1):122-129. doi:10.1016/j.landurbplan.2014.11.003.
- Bassuk, N., Curtis, D.F., Marranca, B.Z., Neal, B. 2009. Recommended urban trees: Site assessment and tree selection for stress tolerance. Available online: <http://www.hort.cornell.edu/uhi/outreach/recurbtree/pdfs/~recurbtrees.pdf> [accessed 8 May 2016].
- Bigsby, K.M., McHale, M.R., and Hess, G.R. 2014. Urban morphology drives the homogenization of tree cover in Baltimore, MD and Raleigh, NC. *Ecosystems.* **17**(2):212-227. doi:10.1007/s10021-013-9718-4.
- Bourne, K.S., and Conway, T.M. 2014. The influence of land use type and municipal context on urban tree species diversity. *Urban Ecosyst.* **17**(1):329-348. doi:10.1007/s11252-0130317-0.
- Brändle, M., and Brandl, R. 2001. Species richness of insects and mites on trees: expanding Southwood. *J. Animal Ecol.* **70**(3):491-504. doi:10.1046/j.13652656.2001.00506.x.

- Burghardt, K., Tallamy, D.W., Shriver, W.G. 2009. Impact of native plants on bird and butterfly biodiversity in suburban landscapes. *Conserv. Biol.* **23**(1): 219–224. doi: 10.1111/j.1523-1739.2008.01076.x.
- Byers, J.E. 2002. Impact of non-indigenous species on natives enhanced by anthropogenic alteration of selection regimes. *Oikos*. **97**(3):449-458. doi:10.1034/j.16000706.2002.970316.x.
- Cardinale, B.J., Duffy, E., Gonzalez A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzing, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., and Naeem, S. 2012. Biodiversity loss and its impact on humanity. *Nature* (London, U. K.). **486**(7401):59-67. doi:10.1038/nature11148.
- Čepelová, B., and Münzbergová, Z. 2012. Factors determining the plant species diversity and species composition in a suburban landscape. *Landscape Urban Plan.* **106**(4):336-346. doi:10.1016/j.landurbplan.2012.04.008.
- Chalker-Scott, L. 2015. Nonnative, noninvasive woody species can enhance urban landscape biodiversity. *Arboric. Urban For.* **41**(4):173-186. Available from <http://joa.isaarbor.com/articles.asp?JournalID=1&VolumeID=41&IssueID=4> [accessed 8 March 2016].
- Chen, J.M., Rich, P.M., Gower, S.T., Norman, J.M., Plummer, S. 1997. Leaf area index of boreal forests: Theory, techniques, and measurements. *J. Geophysical Res.* **102**(24):29429-29443. 10.1029/97JD01107.
- The City of London. London Urban Forest Strategy, 2014. Available online: <https://www.london.ca/residents/Environment/TreesForests/Documents/London%20Urban%20Forestry%20Strategy%20Final.pdf> [accessed on 8 March 2016].
- Clark, K.H., and Nicholas, K.A. 2013. Introducing urban food forestry: A multifunctional approach to increase food security and provide ecosystem services. *Landscape Ecol.* **28**(9):1649-1669. doi:10.1007/s10980-013-9903-z.
- Conway, T.M., and Vecht, J.V. 2015. Growing a diverse urban forest: Species selection decisions by practitioners planting and supplying trees. *Landscape Urban Plan.* **138**:1-10. doi:10.1016/j.landurbplan.2015.01.007.
- Conway, T.M., and Bourne, K.S. 2013. A comparison of neighbourhood characteristics related to canopy cover, stem density and species richness in an urban forest. *Landscape Urban Plan.* **113**:10-18. doi:10.1016/j.landurbplan.2013.01.005.

- Cornelis, J. and Hermy, M. 2004. Biodiversity relationships in urban and suburban parks in Flanders. *Landscape Urban Plan.* **69**:385-401. doi:10.1016/j.landurbplan.2003.10.038.
- Dearborn, D.C., and Kark, S. 2009. Motivations for conserving urban biodiversity. *Conserv.Biol.* **24**(2):432-440. doi:10.1111/j.1523-1739.2009.01328.x.
- Deutschewitz, K., Lausch, A., Kühn, I., Klotz, S. 2003. Native and alien plant species richness in relation to spatial heterogeneity on a regional scale in Germany. *Glob. Ecol. Biogeogr.* **12**: 299–311.
- Dirr, M.A. 1976. Selection of trees for tolerance to salt injury. *J. Arboric.* **2**(11):209-216. Available from <http://www.nswcoa.ca/assets/Uploads/files/Salt-tolerant-trees.pdf> [accessed 8 March 2016].
- Dobbs, C., Kendal, D., Nitschke, C. 2013. The effects of land tenure and land use on the urban forest structure and composition of Melbourne. *Urban For. Urban Green.* **12**(4):417-425. doi:10.1016/j.ufug.2013.06.006.
- Dodds, K.J. and Orwig, D.A. 2011. An invasive urban forest pest invades natural environments — Asian longhorned beetle in northeastern US hardwood forests. *Can. J. For. Res.* **41**: 1729-1742. doi:10.1139/X11-097
- Doody, B.J., Sullivan, J.J., Meurk, C.D., Stewart, G.H., and Perkins, H.C. 2010. Urban realities: the contribution of residential gardens to the conservation of urban forest remnants. *Biodivers. Conserv.* **19**(5):1385-1400. doi:10.1007/s10531-009-9768-2.
- Duinker, P.D., Ordóñez C., Steenberg, J.W.N., Miller, K.H., Toni, S.A., and Nitoslawski, S.A. 2015. Trees in Canadian cities: Indispensable life form for urban sustainability. *Sustainability.* **7**(6):7379-7396. doi:10.3390/su7067379.
- Dwyer, J.F., Nowak, D.J., Noble, M.H., and Sisinni, S.M. 2000. Connecting people with ecosystems in the 21st century: an assessment of our nation's urban forest. USDA For. Serv. Res. Pap. PNW-GTR-490.
- Fahey, R.T., Bowles, M.L., and McBride, J.L. 2012. Origins of the Chicago urban forest: Composition and structure in relation to presettlement vegetation and modern land use. *Arboric. Urban For.* **38**(5):181-193. Available from http://www.mortonarbforestecology.org/uploads/1/1/2/3/11230112/fahey_bowle_mcbrie_2012.pdf [accessed 8 March 2016].
- Farrar, J.L. 1995. *Trees in Canada*; Fitzhenry and Whiteside Limited: Markham, ON, Canada; p. 502.

- Ferenc, M., Sedláček, O., Fuchs, R. 2013. How to improve urban greenspace for woodland birds: site and local-scale determinants of bird species richness. *Urban Ecosyst.* **17**(2). DOI 10.1007/s11252-013-0328-x.
- Florgård, C. Long-term changes in indigenous vegetation preserved in urban areas. *Landscape Urban Plan.* **52**(2-3):101-116. doi:10.1016/S0169-2046(00)00126-2.
- Forsyth, A. 2012. Defining suburbs. *Journal of Planning Literature.* **27**(3):270-281. doi:10.1177/0885412212448101.
- Fraser, E.D.G., and Kenney, W.A. 2000. Cultural background and landscape history as factors affecting perceptions of the urban forest. *J. Arboric.* **26**(2):106-113. Available from http://www.sfrc.ufl.edu/urbanforestry/Resources/PDF%20downloads/Fraser_200.pdf[accessed 8 March 2016].
- Goddard, M.A., Dougill, A.J., and Benton, T.G. 2010. Scaling up from gardens: Biodiversity conservation in urban environments. *Trends Ecol. Evol.* **25**(2):90-98. doi:10.1016/j.tree.2009.07.016.
- Goddard, M.A., Dougill, A.J., and Benton, T.G. 2013. Why garden for wildlife? Social and ecological drivers, motivations and barriers for biodiversity management in residential landscapes. *Ecol. Econ.* **86**:258-273. doi:10.1016/j.ecolecon.2012.07.016.
- Godefroid, S., and Koedam, N. 2007. Urban plant species patterns are highly driven by density and function of built-up areas. *Landscape Ecol.* **22**(8):1227-1239. doi:10.1007/s10980007-9102-x.
- Godefroid, S., and Koedam, N. 2003a. Distribution pattern of the flora in a peri-urban forest: An effect of the city–forest ecotone. *Landscape Urban Plan.* **65**:169-85. doi:10.1016/S01692046(03)00013-6.
- Godefroid, S., and Koedam, N. 2003b. How important are large vs. small forest remnants for the conservation of the woodland flora in an urban context? *Global Ecol. Biogeogr.* **12**:287-298. doi:10.1046/j.1466-822X.2003.00035.x.
- Gong, C., Chen, J., and Yu, S. 2013. Biotic homogenization and differentiation of the flora in artificial and near-natural habitats across urban green spaces. *Landscape Urban Plan.* **120**:158-169. doi:10.1016/j.landurbplan.2013.08.006.
- Gordon, D.L.A., and Shirokoff, I. 2014. Suburban nation? Population growth in Canadian suburbs, 2006-2011. *In Proceedings of the 6th Council for Canadian Urbanism (CanU6) summit: Cities at the Edge, Toronto, Ont., 18-20 September 2014. Edited by Ajay Agarwal, Pierre Filion, Jill Grant, Richard Harris, Paul Hess, Nik Luka, Martin Turcotte, Andrejs Skaburskis and Ian Wight. School of Urban and Regional Planning, Queen's University, Kingston, Ont. pp. 1-32.*

- Hahs, A.K., McDonnell, M.J., McCarthy, M.A., Vesk, P.A., Corlett, R.T., Norton, B.A., Clemants, S.E., Duncan, R.P., Thompson, K., Schwartz, M.W., and Williams, N.S. 2009. A global synthesis of plant extinction rates in urban areas. *Ecol Lett.* **12**(11):1165-1173. doi:10.1111/j.1461-0248.2009.01372.x.
- Hansen, A.J., Knight, R.L., Marzluff, J.M., Powell, S., Brown, K., Gude, P.H., and Jones, K. 2005. Effects of exurban development on biodiversity: Patterns, mechanisms, and research needs. *Land Use Change in Rural America.* **15**(6):1893-1905. doi:10.1890/055221.
- Harrington, R.A., Kujawski, R., Ryan, H.D.P. 2003. Invasive plants and the green industry. *J. Arboric.* **29**(1):42-48.
- Henderson, S.P.B., Perkins, N.H., and Nelischer, M. 1998. Residential lawn alternatives: A study of their distribution, form and structure. *Landscape Urban Plan.* **42**(2-4):135-145. doi:10.1016/S0169-2046(98)00084-X.
- Heynen, N.C., and Lindsey, G. 2003. Correlates of urban forest canopy cover: Implications for local public works. *Public Works Management & Policy.* **8**(1):33-47. doi:10.1177/1087724X03008001004.
- Heynen, N.C., Perkins, H.A., and Parama, R. 2006. The political ecology of uneven urban green space: The impact of political economy on race and ethnicity in producing environmental equality in Milwaukee. *Urban Aff. Rev.* **42**(1):3-25. doi:10.1177/1078087406290729.
- Hobbs, E.R. 1988. Species richness of urban forest patches and implications for urban landscape diversity. *Landscape Ecol.* **1**(3):141-152. doi:10.1007/BF00162740.
- Honnay, O., Endels, P., Vereecken, H., and Hermy, M. 1999. The role of patch area and habitat diversity in explaining native plant species richness in disturbed suburban forest patches in Northern Belgium. *Divers. Distrib.* **5**(4):129-141. doi:10.1046/j.14724642.1999.00047.x.
- Hope, D., Gries, C., Zhu, W., Fagans, W.F., Redman, C.L., Grimm, N., Nelson, A.L., Martin, C., and Kinzig, A. 2003. Socioeconomics drive urban plant diversity. *Proc. Natl. Acad. Sci. U. S. A.* **100**(15): 8788-8792. doi:10.1073/pnas.1537557100.
- Hope, D., Gries, C., Casagrande, D., Redman, C.L., Grimm, N.B., and Martin, C. 2006. Drivers of spatial variation in plant diversity across the central Arizona-Phoenix ecosystem. *Soc. Natur. Resour.* **19**(2):101-116. doi:10.1080/08941920500394469.
- Hostetler, M., Allen, W., Meurk, C. 2011. Conserving urban biodiversity? Creating green infrastructure is only the first step. *Landscape Urban Plan.* **100**(4): 369-371. doi:10.1016/j.landurbplan.2011.01.011.

- HRM. 2013. Halifax Regional Municipality urban forest master plan. Halifax Regional Municipality, Halifax, NS, Canada. pp. 460. Available from <http://www.halifax.ca/property/UFMP/documents/SecondEditionHRMUFMP.pdf> [accessed 8 March 2016].
- Ilkin, K., Knight, E., Lindenmayer, D.B., Fischer, J., and Manning, A.D. 2013. The influence of native versus exotic streetscape vegetation on the spatial distribution of birds in the suburbs and reserves. *Divers. Distrib.* **19**(3):294-306. doi: 10.1111/j.14724642.2012.00937.x.
- Iverson, L.R., and Cook, E.A. 2000. Urban forest cover of the Chicago region and its relation to household density and income. *Urban Ecosyst.* **4**(2):105-124. Doi: 10.1023/A:1011307327314.
- Jim, C.Y. 2004. Green-space preservation and allocation for sustainable greening of compact cities. *Cities* **21**(4):311-320. doi:10.1016/j.cities.2004.04.004.
- Jim, C.Y. 1993. Trees and landscape of a suburban residential neighborhood in Hong Kong. *Landscape Urban Plan.* **23**(2):119-143. doi:10.1016/0169-2046(93)90112Q.
- Jim, C.Y., and Chen, W.Y. 2008. Pattern and divergence of tree communities in Taipei's main urban green spaces. *Landscape Urban Plan.* **84**(3-4):312-323. doi:10.1016/j.landurbplan.2007.09.001.
- Jim, C.Y., and Liu, H.T. 2001. Species diversity of three major urban forest types in Guangzhou City, China. *For. Ecol. and Manage.* **146**(1-3):99-114. doi:10.1016/S03781127(00)00449-7.
- Julien, M., and Zmyslony, J. 2001. Why do landscape clusters emerge in an organized fashion in anthropogenic environments? *Landscape Res.* **26**(4):337-350. doi:10.1080/01426390120090139.
- Kendal, D., Dobbs, C., Lohr, V.I. 2014. Global patterns of diversity in the urban forest: Is there evidence to support the 10/20/30 rule? *Urban For. Urban Green.* **13**(3):411-417. doi:10.1016/j.ufug.2014.04.004.
- Kendal, D., Williams, N.S.G., and Williams, K.J.H. 2012a. Drivers of diversity and tree cover in gardens, parks, and streetscapes in an Australian city. *Urban For. Urban Green.* **11**(3):257-265. doi:10.1016/j.ufug.2012.03.005.
- Kendal, D., Williams, N.S.G., and Williams, K.J.H. 2012b. A cultivated environment: Exploring the global distribution of plants in gardens, parks, and streetscapes. *Urban Ecosyst.* **15**(3):637-652. doi:10.1007/s11252-011-0215-2.

- Kendle, A.D., and Rose, J.E. 2000. The aliens have landed! What are the justifications for 'native only' policies in landscape plantings? *Landscape Urban Plan.* **47**(1-2):19-31. doi:0.1016/S0169-2046(99)00070-5.
- Kenney, W.A., Van Wassenae, P.J.E., and Satel, A.L. 2011. Criteria and indicators for strategic urban forest planning and management. *Arboric. Urban For.* **37**(3):108-117. Available from http://www.isa-arbor.com/events/conference/proceedings/2013/VAN_WASSENAER_article_AF_May_2011.pdf [accessed 8 March 2016].
- Kershaw, L. 2001. *Trees of Ontario*; Lone Pine Publishing: Edmonton, AB, Canada, 2001; p. 240.
- Keys, K., Neily, P., Quigley, E., and Stewart, B. 2010. *Forest Ecosystem Classification for Nova Scotia, Part III: Ecosites*. Halifax, NS: Department of Natural Resources. Available from <http://www.novascotia.ca/natr/library/forestry/reports/Ecosites.pdf>
- Kinzig, A.P., Warren, P., Martin, C., Hope, D., and Katti, M. 2005. The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. *Ecol.Soc.* **10**(1):23. Available from <http://www.ecologyandsociety.org/vol10/iss1/art23/> [accessed 8 March 2016].
- Kirkpatrick, J.B., Daniels, G.D., and Zagorski, T. 2007. Explaining variation in front gardens between suburbs of Hobart, Tasmania, Australia. *Landscape Urban Plan.* **79**(3-4):314-322. doi:10.1016/j.landurbplan.2006.03.006.
- Kirkpatrick, J.B., Daniels, G.D., Davison, A. 2011. Temporal and spatial variation in garden and street trees in six eastern Australian cities. *Landscape Urban Plan.* **101**(3):244-252. doi:10.1016/j.landurbplan.2011.02.029.
- Kirkpatrick, J.B., Davison, A., and Daniels, G.D. 2012. Resident attitudes towards trees influence the planting and removal of different types of trees in eastern Australian cities. *Landscape Urban Plan.* **107**(2):147-158. doi:10.1016/j.landurbplan.2012.05.015.
- Kirkpatrick, J.B., Davison, A., and Daniels, G.D. 2013a. Sinners, scapegoats or fashion victims? Understanding the deaths of trees in the green city. *Geoforum.* **48**:165-176. doi:10.1016/j.geoforum.2013.04.018.
- Kirkpatrick, J.B., Davison, A., and Harwood, A. 2013b. How tree professionals perceive trees and conflicts about trees in Australia's urban forest. *Landscape Urban Plan.* **119**:124-130. doi:10.1016/j.landurbplan.2013.07.009.

- Konijnendijk, C.C., Ricard, R.M., Kenney, A., and Randrup, T.B. 2006. Defining urban forestry – A comparative perspective of North America and Europe. *Urban For. Urban Green*. **4**(3-4):93-103. doi:10.1016/j.ufug.2005.11.003.
- Kowarik, I. 2011. Novel urban ecosystems, biodiversity, and conservation. *Environ. Pollut.* **159**(8-9):1974-1983. doi:10.1016/j.envpol.2011.02.022.
- Kühn, I., Brandl, R., and Klotz, S. 2004. The flora of German cities is naturally species rich. *Evol. Ecol. Res.* **6**(5):749-764. Available from https://www.ufz.de/export/data/1/22488_kuehn.eer1629.pdf [accessed 8 March 2016].
- Landry, S.M., and Chakraborty, J. 2009. Street trees and equity: Evaluating the spatial distribution of an urban amenity. *Environ. Plann. A.* **41**(11):2651-2670. doi:10.1068/a41236.
- LaPaix, R., and Freedman, B. 2010. Vegetation structure and composition within urban parks of Halifax Regional Municipality, Nova Scotia, Canada. *Landscape Urban Plan.* **98**(2):124-135. doi: 10.1016/j.landurbplan.2010.07.019.
- Lacan, I., and McBride, J.R. 2008. Pest Vulnerability Matrix (PVM): A graphic model for assessing the interaction between tree species diversity and urban forest susceptibility to insects and diseases. *Urban For. Urban Green*. **7**(4):291-300. doi:10.1016/j.ufug.2008.06.002.
- Leichenko, R.M., and Solecki, W.D. 2013. Climate change in suburbs: An exploration of key impacts and vulnerabilities. *Urban Climate*. **6**:82-97. doi:10.1016/j.uclim.2013.09.001.
- Loo, J., and Ives, N. 2003. The Acadian forest: Historical condition and human impacts. *For. Chron.* **79**: 462–474.
- Luck, G.W., Smallbone, L.T., and O'Brien, R. 2009. Socio-economics and vegetation change in urban ecosystems: patterns in space and time. *Ecosystems*. **12**(4):604-620. doi:10.1007/s10021-009-9244-6.
- Martin, C.A., Warren, P.S., and Kinzig, A.P. 2004. Neighbourhood socioeconomic status is a useful predictor of perennial landscape vegetation in residential neighbourhoods and embedded small parks of Phoenix, AZ. *Landscape Urban Plan.* **69**(4):355-368. doi:10.1016/j.landurbplan.2003.10.034.
- Martin, P.H. 1999. Norway maple (*Acer platanoides*) invasion of a natural forest stand: Understorey consequences and regeneration pattern. *Biol. Invasions* **1**(2):215-222.

- Marzok, N., Dean, M., and Eastwood, M. 2014. BiodiverCities: A primer on nature in cities. Toronto (ON): ICLEI – Local Governments for Sustainability, Toronto and Region Conservation Authority. 48 p. Available from <http://www.icleicanada.org/component/k2/item/121-biodivercitiesprimer> [accessed 8 March, 2016].
- McKinney, M.L. 2002. Urbanization, biodiversity, and conservation. *BioScience* **52**(10):883- 890. doi:[tp://dx.doi.org/10.1641/00063568\(2002\)052\[0883:UBAC\]2.0.CO;2](http://dx.doi.org/10.1641/00063568(2002)052[0883:UBAC]2.0.CO;2).
- McKinney, M.L. 2006. Urbanization as a major cause of biotic homogenization. *Biol. Conserv.* **127**:247–260. doi:10.1016/j.biocon.2005.09.005.
- McKinney, M.L. 2008. Effects of urbanization on species richness: a review of plants and animals. *Urban Ecosyst.* **11**(2):161-176. Available from <http://www.mit.edu/people/spirn/Public/Granite%20Garden%20Research/Urban%20ecology/McKinney%202008%20Species%20Richness.pdf> [accessed 8 March 2016]
- McPherson, E.G. and Rowntree, R.A. 1989. Using structural measures to compare twenty-two U.S. street tree populations. *Landscape J.* **8**: 13-23. Available from http://www.fs.fed.us/psw/programs/uesd/uep/products/1/psw_cufr745_structuralmeasures.pdf [accessed 10 May 2016].
- McWilliam, W., Brown, R., Eagles, P., and Seasons, M. 2014. Barriers to the effective planning and management of residential encroachment within urban forest edges: A Southern Ontario, Canada case study. *Urban For. Urban Green.* **13**(1):48-62. doi:10.1016/j.ufug.2013.08.002.
- McWilliam, W., Brown, R., Eagles, P., Seasons, M. 2015. Evaluation of planning policy for protecting green infrastructure from loss and degradation due to residential encroachment. *Land Use Policy* **47**: 459–467. doi:10.1016/j.landusepol.2015.05.006.
- Meléndez-Ackerman, E.J., Santiago-Bartolomei, R., Vila-Ruiz, C.P., Santiago, L.E., García Montiel, D., Verdejo-Ortiz, J.C., Manrique-Hernández, H., and Hernández-Calo, E. 2014. Socioeconomic drivers of yard sustainable practices in a tropical city. *Ecol. Soc.* **19**(3):20. doi: 10.5751/ES-06563-190320.
- Motz, K., Sterba, H., Pommerening, A. 2010. Sampling measures of tree diversity. *For. Ecol. Management.* **260**(11):1985-1996. doi:10.1016/j.foreco.2010.08.046.
- Nassauer, J.I., Wang, Z., and Dayrell, E. 2009. What will the neighbours think? Cultural norms and ecological design. *Landscape Urban Plan.* **92**(3-4):282-292. doi:10.1016/j.landurbplan.2009.05.010.

- Nitoslawski, S.A., & Duinker, P.N. 2016. Managing tree diversity: A comparison of suburban development in two Canadian cities. *Forests* **7**(6): 119. doi:10.3390/f7060119.
- Nock, C.A., Paquette, A., Follet, M., Nowak, D.J., and Messier, C. 2013. Effects of urbanization on tree species functional diversity in eastern North America. *Ecosystems* **16**(8):1487–1497. doi:10.1007/s10021-013-9697-5.
- Nowak, D.J. 1994. Understanding the structure of urban forests. *J. Forest.* **92**(10):42-46. Available from http://www.nrs.fs.fed.us/pubs/jrnl/1994/ne_1994_nowak_001.pdf [accessed March 8 2016].
- Nowak, D.J., Rowntree, R.A., McPherson, E.G., Sisinni, S.M., Kerkmann, E.R., and Stevens, J.C. 1996. Measuring and analyzing urban tree cover. *Landscape Urban Plan.* **36**:49-57. Available from http://www.nrs.fs.fed.us/pubs/jrnl/1996/ne_1996_nowak_001.pdf [accessed March 8 2016].
- Olden, J.D., Poff, N.L.R., Douglas, M.R., Douglas, M.E., Fausch, K.D. 2004. Ecological and evolutionary consequences of biotic homogenization. *Trends Ecol.* **19**: 18–24. DOI: <http://dx.doi.org/10.1016/j.tree.2003.09.010>
- Ordóñez, C., & Duinker, P.N. 2012. Ecological integrity in urban forests. *Urban Ecosyst.* **15**(4):863-877. doi:10.1007/s11252-012-0235-6.
- Ordóñez, C., & Duinker, P.N. 2013. An analysis of urban forest management plans in Canada: Implications for urban forest management. *Landscape Urban Plan.* **116**:36-47. doi:10.1016/j.landurbplan.2013.04.007.
- Ordóñez, C., & Duinker, P.N. 2014. Assessing the vulnerability of urban forests to climate change. *Environ. Rev.* **22**(3):311-321. doi: 10.1139/er-2013-0078.
- Ordóñez, C., & Duinker, P.N. 2015. Climate change vulnerability assessment of the urban forest in three Canadian cities. *Climatic Change* **131**(4):531-543. DOI: 10.1139/er-2013-0078.
- Palludan-Müller, G., Saxe, H., Pederson, L.B., and Randrup, T.B. 2002. Differences in salt sensitivity of four deciduous tree species to soil or airborne salt. *Physiol. Plantarum* **114**(2):223-230.
- Pearce, L.M., Davison, A., and Kirkpatrick, J.B. 2015. Personal encounters with trees: The lived significance of the private urban forest. *Urban For. Urban Green.* **14**(1):1-7. doi:10.1016/j.ufug.2014.11.003
- Peckham, S., Duinker, P., and Ordóñez, C. 2013. Urban forest values in Canada: Views of citizens in Calgary and Halifax. *Urban For. Urban Green.* **12**(2):154-162. doi:10.1016/j.ufug.2013.01.001.

- Pedlowski, M.A., Da Silva, V.A.C., Adell, J.J.C., and Heynen, N.C. 2002. Urban forest and environmental inequality in Campos dos Goytacazes, Rio de Janeiro, Brazil. *Urban Ecosyst.* **6**(1):9-20. Available from <http://link.springer.com/article/10.1023%2FA%3A1025910528583> [accessed 8 March 2016].
- Pennington, D.N., Hansel, J.R., and Gorchov, D.L. 2010. Urbanization and riparian forest woody communities: Diversity, composition, and structure within a metropolitan landscape. *Biol. Conserv.* **143**(1):182-194. doi:10.1016/j.biocon.2009.10.002.
- Pincetl, A., Prabhu, S.S., Gillespie, T.W., Jenerette, G.D., and Pataki, D.E. 2013. The evolution of tree nursery offerings in Los Angeles County over the last 110 years. *Landscape Urban Plan.* **11**:10-17. doi:10.1016/j.landurbplan.2013.05.002.
- Point Pleasant Park Plan. 2008. Point Pleasant Park Comprehensive Plan: Introduction. NIPpaysage and Ekistics Planning and Design, Halifax, NS, Canada. pp. 14. Available from <http://www.pointpleasantpark.ca/site/ppp/media/pointpleasantpark/PPP%20CP%20Chapter%201%20Introduction%2010%2008%20web.pdf> [8 March 2016].
- Polakowski, N.R., Lohr, V.I., Cerny-Koenig, T. 2011. Survey of wholesale production nurseries indicates need for more education on the importance of plant species diversity. *Arboric. Urban For.* **37**: 259–264.
- Poland, T.M., and McCullough, D.G. 2006. Emerald ash borer: Invasion of the urban forest and threat to North America's ash resource. *J. Forest.* **104**(3):118-124. Available from http://www.nrs.fs.fed.us/pubs/jrnl/2006/nc_2006_Poland_003.pdf [accessed 8 March 2016].
- Ramage, B.S., Roman, L.A., and Dukes, J.S. 2013. Relationships between urban tree communities and the biomes in which they reside. *Appl. Veg. Sci.* **16**(1):8 20. DOI: 10.1111/j.1654109X.2012.01205.x.
- Ranta, P., and Viljanen, V. 2011. Vascular plants along an urban-rural gradient in the city of Tampere, Finland. *Urban Ecosyst.* **14**(3):361-376. <http://dx.doi.org/10.1007/s11252-011-0164-9>.
- Raupp, M.J., Cumming, A.B., and Raupp, E.C. 2006. Street tree diversity in eastern North America and its potential for tree loss to exotic borers. *Arboric. Urban For.* **32**(6):297-304. Available from <http://naldc.nal.usda.gov/download/27863/PDF> [accessed 8 March 2016].
- Roloff, A., Korn, S., and Gillner, S. 2009. The Climate-Species-Matrix to select tree species for urban habitats considering climate change. *Urban For. Urban Green.* **8**(4):295-308. doi:10.1016/j.ufug.2009.08.002.

- Rostami, M. 2011. Tree species selection for the Halifax urban forest under a changing climate M.E.S. thesis, School for Resource and Environmental Studies, Dalhousie University, Halifax, N.S. Available from <http://dalspace.library.dal.ca:8080/xmlui/handle/10222/14211> [accessed 8 March 2016].
- Rudd, H., Vala, J., and Schaefer, V. 2002. Importance of backyard habitat in a comprehensive biodiversity conservation strategy: A connectivity analysis of urban green spaces. *Restoration Ecol.* **10**(2):368-375. <http://dx.doi.org/10.1046/j.1526-100X.2002.02041.x>.
- Santamour, F.S. 1990. Trees for urban planting: Diversity, uniformity and common sense. Proceedings of the 7th Conference of the Metropolitan Tree Improvement Alliance (METRIA) 7:57-65. Available from <http://www.ces.ncsu.edu/fletcher/programs/nursery/metria/metria07/m79.pdf> [accessed 8 March 2016].
- Savard, J.L., Clergeau, P., and Mennechez, G. 2000. Biodiversity concepts and urban ecosystems. *Landscape Urban Plan.* **48**(3-4):131-42. doi:10.1016/S0169-2046(00)00037-2.
- Seto, K.C., Fragkias, M., Güneralp, B., Reilly, M.K. 2011. A meta-analysis of global urban land expansion. *PLOS ONE* 6(8): e23777. doi:10.1371/journal.pone.0023777
- Siegert, C.M., Levia, D.F., Hudson, S.A., Dowtin, A.L., Zhang, F., and Mitchell, M.J. 2016. Small-scale topographic variability influences tree species distribution and canopy throughfall partitioning in a temperate deciduous forest. *Forest Ecol. Manag.* **359**:109-117. doi:10.1016/j.foreco.2015.09.028
- Smith, R.M., Gaston, K.J., Warren, P.H., and Thompson, K. 2005. Urban domestic gardens (V): Relationships between landcover composition, housing and landscape. *Landscape Ecol.* **20**:235-53. doi:10.1007/s10980-004-3160-0.
- Species at Risk Act, R. S. C. 2003, c.29. Available from http://laws-lois.justice.gc.ca/eng/acts/S_15.3/ [accessed 8 March 2016].
- Steenberg, J.W.N., Duinker, P.N., and Charles, J.D. 2013. The neighbourhood approach to urban forest management: The case of Halifax, Canada. *Landscape Urban Plan.* **117**:135-144. doi:10.1016/j.landurbplan.2013.04.003.
- Stewart, G.H., Ignatieva, M.E., Meurk, C.D., and Earl, R.D. 2004. The re-emergence of indigenous forest in an urban environment, Christchurch, New Zealand. *Urban For. Urban Green.* **2**(3):149-158. doi:10.1078/1618-8667-00031.

- Stone, L., Gabric, A., Berman, T. 1996. Ecosystem resilience, stability and productivity: Seeking a relationship. *The American Naturalist* **148**(5): 892-903.
- Summit, J., and McPherson, E.G. 1998. Residential tree planting and care: A study of attitudes and behaviour in Sacramento, California. *J. Arboric.* **24**(2):89-97. Available from <http://joa.isaarbor.com/request.asp?JournalID=1&ArticleID=2795&Type=2> [accessed 8 March, 2016].
- Sydnor, T.D., Subburayalu, S., and Bumgardner, M. 2010. Contrasting Ohio nursery stock availability with community planting needs. *Arboric. Urban For.* **36**(1):47-54. Available from http://www.nrs.fs.fed.us/pubs/jrnl/2010/nrs_2010_sydnor_001.pdf? [accessed 8 March 2016].
- Tait, C.J., Daniels, C.B., and Hill, R.S. 2005. Changes in species assemblages within the Adelaide metropolitan area, Australia, 1836-2002. *Ecol. Appl.* **15**(1):346-359. DOI: 10.1890/04-0920
- Toni, S.A., & Duinker, P.N. 2015. A framework for urban-woodland naturalization in Canada. *Environ. Rev.* **23**(3):321-336. doi:0.1139/er-2015-0003
- Toronto Municipal Code. 2013. Chapter 813: Trees. City of Toronto, O.N., Canada. pp. 23. Available from http://www.toronto.ca/legdocs/municode/1184_813.pdf [accessed 8 March 2016].
- Tratalos, J., Fuller, R.A., Warren, P.H., Davies, R.G., and Gaston, K.J. 2007. Urban form, biodiversity potential and ecosystem services. *Landscape Urban Plan.* **83**(4):308-317. doi:10.1016/j.landurbplan.2007.05.003.
- Troy, A.R., Grove, J.M., O'Neil-Dunne, J.P.M., and Pickett, S.T.A., Cadenasso ML. 2007. Predicting opportunities for greening and patterns of vegetation on private urban lands. *Environ. Manage.* **40**(3):394-412. doi:10.1007/s00267-006-0112-2
- Turner, K., Lefler, L., and Freedman, B. 2005. Plant communities of selected urbanized areas of Halifax, Nova Scotia, Canada. *Landscape Urban Plan.* **71**(2-4):191-206. doi:10.1016/j.landurbplan.2004.03.003.
- Turcotte, M. 2008. The City/Suburb Contrast: How Can We Measure It? [online] Available from <http://www.statcan.gc.ca/pub/11-008-x/2008001/article/10459-eng.htm> [accessed 8 March 2016].
- Tzoulas, K., Korpela, K., Venn, S., Yli-Pelkonen, V., Kaźmierczak, A., Niemela, J., James, P. 2007. Promoting ecosystem and human health in urban areas using Green Infrastructure: A literature review. *Landscape Urban Plan.* **81**(3):167-178. doi:10.1016/j.landurbplan.2007.02.001.

- United Nations, Department of Economic and Social Affairs, Population Division. 2014. World Urbanization Prospects: The 2014 Revision, Highlights. [online] Available from <http://esa.un.org/unpd/wup/Publications/Files/WUP2014-Highlights.pdf>.
- Van Dyke, F. 2008. Biodiversity: Concept, Measurement, and Challenge. In *Conservation Biology: Foundations, Concepts, Applications*, 2nd ed.; Springer Science + Business Media BV: Dordrecht, The Netherlands; pp. 83–119.
- Vogt, J.M., Watkins, S.L., Mincey, S.K., Patterson, M.S., and Fischer, B.C. 2015. Explaining planted-tree survival and growth in urban neighbourhoods: A social ecological approach to studying recently-planted trees in Indianapolis. *Landscape Urban Plan.* **136**:130-143. doi:10.1016/j.landurbplan.2014.11.021.
- Von Der Lippe, M., and Kowarik, I. 2007. Long-distance dispersal of plants by vehicles as a driver of plant invasions. *Conserv. Biol.* **21**(4):986-996. DOI: 10.1111/j.1523-1739.2007.00722.x.
- Wheeler, S.M. 2015. Built landscapes of metropolitan regions: An international typology. *J. Am. Plann. Assoc.* **81**(3):168-189. DOI:10.1080/01944363.2015.1081567
- Zhao, M., Escobedo, F.J., Staudhammer, C. 2010a. Spatial patterns of a subtropical, coastal urban forest: Implications for land tenure, hurricanes, and invasives. *Urban For. Urban Green.* **9**(3):205-214. doi:10.1016/j.ufug.2010.01.008.
- Zhao, J., Ouyang, Z., Xu, W., Zheng, H., Meng, X. 2010b. Sampling adequacy estimation for plant species composition by accumulation curves – A case study of urban vegetation in Beijing, China. *Landsc. Urban Plan.* **95**: 113–121. doi:10.1016/j.landurbplan.2009.12.008.
- Zipperer, W.C. 2002. Species composition and structure of regenerated and remnant forest patches within an urban landscape. *Urban Ecosyst.* **6**:271-290. DOI:10.1023/B:UECO.0000004827.12561.d4
- Zipperer, W.C. 2008. Applying ecosystem management to urban forestry. In *Ecology, planning, and management of urban forests. Edited by M. Carreiro, Y.C. Song, and J. Wu.* Springer, New York, NY. pp. 97-108.

APPENDIX

Table 3 Contingency table for test “O” (old neighbourhoods).

	Native Trees	Non-Native Trees
Woodland (Halifax)	538	120
Field (London)	445	448

Table A2. Contingency table for test “N” (new neighbourhoods).

	Native Trees	Non-Native Trees
Woodland (Halifax)	98	170
Field (London)	72	86

Table A3. Contingency table for test “H” (all neighbourhoods in Halifax).

	Native Trees	Non-Native Trees
Old	538	120
New	98	170

Table A4. Contingency table for test “L” (all neighbourhoods in London).

	Native Trees	Non-Native Trees
Old	445	448
New	72	86

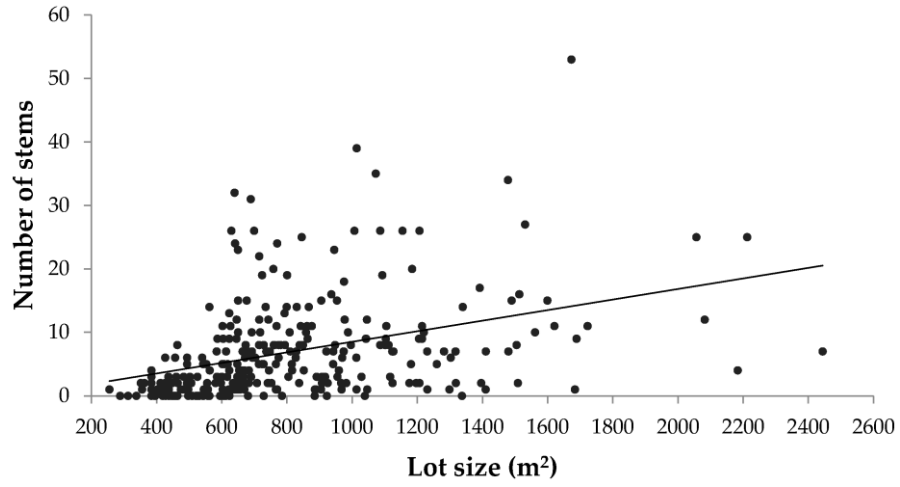


Figure A1 Correlation between private property lot size and number of stems found on each lot ($R^2 = 0.14$, $p < 0.05$, 284 lots sampled in total).

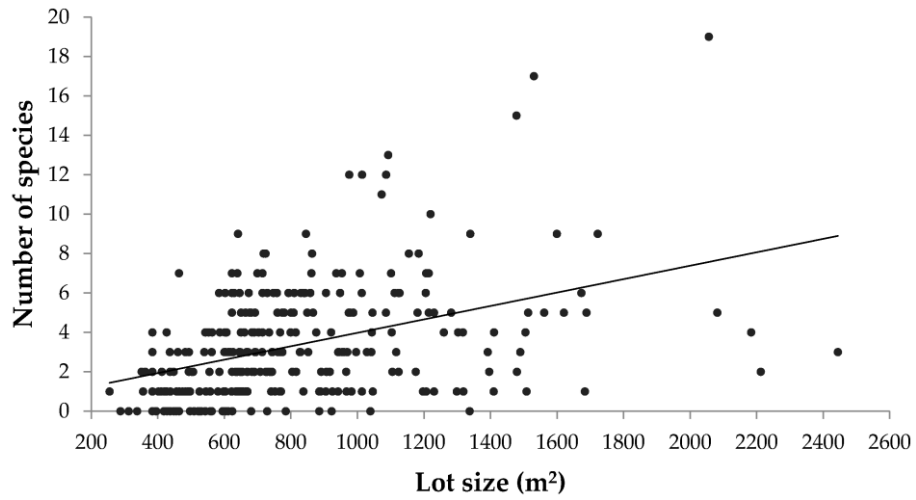


Figure A2 Correlation between private property lot size and number of species found on each lot ($R^2 = 0.17$, $p < 0.05$, 284 lots sampled in total).



Figure A3 Remnant sites sampled in Halifax.

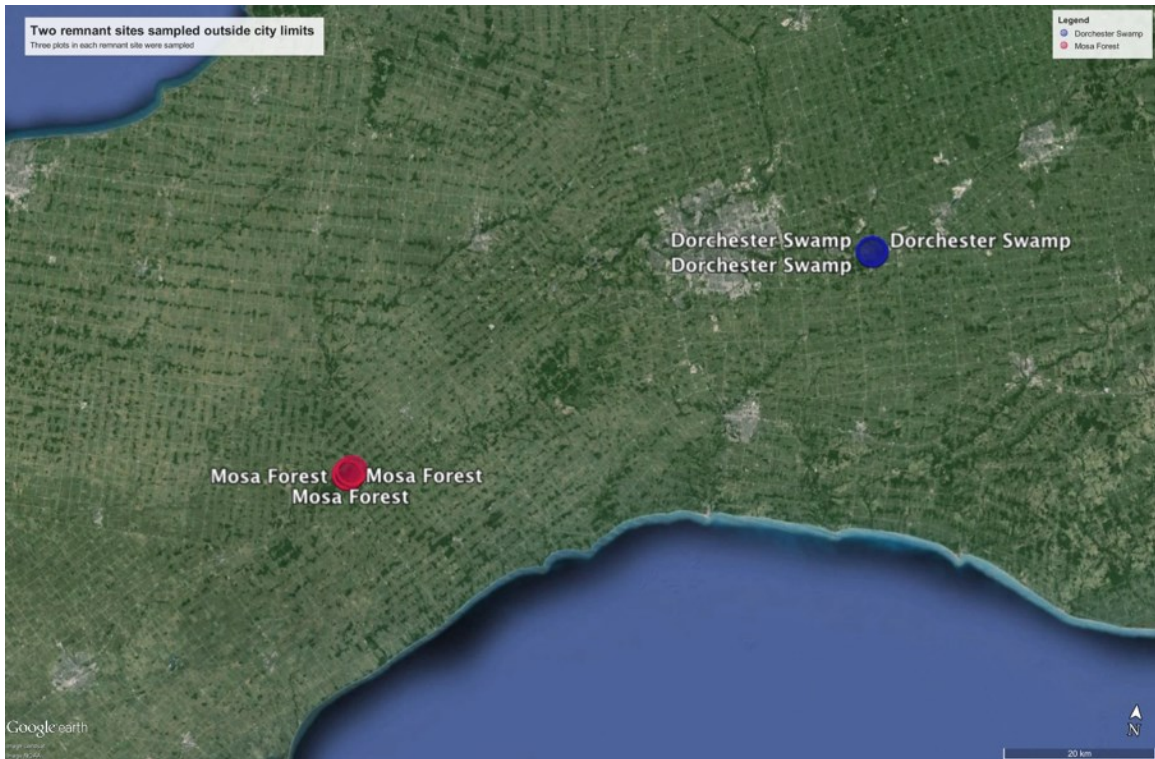


Figure A4 Remnant sites sampled in London.

Native species		Non-native species	
Common name	Latin name	Common name	Latin name
alternate-leaf dogwood	<i>Cornus alternifolia</i>	american linden	<i>Tilia americana</i>
american beech	<i>Fagus grandifolia</i>	amur maple	<i>Acer ginnala</i>
american elm	<i>Ulmus americana</i>	austrian pine	<i>Pinus nigra</i>
balsam fir	<i>Abies balsamea</i>	big leaf linden	<i>Tilia platyphyllos</i>
balsam poplar	<i>Populus balsamifera</i>	bitter cherry	<i>Prunus emarginata</i>
black ash	<i>Fraxinus nigra</i>	black walnut	<i>Juglans nigra</i>
black spruce	<i>Picea mariana</i>	blue ash	<i>Fraxinus quadrangulata</i>
chokecherry	<i>Prunus virginiana</i>	blue spruce	<i>Picea pungens</i>
common juniper	<i>Juniperus communis</i>	butternut	<i>Juglans cinerea</i>
eastern hemlock	<i>Tsuga canadensis</i>	callery pear	<i>Pyrus calleryana</i>
grey birch	<i>Betula populifolia</i>	canada plum	<i>Prunus nigra</i>
ironwood	<i>Ostrya virginiana</i>	common apple	<i>Malus</i> spp.
large tooth aspen	<i>Populus grandidentata</i>	crabapple	<i>Malus</i> spp.
pin cherry	<i>Prunus pensylvanica</i>	eastern red cedar	<i>Juniperus virginiana</i>
red ash	<i>Fraxinus pennsylvanica</i>	europaean alder	<i>Alnus glutinosa</i>
red maple	<i>Acer rubrum</i>	europaean ash	<i>Fraxinus excelsior</i>
red oak	<i>Quercus rubra</i>	europaean copper beech	<i>Fagus sylvatica</i>
red pine	<i>Pinus resinosa</i>	europaean larch	<i>Larix decidua</i>
red spruce	<i>Picea rubens</i>	europaean white birch	<i>Betula pendula</i>
serviceberry	<i>Amelanchier</i> spp.	freeman maple	<i>Acer x freemanii</i>
showy mountain ash	<i>Sorbus decora</i>	ginkgo	<i>Ginkgo biloba</i>
staghorn sumac	<i>Rhus typhina</i>	hackberry	<i>Celtis occidentalis</i>
striped maple	<i>Acer pensylvanicum</i>	hawthorn	<i>Crataegus</i> spp.
sugar maple	<i>Acer saccharum</i>	honey locust	<i>Gleditsia triacanthos</i>
trembling aspen	<i>Populus tremuloides</i>	horse chestnut	<i>Aesculus hippocastanum</i>
white ash	<i>Fraxinus americana</i>	japanese cherry	<i>Prunus serrulata</i>
white birch	<i>Betula papyrifera</i>	japanese dogwood	<i>Cornus kousa</i>
white cedar	<i>Thuja occidentalis</i>	japanese maple	<i>Acer palmatum</i>
white pine	<i>Pinus strobus</i>	japanese maple aureum	<i>Acer shirasawanum</i>
white spruce	<i>Picea glauca</i>	laburnum	<i>Laburnum anagyroides</i>
witch hazel	<i>Hamamelis virginiana</i>	littleleaf linden	<i>Tilia cordata</i>
yellow birch	<i>Betula alleghaniensis</i>	london plane	<i>Platanus x acerifolia</i>
		magnolia	<i>Magnolia</i> spp.
		non-native prunus	<i>Prunus</i> spp.
		norway maple	<i>Acer platanoides</i>
		norway spruce	<i>Picea abies</i>
		paperbark maple	<i>Acer griseum</i>
		peach tree	<i>Prunus persica</i>
		pin oak	<i>Quercus palustris</i>
		sargent's cherry	<i>Prunus sargentii</i>
		scots pine	<i>Pinus sylvestris</i>
		silver linden	<i>Tilia tomentosa</i>
		silver maple	<i>Acer saccharinum</i>
		sitka spruce	<i>Picea sitchensis</i>
		sweet cherry	<i>Prunus avium</i>
		tree lilac	<i>Syringa reticulata</i>
		water birch	<i>Betula occidentalis</i>
		white fir	<i>Abies concolor</i>
		willow	<i>Salix</i> spp.
		yellow cedar	<i>Cupressus nootkatensis</i>

Figure A5 Complete list of species sampled in Halifax.

Native species		Non-native species	
Common name	Latin name	Common name	Latin name
alternate-leaf dogwood	<i>Cornus alternifolia</i>	amur corktree	<i>Phellodendron amurense</i>
american chestnut	<i>Castanea dentata</i>	amur maple	<i>Acer ginnala</i>
american linden	<i>Tilia americana</i>	austrian pine	<i>Pinus nigra</i>
balsam fir	<i>Abies balsamea</i>	black locust	<i>Robinia pseudoacacia</i>
bitternut hickory	<i>Carya cordiformis</i>	blue spruce	<i>Picea pungens</i>
black ash	<i>Fraxinus nigra</i>	callery pear	<i>Pyrus calleryana</i>
black cherry	<i>Prunus serotina</i>	cherry plum	<i>Prunus cerasifera</i>
black maple	<i>Acer nigrum</i>	chinese elm	<i>Ulmus parvifolia</i>
black spruce	<i>Picea mariana</i>	columnar (european) hornbeam	<i>Carpinus betulus</i>
black walnut	<i>Juglans nigra</i>	common apple	<i>Malus</i> spp.
blue beech	<i>Carpinus caroliniana</i>	common pear	<i>Pyrus communis</i>
chinquapin oak	<i>Quercus muehlenbergii</i>	crabapple	<i>Malus</i> spp.
chokecherry	<i>Prunus virginiana</i>	curly willow	<i>Salix matsudana</i>
common juniper	<i>Juniperus communis</i>	dawn redwood	<i>Metasequoia glyptostroboides</i>
cucumbertree	<i>Magnolia acuminata</i>	douglas fir	<i>Pseudotsuga menziesii</i>
eastern hemlock	<i>Tsuga canadensis</i>	english oak	<i>Quercus robur</i>
eastern red cedar	<i>Juniperus virginiana</i>	european beech	<i>Fagus sylvatica</i>
flowering dogwood	<i>Cornus florida</i>	european buckthorn	<i>Rhamnus cathartica</i>
freeman maple	<i>Acer x freemanii</i>	european mountain ash	<i>Sorbus aucuparia</i>
hackberry	<i>Celtis occidentalis</i>	european white birch	<i>Betula pendula</i>
honey locust	<i>Gleditsia triacanthos</i>	field maple	<i>Acer campestre</i>
ironwood	<i>Ostrya virginiana</i>	ginkgo	<i>Ginkgo biloba</i>
kentucky coffee tree	<i>Gymnocladus dioica</i>	greenvase zelkova	<i>Zelkova serrata</i>
large tooth aspen	<i>Populus grandidentata</i>	homestead elm	<i>Ulmus x 'Homestead'</i>
pignut hickory	<i>Carya glabra</i>	horse chestnut	<i>Aesculus hippocastanum</i>
pin cherry	<i>Prunus pensylvanica</i>	japanese cherry	<i>Prunus serrulata</i>
red ash	<i>Fraxinus pennsylvanica</i>	japanese dogwood	<i>Cornus kousa</i>
red maple	<i>Acer rubrum</i>	japanese maple	<i>Acer palmatum</i>
red mulberry	<i>Morus rubra</i>	katsura tree	<i>Cercidiphyllum japonicum</i>
red oak	<i>Quercus rubra</i>	littleleaf linden	<i>Tilia cordata</i>
red pine	<i>Pinus resinosa</i>	london plane	<i>Platanus x acerifolia</i>
red spruce	<i>Picea rubens</i>	manitoba maple	<i>Acer negundo</i>
redbud	<i>Cercis canadensis</i>	northern catalpa	<i>Catalpa speciosa</i>
serviceberry	<i>Amelanchier</i> spp.	norway maple	<i>Acer platanoides</i>
shagbark hickory	<i>Carya ovata</i>	norway spruce	<i>Picea abies</i>
showy mountain ash	<i>Sorbus decora</i>	olive tree	<i>Olea europaea</i>
silver maple	<i>Acer saccharinum</i>	paperbark maple	<i>Acer griseum</i>
slippery elm	<i>Ulmus rubra</i>	pioneer elm	<i>Ulmus x hollandica 'Pioneer'</i>
staghorn sumac	<i>Rhus typhina</i>	prickly juniper	<i>Juniperus oxycedrus</i>
sugar maple	<i>Acer saccharum</i>	russian olive	<i>Elaeagnus angustifolia</i>
sweet birch	<i>Betula lenta</i>	sargent's cherry	<i>Prunus sargentii</i>
tamarack	<i>Larix laricina</i>	scots pine	<i>Pinus sylvestris</i>
trembling aspen	<i>Populus tremuloides</i>	silver linden	<i>Tilia tomentosa</i>
tulip tree	<i>Liriodendron tulipifera</i>	sitka spruce	<i>Picea sitchensis</i>
white ash	<i>Fraxinus americana</i>	magnolia spp.	<i>Magnolia</i> spp.
white birch	<i>Betula papyrifera</i>	sycamore maple	<i>Acer pseudoplatanus</i>
white cedar	<i>Thuja occidentalis</i>	tree lilac	<i>Syringa reticulata</i>
white oak	<i>Quercus alba</i>	venus dogwood	<i>Cornus nuttallii</i>
white pine	<i>Pinus strobus</i>	water birch	<i>Betula occidentalis</i>
white spruce	<i>Picea glauca</i>	white mulberry	<i>Morus alba</i>
witch hazel	<i>Hamamelis virginiana</i>	willow tree	<i>Salix</i> spp.
yellow birch	<i>Betula alleghaniensis</i>	yellow cedar	<i>Cupressus nootkatensis</i>

Figure A6 Complete list of species sampled in London.

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