URBAN FOREST VULNERABILITY AND ITS IMPLICATIONS FOR ECOSYSTEM SERVICE SUPPLY AT MULTIPLE SCALES

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Title: Urban Forest Vulnerability and Its Implications for Ecosystem Service Supply at Multiple Scales

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ABSTRACT

The urban forest is a valuable ecosystem service provider that is garnering increasing attention in environmental research and municipal planning agendas. However, because of its location in heavily built-up and densely-settled environments, the urban forest is vulnerable. The purpose of this dissertation is to conceptualize, assess, and analyze urban forest ecosystems and their vulnerability at multiple spatial and temporal scales. An urban forest ecosystem classification framework that integrates biophysical, built, and human components is developed. Subsequent classification of ecosystems at the neighbourhood scale reveals the spatial arrangement of several social-ecological interactions. Such information is valuable to ecosystem-based decision support while also informing future vulnerability research. The investigation of ecosystem vulnerability began with the development of a theory-based conceptual framework. Urban forest vulnerability is defined as the likelihood of decline in ecosystem service supply and its associated benefits for human populations, urban infrastructure, and biodiversity. It is comprised of exposure, sensitivity, and adaptive capacity, which describe the built environment and associated stressors, urban forest structure, and the human population, respectively. This framework is applied using empirical field research in Toronto, Canada to explore the processes of vulnerability and their influence on ecological change. Results indicate that there are several significant predictors of urban forest decline and mortality, and emphasize the importance of applying diverse metrics to describe the built environment and urban forest structure at fine spatial scales. Vulnerability assessment and analysis at much broader spatial and temporal scales, using a spatially-explicit assessment approach and ecological modelling of alternative management and disturbance scenarios, is further investigated. This latter research emphasizes the importance of aligning scales of management with ecosystem function and the long-term influence of management

intervention on ecological conditions. The multiple scales of investigation and methodological approaches developed in this study provide complementary opportunities to synthesize and apply existing theory in novel settings while also generating new theories pertaining to the processes of urban ecosystem change and decline. The intention of this study is to contribute to the understanding of urban forest ecosystems and their vulnerability, while also providing practical knowledge and tools for the sustainable management of this resource.

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DEDICATION

Dedicated to Jasmine

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CHAPTER 1 INTRODUCTION

1.1. Background

The urban environment is quickly becoming the most common setting in which people worldwide will spend their lives. Over 80% of Canadians and half of the global population now live in urban areas (United Nations, 2014), with the latter expected to increase to 60% by 2030 (United Nations Population Fund, 2007). Moreover, given the corresponding decline in rural populations, it is predicted that all global population growth over the next several decades will be within cities (United Nations Population Fund, 2007). Urban areas are also growing in extent, as urbanization and urban expansion are occurring at a rate that exceeds human population growth (Alig et al., 2004).

Maintaining and enhancing ecosystems like the urban forest are consequently becoming greater priorities for municipalities, as these resources become the primary point of contact in which people will experience and interact with nature (Alberti et al., 2003; Pickett et al., 2011). Moreover, the ecosystem services that urban forests provide to city-dwellers represent a diverse and substantial set of environmental, social, and economic benefits (Dwyer et al., 1992; Bolund & Hunhammer, 1999; Nowak & Dwyer, 2007), and are now recognized as a vital component in the overall sustainability of cities (Grove, 2009; Duinker et al., 2015). With the growing extent and relevance of urban areas to the majority of the global population, the need to understand, quantify, and manage these ecosystem services is becoming increasingly researched, with findings implemented in the form of urban ecosystem management strategies.

The sustainable management of urban forests also brings with it several challenges. It is widely accepted that cities have negative ecological consequences for the surrounding ecosystems that sustain them (Rees, 1996; Collins et al., 2000). However, the densely-settled human populations and frequent alteration and degradation of natural environments that characterize cities also make a difficult setting for forest growth and establishment (Nowak et al., 2004; Trowbridge & Bassuk, 2004; Konijnendijk et al., 2005). There is a growing body of research on the effects of various stressors and disturbances on individual trees and urban forests. However, there is a considerable knowledge gap on their combined effects and interaction with the built environment and human populations, and more importantly, their implications for ecosystem service supply. The growing saliency of trees and forests in cities and their overall role in urban sustainability now demands an expansive discourse that addresses the vulnerability of the urban forest resource and its associated benefits.

1.2. Defining the Urban Forest Ecosystem

Before any substantive discussion of ecosystem services and their vulnerability, it is first important to clearly define and conceptualize urban forest ecosystems. There are some disciplinary and regional variations in the general definition of the urban forest. The most universal definition of the urban forest in North America describes it as all trees and associated vegetation within a city boundary (Miller, 1997). However, there has been considerable evolution in both concept and terminology. In North America, early urban forest definitions had their beginnings in arboriculture and individual tree care, with an emphasis on shade and street trees (McPherson, 2003). Outside of the North American perspective, the term urban forest may have different connotations, where historically it might have referred to municipally-owned and managed woodlots or more recently as part of the broader concept of urban green space (Konijnendijk et al., 2006). Growing recognition of the ecosystem services provided by trees and forests in cities, as well as their susceptibility to pests and other urban stressors, has moved urban forest definitions towards more holistic and ecosystem-based concepts. Urban forests in North American research and practice are now recognized as individual trees, forest stands, and their associated biotic and abiotic components (Kenney et al., 2011). Additionally, urban forests definitions also recognize the people that dwell within them, along with their institutions and infrastructure (Konijnendijk et al., 2006), which aligns more with the modern ecosystem concept.

While the term ecosystem may invoke images of forests, mountains, and other natural settings, the majority of people will spend their lives in urban ecosystems. Urban ecological theory has made several recent advancements in the definition of the ecosystem concept over the past several decades (O'Neill, 2001; Pickett et al., 2011). One such advancement is the recognition that urban ecosystems are not limited to biophysical components and processes, but that social structures and organizations also determine the structure and function of these ecosystems (Pickett & Grove, 2009; Pickett et al., 2011). This integrated approach to investigating and characterizing urban ecosystems is captured in the shift from envisioning ecology in cities to the ecology of cities (Grimm et al., 2000). Research that examines ecology in cities tends to focus on characterizing previously established ecosystem patterns and processes, and how they differ in the urban environment compared to more natural, or undisturbed, settings (Grimm et al., 2000). In contrast, the approach of viewing the ecology of cities explores the structure and function of urban ecosystems, inclusive of their human components and built environments.

Correspondingly, urban forest ecosystems are highly heterogeneous across the urban landscape and are characterized by a complex interaction of social and biophysical processes across spatial and temporal scales (Grove et al., 2006a; Cadenasso et al., 2007). As with hinterland forest ecosystems, urban forest structure is affected by biogeophysical factors, such as soils, geology, topography, and climate (Nowak, 1994). This is especially true in forested parks and residual woodlands. However, unlike hinterland forests, the built environment and infrastructure are a significant determinant of urban forest structure and function (Heynen & Lindsey, 2003; Conway & Hackworth, 2007; Tratalos et al., 2007). The varying density of built structures and impervious surfaces, and the type and intensity of land uses, are all highly influential on urban forest ecology (Grove et al., 2006b; Conway & Hackworth, 2007; Conway & Bourne, 2013; Grove et al., 2014). Moreover, they are also indicative of the presence and magnitude of stressors that affect tree growth and survival (Nowak et al., 2004; Jutras et al., 2010; Koeser et al., 2013).

The influential human component of urban forest ecosystems also sets them apart from more-naturalized and non-urban forest ecosystems. An increasing body of research points to demographic and socioeconomic determinants of the distribution of tree canopy cover and structure of tree communities (Heynen & Lindsey, 2003; Grove et al., 2006a; Troy et al., 2007; Pham et al., 2013). Much of this research points to the close association of urban forest amenities with socioeconomic status, where positive relationships exist between tree cover and wealth, education, and land tenure (Boone et al., 2010; Grove et al., 2014). Moreover, certain demographics and ethnocultural groups may be more likely to plant or maintain trees on their properties or engage in stewardship activities (Fraser & Kenney, 2000; Greene et al., 2011). Social-ecological determinants of urban forest structure and function are often discernible at the neighbourhood scale, where socioeconomic groups tend to cluster (Weiss, 2000) and residents are more likely to maintain their properties in a fashion similar to their neighbours (Grove et al., 2006b; Boone et al., 2010). This diversity of social and biophysical urban forest processes translates not only into a high degree of spatial heterogeneity in structure, but a high variability in the supply of urban forest ecosystem services across the urban landscape (Zipperer et al., 1997; Grimm et al., 2000; Escobedo & Nowak, 2009).

1.3. Ecosystem Services

Ecosystem services are a prominent theme in both urban forest research and policy development (Nowak & Dwyer, 2007; Ordóñez & Duinker, 2013). They derive from a broad and variable range of urban forest ecosystem functions, which are the biotic and abiotic components and processes of ecosystems that are intrinsic to their self-maintenance (Odum & Barrett, 2005). Correspondingly, ecosystem services signify the benefits that individuals and societies derive from these functions (Costanza et al., 1997). In the broader context, the human race is entirely dependent on ecosystem services like climate regulation, water supply, pollination, and habitat provision for subsistence (Costanza et al., 1997). While city populations are invariably dependent upon ecosystems and their services beyond city limits (Rees, 1996; Bolund & Hunhammar, 1999), there is a growing recognition of the variety of benefits provided by urban ecosystem services within cities (Dobbs et al., 2011; Millward & Sabir, 2011). Urban forests are a dominant natural feature in many cities and provide city inhabitants, municipal infrastructure longevity, and the surrounding environment with a remarkably diverse array of ecosystem services. While urban forest ecosystem services are broad and often overlapping or cumulative, they can be conveniently divided into social, economic, and environmental/ecological benefits.

1.3.1. Social benefits

Trees in the city have a positive influence on the psychosocial well-being of urban inhabitants (Ulrich et al., 1991; Kaplan, 1995). In the simplest terms, the aesthetic beauty of trees and forests and the sense of place they impart are highly valued by the citizenry (Peckham et al., 2013). While these ecosystem services are difficult or impossible to quantify, they are recognized and valued by citizens. A recent study suggests that non-material or amenity-based ecosystem services are in fact the most valued (Peckham et al., 2013). There are also various cultural values and benefits associated with urban forests and green spaces (Konijnendijk, 2008). For instance, cultural backgrounds from the English Isles tend to value larger shade trees, while Mediterranean cultures have been shown to prefer food-producing trees and gardens (Fraser &

Kenney, 2000). City trees also have place-based cultural value as sites of personal and historical significance (Peckham et al., 2013).

These psychological and cultural benefits also have a physiological component and can translate to positive health effects. For example, in a landmark study, visual exposure to trees and natural settings were found to reduce hospital recovery times from surgery (Ulrich, 1984). Other examples include shade provisioning to reduce skin cancer risk and reducing health incidents associated with air pollution (Nowak & Dwyer, 2007). Urban forests, especially those with high connectivity, offer recreational benefits (Gobster & Westphal, 2004). Lastly, the urban forest and urban vegetation in general also contribute to a reduction in criminal activity (Kuo & Sullivan, 2001). In a recent study, Troy et al. (2012) found an inverse relationship between canopy cover and crime rates in the City of Baltimore.

1.3.2. Economic benefits

There are several direct and quantifiable economic benefits from the urban forest that have been documented. A number of studies have shown positive contributions to residential property values by trees (Tyrväinen & Miettinen, 2000; Donovan & Butry, 2010). Moreover, residential shade trees can yield economic benefits through home energy conservation, particularly by reducing the demand for air conditioning and its associated costs (Sawka et al., 2013). The visual attractiveness of trees has been shown to positively influence consumers' perceptions of business districts (Joyce et al., 2010). Consumers also admitted they were willing to pay higher prices, travel further and longer, and shop longer and more frequently in areas with green streetscapes (Wolf, 2005). A valuable service that benefits municipal public works budgets is the improved infrastructure longevity associated with urban trees. The shade provided by tree canopies reduces the amount of maintenance and repair required for city streets by slowing the rate of volatilization in asphalt cement, which is responsible for pavement cracking and increased repaving costs (McPherson & Muchnik, 2005).

Within urban forestry, the management focus is primarily on ecosystem services as amenities and not as goods, as is the case with wood fibre in traditional forestry. However, with the growing urban population and increasing public interest in urban forests there exists a small market for goods derived from city trees. Value-added timber products from urban sources are developing a market, especially for salvaged urban wood where an invasive forest pest, such as the emerald ash borer (*Agrilus planipennis*; EAB), has caused widespread mortality (Bratkovich & Bowyer, 2008; Herms & McCullough, 2014). There is also a movement towards urban food production and the value of fruit and nut trees (Clark & Nicholas, 2013). While these are examples of direct economic benefits, the majority of urban forest ecosystem services have some indirect financial value.

1.3.3. Environmental/ecological benefits

The urban forest provides a wide range of ecological and environmental benefits. Healthy and extensive urban forests and naturalized areas provide wildlife habitat and connectivity in cities where habitat is typically degraded (Adams, 2005; Calder et al., 2008). Urban forests help to mitigate air pollution, as they remove pollutants like tropospheric ozone (O_3), carbon monoxide (CO), and sulphur dioxide (SO_2) from the atmosphere and intercept particulate pollutants on the surface of their foliage (Nowak et al., 2006). Urban tree canopies also intercept and slow stormwater and thus help to abate the flow of contaminated runoff into aquatic systems and reduce stress on wastewater infrastructure (Xiao & McPherson, 2002). Foliar transpiration and shading help to cool ambient air temperatures and inhibit surface warming, respectively, and thereby reduce the urban heat-island effect (Solecki et al., 2005; Sawka et al., 2013). Finally, urban forests both help to mitigate climate change through carbon sequestration and storage in woody biomass (Nowak & Crane, 2002) and help cities adapt to climate change through many of the aforementioned environmental services (Ordóñez & Duinker, 2014). The i-Tree Eco suite of models, which were developed by the United States Department of Agriculture (USDA) Forest Service to quantify urban forest structure and function (USDA Forest Service, 2013a), have been predominately used to quantify these environmental benefits.

1.3.4. Ecosystem disservices

City trees can have adverse effects on the urban environment and its inhabitants. Allergens from tree pollen are among the top negative issues that city dwellers associate with the urban forest (Lohr et al., 2004). Trees also emit volatile organic compounds (VOCs), which can lead to O_3 and CO formation (Brasseur & Chatfield, 1991). Tree growth frequently conflicts with built infrastructure. For example, tree root growth can cause significant costs for street pavement and sidewalk repair (McPherson, 2000). Trees can also present a hazard to property and human health due to blowdown during storm events (Lopes et al., 2009). Ice storms present infrequent but hazardous events due to limb breakage, tree failures, and associated damage to utility lines (Hauer et al., 1993; Irland, 2000). Moreover, natural or disturbance-related mortality and the presence of standing deadwood both present a similar hazard or significant costs for removal, as is the case currently with widespread ash (*Fraxinus* spp.) mortality caused by the EAB (Kovacs et al., 2010). Importantly, many of the disservices attributed to urban trees and forests are based on individual preferences (Lohr et al., 2004; Kirkpatrick et al., 2012) and are thus highly variable. For example, some residents claim that the messiness of trees is a negative issue, while others do not (Lohr et al., 2004). The attitudes of residents towards trees have been shown to vary according to land tenure (e.g., ownership conflicts; Kenney & Idziak, 2000), age (Conway & Bang, 2014), and cultural background (Fraser & Kenney, 2000).

1.4. Urban Forest Management

Urban trees and forests represent a considerable challenge in the context of environmental and natural resource management (Alberti et al., 2003; Konijnendijk et al., 2005). Unlike more-naturalized forest settings, much of the urban landscape requires management intervention to maintain function and ensure tree establishment and survival. Urban forest ecosystems require varying intensities and frequencies of management intervention to ensure the supply of beneficial ecosystem services while also mitigating potential threats and disservices (Trowbridge & Bassuk, 2004; Duinker et al., 2015). Furthermore, there are more than just ecological considerations to address in the management of this resource. In addition to the supply of various ecosystem services, it is important to consider social equity in their distribution, as lower-income neighbourhoods and marginalized ethnocultural groups tend to be associated with less green space, degraded urban forest health, and lower canopy cover (Heynen et al., 2006; Troy et al., 2007; Landry & Chakraborty, 2009; Pham et al., 2013).Consequently, the practice of urban forestry is gaining more attention and becoming entrenched in many municipalities.

The management of urban trees and forest ecosystems is a complex and dynamic practice and science that has evolved over the past century (Konijnendijk et al., 2006). Urban forestry in North America became a separate practice and profession from traditional forestry in the 1960s, when it became an early amalgamation of forestry with horticulture, arboriculture, and landscape architecture (Miller, 1997). Arguably, broad establishment and acceptance of urban forestry was cemented by the Dutch elm disease (*Ophiostoma novo-ulmi*), which ravaged the extensively planted elm (*Ulmus* spp.) populations of North American cities and first exposed the public to the consequences of widespread urban canopy loss (Johnston, 1996). While urban forest management in many ways remains reactive and practice-oriented, being driven by immediate disturbance and threats like the Dutch elm disease, more advanced and progressive models of urban forest governance are emerging (Lawrence et al., 2013). Urban forest management today continues to grow beyond operational necessities like tree planting, maintenance, and removal.

The complexity and heterogeneity of urban forest ecosystems and the growing importance of cities have demanded more holistic and interdisciplinary approaches to understanding them. This requires not only maintenance operations and municipal policies, but on-going research from the social, natural, and applied sciences, as well as partnerships between governments, industry, academia, and communities (Konijnendijk, 2004). The Society of American Foresters currently defines urban forestry as “the art, science, and technology of managing trees and forest resources in and around urban community ecosystems for the physiological, sociological, economic, and aesthetic benefits trees provide society” (Helms, 1998, p. 193), which is more representative of modern understandings of the urban forest. In fact, today it is argued that urban forest management and governance can be particularly innovative, as they often involves partnerships with a variety of non-government stakeholders, such as environmental non-governmental organization, citizen associations, landowners, and industry (Lawrence et al., 2013).

Across North America, the urban forest is increasingly becoming an item on municipal planning agendas and many cities are creating policies and strategic plans that address their urban forest resource (Conway & Urbani, 2007; Ordóñez & Duinker, 2013; Steenberg et al., 2013; Gibbons & Ryan, 2015). The design and implementation of tree protection regulations for public and private properties are becoming commonplace in larger cities (Kenney & Idziak, 2000; Conway & Urbani, 2007). Many of these regulations focus on the development process, since development practices and land-use change are among the largest contributors to urban tree mortality and canopy loss (Kenney & Idziak, 2000). Municipalities are also adopting comprehensive and strategic urban forest management plans (Ordóñez & Duinker, 2013; Gibbons & Ryan, 2015), which generally consist of guidelines for tree planting and species selection, tree maintenance (e.g., pruning-cycle establishment), pest management, conservation goals, and performance standards (van Wassenaer et al., 2000). Such planning documents are an important step for communities to acknowledge urban forests as a public good and a key stage of policy development for ensuring explicit and consistent goals for long-term sustainable urban forest management (Clark et al., 1997).

However, the urban forest is a vulnerable resource. Densely-settled and heavily built-up areas represent considerably different conditions from which most tree species have evolved. There is also an abundance of stressors and disturbances that afflict city trees, ranging from soil loss/degradation and pollution to invasive species to poor management practices (Trowbridge & Bassuk, 2004; Konijnendijk et al., 2005; Laćan & McBride, 2008; Jutras et al., 2010; Lu et al., 2010). Indeed, many urban trees suffer disproportionate rates of decline and mortality due to these physical, biological, and social stressors and disturbances (Nowak et al., 1990; Hauer et al., 1994; Randrup et al., 2001; Nowak et al., 2004; Roman & Scatena, 2011; Lawrence et al., 2012; Koeser et al., 2013). A prominent focus in municipal urban forest policy and strategic management in North America is on urban forest ecosystem services and their associated benefits (Ordóñez & Duinker, 2013; Steenberg et al., 2013). Arguably, there is less attention given to potential threats to urban forest ecosystems. Moreover, from a research perspective, there are many studies on ecological disturbance and stressors of urban forests, especially street trees (e.g., Hauer et al., 2011; Roman & Scatena, 2011; Koeser et al., 2013), but little discussion of the overall vulnerability of these social-ecological systems.

1.5. Study Overview and Objectives

The purpose of this dissertation is to conceptualize, assess, and analyze urban forest ecosystems and their vulnerability at multiple spatial and temporal scales. Specific research questions include:

- How might diverse social and biophysical factors describing urban forest structure and function be used to classify urban forest ecosystems at the neighbourhood scale?

- What components of urban forest ecosystems, inclusive of their human population and built environment, represent key vulnerabilities in ecosystem service supply?

— What components of urban forest vulnerability best explain and predict ecosystem change and potential decline?

— What is the spatial and temporal variability in urban forest structure and function under different management and disturbance scenarios?

— What are the implications for, and possible uses of, vulnerability assessment and analysis in urban forest planning and management?

To approach and answer these research questions, this dissertation is organized into four independent but closely-related and cumulative bodies of research in the format of manuscript-style chapters. Prior to the four research contributions to the dissertation in Chapters Two through Five, Chapter One provides background information on cities, urban forest ecosystems, ecosystem services, and management. The first body of research presented in Chapter Two involves the development, proposal, and application of a framework for urban forest ecosystem classification (UFEC) that integrates ecosystem components characterizing the biophysical landscape, built environment, and human population. Despite recent advances in urban ecological theory and the ecosystem concept (e.g., O'Neill, 2001; Pickett et al., 2011), there remain many divergences across disciplines in how both urban landscapes and forest ecosystems are stratified, classified, and modelled. Specifically, there is uncertainty around the stratification and classification of urban forest ecosystems at spatial scales best suited to strategic planning and sustainable urban forest management.

Ecosystem classification is an approach that entails quantifying the processes that shape ecosystem conditions into logical and relatively homogeneous management units, making the potential for ecosystem-based decision support available to planners (Klijn & Udo de Haes, 1994; Hargrove & Hoffman, 2005; Bailey, 2009). The multifactor UFEC developed in this study was developed with such a purpose in mind. It was then applied in Toronto, Canada using spatially-explicit social and biophysical variables and hierarchical cluster analysis to categorize city neighbourhoods into independent urban forest ecosystem classes. The structure of the urban forest ecosystem classes identified is analyzed, including a comparison across the different classes and ecosystem components. Next, the implications of these findings for Toronto and their broader relevance for both urban ecological theory and sustainable management are discussed. This research was an important first stage of the study, as it was critical to first analyze and communicate how urban forest ecosystems will be conceptualized prior to any exploration of their vulnerability.

The second body of research in Chapter Three begins this dissertation's core area of focus on urban forest ecosystem vulnerability. Vulnerability analysis and assessment is an increasingly used concept and method for approaching issues of system sustainability and ecosystem service supply (Turner et al., 2003a; Schröter et al., 2005; Adger, 2006; Eakin & Luers, 2006; Lindner et al., 2010). Modern definitions of vulnerability identify it as an element of social-ecological systems that is an outcome of multiple and interacting social and biophysical properties across spatial and temporal scales (Metzger et al., 2006). It is comprised of: 1) exposure, which are external stressors and disturbances that negatively affect function; 2) sensitivity, which is the internal system structure that magnitude of potential impacts in response to stress; and, 3) adaptive capacity, which is the capacity of a system to shift or alter its state to reduce its vulnerability (Turner et al., 2003a; Adger, 2006; Eakin & Luers, 2006). The complex, heterogeneous, and adaptive interaction of social and biophysical processes that shape urban forests make them a prime candidate for vulnerability science. This Chapter describes the design of a theory-based conceptual framework and set of potential indicators for the assessment and analysis of urban forest ecosystem vulnerability that will be adapted and applied in Chapters Four and Five.

First, a review of both vulnerability in social-ecological systems and of the various bodies of literature that contribute to the study of urban forest ecology and management in a vulnerability context is given. Next, a detailed description of the vulnerability framework and its justification for urban forest research and practice are discussed, including the identification of potential indicators of urban forest exposure, sensitivity, and adaptive capacity. A description of the various approaches, models, metrics, and methods for vulnerability assessment and analysis is then included. Lastly, a discussion of the strengths and weaknesses of a vulnerability approach, alternative approaches to the study of urban social-ecological systems, and implications for urban forest planning and management is provided.

Chapter Four then presents the application of this vulnerability framework in the field in an individual neighbourhood in Toronto. The objective of this exploratory analysis is to both profile the nature of ecological change among public trees in a residential neighbourhood and to analyze the relative influence of the various components of vulnerability thereon. The conceptual framework of urban forest vulnerability from Chapter Three is first adapted according to the study's objectives, scale of analysis, and feasibility of measurement. Ecological change is characterized by tree mortality, condition, diameter growth, and planting rates, which were measured using comparisons with existing tree inventory data collected in 2007 and 2008. Linear and logistic regression analyses are used to explore the explanatory power and predictive capacity of different indicators of vulnerability on ecosystem change at the individual-tree and street-section scale. Inductive approaches to vulnerability analysis based on empirical, observable phenomena at finer spatial and temporal scales, such as this study, are important for advancing the theoretical understanding of vulnerability in complex systems like the urban forest (Adger, 2006; Füssel, 2010). However, they cannot reveal all social and biophysical aspects of system vulnerability, especially long-term risk (Hinkel, 2011). Consequently, more deductive and theory-driven approaches that employ ecological modelling over broader spatial and temporal scales are a common component of vulnerability science (Füssel, 2010; Hinkel, 2011).

Chapter Five is one such approach that combines the theory-based conceptual framework of vulnerability with forward-looking ecological modelling to assess and analyze the spatio-temporal variability and long-term vulnerability of Toronto's urban forest ecosystem service supply. Specific objectives of this final research chapter are to conduct a quantitative, spatially-explicit assessment of urban forest vulnerability in Toronto ecosystems based on current conditions and to model temporal changes in urban forest structure and function under different management and disturbance scenarios using the i-Tree Forecast model (Nowak & Crane, 2000; Nowak et al., 2014). This research was conducted at the ecosystem and municipal scale, using the urban forest ecosystem classes identified in Chapter Two. Lastly, Chapter Six summarizes the main conclusions of this dissertation and their broader significance, followed by a discussion of potential areas for future research.

As the global population continues to concentrate in urban areas (United Nations, 2014) and increasingly relies on the ecosystem services and associated benefits provided by trees and urban forests (Grove, 2009; Duinker et al., 2015), understanding the drivers and processes of urban forest change, and potential loss of function, is vital for strategic planning and sustainable management necessary to minimize future ecosystem vulnerability.

CHAPTER 2 NEIGHBOURHOOD-SCALE URBAN FOREST ECOSYSTEM CLASSIFICATION

Abstract: Urban forests are now recognized as essential components of sustainable cities, but there remains uncertainty concerning how to stratify and classify urban landscapes into units of ecological significance at spatial scales appropriate for management. Ecosystem classification is an approach that entails quantifying the social and ecological processes that shape ecosystem conditions into logical and relatively homogeneous management units, making the potential for ecosystem-based decision support available to urban planners. The purpose of this study is to develop and propose a framework for urban forest ecosystem classification (UFEC). The multifactor framework integrates 12 ecosystem components that characterize the biophysical landscape, built environment, and human population. This framework is then applied at the neighbourhood scale in Toronto, Canada, using hierarchical cluster analysis. The analysis used 27 spatially-explicit variables to quantify the ecosystem components in Toronto. Twelve ecosystem classes were identified in this UFEC application. Across the ecosystem classes, tree canopy cover was positively related to economic wealth, especially income. However, education levels and homeownership were occasionally inconsistent with the expected positive relationship with canopy cover. Open green space and stocking had variable relationships with economic wealth and were more closely related to population density, building intensity, and land use. The UFEC can provide ecosystem-based information for greening initiatives, tree planting, and the maintenance of the existing canopy. Moreover, its use has the potential to inform the prioritization of limited municipal resources according to ecological conditions and to concerns of social equity in the access to nature and distribution of ecosystem service supply.

Keywords: ecosystem classification, urban forest, neighbourhood, management, cluster analysis

2.1. Introduction

Canadian landscapes and ecosystems tend to invoke images of forests, mountains, and other natural settings. However, most Canadians spend their lives in urban ecosystems, with over 80% of Canada's population currently residing in cities, while greater than half of the global population now calls urban centres home (United Nations, 2014). These trends have important implications for how societies interact with nature and, perhaps more importantly, how urban ecosystems can provide sustainable benefits for cities in the future. There is a growing recognition among city inhabitants of urban forests and the diverse array of ecosystem services they provide (Dwyer et al., 1992; Nowak & Dwyer, 2007). For example, stormwater retention and moderation of urban heat island effects by urban forests are highly beneficial for both city infrastructure and human health (Nowak & Dwyer, 2007).

Ecosystem services and their connection to healthy urban forests are now becoming recognized as essential components of sustainable cities and integral to municipal climate change action (Grove, 2009; Demuzere et al., 2014; Duinker et al., 2015). As a result, many municipal governments are investing in the development of urban forest strategies that include management plans and policies that seek to protect and expand urban forest resources (Kenney & Idziak, 2000; Ordóñez & Duinker, 2013). Nevertheless, many important areas of urban forest research and management remain underdeveloped and unexplored. One question that remains unanswered is how to stratify and classify the urban landscape into units of ecological significance and at spatial scales best suited to strategic planning and sustainable urban forest management. The urban landscape is highly heterogeneous and characterized by an especially complex interaction of social and biophysical processes (Grove et al., 2006b; Cadenasso et al., 2007). Consequently, to adequately assess urban forest structure and function, and to inform decision making in urban forestry, some form of landscape stratification and classification is necessary.

Ecosystem classification may present an effective approach for addressing these challenges. It enables researchers and practitioners to evaluate and manage complex multifactor systems. By allowing for the organization of ecological processes into logical and relatively homogeneous management units, classification frameworks make valuable tools for ecosystem-based planning and decision support (Klijn & Udo de Haes, 1994; Hargrove & Hoffman, 2005). Ecosystem classification provides a holistic approach to management by expanding focus from a single resource or amenity (e.g., timber supply in forestry) to the structure and function of the entire ecosystem (Bailey, 2009). Moreover, urban ecosystem classification can address the heterogeneous and hierarchical nature of social-ecological systems (Cadenasso et al., 2013) by enabling the stratification and classification of landscapes at spatial scales identified as important for maintaining desirable ecosystem functions and services. However, ecosystem classification for environmental and natural resource management remains untested in urban settings. Consequently, advancements in urban ecology and the ecosystem concept have not been addressed in ecosystem classification.

The field of urban ecology has made important strides regarding the definition of the ecosystem concept over the past several decades and represents an important knowledge frontier. One such advancement within this discipline is the recognition that urban ecosystems are not limited to biophysical components and processes, but that social workings and practices are also integral to the structure and function of these ecosystems (Lister, 2008; Pickett et al., 2011). This integrated approach to investigating and characterizing urban ecosystems is captured in the shift from envisioning ecology in cities to the ecology of cities (Grimm et al., 2000). Research that examines ecology in cities tends to focus on characterizing previously established ecosystem patterns and processes, and how they differ in the urban environment compared to more natural, or undisturbed, settings (Grimm et al., 2000). In contrast, the approach of viewing the ecology of cities explores the structure and function of urban ecosystems, inclusive of their human components.

Ecosystem classification within traditional ecology and natural resource management in non-urban settings has a much longer history and broader research base (Omernik, 1987). For example, forest ecosystem classification has been employed by forest practitioners for decades in many regions across Canada (e.g., Sims et al., 1996; Keys et al., 2003). This approach to classification at the stand level uses site, soil, and vegetation variables to inform ecosystem-based management decisions in forestry. At broader spatial scales, hierarchical, ecological land classifications based on physiographic, climatic, and biological conditions have been developed for several types of resource management and biodiversity conservation (Omernik, 1987; Klijn & Udo de Haes 1994). Despite their maturity and uptake, these approaches to ecosystem classification are designed for natural landscapes, and as such they are silent on the dynamic social processes and built environments that typify urban areas and fall short when applied to urban forest ecosystems.

Conventional approaches to classification in urban settings contrast sharply with those of more natural landscapes. While some research exists on biophysical classifications of cities (e.g., Brady et al., 1979), most often boundaries are delineated and units are classified by social factors, such as ownership, jurisdiction, and land use (Borgström et al., 2006). Thus, these approaches ignore key biophysical determinants of ecosystem structure and function. Land use has been a ubiquitous approach to classifying urban landscapes for planning purposes (Anderson et al., 1976), and is indeed used often in urban forest planning and management (Steenberg et al., 2013). Land use is highly influential on the structure of city tree communities as it describes the intensity of human activities and often the amount of available growing space (Nowak et al., 1996; Jutras et al., 2010), but it fails to capture important variability in the structure and function of urban forests. For example, within residential land uses there can be considerable amounts of variability in affluence and density, which have been found to be strong predictors of urban vegetation (Berland, 2012; Grove et al., 2014).

Describing and managing urban forest ecosystems, in light of this new paradigm that calls for research that more tightly integrates social and biophysical components, is necessary though challenging (Alberti et al., 2003). Urban ecosystem classification may provide a useful tool for practitioners while also presenting an opportunity to move some of the theoretical advancements of ecology into urban forest practice. The discipline of urban forest science, and the practice of urban forestry, has advanced sufficiently over the past decades to now warrant a classification system tailored to its purposes and not borrowed from forestry or urban planning in a best-fit scenario.

The purpose of this study is to develop and propose a framework for an urban forest ecosystem classification (UFEC). Specifically, the research objectives are to: 1) develop a multifactor classification framework that integrates ecosystem components characterizing the biophysical landscape, built environment, and human population; 2) apply the framework to classify urban forest ecosystems in Toronto, Canada, at the neighbourhood scale using hierarchical cluster analysis; and 3) interpret the biophysical and social structure of the resulting ecosystem classes and its implications for urban forest management. The intent of these research objectives is not only to make a practical contribution to urban forestry with the proposed UFEC, but also to make theoretical contributions to the study of urban forest ecosystems and ecosystem classification systems in urban areas.

2.2. Classification Framework

In conceiving the UFEC framework for application in this study, I propose three broad categories of ecosystem components: the biophysical landscape, the built environment, and the human population (Fig. 2.1). This section describes each component and the reasoning behind its inclusion in the framework. The selection of a suitable variable or variables to measure ecosystem components is dependent on both locally and regionally available data. Variables selected for this study are described in Section 3.2.

The human component and, to a lesser degree, the built environment represent novel components for inclusion in the definition and classification of ecosystems. The recognition that humans are integrated components of urban ecosystems, and not external forces of disturbance, has been widely accepted in urban ecological theory (e.g., Grimm et al., 2000; Alberti et al., 2003; Pickett et al., 2011). However, given the unpredictable nature of human behavior, and the likely challenges for both ethics and measurement feasibility, there are few examples of applied research that take this approach. Cadenasso and colleagues have advanced work on urban land cover classification by integrating natural and built components to better account for urban landscape heterogeneity (Cadenasso et al., 2007; Cadenasso et al., 2013). However, to our knowledge, no studies have developed or applied ecosystem classification methods that have integrated variables describing the human population.

2.2.1. Biophysical Landscape

Biophysical factors have the most prominent lineage within ecosystem classification in natural resource management (Omernik, 1987). As with non-urban forest ecosystems, urban forest structure and function are determined to some degree by pre-existing vegetation, soil type, topography, and climate (Nowak, 1994; Pickett et al., 2001). These influences are especially true in forested urban parks, residual woodlands, and other more naturalized settings. Vegetation is the first biophysical component in the proposed UFEC framework, as it is conceptually and logistically important in the context of sustainable urban forest management. The most common variable used to represent urban trees is canopy cover, owing to the relative ease of data acquisition and modest costs of processing (Nowak et al., 1996). However, more detailed tree inventories are also useful for urban forest classification. In addition to tree canopy cover, open green space (i.e., grass, shrub, and/or herbaceous cover) and relative stocking, are other important variables describing urban vegetation (Cadenasso et al., 2007; Kenney et al., 2011). Stocking is the ratio of canopy cover to total green space, where total green space is defined as tree canopy cover and open green space.

Topography is included in the UFEC as a biophysical ecosystem component because it influences the ecological processes within urban forests through exposure, drainage, light availability, and microclimate (Klijn & Udo de Haes, 1994; Bailey, 2009). Topographic variables are commonly used in ecosystem classifications, most prominent of these variables being slope gradient, slope/topographic position, and aspect (Sims et al., 1996; Keys et al., 2003; Bailey, 2009). In the urban landscape, where our UFEC is concerned, I suggest limiting the representation of topography to slope and/or slope position, given their combined influences on both the natural processes and patterns of urban development, where steeper terrain often remains forested due to higher costs and elevated safety risks (e.g., erosion or slope failure; Heynen & Lindsey, 2003). Soil type is also included as a biophysical component because it is an important driver of tree species suitability and growth; the soil-rooting medium provides essential water and nutrients for above-ground growth (Craul, 1999; Millward et al., 2011). The physical and chemical properties of soil greatly influence forest ecosystem structure and function and thus soil type is commonly used in ecosystem classification (Zhou et al., 2003; Bailey, 2009).

Climate is an important determining factor in ecosystem structure and function at spatial scales broader than what is typically encountered in cities; it is a key driver of tree species distribution (Hargrove & Hoffman, 2005). I have included climate as an ecosystem component in the UFEC framework in anticipation of future UFEC applications in other regions or across multiple cities. Since climate has also been central in the development of the broader-scale ecological landscape classifications for the determination of ecoregions (Omernik, 1987; Host et al., 1996), I suggest that ecoregion data, rather than raw climate data, would be a logical data choice to represent climatic influence on urban forest ecosystems. However, climate variables (e.g., temperature, precipitation, growing degree days) are not included in this study, since these data are unlikely to be highly influential on urban forest ecosystem variation within a single city. Importantly, the urban heat island effect and microclimates associated with building density, topographic variability, and temperature moderation by waterbodies and urban vegetation can produce considerable climatic variability within city limits and can influence ecological processes at much finer spatial scales (Taha, 1997). Data on these effects could be difficult to obtain for an entire city, though they may, in part, be captured in other ecosystem components, including existing canopy cover, building density, and topography.

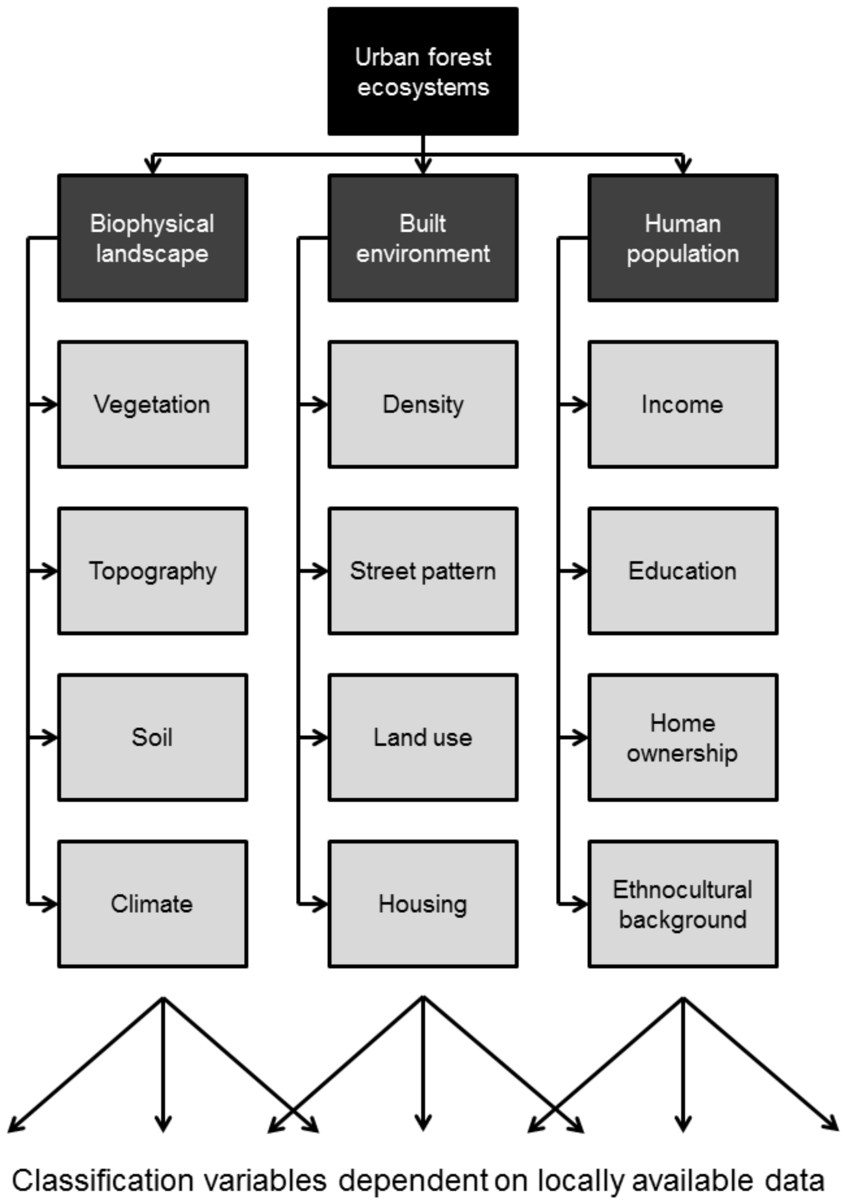

Fig. 2.1. Design of the framework for urban forest ecosystem classification (UFEC).

2.2.2. Built Environment

Unlike non-urban forest landscapes, the built environment and urban infrastructure are significant determinants of ecosystem structure and function within cities (Pickett et al., 2001; Grove, 2009). The type, orientation, and density of buildings and road networks, as well as the extent of impervious surfaces and other grey infrastructure, determine the physical space available for trees and other vegetation. Additionally, urban form and land use in general have been shown to be indicative of the presence and magnitude of stressors that affect tree growth and survival (Nowak et al., 2004; Jutras et al., 2010). Thus, the built environment is also indicative of environmental quality and the vulnerability of urban forest ecosystems.

Density is one of the most frequently used metrics in the assessment of urban form and the built environment (Churchman, 1999), and is included as an ecosystem component in the UFEC framework. However, density is a somewhat contentious term and has a number of definitions and measurements (Churchman, 1999). One important distinction is between building intensity and population density. Building intensity refers to the physical characteristics and coverage of the built environment in a defined area, such as the ratio of building footprints or floor area to parcel area (Forsyth, 2003). Population density is a simple area-based metric of people per geographic unit (e.g., km^2). Much of the existing research on the relationships between urban vegetation and the built environment employ the latter (e.g., Troy et al., 2007; Pham et al., 2013; Grove et al., 2014). However, building intensity and population density are not necessarily correlated in all instances, though they tend to be referred to interchangeably as density (Forsyth, 2003). I therefore recommend that both of these dimensions of density be included in UFEC development and applications.

In addition to density, neighbourhood-based research often incorporates land use mix and street pattern as the three main metrics of urban form (Southworth & Owens, 1993; Krizek, 2003; Conway & Bourne, 2013). As previously mentioned, land use is influential on the physical structure of urban forests (Nowak et al., 1996) and is included as an ecosystem component in the framework. Street pattern is also included as an ecosystem component. City streets represent a highly identifiable spatial network in the urban landscape and have long been used in urban stratification and boundary delineation (e.g., census boundaries, neighbourhoods, city wards, municipal boundaries). Street patterns, like road density and block size, are metrics of urban form that are influential on vegetation patterns (Conway & Bourne, 2013). Additionally, cityowned trees along streets and in public rights of way and their management comprise a significant share of municipal urban forestry (Maco & McPherson, 2002).

Lastly, housing type and age are frequently identified as important drivers of urban ecological processes in residential and mixed-use areas (Grove et al., 2006b; Conway & Hackworth, 2007; Conway & Bourne, 2013). While housing has important implications for socioeconomic background that will be captured in the human population ecosystem components, it is also a determinant of the physical environment. Housing type is highly influential on building intensity and is generally indicative of the type and amount of green space available for urban trees and other vegetation (Grove et al., 2006a; Troy et al., 2007). For example, single-family, detached housing is commonly associated with higher canopy cover (Troy et al., 2007). Additionally, housing age is influential on urban forest structure and is often indicative of tree size and a more extensive, mature canopy (Zipperer et al., 1997; Boone et al., 2010).

2.2.3. Human Population

The influential human component of urban forest ecosystems is what sets them apart from most other forest ecosystems in the context of both classification and management. While nearly all ecosystems on the planet are influenced by humans to varying degrees, urban forests in particular are characterized by dense human settlement, altered physical environments, land use competition, and a myriad of other sociopolitical and socioeconomic influences that shape ecosystem structure and function (Konijnendijk et al., 2006). A growing body of research points to demographic and socioeconomic influences on the distribution and structure of canopy cover and urban vegetation (e.g., Heynen & Lindsey, 2003; Grove et al., 2006a; Troy et al., 2007; Pham et al., 2013). The most consistent theme to emerge in this research is the positive association of socioeconomic status with urban tree cover (Grove et al., 2014). Specifically, income and education represent important determinants of urban forests and are both included in the UFEC framework as ecosystem components.

Homeownership is also included as an ecosystem component. Land tenure can affect urban forests in a number of fashions, especially in residential land uses (Boone et al., 2010). There is certainly correlation with the abundance of renters and housing type and the built environment (Boone et al., 2010). However, residents can directly influence the urban forest through various land management practices, or an absence thereof (Grove et al., 2006b). For one, homeowners, unlike renters, have direct legal control, and often obligation (e.g., tree by-laws), over their landscaping practices (Grove et al., 2014). Also, renters tend to be a more mobile population and less likely to invest resources in maintaining property vegetation (Troy et al., 2007).

Lastly, ethnocultural background is included as an ecosystem component as it can influence urban forest ecosystem structure. Ethnocultural groups differ in their connections to trees, which can result in different landscaping practices and influence the willingness to plant and maintain trees on their properties (Fraser & Kenney, 2000; Greene et al., 2011). A number of studies also address urban forests and ethnocultural background from an environmental justice perspective, where visible minorities and marginalized populations are often situated in neighbourhoods with less abundant green space and tree cover (e.g., Heynen et al., 2006; Landry & Chakraborty, 2009). While there are certainly instances of different ethnocultural groups preferring an absence of tree cover on their property (Fraser & Kenney, 2000), the latter issue highlights the importance of considering equity in the distribution of urban forest ecosystem services.

2.2.4. Ecosystem Boundaries

A core intention in the development of our UFEC framework is that city neighbourhoods, in addition to representing a logical spatial scale for management (Manzo & Perkins, 2006; Steenberg et al., 2013), also represent a consistent unit of analysis and bounding mechanism for urban forest ecosystems. Consequently, for the Toronto application of the UFEC I stratified the study area by neighbourhood. Often in ecosystem classification, landscapes are pre-stratified by climate, biogeography, topography, and other biophysical variables. In other instances, expert-based boundary delineation for stratifying landscapes is used prior to analysis (Hessburg et al., 2000; Hargrove & Hoffman, 2005). Arguably there are elements of both in this approach. Since this classification method attempts to minimize within-class heterogeneity of biophysical variables, but also of the human population and built environment variables, more traditional biophysical stratification and boundary delineation would not be appropriate. Therefore, neighbourhoods were selected to minimize trade-offs in both structural and functional homogeneity. However, it should be noted that, like ecosystems, neighbourhood boundaries are frequently subjective, ephemeral, or continuous (Galster, 2001). Despite their long history in urban planning, neighbourhoods remain difficult to define and map, and the implications and uncertainties around doing so in UFEC applications should be considered.

Neighbourhoods tend to be characterized by comparatively homogenous socioeconomic and structural conditions (Galster, 2001; Boone et al., 2010). Similar physical conditions in the built environment will certainly correspond to more homogenous ecosystem structure, but urban vegetation also tends to reflect the practices and values of the human population (Burgi et al. 2004). Group identity and the neighbourhood effect, where residents will maintain their lawns and trees in ways that conform to their neighbours, will also reinforce vegetation homogeneity within neighbourhoods (Zmyslony & Gagnon 2000; Grove et al., 2006a).

The notion of ecological trade-off between structurally different but functionally similar components is not entirely unique to urban landscapes. Bailey (2009) illustrates the example of floodplain ecosystems, which consist of a number of heterogeneous features (e.g., active and inactive channels, islands, and wetlands) that comprise a single functional unit. That the unit is classified according to system function and not structure alone is important if the purpose of the classification is floodplain management or a related field of research or practice. The research and/or management purpose of classifications must be considered in their design (Bailey, 2009). For example, within a single forest landscape, ecosystem classification for timber management and wildlife management might differ significantly in both classes and boundaries. Classification in forestry tends to focus on tree species composition and age structure, while wildlife management may focus on species distribution ranges and habitat connectivity. The aim of the ecosystem classification in this study is to characterize the urban forest in such a way that accounts for the social-biophysical interaction and spatial heterogeneity that influence the supply of ecosystem services at a scale that is relevant to decision makers and the citizenry.

2.3. Methods

2.3.1. Study Area

The study was conducted in Toronto, Ontario, Canada (Fig. 2.2). Toronto is the provincial capital of Ontario, with a total population of 2,615,060 and population density of 4,151 persons per km^2 as of 2011. Toronto is situated on the northwest shore of Lake Ontario, and is the economic centre of the Greater Toronto Area (GTA). It is a culturally diverse city, with 49.9% of Torontonians born outside of Canada and 46.9% identifying as a visible minority. Population growth rate in Toronto was 4.3% between 2006 and 2011 (City of Toronto, 2013a). The City of Toronto's Social Policy Analysis and Research section has designated 140 discrete neighbourhoods, which have since been used in urban forest policy development (City of Toronto, 2013c).

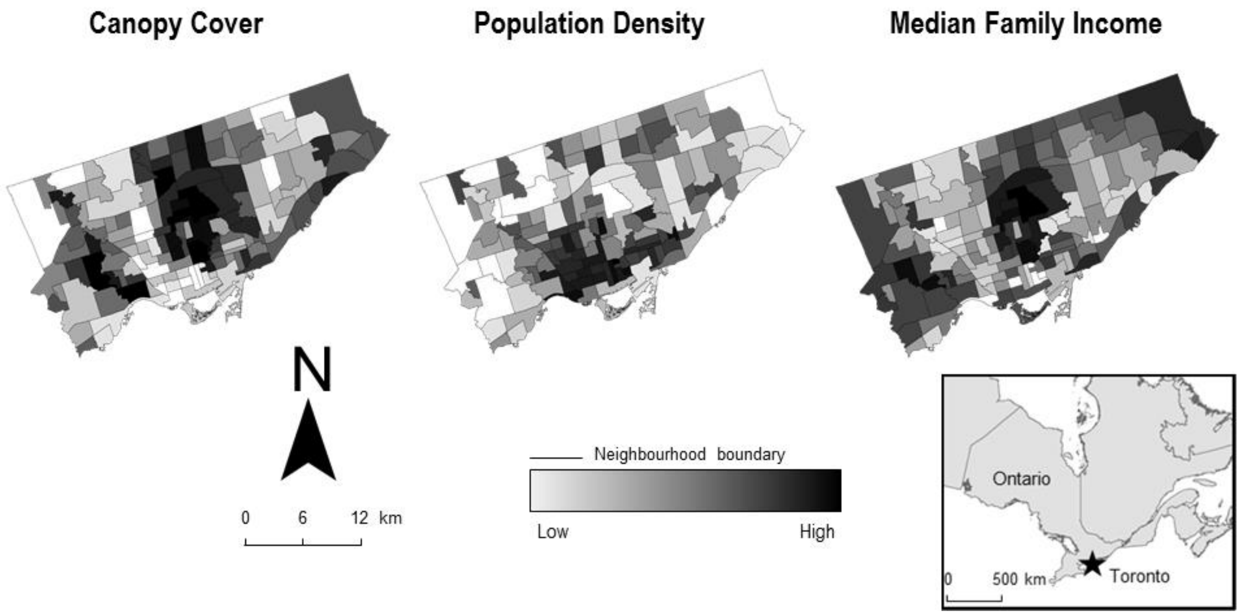

Fig. 2.2. The City of Toronto and its 140 neighbourhoods, showing the distribution of three selected variables from each of the biophysical landscape, built environment, and human population categories of ecosystem components.

Toronto covers an area of 635 km^2 with a mean elevation of 113 m above sea level. It has a continental climate with hot, humid summers and cold winters, though its climate is also influenced by its proximity to Lake Ontario (Roots et al., 1999). The mean annual precipitation is 834 mm, with 710 mm falling as rain (Environment Canada, 2008). Toronto has a mean July temperature of 22.2°C, a mean January temperature of -4.2°C, and a mean annual temperature of 9.2°C (Environment Canada, 2008). Its land area was originally forested before European settlement began in the early nineteenth century. Toronto lies in the Deciduous Forest Region, in the Mixedwood Plains Ecozone, which is typically dominated by sugar maple (*Acer saccharum*), red oak (*Quercus rubrum*), white pine (*Pinus strobus*), and eastern hemlock (*Tsuga canadensis*), and characterized by fertile soils and extensive waterways (Ontario Ministry of Natural Resources [OMNR], 2012). Dominant tree species in Toronto's current urban forest include Norway maple (*A. platanoides*), sugar maple, Manitoba maple (*A. negundo*), green ash (*Fraxinus pennsylvanica*), and white spruce (*Picea glauca*). Urban forest canopy cover is highly heterogeneous across city neighbourhoods, ranging from a minimum of 2.4% to a maximum of 61.9% (City of Toronto, 2010; Nowak et al., 2013a).

2.3.2. Data Sources and Preparation

A variety of spatially explicit, quantitative variables (Table 2.1) were used to measure the 12 urban forest ecosystem components described in Section 2.2. Selection of variables was informed by the literature and, where necessary, restricted by the availability of data for the study area. Percent canopy cover, grass/shrub cover, and percent stocking were measured using land cover data derived from 2007 QuickBird satellite imagery (0.6 m per side pixel resolution) and City of Toronto planimetric data (City of Toronto, 2010). Building footprints were also derived from these land cover data. While there is some associated error in this method given the potential for tree canopy to overhang lower buildings (Pham et al., 2013), these data were found to be more accurate than the building footprint polygons from municipal cadastral datasets.

A digital elevation model (DEM) with 10-m horizontal and 1-m vertical resolution (OMNR, 2006) was used to derive percent slope gradient, which was averaged at the neighbourhood scale, and topographic position index (TPI), which is a measure of the relative position (i.e., elevation) of a given DEM pixel to neighbouring pixels (Weiss, 2001). Percent depression (TPI < -1 standard deviation [SD]), level (-1 SD ≤ TPI ≤ 1 SD), and hilltop (TPI > 1 SD) slope positions were calculated in each neighbourhood following the methods described by Weiss (2001). Soils data were retrieved from georeferenced quaternary/surficial geology maps (OMNR, 1980), which provide a description of typical soil textures (i.e., sand, silt, clay) for each deposit class. While useful, these data are less preferable than soil survey data, which were not available for the City of Toronto. However, soil texture derived from surficial geology has been used previously in ecological landscape classification (e.g., Host et al., 1996) and was considered an acceptable approach for this study.

Table 2.1. Variables selected to measure the 12 ecosystem components in the urban forest ecosystem classification (UFEC) framework for its application in the City of Toronto.

Variable Code	Description
	Biophysical Landscape
	Vegetation
CNPY	Canopy cover (%)
GRSS	Grass/shrub cover (%)
STCK	Stocking (%)
	Topography
SP-1; SP2; SP3	Slope position (% depression, flat, and/or hilltop)
SLOP	Mean slope gradient (%)
	Soil
SL-1; SL-2; SL-3	Dominant soil texture type (% sand, silt, and/or clay)
	Built Environment
	Density
BLDG	Mean building site coverage (%)
POPD	Population density (population/km^2)
	Street Pattern
BLCK	Average block size (m^2)
	Land Use Mix
LU-1; LU-2; LU-3; LU-4	Residential, commercial, government and institutional, and/or resource and industrial (%)
	Housing[1]
SDET	Single-detached house (dwellings/10,000 people)
APT5	Apartment building with five or more storeys (dwellings/10,000 people)
PC-1; PC-2; PC-3	Period of construction before 1946, between 1946 and 1980, and/or between 1981 and 2006 (dwellings/10,000 people)
	Human Population
	Income
INCM	Median family income ($)
	Education
UNIV	Population with a university certificate, diploma, or degree (individuals/10,000 people)
	Home Ownership
OWNR	Percent of owner-occupied private dwellings (%)
	Ethnocultural Background
IMGT	Population with immigrant status (individuals/10,000 people)
MNTY[2]	Population that is a visible minority (individuals/10,000 people)

[1] Selection and aggregation of housing data that was found to be valid when examining patterns of urban vegetation by Conway and Hackworth (2007) and Conway and Bourne (2013).
[2] Statistics Canada defines visible minority as non-Aboriginal people whose race is non-Caucasian and/or whose colour is non-white.

Building site coverage, measured as the mean building footprint to parcel ratio in a given neighbourhood, was the variable selected to represent building intensity (Forsyth, 2003). Floor area to parcel ratio would have been a preferable metric, though again these data are seldom available for municipalities at a spatial scale useful to UFEC. Street pattern was assessed using mean block size, measured as the mean size of contiguous parcel blocks in a given neighbourhood (Krizek, 2003). Land use refers to observed and current use (i.e., not planned land use or zoning), and data were derived from DMTI Spatial Inc., and included residential, commercial, government and institutional, and resource and industrial classes.

All variables describing the human population, as well as population density and housing data, were derived from Statistics Canada 2006 census data. The 2006 census was used in place of the 2011 census given the 2007 vintage of the land cover data. Data were obtained within the Toronto census subdivision at the census tract (CT)-level and were aggregated to the neighbourhood scale. All neighbourhood boundaries correspond to those of existing CTs. Each neighbourhood is comprised of one to 10 CTs, and boundaries were delineated according to former and existing planning areas, service area boundaries, and natural and built features (e.g., rivers and roads). Each census variable was aggregated to the neighbourhood scale by summing total counts of all CTs within a neighbourhood, and standardized as variable counts per 10,000 people. The one exception was median family income, which was aggregated by weighted average, where median family income in each CT was weighted by total CT population.

2.3.3. Analysis and Classification

Cluster analysis refers to a collection of techniques for examining relationships among data and is used to classify similar objects (e.g., neighbourhoods) into mutually exclusive groupings with the purpose of maximizing within-group homogeneity and between-group heterogeneity (Jongman et al., 1995; Hair et al., 2010). The term heterogeneity in this context corresponds to distance in multivariate data space (e.g., Euclidean distance). Cluster analysis is a commonly used tool for classifying ecosystems (Host et al., 1996; McNab et al., 1999; Mora & Iverson, 2002; Hargrove & Hoffman, 2005; Womack & Carter, 2011) and for developing neighbourhood typologies (Reibel & Regelson, 2007; Mikelbank, 2011; Reibel, 2011). I used hierarchical cluster analysis to determine natural groupings in the 140 neighbourhoods based on the 27 input variables (Table 2.1) measuring the 12 ecosystem components in the UFEC framework (Fig. 2.1). Hierarchical cluster analysis was selected given the sample size and the hierarchical nature of ecological systems (Zhou et al., 2003). It is critical that the selection of input variables for cluster analysis have a strong theoretical grounding, as variable selection is highly influential on the final clusters produced (Hair et al., 2010). Moreover, the analysis will always produce clusters regardless of their theoretical validity (Hair et al., 2010). Therefore, a theoretical grounding based on current research on urban forest ecology and management was used to build the UFEC framework first, and then to guide variable selection for the Toronto analysis.

Cluster analysis used Ward's method, which is a hierarchical, agglomerative method that begins with *n* clusters comprised of individual objects that are successively grouped in such a way as to reduce overall variance (Jongman et al., 1995). Ward's method is recommended when continuous variables are used and is supported as valid for the application of cluster analysis in ecosystem classification research (Kent & Coker, 1995; Jongman et al., 1995; McNab et al., 1999). Given their different units of measurement, each variable was standardized prior to analysis using its maximum value, so that values ranged between 0 and 1 (Jongman et al., 1995). Because the Rouge neighbourhood in the northeast corner of Toronto is primarily comprised of a forested national park with some surrounding residential and agricultural land use, it was removed from the analysis as it represented a considerable municipal anomaly.

2.4. Results

In this UFEC application, a total of 11 clusters were identified in the cluster analysis, yielding a total of 12 urban forest ecosystem classes with the omitted Rouge neighbourhood designated as Class 12. Ecosystem classes ranged in area from 4 km^2 to 132 km^2, with an average area of 53 km^2, and consisted of three to 25 city neighbourhoods. The highest tree canopy cover was found in Classes 5 and 9, which were also the wealthiest and least ethnoculturally diverse. The lowest tree canopy cover and highest population and building density were found in Class 11, which covered the commercial and institutional centre of the city. Mapping of the 12 ecosystem classes (Fig. 2.3) showed that most classified neighbourhoods showed some spatial aggregation, especially those associated with the more-densely built urban core (e.g., Classes 6, 9, 10, and 11).

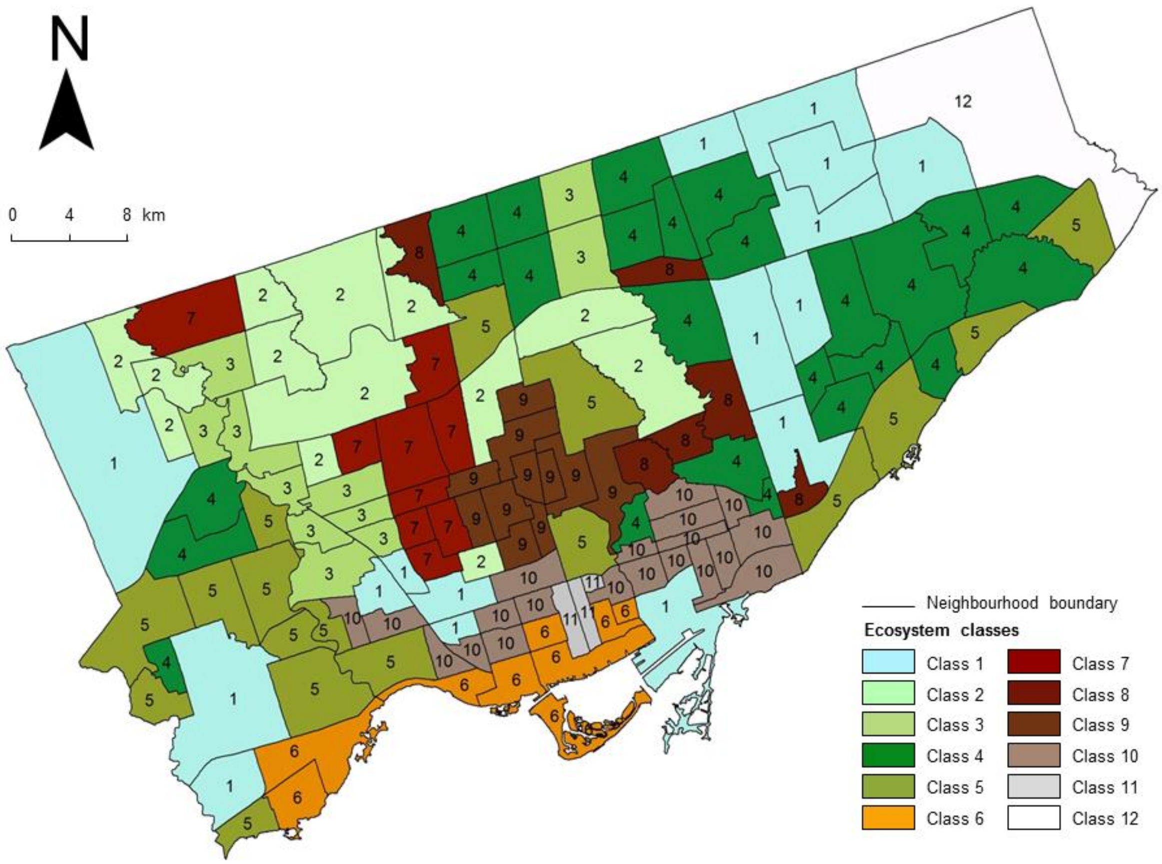

Fig. 2.3. Application of urban forest ecosystem classification (UFEC) to the City of Toronto.

The dendrogram (Fig. 2.4) illustrates the hierarchical and agglomerative grouping of Toronto neighbourhoods into clusters. The analysis initialized with each individual neighbourhood comprising a cluster and then continuously grouped the most similar neighbourhood until a single cluster was formed. Determining the appropriate grouping level to derive clusters and, as such, the final number of ecosystem classes is a heuristic process (Hair et al., 2010). I determined grouping level based on a natural break in the dendrogram (Zhou et al., 2003) and with the objective of maintaining a manageable number of meaningful ecosystem classes.

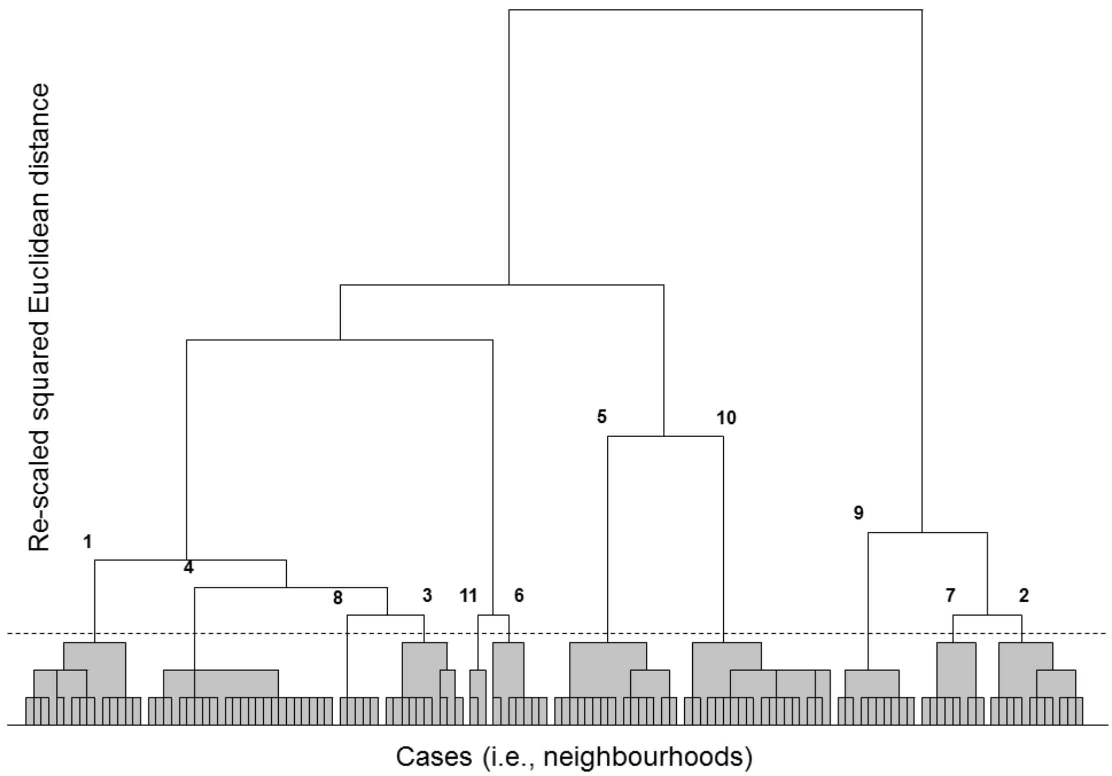

Fig. 2.4. Dendrogram showing the hierarchical clustering of Toronto neighbourhoods and the distance between clusters in multivariate data space (i.e., re-scaled squared Euclidean distance). The dashed line indicates the level at which the 11 clusters were identified.

Each ecosystem class was interpreted and named (Table 2.2) using synoptical tables summarizing variable values within each class (Appendix A). Several of the ecosystem classes were interpreted as variants of a single type and were paired in the naming convention. This was done to improve the organization and clarity of the UFEC where class conditions were complementary along one or more key social-ecological gradients. The Affluent and Forested Neighbourhoods (i.e., Classes 5 and 9) were assigned both a lower and higher density variant based on population density and building site coverage. The Mixed Residential Neighbourhoods (i.e., Classes 2 and 3) differ primarily in biophysical conditions and have a steep terrain variant. Lastly, the Typical Residential Neighbourhoods (i.e., Classes 4 and 10) were found to have both an inner variant with older housing and higher density and an outer variant with newer housing and lower density. These two classes represent 25% of Toronto's geographic area and 33% of its population. Industrial Parkland (i.e., Class 1) was the most extensive ecosystem, covering 20% of the study area.

Table 2.2. Urban forest ecosystem classes identified for the City of Toronto.

Class	Description
Class 1 (*n* = 16)	Industrial Parkland (132 km^2) — Extensive open green space, with low canopy cover and stocking — Level terrain and sand soils — Moderate building site coverage, population density, and block size — Mixed residential and industrial land uses — Newer, owner-occupied, detached housing — Moderate income with lower education and high ethnocultural diversity
Class 2 (*n* = 13)	Mixed Residential Neighbourhood (79 km^2) — Extensive open green space, with moderate canopy cover and stocking — Primarily level terrain with clay soils and infrequent steep and forested slopes — Moderate building site coverage and population density with larger block size — Primarily residential land uses — Mixed detached and apartment housing, with owners and renters — Moderate income and education with high ethnocultural diversity
Class 3 (*n* = 11)	Mixed Residential Neighbourhood – Steep Terrain (38 km^2) — Extensive open green space, with moderate canopy cover and stocking — Clay and sand soils with abundance of forested ravines and valleys — Moderate building site coverage and population density with larger block size — Primarily residential with some industrial/commercial land uses — Mixed detached and apartment housing, with owners and renters — Moderate income with high ethnocultural diversity and low education
Class 4 (*n* = 25)	Typical Residential Neighbourhood – Newer and Outer (122 km^2) — Extensive open green space, with moderate canopy cover and stocking — Level terrain and sand soils — Low building site coverage and moderate population density and block size — Primarily residential land uses — Mixed detached and apartment housing, with owners and renters — Moderate income and education with higher ethnocultural diversity
Class 5 (*n* = 17)	Affluent and Forested Neighbourhood – Lower Density (86 km^2) — Very high canopy cover and stocking — Steep and variable terrain with sand soils — Low building site coverage and population density with moderate block size — Primarily residential land uses — Owner-occupied, detached housing — High income and education with low ethnocultural diversity
Class 6 (*n* = 8)	Waterfront Hardscapes (27 km^2) — Very low vegetation cover and low stocking — Level terrain with silt soils and fill — High building site coverage and population density — Mixed residential and industrial land uses — Abundant apartment tower housing of varying periods of construction with low ownership — Low income with moderate education and ethnocultural diversity

Class	Description
Class 7 (*n* = 9)	High Density Residential Neighbourhood (32 km^2) — Low canopy cover and stocking with moderate open green space — Variable terrain and clay soils — Moderate building site coverage and population density with small block size — Primarily residential with some industrial/commercial land uses — Owner- and renter-occupied detached housing — Moderate income and ethnocultural diversity with low education
Class 8 (*n* = 6)	Towers in the Park (19 km^2) — Moderate canopy cover and high stocking — Primarily level terrain with sand soils and infrequent steep and forested slopes — moderate building site coverage and population density with very large block size — Mixed residential and industrial land uses — Primarily apartment tower housing and low ownership — Low income, moderate education, and very high ethnocultural diversity
Class 9 (*n* = 11)	Affluent and Forested Neighbourhood – Higher Density (25 km^2) — Very high canopy cover and stocking with low open green space — Variable terrain and clay soils — Moderate building site coverage with high population density and small block size — Primarily residential land uses — Mixed, older detached and apartment housing — High income and education with low ethnocultural diversity
Class 10 (*n* = 20)	Typical Residential Neighbourhood – Older and Inner (35 km^2) — Low canopy cover and open green space with high stocking — Variable terrain and sand soils — High building site coverage and population density with small block size — Primarily residential with some institutional land uses — Older, owner- and renter-occupied housing — Moderate income and education with lower ethnocultural diversity
Class 11 (*n* = 3)	The Downtown Core (4 km^2) — Very low canopy cover, open green space, and stocking — Primarily level terrain — Very high building site coverage and population density with small block size — Mixed land use — Primarily apartment tower housing and low ownership — Low income and high education and ethnocultural diversity
Class 12 (*n* = 1)	Peri-Urban Forest (38 km^2) — High canopy cover and open green space — Variable terrain with sand and clay soils — Moderate building site coverage with very low population density and very large block size — Primarily forested with some residential land uses — Newer, owner-occupied, detached housing — High income and ethnocultural diversity with low education

2.5. Discussion and Conclusions

The objective of ecosystem classification is to reduce the structural and functional complexity of ecosystems in models while quantifying key social and ecological processes involved in shaping current ecosystem conditions (Bailey, 2009; Cadenasso et al., 2013). In doing so, ecosystem classification can inform researchers, policy-makers, and practitioners by relating management needs and decision-making to the current conditions of the natural resource in a more holistic and spatially explicit fashion. Thus, the implementation of UFEC could be instrumental in shifting urban forestry towards a model of ecosystem-based management, where, in contrast, the historical focus has tended towards individual tree care or biophysical ecosystem components only (Konijnendijk et al., 2006).

The specific conditions of the ecosystem classes identified in this study can inform urban forest planning and management in a number of ways. From the simplest perspective, the UFEC can provide insight for urban greening, tree planting, and the maintenance of the existing canopy. Canopy cover targets are a common feature in strategic planning in urban forestry (Kenney et al., 2011), such as Toronto's own 40% target. However, given that spatial heterogeneity of canopy cover distribution is common across cities (Cadenasso et al., 2007), assessment and target designation at the ecosystem scale will provide managers with more useful information for greening initiatives. These locally-specific designations will also help provide the detail necessary to create realistic future planting targets. For example, a higher canopy target in Class 1, where open green space was abundant and stocking was low, is more attainable than in Class 11, where building intensity is much higher and limited planting space and opportunities for greening exist. Conversely, in older (i.e., housing age) neighbourhoods where canopy cover and stocking are high, such as Class 9, the maintenance of the existing, aging canopy would most likely become the management priority.

The distribution of canopy cover and associated economic wealth across the 12 Toronto UFEC classes yielded both expected and unexpected findings. Existing research emphasizes the positive spatial relationship between the distribution of canopy cover, and thus associated ecosystem services, and economic wealth (e.g., Grove et al., 2014). Our findings support this observation, as is evident in Classes 5, 9, and 12, which had the three highest canopy cover and income values, respectively. Similarly, low-income residents and minimal tree canopy cover found in Classes 6, 11, and, to a lesser degree, 1. The UFEC classes with wealthy residents and higher canopy cover also tended to be defined by older and detached housing, which is again consistent with existing studies (Boone et al., 2010; Conway & Bourne, 2013). Tree canopy cover was expected to be strongly associated with homeownership given its association with economic wealth (Troy et al., 2007). While this was found in some cases, Class 9 had the second highest canopy cover at 41.8% but a low ownership rate of just 49.1%. In contrast, Class 1, with a low canopy cover of 15.9%, had a higher ownership rate of 66.9%. Similar disparities between canopy cover, income, and education were also evident (i.e., Class 11). These findings suggest potential scale dependencies of some ecological processes, especially those of an anthropogenic origin, where established relationships may be stronger at the parcel scale than at the neighbourhood scale.

Topography, as characterized by both slope and slope position, had similar contrasting relationships with economic wealth and tree canopy cover. Steep terrain was associated with ecosystems with wealthier residents and high tree canopy cover (e.g., Classes 5 and 9) and ecosystems with less wealthy residents, apartment towers, and higher proportions of renters, immigrants, and minorities (e.g., Classes 3, 7, and 8). It was expected that ecosystems with steeper terrain would have higher canopy cover (Heynen & Lindsey, 2003). However, this contrasting socioeconomic pattern may be indicative of some broader socio-ecological trend not captured within the scope of this study. Conversely, it may also be a function of Toronto's extensive ravine system and its own unique development history and topography, which raises issues of UFEC generalizability.

Patterns in the distribution of open green space, contrasted with those of tree canopy cover, appeared to be, in part, a function of building intensity and imperviousness (Berland, 2012) with a greater abundance of open green space in the peripheral ecosystems of Toronto. Open green space, unlike canopy cover, can have a broad range of land uses (e.g., sports fields, institutional grounds, highway rights of way). This difference may explain the variable relationship with socioeconomic background of residents that is not seen with tree canopy cover. This latter point is also important to consider in urban forest planning, where not all open green space can be considered as available planting space (Wu et al., 2008). As expected, stocking also shows these same patterns, though inverted. However, stocking was often found to be low in ecosystems with a high proportion of industrial land uses (e.g., Classes 1 and 6), which highlights industrial properties as potential greening opportunities for municipalities.

Of importance to urban forestry is the relationship of population density and building intensity (collectively referred to hereafter as urban density) with canopy cover, and whether high urban density and the process of densification have negative implications for urban forest ecosystem service supply. Existing theories suggest that canopy cover is likely to decrease as urban density increases due to increases in housing, transportation networks, and other impervious urban features (Cook et al., 2012; Grove et al., 2014). There are certainly instances where this holds true, with Classes 5 and 11 at either extreme. In ecosystems characterized by high urban density, the maintenance and enhancement of public green spaces and parks may be a priority for urban forest management given the lack of space for greening initiatives on other properties. Moreover, these ecosystems (e.g., Class 11; the Downtown Core) will likely have higher tree planting and maintenance costs due to additional infrastructure needs and higher rates of tree mortality (Nowak et al., 2004; Jutras et al., 2010).

The present study's findings suggest that at the neighbourhood scale, there are instances where other ecosystem components, such as socioeconomic background and land use can be more influential on tree canopy cover than urban density. For example, Class 9 has relatively high urban density and high tree canopy cover. The wealthy residents that characterize this class may, in part, explain this phenomenon, given the existing research on the positive association of tree cover and wealth (e.g., Grove et al., 2014). As another example, Classes 1 and 2 both had moderate population density and building intensity values. However, Class 1 had a much lower tree canopy cover of 15.9% and stocking of 41.8% than Class 2, with 30.6% canopy cover and 54.5% stocking. The high proportion of industrial land uses and their associated abundance of open green space in Class 1 may explain this disparity. More empirical research is required to substantiate these patterns and the relationship between tree cover and urban density. Moreover, many of the established relationships between green space and urban form discussed in this paper are from research conducted in American cities.

In addition to aligning management needs and objectives with current ecosystem conditions, ecosystem classification may also be employed in the prioritization of management actions, or 'ecological triage' (Hobbs & Kristjanson, 2003; Hargrove & Hoffman, 2005). Restrictive municipal operating budgets for urban forest management are commonplace across North America (Kenney & Idziak, 2000). Identifying areas where ecosystem service supply is low and opportunities for enhancement exist will help to prioritize strategic urban forest planning initiatives. It is important, however, that this approach is not used to validate underfunding of urban forestry, where challenging and costly sites (e.g., Class 11 – the Downtown Core) are omitted from greening initiatives.

It is also valuable to prioritize limited municipal resources according to other management objectives, such as social equity in the distribution of ecosystem services (Pham et al., 2013). Lower-income neighbourhoods have been shown to be associated with less green space, degraded urban forest health, and lower canopy cover (Troy et al., 2007; Landry & Chakraborty, 2009; Pham et al., 2013). It has also been demonstrated that there are racial and ethnic inequalities in the distribution of urban canopy (Heynen et al., 2006; Landry & Chakraborty, 2009). These inequalities in access to ecosystem services translate into unequal access to the myriad of benefits that the urban forest supplies (Heynen et al., 2006). While Toronto does exhibit exceptions to these patterns (e.g., Class 8 – Towers in the Park), a negative relationship between canopy cover and ethnocultural diversity was observed. In contrast, neighbourhoods with residents having wealthier socioeconomic backgrounds have a greater capacity for self-organization and access to financial resources for urban forest stewardship activities (Grove et al., 2006a; Heynen et al., 2006); these neighbourhoods may be a lower management priority for limited municipal resources. The inclusion of socio-demographic data in the UFEC can aid in the consideration of social equity as far as urban tree canopy distribution is concerned.

Generalizability of the UFEC and the 12 ecosystem classes identified in this study will be important to address in future research and application to other cities. In this study, I adopted current urban ecological theory pertaining to the social and biophysical processes that shape urban forest ecosystems. However, social-ecological systems like cities, and urban forests within, are also legacies of their own unique conditions and histories (Grove, 2009). For example, the biophysical legacy of Toronto's ravine system and its influence on construction, the 1998 amalgamation, and other place-specific phenomena likely influenced the results of this study, most notably the derived ecosystem classes. Moreover, the established relationship of low income with high urban density is not as pronounced in Toronto as many other North American cities (Conway & Hackworth, 2007). Future UFEC research and applications will likely yield novel ecosystem classes specific to individual cities. However, the intent of the UFEC design is to provide a science-based tool that also allows for the inclusion of local knowledge in the identification and interpretation of ecosystem classes. Ecosystem classification is, in part, an inherently subjective process (Sokal, 1974) and it is hoped that this balance could be beneficial for the efficacy of the UFEC framework. Lastly, it is hoped that future UFEC research in other cities, countries, and climatic regions could help refine and advance the tool and strengthen the practice of ecosystem classification in cities.

CHAPTER 3 A CONCEPTUAL FRAMEWORK OF URBAN FOREST ECOSYSTEM VULNERABILITY

Abstract: The urban environment is quickly becoming the most common setting in which people worldwide will spend their lives. Consequently, urban trees and forests, and the ecosystem services they provide, are becoming a priority for municipalities. Approaches to quantifying and communicating the vulnerability of this resource are essential for maintaining a consistent and equitable supply of these ecosystem services. I propose a theory-based conceptual framework for the assessment of urban forest vulnerability that integrates the biophysical, built, and human components of urban forest ecosystems. A subsequent review and description of potential vulnerability indicators are provided. Urban forest vulnerability can be defined as the likelihood of decline in ecosystem service supply and its associated benefits for human populations, urban infrastructure, and biodiversity. It is comprised of: 1) exposure, which is the stressors and disturbances associated with the urban environment that negatively affect tree and ecosystem function, 2) sensitivity, which is determined by urban forest structure and dictates the system response to forcing from exposures and the corresponding magnitude of potential impacts, and 3) adaptive capacity, which is the social and environmental capacity of a system to shift or alter its state to reduce its vulnerability or accommodate a greater range in its ability to function while stressed. Potential impacts, or losses in ecosystem service supply, are temporal in nature and require backward-looking monitoring and/or forward-looking modelling, and can be communicated through the use of vulnerability indicators, aggregated indices, and mapping. A vulnerability approach can communicate complex issues to decision makers and stakeholders and advance the theoretical understanding of urban forest ecosystems.

Keywords: urban forest, vulnerability, social-ecological system, ecosystem service; indicator

3.1. Introduction

The urban environment is quickly becoming the most common setting in which people worldwide will spend their lives (United Nations, 2014). Urban areas are also growing in extent, as urbanization and urban expansion are occurring at a rate that succeeds human population growth (Alig et al., 2004). Maintaining and enhancing urban forest ecosystems are subsequently becoming a priority for municipalities (Clark et al., 1997; Kenney & Idziak, 2000). Moreover, the ecosystem services they provide to urban populations are now recognized as a vital component in the overall sustainability of cities (Grove, 2009; Duinker et al., 2015). These ecosystem services represent a diverse and substantial set of environmental, social, and economic benefits (Nowak & Dwyer, 2007). With this relevance of urban forest ecosystems to the majority of the global population, the qualification, quantification, and management of these ecosystem services are being increasingly researched and implemented.

However, the urban forest is a vulnerable resource. The dense human populations and the alteration and degradation of natural environments that characterize cities lead to harsh growing conditions, which make a difficult scenario for tree growth and forest establishment (Nowak et al., 2004; Trowbridge & Bassuk, 2004; Konijnendijk et al., 2005). Moreover, there is a tremendous diversity and conflict in how urban forests, and more broadly urban ecosystems, are defined, modelled, and managed (Konijnendijk et al., 2006). Many of the divergences fall within disciplinary divides (e.g., arboriculture, forestry, ecology, geography, urban planning) and are perhaps an externality of the interdisciplinary nature of urban forests (Steenberg et al., 2015a). Vulnerability science can provide a framework for integrating key intellectual contributions from these various disciplines while investigating the sustainability of ecosystem service supply from urban forest ecosystems.

Forests in general are vulnerable to environmental change and altered disturbance regimes because the longevity and stationary nature of trees restrict or inhibit necessary adaptations to rapid change (Nitschke & Innes, 2008; Lindner et al., 2010). Urban forests suffer additional vulnerability due to their setting in constantly changing, heterogeneous, and stressed urban environments that are exceedingly different from the environments in which most tree species have evolved (Alberti et al., 2003; Cadenasso et al., 2013). Much of the discourse on urban forests and trees in the city is centered on the effects of various stressors and disturbances on individual trees, with a prominent focus on street trees (e.g., Jutras et al., 2010, Roman &

Scatena, 2011; Koeser et al., 2013). There is a considerable knowledge gap on their combined effects and interaction with urban forest ecosystem structure, inclusive of the built environment and human population, and the associated implications for ecosystem service supply. Consequently, there is a need to synthesize this existing body of research on urban forest stressors and disturbances in the broader context of entire system structure and function.

In this paper, I propose a theory-based conceptual framework for the assessment of urban forest vulnerability that integrates the biophysical, built, and human components of urban forest ecosystems. A review of relevant bodies of literature and subsequent identification of potential vulnerability indicators are provided. The applicability of vulnerability science for complex social-ecological systems and its capacity to shift research away from an impacts-only perspective make it a suitable approach for investigating the urban forest resource. With the complex nature of urban forest ecosystems, integrative approaches and tools for identifying potential losses in function or undesirable changes in structure can be highly valuable for guiding urban forest planning and management and can act as decision-support models. Such approaches to quantifying and communicating vulnerability are essential for maintaining a consistent and equitable supply of ecosystem services.

3.2. Vulnerability in Social-Ecological Systems

Vulnerability analysis and assessment are an increasingly used concept and method for approaching issues of system sustainability and ecosystem service supply (Turner et al., 2003a; Schröter et al., 2005; Adger, 2006; Eakin & Luers, 2006; Lindner et al., 2010). The concept of vulnerability has a long history within a diversity of disciplines and as a structured approach to research it has appeared in the literature at varying levels of formality and complexity. Today, it has evolved into an integrative and effective tool for exploring issues of sustainability in social-ecological systems (Turner et al., 2003a; Eakin & Luers, 2006; Cumming, 2014). Vulnerability can be defined in simple terms as "…the degree to which a system, subsystem, or system component is likely to experience harm due to exposure to a hazard, either a perturbation or a stress/stressor" (Turner et al., 2003a, p. 8074). There remains variability in terminology, concepts, and methodological approaches arising from different lineages. However, these divergences tend to be dependent on the research objectives of a given study (Eakin & Luers, 2006). The important similarity is that vulnerability science shifts research away from just stressors and impacts towards a holistic view of the entire system (Luers et al., 2003; Adger et al., 2004).

The early roots of vulnerability research characterized it either as a lack of entitlement or as vulnerability to natural hazards, as described in the review by Adger (2006). The entitlements approach focused on social aspects of vulnerability, looking at variability in population characteristics that lacked access (i.e., entitlement) to natural resources or ecosystem services due to drought, disease, war, or other disasters (Sen, 1984). While concepts from this background merged with modern definitions of vulnerability, they also diverged into separate areas looking at poverty and often overlooked biophysical processes. The hazard-based approaches were rooted more in the physical sciences and were focused on risk, and examined environmental hazards as well as society's potential for loss (Burton et al., 1993; Eakin & Luers, 2006). However, political ecologists argued that the hazard paradigm disregarded social elements, and did not address why certain marginalized populations were more vulnerable (Cutter, 1996). Certainly, these definitions were not independent of each other, and issues around natural hazards and underlying social vulnerabilities were bridged early on (Blaikie et al., 1994; Adger, 2006). More recently, there has been a growing consensus on conceptual approaches to vulnerability research that have converged within the arena of global environmental change and advancements in sustainability science (Luers et al., 2003; Turner et al., 2003a; Metzger et al., 2006; 2008; Lindner et al., 2010). Vulnerability assessment has since become a core component of several international, collaborative environmental change investigations, including the Intergovernmental Panel on Climate Change (IPCC) assessment reports and the Millennium Ecosystem Assessments.

Latent definitions of vulnerability identify it as an element of social-ecological systems that is an outcome of multiple and interacting social and biophysical properties across spatial and temporal scales (Metzger et al., 2006). Turner et al. (2003a) proposed one of the more widely accepted conceptual frameworks for understanding the vulnerability of social-ecological systems. They argue that the vulnerability of a system is comprised of exposure, sensitivity, and resilience/adaptive capacity. Exposure refers to the magnitude, frequency, duration, and spatial extent of stressors and disturbances that affect a system (Burton et al., 1993). Sensitivity is the relative level of response by a system to stressors or disturbances, and is determined by intrinsic characteristics of the system itself (Turner et al., 2003a). Adaptive capacity is the capacity for a system to shift or alter its state to reduce its vulnerability or accommodate a greater range in its ability to function while stressed (Smit & Wandel, 2006).

Some studies investigating system vulnerability to environmental change make distinctions between adaptive capacity and resilience (Adger et al., 2004; Adger, 2006), while others appear to simply substitute resilience with adaptive capacity (Luers et al., 2003). Adger (2006) and Miller et al. (2010) speak to the compatibility and indeed commonality between resilience and adaptation, though others caution against the unclear and incompatible use of vulnerability, adaptive capacity, and resilience terminology (Gallopin, 2006). Gallopin (2006) suggests that resilience and adaptive capacity are indeed subsets of the overall coping capacity of a system in the vulnerability context. Resilience has recently been gaining popularity as an approach to understanding urban social-ecological systems (Miller et al., 2010), which will be discussed further in Section 3.5. However, most recent studies investigating system vulnerability to environmental change, including ecosystem service vulnerability, adopt the adaptive capacity terminology (Schröter et al., 2005; Metzger et al., 2006; 2008; Lindner et al., 2010; Ordóñez & Duinker, 2014). The Turner et al. (2003a) framework of vulnerability, and similar derivatives, has been successfully applied to a variety of social-ecological systems in the context of global change, including agricultural systems (Luers et al., 2003), Arctic populations and resource extraction (Turner et al., 2003b), and forests and ecosystem service supply (Metzger, et al., 2006; 2008; Lindner et al., 2010). In this paper, I adapt and expand this framework for application in urban forest ecosystems.

3.3. Urban Forest Vulnerability Framework

I define urban forest vulnerability as the likelihood of decline in ecosystem service supply and its associated benefits for human populations, urban infrastructure, and biodiversity. The framework is comprised of exposure, sensitivity, adaptive capacity. The temporal dimension of the framework refers to urban forest ecosystem service supply, where potential impacts are an outcome of vulnerability and are described as losses or undesirable changes in ecosystem service supply. The definition and conceptual design of the framework are derived from research investigating vulnerability to global environmental change in social-ecological systems. Where this study diverges from those addressing global change is that the stressors and disturbances of interest are not climatic variables, but rather those associated with densely-settled urban environments. However, the underlying concern is the decline or loss of system function in response to persistent and/or sudden change (Schröter et al., 2005; Metzger, et al., 2006; 2008; Lindner et al., 2010), where environmental change refers to urban development, alterations to the built environment, and social processes of cities.

The development of a conceptual framework of vulnerability is an important first step prior to the identification of specific indicators for measurement (Adger et al., 2004). The framework of urban forest vulnerability developed for this study (Fig. 3.1) builds on the widely used conceptual approach introduced by Turner et al. (2003a). However, the framework also incorporates elements of the framework from the Advanced Terrestrial Ecosystem Analysis and Modelling (ATEAM) research and Millennium Ecosystem Assessment (e.g., Schröter et al., 2005), which was applied by Metzger et al. (2006; 2008) and Lindner et al. (2010) to investigate the vulnerability of ecosystem services to environmental change. Lastly, the framework incorporates novel elements of vulnerability unique to urban forest ecosystems, according to existing research on the stressors and disturbances of city trees and social-ecological determinants of urban forest ecosystem structure and function.

Deductive approaches to vulnerability assessment involve indicator identification according to existing theory using a defined conceptual framework (Füssel, 2010). A deductive approach is useful for complex social-ecological systems with multiple variables of concern at different spatial and temporal scales (Hinkel, 2011). Conversely, observed data and data-driven inductive approaches tend to be more repeatable and objective. However, they cannot reveal all vulnerability issues, especially long-term variability and risk. Most comprehensive studies on system vulnerability employ elements of both approaches, though it is argued that beginning with a defined framework is valuable (Füssel, 2010). A combination of empirical field data describing urban forest structure and surrounding environmental conditions (Chapter Four), along with a theory-based conceptual framework and ecological modelling (Chapter Five) can offer complementary insight into urban forest ecosystem vulnerability. The following sections describe a conceptual framework of urban forest vulnerability and potential indicators for subsequent empirical assessments and analyses.

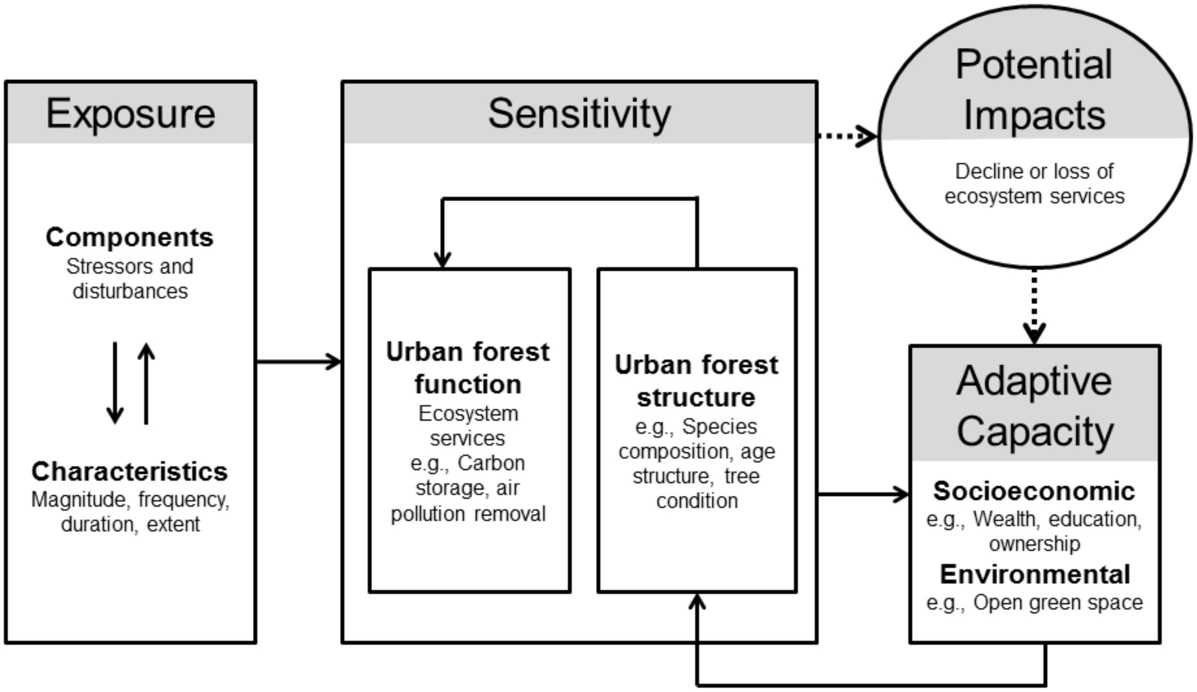

Fig. 3.1. Conceptual framework urban forest ecosystem vulnerability.

3.3.1. Exposure

Exposure refers to the types, magnitude, frequency, duration, and extent of stressors and disturbances that negatively affect system functioning (Burton et al., 1993; Turner et al., 2003a). I define exposure in the context of urban forest vulnerability as the stressors and disturbances associated with the urban environment that negatively affect tree and ecosystem health and/or cause tree mortality, thereby reducing or halting the supply of ecosystem services (Table 3.1). A great deal of the stress on urban trees can be associated with infrastructure and the built environment (Trowbridge & Bassuk, 2004). The geometry and density of buildings and other urban structures affects the irradiation (i.e., sunlight available for photosynthesis and plant growth) and the microclimate of urban areas, which can negatively affect tree growth in heavily built-up areas (Jutras et al., 2010). Moreover, the extent of impervious surfaces (e.g., concrete and asphalt) restricts the land area available for urban forest establishment (Tratalos et al., 2007). Tree proximity to, and potential conflict with, infrastructure can also be an indirect source of stress due to management practices associated with removing conflicts (Trowbridge & Bassuk, 2004). Land use is frequently used as a surrogate for the various elements of urban morphology and the built environment that affect urban forest structure and function, as it has been shown to be influential on both tree mortality and ecosystem structure (Nowak et al., 2004). There are also social exposures associated with both land use intensity and land management practices that cause intentional and unintentional physical damage to trees (Lu et al., 2010).

Pollution and environmental contaminants negatively affect tree and urban forest health. Despite the amelioration of urban air pollution by trees (Nowak & Dwyer, 2007), tree physiology is simultaneously degraded by airborne pollutants. For example, tropospheric or ground-level ozone reduces plant photosynthetic rates and hinders biomass accumulation (Sitch et al., 2007). The chemical properties of urban soils are also commonly altered to varying degrees in cities. Soil contamination with heavy metals and de-icing salts, low nutrient availability due to leaf-litter removal, and altered pH levels are all common urban stressors of trees (Craul, 1992; Zimmerman et al., 2005). However, the relationship between urban forest health and urban soils is far more complex. Soil degradation and loss is a frequent scenario in urban areas do to rapid development and poor practices like grading and topsoil removal (Craul, 1999; Millward et al., 2011). Soils are vital for sustaining urban trees, as they provide the rooting medium and essential water and nutrients for above-ground growth (Craul, 1992; Craul, 1999). Moreover, physical soil properties are often negatively affected by urbanization due to the loss of soil structure caused by compaction and surface sealing (Craul, 1992; Craul, 1999). The loss of soil structure can result in restricted root growth and degraded water infiltration, hindering overall tree health and growth (Hanks & Lewandowski, 2003). Insufficient soil volumes to sustain proper root growth are also a common occurrence in land uses with an abundance of development and impervious surfaces (Trowbridge & Bassuk, 2004). However, quantifying and predicting the influence of soil quality and quantity on urban forest ecosystems are difficult, as soil conditions are heterogeneous and highly spatially variable.

The primary biological threats to urban trees are from insects and pathogens (Konijnendijk et al., 2005; Laćan & McBride, 2008). Both urban and hinterland forests are subject to insects and pathogens. However, trees that are stressed, as many are in the urban environment, are more susceptible to infestation and decline (Armstrong & Ives, 1995). Moreover, urban areas are frequently subject to invasive forest pests and diseases that have been introduced as a result of global trade and the warming climate (Dukes et al., 2009). A well-known example that decimated urban tree populations is the Dutch elm disease, and more recently the emerald ash borer, which is currently afflicting ash populations in Canada and the

United States (Herms & McCullough, 2014). Importantly, exposure is scale dependent, ranging from site-scale environmental degradation (e.g., soil compaction, construction activity, and proximity to infrastructure; Koeser et al., 2013) to ecosystem-scale stress from the combined effects of density and land use (Konijnendijk et al., 2005).

Table 3.1. Potential indicators of urban forest exposure (see Appendix B for indicator sources).

Category	Indicator	Description	Source
Built environment	Land use	Land uses have variable intensities of use, population densities, and building intensities, and are a broad-scale indicator of environmental quality and of potential social stressors. Commercial, industrial, utility, and transportation land uses tend to have lower canopy cover and higher mortality Residential and institutional land uses tend to have higher canopy cover and lower mortality rates Parks, cemeteries, and other green spaces typically represent the most forested areas within cities	1, 2, 3, 4, 5, 6
	Population density	The density of people in a geographic unit is a broad-scale indicator of environmental quality and the potential for social stressors on trees as densities increase	6, 7, 8, 9, 10, 11
	Light availability	Low light availability limits photosynthetic activity and plant growth	2, 4, 12, 13, 14
	Building intensity	Building intensity refers to the density and relative size of buildings in an area and is a broad-scale indicator of growing space, light availability, and microclimate	6, 7, 8, 10, 11, 13, 15
	Building height	The height of surrounding buildings influences light availability and microclimate	6, 9, 10, 13
	Building type	Building type is a finer-scale metric than land use and indicates available growing space, land use intensity, and overall environmental quality	3, 6, 7, 9, 10, 13
	Conflict with infrastructure	Conflicts with infrastructure, especially overhead utility wires, frequently lead to excessive pruning and premature tree removals	2, 6, 12, 13
	Distance from nearest building	Trees with shorter distances from buildings tend to have less growing space and more conflicts with infrastructure	2, 6, 13
	Distance from street	Trees with shorter distances from streets tend to have a higher exposure to pedestrian and vehicular traffic and pollution associated with roadways (e.g., de-icing salts)	2, 6, 13, 16
	Imperviousness	Impervious surfaces limit the availability of space for tree establishment and growth, restrict water infiltration into soils, and increase urban temperatures	3, 6, 7, 12, 13
	Site size	Site size can restrict both above- and below-ground tree growth and is often an indicator of future conflicts with infrastructure	3, 6, 12, 17
	Site type	The type of site where trees are established is influential on its overall level of exposure to social and physical stressors (e.g., higher exposure in sidewalk tree pits versus wide grass medians)	2, 3, 6, 12, 16

Category	Indicator	Description	Source
	Street width	Wider streets are indicative of higher stress from the built environment, especially vehicular traffic and associated pollutants	2, 6, 16
Biological stressors	Signs of infestation	Trees often have signs (e.g., leaf wilting, exit holes in bark) when infested with insects and pathogens, which can frequently be identified and differentiated in the field	12, 13, 18, 19, 20, 21, 22
	Presence of pest/disease	Insects and pathogens that are identified can be used to estimate future risk for trees and adjacent areas, based on known forest composition and structure	12, 13, 18, 19, 20, 21, 22
Social stressors	Construction	Construction activities frequently damage trees and soils, especially root systems during excavations	12, 17, 23, 24
	Pollution	Pollution is a common occurrence in urban environments, including emission-related air pollution, acid rain, and soil and surface water contamination, and is a source of stress for trees	12, 13, 14, 25, 26
	Poor management	Poor management can physically damage trees (e.g., improper pruning) and affect their future growth and longevity (e.g., species selection and planting location)	3, 12, 13, 14, 27
	Vandalism	Vandalism (e.g., torn limbs) includes physical damage to trees, which is especially common among young street trees	3, 6, 27
	Vehicular/ pedestrian traffic	High levels of traffic are associated with greater stress on urban trees, such as soil compaction and vandalism associated pedestrian traffic and air pollutants and de-icing salts associated with vehicular traffic	2, 3, 12
Soils	Compaction	Loss of soil structure due to compaction and surface sealing can result in restricted root growth and degraded water infiltration	2, 3, 4, 12, 13, 14, 28, 29, 30
	Contamination	Soil contamination from polluted runoff and de-icing salts alters soil pH and adversely affects plant growth	2, 4, 12, 13, 14, 25, 28, 29, 30
	Nutrients/ organic matter	Low nutrient availability and organic matter content can result from leaf-litter removal and soil alterations, which adversely affects plant growth	2, 4, 12, 13, 14, 25, 27, 28, 29, 30
	Volume	Insufficient soil volumes restrict proper root growth and limit tree size at maturity	12, 13, 14, 28, 29, 30
Climate	Temperature	Variable urban microclimates and heat islands stress and damage urban trees; global climate warming and increasing freeze-thaw events adversely influence tree health	12, 13, 14, 31, 32
	Precipitation	Both drought events and excessive precipitation adversely affect tree health and cause mortality, especially among newly-established trees	12, 13, 14, 31, 32
	Storm events	Severe storm events can cause broken limbs and windthrow, with structural damage possible both above and below the ground	12, 13, 14, 33, 34, 35, 36

3.3.2. Sensitivity

Sensitivity is the relative system response to forcing from a given stressor or disturbance in the urban environment and determines the magnitude of potential impacts (i.e., loss of ecosystem services) in response to exposure (Turner et al., 2003a). I define urban forest sensitivity as being determined by forest structure, inclusive of species composition, age structure, and tree condition (Table 3.2). Ecosystem, species, and genetic diversity are key determinants of forest and urban forest sensitivity to insects and pathogens (Laćan & McBride, 2008). Furthermore, trees in poor condition that are already under stress are more susceptible to insects and pathogens (Armstrong & Ives, 1995). While urban forests tend to have higher species richness than pre-settlement forests, there is frequently poor spatial distribution of species diversity and a tendency for single-species dominance in localized pockets (McBride & Jacobs, 1979). This spatial aggregation of tree species and localized lack of diversity will influence sensitivity and potentially lead to highly localized impacts. Tree species are also highly variable in their tolerance to urban conditions and many are commonly found in poor condition due to their planting/establishment location (Trowbridge & Bassuk, 2004).

Table 3.2. Potential indicators of urban forest sensitivity (see Appendix B for indicator sources).

Category	Indicator	Description	Source
Structure	Diameter at breast height	Smaller, newly-established trees have higher rates of mortality; Larger, mature trees are frequently in poor condition and sensitive to storm damage	2, 3, 4, 6, 13, 16, 17, 33, 35, 37
	Structural diversity	Even-aged, immature urban forests are sensitive to higher mortality rates; Even-aged, overmature urban forests are sensitive to widespread senescence, age-related decline, storm disturbance, and mortality	12, 13, 18, 20, 33, 34, 35, 37
Composition	Species	Tree species have variable sensitivities to urban conditions (e.g., air pollution, de-icing salts, restricted growing space; microclimate effects)	2, 3, 4, 6, 12, 13, 17, 33, 35, 37
	Species diversity	Low species diversity, especially in localized pockets, increases sensitivity to species-, genus-, and family-specific pests and other stressors	12, 13, 18, 20, 33, 35, 37
Condition	Tree condition	Trees in poor condition are more sensitive to other stressors and disturbances and have higher rates of decline and mortality	3 ,6, 13, 17, 33, 34, 35, 37, 38

Urban forest age and structural diversity are also an important component of sensitivity, as an abundance of overmature trees can result in widespread tree senescence and mortality in a short time period. Older trees and even-aged forests are also more susceptible to storm damage and windthrow (Mitchell, 1995; Lopes et al., 2009). Conversely, younger and newly-planted urban trees are sensitive and have far higher associated mortality rates (Roman & Scatena, 2011). Since tree age measurement is time consuming and damaging to the tree (e.g., increment borers), stem diameter measurements are frequently used as a proxy to assess urban forest age structure. Arguably, ecosystem-scale urban forest sensitivity to various urban stressors and disturbances is an understudied phenomenon.

3.3.3. Adaptive Capacity

The adaptive capacity of a social-ecological system is determined by both inherit environmental and social components (Lindner et al., 2010). I define the social dimension of adaptive capacity within urban forest ecosystems as economic wealth, education, and the likelihood to engage in urban forest stewardship activities (Table 3.3). Populations with a greater access to resources, a greater capacity to self-organize, and a higher level of education will have greater adaptive capacity (Grove et al., 2006b; Manzo & Perkins, 2006; Boone et al., 2010; Pham et al., 2013). Neighbourhoods with higher levels of wealth, homeownership, education will therefore likely have a greater capacity to maintain, improve, and prevent decline in the supply of urban forest ecosystem services (Martin et al., 2004; Grove et al., 2006b; Troy et al., 2007). Additionally, neighbourhoods with residents associations, community groups, business improvement areas, and other social structures that are aware of, and active in, urban forest stewardship are more likely to engage in stewardship and lobby municipal governments to enhance their urban forest (Martin et al., 2004; Manzo & Perkins, 2006). Policies and institutions also influence urban forest structure and function through regulation, incentive programs, and public education and outreach designed to protect and/or enhance trees and green spaces (Conway & Urbani, 2007).

I describe environmental adaptive capacity as a function of tree canopy cover, open green space, and continuous forested area. Existing tree canopy cover characterizes the existing level of ecosystem services and therefore a greater potential of maintaining desirable levels of ecosystem service supply through active management (Troy et al., 2007; Nowak & Greenfield, 2012; Pham et al., 2013). The total area of open green space that is available for new tree establishment, either by planting or natural regeneration, is indicative of the capacity for greening initiatives and increasing ecosystem service supply (Troy et al., 2007). However, ecosystem service supply in urban forests is disproportionately attributed to the extent of continuous forest ecosystems located within a city's parks and undeveloped land (Nowak & Greenfield, 2012). Moreover, where natural regeneration is possible, the maintenance and enhancement of ecosystem service supply without management intervention (i.e., tree planting) may be possible (Nowak, 2012; Nowak & Greenfield, 2012). Therefore, the area of continuous forest cover can be seen as an influential component of environmental adaptive capacity.

Table 3.3. Potential indicators of urban forest adaptive capacity (see Appendix B for indicator sources).

Category	Indicator	Description	Source
Social	Income	More affluent Individuals have more resources to invest in stewardship activities; Income is positively correlated with urban forest amenities across cities	6, 8, 9, 10, 11, 39, 40, 41
	Housing value	Housing value is often indicative of affluence, but also of property size and available space for tree establishment and growth	8, 11, 41
	Homeownership	Homeowners have direct legal control over the landscaping and management practices on their properties	6, 8, 9, 10, 11, 40, 41
	Education	Higher education is associated with affluence and engagement in stewardship activities, and is positively correlated with urban forest amenities across cities	6, 8, 10, 41
	Stewardship	Local organizations, residents associations, and households that engage in stewardship activities contribute to the maintenance and enhancement of urban trees and forests	3, 4, 8, 13, 39, 41
	Municipal policies	Municipal tree protection and conservation by-laws, strategic and operational management plans, and public education and outreach contribute to the maintenance and enhancement of the urban forest resource	12, 13, 42, 43
Environmental	Open green space	The amount of open green space is indicative of the total area available for tree establishment and urban greening initiatives	6, 7, 8, 9, 10, 11, 13, 40
	Existing tree canopy	Extensive tree canopy cover is indicative of an intact urban forest and higher levels of ecosystem service supply	6, 7, 8, 9, 10, 11, 13, 39, 40
	Forested area	Continuous and naturalized forested areas have high levels of ecosystem service supply and require fewer management interventions	1, 7, 13, 30, 44

3.4. Assessing and Analyzing Vulnerability

Quantitative, indicator-based vulnerability assessment frameworks such as the one described in this study are one such approach to assessment that have been used at multiple scales and in multiple regions to explore potential threats to ecosystem service supply in social-ecological systems (Luers et al., 2003; Turner et al., 2003b; Schröter et al., 2005; Metzger et al., 2006; 2008; Lindner et al., 2010). Indicator selection and design for urban forest vulnerability assessment will be scale, context, and place dependent (Adger et al., 2004; Birkmann, 2007; Hinkel, 2011). Correspondingly, while Section 3.3 provides a comprehensive, though certainly incomplete, review and selection of possible indicators of urban forest exposure, sensitivity, and adaptive capacity, future applications of these indicators may be limited by place and relevance, but also by the availability of data and feasibility of measurement.

Our intent was to target readily available data sources and established tools and models, where possible. The suggested adaptive capacity indicators are all targeted towards national census data and satellite-derived land cover data for social and environmental adaptive capacity, respectively. Conversely, sensitivity indicators are relatively dependent on field data and the availability of tree inventories. Because of the numerous, cumulative, and interactive nature of the stressors and disturbances associated with urban forest exposure, the data needs and measurement feasibility of exposure indicators will likely be the greatest challenge for assessment and analysis. Several exposure indicators can utilize the previously mentioned and widely available data sources, such as land cover data (e.g., imperviousness) and census data (e.g., housing density). However, careful consideration of specific indicator selection and design, data needs, and spatial scale of assessment should be undertaken prior to the application of this urban forest vulnerability framework.

Vulnerability is a temporal phenomenon (Adger, 2006), and in the case of urban forest vulnerability, it is dependent on the supply of ecosystem services over time. Potential impacts are the final component described in the conceptual framework of vulnerability (Fig. 3.1). They are defined as declines or undesirable and destabilizing changes in ecosystem service supply resulting from exposure to external forcing and internal system sensitivity (Lindner et al., 2010). Since potential impacts are a temporal function of assessment and analysis, and thus an outcome of vulnerability, they require either forward-looking ecological modelling (e.g., Chapter Five) or backward-looking monitoring (e.g., Chapter Four) for quantification. Drawing from established models and tools for managers, such as the i-Tree suite of models, can offer one prospective method for approaching spatial and temporal variability in ecosystem service supply.

The i-Tree models were developed by the United States Department of Agriculture (USDA) Forest Service to estimate the structure and condition of the urban forest, as well as several key ecosystem services and their financial value, based on locally-collected field data and spatial data (USDA Forest Service, 2013a). Specifically, the i-Tree Eco model, formerly known as the Urban Forest Effects (UFORE) model, is well established in urban forestry and has been used globally by hundreds of communities (Nowak & Crane, 2000). Consequently, these models can provide some insight into the temporal nature of ecosystem service supply and overall vulnerability. This can be done both through the establishment of permanent sample plots and monitoring of ecosystem change. This can also be done through the use of the newly developed i-Tree Forecast model, which simulates future changes in urban forest structure and function based on initial forest conditions and user-defined mortality and establishment rates (Nowak et al., 2014). Moreover, i-Tree data can be used to assess most urban forest sensitivity indicators and several exposure indicators (e.g., pest detection). However, there are many other established indicator-based models and measurement protocols that can be employed in vulnerability research depending on local availability and study design (e.g., Clark et al., 1997; Kenney & Puric-Mladenovic, 2001; Dobbs et al., 2011; Kenney et al., 2011).

The overarching purpose of a vulnerability approach in urban forestry is to communicate complex issues to practitioners, policy makers, and communities in accessible ways. In addition to analyzing individual vulnerability indicators to explore the sources and internal structure of vulnerability in a given study area, some form of indicator aggregation is commonly used (Adger et al., 2004). Indicator aggregation can range from standardization and simple linear combination to more complex methods using fuzzy logic or even expert-derived weights (Tran et al., 2002; Eakin & Luers, 2006; Birkmann, 2007). However, caution should be taken around the loss of transparency and validity with excessive aggregation and the assumptions involved (Adger et al., 2004; Hinkel, 2011). There are arguments both for and against aggregation that will be discussed in Section 3.5.

Lastly, mapping especially has been shown to be an effective means for communicating vulnerability (O'Brien et al., 2004; Eakin & Luers, 2006). This might entail the mapping of individual indicators or overall aggregated indices of vulnerability and its core components (i.e., exposure, sensitivity, adaptive capacity). Moreover, with ecosystem service monitoring and modelling, the mapping of vulnerability outcomes (e.g., potential impacts) is also feasible (Metzger et al., 2006). The growing availability and accessibility of data and increasing sophistication of geographic information systems (GIS) and tools for spatial analysis have increased the possibility for the spatial communication of ecosystem vulnerability (Eakin & Luers, 2006).

3.5. Discussion and Conclusions

A prominent focus in municipal urban forest policy and strategic management in North America is on urban forest ecosystem services and their associated benefits (Ordóñez & Duinker, 2013; Steenberg et al., 2013). Arguably, there is less attention on potential threats to urban forest ecosystems, and little discussion of overall system vulnerability. There are some exceptions, such as recent attention to major invasive pests like the emerald ash borer (Herms & McCullough, 2014). From a research perspective, there are many studies on ecological disturbance and stressors of urban forests, especially street trees (e.g., Jutras et al., 2010; Hauer et al., 2011; Koeser et al., 2013). Laćan and McBride (2008) created a vulnerability model for urban forests pests. More recently, Ordóñez and Duinker (2014) investigated the vulnerability of urban forests to climate change. However, attention to urban forest threats in policy and management fails to adequately integrate them with the built environment and social processes. Moreover, there is a lack of attention to the spatial heterogeneity of urban forests and their ecosystem services (Berland, 2012; Cadenasso et al., 2013). A vulnerability approach can provide such a methodology to examine the baseline socioeconomic and biophysical conditions of urban forest ecosystems and the potential for loss while moving away from an impacts-only and/or benefits-only discourse.

The assessment and analysis of vulnerability can also shed light on longer-term processes and unexpected, multi-faceted relationships between ecosystem service supply and vulnerability (Metzger et al., 2006). Exceptions to an assumed negative relationship between vulnerability and ecosystem service supply may exist, depending on the spatial and temporal scales of assessment. For instance, residential neighbourhoods with older housing and higher affluence are frequently characterized by large, mature trees and correspondingly higher levels of ecosystem service supply (Zipperer et al., 1997; Boone et al., 2010). Despite this adaptive capacity, widespread pest-related decline and mortality are still possible where species diversity is low (Laćan & McBride, 2008). Moreover, widespread senescence and age-related mortality is a likely scenario in these older neighbourhoods (Kenney et al., 2011; Steenberg et al., 2013). Newly-constructed suburban housing developments often have higher affluence and an abundance of open green space where tree establishment is possible (Chapter Two), and thus high social and environmental adaptive capacity. However, as new development typically involves land clearing, trees may be absent or only recently established (Puric-Mladenovic et al., 2000), presenting a scenario of low vulnerability and low levels of ecosystem service supply. These latter examples not only stress the internal variability and complexity of vulnerability, but also the importance of temporal dynamics and the potential threat of time-lag effects in forest ecosystems associated with disturbance and environmental change.

Vulnerability is one of a large number of theoretical frameworks in the body of research on urban social-ecological systems (Grove, 2009; Cumming, 2014). Yet, while there is an increasing need for frameworks to understand and predict the outcomes of intervention through management and policy in these systems, there is a lack of consensus on which are the most effective (Cumming, 2014). The sustainability approach was commonly used in urban planning. Early conceptions of sustainability in urban planning, despite theoretical advancements in ecology, saw sustainability as an achievable and persistent state for cities (Ahern, 2011). Resilience theory has since become more prominent and has begun to both replace and supplement this mode of sustainability (Ahern, 2011). Moreover, resilience is a commonly used term and framework for researching urban social-ecological systems (Miller et al., 2010). Resilience is a system's ability to recover from a disturbance and change back to a reference state and/or to maintain that reference state or states while stressed, and has a longer tradition in the natural sciences, particularly in ecology (Turner et al., 2003a).

However, both vulnerability and resilience are fundamentally concerned with the response of complex systems to change and arguably some of their biggest differences are in their disciplinary backgrounds and lexicons (Miller et al., 2010). Importantly, more recent vulnerability research in the arena of sustainability science integrates resilience concepts into a broader definition and conceptual framework of vulnerability. The framework developed by Turner et al. (2003a) and used in this paper employs the concept of resilience to describe the attributes and processes that have since been termed adaptive capacity in more recent applications (Metzger et al., 2006; 2008). Arguably, a vulnerability approach to addressing change in social-ecological systems therefore provides a broader and more holistic system picture by explicitly addressing the causes/types of change and not just the system's response to them.

There are also several challenges and limitations associated with vulnerability assessment and analysis. Vulnerability is an abstract concept that cannot be measured directly (Turner et al., 2003a). Consequently, vulnerability assessment and analysis are nearly always limited by a lack of metrics and available data (Luers et al., 2003). However, for the sake of sustainable management and the amelioration of the negative consequences associated with vulnerable systems, it is necessary to operationalize the concept in some way (Eakin & Luers, 2006). Since it is essentially impossible to characterize the entirety of a system in a research or management context, systems must be generalized through assumption, abstraction, and aggregation (Jørgensen & Bendoricchio, 2001; Turner et al., 2003a).

This latter necessity of the omission and reduction of information brings with it several critiques of vulnerability assessment and how its findings can be used. A prominent critique pertains to the use of vulnerability indicators and aggregated indices (Adger et al., 2004; Hinkel, 2011). Indicators and indices are the primary way in which vulnerability is communicated to policy makers and in which the effectiveness of policy measures are monitored (Hinkel, 2011). However, there is often confusion and even overstatement on what vulnerability indicators can do and a lack of transparency in how they are developed and applied (Eriksen & Kelley, 2007). Whether indicators are deductive and based on existing theoretical knowledge, inductive and based on measured observable phenomena, or some combination of these latter two, documentation and full transparency on their selection and application is vital for communicating vulnerability (Eriksen & Kelly, 2007; Füssel, 2010; Hinkel, 2011). Vulnerability indicators are valuable tools for reducing complexity to inform policy, but the spatial, temporal, and analytical scale of reduction must also be weighed (Hinkel, 2011). For example, the knowledge omission in reducing a broad-scale and complex phenomenon like global climate change to a single indicator in order to determine international resource allocation policies would most likely be ineffective if not unjust and lack transparency. Ultimately, scientifically valid and transparent indicators are one set of tools for urban forestry that can be used to operationalize complex phenomena like vulnerability in order to inform policy, but they cannot and should not remove all subjectivity and complexity from the decision-making process.

Urban forest ecosystems and their management are now prominent both as a topic of research and as a source of beneficial ecosystem services for citizens, municipal governments, and biodiversity. There is a need for comprehensive frameworks for understanding and assessing potential threats and losses in urban forests. Vulnerability assessment in urban forests can not only identify risk but also address social equity in the distribution of this public amenity (Boone, 2010; Dunn, 2010). From a municipal planning and management perspective, neighbourhoods with inequalities in the access to urban forest ecosystem services could be prioritized to build adaptive capacity and thereby ensure equitable access to environmental amenities (Heynen et al., 2006). It will be important, however, in future interdisciplinary vulnerability research and assessments in urban forests to include social perspectives, both in the methodologies and definitions used. Quantitative, indicator-based frameworks have the benefits of measurability, comparability, and generalizability. However, qualitative approaches, such as narrative and scenario analysis, public engagement, and participatory research, can be used to approach the more subtle, subjective, and perceived nature of urban forest vulnerability (Cutter, 2003). For instance, cultural values and benefits from urban trees and forests are both difficult to measure and variable in nature (Konijnendijk, 2008). Ultimately, the two most important functions of vulnerability frameworks are to communicate complex issues to decision makers and stakeholders and to advance the theoretical understanding around the biophysical, built, and social dimensions of urban forest ecosystems.

CHAPTER 4 PUBLIC TREE VULNERABILITY AND ANALYSIS OF ECOLOGICAL CHANGE IN A RESIDENTIAL URBAN FOREST

Abstract: The urban forest is a valuable ecosystem service provider, though cities are frequently-degraded environments, with a myriad of stressors and disturbances affecting trees. Vulnerability science is increasingly used to explore issues of sustainability and ecosystem service supply in complex social-ecological systems, and can be a useful approach for researching urban forest decline. The purpose of this study is to identify and explore drivers of urban forest vulnerability and their influence on ecosystem change. A series of quantitative indicators of exposure, sensitivity, and adaptive capacity that describe the built environment and associated stressors, urban forest structure, and a neighbourhood's human population, respectively, were assessed for 806 public trees in a residential neighbourhood in Toronto, Canada. Ecosystem change was characterized by tree mortality, condition, diameter growth, and planting rates, which were derived from an existing 2007/2008 inventory. Linear and logistic regression approaches were used to explore the relative influence and predictive capacity of vulnerability indicators on ecosystem change at the individual-tree and street-section scale. Mortality models showed high prediction accuracy and several significant explanatory variables, most notably smaller trees and poor tree condition, as well as proximity to commercial buildings and other indicators of heavily built-up environments. Tree condition was similarly influenced by these latter stressors, but in contrast to mortality declined notably with tree size and environments associated with larger trees. Diameter growth models were weak, though tree condition, species, and size were all significant predictors. Tree planting rates were not as influenced by socioeconomic indicators as was expected, although they were positively related to homeownership. Understanding the causes of urban forest change and decline are essential for developing planning strategies to reduce long-term system vulnerability.

Keywords: condition, growth, i-Tree Eco, mortality, Neighbour*woods*, tree planting

4.1. Introduction

The urban forest is a valuable ecosystem service provider and represents essential green infrastructure for many cities. However, cities are highly-altered, densely-settled, and frequently-degraded environments, with a myriad of stressors and disturbances that create difficult conditions for tree establishment and growth (Nowak et al., 2004; Trowbridge & Bassuk, 2004). Consequently, urban trees are often in poor condition (Koeser et al., 2013) and frequently have reduced longevity (Roman & Scatena, 2011), both of which translate to a reduction in ecosystem services (Nowak & Dwyer, 2007). Cases and causes of urban forest decline need to be identified, assessed, and modelled. Such research can inform the processes of urban design and policy development, as well as urban forest management, so that unnecessary tree decline and mortality are avoided and the benefits that urban inhabitants receive from trees are maximized.

The built environment is a source of stress for urban trees, especially in higher-density neighbourhoods. Building density, height, and type affect the irradiation (i.e., sunlight available for photosynthesis and plant growth), the physical growing space for trees, and the microclimate of urban areas (Jutras et al., 2010). Moreover, construction activities and conflicts with above- and below-ground utilities, and other grey infrastructure, are common sources of urban tree decline and mortality (Hauer et al., 1994; Randrup et al., 2001). Land use is highly influential on urban forest ecosystems, and is indeed indicative of the presence of many of these stressors. Land uses with higher populations and building densities, as well as abundant impervious surfaces with more vehicular and pedestrian traffic (e.g., commercial land uses), have higher rates of tree mortality and urban forest decline (Nowak et al., 2004; Lu et al., 2010). Soil pollution, compaction, and loss are frequent scenarios in urban areas, which can restrict root growth, limit water infiltration, and/or hinder tree health and above-ground growth (Craul, 1999). Cities are also characterized by high rates of commercial trade, exposing urban trees and forests to invasive insects and pathogens (Laćan & McBride, 2008), such as the emerald ash borer (*Agrilus planipennis*; EAB) and Asian longhorned beetle (*Anoplophora glabripennis*; ALB). These stressors and disturbances can be interactive and cumulative, and their ultimate effect of individual trees and urban forest ecosystems is dependent on tree condition, species, age, and overall species and structural diversity.

The influences of the human population and socioeconomic variability on urban forest structure and function are complex, dynamic, and uncertain. There are a number of social stressors, ranging from vandalism and poor management practices affecting individual trees (Lu et al., 2010; Koeser et al., 2013) to citywide issues of urban forest policy and governance affecting the maintenance of the entire urban forest resource (Conway & Urbani, 2007). Furthermore, there is a growing body of research that has investigated the influence of the socioeconomic characteristics of residents and their association with urban forest condition as well as the spatial distribution of city trees and their provision of benefits (Heynen & Lindsey, 2003; Grove et al., 2006b; Troy et al., 2007; Pham et al., 2013; Shakeel & Conway, 2014). This research points to strong positive relationships between resident affluence and urban tree cover, where higher levels of resident income, education, and homeownership are spatially associated with urban tree cover. Moreover, several studies highlight direct relationships of these resident socioeconomic attributes with participation in urban forest stewardship activities (Conway et al., 2011; Greene et al., 2011).

Research investigating rates and causes of tree mortality and urban forest decline is an important resource for urban forest practitioners. Moreover, the disciplines of ecology, urban planning, and geography continue to explore the dynamics of urban forest ecosystem change and its relationship with human populations. However, there is a considerable knowledge gap on the combined effects of these stressors and their interaction with urban forest structure. Moreover, there is little research investigating the effects of socioeconomic variability on urban forest ecosystem decline. Vulnerability science can offer a useful theoretical approach for addressing these gaps in a research context and for bridging the potential contributions of different disciplines that investigate urban forests and their benefits.

The term and concept of vulnerability has a number of definitions and is used in research in variety of contexts at different levels of formality and complexity (Adger, 2006). Current vulnerability science in social-ecological systems is widely held to be a useful approach for exploring issues of sustainability and environmental change in both theoretical and applied research (Turner et al., 2003a; Adger et al., 2004; Füssel, 2010). It was used in the development of an urban forest vulnerability framework (Chapter Three), where vulnerability is defined as the likelihood of decline in urban forest ecosystem service supply in response to stress, and is comprised of exposure, sensitivity, and adaptive capacity.

Exposure refers to the magnitude, frequency, duration, and spatial extent of stressors and disturbances that affect a system (Burton et al., 1993; Adger, 2006). These are the external causes of tree decline and mortality associated with the urban environment. Sensitivity is the relative level of response by a system to stressors or disturbances, and is determined by intrinsic characteristics of the system itself (Turner et al., 2003a). This is the internal structure of urban tree communities, such as species, size/age, condition, and diversity. Adaptive capacity is the capacity for a system to shift or alter its state to reduce its vulnerability or accommodate a greater range in its ability to function while stressed (Adger et al., 2004). This refers to associated human populations and their behaviours regarding urban forest stewardship, as well as the environmental capacity for urban forest enhancement (e.g., tree planting space). By shifting research focus away from external agents of stress and disturbance only, vulnerability analysis allows for more a comprehensive and integrative mechanism for modelling urban forest structure, function, and change.

The purpose of this study is to explore the processes of urban forest vulnerability and their influence on ecosystem change. Specifically, the conceptual framework of urban forest vulnerability (Chapter Three) is used to assess 2014 data describing 806 public trees in a residential neighbourhood in Toronto, Canada. The framework consists of a series of quantitative indicators of exposure, sensitivity, and adaptive capacity that describe the built environment and associated stressors, urban forest structure, and the neighbourhood's human population, respectively. Ecosystem change is characterized by tree mortality, condition, diameter growth, and planting rates, which were measured using comparisons with existing tree inventory data collected in 2007 and 2008. Regression analysis is used to explore the relative influence and predictive capacity of different indicators of vulnerability on ecological change at the individual-tree and street-section scale. With much of the global population increasingly living in cities and urbanization rates on the rise (United Nations, 2014), on-going research and science-based tools for understanding the causes of urban forest change and decline are essential for developing planning strategies to reduce long-term system vulnerability.

4.2. Methods

4.2.1. Study Area

The study was conducted in a centrally located residential neighbourhood, Harbord Village, in Toronto, Canada. As of 2011, Harbord Village had a population of 8,583, population density of 13,484 persons/km^2, and total area of 0.6 km^2, and is predominately comprised of semi-detached residential dwellings, with approximately 1,600 households (Keller, 2007; Statistics Canada, 2012). There are commercial land uses along main street sections, with several larger multi-unit and institutional parcels, and three small public parks. Urban forest researchers and Harbord Village residents conducted a tree inventory in 2007 and 2008 in order to inform their strategic urban forest management plan (Keller, 2007). Dominant tree species in the neighbourhood include *Acer platanoides*, *Fraxinus pennsylvanica*, *Gleditsia triacanthos*, *Thuja occidentalis*, *Acer saccharinum*, and *Aesculus hippocastanum*. In-grown *Morus alba*, *Ailanthus altissima*, and *Acer negundo* are also common. Toronto has a continental climate with hot, humid summers and cold winters. The city is within the Deciduous Forest Region and Mixedwood Plains Ecozone (Ontario Ministry of Natural Resources [OMNR], 2012).

4.2.2. Data Collection and Processing

Data collection took place during the growing season of 2014. A total of 806 public street trees and public trees in front-yard rights of way, parks, and schools were inventoried and matched with data from the existing 2007/2008 tree inventory. Residential back-yard trees were omitted from the study due to access constraints. In addition to the standard tree inventory metrics of species, diameter at breast height (DBH), and location (i.e., geographic location and civic address), a series of indicators of urban forest vulnerability were assessed for each tree. The design of the urban forest vulnerability assessment framework and selection of indicators is described in Chapter Three. Specific indicator selection and design were further refined according to the study's scale of assessment (i.e., individual trees and street sections), data availability, and feasibility. Ecosystem service supply of the measured trees was quantified using the i-Tree Eco model. Formerly known as the Urban Forest Effects (UFORE) model, i-Tree Eco is the flagship model of the i-Tree software suite that synthesizes a large body of research to quantify several key ecosystem services and their value based on the structure of the urban forest (Nowak & Crane, 2000).

Indicators in the vulnerability assessment framework are assigned to the sub-categories of exposure, sensitivity, or adaptive capacity, depending on the assumed relationship of the phenomena being measured with urban forest structure and function. Exposure indicators (Table 4.1) represent external stressors and disturbances that cause tree decline and mortality, and subsequently a decline in ecosystem service supply. While some of the exposure indicators represent direct stressors (e.g., vandalism), most characterize indirect relationships between stress and the surrounding environment, all of which have been previously identified as important causes and correlates of tree decline and/or mortality (Nowak et al., 1990; Hauer et al., 1994; Randrup et al., 2001; Nowak et al., 2004; Trowbridge & Bassuk, 2004; Jutras et al., 2010; Lu et al., 2010; Lawrence et al., 2012; Koeser et al., 2013). The main data source for exposure indicators was field data collected during this study. Additionally, 2011 census data were used to measure population density and a combination of 2013 orthorectified aerial photography and 2013 City of Toronto property map data were used to measure built area intensity (assessed as building site coverage; the ratio of building footprint to parcel area; Forsyth, 2003), distances to nearest buildings, and widths of streets. The latter two exposure indicators were not calculated in the field for both access and safety reasons. The binary exposure indicators resulting from the presence/absence of conflicts with infrastructure (Kenny & Puric-Mladenovic, 2001), vandalism, and poor management were measured in the field.

Table 4.1. Description of urban forest vulnerability indicators used to assess system exposure.

Indicator Description	Vulnerability Assumption	Mean/Count* (Standard Deviation)
Population density (persons/km^2)	Positive	14,834 (±8,146)
Built area intensity (%)	Positive	50.2 (±21.2)
Land use[1] (categorical)	-	-
Site type (categorical)	-	-
Site size (m^2)	Negative	136.7 (±383.4)
Type of nearest building (categorical)	-	-
Height of nearest building (storeys)	Negative	4.1 (±4.5)
Distance to nearest building (m)	Negative	6.7 (±14.2)
Distance to street (m)	Negative	4.1 (±3.0)
Width of sidewalk (m)	Positive	2.7 (±1.9)
Width of street (m)	Positive	11.2 (±6.7)
Impervious cover (%)	Positive	47.3 (±32.1)
Light availability[2] (ordinal rank; 0-5)	Negative	2.7 (±1.1)
Conflict of overhead utilities (0/1)	Positive	416*
Conflict with sidewalk (0/1)	Positive	76*
Conflict with buildings (0/1)	Positive	259*
Conflict with building foundation (0/1)	Positive	47*
Conflict with other infrastructure (0/1)	Positive	294*
Poor management (0/1)	Positive	172*
Vandalism (0/1)	Positive	92*

[1] Land-use designation is based on categories described in the i-Tree Eco manual. Land uses present in Harbord Village include commercial/industrial, institutional, multi-unit residential, park, residential, and vacant.
[2] Light availability was measured using crown light exposure, which is a component of the i-Tree Eco measurement protocol.

Sensitivity indicators (Table 4.2) represent the internal structure of the system, in this case the tree communities measured in the study, and its relative response to exposures. In other words, they are elements of urban forest structure that increase or decrease the likelihood of tree decline and mortality in response to stress. Species and DBH class were included to account for potential variation in the vulnerability of tree species and sizes (i.e., ages). A number of studies have found that mortality rates fluctuate by species and are elevated in younger and newly-planted urban trees (e.g., Nowak et al., 1990; Roman & Scatena, 2011). Tree condition is another predictor of urban tree mortality (Koeser et al., 2013) and is itself an indicator of sensitivity to stress (Trowbridge & Bassuk, 2004).

In this study, I derive tree condition using an aggregated index calculated from data collected as part of the Neighbour*woods* assessment protocol. This aggregate index has a maximum value of 1.0 indicating the extremely poor tree condition. Neighbour*woods* is a tool for community-based urban forest stewardship that was developed by Kenney and Puric-Mladenovic (2001), which describes a standardized procedure for community members to inventory and monitor the location, composition, and condition of their urban trees. The protocol describes 15 ordinal metrics of tree condition, ranging from 0 (best condition) to 3 (worst condition), as well as one presence/absence metric (i.e., 0 = no conks present, 1 = conks present). The latter metric was omitted since only one conk was observed in the study area, giving a total possible score of 45, which was then standardized to produce the condition index. A Neighbour*w*oods assessment was conducted during the 2007/2008 Harbord Village tree inventory and was again conducted for all trees measured in 2014. The tree condition index was calculated for both 2007/2008 and 2014 data.

Species and structural diversity were also measured at the street-section scale using the Shannon-Wiener index, using both species data and DBH data in 5-cm classes (Staudhammer & LeMay, 2001). Diversity indices are commonly used in forestry and forest ecology as a measure of ecosystem condition (Staudhammer & LeMay, 2001) and are important metrics of urban forest sensitivity to a variety of species- and age-specific stressors and disturbances (Laćan & McBride, 2008; Lopes et al., 2009). All sensitivity indicators were measured using field data.

Table 4.2. Urban forest vulnerability indicators used to assess system sensitivity.

Indicator Description	Vulnerability Assumption	Mean/Count* (Standard Deviation)
Species (categorical)	-	-
DBH class (categorical)	-	-
Tree condition index (Neighbour*woods* protocol)	Positive	0.30 (±0.17)
Species diversity (Shannon-Wiener index; tree species)	Negative	1.42 (±0.75)
Structural diversity (Shannon-Wiener index; DBH)	Negative	1.60 (±0.65)
In-grown tree (0/1)	Positive	41*

Adaptive capacity indicators (Table 4.3) represent components of the urban forest that enable it to reduce its own vulnerability or increase its capacity to tolerate greater change without adverse effects (Adger, 2006). In the context of this study, indicators of adaptive capacity measure socioeconomic variables that are likely to decrease the likelihood of a decline in ecosystem service supply or environmental ones that are likely to increase supply. All social adaptive capacity indicators were measured using 2011 National Household Survey data at the dissemination-area level, excluding presence/absence indicators that were assessed in the field. The environmental adaptive capacity indicators were measured using 2007 land cover data derived from QuickBird satellite imagery with 0.6-m resolution, quantified at the parcel scale (City of Toronto, 2010).

Table 4.3. Urban forest vulnerability indicators used to assess social and environmental adaptive capacity of the system.

Indicator Description	Vulnerability Assumption	Mean/Count* (Standard Deviation)
Social adaptive capacity	-	-
Median family income ($)	Negative	54,194 (±11,676)
Average dwelling value ($)	Negative	734,451 (±152,682)
Homeownership (%)	Negative	44.0 (±14.8)
Population with a university degree (individuals/10,000 people)	Negative	4,313 (±1,130)
Signs of stewardship[1] (0/1)	Negative	162*
Replanted site[2] (0/1)	Negative	19*
Environmental adaptive capacity	-	-
Open green space (%)	Negative	16.7 (±13.4)
Existing canopy cover (%)	Negative	18.0 (±20.3)

[1] Signs of stewardship include direct and obvious actions taken to protect trees or enhance growth (e.g., mulch, bicycle guards, pest protection; Lu et al., 2010).
[2] This indicator measured locations where tree mortality had occurred and the site had been replanted since the 2007/2008 inventory.

4.2.3. Analysis

4.2.3.1. Ecosystem Change

Four metrics of urban forest ecosystem change were assessed by comparing field data collected for this study in 2014 with the existing 2007/2008 tree inventory. Tree mortality was measured as presence/absence at the individual-tree scale using matched tree comparisons. Annual mortality rates (Eq. 4.1) were measured at the street-section scale with the equation used by Nowak et al. (2004) and adapted by Lawrence et al. (2012):

$$m = 1 - (N_1/N_0)^{1/t} \qquad \text{(Eq. 4.1)}$$

Where m is the annual mortality rate (%), N_0 is the number of living trees at the time of the first inventory, N_1 is the number of living trees at the time of the second inventory, and t is the number of years between inventories. Diameter growth rates (cm yr^{-1}) were measured by dividing the difference in DBH between matched trees by the time interval between inventories. Tree planting rates (trees ha^{-1} yr^{-1}) were measured at the street-section scale. The fourth and final ecosystem change variable was the Neighbour*woods*-derived 2007/2008 and 2014 tree condition indices. However, change in tree condition between inventories was not measured due to the likelihood of assessment subjectivity among different researchers collecting data at the two time instances.

4.2.3.2. Vulnerability Analysis

Regression analysis was used to evaluate the predictive capacity and explanatory power of the vulnerability indicators on urban forest ecosystem change in Harbord Village. Tree mortality, condition, growth, and planting were used as dependent response variables in separate regression models, using the vulnerability indicators as independent, predictor variables. Given the large number of independent variables, backwards regression was used to eliminate non-contributing variables and improve model parsimony (Hair et al., 2010). Backward elimination has been used successfully in parcel-scale urban forest research that is of an exploratory nature (Shakeel & Conway, 2014), and was accordingly selected for the vulnerability assessment conducted herein.

Multiple linear regression analysis was conducted at two spatial scales: individual trees (n = 806) and street sections (n = 56). For analysis at the street-section scale, the means of continuous variables and percentage presence of binary variables were calculated. All categorical variables were omitted except for land use, which was re-assessed to classify each street section as a single land use. The tree planting rate dependent variable was only modelled at the street-section scale and tree condition was omitted as an independent variable in models predicting condition. Binary logistic regression was used in place of linear regression to predict mortality at the individual-tree scale (live tree = 0, dead tree = 1). The 2007/2008 tree condition index was used in place of the 2014 index in the mortality models, since condition can only be assessed on living trees.

The site size, height of nearest building, distance to nearest building, distance to street, width of street, width of sidewalk, and tree planting rate variables were log transformed to meet normality assumptions for regression analysis. Tolerance values indicated no issues with multicollinearity at the street-section scale (i.e., tolerance values above 0.1; Hair et al., 2010), while the land use dummy variables showed some multicollinearity at the individual-tree scale, which was to be expected. A total of seven final regression models were constructed to predict and analyze the four ecosystem change variables at two spatial scales.

4.3. Results

4.3.1. Ecosystem Change

The change in size-class distribution between the 2007/2008 and 2014 inventories (Fig. 4.1) showed a decline in the presence of smaller (i.e., younger) trees and an increase in medium and larger (i.e., older) trees, while the largest size class remained relatively stable. This shift towards a larger, older urban forest ecosystem is also reflected in size-class mortality rates (Table 4.4), where higher mortality rates are seen in the smaller size classes. When planted trees are incorporated, the total number of trees in the > 0 cm to 10 cm DBH size class in 2014 exceeded the 2007/2008 inventory.

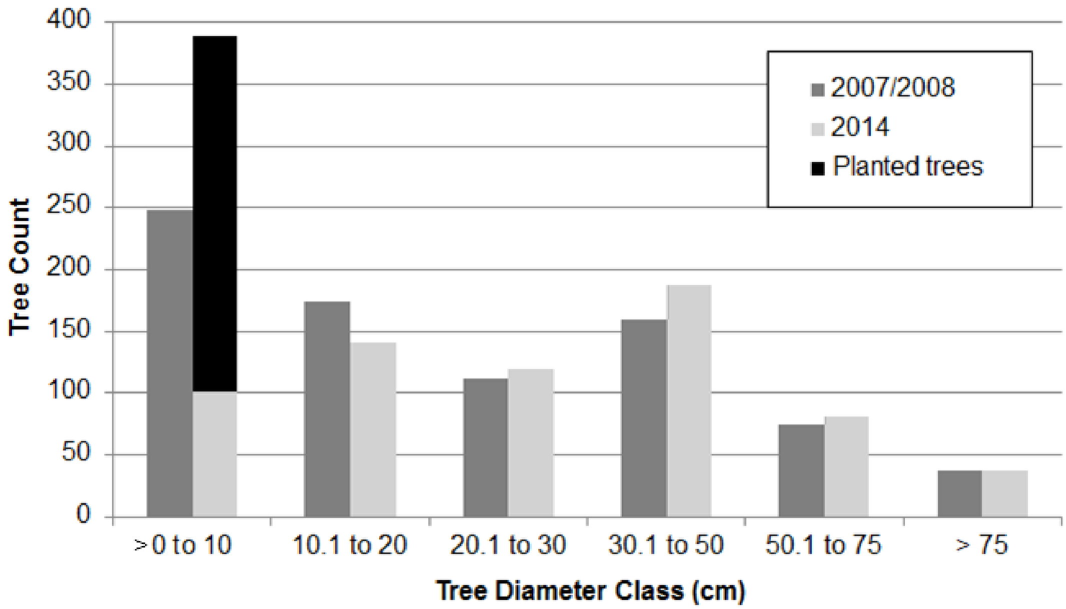

Fig. 4.1. Change in size class distribution of measured trees between the 2007/2008 and 2014 inventories.

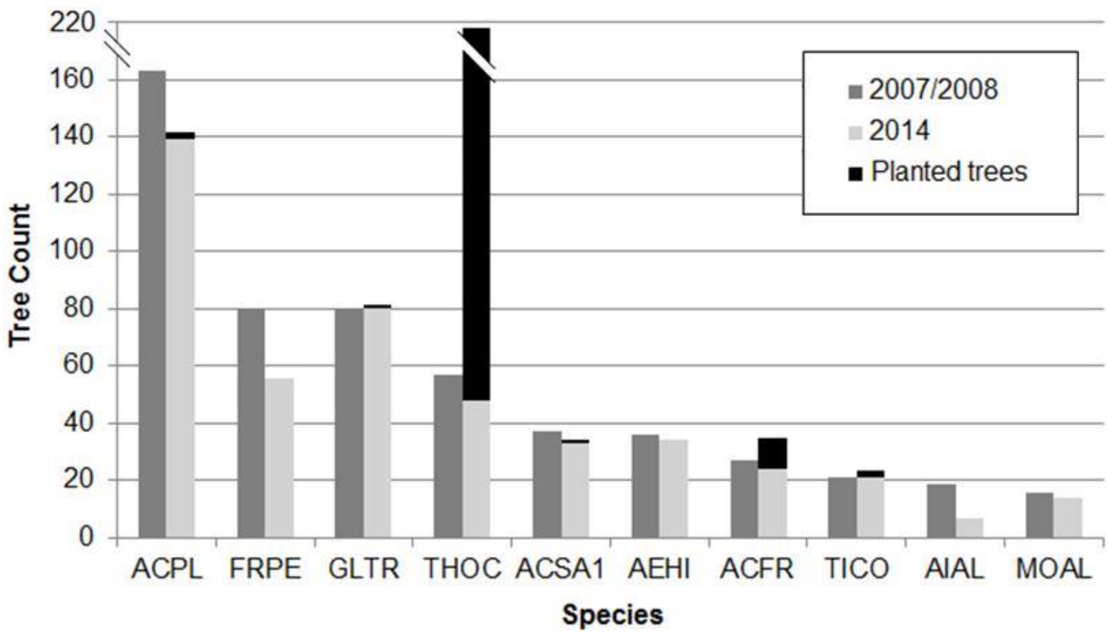

Fig. 4.2. Change in tree species distribution of the 10 most abundant species measured between the 2007/2008 and 2014 inventories. ACPL: *Acer platanoides*; FRPE: *Fraxinus pennsylvanica*; GLTR: *Gleditsia triacanthos*; THOC: *Thuja occidentalis*; ASCA1: *Acer saccharinum*; AEHI: *Aesculus hippocastanum*; ACFR: *Acer x freemanii*; TICO: *Tilia cordata*; AIAL: *Ailanthus altissima*; MOAL: *Morus alba*.

The most abundantly planted trees were *Thuja occidentalis*, *Acer palmatum*, *Amelanchier* spp., *Acer* x *freemanii*, *Cornus* spp., *Juniperus virginiana*, and *Pinus mugo*, which are considerably different from the current dominant species and nearly all smaller-sized trees at maturity. The total tree planting rate in the study area was 1.42 trees ha^{-1} yr^{-1} and *Thuja occidentalis,* which was frequently planted along fencelines, represented 43% of all new trees planted. Diameter growth rates slowed with increases to tree size; the condition of measured trees also consistently worsened with greater tree size. However, the lower diameter growth rate of the > 0 cm to 10 cm tree size class was anomalous. It should be noted that multiple-year DBH measurements and growth rates derived therefrom are likely to have high measurement error, which is a potential explanation for this anomaly.

The dominant species in the study area in both the 2007/2008 and 2014 inventories (Fig. 4.2) was *Acer platanoides*. *Thuja occidentalis* exceeded *Acer platanoides* in 2014 in abundance. When planted trees are incorporated, *Gleditsia triacanthos, Thuja occidentalis, Acer* x *freemanii,* and *Tilia cordata* all increased in population size, while *Acer platanoides, Fraxinus pennsylvanica, Acer saccharinum, Aesculus hippocastanum, Ailanthus altissima,* and *Morus alba* decreased. No planted *Fraxinus pennsylvanica*, *Aesculus hippocastanum*, *Ailanthus altissima*, or *Morus alba* were observed. *Ailanthus altissima* had a substantially higher mortality rate than other trees (Table 4.4), followed by *Fraxinus pennsylvanica*, both of which were higher than the study area average annual mortality rate of 2.4%. No dead/absent *Gleditsia triacanthos* or *Tilia cordata* were observed. *Fraxinus pennsylvanica* were in the worse condition, which was likely attributable to the on-going EAB infestation in the study area, while *Thuja occidentalis* were consistently in better condition. Tree condition of other species was generally reflective of tree size, where consistently larger, older species (e.g., *Acer saccharinum* and *Aesculus hippocastanum*) were in worse condition. Diameter growth rates were as expected, given both species-specific growth rates and size-class distribution of species in the study area. The 672 trees alive in the study area in 2014 had a replacement value of $1,794,671 and total ecosystem service supply value of $20,272, with carbon storage being the most valuable, followed by the energy savings from reduced cooling in buildings attributable to tree shading (Table 4.5).

Table 4.4. Annual mortality rate (%), mean diameter growth rate (cm yr^{-1}), and mean condition index value of measured trees, stratified by diameter class and 10 most abundant species.

Category	N	Annual Mortality Rate	Mean growth rate	Mean Condition Index Value
All trees	806	2.40	0.59	0.30
Size class				
> 0 cm – 10 cm DBH	168	6.56	0.27	0.23
10.1 cm – 20 cm DBH	174	2.67	0.64	0.28
20.1 cm – 30 cm DBH	133	1.36	0.72	0.29
30.1 cm – 50 cm DBH	200	0.75	0.69	0.29
50.1 cm – 75 cm DBH	89	1.09	0.56	0.39
> 75 cm DBH	42	1.33	0.37	0.43
Species				
Acer platanoides	163	2.10	0.46	0.36
Fraxinus pennsylvanica	80	4.64	0.50	0.40
Gleditsia triacanthos	80	0	0.59	0.28
Thuja occidentalis	57	2.27	0.59	0.17
Acer saccharinum	37	1.51	0.48	0.39
Aesculus hippocastanum	36	0.76	0.28	0.37
Acer x *freemanii*	27	1.56	1.10	0.23
Tilia cordata	21	0	0.82	0.29
Ailanthus altissima	19	12.47	1.11	0.18
Morus alba	16	1.75	1.05	0.33
Other	270	2.93	0.65	0.26

Table 4.5. Urban forest ecosystem services provided by living trees (n = 672) measured in 2014 and their financial value, as estimated by the i-Tree Eco model.

Ecosystem service	Metric	Total	Value ($ CAD)
Air pollution removal	Pollutant removal (g yr^{-1})	167,556	1,362
	Carbon monoxide (g CO yr^{-1})	165	0
	Ground-level ozone (g O_3 yr^{-1})	125,879	1,059
	Nitrogen dioxide (g NO_2 yr^{-1})	31,978	269
	Sulphur dioxide (g SO_2 yr^{-1})	5,290	11
	Particulate matter (g $PM_{2.5}$ yr^{-1})	4,243	24
Rainfall interception	Avoided runoff (m^3 yr^{-1})	525	1,219
Building energy effects	Annual energy savings – heating (MBTU)	236	2,536
	Annual energy savings – heating (kWh)	20,942	1,570
	Annual energy savings – cooling (kWh)	55,951	4,196
Carbon storage and sequestration[1]	Gross carbon sequestration (kg C yr^{-1})	7,935	238
	Total carbon storage (kg C)	259,582	7,788
Urban forest replacement value	Total tree compensatory value ($)	-	1,794,671

[1] A value of $30 CDN per metric tonne of carbon was used (Pothier & Millward, 2013).

4.3.2. Vulnerability Analysis

Regression modelling yielded several significant tree mortality predictors and explained 62% of the variation in mortality at the individual-tree scale (Table 4.6) and 73% at the street-section scale (Table 4.9). The strongest predictor of mortality at both scales was tree condition in 2007/2008. Trees were more likely to survive the period between inventories in all land uses excluding commercial/industrial uses, though multi-unit residential and park land uses were not statistically significant. Within residential land uses, trees adjacent to row houses versus other housing types were more likely to experience mortality. *Acer platanoides* and trees within the >0 cm to 10 cm size class had higher odds of mortality, while *Gledistia triacanthos* and *Morus alba* had lower odds. At the street-section scale, significant mortality predictors were higher impervious cover and wider street widths, while homeownership and larger distances from adjacent buildings explained lower mortality rates. Contrary to vulnerability assumptions, individual trees in conflict with overhead utilities, buildings, and other pieces of infrastructure were less likely to die. Also contrary to assumptions, high species diversity and low built area intensity were significant predictors of mortality at the street-section scale. The model accuracy of classifying trees as live or dead (Table 4.7) shows relatively high prediction accuracies and indicates that, on average, predicting live trees had a higher accuracy than dead trees. However, these classification rates require further validation in future research with external datasets.

Table 4.6. Beta coefficients and *p*-values for the backward logistic regression analysis predicting individual tree mortality (0 = live; 1 = dead) using the urban forest vulnerability indicators.

Variable	Coefficient	Odds Ratio	*p*
Conflict with overhead utilities	-3.392	0.034	< 0.001
Conflict with buildings	-2.835	0.059	< 0.001
Conflict with other	-1.629	0.196	< 0.001
Land use - Institutional	-2.141	0.118	< 0.001
Land use - Multi-unit residential	-21.56	0	0.997
Land use - Park	-21.564	0	0.998
Land use - Residential	-1.334	0.263	0.002
Building type - Row house	0.939	2.556	0.027
Site type - Raised wooden planter	-1.625	0.197	0.007
2007/2008 tree condition	3.365	28.933	0.014
0 cm - 10 cm DBH	1.309	3.704	< 0.001
Acer platanoides	0.682	1.978	0.074
Gleditsia triacanthos	-20.272	0	0.995
Acer x *freemanii*	-1.805	0.164	0.026
Morus alba	-2.343	0.096	0.013
In-grown tree	2.299	9.96	< 0.001
Pseudo-R^2	0.620		

Table 4.7. Classification accuracies (%) of backward logistic regression models predicting tree mortality by diameter class and three most abundant species.

	N	Live Trees	Dead Trees	Total
> 0 cm – 10 cm DBH	168	87.1	74.6	82.1
10.1 cm – 20 cm DBH	174	94	34.3	84.3
20.1 cm – 30 cm DBH	133	87.1	65.3	83.2
30.1 cm – 50 cm DBH	200	100	100	100
50.1 cm – 75 cm DBH	89	98.8	71.4	96.6
> 75 cm DBH	42	97.4	75	95.2
Acer platanoides	163	100	95.8	99.4
Fraxinus pennsylvanica	80	83.9	91.7	86.3
Thuja occidentalis	57	97.9	66.7	93
All trees	806	95.1	62.7	89.7

Modelling tree condition also yielded significant predictors, though explained less variation in condition at the individual-tree scale (37%; Table 4.8) compared to the street-section scale (65%; Table 4.9). Evidence of poor tree management (e.g., improper pruning practices) and trees in the largest two size classes were strong predictors of poorer condition for individual trees. Median family income was significant at both the individual-tree and street-section scale, with higher income predicting better tree condition. Commercial buildings at the individual-tree scale and commercial/industrial land uses at the street-section scale also explained poor tree condition. Other significant predictors at the street-section scale were high existing canopy cover and open green space, higher homeownership, higher education, larger site sizes, and greater distances to adjacent buildings. All of these latter findings run counter to *a priori* assumptions of vulnerability and were likely heavily influenced by declining condition with tree size/age, as these variables were all indicative of sites that tended to have larger trees.

Table 4.8. Beta coefficients and *p*-values (in brackets) for the backward linear regression analysis predicting individual 2014 tree condition index values and diameter growth rates (cm yr^{-1}) of individual trees using the urban forest vulnerability indicators.

Independent Variable	Condition Model[1]	Independent Variable	Diameter Growth Model
Light availability	-0.091 (0.007)	2014 tree condition	-0.134 (0.000)
Poor management	0.282 (0.000)	Population density	-0.076 (0.095)
Vandalism	0.139 (0.000)	Conflict with buildings	0.132 (0.000)
Building type - Commercial at grade, apartment tower[2]	0.160 (0.000)	Conflict with other	0.096 (0.009)
Building type - Commercial at grade, residential	0.141 (0.000)	Distance to street	0.097 (0.018)
Building type - Commercial building	0.161 (0.000)	Width of street	-0.184 (0.000)
Building type - Institutional building	0.074 (0.029)	Poor management	-0.078 (0.030)
Building type - Row house	-0.087 (0.012)	Land use - Institutional	0.35 (0.000)
Site type - Grass median	0.088 (0.012)	Land use - Multi-unit residential	0.151 (0.003)
Site type - Raised concrete planter	0.112 (0.112)	Building type - Institutional building	-0.216 (0.001)
> 0 cm - 10 cm DBH	-0.149 (0.000)	Building type - Row house	-0.083 (0.018)
10.1 cm - 20 cm DBH	-0.071 (0.053)	Site type - Driveway/fenceline	-0.138 (0.000)
50.1 cm - 75 cm DBH	0.238 (0.000)	> 0 cm - 10 cm DBH	-0.327 (0.000)
> 75 cm DBH	0.238 (0.000)	10.1 cm - 20 cm DBH	-0.104 (0.006)
Acer platanoides	0.159 (0.000)	> 75 cm DBH	-0.105 (0.010)
Fraxinus pennsylvanica	0.161 (0.000)	*Acer platanoides*	-0.253 (0.000)
Thuja occidentalis	-0.071 (0.038)	*Acer saccharinum*	-0.086 (0.036)
Morus alba	0.060 (0.065)	*Aesculus hippocastanum*	-0.169 (0.000)
Median family income	-0.382 (0.000)	*Acer* x *freemanii*	0.152 (0.000)
Homeownership	0.215 (0.000)	*Morus alba*	0.077 (0.037)
Population with a university degree	0.141 (0.003)	In-grown tree	0.094 (0.008)
Replanted site	-0.137 (0.000)		
R^2	0.372		0.312

[1] The condition index ranges between 0 and 1, where 1 indicates extremely poor tree condition.

[2] Buildings with commercial space at street level with residential units or apartment towers (i.e., greater than five storeys) were classified as commercial at grade, residential, or commercial at grade, apartment tower, respectively.

The regression models predicting diameter growth rates did not perform as well as the mortality and condition models, explaining only 31% and 38% of the variation in growth rates at the individual-tree (Table 4.8) and street-section (Table 4.9) scale, respectively. At the individual-tree scale, significant predictors with the strongest explanatory power were institutional land uses predicting faster growth rates and the > 0 cm to 10 cm size class predicting slower growth rates. Additionally, 2014 tree condition was a significant predictor of growth rates at the individual-tree and street-section scale, with better condition explaining faster growth rates. Lastly, modelling tree planting rates at the street-section scale explained 53% of variation in rates (Table 4.9). High rates of homeownership, high existing canopy cover, high built area intensity, and good 2014 tree condition were all found to be significant predictors of tree planting rates.

Table 4.9. Beta coefficients and *p*-values (in brackets) for backward linear regression analyses predicting annual tree mortality rate (%), mean 2014 tree condition index value, mean diameter growth rate (cm yr^{-1}), and tree planting rate (trees ha^{-1} yr^{-1}) at the street-section scale using the urban forest vulnerability indicators.

Independent Variable	**Mortality Model**	**Condition Model**[1]	**Diameter Growth Model**	**Tree Planting Model**
Built area intensity	-0.236 (0.069)	0.339 (0.036)	-0.313 (0.017)	0.542 (0.002)
Land use - Commercial/industrial	-	0.736 (0.001)	-	-
Land use - Institutional	-	-	0.271 (0.025)	-
Land use - Park	-	0.359 (0.002)	-	-
Site size	-	0.469 (0.008)	-	-
Height of nearest building	-	-	-0.242 (0.043)	-
Distance to nearest building	-0.152 (0.091)	0.332 (0.007)	-	-
Width of street	0.171 (0.092)	-	-	-
Impervious cover[2]	0.262 0.040)	-	-	-
2014 tree condition	-	-	-0.407 (0.002)	-0.318 (0.039)
2007/2008 tree condition	0.889 (< 0.001)	-	-	-
Species diversity	0.240 (0.016)	-	-	-
Structural diversity	-	-0.395 (0.001)	-0.270 (0.049)	-
Median family income	-	-0.539 (0.008)	-	-
Homeownership	-0.261 (0.002)	0.337 (0.030)	-	0.299 (0.038)
Population with a university degree	-	0.354 (0.022)	-	-
Open green space	-	0.407 (0.021)	-	-
Existing canopy cover	-	0.435 (0.003)	-	0.640 (0.001)
R^2	0.733	0.646	0.383	0.533

[1] The condition index ranges between 0 and 1, where 1 indicates extremely poor tree condition.
[2] Impervious cover was measured at the parcel scale (mean = 65.3%; standard deviation = 21.5%) for the tree mortality analysis, since it could not be estimated in crown projections for dead trees.

4.4. Discussion and Conclusions

The findings of this study suggest that the highest exposure and corresponding levels of tree decline and mortality were most influenced by the intensity of land use and the conditions encountered in the built environment. Commercial land uses in street sections and commercial buildings adjacent to individual trees consistently explained higher mortality rates and poor tree conditions. While studies have found varying effects of commercial land uses on urban trees (e.g., Lawrence et al., 2012), it is generally established that they are among the most detrimental for tree health (Nowak et al., 2004; Jutras et al., 2010). However, at finer spatial scales it is important to differentiate between different causes and correlates of urban forest decline within commercial land uses. For example, street width (i.e., wider streets) can be a correlate of tree stress (Berrang et al., 1985; Nagendra and Gopal, 2010) and was found to be a significant predictor of mortality at the street-section scale in this study. However, several residential street sections in the study area had wider streets than commercial sections. While land use is a fairly established mechanism for stratifying urban landscapes and conducting urban forest research (Nowak et al., 1996; Steenberg et al., 2013), the results of this study suggest that at the household scale, differentiated indicators (e.g., building type, impervious cover, street geometry) are necessary for accurately predicting mortality and declining tree condition. Moreover, socioeconomic conditions have the capacity to be highly heterogeneous at the household scale compared to the neighbourhood scale.

There is a myriad of physical, biological, and social stressors and disturbances that afflict urban trees and forests (Trowbridge & Bassuk, 2004). In this study, exposure indicators were mainly limited in scope to those stressors associated with the built environment and urban form. Nor did this study explicitly address discrete and severe exposure events (e.g., storms). However, the intent was that the sensitivity indicators would, in part, address these other dimensions of exposure for which quantification and/or data availability were limiting factors for measurement. For example, vulnerability to biological threats (e.g., EAB) or storm events can be captured in the sensitivity metrics of species composition (e.g., ash abundance and species diversity; Laćan & McBride, 2008) and age structure (e.g., structural diversity and over-mature canopies; Staudhammer & LeMay, 2001; Lopes et al., 2009). Nonetheless, the study's findings suggest that quantifying biological exposures would be beneficial in future vulnerability assessment given the high levels of decline and mortality of *Fraxinus pennsylvanica* attributable to the EAB.

Despite this latter point, urban forest structural elements that characterize sensitivity were found to be valuable in examining vulnerability. Specifically, tree condition was a highly influential predictor of mortality and diameter growth, both for individual trees and street sections. This confirms existing research supporting tree condition as an effective predictor of mortality (Koeser et al., 2013). However, findings also highlight important drivers of condition decline, such as poor management and vandalism, where poor management was most often identified as improper pruning practices and vandalism as torn branches on smaller trees (Lu et al., 2010). Decline, mortality, and vulnerability of different tree species were likely a function of the composition and age distribution of the neighbourhood and tolerance of individual species to urban conditions (e.g., high tolerance of *Gleditsia triacanthos* and therefore low sensitivity and minimal mortality; Burns & Honkala, 1990). One notable species-level effect was the much higher likelihood and predictive capacity of mortality for in-grown species (e.g., *Ailanthus altissima*), which emphasizes the importance of differentiating between planted and in-grown trees in urban forest vulnerability assessment.

Tree size was a highly influential metric of urban forest sensitivity, both in its interaction with exposures and as a predictor of tree condition. Trees in the smallest size class had by far the highest rates and predictive capacity of mortality, as was expected (Nowak et al., 1990; Roman & Scatena, 2011). Measured trees were consistently in poor condition with size/age. In fact, at the street-section scale it appeared that sites and environments where large trees were typically present were the most important factors for explaining poor condition. For example, larger sites with higher canopy cover and/or open green space that were further away from adjacent buildings were more likely to have larger trees and, therefore, trees in poor condition. These findings, to some degree, highlight the influence of specific conditions in Harbord Village and subsequent limits to their generalizability. However, declining tree condition with age is an established pattern (Nowak et al., 2004), which suggests higher sensitivity and subsequent vulnerability of mature urban forest ecosystems. Importantly, it may also suggest that the processes driving decline in tree condition may sometimes differ from those driving mortality.

The adaptive capacity dimension of this vulnerability analysis was limited by the scale of available socioeconomic data (i.e., census dissemination areas as opposed to households). However, the findings did reveal some important aspects of urban forest adaptive capacity. Several studies support a strong positive relationship of both tree canopy cover and urban forest stewardship activities with socioeconomic status (e.g., Grove et al., 2006b; Troy et al., 2007; Conway et al., 2011), suggesting higher adaptive capacities in neighbourhoods with wealthier households. High median family income was found to be a significant predictor of better condition of individual trees and street sections. However, income was not influential on mortality as expected, though homeownership appeared to be an important driver of ecosystem change. Interestingly, higher rates of homeownership were significant predictors of lower mortality rates but poor tree condition. Based on the ecosystem change data and field observations, this could be explained by the abundance of in-grown trees on multi-unit (i.e., rental) properties, as in-grown trees were found to have consistently higher mortality rates and better condition. However, this needs to be substantiated with further field research with household-scale socioeconomic data.

Of specific interest to vulnerability research was the relationship of adaptive capacity to tree planting rates. Tree planting is more a social phenomenon than an ecological one, since it involves direct and deliberate action by individuals. Consequently, it was expected that adaptive capacity indicators describing the human population would be influential on tree planting rates (Greene et al., 2011). The tree planting model suggests that rates were much higher in residential areas with higher existing canopy cover and poor tree condition (i.e., larger, older trees). The only significant socioeconomic adaptive capacity indicator again was homeownership, which explained higher rates of tree planting, as expected (Greene et al., 2011). Overall, adaptive capacity indicators were less influential on urban forest ecosystem change than exposure and sensitivity indicators. However, this does not preclude them as being important in long-term urban forest vulnerability. Füssel (2010) emphasizes that while observed empirical data are more objective and reliable, they cannot reveal all aspects of system vulnerability, especially long-term risks. It is likely that the shorter timespan between tree inventories in this study, as well as the scale of data used in the analysis, might explain this lower influence of adaptive capacity on ecosystem change. Household-scale, qualitative research may also provide valuable insight into these social processes in future work.

Given their longevity and stationary nature, trees and forests are in general vulnerable to environmental change (Lindner et al., 2010), where urban forest structure and function may lag considerable in their response to drivers of change (e.g., changes in management practices). The disparity between commonly-planted tree species and overstory species composition in Harbord

Village, coupled with on-going decline of *Fraxinus pennsylvanica*, and its removal from tree planting schedules, suggest that considerable future change in ecosystem conditions is likely. Moreover, *Acer platanoides*, which was the dominant overstory species, was an extremely popular urban tree in previous decades but is now rarely planted in Toronto because of its invasive nature (City of Toronto, 2013b). In addition to these potential lag effects in species composition, species-specific mortality and planting rates and the shifts towards smaller, ornamental species in the neighbourhood may also correspond to declines in future ecosystem service supply. The i-Tree Eco model results, as well as existing research (Nowak & Dwyer, 2007), indicate that most ecosystem services are strongly associated with larger, longer-lived tree species with large leaf areas. These issues reinforce the temporal nature of vulnerability and associated impacts (i.e., losses in ecosystem service supply; Adger, 2006; Chapter Five). Urban forest vulnerability assessment therefore requires both hindsight, in the form of monitoring, and foresight, in the form of ecological modelling.

Vulnerability science offers an integrative lens through which to explore risk and loss of function in highly complex, social-ecological systems like the urban forest (Turner et al., 2003a; Adger, 2006; Grove, 2009). Most research investigating mortality and decline in urban forests focuses primarily on stressors and disturbances. This study suggests there is a need to investigate how these stressors interact with urban forest structure and surrounding human populations in order to reliably predict the likelihood of potential loss. Moreover, most of the established relationships between urban forests and socioeconomic variability are based on two-dimensional tree canopy cover data at broader spatial scales. There are few studies (e.g., Shakeel & Conway, 2013) investigating urban forest ecological processes at finer scales using empirical field data. However, the findings from Harbord Village will be limited to some degree in there transferability to different neighbourhoods, cities, and scales. For instance, regression analysis revealed some counterintuitive relationships between vulnerability indicators and ecosystem change variables (e.g., conflicts with infrastructure and mortality). Further research is needed that tests both the reliability and validity of indicator design and the generalizability of the findings in the study area.

This study is part of an on-going research initiative to understand the complex, interacting, and cumulative nature of urban forest vulnerability and to develop frameworks for vulnerability assessment. With the increasing attention to urban forests from municipalities

(Ordóñez & Duinker, 2013), community groups (Conway et al., 2011), and individuals (Sawka et al., 2013), the demand for management tools that quantify ecosystem structure, function, and change and inform management is high. The i-Tree models have been used globally as science-based tools for quantifying urban forest structure and function, while Neighbour*woods* is growing in popularity as a community-oriented tool for inventorying trees. Urban forest vulnerability assessment may provide a tool to supplement the understanding of urban forest benefits (i.e., i-Tree Eco) and condition (i.e., Neighbour*woods*) by quantifying potential threats and the likelihood of urban forest decline.

CHAPTER 5 SUPPLY OF URBAN FOREST ECOSYSTEM SERVICES AND THEIR VULNERABILITY: A PROSPECTIVE ANALYSIS

Abstract: The benefits derived from urban forest ecosystem services are garnering increasing attention in both environmental research and municipal planning agendas. However, because of their location in cities, urban trees and forests are vulnerable and commonly suffer disproportionate levels of stress and disturbance in comparison to more naturalized forest ecosystems. The purpose of this study is to assess and analyze spatial and temporal changes, and potential vulnerability, of the urban forest resource in Toronto, Canada. This was done using a spatially-explicit, indicator-based assessment of vulnerability and i-Tree Forecast modelling of temporal changes in forest structure and function. Nine different management and disturbance scenarios were simulated for 45 years and model output was analyzed at the ecosystem and municipal scale. Substantial mismatches in ecological processes between spatial scales of assessment were found, which can translate in to unanticipated loss of function as well as social inequities in the spatial distribution of urban forest benefits if scales of management are misaligned. At the municipal scale, the effects of the Asian longhorned beetle and ice storm disturbance were far less influential on urban forest structure and function than forecast changes in management actions. For instance, the strategic goal of removing invasive species and increasing tree planting rates to meet new canopy cover targets resulted in a decline in ecosystem service supply. Introducing vulnerability parameters in this modelling experiment increased the spatial heterogeneity in urban forest structure and function while expanding the spatial disparities of resident access to ecosystem services across the urban landscape. Also, while lower levels of ecosystem service supply were associated with higher urban forest vulnerability and vice versa, there was a variable and uncertain relationship between vulnerability and potential impacts to urban forest benefits (i.e., relative change in leaf biomass). Vulnerability assessment and analysis of urban forest ecosystems can provide strategic planning initiatives with valuable insight into the processes of, and potential risks for, structural and functional change resulting from management intervention.

Keywords: vulnerability, urban forest, ecosystem service, i-Tree Forecast, social-ecological system, scale

5.1. Introduction

The benefits derived from urban forest ecosystem services are garnering increasing attention in both environmental research and municipal planning agendas. City trees help to improve energy efficiency by shading buildings (Sawka et al., 2013), reduce the urban heat island effect (Solecki et al., 2005), and ameliorate environmental quality by removing air pollution and increasing stormwater retention (Xiao & McPherson, 2002; Nowak et al., 2006). The diverse array of ecological, social, and economic benefits provided by urban forest ecosystems (Nowak & Dwyer, 2007) have prompted a growing number of municipalities to develop tree protection policies and strategic urban forest management plans (Conway & Urbani, 2007; Ordóñez & Duinker, 2013; Gibbons & Ryan, 2015). However, because of their location in cities, urban trees and forests, and the ecosystem services they provide, are inherently vulnerable to a myriad of stressors. Urban landscapes are highly altered, frequently changing, and densely-settled environments with fragmented ownership and high levels of competition for space (Kenney & Idziak, 2000; Trowbridge & Bassuk, 2004; Konijnendijk et al., 2005). Consequently, urban forests commonly suffer disproportionate levels of stress and disturbance in comparison to more naturalized forest ecosystems.

The body of research on these urban forest stressors and disturbances continues to grow. For instance, there is considerable attention paid to biological threats to urban trees (e.g., Laćan & McBride, 2008). These include long-standing biological stressors like the Dutch elm disease (*Ophiostoma novo-ulmi*; Smalley & Guries, 1993), the damage of which helped to initiate the modern practice of urban forestry in North America (Johnston, 1996). However, much of the research and policy development (Haack et al., 2010; Herms & McCullough, 2014) is now focused on more-recently introduced invasive forest pests like the emerald ash borer (*Agrilus planipennis*; EAB) and Asian longhorned beetle (*Anoplophora glabripennis*; ALB). Climatic disturbances have also gained attention as detrimental impacts on urban forest ecosystems. This includes the changing climate (Ordóñez & Duinker, 2014), as well as isolated severe weather events (Lopes e t al., 2009; Hauer et al., 1993; Hauer et al., 2011; Staudhammer et al., 2011). A notable body of urban forest research is focused on tree decline and mortality attributable to various elements of the built environment (Jutras et al., 2010; Lu et al., 2010; Roman & Scatena, 2011; Koeser et al., 2013).

In addition to identifying and assessing the impacts of these various stressors and disturbances on city trees, it is crucial to understand their implications for the overall structure and function of urban forest ecosystems. The urban forest is a highly complex, heterogeneous, dynamic social-ecological system (Cadenasso et al., 2007; Grove, 2009; Mincey, 2012). Consequently, there is a substantial amount of uncertainty around the implications of these threats across spatial and temporal scales (Borgström et al., 2006; Cumming et al., 2006). Vulnerability science can provide a mechanism for integrating the biophysical, social, and built dimensions of urban forest stress and disturbance in a research context, shifting the focus beyond an impacts-only perspective to a more holistic view of the entire system and the ecosystem services it provides (Turner et al., 2003a; Adger, 2006; Chapter Three).

Ecosystem service supply is commonly employed as a focal point in vulnerability research. For instance, Luers et al. (2003) examined the vulnerability of agricultural ecosystem services to climatic change. Metzger et al. (2008) investigated the vulnerability of ecosystem service supply in Europe using carbon storage as a metric. Quantitative, indicator-based vulnerability assessments and analyses have been used at multiple scales and in multiple regions to explore potential threats to managed ecosystems and ecosystem service supply in social-ecological systems (Luers et al., 2003; Turner et al., 2003b; Schröter et al., 2005; Metzger et al., 2006; 2008; Lindner et al., 2010). The framework of system vulnerability developed by Turner et al. (2003a) is one of the more prominent employed in such research. I adapted and refined this framework in Chapter Three to define and conceptualize urban forest ecosystem vulnerability. I define it as "…the likelihood of decline in ecosystem service supply and its associated benefits for human populations, urban infrastructure, and biodiversity" (Chapter Three).

The temporal nature of vulnerability frequently necessitates some form of ecological modelling to forecast potential future scenarios of change (Eakin & Luers, 2006). Moreover, ecological modelling in highly complex and uncertain social-ecological systems like the urban forest enables the simulation of alternative experimental scenarios at spatial and temporal scales that would not otherwise be feasible (Jørgensen & Bendoricchio, 2001; Landsberg, 2003). This latter capacity of modelling in vulnerability research can therefore be highly useful in informing decision making and helping to shape longer-term strategic directions for municipal urban forest management. The i-Tree suite of ecological models developed by the United States Department of Agriculture (USDA) Forest Service provides a number of tools and methodologies for quantifying and assessing the structure and function of urban forest ecosystems. The i-Tree Eco model in particular has been used by a large number of municipalities to assess their urban forest resource and inform policy development (Ordóñez & Duinker, 2013; USDA Forest Service, 2013a). More recently, the i-Tree Forecast model was developed (Nowak et al., 2008; Nowak et al., 2014). It uses the methods from i-Tree Eco to simulate changes in urban forest structure and function over time and, therefore, is a valuable tool for investigating future urban forest vulnerability.

The purpose of this study is to assess and analyze spatial and temporal changes and potential vulnerability of the urban forest resource in Toronto, Canada. Specific research objectives include: 1) conduct a quantitative, spatially-explicit, indicator-based assessment of urban forest vulnerability at the ecosystem scale based on current conditions; 2) model temporal changes in urban forest structure and function under different management and disturbance scenarios using the i-Tree Forecast model; and, 3) identify potential future losses in ecosystem service supply to assess overall, long-term system vulnerability at the ecosystem and municipal scale. As the global population continues to concentrate in urban areas (United Nations, 2014), reliance on ecosystem services, and their associated benefits, provided by city trees and urban forests will expand. Understanding the drivers and processes of urban forest change and potential loss of function is vital for strategic planning and decision-support in the reduction of long-term vulnerability.

5.2. Methods

5.2.1. Study Area

The City of Toronto (Fig. 5.1) is the provincial capital of Ontario, Canada, and is situated on the northwest shore of Lake Ontario. Toronto is the fifth largest city in North America, and has a total area of 635 km^2, total population of 2,615,060, and a population density of 4,151 persons per km^2. Toronto has a continental climate with hot, humid summers and cold winters, with a mean July temperature of 22.2^{o}C, mean January temperature of -4.2^{o}C, and mean total annual precipitation of 834 mm (Environment Canada, 2008). The city is situated in the Deciduous Forest Region, where pre-settlement forests and residual forests within the city's larger parks and ravine system are characterized by sugar maple (*Acer saccharum*), red oak (*Quercus rubrum*), white ash (*Fraxinus americana*), white pine (*Pinus strobus*), and eastern hemlock (*Tsuga canadensis*; Ontario Ministry of Natural Resources [OMNR], 2012). Dominant urban forest species in more built-up and densely-settled areas include Norway maple (*Acer platanoides*), white cedar (*Thuja occidentalis*), Manitoba maple (*Acer negundo*), and green ash (*Fraxinus. pennsylvanica*; Nowak et al., 2013a). The study area was stratified using the 12 urban forest ecosystem classes (Chapter Two), who applied an integrated urban forest ecosystem classification (UFEC) framework to classify ecosystems at the neighbourhood scale in Toronto according to their biophysical conditions, built environment, and human population. For a more detailed description of the social-ecological conditions in these 12 urban forest ecosystem classes, see Chapter Two.

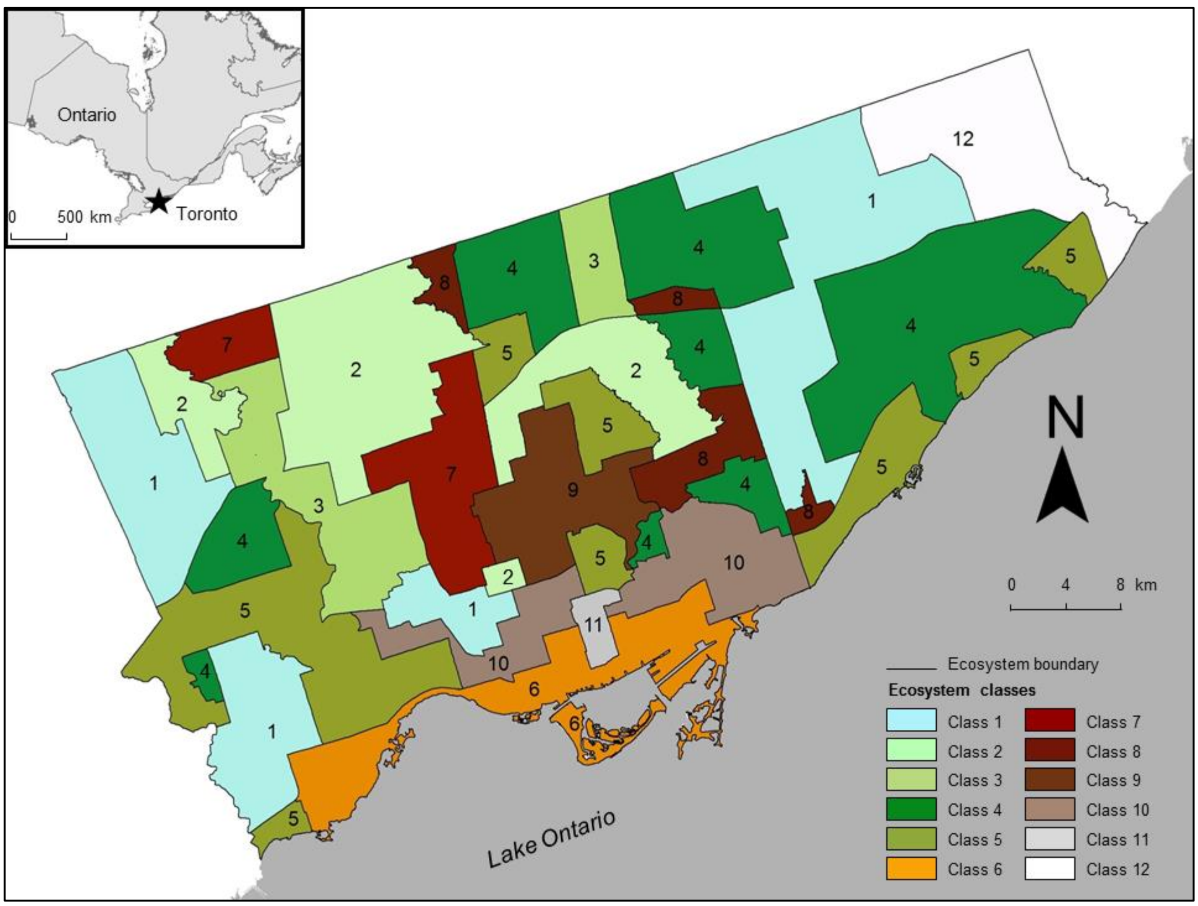

Fig. 5.1. The City of Toronto, Canada, showing the 12 urban forest ecosystem classes.

5.2.2. Vulnerability Assessment Framework

A quantitative, spatially-explicit assessment of urban forest vulnerability was conducted in each of the 12 ecosystem classes using a series of indicators of urban forest exposure, sensitivity, and adaptive capacity (Table 5.1). Indicator selection and design was guided by the conceptual framework of urban forest ecosystem vulnerability (Fig. 5.2) described in Chapter Three and refined further according to the spatial scale of assessment and data availability. Exposure indicators assess causes and correlates of stress and disturbance afflicting urban trees and forests. These indicators are primarily associated with the intensity of the built environment and density of human settlement, which have been found to be important drivers of urban forest structure, tree condition, and tree mortality (Nowak et al., 2004; Trowbridge & Bassuk, 2004; Konijnendijk et al., 2005; Jutras et al., 2010; Lu et al., 2010; Koeser et al., 2013). For instance, an abundance of impervious surfaces is indicative both harsh growing conditions and restrictive settings for tree establishment (Trowbridge & Bassuk, 2004; Tratalos et al., 2007). High-intensity commercial and industrial land uses have also been associated with poor tree condition, high mortality, and lower canopy cover (Nowak et al., 2004; Jutras et al., 2010; Lawrence et al., 2012). Urban form/morphology is characterized by indicators measuring building intensity, building height, and street width, which have all been found to have positive relationships with declining tree condition and/or increasing tree mortality (Trowbridge & Bassuk, 2004; Tratalos et al., 2007; Jutras et al., 2010; Nagendra & Gopal. 2010; Chapter Four). Finally, the density of human settlement has negative associations with urban forest health (Troy et al., 2007; Lu et al., 2010) and is measured using population density, pedestrian traffic, and vehicular traffic indicators.

Urban forest sensitivity refers to structural elements of tree communities within the ecosystem that are influential on their response to external stressors and disturbances. In this study, there are indicators that have both negative and positive relationships with vulnerability. The former includes species diversity and structural diversity, which are important factors in urban forest vulnerability to species- and age/size-specific exposures, such as invasive pests and storm events (Laćan & McBride, 2008; Lopes et al., 2009; Ordóñez & Duinker, 2010). Diversity indices are frequently used to assess forest ecosystem health, and were measured with the Shannon-Wiener Index (H’) using tree species composition and diameter at breast height (DBH) in 5-cm classes (Staudhammer & LeMay, 2001). Conversely, poor tree condition is associated with higher vulnerability, as it influences both the level of ecosystem service supply and the likelihood of mortality in response to stress (Armstrong & Ives, 1995; Koeser et al., 2013). Tree condition is measured as percent crown dieback according to the i-Tree Eco/Forecast protocol, where increasing dieback equates to declining condition (USDA Forest Service, 2013b; Nowak et al., 2014).

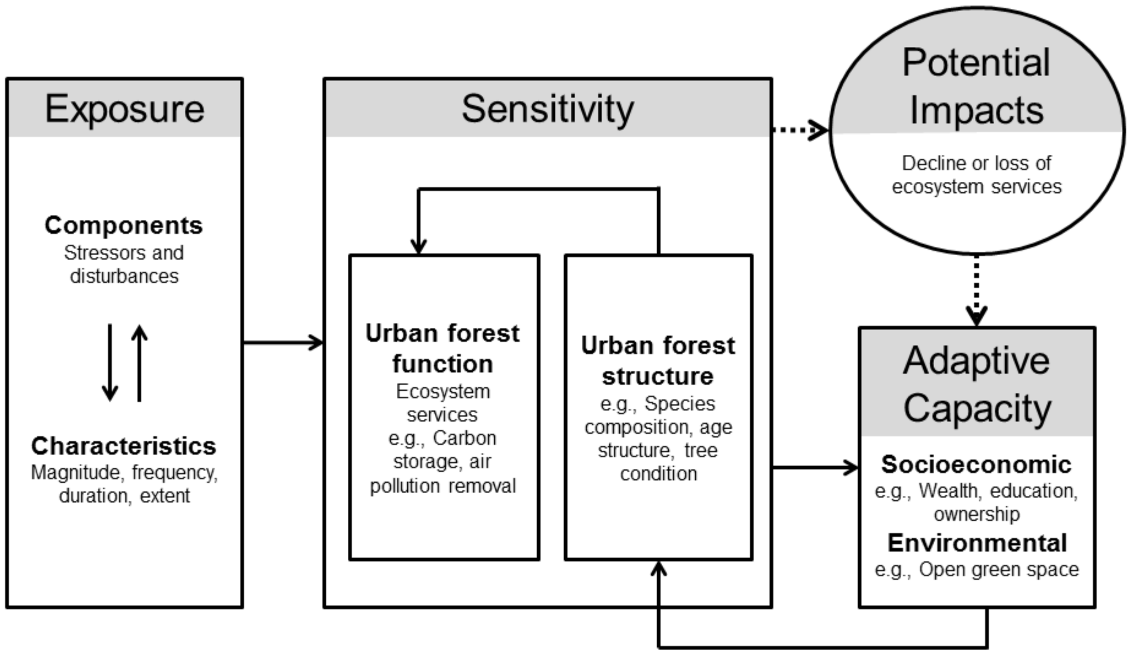

Fig. 5.2. Conceptual framework of urban forest vulnerability.

Adaptive capacity in urban forest ecosystems refers to both social and environmental elements that reduce vulnerability or increase capacity to tolerate stress (Adger, 2006; Chapter Three). Social adaptive capacity indicators measure socioeconomic conditions that are influential on both urban forest structure and function. Indicators used in this study include family income, dwelling value, homeownership, and education, which have all been found to have positive relationships with the density and size of trees and the extent of tree canopy cover (Grove et al., 2006b; Troy et al., 2007; Boone et al., 2010; Pham et al., 2013). These variables have also been associated with a greater likelihood of individuals to partake in stewardship activities (Manzo & Perkins, 2006; Conway et al., 2011; Greene et al., 2011). Environmental adaptive capacity indicates where tree planting and urban greening opportunities are more feasible (i.e., open green space), where urban forests are highly established (i.e., tree canopy cover), and where natural regeneration is likely to maintain forest conditions with fewer management needs than built-up areas (i.e., forested area; Troy et al., 2007; Nowak, 2012; Nowak & Greenfield, 2012).

In order to map and further communicate the assessment results, all indicators were standardized according to the maximum values found in the study area and aggregated into indices of exposure, sensitivity, and adaptive capacity ranging between 0 and 1. This was necessary given the different units of measurement across indicators as well as to calculate an index of overall urban forest vulnerability to be used in the i-Tree Forecast modelling. Individual indicators were not weighted prior to aggregation in this study. Standardized, aggregated indices of vulnerability and its components are a commonly-used tool in vulnerability research and assessments (Adger et al., 2004; Metzger et al., 2006; Birkmann, 2007). Exposure, sensitivity, adaptive capacity, and vulnerability indices were subsequently mapped at the ecosystem scale. Potential impacts are an outcome of urban forest vulnerability and refer to the likelihood of decline in ecosystem service supply resulting from the interaction of exposure and sensitivity (Lindner et al., 2010). This temporal component of urban tree vulnerability will be explored and assessed using simulation modelling under different management and disturbance scenarios described in Section 5.2.4.

Table 5.1. Exposure, sensitivity, and adaptive capacity indicators used to assess the vulnerability of the 12 urban forest ecosystem classes in Toronto.

Vulnerability Component	Indicator	Description
Exposure	Imperviousness[1]	Percent cover (%) of impervious surfaces
	Population density[2]	Density of humans living in a geographic area (population/km^2)
	Building intensity[1,3]	Intensity of the built area (%), estimated as the mean ratio of building footprint (m^2) to parcel area (m^2)
	Commercial/ industrial land use[4]	Percent area (%) that is commercial/industrial land use, as defined by the i-Tree measurement protocol
	Building height[3]	Mean building height (m)
	Street width[1,3]	Mean width of streets, estimated as the ratio of total street length (m) to total street area (m^2)
	Vehicular traffic[3]	24-hr volume of vehicular traffic at observation points
	Pedestrian traffic[3]	24-hr volume of pedestrian traffic at observation points
Sensitivity	Species diversity[5]	Shannon-Wiener index (H'), calculated using tree species data
	Structural diversity[5]	Shannon-Wiener index (H'), calculated using tree DBH data in 5-cm classes
	Tree condition[5]	Condition of existing trees, estimated as mean crown dieback (%)
Adaptive capacity	*Social*	
	Income[2]	Median family income ($), weighted by total population
	Housing value[2]	Average dwelling value ($), weighted by total number of dwellings
	Homeownership[2]	Percent of owner-occupied private dwellings (%)
	Education[2]	Population with a university certificate, diploma, or degree (individuals/10,000 people)
Adaptive capacity	*Environmental*	
	Open green space[1]	Percent grass and shrub cover (%)
	Tree canopy[1]	Percent tree canopy cover (%)
	Forested area[6]	Percent area (%) that is classified as undeveloped forest cover

[1] Land cover data derived from 2007 QuickBird satellite imagery
[2] Statistics Canada 2006 census data, aggregated from the census tract level
[3] City of Toronto municipal database
[4] DMTI Spatial Inc. land use data
[5] Toronto i-Tree Eco 2008 plot data
[6] Ontario Ministry of Natural Resources 2010 Forest Cover data

5.2.3. Model Description and Parameterization

The i-Tree Forecast model was designed by the USDA Forest Service to simulate changes in urban forest structure and function using the methods developed for the i-Tree Eco model (Nowak & Crane, 2000; Nowak et al., 2014). The model estimates the future changes in the number, size, species, and condition of trees from the initial structure at an annual time-step with user-defined simulation lengths. Three core model processes calculate changes from initial conditions: tree growth, tree mortality, and tree establishment. Annual diameter growth rates were estimated by the model based on growing-season length, species-specific growth rates sorted into slow-, moderate-, and fast-growing categories, and the level of tree competition as determined by light availability (i.e., crown light exposure). These base growth rates were adjusted according to tree condition (i.e., crown dieback) and maturity (i.e., ratio of tree height to species-specific maximum height), whereby diameter growth was slowed as condition worsened and/or trees matured.

Tree mortality was simulated using both user-input mortality rates and fixed rates, according to tree condition (Nowak et al., 2014). Trees with 50-75% crown dieback were assigned a fixed annual mortality rate of 13.1%, trees with 76-99% crown dieback were assigned a fixed annual mortality rate of 50%, and trees with 100% crown dieback were assigned a fixed annual mortality rate of 100%. Trees with 0-49% crown dieback have user-defined annual mortality rates. For this study, tree species were aggregated into seven groups based on species abundance and/or functional/ecological similarity, which were then assigned annual mortality rates based on averages of published data (Table 5.2). White cedar, sugar maple, and Norway maple are the three most abundant tree species in Toronto (Nowak et al., 2013a) and were assigned species-specific mortality rates from the literature. Mortality for the ash genus was fixed at 10% of the initial population in order to completely remove the species within 10 years of model initialization, given the current EAB infestation, high tree mortality rates after infestation, and the City of Toronto's ash removal strategy (City of Toronto, 2013b; Herms & McCullough, 2013). Invasive tree species have been found to have higher mortality rates and conifers have been found to have lower rates, so these two groups of species were assigned separate values (Nowak et al., 2004; Nowak et al., 2013b; Chapter Four). An average of all published total annual mortality rates was used for the remaining 90 species of broadleaved trees. Lastly, all user-input mortality rates are internally adjusted by the model according to diameter class, with mortality rates increasing with tree size and a higher rate for the smallest diameter class.

Tree species establishment rates are user-defined for each species at a fixed rate for each time-step. These rates were set at either replacement rates or rates identified in Toronto's strategic urban forestry plan (City of Toronto, 2013b), depending on the model scenario (Section 5.2.4). After the model was parameterized and initialized, tree total height, crown height, crown width, and leaf area were subsequently estimated at each time-step using allometric equations predicting their relationship to DBH at the species, genus, order, or family level. The level of total carbon storage and leaf biomass are also estimated throughout the simulation. For a more detailed description of model processes, see Nowak et al. (2008) and Nowak et al. (2014).

Table 5.2. Annual mortality rates (%) assigned to the seven tree species groups in Toronto for input in the i-Tree Forecast model.

Group	N	Annual Mortality Rate	Species	Sources
White cedar	1,675,008	2.97	*Thuja occidentalis*	5, 9
Sugar maple	1,025,378	2.39	*Acer saccharum*	5
Norway maple	694,237	4.09	*Acer platanoides*	4, 5, 9, 10
Ash	933,978	Eliminated within 10 years	*Fraxinus americana; Fraxinux pennsylvanica; Fraxinus excelsior*	1, 6
Main invasive species	942,875	13.00	*Acer negundo; Ailanthus altissima; Morus alba; Ulmus pumila*	4, 5, 9
Other conifers	1,350,986	1.82	All remaining conifer species	4, 5
Other broadleaves	3,786,245	4.52	All remaining broadleaved species	2, 3, 4, 5, 7, 8, 9

1: City of Toronto, 2013, 2: Lawrence et al., 2012, 3: McPherson & Simpson, 1999, 4: Nowak et al., 2004, 5: Nowak et al., 2013, 6: Poland & McCullough, 2006, 7: Roman & Scatena, 2011, 8: Staudhammer et al., 2011, 9: Chapter Four, 10. Sydnor, 1999

In this study, initial conditions were described using 407 randomly-sampled 0.04 ha plots measuring urban forest structure collected by the City of Toronto in 2008 following the i-Tree Eco measurement protocol (City of Toronto, 2010; Nowak et al., 2013a). The i-Tree Eco plot-based estimates for the total tree population in the study area were then converted to unique species-diameter cohorts for input to i-Tree Forecast. In the original i-Tree Eco analysis, the city was post-stratified after random sampling by land use (City of Toronto, 2010; Nowak et al., 2013a). In the present study, the City of Toronto was re-stratified using the 12 urban forest ecosystem classes (Chapter Two). This was done to capture a wider range of the social and ecological processes that shape urban forest ecosystems than is possible with land use alone. According to the i-Tree Eco measurement protocols, a minimum of 20 plots should be measured within each stratum in order to minimize sampling error (USDA Forest Service, 2013b). This minimum was met in all ecosystem classes except for Class 9 (18 plots), Class 8 (12 plots), and Class 11 (3 plots). While this is a study limitation, it is expected that sampling error will not be excessive given that these are the three smallest ecosystem classes, respectively. However, given the very few plots in Class 11, three additional cohorts of honeylocust (*Gleditsia triacanthos*), Norway maple, and little-leaf linden (*Tilia cordata*) were added to the initial structure based on their abundance in an existing street tree inventory (City of Toronto, 2012).

5.2.4. Experimental Design and Analysis

Three management scenarios and three disturbance scenarios where all simulated for 45 years, giving nine final experimental scenarios (Table 5.3). The management scenarios included: 1) a control, with replacement establishment rates and mortality rates derived from the literature (Table 5.2); 2) a vulnerability scenario, with replacement establishment rates and mortality rates weighted using the ecosystem-scale urban forest vulnerability index that was derived from the assessment described in Section 5.2.2 (Table 5.4); and 3) a strategic planning scenario, with mortality rates again weighted by vulnerability and establishment fixed at 570,000 trees annually, which is the rate identified as necessary to achieved Toronto's strategic planning goals (City of Toronto, 2013b). Additionally, Norway maple and the invasive species group were eliminated by the end of the simulations, again according to strategic planning goals (City of Toronto, 2013b). The disturbance scenarios included: A) a no disturbance scenario; B) an ALB scenario, where the maple, birch (*Betula* spp.), poplar (*Populus* spp.), and willow (*Salix* spp.) genera were eliminated within 10 years in a single ecosystem class and restricted from further establishment (Haack et al., 2010); and C) an ice storm scenario, where mortality and declines in tree condition (i.e., increases in dieback) were introduced at simulation-year zero to simulate the effects of an ice storm event (Hauer et al., 1993; Hauer et al., 2011). The ALB was introduced to Ecosystem Class 1, where shipping and trade are more abundant and adjacent to where an existing ALB outbreak was identified and contained (Haack et al., 2010). The ash genus was removed within 10 years in all nine scenarios to simulate the effects of the already-present EAB and management control thereof (City of Toronto, 2013b; Herms & McCullough, 2014).

Table 5.3. Experimental design of the three management and three disturbance scenarios for modeling with i-Tree Forecast. Each of the nine final scenarios represent all possible management and disturbance combinations.

Scenario	Tree Mortality	Tree Establishment
1 – Control	— Annual mortality rates derived from the literature — *Fraxinus* genus eliminated within 10 years	— Annual establishment rate equal to species-specific mortality at year one — No *Fraxinus* genus established
2 – Vulnerability	— Annual mortality rates derived from the literature and weighted by the vulnerability index in each ecosystem class — *Fraxinus* genus eliminated within 10 years	— Annual establishment rate equal to control scenario — No *Fraxinus* genus established
3 – Strategic planning	— Annual mortality rates derived from the literature and weighted by the vulnerability index in each ecosystem class — *Fraxinus* genus eliminated within 10 years — *Acer platanoides, Acer negundo, Ailanthus altissima, Morus alba,* and *Ulmus pumila* eliminated within 45 years	— Annual establishment rate set at 570,000 trees yr^{-1}, based on rate needed to reach 40% canopy cover — Species-specific establishment rates based on initial composition ratios — No *Fraxinus* genus established — No *Acer platanoides, Acer negundo, Ailanthus altissima, Morus alba,* or *Ulmus pumila* established
A – No disturbance	— No change to annual mortality rates	— No change to annual establishment rates
B – Asian longhorned beetle	— *Acer, Betula, Populus,* and *Salix* genera eliminated within 10 years in Class 1	— No *Acer, Betula, Populus,* or *Salix* genera established in Class 1
C – Ice storm event	— Mortality of 1.4% of trees less than 30 cm DBH, 2.3% of trees 30-60 cm DBH, and 1.9% of trees greater than 60 cm DBH at year zero — Critical condition assigned to 1.3% of trees less than 30 cm DBH, 6.5% of trees 30-60 cm DBH, and 17.1% of trees greater than 60 cm DBH at year zero	— No change to annual establishment rates

To analyze changes in urban forest structure and function, several response variables were selected for comparison. At the municipal scale, changes in species composition were assessed through comparisons of the percent of the total tree population for each of the seven species groups at simulation-year zero and 45. Changes in structure and function were quantified annually over the 45-year scenarios at the municipal scale. Response variables included mean tree DBH (cm), the total number of trees, total leaf biomass (kg), and total carbon storage (kg C). At the ecosystem scale, changes in total carbon storage, leaf biomass, and the total number of trees were assessed in each of the 12 ecosystem classes between simulation-year zero and 45. Lastly, to assess and communicate potential impacts resulting from vulnerability-weighted mortality, disturbance, and management intervention, ecosystem service supply and its relative change between simulation-year zero and 45 were quantified and mapped at the ecosystem scale. Ecosystem service supply was assessed using total leaf biomass, since leaf biomass is positively associated with the overall level of supply (Nowak et al., 2008). Ecosystem service supply and potential impacts were mapped for Scenario 2A only.

Table 5.4. Annual mortality rates (%) of the seven tree species groups adjusted for urban forest vulnerability in each ecosystem class.

Ecosystem Class	White Cedar	Sugar Maple	Norway Maple	Ash[1]	Main Invasive Species	Other Conifers	Other Broadleaves
Unweighted	2.97	2.39	4.09		13.00	1.82	4.52
1	2.23	2.77	3.81	-	12.13	1.70	4.22
2	2.44	3.03	4.18	-	13.28	1.86	4.62
3	2.59	3.22	4.43	-	14.08	1.97	4.89
4	2.66	3.31	4.55	-	14.48	2.03	5.03
5	1.78	2.22	3.05	-	9.70	1.36	3.37
6	2.44	-	4.18	-	13.29	1.86	4.62
7	2.34	2.91	4.00	-	12.72	1.78	4.42
8	3.29	4.09	5.64	-	17.92	2.51	6.23
9	1.96	2.44	3.36	-	10.68	1.50	3.71
10	2.18	-	3.74	-	11.88	1.66	4.13
11	-	-	7.60	-	24.17	-	8.40
12	1.69	2.10	2.89	-	9.20	1.29	3.20

[1] The ash genus is eliminated within 10 years in all scenarios.

5.3. Results

5.3.1. Vulnerability Assessment

The vulnerability assessment revealed several patterns across the 12 urban forest ecosystem classes, as shown by the indicator values (Appendix C) and vulnerability indices (Fig. 5.3; Fig. 5.4). Vulnerability was by far the highest in Class 11, which is the smallest ecosystem class and situated in the high-density downtown core, where population density, building height, and pedestrian traffic were all at least twice the value of other ecosystem classes. This class had the highest levels of exposure and sensitivity, as well as the second lowest adaptive capacity. Class 12 was on the opposite extreme with the lowest vulnerability, and is situated on the outer edge of Toronto and includes extensive forested lands and open green space. Class 12 had the lowest exposure and highest adaptive capacity, but moderate sensitivity due to poorer tree condition. Class 8 was the second most vulnerable ecosystem, owing to its higher levels of exposure and sensitivity. However, despite low social adaptive capacity, the apartment towers and industrial areas that characterize this ecosystem are commonly adjacent to open green spaces and forested areas, giving moderate adaptive capacity levels. Class 1, which is an extensive ecosystem with mixed residential and industrial land uses, was somewhat anomalous with high levels of exposure but very low sensitivity due to high species and structural diversity.

Ecosystem Class 6 is characterized by high-density residential areas and waterfront industrial land uses, and despite higher levels of exposure, it had moderate sensitivity, adaptive capacity, and overall vulnerability. Classes 4 and 10 are both characterized by typical, residential neighbourhoods and together cover 25% of Toronto's land area. Despite the higher level of exposure of the higher-density Class 10, Class 4 had higher overall vulnerability due to its abundance of trees in poor condition and high level of sensitivity. Classes 5 and 9 are both associated with highly-affluent populations and extensive tree canopy cover. Consequently, both had very low vulnerability index values. Lastly, Classes 2, 3, and 7 are all characterized by mixed-residential neighbourhoods with moderate levels of exposure, sensitivity, and adaptive capacity, though the higher-density Class 7 had higher levels of exposure and lower adaptive capacity.

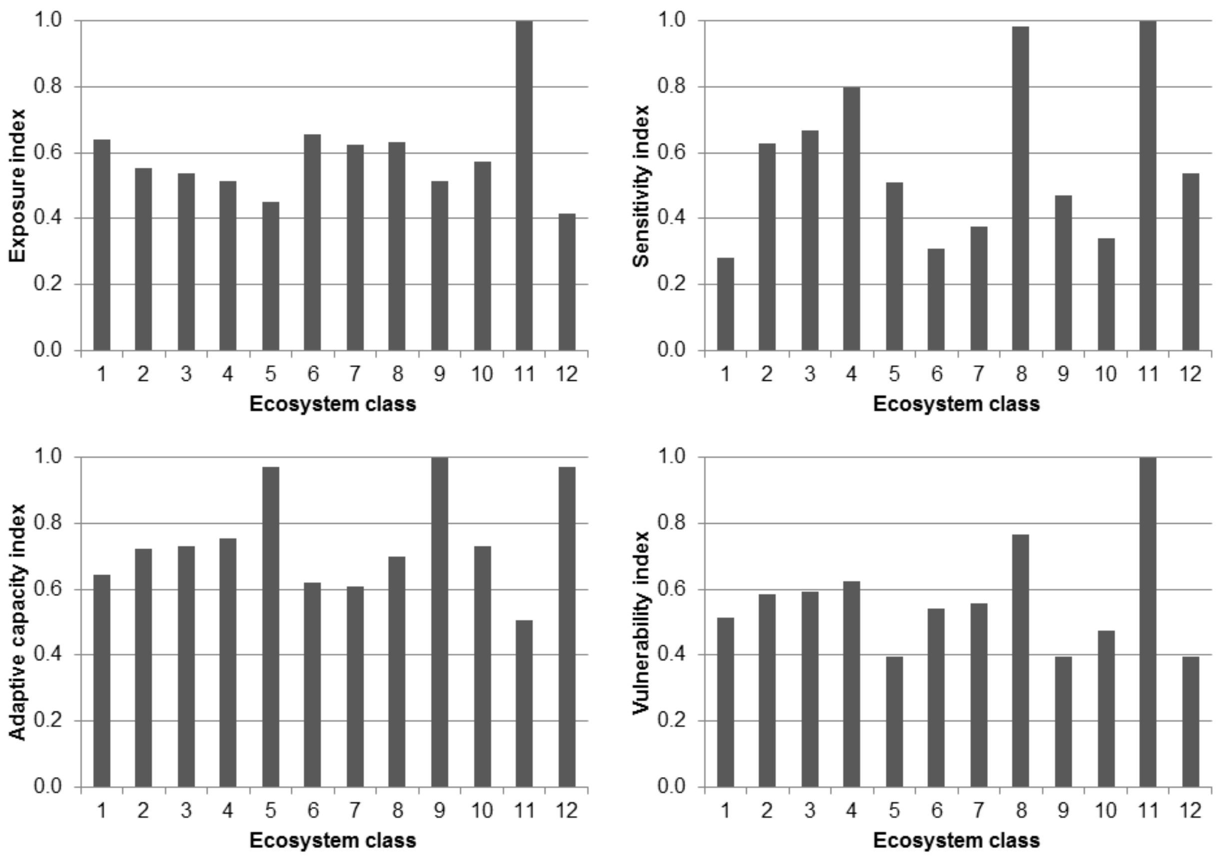

Fig. 5.3. Exposure, sensitivity, adaptive capacity, and vulnerability index values in each of the 12 urban forest ecosystem classes.

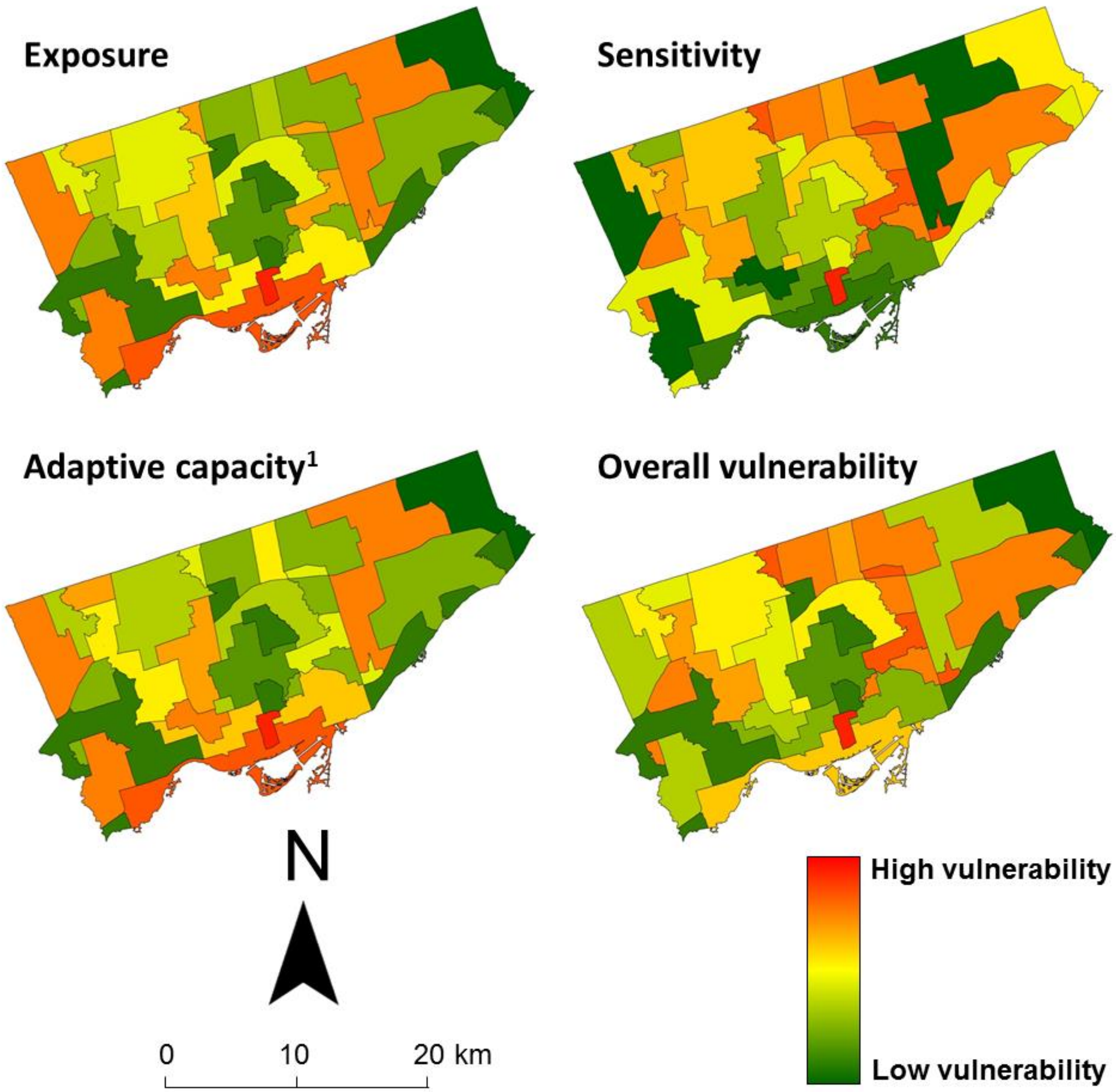

Fig. 5.4. Urban forest vulnerability assessment results for Toronto, Canada showing indices of exposure, sensitivity, adaptive capacity, and overall vulnerability mapped in the 12 ecosystem classes. [1] The colour scheme for adaptive capacity has been inverted to be consistent with the other maps, so that red indicates low adaptive capacity.

5.3.2. Model Output and Analysis

At the municipal scale, the influence of the disturbance scenarios on urban forest species composition was marginal (Fig. 5.5). Small declines of the invasive species, Norway maple, and sugar maple groups attributable to ALB were observed in Scenario 1B, which were slightly exacerbated in Scenario 2B when vulnerability-weighted mortality rates were introduced. The strategic planning scenarios (i.e., 3A, 3B, and 3C) were highly influential on species composition over the 45-year simulations. In accordance with the strategic planning goals in the experimental design of eliminating invasive species and Norway maple, the loss of these species by the end of the simulation was observed. However, what was not expected was the marked decline in the percent of sugar maple in the municipal tree population, which was most likely due to error in the modelling and is thus a limitation of the study. Corresponding increases in the other broadleaves, other conifers, and white cedar groups were observed, though there was a proportionally smaller increase of other broadleaf trees. At the municipal scale, the ice storm disturbance had no noticeable effect on species composition in Scenarios 1C, 2C, and 3C. As expected, the ash genus was absent in the final simulation-year of all scenarios.

The temporal dynamics of urban forest structure and function over the 45-year simulations showed a number of trends (Fig. 5.6). In all three control scenarios and all three vulnerability scenarios, the urban forest structure tended towards fewer but larger (i.e., DBH, leaf biomass, carbon storage) trees. As with urban forest composition, the strategic planning scenarios were markedly different in their structural dynamics. Unlike the control and vulnerability scenarios, a net gain in the total number of trees was observed. Correspondingly, a net loss in total carbon storage was also observed. This latter finding in the strategic planning scenarios is reflective of a shift in size-class distribution towards a smaller (i.e., younger) tree population with a much lower carbon storage. Leaf biomass showed considerable increases in values over the simulation in all scenarios, with greater rates of increase in the control and vulnerability scenarios. Again there was no discernible influence of the ice storm disturbance at this scale, though marginally lower values in all response variables were present. At the municipal scale, the only observable influence of introducing vulnerability-weighted mortality rates in the different ecosystem classes was on total carbon storage. The vulnerability scenarios showed higher carbon storage values than the controls. A change in the structural dynamics in all scenarios is observable at simulation-year 10 when the removal of the ash population is complete. For instance, mean DBH and total carbon storage declined until year 10 before beginning to increase for the remainder of the simulations.

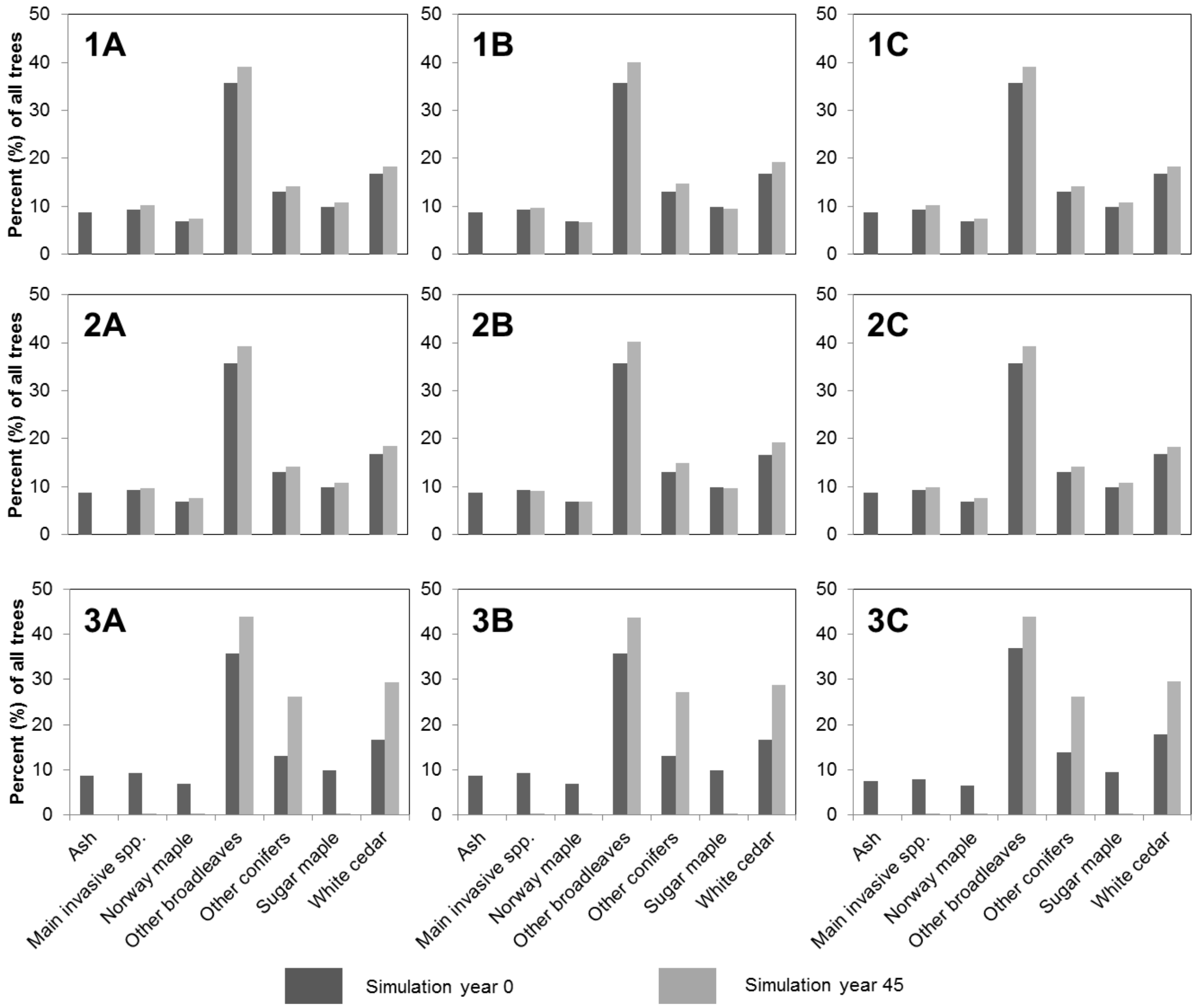

Fig. 5.5. Changes in the percent of the seven species groups at the municipal scale between simulation-year zero and simulation-year 45 in all nine scenarios.

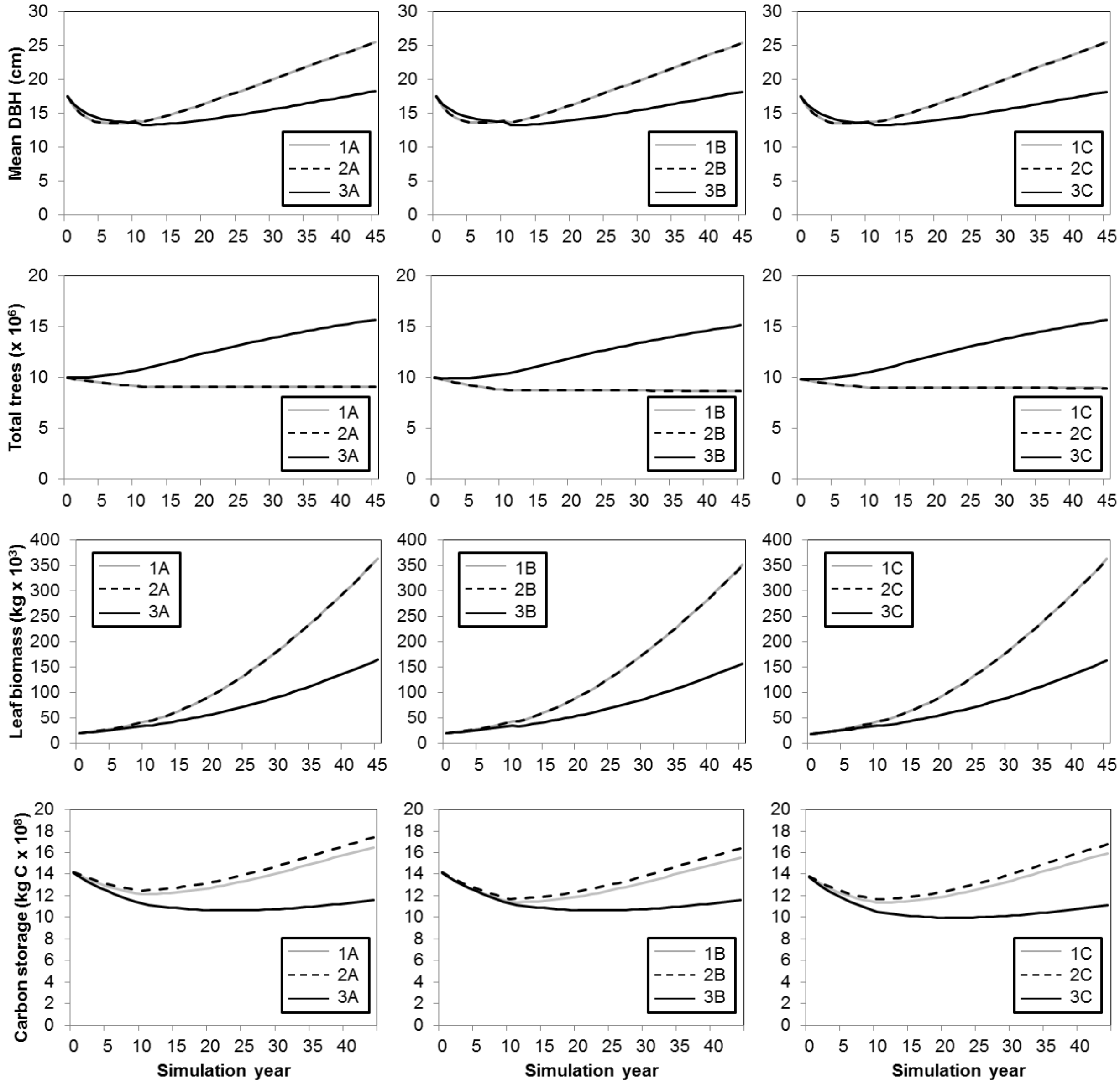

Fig. 5.6. Temporal dynamics of mean diameter at breast height (DBH; cm), total number of trees, leaf biomass (kg), and total carbon storage (kg C) at the municipal scale over the 45-year simulation in all nine scenarios.

While no major differences in urban forest structure and function were observed between the beginning and end of simulation at the municipal scale between control and vulnerability scenarios, substantial variability existed across classes at the ecosystem scale. Variability in carbon storage (Table 5.5) was strongly influenced by the level of vulnerability assessed in each ecosystem class (Section 5.3.1). Where vulnerability was higher, net losses in carbon storage were exacerbated (e.g., Classes 2 and 6) or net gains were subdued (e.g., Classes 3, 4, 8, and 11). The same pattern in inverse was observed in ecosystem classes with lower vulnerability (e.g., Classes 1, 7, 9, 10, and 12). In Class 5, which had low vulnerability, a shift from a net loss to a net gain in carbon storage was observed between the control and vulnerability scenarios, respectively. Unlike carbon storage, leaf biomass increased considerably in all ecosystem classes (Table 5.6). However, a similar trend in the relationship between leaf biomass and vulnerability was observed, albeit much less pronounced. Leaf biomass accumulation was much less in Class 11 and marginally less in Class 8, while a small increase in accumulation was observed in Class 5.

Table 5.5. Change in tree carbon storage (kg C x 10^3) in each ecosystem class between simulation-year zero and simulation-year 45 in all nine scenarios.

Class	1A	1B	1C	2A	2B	2C	3A	3B	3C
1	35,140	-51,584	32,536	46,628	-48,411	43,448	-34,509	-58,910	-34,649
2	-59,180	-59,180	-58,120	-62,546	-62,546	-61,370	-107,364	-107,364	-104,928
3	57,876	57,876	57,137	50,567	50,567	50,061	-25,585	-25,585	-24,045
4	128,254	128,254	126,842	95,162	95,162	94,683	-16,959	-16,959	-14,796
5	-22,948	-22,948	-29,707	45,634	45,634	34,738	-81,744	-81,744	-89,773
6	-22,591	-22,591	-21,738	-23,414	-23,414	-22,544	-32,575	-32,575	-31,244
7	26,666	26,666	26,692	28,206	28,206	28,223	4,886	4,886	5,066
8	42,895	42,895	42,841	27,091	27,091	26,980	-1,568	-1,568	-1,556
9	-22,946	-22,946	-25,593	-11,278	-11,278	-15,476	-49,619	-49,619	-49,703
10	-29,825	-29,825	-28,766	-21,911	-21,911	-20,993	-57,051	-57,051	-54,822
11	936	936	924	6	6	8	-218	-218	-213
12	97,777	97,777	92,093	145,386	145,386	136,721	151,299	151,299	144,281
Toronto	232,055	145,331	215,140	319,532	224,493	294,479	-251,007	-275,408	-256,384

As observed at the municipal scale, lower carbon storage and leaf biomass were associated with the strategic planning scenarios at the ecosystem scale. Less accumulation of leaf biomass was observed in all ecosystem classes, though there was little variability across classes. However, the net change in carbon storage was variable and observed as both net gains and losses. Where a net gain in carbon storage was observed in control scenarios, smaller gains were observed in strategic planning scenarios (e.g., Class 7). Conversely, where net losses were observed in control scenarios, they were exacerbated in strategic planning scenarios (e.g., Classes 2, 5, 6, 9, and 10). However, another common finding was a shift from a net gain to a net loss in carbon storage (e.g., Classes 1, 3, 4, 8, and 11). The sole anomaly in these trends in strategic planning scenarios was Class 12, which was the least vulnerable ecosystem. In Class 12, both the carbon storage and total number of trees were observed as net increases.

Table 5.6. Change in tree leaf biomass (kg) in each ecosystem class between simulation-year zero and simulation-year 45 in all nine scenarios.

Class	1A	1B	1C	2A	2B	2C	3A	3B	3C
1	29,690	16,804	29,644	29,690	16,808	29,644	13,797	6,263	13,725
2	34,309	34,309	34,287	34,309	34,309	34,287	15,966	15,966	15,902
3	25,073	25,073	25,006	25,071	25,071	25,006	9,618	9,618	9,499
4	53,473	53,473	53,386	53,469	53,469	53,381	25,849	25,849	25,715
5	44,741	44,741	44,758	44,780	44,780	44,797	19,709	19,709	19,680
6	19,515	19,515	19,515	19,515	19,515	19,515	8,005	8,005	7,976
7	24,769	24,769	24,762	24,769	24,769	24,762	8,875	8,875	8,865
8	24,171	24,171	24,144	23,893	23,893	23,866	9,107	9,107	9,076
9	23,277	23,277	23,194	23,277	23,277	23,194	9,195	9,195	9,099
10	26,325	26,325	26,312	26,325	26,325	26,309	6,543	6,543	6,503
11	1,655	1,655	1,655	929	929	929	368	368	368
12	36,664	36,664	36,560	36,664	36,664	36,548	17,366	17,366	17,244
Toronto	343,663	330,777	343,223	342,690	329,808	342,238	144,398	136,864	143,654

Again similar to the municipal scale, the most consistent theme at the ecosystem scale in the net change in the number of trees (Table 5.7) across all classes and scenarios was an inverse relationship between carbon and trees, where fewer trees were associated with higher carbon storage. However, exceptions to this theme existed at either extreme in the level of urban forest vulnerability. In the two most vulnerable ecosystems (i.e., Classes 8 and 11) and in the least vulnerable ecosystem (i.e., Class 12), this latter relationship was inversed. A decline in the total number of trees was associated with a decline in carbon storage in Classes 8 and 11, while an increase in the total number trees was associated with an increase in carbon storage in Class 12.

Differences in carbon storage, leaf biomass, and the total number of trees were more notable across the management scenarios than the disturbance scenarios. The ice storm was associated with consistently, though marginally, lower levels of carbon storage and leaf biomass, and higher total numbers of trees compared to other scenarios. With the introduction of ALB in Scenarios 1B, 2B, and 3B, the decline in carbon storage in Class 1 in the strategic planning scenarios was exacerbated, while carbon storage saw a considerable shift from a net gain to a net loss in the control and vulnerability scenarios. Correspondingly, declines in the total number of trees in Class 1 in the control and vulnerability scenarios were exacerbated while the strategic planning scenarios saw a shift from a net gain to a net loss. Leaf biomass had approximately half the net accumulation of the control scenario in Class 1 with the introduction of ALB.

Table 5.7. Change in the total number of trees (x 10^3) in each ecosystem class between simulation-year zero and simulation-year 45 in all nine scenarios.

Class	1A	1B	1C	2A	2B	2C	3A	3B	3C
1	-217	-617	-214	-218	-623	-215	243	-325	256
2	-58	-58	-57	-58	-58	-57	1,090	1,090	1,103
3	-29	-29	-28	-49	-49	-28	-320	-320	-309
4	-172	-172	-170	-173	-173	-170	1,253	1,253	1,283
5	-153	-153	-150	-155	-155	-152	1,267	1,267	1,277
6	-4	-4	-4	-4	-4	-4	204	204	207
7	-6	-6	-6	-6	-6	-6	136	136	140
8	-70	-70	-69	-93	-93	-91	-162	-162	-157
9	-9	-9	-16	-9	-9	-15	532	532	536
10	-7	-7	-6	-7	-7	-8	-1	-1	3
11	0	0	0	-3	-3	-3	-3	-3	-2
12	-158	-158	-157	-159	-159	-164	1,475	1,475	1,481
Toronto	-884	-1,284	-877	-935	-1,340	-913	5,713	5,145	5,816

Lastly, mapping total leaf biomass, leaf biomass per hectare, relative change in leaf biomass over scenario simulation (i.e., potential impacts), and vulnerability at the ecosystem scale (Fig. 5.7) revealed the variable relationship between vulnerability and ecosystem service supply. Leaf biomass and leaf biomass per hectare frequently had positive relationships with the level of urban forest vulnerability, where high levels of ecosystem service supply were associated with low vulnerability and vice versa (e.g., Classes 3, 5, 7, 10, 11, and 12). However, the relationship between potential impacts and vulnerability was more variable, where several ecosystem classes had much higher net gains in leaf biomass despite higher vulnerability, and vice versa.

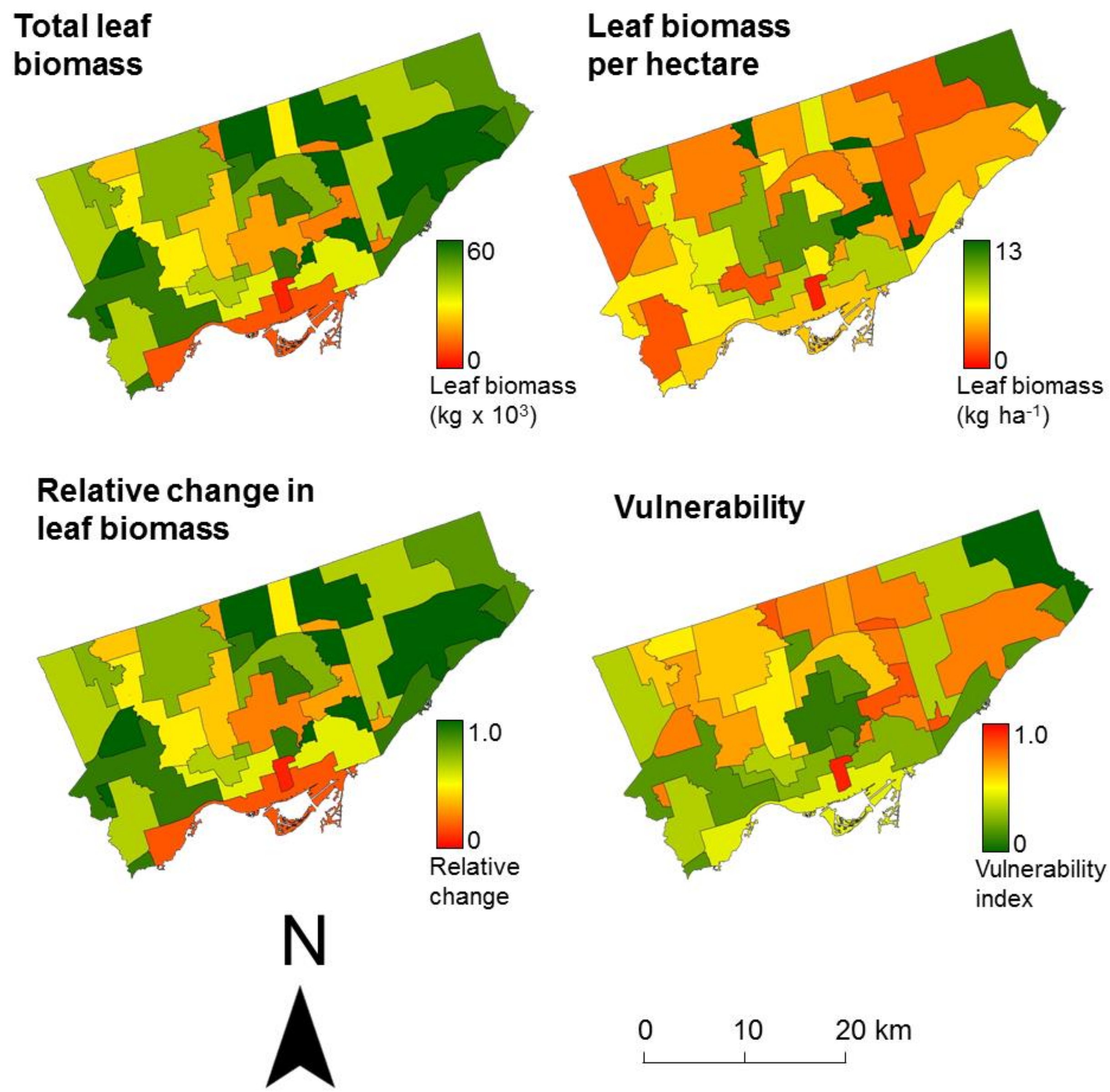

Fig. 5.7. Top: urban forest ecosystem service supply, represented by total leaf biomass (kg x 10^3) and leaf biomass per hectare (kg ha^{-1}), mapped at simulation-year 45 in Scenario 2A in the 12 ecosystem classes. Bottom: the urban forest vulnerability index and potential impacts, represented by the relative change in leaf biomass between simulation-year zero and 45 in Scenario 2A in the 12 ecosystem classes.

5.4. Discussion

A core focus of this study and prominent theme in its findings was variability in the spatial and temporal nature of vulnerability. Spatial scale is a critical but often ignored or incorrectly handled dimension of urban environmental management (Folke et al., 1998; Lovell et al., 2003; Borgström et al., 2006), including urban forestry (Steenberg et al., 2013). In particular, a challenge in social-ecological systems arises from mismatches in spatial scales (Lee, 1993; Cumming et al., 2006). Specifically, these mismatches occur when organizational structures and institutions of environmental management are misaligned with the scales of social-ecological processes of the system under management (Cumming et al., 2006). This inattention to spatial scale or mismatches between management and ecological processes may cause a loss in the effectiveness of management intervention and a corresponding decline in ecosystem function (Borgström et al., 2006; Cumming et al., 2006).

The findings of this study emphasize these latter points, where changes in urban forest structure and function (i.e., tree abundance, leaf biomass, and carbon storage) frequently differed at the ecosystem scale from overall trends at the municipal scale. For instance, despite the citywide trend of increasing carbon storage over the 45-year simulations in both control and vulnerability scenarios, there was considerable difference in both the magnitude and direction of net changes in carbon storage and total tree numbers at the ecosystem scale. Additionally, a pervasive trend at the municipal scale and in most individual ecosystem classes was increasing tree numbers being associated with less stored carbon and vice versa, given the shift towards a smaller and younger size-class distribution. However, in the most (i.e., Classes 8 and 11) and least (i.e., Class 12) vulnerable ecosystems, the total number of trees and total carbon storage were positively correlated. This suggests that these mismatches between both spatial and temporal scales could potentially be exacerbated at both high and low levels of ecosystem vulnerability.

The experimental design and model parameterization of this study dictated many of these incongruences between scales. The modifiable areal unit problem also adds uncertainty to these findings, where the units of analysis (i.e., bounding and spatial scale), and subsequent aggregation of the social-ecological data, can bias the analysis results (Openshaw, 1984). However, it is reasonable to assume such spatial variability in tree mortality rates within municipal boundaries depending on specific biophysical, built, and socioeconomic conditions

(Nowak et al., 2004; Lawrence et al., 2012). This would subsequently drive variability in the level of ecosystem service supply, which can translate to both unanticipated loss of function and social inequities in the distribution of benefits if the scale of management is misaligned. Moreover, Borgström and colleagues (2006) found that meso-scales, between the site- and operational-scale and the much broader strategic scale, are often absent in urban environmental management. Research on urban forest ecology and management, and in this case vulnerability, at meso-scales like the ecosystem classes adopted in this study is arguably an important but under-utilized area in science-based urban forest planning and policy development.

Introducing vulnerability parameters in this modelling experiment increased the spatial heterogeneity in urban forest structure and function while expanding the spatial disparities of resident access to ecosystem services across the urban landscape. With regards to the spatial variability of urban forest vulnerability across the city, it was clear that the more densely-settled and heavily built-up urban core had the highest vulnerability. Specifically, the downtown core ecosystem (i.e., Class 11) and the mixed industrial and high-density residential ecosystem (i.e., Class 8) were the most vulnerable. However, they were also the two smallest in spatial extent, so potential impacts in these ecosystems would be less substantive to the ecosystem service supply of the entire city. The more extensive, affluent, and less-densely populated residential neighbourhoods and peri-urban forests outside of the urban centre (e.g., Classes 5 and 12) tended towards lower vulnerability and higher levels of ecosystem service supply. From both an urban forest benefits and environmental justice perspective, it is valuable to make the distinction between population density and the spatial extent of these ecosystems classes (Troy et al., 2007; Landry & Chakraborty, 2009; Pham et al., 2013). While the most vulnerable ecosystem classes represented a relatively small proportion of Toronto's spatial extent, they are among the most densely populated and least affluent.

Temporal scale and variability in system structure and function is another important component of vulnerability, especially given the more complex and longer-term socioeconomic dimensions of vulnerability (Eakin & Luers, 2006; Füssel, 2010). Moreover, trees and forests are especially vulnerable to environmental change and altered disturbance regimes, given the longevity of trees and slow rate of ecological responses in forests (Nitschke & Innes, 2008; Steenberg et al., 2011). The strategic planning scenarios, as well as the effects of the EAB and loss of the municipal ash population, demonstrate these latter issues. An initial decline in both tree size and abundance was observed in all nine scenarios, followed by a subsequent increase after complete ash mortality. While the strategic planning scenarios show a net loss in carbon storage due to this widespread mortality, there is an increasing trend towards the end of the simulations, suggesting future net increases. Conversely, while the control and vulnerability scenarios showed net increases in carbon storage, the net loss in total tree numbers could potentially generate an age-class imbalance over time, with a disproportionate amount of overmature trees and insufficient younger cohorts to replace them. The increased removal (i.e., mortality) of invasive species and increased rates of tree planting in the strategic planning scenarios, while corresponding to initially lower levels of ecosystem service supply, may yield a more structurally diverse and resilient urban forest. These findings reinforce the importance of longer-term considerations in urban forest planning and management necessary to avoid such undesirable time-lag effects.

The internal structure vulnerability was influential on simulated urban forest structure and function. The level of exposure and adaptive capacity tended to have an inverse relationship; where exposure was high, adaptive capacity was low. Based on the selected exposure and adaptive capacity indicators, this relationship supports existing research on the strong positive relationship between resident affluence and tree cover, as well as poor environmental quality in less-affluent, higher-density neighbourhoods (Martin et al., 2004; Grove et al., 2006b; Troy et al., 2007; Pham et al., 2013; Grove et al., 2014). However, the level of sensitivity was less predictable with regards to overall vulnerability. For instance, Class 4 is comprised of typical residential neighbourhoods outside of the urban core (Chapter Two) and had slightly above average adaptive capacity and below average exposure. However, Class 4 also had a high degree of sensitivity due to an abundance of trees in poor condition. While there was certainly some citywide correlation between the level of exposure and sensitivity (e.g., Classes 8, 11, and 12), the sensitivity dimension of vulnerability stresses the complex and uncertain nature of urban forest structure, and the potential for unanticipated loss of function. While the structure of urban tree communities and forest ecosystems are reflective of current biophysical, built, and social conditions (Nowak, 1994), they are also a legacy of past disturbance, planning and management decisions, and development history (Grove, 2009; Boone et al., 2010). It should also be noted that unlike the exposure and adaptive capacity indicators, the sensitivity indicators are derived from plot-based field data, so there is an inherit level of sampling error and structural heterogeneity.

The relationship between urban forest vulnerability, potential impacts, and ecosystem service supply revealed some expected and unexpected findings. It was expected that more vulnerable urban forest ecosystems would be more likely to experience potential impacts (i.e., lower levels or declines of ecosystem service supply). Most often it was found that less vulnerable ecosystem classes had higher levels of ecosystem service supply, as represented by leaf biomass, and vice versa. However, it was frequently found that relative changes in ecosystem service supply over the simulated scenarios were in contrast to this assumed relationship. For instance, both Classes 5 and 9 were associated with higher resident affluence, higher levels of ecosystem service supply, and low vulnerability. Class 5 was found to have high ecosystem service supply and a larger relative increase in supply in the vulnerability scenarios. Conversely, Class 9, which had lower total leaf biomass but high leaf biomass per hectare, showed a considerably smaller relative increase in ecosystem service supply. These findings are a source of uncertainty in the study, but may point to the importance of stocking and environmental adaptive capacity. Despite very similar levels of exposure, sensitivity, and overall adaptive capacity, Class 5 had more open space and forested areas and thus greater capacity for increasing ecosystem service supply.

Future research into urban forest vulnerability might examine alternate approaches to indicator aggregation, such as expert-weighting, which would be valuable in such theory-driven assessment. A limitation of the i-Tree Forecast model is both the ecological and social uncertainty around the loss of entire species and genera from urban forests, such as the ash genus in all model scenarios or the invasive species and Norway maple in the strategic planning scenarios. As these species are removed, it is likely that competitive release and changes in competitive relationships will occur in more naturalized forested areas (Gustafson et al., 2010), which might lessen the decline in ecosystem service supply in the strategic planning scenarios. Moreover, the social components of tree species selection in both public and private tree planting are difficult to predict and model (Greene et al., 2011; Conway & Vander Vecht, 2015). Importantly, this study also makes the assumption of no climate change or alteration to the morphological and socio-demographic conditions of the city in the simulation of these nine scenarios. These are certainly critical considerations for the future planning and management of the urban forest resource (Kenney et al., 2011). However, the intent of this modelling was not to predict future conditions in the City of Toronto, but rather to conduct vulnerability experiments focused on structural and functional changes in urban trees and forests. This can be done through the assumption, abstraction, and aggregation afforded by ecological models that would otherwise be unfeasible (Jørgensen & Bendoricchio, 2001; Landsberg, 2003), which can yield unique perspectives for longer-term strategic planning at multiple spatial scales.

5.5. Conclusions

Urban forest strategic planning, and the development of municipal urban forest management plans, is increasing across North America (Steenberg et al., 2013, Ordóñez & Duinker, 2013; Gibbons & Ryan, 2015). Such advancements are important stages in policy development for ensuring explicit and consistent goals for long-term sustainable urban forest management (Clark et al., 1997). While the experimental scenarios in this study were derived from Toronto's strategic management plan (City of Toronto, 2013b), they represent goals that are widely adopted in urban forest planning and management, such as canopy targets, tree planting goals, and invasive species management (Clark et al., 1997; van Wassenaer et al., 2000; Kenney et al,. 2011). The findings of this study not only reinforce the importance of long-term planning in urban forest research and management, but also the high capacity for management actions to influence the structure and function of urban forest ecosystems. For instance, in the time frame selected for this study, the management goal of increasing tree planting to expand the urban forest resource and remove undesirable invasive species had the effect of lowering ecosystem service supply. Vulnerability assessment and analysis of urban forest ecosystems can provide the strategic planning process with valuable insight in the processes of, and potential risks for, structural and functional change resulting from management intervention.

CHAPTER 6 CONCLUSIONS

6.1. Research Summary and Conclusions

This study investigated the nature of urban forest ecosystem vulnerability at multiple spatial and temporal scales in Toronto, Canada. The approach to assessing and analyzing vulnerability included the development of a conceptual framework, followed by its application using ecological modelling, spatial analysis, multivariate statistics, and empirical field research. Prior to these vulnerability investigations, a framework of urban forest ecosystem classification (UFEC) that integrates ecosystem components characterizing the biophysical landscape, built environment, and human population was developed. It was subsequently applied in the City of Toronto using hierarchical cluster analysis and geographic data to classify city neighbourhoods as urban forest ecosystems. The identification of novel ecosystem classes and their analysis revealed a number of findings and management implications:

- A positive spatial relationship exists between the distribution of tree canopy cover with economic wealth and older, detached housing.
- Some established social-ecological processes, such as the positive association of homeownership and education with urban forest amenities appear to be less pronounced.
- Unlike tree canopy cover, patterns in the distribution of open green space appear to be more a function of building intensity and imperviousness rather than affluence.
- At the neighbourhood scale, there are instances where socioeconomic status and land use appear more influential on tree canopy cover than urban density.
- Ecosystem classification can be employed in the prioritization of management actions, or ‘ecological triage’, including those concerned with social equity in the distribution of ecosystem services.

There remains uncertainty around how to stratify and classify the urban landscape into units of ecological significance at spatial scales best suited to strategic planning and sustainable urban forest management. Neighbourhood-scale ecosystem classification can aid managers with ecosystem-based planning and decision support while also integrating advancements of urban ecology into practice. This could foster a shift towards a model of ecosystem-based management, where in contrast the historical focus in urban forestry has tended towards individual tree care or biophysical ecosystem components only. This stream of research, although not directly concerned with vulnerability, was important for delineating the scope of urban forests as social-ecological systems for the remainder of the study.

The research on urban forest ecosystem vulnerability subsequently began with the development of a theory-based conceptual framework of vulnerability. This framework was derived from research investigating the sustainability of ecosystem service supply in social-ecological systems under conditions of environmental change (Turner et al., 2003a; Adger, 2006), and further refined and adapted for applicability in the urban forest. Review and synthesis of the diverse bodies of literature relevant to urban forest ecology and management, and the succeeding vulnerability framework development and proposal of potential indicators, can be summarized as follows:

- Urban forest vulnerability is defined as the likelihood of decline in ecosystem service supply and its associated benefits for human populations, urban infrastructure, and biodiversity.

- Exposure is a component of vulnerability, and is defined as the stressors and disturbances associated with the urban environment that negatively affect urban tree and ecosystem health.

- Sensitivity is another component of vulnerability, which dictates relative system response to forcing by exposures and is determined by urban forest structure, inclusive of species composition, structure, and tree condition.

— Adaptive capacity is the final component of vulnerability, which is the capacity for a system to shift or alter its state to reduce its vulnerability and is determined by both social and environmental factors.

— Potential impacts are an outcome of vulnerability and are defined as declines or undesirable and destabilizing changes in ecosystem service supply over time.

Maintaining and enhancing urban forest ecosystems are becoming a priority for municipalities as global populations continue to concentrate in cities and urban areas continue to expand (Duinker et al., 2015). The dense human populations and the alteration and degradation of natural environments that characterize cities lead to harsh growing conditions, which make a difficult scenario for tree growth and forest establishment (Konijnendijk et al., 2005). With the complex, heterogeneous, and dynamic nature of urban forest ecosystems, integrative approaches and tools for identifying potential losses in function or undesirable changes in structure like those provided by vulnerability science can be highly valuable for guiding urban forest planning and management. Additionally, the proposed framework can act as a decision-support model for practitioners.

This vulnerability framework was then applied in empirical field research in a residential Toronto neighbourhood to explore the processes of urban forest vulnerability and their influence on ecosystem change among public trees. Linear and logistic regression analyses were used to test the relative influence and predictive capacity of the exposure, sensitivity, and adaptive capacity indicators on tree mortality, condition, diameter growth, and planting rates. The exposure indicators revealed several predictors of ecosystem change and decline:

— The highest exposure and corresponding levels of tree decline and mortality were most influenced by the intensity of land use and the built environment.

— Commercial streets and adjacency to commercial buildings consistently explained higher mortality rates and poor tree condition.

— At this fine spatial scale of assessment, metrics of heavily built-up environments other than land use (e.g., imperviousness, street geometry, and building type) are also important predictors of city tree decline and mortality.

The sensitivity of the urban forest in this neighbourhood was also influential on ecosystem change, while adaptive capacity indicators consistently explained less variation:

— Both smaller trees and in-grown trees were more likely to experience mortality, while poor tree condition was a consistent predictor of tree mortality and slower diameter growth at multiple scales.

— Mature urban forest ecosystems with larger trees may have higher levels of sensitivity and vulnerability, as the strong association of declining condition with increasing tree size was evident throughout the analyses.

— It was expected that adaptive capacity would be more influential on ecosystem change, especially tree planting, though higher rates of planting were associated with homeownership.

— The substantial shift in species composition of recently established trees away from dominant overstory species, combined with the much higher planting rates of smaller ornamental species, suggests that a considerable shift in ecosystem structure and decline in ecosystem service supply is likely.

Much of the understanding of the determinants of urban forest structure and function is derived from broad-scale research investigating variability in two-dimensional tree canopy cover, socioeconomic status, and the built environment (e.g., Grove et al., 2006b; Troy et al., 2007). There is a shortage of studies that examine the processes of ecological change at finer scales using empirical field data. From a vulnerability perspective, there is a need among municipalities, community groups, and individual residents alike to understand how these fine-scale processes interact with urban forest structure in order to reliably predict the likelihood of potential loss in valuable ecosystem services.

The final body of research in this dissertation involved the application of the vulnerability framework at much broader spatial and temporal scales using spatial analysis and ecological modelling. Given the complex social-biophysical interactions and spatial heterogeneity that characterize the urban forest, there are high levels of uncertainty around the impacts of urban forest threats across spatial and temporal scales. This research involved the spatially-explicit assessment of urban forest vulnerability in the UFEC-derived ecosystems and i-Tree Forecast modelling of temporal changes in structure and function under different management and disturbance scenarios. Vulnerability assessment and subsequent modelling produced several notable outcomes:

- The heavily built-up urban core and mixed high-density residential and industrial urban forest ecosystems were the most vulnerable, while the more extensive ecosystems associated with resident affluence and abundant tree cover and open green space were the least.

- At the municipal scale, the effects of the Asian longhorned beetle (ALB) and ice storm disturbance were far less influential on urban forest structure and function than the strategic planning goals of increased tree planting rates and invasive tree species removal.

- The introduction of vulnerability-weighted mortality and strategic planning goals generated substantial mismatches in ecological processes between spatial and temporal scales of assessment. This can translate to both unanticipated loss of function and social inequities in the distribution of benefits if scales of management are misaligned.

- Introducing vulnerability parameters to this prospective modelling increased the spatial heterogeneity in urban forest structure and function while expanding the spatial disparities of resident access to ecosystem services across the urban landscape.

— An inverse relationship changes in total number of trees and total carbon storage was observed during simulation, such that the higher rates of tree planting under the strategic planning scenarios corresponded to declines in stored carbon.

— While lower levels of ecosystem service supply were associated with high urban forest vulnerability and vice versa, there was a variable and uncertain relationship between vulnerability and potential impacts.

Vulnerability assessment and analysis of urban forest ecosystems can provide strategic planning initiatives with valuable insight into the processes of, and potential risks for, structural and functional change resulting from management intervention. Ecological modelling in highly complex and vulnerable social-ecological systems like the urban forest enables the simulation of alternative experimental scenarios at spatial and temporal scales that would not otherwise be feasible (Jørgensen & Bendoricchio, 2001).

These latter two applied bodies of vulnerability research conducted at the spatial scale of individual trees, street sections, ecosystems, and the entire municipality employed both backward-looking monitoring and forward-looking ecological modelling. Moreover, a combination of observation-based, inductive vulnerability analysis in the field with more deductive ecological modelling and vulnerability assessment across the entire city were used. Such a combination of approaches offered complementary opportunities to synthesize and apply existing theory in novel settings while also generating new theories pertaining to the processes of ecosystem change and decline.

6.2. Future Directions

There are several areas of potential future research for the advancement and application of UFEC. An objective of this research was to both make theoretical contributions to the study of urban forest ecosystems and to make a practical contribution to ecosystem-based urban forestry. With regards to the latter, the many unique biophysical, socio-demographic, and historical characteristics of Toronto were evident in the application and classification of ecosystem classes in this study. To increase the generalizability of the UFEC framework, it will be important to refine and apply the framework in cities of different size, cultural makeup, and biogeoclimatic conditions. Individual urban social-ecological systems have many unique qualities (Grove, 2009), and both ecosystem classification (Sokal, 1974) and neighbourhood classification (Reibel, 2011) are inherently subjective processes. Consequently, it will be valuable to embed practitioners and include local knowledge in the identification and interpretation of ecosystem classes in future work.

Given the more novel and exploratory nature of the research on urban forest ecosystem vulnerability, the potential areas for future study and methodological refinement are many. Vulnerability is an abstract concept that cannot be measured directly, and research is nearly always limited by a lack of metrics and available data (Luers et al., 2003). However, for the sake of sustainable management and the amelioration of the negative consequences associated with vulnerable systems, it is necessary to operationalize the concept in some way (Eakin & Luers, 2006). The theory-based conceptual framework and series of quantitative indicators developed in this study are one such approach to operationalization. However, future applications and refinement of these vulnerability metrics could improve their utility. For instance, further experimentation with methods of indicator aggregation, such as expert-weighting, would be valuable in theory-driven assessment and analyses using vulnerability indicators and ecological modelling. Conversely, the empirical vulnerability analyses revealed a number of counter-intuitive relationships between vulnerability indicators and ecological change. Future research might differentiate between potential causes of these discrepancies, such as inadequate indicator design, anomalous conditions in the study area, or potentially untested and scale-dependent social-ecological processes that warrant further investigation. Validation of both the vulnerability indicators and regression models with external datasets from other neighbourhoods and cities, as well as longer-term field monitoring, could also refine this framework further.

It would be advantageous to explore the vulnerability of urban forests with qualitative approaches, such as narrative and scenario analysis, public engagement, and participatory research, which could be used to approach the more subtle and perceived nature of urban forest vulnerability (Cutter, 2003). Moreover, this study adopted a scope of urban forest ecosystem services limited to those that are quantified by the i-Tree models, which are more focused on environmental and economic benefits. An interdisciplinary, mixed-methods vulnerability research agenda might broaden its scope to include more ecological benefits (e.g., habitat provision and soil conservation) and social benefits (e.g., recreation and mental wellbeing).

Lastly, the use of both alternative methods and additional models to address changes in climate, urban morphology, and socio-demographics would be a logical next stage of urban forest vulnerability research.

Given that much of this dissertation is centered on the development of conceptual models/frameworks for urban forest research and their subsequent application in the City of Toronto, it is important to discuss their transferability and generalizability. From the simplest perspective, the conceptualization, assessment, and analysis of vulnerability can be substantively influenced by local biogeoclimatic conditions in a given city. For instance, in grassland or arid ecoregions both drought as a source of exposure and irrigation as a source of adaptive capacity can influence urban forest ecosystem structure and function. However, costs associated with the irrigation of trees in such regions can outweigh energy savings associated with tree shading (Nowak & Dwyer, 2007). This latter point has implications for the overall definition of ecosystem service supply, associated benefits, and vulnerability. Moreover, many benefits and vulnerabilities associated with urban forest ecosystem services are also socially constructed and dependent upon personal values and ethnocultural backgrounds. Since much of the research and theory used in the development of this vulnerability research was derived from the Canadian and American literature base, the transferability of this study will also be limited to some degree in other global regions. Future research, therefore, could adapt and advance the urban forest vulnerability discourse according to cultural and biogeoclimatic variability.

6.3. Concluding Comments

With much of the global population shifting toward city living, and urbanization rates on the rise, an understanding of urban ecosystems and their interaction with society is critical, though challenging (Alberti et al., 2003). The urban forest is a social-ecological system that provides city inhabitants with a substantial array of environmental, social, and economic benefits (Nowak & Dwyer, 2007). However, the complexity and uncertainty surrounding its sustainable management necessitate continued research on the diverse array of social and biophysical drivers of ecosystem change and potential decline in function. This study addresses these needs through the classification of urban forest ecosystems and the subsequent exploration of possible sources of vulnerability arising from their biophysical, built, and human components. Ultimately, it is hoped that this study can contribute to the advancement in understanding of urban forest ecosystems and their vulnerability, while also providing practical knowledge and tools for the sustainable management of this vital resource.

APPENDICES

Appendix A – Urban forest ecosystem classification (UFEC) synoptical tables describing conditions in the 12 ecosystem classes in the City of Toronto

Table A1. Mean value and standard deviation of variables in the biophysical landscape category across the 12 ecosystem classes.

Variable	Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7	Class 8	Class 9	Class 10	Class 11	Class 12[1]
CNPY	15.9 (±4.3)	30.6 (±10.2)	30.8 (±8.7)	31.7 (±6.6)	42.7 (±10.0)	15.1 (±2.9)	19.0 (±7.6)	31.5 (±6.7)	41.8 (±8.5)	28.2 (±9.4)	10.3 (±3.8)	34.3
GRSS	22.2 (±5.3)	24.9 (±7.1)	23.6 (±5.6)	24.9 (±3.7)	23.1 (±6.0)	19.1 (±6.1)	21.0 (±2.3)	20.1 (±2.8)	18.6 (±3.5)	17.6 (±2.9)	10.7 (±5.2)	25.2
STCK	41.8 (±9.2)	54.5 (±14.1)	56.2 (±9.0)	55.6 (±8.1)	64.4 (±10.4)	45.0 (±8.5)	45.8 (±7.8)	60.8 (±3.5)	68.7 (±6.7)	60.1 (±9.5)	49.9 (±4.7)	57.6
SP-1	9.6 (±8.3)	5.4 (±4.5)	13.1 (±10.5)	5.2 (±5.7)	15.7 (±22.2)	3.9 (±3.3)	22.4 (±14.7)	2.7 (±4.6)	13.7 (±10.1)	9.6 (±10.4)	13.3 (±11.5)	18.8
SP-2	80.0 (±17.2)	93.8 (±5.0)	72.9 (±14.9)	92.1 (±7.1)	78.2 (±23.8)	91.5 (±5.9)	52.0 (±21.2)	91.6 (±12.9)	71.1 (±17.8)	74.5 (±24.1)	78.2 (±8.2)	64.7
SP-3	10.4 (±10.3)	0.8 (±1.4)	13.9 (±12.8)	2.7 (±3.5)	6.5 (±8.4)	4.6 (±4.4)	25.6 (±9.6)	5.6 (±8.8)	15.2 (±8.3)	15.9 (±17.3)	8.5 (±3.6)	16.5
SLOP	2.9 (±0.6)	4435 (±0.9)	5.6 (±1.0)	4.4 (±2.0)	7.4 (±4.7)	2.6 (±0.6)	4.0 (±1.3)	7.2 (±1.9)	4.7 (±1.3)	3.75 (±1.7)	2.1 (±0.1)	6.4
SL-1	92.5 (±13.7)	13.2 (±15.8)	44.9 (±22.4)	90.5 (±10.7)	60.4 (±27.8)	9.4 (±9.8)	6.45 (±15.6)	86.3 (±8.5)	6.9 (±7.8)	77.6 (±20.7)	63.7 (±35.6)	42.3
SL-2	2.3 (±5.7)	0	0	0.3 (±1.2)	6.0 (±9.4)	62.1 (±30.9)	0	0	0	15.0 (±18.6)	26.9 (±23.4)	0
SL-3	2.4 (±8.7)	70.3 (±19.1)	17.2 (±17.5)	1.1 (±4.1)	8.7 (±16.7)	0	92.0 (±20.1)	1.7 (±3.3)	91.0 (±12.4)	0	0	35.1

[1] Class 12 is comprised of a single neighbourhood.

Table A2. Mean value and standard deviation of variables in the built environment category across the 12 ecosystem classes.

Variable	Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7	Class 8	Class 9
BLDG	35.2	30.3	31.4	27.9	26.2	43.4	36.0	30.4	30.6
	(±3.2)	(±5.2)	(±4.2)	(±3.3)	(±3.6)	(±7.8)	(±8.3)	(±5.0)	(±3.2)
POPD	4,452.9	4,320.1	4,290.8	5,233.8	3,088.4	7,993.2	5,465.6	6,069.5	7,020.4
	(±2,514.1)	(±2,081.5)	(±1,561.9)	(±2,522.8)	(±1,009.0)	(±5,499.8)	(±2,718.9)	(±1,905.3)	(±3,932.5)
BLCK	45,100.4	54,073.7	52,255.3	47,120.7	42,675.3	26,358.1	31,980.1	109,333.0	21,992.3
	(±27,640.0)	(±24,262.5)	(±26,599.1)	(±16,869.0)	(±18,154.2)	(±13,952.1)	(±27,651.9)	(±77,279.2)	(±5,253.6)
LU-1	44.5	54.6	47.5	62.4	61.	38.0	60.6	27.7	76.6
	(±10.7)	(±17.0)	(±13.7)	(±11.5)	(±12.1)	(±21.2)	(±19.1)	(±15.3)	(±8.3)
LU-2	3.4	2.2	2.8	3.0	0.6	3.7	1.7	1.2	0.8
	(±3.1)	(±2.2)	(±2.1)	(±2.3)	(±0.5)	(±7.6)	(±2.4)	(±2.0)	(±1.3)
LU-3	4.2	6.9	5.5	7.7	5.1	8.6	6.1	2.6	5.5
	(±2.3)	(±5.5)	(±3.1)	(±4.5)	(±3.9)	(±4.7)	(±5.6)	(±2.5)	(±2.5)
LU-4	34.2	14.8	17.2	7.8	5.1	26.5	17.8	32.9	5.1
	(±8.0)	(±11.2)	(±12.5)	(±6.9)	(±5.1)	(±10.5)	(±19.0)	(±9.3)	(±3.4)
SDET	114.8	111.7	114.3	110.8	202.4	33.9	124.5	38.1	138.2
	(±59.8)	(±63.2)	(±43.5)	(±46.7)	(±52.2)	(±51.1)	(±44.9)	(±34.0)	(±69.9)
APT5	76.4	129.0	130.6	173.6	99.9	249.6	79.1	269.6	181.7
	(±48.8)	(±53.5)	(±69.0)	(±67.0)	(±67.5)	(±150.3)	(±52.0)	(±54.7)	(±145.2)
PC-1	594.5	296.3	351.6	166.9	717.5	1,112.0	693.2	158.5	1,838.1
	(±862.8)	(±503.8)	(±301.4)	(±202.6)	(±767.5)	(±629.1)	(±695.6)	(±139.4)	(±646.1)
PC-2	1,811.2	2,488.8	2,438.3	2,622.7	2,496.7	1,572.9	2,328.9	2,763.2	2,150.7
	(±818.0)	(±546.8)	(±324.5)	(±655.9)	(±737.7)	(±726.3)	(±459.9)	(±180.3)	(±951.6)
PC-3	1,039.7	812.2	832.8	870.8	722.2	1,884.5	641.5	852.2	723.3
	(±577.5)	(±381.7)	(±327.3)	(±667.7)	(±392.8)	(±1,568.6)	(±322.1)	(±264.2)	(±371.4)

Table A2 (Continued). Mean value and standard deviation of variables in the built environment category across the 12 ecosystem classes.

Variable	Class 10	Class 11	Class 12[1]
BLDG	41.7 (±8.1)	53.2 (±10.0)	30.4
POPD	8,137.3 (±1,715.4)	22,393.5 (±16,616.3)	1,165.8
BLCK	13,979.0 (±3,445.8)	15,138.1 (±6,070.1)	142,349.3
LU-1	69.5 (±14.0)	55.6 (±17.0)	16.20
LU-2	1.0 (±2.0)	6.2 ±6.1)	0.3
LU-3	10.4 (±12.3)	19.5 (±6.4)	4.5
LU-4	8.4 (±7.6)	10.4 (±1.5)	14.3
SDET	85.3 (±72.6)	0.3 (±0.3)	167.5
APT5	85.1 (±87.3)	523.5 (±64.5)	20.7
PC-1	2,463.4 (±438.7)	473.1 (±202.9)	16.2
PC-2	1,422.9 (±502.2)	2,796.0 (±1,011.3)	620.5
PC-3	482.4 (±384.7)	2,392.6 (±1,544.4)	2,204.2

[1] Class 12 is comprised of a single neighbourhood.

Table A3. Mean value and standard deviation of variables in the human population category across the 12 ecosystem classes.

Variable	Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7	Class 8	Class 9
INCM	54,718.7	56,564.2	51,263.8	52,926.0	84,873.1	42,729.0	50,291.5	42,409.2	80,095.1
	(±6,668.0)	(±18,849.9)	(±12,572.3)	(±9,693.6)	(±39,069.9)	(±15,411.8)	(±3,9797)	(±8,021.3)	(±24,666.8)
UNIV	2,221.1	2,299.6	1,833.0	2,774.4	3,387.5	3,067.3	1,906.9	2,980.0	5,001.7
	(±596.4)	(±1,333.2)	(±1,255.8)	(±878.2)	(±1,046.9)	(±1,211.9)	(±660.5)	(±710.0)	(±501.6)
OWNR	66.9	52.7	56.4	57.5	70.2	40.1	58.1	33.9	49.1
	(±12.7)	(±10.7)	(±15.0)	(±14.2)	(±14.5)	(±20.0)	(±10.4)	(±13.2)	(±18.7)
IMGT	5,633.7	5,165.5	5,480.3	5,687.5	3,409.8	4,090.4	5,502.8	6,487.0	2,702.1
	(±1,176.6)	(±1,113.9)	(±526.3)	(±734.7)	(±716.4)	(±1,036.9)	(±580.6)	(±701.2)	(±739.0)
MNTY	5,677.4	4,872.7	5,401.2	5,690.6	1,933.3	4,286.7	3,717.4	6,297.2	1,548.2
	(±2,680.6)	(±2,164.6)	(±1,110.6)	(±1,410.1)	(±716.0)	(±1,921.2)	(±838.5)	(±1,814.1)	(±680.0)

Table A3 (Continued). Mean value and standard deviation of variables in the human population category across the 12 ecosystem classes.

Variable	Class 10	Class 11	Class 12[1]
BLDG	58,099.4 (±9,713.0)	41,393.2 (±10,598.3)	76,782.2
POPD	3,333.1 (±957.0)	4,757.3 (±1,553.0)	2,311.8
BLCK	55.3 (±12.6)	21.3 (±15.4)	86.6
LU-1	3,445.0 (±731.7)	4,589.7 (±1,235.1)	5,522.0
LU-2	2,500.3 (±834.8)	5,119.6 (±1,668.4)	7,505.0

[1] Class 12 is comprised of a single neighbourhood.

Appendix B – Indicator Sources

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Appendix C – Vulnerability Indicator and Index Values

Table C1. Exposure indicator values and aggregated exposure index assessed in each of the 12 ecosystem classes.

Indicator	Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7	Class 8	Class 9	Class 10	Class 11	Class 12
Imperviousness (%)	59.0	44.5	43.0	42.5	32.2	58.4	61.1	47.1	37.0	52.9	82.2	21.1
Population density (population/km^2)	2,995	3,366	4,010	4,589	2,748	4,549	4,219	5,703	6,001	7,728	15,725	1,148
Building intensity (%)	34.5	30.2	31.9	27.9	26.7	41.9	37.2	31.2	29.9	41.3	56.2	30.4
Commercial/industrial land use (%)	39.2	19.9	19.4	11.4	6.1	34.3	27.6	33.8	5.5	9.9	18.1	14.6
Building height (m)	5.3	5.3	5.2	5.0	4.9	7.8	5.2	6.4	6.6	5.8	30.9	5.8
Street width[1]	0.93	0.92	0.92	0.92	0.91	0.92	0.93	0.93	0.89	0.90	0.93	0.94
Vehicular traffic (vehicles/24 hr)	34,456	38,276	33,269	35,549	33,054	25,369	34,912	36,649	34,369	25,647	30,267	22,298
Pedestrian traffic (individuals/24 hr)	1,505	1,863	1,593	2,008	2,221	5,986	2,335	1,848	4,103	5,460	22,178	704
Exposure index	0.64	0.55	0.54	0.51	0.45	0.66	0.62	0.63	0.51	0.57	1.00	0.42

[1] Mean width of streets, estimated as the ratio of total street length (m) to total street area (m^2).

Table C2. Sensitivity indicator values and aggregated sensitivity index assessed in each of the 12 ecosystem classes.

Indicator	Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7	Class 8	Class 9	Class 10	Class 11	Class 12
Species diversity (H’)	2.98	2.95	2.35	3.35	3.05	2.75	3.04	2.56	2.38	2.81	0.69	3.22
Structural diversity (H’)	2.52	2.35	1.88	1.99	2.13	2.06	1.99	1.60	1.75	2.46	0.69	2.00
Tree condition (% dieback)	5.1	12.4	7.4	16.1	8.6	1.8	4.3	14.3	1.9	5.4	0.0	9.2
Sensitivity index	0.28	0.63	0.67	0.80	0.51	0.31	0.37	0.98	0.47	0.34	1.00	0.54

Table C3. Adaptive capacity indicator values and aggregated adaptive capacity index assessed in each of the 12 ecosystem classes.

Indicator	Class 1	Class 2	Class 3	Class 4	Class 5	Class 6	Class 7	Class 8	Class 9	Class 10	Class 11	Class 12
Income ($)	62,236	68,456	58,914	60,416	103,097	58,089	60,630	48,300	116,721	78,975	57,081	80,877
Housing value ($)	406,289	372,127	425,865	421,619	552,587	458,872	463,730	409,493	952,521	573,562	554,017	425,544
Homeownership (%)	66.8	52.2	54.9	57.2	67.7	42.9	56.1	34.6	45.7	52.7	23.1	86.6
Education (individuals/ 10,000 people)	2,082	2,109	1,737	2,635	3,236	3,061	1,871	2,817	4,748	3,294	4,298	2,099
Open green space (%)	24.7	26.0	23.7	25.1	23.9	22.2	20.8	20.1	18.9	17.5	8.9	25.2
Tree canopy (%)	14.9	28.6	32.0	31.6	42.3	15.5	17.8	31.9	44.0	29.1	8.9	34.3
Forested area (%)	1.8	7.1	10.5	6.8	12.9	2.0	2.3	12.3	6.0	2.6	0.0	24.2
Adaptive capacity index	0.64	0.72	0.73	0.75	0.97	0.62	0.61	0.70	1.00	0.73	0.51	0.97

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