A Conceptual Framework of Urban Forest Ecosystem Vulnerability

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**A CONCEPTUAL FRAMEWORK OF URBAN FOREST
ECOSYSTEM VULNERABILITY**

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Abstract: The urban environment is becoming the most common setting in which people worldwide will spend their lives. Urban forests, and the ecosystem services they provide, are becoming a priority for municipalities. Quantifying and communicating the vulnerability of this resource are essential for maintaining a consistent and equitable supply of these ecosystem services. We 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 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 refers to the stressors and disturbances associated with the urban environment that negatively affect ecosystem function, 2) sensitivity, which is determined by urban forest structure and dictates the system response to forcing from exposures and the magnitude of potential impacts, and 3) adaptive capacity, which is the social and environmental capacity of a system to shift or alter its conditions to reduce its vulnerability or to improve 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 to be measured and assessed. Vulnerability can be communicated through the use of indicators, aggregated indices, and mapping. A vulnerability approach can communicate complex issues to decision-makers and advance the theoretical understanding of urban forest ecosystems.

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

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 exceeds human population growth (Alig et al. 2004). Municipalities and city residents are consequently directing their focus on maintaining and enhancing urban forest ecosystems and the array of beneficial ecosystem services they provide (Clark et al. 1997; Kenney and Idziak 2000). Urban trees and forests are consequently being recognized as a vital component in the overall sustainability of cities (Grove 2009; Duinker et al. 2015). Documented ecosystem services generated by the urban forest provide a diverse and substantial set of environmental, social, and economic benefits (Nowak and Dwyer 2007). These ecosystem services range from air pollution removal and urban heat moderation to increased real estate values and human health benefits (Ulrich et al. 1991; Nowak and Dwyer 2007; Donovan and Butry 2010). With this growing importance of urban forests to the majority of the global population, both researchers and communities are increasingly focusing on the qualification, quantification, and management of these ecosystem services.

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 tree growth and forest establishment difficult (Nowak et al. 2004; Trowbridge and Bassuk 2004; Konijnendijk et al. 2005). Moreover, there is diversity and conflict in how urban forests, and more broadly urban ecosystems, are defined, modelled, and managed (Konijnendijk et al. 2006). This is largely due to disciplinary divides (e.g., arboriculture, forestry, ecology, geography, urban planning) and the interdisciplinary nature of urban forests in general (Steenberg et al. 2015). 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 forests. For this paper, we define urban forests as the individual trees, forest stands, and associated biotic and abiotic components in a given urban landscape (Miller 1997; Kenney et al. 2011). Our definition also includes the influences of human populations and the built environment on urban forest structure and function (Konijnendijk et al. 2006), which aligns more with the modern ecosystem concept (Pickett & Grove 2009).

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

The purpose of this paper is to adopt a vulnerability science approach to review and synthesize key contributions from disciplines that directly and indirectly address threats to urban forest ecosystems. We 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. We also provide a review of relevant bodies of literature and subsequently identify potential vulnerability indicators that have been applied in past research. Lastly, we review various methods of assessing and analyzing vulnerability, with an emphasis on quantitative, indicator-based approaches. 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.

2. Vulnerability in Social-Ecological Systems

Social-ecological systems are multi-scaled, dynamic systems whose structure and function are shaped by both biophysical processes and human institutions and activities (Berkes and Folke 1998). Most of Earth’s ecosystems are influenced by human populations and social processes to some degree. The concept of social-ecological systems is as much a framing mechanism for interdisciplinary research and environmental problem solving (Grove, 2009; Binder et al. 2013). Correspondingly, the study of social-ecological systems often entails direct focus on linkages between social and ecological processes, the supply of natural resources and ecosystem services, and complex environmental problems (Binder et al. 2013). The latter focus on environmental problems (e.g., climate change) has created a logical intersection with vulnerability science (Turner et al. 2003a).

Vulnerability science is an increasingly used concept and method for approaching issues of sustainability and ecosystem service supply in social-ecological systems (Turner et al. 2003a; Schröter et al. 2005; Adger 2006; Eakin and Luers 2006; Lindner et al. 2010). 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). The concept of vulnerability has a long history within a diversity of disciplines, and there remains variability in terminology, concepts, and methodological approaches arising from the different lineages (Turner et al. 2003a; Eakin and Luers 2006; Cumming 2014). However, these divergences tend to be dependent on the research objectives of a given study (Eakin and 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 (Adger 2006). 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 and 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 (Adger 2006), and issues around natural hazards and underlying social vulnerabilities were bridged early on (Blaikie et al. 1994). 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 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.

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). 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 conditions to reduce its vulnerability or to improve its ability to function while stressed (Adger 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 (Gallopín 2006). Gallopín (2006) suggests that resilience and adaptive capacity are indeed subsets of the overall coping capacity of a system under stress. Resilience is also gaining popularity as an approach to understanding urban social-ecological systems (Miller et al. 2010), which will be discussed further in Section 5. However, most recent studies investigating 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 and 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 environmental 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, we adapt and expand this framework for application in urban forest ecosystems.

3. Urban Forest Vulnerability Framework

Developing a conceptual framework of vulnerability is an important first step prior to the identification of specific metrics or indicators (Adger et al. 2004). The framework of urban forest vulnerability developed for this study (Fig. 1) builds on the widely used approach introduced by Turner et al. (2003a). In their conceptualization of vulnerability in coupled human-environment systems (i.e., social-ecological systems), they propose a local-level framework comprised of exposure, sensitivity, and resilience with external and multi-scale (e.g., local-global) linkages.

Our framework also incorporates concepts from the Advanced Terrestrial Ecosystem Analysis and Modelling (ATEAM) research (e.g., Schröter et al. 2005). The ATEAM project was an international, interdisciplinary research collaboration funded by the European Commission with the purpose of identifying and assessing global change impacts on ecosystem service supply (Schröter et al. 2005; Metzger et al. 2006). Their quantitative, spatially-explicit vulnerability framework was applied by Metzger et al. (2006; 2008) and Lindner et al. (2010) to investigate the vulnerability of ecosystem services in Europe. Lastly, our framework incorporates novel elements of vulnerability unique to urban forest ecosystems that are described throughout the remainder of this section.

We define urban forest vulnerability as the likelihood of decline in ecosystem service supply and its associated benefits for human populations, urban infrastructure, and biodiversity. Building on the aforementioned existing frameworks, urban forest vulnerability is similarly comprised of exposure, sensitivity, and adaptive capacity. Potential impacts are an outcome of system exposure and sensitivity and are described as losses or undesirable changes in ecosystem service supply. For example, a city street lined entirely with ash species (*Fraxinus* spp.) will be more sensitive to an exposure to the emerald ash borer (*Agrilus planipennis*) than a street with greater species diversity. The potential impacts of this exposure to stress are widespread dieback and mortality, corresponding to a loss of the ecosystem services provided by these trees.

Our definition and conceptual framework are derived from research investigating vulnerability to global environmental change in social-ecological systems (e.g., Turner et al. 2003a; Schröter et al. 2005). Where this study differs is that the stressors and disturbances of interest are not climatic variables, but rather those associated with densely-settled urban environments. These might include typical forest disturbances (e.g., wind damage), but also

urban development, alterations to the built environment, and social processes of cities (e.g., policy development and management intervention). 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). The following sections describe the conceptual framework of urban forest vulnerability. We also review and summarize several key determinants of urban forest structure and function from the literature that may represent suitable indicators of vulnerability (Table 1, 2, and 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). Urban forest exposure therefore refers to the stressors and disturbances associated with the urban environment that negatively affect tree condition and ecosystem function and/or cause tree mortality, thereby reducing ecosystem service supply (Table 1). These might range from site-level environmental degradation (e.g., soil compaction, construction activity, and proximity to infrastructure; Koeser et al. 2013) to ecosystem-level stress from the combined effects of density and land use (Konijnendijk et al. 2005).

A great deal of the stress on urban trees can be associated with infrastructure and the built environment (Trowbridge and 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 (e.g., overhead wires) can also be an indirect source of stress due to management practices associated with removing conflicts (Trowbridge and Bassuk 2004). Land use has also been found to be highly influential on both tree mortality and ecosystem structure (Nowak et al. 2004), especially those with greater intensity of use and higher density (e.g., commercial and industrial land uses). Exposure to social stressors associated with both land use intensity and land management practices also cause intentional and unintentional physical damage to trees (Lu et al. 2010).

Pollution and environmental contaminants negatively affect tree biology and urban forest ecological processes. Despite the amelioration of urban air pollution by trees (Nowak and 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 function and urban soils is far more complex. Soil degradation and loss is a frequent scenario in urban areas due 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 condition and growth (Hanks and 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 and Bassuk 2004).

Outside of a city's human population, the primary biological threats to urban trees are from insects and pathogens (Konijnendijk et al. 2005; Laćan and 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 and 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 (*Ophiostoma novo-ulmi*), and more recently the emerald ash borer, which is currently afflicting ash populations in Canada and the United States (Herms and McCullough 2014). The frequency and severity of these biological invasions in urban areas is also projected to increase in the near future.

3.2. Sensitivity

Sensitivity is the degree of system response to forcing from a 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). Urban forest sensitivity is influenced by a variety of factors, including species composition, age structure, and tree condition (Table 2). Ecosystem, species, and genetic diversity are key determinants of urban forest sensitivity to insects and pathogens (Laćan and McBride 2008). Furthermore, trees in poor condition that are already under stress are more susceptible to the effects of insects and pathogens (Armstrong and 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 and Jacobs 1979; Nock et al. 2013). Moreover, tree species are highly variable in their tolerance to urban conditions and poor tree condition due to improper site selection is a common phenomenon (Trowbridge and Bassuk 2004).

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 have far higher associated mortality rates (Roman and Scatena 2011). Arguably, ecosystem-scale urban forest sensitivity to various urban stressors and disturbances is an understudied phenomenon.

3.3. Adaptive Capacity

The adaptive capacity of a social-ecological system is its ability to function while stressed or to adapt its conditions to reduce vulnerability (Adger 2006). It is determined by both inherent environmental and social components (Lindner et al. 2010). The social dimension of adaptive capacity within urban forest ecosystems is in-part a function of the economic wealth and education of city residents and their likelihood of engaging in stewardship activities (Table 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. 2006; Manzo and Perkins 2006; Boone et al. 2010; Pham et al. 2013; van Heezik 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. 2006; Troy et al. 2007). However, while wealth, homeownership, and education are frequently correlated, homeownership and education have a more variable relationship with tree cover and stewardship activities (Pham et al. 2013; Steenberg et al. 2015).

Neighbourhoods with resident associations, community groups, business improvement areas, and other social structures that are aware of urban forest issues are also more likely to engage in stewardship and lobby municipal governments to enhance their urban forest (Martin et al. 2004; Manzo and Perkins 2006; Conway et al. 2011). Homeownership again may also present a more nuanced example of adaptive capacity, as homeowner behaviour regarding landscaping practices can be influenced by neighbourhood-wide trends (i.e., the neighbourhood effect) and the presence of residence associations (Grove et al. 2006; Conway et al. 2011).

With regards to social adaptive capacity, an important distinction exists between citizen- and community-led, bottom-up processes and government-led, top-down processes that also influence urban forests. Government policies and practices also influence the spatial distribution and structure of urban forests through management, regulation, incentive programs, and public education and outreach designed to protect and/or enhance trees and green spaces (Heynen et al. 2006; Conway and Urbani 2007; Kendal et al. 2012). For instance, municipal tree protection by-laws/ordinances that regulate tree removal on private land are in place in many large municipalities (Conway and Urbani 2007). The existence of a municipal urban forestry program and corresponding public investment in urban forests (e.g., tree planting, maintenance, and removal) are especially influential drivers on public property (e.g., streets and parks; Heynen et al. 2006; Kendal et al. 2012). As a result, there can be both spatial heterogeneity and inequalities in the access to urban forest amenities in areas with less public space (Heynen et al. 2006).

We describe environmental adaptive capacity in urban forests 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 higher levels of ecosystem service supply through active management (Troy et al. 2007; Nowak and Greenfield 2012; Pham et al. 2013). Conversely, the 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). Overall ecosystem service supply in a given city is highly influenced by the extent of continuous forest cover located within a city’s parks and undeveloped land (Nowak and 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 and Greenfield 2012). Therefore, the area of continuous forest cover can be seen as an influential component of environmental adaptive capacity.

4. Assessing and Analyzing Vulnerability

There are numerous quantitative and qualitative approaches to assessing and analyzing vulnerability. Qualitative approaches address the more subjective and perceived nature of urban forest vulnerability where quantification is not feasible or desirable (Cutter 2003; Ordóñez and Duinker 2014; Kok et al. 2015). For instance, scenario analysis is a participatory research tool to explore possible future scenarios and has been used in qualitative vulnerability research (Swart et al. 2004). Quantitative, indicator-based vulnerability assessment frameworks are arguably the more common approach and have been used at multiple scales and in multiple regions to assess

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).

Deductive, indicator-based assessments of vulnerability 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, observation-based, data-driven inductive approaches to vulnerability analysis focus on measurable cause-and-effect relationships between stressors and system components. Inductive approaches tend to be more repeatable and objective than deductive approaches. However, they are limited in scale and cannot reveal all vulnerabilities and potential impacts, especially long-term variability and risk. Most comprehensive studies on system vulnerability employ elements of both approaches, though it is valuable to always begin with a defined conceptual framework (Füssel 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). For instance, broad-scale assessments of urban forest vulnerability and inequality might focus on socioeconomic indicators while more localized assessments of invasive species introductions might focus on species composition and diversity indicators. The comprehensive review of determinants of urban forest structure and function in Section 3 (Table 1, 2, and 3) provides possible examples of urban forest vulnerability indicators. However, these are not intended to be a complete set of indicators for vulnerability assessment, as many of them are closely related or scale dependent.

Data availability and measurement feasibility are also important considerations for urban forest vulnerability assessment. Targeting readily available data sources during indicator

selection and design is an important consideration if vulnerability frameworks are to be transferable to practitioners. For instance, indicators of the urban forest adaptive capacity concepts discussed earlier would be well suited to national census data and satellite-derived land cover data for social and environmental adaptive capacity, respectively. Sensitivity indicators would be relatively dependent on field data and the availability of tree inventories. Given the numerous, cumulative, and interactive nature of the stressors and disturbances associated with urban forest exposure, the data needs of associated indicators will likely be more challenging. Several exposure indicators might 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, *a priori* consideration of specific indicator selection and design, data needs, and spatial scale of assessment is important.

Vulnerability is a temporal phenomenon (Adger 2006), and in the case of urban forest vulnerability it relates to the supply of ecosystem services over time. Potential impacts refer to declines or undesirable and destabilizing changes in ecosystem service supply resulting from exposure to external forcing and internal system sensitivity (Lindner et al. 2010). They therefore require either forward-looking ecological modelling or backward-looking monitoring for quantification. Drawing from established tools for managers that are used for monitoring and modelling can assist in approaching spatial and temporal variability in vulnerability and ecosystem service supply.

The selection, design, and implementation of indicators for the purposes of monitoring has a rich history in both research and practice (e.g., forestry, environmental assessment, ecological restoration). Indicator-based monitoring is especially useful for practitioners and policy makers as a more feasible and cost effective way of evaluating temporal change and

trends in managed systems (Rametsteiner et al. 2011). In principle, indicators are variables that are selected for monitoring because they are highly representative of overall system conditions and/or highly sensitive to changes in system conditions (Noss 1990). For example, top predators that require extensive, intact habitats are used as indicator species of broader ecosystem integrity in ecological monitoring (Noss 1990). In the urban forest context, canopy cover and leaf area are often used as an indicators of ecosystem service supply (Kenney 2000). The objective is for indicators to provide insight into the state of a system of interest without having to measure its entirety and to potentially yield an early warning of adverse environmental changes. In industrially-managed forests, criteria and indicators are used to monitor performance-based progress towards sustainability goals (Hall 2001). In criteria and indicator frameworks, indicators are aligned with different criteria of sustainability values and goals relating to the ecological, social, and economic conditions of forests and the forest sector (Hall 2001). The criteria and indicator model could be highly applicable to urban forest vulnerability assessment and monitoring, and indeed Kenney et al. (2011) have developed a criteria and indicator framework for strategic urban forest planning.

Forward-looking modelling is the complement to monitoring and can be valuable for examining potential futures under complex and uncertain conditions, such as those found in cities. Ecological modelling involves assumption, abstraction, and aggregation of system conditions using computer-based simulation models so that management and disturbance experiments can be done at broad spatial and temporal scales (Jørgensen & Bendoricchio, 2001). One model that is applicable to urban forest vulnerability is i-Tree Forecast, which is part of the i-Tree suite of models developed by the United States Department of Agriculture (USDA) Forest Service and simulates future changes in urban forest structure and function based on user

defined-mortality and establishment rates (USDA Forest Service 2013). This model has been used by municipalities to estimate tree planting requirements to meet long-term canopy cover targets under different mortality scenarios (Nowak et al. 2013; Nowak et al. 2014).

The overarching purpose of a vulnerability approach in urban forestry is to communicate complex issues to practitioners, policy makers, and communities in accessible ways.

Consequently, some form of indicator aggregation is commonly used in addition to analyzing individual vulnerability indicators (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 and 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 5.

Lastly, mapping has been shown to be an effective means for communicating vulnerability (O'Brien et al. 2004; Eakin and 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 monitoring and modelling tools, 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 and Luers 2006).

5. Discussion and Conclusions

A prominent focus in municipal urban forest policy and management in North America is on urban forest ecosystem services and their associated benefits (Ordóñez and Duinker 2013; Steenberg et al. 2013). There is less attention on potential threats to urban forest ecosystems, and little discussion of overall system vulnerability (Ordóñez and Duinker 2013). In contrast, there are many studies on ecological disturbance and stressors of urban forests, especially street trees in the research literature (e.g., Jutras et al. 2010; Hauer et al. 2011; Koeser et al. 2013). For instance, 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. Integrating a vulnerability approach into municipal urban forestry programs and policy development could help to bridge some of this gap between research and practice.

The assessment and analysis of vulnerability can also shed light on longer-term processes and unexpected, multi-faceted relationships between ecosystem service supply and risk (Metzger et al. 2006). For instance, residential neighbourhoods with older housing and higher levels of affluence are frequently characterized by large, mature trees and correspondingly high 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 and 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). Conversely, newly-constructed suburban housing developments often have higher affluence and an abundance of open green space where tree establishment is possible (Steenberg et al. 2015), and thus high social and environmental adaptive capacity. However, as new development typically involves land clearing, trees may be absent, small, and/or sparse (Puric-Mladenovic et

455 al. 2000), presenting a scenario of low vulnerability and low levels of ecosystem service supply.
456 These latter examples not only stress the internal variability and complexity of urban forest
457 vulnerability, but also the importance of temporal dynamics and the potential threat of time-lag
458 effects in forest ecosystems associated with disturbance and environmental change.

459 Vulnerability is one of a large number of theoretical frameworks in the body of research
460 on urban social-ecological systems (Grove 2009; Cumming 2014). While there is an increasing
461 need for frameworks to understand and predict the outcomes of intervention through
462 management and policy in these systems, there is a lack of consensus on which are the most
463 effective (Cumming 2014). The sustainability approach is commonly used in urban planning.
464 Early conceptions of sustainability in urban planning saw sustainability as an achievable and
465 persistent state for cities (Ahern 2011). Resilience theory, which recognizes the more dynamic
466 nature of cities, has since become more prominent and has begun to both replace and supplement
467 this mode of sustainability (Ahern 2011). Moreover, resilience is a commonly used term and
468 framework for researching urban social-ecological systems (Carpenter et al. 2005; Miller et al.
469 2010). Resilience is a system's ability to recover from a disturbance and change back to a
470 reference state and/or to maintain that reference state or states while stressed, and has a longer
471 tradition in the natural sciences (Turner et al. 2003a).

472 However, both vulnerability and resilience are fundamentally concerned with the
473 response of complex systems to change and arguably some of their biggest differences are in
474 their disciplinary backgrounds and lexicons (Miller et al. 2010). Importantly, more recent
475 vulnerability research in the arena of global environmental change integrates resilience concepts
476 into a broader definition and conceptual framework of vulnerability. The framework developed
477 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 certainly 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 and Luers 2006). Since it is essentially impossible to characterize the entirety of a system in a research or management context, systems must be generalized and abstracted using tools like indicators and ecological models (Jørgensen and 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 management interventions 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 and 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 and Kelly 2007; Fussel 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 for adaptation 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. 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). However, it will be important to include social perspectives and methodologies in future interdisciplinary vulnerability research and assessments. Quantitative, indicator-based frameworks have the benefits of measurability, comparability, and generalizability. However, qualitative approaches, such as scenario analysis, public engagement, and participatory research, can be used to approach the more subtle, subjective, and perceived nature of urban forest

524 vulnerability (Cutter 2003). Ultimately, the two most important functions of vulnerability
525 frameworks are to communicate complex issues to decision makers and stakeholders and to
526 advance the theoretical understanding around the biophysical, built, and social dimensions of
527 urban forest ecosystems.

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Table 1. Potential indicators of urban forest exposure.

Category	Indicator	Description	Source*
Built environment	Land use	Land uses have variable intensities of use, population densities,	1, 2, 3, 4,
		and building intensities, and are a broad-scale indicator of	5, 6
		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	
	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

Category	Indicator	Description	Source*
	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
	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
	Known existing infestations	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

Category	Indicator	Description	Source*
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 with 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

Category	Indicator	Description	Source*
	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 condition	12, 13, 14, 31, 32
	Precipitation	Both drought events and excessive precipitation adversely affect tree condition 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

*See Appendix A for sources.

Table 2. Potential indicators of urban forest sensitivity.

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

*See Appendix A for sources.

Table 3. Potential indicators of urban forest adaptive capacity.

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
Environmental	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
	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

Category	Indicator	Description	Source*
	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

*See Appendix A for sources.

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Fig. 1. Conceptual framework of urban forest ecosystem vulnerability, adapted from Turner et al. (2003a).

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