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RESEARCH

Forecasting Urban Forest Ecosystem Structure, Function, and Vulnerability

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Abstract The benefits derived from urban forest ecosystems are garnering increasing attention in ecological research and municipal planning. However, because of their location in heterogeneous and highly-altered urban landscapes, urban forests are vulnerable and commonly suffer disproportionate and varying levels of stress and disturbance. The objective of this study is to assess and analyze the spatial and temporal changes, and potential vulnerability, of the urban forest resource in Toronto, Canada. This research was conducted using a spatially-explicit, indicator-based assessment of vulnerability and i-Tree Forecast modeling of temporal changes in forest structure and function. Nine 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 were found, which can translate into unanticipated loss of function and social inequities if not accounted for in planning and management. At the municipal scale, the effects of Asian longhorned beetle and ice storm disturbance were far less influential on structure and function than changes in management actions. The strategic goals of removing invasive species and increasing tree planting resulted in a decline in carbon storage and leaf biomass. Introducing vulnerability parameters in the modeling increased the spatial heterogeneity in structure and function while expanding the disparities of resident access to ecosystem services. There was often a variable and uncertain relationship between vulnerability and ecosystem structure and function. Vulnerability assessment and analysis can provide strategic planning initiatives with valuable insight into the processes of structural and functional change resulting from management intervention.

Keywords Vulnerability assessment · i-Tree forecast · Ecological modeling · Strategic planning · Disturbance · Scale

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Introduction

The benefits derived from urban forest ecosystems are garnering increasing attention in both environmental research and municipal planning agendas (Pincetl 2009; Duinker et al. 2015). City trees help to improve energy efficiency by shading buildings (Sawka et al. 2013), reduce the urban heat island effect (Solecki et al. 2005; Millward et al. 2014), and ameliorate environmental quality by removing air pollution and increasing stormwater retention

(Xiao and McPherson 2002). The diverse array of ecological, social, and economic benefits provided by urban forest ecosystems (Nowak and Dwyer 2007) have prompted a growing number of municipalities to develop tree protection policies and strategic urban forest management plans (Ordóñez and Duinker 2013; Steenberg et al. 2013; Gibbons and 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 fragmented, frequently changing, and densely-settled environments with complex ownership regimes and high levels of competition for space (Trowbridge and Bassuk 2004; Konijnendijk et al. 2005). Consequently, urban forests commonly suffer high levels of stress and disturbance.

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 and McBride 2008; Berland and Elliot 2014). These include long-standing biological stressors like the Dutch elm disease (*Ophiostoma novo-ulmi*; Smalley and 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; Berland and Elliot 2014; Herms and 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 and Duinker 2014), as well as isolated severe weather events (Lopes et al. 2009; Hauer et al. 1993, 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 and 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 (Grove 2009). 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). The temporal dynamics of urban forest structure and function in particular remain an area of uncertainty. Several studies have investigated temporal changes in structure and function using sample plot re-measurements from multiple time periods (Nowak et al. 2004, 2013a; Lawrence et al. 2012; Tucker Lima et al. 2013). Such studies continue to increase the understanding of social-ecological determinants of structure and function, including tree mortality, establishment, and growth rates. However, the temporal dynamics of urban forest ecosystems remains a critical area of study, especially regarding the effects of multiple and interacting stressors and disturbances.

Vulnerability science can provide an approach for integrating the biophysical, social, and built dimensions of urban forest stress and disturbance, shifting the focus beyond an impacts-only perspective to a more holistic view of the entire ecosystem and its structure and function (Wickham et al. 2000; Turner et al. 2003a; Adger 2006; Steenberg et al. 2016). The widely-adopted Turner et al. (2003a) framework defines vulnerability 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). Quantitative, indicator-based vulnerability assessments in particular 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). More recently, the Turner et al. (2003a) framework was adapted and refined by Steenberg et al. (2016) to define and conceptualize urban forest ecosystem vulnerability, which is defined as "…the likelihood of decline in ecosystem service supply and its associated benefits for human populations, urban infrastructure, and biodiversity" (Steenberg et al. 2016, p. 2).

The temporal nature of vulnerability frequently necessitates some form of ecological modeling to forecast potential future scenarios of change (Eakin and Luers 2006). Moreover, ecological modeling 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 and Bendoricchio 2001; Landsberg 2003). This latter capacity of modeling 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 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 and Duinker 2013; USDA Forest Service 2013). The i-Tree Forecast model has been developed to simulate temporal changes in urban forest structure and function and can therefore be used to investigate future urban forest vulnerability. It is driven by three core model processes that simulate tree growth,

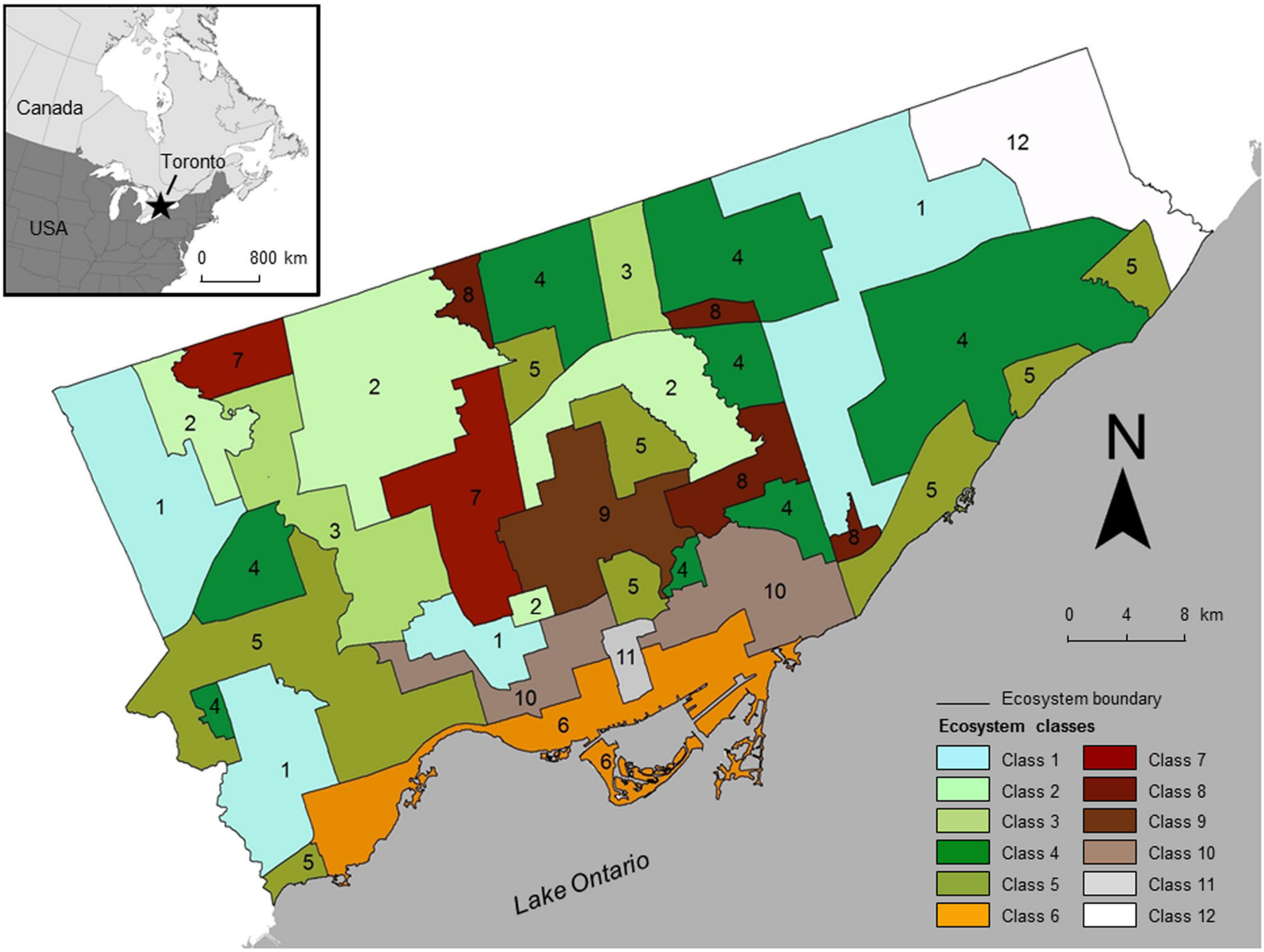

Fig. 1 The City of Toronto, Canada, showing the 12 urban forest ecosystem classes (Steenberg et al. 2015)

mortality, and establishment rates, and uses i-Tree Eco methods (Nowak et al. 2008, 2014) to estimate annual structure and function of the urban forest.

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 based on current conditions; (2) model temporal changes in urban forest structure and function under different management and disturbance scenarios using the newly-developed i-Tree Forecast model; and, (3) identify potential future losses in function to assess overall, long-term system vulnerability at the ecosystem and municipal scale. As the global population continues to concentrate in cities (United Nations 2014), reliance on ecosystem services, and their associated benefits, provided by urban trees and 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.

Methods

Study Area

The City of Toronto (Fig. 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 4151 persons per km^2. In recent years, Toronto has been experiencing both steady population growth (e.g., 4.3 % between 2006 and 2011) and increasing development and densification, both of which are expected to continue into the near future (City of Toronto 2015). Toronto has a continental climate, with a mean July temperature of 22.2 °C, mean January temperature of −4.2 °C, and mean total annual precipitation of 834 mm (Environment Canada 2015). 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*),

Table 1 The 12 urban forest ecosystem classes in the City of Toronto defined by Steenberg et al. (2015)

Class	Description	Canopy cover (%)	Population density (persons km^{-2})	Median family income ($ CAD)
Class 1	Industrial Parkland	14.9	4452	54,718
Class 2	Mixed residential neighborhood	28.6	4320	56,564
Class 3	Mixed residential neighborhood—steep terrain	32.0	4290	51,263
Class 4	Typical residential Neighborhood—newer and outer	31.6	5233	52,926
Class 5	Affluent and forested neighborhood—lower density	42.3	3088	84,873
Class 6	Waterfront hardscapes	15.5	7993	42,729
Class 7	High density residential neighborhood	17.8	5465	50,291
Class 8	Towers in the park	31.9	6069	42,409
Class 9	Affluent and forested neighborhood—higher density	44.0	7020	80,095
Class 10	Typical Residential Neighborhood – Older and Inner	29.1	8137	58,099
Class 11	The downtown core	8.9	22,393	41,393
Class 12	Peri-urban forest	34.3	1165	76,782

Canopy cover (%), population density (persons per km2), and median family income ($ CAD) are included for context

white pine (*Pinus strobus*), and eastern hemlock (*Tsuga canadensis*; Ontario Ministry of Natural Resources 2012). Dominant urban forest species in more built-up and densely-settled areas include Norway maple (*A. platanoides*), white cedar (*Thuja occidentalis*), Manitoba maple (*A. negundo*), and green ash (*F. pennsylvanica*; Nowak et al. 2013b). Toronto's urban forest has recently experienced some major disturbances that are relevant to this study, including a severe ice storm in 2013, an on-going EAB infestation, and an introduction and subsequent eradication of ALB.

Analysis in this study was done at both the municipal and ecosystem scale for the purpose of comparison, where urban forest structural variables were calculated for the entire City of Toronto and for each of its ecosystems, respectively. To analyze at the ecosystem scale, the study area was stratified using the 12 urban forest ecosystem classes developed by Steenberg et al. (2015). They applied an integrated urban forest ecosystem classification (UFEC) framework to classify ecosystems at the neighborhood scale in Toronto according to their biophysical conditions, built environment, and human population (Table 1). See the original publication for a more detailed description of the social-ecological conditions in these 12 urban forest ecosystem.

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 (Table 2). Indicator selection and design was guided by the conceptual framework of urban forest ecosystem vulnerability described by Steenberg et al. (2016) and refined further according to the spatial scale of assessment and data availability. The vulnerability framework applied in this study defines vulnerability as being comprised of exposure, sensitivity, and adaptive capacity (Turner et al. 2003a; Schröter et al. 2005; Steenberg et al. 2016). 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 and 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 and 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 and Bassuk 2004; Tratalos et al. 2007; Jutras et al. 2010; Nagendra and Gopal 2010). 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.

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

[a] Land cover data derived from 2007 QuickBird satellite imagery with 0.6 m per side pixel resolution

[b] Statistics Canada 2006 census data, aggregated from the census tract to ecosystem scale

[c] City of Toronto municipal database, aggregated to the ecosystem scale

[d] DMTI Spatial Inc. 1:1000 land use data

[e] Toronto i-Tree Eco 0.04 ha 2008 plot data, aggregated to the ecosystem scale

[f] Ontario Ministry of Natural Resources 2010 Forest Cover data, derived from 1:10,000 aerial photography

Urban forest sensitivity refers to structural elements of trees and tree communities that influence the magnitude of their response to external stressors and disturbances. For instance, species diversity and structural diversity are important factors influencing the level of response to a number of exposures, such as invasive pests and storm events (Laćan and McBride 2008; Lopes et al. 2009). Diversity indices are frequently used to assess forest ecosystem resilience, and were measured with the Shannon-Wiener Index (H′) using tree species composition and diameter at breast height (DBH) in 5-cm classes (Staudhammer and LeMay 2001). Poor tree condition is associated with higher sensitivity and vulnerability, as it influences both the level of ecosystem service supply and the likelihood of mortality in response to stress (Armstrong and Ives 1995; Koeser et al. 2013). Tree condition is measured as percent crown dieback, where increasing dieback equates to declining condition (Nowak et al. 2008, 2014). The indicator-based assessment of sensitivity to pests, storms, and other disturbances was limited to tree condition and the two diversity indicators. Instead, species- and size-specific sensitivity was addressed in the modeling component of the study. However, future vulnerability research might explore the value of species- and size-specific sensitivity indicators for assessments (Mitchell 1995; Laćan and McBride 2008).

Adaptive capacity in urban forest ecosystems refers to both social and environmental elements that reduce vulnerability or increase capacity to tolerate stress (Luers et al. 2003; Adger 2006; Lindner et al. 2010; Steenberg et al. 2016). Social adaptive capacity indicators measure socio-economic conditions that are influential on 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. 2006; 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 and 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).

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 modeling. Individual indicators were not weighted prior to aggregation in this study. Standardized, aggregated indices of vulnerability and its components, while not without their limitations, are a commonly-used tool in vulnerability research and assessments (Metzger et al. 2006; Birkmann 2007; Steenberg et al. 2016). 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 function 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 modeling under different management and disturbance scenarios described in the Experimental Design and Analysis section.

Model Description and Parameterization

The i-Tree Forecast model was designed by the USDA Forest Service to simulate future changes in urban forest structure and function. The model estimates future changes in the number, size, and species 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. To model tree growth, annual diameter growth rates are estimated by the model based on growing-season length, species-specific growth rates, the level of tree competition, tree condition, and tree maturity. These growth rates are determined by several factors and are used to estimate tree growth and carbon sequestration in the i-Tree Eco model (Nowak and Crane 2000; Nowak et al. 2008). As outlined by Nowak et al. (2008), different base growth rates derived from the literature are first assigned to open-grown and street trees (0.83 cm year^{-1}), park trees (0.61 cm year^{-1}), and forest trees (0.38 cm year^{-1}). Trees are assigned to one of these three categories according to crown light exposure (i.e., number of sides of the tree crown exposed to sunlight, ranging from 0 to 5). The crown light exposure values are 4–5 for open/street trees, 2–3 for park trees, and 0–1 for forest trees (Nowak et al. 2008). This approach was used to account for variability in competition levels across different urban land types. However, it will be important for future research to investigate other approaches to incorporating site and land-type effects on growth rates into the model, such as land use (Lawrence et al. 2012).

In order to adjust open/street, park, and forest growth rates for growing season lengths that differ from the 153 frost-free days used in the i-Tree Eco model, the rates are multiplied by the number of study area frost-free days divided by 153 frost-free days. With the growing season length of 160 frost-free days in the City of Toronto (Nowak et al. 2013b), these growth rates are adjusted to open/street trees: 0.87 cm year^{-1}, park trees: 0.64 cm year^{-1}, and forest trees: 0.40 cm year^{-1}. These growth rates are then adjusted according to tree condition and maturity. Growth rates are adjusted for tree condition whereby trees with less than 25 % dieback are unadjusted, 26–50 % dieback are multiplied by 0.72, 51–75 % dieback are multiplied by 0.42, 76–99 % dieback are multiplied by 0.15, and 100 % dieback (i.e., dead) are multiplied by 0 (Nowak et al. 2008). Growth rates are adjusted for tree maturity so that when tree height reaches 80 % of the average height at maturity for a given species, the growth rate is proportionally reduced each year until it reaches half its original value at the average height at maturity. If a tree is not removed by mortality, growth rates are set to 0 cm year^{-1} when 125 % of the average height at maturity is reached.

Lastly, there is both a limited availability of data describing species-specific growth rates in cities and a high number of species present in cities compared to non-urban forests. Consequently, species-specific variation in growth rates are only accounted for by adjusting the above growth rates into three categories: slow-growing, moderate-growing, and fast-growing species. The growth rates described above are used for the moderate-growing species category. The slow-growing species are 1.67 times less and the fast-growing species are 1.4 times more than the moderate-growing species. While this is a limitation of the study, and of the accuracy of findings relating to diameter growth rates (e.g., carbon storage), these adjustments were made to capture at least some variability in growth rates across species.

Tree mortality was simulated using both user-input mortality rates and fixed rates, according to tree condition. Trees above 50 % dieback are assigned fixed annual mortality rates, so that trees with 50–75 % crown dieback have a

Table 3 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	White cedar	4, 8
Sugar maple	1,025,378	2.39	Sugar maple	4
Norway maple	694,237	4.09	Norway maple	3, 4, 8, 9
Ash	933,978	Eliminated within 10 years	White ash; green ash; European ash (*F. excelsior*)	1, 5
Main invasive species	942,875	13.00	Manitoba maple; tree of heaven (*Ailanthus altissima*); white mulberry (*Morus alba*); Siberian elm (*Ulmus pumila*)	3, 4, 8
Other conifers	1,350,986	1.82	All remaining conifer species	3, 4
Other broadleaves	3,786,245	4.52	All remaining broadleaved species	2, 3, 4, 6, 7, 8

1. City of Toronto (2013), 2. Lawrence et al. (2012), 3. Nowak et al. (2004), 4. Nowak et al. (2013a, b), 5. Poland and McCullough (2006), 6. Roman and Scatena (2011), 7. Staudhammer et al. (2011), 8. Steenberg (2016), 9. Sydnor et al. (1999)

mortality rate of 13.1 %, trees with 76–99 % crown dieback have a mortality rate of 50 %, and trees with 100 % crown dieback have a mortality rate of 100 %. Trees with 0–49 % crown dieback have user-defined annual mortality rates. Mortality rates that are fixed in the model are currently derived from maple street trees in Syracuse, New York (Nowak 1986), which is a limitation (Roman et al. 2016). However, 98 % of the trees in this study had less than 50 % dieback, meaning that simulation was primarily driven by user-input mortality derived from a broader literature base.

For this study, all 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 3). White cedar, sugar maple, and Norway maple are the three most abundant tree species in Toronto (Nowak et al. 2013b) 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 2013; Herms and McCullough 2014). Invasive tree species have been found to have higher mortality rates and conifers have been found to have lower rates (Nowak et al. 2004, 2013a; Steenberg 2016), so invasive species and conifers were assigned mortality rates of 13.0 and 1.82 %, respectively. 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 adjusted by the model according to diameter class to account for size-specific mortality (Nowak 1986), with mortality rates increasing with tree size and a higher rate for the smallest diameter class (i.e., establishment-related mortality). Trees are assigned to one of seven diameter classes based the percentage of the maximum DBH for a given species (Table 4).

Table 4 Adjustment factors for size-specific mortality rates, where diameter class ranges are percentages of species-specific maximum DBH

Diameter class	Adjustment factor	5 % Mortality
0–9.9	0.97	4.85
10–19.9	0.80	4.00
20–39.9	0.73	3.65
40–59.9	0.73	3.65
60–79.9	0.97	4.85
80–99.9	1.00	5.00
100	1.80	9.00

An example of 5 % annual mortality is given

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 2013), depending on the model scenario. Replacement-rate establishment in Toronto equates to 6.65 trees ha^{-1} $year^{-1}$, which is within the range of published, empirical establishment rates. For instance, Nowak et al. (2013a) found an establishment rate of 6.03 trees ha^{-1} $year^{-1}$ in Syracuse, New York, Broshot (2011) found 4.40 trees ha^{-1} $year^{-1}$ in Portland, Oregon, and Lawrence et al. (2012) found a range between approximately 1.25–13.13 trees ha^{-1} $year^{-1}$ across land uses in Gainesville, Florida. After the model was parameterized with the necessary user-input variables described above and the initial conditions were defined, tree total height, crown height, and crown width were subsequently estimated at each time-step using regression equations predicting their relationship to DBH (Online Resource 1: DBH Equations). If no equation exists at the species level, the genus, family, or order level are used as necessary. If the order does not exist, an average of all orders within a class are used. The level of total carbon storage and

Table 5 Experimental design of the three management and three disturbance scenarios for modeling with i-Tree forecast

Scenario	Tree mortality	Tree establishment
1—Control	Annual mortality rates derived from the literature	Annual establishment rate set to species-specific mortality at year one, equaling 426,572 trees $year^{-1}$
	Ash genus eliminated within 10 years[a, b]	No ash genus established[b]
2—Vulnerability	Annual mortality rates derived from the literature and weighted by the vulnerability index in each ecosystem class	Annual establishment rate equal to control scenario
	Ash genus eliminated within 10 years[a, b]	No ash genus established[b]
3—Strategic planning	Annual mortality rates derived from the literature and weighted by the vulnerability index in each ecosystem class	Annual establishment rate set at 570,000 trees $year^{-1}$, based on rate needed to reach 40 % canopy cover[b]
	Ash genus eliminated within 10 years[a, b]	Species-specific establishment rates based on initial composition ratios
	Norway maple, Manitoba maple, tree of heaven, and Siberian elm eliminated within 45 years[b]	No ash genus established[b]
		No Norway maple, Manitoba maple, tree of heaven, or Siberian elm established[b]
A—No disturbance	No change to annual mortality rates	No change to annual establishment rates
B – Asian longhorned beetle	Maple, birch, poplar, and willow genera eliminated within 10 years in Class 1[c]	No maple, birch, poplar, and willow genera established in Class 1[c]
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[d,e]	No change to annual establishment rates
	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[d,e]	

Each of the nine final scenarios represent all possible management and disturbance combinations

[a] Herms and McCullough (2014)

[b] City of Toronto (2013)

[c] Haack et al. (2010)

[d] Hauer et al. (1993)

[e] Hauer et al. (2011)

Table 6 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[a]	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

Dashes indicate that a species group was not present in a given ecosystem class, excluding those used for the ash genus

[a] No annual mortality rates were used for the ash genus, as it was eliminated within 10 years in all scenarios

leaf biomass are estimated using the i-Tree Eco methods described in Nowak et al. (2008).

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 (Nowak et al. 2013b) following the i-Tree Eco measurement protocol (Nowak et al. 2008). The i-Tree Eco plot-based samples were converted to total tree population estimates for each ecosystem class based on stem densities, total plot area in the class, and total area of the class. For model simulation, tree population estimates are grouped by species and then by 2.54-cm diameter class, so that the minimum analytical scale and format of model input/output are unique species-diameter cohorts. In the original i-Tree Eco analysis, the city was post-stratified by land use after random sampling (Nowak et al. 2013b). In the present study, the City of Toronto was re-stratified using the 12 urban forest ecosystem classes (Steenberg et al. 2015). This was done to capture a wider range of the social and ecological processes driving heterogeneity in the urban landscape than is possible with land use alone. A minimum of 20 plots per stratum (i.e., ecosystem class) was targeted. This minimum was met in all ecosystem classes except for Class 9 (18 plots), Class 8 (12 plots), and Class 11 (3 plots), which had the three smallest class areas, respectively. While this is a study limitation, it is expected that sampling error will not be excessive given their smaller area. 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 municipal street tree inventory (City of Toronto 2012).

Experimental Design and Analysis

Three management scenarios were simulated for 45 years. Within each management scenario, three different disturbance scenarios were also simulated, giving nine final experimental scenarios (Table 5). The management scenarios included: (1) a control, with replacement establishment rates and mortality rates derived from the literature (Table 3); (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 (Table 6); 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 2013). 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 2013). 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 (i.e., Class 1) 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, 2011). The ALB was introduced to Ecosystem Class 1, where shipping and trade are more abundant and where an existing ALB outbreak was identified and contained (Haack et al. 2010). The ash genus was

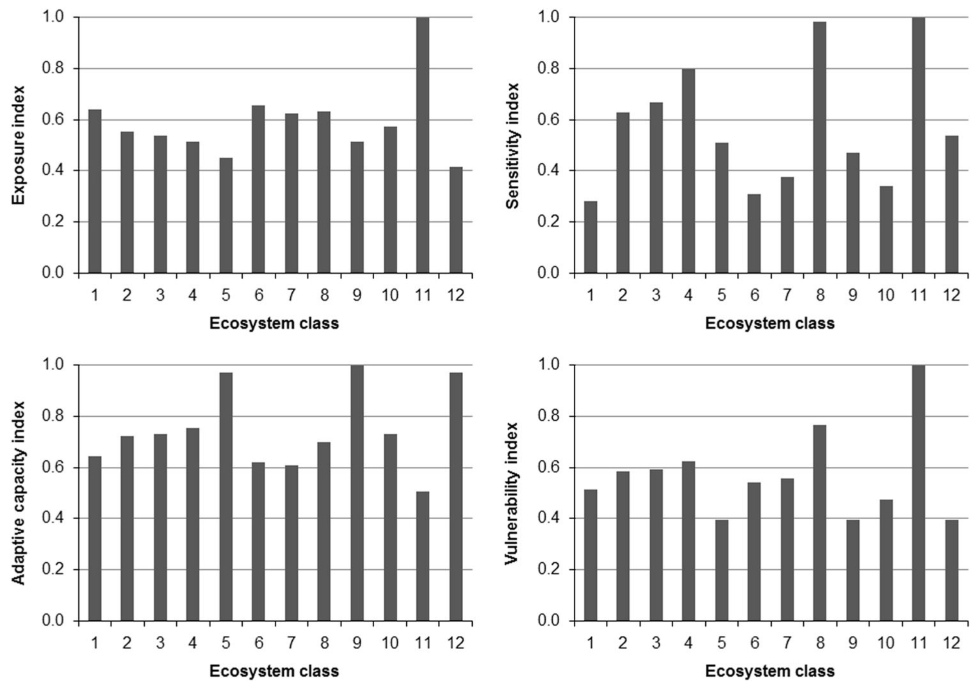

Fig. 2 Exposure, sensitivity, adaptive capacity, and vulnerability index values in each of the 12 urban forest ecosystem classes

removed within 10 years in all nine scenarios to simulate the effects of the already-present EAB and management control thereof (City of Toronto 2013; Herms and McCullough 2014).

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, leaf biomass and its relative change between simulation-year zero and 45 were quantified and mapped at the ecosystem scale. Structural attributes of urban forests like leaf biomass are commonly used to model ecosystem service supply, since leaf biomass is positively associated with the overall level of supply of several noted ecosystem services (e.g., stormwater retention and air pollution removal; Nowak et al. 2008). Leaf biomass and potential impacts were mapped for Scenario 2A only.

Results

Vulnerability Assessment

The vulnerability assessment revealed several patterns across the 12 urban forest ecosystem classes, as shown by the indicator values and vulnerability indices (Figs. 2, 3; Online Resource 2: Indicator Values). The vulnerability index by far had the highest value 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 (i.e., socioeconomic indicators that are positively associated with canopy cover and stewardship), the apartment towers and industrial areas that characterize this ecosystem are commonly adjacent to open green spaces and forested areas, giving moderate overall adaptive capacity index values. Class 1, which is an extensive ecosystem with mixed residential and industrial land uses, was somewhat

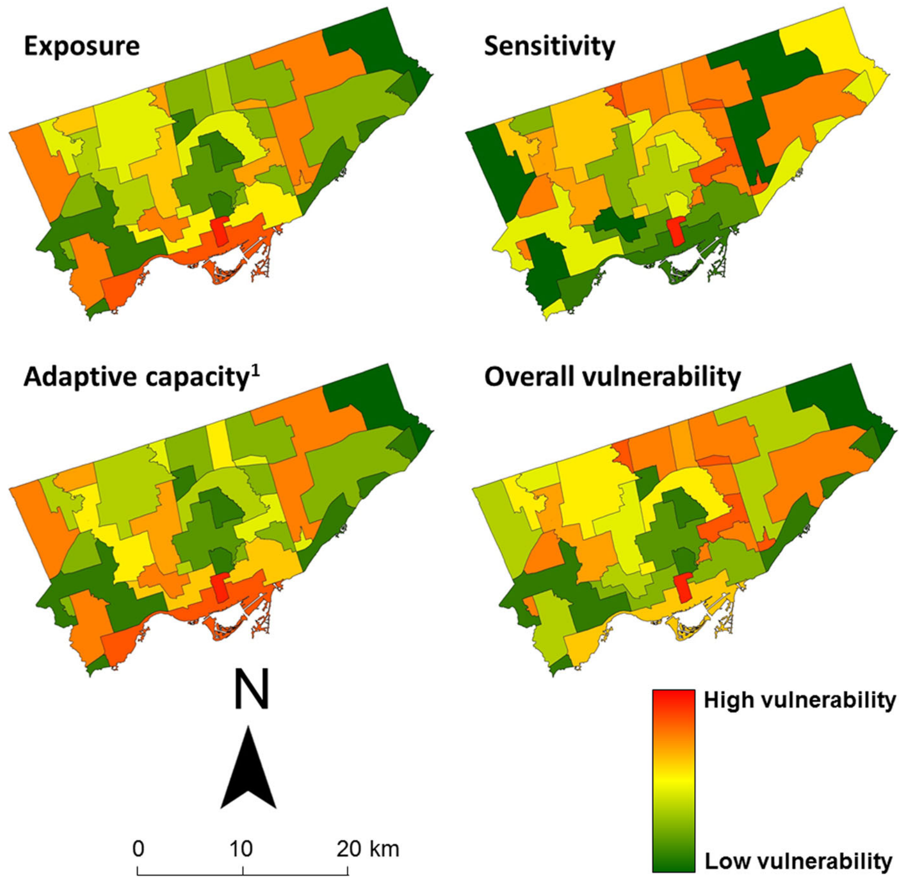

Fig. 3 Urban forest vulnerability assessment results for Toronto, Canada showing indices of exposure, sensitivity, adaptive capacity, and overall vulnerability mapped in the 12 ecosystem classes.[1] The color scheme for adaptive capacity has been inverted to be consistent with the other maps, so that red indicates low adaptive capacity

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 neighborhoods 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 neighborhoods with moderate levels of exposure, sensitivity, and adaptive capacity.

Model Output and Analysis

At the municipal scale, the influence of the disturbance scenarios on urban forest species composition was marginal (Fig. 4). 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. Corresponding increases in the sugar maple, other broadleaves, other conifers, and white cedar groups were observed. 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). 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

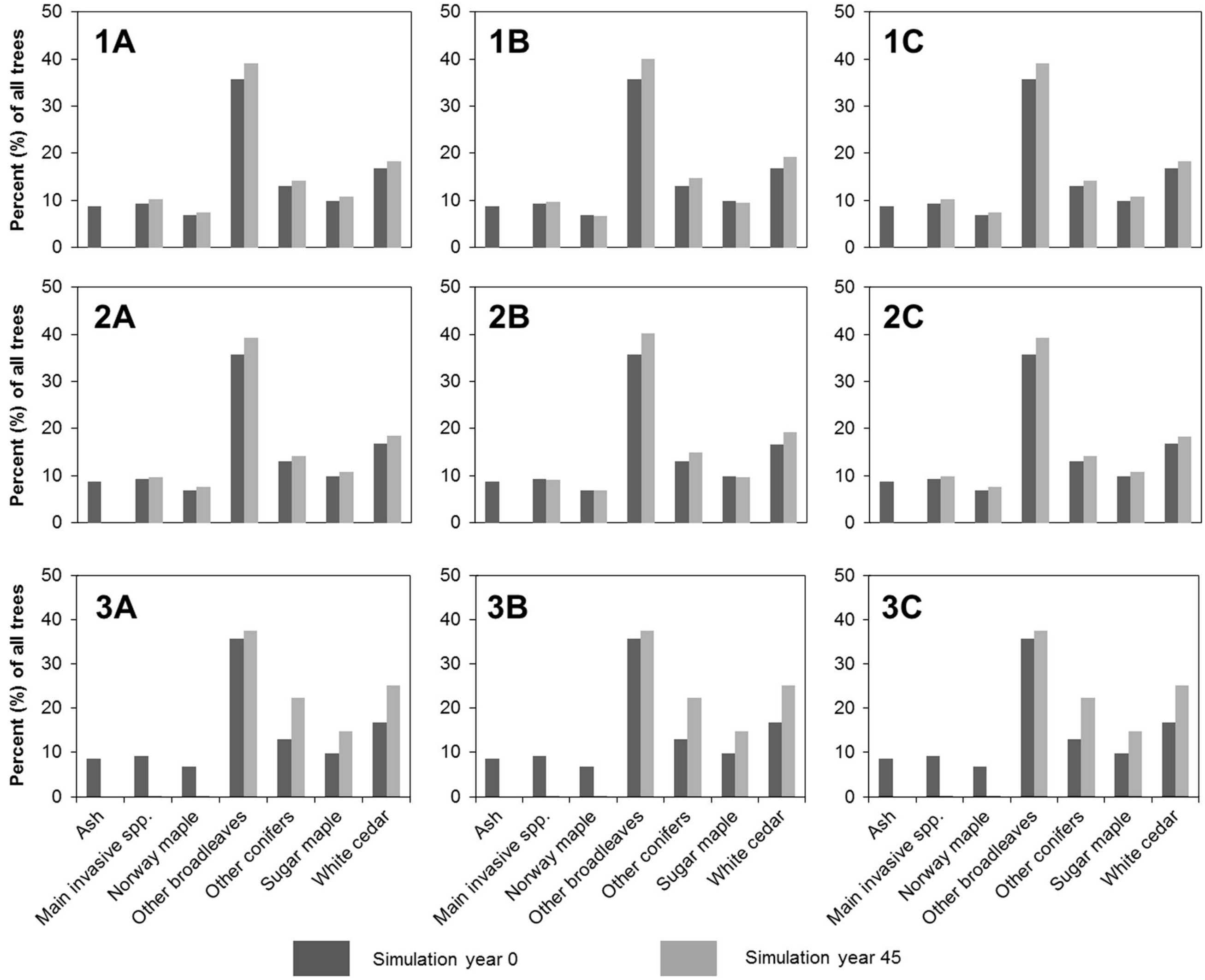

Fig. 4 Changes in the percent of the seven species groups at the municipal scale between simulation-year zero and simulation-year 45 in all nine scenarios. Scenario **1a**: control/no disturbance; Scenario **1b**: control/Asian longhorned beetle; Scenario **1c**: control/ice storm event; Scenario **2a**: vulnerability/no disturbance; Scenario **2b**: vulnerability/Asian longhorned beetle; Scenario **2c**: vulnerability/ice storm event; Scenario **3a**: strategic planning/no disturbance; Scenario **3b**: strategic planning/Asian longhorned beetle; Scenario **3c** strategic planning/ice storm event

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.

While no major differences in urban forest structure and function were observed between control and vulnerability scenarios at the municipal scale, substantial variability existed across classes at the ecosystem scale. Variability in carbon storage (Table 7) was strongly influenced by the level of vulnerability assessed in each ecosystem class. 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

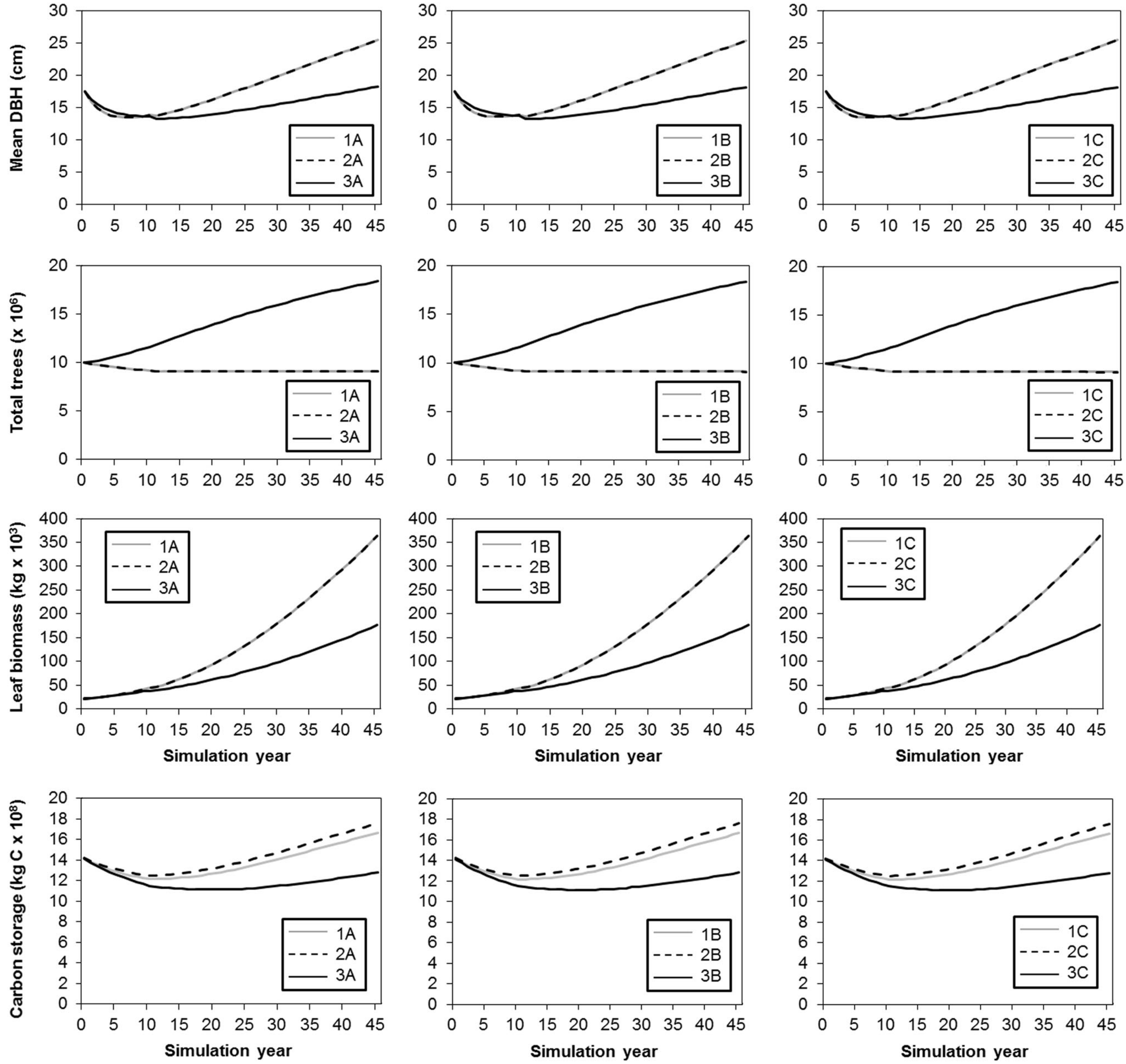

Fig. 5 Temporal dynamics of mean diameter at breast height (DBH; cm), total number of trees, leaf biomass (kg), and total carbon storage (kg C) at the municipal scale over the 45-year simulation in all nine scenarios. Scenario **1a**: control/no disturbance; Scenario **1b**: control/Asian longhorned beetle; Scenario **1c**: control/ice storm event; Scenario **2a**: vulnerability/no disturbance; Scenario **2b**: vulnerability/Asian longhorned beetle; Scenario **2c**: vulnerability/ice storm event; Scenario **3a**: strategic planning/no disturbance; Scenario **3b**: strategic planning/Asian longhorned beetle; Scenario **3c** strategic planning/ice storm event

and vulnerability scenarios, respectively. Unlike carbon storage, leaf biomass increased considerably in all ecosystem classes (Table 8). However, a similar trend in the relationship between leaf biomass and vulnerability was observed, albeit much less pronounced.

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,

Table 7 Change in tree carbon storage (kg C × 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	−16,234	−58,910	−16,299
2	−59,180	−59,180	−58,120	−62,546	−62,546	−61,370	−100,155	−100,155	−97,883
3	57,876	57,876	57,137	50,567	50,567	50,061	−17,894	−17,894	−16,817
4	128,254	128,254	126,842	95,162	95,162	94,683	−3913	−3913	−3414
5	−22,948	−22,948	−29,707	45,634	45,634	34,738	−47,581	−47,581	−52,254
6	−22,591	−22,591	−21,738	−23,414	−23,414	−22,544	−31,450	−31,450	−30,165
7	26,666	26,666	26,692	28,206	28,206	28,223	6498	6498	6738
8	42,895	42,895	42,841	27,091	27,091	26,980	92	92	92
9	−22,946	−22,946	−25,593	−11,278	−11,278	−15,476	−35,940	−35,940	−36,001
10	−29,825	−29,825	−28,766	−21,911	−21,911	−20,993	−55,624	−55,624	−53,450
11	936	936	924	6	6	8	−218	−218	−213
12	97,777	97,777	92,093	145,386	145,386	136,721	170,446	170,446	162,540
Toronto	232,055	145,331	215,140	319,532	224,493	294,479	−131,971	−174,446	−134,798

Scenario 1A: control/no disturbance, Scenario 1B: control/Asian longhorned beetle, Scenario 1C: control/ice storm event; Scenario 2A: vulnerability/no disturbance, Scenario 2B: vulnerability/Asian longhorned beetle, Scenario 2C: vulnerability/ice storm event; Scenario 3A: strategic planning/no disturbance, Scenario 3B: strategic planning/Asian longhorned beetle, Scenario 3C strategic planning/ice storm event

which was the least vulnerable ecosystem. In Class 12, both the carbon storage and total number of trees were observed as net increases.

Again similar to the municipal scale, the most consistent theme at the ecosystem scale in the net change in the number of trees (Table 9) 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.

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. 6) revealed the variable relationship between vulnerability and ecosystem structure. As was expected, leaf biomass and leaf biomass per hectare frequently had negative relationships with the level of urban forest vulnerability, where high leaf biomass values 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.

Discussion

A core focus of this study was variability in the spatial and temporal nature of vulnerability. 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

Table 8 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	14,874	6263	14,796
2	34,309	34,309	34,287	34,309	34,309	34,287	17,116	17,116	17,047
3	25,073	25,073	25,006	25,071	25,071	25,006	10,471	10,471	10,341
4	53,473	53,473	53,386	53,469	53,469	53,381	27,167	27,167	27,026
5	44,741	44,741	44,758	44,780	44,780	44,797	21,026	21,026	20,995
6	19,515	19,515	19,515	19,515	19,515	19,515	8310	8310	8280
7	24,769	24,769	24,762	24,769	24,769	24,762	9851	9851	9840
8	24,171	24,171	24,144	23,893	23,893	23,866	10,114	10,114	10,080
9	23,277	23,277	23,194	23,277	23,277	23,194	10,167	10,167	10,060
10	26,325	26,325	26,312	26,325	26,325	26,309	6659	6659	6618
11	1655	1655	1655	929	929	929	380	380	380
12	36,664	36,664	36,560	36,664	36,664	36,548	18,597	18,597	18,467
Toronto	343,663	330,777	343,223	342,690	329,808	342,238	154,732	146,121	153,931

Scenario 1A: control/no disturbance, Scenario 1B: control/Asian longhorned beetle, Scenario 1C: control/ice storm event; Scenario 2A: vulnerability/no disturbance, Scenario 2B: vulnerability/Asian longhorned beetle, Scenario 2C: vulnerability/ice storm event, Scenario 3A: strategic planning/no disturbance, Scenario 3B: strategic planning/Asian longhorned beetle, Scenario 3C: strategic planning/ice storm event

11) and least (i.e., Class 12) vulnerable ecosystems, the total number of trees and total carbon storage were positively correlated. These findings emphasize the importance of aligning spatial scales of management and planning with appropriate scales of ecological function to avoid unexpected or undesirable ecosystem change (Borgström et al. 2006; Cumming et al. 2006).

The experimental design and model parameterization of this study dictated many of these mismatches in structure and function between the ecosystem and municipal scales. However, it is reasonable to assume such spatial variability in tree mortality rates within municipal boundaries depending on spatial heterogeneity in specific biophysical, built, and socioeconomic conditions. For instance, a high level of variability in mortality rates has been observed across different land uses and land cover types (Nowak et al. 2004; Lawrence et al. 2012; Tucker Lima et al. 2013). This difference would subsequently drive variability in urban forest structure and the corresponding 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 modeling experiment increased the spatial heterogeneity in structure and function while expanding the spatial disparities of resident access to the urban forest across the city. 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 carbon storage and leaf biomass of the entire city. The more extensive, affluent, and less-densely populated residential neighborhoods and peri-urban forests outside of the urban center (e.g., Classes 5 and 12) tended towards lower vulnerability and higher levels of carbon storage and leaf biomass. 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 and 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 and 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 (Boone et al. 2010; Tucker Lima et al. 2013). The strategic planning scenarios,

Table 9 Change in the total number of trees (×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	415	−325	437
2	−58	−58	−57	−58	−58	−57	1217	1217	1231
3	−29	−29	−28	−49	−49	−28	−150	−150	−145
4	−172	−172	−170	−173	−173	−170	1518	1518	1555
5	−153	−153	−150	−155	−155	−152	1488	1488	1499
6	−4	−4	−4	−4	−4	−4	220	220	224
7	−6	−6	−6	−6	−6	−6	177	177	182
8	−70	−70	−69	−93	−93	−91	−83	−83	−80
9	−9	−9	−16	−9	−9	−15	582	582	587
10	−7	−7	−6	−7	−7	−8	39	39	−116
11	0	0	0	−3	−3	−3	−1	−1	−1
12	−158	−158	−157	−159	−159	−164	1621	1621	1628
Toronto	−884	−1284	−877	−935	−1340	−913	7043	6303	7170

Scenario 1A: control/no disturbance, Scenario 1B: control/Asian longhorned beetle, Scenario 1C: control/ice storm event; Scenario 2A: vulnerability/no disturbance, Scenario 2B: vulnerability/Asian longhorned beetle, Scenario 2C: vulnerability/ice storm event, Scenario 3A: strategic planning/no disturbance, Scenario 3B: strategic planning/Asian longhorned beetle, Scenario 3C strategic planning/ice storm event

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 carbon storage and leaf biomass, 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 variable levels of exposure, sensitivity, and adaptive capacity (i.e., the internal composition of vulnerability) were 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 neighborhoods (Martin et al. 2004; Grove et al. 2006; Troy et al. 2007; Pham et al. 2013). However, the level of sensitivity was less predictable with regards to overall vulnerability. For instance, Class 4 is comprised of typical residential neighborhoods outside of the urban core (Steenberg et al. 2015) 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 variability.

The relationship between urban forest vulnerability, potential impacts, and leaf biomass 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 amounts or declines of leaf biomass). Most often it was found that less vulnerable ecosystem classes had higher amounts of leaf biomass, and vice versa. However, it was frequently found that relative changes in leaf biomass over the simulated scenarios were in contrast to this assumed relationship. For

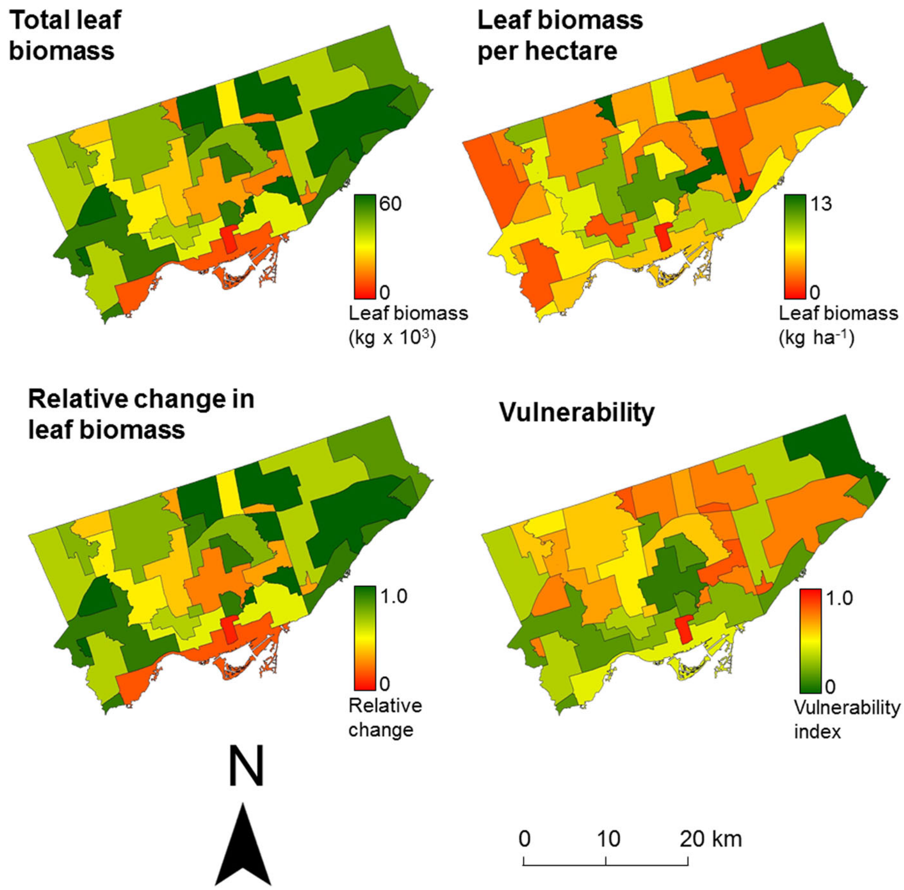

Fig. 6 Top: Total leaf biomass (kg × 10^3) and leaf biomass per hectare (kg ha^{-1}), mapped at simulation-year 45 in Scenario 2A in the 12 ecosystem classes. *Bottom*: the urban forest vulnerability index and potential impacts, represented by the relative change in leaf biomass between simulation-year zero and 45 in Scenario 2A in the 12 ecosystem classes

instance, both Classes 5 and 9 were associated with higher resident affluence, higher amounts of leaf biomass, and low vulnerability. Class 5 was found to have high amounts of leaf biomass and a larger relative increase in biomass in the vulnerability scenarios. Conversely, Class 9, which had lower total leaf biomass but high leaf biomass per hectare, showed a considerably smaller relative increases in biomass. 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 leaf biomass and the overall urban forest resource.

Future research into urban forest vulnerability might examine alternate approaches to indicator aggregation, such as weighting methods, which would be valuable in such theory-driven assessments. Additionally, establishing relative indicator importance in terms of influence on structure and function could help inform future modeling research. One 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 will likely enhance natural regeneration and lessen the decline in carbon storage and leaf biomass 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 and Vander Vecht 2015). Also in regards to the i-Tree Forecast model, total carbon storage in Toronto ranged from 1.17×10^9 kg C to 1.74×10^9 kg C at the end of simulation, depending on the scenario. This range of values is towards the upper end and above urban forest carbon storage values reported for other medium-to-large American cities, which range from 1.93×10^7 kg C in Jersey City to 1.23×10^9 kg C in New York (Nowak and Crane 2002; Timilsina et al. 2014). This suggests that future calibration and refinement of model growth rates would be valuable, although Toronto is a larger city with comparatively high canopy cover.

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 processes are both realities and certainly critical considerations for the future planning and management of the urban forest resource (Kenney et al.

2011). However, the intent of this modeling was not to predict future socioeconomic conditions in the City of Toronto, but rather to conduct vulnerability experiments focused on structural and functional changes in urban trees and forests in response to disturbance and management. This can be done through the assumption, abstraction, and aggregation afforded by ecological models that would otherwise be unfeasible (Jørgensen and Bendoricchio 2001; Landsberg 2003), which can yield unique perspectives for longer-term strategic planning at multiple spatial scales.

Conclusions

Urban forest strategic planning and the development of municipal urban forest management plans are increasing across North America. Such advancements are important stages in policy development for ensuring explicit and consistent goals for long-term sustainable urban forest management. While the experimental scenarios in this study were derived from Toronto's strategic management plan, they represent goals that are widely adopted in urban forest planning and management, such as canopy targets, tree planting goals, and invasive species management. 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 carbon storage and leaf biomass. 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.

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Compliance with Ethical Standards

Conflict of Interest The authors declare that they have no competing interests.

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