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

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

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

**1. Introduction**

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

The built environment is a source of stress for urban trees, especially in higher-density neighbourhoods. Building density, height, and type affect the irradiation (i.e., sunlight available for photosynthesis and plant growth), the physical growing space for trees, and the microclimate of urban areas (Jutras et al., 2010). Moreover, construction activities and conflicts with above- and below-ground utilities and other grey infrastructure are common sources of urban tree decline and mortality (Randrup et al., 2001; Koeser et al., 2013). Land use is highly influential on urban forest ecosystems, and is indeed indicative of the presence of many of these stressors. Land uses with higher populations and building densities, as well as abundant impervious surfaces (e.g., commercial land uses), have higher rates of tree mortality and urban forest decline (Nowak et al., 2004; Lu et al., 2010). Cities are also characterized by high rates of commercial trade, exposing urban trees and forests to invasive insects and pathogens (Laćan & McBride, 2008), such as the emerald ash borer (*Agrilus planipennis*; EAB) and Asian longhorned beetle (*Anoplophora glabripennis*; ALB). These stressors and disturbances can be interactive and cumulative, and their ultimate effect of individual trees and urban forest ecosystems is dependent on tree condition, species, age, and overall species and structural diversity.

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

Research investigating rates and causes of tree mortality and urban forest decline is an important resource for urban forest practitioners. Moreover, the disciplines of ecology, urban planning, and geography continue to explore the dynamics of urban forest ecological change and its relationship with human populations. However, there is a considerable knowledge gap on the combined effects of these stressors and their interaction with urban forest structure. Moreover, there is little research investigating the effects of socioeconomic variability on urban forest ecosystem decline.

Vulnerability science can offer a useful theoretical approach for addressing these gaps in a research context and for bridging the potential contributions of different disciplines that investigate urban forests and their benefits. Current vulnerability science in social-ecological systems is widely held to be a useful approach for exploring issues of sustainability and environmental change in both theoretical and applied research (Turner et al., 2003; Füssel, 2010). It was used in the development of an urban forest vulnerability framework (Steenberg et al., 2015b), where vulnerability is defined as the likelihood of decline in urban forest ecosystem service supply in response to stress, and is comprised of exposure, sensitivity, and adaptive capacity.

Exposure refers to the magnitude, frequency, duration, and spatial extent of stressors and disturbances that affect a system (Burton et al., 1993; Adger, 2006). These are the external causes of tree decline and mortality associated with the urban environment. Sensitivity is the relative level of response by a system to stressors or disturbances, and is determined by intrinsic characteristics of the system itself (Turner et al., 2003). This is the internal structure of urban tree communities, such as species, size/age, condition, and diversity. Adaptive capacity is the capacity for a system to shift or alter its state to reduce its vulnerability or accommodate a greater range in its ability to function while stressed (Adger, 2006; Füssel, 2010). This refers to associated human populations and their behaviours regarding urban forest stewardship, as well as the environmental capacity for urban forest enhancement (e.g., tree planting space). By shifting research focus away from external agents of stress and disturbance only, vulnerability analysis allows for more a comprehensive and integrative mechanism for modelling urban forest structure, function, and change.

The purpose of this study is to explore the processes of urban forest vulnerability and their influence on ecological change. Specifically, the conceptual framework of urban forest vulnerability (Steenberg et al., 2015b) is used to assess 2014 data describing 806 public trees in a residential neighbourhood in Toronto, Canada. The framework consists of a series of quantitative indicators of exposure, sensitivity, and adaptive capacity that describe the built environment and associated stressors, urban forest structure, and the neighbourhood’s human population, respectively. Ecological change is characterized by tree mortality, condition, diameter growth, and planting rates, which were measured using comparisons with existing tree inventory data collected in 2007 and 2008. Regression analysis is used to explore the relative influence and predictive capacity of different indicators of vulnerability on ecological change at the individual-tree and street-section scale. With much of the global population increasingly living in cities and urbanization rates on the rise, on-going research and science-based tools for understanding the causes of urban forest change and decline are essential for developing planning strategies to reduce long-term system vulnerability.

**2. Methods**

*2.1. Study Area*

The study was conducted in a centrally located residential neighbourhood, Harbord Village, in Toronto, Canada. As of 2011, Harbord Village had a population of 8,583, population density of 13,484 persons/km2, and total area of 0.6 km2, and is predominately comprised of semi-detached residential dwellings, with approximately 1,600 households (Keller, 2007; Statistics Canada, 2012). There are commercial land uses along main street sections, with several larger multi-unit and institutional parcels, and three small public parks. Urban forest researchers and Harbord Village residents conducted a tree inventory in 2007 and 2008 in order to inform their strategic urban forest management plan (Keller, 2007). Dominant tree species in the neighbourhood include *Acer platanoides*, *Fraxinus pennsylvanica*, *Gleditsia triacanthos*, *Thuja occidentalis*, *Acer saccharinum*, and *Aesculus hippocastanum*. In-grown *Morus alba*, *Ailanthus altissima*, and *Acer negundo* are also common. Toronto has a continental climate with hot, humid summers and cold winters. The city is within the Deciduous Forest Region and Mixedwood Plains Ecozone (Ontario Ministry of Natural Resources [OMNR], 2012).

*2.2. Data Collection and Processing*

Data collection took place during the growing season of 2014. A total of 806 public street trees and public trees in front-yard rights of way, parks, and schools were inventoried and matched with data from the existing 2007/2008 tree inventory. Residential back-yard trees were omitted from the study due to access constraints. In addition to the standard tree inventory metrics of species, diameter at breast height (DBH), and location, a series of indicators of urban forest vulnerability were assessed for each tree. The design of the urban forest vulnerability assessment framework and selection of indicators is described Steenberg et al. (2015b). Specific indicator selection and design were further refined according to the study’s scale of assessment (i.e., individual trees and street sections), data availability, and feasibility. Ecosystem service supply of the measured trees was quantified using the i-Tree Eco model. Formerly known as the Urban Forest Effects (UFORE) model, i-Tree Eco is the flagship model of the i-Tree software suite that synthesizes a large body of research to quantify several key ecosystem services and their value based on the structure of the urban forest (Nowak & Crane, 2000).

Indicators in the vulnerability assessment framework are assigned to the sub-categories of exposure, sensitivity, or adaptive capacity, depending on the assumed relationship of the phenomena being measured with urban forest structure and function. Exposure indicators (Table 1) represent external stressors and disturbances that cause tree decline and mortality, and subsequently a decline in ecosystem service supply. While some of the exposure indicators represent direct stressors (e.g., vandalism), most characterize indirect relationships between stress and the surrounding environment, all of which have been previously identified as important causes and correlates of tree decline and/or mortality (Randrup et al., 2001; Nowak et al., 2004; Trowbridge & Bassuk, 2004; Jutras et al., 2010; Lu et al., 2010; Lawrence et al., 2012; Koeser et al., 2013). The main data source for exposure indicators was field data collected during this study. Additionally, 2011 census data were used to measure population density and a combination of 2013 orthorectified aerial photography and 2013 City of Toronto property map data were used to measure built area intensity (assessed as building site coverage; the ratio of building footprint to parcel area), distances to nearest buildings, and widths of streets. The binary exposure indicators resulting from the presence/absence of conflicts with infrastructure (Kenny & Puric-Mladenovic, 2001), vandalism, and poor management were measured in the field.

**Table 1.** Description of urban forest vulnerability indicators used to assess system exposure.

|  |  |  |
| --- | --- | --- |
| **Indicator Description** | **Vulnerability Assumption** | **Mean/Count\*****(Standard Deviation)** |
| Population density (persons/km2) | Positive | 14,834 (±8,146) |
| Built area intensity (%) | Positive | 50.2 (±21.2) |
| Land use1 (categorical) |  |  |
| Site type (categorical) |  |  |
| Site size (m2) | Negative | 136.7 (±383.4) |
| Type of nearest building (categorical) |  |  |
| Height of nearest building (storeys) | Negative | 4.1 (±4.5) |
| Distance to nearest building (m) | Negative | 6.7 (±14.2) |
| Distance to street (m) | Negative | 4.1 (±3.0) |
| Width of sidewalk (m) | Positive | 2.7 (±1.9) |
| Width of street (m) | Positive | 11.2 (±6.7) |
| Impervious cover (%) | Positive | 47.3 (±32.1) |
| Light availability2 (ordinal rank; 0-5) | Negative | 2.7 (±1.1) |
| Conflict of overhead utilities (0/1) | Positive | 416\* |
| Conflict with sidewalk (0/1) | Positive | 76\* |
| Conflict with buildings (0/1) | Positive | 259\* |
| Conflict with building foundation (0/1) | Positive | 47\* |
| Conflict with other infrastructure (0/1) | Positive | 294\* |
| Poor management (0/1) | Positive | 172\* |
| Vandalism (0/1) | Positive | 92\* |

1 Land-use designation is based on categories described in the i-Tree Eco manual. Land uses present in Harbord Village include commercial/industrial, institutional, multi-unit residential, park, residential, and vacant.

2 Light availability was measured using crown light exposure, which is a component of the i-Tree Eco measurement protocol.

Sensitivity indicators (Table 2) represent the internal structure of the system, in this case the tree communities measured in the study, and its relative response to exposures. In other words, they are elements of urban forest structure that increase or decrease the likelihood of tree decline and mortality in response to stress. Species and DBH class were included to account for potential variation in the vulnerability of tree species and sizes (i.e., ages). A number of studies have found that mortality rates fluctuate by species and are elevated in younger and newly-planted urban trees (e.g., Roman & Scatena, 2011). Tree condition is another predictor of urban tree mortality (Koeser et al., 2013) and is itself an indicator of sensitivity to stress (Trowbridge & Bassuk, 2004).

In this study, we derive tree condition using an aggregated index calculated from data collected as part of the Neighbour*woods* assessment protocol. This aggregate index has a maximum value of 1.0 indicating the extremely poor tree condition. Neighbour*woods* is a tool for community-based urban forest stewardship that was developed by Kenney and Puric-Mladenovic (2001), which describes a standardized procedure for community members to inventory and monitor the location, composition, and condition of their urban trees. The protocol describes 15 ordinal metrics of tree condition, ranging from 0 (best condition) to 3 (worst condition), giving a total possible score of 45, which was then standardized to produce the condition index. A Neighbour*w*oods assessment was conducted during the 2007/2008 Harbord Village tree inventory and was again conducted for all trees measured in 2014. The tree condition index was calculated for both 2007/2008 and 2014 data.

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

**Table 2.** Urban forest vulnerability indicators used to assess system sensitivity.

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

Adaptive capacity indicators (Table 3) represent components of the urban forest that enable it to reduce its own vulnerability or increase its capacity to tolerate greater change without adverse effects (Adger, 2006). In the context of this study, indicators of adaptive capacity measure socioeconomic variables that are likely to decrease the likelihood of a decline in ecosystem service supply or environmental ones that are likely to increase supply. All social adaptive capacity indicators were measured using 2011 National Household Survey data at the dissemination-area level, excluding presence/absence indicators that were assessed in the field. The environmental adaptive capacity indicators were measured using 2007 land cover data derived from QuickBird satellite imagery with 0.6-m resolution, quantified at the parcel scale (City of Toronto, 2010).

**Table 3.** Urban forest vulnerability indicators used to assess social and environmental adaptive capacity of the system.

|  |  |  |
| --- | --- | --- |
| **Indicator Description** | **Vulnerability Assumption** | **Mean/Count\*****(Standard Deviation)** |
| *Social adaptive capacity* |  |  |
| Median family income ($) | Negative | 54,194 (±11,676) |
| Average dwelling value ($) | Negative | 734,451 (±152,682) |
| Homeownership (%) | Negative | 44.0 (±14.8) |
| Population with a university degree (individuals/10,000 people) | Negative | 4,313 (±1,130) |
| Signs of stewardship1 (0/1) | Negative | 162\* |
| Replanted site2 (0/1) | Negative | 19\* |
| *Environmental adaptive capacity* |  |  |
| Open green space (%) | Negative | 16.7 (±13.4) |
| Existing canopy cover (%) | Negative | 18.0 (±20.3) |

1 Signs of stewardship include direct and obvious actions taken to protect trees or enhance growth (e.g., mulch, bicycle guards, pest protection; Lu et al., 2010).

2 This indicator measured locations where tree mortality had occurred and the site had been replanted since the 2007/2008 inventory.

*2.3. Analysis*

2.3.1. Ecological Change

Four metrics of ecological change were assessed by comparing field data collected for this study in 2014 with the existing 2007/2008 tree inventory. Tree mortality was measured as presence/absence at the individual-tree scale using matched tree comparisons. Annual mortality rates (Eq. 1) were measured at the street-section scale with the equation used by Nowak et al. (2004) and adapted by Lawrence et al. (2012):

 $m=1-(N1/N0)1/t$ (Eq. 1)

Where *m* is the annual mortality rate (%), *N*0 is the number of living trees at the time of the first inventory, *N*1 is the number of living trees at the time of the second inventory, and *t* is the number of years between inventories. Diameter growth rates (cm/yr) were measured by dividing the difference in DBH between matched trees by the time interval between inventories. Tree planting rates (trees/ha/yr) were measured at the street-section scale. The fourth and final ecological change variable was the Neighbour*woods*-derived 2007/2008 and 2014 tree condition indices. However, change in tree condition between inventories was not measured due to the likelihood of assessment subjectivity among different researchers collecting data at the two time instances.

2.3.2. Vulnerability Analysis

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

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

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

**3. Results**

*3.1. Ecological Change*

The change in size-class distribution between the 2007/2008 and 2014 inventories (Fig. 1) showed a decline in the presence of smaller (i.e., younger) trees and an increase in medium and larger (i.e., older) trees, while the largest size class remained relatively stable. This shift towards a larger, older urban forest ecosystem is also reflected in size-class mortality rates (Table 4), where higher mortality rates are seen in the smaller size classes. When planted trees are incorporated, the total number of trees in the >0-10 cm DBH size class in 2014 exceeded the 2007/2008 inventory.



**Fig. 1.** Change in size-class distribution of measured trees between the 2007/2008 and 2014 inventories.



**Fig. 2.** Change in tree species distribution of the 10 most abundant species measured between the 2007/2008 and 2014 inventories. ACPL: *Acer platanoides*; FRPE: *Fraxinus pennsylvanica*; GLTR: *Gleditsia triacanthos*; THOC: *Thuja occidentalis*; ASCA1: *Acer saccharinum*; AEHI: *Aesculus hippocastanum*; ACFR: *Acer x freemanii*; TICO: *Tilia cordata*; AIAL: *Ailanthus altissima*; MOAL: *Morus alba*.

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

The dominant species in the study area in both the 2007/2008 and 2014 inventories (Fig. 2) was *Acer platanoides*. *Thuja occidentalis* exceeded *Acer platanoides* in 2014 in abundance. When planted trees are incorporated, *Gleditsia triacanthos*, *Thuja occidentalis*, *Acer* x *freemanii*, and *Tilia cordata* all increased in population size, while *Acer platanoides*, *Fraxinus pennsylvanica*, *Acer saccharinum*, *Aesculus hippocastanum*, *Ailanthus altissima*, and *Morus alba* decreased. No planted *Fraxinus pennsylvanica*, *Aesculus hippocastanum*, *Ailanthus altissima*, or *Morus alba* were observed. *Ailanthus altissima* had a substantially higher mortality rate than other trees (Table 4), followed by *Fraxinus pennsylvanica*, both of which were higher than the study area average annual mortality rate of 2.4%. No dead/absent *Gleditsia triacanthos* or *Tilia cordata* were observed. *Fraxinus pennsylvanica* were in the worse condition, which was likely attributable to the on-going EAB infestation in the study area, while *Thuja occidentalis* were consistently in better condition. Tree condition of other species was generally reflective of tree size, where consistently larger, older species (e.g., *Acer saccharinum* and *Aesculus hippocastanum*) were in worse condition. Diameter growth rates were as expected, given both species-specific growth rates and size-class distribution of species in the study area. The 672 trees alive in the study area in 2014 had a replacement value of $1,794,671 and total ecosystem service supply value of $20,272, with carbon storage being the most valuable, followed by the energy savings from reduced cooling in buildings attributable to tree shading (Table 5).

**Table 4.** Annual mortality rate (%), mean diameter growth rate (cm/yr), and mean condition index value of measured trees, stratified by diameter class and 10 most abundant species.

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

**Table 5.** Urban forest ecosystem services provided by living trees (*n*=672) measured in 2014 and their financial value, as estimated by the i-Tree Eco model.

| **Ecosystem service** | **Metric** | **Total** | **Value** **($ CAD)** |
| --- | --- | --- | --- |
| Air pollution removal | Pollutant removal (g/yr) | 167,556 | 1,362 |
|  | Carbon monoxide (g CO/yr) | 165 | 0 |
|  | Ground-level ozone (g O3/yr) | 125,879 | 1,059 |
|  | Nitrogen dioxide (g NO/yr) | 31,978 | 269 |
|  | Sulphur dioxide (g SO/yr) | 5,290 | 11 |
|  | Particulate matter (g PM2.5/yr) | 4,243 | 24 |
| Rainfall interception | Avoided runoff (m3/yr) | 525 | 1,219 |
| Building energy effects | Annual energy savings – heating (MBTU) | 236 | 2,536 |
|  | Annual energy savings – heating (kWh) | 20,942 | 1,570 |
|  | Annual energy savings – cooling (kWh) | 55,951 | 4,196 |
| Carbon storage and sequestration1 | Gross carbon sequestration (kg C/yr) | 7,935 | 238 |
|  | Total carbon storage (kg C) | 259,582 | 7,788 |
| Urban forest replacement value | Total tree compensatory value ($)  | - | 1,794,671 |

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

### 3.2. Vulnerability Analysis

Regression modelling yielded several significant tree mortality predictors and explained 62% of the variation in mortality at the individual-tree scale (Table 6) and 73% at the street-section scale (Table 9). The strongest predictor of mortality at both scales was tree condition in 2007/2008. Trees were more likely to survive the period between inventories in all land uses excluding commercial/industrial uses, though multi-unit residential and park land uses were not statistically significant. Within residential land uses, trees adjacent to row houses versus other housing types were more likely to experience mortality. *Acer platanoides* and trees within the >0-10 cm size class had higher odds of mortality, while *Gledistia triacanthos* and *Morus alba* had lower odds. At the street-section scale, significant mortality predictors were higher impervious cover and wider street widths, while homeownership and larger distances from adjacent buildings explained lower mortality rates. Contrary to vulnerability assumptions, individual trees in conflict with overhead utilities, buildings, and other pieces of infrastructure were less likely to die. Also contrary to assumptions, high species diversity and low built area intensity were significant predictors of mortality at the street-section scale. The model accuracy of classifying trees as live or dead (Table 7) shows relatively high prediction accuracies and indicates that, on average, predicting live trees had a higher accuracy than dead trees. However, these classification rates require further validation in future research with external datasets.

**Table 6.** Beta coefficients and *p*-values for the backward logistic regression analysis predicting individual tree mortality (0=live; 1=dead) using the urban forest vulnerability indicators.

|  |  |  |  |
| --- | --- | --- | --- |
| **Variable** | **Coefficient** | **Odds Ratio** | ***p*** |
| Conflict with overhead utilities  | -3.392 | 0.034 | <0.001 |
| Conflict with buildings | -2.835 | 0.059 | <0.001 |
| Conflict with other  | -1.629 | 0.196 | <0.001 |
| Land use - Institutional | -2.141 | 0.118 | <0.001 |
| Land use - Multi-unit residential | -21.56 | 0 | 0.997 |
| Land use - Park | -21.564 | 0 | 0.998 |
| Land use - Residential | -1.334 | 0.263 | 0.002 |
| Building type - Row house | 0.939 | 2.556 | 0.027 |
| Site type - Raised wooden planter | -1.625 | 0.197 | 0.007 |
| 2007/2008 tree condition | 3.365 | 28.933 | 0.014 |
| 0-10 cm DBH | 1.309 | 3.704 | <0.001 |
| *Acer platanoides* | 0.682 | 1.978 | 0.074 |
| *Gleditsia triacanthos* | -20.272 | 0 | 0.995 |
| *Acer* x *freemanii* | -1.805 | 0.164 | 0.026 |
| *Morus alba* | -2.343 | 0.096 | 0.013 |
| In-grown tree | 2.299 | 9.96 | <0.001 |
| Pseudo-R2 | 0.620 |  |  |

**Table 7.** Classification accuracies (%) of backward logistic regression models predicting tree mortality by diameter class and three most abundant species.

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

Modelling tree condition also yielded significant predictors, though explained less variation in condition at the individual-tree scale (37%; Table 8) compared to the street-section scale (65%; Table 9). Evidence of poor tree management (e.g., improper pruning practices) and trees in the largest two size classes were strong predictors of poorer condition for individual trees. Median family income was significant at both the individual-tree and street-section scale, with higher income predicting better tree condition. Commercial buildings at the individual-tree scale and commercial/industrial land uses at the street-section scale also explained poor tree condition. Other significant predictors at the street-section scale were high existing canopy cover and open green space, higher homeownership, higher education, larger site sizes, and greater distances to adjacent buildings. All of these latter findings run counter to *a priori* assumptions of vulnerability and were likely heavily influenced by declining condition with tree size/age, as these variables were all indicative of sites that tended to have larger trees.

**Table 8.** Beta coefficients and *p*-values (in brackets) for the backward linear regression analysis predicting individual 2014 tree condition index values and diameter growth rates

(cm/yr) of individual trees using the urban forest vulnerability indicators.

|  |  |  |  |
| --- | --- | --- | --- |
| **Independent Variable** | **Condition Model1** | **Independent Variable** | **Diameter Growth Model** |
| Light availability | -0.091 (0.007) | 2014 tree condition | -0.134 (0.000) |
| Poor management | 0.282 (0.000) | Population density | -0.076 (0.095) |
| Vandalism | 0.139 (0.000) | Conflict with buildings | 0.132 (0.000) |
| Building type - Commercial at grade, apartment tower2 | 0.160 (0.000) | Conflict with other  | 0.096 (0.009) |
| Building type - Commercial at grade, residential | 0.141 (0.000) | Distance to street | 0.097 (0.018) |
| Building type - Commercial building | 0.161 (0.000) | Width of street | -0.184 (0.000) |
| Building type - Institutional building | 0.074 (0.029) | Poor management | -0.078 (0.030) |
| Building type - Row house | -0.087 (0.012) | Land use - Institutional | 0.35 (0.000) |
| Site type - Grass median | 0.088 (0.012) | Land use - Multi-unit residential | 0.151 (0.003) |
| Site type - Raised concrete planter | 0.112 (0.112) | Building type - Institutional building | -0.216 (0.001) |
| >0-10 cm DBH | -0.149 (0.000) | Building type - Row house | -0.083 (0.018) |
| 10.1-20 cm DBH | -0.071 (0.053) | Site type - Driveway/fenceline | -0.138 (0.000) |
| 50.1-75 cm DBH | 0.238 (0.000) | >0-10 cm DBH | -0.327 (0.000) |
| >75 cm DBH | 0.238 (0.000) | 10.1-20 cm DBH | -0.104 (0.006) |
| *Acer platanoides* | 0.159 (0.000) | >75 cm DBH | -0.105 (0.010) |
| *Fraxinus pennsylvanica* | 0.161 (0.000) | *Acer platanoides* | -0.253 (0.000) |
| *Thuja occidentalis* | -0.071 (0.038) | *Acer saccharinum* | -0.086 (0.036) |
| *Morus alba* | 0.060 (0.065) | *Aesculus hippocastanum* | -0.169 (0.000) |
| Median family income | -0.382 (0.000) | *Acer* x *freemanii* | 0.152 (0.000) |
| Homeownership | 0.215 (0.000) | *Morus alba* | 0.077 (0.037) |
| Population with a university degree | 0.141 (0.003) | In-grown tree | 0.094 (0.008) |
| Replanted site | -0.137 (0.000) |  |  |
| R2 | 0.372 |  | 0.312 |

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

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

The regression models predicting diameter growth rates did not perform as well as the mortality and condition models, explaining only 31% and 38% of the variation in growth rates at the individual-tree (Table 8) and street-section (Table 9) scale, respectively. At the individual-tree scale, significant predictors with the strongest explanatory power were institutional land uses predicting faster growth rates and the >0-10 cm size class predicting slower growth rates. Additionally, 2014 tree condition was a significant predictor of growth rates at the individual-tree and street-section scale, with better condition explaining faster growth rates. Lastly, modelling tree planting rates at the street-section scale explained 53% of variation in rates (Table 9). High rates of homeownership, high existing canopy cover, high built area intensity, and good 2014 tree condition were all found to be significant predictors of tree planting rates.

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

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Independent Variable** | **Mortality Model** | **Condition Model1** | **Diameter Growth Model** | **Tree Planting Model** |
| Built area intensity | -0.236 (0.069) | 0.339 (0.036) | -0.313 (0.017) | 0.542 (0.002) |
| Land use - Commercial/industrial |  | 0.736 (0.001) |  |  |
| Land use - Institutional |  |  | 0.271 (0.025) |  |
| Land use - Park |  | 0.359 (0.002) |  |  |
| Site size |  | 0.469 (0.008) |  |  |
| Height of nearest building |  |  | -0.242 (0.043) |  |
| Distance to nearest building | -0.152 (0.091) | 0.332 (0.007) |  |  |
| Width of street | 0.171 (0.092) |  |  |  |
| Impervious cover2 | 0.262 0.040) |  |  |  |
| 2014 tree condition |  |  | -0.407 (0.002) | -0.318 (0.039) |
| 2007/2008 tree condition | 0.889 (< 0.001) |  |  |  |
| Species diversity | 0.240 (0.016) |  |  |  |
| Structural diversity |  | -0.395 (0.001) | -0.270 (0.049) |  |
| Median family income |  | -0.539 (0.008) |  |  |
| Homeownership | -0.261 (0.002) | 0.337 (0.030) |  | 0.299 (0.038) |
| Population with a university degree |  | 0.354 (0.022) |  |  |
| Open green space |  | 0.407 (0.021) |  |  |
| Existing canopy cover |  | 0.435 (0.003) |  | 0.640 (0.001) |
| R2 | 0.733 | 0.646 | 0.383 | 0.533 |

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

2 Impervious cover was measured at the parcel scale (mean=65.3%; standard deviation=21.5%) for the tree mortality analysis, since it could not be estimated in crown projections for dead trees.

**4. Discussion and Conclusions**

The findings of this study suggest that the highest exposure and corresponding levels of tree decline and mortality were most influenced by the intensity of land use and the conditions encountered in the built environment. Commercial land uses in street sections and commercial buildings adjacent to trees consistently explained higher mortality rates and poor tree conditions. While studies have found varying effects of commercial land uses on urban trees (e.g., Lawrence et al., 2012), it is generally established that they are among the most detrimental for tree health (Nowak et al., 2004; Jutras et al., 2010). However, at finer spatial scales it is important to differentiate between different causes and correlates of urban forest decline within commercial land uses. For example, street width (i.e., wider streets) can be a correlate of tree stress (Nagendra and Gopal, 2010) and was found to be a significant predictor of mortality at the street-section scale in this study. However, several residential street sections in the study area had wider streets than commercial sections. While land use is a fairly established mechanism for stratifying urban landscapes and conducting urban forest research (Nowak et al., 1996; Steenberg et al., 2015a), the results of this study suggest that at the household scale, differentiated indicators (e.g., building type, impervious cover, street geometry) are necessary for accurately predicting mortality and declining tree condition.

There is a myriad of physical, biological, and social stressors and disturbances that afflict urban trees and forests (Trowbridge & Bassuk, 2004). In this study, exposure indicators were mainly limited in scope to those stressors associated with the built environment and urban form. However, the intent was that the sensitivity indicators would, in part, address these other dimensions of exposure for which quantification and/or data availability were limiting factors for measurement. For example, vulnerability to biological threats (e.g., EAB) or storm events can be captured in the sensitivity metrics of species composition (e.g., ash abundance and species diversity; Laćan & McBride, 2008) and age structure (e.g., structural diversity and over-mature canopies; Staudhammer & LeMay, 2001; Lopes et al., 2009). Nonetheless, the study’s findings suggest that quantifying biological exposures would be beneficial in future vulnerability assessment given the high levels of decline and mortality of *Fraxinus pennsylvanica* attributable to the EAB.

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

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

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

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

Given their longevity and stationary nature, trees and forests are in general vulnerable to environmental change, where urban forest structure and function may lag considerable in their response to drivers of change (e.g., changes in management practices). The disparity between commonly-planted tree species and overstory species composition in Harbord Village, coupled with on-going decline of *Fraxinus pennsylvanica*, and its removal from tree planting schedules, suggest that considerable future change in ecosystem conditions is likely. Moreover, *Acer platanoides*, which was the dominant overstory species, was an extremely popular urban tree in previous decades but is now rarely planted in Toronto because of its invasive nature (City of Toronto, 2013). In addition to these potential lag effects in species composition, species-specific mortality and planting rates and the shifts towards smaller, ornamental species in the neighbourhood may also correspond to declines in future ecosystem service supply. The i-Tree Eco model results, as well as existing research (Nowak & Dwyer, 2007), indicate that most ecosystem services are strongly associated with larger, longer-lived tree species with large leaf areas. These issues reinforce the temporal nature of vulnerability and associated impacts (i.e., losses in ecosystem service supply; Adger, 2006; Steenberg et al., 2015b). Urban forest vulnerability assessment therefore requires both hindsight, in the form of monitoring, and foresight, in the form of ecological modelling.

Vulnerability science offers an integrative lens through which to explore risk and loss of function in highly complex, social-ecological systems like the urban forest (Turner et al., 2003; Adger, 2006; Grove, 2009). Most research investigating mortality and decline in urban forests focuses primarily on stressors and disturbances. This study suggests there is a need to investigate how these stressors interact with urban forest structure and surrounding human populations in order to reliably predict the likelihood of potential loss. Moreover, most of the established relationships between urban forests and socioeconomic variability are based on two-dimensional tree canopy cover data at broader spatial scales. There are few studies (e.g., Shakeel & Conway, 2013) investigating urban forest ecological processes at finer scales using empirical field data. However, the findings from Harbord Village will be limited to some degree in there transferability to different neighbourhoods, cities, and scales. For instance, regression analysis revealed some counterintuitive relationships between vulnerability indicators and ecosystem change variables (e.g., conflicts with infrastructure and mortality). Further research is needed that tests both the reliability and validity of indicator design and the generalizability of the findings in the study area. With the increasing attention to urban forests from municipalities (Ordόñez & Duinker, 2013) and community groups (Conway et al., 2011), the demand for approaches to quantify ecosystem structure, function, and vulnerability to inform management is high.

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