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CONTENTS

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Key Words. *Acer* spp.; Diversity Indices; Ecosystem Services; New Jersey; New York; Pennsylvania; Resilience; Street Trees.

Abstract. Advances in control can help municipal foresters save ash trees from emerald ash borer (EAB) [*Agrilus planipennis* (Fairmaire)] in urban forests. Although ash trees of any size can be protected from this pest, cities often do not implement programs because they fail to recognize and act on incipient populations of EAB. In this study, researchers develop a model for predicting ash mortality over an eight-year period, and validated with data from the removal of >14,000 ash trees killed by EAB in Fort Wayne, Indiana, U.S. researchers then developed a sampling scheme to help foresters map their ash trees along the expected progression of ash decline. This model was then used to modify a web-based EAB cost calculator that compares discounted annual and cumulative costs of implementing a variety of management strategies. It was determined that strategies that most heavily relied on saving ash trees were less expensive and produced a larger forest than those strategies that mostly removed and replaced ash trees. Ratios of total discounted costs to discounted cumulative benefits of strategies that saved most ash trees were over two-thirds lower than strategies of proactive tree removal and replacement. Delaying implementation of an ash management program until damage would be visible and more obvious to the community (Year 5 of the model) decreased the cost–benefit ratio by <5%. Thus, delays that rely on the abundance of locally damaged trees to bolster community support do not necessarily diminish the utility of implementing a control strategy.
Key Words. *Agrilus planipennis*; Ash; Ash Tree Decline Model; EAB Cost Calculator; Emerald Ash Borer; Indiana; Pest Management; Projection.

Abstract. Tree lodging in landscapes during storms has been attributed to root architecture in nurseries. Objectives of this study were to evaluate influence of retention time in three progressively larger nursery containers, and root pruning at landscape planting, on establishment, anchorage, and root architecture in the first four post-planting years. All trees were retained in three progressively larger containers (11, 57, and 170 L) for a total of 32 months, with varying retention times in each. Retention time had little influence on post-planting xylem water potential for *Acer rubrum* and *Ulmus parvifolia*. There were few differences in aboveground growth among retention times. Except for *Acer*, retention time had a negligible influence on anchorage. Root pruning by shaving 170 L root ball periphery when planting had no impact on growth except for one post-planting year. However, root pruning invoked a dramatic reduction in circling and descending roots four years after planting caused by root deflection in the final nursery container. Although root pruning had no influence on bending stress required to winch *Magnolia* trunks to any degree of trunk tilt, approximately 10% more bending stress was required to winch *Acer* trunks up to five degrees tilt when root balls were shaved at planting.
Key Words. *Acer rubrum*; Anchorage; Bending Stress; *Magnolia grandiflora*; Planting; Post Planting; Root Architecture; Transplanting; *Ulmus parvifolia*; Xylem Water Potential.

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Abstract. Practices to promote tree root growth have been sought in arboriculture for many years, especially as a treatment for trees that are stressed or in decline. Tree fertilization is a common practice in arboriculture. A few research reports have shown that nitrogen fertilization can increase fine-root density in localized areas of soil where the fertilizer has been applied, but the effect on the whole root system has not been investigated. Research has reported fine-root stimulating effects from basal application of the growth regulator paclobutrazol, but results have not been consistent. More information is needed. A slow-release granular formulation of nitrogen was broadcast over the entire root system of mature oaks for four consecutive years, with or without a paclobutrazol basal drench in the first year. Nitrogen was also applied to younger green ash and black maple as granular broadcast or subsoil liquid soil injection for two consecutive years. There was no overall increase in fine-root development from any of the treatments, and no localized increase from soil injection application of nitrogen. Growth response of the crowns was minimal. Soil profiles were undisturbed with moderate natural nitrogen availability. The results suggest that routine fertilization of trees at standard recommended rates may be ineffective if soil fertility is moderate.

Key Words. *ANSI A300 Standard for Tree Care Operations*; Fertilization; Growth Regulator; Injection; Nitrogen; Paclobutrazol; Root Growth; Root Stimulation.

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CONVERSION CHART (METRIC TO IMPERIAL)

Metric unit	Multiply by	To obtain English unit
Length		
millimeters (mm)	0.04	inches (in)
centimeters (cm)	0.4	inches (in)
meters (m)	3.3	feet (ft)
meters (m)	1.1	yards (yd)
kilometers (km)	0.6	miles (mi)
Area		
square centimeters (cm^2)	0.16	square inches (in^2)
square meters (m^2)	1.2	square yards (yd^2)
square meters (m^2)	10.8	square feet (ft^2)
square kilometers (km^2)	0.4	square miles (mi^2)
hectares (ha)	2.5	acres (ac)
Mass (weight)		
grams (g)	0.035	ounces (oz)
kilograms (kg)	2.2	pounds (lb)
metric tonnes (t)	1.1	short tons
Volume		
milliliters (mL)	0.03	fluid ounces (fl oz)
milliliters (mL)	0.06	cubic inches (in^3)
liters (L)	2.1	pints (pt)
liters (L)	1.06	quarts (qt)
liters (L)	0.26	gallons (gal)
cubic meters (m^3)	35	cubic feet (ft^3)
cubic meters (m^3)	1.3	cubic yards (yd^3)
Temperature (exact)		
degrees Celsius (°C)	multiply by 9/5, then add 32	degrees Fahrenheit (°F)

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Street Tree Diversity in Three Northeastern U.S. States

F.D. Cowett and Nina Bassuk

Abstract. Street tree diversity is widely viewed as a key component in the resilience of street tree populations to pests, diseases, and climate change. Assessment of street tree diversity is considered integral to sustainable street tree management and preservation of the ecosystem services and social benefits that street trees provide. This paper assesses street tree diversity in three northeastern U.S. states—New Jersey, New York, and Pennsylvania—by analyzing municipal street tree inventory data stratified by the 2012 USDA Plant Hardiness Zones. Despite the lesson learned from the historical devastation of overplanted American elms (*Ulmus americana*) by Dutch elm disease, and awareness of the contemporary threats posed to ashes (*Fraxinus* spp.) by the emerald ash borer (*Agrilus planipennis*) and to maples (*Acer* spp.), and other tree genera by the Asian longhorned beetle (*Anoplophora glabripennis*), results presented here indicate a current concentration of street trees among a relatively small number of species and genera, and in particular the dominance of maples as street trees. Results also show a positive relationship between street tree diversity and warmer average minimum winter temperatures. Consequently, there is a clear need in all three states for greater species and genus diversity in statewide and municipal street tree populations. However, meaningful impediments exist to increasing street tree diversity, especially in the short term.

Key Words. *Acer* spp.; Diversity Indices; Ecosystem Services; New Jersey; New York; Pennsylvania; Resilience; Street Trees.

Biological diversity at the species level has been associated with the stability and productivity of non-urban vegetated ecosystems (Bezemer and van der Putten 2006; Tilman et al. 2006; Cadotte et al. 2012), including non-urban forest ecosystems (Thompson et al. 2009). Urban forest ecosystems differ meaningfully from non-urban forest ecosystems, since factors typically found in urban environments, including but not limited to buildings, impervious surfaces, anthropogenic soils, pollution, and the urban heat island, may not be found in non-urban environments or at least not to the same degree (Nock et al. 2013). Despite these differences, however, and similar to the non-urban forest, urban tree diversity has been associated with the stability and productivity of the urban forest and with continued provision of ecosystem services and social benefits (Manes et al. 2012).

Street trees growing in public street rights-of-way typically comprise a minority of the urban forest and the services and benefits they provide (Dwyer et al. 2000). Nevertheless, they have received special attention and have been a focus of urban forestry due to their public function (Clark et al. 1997; Cumming et al. 2008). Because of the devastation wrought many years ago by Dutch elm disease on American elms (*Ulmus americana*) (Campanella 2003) and more current threats, such as those posed to ashes (*Fraxinus* spp.) by the emerald ash borer (*Agrilus planipennis*) (Poland and McCullough 2006) and to maples (*Acer* spp.) and other genera by the Asian longhorned beetle (*Anoplophora glabripennis*) (Smith and Wu 2008), diversity in street tree plantings is widely viewed as a key component in the resilience of street tree populations to pests, diseases, and climate change, and assessment of street tree diversity is considered integral to sustainable street tree management (Raupp et al. 2006; Sjöman et al. 2012).

Street tree diversity has been assessed at a variety of geographic levels. It has most commonly been assessed at the level of the individual municipality from data collected in a complete or sample street tree inventory (Clark et al. 1997). In Syracuse, New York, U.S., for example, Sanders (1981) assessed street tree diversity for the city as a whole and for 17 city neighborhoods from a complete inventory of 32,517 street trees. In Davis, California, U.S., Maco and McPherson (2003) assessed street tree diversity for the city as a whole and for 11 sampling zones from a sample of 3,089 public and privately managed trees located within the city's right-of-way.

Street tree diversity also has been assessed at broader geographic levels, such as a region or state. Lesser (1996) assessed street tree diversity in southern California based on inventory data from 21 cities, Ball et al. (2007) assessed street tree diversity in South Dakota, U.S., based on inventory data from 34 communities statewide, and Raupp et al. (2006) assessed street tree diversity in the temperate zone of eastern North America based on inventory data from 12 cities and one college campus.

Finally, street tree diversity has been assessed at still broader geographic levels involving multiple countries and continents. Sjöman et al. (2012) assessed Nordic street tree diversity based on inventory data from 10 cities in four countries, and Kendal et al. (2014) assessed global street tree diversity based on tree species lists from 108 cities in six continents. In all the studies cited here, deficiencies were found in street tree diversity using a variety of metrics.

One common metric employed to assess street tree diversity is frequency distribution, where the relative abundances of street trees belonging to botanic species, genera, and families are calculated as percentages of the population as a whole. Relative abundance metrics were popularized by Santamour (1990), who advocated for more even distributions of street tree species, genera, and families in municipal street tree populations. Santamour proposed as a general rule that no tree species should comprise more than 10%, no tree genus should comprise more than 20%, and no tree family should comprise more than 30% of a municipality's street tree population. Thus, Sanders (1981) found Norway maple (*Acer platanoides*) to comprise 37.5% of all street trees in Syracuse, New York, U.S.; Lesser (1996) found American sweetgum (*Liquidambar styraciflua*) to comprise 14.27% of all street trees in 21 southern California cities; and Ball et al. (2007) found ashes (*Fraxinus* spp.) to comprise 36.3% of all street trees in 34 South Dakota communities. Santamour's 10-20-30 benchmarks have become a widely accepted rule-of-thumb, even though there is a lack of scientific or empirical evidence to validate those numbers as effective thresholds. Additionally, it has been argued that applying the 10-20-30 rule in some urban landscapes may be counterproductive to street tree management by replacing well-adapted tree species with underperforming ones (Richards 1993; Kendal et al. 2014). Conversely, more stringent standards than Santamour's have been offered before and since. Barker (1975) proposed that no tree species should comprise more than 5% of the street tree population, Bassuk et al. (2009) proposed limiting any one street tree species to between 5% and 10% of the street tree population, and Ball (2015) proposed that no tree genus should comprise more than 5% of the street tree population.

A diversity index is another metric employed in assessing street tree diversity. Many indices have been utilized in ecology and environmental science to make comparisons between biological populations. These indices usually consider more than simply relative abundance and include such factors as population size and species richness (the number of species in the population) in their calculation. Two indices often used in assessing street tree diversity are Simpson's Diversity Index (Simpson 1949) and the Shannon-Wiener Diversity Index (Shannon 1948). Simpson's Diversity Index calculates the proportion of species i relative to the total number of species (p_i), sums the squared proportions for all the species, and then takes the reciprocal, according to the formula:

$$[1] \qquad D = 1/\sum_{i=1}^{s} p_i^2$$

The Shannon-Wiener Diversity Index calculates the proportion of species i relative to the total number of species (p_i), which is then multiplied by the natural logarithm of this proportion ($\ln p_i$), summed across all species, and then multiplied by -1, according to the formula:

$$[2] \qquad H = -\sum_{i=1}^{s} p_i \ln p_i$$

Simpson is sometimes preferred to Shannon-Wiener because it gives more weight to the more abundant species and is therefore more sensitive to the distribution evenness advocated by Santamour than to species richness; Shannon-

Wiener is sometimes preferred to Simpson because it gives more weight to less abundant species and is more sensitive to sample size (Barbour et al. 1987; Colwell 2009). Despite these distinctions, Simpson and Shannon-Wiener have been applied somewhat interchangeably in assessing street tree diversity. For example, Maco et al. (2005) used Simpson's Diversity Index to assess street tree diversity in Berkeley, California, Dobbs et al. (2013) used the Shannon-Wiener Diversity Index to compare tree species composition between streets, parks, and private property in Melbourne, Australia, and Jim and Chen (2009) and Kara (2012) used both the Simpson and Shannon-Wiener indices to assess street tree diversity in Taipei, Taiwan, and Aydin, Turkey, respectively. Additionally, Sun (1992) and Sreetheran et al. (2011) used the inverse of Simpson's Diversity Index (1/SDI) to equate an inverse SDI value of 10 with Santamour's 10% rule for species and an Inverse SDI value of 20 with a 5% benchmark for species; Subburayalu and Sydnor (2012) used a Simpson Diversity Index weighted by environmental benefits, pest vulnerability, and taxon adaptability to identify areas requiring increased street tree diversity in four Ohio, U.S., communities. As with Santamour's 10-20-30 rule, results reported for the Simpson and Shannon-Wiener indices have not been equated scientifically with effective thresholds for street tree diversity. However, the Simpson and Shannon-Wiener indices and relative abundance metrics have been employed not only to make quantitative comparisons for street tree diversity between neighborhoods, municipalities, regions, and other geographic levels, but also to explore explanatory factors. Thus, McPherson and Rowntree (1989) and Pauleit (2002) found greater street tree diversity to be associated with warmer climate, Jim and Chen (2009) found greater street tree diversity in older neighborhoods, and Kara (2012) found differences in street tree diversity based on land use and street type.

This paper assesses street tree diversity for three states in the northeastern United States—New Jersey, New York, and Pennsylvania—based on street tree inventory data obtained from municipalities in these states. It quantifies differences in diversity, between municipalities and between states, and considers explanatory factors. Finally, this paper makes recommendations for increasing diversity so as to enhance the resilience of street tree populations to pests, diseases, and climate change, and ensure the continued provision of ecosystem services and social benefits associated with street trees.

METHODS

Street tree inventory data were obtained for 57 municipalities in New Jersey, 164 municipalities in New York, and 54 municipalities in Pennsylvania (Figure 1). All municipalities from which data were obtained are Census Places. The United States Census Bureau defines a Place as a legally bounded and incorporated concentration of population, such as a city, town, village, or borough, or an unincorporated concentration of population identifiable by name whose boundaries may change from one decennial census to the next (U.S. Census Bureau 2010). In 2014, there were 545 Census Places in New Jersey, 1,196 in New York, and 1,762 in Pennsylvania (U.S. Census Bureau 2015). Therefore, street tree inventory data were obtained from 10.5% of all Census Places in New Jersey, 13.7% of all Census Places in New York, and 3.1% of all Census Places in Pennsylvania.

These data were used to calculate the relative abundance percentages of street tree species and genera comprising each inventory. Additionally, sta-

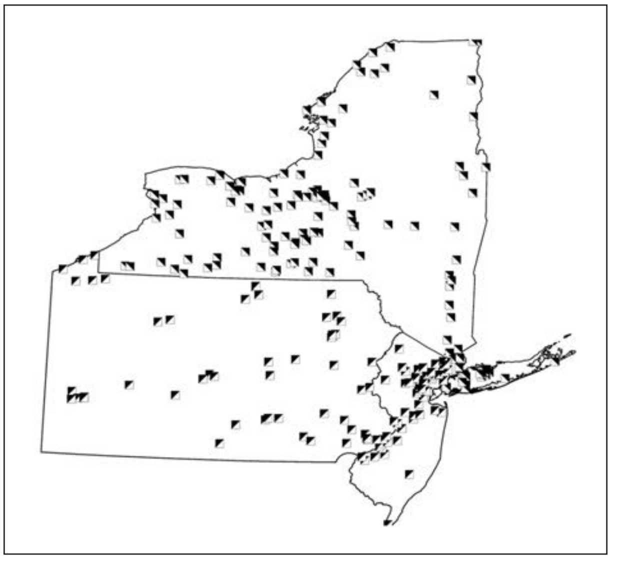

Figure 1. Street tree inventories obtained in New Jersey, New York, and Pennsylvania, U.S.

tistics for Simpson's Diversity Index, the Shannon-Wiener Diversity Index, and distribution evenness (Buzas and Gibson 1969) at species and genus levels were calculated using PAST Paleontological Statistics software Version 3.0 (Hammer et al. 2001). Statistics for the inverse of Simpson's Diversity Index were also determined. Some municipalities either collected data for genus and not species, or collected data at the species level for some but not all genera. Therefore, the number of municipalities for which relative abundance percentages and diversity-index statistics can be calculated at the genus level exceeds the number of municipalities for which relative abundance percentages and diversity-index statistics can be calculated at the species level.

Because the relative abundance percentages and diversity-index statistics mentioned comprise a non-random sample, there is a potential for selection bias and geographic variability to compromise the accuracy of further statistical analysis. Post-stratification of data and weighting with auxiliary information is a technique often used to correct for selection bias due to non-random sampling (Bethlehem 2010). A New York State street tree assessment conducted previously by the authors (Cowett and Bassuk 2014), based on a non-random sample, stratified data by the 1990 USDA Plant Hardiness Zones (U.S. National Arboretum 1990) and then weighted the stratified data by a measure of street length contained within each zone. To correct for potential selection bias due to non-random sampling, a similar technique was considered for this assessment.

A Geographic Information System (GIS) shapefile of the 2012 Plant Hardiness Zones for the Mid-Atlantic region was purchased from Climate Source (Corvallis, Oregon, U.S.), the exclusive public distributor of the Plant Hardiness Zone GIS data sets. The 2012 Plant Hardiness Zones account for a general warming trend and changes in zone boundaries since the 1990 version (Daly et al. 2008; U.S. Department of Agriculture 2012). GIS software was used to clip the zones to the boundaries of New Jersey, New York, and Pennsylvania (Figure 2). Each municipality was assigned to a zone based on the location of the municipality's inner centroid (i.e., a geometrically calculated center point within a municipality's boundaries). The relative abundance percentages for street tree species and genera found in inventoried municipalities were then averaged. The means for prevalent species and genera were regressed on the 2012 USDA Plant Hardiness Zones in a one-way analysis of variance (ANOVA). Significant effects (α = 0.05) that satisfied statistical assumptions for normality of residuals and homoscedasticity were found for many, but not all species and genera. Effects were generally greater for New York and Pennsylvania than for New Jersey. Based on these findings, it was decided to stratify data by the 2012 Plant Hardiness Zones.

Auxiliary information used for weighting purposes in the 2014 New York statewide assessment was a measure of street length contained within each 1990 USDA Plant Hardiness Zone. This measure reflected, first, obtaining a GIS shapefile of all street centerlines statewide from New York State; second, deleting street types, such as driveways, interstate highways, and divided highway segments, unlikely to contain street trees using New York Accident Location Information System (ALIS) codes; third, selecting centerlines contained within cities, villages, and Census Designated Places (CDPs) as well as Census Blocks with a population density of at least 500 persons per square mile (ppsm); and, fourth, calculating the percentage of selected street length contained within each Plant Hardiness Zone as a percentage of the selected statewide whole. Due to differences found in the coding and formatting of each state's most current street centerline data, GIS shapefiles provided by the states could not be used as auxiliary information for weighting purposes. Instead, U.S. Census TIGERLine All Roads GIS shapefiles (U.S. Census Bureau 2014) composed of data coded and formatted similarly for all three states, were used; first, to select street types by MTFCC (MAF/TIGER Feature Class Codes) codes in a manner matching as closely as possible the 2014 New York statewide assessment, and second to select streets contained within all Census Places, Census Urbanized Areas, and Census Blocks with a population density of at least 250 ppsm located in each state (Figure 3). The 250 ppsm threshold, a less stringent threshold than the 500 ppsm used in the 2014 New York statewide assessment, was deemed necessary to select streets contained within unincorporated communities and population concentrations in rural and suburban areas where inventories did not exist, but street trees could be expected to be found.

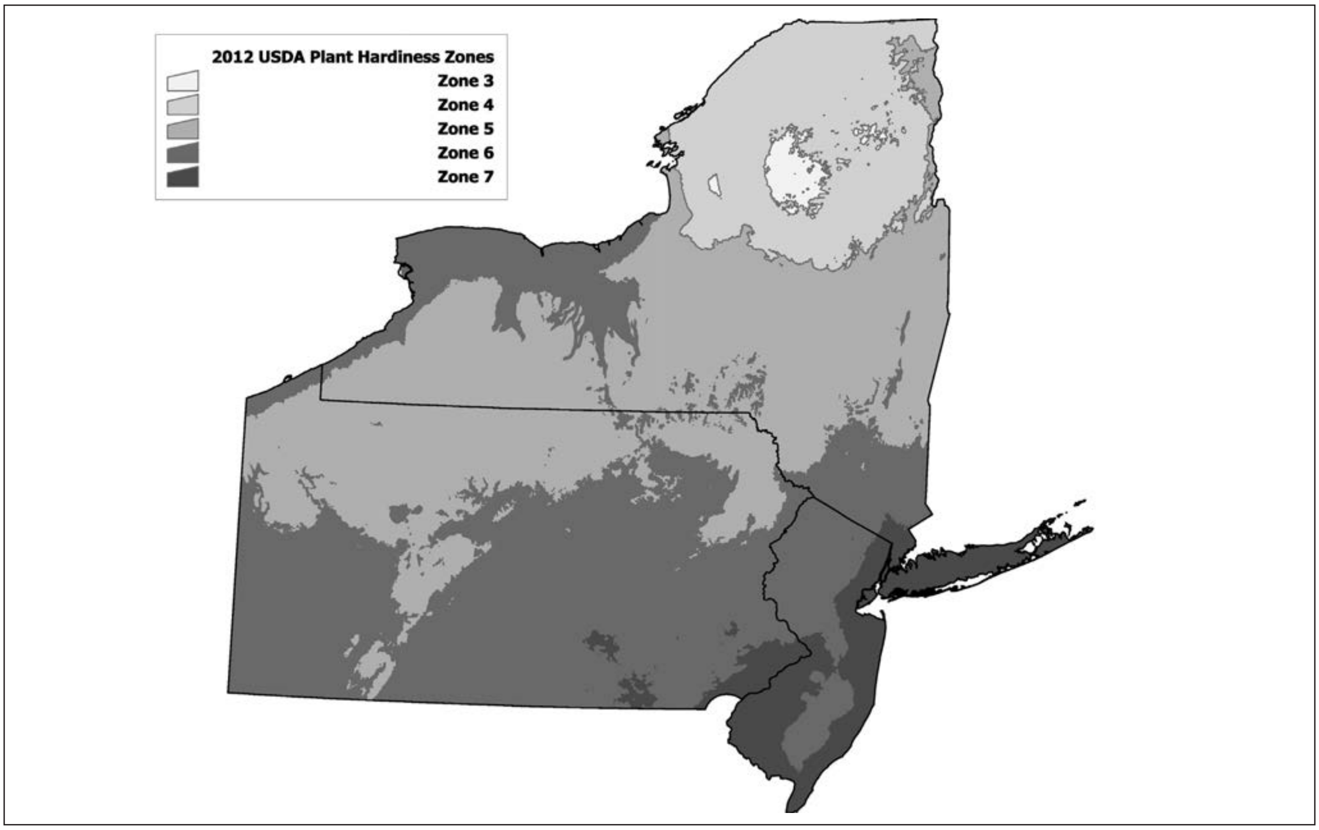

Figure 2. 2012 USDA Plant Hardiness Zones for New Jersey, New York, and Pennsylvania, U.S.

The percentage of selected street length contained within each 2012 Plant Hardiness Zone as a percentage of the selected statewide whole was calculated (Table 1). Substantial differences between zones and states were found. For example, Zone 7 contained a majority of street length in New Jersey and the most street length in New York, whereas Zone 6 contained a majority of street length in Pennsylvania. These percentages were used to create weights for each state according to the formula:

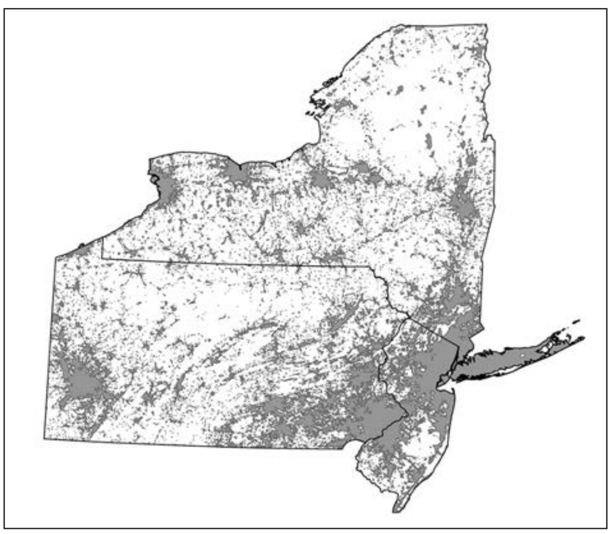

Figure 3. Shaded areas represent 2010 U.S. Census Places, Urbanized Areas, and Census Blocks with a population density of at least 250 persons per square mile (ppsm) in New Jersey, New York, and Pennsylvania, U.S.

[3] $[(w1 \bullet m1) + (w2 \bullet m2) + (w3 \bullet m3) + (w4 \bullet m4)] / (w1 + w2 + w3 + w4)$

Where $m1$, $m2$, $m3$, and $m4$ denote the group means (i.e., means for species and genus composition in the 2012 USDA Plant Hardiness Zones) and $w1$, $w2$, $w3$, and $w4$ denote the different weights for each group (i.e., the relative percentage of summed selected street length in each 2012 USDA Plant Hardiness Zone). Zones 5, 6, and 7 can be found in all three states. The Zone 5 area in New Jersey is very small and is not associated with any municipal street tree data, containing only 0.006% of selected New Jersey street length. Therefore, for weighting purposes, selected New Jersey street length contained in Zone 5 was aggregated with selected street length contained in Zone 6. Zones 3 and 4 are found

Table 1. Summed selected street length contained within the 2012 USDA Plant Hardiness Zones in New Jersey, New York, and Pennsylvania, U.S.

2012 Plant Hardiness Zone	Zone 3	Zone 4	Zone 5	Zone 6	Zone 7
New Jersey					
Street Length in Meters	0	0	3,394	16,915,805	39,159,020
Percent of Statewide Total	0.000%	0.000%	0.006%	30.165%	69.829%
New York					
Street Length in Meters	51,687	3,043,301	24,698,158	25,043,999	35,072,884
Percent of Statewide Total	0.059%	3.462%	28.095%	28.488%	39.896%
Pennsylvania					
Street Length in Meters	0	0	7,830,965	70,396,700	17,736,544
Percent of Statewide Total	0.000%	0.000%	8.160%	73.357%	18.482%

only in New York State. Zone 3 is sparsely populated and not associated with any municipal street tree data, containing only 0.059% of all selected New York State street length. Therefore, for weighting purposes, selected New York State street length contained in Zone 3 was aggregated with selected street length contained in Zone 4. Finally, a regional weighted mean was created from the statewide weighted means based on the percentage of each state's selected street length relative to the sum of selected street length found in all three states.

RESULTS

Species and Genus Composition

Weighted statewide relative abundance percentages were calculated for street tree species and genera from collated street tree inventory data (Table 2; Table 3). *Acer platanoides* (Norway maple) was found to be the most prevalent street tree species in all three states, with a regional weighted mean of 16.34% (14.63% in New Jersey, 19.80% in New York, and 15.08% in Pennsylvania). *Acer* spp. (maple) was found to be the most prevalent street tree genus in all three states, with a regional weighted mean of 38.94% (36.72% in New Jersey, 40.91% in New York, and 38.96% in Pennsylvania). On both regional and statewide levels, these results exceed Santamour's 10% rule for species, and his 20% rule for genus, and reflect abundance percentages found on the municipal level. Municipal species composition revealed that, for those municipalities from which street tree inventory data at the species level were obtained, 47 of 50 municipalities in New Jersey (94.0%), 152 of 153 municipalities in New York (99.3%), and 38 of 43 municipalities in Pennsylvania (88.4%) exceeded the 10% rule proposed by Santamour. In most but not all cases, this was due to the percentage of street trees that were *Acer platanoides* (Norway maple), although in many municipalities the percentages of *Acer rubrum* (red maple), *Acer saccharum* (sugar maple), *Gleditsia triacanthos* (honeylocust), *Platanus* × *acerifolia* (London planetree), *Pyrus calleryana* (Callery pear), *Quercus palustris* (pin oak), and/or *Quercus rubra* (northern red oak) surpassed 10%. Municipal genus composition revealed that, for those municipalities from which street tree inventory data at the genus level were obtained, 56 of 57 municipalities in New Jersey (98.2%), 162 of 164 municipalities in New York (98.8%), and 53 of 54 municipalities in Pennsylvania (98.1%) exceeded the 20% rule proposed by Santamour. In most but not all cases, this was due to the percentage of street trees belonging to the *Acer* genus, although in many municipalities the percentages of *Gleditsia* spp., *Malus* spp. (crabapple), *Platanus* spp. (planetree), *Pyrus* spp. (pear), *Quercus* spp. (oak), and/or *Tilia* spp. (linden) surpassed 20%.

Minimum average winter temperature as represented by the 2012 USDA Plant Hardiness Zones appeared to impact species and genus composition, as substantial differences were found between the zones for many but not all species and genera. Differences were more pronounced in New York and Pennsylvania than in New Jersey. For example, in New Jersey, differences were found for some, but not all, street tree genera (Figure 4); in New York, differences were found for most street tree species (Figure 5); and in Pennsylvania, differences were found for most street tree genera (Figure 6). In addition, the mean number of street tree species and genera per zone for the three states was found to increase as minimum average winter temperature increased (Figure 7).

Table 2. Weighted, statewide, relative abundance percentages for street tree species in New Jersey, New York, and Pennsylvania, U.S.

Species	NJ	NY	PA	Mean
Acer platanoides	14.63	19.80	15.08	16.34
Acer rubrum	10.79	5.23	8.78	8.27
Acer saccharum	4.74	8.47	9.43	7.92
Pyrus calleryana	6.59	5.63	7.68	6.80
Quercus palustris	8.92	3.87	5.85	6.08
Platanus × acerifolia	6.14	6.09	4.20	5.26
Gleditsia triacanthos	3.02	5.03	4.29	4.17
Acer saccharinum	3.05	4.94	3.82	3.94
Quercus rubra	3.31	1.91	3.02	2.77
Tilia cordata	1.79	2.61	1.85	2.14
Malus spp.	0.85	2.28	2.80	2.06
Fraxinus pennsylvanica	1.56	1.98	1.30	1.57
Zelkova serrata	2.11	1.13	1.37	1.50
Liquidambar styraciflua	1.50	0.80	1.11	1.12
Fraxinus americana	1.48	0.85	0.75	0.97
Picea abies	0.70	1.18	0.88	0.92
Pinus strobus	0.91	1.22	0.54	0.91
Ginkgo biloba	0.71	0.88	1.05	0.84

Table 3. Weighted, statewide, relative abundance percentages for street tree genera in New Jersey, New York, and Pennsylvania, U.S.

Genus	NJ	NY	PA	Mean
Acer	36.72	40.91	38.96	38.94
Quercus	15.12	8.44	9.73	10.77
Pyrus	6.39	6.36	7.95	7.08
Platanus	6.92	6.30	4.61	5.71
Gleditsia	2.78	4.76	3.98	3.89
Prunus	3.75	3.09	4.07	3.70
Tilia	3.97	4.09	3.14	3.63
Fraxinus	4.18	3.29	2.62	3.23
Picea	1.57	2.70	2.64	2.38
Malus	0.78	2.26	3.03	2.22
Zelkova	2.02	1.16	1.40	1.49
Pinus	1.27	1.71	1.19	1.36
Ulmus	1.15	1.38	1.30	1.29
Cornus	1.18	0.98	1.32	1.19
Liquidambar	1.38	0.84	0.99	1.05
Ginkgo	0.73	0.81	1.12	0.93
Robinia	0.74	1.08	0.32	0.65

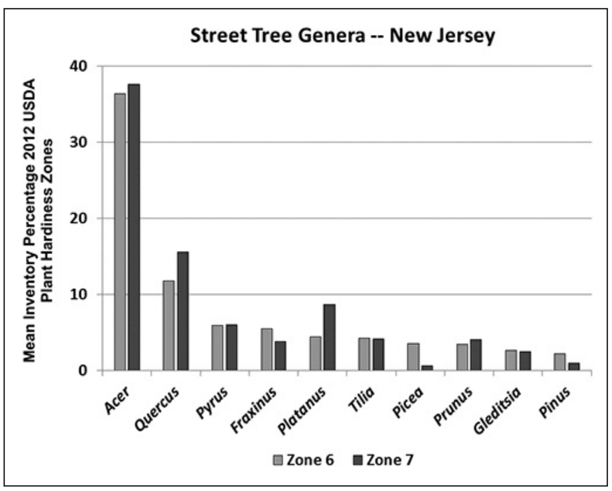

Figure 4. Mean inventory percentages for New Jersey street tree genera by 2012 USDA Plant Hardiness Zone.

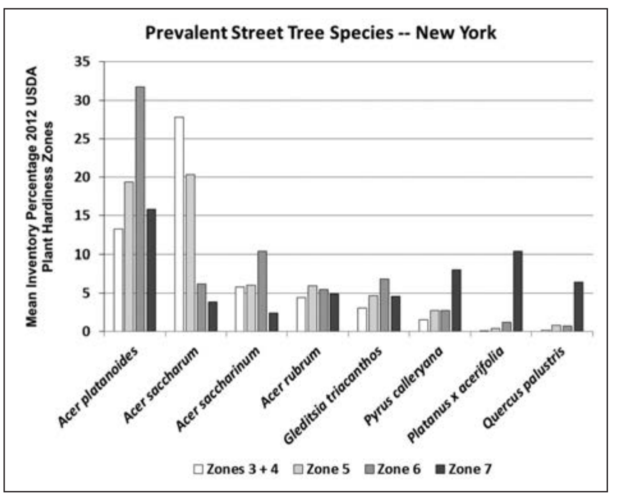

Figure 5. Mean inventory percentages for New York street tree species by 2012 USDA Plant Hardiness Zone.

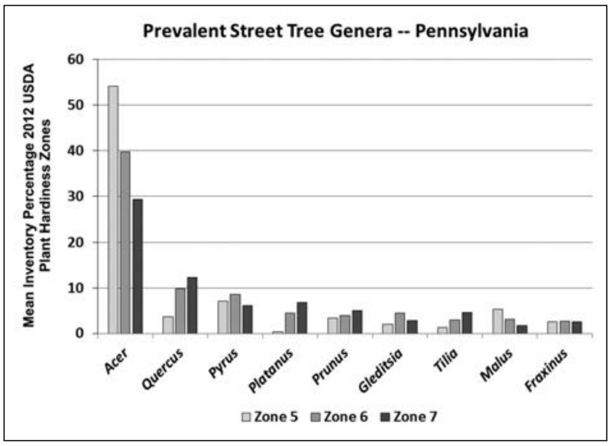

Figure 6. Mean inventory percentages for Pennsylvania street tree genera by 2012 USDA Plant Hardiness Zone.

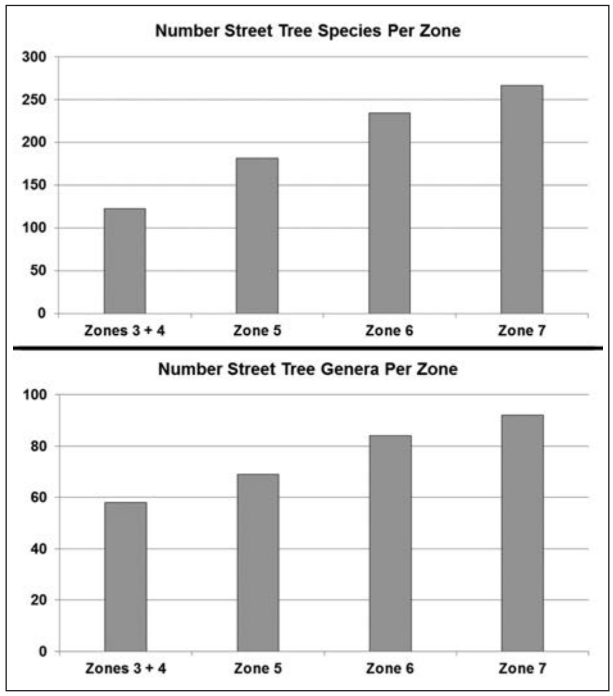

Figure 7. Mean number of street tree species and genera for New Jersey, New York, and Pennsylvania per 2012 USDA Plant Hardiness Zone.

Diversity Indices

Statistics were generated at the species and genus levels for the inverse of Simpson's Diversity Index and the Shannon-Wiener Diversity Index. For both the Inverse SDI and Shannon-Wiener, a larger value indicates greater diversity, and a smaller value indicates less diversity. Differences in values appear greater for the Inverse SDI than for Shannon-Wiener because the latter index is logarithmic. At both the species and genus levels, and for both diversity indices, street tree diversity was found to be greatest in New Jersey and least in New York State (Table 4). Street tree diversity also appeared impacted by minimum average winter temperature. At both the species and genus levels, and for both diversity indices excepting the Inverse SDI for genus in Zones 3 + 4 and 5, street tree diversity increased as minimum average winter temperature increased (Figure 8).

Finally, statistics were generated for distribution evenness at the species and genus levels. Species diversity was found to be positively correlated more with the number of species in each municipality than with the evenness of the municipality's species distribution or the number of municipal trees for both the Inverse SDI and the Shannon-Wiener Diversity Index. Genus diversity was found to be positively correlated more with the evenness of the genera distribution than with the number of genera or the number of municipality trees for the Inverse SDI, and to be positively correlated more with the number of genera than with the evenness of the genera distribution or the number of municipality trees for the Shannon-Wiener Diversity Index (Table 5). Thus, with the exception of genus diversity for the Inverse

Table 4. Mean and Median Inverse SDI (Inverse of Simpson's Diversity Index) and Shannon-Weiner Diversity Index values at species and genus levels for New Jersey, New York, and Pennsylvania, U.S.

	New Jersey	New York	Pennsylvania
Species			
Mean INV SDI	13.03	8.80	12.17
Median INV SDI	11.55	7.85	11.25
Mean Shannon-Wiener	3.00	2.67	2.89
Median Shannon-Wiener	2.99	2.68	2.84
Genus			
Mean INV SDI	5.52	4.08	5.23
Median INV SDI	5.20	3.44	4.97
Mean Shannon-Wiener	2.22	1.91	2.15
Median Shannon-Wiener	2.19	1.91	2.17

Table 5. Correlations for Mean Inverse SDI (Inverse of Simpson's Diversity Index) and Shannon-Weiner Diversity Index values, number of species and genera, number of municipality trees, and distribution evenness (Pearson's r, $P < 0.0001$).

	Number of species	Number of trees	Evenness
Species diversity			
Inverse SDI	0.5865	0.0954	0.3821
Shannon-Wiener	0.6834	0.1448	0.2817
Genus diversity			
Inverse SDI	0.5429	0.2114	0.5741
Shannon-Wiener	0.6431	0.1864	0.5027

SDI, increased street tree diversity was associated more with a greater number of less abundant species and genera than with more even distributions or street tree population size. Furthermore, the percentage of *Acer* spp. in a municipal street tree population was found to be negatively correlated with the Inverse SDI (r = -0.8584) and the Shannon-Wiener Diversity Index (r = -0.8412), meaning street tree diversity increased as the percentage of *Acer* spp. decreased—not surprising given the wide-ranging dominance of *Acer* spp. in species and genus composition.

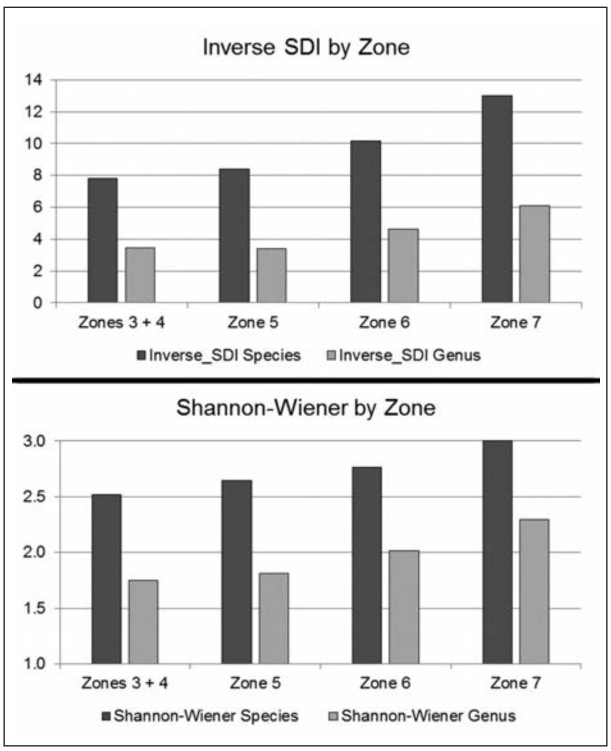

Figure 8. Mean Inverse SDI (Inverse of Simpson's Diversity Index) and Shannon-Weiner Diversity Index values for street tree species and genera by 2012 USDA Plant Hardiness Zones.

DISCUSSION

Relative abundance percentages for prevalent street tree species and genera indicate non-conformance in New Jersey, New York, and Pennsylvania, and in most inventoried municipalities with Santamour's 10% rule for street tree species and 20% rule for street tree genera. Statistics for the Inverse SDI, where an Inverse SDI value of 10 equates with Santamour's 10% rule for species (Sun 1992; Sreetheran et al. 2011) and an Inverse SDI value of 5 equates with Santamour's 20% rule for genera, suggest a satisfactory level of diversity for street tree species and genera in New Jersey and Pennsylvania, but not in New York, with the mean Inverse SDI in New Jersey and Pennsylvania exceeding 10 for species and 5 for genus. However, these diversity-index statistics may be misleading since, for the Inverse SDI and Shannon-Wiener Diversity Index, species diversity is correlated more with the number of species than with evenness in the species distribution, and, for the Shannon-Wiener Diversity Index, although not for the Inverse SDI, genus diversity is correlated more with the number of genera than with the evenness of genera distribution. In other words, the richness of street tree species and genera may be adequate, but street trees are concentrated in the more prevalent species and genera, and the less prevalent species and genera inflate statistics for the Inverse SDI and Shannon-Wiener Diversity Index. These statistics are also influenced by average minimum winter temperature as increases in species and genus diversity for the Inverse SDI and Shannon-Wiener Diversity Index were found to correspond with temperature increases in the 2012 USDA Plant Hardiness Zones. In fact, most municipalities where relative abundance percentages for prevalent street tree species and genera conform to Santamour's 10% rule for street tree species and 20% rule for street tree genera proved to be located in Plant Hardiness Zone 7, which has the warmest average minimum winter temperatures of all zones associated with New Jersey, New York, and Pennsylvania.

Thus, the results suggest insufficient species and genus diversity for street trees at regional, statewide, and municipal levels in three northeastern states. Lack of street tree diversity appears to be more of a pressing concern in New York than in New Jersey or Pennsylvania, owing at least in part to New York's greater preponderance of *Acer* spp., less species and genera richness, and colder average minimum winter temperatures. However, street tree populations in New Jersey and Pennsylvania would also benefit from greater species and genus diversity. Unfortunately, increasing street tree diversity is easier said than done due to a variety of issues discussed in the following, many of which have been addressed by Polakowski et al. (2011) and Lohr (2013).

(1) The public street right-of-way, where most street trees are found, is typically an unhospitable environment for tree growth, health, and longevity. Air pollution, urban heat-island effect, drought, flooding, soil compaction, inadequate soil volume, nutrient imbalances and deficiencies, winter street salt, and utility pruning are some of the many stressors associated with public street right-of-way planting locations. These stressors not only negatively impact street tree growth and accelerate tree mortality but also limit the number of tree species and genera adaptable to such conditions.

(2) Not all tree species adaptable to tough urban conditions make good street trees. Some tree species are poorly suited to be street trees due to their growth and branching habits. For example, silver maples (*Acer saccharinum*) are a fast-growing species tolerant of wet and dry soils and easy to transplant. However, they are also "weak-wooded," prone to rot and decay, and vulnerable to sudden catastrophic branch failure capable of harming persons and property even when in apparently good condition, due to their characteristically narrow, v-shaped branch unions.

(3) Planting evenly spaced, even-aged trees of the same species along streets, avenues, and boulevards to achieve an aesthetically pleasing visual uniformity is a formal planting scheme dating back to sixteenth century Europe (Couch 1992), which continues to be recommended (Gerhold and Porter 2000; Simons and Johnson 2008) and remains operative today. For example, the Monumental Core Framework Plan to preserve Washington, D.C.'s National Mall calls for allées of Dutch-elm-disease-resistant *Ulmus americana* to be planted along major streets and park roads as a unifying landscape element (Sherald 2009), such as 88 'Princeton' American elms planted in 2005 along Pennsylvania Avenue in front of the White House.

(4) Recognizing biodiversity as an important ecological concept does not necessarily translate into actions increasing plant diversity. For example, in a survey of plant nursery workers, Polakowski et al. (2011) found general acknowledgement among respondents that diversity of plant species in landscapes is ecologically important, but also insufficient understanding as to why this is so or about the ways in which diversity relates to landscape practices, such as implementation of Santamour's 10-20-30 rule.

(5) Less prevalent street tree species and genera are often unavailable from local suppliers or are unavailable in large enough numbers to significantly increase diversity. For example, Iles and Vold (2003) found that Iowa, U.S., nurseries overproduce, and Iowa landscape professionals specify a disproportionately small number of species and cultivars; Ries (2009) interviewed numerous municipal foresters who were either unable to obtain from local suppliers less prevalent tree species for new plantings or were forced to obtain these tree species from more distant, non-local suppliers. Sydnor et al. (2010) found that only 3% to 5% of the trees desired by Ohio urban foresters were available from Ohio nurseries.

Many of these issues can be mitigated. Planting conditions for street tree species and genera can be improved by practices such as selecting the correct tree for a planting location based on above- and belowground conditions, watering newly planted trees during establishment and during extreme heat and/or drought, converting tree pits to continuous soil trenches, and using structural soil to facilitate root growth under paving into adjacent lawn areas. Visual uniformity can be accomplished while also satisfying the need for species diversity by grouping trees that are visually compatible based on size, shape, branching density, and foliage texture (Bassuk et al. undated). Better understanding of diversity importance can be achieved through increased education (Polakowski et al. 2011) and perhaps through a change in terminology by stressing the reduction of species and genera overuse (Ball et al. 2007).

A more difficult issue to mitigate may be the availability of less prevalent street tree species and genera. One possible strategy for improving availability is to implement forward contracting, in which the municipality enters into a formal agreement with a tree supplier to grow a stipulated number of trees and tree species at a predetermined price for delivery at a future date. For example, to meet the demands of its MillionTreesNYC tree planting campaign and increase street tree diversity, New York City contracted with a number of nurseries to grow less prevalent tree species and genera (Ries 2009). Another possible strategy is for a municipality to grow its own street trees. For example, in the 1990s, unable to obtain a desired mix of new plantings, Columbus, Ohio, decided to expand its municipal nursery to produce the trees required (Sydnor et al. 2010). However, while forward contracting and municipal tree production may be successful strategies for improving the availability of less prevalent street tree species and genera, as well as increasing street tree diversity, they may not be feasible for many if not most municipalities, especially municipalities of smaller size who must buy from the existing stock of wholesale tree nurseries. This stock was depleted by the "great recession" of 2008–2009, when many tree growers either went out of business or downsized. As a result, there is currently a shortage in the United States of 7.6 cm caliper trees for new plantings until at least 2017 (McClellan 2014; KAT 2015). Nevertheless, the demand for less prevalent street tree species and genera exists and may in fact be increasing, due at least in part to shortages of previously overproduced species, such as red maple, zelkova, and pin oak (Rodda 2014), and also to the need for alternatives to ash in response to emerald ash borer (Zawislak 2015). Therefore, there is a need to better understand the factors influencing nursery growing decisions and how to best encourage nurseries to make diverse species and genera available for sale (Conway and Vander Vecht 2015).

Implicit in this discussion are issues of time and scale. Just as it takes four to five years for growers to get trees to selling size (KAT 2015), increases in municipal street tree diversity cannot be achieved overnight. Apart from preemptive removals to deal with an invasive pest or disease, municipalities are not going to remove healthy, well-performing street trees simply to increase diversity. Therefore, although most municipalities have no shortage of vacant sites where new street trees can be planted, significant structural change in municipal street tree species and genus composition can likely only take place through a consistent, long-term strategy of replacing more prevalent species and genera that have reached the end of their life cycles with less prevalent species and genera appropriate to planting conditions. For municipalities where the street tree population is more aged, the transition to greater diversity may be accomplished more quickly than in municipalities where the street tree population is younger.

Scale will also play a role, since, strictly in terms of the number of trees required, it is an easier task to increase species and genus diversity in a smaller sized municipality that has fewer street trees overall than in a larger sized municipality with more trees. For example, the percentages of *Acer* spp. in the street tree populations of Syracuse, New York, and Hastings-on-Hudson, New York, based on 2014 and 2013 street tree inventories, respectively, are relatively similar, with 35% in Syracuse and 39% in Hastings-on-Hudson. Syracuse is a much larger municipality with a much larger street tree population. Lowering the percentage of maples in Syracuse to 20% of the overall street tree population would entail replacing 4,800 street trees with other street tree genera, whereas to accomplish the same goal in Hastings-on-Hudson would entail replacing only 200 street trees. Lowering the percentage of maples in Syracuse to 10% of the overall street tree population would entail replacing 8,500 street trees with other street tree genera, whereas to accomplish the same goal in Hastings-on-Hudson would entail replacing only 300 street trees.

CONCLUSION

Increasing street tree diversity is not a panacea for maintaining municipal street tree populations at their existing levels and thereby preserving the ecosystem services and social benefits they provide. Many other factors have impacted or will impact street tree populations, including urban development, state and municipal budgets, and climate change. Even so, increasing street tree diversity by making the most common species a little less prevalent is a big step in the right direction toward sustainable street tree management. Based on the results presented here, indicating a current concentration of New Jersey, New York, and Pennsylvania street trees among a relatively small number of species and genera, and in particular the dominance of maples (*Acer* spp.), which are vulnerable to the Asian longhorned beetle, there is a clear need in these states for greater species and genus diversity in statewide and municipal street tree populations. While meaningful impediments exist to increasing diversity, especially in the short-term, it is a policy well worth pursuing at both state and municipal levels.

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F.D. Cowett (corresponding author)
Post-doctoral Associate
Department of Horticulture
Cornell University
134 Plant Sciences Bldg.
Ithaca, New York 14853, U.S.
fdc2@cornell.edu

Nina Bassuk
Cornell University
Horticulture Section, School of Integrative Plant Science
Ithaca, New York 14853, U.S.

Résumé. La diversité des arbres de rues est tenue comme étant un élément clé dans la résistance des populations d'arbres d'alignement face aux ravageurs, aux maladies et aux changements climatiques. L'analyse de la diversité des arbres de rues est considérée comme une constituante de la gestion durable des arbres d'alignement et de la préservation des services écosystémiques et des avantages sociaux que procurent ces arbres. Cet article analyse la diversité des arbres d'alignement de trois États du nord-est des États-Unis, soit le New Jersey, New York et la Pennsylvanie, en examinant les données d'inventaire municipal d'arbres de rues en superposition avec la carte des zones de rusticité des végétaux 2012 du Département de l'agriculture des États-Unis (USDA). Malgré la leçon tirée de la dévastation historique des ormes d'Amérique (*Ulmus americana*), plantés en très grand nombre, par la maladie hollandaise de l'orme, la prise de conscience toute contemporaine de la menace posée aux frênes par l'agrile du frêne (*Agrilus planipennis*), ainsi que les risques que présentent aux érables (*Acer* spp.) et à certaines autres essences d'arbres, le longicorne asiatique (*Anoplophora glabripennis*), les résultats de ces données démontrent une actuelle concentration d'un nombre relativement petit d'espèces et de genres parmi les arbres de rue, en particulier la dominance des érables chez les arbres d'alignement. Les résultats établissent également une indéniable relation entre la diversité des arbres de rues et les températures minimales moyennes plus chaudes durant l'hiver. Par conséquent, il y a une nécessité évidente pour ces trois États d'accroître la diversité des espèces et des genres dans les populations municipales d'arbres de rue ainsi qu'à l'échelle de l'état. Cependant, il existe des obstacles significatifs à l'accroissement de la diversité des arbres d'alignement, particulièrement à court terme.

Zusammenfassung. Die Artenvielfalt von Straßenbäumen wird weitgehend als Schlüsselkomponente für die Widerstandkraft von Straßenbaumpopulationen gegenüber Schädlingen, Krankheiten und klimatischen Veränderungen betrachtet. Die Untersuchung der Artenvielfalt bei Straßenbäumen wird als integrativ in einem nachhaltigen Straßenbaummanagement und zur Erhaltung der Ökosystemleistungen und auch der sozialen Vorteile, die Straßenbäume liefern, eingeschätzt. Diese Studie untersucht die Artenvielfalt bei Straßenbäumen in drei nordöstlichen Staaten der USA—New Jersey, New York, and Pennsylvania— durch eine Analyse der kommunalen Baumkatasterdaten, die stratifiziert sind durch die 2012 USDA Plant Hardiness Zones. Ungeachtet der Erfahrungen aus dem historischen Schwund der überpflanzten Amerikanischen Ulmen durch die Holländische Ulmenkrankheit und dem Bewusstsein gegenüber zeitgenössischer Bedrohungen von Eschen durch den Eschenbohrer (EAB) und von Ahornen und anderen Baumarten durch den Asiatischen Laubbock zeigen die hier präsentierten Ergebnisse auf eine gegenwärtige Konzentration von Straßenbäumen innerhalb einer relativ kleinen Anzahl von Arten und Gattungen und besonders eine Dominanz von Ahornen bei Straßenbäumen. Die Ergebnisse zeigen auch eine positive Beziehung zwischen Straßenbaumvielfalt und wärmeren Durchschnittstemperaturen im Winter. Schlussendlich gibt es einen klaren Bedarf in allen drei Staaten nach größerer Arten- und Gattungsvielfalt in bundesweiten und kommunalen Straßenbaumpopulationen. Dennoch existieren bedeutsame Hemmschwellen, die Vielfalt von Straßenbäumen, insbesondere kurzfristig, zu erhöhen.

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Tools for Staging and Managing Emerald Ash Borer in the Urban Forest

Clifford S. Sadof, Gabriel P. Hughes, Adam R. Witte, Donnie J. Peterson, and Matthew D. Ginzel

Abstract. Advances in control can help municipal foresters save ash trees from emerald ash borer (EAB) [*Agrilus planipennis* (Fairmaire)] in urban forests. Although ash trees of any size can be protected from this pest, cities often do not implement programs because they fail to recognize and act on incipient populations of EAB. In this study, researchers develop a model for predicting ash mortality over an eight-year period, and validated with data from the removal of >14,000 ash trees killed by EAB in Fort Wayne, Indiana, U.S. researchers then developed a sampling scheme to help foresters map their ash trees along the expected progression of ash decline. This model was then used to modify a web-based EAB cost calculator that compares discounted annual and cumulative costs of implementing a variety of management strategies. It was determined that strategies that most heavily relied on saving ash trees were less expensive and produced a larger forest than those strategies that mostly removed and replaced ash trees. Ratios of total discounted costs to discounted cumulative benefits of strategies that saved most ash trees were over two-thirds lower than strategies of proactive tree removal and replacement. Delaying implementation of an ash management program until damage would be visible and more obvious to the community (Year 5 of the model) decreased the cost–benefit ratio by <5%. Thus, delays that rely on the abundance of locally damaged trees to bolster community support do not necessarily diminish the utility of implementing a control strategy.

Key Words. *Agrilus planipennis*; Ash; Ash Tree Decline Model; EAB Cost Calculator; Emerald Ash Borer; Indiana; Pest Management; Projection.

Since its detection in Detroit, Michigan, U.S., in 2002, emerald ash borer (EAB), [*Agrilus planipennis* (Fairmaire)] has spread to 25 states and two Canadian provinces, killing hundreds of millions of ash trees in its wake (Emeraldashborer.info 2015). EAB attacks and kills most North American ash species. Adult beetles lay eggs on the tree bark. Neonate larvae bore into the phloem tissue, and as they develop, consume greater amounts of active xylem tissue of this ring-porous tree species. Beetles take one to two years to complete their life cycle, and with repeated attack, they can functionally girdle and kill their host trees (Cappaert et al. 2005; Wei et al. 2007; Tluczek et al. 2011). With the exception of blue ash, *Fraxinus quadrangulata* (Tanis and McCullough 2012; Tanis and McCullough 2015), all healthy North American species of *Fraxinus* can experience high rates of mortality from this pest. With over eight billion ash trees in North America, the potential for continued devastation will likely make EAB the most destructive pest to invade the forests of this continent (Herms and McCullough 2014).

Ash trees contribute significantly to the canopy of urban forests, with 38 million trees estimated to be present in eastern North America (Kovacs et al. 2010). While ash species account for between 20% and 30% of the urban forest in many cities, it is not uncommon for cities in some regions of the United States (e.g., Colorado and Iowa) to have an ash component of >50% (Raupp et al. 2006; Ball et al. 2007; Sydnor et al. 2007; Sydnor et al. 2011). Thus, the spread of EAB threatens a substantial portion of the urban forest and will cost North American cities well over USD $10 billion to manage (Kovacs et al. 2010; McKenny et al. 2012). The availability of highly effective insecticides has now made it possible to protect trees from EAB with applications of a variety of active ingredients even after damage has reduced canopy density by 50% (Herms et al. 2014). In practice, however, few trees with >30%

canopy thinning are selected to be saved due to a potential loss of structural integrity and aesthetic value after the damaged portion has been removed. Protection provided by a single insecticide application ranges from one to four years and depends on tree size, compound, dose, and solubility of the insecticide formulation. Of these products, a single injection of emamectin benzoate is highly toxic to adult EAB and larvae and can protect ash trees for two to three years (Smitley et al. 2010; McCullough et al. 2011; Flower et al. 2015; Poland et al. 2015). Ongoing studies indicate that even large trees (dbh > 120 cm) can be protected (CSS, MDG pers. obs.).

The spread of EAB and its damage through a forest has been described using a wave analogy (Burr and McCullough 2014). At the cusp of the wave, during the first few years after detection, the density of EAB is low (<10/m^2) and mortality rates of EAB larvae in trees are high (Chen et al. 2012; MacQuarrie and Scharback 2015). During this phase, most ash trees appear healthy and are largely asymptomatic. As densities of larvae increase, the added stress diminishes the capacity of trees to defend themselves, and larval mortality rates decline (Villari et al. 2016). Populations of EAB then begin to grow exponentially as the invasion wave swells to its crest. During the crest phase, enough phloem has been consumed to cause most of the ash trees to express symptoms of canopy thinning (Anulewicz et al. 2007). After EAB has consumed most of the available ash phloem, local EAB populations begin to decline as beetles disperse in search of more suitable ash hosts. Strategies that have been proposed to slow the spread of EAB and its wave of destruction in a forest rely on applying consistent protective measures soon after its detection in an area during the cusp phase of the invasion (Kovacs et al. 2011; McCullough and Mercader 2012; McCullough et al. 2015).

Recent cost–benefit analyses indicate that protecting healthy trees from EAB with insecticides can be more cost-effective than simply removing trees as they die and replanting with resistant trees. Investigations that seek to optimize the net present value of past funds spent on tree maintenance and ecosystem services provided by trees, suggest that cities should focus management efforts on trees with a dbh of at least 30 cm (Kovacs et al. 2010). Attempts to optimize limited monetary resources available for managing trees in a metropolitan area suggest that most of the funds be allocated to protecting trees, and that resources should be pooled across political boundaries to allow cities to benefit from the economy of scale (Kovacs et al. 2014). Several interactive web-based tools have been developed to allow users to customize local cost estimates for both individual trees (McKenny and Pedlar 2012) and urban forests (Vannatta et al. 2012), and the output from these models suggest a similar course of action. Despite this emerging consensus on the utility of protecting ash trees, many municipalities still believe the costs to protect trees are prohibitive, and elect to replace trees after they are killed by EAB.

Clearly, there is a gap in knowledge between the course of action suggested by recent theoretical advances and the practice of EAB management. In this study, researchers characterize how cities currently experiencing EAB outbreaks are managing their ash resource to test the assumption that few cities are opting to protect substantial numbers of trees. The following describes how researchers modified the web-based EAB Cost Calculator (Sadof et al. 2011) with a model to better predict long- and short-term costs of various management strategies at specific stages of the invasion wave. Finally, representative cost estimates are used to predict total discounted costs and forest size resulting from different management strategies implemented before and after damage from EAB is likely to be detected. The goal is to outline a process for systematically assessing the stage of an EAB invasion, and predicting management costs to inform the decision-making process of cities with substantial numbers of ash trees.

MATERIALS AND METHODS

Assessing Current Municipal Management Practices for EAB

On 04 March 2014, a Google™ web search was conducted to determine the number of cities whose EAB programs were highlighted in the news during the preceding 12-month period. The search was conducted using the following key terms: emerald ash borer city protect, or emerald ash borer city management. Management practices were placed into five categories that describe the extent to which cities chose to protect rather than remove trees (Table 1).

Table 1. Strategies employed by municipalities to manage emerald ash borer and their rationales in cities found in a web search covering a 12-month period ending 04 March 2014.

Strategy	Rationale	Percentage of cities (n = 40)
Reactively remove ash	Remove dying ash to prevent hazard	20.0
Proactively removing all ash	Removing ash over time to reduce annual cost	17.5
Protecting only legacy ash	Only healthy trees of historic or significant landscape importance are protected	5.0
Protecting <50% of healthy ash	A substantial proportion beyond legacy ash trees are protected	40.0
Protecting >50% of healthy ash	Most of the healthy ash are protected	17.5

Estimating Large- and Small-Scale Treatment Costs

In order to better understand the variation in costs associated with emamectin benzoate treatments, researchers sought to determine if there was a relationship between the cost of application and the number of trees treated. From September through December 2014, information was gathered from public records on the prices paid per 2.54 cm dbh to treat municipal ash trees in 27 municipalities in Indiana, Illinois, Missouri, Minnesota, and Ohio, U.S. Researchers also gathered information on the number of trees in each bid and then assessed the relationship between the actual bid price for an emamectin benzoate treatment and the number of trees.

Development and Validation of a Model of Ash Forest Decline

Researchers developed a model (Figure 1) that used percentages of ash mortality to predict the accumulated number of ash trees to be removed because they were in poor condition (i.e., losing more than 30% of their canopy to EAB). The default settings of the calculator assumed the city was early in the cusp phase of the invasion wave, with 1% of ash trees in the poor category. Each year the percentage of affected ash trees doubled, so that by the fifth year, it reached 16%, or approximately one out of six ash trees. The percentage peaked in year eight, with 100% of the trees characterized as being in poor condition. In the last three years of the model, 84% of the trees reached this level of decline. This pattern of ash destruction is premised on a hypothetical rise and fall of the maximum EAB population that responds to the available ash resource (Figure 1).

Researchers tested the ability of the model to predict ash tree decline in two ways. First the decline of untreated urban ash trees was compared, from 2010 through 2015, in Lafayette, Indiana, U.S., where EAB was first detected in 2011, and on the north side of Indianapolis, Indiana, where EAB was detected in 2006. In each city, approximately 100 ash trees with a dbh between 14 to 40 cm were selected. Each summer, 50 ash trees were visually assessed as good (<10% canopy decline) and 50 were ranked as fair (10% to <30% canopy decline) (Hughes et al. 2015). Trees were ranked as poor if they had more than 30% canopy thinning, and those in the critical category exceeded >80% canopy thinning. The capacity of the model to predict

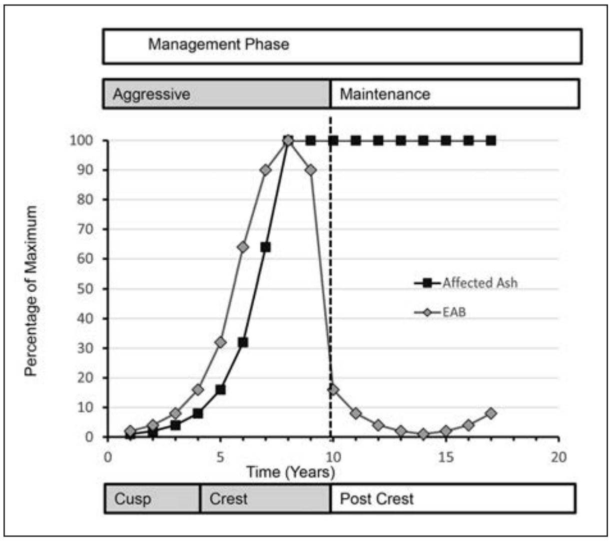

Figure 1. An invasion wave model to predict borer density and the decline of ash trees and guide management intensity in different stages of the initial invasion of the emerald ash borer. Affected ash trees represent those trees that are too damaged to be saved (>30% canopy thinning) with an insecticide treatment.

rates of ash decline in each forest at each survey date was determined by comparing observed and predicted numbers of ash trees entering at least the poor category with a Kolmogorov-Smirnov test (Gotelli and Ellison 2004). This method was also used to determine the ability of the model to predict the number of ash trees removed in a city during the initial EAB invasion. Here, predicted values were compared with actual removals for the City of Fort Wayne, Indiana, from the first year EAB was detected (2006) through the year when the last of the 14,403 untreated ash trees were removed (2013).

Modification of a Web-Based Cost Calculator

The modified EAB Cost Calculator (Sadof 2016) is based on a previous version (Sadof et al. 2011) that used a local inventory of ash trees and local estimates of pricing for treatment, removal, and replacement of trees based on tree size. Users can choose predefined strategies or create their own strategy that chooses the percentage of trees in each size class that will be removed, replaced, or protected with insecticides. Annual and cumulative costs of up to three management strategies are plotted for 25 years. Benefits of each management strategy are compared by plotting the expected total dbh of all surviving ash trees resulting from each management strategy. To account for the time value of money, the modified calculator uses the same formula as in the first version to calculate the present value of costs (Rose et al. 1988).

The local tree inventory is used to create a matrix of trees that is applied in an iterative approach to simulate annual tree growth and costs for 25 years. Trees within each size class are assigned equally spaced starting sizes that are approximated by dividing the span width by the number of trees in a size class. So, if there are 600 trees in the 15–30 cm category, the calculator creates a matrix of 600 trees with sizes 0.024 cm apart. Annual growth of surviving trees is approximated by a linear model that adds 1.143 cm of dbh per year based on a linear estimate of growth quantified for ash trees (Peper et al. 2014). When trees are "killed" by EAB or through planned removal, the model randomly selects individuals in each size class that will be removed and replaced. Two additional 15 cm growth spans are built into the model to receive trees that grow beyond the last size class provided by the original tree inventory.

Cities whose trees have already begun to show damage can stage their infestation from the percentage of ash trees in the poor category and start the simulation at a more relevant point in the eight-year ash forest decline model. The EAB Cost Calculator allows cities to start as late as six-years into the invasion. By the seventh year of the cycle, when 64% of the trees are beyond saving, ash management options are restricted to removal and replacement. During this phase, the model's predictions are not likely to be accurate because it cannot predict the distribution of live trees left to treat.

Municipal arborists can adjust the frequency of pesticide applications to be most aggressive during the cusp and crest phases of the invasion as EAB populations are building and threatening tree heath. After 10 years, two years after all untreated trees are rendered beyond saving, there is little ash phloem to support the beetles. As such, populations of EAB are presumed to be present, but at a much lower level. For this reason, the EAB Cost Calculator switches from an aggressive to a maintenance phase of management after this time (Figure 1). Operationally, in the aggressive phase, all trees designated for protection are treated frequently enough to provide maximum protection. In the post-crest phase, pesticide applications are replaced by an integrated pest management approach that includes monitoring annually for fresh symptoms of EAB attack, such as woodpeckers or bark splits. Detection of these symptoms triggers a round of insecticide application, before substantial, additional canopy thinning occurs. Reduced costs are approximated in the model by reducing the frequency of pesticide application in this maintenance phase.

Defining Management Strategies

A fictitious forest, composed of 1,600 ash trees, was used to estimate management costs and forest growth over time (Table 2). The size class of this forest was skewed toward larger trees to account for fewer ash trees being planted after EAB was detected in 2002. Approximately two thirds of the ash trees in this forest had a dbh > 30 cm. The costs of six common management strategies (Table 3) represent a range of management combinations of tree removal, replacement, and treatment with insecticides.

It was assumed that all trees treated in this simulation would be treated with emamectin benzoate once every three years during the aggressive management phase and once every five years in the maintenance phase. Researchers assumed that over the 25-year period, only 2% of the ash trees treated with this pesticide would die due to insecticide failure because of the high efficacy of this product (Herms et al. 2014). Trees dying due to insecticide failure were removed and replaced. Researchers chose the commonly used mortality rate of 5% to estimate loss of replacement trees due to transplant failure (McPherson et al. 2006). An annual mortality rate of 2% was applied to all trees to approximate normal loss. The cost to plant, stake, and mulch a new 3.2 cm dbh tree was set to $400. Rates approved for the City of Indianapolis in December 2014 were used to estimate the costs of removing a tree and grinding the stump (Table 2). A 3% discount rate was used to estimate the present value of costs.

Table 2. Size class distribution of ash forest and cost ($USD per cm dbh) to remove and grind the stump of an ash tree of each size used in the model simulation. Costs were based on rates for the City of Indianapolis, Indiana, U.S., in 2014.

Size span (cm dbh)	Ash trees in forest	Removal and grinding cost
3–8	50	$14.00
8–15	200	$14.00
15–30	300	$14.75
30–46	400	$18.00
46–61	300	$21.75
61–76	200	$25.10
76–91	100	$30.50
>91	50	$36.00

COSTS AND BENEFITS OF MANAGEMENT PROGRAMS

Researchers ran simulations early and late in the wave of ash decline in order to predict discounted costs from a current inventory and assessment of tree quality 25 years into the future. The simulation of an early intervention began during the first year of the ash-decline wave when EAB would be difficult to detect with only one percent of the trees exhibiting the obvious symptom of losing >30 % of their canopy. This is the scenario of a city that would be able to initiate a program before symptomatic trees were apparent in a community. Simulation of a late intervention was conducted during the fifth year of the ash-decline wave, when the presence of EAB would be easy to detect with 16% of trees expressing obvious symptoms. Here, the same inventory of trees in Table 2 were used, but with the infestation staged at Year 5 in order compress the time available for the city to remove dying trees. Both the early and late simulations were conducted using a cost of $3.94/cm dbh to represent the cost for treating low numbers of trees, and at $1.94/cm dbh to represent discounts given for bulk purchase of treatment services.

Necessary adjustments were made to some of the EAB Cost Calculator outputs in order to compare costs incurred when initiating a management program either early or late in the ash-decline wave. The early simulation, starting in Year 1, provided an accurate estimate of the present value of costs and tree growth throughout the 25-year cycle of each management scenario in an urban forest. For the late simulation, researchers assumed no added cost for EAB management during the first

Table 3. Management strategies used to compare costs and benefits in the EAB Cost Calculator 3.0 simulation. See text for more details about how costs were calculated.

Management strategy	Detail
Proactively replace ash	Proactively remove and replace all ash trees before EAB has damaged them beyond the point of rescue.
Reactively replace ash	Remove and replace ash trees as the model of ash decline predicts that emerald ash borer will damage them beyond the point of rescue.
Save 50%	Treat half of the ash trees with insecticide and proactively remove and replace the rest.
Save 80%	Treat 80% of the ash trees with insecticide and proactively remove and replace the rest.
Treat <30 cm dbh	Treat ash trees with insecticides when dbh > 30 cm. Remove and replace the rest. This strategy optimizes previous municipal investment in larger trees and the benefits of ecosystem services they provide (Kovacs et al. 2010).
Treat all	All ash trees are protected by insecticide treatment.

four years and started projecting costs in Year 5 when 16% of ash trees were damaged beyond saving and thus removed. The size of the standing ash forest during these first four years was approximated using the size of treated ash trees calculated for the early simulation. In this way, the starting size of the forest in the late simulation could account for the modest growth and limited removal of trees that would occur before implementing an EAB management plan. This limited removal early in the invasion wave would compress tree removal costs in Years 5–8 of the simulation.

Benefits of each management plan were presented in two ways: First, the forest size was tracked and plotted as trees grew over 25 years under each management regime for the early- and late-intervention scenarios. These plots can be useful for managers who make decisions based on forest size. Second, researchers measured the sum of the total tree dbh discounted annually at 3% over the 25 years for the early and late scenarios. This summation represents the accumulated discounted benefits of each management strategy over time in terms of discounted total tree diameters. The total discounted costs and accumulated discounted benefits were used to calculate a cost–benefit ratio that was expressed in dollars/m dbh.

RESULTS

Assessing Current Municipal Management Practices for EAB

Of the 40 cities encountered in the web search, 37.5% elected to remove rather than save any ash trees. Five percent of cities chose to save only ash trees of historic or landscape importance, whereas 40% of cities saved less than half their ash trees. Articles that provided reasons for lack of treatment used words like: lack of guarantee, high price, lack of confidence that product will save trees, trees are already too damaged. Only 17.5% of cities chose to save more than half of their healthy ash trees (Table 1).

Estimating Large- and Small-Scale Treatment Costs

Of the 27 cities contacted for this study, 12 used emamectin benzoate to treat their trees (Figure 2). Six cities with bids that included <150 trees paid an average of \$3.29 + 0.56/cm dbh to have a contractor treat the trees. Six cities with >150 trees in their bid paid an average of \$1.82 + 0.09/cm dbh to have trees treated. This average bid was 44.7% lower than bids to treat less than 150 trees.

Validation of Ash-Decline Model

The ash-decline model accurately predicted the annual number of ash trees removed from Fort Wayne, Indiana, in five out of eight years (Figure 3a) (K-S = 0.0136, $P < 0.01$). In contrast, the model accurately predicted the decline of those trees ranked as good or fair condition to poor in all but one year of monitoring efforts in Lafayette (K-S = 0.146, $P < 0.01$) and Indianapolis (K-S = 0.156, $P < 0.01$) (Figure 3b; Figure 3c). Higher rates of ash tree decline in 2012 may have been due to the historic drought that occurred during that year in the Lafayette and Indianapolis area.

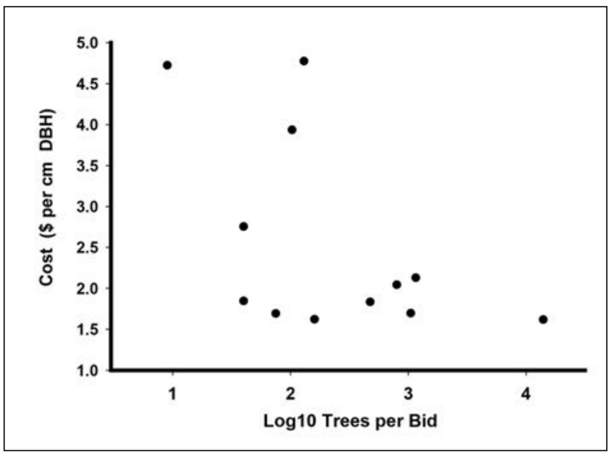

Figure 2. Cost per 2.5 cm dbh of bids for treating ash trees with emamectin benzoate and number of trees per bid determined by surveying the public records of 27 cities in Illinois, Indiana, Minnesota, Missouri, and Ohio, U.S., from September to December of 2014. Currency is in \$USD.

Simulated Costs and Benefits of Management Programs

The highest annual costs for managing ash trees were incurred when trees were removed as they became unsalvageable (reactive removal), followed by proactive removal of ash trees (Figure 4a). Lower annual costs were predicted for treating all the ash trees or trees

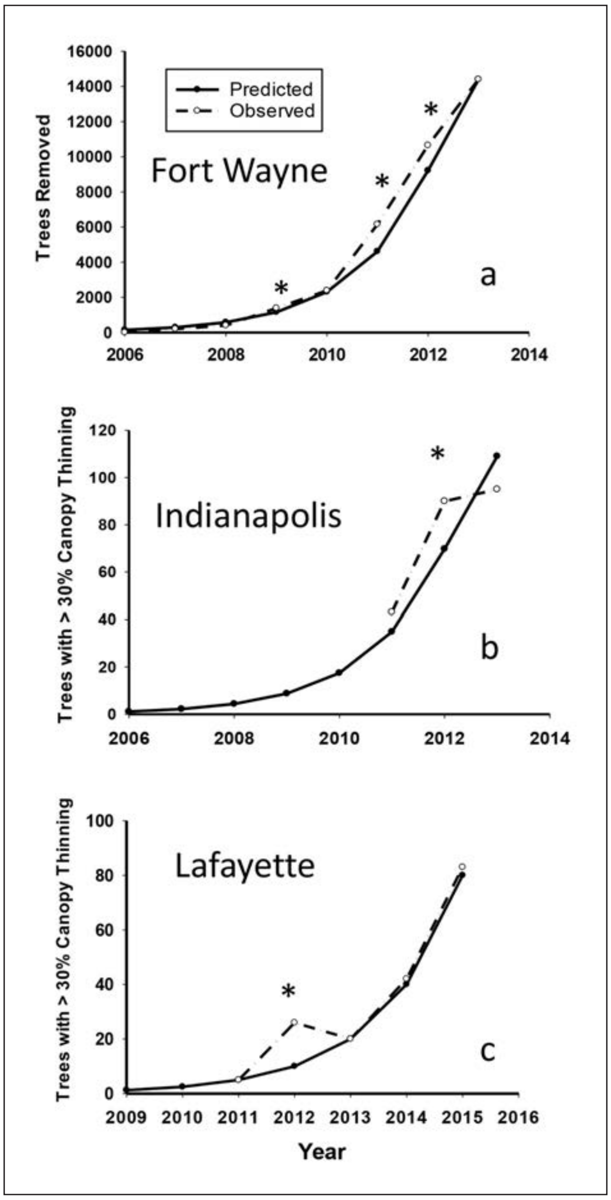

Figure 3. Ash tree decline predicted by the invasion wave model plotted with a) observed rights-of-way tree removals by the City of Fort Wayne, Indiana, U.S., b) numbers of good and fair trees declining to poor quality (>30% canopy thinning) in Indianapolis, Indiana, and c) Lafayette, Indiana. An asterisk (*) indicates years with significant difference from the predicted distribution *(P* < 0.05) with a Kolmogorov Smirnov test.

with a dbh > 30 cm. Delaying the implementation of a management strategy to Year 5 of the cycle (Figure 4b) compressed the costs of tree removal and replacement, resulting in substantially higher peak annual costs for proactively (107.8%) and reactively (39.4%) removing and replacing trees. Delaying the management strategies caused less of an increase (16.0%) for the peak annual cost of treating all trees with a dbh > 30 cm because fewer tree removals and replacements were compressed into the remaining years of the ash-decline wave.

After 25 years, the projected size of forests whose ash trees were removed and replaced were less than one-third the size of those whose ash trees were treated early (Figure 5a) or late (Figure 5b) in the wave of ash tree decline. Ratios of total discounted costs to total discounted tree diameters were also greatly lower in management strategies that saved trees (Table 4). When the cost of treating an ash tree was \$3.94/cm, the cost–benefit ratios of protecting most ash trees was roughly half of those from proactively removing and replacing all ash trees. Cutting the price of treatment in half

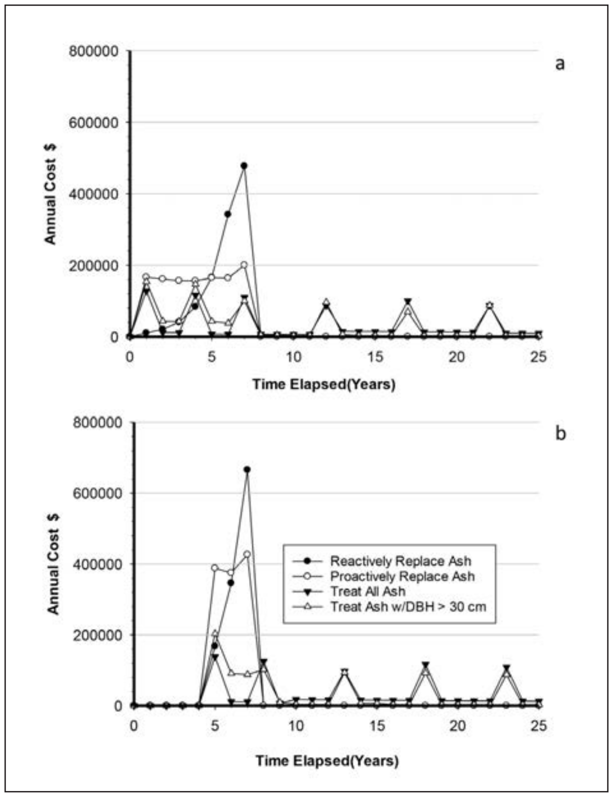

Figure 4. Annual costs of implementing selected management programs of a 1,600 ash tree forest predicted by the EAB Cost Calculator 3.0 when initiated in the a) first year of the invasion cycle (1% of ash trees beyond saving) and b) fifth year of the invasion cycle (16% of ash trees beyond saving). Cost of treatment assumes a bulk price for emamectin benzoate of \$1.94/cm, and default calculator values for the cost of tree removal and replacement based on Indianapolis estimates. Currency is in \$USD.

($1.97/cm) reduced the cost–benefit ratios of protecting most ash trees by over two-thirds. However, delaying the implementation of management strategies to Year 5 of the ash-decline cycle had little effect on the cost–benefit ratios for strategies requiring treatment. These delays reduced the cost–benefit ratio by <5%.

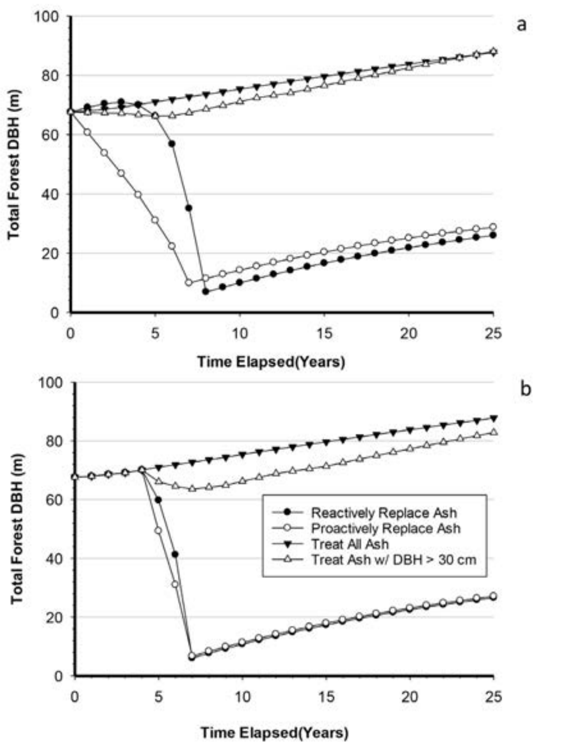

Figure 5. Tree growth (total dbh) of 1,600 ash forests managed under selected regimes predicted by the EAB Cost Calculator 3.0 when initiated in the a) first year of the invasion cycle (1% of ash trees beyond saving) and b) fifth year of the invasion cycle (16% of ash trees beyond saving).

DISCUSSION

The investigation of ash decline in the wake of EAB invasion support the model used by the EAB Cost Calculator to predict decline in urban ash forests during the initial EAB invasion. Although the model over or underestimated tree removals in Fort Wayne 37.5% of the time, it was accurate in Year 5, when nearly 84% of ash trees were still standing, and in Year 8, when the last of the untreated ash trees were removed. As such, the model approximated the compression of removal costs in the last three years of the removal cycle in Fort Wayne. Thus, the rate at which the EAB Cost Calculator anticipates the removal of untreated ash trees closely approximates the actual costs incurred by one city with a substantial number ash trees in its urban forest.

In contrast, the model of ash-tree decline accurately predicted when an equal number of ash trees ranked as good and fair had deteriorated to the ranking of poor (>30% canopy thinning) in all years but 2012, when there was an historic drought in Indiana. From an operational perspective, this model gives managers a tool to map their position in time on the ash-decline curve. This focus on the accumulation of ash trees with >30% canopy thinning can help municipal foresters gather support for treatment efforts before most of the ash trees become unsalvageable.

Investigations of public responses to pest injury on plants suggest that the public could easily detect 30% canopy thinning on an individual tree. Consumer surveys show that as little as 10% defoliation, distortion, or discoloration render plants aesthetically unacceptable to the general public (Sadof and Raupp 1997; Sadof and Sclar 2002). Studies of street trees in particular found that as little as 5%

Table 4. Ratio of total discounted costs associated with emerald ash borer management per meter of trunk diameter of standing trees after implementing selected emerald ash borer management strategies for 25 years in a 1,600-tree forest. The model assumes a 3% discount rate and treating ash trees every three years through the crest of the EAB invasion wave and every five years thereafter. Reduction is the decrease in the ratio when the cost of treatment is reduced from $3.94 to $1.97 per cm dbh. Currency is in $USD.

Time step of initiation	Year 1			Year 5		
Cost ($USD) per cm for treatment	$3.94	$1.97	Reduction (%)	$3.94	$1.97	Reduction (%)
Reactively replace	$1,758.28	$1,758.28	0.00	$1,933.32	$1,933.32	0.00
Proactively replace	$2,178.42	$2,178.42	0.00	$1,983.20	$1,983.20	0.00
Treat >30 cm dbh	$973.60	$641.19	34.14	$981.56	$625.40	36.28
Treat 50%	$1,176.18	$883.30	24.90	$1,148.08	$886.33	22.80
Treat 80%	$1,050.47	$677.67	35.49	$962.17	$648.75	32.57
Treat all	$1,056.88	$592.97	43.89	$952.74	$579.32	39.19

defoliation by orange striped oakworm [*Anisota senatoria* (J.E. Smith)] (Lepidoptera: Saturniidae) to an individual oak tree was sufficient to trigger a complaint call and a request for pesticide treatment (Coffelt and Schultz 1990). Furthermore, other studies of forest vistas suggest that the general public could discern between forests with as little as 10% difference in trees with significant canopy dieback (Buyhoff et al. 1992). Thus, it is quite likely that the public will notice the presence of declining ash trees in Year 5 of the eight-year cycle when the model predicts that only 16% of trees would have been rendered unsalvageable.

It is very difficult to gain support for managing new invasive insects, like EAB, in urban forests, despite effective and proven treatment options. Indeed, the examination of news coverage of municipal responses to EAB indicates that nearly four of five cities elected to remove all their ash or save less than half of the healthy ash trees. This low rate of ash protection may in part be explained by a failure to adequately communicate the risks EAB bring to a community in the absence of a treatment program. This has been the case for gypsy moth (*Lymantria dispar*), where public opposition to area wide management approaches can stem from an inability to communicate risks and benefits of managing this serious forest defoliator (Nealis 2009; Tobin et al. 2012; Bigsby et al. 2014). The current study's procedure for estimating the decline of good and fair ash trees could be used with the ash-decline model described herein to communicate current and future risk of ash tree destruction in a local community. When used with the EAB Cost Calculator, this information can inform discussions early in the invasion process while there is still time to save healthy ash trees.

From a safety perspective, tracking the decline of good and fair ash trees to the poor level focuses attention on ash trees as they become more likely to lose limbs. Recent demographic studies of ash trees in EAB-infested areas indicate that ash trees reach 30% canopy thinning before they become hazard trees and this measure of decline is a good predictor of EAB presence (Hughes et al. 2015; Persad and Tobin 2015). Thus, framing management objectives in terms of reducing the accumulation of poor ash can also prevent hazards associated with failure of ash tree in rights-of-way and other public spaces.

The current survey of public ash treatment records indicate that cities could substantially lower the treatment price paid per dbh of ash by pooling their efforts to treat more trees. Simulations run with the EAB Cost Calculator suggest that lowering treatment costs reduces both annual and total discounted costs of plans focusing on saving ash trees. These findings are consistent with others (Kovacs et al. 2014) that go so far as to suggest that municipalities consider crossing political boundaries to benefit from economies of scale.

It is not surprising that after 25 years, the management plans that save all ash trees produce substantially larger forests than those that remove and replace all. Cost–benefit ratios associated with protecting ash trees from EAB in these forests can be over two-thirds lower than for proactively removing and replacing ash trees. These advantages are not likely to be lost if treatment is delayed to Year 5 of the eight-year progression of ash decline, and results in <5% increase in the cost–benefit ratio. Thus, the results of the current study support the findings of others that show saving ash trees is more cost-effective than removing and replacing them (McCullough and Mercader 2012; Vannatta et al. 2012; Kovacs et al. 2014). Moreover, even several years after the initial EAB invasion, there are enough healthy ash to retain advantages of an intervention program (Epanchin-Niell and Wilen 2012). Economic advantages of protecting ash trees can only increase as area-wide approaches are developed that lower costs of protection by treating only a fraction of the urban ash trees (McCullough and Mercader 2012).

In conclusion, in the absence of pesticide treatment, the arrival of EAB into an urban forest will destroy ash trees in a predictable manner that can be described by an eight-year model of ash decline. Municipal foresters can stage the level of EAB infestation by monitoring a subsample of trees, ranked as good and fair, and then use the model to inform management decisions. This information, along with a local tree inventory and cost estimates, can be used with the EAB Cost Calculator to help convince communities of the advantages of treating ash trees to save them from a destructive pest.

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Clifford S. Sadof (corresponding author)
Department of Entomology
Purdue University
West Lafayette, Indiana 47907-2089, U.S.
csadof@purdue.edu

Gabriel P. Hughes
Department of Entomology
Purdue University
West Lafayette, Indiana 47907-2089, U.S.

Adam R. Witte
Department of Entomology
Purdue University
West Lafayette, Indiana 47907-2089, U.S.

Donnie J. Peterson
Department of Entomology
Purdue University
West Lafayette, Indiana 47907-2089, U.S.

Matthew D. Ginzel
Department of Entomology
Purdue University
West Lafayette, Indiana 47907-2089, U.S.

Résumé. Les progrès en matière de contrôle peuvent aider les arboriculteurs municipaux à sauver les frênes de l'agrile du frêne (*Agrilus planipennis* [Fairmaire]) dans les forêts urbaines. Bien que les frênes de toutes dimensions peuvent être protégés contre ce ravageur, les villes mettent rarement en place des programmes parce qu'elles ne reconnaissent pas l'urgence d'agir sur les populations naissantes de l'agrile. Dans cette étude, les chercheurs ont élaboré un modèle pour prédire la mortalité des frênes sur une période de huit ans, modèle validé par des données provenant de l'abattage de plus de 14 000 frênes tués par l'agrile à Fort Wayne en Indiana, États-Unis. Les chercheurs ont par la suite développé un procédé d'échantillonnage afin d'aider les forestiers urbains à dresser la cartographie de leurs frênes en considérant la progression anticipée de leur déclin. Ce modèle a alors été utilisé pour modifier un calculateur en ligne des coûts liés à l'agrile, comparant les coûts actualisés annuels et cumulatifs de mise en œuvre d'une variété de stratégies de gestion. Il a été déterminé que les stratégies qui misaient le plus fortement sur la sauvegarde des frênes étaient moins onéreuses et produisaient des arbres plus gros que les stratégies qui se contentaient surtout d'abattre et de remplacer les frênes. Les ratios des coûts totaux actualisés par rapport aux avantages cumulatifs actualisés des stratégies qui sauvegardaient la plupart des frênes étaient inférieurs de plus des deux tiers par rapport aux stratégies proactives d'élimination et de remplacement des arbres. Le fait de retarder la mise en œuvre d'un programme de gestion des frênes jusqu'à ce que les dégâts soient visibles et plus évidents par la communauté (année 5 du modèle) a diminué le rapport coût-bénéfice de 5 %. Par conséquent, les délais qui se fondent sur l'abondance d'arbres endommagés localement pour recevoir le soutien des communautés ne diminuent pas nécessairement l'utilité implanter une stratégie de contrôle.

Zusammenfassung. Fortschritte bei der Kontrolle können kommunalen Förstern helfen, die Eschen in urbanen Wäldern vor dem Befall mit dem Eschenbohrer (EAB) zu retten. Obwohl Eschen jeder Größe vor diesem Schädling geschützt werden können, haben Städte oft keine Programme implementiert, weil sie nicht in der Lage sind, die beginnende Käferpopulationen von EAB zu erkennen und zu handeln. In dieser Studie entwickeln Forscher ein Modul zur Vorhersage von Eschen-Mortalität über eine Periode von acht Jahren, welches durch die Daten aus der Beseitigung von > 14.000 abgestorbenen Eschen in Fort Wayne, Indiana, U.S. validiert wurde. Die Forscher entwickelten dann ein Probennahmesystem, um Förstern zu helfen, ihre Eschen entlang der erwarteten Progression des Eschenrückgangs zu kartieren. Dieses Modell wurde dann verwendet, ein web-basiertes EAB-Kosten-Kalkulationsprogramm zu modifizieren, welches die herabgesetzten jährlichen und die kumulativen Kosten der Implementierung einer Auswahl von Managementstrategien miteinander vergleicht. Es war bestimmt, dass die Strategien, welche stark auf der Rettung von Eschen basieren, weniger Kosten und größere Waldflächen produzieren als solche Strategien, die hauptsächlich befallene Eschen entfernen und ersetzen. Die Verhältnisse der totalen diskontierten Kosten zu den diskontierten kumulativen Vorteilen derjenigen Strategien die die meisten Eschen retteten, waren über zwei Drittel niedriger als Strategien zur proaktiven Eschenbeseitigung und Ersatzpflanzung. Eine verzögerte Implementierung eines Eschen-Managementprogramms erst bei der ersten Sichtbarwerdung von Schäden und damit deutlicher erkennbar für die Kommune (Jahr 5 des Modellversuchs) verringerte das Kosten-Nutzen-Verhältnis um < 5 %. Daher können Verzögerungen, die zunächst ein Massenaufkommen von geschädigten Bäumen brauchen, um eine Unterstützung in der Kommune zu erhalten, nicht unbedingt die Nützlichkeit der Implementierung solcher Programme geringer werden lassen.