



Management slows down invasion by non-native plants but does not prevent community change over 35 years in urban forests of the Midwestern USA



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ABSTRACT

Urban forests are threatened by land cover change and invasive species but little is known about how they are changing over time. Furthermore, although management is sometimes used to maintain native communities, we have little information about the long-term outcomes of forest management actions. In this study, we examined current status and long-term changes in 35 urban forest plots in DuPage County, Illinois (USA). From 1979 to 2014, in five-year intervals, presence/absence of all woody species was recorded in the plots. Environmental conditions in and surrounding the plots were measured in the field or with GIS. With these data, we analyzed trends and changes in the community, looked for evidence of biotic homogenization, and examined drivers of non-native species. We found a significant change in community composition over time but no evidence of biotic homogenization in either managed or unmanaged plots. There was a significant increase in non-native species richness over time but no change in richness of other species groups. Vegetation management such as fire prescription and clearing seemed to somewhat decrease the number of non-native species, but did not increase the number of more desirable species. Distance to agriculture and deer control frequency determined the variation in both non-native species abundance and richness. Overall, we conclude that the management activities were able to slow down the spread of some undesirable species but did not prevent their increase over time. More consistent management, as well as collaboration with stakeholders outside the forest preserves, is likely needed to maintain native communities.

1. Introduction

Understanding variation in plant community composition across space and time is a central goal of vegetation ecology (Anderson et al., 2011). This topic is particularly crucial in urban forests due to the worldwide trend toward urbanization and threats to urban green spaces (McPherson et al., 1997). Urban forests include street trees, trees in city parks, green belts, and forest patches inside or around urban areas (Moeller, 1977). These forests provide a variety of critical ecosystem services, harbor a high diversity of species, and play an important recreational, spiritual, and cultural role in the lives of city residents (Bolund and Hunhammar, 1999). Evaluating long-term dynamics of urban forests will help determine whether forests are resilient to disturbances and inform choices between competing demands of anthropogenic use versus conservation of biodiversity.

Many factors contribute to changes in urban forest structure and composition. These include direct human modification of the forest and

‘natural’ factors such as disturbances, herbivore populations, and climate change (Keleş et al., 2017). The latter set of factors can be the direct or indirect result of human activity as well. For example, forest fires are often directly suppressed due to safety concerns, leading to dense-canopied forests dominated by shade-tolerant species (Nowacki and Abrams, 2008). Indirectly, fire regimes can be altered as a result of non-native species spread (Flory et al., 2015). Another indirect impact of humans can be seen in the increased deer populations resulting from intentional predator elimination in most ecosystems (Terborgh and Estes, 2010), including the north-central and eastern temperate forests of the United States (Hobbs, 1996; Russell et al., 2001). Dense deer populations have profound effects on forest regeneration and composition (Tilghman, 1989) and pose significant challenges to the conservation of native plant communities (Griggs et al., 2006) by facilitating the abundance and spread of non-native invasive species (Shen et al., 2016).

Urbanization and other land conversion can also drive forest

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change. Many native canopy species are sensitive to impacts associated with urbanization (e.g., fragmentation, competition with non-native plants, hydrological changes; [Didham et al., 2007](#)) and agricultural land use (e.g., high N deposition due to fertilizer use and fossil fuel combustion; [Frey et al., 2004](#)). Land use changes may lead to loss of native species ([Czech et al., 2000](#)) and invasion by non-native species ([Pennington et al., 2010](#)).

The loss of native species and increase in non-native species can lead to biotic homogenization—the increase in compositional similarity among formerly distinct communities that reduces the ecological uniqueness of local ecosystems ([Blair, 2001](#)). Because homogenization often replaces a number of local species with a smaller number of non-native and widespread (ruderal) species, it reduces spatial diversity ([McKinney and Lockwood, 1999](#)). Although many of these non-native species thrive in disturbed environments (e.g. urban forests), some also invade, and thus homogenize, relatively undisturbed natural areas ([Williamson, 1996](#)). Homogenization has been documented in forests over the entire northern USA Great Lakes region ([Schulte et al., 2007](#)) and in other parts of the world ([Naaf and Wulf, 2010](#); [Keith et al., 2009](#)). In temperate forests of the Midwestern USA, some important drivers of homogenization include fire suppression ([Hanberry et al., 2012](#)), deer overpopulation ([Rooney and Waller, 2004](#)), and invasion by non-native plants ([McKinney and Lockwood, 1999](#)). Fire suppression can lead to mesophication, which in turn can lead to biological homogenization ([Hanberry et al., 2012](#)). Mesophication of broadleaf forests throughout the eastern United States is occurring via replacement of fire-stabilized oaks and pine by fire-sensitive species, specifically shade-tolerant mesophytic species ([Nowacki and Abrams, 2008](#)). Management techniques may be able to counteract homogenization of forests but little research has directly addressed their effectiveness in doing such.

Forest management is often used to impede loss of native species, invasion by non-native species, and biotic homogenization. Restoration-focused management regimes include prescribed fire ([Bowles et al., 1994](#)), understory thinning, and non-native shrub removal or clearing ([Lorimer et al., 1994](#)). However, vegetation responses to management can vary widely, depending on both the nature of the management and initial stand condition ([McRae et al., 2001](#)). Long-term effects of management on forest structure and composition are largely unknown ([Arthur et al., 2015](#)) or can be counter-intuitive. For example, some research shows that even with multiple fires, changes in stand structure are often too minor to significantly alter the course of succession ([Hutchinson et al., 2012](#)). Other studies show that, even within fire-tolerant vegetation, fire can promote non-native species ([D'antonio and Meyerson, 2002](#)) if it is intense enough to reveal bare mineral soil and open previously closed canopies ([Symstad et al., 2014](#)). On the other hand, deer management may be more effective than fire, as studies show that non-native vegetation cover, abundance, growth, and population growth rates decline when deer are controlled or eliminated ([Dávalos et al., 2015](#); [Nuzzo et al., 2017](#)).

Most studies to-date have explored the role of individual factors such as urbanization ([Pennington et al., 2010](#)), management ([McGarvey et al., 2013](#)) or local environmental conditions ([Oliveira-Filho et al., 2001](#)) on forest communities. Furthermore, most ecological studies are limited to a relatively short time period, often just a few years ([Hutchinson et al., 2012](#)). However, managers need information about the impact of all of these factors over longer time periods to make informed decisions about the future of urban forests. This research attempts to fill that gap in knowledge by examining changes in community composition over a 35-year period (1979–2014) in forests in DuPage County, Illinois. Using data from 35 forest plots in an urbanizing landscape, we explore community change, homogenization, and changes in individual species and groups of species (e.g. non-native, ruderal, and shade-tolerant species, which all contribute to biotic homogenization in our study area). Specifically, we ask the following questions:

1. Has the woody plant community in DuPage County Forest Preserves changed over 35 years? Have the communities become more homogenized?
2. What are the trends and patterns in individual species occurrence over time? How have particular woody species or groups of woody species, such as native and non-native species, shade tolerant and intolerant species, and habitat specialists, changed over time?
3. What are the most important drivers of invasion by non-native species?
4. What is the role of management in the changing forest? Is management effective in controlling the non-native species and maintaining the conservation value of plots? Could management techniques prevent biotic homogenization?

Our work contributes unique insight into long-term forest management and plant invasions in an urban landscape.

2. Methods

2.1. Study area

This study makes use of a dataset collected over a 35-year period in DuPage County in northern Illinois (USA) ([Fig. 1](#)). DuPage County is one of the “collar” counties of the Chicago metropolitan area. It has a population of 916,924 residents according to the 2010 decadal census and is the second most populated county in Illinois; its population is expected to grow in the future because of its proximity to Chicago ([DuPage County Statistical Profile, 2011](#)). The study region has a humid continental climate with mean temperatures of 23 °C in July and –10 °C in January, and mean annual precipitation of 92 cm (based on climate normals from Illinois State Climatology). Soil is mostly very

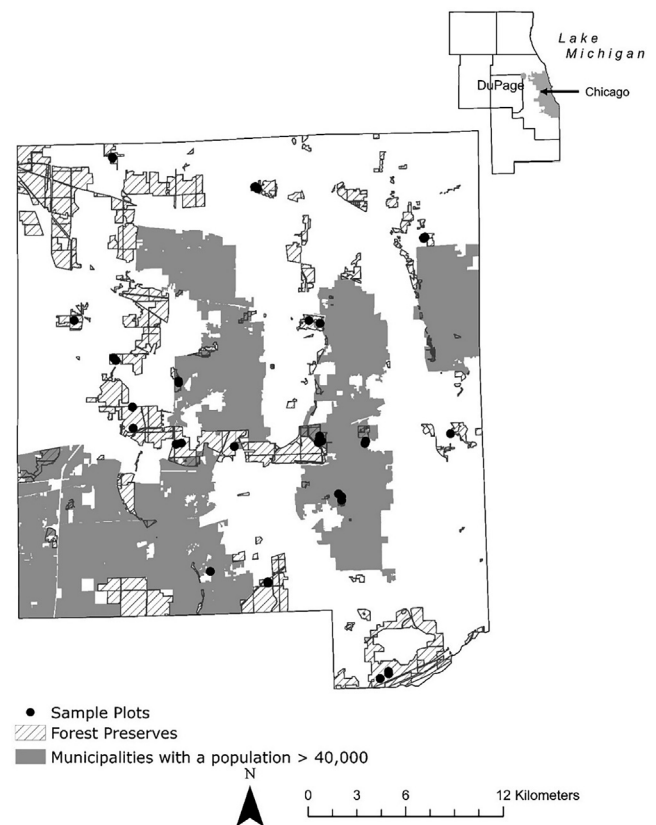


Fig. 1. Distribution of forest preserves, populated municipalities (> 40,000 people), and our sample locations in DuPage County, Illinois, USA. Population data from <http://dupage.maps.arcgis.com>.

deep and moderately well drained or well drained (Kelsey 2000).

Prior to European habitation, about 85% of DuPage County was tallgrass prairie and the remaining 15% was oak-maple forest (Thompson, 1985). The vegetation of this region was historically controlled by an interaction between landscape fire breaks and fire processes related largely to Native American activity (McBride and Bowles, 2001). However, with the cessation of annual fires at the time of European settlement, these open communities began to fill in rapidly and develop closed-canopy forests (Fahey et al., 2012). The county manages many of these forests in their Forest Preserve District.

In total, 17 forest preserves around the county were selected for long-term monitoring. Thirty-five sample plots were placed inside those preserves to represent typical woodlands in the county. The plots were mostly comprised of white oak (*Quercus alba*), red oak (*Q. rubra*), bur oak (*Q. macrocarpa*), wild black cherry (*Prunus serotina*), white ash (*Fraxinus americana*), American elm (*Ulmus americana*), sugar maple (*Acer sacharum*), and basswood (*Tilia americana*). Each plot is 47,000 square feet (approximately 0.45 Ha).

No management occurred in the plots before the monitoring period began. In subsequent years, vegetation management occurred in 26 of the 35 plots. Vegetation management usually comprised prescribed fire and/or understory clearing, but sometimes included herbicide as well. Fire was usually applied between November until early December, or between late March and mid-April before green-up. Fires were managed to advance slowly against the wind in a controlled fashion. Clearing focused on preserving desirable trees in the understory by removing or killing other plants that threatened their survival or development. This generally included removal of non-native trees and shrubs and thinning of abundant native tree species through cutting and herbicide application.

In addition to vegetation management at the plot level, deer management was applied at regular intervals (often annually) across 11 forest preserves that included 25 of the plots. Deer management was lethal removal with the goal of reducing the deer population to 6–8 individuals per square kilometer before fawning. Vegetation and deer management decisions are based on the forest preserve district's priorities, logistical constraints, and are intended to maximize the “return on investment” for the district.

2.2. Data collection

For the purpose of this study, we focus on change in woody vegetation (including mature trees, small trees, and shrubs) between two time periods: 1979 and 2014. The sample sites were monitored every 5 years (or more frequently) during this time period by scientists and staff of the Forest Preserve District of DuPage County. At each time interval, investigators measured all mature trees (diameter at breast height > 3.5 in. [9 cm]) in each plot. To survey small trees and shrubs, 5 (4 m * 4 m) quadrates were randomly placed in each plot. Presence/absence of all woody species (trees and shrubs) was recorded since 1979, and abundance of mature trees was recorded over the same time period, but abundance of small trees and shrubs was only measured starting in 2002.

Each woody species was classified according to several aspects of its biology. Plant species native to the Chicago Region were identified according to Swink and Wilhelm (1994). Native species were classified as specialists or ruderals based on their coefficient of conservatism. This coefficient, ranging from 0 to 10, indicates a species' tolerance to disturbance and fidelity to habitat integrity (Taft et al., 1997). Native species with coefficients < 4 are considered ruderal species that can establish in disturbed habitats, while those with coefficients of 4 or greater specialize on ‘unaltered landscapes’ and are found in more intact native habitats. Non-native species were not assigned a numerical value. Species were also classified according to their shade tolerance. Shade tolerance/intolerance was determined using Niinemets and Valladares (2006) classification and some online resources including

the Lady Bird Johnson Wildflower center (wildflower.org), the University of Minnesota extension office (extension.umn.edu) and the Missouri Botanical Garden (missouribotanicalgarden.org), as there was no single resource that included all species in the study area. All native and non-native species that were described to tolerate shade, partial shade, or ‘full sun to full shade’ were classified as shade tolerant. Species that could tolerate full sun only were classified as shade intolerant. All species, and their classifications, are listed in Appendix A.

We gathered a set of variables that described the environment at each plot, most of which were only measured once during the 35-year period. At each random quadrat, the amount of light was determined using a standard photographer's light meter. Readings were taken in 2014 on cloudless summer days between 10 a.m. and 2 p.m. and expressed as a percentage of the available light. Slope and aspect of the plot were measured using Illinois Statewide 30-Meter Digital Elevation Model (<http://www.isgs.illinois.edu>). Clay and organic matter in the soil were calculated using Soil Survey Geographic database (SSURGO). Initial (1979) native woody species richness was measured to test if more diverse native communities are less invasible (i.e., biotic resistance potential). Pre-settlement vegetation was defined as ‘forest’ or ‘non-forest’ using the map created by Bowles et al. (1998). Vegetation management was described in terms of number of unique events (e.g., number of fires) but also in binary terms (managed or not managed). Deer management was described as the number of deer control events in a plot throughout 35 years. Finally, we measured a number of larger-scale variables around each plot using ArcGIS 9.3 (ESRI, 2016). These include percent tree canopy cover in a 500 m radius, distance from the plot to the edge of the forest, distance to the closest paved road or trail, distance to the closest residential area, distance to a river, and distance to agriculture. All variables and their sources are listed in Appendix B.

2.3. Data analysis

2.3.1. Community-level analyses

To examine change in the entire forest community between 1979 and 2014, we used permutational multivariate analysis of variance (i.e., PERMANOVA) and looked for an effect of year, plot, and vegetation management on community composition. Because abundance of small trees and shrubs was not measured until 2002, we used presence/absence data to describe the community in the two years, and Sorensen distance to measure pairwise compositional dissimilarity between sites and years. The test was conducted in the “vegan” package in R (Oksanen et al., 2017) with 999 permutations.

Changes in community composition were visualized by performing non-metric multidimensional scaling (NMDS) on the Sorensen distances. This can illustrate shifts in the communities, direction of compositional change, and changes in dispersion. The axes of an NMDS graph do not indicate specific ecological variables, but are created to best represent the similarities and differences between plots. We plotted the vegetation in both years for all plots combined, as well as managed and unmanaged plots separately. Successional vectors were added to visualize the pairwise trend between plots from 1979 to 2014 (Appendix C). NMDS was performed in PC-ORD (McCune and Mefford, 2011) in two dimensions and with a maximum of 250 random starts to achieve convergence.

To test for biotic homogenization in the study sites, we used a distance-based test for homogeneity of multivariate dispersions (PERMDISP; Anderson et al., 2008), comparing dispersion of communities in 1979 to dispersion of communities in 2014. A smaller dispersion in 2014 would indicate reduced beta diversity and biotic homogenization. Managed and unmanaged plots were examined separately. We applied the function “betadisper” in package “vegan” in R, which is a multivariate analogue of Levene's test for homogeneity of variances. Differences among sample locations in each time period were calculated with the Sorensen index. To test the null hypothesis that beta diversity does not differ between the two years for managed and unmanaged plots, we

calculated an F-statistic to compare the average distance of each sample to the centroid of the group. We calculated a P-value from 999 permutations of least-squares residuals.

2.3.2. Analyses of species groups

To examine changes in particular groups of woody species between 1979 and 2014 within managed and unmanaged plots, we tested six nested generalized linear mixed models (GLMMs) in the package “lme4” (Bates et al., 2015) in R. For this analysis, “management” refers to plot-level vegetation management such as prescribed fire and clearing. First, we examined changes in number of native and non-native species in each plot. Next, we looked more closely at the native species by examining changes in ruderal and specialist species (i.e., species with a coefficient of conservatism < 4 or ≥ 4 , respectively). Finally, we examined if the number of shade-tolerant and intolerant species changed within 35 years. We applied the Poisson distribution for native, specialist, generalist, shade-tolerant, and shade-intolerant species, and we applied the negative binomial distribution for non-native species. Because several sample plots were contained in the same forest preserve, we specified plot as a random effect nested in forest preserves, while year and management were fixed effects.

To further understand invasion by non-native species, we used hierarchical linear models in the form of nested generalized linear mixed models (GLMMs) to identify potential drivers of richness and abundance of non-native species in 2014 (the most recent year that data were collected). Potential drivers included the variables we measured for each plot (i.e., the local environment, management activities, and landscape-scale factors listed in Appendix B). Distance to river was excluded because most plots were > 100 m away from river. We used the negative binomial distribution to model non-native species abundance and the Poisson family to model non-native species richness. As before, plot was a random effect nested in forest preserve. Model selection was based on a stepwise “backward” method that started with the full model (i.e., included all possible predictor variables). We removed the least significant explanatory variable first and continued removing non-significant variables ($p > 0.05$) one at a time until the model contained nothing but significant variables.

2.3.3. Analyses of individual species

To explore trends in occurrence of individual woody species over time, we used nested GLMMs with binomial distribution for each species found in the plots. We used the matrix of presence and absence of each species in 35 plots over 35 years as the response variable, and set time as the explanatory variable. Time was in five-year intervals from 1979 to 2014. Plot was a random factor nested in forest preserve. We identified species that showed significant increases or decreases in their distribution over time. We also predicted the number of occurrences for each species in year 2029 (representing a 50-year time period) using each species' model slope, intercept and the function “plogis” in R.

3. Results

3.1. Community-level analyses

Over the 35-year sampling period, we found 57 tree species in the plots. Forty-three tree species were native and 14 were non-native. Over the same period, 43 species of shrubs were identified in the plots, including 26 native and 17 non-native species. According to the PERMANOVA, the forest community changed significantly between 1979 and 2014 ($F = 9.08$, $P = 0.001$). Plot, time and vegetation management explained 61%, 11% and 8% of variation in the woody species community, respectively (Table 1). The NMDS showed a slight shift of the community in the managed plots and a larger shift in the unmanaged plots (Fig. 2, Appendix C). The PERMDISP test for managed plots ($F = 1.821$, $P = 0.2$) and unmanaged plots ($F = 0.561$, $P = 0.461$) did not show any sign of homogenization in either set of communities.

Table 1

Results from the multivariate permutational analysis (PERMANOVA) of differences in woody plant communities based on plot ($n = 35$), year (1979 vs. 2014), and vegetation management (managed vs. unmanaged).

PERMANOVA	df	SS	MS	F	R ²	P
Plot	34	3.716	0.109	2.962	0.613	0.001
Year	1	0.679	0.679	18.403	0.112	0.001
Management	3	0.519	0.173	4.689	0.085	0.001
Residuals	31	1.143	0.036		0.188	
Total	69	6.058			1	

3.2. Analyses of species groups

The only group of woody species that showed a significant change in richness over time or with management were non-native species, which increased over time and decreased with management (Table 2). There was not a significant interaction between time and management in any of the models.

The selected model for abundance of non-native species showed that distance to agriculture, deer control, light, distance to road, and native richness were important in explaining non-native abundance (Table 3). Non-native species abundance was lower in plots that were farther from agriculture and roads and had more deer control, while it was higher in plots with more native species richness and light. The selected model for richness of non-native species showed that deer control and distance to agriculture both had negative effects on non-native richness (Table 3).

3.3. Analysis of individual species

The GLMMs for individual species showed that nine species experienced significant declines and nine species experienced significant expansions throughout 35 years (Table 4). All species that declined were native, while two thirds of the species that increased were non-native and the rest were native but ruderal.

4. Discussion

We examined plant community changes over a 35-year period in urbanizing woodlands. Our results indicate a clear shift in the woody species community but no evidence of biotic homogenization. Furthermore, we found a general trend towards more non-native species over time. Management of vegetation seemed to somewhat decrease the number of non-native species but did not increase the number of more desirable species. Our analyses provide quantitative information that will allow land managers to better predict how forest communities will respond to management and urbanization.

In general, we found a significant change in species composition over time. There was a clear increase in number of non-native species. Although we found no significant change in native, ruderal, or shade-intolerant species richness over the same period, there was a trend toward fewer specialist species and more shade-tolerant species over time (Fig. 3, Table 2) that merits attention and should be considered for appropriate management actions in the future. Some individual species showed a significant change in their distribution. For instance, honey locust (*Gleditsia triacanthos*), a ruderal, shade-intolerant, native species was rare in 1979 but has increased significantly since then. In contrast, the native black maple (*Acer saccharum* subsp. *nigrum*) and silver maple (*A. saccharinum*) have declined over that time. Furthermore, wahoo (*Euonymus atropurpureus*) and wild black currant (*Ribes americanum*), which are specialist shrubs with high conservation value in DuPage County, dramatically decreased over time. By 2029, we predict that some species such as wahoo and wild black currant will be lost from the community, while others such as Japanese crab (*Malus sieboldii*) will be established. This calls for action toward targeted non-native species

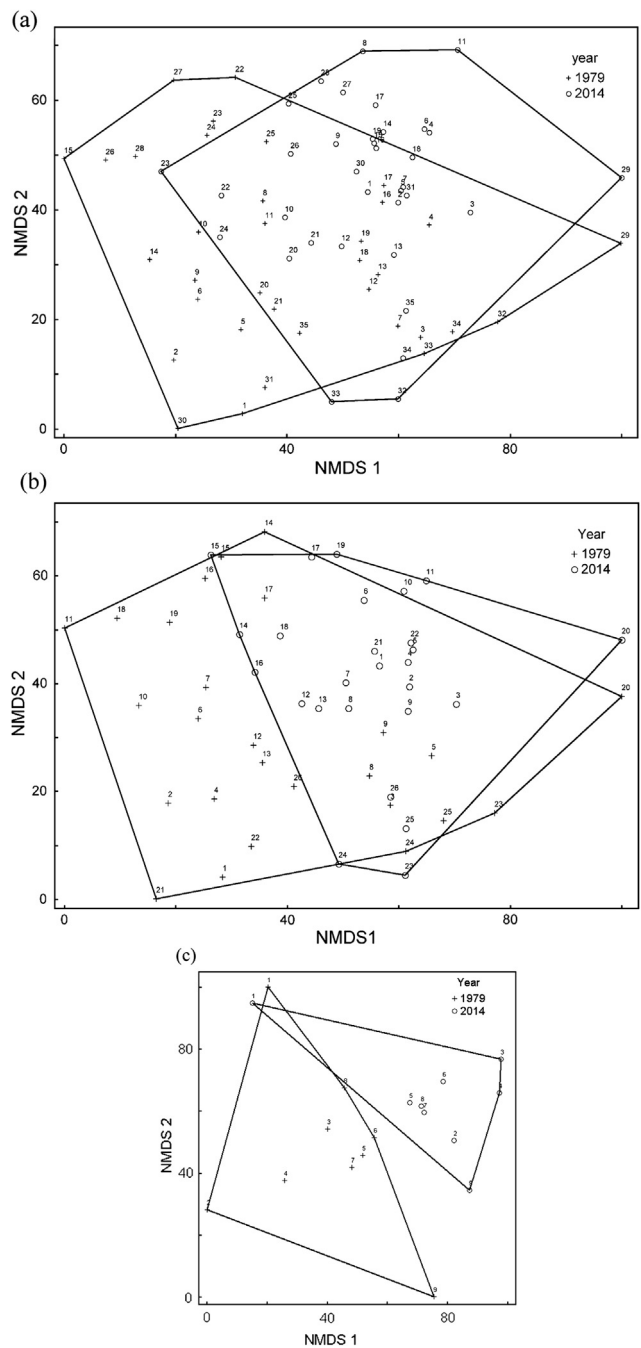


Fig. 2. Non-metric multidimensional scaling (NMDS) of woody species community composition in (a) all plots combined, (b) managed plots, and (c) unmanaged plots in year 1979 and 2014. Plots are numbered (with consecutive numbers from 1 to 35 for all plots, 1 to 26 for managed plots, and 1 to 9 for unmanaged plots) to facilitate comparisons between years. Successional vectors are shown in Appendix C.

along with protecting both highly specialized native species and also some native species with a medium coefficient of conservatism that will promote native diversity in forest habitats.

Interestingly, forest communities did not become more homogenized over 35 years. This holds true for both managed and non-managed plots. Our results indicate that an increase in non-native species does not necessarily lead to biotic homogenization. Moreover, heterogeneous communities may not indicate high native diversity or healthy ecosystems. One important factor that might have contributed to heterogeneity in the unmanaged plots is the invasion history of the

Table 2

Nested generalized linear mixed effects models for particular groups of woody species (native, non-native, specialist, ruderal, shade-tolerant, shade-intolerant species) and how they have changed over time (1979–2014) and with vegetation management. Sample plots are a random variable nested within forest preserves. Significant variables (at $p \leq 0.05$) are shown in bold text. Non-significant variables that show a trend ($p \leq 0.1$) are shown in *italics*.

Response variable	Explanatory variable	B	S.E.	Z-value	P value
Native species richness	Intercept	3.067	0.048	68.001	< 0.001
	Time	0.059	0.060	−0.975	0.330
	Management	0.018	0.099	−0.189	0.850
	Time*Management	0.132	0.121	−1.094	0.274
Non-native species richness	Intercept	0.649	0.171	3.790	< 0.001
	Time	1.049	0.209	5.005	< 0.001
	Management	0.894	0.290	3.079	0.002
	Time*Management	−0.436	0.352	−1.239	0.215
Specialist species richness	Intercept	2.568	0.063	40.685	< 0.001
	<i>Time</i>	−0.138	0.079	−1.745	0.081
	Management	0.145	0.129	−1.121	0.262
	Time*Management	−0.103	0.165	−0.623	0.533
Ruderal species richness	Intercept	2.112	0.068	30.976	< 0.001
	Time	0.058	0.095	0.618	0.536
	<i>Management</i>	−0.222	0.124	−1.795	0.072
	Time*Management	−0.196	0.179	−1.098	0.272
Shade-tolerant species richness	Intercept	3.099	0.047	65.861	< 0.001
	<i>Time</i>	0.099	0.057	1.729	0.083
	Management	−0.110	0.093	−1.183	0.236
	Time*Management	−0.112	0.109	−1.033	0.301
Shade-intolerant species richness	Intercept	0.479	0.181	2.646	0.008
	Time	−0.295	0.231	−1.274	0.202
	Management	0.725	0.431	−1.682	0.092
	Time*Management	0.518	0.525	0.987	0.323

Table 3

Nested generalized linear mixed models for non-native species abundance and richness for 35 forest plots in DuPage County, Illinois in 2014. Final model was identified using a backward stepwise approach; significant ($p \leq 0.05$) variables are shown in alphabetical order. Coefficients are standardized.

Response variable	Explanatory variable	B	S.E.	Z-value	P value
Non-native species abundance	Intercept	2.994	1.760	1.701	0.088
	Distance to Agriculture	−1.036	0.312	−3.312	< 0.001
	Deer control	−0.045	0.014	−3.118	0.002
	Light	0.036	0.013	2.704	0.007
	Native richness	0.072	0.036	1.947	0.050
	Distance to Road	−0.813	0.408	−1.992	0.046
Non-native species richness	Intercept	3.560	0.487	7.312	< 0.001
	Distance to Agriculture	−0.477	0.152	−3.130	0.002
	Deer control	−0.029	0.008	−3.414	< 0.001

studied area. Specifically, non-native species that were introduced more recently and have lower residence time might have not achieved their potential range; they therefore increase floristic differentiation as opposed to non-native species introduced earlier (Lososová et al., 2012). In our case, we inform that biotic impoverishment is taking place despite the heterogeneous community composition.

Distance to agriculture and deer control frequency were important variables related to both non-native species abundance and richness. Although some studies have shown that roads and residential areas are important sources of non-native plants (Gelbard and Belnap, 2003; Borgmann and Rodewald, 2005; Maheu-Giroux and de Blois, 2007), agriculture was more influential than other land uses in our study. Agriculture provides resources such as nitrogen, which favors fast-growing species and can lead to their invasion or increased dominance (Huenneke et al., 1990). Furthermore, the agricultural lands near our forest preserves are older than the nearest roads and residential areas,

Table 4

Woody species that have declined or expanded significantly in the number of plots in which they occurred between 1979 and 2014, along with a prediction of number of plots in which they will occur in the future (2029). Results are based on nested generalized linear mixed models, with sample plots nested within forest preserves. * indicates nonnative species.

Species	Explanatory variable	B	S.E.	Z-value	P value	Number of plots occupied		
						1979	2014	2029 (predicted)
<i>Declining species</i>								
Black maple	Intercept	− 1.522	0.689	− 2.207	0.027	10	3	0
(<i>Acer saccharum</i> ssp. <i>Nigrum</i>)	Time	− 0.065	0.019	− 3.329	< 0.001			
Dotted Hawthorn	Intercept	− 0.860	0.639	− 1.345	0.178	12	5	2
(<i>Crataegus punctate</i>)	Time	− 0.052	0.017	− 2.990	0.002			
Downy Arrow wood	Intercept	1.483	0.716	2.070	0.038	21	2	0
(<i>Viburnum rafinesquianum</i>)	Time	− 0.203	0.033	− 6.151	< 0.001			
Frosted Hawthorn	Intercept	− 1.907	0.627	− 3.039	0.002	4	0	0
(<i>Crataegus pruinosa</i>)	Time	− 0.145	0.048	− 3.020	0.002			
Iowa crab	Intercept	− 0.670	0.64	− 1.034	0.301	12	4	0
(<i>Malus ioensis</i>)	Time	− 0.091	0.020	− 4.400	< 0.001			
Hill oak	Intercept	− 8.372	2.056	− 4.071	< 0.001	12	3	0
(<i>Quercus ellipsoidalis</i>)	Time	− 0.362	0.113	− 3.192	0.001			
Silver Maple	Intercept	0.072	0.704	0.102	0.918	18	3	0
(<i>Acer saccharinum</i>)	Time	− 0.147	0.025	− 5.731	< 0.001			
Wahoo	Intercept	− 3.081	1.441	− 2.138	0.032	7	3	0
(<i>Euonymus atropurpureus</i>)	Time	− 0.093	0.029	− 3.176	0.001			
Wild black currant	Intercept	− 1.006	0.596	− 1.689	0.091	13	6	3
(<i>Ribes americanum</i>)	Time	− 0.051	0.018	− 2.774	0.005			
<i>Expanding species</i>								
Amur honeysuckle*	Intercept	− 2.727	1.122	− 2.431	0.015	10	29	35
(<i>Lonicera maackii</i>)	Time	0.217	0.040	5.319	< 0.001			
Burning bush*	Intercept	− 3.557	0.860	− 4.136	< 0.001	1	11	16
(<i>Euonymus alatus</i>)	Time	0.068	0.021	3.265	0.001			
Common blackberry	Intercept	− 0.498	0.358	− 1.391	0.164	12	24	33
(<i>Rubus allegheniensis</i>)	Time	0.072	0.015	4.709	< 0.001			
Common buckthorn*	Intercept	0.733	0.361	2.025	0.042	16	29	34
(<i>Rhamnus cathartica</i>)	Time	0.064	0.017	3.729	< 0.001			
Hackberry	Intercept	− 2.903	0.907	− 3.199	0.001	10	26	34
(<i>Celtis occidentalis</i>)	Time	0.127	0.025	5.090	< 0.001			
Honey locust	Intercept	− 4.754	0.860	− 5.523	< 0.001	1	9	18
(<i>Gleditsia triacanthos</i>)	Time	0.096	0.026	3.581	< 0.001			
Japanese crab*	Intercept	− 10.079	2.558	− 3.939	< 0.001	0	12	35
(<i>Malus sieboldii</i>)	Time	0.231	0.057	4.023	< 0.001			
Multiflora rose*	Intercept	0.129	0.400	0.322	0.747	15	29	34
(<i>Rosa multiflora</i>)	Time	0.079	0.017	4.599	< 0.001			
Smooth arrow wood*	Intercept	− 2.282	0.576	− 3.956	< 0.001	0	14	30
(<i>Viburnum recognitum</i>)	Time	0.081	0.017	4.755	< 0.001			

and therefore may have had more time to contribute to the spread of non-native species. Jodoin et al. (2008) found that road age is an important factor in common reed (*Phragmites australis*) invasion in Canada.

The positive effect of deer control in reducing non-native abundance and richness is in accordance with other studies that show non-native plant cover decreased in deer-proof plots (Nuzzo et al., 2017). Deer control can prevent dominance of the unpalatable, browse-tolerant species (Goetsch et al., 2011) and non-native species that deer tend to avoid (Tesoriere et al., 2007) and can enhance native species vitality (Nasiri et al., 2018). In 25 of our research plots, deer control occurred more regularly and consistently compared to fire and clearing, which varied greatly from year to year. This may account for the greater contribution of deer management in controlling the non-native species. However, deer control may not be as effective in maintaining native species if the area has already been intensely browsed (Nuttall et al., 2014). Furthermore, lethal deer control can be limited by logistical constraints such as proximity to homes or other areas that are heavily used by humans.

5. Conclusion

Our study has implications for management and conservation. A positive finding is that biotic homogenization has not occurred in these urbanizing forest patches. In addition, the number of native species has

not decreased. The vegetation management in our plots was able to slow down the spread of non-native species but did not prevent their increase over time. As shown in the NMDS results, managed plots appear to maintain the original community composition better than unmanaged plots. Vegetation management did not have a significant effect on shade-intolerant or ruderal species, but our data suggest a trend that might become significant with more time or more regular management. Therefore, we can conclude that fire and clearing are promising tools for preserving the woody plant communities in this region.

We suggest that if vegetation management was more consistent (repeated at regular intervals) it might have been even more effective in controlling the non-native species. It takes consecutive, repeated fires to stop seed production by killing existing individuals and eliminating plants that arise from the seed bank or from vegetative structures, which often are stimulated by the initial fire (Dey and Kabrick, 2015). Wilhelm and Rericha (2007) show that implementation of fire and managed thinning on a regular basis resulted in a marked increase in native plant densities and diversity in Iowa (USA). Propagule re-introduction of native species may also help control the non-native species, as many native species may naturally be seed limited (Hartman and McCarthy, 2004). This method fills in the niches available after successful removal of non-native plants and prevents colonization and expansion of undesirable species (Kettenring and Adams, 2011). Nonetheless, non-native species control would not be fully effective in such a developed landscape unless it is extended to the neighboring

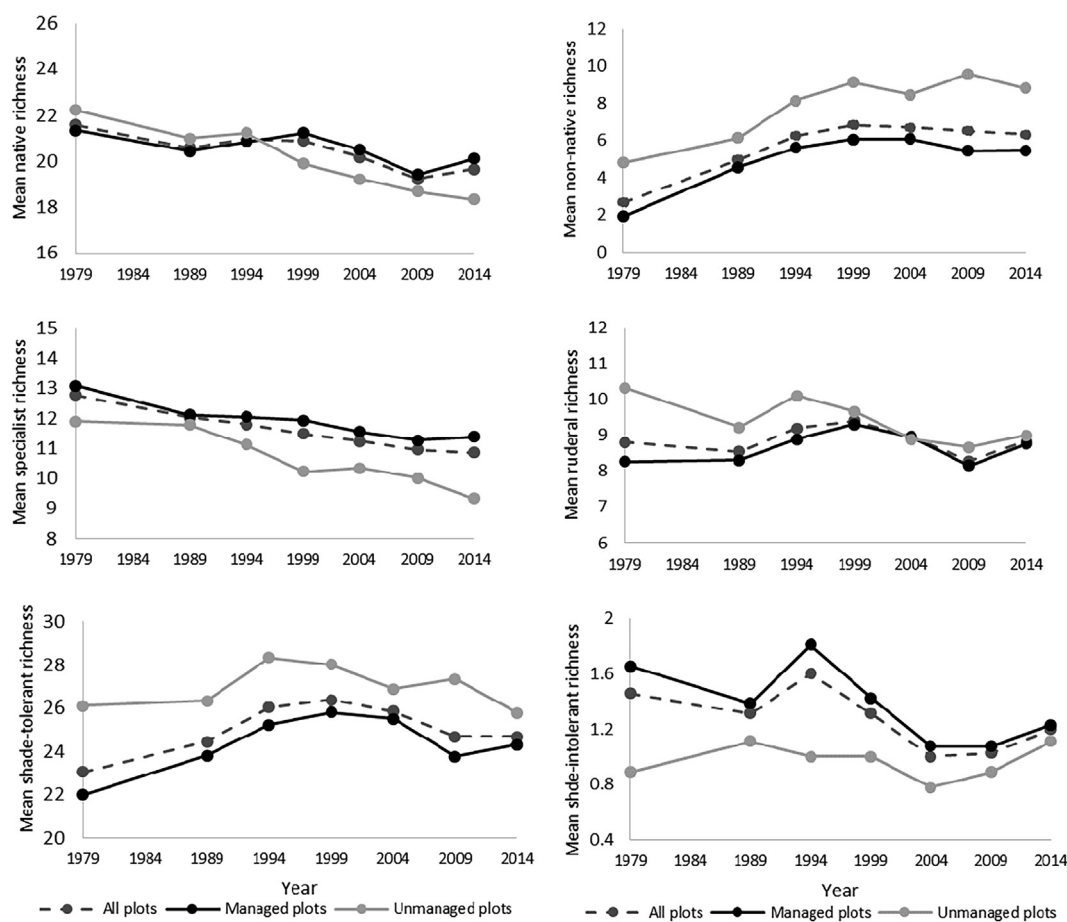


Fig. 3. Temporal trends in mean richness per sample plot of particular groups of woody species: native, non-native, specialist, ruderal, shade-tolerant, shade-intolerant species.

environments (e.g., residential areas, city parks, street trees), so a collaborative public-private approach will be necessary (Webster et al., 2006).

Maintaining the diversity of native communities and controlling the spread of undesirable species have been recognized as priorities at the global level (Andersen et al., 2004). Our study provides insight into these important issues. Many different agents come into play for maintaining the native community; reaching this goal requires collaboration between forestry, research institutes and residents. By conducting more long-term studies and, ideally, more controlled and consistent management experiments, we will better understand the temporal dynamics of urban forests.

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Appendix. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.foreco.2019.06.028>.

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