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Changes in Plant Species Composition and Structure in Two Peri-urban Nature Preserves over 10 Years

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Abstract.—Peri-urban natural areas, at the boundaries of cities and adjacent agricultural/rural land, are subject to ecological threats endemic to both land use types. We used permanent plots to document changes in habitat quality by monitoring herbaceous-layer plant species presence and cover over a decade (1996/97 and 2007) in two peri-urban nature preserves in central Indiana, U.S.A. The preserves are comprised of different forest community types: wet-mesic depressional forest and mesic upland forest. Habitat characteristics, based on Floristic Quality Assessment parameters, showed only a single change for either preserve between survey years: wetness values were lower in the wet-mesic depressional site in 2007 than in 1996, indicating more plants with affinity for wet soil. No changes in community structure (total species richness, evenness, and diversity) were found. The number of nonnative species increased between survey years, especially in the wet-mesic depressional forest, where numbers went from zero to six, five of which are classified as invasive. There was considerable turnover in individual species presence, with perennial forb species the most likely species to be found in only 1 y or the other. Species did not rearrange themselves within plots, but completely appeared or disappeared from all plots within a preserve between the sample years, suggesting that species composition of the flora is dynamic. Management recommendations, including those related to evidence of heavy deer browse, are presented based on our findings. Repeat monitoring of our plots in future decades will allow quantification of any extinction debt that may now be in place due to the increased presence of nonnative species, especially invasive shrubs escaped from landscaping.

Introduction

The conversion of global land cover to urban use has increased by 58,000 km^2 from 1970–2000 with the largest change occurring in North America (Seto *et al.*, 2011). This change in land use is generally attributed to 'urban sprawl' caused by rapid population growth and/or the changing patterns of commercial life in urban centers (Mieszkowski and Mills, 1993; Brueckner, 2000). In comparison to long established urban areas, peri-urban habitats, defined as areas surrounding formal urban boundaries, are undergoing rapid land use changes often resulting in habitat loss (McKinney, 2008) and decreases in biodiversity (McKinney, 2008; Stehlik *et al.*, 2007). This is unfortunate as these peripheral habitats can act as dispersal corridors between rural and urban environments (Snep *et al.,* 2006).

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Additionally, peri-urban areas may act as future designated green spaces used for human interactions with the natural environment (Brueckner, 2000).

Recent studies spanning rural to urban habitats, using transects or other spatial gradients, have documented variation in species richness across a variety of taxonomic groups at these rural-urban boundaries, with peaks in regional plant species in peri-urban areas (McKinney, 2008). Moderate disturbance intensity, along with the small-scale spatial heterogeneity present in a landscape matrix of agriculture, residential developments of different ages and stages, and remnant natural areas are the most likely reason for this peak in species richness (Rebele, 1994; Zerbe *et al.*, 2004; Wania *et al.*, 2006).

While plant communities within peri-urban habitats may benefit from habitat heterogeneity and moderate disturbance, they also must cope with the effects of habitat fragmentation, edge effects, and invasive species. These effects can be exacerbated in and near cities where close contact with humans leads to greater opportunity for escape of invasive ornamental species (La Sorte *et al.*, 2007; Winter *et al.*, 2009). For example nonnative plants are often intentionally and unintentionally introduced into peri-urban areas from landscaping, food waste, and other human activities (Godefriod and Koedam, 2003; McKinney, 2008; Walker *et al.*, 2009). These nonnative introductions may contribute to overall species richness but often do not provide the ecological services of native species and, in the case of invasive nonnatives, actively contribute to habitat degradation (Niinemets and Penuelas, 2008; Walker *et al.*, 2009).

Given these characteristics of peri-urban areas, they are locations likely to exhibit contemporary changes in species composition that may affect the quality of natural areas. One way to monitor ecological integrity in peri-urban natural areas is to track structural and compositional changes in vegetation using permanent plots. As changes are detected, appropriate management action can be taken to mitigate damage (Godefriod and Koedam, 2003). We monitored herb-layer vegetation in two peri-urban nature preserves with different forest community types in Indianapolis, Indiana, U.S.A. using permanent plots sampled in 1996 or 1997 and again in 2007 to assess how plant composition has changed. We looked for changes in habitat quality over the decade by quantifying abundance of native, nonnative and invasive nonnative species. Our resampling allowed for quantification of vegetation, changes based on direct comparison of data sets. Peri-urban land use changes are predicted to continue as urban areas expand. Understanding habitat change that correlates with rapid anthropogenic habitat modification is vital in order to adequately preserve natural areas.

METHODS

STUDY SITES

Indianapolis, Indiana, located in the Central Till Plain Natural Region (Homoya *et al.*, 1985), is the 12th largest city in the U.S.A. and home to approximately one million people. The surrounding peri-urban area continues to expand as an additional 231,000 residents have been added within the last 10 y (2000–2010), which amounts to 57% of the state's total growth (http://www.stats.indiana.edu/). Additionally, land use change tends to be most severe in peri-urban Indianapolis areas (Fig. 1; for larger scale land use changes around Indianapolis see Fry *et al.*, 2009). Historical records from the 1820s document the area was 98% forested in presettlement times, with beech-maple association dominating 76% of the landscape (Barr *et al.*, 2002). Our study was conducted at two state dedicated nature preserves within a 1600 ha park owned and managed by the city. State dedication provides the highest level of protection from development and from activities disruptive to the vegetation including mountain biking or horseback riding, along with a level of

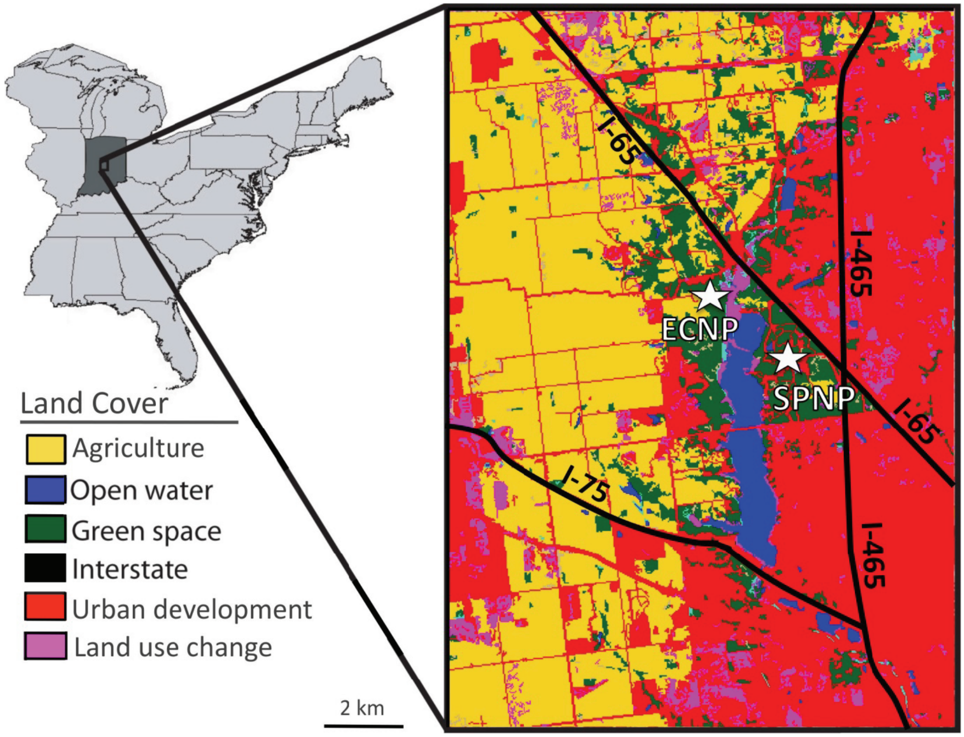

FIG. 1.—Map of land use change and land cover types in the northwest corner of Indianapolis, Marion County, Indiana USA. Study areas Spring Pond Nature Preserve (SPNP) and Eagle's Crest Nature Preserve (ECNP) are indicated by stars. Note that pink indicates change in land use between 1992–2001 all other land cover types remained unchanged during that time period. The 1992/2001 Retrofit Land Cover Change data is from the USGS National Land Cover Database (Fry *et al.* 2009)

environmental management and oversight of trails above and beyond that provided by the city.

Spring Pond Nature Preserve (GPS Lat. N 39 51.723, Long. W 86 16.973) is a 17.8 ha remnant of wet-mesic depressional forest located on deep, nearly level, very poorly drained soils on broad, gently undulating till plains. Scattered wet-mesic depressional forests were once common in Indianapolis, but few large wooded tracts remain due to decades of draining and ditching. Those that do remain are often very wet. Aerial photographs document almost all of Spring Pond has been continually wooded since at least 1941, making it one of the few remaining older-growth tracks of wet-mesic forest in the area. The wet-mesic forest at Spring Pond is primarily mature second growth dominated by *Fagus grandifolia* (beech), *Acer saccharum* (sugar maple), *Platanus occidentalis* (sycamore), *Quercus macrocarpa* (burr oak) and *Fraxinus* spp. (ash) (I-DNR DNP, 1988). Due to its representation of presettlement forests, the Indiana Department of Natural Resources Division of Nature Preserves dedicated it as a Nature Preserve in 1987.

Our second study site is the 18.6 ha Eagle's Crest Nature Preserve (GPS Lat. N 39 53.027, Long. W 86 18.755). Terrain here is flat ridge tops and steep slopes with older second-growth mesic upland forest dominated by *Quercus rubra* (red oak), *Acer saccharum* (sugar maple), *Fagus grandifolia* (beech), *Quercus alba* (white oak), *Carya ovata* (shagbark hickory), and *Fraxinus* spp. (ash) (I-DNR DNP, 1988). The fairly narrow, flat areas between drainages at Eagle's Crest are unsuitable for agriculture and are probably the reason why the mature forest remains today. This site had been privately owned until it became a state-dedicated nature preserve in 1987. A single hiking trail bisects the preserve from northwest to southeast.

SAMPLE METHODS AND DATA ANALYSIS

We sampled vegetation at Spring Pond in August, 1996, and at Eagle's Crest in June, 1997. We resampled each site during the same months in 2007 to minimize the chances of seasonal phenology differences affecting our ability to detect species' presence. Our basic sampling unit (site) was a 3 × 3 grid system of plot center points (nine per site), separated by 20 m, established along ordinal axis points using a compass and meter tape. We surveyed herb-layer vegetation (herbaceous plants and woody plants with stems less than 1 cm in diameter) in 10 m^2 circular plots around each center point (plot). Species presence was recorded, along with an estimate of the percentage aerial cover for each species. Cover classes were assigned as 1 = 1–7%, 2 = 8–25%, 3 = 26–50%, 4 = 51–75%, 5 = 76–93%, 6 = 94–100%.

We used haphazard sampling (McCune and Grace, 2002), meaning not systematically nor randomly located, to select five sites for study at Spring Pond and seven at Eagle's Crest. Within each site, plot centers were marked with recycled, solid high-density polyethylene stakes (0.08 m wide × 1 m long) placed in the ground to 10 cm above soil level with a post-hole digger. Accurate Global Positioning Software (GPS) was not available to us at the start of the study in 1990s but was used to map plot center points in 2007 to help with locating plots in the future.

To assess plant species diversity in our plots, richness, evenness, and Shannon-Weaver diversity were calculated using PC-ORD (McCune and Mefford, 1999). We calculated relative importance values (RIV) by averaging relativized frequency (relative frequency of one species as a percentage of total plant frequency) and cover (based on cover classes as defined above) for all species at each preserve. Coefficients of conservatism (C-values; Swink and Wilhelm 1994) were used to quantify species' fidelity to high quality habitats and tolerance of disturbance, as an indicator of overall floristic quality. C-values were chosen for analysis of our data because they provide a numerical value for species' habitat preferences that can be used to make statistical comparisons between sites and through time. C-values rank native species (those thought to have been present before European settlement) from 0–10 based on fidelity to high-quality habitats, with higher numbers indicating greater preference for high-quality habitat and less tolerance of disturbance. Because fidelity to high quality habitats can vary across species' ranges, we used C-values developed specifically for the Indiana flora by Rothrock (2004).

For each preserve and each sample year, we calculated Floristic Quality Index (FQI) values derived from C-values using Floristic Quality Assessment software (Wilhelm and Masters, 2004). FQI is calculated as $FQI = \Sigma(CC_i)/\sqrt{(N_{native})}$, where CC_i = the coefficient of conservatism of plant species i, and N_{native} = the total number of native species occurring in the community being evaluated.

We also used Floristic Quality Assessment software (Wilhelm and Masters, 2004) to assign each species a regional numeric 'wetness' value corresponding to U.S. Fish and Wildlife Wetland Category type (http://plants.usda.gov/wetinfo.html). Obligate wetland species score −5, obligate upland species 5.

We examined relationships between mean FQI, native C-value, mean C with nonnatives (nonnatives are assigned a C-value of zero to calculate this parameter), wetness, and other site parameters (richness, evenness, and diversity). In addition paired *t*-tests (all variables were normally distributed) were conducted in the R v2.6.2 statistical software environment to look for differences between years (R Development Core Team, 2008). Jaccard's matching coefficients were calculated to compare floristic similarity between sample years. Lastly, Floristic Quality Assessment software was also used to assign physiognomic class (*e.g.*, tree, herbaceous perennial, annual perennial) to our species to allow structural comparisons of the vegetation present in our plots during each sample year. Nomenclature follows Rothrock (2004), which is based largely on the Flora of North America (www.floranorthamerica.org). Indiana has not had a state-wide treatment of the entire flora in over 70 y.

RESULTS

For Spring Pond, a total of 79 species were recorded in the five sample sites during the two inventory years (Table 1). In 1996, 51 species were present, all native. In 2007, 57 species were present including six nonnatives (10.7%). Nonnatives were in the lowest cover classes of 1 or 2 and generally occurred in low frequency, with the exception of *Lonicera maackii*, which was present in five plots occurring in two of the five sample sites (Table 2).

A greater overall number of species were present in the seven sites sampled at Eagle's Crest, a total of 106 (Table 1). As at Spring Pond, few nonnatives were recorded: only four of 73 (5.5%) in 1997 and six of 92 (6.6%) in 2007. *Alliaria petiolata* was the most frequently encountered nonnative, found in seven plots scattered among three sites (Table 2). Its average cover class indicated it covered 25–40% of plots in which it occurred. *Lonicera tatarica* was found in only one plot, but covered 26–50% of the plot (Table 2).

No plants listed as rare, threatened, or endangered by the state (http://www.in.gov/dnr/naturepreserve/files/etrplants.pdf) were found during either of our surveys. In addition no plants with C-values of 9 or 10, indicative of species restricted to remnant landscapes that appear to have suffered very little post-settlement trauma (Rothrock and Homoya 2005), were detected. For Spring Pond in 1996, two species with C-value of 8 (*Anemone acutiloba* and *Fagus grandifolia*) and four with C-values of 7 (*Galium asprellum, Actaea pachyoda, Hydrangea arborescens*, and *Jeffersonia diphylla*) were identified. In 2007 the only species with a C-value of 8 was *F. grandifolia.* Two new species with C-values of 7 were present: *Solidago caesia* and *Carex laxiflora*, and the four recorded in 1996 were absent. C-values of 7 or 8 indicate species of high-quality remnant plant communities that appear to endure some disturbance. More species with C-values of 7 or 8 were seen in our surveys of Eagle's Crest than of Spring Pond. Those present at Spring Pond were also present at Eagle's Crest one or both years. Additionally, *Dryopteris marginalis* and *Epifagus virginiana*, both with C-value of 8, and seven other plants with C-values of 7 were present at Eagle's Crest (Table 1). These plant with C-value of 7 and 8 at Eagle's Crest were mostly present during both sample years (for C-value 8, three plants were present both years, none in only 1997 and two only in 2007; for C-value 7, six plants were present both sample years, four only in 1997 and five only in 2007) and do not show a clear trend of increasing or decreasing over time.

There were no significant differences in vegetation structure based on species richness, evenness, or diversity and no differences in floristic quality as indicated by C-value and FQI in plots between the two sample years for either nature preserve (Table 3). The addition of nonnative plants to the flora at Spring Pond, from none in 1996 to six in 2007 is reflected in lower native mean C-values with nonnatives and lower FQI with nonnatives; however values

TABLE 1.—Species present across the five sites at Spring Pond Nature Preserve and the seven sites at Eagle's Crest Nature Preserve in 1996/97 and 2007. Nonnative species are in capital letters. C-values for each species were developed specifically for Indiana flora (Rothrock, 2004). Species with C-values of 7 or greater are in bold

		Spring Pond		Eagle's Crest	
Species	C-value	1996	2007	1997	2007
Acer negundo	1		x		
Acer saccharum ssp. *saccharum*	4	x	x	x	x
Actaea pachypoda	**7**	**x**		**x**	**x**
Aesculus glabra	5	x	x	x	x
Ageratina altissima	2	x	x	x	x
Agrimonia parviflora	4		x	x	x
ALLIARIA PETIOLATA	-				x
Allium tricoccum	**7**			**x**	**x**
Anemone acutiloba	**8**	**x**		**x**	**x**
Antennaria neglecta	3			x	x
Arisaema dracontium	5				x
Arisaema triphyllum	4	x		x	x
Asarum canadense	5			x	x
Asimina triloba	6	x	x	x	x
Aspenium platyneuron	3				x
BERBERIS THUNBERGII	-		x		
Boehmeria cylindrica	3	x	x	x	x
Brachyelytrum erectum	6				x
Campanulastrum americanum	4	x			
Campsis radicans	1	x			
Cardamine concatenata	5			x	
Carex grayi	5		x		
Carex jamesii	4				x
Carex laxiflora	**7**		**x**	**x**	**x**
Carex tribuloides var. *tribuloides*	5		x		
Carpinus caroliniana ssp. *virginiana*	5		x		x
Celtis occidentalis	3	x		x	x
Cercis candensis	3			x	x
Cinna arundinacea	4		x	x	x
Circaea lutetiana ssp. *canadensis*	2	x	x		x
Claytonia virginica	2			x	
Cornus florida	4			x	
Cryptotaenia canadensis	3	x	x		x
Cystopteris protrusa	4	x		x	x
Dichanthelium clandestinum	3				x
Dicentra cucullaria	6	x		x	
Discorea quarternata	5				x
Dryopteris marginalis	**8**				**x**
ELAEAGNUS UMBELLATA	-		x		
Elymus hystrix	5				x
Enemion biternatum	5			x	x
Epifagus virginiana	**8**				**x**
Erigenia bulbosa	5			x	
Erigeron annuus	0		x		x
Erythronium americanum	5			x	

TABLE 1.—Continued

Species	C-value	Spring Pond 1996	Spring Pond 2007	Eagle's Crest 1997	Eagle's Crest 2007
EUONYMOUS ALATA	-				x
Eupatorium perfoliatum	4		x		
Eupatorium serotinum	0	x	x		
Fagus grandifolia	**8**	**x**	**x**	**x**	**x**
Festuca subverticillata	4				x
Fraxinus pennsylvanica var. *lanceolata*	1	x			
Fraxinus quadrangulata	**7**			**x**	**x**
Galium aparine	1			x	x
Galium asprellum	**7**	**x**			
Galium circaezans var. *hypomalacum*	5	x	x	x	
Galium concinnum	5	x			
Galium tinctorium	6	x		x	x
Galium triflorum	5	x	x		
Geranium maculatum	4			x	x
Geum canadense	1	x	x	x	x
Geum virginianum	5	x	x		
Gleditsia triacanthos	1		x		
Glyceria striata	4		x		
Hackelia virginiana	0		x		
Helianthus decapetalus	5			x	
Helianthus divaricatus	5			x	
Hydrangea arborescens	**7**	**x**			**x**
Hydrophyllum appendiculatum	6			x	x
Hydrophyllum canadense	**8**			**x**	**x**
Hydrophyllum macrophyllum	**7**			**x**	**x**
Hydrophyllum virginianum	4		x		
Hypericum punctatum	3			x	
Impatiens capensis	2	x		x	x
Jeffersonia diphylla	**7**	**x**		**x**	
Juncus tenuis	0		x		
Laportea canadensis	2	x		x	x
Leersia virginica	4		x		
Lindera benzoin	5	x	x	x	x
Liriodendron tulipifera	4			x	x
LONICERA JAPONICA	-		x	x	x
LONICERA MAACKII	-		x		
LONICERA TATARICA	-			x	x
Lycopus virginicus	5	x			
Maianthemum racemosum	3			x	x
Medeola virginica	**7**			**x**	
Menispermum canadense	3	x	x	x	x
Mimulus ringens	4		x		
Nyssa sylvatica	5	x			
Osmorhiza claytonii	3	x	x	x	x
Ostrya virginica	5				x
Oxalis stricta	0	x	x	x	x
Packera obovata	**7**			**x**	
Parietaria pensylvanica	1		x		
Parthenocissus quinquefolia	2	x	x	x	x

TABLE 1.—Continued

Species	C-value	Spring Pond 1996	Spring Pond 2007	Eagle's Crest 1997	Eagle's Crest 2007
Phacelia bipinnatifida	6			x	x
Phlox divaricata	5			x	x
Phryma leptostachya	4	x	x		
Phytolacca americana	0	x	x		
Pilea pumila	2	x	x		
POA PRATENSIS	-		x		
Podophyllum peltatum	3	x		x	x
Polemonium reptans	5			x	x
Polygonatum biflorum	4	x		x	x
Polymnia canadensis	3				x
Polystrichum acrostichoides	5				x
Prenanthes altissima	5			x	x
Prunella vulgaris ssp. *lanceolata*	1		x		
Prunus serotina	1			x	x
Quercus rubra	4	x			
Ranunculus hispidus var. *nitidus*	5		x		
Ribes cynosbati	4			x	x
ROSA MULTIFLORA	-		x	x	x
Rubus allegheniensis	2		x		
Rubus occidentalis	1	x	x		x
Sambucus nigra	2				x
Sanguinaria candensis	5			x	x
Sanicula marilandica	6	x		x	x
Sanicula odorata	2	x	x		
Scutellaria galericulata	4	x	x		
Scutellaria lateriflora	4	x	x		
Smilax hispida	3	x	x	x	x
Solidago altissima	0				x
Solidago caesia	**7**		**x**		**x**
Solidago flexicaulis	6			x	x
Staphylea trifolia	5			x	x
Stellaria pubera	**7**			**x**	
Stylophorum diphyllum	**7**				**x**
TARAXACUM OFFICINALE	-			x	x
Thalictrum thalictroides	**7**				**x**
Tilia americana	5				x
Tovara virginiana	3	x	x	x	x
Toxicodendron radicans ssp. *radicans*	1	x	x	x	x
Tradescantia subaspera	5			x	x
Trillium flexipes	5				x
Trillium recurvatum	4			x	x
Trillium sessile	4				x
Triosteum aurantiacum	5				x
Ulmus rubra	3				x
Uvularia grandiflorum	**7**			**x**	**x**
Valeriana pauciflora	**7**				**x**
Viola sagittata	6				x
Viola striata	4			x	x
Verbena urticifolia var. *urticifolia*	3		x		
Viburnum dentatum	6	x	x		

TABLE 2.—Frequency of occurrence and mean cover class of nonnative species at Spring Pond Nature Preserve and Eagle's Crest Nature Preserve. Nonnatives at Spring Pond Nature Preserve were sampled in 45 plots across 5 sites (see methods) in 1996 and 2007. Nonnatives at Eagle's Crest Nature Preserve were sampled in 63 plots across seven sites (see methods) in 1997 and 2007. Frequency of occurrence is total number of plots in which each plant was observed, followed by the number of sites in parentheses, to indicate degree of dispersion across the preserves. All are considered invasive in Indiana except *Poa pratensis* and *Taraxacum officinale*

	Spring Pond				Eagle's Crest			
	1996		2007		1997		2007	
Species	Freq.	Mean cover class	Freq.	Mean cover class	Freq.	Mean cover class	Freq.	Mean cover class
Alliaria petiolata	-	-	-	-	-	-	7(3)	2.4
Berberis thunbergii	-	-	1(1)	1.0	-	-	-	-
Elaeagnus umbellata	-	-	2(1)	1.5	-	-	-	-
Euonymous alata	-	-	-	-	-	-	2(1)	1.5
Lonicera japonica	-	-	2(2)	1.0	1(1)	2.0	1(1)	2.0
Lonicera maackii	-	-	5(2)	2.0	-	-	-	-
Lonicera tatarica	-	-	-	-	2(1)	1.5	1(1)	3.0
Poa pratensis	-	-	1(1)	1.0	-	-	-	-
Rosa multiflora	-	-	2(1)	2.0	1(1)	1.0	2(2)	1.0
Taraxacum officinale	-	-	-	-	5(2)	1.2	1(1)	1.0

are not lowered enough to be statistically significantly different. Wetness values were greater for plants surveyed at Spring Pond in 1996 than 2007, reflecting more plants indicative of wetland conditions were present in 2007.

Although community structural parameters did not change during the 10 or 11 y between our inventories, there was considerable species turnover. Counts reveal approximately 50 and 100 native plants were present at Spring Pond and Eagle's Crest for both inventory years, respectively. However, the taxa present were not the same taxa for both years. Jaccard's matching (binary similarity) coefficients were only 0.43 for the two sample years at Spring Pond and 0.56 at Eagle's Crest. Most of the species that turned over between sample years had low RIV in the beginning of the study; species with the highest RIV (those with

TABLE 3.—Paired *t*-tests results that compare vegetation structure and Floristic Quality Assessment parameter values among sites (arrays of 9 plots) inventoried at Spring Pond Nature Preserve in 1996 and resampled in 2007 and at Eagle's Crest Nature Preserve sampled in 1997 and again in 2007. Significance: * $P < 0.05$

	Spring Pond (N = 5)				Eagle's Crest (N=7)			
Mean values/plot	1996	2007	*t*	P	1997	2007	*t*	P
Richness	10.6	10.4	0.247	0.817	12.2	11.0	0.092	0.930
Evenness	0.95	0.93	0.151	0.443	0.96	0.97	-0.015	0.510
Diversity	2.21	2.12	0.076	0.472	2.34	2.26	0.354	0.368
Native mean C-value	3.36	3.40	-1.571	0.191	4.20	4.40	-0.582	0.582
Native mean C-value with nonnatives	3.36	3.20	0.741	0.500	4.10	4.30	0.326	0.378
FQI	17.2	16.8	-0.761	0.489	13.2	12.8	-0.476	0.651
FQI with nonnatives	17.2	16.3	-0.418	0.697	13.1	12.7	0.532	0.307
Wetness value	0.4	0.2	2.539	0.038*	2.2	2.0	1.247	0.259

values > 5) tended to be consistent between years (unpublished data). Examination of the physiognomy of species present during both sample years with those present only in 1996/97 or 2007 reveals perennial forbs were most labile during the ten or 11 y between vegetation sampling events. They comprised 59% of all plants seen both sample years at Spring Pond and 58% at Eagle's Crest and 45% or 57% of all plants present in only one or the other of the sample years at Spring Pond and Eagles' Crest, respectively.

Discussion

We did not find evidence of significant changes in habitat quality at either preserve as measured by fidelity to high-quality habitat (mean C-values and FQI, with or without nonnatives), with the exception of wetland values for Spring Pond. Although no plants with the highest C-value rankings of 9 or 10 were present in either year, this is not unusual for habitat remnants in the Central Till Plain Natural Region. Our recent city-wide study found only three extant taxa with C-values of 9 or 10 (Dolan *et al.*, 2011a). As noted by Rothrock and Homoya (2005), this region is home to few rare plants, likely due to its relatively recent glacial history and few specialized habitats, as well as historical disturbance. Plants with C-values of 7 or 8, those that are reflective of high quality remnant plant communities that appear to endure some disturbance, did not show a trend of decreasing in number at either preserve during the timeframe of our study. We also did not find evidence of changes in community structure as measured by richness, evenness, and diversity at either preserve.

Another change in the flora at Spring Pond and Eagle's Crest that might indicate decline in habitat quality would be an increase in nonnative species. An estimated 31% of plants growing outside of cultivation in Indiana are nonnative (K. Yatskievych, pers. comm.), while recent estimates put the percentage at 27% in Indianapolis (Dolan *et al.*, 2011b), providing an ample source of nonnative species propagules to the preserves. At the time we began our study in 1996, no nonnatives occurred in plots at Spring Pond, a sign that the nature preserve was a high quality site from this perspective. By 2007 six nonnative plants, comprising 10.7% of the flora, were present. All but one, *Poa pratensis*, a grass, are woody species considered highly invasive in Indiana (http://www.entm.purdue.edu/IISC/invasiveplants.php). Invasive species are generally defined as those that negatively impact habitats by reducing biodiversity and/or degrading habitat (Weber, 2003).

Nonnative species at Eagle's Crest remained fairly constant as a percent of the taxa present between 1997 and 2007 (5.5% vs. 6.6%). As with Spring Pond, most were invasive woody shrubs or vines that remained present in low frequency and fairly low cover class. The exception was the highly invasive herb *Alliaria petiolata*, the most broadly distributed (that is, found in the largest number of sites) nonnative plant found in either preserve. *Taraxacum officinale*, a nonnative species not classified as invasive in Indiana, declined in frequency between sample years.

Although much ecological damage is caused by invasive herbaceous plants, woody plants also pose an increasing threat in many areas (Richardson and Rejmánek, 2011). Invasive nonnative woody species pose threats to biodiversity and ecosystem function, especially in urban natural areas through effects ranging from increased populations of nonnative worms that alter decomposition rates (Poulette and Arthur, 2012) to reduced nesting success for birds (Nemec *et al.*, 2011).

Peri-urban areas tend to be susceptible to invasives and nonnatives as these areas are often influenced by human disturbance and are in close proximities to established nonnative species within urban centers (Godefriod and Koedam, 2003; McKinney, 2008; Walker *et al.*, 2009). Fortunately, Spring Pond receives little human disturbance at the present time.

Although there are several narrow unimproved trails within the preserve, there is no close-by parking and no visitors were seen during our surveys; therefore, trampling effects are minimal and there is little human-vectored seed introduction from outside the preserve. This 'isolation' effect has most likely contributed to the small degree of species composition change over the 10 y of this study and to the preserve's rank as one of the highest quality sites among 14 natural areas surveyed in Indianapolis over the last 20 y (Dolan *et al.*, 2011a), based on percent of nonnative species, mean native C-values, and total mean C-value. Sites in that study with low mean native C-values tended also to have a high percentage of nonnatives present and were those that had the highest degree of human disturbance. Drayton and Primack (1996) suggested a policy of preventing new trails from being developed and closing existing ones to help stop the loss of native species and introduction of nonnatives in an isolated conservation area in Boston. This is advisable for Spring Pond. Despite little human traffic, we initially noted all of the invasive shrubs found inside Spring Pond in 2007 to be present in additional plots sampled in woods adjacent to the preserve in 1996. These species produce seed in berries that are dispersed by birds, the most common dispersal mode for woody invasive species globally (Richardson and Rejmánek, 2011). In addition to limiting trails, scouting of surrounding woods to identify and remove invasive nonnative plants before they become established may help reduce sources of introduction into the preserve (Dickson, 1998; Godefriod and Koedam, 2003). Currently, IndyParks has high management intensity for natural preserves with the goal of eradicating invasives. The combination of management intensity and little human disturbance may also explain the relatively low rates of invasives in both parks.

An additional feature of peri-urban habitats that can influence loss of native species and lead to the introduction and spread of nonnatives is the presence of an over-population of *Odocoileus virginianus* Zimm. (white-tailed deer). These peri-urban habitats tend to have higher deer densities than surrounding areas. This increase is most likely due to habitat heterogeneity at urban fringes, where deer use forest habitats for shelter and use nearby agricultural or horticultural patches for food (Alverson *et al.,* 1988; Pickett *et al.,* 2001). Additional contributing factors to deer densities are the restriction on hunting in urban areas and a decrease in predator pressure (Pickett *et al.,* 2001). Herbivores such as deer, through their preference for palatable plants, may be responsible in part for the large turnover in species between survey years in our study. Deer are selective foragers, and their preferred food plants decline before overall reductions in the herbaceous community can be detected (Webster *et al.*, 2001). Unpalatable species can also decline due to impacts of deer via soil compaction and loss of leaf litter (Heckel *et al.,* 2010). Specifically, white-tailed deer are known to alter structure and composition of forest communities in Indiana (Webster and Parker, 1997). They have been referred to as keystone herbivores throughout the Eastern United States (Waller and Alverson, 1997) and can decrease native tree species regeneration and negatively impact rare plants and bird communities.

Between our survey years, we saw a reduction in importance value and frequency of three species used as indictors of deer damage in Indiana at Spring Pond, suggesting significant deer browse is occurring at this site. These species are palatable to deer and individual size is correlated with intensity of deer browse (Webster *et al.*, 2001): *Actea pachypoda* was present in one plot with a RIV of 1 in 1996 but was not found in 2007. *Arisaema triphyllum* also disappeared in 2007, after having been present in seven plots and having a RIV of 2.6 in 1996. *Osmorhiza claytonii* presence declined from seven to one plot over the 11 y and its RIV halved, from 1.6 to 0.8. White-tailed deer may act synergistically with invasive nonnative plants to degrade habitat. Deer eat seeds and facilitate dispersal of Asian bush honeysuckles

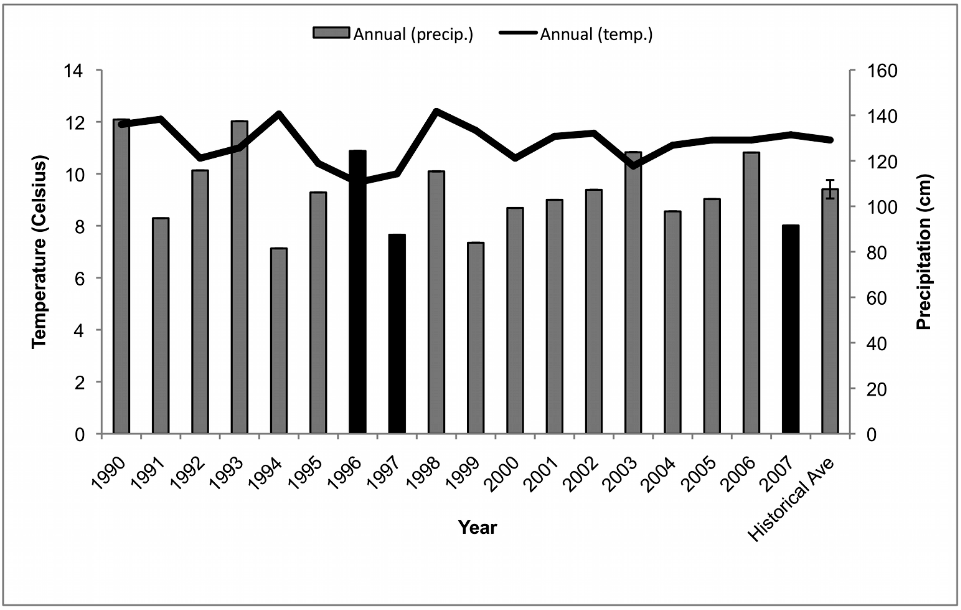

FIG. 2.—Annual precipitation and temperature data, 1990–2007, for weather stations located near the preserve study areas. Black bars represent the years of sampling for this study (1996/97 and 2007). Data are from the National Oceanographic and Atmospheric Administration National Climate Data Center (http://www.ncdc.noaa.gov/cdo-web/search)

like *Lonicera maackii* (Vellend, 2002) and have been documented to increase presence of *Alliaria petiolata* (Knight *et al.,* 2009). Through these direct and indirect effects, ubiquitous herbivores such as white-tailed deer can contribute to the homogenization of urban floras.

At Eagle's Crest, trends in deer indicator species between sample years were more ambiguous. *Osmorhiza claytonii* declined in frequency from ten to three plots and its RIV decreased from 1.0 to 0.5. However, *Actea pachypoda* increased in frequency from two to four plots and its RIV doubled from 0.3 to 0.6, and *Arisaema triphyllum* was found in 17 plots in 1997 and 20 in 2007 while its RIV increased from 2.3 to 3.1 between sample years. These results are in agreement with directed studies of deer browse at the two preserves from 2002 to 2007. The deer population exceeds capacity of the habitat at both sites, with browse at Spring Pond ranging from heavy to severe and browse at Eagle's Crest classified as heavy (G. Parker, pers. comm.).

Our findings of no significant changes in community structure yet high species turnover in plots sampled approximately a decade apart are similar to results found for plots in two areas of hardwood forest vegetation inventoried seven years in Quebec, Canada (Holland, 1978). Between 1969 and 1976, the mean number of species in quadrats remained nearly constant, but the species composition changed by an average of 17%. Changes did not alter qualitative characteristics of the flora such as life form, but did reflect ''the operation of a system of continuous rearrangement of species in the small quadrats of both sample areas'' (Holland, 1978). In our case species were not just rearranging themselves within plots but completely appearing or disappearing from all plots within a preserve between the sample years, suggesting species composition of the flora is surprisingly dynamic, even among perennial plants.

These changes in species composition may have been influenced by management practices in the preserves, including invasive species removal, but these efforts are not well documented or easily quantified. Deer browse, acting directly on palatable plants or indirectly through assisted spread of invasive species as discussed above, may be a contributing factor. Finally, climatological factors such as differences in precipitation and temperature between sample years, along with trends in these parameters in the years preceding sampling, may all have contributed to the large turnover in species. Previous studies examining precipitation influences on species composition have found a positive relationship with precipitation and species diversity (Pugnaire and Lázaro, 2000; Knapp *et al.*, 2002). This has been largely attributed to variability in soil moisture thresholds for germination among species (Pake and Venables, 1996). For our study sites, 1996 was the wettest year from 1990–2007, with an additional 20 cm of precipitation more than the historical annual average (Fig. 2). However, the signal of this wet year was not reflected in wetness values of plants present in our sample plots (Table 3), indicating the complexity of these interactions.

CONCLUSION

Peri-urban environments around Indianapolis are expected to expand into the future, extending into adjacent counties (http://www.stats.indiana.edu/). The expansion of urban and associated peri-urban land use is a global phenomenon, with wide-ranging implications for biodiversity. In 2007 the United Nations reported for the first time in history more than half of the world's human population lived in cities (UNFPA, 2007). Continued population increase is predicted to expand geographic coverage of urban areas to 1.2 million km^2 by 2030, triple the coverage in 2000 (Seto *et al.,* 2012). This increase will add to the unique challenges facing flora and fauna in and around cities. The expected increase in global urbanization is predicted to have especially negative consequences for biodiversity during the next few decades (Knapp *et al.,* 2012). This challenge to biodiversity and the ecological services ecosystems provide will likely be further exacerbated by global climate change (Seto *et al.,* 2012). Ecosystems with reduced diversity, in terms of both α diversity and phylogenetic diversity, will be challenged to respond to changing climatic regimes (Willis *et al.*, 2008).

Previous urban area longitudinal studies examining changes over longer periods of time have observed high losses of native species and increases in nonnatives. Our city-wide analysis of floristic changes in Indianapolis, Indiana, U.S.A. over the last 70 y found about 700 species before 1940 and in recent floristic surveys; however, they were not the same 700 species (Dolan *et al.,* 2011b). The recent total reflects a loss of native species of 2.4 per year, with a gain of 1.4 nonnative species each year. A study of floristic changes over 100 y in a woodland park in Boston, Massachusetts, U.S.A. found a loss of native species of 0.36% per year and a gain of nonnative of 0.18% per year (Drayton and Primack, 1996). Native species declined over 70% in Central Park in New York City, New York, U.S.A. over the last 100 years while nonnative species, mostly species widespread throughout the city, increased (DeCandido *et al.,* 2007). Over 20% of native vascular plants species present in a peri-urban area in Switzerland in 1915 were extirpated by 2003 (Stehlik *et al.,* 2007).

Our data do not support a change in vegetation quality in our peri-urban preserves, as measured by loss of rare species or changes in structural composition or Floristic Quality Assessment parameters between the 10 and 11 y between our inventories. However, we do document an increase in invasive nonnative species and surprisingly high turnover of individual species present. The invasive shrubs now present may show their effects on native flora in future years via extinction debt (Hanski and Ovaskainen, 2002). Native species may be on the pathway to extirpation due to ecological changes brought about by the introduction of nonnative shrubs; populations now present may not be viable in the future. Invasive species have the potential to rapidly alter and degrade habitat in ways that may lead to loss of native species (*e.g.*, Brown and Mitchell 2001; Stinson *et al.,* 2006) and increased opportunity for colonization by other nonnative plants (Gordon, 1999). Through these effects, invasive species are likely to promote habitat quality decline. Permanent plots used in this study can be reinventoried in coming decades to continue to quantify floristic changes through time and to determine if the spreading invasive species we document here have indeed created an extinction debt. Studies are also needed that directly address the role of varying levels of natural resource management and human use of sites in peri-urban areas to assess how these factors influence species changes in peri-urban habitats.

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