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Effects of vegetation removal on native understory recovery in an exotic-rich urban forest[1]

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VIDRA R. L. (University Writing Program, Duke University. Durham, NC 27708). T. H. SHEAR (Department of Forestry and Environmental Resources. North Carolina State University, Raleigh, NC 27695-8002), AND J. M. STUCKY (Department of Botany, North Carolina State University, Raleigh, NC 27695-7612). Effects of vegetation removal on native understory recovery in an exotic-rich urban forest. J. Torrey Bot. Soc. 134: 410419. 2007. ---Urban forests represent patches of biodiversity within otherwise degraded landscapes, yet these forests are threatened by invasion by exotic plant species. We investigated the response of a forest understory to removal of four common exotic species: *Elaeagnus umbellata* Thunb., *Lonicera japonica* Thunb., Ligustrum *sinense*, Laur., and *Microstegium vimineum* (Trio.) A. Camus in a forest within the city of Raleigh, NC, USA. In the summer of 2001, we initiated a removal experiment with three treatments. In the "repeated removal" treatment, all understory vegetation was initially removed by clipping and new exotic seedlings were repeatedly removed every 2 weeks throughout the study period. The "initial removal" treatment involved a one-time understory vegetation removal with no further weeding. Control plots had no intervention throughout the study period. We conducted vegetation surveys of the plots prior to treatment initiation and in April and August of 2002 and 2003. With a non-metric multidimensional scaling (NMS) ordination, we were able to discern differences in species composition between the repeated removal treatment and the other two treatments. However, using repeated measures ANOVA. we found no significant differences in native species richness, cover, and abundance among treatments during most sampling periods. We also used a seedbank study to determine that while some early successional species were present, no native shrubs and few native trees emerged from the seedbank. These results suggest that (It repeated removal is required to decrease the importance of exotic species, especially if the site is in close proximity to a source of exotic propagules: and (2) subsequent to exotic removal, native species may not recover sufficiently without supplemental plantings. Therefore, restoration plans for urban forests should incorporate both long-term monitoring and native plant re-introduction to achieve a diverse native community.

Key words: exotic species removal, forest restoration, seedbank study, urban forests.

Urbanization is transforming landscapes worldwide. The fragmented natural areas that remain within these urban matrices may represent pockets of biodiversity (Miller and Hobbs, 2002). Urban forests, in particular. can provide wildlife habitat, pollution filtration, and many other ecological services as well as recreational opportunities. Yet, the species compositions of these forests are often significantly altered by exotic plant invasions (e.g., Pyle 1995, Ehrenfcld 1997. Hutchinson and Vankat 1997, Kloor 1999).

Few studies have addressed the effects of exotic species on urban ecosystems (Byers et al. 2002, Miller and Hobbs 2002, Loeb 2006, Vidra et al. 2006). Exotic plants could displace native species and alter long-term successional trajectories (Fikes and Niering 199). Collier et al. 2002, Gorchov and Trisel 2003), although they could also enhance the biodiversity of these forests (Kloor 1999). Exotics could alter forest structure by providing more complex. continuous understory and shrub layers, affecting wildlife species both positively and negatively (e.g.. Schmidt and Whelan 1999.

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Mason et al. 2007). Because the effects of most invasive exotic species on species diversity and ecosystem function remain unknown, most management decisions to control exotics are supported only by anecdotal evidence (Byers et al. 2002). Even if exotic plants are adding to the overall species richness of a site, their presence could have long-term implications for forest health (Fikes and Niering 1999, Gorchov and Trisel 2003).

The impacts of exotic species invasion on native species diversity have primarily been studied by comparing invaded to uninvaded areas (see Levine et al. 2003 for review). Comparing the vegetation of a specific area pre- and post-invasion is one preferable method for determining whether the introduction of exotic species alters the native species composition. However, detailed historical records, other than species lists, are often unavailable for specific sites. Traditional weed addition/removal studies lend insight on potential competitive interactions but may be limited by the manipulations of investigators and unnatural laboratory conditions. Instead, in situ removal studies can be used to investigate the potential that exotics are suppressing native species (Luken et al. 1997, Meekins and McCarthy 1999, Gould and Gorchov 2000). These studies can then be used to generate specific hypotheses about competitive interactions between native and exotic species.

Even without specific evidence that exotic species threaten native species, removing exotics is often a primary management objective for restoration of urban forests. Removal of exotic species is used to maintain native species composition, enhance wildlife habitat, and improve the aesthetic appeal of forests.

Removing exotic species from urban natural areas requires tremendous time and effort (e.g.. Cramer 1993, Holloran 1996). Therefore, the potential benefits of removing exotic species should be evaluated to better inform management and restoration decisions (Westman 1990, Myers et al. 2000). We are interested in the response of urban forest communities to removal of exotic plant species. particularly in terms of native species recovery. We asked the following questions: (1) Does removal of exotic species significantly increase native species richness, cover, and abundance in urban forests? (2) Is a one-time vegetation removal sufficient for native species recovery? (3) Do short-term responses to exotic species removal indicate potential for recovery of native forest communities?

Materials and Methods. STUDY SITE. This study was conducted on the grounds of the North Carolina Museum of Art in Raleigh, NC. Within a heavily urbanized area, the 164 acre museum property includes museum buildings. forest, and pasture. We are currently assisting museum staff in developing restoration plans for an Art and Environment Park. One element of the restoration effort could involve removal of an extensive exotic plant understory. dominated by *Ligustrum sinense* (Lour.) (Chinese privet) and *Elaeagnus* (Thunb.) (silverberry), throughout approximately 18 ha (45 ac) of forested land.

This mixed-hardwood forest is representative of the remaining forest fragments in the Raleigh metropolitan area. Urbanization has transformed the landscape from mixed hardwood-pine forests to a matrix of development surrounding isolated forest patches. The remaining secondary successional forest patches suffer from recent anthropogenic disturbances. including heavy recreational use, runoff from surrounding developments, and invasion by a suite of exotic plant species. Several exotic shrubs, such as *Ligustrum sinense, Elueagnus umbellata*, and *Rosa multiflora* (Thunb. ex Murr) (multiflora rose) often dominate the understory of these forests (Merriam 2003). In addition. *Microstegium vimineum* (Trin.) A. Camus (stilt grass) is quickly spreading throughout floodplains and uplands of this region (Miller 2003).

The Art Museum forest is a typical urban forest patch. surrounded by major roads and industrial and residential development. This urban forest represents many of the typical challenges in urban forest restoration: isolation from less disturbed intact forests, close proximity to sources of exotic propagules: extensive invasion of the understory by a suite of exotic species. and continued disturbance from recreational use.

The canopy of this midslope forest is composed of native mixed hardwoods, including *Liriodendron tulipifera* L. (tulip poplar), *Diospyros virginiana* L. (persimmon). *Liquidambar stvraciflua* L. (sweetgum), *Quercus alba* L. (white oak), *Q. phellos* L. (willow oak), and *Q. rubra* L. (red oak). The understory is dominated by exotic shrubs, which create an almost continuous thicket throughout large parts of this forest. The understory does have seedlings of several native species, including *Asplenium platyneuron* (L.) Oakes. *Botrychium virginiamum* (L.) Sw., *Cornus florida* L., *Dio spyros virginiana* L., *Parthenocissus quinquefolia* (L.) Planch., *Prunus serotina* (Ehrh.). *Rubus sp., Toxicodendrom radicans* (L.) Kuntze, and *Vitis rotundifolia* (Michx.). However, there are no established native plants taller than I in. In fact, the majority of native seedlings in the understory are no older than I growing season and are hidden beneath the almost continuous exotic shrub layer (Table 1).

This forest also bears some evidence of recent human settlement, including established ornamental plantings and small waste dumps. The Cecil silt loam soils have been moderately to severely eroded throughout the study area.

STUDY DESIGN. Within this forested area, we delineated four sites of approximately 100 m' and located at least 30 in apart from one another. Each site featured a closed forest canopy, with no discernible canopy gaps, and was located at least 100 m from the forest edge and nearest stream corridor to minimize edge effects (e.g.. Cadenasso and Pickett 2001). Exotic understory density was visually consistent throughout these sites.

We established a replicated complete block experiment with three treatments: repeated removal, initial removal, and control. These treatments were applied to 2 × 2 m plots in three blocks per site, for a total of 36 plots. Plot corners were marked with PVC pipes and flagging tape for relocation. Individual plots were separated by at least I m and the sites were marked off with caution tape to reduce public interference with the experiment.

In July 2001, prior to experimental manipulations, we conducted vegetation surveys of each plot. We first divided each plot into four 1 x 1 m sections and then recorded the species. number of stems per species, and percentage cover of each species in the four sections. Nomenclature follows Radford et al. (1968). with exotic species defined as introduced to the United States according to the USDA PLANTS database. We then summed the number of sterns by species and averaged the percentage cover of species for each 2 x 2 m plot.

We initiated the three treatments on July 9, 2001. Control plots were left unaltered. The other plots were cleared of all vegetation, except native trees greater than 5 cm DBH (diameter at breast height), by clipping stems to ground level and removing this vegetation from the site. Given the small size of the plots. we did not pull up the roots of the standing vegetation to avoid significant soil disturbance (McClellan et al. 1995). We chose to remove both native and exotic plants for two reasons: (1) to eliminate initial among-plot differences in native species richness, abundance, and cover: and (2) to minimize above- and below-ground competition within plots. Prior to the treatments, the majority of standing biomass in these plots was of the two common exotic shrubs. *Ligustrum sinense* and *Elaeagnus umbellata* and there were no existing native plants greater than I m tall in any of the plots.

To investigate whether a one-time vegetation removal would be sufficient for native species to recover, we used an "initial removal" treatment that allowed for regeneration of all vegetation. The only plot maintenance performed was clipping of stump sprouts on some of the larger remnant stumps of *Ligustrum sinense* and *Elaeagnus umbellata* throughout the 2002 growing season. Previous studies indicated that clipping of the exotic shrub. *Lonicera maackii,* resulted in complete mortality of new stems on the shrub after 2 years (Luken 1988, Luken and Mattimiro 1991). In our study, resprouting of clipped shrubs was minimal by the second growing season.

The "repeated removal" treatment involved repeatedly clipping new exotic plant seedlings (*Elaeagnus umbellata, Ligustrum sinense, Rosa multiflora, Lonicera japonica* (Thumb.), and *Microstegium vimineum*) that emerged in the plots. These exotics were removed every 2 weeks during the growing season from the time the study was initiated (July 2001) until November 2003. We also cleared a I in border around all plots, including the controls, to minimize above- and below-ground competition from neighboring plants. We repeated the vegetation surveys of all plots in April and August of 2002 and 2003.

SEEDBANK STUDY. Because of the short duration of our study, we investigated potential future recovery of native plants from the seedbank of the site. We conducted a seedbank study to determine the presence of potential native plants whose germination may be currently suppressed by either the existing

Table 1. Pre-treatment presence and range of cover of understory species prior to treatments (July 2001) and during last sampling period (August 2003). Percent covers were converted to categorical cover classes for subsequent analyses ([1] indicates exotic species).

Species	Presence in plots July 2001	Range of cover July 2001	Presence in plots August 2003	Range of cover August 2003
Woody Plants				
Amelanchier arborea	1	<1		
Carya sp.			2	<1–1.75
Cornus florida	14	<1–2.5	9	<1–10.5
Diospyros virginiana	7		9	<1–6.75
Elaeagnus umbellata[1]	36	5.25–55	22	<1–50
Fagus grandifolia	1	1.25	1	<1
Ilex opaca	3	<1	3	<1–1.25
Ligustrum sinense[1]	36	<1–28.75	18	<1–63.75
Lindera benzoin			2	<1
Liquidambar styraciflua	12	3.75–21.25	7	<1–21.25
Liriodendron tulipifera			16	<1–3.5
Nandina domestica[1]			1	<1
Nyssa sylvatica	1	<1		
Pinus taeda			1	<1
Prunus serotina	29	<1–1	22	<1–13.5
Quercus alba	4	<1–2.5	4	<1–3
Quercus phellos	18	<1–1.75	8	<1
Quercus sp.	9	<1	3	<1
Rosa multiflora[1]	20	<1–8.75	9	1–8.75
Ulmus rubra	6	<1–6.25	6	<1–23.75
Viburnum dentatum	1	<1	1	<1
Herbs and Ferns				
Asplenium platyneuron	16	<1–10	17	<1–7.25
Aster sp.			8	<1–1.25
Botrychium virginianum	13	<1–1.5	18	<1–2.75
Duchesnea indica[1]	12	<1–8.75	14	<1–8.75
Ophioglossum vulgatum	1	<1		
Galium sp.			1	<1
Phytolacca americana			5	<1
Polygonum sp.			6	1–1.5
Ranunculus sp.	23	<1–4	24	<1–5
Tipularia discolor			1	<1
Viola sp.	13	<1–2	7	<1–1.5
Unknown 1			1	1.25
Unknown 2			1	<1
Grasses				
Liriope muscari[1]			1	<1
Microstegium vimineum[1]	17	<1–7.5	13	<1–25.5
Vines				
Campsis radicans	5	<1	6	<1–2.25
Gelsemium carolina	1	<1		
Hedera helix[1]	2	<1		
Ipomoea purpurea[1]			4	<1–1
Jasminum nudiflorum[1]	2	<1		
Lonicera japonica[1]	36	<1–30	21	<1–18
Parthenocissus quinquefolia	24	<1–2.75	27	<1–8.25
Rubus sp.	8	<1–3.5	11	<1–7.5
Smilax glauca			12	<1–3
Smilax rotundifolia	14	<1–1.25	11	<1–1.25
Toxicodendron radicans	19	<1–12.5	20	<1–11.25
Vitis rotundifolia	16	<1–2.25	36	<1–10

vegetation or by environmental conditions. In February 2003, we collected soil samples from each of the four sites. Since we intend to continue monitoring recovery, we collected our soil samples from the area immediately adjacent to each plot instead of directly from the plot. This area was kept clear of vegetation throughout the study period, so that conditions were similar within the plot and in this border zone. We collected four soil cores, each 2.5 cm in diameter and 5 cm deep, from each side of every plot, for a total of 16 cores per plot and

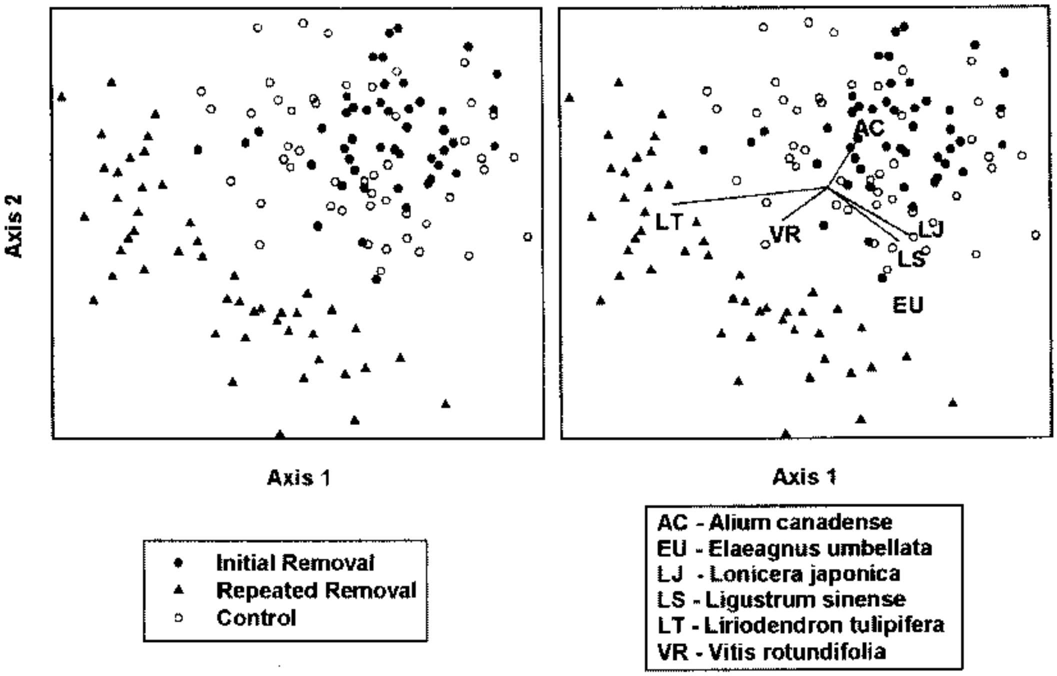

FIG. 1. Non-metric multidimensional scaling (NMS) two-axis solution for species composition of three experimental treatments and with species vectors overlaid.

144 cores per site. The soil substrate is quite rocky, which prohibited deeper soil samples.

We pooled the soil samples for each site and placed them in a refrigerator for 8 weeks to ensure cold stratification. In April 2003, we transferred the soil samples to an unheated greenhouse. One inch of soil was spread over a two inch substrate of Metromix, a commercially available potting soil. Six flats were used per site and an additional four flats of Metromix were interspersed between the study flats to measure any germination due to greenhouse contamination. An automatic watering system was used to ensure moist soil conditions. From April 2003 through April 2004, emerging seedlings were identified, counted, and removed from the flats.

DATA ANALSIS. We first used non-metric multidimensional scaling (NMS) with the Sorensen distance measure in PC-ORD (McCune and Mefford 1999) to detect differences in species composition among treatments and among time periods. NMS analysis was run using the "autopilot" option to determine the best number of axes and then manually rerun 20 times. The scores were plotted with varimax rotation. For this analysis. we used categorical cover classes (Peet et al. 1998) instead of raw percentage cover to minimize variability in visual percent cover estimates made over two years.

Next, we used mixed model repeated measures analysis of variance (ANOVA) (SAS/STAT software, Version 8.1 of the SAS System for Windows, copyright 2001. SAS Institute, Inc.) to test for the effect of treatment and time on the following response variables: total species richness. exotic species richness, native species richness, native species cover, native species abundance (or number of stems), native tree seedling richness, and native tree seedling abundance. The mixed model allowed site and block to be treated as random effects and included an interaction between treatment and time. Differences of least square means, significant at the $P < 0.05$ level, were compared to detect treatment and time effects.

Results. TREATMENT EFFECTS. We selected a two-axis solution for ordinating species composition of plots (Fig. 1), with the distance between plots representing the difference in species composition. The cluster of repeated removal plots is clearly separate from plots of the initial removal and control treatments.

Table 2. Mixed model repeated measures ANOVA results for effects of treatment, time, and the treatment by time interaction on individual response variables.

Response variable	Treatment	Time	Treatment*time
Total species richness	$F = 15.23$, $P < 0.0001$	$F = 34.14$, $P < 0.0001$	$F = 2.74$, $P = 0.02$
Exotic richness	$F = 157.69$, $P < 0.0001$	$F = 7.64$, $P = 0.0001$	$F = 4.41$, $P = 0.0005$
Native richness	$F = 0.53$, $P = 0.59$	$F = 27.38$, $P < 0.0001$	$F = 2.87$, $P = 0.01$
Native tree richness	$F = 3.36$, $P = 0.04$	$F = 3.12$, $P = 0.03$	$F = 1.57$, $P = 0.16$
Native tree abundance	$F = 2.73$, $P = 0.07$	$F = 7.79$, $P = 0.0001$	$F = 4.89$, $P = 0.0002$
Native cover	$F = 0.08$, $P = 0.93$	$F = 7.73$, $P = 0.0002$	$F = 1.10$, $P = 0.37$
Native species abundance	$F = 0.09$, $P = 0.42$	$F = 14.43$, $P < 0.0001$	$F = 6.66$, $P < 0.0001$

suggesting differences in species composition Table I).

The repeated measures ANOVA results confirmed differences among treatments and among time periods for most response variables (Table 2). Prior to initiation of the treatments, the plots were not significantly different, in tither exotic species richness or native species richness. Throughout time, the control and initial removal treatments had similar exotic species richness, while the repeated removal treatment had no exotic species (Fig. 2a). There were no significant differences in the number of native species (Fig. 2a) or native species cover (Fig. 2b) among treatments throughout this experiment. Native tree seedling richness was significantly higher in the control treatment until the August 2003 survey, when there were no significant differences detected among treatments (Fig. 2d). One native tree species in particular. *Liriodendron tulipifera,* achieved higher percentage cover in the repeated removal treatment (Fig. I).

In August 2003, native species abundance. or the number of native stems, was significantly higher (mean = 75.9. SD = 26.5) in the repeated removal treatment than either the initial removal (mean = 44.9, SD = 31.3) or control (mean = 35.2, SD = 27.7) treatments (Fig. 2c). While one species. *Allium canadense* L. was ubiquitous in all plots, the most common native species in the repeated removal treatment were *Vitus rotundifolia* and *Liriodendron tulipiferu* (Fig. 1). The suite of common exotic plants. *Elaeagnus umbellata* , *Ligustrum sinense* and *Lonicera , japonica,* dominated both the initial removal and control treatments (Fig. 1). Tree seedling abundance peaked in the repeated removal treatment in April 2003 but was not significantly different from the other treatments in August 2003 (Fig. 2e). INTERANNUAL AND SEASONAL VARIABILITY. The effect of time was significant in all repeated measures ANOVA models (Table 2). Mean native species richness, native species cover, and native species abundance all declined in all treatments during the August 2002 period (Figs. 2 a c). Most response variables recovered or increased during the April and August 2003 surveys (Figs. 2 a e).

SEEDBANK RESULTS. Several early successional species sprouted in the greenhouse flats from April to December 2003 (Table 3). Some tree seedlings emerged. reflecting the canopy composition: *Liriodendron tulipifera. Prunus serotina* and *Liquidambar styraciflua*. One native shrub seedling. *Myrica cerifera (L.)* Small, and several exotic plant seedlings *(Ligustrum sinense, Lonicera japonica.* and *Microstegium vinuneum)* sprouted as well. Fifteen of the twenty species found in the seed bank were also present in the study plot(s) during one or more sampling period.

Discussion. We conducted this study in a forest community whose composition is increasingly common in urban areas: native overstories with understory dominated by exotic shrubs (Vidra 2004). While there are seedlings of native shrubs and trees in the understory, it is rare to find any larger specimens throughout the forest (Table 1).

Preventing the establishment of exotic plants in the understory of an urban forest, as we did in our "repeated removal" treatment, does lead to a different species composition. Yet, we attribute this to an absence of exotic species. not to an increase in native species. While some new native species did emerge post-treatment (Table 1). we found no greater native species richness in either removal treatment compared to the control. The control plots maintained occurrences of scv-

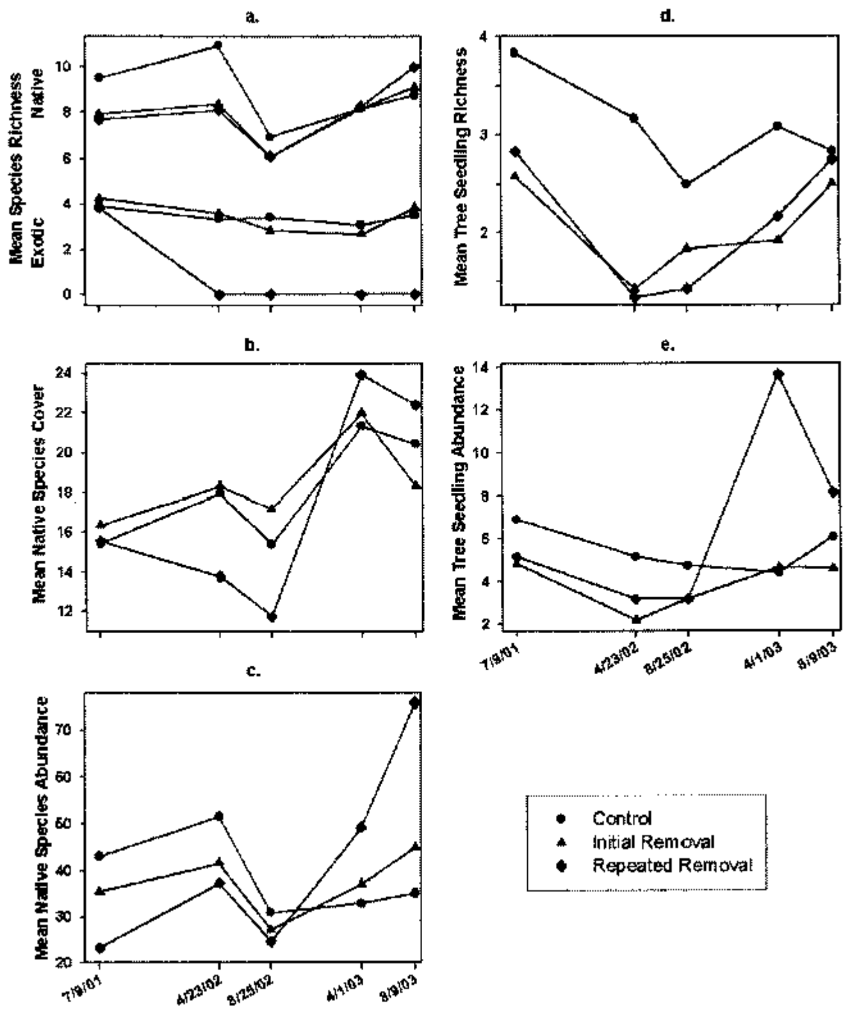

FIG. 2. Mean values for response variables for each sampling period.

eral native seedlings throughout the study period, including *Prunus serotina, Acer rubrum. Liriodendron tulipifera, Parthenocissus quinquefolia, and Toxicodendrom radicans, so* there may be longer-term differences in species composition as the native species in the treatment plots are allowed to flourish while those in the control plots will continue to compete with the thick exotic understory. In August 2003, the number of native stems per plot in the repeated removal treatment was significantly higher than those in either of the other two treatments, indicating potential for long-term increases in native plants once exotics are removed.

There are several possible reasons for the lack of native species recovery in the repeated removal treatment plots. First, environmental conditions may not be amenable to native germination. The seedbank is dominated by shade intolerant species, more typical of old fields than of mature forest understory. It is therefore reasonable that we did not see a significant flush of new growth from these species. Interannual variability was also significant during this study period, with a severe drought affecting the region on August 2002. During the 2003 growing season, both cover and abundance *of Liriodendron tulipifera* increased within the plots, suggesting that under normal precipitation conditions, recovery may be more significant. Yet, throughout the floodplains and mid-slopes of this study forest. *Microstegium vimineum* occupied large areas in 2003, further outcompeting understory seedlings (pers. obs.).

Table 3. Number of seedbank seedlings by species and site that emerged between April 2003 and January 2004.

Species*	Site 1	Site 2	Site 3	Site 4
Duchesnea indica[1,2]	8	7	5	14
Rubus sp.[2]	7	13	2	6
Lonicera japonica[1,2]	6	3		1
Liriodendron tulipifera[2]	5	5	6	6
Microstegium vimineum[1,2]	5	5		
Liquidambar styraciflua[2]	4	6	3	1
Cyperus odoratus	4	12		11
Eupatorium capillifolium	4	1		1
Viola sp.[2]	3	4	3	1
Phytolacca americana[2]	3	4	1	6
Vitis rotundifolia[2]	3			1
Huechera sp.	2	2	1	1
Ipomea sp.[2]	2	2	5	3
Gnaphalium obtusifolium	1	1	5	12
Sida rhombifolia	2	1	1	2
Ligustrum sinense[1,2]	1	1		
Pinus taeda[2]	1		4	3
Ambrosia artmesiifolia	1			2
Plantago sp.	1			
Carex crinita		6	5	12
Baccharis halimifolia		5	3	3
Scutellaria elliptica		2		2
Eragrostis hirsute		2		
Geranium carolinianum[2]		1	1	
Campsis radicans[2]		1		2
Myrica cerifera		1		
Prunus serotina[2]		1		
Lespedeza cuneata[1]		1		
Erigeron canadensis			1	
Dichanthelium sp.				1
Solanum americanum				1
Total species richness	20	24	16	21

* ([1] indicates exotic species, [2] indicates species present in plots during one or more sampling periods).

The second explanation for the lack of native species recovery is that exotic species continue to outcompete native species. Large numbers of *Elacagnus umbellata* and *Ligustrum sinense* seedlings continually emerged throughout the growing seasons in all plots that initially contained smaller exotic shrubs, under I in tall. This phenomenon is consistent with. Webb et al's (2001) finding of large increases of seedlings of the exotic *Acer platanoides L.* when relatively small seedlings of this species were removed. The new plants could be from germinating seeds, provided by the abundant exotic shrubs found in the rest of this forest patch, or from bird dispersal of exotic seeds from nearby sites (Vidra 2004). These abundant exotic seedlings may be competitively excluding the native seedlings from the plots. Further study of this competitive interaction, however, is needed.

Third, dispersal of viable native propagules into the study plots may be limited. An informal survey of the site found no native shrubs over I m tall (pers. obs.). Recruitment of native propagules from outside the site is also limited by the isolation of the NC Art Museum forest within a heavily urbanized matrix. A concurrent study suggests that landscape context in urban areas can significantly influence the extent of exotic species invasion, as adjacent neighborhoods provide a source of exotic propagules (Vidra 2004). The potential for native propagules to disperse to this site, via animals, wind, or water, should be examined more closely in a future study.

Finally, the lack of native woody shrubs in the seedbank may be the result of "ghost of competition past." While we have no historical records that document the time since invasion, the exotic understory may have persisted long enough to displace native species from the seedbank as well as the understory. To explore this explanation, future studies could compare the seedbank at this site with both uninvaded (or less invaded) sites and those sites that have only recently been invaded.

This study took place over two complete growing seasons. Other experiments have shown responses of annual plants to removal over the course of just one growing season (Meekins and McCarthy 1999, Gould and Gorchov 2000). Woody plants will likely take longer to recover and may require changes in environmental conditions as well, such as an increase in light availability. While the native species richness has not significantly increased in our treatment plots, we are encouraged by the increasing number of native stems in the repeated removal plots. While the initial results of this study are presented here, we plan to continue monitoring these plots for native species recovery over a longer time frame.

If exotic species removal is integrated into restoration plans, supplemental plantings of desirable species may be necessary to restore a diverse forest community. These plantings may also help to resist future invasion by occupying space and resources. thereby edging out aggressive exotic species. This is particularly important in urban areas, where native species recruitment may be minimized by extreme habitat fragmentation (Miller and Hobbs 2002, Duguay et al. 2007). The landscape context, or surrounding land use, plays an important role in invasion dynamics of forest fragments in urban areas (Vidra 2004). Therefore, attention must be paid to both sources of exotic seeds and native seeds. In this study, there were likely plenty of exotic seeds due to the abundance of exotic plants left in the surrounding forest. However, native shrubs were absent at this site and in the seedbank, suggesting that recruitment from outside the site or intentional planting of native shrubs may be necessary to create a diverse forest understory.

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