Effect of invasive species removal on the understory of an urban forest

By

Patrick Chess

Submitted to the Graduate Faculty of
North Carolina State University
in partial fulfillment of the
requirements for the Degree of
Master of Natural Resources

Ecological Restoration

Raleigh, North Carolina

2011

Approved by advisory committee:

Dr. Theodore Shear (Chair)
Dr. Joseph Neal
Dr. Christopher Moorman

August 9, 2011

Abstract

Chess, Patrick. Master of Natural Resources Ecological Restoration. Effect of invasive species removal on the understory of an urban forest

In 2002, a 5-acre section of forest at the North Carolina Museum of Art Park underwent an invasive species removal treatment to prevent ecological damage caused by these invasive species and to aesthetically improve the park. Treatment was targeted specifically at non-native woody shrubs, especially *Elaeagnus umbellata* and *Ligustrum sinense*. This forested area was used as a case study to examine the effects of invasive species removal on the understory of the forest. When compared to an adjacent nontreated area of forest, the treatment was effective at decreasing the cover of both *Elaeagnus umbellata* and *Ligustrum sinense*. The reduction in invasive shrub cover also increased park sightlines, accomplishing the aesthetic goal. Despite the success of the treatment in removing the target invasive species, eight years later the total cover of invasive species was not different between the two areas. This was largely due to a much higher average cover of *Microstegium vimineum* in the treated area. Possible explanations for this difference include competitive release and the introduction of disturbance. There was no observed difference between the understory communities of the nontreated and treated areas. Also, invasive shrub cover was correlated with both canopy cover and proximity to the forest edge. This study shows that successful invasive species removal can lead to further invasion by other species. Furthermore, removal of invasive species may not lead to an increase in native species cover or richness. Finally, this study supports the observation that forest edges and open canopies are often associated with invasion processes.

Acknowledgements

I would first like to thank the North Carolina State University Department of Forestry and Environmental Resources for accepting me into the graduate program and making me feel at home for the last two years. Additionally, I want to thank the North Carolina Museum of Art for allowing me to survey their land and perform this research project. I also want to thank the members of my committee for all of their support. Dr. Neal and Dr. Moorman came into my project late and helped me to succeed. Dr. Shear provided me with guidance for two years and the project never would have started without his assistance and time. I cannot thank the three of you enough. I would like to thank Dr. Hoffmann for loaning me his light meter on such short notice, and recognize Jenny Schafer for arranging time for me to pick it up. Special thanks go out to Joe Sullivan and Ryan Unks for helping me perform my research in the field. Similarly I would like to recognize and show my appreciation for all the members of my research group over the past two years. You all helped me with this process, even if it was just by providing a friendly face. Finally, I would like to thank my family and friends for their support. Without them I would be nothing.

Table of Contents

Introduction

Invasive species removal is often used in ecological restoration. It is frequently assumed that the eradication of an exotic species will lead to the recovery of the natural community. However, few studies examine the vegetative community after eradication and there is little consensus among those that have. Treatment method and timing can have a profound impact on the community response (Flory and Clay, 2009). Invasive species removal has been reported to either increase or have no effect on native species richness (DeMeester and Richter, 2010; Swab *et al*., 2008; Stinson *et al*., 2007). In some cases, the removal of one exotic species promoted the invasion of a second species. For example, the removal of dame's rocket (*Hesperis matronalis*) has resulted in the proliferation of multiflora rose (*Rosa multiflora*) and burning bush (*Euonymus alatus*) (Pavlovic *et al*., 2009). Similarly, the control of an invasive fennel (*Foeniculum vulgare*) resulted in the proliferation of a suite of exotic Mediterranean grasses (Ogden and Rejmanek, 2005).

The lack of consensus among the aforementioned studies makes case studies an important tool for discovering the effect of invasive species removal on the resulting vegetative community. Invasive species removal at the North Carolina Museum of Art (NCMA) properties offers an opportunity for such a case study. The NCMA owns a 160 acre Museum Park that is composed of a forest and prairie. The forest is heavily invaded with exotic shrubs, the two most common species being *Elaeagnus umbellata* (autumn olive) and *Ligustrum sinense* (Chinese privet).

In 2002, a section of the forest was treated for the removal of these invasive shrubs to improve aesthetics and provide ecological benefits. The shrubs restricted visibility in the forest and the NCMA was concerned with the visitor experience. Works of art are placed in the forests along a trail system and the thick invasive shrub layer reduced visitor sightlines from the paths and obstructed views of these sculptures.

Invasive species impact the environment in many ways and have been linked to changes in decomposition rates, soil characteristics, forest structure and nutrient cycling (Blair and Stowasser, 2009; Weidenhamer and Callaway, 2010; Asner *et al*., 2008; Scharfy *et al*., 2009). Exotic species invasion is also commonly assumed to be one of the leading causes of biodiversity loss (Pavlovic *et al*., 2009). For example, *Ligustrum sinense* invasion has been found to reduce the growth and survival of native herbaceous species (Greene and Blossey, 2011).

In 2011, the NCMA began considering invasive plant removal in other sections of the forest. This expansion would be implemented with the goals of the initial treatment in mind. The previous restoration effort was examined to help answer the following questions to determine (1) if the treatment was successful in accomplishing its goals; and (2) what effects invasive shrub species removal had on the understory community?

Methods

Site Characteristics

The 160-acre Museum Park at the NCMA contains a network of trails and artwork interspersed among an urban forest and prairie. The Reedy Creek Greenway system also runs through the Museum Park, connecting Meredith College with the Carl Alwin Schenck Memorial Forest. This section of the multipurpose paved trail was completed in 2000.

A 5-acre section of forest was selected for invasive species removal in late September of 2002. Treatment was conducted through the application of a tank mixture of 1% triclopyr and 1.5% glyphosate. Treatment was targeted specifically at non-native woody invasive species, especially *Elaeagnus umbellata* and *Ligustrum sinense*, by trained professionals using back-pack sprayers. All treatments were foliar applications with care to avoid non-target species.

A 9.5-acre section of adjacent forest was chosen to serve as an approximation of a non-treated area for comparison. Both areas are bordered by the Reedy Creek Greenway along their western edges and are located on a north facing slope. Slopes range from 0-45% in the nontreated area and from 0-20% in the treated area. Soils in the nontreated area are Worsham sandy loams, Pacolet sandy loams and Cecil sandy loams. In the treated area the soils are Worsham sandy loams, Pacolet clay loams and Cecil clay loams. (Soil Survey Staff, NRCS, 2011).

Data Collection

There were 40 plots in the treated area and 71 plots in the nontreated area. The nontreated area had 24 edge plots (33.8%) while the treated area had 11 edge plots (27.5%). Plot locations were established in a 25-meter grid pattern in both the treated and nontreated areas. In addition to the plots established by the grid pattern, ten plot locations were placed along the Reedy Creek Greenway in the nontreated area. These ten plots were established in an effort to adequately sample the edge of the nontreated area. Edge plots were those within 15-meters of the paved or graveled trails, or the edge between treated and nontreated sections of the forest in the treated area. Beyond 15-

meters forests retain characteristics of the interior and lose any edge effect (Yates *et al*., 2003). All plot locations were marked by the placement of a single pin flag.

Plots were created by placing a 2x2m PVC pipe frame at each location. The initial pin flag was used as the northwest corner and the plots were aligned in a north-south direction. A second pin flag was placed in the southwest corner for re-sampling purposes in the second year.

Data were collected from July to September of 2010. At each plot, the absolute percent cover of each species present in the plot was estimated. Species were identified according to the nomenclature provided by Radford *et al*. (1968). Cover values were estimated to the nearest percent below fifteen percent. Values over fifteen percent were estimated in five percent intervals. Overhanging vegetation (vegetation not rooted in the plot) was included in this measurement. Additionally, a visual estimation of the canopy cover at each plot location was recorded by the same observer. Canopy cover was defined as the relative percent cover of all trees above three meters in height. Re-sampling occurred in May of 2011 to determine if any species were missed in the prior sample due to the sampling period occurring very close to the end of the growing season.

Photon irradiance was measured during July of 2011. Data were collected using a LI-COR model LI-1400 Data Logger. This was attached to a LI-COR model Quantum light sensor. Three measurements of irradiance were taken across each plot. These measurements were used to find the average irradiance of each plot to account for the effects of sunflecks.

Data Analyses

Analyses of community data were performed in PC-ORD version 5.32 (McCune and Mefford, 2006). Ordination was used to determine if the treated and nontreated plots were a part of the same community. Three dimensional non-metric multidimensional scaling (NMS) was performed on both the 2010 and 2011 data.

Multi-response permutation procedures (MRPP) were then used to test for differences in community composition between the treatments. MRPP were also used to compare the edge plots with the interior plots. MRPP results in the derivation of a test statistic (t) that "is the difference between the observed and expected deltas divided by the square root of the variance in delta" and "describes the separation between the groups" (McCune and Mefford, 2006). It also generates a statistic that describes the within-group homogeneity (A). This value equals one when there is no variation among members of a single group. It equals zero when "heterogeneity within groups equals expectation by chance" and can be negative if there is more heterogeneity within groups than predicted by chance (McCune and Mefford, 2006). Statistical significance can occur even when dealing with a low A statistic due to a large enough sample size. Therefore, one must "carefully consider the ecological significance of the result, not just the statistical significance" (McCune and Mefford, 2006). Significance for this, and all following tests, was determined as having a p-value of 0.05 or less.

Indicator species analysis (ISA) was performed to detect the value of species in indicating both the treatments and the location of plots along the edge or interior of the forest. ISA tests calculate both the relative abundance and frequency of each species to compute an indicator value. This indicator value is then subjected to a randomization test to determine statistical significance compared to the frequency and abundance expected by chance alone.

Correlation between canopy cover and average vegetative cover per plot was performed in Microsoft Excel (Microsoft Corporation, 2003) using a linear regression approach. Additional correlations were performed between canopy cover and the average cover per plot of individual species as well as with photon irradiance. The relationship between canopy cover and photon irradiance was also examined.

Data taken in 2010 was compared to the 2011 data using paired t-tests in Microsoft Excel (Microsoft Corporation, 2003). T-tests were also used to compare values that used plots as the sample. This includes comparisons of average light levels and canopy cover. T-tests could not be performed to test the difference between treatments because there was a sample size of one. Equivalence of variance was assumed and all tests were conducted with two tails.

Average cover per plot was also examined by grouping the species into growth forms. These growth forms included trees, shrubs, graminoids, vines and forbs. Growth form selection followed the growth habit designation in the United States Department of Agriculture PLANTS Database (NRCS, 2011). The PLANTS Database was also used to determine the status of each species as exotic or native to the state of North Carolina.

Results

The nontreated plots contained 63 species in both 2010 and 2011. There were 52 species observed in the treated plots in 2010, but 60 species were encountered in 2011. Four species were observed in 2010 that were not measured in 2011. These species combined for an average cover per plot of 0.02 percent. In 2011, three new species were measured that combined for an average cover per plot of 0.46 percent. These values are negligible when compared to the average total cover per plot in 2010 (37%) and 2011 (39%). There was no significant difference in total cover between years ($t_{110} = 0.81$, $p = 0.42$).

Examination of the cover by treatment yielded many significant differences between the two years (Table 1). The cover of treated plots was lower in 2011 than in 2010 ($t_{39} = -2.48$, $p = 0.02$) while the cover in the nontreated plots was higher ($t_{70} = 3.46$, $p = 0.001$). The cover of native species in the treated plots remained the same between the two years but was greater in the untreated plots in 2011 than 2010 ($t_{70} = -3.76$, $p < 0.001$). The cover of exotic species was greater in 2010 than 2011 in the treated plots ($t_{39} = 2.74$, $p = 0.009$) but was the same between years in the untreated plots.

An indicator species analysis was run for both the 2010 and 2011 data. The 2010 data contained three significant indicators of the nontreated plots and seventeen indicators of the treated plots. The nontreated plots were indicated by *Elaeagnus umbellata*, *Lonicera japonica* (Japanese honeysuckle) and *Carya tomentosa* (mockernut hickory). The indicators of the treated plots that contained higher than 0.4% cover per plot were *Hedera helix* (English ivy), *Microstegium vimineum* (Nepalese browntop/Japanese stiltgrass), *Rubus argutus* (sawtooth blackberry), *Vitis rotundifolia* (muscadine grape), *Toxicodendron radicans* (poison ivy), *Parthenocissus quinquefolia* (Virginia creeper), and *Acer rubrum* (red maple). The ISA of the 2011 data yielded very similar results. Nontreated plots were still indicated by *Elaeagnus umbellata*. The treated plots were indicated by fifteen species, thirteen of which overlapped with the

2010 list. The indicator species with large amounts of cover were chosen for further examination in order to discover any differences that might exist between the two treatments.

The differences in cover between treatments of four native indicator species were about 9% in 2010 and 7% in 2011 (Table 2). This corresponds to the total differences in all native cover observed between treatments. It is also apparent that even though the total average cover of exotic species remained constant between treatments in 2010, the composition of that cover did not. There was much less cover in the nontreated plots than the treated plots of *Elaeagnus umbellata*, *Ligustrum sinense*, and *Lonicera japonica* (Table 3). This difference in exotic woody cover was also observed in 2011. However, in 2010, the average cover of *Microstegium vimineum* in the treated plots was much greater than in the nontreated plots. This difference in cover of *Microstegium vimineum* was not observed in May 2011 because of *Microstegium*'s life cycle. This species continues to grow throughout the growing season and can flower as late as September (Mehrhoff, 2000; Gibson *et al*., 2002).

Due to the similarity between the 2010 and 2011 data, further analyses were only performed on the 2010 data.

Growth Form

The average cover of forbs and trees did not differ between the treatments (Table 4). However, differences in cover were observed for graminoid, vine and shrub cover. Graminoid cover was higher in the treated area with *Microstegium vimineum* being the dominant species. Cover of the native vines increased in the treated area (*Parthenocissus* *quinquefolia*, *Toxicodendron radicans*, and *Vitis rotundifolia*). Shrub cover, especially *Elaeagnus umbellata* and *Ligustrum sinense*, was higher in the nontreated area than the treated area.

Ordination

Three dimensional NMS ordination revealed no clear separation between the treated and nontreated plots in either 2010 (Fig. 1). The results of the MRPP when grouped by treatment also showed there to be no difference in community between the nontreated and treated areas. The low A values derived ($t = -10.21$, $A = 0.0399$, $p < 0.0001$) suggest that there is a strong likelihood that the homogeneity observed within the groups is due to chance. The significance of the test was likely a result of the large sample size, rather than any true separation of the data.

Canopy Cover

There was more canopy cover in the treated area (60%) than the nontreated area (51%) ($t_{109} = -2.98$, $p = 0.004$). Canopy cover was correlated to total understory cover ($F_{1, 109} = 25.05$, $p<0.001$). While the relationship was significant, it was not that strong ($r=0.43$). This correlation was observed in both the treated area ($F_{1, 38} = 10.47$, $p=0.003$) and nontreated areas ($F_{1, 69} = 28.92$, $p<0.001$).

The data were then examined for relationships between canopy cover and growth form. Shrubs were the only growth form to be significantly correlated to canopy cover ($F_{1, 109} = 67.48$, $p<0.0001$). This relationship was then examined by treatment. While shrub cover was correlated to canopy cover in the nontreated area ($F_{1, 69} = 53.00$, p<0.001) they were not correlated in the treated area ($F_{1, 38} = 1.85$, p=0.182). Individual species were then examined to discover if the difference in canopy could be the reason for any difference in cover (Table 5). The exotic shrubs *Elaeagnus umbellata* and *Ligustrum sinense* were significantly correlated to canopy cover. This corresponded to the relationship observed with the shrub growth form. However, none of the other species tested, except *Lonicera japonica*, were significantly correlated to canopy cover. Canopy cover was also correlated to photon irradiance ($F_{1, 109} = 71.62$, p<0.0001). This correlation was observed in both the treated ($F_{1, 38} = 6.40$, p=0.016) and nontreated areas($F_{1, 69} = 28.03$, p<0.001). The regression that best described the relationship between light and canopy cover was a second order polynomial ($R^2 = 0.51$) (Fig. 2). Additionally, photon irradiance was correlated to total shrub cover ($F_{1, 109} = 91.13$, p<0.0001) and to the shrub cover in the nontreated area ($F_{1, 69} = 81.54$, p<0.001). Photon irradiance was not correlated to the shrub cover in the treated area ($F_{1, 38} = 0.82$, p=0.371). There were also strong correlations between photon irradiance and *Elaeagnus umbellata* (r = 0.60) and *Ligustrum sinense* (r = 0.48).

Edge Effect

There was greater vegetative cover in edge plots (50%) than interior plots (31%) ($t_{109} = -3.82$, p < 0.001). A MRPP grouping of the plots by location found that this classification was not robust in 2010 (t = -5.22, A = 0.02, p = 0.001). Again, the significant result of the test was most likely due to the large sample size. The extremely low A value was interpreted as the within-group homogeneity being the result of chance alone.

An indicator species analysis found no significant indicators of interior plots in 2010. There were seven indicators of edge plots including *Elaeagnus umbellata*, *Ligustrum sinense*, *Rosa multiflora* (multiflora rose), and *Rubus argutus*. Analyses of these four species by average cover value did display this preference for an edge position in the landscape (Table 6). Of these species, *Elaeagnus umbellata* and *Ligustrum sinense* had been shown to be correlated to canopy cover and photon irradiance.

The canopy cover in the edge plots (47%) was significantly lower than the interior plots (57%) ($t_{109} = 3.19$, $p = 0.002$). Similarly, the photon irradiance was significantly lower in the interior plots (51 μmol/m^2/s) than the edge plots (286 μmol/m^2/s) ($t_{109} = -5.43$, $p < 0.001$).

Discussion

Treatment Success

The first question posed was whether or not invasive shrub removal successfully achieved the goals established by the NCMA. First, herbicide treatment did reduce cover of woody exotic vegetation. Percent cover of *Elaeagnus umbellata* and *Ligustrum sinense* was less in the treated area compared to the nontreated area. However, removal of these invasive shrubs was correlated with an increase in *Microstegium vimineum*. This observation is consistent with previous reports. Hanula *et al*. (2009) reported that the removal of *Ligustrum sinense* could increase cover of *Microstegium vimineum*. If the criterion for treatment success was that the total exotic cover was reduced, then there was evidence that this particular treatment failed.

Second, sightlines in the forest were improved by the removal of *Elaeagnus umbellata* and *Ligustrum sinense*. Although the cover of *Microstegium vimineum* had increased, this plant is short in stature and does not impede sightlines. Therefore, if success is measured based on the aesthetic of sightline, then the treatment would be deemed successful.

Effects on the Vegetative Community

The native community may not have suffered much from the invasion of exotic shrubs. Native species were still present and regenerating under this shrub layer and the removal of these exotic species had little impact on the native cover. The foliar application of the herbicide put these non-target species at risk, but the forb and tree cover remained constant between the two areas. The equal cover of these native species suggests that they may not have been adversely impacted by the treatment application or the presence of the invasive shrubs. This explanation is unlikely however, because the cover of native vines was higher in the treated area, indicating some treatment effect.

The native community may have been impacted by the exotic shrub cover, but had not recovered due to the introduction of a new competitor. The effective replacement of *Elaeagnus umbellata* and *Ligustrum sinense* cover with *Microstegium vimineum* cover suggests that the competitive pressure from exotic species may be equal in the two areas. Native vines may have increased their cover in the treated area due to some form of competitive release following the removal of the exotic shrubs.

It has also been observed that *Microstegium* suppresses forest succession by reducing tree regeneration due to competition (Flory and Clay, 2010). This competition between native and exotic species is often due to the influence of invader control over resources, rather than propagule pressure (Yurkonis *et al*., 2005). This places a limit on species establishment but does not result in an increase in species extinction (Yurkonis *et al*., 2005), which would explain the similar richness and cover values recorded between the treatments. It is possible that this proposed resource domination by *Microstegium* is a direct replacement of the control over resources that the invasive shrubs may have exhibited.

In addition to competitive pressure, it is also possible that the community has not recovered due to the surrounding area being heavily invaded. If the entire forest had suffered a decrease in native species, there may have been no available corridor for new seeds to migrate into the area. This would signify that the area had passed an ecological barrier to restoration and that more intensive methods of restoration must be applied.

The origin of the *Microstegium vimineum* was unknown. Studies have shown that *Microstegium* cover decreases dramatically under the cover of midstory trees due to the decrease in light levels (Cole and Weltzin, 2005). *Elaeagnus umbellata* has also been demonstrated to produce a large amount of shade in the understory (Brantley and Young, 2010). Therefore, it is possible that the loss of light competition, due to the removal of the invasive shrub layer, resulted in the large increase in *Microstegium* cover in the treated area.

However, *Microstegium* was not present in the interior of the nontreated area where the cover of the invasive shrubs was already low. Some studies have shown that *Microstegium* was absent from many suitable sites and that determining good indicators is difficult (Gibson *et al*., 2002; Cole and Weltzin, 2004). One possible explanation for the difference in cover between the two areas was the presence of site disturbance. *Microstegium* invasions often rely upon disturbed soils and vegetation as it is a poor invader on its own (Barden, 1987; Nord *et al*., 2010; Rauschert *et al*., 2010). It is possible that *Microstegium* was unable to compete with the vegetation present on the nontreated side. It has been demonstrated that *Microstegium* does not establish successful populations in extant populations of *Lonicera japonica*, a significant indicator of the nontreated area (Barden, 1987). In addition to vegetative disturbance, Microstegium has been associated with soil disturbance (Mortensen *et al*., 2009; Christen and Matlack, 2009). The soils in the treated area may have been disturbed during the construction of the trail system present in that area. The soil series present in the treated area were the same as in the nontreated area, but they differed in surface texture. It is possible that the surface texture was changed during trail creation or subsequent erosion, providing enough disturbance for *Microstegium* to invade. Additionally, it is possible that the trail system provided a means for human dispersal of *Microstegium* seeds into the treated area. Human dispersal is considered the most important catalyst of *Microstegium* invasion (Mortensen *et al*., 2009) and *Microstegium* has been documented as ubiquitous along trails (Cole and Weltzin, 2004; Christen and Matlack, 2009).

Regardless, the fact stands that the treated area contained largely the same understory community as the nontreated area. The successful removal of the target species, may encourage the NCMA to extend treatment into other sections of the forest. This expansion should come with the expectation of maintaining the same cover of forbs and tree species that were present before treatment. Additionally, treatment should increase the cover of native vine species but may also increase the cover of

Microstegium. Therefore, treatment expansion likely depends on the NCMA's preference for the invasive shrubs over the invasive grass.

Canopy and Edge

Invasion processes often occur along forest edges (McDonald and Urban, 2006; Mortensen *et al*., 2009) as urban forests are often bordered by a system of paved trails or roads. The NCMA forest is bisected by the Reedy Creek Greenway. These kinds of transportation grids act as a network for invasion (Nemec *et al*., 2011). The long linear paths promote the movement of people and have been theorized to receive greater seed inputs. Additionally, these trails break open the canopy of the forest. Canopy opening is one of the strongest factors influencing invasion (Gurevitch *et al*., 2008) and may be an influencing factor in the edge effect. It is also possible that these edge areas foster invasions due to soil disturbance caused by the construction of the pathways. Soil bulk density, texture, structure, and pH could all be influenced by the construction of a paved greenway system.

Edge plots were found to have much lower canopy cover and much higher photon irradiance than interior plots. These relationships are very intuitive since the edge plots are bordered by the open canopy over the paths on one side. One obvious effect of canopy was on the average total cover of vegetation. As the canopy cover decreased, the total understory cover increased. However, only three species, all exotic, were found to have significant correlations with canopy cover. Of these three species, *Elaeagnus umbellata* and *Ligustrum sinense* were also significant indicators of edge plots. Other indicators of edge plots included the exotic *Rosa multiflora* and the native *Rubus argutus*, a significant indicator of forest edges in the North Carolina Piedmont (McDonald and Urban, 2006).

Previous research has found *Elaeagnus umbellata* to exhibit a significant edge effect and an association with site disturbance (Ibáñez *et al*., 2009). Another survey found both *Elaeagnus umbellata* and *Ligustrum sinense* to be associated with edges (Drake *et al*., 2003). This corroborates the preference of this species for edges in the NCMA Park. One study found *Elaeagnus umbellata* to have no significant difference in density between edge and interior areas (Yates *et al*., 2003). However, the density was still higher at the edges. It is possible that *Elaeagnus umbellata* establishes itself along the edges of forests and then has the ability to invade to the interior. This process may be happening at the NCMA, but the interior population was not yet significant.

Conclusions

Exotic species removal has had several interesting effects on the North Carolina Museum of Art Park forest. The treatment was effective at removing the target species, and they have not regenerated in large abundances in the eight years since treatment. The museum also accomplished one of their primary goals in clearing out the understory shrubs to improve sightlines. This will provide visitors with a better view of the works of art placed along the paths. However, the treatment was not successful at decreasing the total cover of exotic species. Rather the species composition was shifted from a tall shrub layer to a lower-growing layer of graminoids.

Of note to restoration practitioners is that the introduction of edge areas with low canopy levels will encourage invasion by non-native species. The construction of large, paved greenways creates strips of land that can harbor invasive species and should be avoided or reduced in impact. It is also possible that maintaining a closed canopy above the path could reduce the invasibility of these edges. For the NCMA, the edge areas of the forest present a logical location for invasive species removal treatments in the future. These areas contained the largest cover of exotic species and their location near the greenway means that clearing the shrub layer would increase sightlines the most.

Removal of one exotic species only to see it replaced by another is becoming a common issue in restoration. This is also the biggest issue that the museum faces if it wants to expand treatments into further areas of the forest. More studies need to be done that examine the timing and technique of invasive species removal in order to find combinations that best allow native species to recover and prevent different exotic species from invading. It is also possible that this progression from exotic shrubs to exotic grasses becomes a routine step in restoration and both stages must be treated separately before the problem of invasive species can be resolved. If this is the case, the NCMA must carefully select their goals in the future. If the removal of exotic shrub cover is enough to increase sightlines and they are not concerned with the potential ecological impacts of a possible *Microstegium* invasion, then continuing the current treatment regime seems to be the right decision. However, if they want to guarantee the removal of exotic species and restore the forest to a more native state, then they may have to commit to a more intensive, long-term management plan.

Tables and Figures

Table1: Comparison of cover values between years

	2010		2011	
Cover	Treated	Nontreated	Treated	Nontreated
Total	44	33	36*	40*
Native	18	9	19	12*
Exotic	24	24	14*	28

Values presented are the average percent cover of all plant species per plot. Significant differences observed between the two years ($p < 0.05$) are marked by an asterisk (*) by the 2011 value.

Table 2: Average percent cover per plot of native species

	2010		2011	
Species	Treated	Nontreated	Treated	Nontreated
Toxicodendron radicans	5	1	7	3
Vitis rotundifolia	4	1	3	1
Parthenocissus quinquefolia	2	1	3	2
Acer rubrum	1	trace	1	trace
Total native cover	18	9	19	12

Species listed were significant indicators of treatment and were present in the largest average cover among all indicator species. Trace values are those that are less than one percent. The common trend was a higher cover in the treated area when compared to the nontreated area.

Table 3: Average percent cover per plot of exotic species

	2010		2011	
Species	Treated	Nontreated	Treated	Nontreated
Elaeagnus umbellata	4	13	4	16
Ligustrum sinense	1	4	1	3
Lonicera japonica	1	2	3	5
Microstegium vimineum	17	3	4	trace
Total invasive cover	24	24	14	27

All species listed were significant indicators of treatment and were present in the largest average cover among indicator species. Trace values are those that are less than one percent. The common trend was the higher cover in the nontreated area when compared to the treated area with the notable exception of *Microstegium vimineum*.

Table 4: Average percent cover per plot of vegetative growth forms

Growth Form	Treated	Nontreated
Graminoid	18	4
Vine	14	5
Shrub	7	18
Tree	3	2
Forb	3	4

Tree and forb cover remained the same between treatments while large differences were observed in shrub, vine and graminoid cover. All values are from 2010 data.

Table 5: Correlations between canopy cover and species

Species	F	p	r
Ligustrum sinense	46.95	<0.0001	0.55
Elaeagnus umbellata	30.10	<0.0001	0.46
Lonicera japonica	14.52	<0.0001	0.35
Vitis rotundifolia	1.90	0.17	
Toxicodendron radicans	1.89	0.17	
Acer rubrum	1.64	0.20	
Rubus argutus	0.95	0.33	
Parthenocissus quinquefolia	0.58	0.45	
Rosa multiflora	0.29	0.59	
Microstegium vimineum	0.001	0.97	

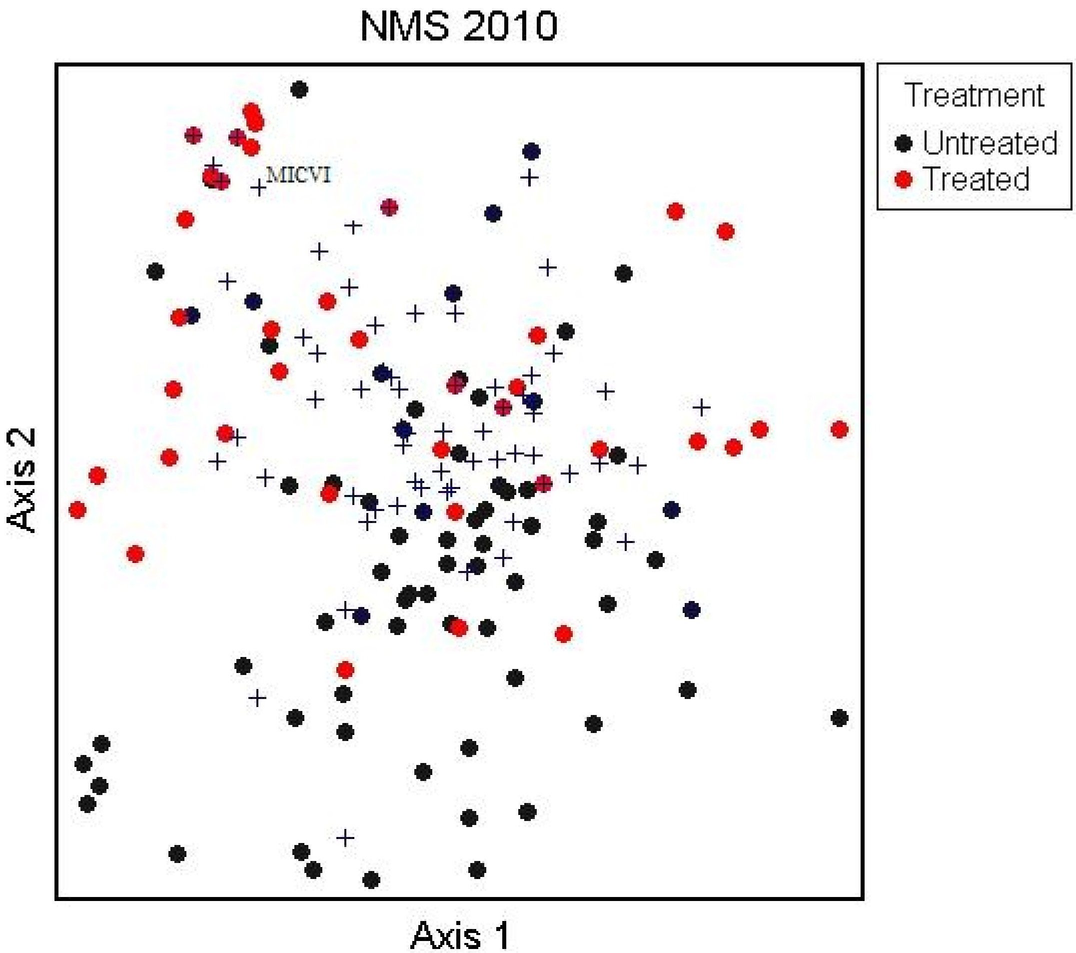
Species listed are those that had the largest average cover values among all species. Degrees of freedom of the F statistic are 1 and 109. Significance is determined as $p < 0.05$. The r values are only given for significant relationships. All values were taken from 2010 data.

Table 6: Average percent cover per plot in edge plots

Species	Edge	Interior
Elaeagnus umbellata	17	6
Ligustrum sinense	7	1
Rosa multflora	2	trace
Rubus argutus	1	trace

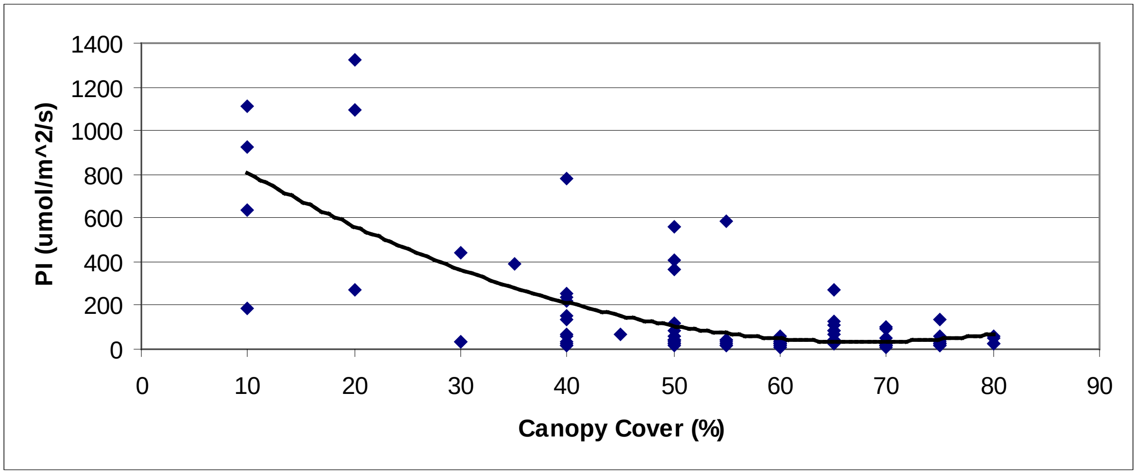
Species listed are significant indicators of edge plots. All species displayed larger cover in the edge plots than the interior plots. Trace values are those that are less than one percent. All values were taken from 2010 data.

Figure 1: NMS of 2010 community data



Red circles represent treated plots and black circles represent nontreated plots. Species are represented as blue crosses. There is no clear separation of plots by treatment. Rather it appears that the treated plots represent a subsample of the nontreated plots. The species are similarly clumped with no clear separation. *Microstegium vimineum* is labeled as a possible driving force behind the concentration of the treated plots along the upper part of axis 2.

Figure 2: Relationship between canopy cover and photon irradiance



Canopy cover is related to photon irradiance by the following expression: PI = $0.23(\text{Canopy})^2 - 31.35(\text{Canopy}) + 1096.5$. This relationship explains roughly half of the data ($R^2 = 0.51$). All data were taken from 2011.

Works Cited

Asner, G.P., R.F. Hughes, P.M. Vitousek, D.E. Knapp, T. Kennedy-Bowdoin, J. Boardman, R.E. Martin, M. Eastwood, and R.O. Green. 2008. Invasive plants transform the three-dimensional structure of rain forests. Proceedings of the National Academy of Sciences of the United States of America, 105(11): 4519-4523.

Barden, L.S. 1987. Invasion of *Microstegium vimineum* (Poaceae), an exotic, annual, shade-tolerant, C4 grass, into a North Carolina floodplain. The American Midland Naturalist, 118: 40-45.

Blair, B.C. and A. Stowasser. 2009. Impact of *Lonicera maackii* on decomposition rates of native leaf litter in a southwestern Ohio woodland. Ohio Journal of Science, 109(3): 43-47.

Brantley, S.T. and D.R. Young. 2010. Linking light attenuation, sunflecks, and canopy architecture in mesic shrub thickets. Plant Ecology, 206: 225-236.

Christen, D.C. and G.R. Matlack. 2009. The habitat and conduit functions of roads in the spread of three invasive plant species. Biological Invasions, 11: 453-465.

Cole, P.G. and J.F. Weltzin. 2004. Environmental correlates of the distribution and abundance of *Microstegium vimineum*, in east Tennessee. Southeastern Naturalist, 3(3): 545-562.

---. 2005. Light limitation creates patchy distribution of an invasive grass in eastern deciduous forests. Biological Invasions, 7: 477-488.

DeMeester, J.E. and D.deB. Richter. 2010. Restoring restoration: removal of the invasive plant *Microstegium vimineum* from a North Carolina Wetland. Biological Invasions, 12: 781-793.

Drake, S.J., J.F. Weltzin and P.D. Parr. 2003. Assessment of non-native invasive plant species on the United States Department of Energy Oak Ridge National Environmental Research Park. Castanea, 68(1): 15-30.

Flory, S.L. and K. Clay. 2010. Non-native grass invasion suppresses forest succession. Oecologia, 164: 1029-1038.

---. 2009. Invasive plant removal method determines native plant community responses. Journal of Applied Ecology, 46: 434-442.

Gibson, D.J., G. Spyreas, and J. Benedict. 2002. Life History of *Microstegium vimineum* (Poaceae), an invasive grass in southern Illinois. Journal of the Torrey Botanical Society, 129(3): 207-219.

Greene, B.T. and B. Blossey. 2011. Lost in the weeds: *Ligustrum sinense* reduces native plant growth and survival. Biological Invasions, published online 13 April 2011.

Gurevitch, J., T.G. Howard, I.W. Ashton, E.A. Leger, K.M. Howe, E. Woo, and M. Lerdau. 2008. Effects of experimental manipulation of light and nutrients on establishment of seedlings of native and invasive woody species in Long Island, NY forests. Biological Invasions, 10:821-831.

Hanula, J.L., S. Horn and J.W. Taylor. 2009. Chinese privet (*Ligustrum sinense*) removal and its effect on native plant communities of riparian forests. Invasive Plant Science and Management, 2(4): 292-300.

Ibáñez, I., J.A. Silander Jr., J.M. Allen, S.A. Treanor and A. Wilson. 2009. Identifying hotspots for plant invasions and forecasting focal points of further spread. Journal of Applied Ecology, 46: 1219-1228.

McCune, B. and M.J. Mefford. 2006. PC-ORD. Multivariate Analysis of Ecological Data. Version 5.32. MjM Software, Glenedon Beach, Oregon, U.S.A.

McDonald, R.I. and D.L. Urban. 2006. Edge effects on species composition and exotic species abundance in the North Carolina Piedmont. Biological Invasions, 8: 1049-1060.

Mehrhoff, L.J. 2000. Perennial *Microstegium vimineum* (Poaceae): an apparent misidentification? Journal of the Torrey Botanical Society, 127(3): 251-254.

Microsoft Corporation. 2003. Microsoft Office Excel 2003. Microsoft Office Standard Edition 2003 for Students and Teachers.

Mortensen, D.A., E.S.J. Rauschert, A.N. Nord and B.P. Jones. 2009. Forest roads facilitate the spread of invasive plants. Invasive Plant Science and Management, 2(3): 191-199.

Nemec, K.T., C.R. Allen, A. Alai, G. Clements, A.C. Kessler, T. Kinsell, A. Major, and B.J. Stephen. Woody invasions of urban trails and the changing face of urban forests in the Great Plains, USA. The American Midland Naturalist, 165(2): 241-256.

Nord, A., D.A. Mortensen and E.S.J. Rauschert. 2010. Environmental factors influence early population growth of Japanese stiltgrass (*Microstegium vimineum*). Invasive Plant Science and Management, 3(1): 17-25.

Ogden, J.A. and M. Rejmanek. 2005. Recovery of native plant communities after the control of a dominant invasive plant species, *Foeniculum vulgare*: Implications for management. Biological Conservation, 125: 427-439.

Pavlovic, N.B., S.A. Leicht-Young, K.J. Frohnapple, and R. Grundel. 2009. Effect of removal of *Hesperis matronalis* (dame's rocket) on species cover of forest understory vegetation in NW Indiana. The American Midland Naturalist, 161(1): 165-176.

Radford, A.E., H.E. Ahles and C.R. Bell. 1968. Manual of the vascular flora of the Carolinas. Chapel Hill: University of North Carolina Press.

Rauschert, E.S.J., D.A. Mortensen, O.N. Bjørnstad, A.N. Nord and N. Peskin. 2010. Slow spread of the aggressive invader, *Microstegium vimineum* (Japanese stiltgrass). Biological Invasions, 12: 563-579.

Scharfy, D., H. Eggenschwiler, H. Olde Venterink, P.J. Edwards, and S. Gusewell. 2009. The invasive alien plant species *Solidago gigantea* alters ecosystem properties across habitats with differing fertility. Journal of Vegetation Science, 20: 1072-1085.

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available online at http://websoilsurvey.nrcs.usda.gov/ accessed July 26, 2011.

Stinson, K., S. Kaufman, L. Durbin, and F. Lowenstein. 2007. Impacts of garlic mustard invasion on a forest understory community. Northeastern Naturalist, 14(1): 73-88.

Swab, R.M., L. Zhang, and W.J. Mitsch. 2008. Effect of hydrologic restoration and *Lonicera maackii* removal on herbaceous understory vegetation in a bottomland hardwood forest. Restoration Ecology, 16(3): 453-463.

USDA, NRCS. 2011. The PLANTS Database (http://plants.usda.gov, 28 June 2011). National Plant Data Team, Greensboro, NC 27401-4901 USA.

Weidenhamer, J.D. and R.M. Callaway. 2010. Direct and indirect effects of invasive plants on soil chemistry and ecosystem function. Journal of Chemical Ecology, 36: 59-69.

Yates, E.D., D.F. Levia Jr. and C.L. Williams. 2003. Recruitment of three non-native invasive plants into a fragmented forest in southern Illinois. Forest Ecology and Management, 190: 119-130.

Yurkonis, K.A., S.J. Meiners, and B.E. Wachholder. 2005. Invasion impacts diversity through altered community dynamics. Journal of Ecology, 93: 1053-1061.