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Recovery of Native Plant Species After Initial Management of Non-Native Plant Invaders

Vegetation Monitoring in an Exclosure in Morristown National Historical Park

Natural Resource Report NPS/MORR/NRR—2020/2168

ON THE COVER

Fenced exclosure 2003, with vegetation returning on the left (inside) compared to the unfenced area (right side) (MANISHA PATEL)

Recovery of Native Plant Species After Initial Management of Non-Native Plant Invaders

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August 2020

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

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Please cite this publication as:

Ehrenfeld, J. G., K. A. Ross, M. Patel, J. N. Epiphan, and S. N. Handel. 2020. Recovery of native plant species after initial management of non-native plant invaders: Vegetation monitoring in an exclosure in Morristown National Historical Park. Natural Resource Report NPS/MORR/NRR—2020/2168. National Park Service, Fort Collins, Colorado. https://doi.org/10.36967/nrr-2278124

NPS 337/173636, August 2020

Contents

Contents (continued)

Figures

Figures (continued)

Page

Tables

Abstract

This report summarizes research conducted in the Native Plant Demonstration Project, within a 0.8 hectare (2 acre) exclosure in the Jockey Hollow section of Morristown National Historical Park between 2002 and 2008. To establish the experiment, a 0.8 hectare area was cleared of the dense infestation of Japanese barberry (*Berberis thunbergii*) and other invasive shrubs. A 2.4m (8ft) deer exclosure was erected around the cleared 0.8 ha area. Canopy trees were inventoried in 2003. Percent cover of understory vegetation was monitored within the exclosure in 2003, 2004, and 2008 to determine if and how restoration would proceed without further intervention. For comparison, understory vegetation outside the exclosure was surveyed along four transects in 2004.

Removal of the uniform and dense Japanese barberry population allowed other invasive species to flourish as the understory received higher light levels. In 2003–2004 Japanese stilt grass (*Microstegium vimineum*) formed a dense carpet across the exclosure. In July 2003, volunteers and project staff manually removed the stiltgrass and many remaining barberry resprouts. Native plant diversity increased slowly during the five years of monitoring. Even though invasive woody vegetation initially decreased, invasive plants dominated in cover after five years. Native woody plant percent cover increased slowly over time, but it was not sufficient to establish a native woody plant stratum five years after the initial management.

In April 2003, soil manipulations were undertaken to determine whether restoration could be enhanced by modifying soil conditions. The soil amendment treatments were applied to sixteen 3m x 3m quadrats established within the exclosure following a Latin square design, and included: 1) removal of the top 5cm of organic-rich material (enriched by non-native earthworm casts); 2) addition of wood chips to increase organic matter, particularly of lignin-rich, recalcitrant, slowly-decomposing material; 3) addition of aluminum sulfate to reduce soil pH, in an attempt to limit N availability and alter habitat for exotic earthworms; and 4) control plots that were similarly disturbed but no soil amendments were applied. While the treatments were successful in modifying some soil properties in the directions desired, there was little evidence that these changes affected vegetation in the years samples were surveyed.

In summary, removal of non-native, invasive species without further vegetation management allows other invasive species to become established when light conditions are altered and the invasives are released from intensive deer browse. However, native woody and herbaceous plants do become established after initial intensive planting efforts and manual weeding, suggesting that over time, a native understory vegetation community will re-establish in areas protected from deer browse. We found that soil manipulations were effective in altering soil properties, but these changes have little apparent effect on plant establishment in the first few years after treatment.

Acknowledgments

We are indebted to Robert Masson, the biologist of MORR, for all his help with the logistics of setting up and maintaining the exclosure fence. Without his enthusiastic and energetic help, this project would not have been possible. Thank you to Jack Siegrist for his assistance collating data sets and running them in the statistical program R. We also thank the countless Rutgers students, MORR park staff, and volunteer groups who helped with the removal of invasives species and vegetation surveys. We thank the National Park Service for providing funding for the study.

This study is dedicated to Professor Joan G. Ehrenfeld whose untimely death removed a brilliant and dynamic ecologist from the scene. Jean N. Epiphan and Steven N. Handel edited this final version for submission to the NPS.

Introduction

Restoration of natural areas that are heavily infested with non-native invasive plant species poses several challenges. Methods of removing undesirable species are varied and optimal procedures are frequently researched but can have mixed results unless long term efforts continue after initial removal or treatments (Flory 2010; Love and Anderson 2009; Ward et al. 2009). Following removal, decisions must be made as to the optimal approach for restoring native species. In active restoration, desirable species may be planted (Benayas et al. 2009; Middleton et al. 2010), but this requires substantial resources of personnel and money to accomplish on any but very small tracts of land. An alternative is to allow native species to recolonize by themselves; this approach may be necessary when resources for re-planting are limited and/or areas to be restored are large. In this case, dispersal limitations may restrict the composition of the final community. Alternatively, dispersal of desirable species may result in a higher diversity than can be managed by planters. In both cases, individuals of desirable native species may have been present and are potentially released from competition by the removal of the invasive species (Runkle et al. 2007; Swab et al. 2008). A bud or propagule bank of desirable species may be present, providing a process by which pre-existing individuals of native plant species can be encouraged to re-occupy a site following invasive plant removal. However, removal of the invasive species may open up space and allow light penetration that promotes the re-establishment of those invasive species or the colonization of new ones (Swab et al. 2008).

The development of a restored community following the removal of invasive species will be affected by a number of possible processes, sometimes referred to as community assembly rules. Priority effects are due to the ability of an early-establishing species to limit or prevent the establishment of later-arriving species (Grman and Suding 2009). Priority effects may result from greater competitive ability or from size-asymmetry, in which the first-arriving species commandeers resources because of its greater size than later arrivals. Grman and Suding (2009) showed that alteration of soil conditions by a first-arriving species can leave a legacy that determines subsequent establishment of other species. When invasive species can exert both size-based and soil legacy effects, restoration may be particularly difficult. Another aspect of community assembly is the niche similarity of the desirable (native) and undesirable (invasive) species (Funk et al. 2008). When the undesirable invasive is similar in growth form, size, and resource use to the native species, it may be easier to suppress the invasive with native species than when the invasive occupies a different niche or has different life history traits (Funk et al. 2008). Finally, the management regimes utilized to complement successional processes are often important considerations in forest restoration (McClain et al. 2010). McClain et al. (2010) found that initial floristic condition prior to restoration efforts can determine the natural succession, which can serve to restore some generalist native understory species. However, active long-term management and direct planting are needed to restore entire plant communities.

We report a study that combined passive and active approaches to restoration in a 0.8 hectare (2 acre) fenced area (hereafter, exclosure) in the Jockey Hollow section of Morristown National Historical Park. The vegetation of the park has been well documented (Ehrenfeld 1982; Dibeler and Ehrenfeld 1990; NatureServe 2020) and consists of mixed-hardwood forests. There were limitations in the funding available for the project, therefore the restoration involved planting a relatively small number of native species and otherwise allowing vegetation to recruit by itself following removal of invasive plants. The dominant understory species prior to restoration was Japanese barberry (*Berberis thunbergii*); in the area designated for restoration, a continuous and dense thicket (approximately 10 stems m^2) was present (Figure 1). With the installation of a native plant demonstration exclosure, we sought to answer three questions:

1. What are the trajectories of abundance of both individual invasive and dominant native species over time, following the invasive removal?
2. What are the trajectories over time of species groups (all invasives, all herbs, shrubs, tree seedlings)?
3. Can soil conditions be manipulated to limit invasive species growth and enhance native woody seedling growth?

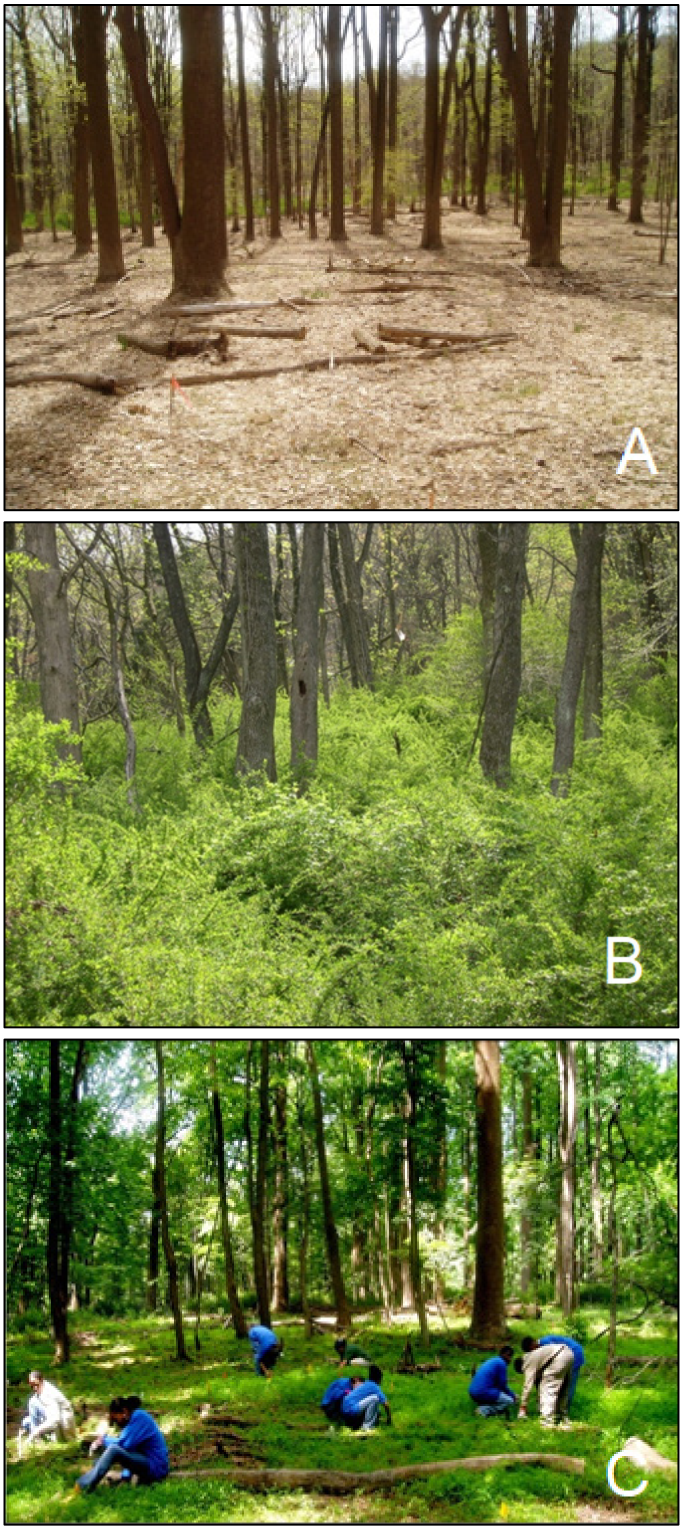

Figure 1. View of Japanese barberry (*Berberis thunbergii*) populations in the exclosure prior to clearance and exclosure installation in April 2002 (A); view of exclosure in early May 2003 after exclosure installation (B); July 2003, Youth Conservation Corps volunteers hand pulling Japanese stiltgrass (*Microstegium vimineum*) which is most of the vegetation visible on the ground (C). (KRISTEN A. ROSS).

Methods

Study Site Establishment

Morristown National Historical Park (MORR) is located in northern New Jersey in the central portion of the Northeastern Interior Dry-Mesic Oak Forest range, which extends from West Virginia to Massachusetts and Rhode Island (NatureServe 2020). It is the dominant ecological cover-type (over 80%) in MORR and representative of the geographical area (Figure 2). In a previous survey, it was found that more than half of MORR is heavily colonized by several, regionally common, invasive species, including Japanese barberry and Japanese stiltgrass (*Microstegium vimineum*), wineberry (*Rubus phoenicolasius*), multiflora rose (*Rosa multiflora*), and others (Dibeler and Ehrenfeld 1990). In addition, the forest has been subject to very high white-tailed deer (*Odocoileus virginianus*) populations, 75 to 100 per square mile (0.3–0.4 per hectare), since the 1980s (R. Masson, National Park Service, personal communication, 2002) as much of the developed portions of the northeastern United States.

The National Park Service established a Native Plant Demonstration Project carried out by MORR park staff and Rutgers University. Selection of an appropriate site began in November 2001. The original project plan extended through December 2005, but additional monitoring was done in the summer of 2008. The selected experiment site within MORR is located (lat 40.771663, long −74.543626) 56 meters west of Cemetery Road and 63 meters from the nearest pedestrian trail (Figure 3). This specific location was selected because:

1. It was representative of intact forest canopy within the dominant forest type at MORR, Northeastern Interior Dry-Mesic Oak Forest.
2. It contained extremely thick patches of the dominant invasive plant species including Japanese barberry and Japanese stiltgrass which represented a worse-case scenario of full understory invasion (Figure 1A). Density of Japanese barberry in this area was similar to or higher than adjacent areas within the Jockey Hollow section of MORR (Ehrenfeld 1982).
3. It was subject to high deer populations that influence vegetation composition, just as the rest of the park.
4. The site was neither subject to edge effects nor in the vicinity of active management such as mowed roadway edges and fields or near invasive species control areas.
5. The site has consistent terrain; it lacked variable slopes or obtrusive landforms.

Initial Vegetation Management

In April 2002, a 0.8-hectare area was brush-cut by volunteer staff from the Park and over the next 3.5 months, remaining slash and barberry roots and stumps were removed by project personnel and many volunteers (Figure 1B). Park staff used a brush method to herbicide cut barberry stumps with triclopyr (Garlon® 3A, Dow AgroSciences). In August 2002, 14 black locust (*Robinia pseudoacacia*) were logged from the site due to their ability to fix nitrogen and create nutrient-rich conditions that could facilitate further invasion. One invasive princess tree (*Paulownia tomentosa*) was also removed. On August 19, 2002 a 2.4m (8ft) deer exclosure was installed around the cleared area to prevent deer browse, a significant issue in the rest of the park.

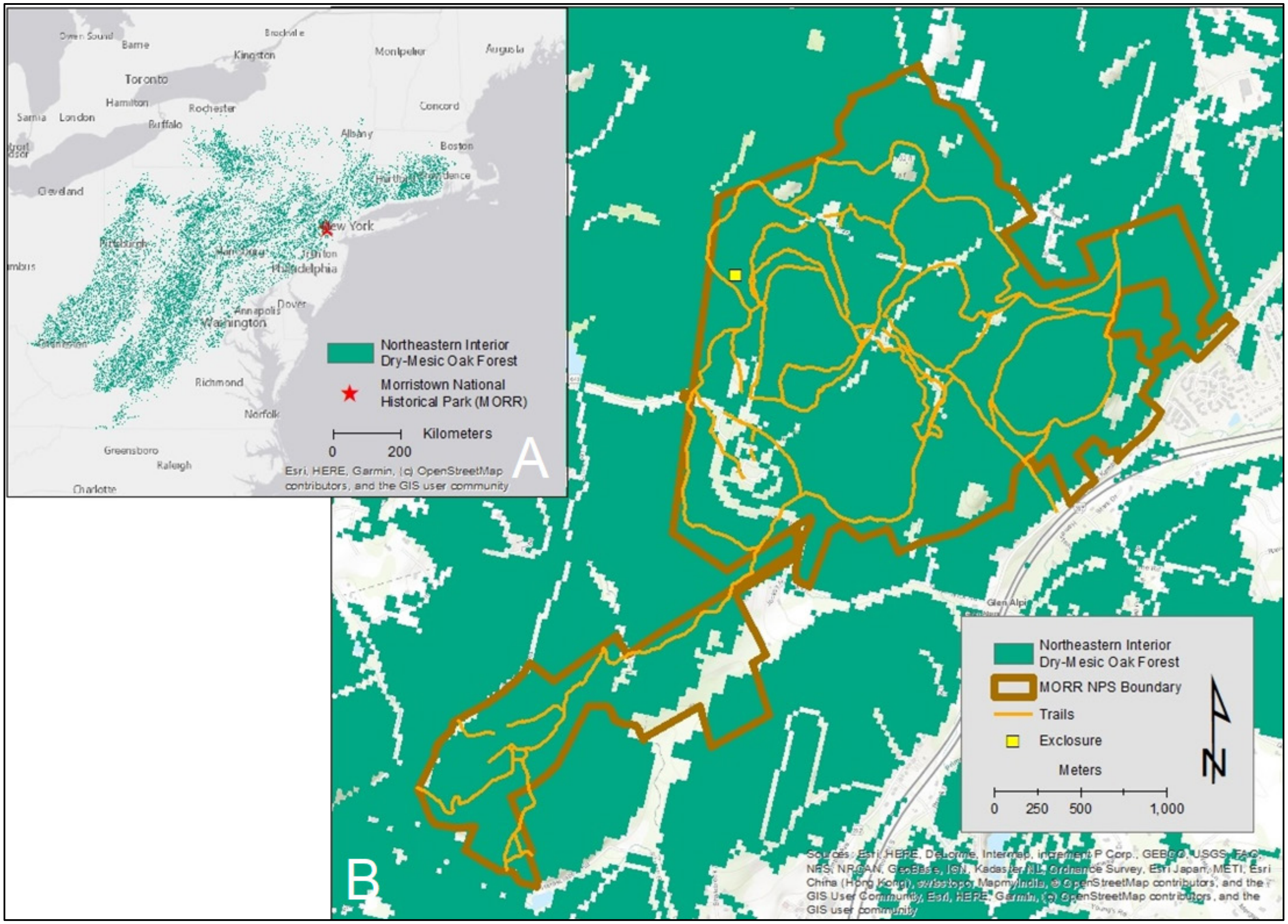

Figure 2. Map A: Range of the northern interior dry-mesic oak forest national vegetation classification (NatureServe 2020) and the location of Morristown National Historical Park represented by the red star. Map B: Morristown National Historical Park boundary, trails, and the exclosure location (yellow square) within the northern interior dry-mesic oak forest national vegetation classification.

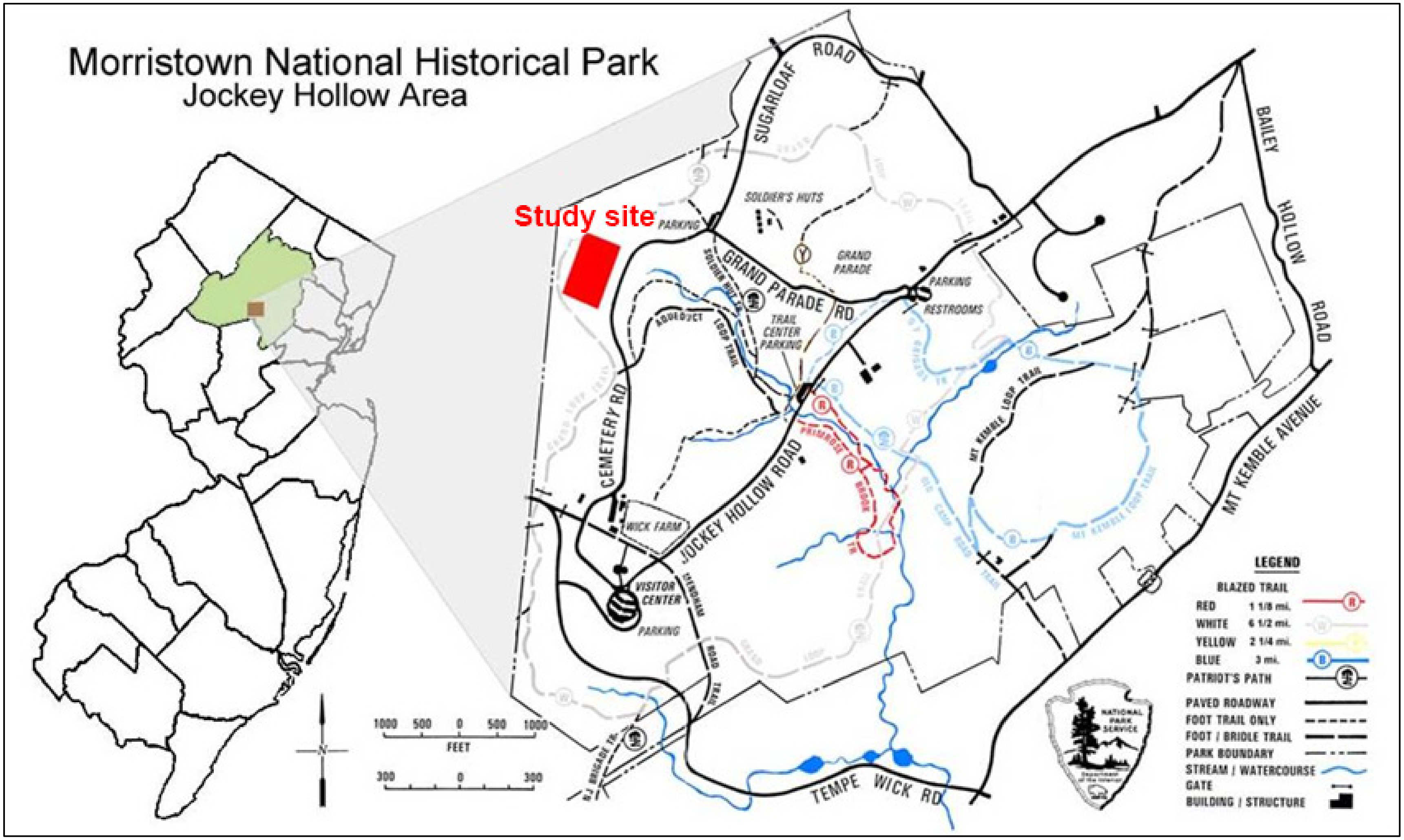

Figure 3. Study site (in red) location within the Jockey Hollow section of Morristown National Historical Park in Morris County, New Jersey. The study site contains the exclosure and control areas (image modified by K. Ross with permission from NPS).

Soil Sampling

To understand initial soil conditions and variability within the exclosure prior to establishing experimental soil quadrats, we collected 45 soil samples from the top 10cm bare soil (with leaf litter removed) using a 5cm diameter hand soil corer (Figure 4) in September 2002, from three randomly chosen 3 x 3m quadrats each 15m inside the perimeter of the fence. We subsampled each quadrat fifteen times and processed the soil for pH, percent moisture, percent organic matter (loss on ignition), inorganic N (NO_3^--N, NH_4^+-N using KCl extraction). A subset of these samples was sent to the Soil Testing Lab at Rutgers University for texture analysis. These analyses allowed us to calculate how many soil sampling quadrats to establish across the exclosure and the quantity of subsampling necessary to capture soil variability within quadrats.

Figure 4. Soil corer (i.e. bulb planter) used to sample top 10cm of soil in treatment quadrats. (KRISTEN A. ROSS).

In late fall 2002, sixteen 3 x 3m quadrats were established in a Latin square design (an experimental design that restricts randomization where each soil treatment is applied in each row and each column) for long-term sampling within the exclosure (Figure 5). Each quadrat was separated by 10m and no closer than 15m to fenced perimeter.

In early spring 2003, three initial soil samples were taken from the top 10 cm of bare soil in each of the sixteen soil quadrats (total of 48 samples) in the Latin square to determine pH, percent moisture, percent organic matter, and inorganic N (NO_3^--N, NH_4^+-N) prior to soil treatment application to the quadrats and native vegetation installation. We applied all soil treatments to the sixteen quadrats in early April 2003. The three different soil treatments were applied to determine whether soil conditions that might constrain vegetation development could be manipulated and if the treatments would affect subsequent plant colonization.

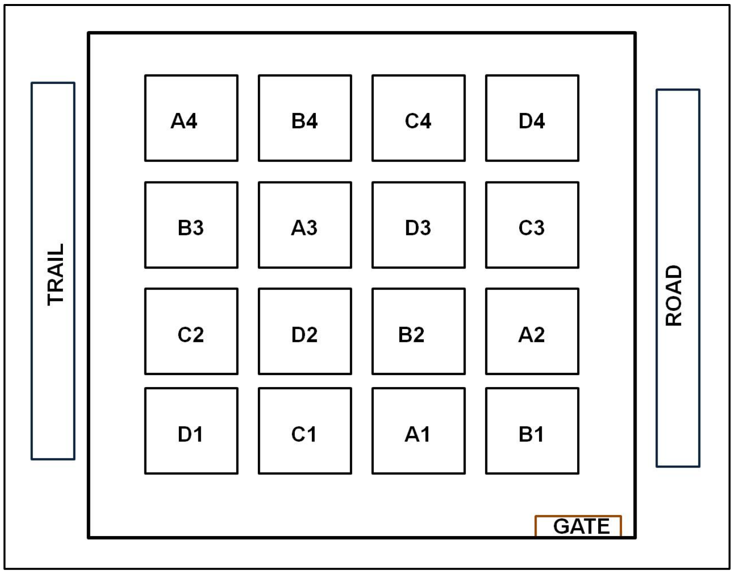

Figure 5. Soil manipulation schematic Latin square layout inside the exclosure. Each quadrat is 9m^2 and approximately 15m in from the fence perimeter and approximately 10m apart. Each letter represents a different soil treatment and each number is a replicate of that treatment. A=topsoil removal; B=woodchips added to quadrat; C=control quadrats; D=aluminum sulfate addition.

We applied the following treatments:

1. Treatment A: we cleared the top 5–10 centimeters of organic-rich soil from four of the sixteen soil quadrats in the first year of the experiment (2003). In preliminary observations, we found the top soil layer consisted mainly of earthworm castes.
2. Treatment B: we added 18–23 kilograms of hardwood mulch to four of the sixteen soil quadrats for each of the three years to immobilize nitrogen and increase soil C:N.
3. Treatment C: in the control quadrats we cleared the leaf litter and then replaced it.
4. Treatment D: we added 6.8 kilograms of aluminum sulfate to four of the sixteen soil quadrats to lower the pH.

All treatments were applied in the spring for three consecutive years except for the topsoil removal, treatment A. Three subsamples of soil from the top 10cm were collected using a 5cm diameter hand soil corer (Figure 4) from each quadrat for a total of 48 samples per sampling period in the spring, summer, and fall of 2003, 2004, and 2005 to track changes in soil properties. One final soil collection occurred fall 2008.

Soil Analysis

Collected samples were stored in closed plastic bags at about 5°C until analyzed. Initial pH readings were taken using a portable UP-5 meter in a 1:5 soil:distilled water slurry (Denver Instruments, Denver, CO). Each sample was stirred for five minutes, settled for ten minutes, and then read with the pH meter. Percent moisture was gravimetrically determined on fresh soil (1–2g) at 105°C for forty-eight hours. Loss on ignition (LOI) was determined at 500°C in a muffle furnace for a minimum of three hours. Soluble soil forms of inorganic N (NO_3^--N, NH_4^+-N) were extracted from

10g of fresh soil with 50 mL of a 2M KCl solution, shaken for one hour, filtered, and frozen at 4°C until analyzed. Extracts were analyzed on a Lachat QuikChem FIA+ (Lachat Instruments, Hach Co. Loveland, CO) for NO_3^--N and NH_4^+-N (QuikChem Systems 1986, 1987). We analyzed soil texture and C:N ratio for baseline understanding of soil properties. Statistical analysis of soil variables is described in Appendix B.

Vegetation Installation

Six species of native woody seedlings purchased from Pinelands Nursery (Columbus, NJ) and the New Jersey Forest Nursery (Jackson, NJ) were planted within the sixteen 3 x 3m (9m^2) soil quadrats to establish woody plant recovery after invasives species removal and to monitor survival after the first year of soil treatments. The following species were planted in the spring of 2003 in each soil quadrat: one witch hazel (*Hamamelis virginiana*); two spicebush (*Lindera benzoin*); two low sweet blueberry (*Vaccinium angustifolium*); one white oak (*Quercus alba*); one chestnut oak (*Quercus montana*); one northern red oak (*Quercus rubra*). Therefore, in each soil manipulation treatment there were eight spicebush planted, eight low sweet blueberry, and four of each of the other species: witch hazel; white oak; chestnut oak; and northern red oak. Survival of the planted seedlings was recorded in July 2004 and April 2005.

In addition, over 500 native shrubs and trees (same species as listed above) were planted all around the exclosure outside of the Latin square quadrats in spring 2003 with the help of more than 40 volunteers from local companies and school groups. As visible in Figure 1B after barberry was removed, there was little woody understory structure. The existing few sub-canopy species such as flowering dogwood (*Cornus florida*) and spicebush were substantially damaged by deer browse. These individuals planted through volunteer efforts were not surveyed for survival.

In summer 2003, it was apparent that Japanese stiltgrass had taken advantage of the open understory (Figure 1C). Hand pulling of stiltgrass was necessary throughout the exclosure to increase survivability of the planted woody vegetation (Figure 6). In July 2003, stiltgrass cover was recorded inside each soil treatment quadrat and then removed by hand.

In 2004, we censused woody seedlings planted within the soil treatment quadrats. Percent seedling survival was calculated on a per soil treatment quadrat basis based on the total number originally planted in 2003.

Figure 6. New Japanese stiltgrass (*Microstegium vimineum*) seedlings emerging in late spring 2003 (left); planted spicebush (*Lindera benzoin*) surrounded by Japanese stiltgrass which has been carefully hand-pulled in the area directly adjacent to the spicebush (right). (KRISTEN A. ROSS).

Vegetation Sampling

Vegetation surveys were conducted within the exclosure in June of 2003, 2004, and 2008 using stratified random quadrat sampling (Figure 7). Six transects with ten 3 x 3m ($9m^2$) quadrats (except in 2008 five to nine quadrats per transect were sampled) were randomly placed along each transect running perpendicular to Cemetery Road fence line. The transects were each begun 10m inside the fence perimeter then, using a random number table, the quadrats were surveyed at random distances between 1–9m apart along the transect for percent cover of all species present. Each transect was set about 15m apart to cover most of the exclosure. The transect closest to the gate was set 20m in as this area was the most highly impacted by foot traffic during restoration. We used a similar sampling regime in subsequent years (2004, 2008), but quadrats were not necessarily placed in the exact same place each year along each transect.

In 2004, similar stratified sampling was done outside of the exclosure using four transects running parallel to the fence with one transect placed on each side of the fence. Each transect was placed 10m away from the fence and contained between six to eight $9m^2$ quadrats placed 12m apart; there were twenty four quadrats outside the exclosure. In 2004, percent cover surveys were performed in these quadrats outside the exclosure, which served as a control comparison.

In 2003, we performed an inventory of all canopy trees inside the exclosure; we recorded species and diameter at breast height (DBH). In 2008, we assessed vertical structure of woody seedlings inside the exclosure by placing them into height classes categories (<0.25m to 2.0m in height). Vegetation was sampled in June of each year of the project.

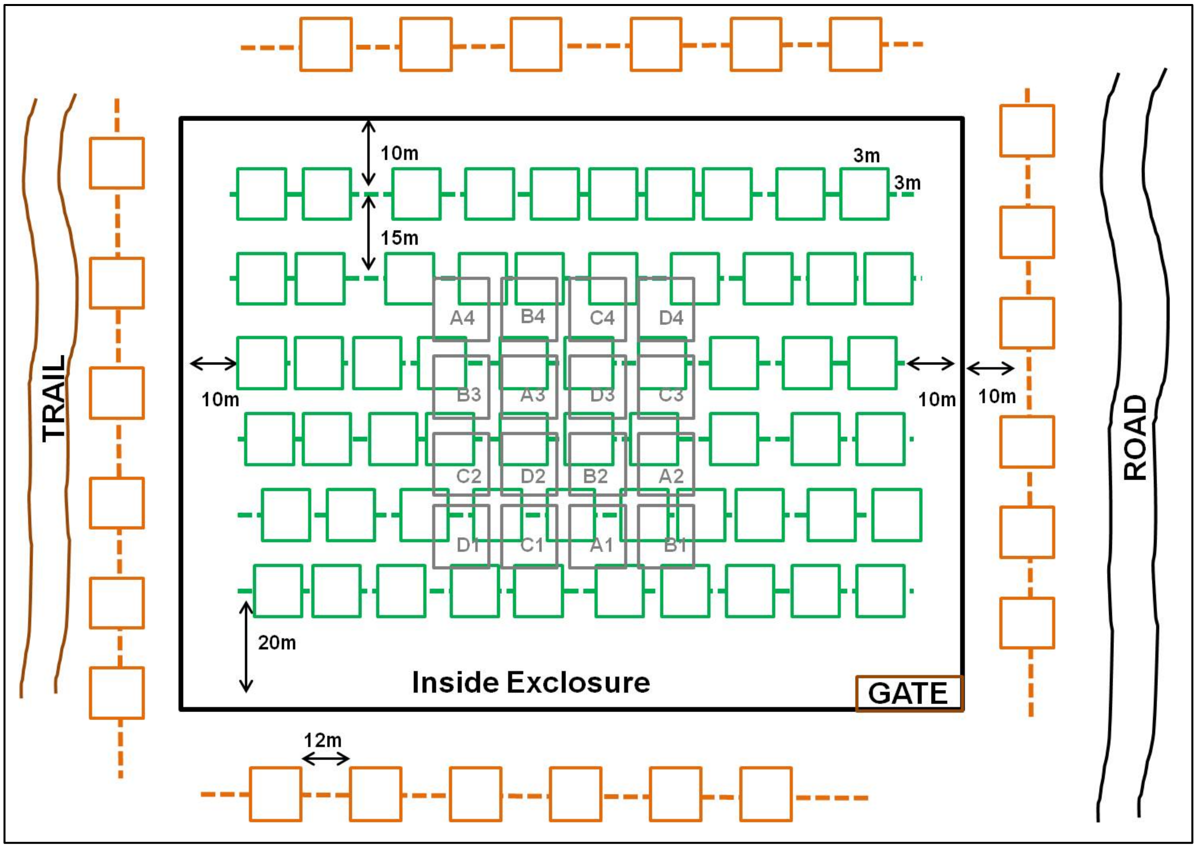

Figure 7. Schematic vegetation survey sampling design inside and outside of the exclosure. The black rectangle represents the 0.8ha exclosure fence. The green dotted lines represent six parallel transects 15m apart. The transects were established 10m from the Cemetery Road fence line each sampling year except along the fence line with the gate. The southern-most transect was placed 20m inside the southern fence line due to greater foot traffic near the gate. Ten 9m^2 percent cover survey quadrats represented by green squares were sampled along each transect at random distances (between 1–9m) from each other along the transect. Outside of the exclosure, one transect, shown in orange, was placed parallel to, but 10m away from, the fence line on each side of the fence to serve as an unfenced control comparison. The orange squares represent the twenty-four control 9m^2 vegetation survey quadrats sampled in 2004 outside the exclosure placed 12 m apart. This diagram is not to scale.

Vegetation Data Analysis

Species percent cover data were analyzed using Non-metric Multidimensional Scaling (NMDS) as it is considered to be the most robust unconstrained ordination method (Minchin 1987) and provides a "map" of the relative similarities of community components, in this case plant species cover based on rank order of the distances between each component (Clarke 1993). The stress coefficient indicates the level of agreement between the rank orders. NMDS is a commonly used method to depict similarities among community relationships (Clarke 1993). NMDS was performed using the metaMDS function in the package vegan based on 1000 permutations (Oksanen et al. 2010) in R (R Development Core Team 2010) using the default methods recommended by Minchin (1987). These defaults include the use of Bray-Curtis similarity distance matrix (Faith et al. 1987; Bray and Curtis 1957) with square root transformation because the data have a large range of values. This was followed by Wisconsin double standardization of plant cover values (Bray and Curtis 1957), which relativizes the species by their maxima, and the sites by their totals, giving equal weighting to all years and species. These default methods gave a readily interpretable result and so were retained. The goodness of fit of year and fenced and not fenced factors to the NMDS ordination scores were assessed by 1000 permutations of the environmental variables using the function envfit in the vegan package (Oksanen et al. 2010) for R (R Development Core Team 2010). The $r^2 = 1 - ss_w / ss_t$, where ss_w and ss_t are within-group and total sums of squares, respectively.

Species diversity

Average species richness across quadrats inside and outside of the exclosure was determined using an analysis of variance performed on these means and all pairwise comparisons were computed post hoc using Tukey's honestly significant difference (Sokal and Rohlf 2000). Beta diversity was computed as the average Sørenson distance (Pielou 1974) among quadrats within each treatment group. Shannon diversity and evenness (Pielou 1984) were computed as summary statistics for each treatment group.

Functional group diversity

Vegetation structure was represented as the fraction of total cover within a quadrat for each of three life forms or functional groups: herb, shrub, and tree. We assessed the diversity of functional groups using Pielou's (1984) diversity measure with the proportional abundances of the three functional groups.

Results

Soil Manipulations

Figure 8 summarizes the effects of four years of soil treatments on soil properties. Although soil was collected each year following treatment, the data presented here are from 2008. Analyses of all the data show that while there were significant changes over time (within years and between years), these changes were small, and are summarized by the net effect of the manipulations as represented in the 2008 data. The summary of soil treatment results in Figure 8 from the analysis of variance and Kruskal-Wallis tests (and the supporting statistical analyses in Appendix B) show that while the manipulations were effective in modifying pH ($F_{3,36}$=3.6, p=0.02), percent soil moisture ($F_{3,36}$=12.8, p<0.001), and percent organic matter ($F_{3,36}$=13.1, p<0.001), the manipulations had no effect on available inorganic nitrogen (N-NO_3, N-NH_4) concentrations (NO_3: H=1.85, df=3, p=0.60; NH_4: H=0.83, df=3, p=0.84). Changes seen in the moisture and organic matter were driven by the addition of mulch as predicted (Appendix B). Soil pH was also altered through the removal of the topsoil. No soil treatment, however, impacted the survival of the planted woody species.

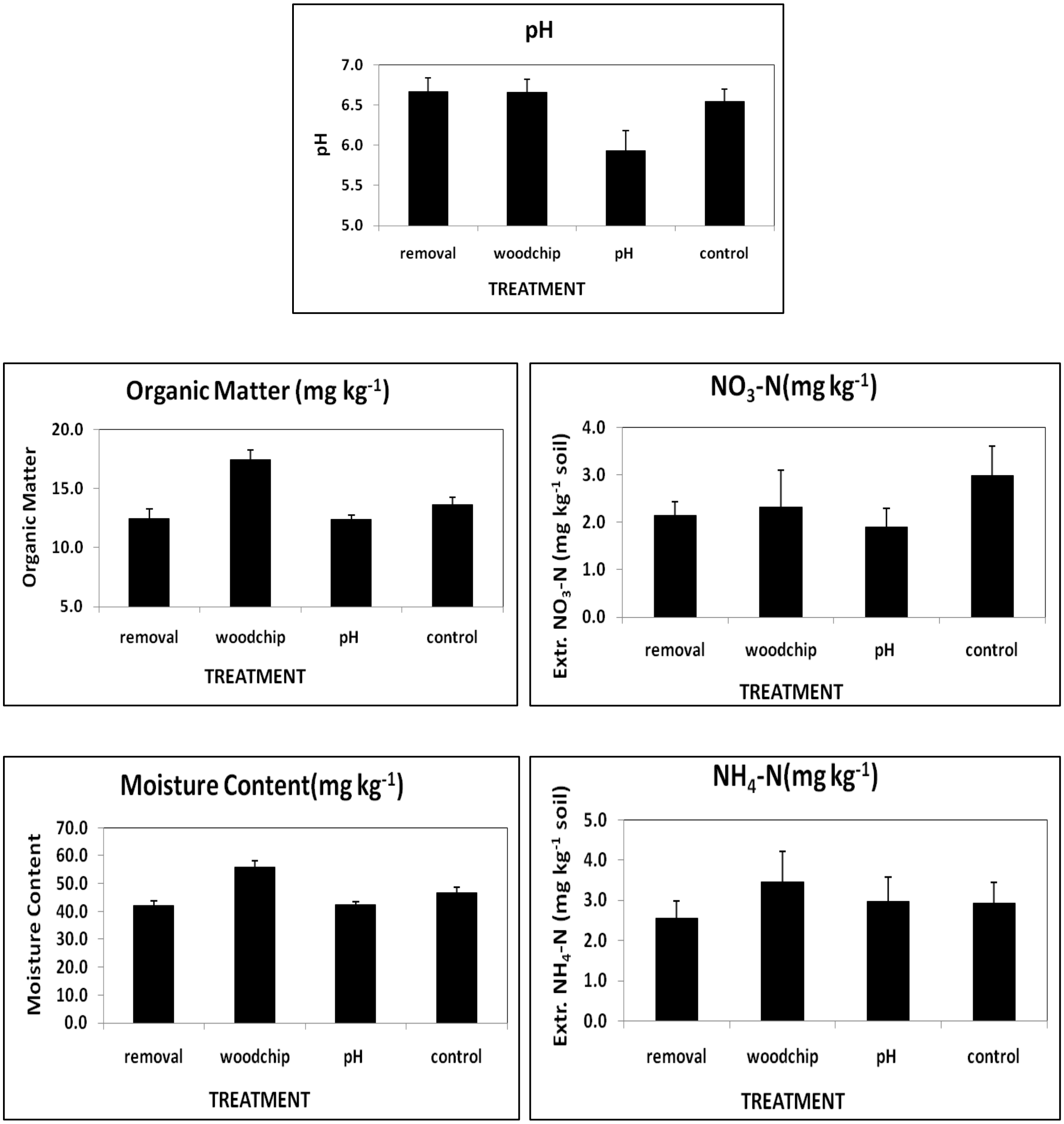

Figure 8. Summary of the effects of four years of soil manipulations in experimental quadrats. Data are from soil collected in 2008.

Survivorship of Vegetation Installation

No significant difference (df = 3, $F = 0.117$, $p = 0.9491$) was found in the mean number of surviving transplanted seedlings in any of the soil manipulation treatments. Overall, lowbush blueberry (*Vaccinium angustifolium*) and spicebush (*Lindera benzoin*) had the most surviving individuals one year after planting in the four soil treatments (Figure 9), however twice as many of these two species were planted in comparison to the others. We noticed that many of the oak seedlings had signs of small mammal herbivory.

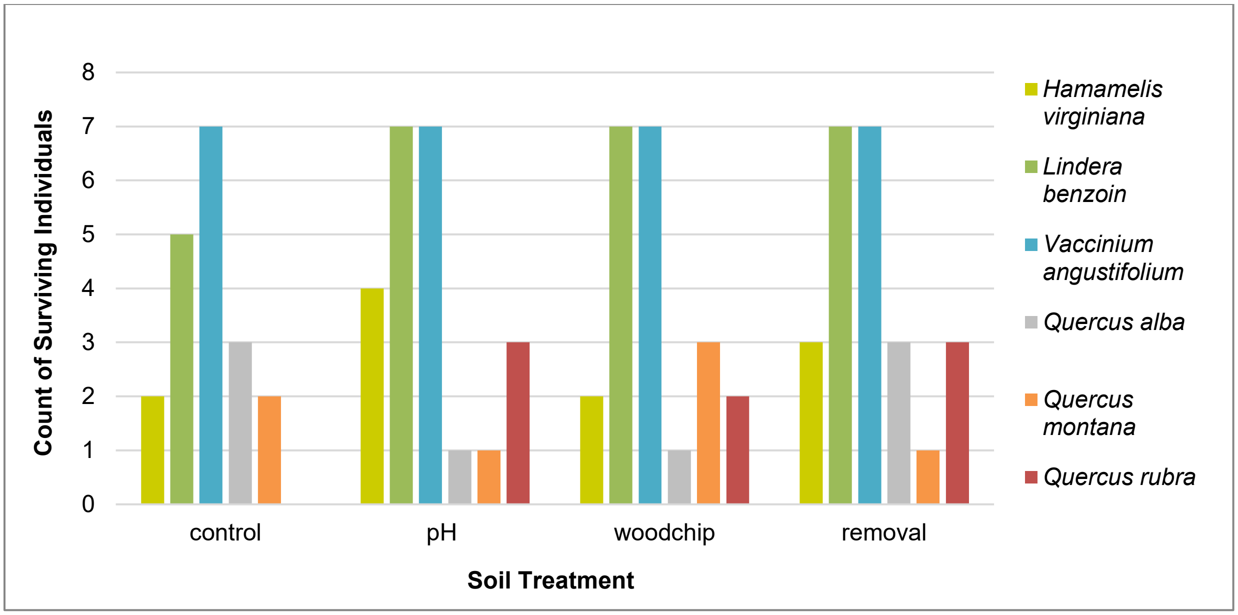

Figure 9. Survival per species one year after planting (2004) in each of the four soil manipulation quadrats: control; pH alteration where ammonium sulfate was added; woodchip addition; and removal of topsoil with earthworm castings. Eight individuals of the following six planted species were installed in each soil manipulation quadrat in 2003: one witch hazel (*Hamamelis virginiana*); two spicebush (*Lindera benzoin*); two lowbush blueberry (*Vaccinium angustifolium*); one white oak (*Quercus alba*); one chestnut oak (*Quercus montana*); one northern red oak (*Quercus rubra*).

Plant Occurrence

Table 1 lists the most commonly occurring species (percentage of all quadrats) observed inside the exclosure in 2003 (first year after clearing) and 2008, and outside the exclosure in 2004. Within one year after clearing, the area within the exclosure was nearly completely taken over by the two most abundant invasives, Japanese barberry (*Berberis thunbergii*) and Japanese stilt grass (*Microstegium vimineum*; Table 1), similar to the vegetation growing outside the exclosure, the control area. The remaining common species included invasives and generalist native species. Subtle differences in species composition are shown by the ordination results discussed below.

Table 1. Frequency (percentage of quadrats containing the species) of the twenty most frequently encountered species sampled within the exclosure and outside the exclosure (control).

Scientific Name	Common Name	Frequency 2003 Exclosure	Frequency 2008 Exclosure	Frequency 2004 Control
Microstegium vimineum [a]	Japanese stiltgrass	98.33	95.35	60.00
Berberis thunbergii [a]	Japanese barberry	93.33	90.70	70.00
Arisaema triphyllum	Jack-in-the-pulpit	78.33	37.21	62.50
Lonicera japonica [a]	Japanese honeysuckle	75.00	81.40	57.50
Allium vineale [a]	Wild garlic	58.33	0	0
Euonymus alatus [a]	Burning bush	58.33	18.61	20.00
Oxalis stricta	Wood sorrel	56.67	79.07	47.50
Celastrus orbiculatus [a]	Asiatic bittersweet	53.33	88.05	37.50
Ageratina altissima	White snakeroot	50.00	58.14	20.00
Rubus phoenicolasius [a]	Wineberry	43.33	76.74	47.50
Fraxinus americana	White ash	40.00	88.37	37.50
Alliaria petiolata [a]	Garlic mustard	38.33	30.23	32.50
Vitis spp.	Wild grape	38.33	44.19	30.00
Rosa multiflora [a]	Multiflora rose	18.33	23.26	37.50
Viburnum prunifolium	Blackhaw viburnum	18.33	(11.63) [b]	0 [b]
Viola sp.	Violet	18.33	27.91	12.50
Cardamine impatiens [a]	Narrowleaf bittercress	11.67	55.81	30.00
Toxicodendron radicans	Poison ivy	11.67	34.88	(10) [b]
Nyssa sylvatica	Black gum	10.00	(11.63) [b]	(7.5) [b]
Polystichum acrostichoides	Christmas fern	8.33	(7.00) [b]	0 [b]

[a] denotes an invasive species

[b] frequency in parentheses for species not among top 20 species in 2004 or 2008

Table 2 lists the mature trees, saplings, and shrubs that were present in the exclosure at the time of establishment (2003). The exclosure area canopy was highly dominated by mature tulip trees (*Liriodendron tulipifera*) which is typical of successional areas within the Northern Interior Dry-Mesic Oak Forest classification (NatureServe 2020). The next most common mature tree species encountered was black gum (*Nyssa sylvatica*) which is noted to occur in moist pockets within the forest classification. There were few sapling size trees (2.5–10cm DBH) of the species that could grow to reach the canopy in the future; this is most likely due to the high deer pressure and subsequent browse of seedlings pre-fence installation. The understory did maintain taller shrubs such as spicebush (*Lindera benzoin*) and blackhaw viburnum (*Viburnum prunifolium*) which had survived the deer browse. The black locust (*Robinia pseudoacacia*) and princess tree (*Paulownia tomentosa*), species not native to the area, were removed when the Japanese barberry were cleared.

Table 2. Composition of woody plants (trees and shrubs) greater than 2.5cm DBH growing in the exclosure at the time of establishment, 2003. DBH (diameter at breast height) is in centimeters. SE is standard error.

Scientific Name	Common Name	Mean DBH ± SE of all woodies	Counts by size class: 2.5–10cm DBH Saplings/shrubs	10–30cm DBH Mid-story	>30cm DBH Canopy
Acer rubrum	Red maple	19.3 ± 2.61	2	16	2
Betula lenta	Black birch	33.6 ± 2.29	0	9	12
Cornus florida [b]	Flowering dogwood	8.2 ± 0.69	10	4	0
Fraxinus americana	White ash	26.0 ± 7.85	2	2	3
Lindera benzoin [a]	Spicebush	7.1 ± 0.77	15	5	0
Liriodendron tulipifera	Tulip tree	65.6 ± 3.03	1	9	66
Populus grandidentata	Big-tooth aspen	25.4	0	1	0
Nyssa sylvatica	Black gum	18.2 ± 2.06	6	22	5
Quercus rubra	Northern red oak	30.5	0	0	1
Sassafras albidum	Sassafras	20.3	0	1	0
Ulmus sp.	Elm	24.2	0	1	0
Viburnum prunifolium [a]	Blackhaw viburnum	6.1 ± 0.72	17	1	0

[a] shrub species

[b] understory tree at maturity

Changes in Species Composition

In Figure 10, the non-metric multidimensional scaling ordination shows a clear separation of quadrats by year along axis 2 and between inside and outside the exclosure along axis 1. In the ordination diagram, numbers indicate the last digit of the year (2003, 2004, 2008) the site was sampled and a blue circle with a small “c” in front of the year number indicates the quadrat was a control (= outside the exclosure). Plant community composition inside the exclosure in 2003 was less similar to that in 2004 as those quadrats are clearly separated. Community composition outside the exclosure sampled in 2004 is more similar to that sampled in 2003 which was the first year after restoration. Community composition measured in 2008 is not very similar to either that in 2003 or 2004 separating along axis 2. Both year ($R^2 = 0.60$ $p<0.001$) and fenced/unfenced ($R^2 = 0.06$, $p<0.001$) are significant explanatory variables for the differences in vegetation composition among the groups (Final stress = 27%).

In the control area the total percent cover of the most abundant invasive plants was off the charts at 141%, half of which was Japanese barberry cover, while the most abundant native species totaled to less than 20% (Figure 11). From 2003 to 2008, within the exclosure quadrats, the total percent cover of the most abundant natives increased gradually by species, but tripled in native percent cover collectively. The most abundant invasive plants collectively covered 75% in 2003, and grew to almost 100% in 2008 mainly due to the resurgence in wineberry (*Rubus phoenicolasius*). Therefore, the exclosure installation and barberry removal allowed for the increase of native plant cover, but it did not impede the collective regrowth of invasive plants. Installation of the exclosure and the removal of Japanese barberry resulted in increased total species richness, driven by increased native species richness, while invasive species richness was unchanged (Figure 12). For all groupings, an analysis of variance found a significant difference among some group means. While exclosure quadrats from 2004 and 2008 had significantly higher mean total richness and native richness than the control quadrats or exclosure quadrats from 2003, the mean richness for invasives in the exclosure quadrats is not significantly different from the control.

Removal of invasive species had no apparent effect on diversity measures (Figure 13), despite observing a significant increase in mean species richness for all species and for native species in the latter years in the exclosure.

Changes in Functional Group Diversity

The exclosure and the removal of Japanese barberry had a significant effect on functional group diversity (Figure 12, df = 3, $F = 10.508$, $p < 0.001$). Vegetation structure was represented as the fraction of total cover for each of three basic life forms or functional groups (i.e. herb, shrub, and tree; Figure 14). Functional group diversity was significantly different in the treatment quadrats in comparison to the control quadrats. In 2003 and especially in 2004, herbs made up a greater portion of the community than in the quadrats outside of the exclosure (i.e. the control), but this shifted in 2008 where greater tree species abundance was observed.

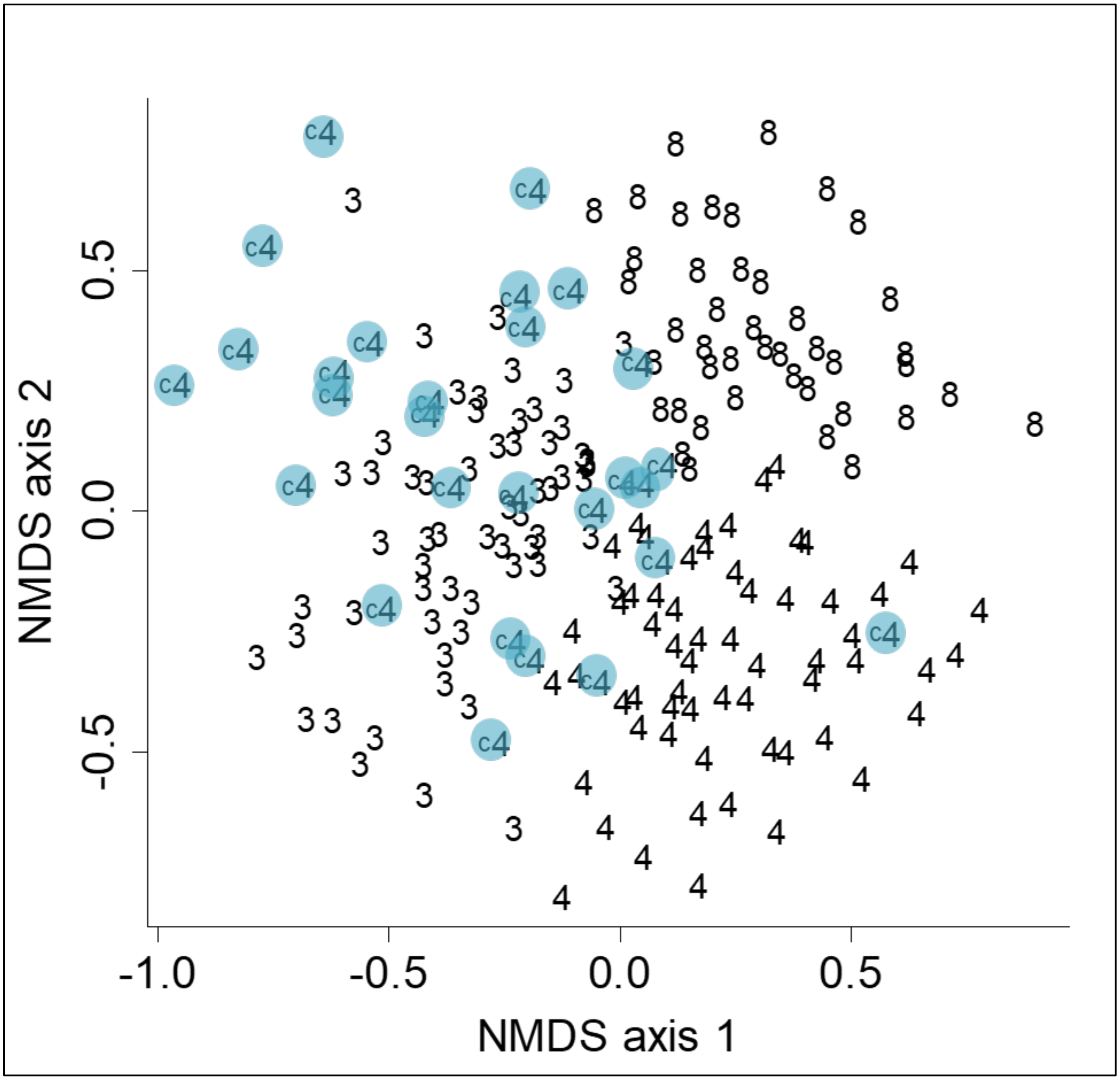

Figure 10. Non-metric multidimensional scaling ordination of vegetation survey quadrats. Numbers indicate the last digit of the year (2003–2008) the study site was sampled; a blue circle and a small “c” in front of the year number indicates the quadrat was a control (outside the exclosure).

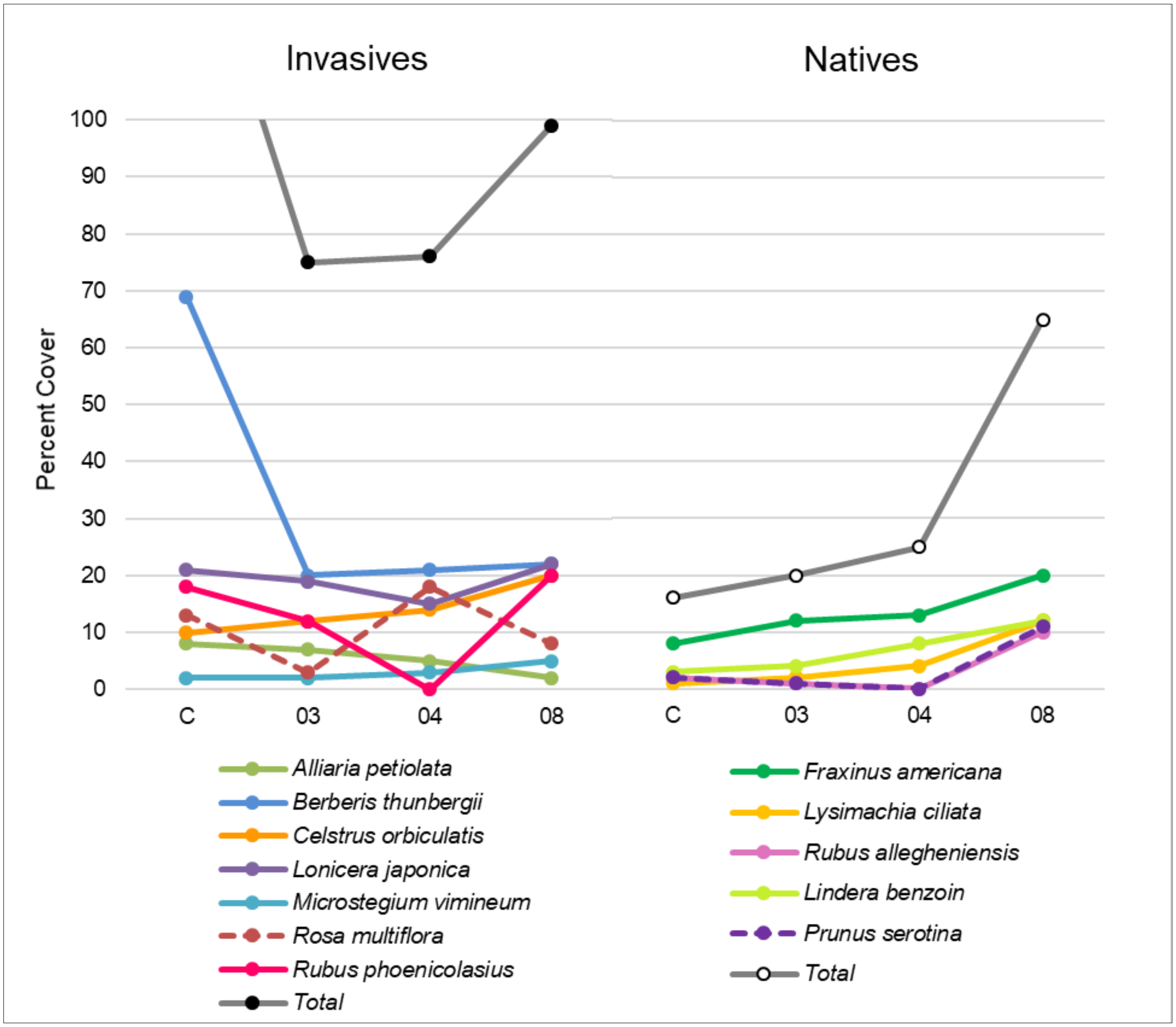

Figure 11. Changes in the abundances of the most abundant invasive and native species in the control quadrats (C) and exclosure quadrats each vegetation survey year (2003, 2004, and 2008). In the control, the total invasive cover was off the chart at 141%. Invasive species include: Japanese barberry (*Berberis thunbergii*); asiatic bittersweet (*Celastrus orbiculatus*); Japanese stiltgrass (*Microstegium vimineum*); Japanese honeysuckle (*Lonicera japonica*); wineberry (*Rubus phoenicolasius*); multiflora rose (*Rosa multiflora*); and garlic mustard (*Alliaria petiolata*). Natives species include: white ash (*Fraxinus americana*); fringed loosestrife (*Lysimachia ciliata*); allegheny blackberry (*Rubus allegheniensis*); spicebush (*Lindera benzoin*); black cherry (*Prunus serotina*).

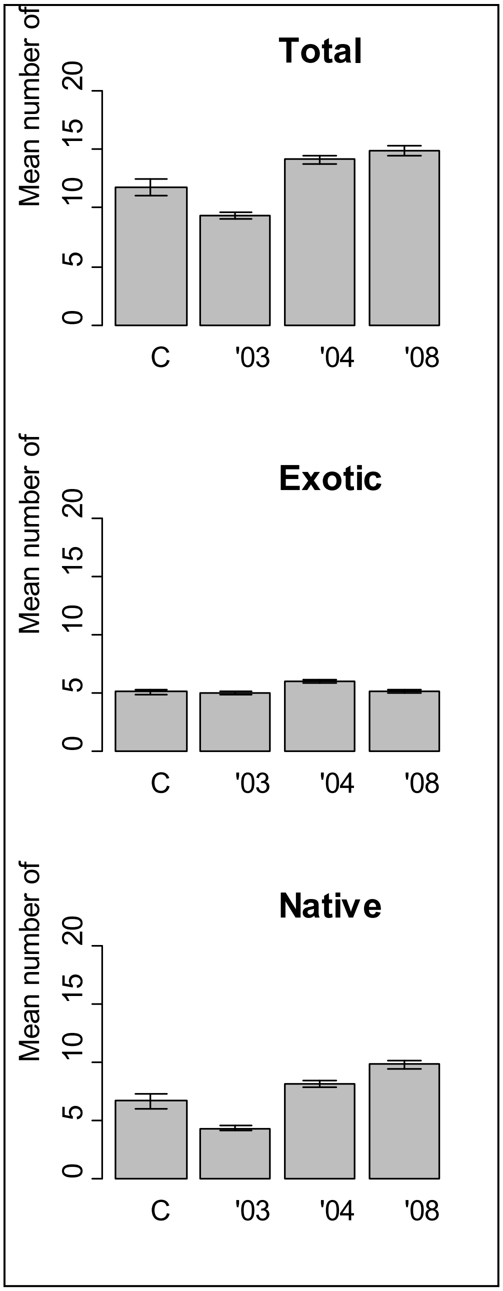

Figure 12. Effects of removal of invasives and exclosure on species richness in comparison to the control area. Columns are labelled as C for the control area (outside the exclosure), and the years 2003, 2004, and 2008. Error bars denote standard error.

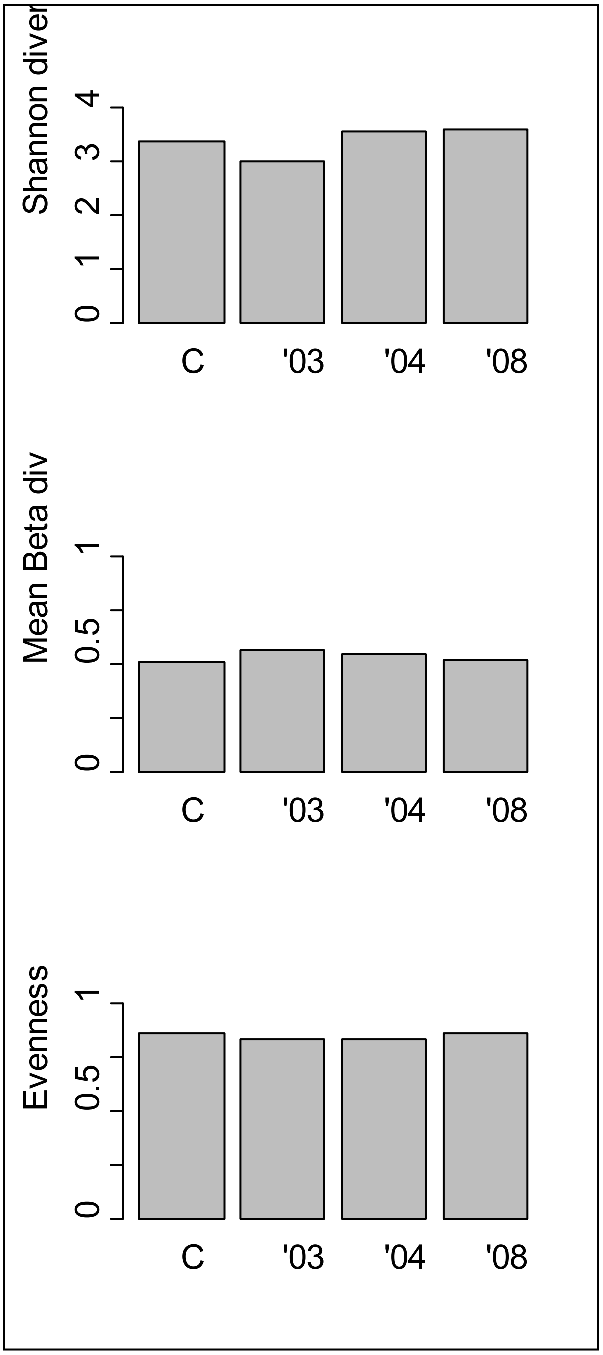

Figure 13. Effects of removal of invasive species on diversity measures in the exclosure. Columns are labelled as C for quadrats outside the exclosure (control area), and the years 2003, 2004, and 2008 sampled inside the exclosure.

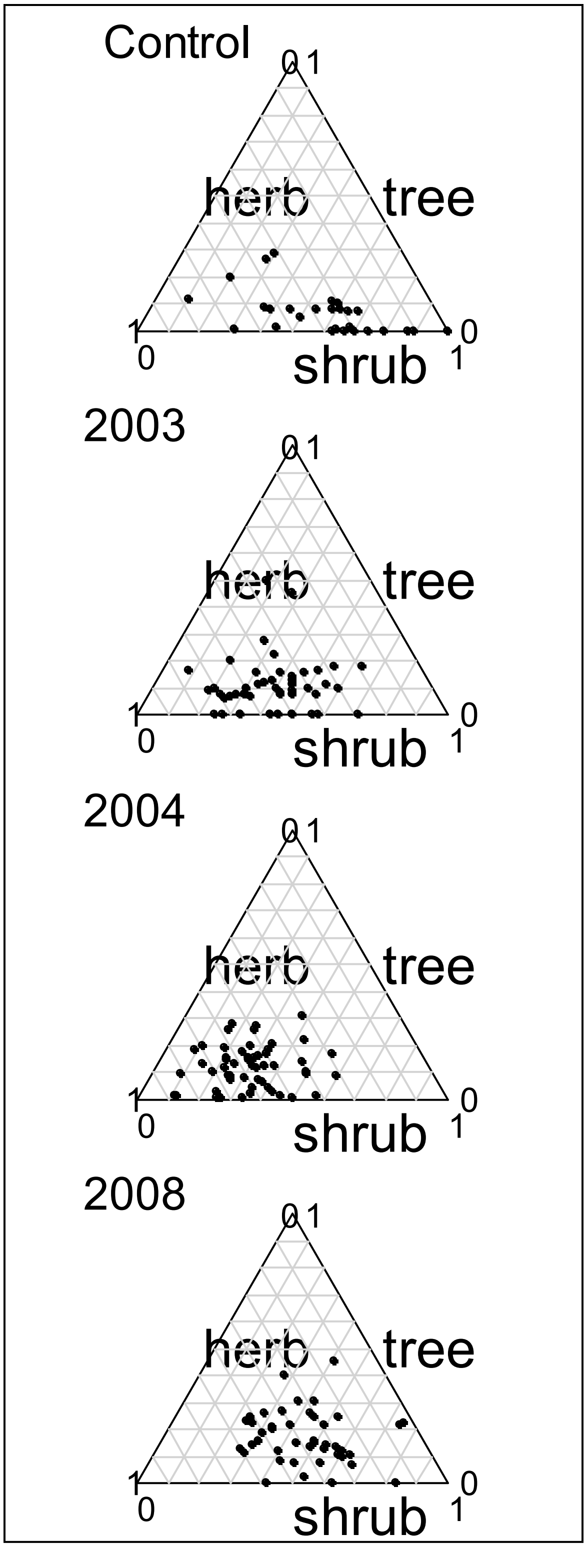

Figure 14. Functional groups represented as the fraction of total cover for each of three basic life forms in the exclosure in years after invasive plant management compared to the control area (outside the exclosure).

Notably, mapleleaf viburnum (*Viburnum acerifolium*) is missing from all the species lists. This plant was a characteristic understory plant of the Jockey Hollow upland forests prior to the increase in deer and the spread of Japanese barberry (Ehrenfeld 1977). However, we did not observe any individuals of this species within the exclosure, either in the sampled quadrats or by observation between the transects. Mapleleaf viburnum has been found in other exclosures in the greater region (Abrams and Johnson 2012; Ward et al. 2018). Since we did not observe this species colonizing the exclosure, we surmise that the length of time that this part of MORR has been overbrowsed has eliminated either seed banks or nearby seed sources for mapleleaf viburnum.

Discussion

We sought to answer three main questions, as follows:

1. What are the trajectories of abundance of both individual invasive and dominant native species over time, following the invasive removal and fencing? It is clear that despite their initial removal, the dominant invasives returned very rapidly without continuous management (e.g., removal, control, substitution with native species). While there were significant rearrangements of the plant communities, as shown by the ordination results, the overall pattern was that the invasives maintained dominance both within the exclosure and outside it. Notably, the vegetation outside the exclosure contained a similar array of native species (most were generalists or commonly found in pioneering woodland locations) to those that colonized the exclosure following the removal of the barberry. Overall, removal of the barberry made room for a typical suite of both invasive species and generalist native species.

2. What are the trajectories over time of species groups (all invasives, all herbs, shrubs, tree seedlings)? The trajectory diagrams showed that there were rearrangements, with an increasing emphasis on woody plants over time. However, this movement towards more woody plants consisted of both invasive and native generalist plants. There was little evidence that mid- to late successional native woody trees, such as oaks, hickories, maples, or beeches were colonizing the exclosure at densities sufficient to establish a canopy. Curiously, the dominant canopy tree throughout the site was tulip tree (*Liriodendron tulipifera*); however, we did not find any natural recruitment of this species within the exclosure.

   Other native tree and shrub seedlings recruited naturally, however, they were present at densities too low to establish a next generation forest with the exception of white ash (*Fraxinus americana*), the only abundant tree seedling and sapling observed in the exclosure. Exceptionally high numbers of ash seedlings in comparison to other species have been observed in other studies within Morristown National Historical Park (Epiphan and Handel 2017, unpublished data). The composition of seedling recruitment could be the response of a collection of site influences: historic clear cutting and agricultural soil disturbance (Ehrenfeld 1977), earthworm infestation (Kourtev et al. 1999), and Japanese barberry litter accumulation, which increases pH (Ehrenfeld et al. 2001; Kourtev et al. 2003). These factors create a scenario which white ash may tolerate as it responds well to elevated levels of nitrogen and its optimal soil pH range exceeds neutral (Schlesinger 1990; Robin-Abbott and Pardo 2017). In contrast, the majority of the other tree cohort species found in the Northeastern Interior Dry-Mesic Oak Forest, do not respond well to increased nitrogen or soil pH above 6.5 or 7. However, the rapid spread of the emerald ash borer beetle (*Agrilus planipennis*) may overwhelm these forces and ash may disappear.

   Therefore, the expected trajectory of the vegetation at this site is a native forest with high dominance of generalist species, relatively low diversity of native woody plants, but an increasing abundance of these native plants compared to the invasives. Since change in species composition was relatively small between 2003, when the invasives were removed, and the five year period of sampling to 2008, we anticipate the trajectory of the forest within the exclosure will continue to change very slowly as the native vegetation becomes more established. If the exclosure remains intact, one possibility is that invasive cover will decrease over time as native woody plants increase richness, cover, and develop shadier understory conditions. However, if white ash remains as the dominant tree regenerating, it will not provide a dense enough canopy to limit sunlight for pioneer invasive species. White ash casts light shade but is also highly susceptible to premature mortality from infestation of the invasive emerald ash borer which is present throughout MORR (Robert Masson, personal communication, 5/14/20). Therefore, as the canopy is projected to cast light shade or facilitate canopy gaps, the invasives are likely to remain within the exclosure.

3. Can soil conditions be manipulated to limit invasive species growth and enhance native woody seedling growth? No significant differences were found in the mean number of surviving transplants among soil manipulation treatments. The findings suggest that the removal of invasive species may not have been mediated by changes in soil factors as originally hypothesized. No other differences in plant communities among the treatment quadrats were observed, suggesting that tested soil factors do not have a strong effect on the facilitating establishment of transplanted plants to the area. However, it is still unknown whether long term native woody seedling growth is enhanced by the initial soil treatment effects.

These results based on the monitoring of an intensive forest recovery demonstration after five years of growth can be seen as the beginning of our understanding of management needs to return the forest to a more historic floristic state. Long-term monitoring and continual surveillance and repair of the exclosure will be needed until that time when the deer population declines dramatically. This may happen with changing of rules to permit deer culling, natural collapse of the deer population due to disease, or new genetics-based population control measures, such as CRISPR technology (Shope et al. 1960; MNHP 2017; Moro et al. 2018).

The descriptive statistics reported here are a contribution to a wider understanding of plant community change as stressors to the native plants are eliminated or decreased. The senior author, J.G. Ehrenfeld, suggested three additional lines of inquiry for this forest and for other regional stands having similar insults to the vegetation structure. First, at what rate does community composition change? This “composition” could be defined in several ways such as the species richness, the equitability, or the proportion of native to on-native biomass. Management decisions would be different dependent on the level of concern for any of these ecological options.

Second, how rapidly does the restored vegetation depart in structure from the unmanaged, invasive plant infested stands? We do not know yet whether the types of restoration action reported here result in a long-term improvement to forest quality. Some regional programs, such as in the City of New York’s public parks, show that removal of invasives followed by addition of local canopy species will persist for at least twenty years (Johnson and Handel 2016). Local conditions may change this optimistic finding, but more long-term studies are needed to gather a consensus of what a park manager can expect.

Third, does restoration of a native community result from some combination of natural mechanisms, as mentioned above, without active management of remnant invasive populations? The ability of local forests to be resilient, in the sense of returning to their historic state despite the onslaught of new invasive plants and insects, seems doubtful. The loss of chestnut, *Castanea dentata*, from eastern forests is the well-known example of a major tree that was attacked by an invasive pest and has not recovered in over a century (Loo 2008). The trajectory of forests hit by other invasive pests is generally grim, and new invaders keep coming (such as emerald ash borer and emerging invasive plants including *Viburnum sieboldii* and *Aralia elata*). Passive management, letting current problems unfold without proactive efforts by managers, has very little support, but some invasive species may fade with time.

The study done here is one prong of what must be an ongoing comprehensive effort to save biodiversity and mitigate against the "homogenization" of habitats by non-native, invasive species in eastern forest stands (McKinney and Lockwood 2001; Rooney et al. 2004).

Towards understanding local long-term trajectories, we revisited the exclosure site in May 2020 to gather general observations. Notably, the fence was compromised in a couple locations from storm damage which allowed deer to re-enter; deer browse was evident and tracks were observed in the soil. There was some storm damage and a large tipped-up tulip tree that created new openings in the canopy. The canopy gaps may have helped facilitate understory growth. In particular, several species of woody invasive plants were proliferating successfully (Table A-1). The native understory assembly included a variety of ferns, sedges, woody seedlings, saplings, and large shrubs. However, the deer have stunted the growth of the native woody species between 30–150cm in height. The deer pressure now inside the exclosure will decrease the amount of future tree saplings as the seedlings cannot reach their next growth stage when subject to browse. A glimpse of hope was found in the few oaks planted in 2004 (chestnut oak and northern red oak) that were observed persisting (Figures 15 & 16).

Figure 15. Chestnut oak (*Quercus montana*) seedling with basal sprouts 30cm in height, still persisting 17 years after planted as part of the vegetation installation in 2003. Note the old orange flagging tape at the base of the stem and the original main leader, found dead and fallen over. This photo was taken on May 14, 2020, when the leaves were still emerging. (JEAN N. EPIPHAN).

Figure 16. Northern red oak (*Quercus rubra*) sapling over 3 meters in height, observed in 2020, that has escaped deer browse and is presumably one of the planted individuals from the vegetation installation in 2003. This photo was taken on May 14, 2020. (JEAN N. EPIPHAN).

Overall, in 2020, there was a distinct observational difference between the outside area and inside the exclosure, even though both now are subject to deer herbivory. Outside of the exclosure, the Japanese barberry dominates the understory as dense, expansive colonies while remnant native plant occurrences are sparse (Figure 17). In comparison, the barberry inside the exclosure is much shorter, many other invasive species have taken off, and many native species are still present (Table A-1).

Figure 17. Observational comparison in 2020 of vegetation outside the exclosure (A) and vegetation inside the exclosure (B). Photos were taken on May 14, 2020. (JEAN N. EPIPHAN).

Conclusions and Management Implications

This study found that while soil legacy effects may be modifiable with manipulations, they do not appear to affect the short-term trajectory of restoration. Restoration of native vegetation is happening, albeit slowly, within the exclosure, and this is primarily based on the establishment of native species that can capitalize on space and light available after invasive shrub removal. These conditions allow for a large component of invasive species to persist within the restored area. Although the data show that there is an increase in diversity due to native species, and that the vegetation is moving towards a more tree-dominated structure, eventual restoration of native forest, without further intervention to control the enormous deer population will be a slow process. Addition of native species by planting can speed local biodiversity improvement as this action supplements natural seed dispersal which may be quite slow in this degraded woodlands.

In summary, removal of invasives without further vegetation management allows other invasive species to become established together with more native species. However, native woody and herbaceous plants do become established, suggesting that slowly, a more native vegetation will re-establish in areas protected from deer browse. Soil manipulations are effective in altering soil properties, but these changes have little apparent effect on plant establishment in the first few years after treatment.

Implications for Practice

Removal of the invasive dominant plant, *Berberis thunbergii*, with initial deer exclusion significantly increased native species richness over time, despite an initial drop, but invasive species richness remained the same.

The treatments had little effect on the diversity of *species*, but increased the diversity of *life forms* over time, resulting in a more even representation of herbs, shrubs, and trees.

The quick appearance of native species suggests that priority effect may be a more important control on community composition than soil legacy effects. This is important because the manipulation of the abundance of adult plants is easier and more cost effective than the manipulation of soil chemistry.

Actions taken against invasive species using adequate initial resources may result in forest restoration benefits even if the invasive species are not eliminated from the system. However, park staff must galvanize local volunteer and friends of the park groups to regularly and continually participate in species removal and planting activities for these improvements to persist. A fundamental and necessary action is the control of the deer population which is the major stress challenging long-term improvements in many National Park Service properties in the region. Policy and operational actions have been studied in detail (MNHP 2017). Without a solution to the deer herbivory pressure, all other actions may never result in a sustainable biodiversity improvement.

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Appendix A. Vegetation records in the 0.8ha exclosure and outside in the control quadrats.

Table A-1. Presence of vegetation by species inside the exclosure in 2003, 2004, outside the exclosure in 2004, and inside the exclosure in 2020. P denotes present; “–” denotes absence. In 2003 and 2004, the records are from quantitative surveys. In 2020, an informal site visit was performed for qualitative observations; a quantitative survey was not performed, therefore, the “not observed” species are not definitively absent or present.

Species name	Common name	2003 exclosure	2004 exclosure	2004 outside control	2020 inside exclosure observations
Abutilon theophrasti [a]	Indian mallow	–	P	–	Not observed
Acalypha rhomboidea	Three-seeded mercury	–	P	–	Not observed
Acer platanoides [a]	Norway maple	–	P	–	Not observed
Acer rubrum	Red maple	P	P	P	Few mature trees
Acer saccharum	Sugar maple	P	P	P	Few mature trees
Achillea millifolium	Yarrow	–	P	–	Not observed
Ageratina altissima	White snakeroot	P	P	P	Scattered clusters
Agrostis spp. (2 species)	Bentgrasses	P	P	P	Not observed
Alliaria petiolata [a]	Garlic mustard	P	P	P	Scattered clusters
Allium vineale	Wild garlic	P	–	–	Not observed
Amphicarpaea bracteata	Hog peanut	–	P	–	Not observed
Apocynum cannabinum	Dogbane	P	P	P	Not observed
Arisaema triphyllum	Jack-in-the-pulpit	P	P	P	Scattered throughout

[a] denotes an invasive species

[b] denotes species presence only because it was planted in the exclosure

Table A-1 (continued). Presence of vegetation by species inside the exclosure in 2003, 2004, outside the exclosure in 2004, and inside the exclosure in 2020. P denotes present; "–" denotes absence. In 2003 and 2004, the records are from quantitative surveys. In 2020, an informal site visit was performed for qualitative observations; a quantitative survey was not performed, therefore, the "not observed" species are not definitively absent or present.

Species name	Common name	2003 exclosure	2004 exclosure	2004 outside control	2020 inside exclosure observations
Berberis thunbergii [a]	Japanese barberry	P	P	P	Prolific; height is much less than barberry outside the fence
Betula lenta	Black birch	P	P	P	Few mature trees
Botrypus virginianus	Rattlesnake fern	–	–	–	Three small clusters found
Cardamine impatiens [a]	Narrowleaf bittercress	–	P	P	Scattered
Carex spp. (5 species)	Sedges	P	P	P	Scattered
Carya glabra	Pignut hickory	–	P	–	Not observed
Carya ovata	Shagbark hickory	–	P	–	Not observed
Celastrus orbiculatus [a]	Asiatic bittersweet	P	P	P	Throughout; climbing up saplings & shrubs
Circaea lutetiana	Enchanter's nightshade	P	P	P	Scattered
Cirsium vulgare	Bull thistle	–	P	–	Not observed
Cornus florida	Flowering dogwood	P	P	–	Few seedlings & trees
Cyperus sp.	Flat sedge	P	–	–	Not observed
Dennstaedtia punctilobula	Hay-scented fern	P	P	–	Scattered clusters
Desmodium sp.	Trefoil	–	P	–	Not observed
Dioscorea villosa	Wild yam	–	P	–	Not observed
Dryopteris carthusiana	Spinulose wood fern	–	–	P	Very few individuals

[a] denotes an invasive species

[b] denotes species presence only because it was planted in the exclosure

Table A-1 (continued). Presence of vegetation by species inside the exclosure in 2003, 2004, outside the exclosure in 2004, and inside the exclosure in 2020. P denotes present; "–" denotes absence. In 2003 and 2004, the records are from quantitative surveys. In 2020, an informal site visit was performed for qualitative observations; a quantitative survey was not performed, therefore, the "not observed" species are not definitively absent or present.

Species name	Common name	2003 exclosure	2004 exclosure	2004 outside control	2020 inside exclosure observations
Euonymus alatus [a]	Burning bush	P	P	P	Scattered clusters of tall shrubs and seedlings
Fagus grandifolia	American beech	P	–	P	Scattered seedlings
Fallopia japonica	Japanese knotweed	–	P	–	Not observed
Fraxinus americana	White ash	P	P	P	Scattered seedlings
Galium asprellum	Rough bedstraw	–	P	P	Scattered clusters
Galium circaezans	Wild licorice	–	P	P	Not observed
Galium triflorum	Sweet-scented bedstraw	–	P	–	Not observed
Geum canadense	Smooth avens	–	P	P	Not observed
Geum laciniatum	Rough avens	–	P	–	Not observed
Hackelia virginiana	Stickseed	P	P	P	Not observed
Hamamelis virginiana [b]	Witch hazel	–	–	–	–
Hypericum punctatum	Spotted St.John's wort	–	P	–	Not observed
Ilex verticillata	Winterberry holly	–	P	–	Not observed
Juniperus virginiana	Eastern red cedar	–	–	P	Not observed
Lactuca sp.	Wild lettuce	–	P	–	Few
Ligustrum vulgare [a]	Common privet	P	P	P	scattered
Lindera benzoin	Spicebush	P	P	P	Large clusters scattered throughout

[a] denotes an invasive species

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Table A-1 (continued). Presence of vegetation by species inside the exclosure in 2003, 2004, outside the exclosure in 2004, and inside the exclosure in 2020. P denotes present; "–" denotes absence. In 2003 and 2004, the records are from quantitative surveys. In 2020, an informal site visit was performed for qualitative observations; a quantitative survey was not performed, therefore, the "not observed" species are not definitively absent or present.

Species name	Common name	2003 exclosure	2004 exclosure	2004 outside control	2020 inside exclosure observations
Liriodendron tulipifera	Tulip tree	P	P	P	Dominant canopy tree, few seedlings and saplings
Lonicera japonica [a]	Japanese honeysuckle	P	P	P	Throughout
Lysimachia ciliata	Fringed loosestrife	–	P	–	Not observed
Maianthemum canadense	Canada mayflower	–	P	–	few
Maianthemum racemosum	False Solomonseal	P	P	P	One observed
Medeola virginiana	Indian cucumber	–	P	P	Not observed
Microstegium vimineum [a]	Japanese stiltgrass	P	P	P	Few clusters emerging
Mitchella repens	Partridgeberry	–	P	–	Few patches near other native vegetation
Monotropa uniflora	Indian pipe	–	–	P	Not observed
Nyssa sylvatica	Black gum	P	P	P	Clusters of saplings and trees
Onoclea sensibilis	Sensitive fern	P	P	–	Scattered; more near native vegetation
Osmunda cinnamomea	Cinnamon fern	P	P	P	One cluster found
Oxalis stricta	Wood sorrel	P	P	P	Not observed
Parthenocissus quinquefolia	Virginia creeper	P	P	P	Scattered throughout
Paulownia tomentosa	Princess tree	P	P	–	One tree
Pilea pumila	Canada clearweed	–	P	P	Not observed

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Table A-1 (continued). Presence of vegetation by species inside the exclosure in 2003, 2004, outside the exclosure in 2004, and inside the exclosure in 2020. P denotes present; "–" denotes absence. In 2003 and 2004, the records are from quantitative surveys. In 2020, an informal site visit was performed for qualitative observations; a quantitative survey was not performed, therefore, the "not observed" species are not definitively absent or present.

Species name	Common name	2003 exclosure	2004 exclosure	2004 outside control	2020 inside exclosure observations
Photinia villosa	Christmas berry	–	P	–	Scattered throughout
Photinia sp.	Unknown photinia	–	P	–	Growing colonies spreading
Phryma lepstachya	Lopseed	–	P	–	Not observed
Plantago sp.	Plantain	P	P	P	Not observed
Polygonum virginianum	Jumpseed	–	P	–	Few scattered plants
Polygonum sp.	Smartweed	P	P	P	Not observed
Polystichum acrostichoides	Christmas fern	P	P	P	Scattered clusters near other ferns
Populus grandidentata	Bigtooth aspen	P	P	–	Not observed
Potentilla simplex	Common cinquefoil	P	–	P	Not observed
Prunella vulgaris	–	–	P	–	Not observed
Prunus avium	Bird cherry	–	P	–	Few trees, saplings and seedlings in clusters
Prunus serotina	Black cherry	P	P	P	Scattered seedlings and saplings
Quercus montana	Chestnut oak	P	P	–	Few seedlings found
Quercus rubra	Northern red oak	P	P	–	Few seedlings and one sapling found
Quercus veluntina	Black oak	–	P	–	Not observed
Ranunculus sp.	Buttercup	P	P	–	Not observed
Robinia pseudoacacia [a]	Black locust	P	P	–	Few seedlings

[a] denotes an invasive species

[b] denotes species presence only because it was planted in the exclosure

Table A-1 (continued). Presence of vegetation by species inside the exclosure in 2003, 2004, outside the exclosure in 2004, and inside the exclosure in 2020. P denotes present; "–" denotes absence. In 2003 and 2004, the records are from quantitative surveys. In 2020, an informal site visit was performed for qualitative observations; a quantitative survey was not performed, therefore, the "not observed" species are not definitively absent or present.

Species name	**Common name**	**2003 exclosure**	**2004 exclosure**	**2004 outside control**	**2020 inside exclosure observations**
Rosa multiflora [a]	Multiflora rose	P	P	P	Widespread throughout
Rubus allegheniensis	Allegheny blackberry	–	P	–	Scattered clusters
Rubus hispidus	Swamp dewberry	–	P	–	Few clusters
Rubus occidentalis	Common blackberry	–	P	–	Few individuals
Rubus phoenicolasius [a]	Wineberry	P	P	P	Widespread throughout
Sassafras albidum	Sassafras	P	P	–	Clusters of young saplings
Scutellaria lateriflora	Mad dog skullcap	–	P	–	Not observed
Solidago caesia	Blue-stemmed goldenrod	–	P	–	One individual found
Solidago canadensis	Canada goldenrod	P	P	–	Not observed
Solidago sp.	Goldenrod	–	P	–	Not observed
Symphyotrichum sp.	Aster	–	P	–	Not observed
Thalictrum thalictroides	Rue anemone	P	P	–	Not observed
Thelypteris noveboracensis	New York fern	–	P	P	Few scattered patches with other ferns
Trifolium pratense	Red clover	P	–	–	Not observed
Toxicodendron radicans	Poison ivy	P	P	P	Throughout
Uvularia perfoilata	Bellwort	–	P	–	Not observed
Vaccinium angustifolium [b]	Lowbush blueberry	P	P	–	Not observed

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Table A-1 (continued). Presence of vegetation by species inside the exclosure in 2003, 2004, outside the exclosure in 2004, and inside the exclosure in 2020. P denotes present; "–" denotes absence. In 2003 and 2004, the records are from quantitative surveys. In 2020, an informal site visit was performed for qualitative observations; a quantitative survey was not performed, therefore, the "not observed" species are not definitively absent or present.

Species name	Common name	2003 exclosure	2004 exclosure	2004 outside control	2020 inside exclosure observations
Veronica officinalis [a]	–	–	P	–	Not observed
Viburnum diliatum [a]	Linden viburnum	–	–	–	Scattered throughout
Viburnum prunifolium	Blackhaw viburnum	P	P	P	Scattered throughout, but browsed
Viburnum seiboldii [a]	Seibold viburnum	P	P	P	Scattered throughout
Viola sp.	Violet	P	P	P	Scattered throughout
Vitis aestivalis	Summer grape	–	P	–	Not observed
Vitis labrusca	Fox grape	–	P	–	Not observed
Vitis sp.	Grape	P	P	–	Not observed

[a] denotes an invasive species

[b] denotes species presence only because it was planted in the exclosure

Appendix B. Statistical analyses of 2008 soils data; analyses of variance for each variable.

In tables B-1 through B-8, all analyses were carried out in Statview (Statview 1999). Not normally distributed variables were analyzed using non-parametric Kruskal-Wallis tests.

Table B-1. Analysis of variance results for differences in pH among 4 soil treatment types.

Source of Variation	DF	SS	MS	F	P
Between Groups	3	3.69	1.23	3.6	0.023
Residual	36	12.299	0.342	–	–
Total	39	15.989	–	–	–

Table B-2. Results of post-hoc Tukey tests comparing pH results found within soil treatment types.The only significant changes in pH levels were found when comparing removal plots to the pH reduction plots (aluminum sulfate additions) and when comparing the woodchip addition plots to the pH reduction plots.

Comparison	Diff of Means	t	Unadjusted P	Critical Level	Significant?
removal vs. pH	0.734	2.81	0.008	0.009	Yes
woodchip vs. pH	0.729	2.79	0.008	0.01	Yes
control vs. pH	0.614	2.349	0.024	0.013	No
removal vs. control	0.121	0.461	0.648	0.017	No
woodchip vs. control	0.115	0.441	0.662	0.025	No
removal vs. woodchip	0.00517	0.0198	0.984	0.05	No

Table B-3. Analysis of variance results for differences in percent organic matter among 4 soil treatment types.

Source of Variation	DF	SS	MS	F	P
Between Groups	3	170.05	56.683	13.068	<0.001
Residual	36	156.154	4.338	–	–
Total	39	326.204	–	–	–

Table B-4. Results of post-hoc Tukey tests comparing percent organic matter results found within soil treatment types. The only significant changes in percent organic matter were found when comparing woodchip addition plots to the pH reduction plots (aluminum sulfate additions) and when comparing the woodchip addition plots to the removal plots.

Comparison	Diff of Means	t	Unadjusted P	Critical Level	Significant?
woodchip vs. pH	5.077	5.451	<0.001	0.009	Yes
woodchip vs. removal	4.974	5.34	<0.001	0.01	Yes
woodchip vs. control	3.817	4.098	<0.001	0.013	Yes
control vs. pH	1.26	1.353	0.184	0.017	No
control vs. removal	1.157	1.242	0.222	0.025	No
removal vs. pH	0.103	0.111	0.912	0.05	No

Table B-5. Analysis of variance results for differences in percent moisture among 4 soil treatment types.

Source of Variation	DF	SS	MS	F	P
Between Groups	3	1251.853	417.284	12.814	<0.001
Residual	36	1172.368	32.566	–	–
Total	39	2424.221	–	–	–

Table B-6. Soil moisture content comparisons for factor: treat.

Comparison	Diff of Means	t	Unadjusted P	Critical Level	Significant?
woodchip vs. removal	13.805	5.409	<0.001	0.009	Yes
woodchip vs. pH	13.599	5.329	<0.001	0.01	Yes
woodchip vs. control	9.118	3.573	0.001	0.013	Yes
control vs. removal	4.687	1.837	0.075	0.017	No
control vs. pH	4.482	1.756	0.088	0.025	No
pH vs. removal	0.205	0.0804	0.936	0.05	No

Table B-7. Soil nitrate analysis; Kruskal-Wallis one-way analysis of variance on ranks. H = 1.848 with 3 degrees of freedom. (P = 0.604). The differences in the median values among the treatment groups are not great enough to exclude the possibility that the difference is due to random sampling variability; there is not a statistically significant difference (P = 0.604).

Group	N	Missing	Median	25%	75%
removal	10	0	1.811	1.574	2.163
woodchip	10	0	1.575	1.323	3.409
control	10	0	2.046	1.363	4.861
pH	10	0	1.573	0.91	2.849

Table B-8. Soil ammonium analysis; Kruskal-Wallis one-way analysis of variance on ranks. H = 0.827 with 3 degrees of freedom. (P = 0.843). The differences in the median values among the treatment groups are not great enough to exclude the possibility that the difference is due to random sampling variability; there is not a statistically significant difference (P = 0.843).

Group	N	Missing	Median	25%	75%
removal	10	0	2.25	1.822	3.298
woodchip	10	0	2.669	2.252	3.813
control	10	0	2.334	1.978	4.494
pH	10	0	2.688	1.659	3.431

NPS 337/173636, August 2020

National Park Service
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