Buck's Hollow
Vegetation Monitoring Report

La Tourette Park, Staten Island, NY

Pre- and Post-Restoration Vegetation
Assessment Report

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NYC Parks

City of New York
Parks & Recreation
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Executive Summary

In 2014 and 2015, the New York City Department of Parks and Recreation (NYC Parks) restored a degraded forest and wetland in La Tourette Park, Staten Island in an area known as Buck's Hollow. Restoration included both chemical and mechanical removal of introduced vines, shrubs, and trees, in conjunction with native plantings to encourage the development of a healthy native plant community. The site was subsequently fenced to protect the plantings from damage by white-tailed deer browse.

To assess restoration success, the Treatment site was monitored in conjunction with two nearby reference sites that were not included in restoration efforts. This included a negative reference site, or Control site, which was dominated by introduced vines, and a positive Reference site, which was composed of healthy, mature forest. All three sites were monitored prior to restoration, in 2012, and following restoration, in 2017, to determine whether the plant community at the Treatment site deviated from the Control and developed towards the Reference.

Restoration efforts successfully reduced introduced vines, shrubs, and trees at the Treatment site. Several planted native tree species increased in abundance, including *Acer rubrum, Liquidambar styraciflua,* and *Platanus occidentalis*. Overall tree abundance did not increase significantly, however, and shrub abundance declined, indicating that there was a high planting mortality rate despite the survival of some planted individuals. This mortality rate may have been the combined result of (1) deer browse pressure, when deer were able to breach the protective fencing, and (2) high herbaceous growth, particularly of tall, dense, introduced forbs and grasses, which overwhelmed the site following restoration and may have outcompeted planted individuals for light and other resources. By 2017, the Treatment site had successfully deviated from the Control site, but future monitoring, particularly if supplemental plantings occur, will be necessary to confirm that the Treatment site will converge towards the Reference site.

Based on the findings presented here, particularly the challenges that white-tailed deer browse and introduced species invasions presented to the project, we recommend:

- Continued monitoring of the Treatment site (and Control and Reference sites) at regular intervals to ensure that the canopy closes and ecosystem function is restored.
- Investments in high-quality deer fencing for restoration efforts in Staten Island, at least until deer densities decline.
- Returning to restored sites after restoration for regular introduced species removal.

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Introduction

In this study, NRG staff monitored ecological characteristics at three sites in La Tourette Park, Staten Island, NY, including a site that was restored (Treatment), a site dominated by *Rosa multiflora* and various woody vines that was not restored (Control), and a positive reference site of healthy forest (Reference). We monitored each site once prior to restoration efforts (2012) and once following restoration (2017). The study included a survey of red-backed salamanders and birds, but this report focuses on the vegetation monitoring. Between sampling years, the white-tailed deer population on Staten Island also expanded dramatically. In 2014, a Forward-Looking Infrared Radar (FLIR) aerial survey estimated 763 deer in Staten Island natural areas (NYC Parks, unpublished data). In January 2017, a camera survey estimated that there were 2,000 deer, or approximately 100 deer/mi^2 of suitable habitat, on Staten Island (DeNicola 2017). For this restoration project, the Treatment site was fenced to protect plantings from browse damage while the other sites (Reference and Control) were not. The objective of this report is to assess the impacts of restoration as well as deer browse on forest health at these sites over the timescale of the project.

The completion of this report was delayed by personnel changes and by the COVID-19 pandemic; thus, many of the report's practical findings have already been incorporated into NRG's forest restoration work – e.g., the new deer fencing specifications were amended to increase height and improve materials based on the experience with the fence used in the Buck's Hollow restoration.

Methods

<u>Site Description and Restoration Work</u>

The restoration area in Buck's Hollow was determined by the extent of canopy gaps and degraded forest. The planting areas are not fully contiguous because two heavily used hiking trails run through the site, dividing it into sections. Each planting area was fenced separately in order to maintain the continuity of hiking trails. Three sites were selected based on variation in treatment and forest quality and proximity to one another (Figure 1). Preceding restoration, Treatment and Control sites were similarly dominated by native and introduced vines, as well as introduced shrubs such as *Rosa multiflora*. The Treatment site was scheduled for upcoming forest restoration efforts, while the Control site was located outside of the restoration plan. The Reference site was a relatively healthy nearby area dominated by native herbaceous and woody plants. All sites are within 0.75 kilometers of one another. A wetland runs through much of the Treatment site. The Control site is adjacent and northwest of the Treatment site.

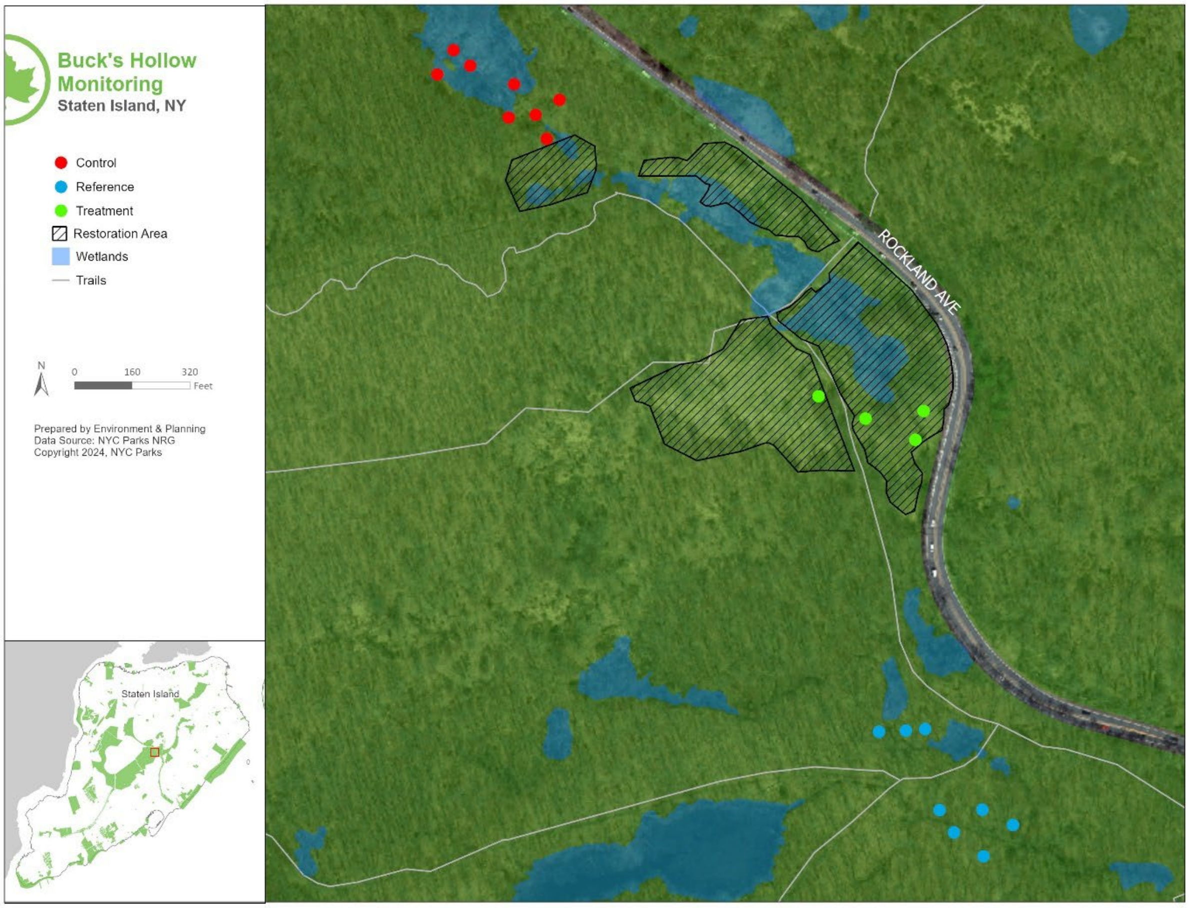

Figure 1. A map of the study area, including Treatment, Control, and Reference sites. Each point represents a plot that was assessed two years preceding and two years following restoration efforts. The hatched Restoration Areas were fenced.

Beginning in spring 2014, NYC Parks' Natural Resources Group began restoration of the degraded forest and wetland via a Capital contract (Figure 2). Since the work was conducted in and adjacent to freshwater wetland areas, contract specifications were modified to reflect the need to protect the wetland resources. Prior to the start of construction, straw bales and silt fences were installed along the stream bed to prevent sedimentation as well as an added precaution to prevent surface runoff from moving herbicide into the stream. Coconut fiber logs were placed along steeper slopes. A temporary stream crossing structure was constructed to allow equipment access from the contractor staging area on Rockland Ave across the stream to the rest of the site. This was the first time this kind of a provision was included in a Natural Resources Group contract. The crossing remained in place for the life of the contract before being removed at final acceptance. This contract also required the use of hand tools and no heavy equipment near the stream, under the dripline of mature trees, and anywhere else that was deemed too sensitive for heavy equipment.

Figure 2. Pre-restoration conditions at the site. Top left, the restoration area to the west of the trail. Top right: the restoration area including the freshwater wetland to the east of the trail. Bottom: the trail that bisected the restoration area. Photos from May 2, 2013.

Introduced herbs and vines, as well as introduced trees and shrubs with a DBH <10-cm, were manually removed or treated with herbicide. Herbicides were applied via a foliar spray method to treat understory herbaceous plants and groundcover vines. Larger woody vines, shrubs and trees were treated via the cut-stump treatment method, whereby stems are cut flush with the ground and herbicide is applied directly to the cut stem. Re-sprouts were subsequently retreated.

Following the removal of introduced species, 2-gallon-size native trees and shrubs were planted at the restoration site, in spring and fall of 2015, to assist with reforestation and biodiversity enhancement (see Appendix A for the full list and quantities). They were planted at four feet on-center, or approximately 65 trees per $100m^2$. The most commonly planted species included *Nyssa sylvatica* and *Platanus occidentalis*, along with a mix of *Quercus* spp. (Table 1). In addition, a native seed mix of herbaceous species was broadcast seeded (see Appendix A for the full list and quantities). Deer fencing had previously been installed in 2013 to protect plantings from damage by deer browse, and holes in the deer fence from normal wear-and-tear were repaired regularly. Fencing was 7 feet in height and posts were connected by a 1.76” x 1.96” horizontal (square) heavy duty polyethylene mesh (see Appendix B for full details).

Table 1. Species removed manually and through herbicide application at Buck's Hollow as part of the restoration efforts in 2014 and 2015. Species planted in and around the Treatment site in spring and fall of 2015.

Introduced Species Removed	**Native Species Planted**	
Woody and herbaceous	*Acer rubrum*	*Quercus alba*
Ailanthus altissima	*Amelanchier arborea*	*Quercus bicolor*
Aralia elata	*Celtis occidentalis*	*Quercus coccinea*
Artemesia vulgaris	*Cercis canadensis*	*Quercus ilicifolia*
Berberis thunbergii	*Clethra alnifolia*	*Quercus macrocarpa*
Chenopodium album	*Cornus amomum*	*Quercus muehlenbergii*
Phragmites australis	*Cornus racemosa*	*Quercus palustris*
Reynoutria japonica	*Diospyros virginiana*	*Quercus phellos*
Rubus phoenicolasius	*Eubotrys racemosa*	*Quercus prinus*
Rosa multiflora	*Ilex verticillata*	*Quercus stellata*
	Juglans nigra	*Quercus velutina*
Vines	*Lindera benzoin*	*Rhus copallina*
Ampelopsis brevipedunculata	*Liquidambar styraciflua*	*Rosa palustris*
Celastrus orbiculatus	*Magnolia virginiana*	*Rubus pensylvanica*
Lonicera japonica	*Nyssa sylvatica*	*Sambucus canadensis*
Persicaria perfoliata	*Photinia melanocarpa*	*Sassafras albidum*
Wisteria japonica	*Photinia pyrifolia*	*Viburnum dentatum*
	Platanus occidentalis	

Our study design compares the Treatment site with two reference sites: the Control site where no restoration was implemented, and the Reference site. Because the Treatment site was protected by deer fencing while the Control and Reference sites were not, we also use these data to assess the effects of deer browse over the five years of this study.

Vegetation Sampling

Twenty 10-m radius circular plots were established across all sites: 8 plots were placed each at the Control and Reference sites, while only 4 plots were placed at the Treatment site due to difficulty accessing the interior of the Treatment site prior to restoration. The circular plots were randomly selected by using a random number generator to place plots along four 100-m transects spaced 10-m apart within each site (Figure 3a).

Three layers of vegetation were sampled in each plot, including the canopy layer, midstory layer, and groundlayer (also referred to as herbaceous layer). Overstory trees were defined as trees with a DBH greater than or equal to 10 cm. Each overstory tree inside the circular plot was identified to species and the DBH was recorded. DBH was then converted to basal area/m^2 for analysis. For the midstory and herb layer assessment, two 10-m-long transects were assembled inside each plot, one oriented north-south and the other oriented east-west. The tapes were ½ meter above the ground. For the midstory, the number of decimeters where each species was present above the tape was tallied. For the groundlayer, the number of decimeters where each species was present below the tape was tallied (Figure 3b).

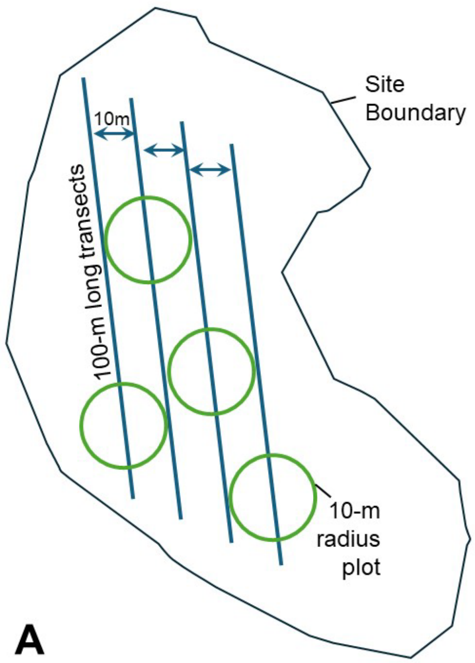

Figure 3. Figure 3A shows how the plots were established within the sites; Figure 3B shows the establishment of the plots within which all trees greater than 10-cm in DBH were recorded.

Vegetation was sampled in 2012 (two years prior to restoration) and 2017 (two years after restoration) from the same plots using the same protocol.

<u>*Data Analysis*</u>

Data from 2012 and 2017 were analyzed to determine the impact of the restoration work and deer fencing at Treatment compared to the Control and the Reference sites. All analysis was completed using R v.4.4.3 (R Core team, Vienna, Austria). Mixed effects models were used to examine the effects of site and year on species richness and relative abundance. Plot was used as the random effect. These models were created using the lmer() and glmer() function in the *lme4* package, and the best fitting model with the lowest Aikake Information Criterion (AIC) for each response variable was selected from an ANOVA table (Bates et al. 2019). Post-hoc tests were run to calculate the estimated marginal means and p-values for the best fitting models using the joint_tests() and emmeans() functions in the *emmeans* package (Lenth 2019).

Midstory and herbaceous layer plant species composition were also assessed across sites using a nonmetric multidimensional scaling ordination plot based on species relative abundance within each sampling plot. This method is often used to analyze changes in community composition. The midstory and groundlayer were plotted separately using the metaMDS() function in the *vegan* package, using the Bray-Curtis method for calculating distance. The impacts of site and year on community composition were calculated with a permutational ANOVA or permANOVA using the adonis() function from the same package. Species that significantly drove the differences between sites and years were identified using the envfit() function from the same package (Oksanen et al. 2019).

Results

Overstory Layer (DBH ≥10-cm)

Because restoration activities were focused on the removal of introduced vines and shrubs and planting of midstory-sized trees, we did not expect the overstory layer to change after the restoration, though differences between the Reference site and the Treatment and Control sites were expected prior to restoration. The Treatment site, for example, was selected for restoration because of pre-existing canopy gaps.

Tree Basal Area

As anticipated, generalized linear mixed models show that total tree basal area did not change significantly at any of the sites between years ($p = 0.7003$). There were slight decreases at all sites between 2012 and 2017, perhaps due to blow downs in large storms such as Hurricane Sandy, which occurred between sampling years. The Treatment site had lower tree basal area than the Reference site in both years ($p = 0.0007$), and native trees accounted for the vast majority of tree basal area at all three sites (Figure 2). The Treatment site had similar basal area to the Control site prior to restoration.

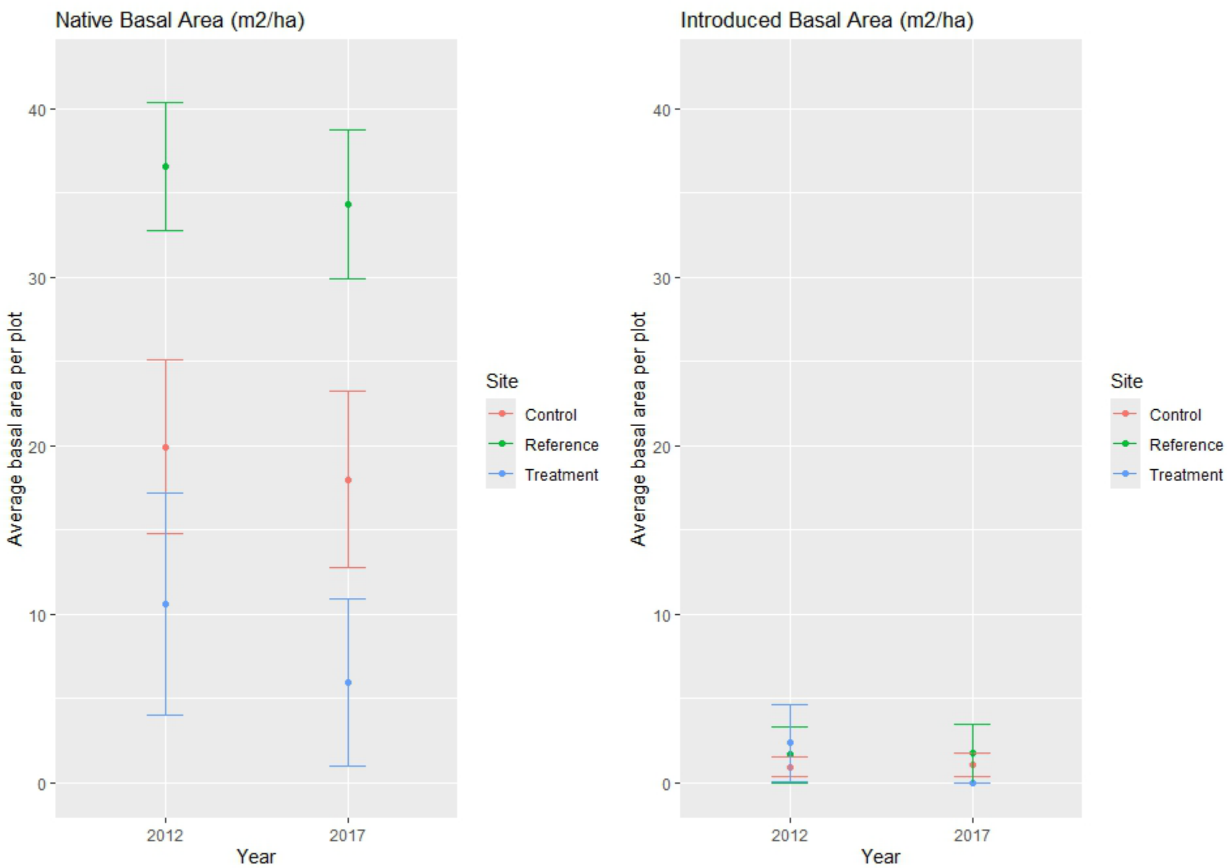

Figure 4. Native and introduced stand basal area in square meters per hectare at all three study sites preceding and following restoration at Treatment.

Species Richness and Composition

The Treatment site had the lowest overstory tree species richness in both years, including six native species and one introduced species (*Acer platanoides*) in 2012 and four native species in 2017. Species richness was slightly lower in 2017 than 2012 in both the Treatment and Control sites. These reductions may be due to blowdowns from large storms in the Treatment and Control sites between sampling years. The Reference site had the highest number of overstory trees species in both years, and only one introduced tree was detected (*Alnus glutinosa*) (Table 2).

There were also more native species than introduced species at all three sites. *Acer rubrum* was the most common tree at the Treatment and Reference sites in both years. *Acer rubrum* had the highest stem count in Control as well, but a few large *Quercus bicolor* trees had the highest basal area at that site. Introduced *Aralia elata* was common at Control in both years, but because of its slender stems it was not an important source of basal area.

Table 2. Total overstory species richness by site and year at all three study sites preceding (2012) and following (2017) restoration at Treatment.

Site	Native Species Richness	Introduced Species Richness	Total Species Richness

	2012	2017	2012	2017	2012	2017
Control	9	8	3	2	12	10
Treatment	6	4	1	0	7	4
Reference	12	12	1	1	13	13

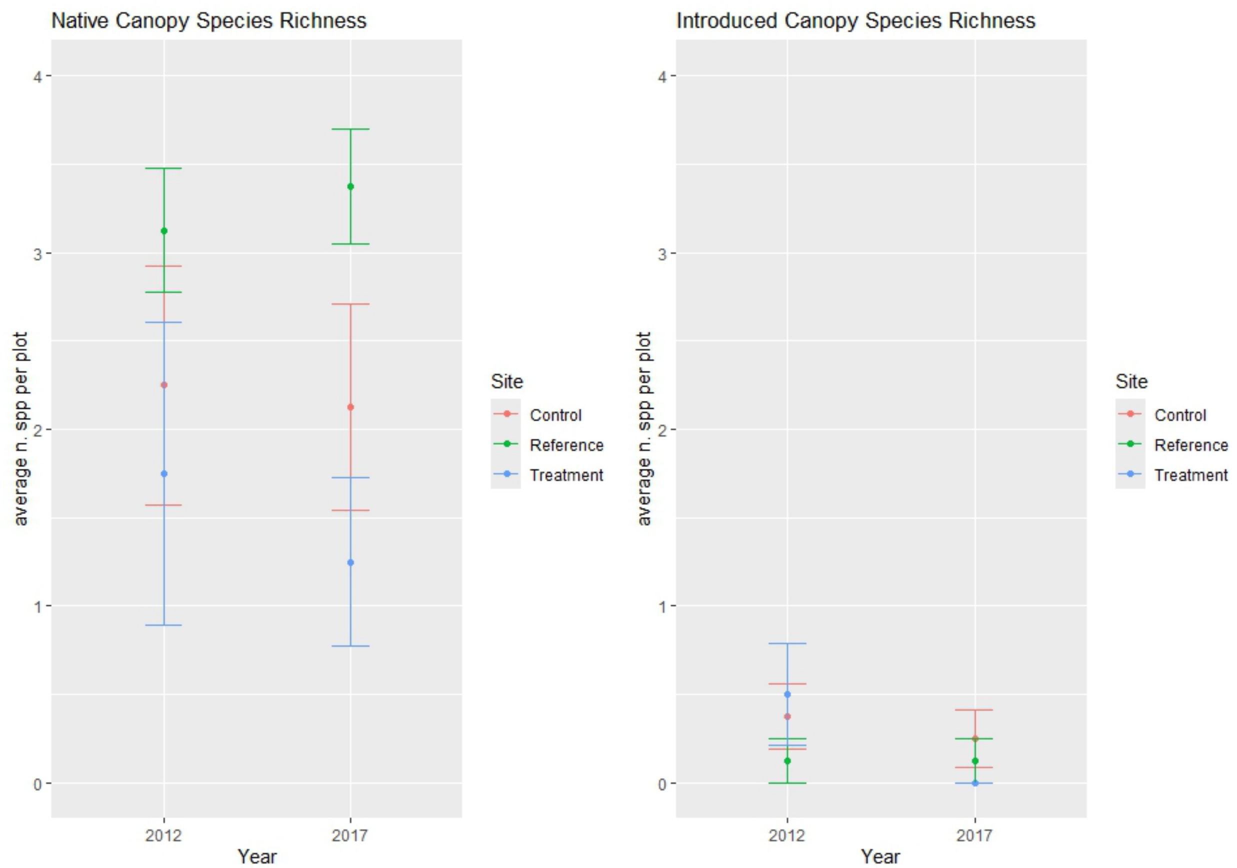

Figure 3. Average native and introduced species richness in the tree canopy per plot at all three sites before (2012) and after (2017) restoration at the Treatment Site.

When total species richness was analyzed at the plot level, we found no significant change in total overstory tree richness between years at any of the sites, and no site had significantly higher species richness per plot than another site.

Midstory Results (Trees, Shrubs and Vines <10-cm DBH, ≥0.5-m height)

Species Richness

After restoration, overall midstory species richness at the Treatment site declined from 21 species in 2012 (15 native and 6 introduced) to 6 species in 2017 (5 native only one introduced species, *Malus sp.*). Total species richness was relatively stable at the reference sites (Table 3).

Table 3. Native, introduced, and total midstory species richness at each study site preceding and following forest restoration at Treatment.

Site	Native Species Richness		Introduced Species Richness		Total Species Richness	
	2012	2017	2012	2017	2012	2017
Control	16	10	7	9	23	19
Treatment	15	5	6	1	21	6
Reference	16	15	1	1	17	16

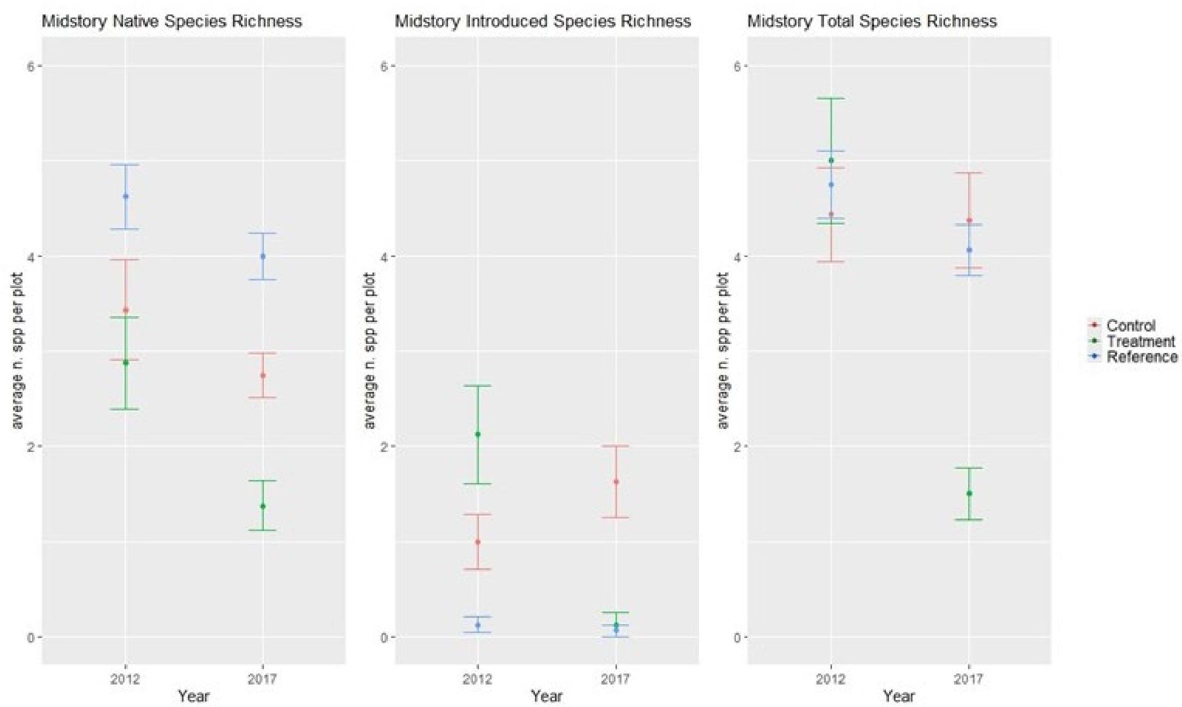

Figure 4. Average midstory species richness per plot at each study site (Total, native, and introduced) preceding and following forest restoration at Treatment.

Following restoration efforts at the Treatment site, total species richness per plot declined significantly ($p < 0.001$) (Figure 4). Introduced richness per plot was significantly higher at the Treatment than Reference site in 2012 ($p = 0.0430$), but the Treatment site converged towards the Reference site in 2017 ($p = 0.9963$). Native species richness per plot was statistically similar among all sites.

Midstory Species Composition

Midstory species composition was visualized using a nonmetric multidimensional scaling ordination plot based on species relative abundances (Figure 5). There is no overlap between plots in the Treatment site from 2012 and 2017, indicating that the plant community in the restoration site changed significantly after restoration. Although there was some overlap in plant composition between Treatment and Control plots in

2012, site was a strong predictor of differences in midstory plant community composition across both years (p = 0.001, Table 4).

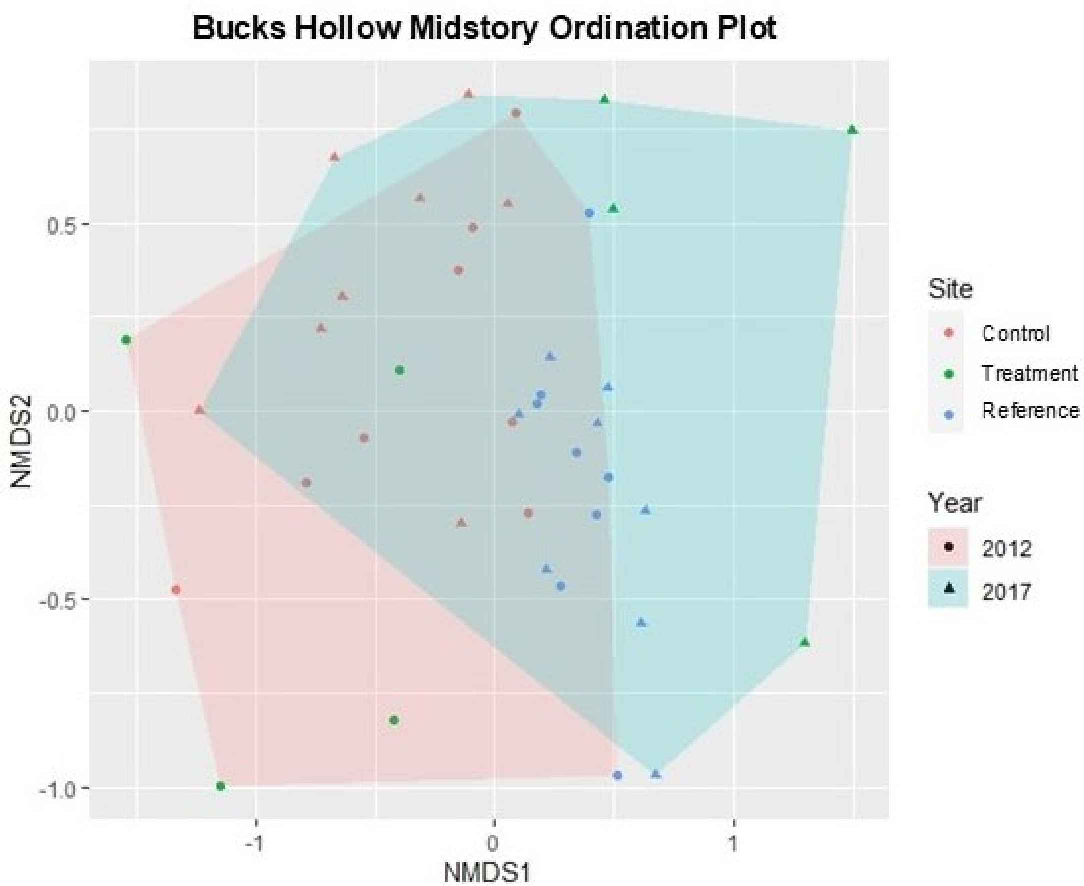

Figure 5. An ordination plot of midstory species composition at Treatment, Control, and Reference preceding and following restoration (stress = 0.201, R^2 = 0.960). Each point is a plot, and points are close together when they contain similar species composition.

Table 4. F-statistic and p-value of repeated measures permANOVA to identify main drivers of midstory species composition.

Explanatory Variable	F	P
Year	0.7849	0.643
Site	4.4590	0.001***
Year * Site	0.8773	0.593

Ordination analysis identified midstory species that were the strongest drivers (p>0.05 and r^2 > 0.2) of the differences between the sites and sampling years (Table 5). The pattern of high native species richness and abundance differentiated the Reference site from the other two, while the increase in native species in the Treatment site between sample dates differentiated Treatment from Control. The higher abundance of invasive species differentiated the Control site. One exception was *Vitis labrusca*, a native vine that was only found in Control plots. Another exception was *Nyssa sylvatica*, which was a planted species but was not observed in any of the plots within the restoration site and only exists in the Reference plots. Nonnative *Phragmites australis* was found in small numbers only in one Control plot.

Table 5. Differences in midstory species composition between years at the three sites, as determined by the average number of midstory transect points (± standard error) occupied by species that drove that difference (r2>0.2; p < 0.05). Trees and shrubs in green were planted at Treatment, and species in red are introduced.

Midstory Species	r^2 (p-value)	Control		Treatment		Reference	
		2012	2017	2012	2017	2012	2017
Acer rubrum	0.5924 (0.001)	27.6 ±50.6	16.5 ±45.9	25.0 ±50.0	37.8 ±75.5	129.0 ±59.1	151.1 ±41.5
Celastrus orbiculatus	0.2045 (0.015)	0.375 ±1.06	6.75 ±11.1	19.8 ±20.2	0	0	0
Lonicera japonica	0.5172 (0.001)	10.3 ±20.9	5.8 ±9.5	26.5 ±12.6	0	0	0.1 ±0.4
Nyssa sylvatica	0.2397 (0.008)	0	0	0	0	14.10 ±40.0	14.9 ±42.1
Platanus occidentalis	0.2447 (0.001)	0	0	0	10.5 ±15.1	0	0
Phragmites australis	0.2120 (0.023)	0.625 ±1.77	0.375 ±1.06	0	0	0	0
Rosa multiflora	0.3249 (0.001)	15.1 ±37.4	26.1 ±45.1	6.0 ±12.0	0	0	0
Vitis labrusca	0.2202 (0.014)	14.5 ±41.0	19.8 ±50.4	0	0	0	0

Three functional groups were present in the midstory: vines, shrubs, and tree saplings (DBH <10 cm). Frequency of several functional groups changed significantly between sampling years (Figure 6).

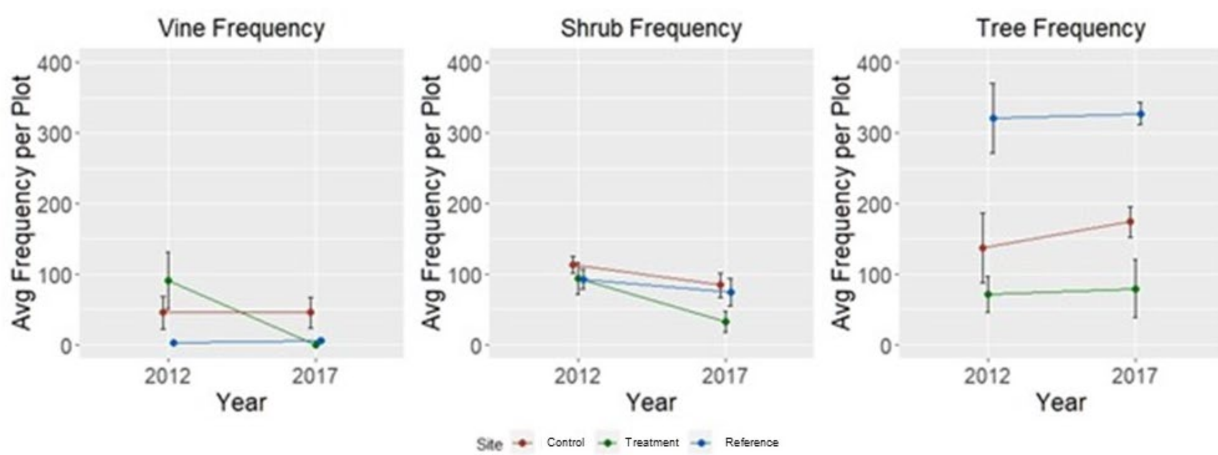

Figure 6. Vine, shrub, and tree frequency in the midstory at each study site preceding and following forest restoration at Treatment.

In 2012, before restoration, introduced woody vines such as *Ampelopsis brevipedunculata, Celastrus orbiculatus,* and *Lonicera japonica* were common at the

Treatment site. Following restoration, no midstory vines were encountered in any of the plots within the Treatment site. As a result, vine frequency dropped significantly at the Treatment site between sampling years ($p < 0.0001$). Vines were common at the Control site in both sampling years. They were also present at the Reference site, but they were uncommon and were largely the native species *Smilax rotundifolia* and *Toxicodendron radicans*.

Shrub frequency declined at all three sites between sampling years (Control: $p < 0.0001$; Treatment: $p < 0.0001$; Reference: $p = 0.0007$). At the Treatment site, as would be expected, the proportion of introduced shrubs declined due to removal and native shrubs increased, due to planting. However, total abundance of shrubs declined between sampling dates at all three sites. For example, *Lindera benzoin* was the most common shrub at all three sites in 2012, but it declined by 34%, 42% and 15% between sampling years at Control, Treatment, and Reference sites respectively. Other woody species also declined in the midstory, but to a lesser extent, such as *Prunus serotina* (increased in 2017 in Control, but was not detected in Treatment and Reference), *Rubus allegheniensis* (reduced by 87% in Control between 2012 and 2017, not detected in Treatment in 2017), and *Viburnum dentatum* (not detected in any site in 2017). In contrast, the invasive shrub *Rosa multiflora*, increased by 73% between the sampling years at Control.

Midstory trees were uncommon at the Treatment site in both years compared to Reference and Control areas, and tree abundance in Treatment did not change significantly after restoration ($p = 0.7570$). This reflects a decline in introduced trees, such as *Alnus glutinosa*, and an increase in planted trees, including *Acer rubrum, Liquidambar styraciflua*, and *Platanus occidentalis*, indicating the survivorship of planted trees two years post-restoration (Table 6). The invasive tree *Aralia elata* nearly doubled in frequency at Control between sampling years, which accounts for the majority of the increase in tree frequency at that site. Neither *Aralia elata* nor *Rosa multiflora* were detected at Treatment or Reference areas in 2017.

Table 6. Tree species frequency observed in the midstory at each site and year.

	Control		Treatment		Reference	
	2012	2017	2012	2017	2012	2017
Native Species						
Acer rubrum	221	132	100	151	1032	1209
Betula lenta	0	0	0	0	26	7
Carpinus caroliniana	0	0	0	0	49	136
Carya tomentosa	0	0	0	0	164	134
Fagus grandifolia	0	0	0	0	133	78
Fraxinus sp.	15	52	112	115	189	105
Liquidambar styraciflua	47	0	0	8	449	405
Liriodendron tulipifera	135	200	0	0	0	0
Nyssa sylvatica	0	0	0	0	113	119
Platanus occidentalis	0	0	0	42	0	0
Prunus serotina	30	72	20	0	5	0
Quercus alba	0	0	0	0	0	15

Quercus bicolor	387	545	0	0	241	317
Quercus rubra/velutina	0	0	0	0	0	81
Sassafrass albidum	0	0	0	0	113	0
Ulmus americana	124	50	3	0	44	14
Introduced Species						
Acer platanoides	0	0	48	0	0	0
Alnus glutinosa	0	0	0	0	16	0
Aralia elata	140	272	0	0	0	0
Morus alba	0	73	0	0	0	0

Groundlayer Results (All Plants <0.5-m height)

Groundlayer Species Richness

Groundlayer (herbaceous) species richness doubled at the Treatment site from 2012 to 2017, due to an increase in both native and introduced species richness, and richness declined at both Reference and Control sites between years (Table 7, Figure 9). This decline was particularly steep at the Reference site, where native groundlayer species richness went from 27 species in 2012 to 13 species in 2017. Several woody species were observed at the Reference site in 2012 but not in 2017, including *Acer rubrum, Clethra alnifolia*, *Fraxinus* spp., *Liquidambar styraciflua*, *Prunus serotina*, *Sassafras albidum*, *Viburnum dentatum*, and *V. prunifolium* (Table 7).

Table 7. Native, introduced, and total herbaceous layer species richness at each study site preceding and following forest restoration at Treatment.

Site	Native Species Richness		Introduced Species Richness		Total Species Richness	
	2012	2017	2012	2017	2012	2017
Control	24	18	10	12	34	30
Treatment	15	26	9	22	24	48
Reference	27	13	8	5	35	18

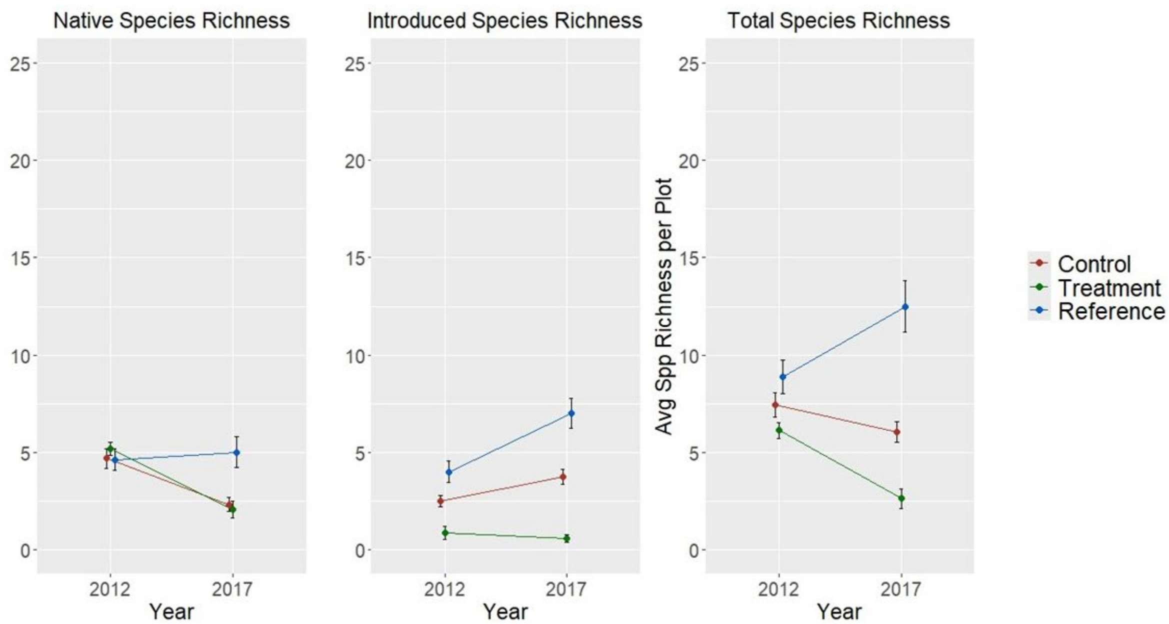

Figure 9. Native, introduced, and total species richness in the groundlayer per plot at each study site preceding and following forest restoration at Treatment.

Total groundlayer species richness per plot increased significantly at the Treatment site following restoration ($p = 0.0001$), although neither native nor introduced richness increased significantly independently (native: $p = 0.5514$; introduced: $p = 0.1302$) (Figure 9). Groundlayer species richness per plot at the Treatment site did not resemble either reference sites in 2017, and diverged significantly from both in all three richness categories. Notably, native species richness declined significantly at both Control ($p = 0.0007$) and Reference ($p < 0.0001$) between sampling years.

Groundlayer Species Composition

Like the midstory, groundlayer species composition was analyzed with a nonmetric multidimensional scaling ordination plot based on species abundances (Figure 10). Unlike the midstory, year, site, and the interaction between year and site were all significant drivers of the groundlayer plant communities ($p = 0.0001$, $p = 0.001$, and $p = 0.001$, respectively). This indicates that each site started with slightly distinct groundlayer plant communities and continued to change in different ways between sampling years. The significant interaction suggests that the Treatment site changed more than the reference sites, which would be expected considering that it was the only site that was managed and protected by fence between sampling years. At the Treatment site, groundlayer plots were relatively similar to the negative reference site in 2012, but shifted considerably after restoration and did not resemble either reference site in 2017. The Treatment site in 2017 may not have had enough time to approach the Reference site characteristics. For Control and Reference sites, change was more subtle between 2012 and 2017.

Additionally, ordination analysis identified groundlayer species that were significant drivers of the differences between sites and sampling years ($p < 0.05$; Table 8). Ruderal herbs such as *Artemesia vulgaris, Festuca* spp., *Poa* spp., and *Ranunculus ficaria* characterized the Treatment site in 2017. *Lindera benzoin* was present in all three sites, and particularly the Reference site, in 2012. *Microstegium vimineum* characterized the Control and Treatment sites in 2017.

Table 8. Average number of groundlayer transect points (± standard deviation) occupied by select plants that were shown to be significant divers of the differences between study sites and years (p < 0.05). Species in red are introduced.

Groundlayer Species	Control		Treatment		Reference	
	2012	2017	2012	2017	2012	2017
Artemesia vulgaris	0	0	0	1.5 ±1.9	0	0
Dennstaedtia punctilobula	0	0	0	0	23.4 ±35.3	0
***Festuca* & *Poa* spp.**	1.3 ±3.5	8.4 ±16.0	0	74.8 ±22.4	0	0
Lindera benzoin	39.6 ±29.8	0.9 ±0.8	10.8 ±8.5	0	47.8 ±14.9	3.5 ±4.0
Liquidambar styraciflua	1.0 ±1.6	0.1 ±0.4	0	1.0 ±0.8	3.0 ±3.8	0
Maianthemum canadense	1.0 ±1.3	0.1 ±0.4	0.3 ±0.5	0	2.4 ±3.4	0.4 ±0.7
Microstegium vimineum	0.3 ±0.7	50.4 ±76.1	0	47.3 ±67.9	0.5 ±1.1	1.3 ±3.5
Ranunculus ficaria	0	0	0	0.8 ±1.5	0	0

Repeated Measures Permanova

Explanatory Variable	F	P
Year	9.1800	0.001***
Site	4.8010	0.001***
Year * Site	2.9467	0.001***

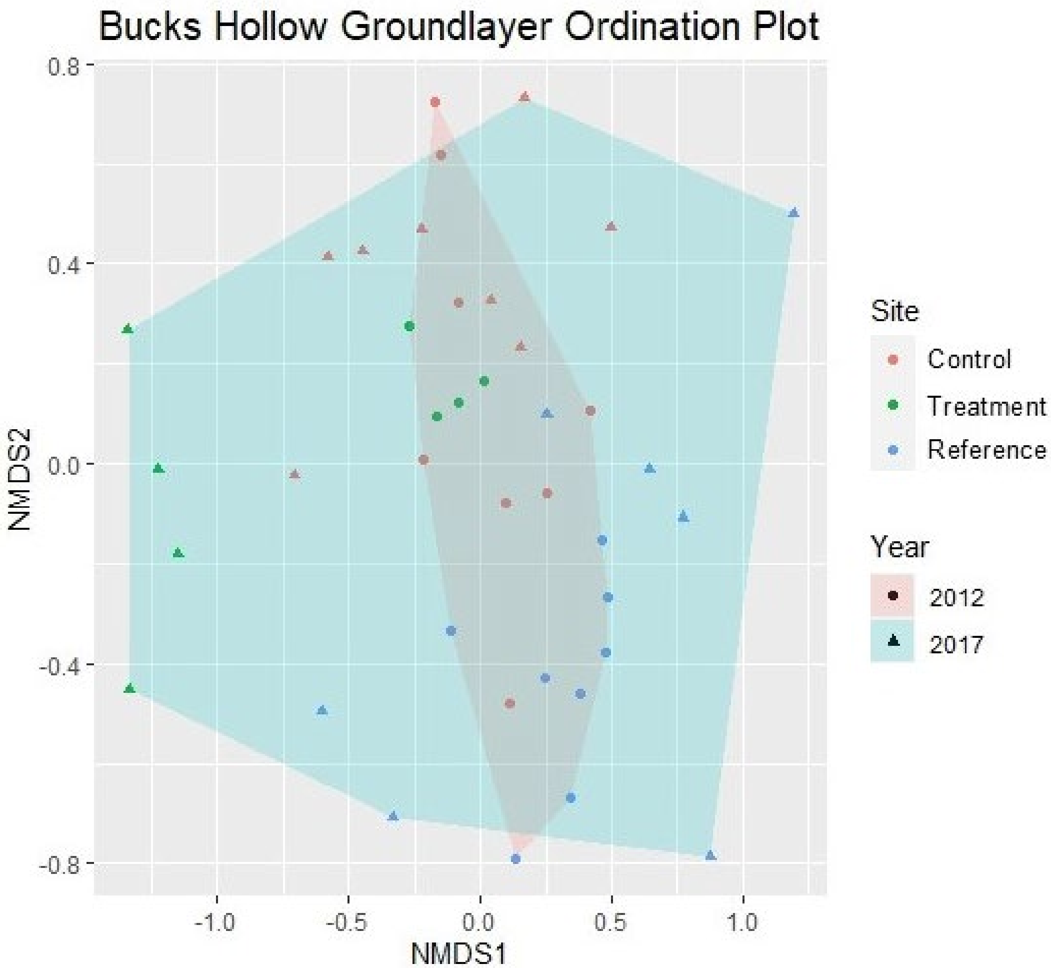

Figure 10. An ordination plot and Permanova based on groundlayer species abundance at Treatment, Control, and Reference preceding and following restoration (stress = 1.195, R^2 = 0.962). Each point is a plot, and points are close together when they contain similar species composition. Both the plot and the Permanova show that plant communities at all three sites shifted significantly between 2012 and 2017.

Five functional groups were present in the groundlayer: graminoids, forbs, vines, shrubs, and trees. Just as in the midstory, frequency of these functional groups changed at each site between sampling years (Figure 11).

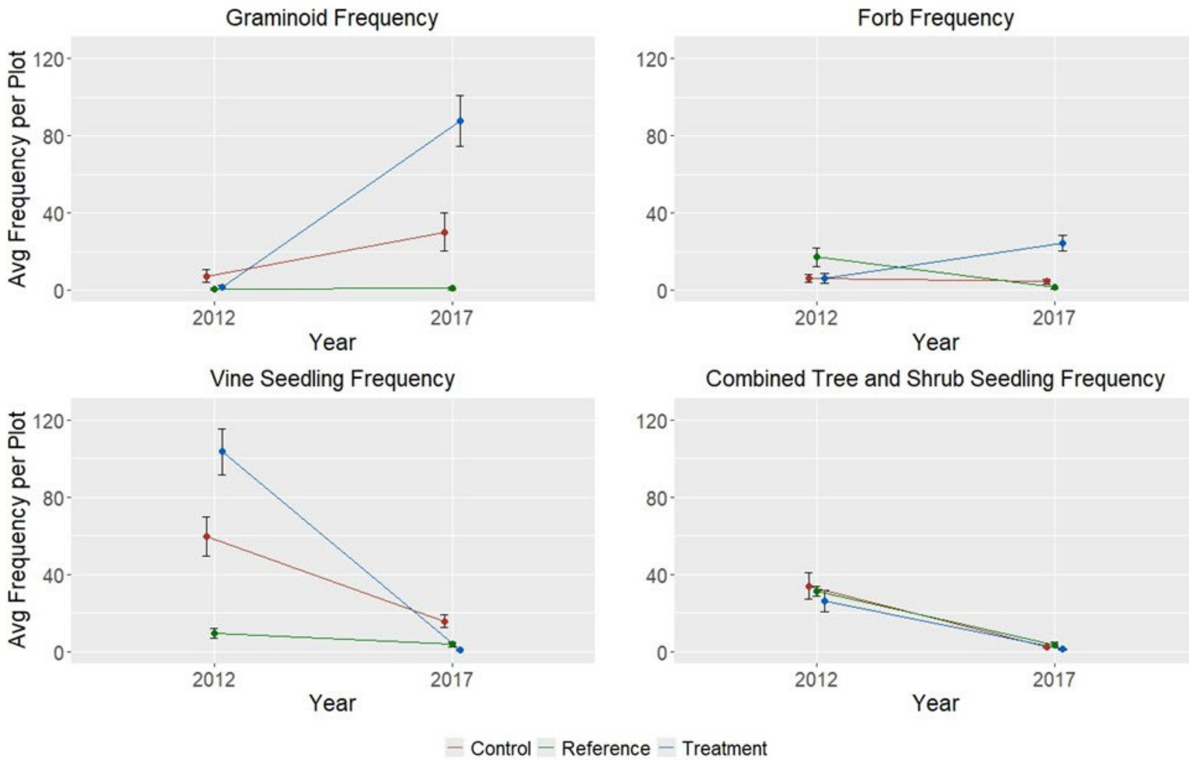

Figure 11. Graminoid, herb, vine, and combined shrub and tree seedling frequency in the groundlayer at each study site preceding and following forest restoration at Treatment.

The frequency of graminoids and forbs was similar among all three sites in 2012. Following restoration, however, graminoid frequency increased significantly at the Treatment site (difference between 2012 and 2017, $p = 0.0010$). In 2017, the frequency of graminoids at Treatment was significantly different from the Reference site ($p = 0.0004$), although it was still similar to Control ($p = 0.2779$). While forb frequency did not increase significantly at the Treatment site between years, this site did have a significantly higher herb frequency than either the Control ($p = 0.0091$) or Reference sites ($p < 0.0001$) in 2017. Overall, then, herb frequency increase in Treatment is mostly driven by graminoids.

The increase in herbaceous productivity at the Treatment site was likely due to an increase in ruderal forbs and graminoids, both native and introduced species, after restoration, which included seeding (and fencing). In addition to the most common herbaceous species listed in Table 8, this pattern was also exhibited by *Allium vineale*, *Articum* spp., *Cirsium* spp., *Daucus carota*, *Nasturtium officinale, Phragmites australis, Settaria* spp., *Stellaria* spp., *Taraxacum officinale,* and *Trifolium* spp.

Despite an increase in total herbaceous frequency, native understory herbaceous species declined in frequency at all three sites between sampling years. While the native forbs and ferns such as *Dennstaedtia punctilobula, Maianthemum canadense,* and *Osmunda cinnamomeum* were infrequent but present at the Reference site in 2012, these species had declined significantly by 2017 (Table 8). In fact, *Maianthemum canadense* was present at all three sites in 2012, and was almost absent from all by 2017 (Figure 12).

In contrast, introduced herbs were detected at the Control site in both sampling years. In 2012, the invasive *Microstegium vimineum*, which is not eaten by deer, was present but uncommon at both reference sites. By 2017, this plant had increased its frequency in the Control site and had spread to the Treatment site. *M. vimineum* did not increase in frequency significantly between 2012 and 2017 in the Reference site (Figure 12).

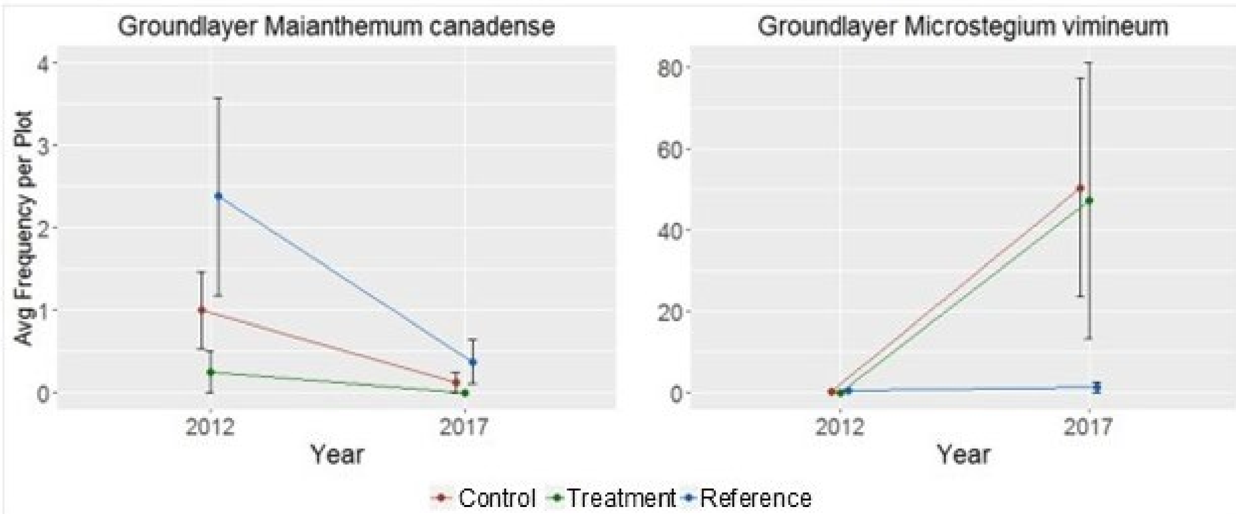

Figure 12. Native *Maianthemum canadense* and introduced invasive *Microstegium vimineum* frequency per plot at each study site preceding and following restoration at Treatment. Note that the y-axis scale is different for each plot.

In 2012, vine seedlings were an important component of the groundlayer at the Treatment and Control sites, and both sites had an abundance of native and introduced vines. Following restoration, groundlayer vine frequency declined significantly at the Treatment site ($p < 0.0001$). Interestingly, this metric also declined significantly at the Control site despite the lack of restoration work ($p < 0.0001$). Just as in the midstory, there were a few vines present at the Reference site in both sampling years, but these were mainly the native *Smilax rotundifolia* and *Toxicodendron radicans*. By 2017, groundlayer vine frequency was similar in all sites. For example, *Lonicera japonica* was a particularly dominant invasive introduced vine at both the Control and Treatment sites in 2012, but by 2017 it had declined considerably at both sites (Figure 13).

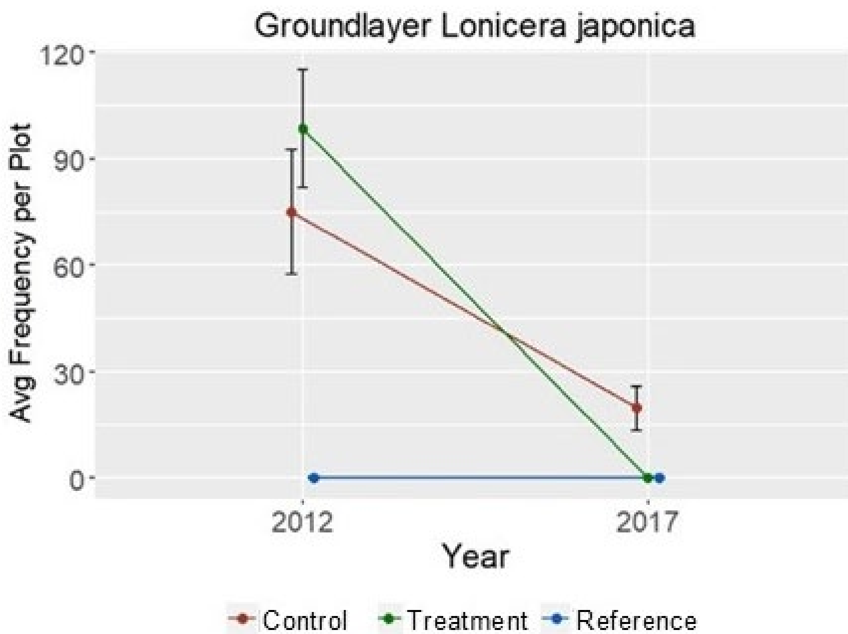

Figure 13. *Lonicera japonica* seedling frequency per plot at each study site preceding and following restoration at Treatment.

Combined groundlayer woody shrub and tree seedling frequency declined significantly at all three sites between sampling years ($p < 0.0001$). Native and introduced shrub species that were dominant understory plants at multiple sites in 2012, including *Lindera benzoin*, *Rubus allegheniensis*, *Rosa multiflora*, and *Viburnum dentatum* were dramatically less common in 2017. In addition to the shrubs featured in Table 8, the NYS endangered (S1) shrub *Euonymus americanus* was present in small numbers in the understory at the Reference site in 2012 but was not encountered in 2017. It is unlikely that these seedlings increased size class and became midstory plants, because midstory shrub frequency also declined drastically.

Despite the overall decline in woody seedlings, there was an increase in tree seedlings at the Treatment site in 2017. No tree seedlings were encountered at this site preceding restoration, but *Fraxinus* spp., *Liquidambar styraciflua,* and *Ulmus* spp. seedlings were all present following restoration, potentially due to a release from both deer herbivory and competition with introduced vines following fencing and restoration.

Discussion

Overstory Tree Stand Dynamics

Restoration efforts did not include any modification of the tree canopy layer, so we did not expect to see any significant changes in overstory tree stand dynamics after restoration. As expected, the overstory tree basal area and species richness did not change significantly at any site between sampling years. There was a decline in introduced canopy tree species at the Treatment site, which may have been the result of natural blow downs during large storm events (such as Hurricane Sandy) that took place between sampling years and before restoration. Trees at the Treatment site before restoration may have been more likely to fall than trees at other sites due to dense vine growth in the canopy, which could have pulled trees down prior to the start of restoration activities; or due to inundated soils in the wetland, where soil was potentially less firm around tree bases. Regardless, this decline in introduced trees did not statistically impact total canopy richness or basal area at this site.

The Reference site had greater total tree basal area than Treatment, which is expected for a positive reference site. In addition, the Treatment site was selected due to the existence of canopy gaps. Small trees planted during the restoration efforts in 2014 had not yet reached the canopy layer (DBH ≥10-cm) by field sampling in 2017. However, continued monitoring of these plantings will allow us to track tree growth and gauge whether restoration will enhance the robustness of the tree layer at the Treatment site to become comparable to the Reference site.

Impacts of Restoration

Restoration had a strong impact on the midstory and groundlayer communities at the Treatment site. Ordination plots show that both layers resembled the negative reference site in 2012 but then deviated following restoration in 2017. Although the Treatment site had not converged towards the Reference site by 2017, we did not expect it to resemble a healthy, mature forest just two years after restoration.

In the Treatment site's midstory, there was a strong decline in introduced species and vine abundance as a direct result of restoration efforts. Tree abundance remained constant, which reflects the removal of introduced midstory trees along with an increase in abundance of several planted trees, including *Acer rubrum, Liquidambar styraciflua,* and *Platanus occidentalis.* However, not all tree species that were planted at the Treatment site increased in abundance in our plots by 2017. Some of these species may have been planted outside of our sampling plots, and some of these trees may have been browsed on by deer since the deer fencing was breached multiple times post restoration. Species that did increase in abundance were planted at a higher rate than other species. In particular, over 450 *Platanus occidentalis* individuals were planted throughout the Treatment site, which was one of the highest planting rates of any species, and this tree abundance increased from 0 to 42 in the midstory plots (Table 6). In contrast to tree abundance results, shrub abundance declined after restoration. This reflects the removal of many *Rosa multiflora* shrubs during restoration, as well as the fact that far fewer shrubs were planted than trees. It is worth noting, however, that the lack of significant increase in either trees or shrubs (Figure 6) may also indicate a low survival rate following planting, which was noted anecdotally by staff and can be attributed to fencing failures and deer herbivory (Kip Stein, pers. comm.).

In the Treatment site's groundlayer, there was an increase in total species richness, graminoid frequency, and a small increase in forb frequency due to the introduction of both native and introduced ruderal species. This herbaceous productivity distinguished the Treatment site from both reference sites, where species richness and herb frequency declined between sampling years. As a result, the groundlayer is the only layer where the Treatment site had greater total, native, and introduced species richness than either reference site in 2017. Ruderal herbaceous growth is characteristic of recent restoration sites, where there is greater light availability and less competition (Sullivan et al. 2009). The majority of the herbs that we saw in 2017 were native shade-intolerant species that will likely decline as the canopy closes. However, we also detected a few individuals of several problematic species, including mugwort (*Artemesia vulgaris*), common reed (*Phragmites australis*), and lesser celandine (*Ficaria verna*).

In conjunction with increases in herbaceous richness and frequency, there was a significant decline in groundlayer vine seedling frequency at the Treatment site, which is likely a direct result of restoration efforts at this site.

This restoration contract included several provisions (e.g., erosion control measures) intended to protect the freshwater wetland resources on the site. Many of these approaches have since become standard practices in subsequent NYC Parks contracts in freshwater wetlands.

Introduced Plant Invasions

We detected a few notable introduced plant expansions when we compared the 2012 and 2017 data.

Aralia elata stem frequency doubled in the midstory at the Control site between sampling years, where it rose from an average of 17.5 points per plot in 2012 to an average of 34.0 points in 2017. *Aralia elata* is a relatively new problematic species in the northeastern US. Its ability to expand clonally via suckers allows it to spread rapidly in canopy gaps (Moore et al. 2009), which may give it an advantage over slower-growing native trees, especially in early successional habitat like the Treatment site. In addition, deer do not tend to browse on *A. elata*, so it is spared from that pressure unlike other shrub species. Although this plant was not detected at the Treatment or Reference site in 2017, this tree population has demonstrated its ability to spread quickly, and it may appear and expand at those sites rapidly.

Additionally, the introduced annual grass *Microstegium vimineum* showed similar invasive tendencies at both the Control and Treatment sites. This grass was only present in one plot each at the Control and Reference sites in 2012 and none at the Treatment site in 2012 or during restoration. However, in 2017, its abundance increased dramatically at the Control site and became highly abundant at the Treatment site. Often, *M. vimineum* populations first appear along roadsides and edges, and subsequently spread to the forest interior (Cole and Weltzin 2004). Mountain biking and hiking are common on the trails at Buck’s Hollow, and have been shown to be important vectors for seed dispersal (Pickering and Mount 2010). Continuous trails and frequent mountain biking may have allowed this species to expand rapidly at Buck’s Hollow. Additionally, *M. vimineum* is associated with deer, so herbivory may have further contributed to its invasion. Deer do not generally consume this grass but may facilitate its spread by consuming competing plant species, or by manipulating abiotic conditions to create favorable microhabitats for its seed (Knight et al. 2009). If these combined pressures increased this plant’s abundance at the Control site and adjacent to exclosures at the Treatment site, high propagule pressure could have allowed this plant to spread into the Treatment site despite the lack of trails and deer herbivory within the fencing.

Microstegium vimineum is a well-studied, problematic grass that is associated with reduced native species richness (Adams and Engelhardt 2009) and reduced growth in some associated species (Bauer and Flory 2011) because of its ability to outcompete planted and naturally occurring woody seedlings for moisture and light (Aronson and

Handell 2011). Furthermore, it is shade-tolerant and can persist abundantly under a closed canopy (Flory et al. 2007), so it may continue to disrupt ecosystem function at the Treatment site after the canopy closes.

Introduced plants were uncommon at the Reference site in both sampling years, potentially due to a combination of factors. Introduced plant invasions are often associated with historical anthropogenic land-use, which can introduce non-native plant seeds or alter soil conditions in a way that is advantageous to invasive plants (Beauséjour et al. 2015). Variable histories between sites would provide a potential explanation for why the Treatment and Control sites seem to be at an increased risk for plant invasions compared to the Reference site, despite the fact that all three are in close proximity. Visual inspection of aerial photographs from 1924 show that the Reference site had remnant forested vegetation at that time while Control and Treatment were both active farmland. Additionally, the Reference site generally had the highest native and overall species richness across forest layers. All else being equal, highly diverse plant communities tend to be more resistant to plant invasions because high species diversity often leads to low light and nutrient availability (e.g., Elton 1958, Naeem et al. 2000). However, native richness declined in the midstory and groundlayer between 2012 and 2017 and may continue to decline. Introduced species propagule pressure is concurrently increasing as invasive plants spread through the nearby Treatment and Control sites. These combined factors could increase the Reference site's vulnerability to introduced plant invasions in the future (Von Holle and Simberloff 2005).

Impacts of White-tailed Deer Browse

Deer densities are persistently above ecological carrying capacity throughout the eastern United States (McShea 2012). Unlike the majority of the northeastern USA, however, Staten Island does not have a history of chronic deer overpopulation, and deer were not present before the late 1990's (E. Pehek, pers. obs.). By January 2017, the deer management consulting company White Buffalo estimated that there were approximately 100 deer/mi^2 of suitable habitat on the island (DeNicola 2017). While our study did not explicitly examine browse in Buck's Hollow, the data indicate that there were community shifts at all three sites between sampling years that could be attributed to the new and rapidly expanding deer population in our study area.

Deer fencing provided some protection for plantings at the Treatment site, allowing for high herbaceous productivity and some planted tree species survival. The site topography rendered fencing less effective than expected (i.e., a hill meant that deer could jump into the site). In addition, downed trees, car accidents, and other disturbances created large, temporary gaps in the fences, which enabled deer to enter some exclosures for long periods of time. The fences kept deer out enough to partially protect the planting investments at the Treatment site, but there may have been a much higher survival rate of planted trees and shrubs had exclosures been more effective. The difficulty of fence repair and repeated fence failures led to the fence being removed in 2021.

Both reference sites showed signs of damage by deer browse. In our study, shrub frequency declined at all sites in both the midstory and groundlayer, despite the lack of disturbance from restoration work at the reference sites. Many dominant shrubs declined drastically at reference sites, and many native shrubs that were present but uncommon in 2012 were not encountered again in 2017. This includes the NYS Endangered (S1) shrub *Euonymus americanus,* which was present in one plot at the Reference site in 2012 but was not detected in 2017. *E. americanus* is sometimes called “deer candy” because they are so preferred by deer. Additionally, many native herbs also declined across all sites between sampling years, including *Dennstaedtia punctilobula, Maianthemum canadense,* and *Osmunda cinnamomeum*. *Maianthemum canadense*, in particular, is preferred by deer and is considered an indicator species for deer browse damage (Rooney 1997). The decline of this plant, along with other herbs, at reference sites also points to high deer browse pressure.

Taken together, these changes suggest that deer browse is a major driver of community composition at Buck’s Hollow both directly (by consuming planted individuals) and indirectly (by accelerating the spread of *M. vimineum*). Furthermore, our results from the reference sites suggest that even if the Treatment site matures to a native, closed canopy forest, as long as deer densities are above the carrying capacity of the forest, native species richness and forest regeneration are at risk without fencing. Thus, high densities of white-tailed deer present a long-term obstacle to forest restoration at Buck’s Hollow and elsewhere on Staten Island, whether or not they are successfully excluded from the restoration site.

Recommendations

Given our results, we have several recommendations for the future of this site, as well as other forest restoration projects in Staten Island.

First, NYC Parks should continue to monitor these sites at regular intervals to ensure that the canopy closes successfully. We believe it is critical to monitor the Treatment and Reference sites to detect any unforeseen issues that seriously threaten the ecological development of the site. Regular inspections by staff could identify emerging threats to these sites’ trajectories. Continued monitoring, for example through the Forest Management Framework’s Rapid Site Assessment protocol, will allow NYC Parks to assess whether this round of planting is more successful than the last. Additionally, future inspections and monitoring can continue to track introduced species expansions to assess whether the spread of invasive species such as *Microstegium vimineum* are a threat to successional development at the Treatment site.

Furthermore, deer and invasive species presented major challenges to this project. Considering our data on how these obstacles impacted the Treatment site in the first two years after restoration, we recommend:

- **Invest in high-quality deer fencing for reforestation efforts in Staten Island, at least until deer densities decline.** While deer fencing was not entirely effective in this project, it did allow some planted trees to survive. Our fencing standards have improved since this restoration project and we have moved away from the less durable materials used in this contract, so NRG has already learned from this project. It is critical that fence lines are adjusted for natural topographical gradients so that deer cannot jump into the exclosures.

- **Remove invasive plant species in a buffer around restoration sites to decrease propagule pressure after restoration.** Restoration sites are particularly vulnerable to nonnative plant invasions immediately following restoration, when there is greater light availability and less competition. If adjacent sites have large introduced and small native plant populations, then introduced plants may spread into Treatment sites quite easily. In this case, for example, the extremely damaging introduced grass *M. vimineum* may not have spread to Treatment sites if it had not been present at nearby sites prior to restoration. Some published studies on urban forest restoration have recommended removing invasive plant species in a buffer around restoration sites to improve the quality of propagules there (Sullivan et al. 2009). Although this strategy would be expensive and time-consuming, it may help to prevent site degradation in the years following restoration. Given limitations within our standard contracting specifications, this would have to be accomplished by in-house or volunteer crews and would likely need to be constrained to a 50-foot buffer around the restoration site for feasibility.

Return after restoration for regular introduced species removal. Additionally, regular introduced plant removal at restoration sites following the initial restoration effort has been shown to increase native plant abundance (Vidra et al. 2007) and diversity (Simmons et al. 2015, Johnson and Handel 2019). Introduced species that were removed from the Treatment site did not return. As discussed, however, the introduction and expansion of *M. vimineum* may have long-term impacts on ecosystem processes at Treatment, even after the canopy closes. In order to ensure that the plant community at this site converges towards the healthy, mature Reference site, it may be helpful to remove this species, and other introduced species, at regular intervals following restoration.

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Appendix A

Planting Palettes and Seed Mix used in Buck's Hollow Restoration

Trees and Shrubs planted	**Quantity**
Acer rubrum	92
Amelanchier arborea	425
Celtis occidentalis	244
Cercis canadensis	50
Clethra alnifolia	198
Cornus amomum	527
Cornus racemosa	300
Diospyros virginiana	310
Eubotrys racemosa	230
Ilex verticillata	99
Juglans nigra	390
Lindera benzoin	250
Liquidambar styraciflua	204
Magnolia virginiana	50
Nyssa sylvatica	753
Photinia melanocarpa	50
Photinia pyrifolia	54
Platanus occidentalis	460
Quercus alba	136
Quercus bicolor	400
Quercus coccinea	777
Quercus ilicifolia	249
Quercus macrocarpa	152
Quercus muehlenbergii	101
Quercus palustris	202
Quercus phellos	150
Quercus prinus	442
Quercus stellata	646

Quercus velutina	181
Rhus copallina	50
Rosa palustris	125
Rubus pensylvanica	125
Sambucus canadensis	275
Sassafras albidum	176
Viburnum dentatum	300
Grand Total	**9173**

Herbaceous seed mix	**Lbs. Per Acre**	**Total pounds**
Asclepias incarnata	0.45	1.6
Eupatorium fistulosum	0.45	1.57
Euthamia graminifolia	1.35	4.72
Panicum virgatum	2.4	8.4
Schizycharium scoparius	2.4	8.4
Solidago juncea	0.4	1.4
Solidago nemoralis	0.4	1.4
Solidago rugosa	1	4.5
Sorghastrum nutans	2.4	8.4
Symphyotrichum laeve	0.4	1.4
Symphyotrichum novae-angliae	0.55	1.9
Symphyotrichum pilosus	0.4	1.4
Tridens flavus	2.4	8.4

Appendix B

Deer fencing specifications for the Buck’s Hollow Restoration. (to be added to finalized PDF: Deer Fence from CNYG-1512M.pdf)