LONG-TERM OUTCOMES OF ECOLOGICAL RESTORATION AND MANAGEMENT IN URBAN FOREST PATCHES

by

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ABSTRACT OF THE DISSERTATION

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AND MANAGEMENT IN URBAN FOREST PATCHES

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Urbanization transforms ecological systems, altering soils, hydrology, climate, species pools, and landscape patterns. Municipalities are turning to ecological restoration of urban forests to provide essential ecosystem services. This dissertation examines long-term effects of ecological restoration of forest patches invaded by woody invasive plants within urban park natural areas in New York City, New York, USA. I compared invaded sites where restoration was initiated 15-20 years prior with similarly invaded urban park forests that had not been restored. Significantly lower invasive species abundance, more complex vertical forest structure, and greater native tree recruitment indicated that invasive species removal followed by planting resulted in divergent successional trajectories and achievement of the central goals of the restoration. However, regenerating species indicated novel future assemblages, and restored sites varied in degree of reinvasion. To examine sources of this variability and test the importance of management effort to success of ecological restoration in urban forest remnants, I compared plant communities, management records, indicators of disturbance, and site characteristics among and between restored and unrestored invaded patches and a less-disturbed urban forest remnant. Differences among restored plant communities were associated with total restoration effort and with soil surfaces impacted by urban conditions, indicating the importance of urban context and ongoing management effort to outcomes of ecological restoration in urban areas. To examine these soil effects and to test whether impacts of urbanization on soils affect long-term outcomes of ecological restoration in urban forest patches, I compared plant community composition of restored, unrestored and less-disturbed sites with soil physical and chemical characteristics and urban soil classification maps. No single soil impact dominated effects on plant community composition, but all sites were impacted by anthropogenic factors known to reduce plant growth, change distributions of soil biota, and alter nutrient cycles. I present an urban perspective on the use of succession theory in ecological restoration and introduce adaptive successional phasing as a tool, emphasizing site-specificity, long-term processes, and the importance of the urban environment's effects on soils, species pools and disturbance regimes, and suggest that native species persisting and thriving in cities should be used in urban ecological restoration.

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INTRODUCTION

More than half of all human beings now live in cities, and both urban populations and urban land cover are expected continue to increase (United Nations 2012). As cities and their proportion of global land cover expand, improving the ability of urban land to provide ecological benefits is increasingly critical. Urbanization introduces a suite of stressors to ecological systems, transforming the biophysical landscape (Williams et al. 2009; Gaston 2010; Niemelä et al. 2011). Urban landscapes are heterogeneous, fragmented, and frequently disturbed (Cadenasso, Pickett, and Schwartz 2007). Their streams are ditched, buried and rerouted, their soils disturbed, dumped on and built over (Walsh et al. 2005; Ehrenfeld 2004), their climates and hydrologies altered (Niemelä et al. 2011). Shifts in social and cultural patterns subject urban soils to numerous physical and chemical changes, from excavation and sealed surfaces to introduced soil biota and atmospheric deposition of pollutants, resulting in urban soils that are more spatially and temporally variable than non-urban soils (Effland and Pouyat 1997; Cadenasso, Pickett, and Schwartz 2007). Cities are also sites of frequent species introductions, with high proportions of non-native and invasive species (Sukopp, Hejný, and Kowarik 1990).

These urban effects alter ecosystem patterns and processes (McDonnell et al. 1997; Bolund and Hunhammar 1999; Alberti 2005; Pouyat et al. 2002), and cities fail to provide many ecosystem services (United Nations 2012; Bolund and Hunhammar 1999). Many of these services, such as food and building materials, can be imported, but not all ecosystem services can be outsourced. Clean air, local climate, and other amenities of urban green space must be provided at the local level. In this context, fragments of remnant or regenerating habitat become disproportionately important to local biodiversity, habitat, psychological wellbeing, climate amelioration, and other environmental benefits (Barton and Pretty 2010; Ehrenfeld 2000; Sadler et al. 2010).

Recognizing that some ecosystem services must be provided at the local level, an array of strategies are being developed to provide essential environmental benefits to urban residents, from risk assessment and ecosystem service valuation to preservation and restoration. Municipalities are turning to the restoration of urban forests for their potential to provide multiple benefits such as pollutant filtration, urban heat island amelioration, social and health benefits, and carbon sequestration (Bolund and Hunhammar 1999; Gaston, Davies, and Edmonson 2010; Alfsen, Duval, and Elmqvist 2011). Strategies include both trees along streets and patches of original or re-grown habitat in the urban matrix. These habitat fragments may be in remnant protected areas with relatively intact original soils, or in places with long histories of repeated transformation. Restoration efforts often focus on reestablishment of native plant communities, both in areas devoid of plant cover and in existing patches invaded by exotic species. To understand the factors influencing outcomes of ecological restoration, and to improve the environmental health of cities, urban restoration efforts need to be evaluated.

Examining a Restoration Legacy

In New York City, forest restoration and reforestation have been adopted as a measure to address urban environmental problems (City of New York 2007). The New York City Department of Parks and Recreation's Natural Resources Group (NRG) has been engaged in ecological restoration of urban forest fragments since its formation in 1984

(NRG 1991; Sisinni and Anderson 1993). The New York City Department of Parks and Recreation manages over 29,000 acres (11,700 ha) of land, comprising 14% of the area of New York City. This includes more than 10,000 acres (4,000 ha) of forest, woodland, freshwater wetland and salt marsh ecosystems managed by NRG (Natural Resources Group 2013). Early NRG forest restoration efforts focused on areas invaded by a suite of woody exotic species including porcelain berry (*Ampelopsis brevipedunculata*), oriental bittersweet (*Celastrus orbiculatus*), and multiflora rose (*Rosa multiflora*). These studies examine the fate of forests invaded by these species that were the focus of NRG's initial efforts to improve the health of mixed oak upland forests in parks by ecological restoration.

Organization of the Dissertation

To test whether native forest structure, composition and successional trajectories can be restored in urban forests by removing woody invasive species and planting native trees, I assessed current ecological conditions in restored areas, as expressed in their plant communities 15-20 years after initiation of restoration activity (Chapter 1). I compared restored forests with other New York City Park forests that were invaded but not restored, and predicted that successful restoration would result in differences in community composition and structure between restored and unrestored sites. If removal and planting were successful, I expected to see reduction in targeted invasive species, a higher proportion of native tree species, a more complex physical structure, naturally-regenerating seedlings and saplings of native woody species in restored sites compared to sites that were invaded but not restored. These differences would suggest divergent successional trajectories resulting from restoration.

To test the hypothesis that effects of urbanization and management effort are important to long-term outcomes of ecological restoration in urban environments, I compared plant community outcomes to site characteristics and a 20-year database of restoration management activity (Chapter 2). I predicted that management frequency following the initial restoration would be important to plant community composition.

Given the importance of soil to conditions for plant germination, establishment, and growth, and evidence that urban environments affect soils, I tested whether effects of urban environments on soils affected restoration outcomes by comparing plant community composition with both field-collected soil samples and recent New York City soil maps that identify both natural and anthropogenic parent materials (Chapter 3). I predicted that soils of urban forest patches would exhibit effects of urban environmental conditions, and that these effects would be associated with differences in restoration outcomes.

The concluding chapter discusses recommendations for management based on the findings of these studies. I outline a possibility-centered approach to urban restoration focusing on urban soils, species pools, and disturbance regimes. I conclude by describing adaptive successional phasing, an approach to ecological restoration using time as a tool.

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CHAPTER 1

Long-term outcomes of ecological restoration in urban park forests

ABSTRACT

Local environmental quality is an urgent concern in urban areas, as global urban populations and urban land area are projected to double by 2050. Municipalities are turning to ecological restoration of urban forests as a measure to improve air quality, ameliorate urban heat island effects, improve storm water infiltration, and to provide other social and ecological benefits. This study examines the long-term effects of restoration undertaken in New York City, NY, USA to restore forests in urban park natural areas invaded by woody invasive plants. In 2009 and 2010, I sampled vegetation in 30 invaded sites in 3 large parks that were restored in the early 1990s, and 30 sites in 3 large parks that were similarly invaded but had not been restored. This restoration project achieved its central goals. After 15-20 years, vegetation composition and structure indicated that invasive species removal followed by planting resulted in persistent structural and compositional shifts, significantly lower invasive species abundance, a more complex forest structure, and greater native tree recruitment. Together, these findings indicate that successional trajectories of vegetation development have diverged between restored forests and invaded forests that were not restored. However, these findings also suggest that future composition of these urban forest patches will be novel assemblages in both cases. I present an urban perspective on the use of succession theory in ecological restoration, and present adaptive successional phasing as a new approach to planning ecological restoration in the highly disturbed and heterogeneous urban environment. By anticipating urban disturbances

and ecological succession, management can be targeted as ecological processes unfold. Models of ecological restoration developed in more pristine environments must be modified for use in cities. An urban approach to ecological restoration must value existing habitats in order to preserve and enhance urban biodiversity for both short-term benefits and long-term sustainability.

KEYWORDS

Ampelopsis brevipedunculata, *Celastrus orbiculatus*, adaptive management, ecological restoration, ecological succession, invasive species, New York City, novel assemblage, *Rosa multiflora,* urban ecology, urban forest, urban parks

INTRODUCTION

Societies profoundly change ecosystems to fulfill human needs and desires, and in few places are these changes more pervasive and persistent than in cities. At a landscape scale, urban ecosystems are heterogeneous, fragmented, and frequently disturbed compared to the ecosystems they replace (Cadenasso, Pickett, and Schwartz 2007). Their streams are ditched and buried, their soils disturbed, dumped on and built over (Walsh et al. 2005; Ehrenfeld 2004). Plant and animal communities are extirpated and new species introduced, pollutants accumulate and the atmosphere is heated. These and other stressors alter ecosystem patterns and processes (McDonnell et al. 1997; Bolund and Hunhammar 1999; Alberti 2005; Pouyat et al. 2002). As a result, fragments of remnant or regenerating habitat become disproportionately important reservoirs of biodiversity and sources of valuable ecological functions (Ehrenfeld 2000). As the number and size of cities increases rapidly across the globe, an array of strategies is being developed to provide essential environmental benefits to the world's urbanizing population, including risk assessment, ecosystem service valuation, habitat preservation, and ecological restoration. Many municipalities are turning to the restoration of urban forests for its potential to provide multiple benefits such as pollutant filtration, urban heat island amelioration, social and health benefits, and carbon sequestration (Bolund and Hunhammar 1999; Gaston, Davies, and Edmonson 2010; Alfsen, Duval, and Elmqvist 2011). These strategies include both trees along streets and patches of original or re-grown habitat in the urban matrix.

The ability of a forested fragment to provide these benefits depends upon the effectiveness of restoration treatments. The effectiveness of restoration treatments is in turn partly dependent on the applicability of the models used to develop the treatments to the system in which they are applied. The urban matrix of residential, industrial, and infrastructural uses surrounding these fragments provides highly contrasting conditions that limit dispersal and alter species pools (Grimm et al. 2000). Habitat fragments in urban landscapes tend to be small, and their high edge-to-interiors make them permeable to the frequent species introductions that are common in urban areas (Cadenasso 2001; Harper et al. 2005; Soulé 1991). Models of forest processes developed for more pristine environments may do a poor job of predicting the outcomes of ecological interventions in the altered urban environment, and goals set using these models may lead to unintended outcomes (Carreiro and Tripler 2005). Ecological restoration interventions in cities need to be evaluated to improve the quality of their outcomes and to test their underlying assumptions. In this study I examine the long-term fate of some of the first municipal-scale efforts to restore forest fragments in urban parks.

Ecological restoration is an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability, with the aim of returning a system to its historical trajectory toward resilience and ability to recover from disturbance (SERI 2004; Falk, Palmer, and Zedler 2006). Since ecological restoration seeks to direct change in the structure and composition of communities of organisms over time, ecological restoration is also the practice of directing ecological succession. The factors directing trajectories of ecological succession are altered in urban environments (Pickett and McDonnell 1989; Luken 1990; Pickett, Meiners, and Cadenasso 2011). These urban factors include multiple, frequent disturbances differ in type and magnitude from historic disturbance regimes (Alberti 2005), resulting in heterogeneous patches with different biological legacies. The organisms available to colonize these disturbed sites may differ in both their identity and abundance, and include many non-native, introduced species (Sukopp, Hejný, and Kowarik 1990; Kowarik 2008). Modified urban soils change the stage on which competitive interactions between these novel assemblages of organisms play out (Pavao-Zuckerman 2008). Under all of these changed conditions, trajectories of composition and function may be quite different from those of more intact habitats; thresholds may have been crossed in soil properties, species abundance, or other ecosystem properties such that interactions or species essential to previous patterns of recovery from disturbance are absent, and novel feedbacks may result in the persistence of alternative stable states (Suding, Gross, and Houseman 2004). Interactions between fast, individual and population-level processes and slow, regional or larger-scale changes in the environment may further influence these trajectories of change (Pickett and McDonnell 1989).

Ecological restoration, as an effort to reassemble properties and functions of dynamic natural systems, provides opportunities to test ecological theory (Bradshaw 1987). In urban environments, ecological restoration has the potential to inform understanding of how the constraints and feedbacks of the urban environment affect ecological processes like successional trajectories (Suding and Gross 2006). However, few restoration efforts are evaluated to understand how treatments relate to outcomes (Benayas et al. 2009). To better understand the factors influencing the outcomes of urban ecological restoration, and to improve the environmental health of cities, urban restoration efforts need to be evaluated. Here, I examine the outcomes of ecological restoration of forest patches in New York City parks after more than 15 years.

Study Context

New York City (40°47' N, 73°58' W) is located at the southeastern tip of the State of New York. The geologic and geographic diversity of its location allows for the presence of a high diversity of plant species for its area; 60% of plant species ever recorded in the State of New York have been recorded in New York City (DeCandido, Muir, and Gargiullo 2004). It is bordered by the Atlantic Ocean, Long Island Sound, and the estuary of the Hudson River. The city is home to 8.3 million people in an area of 302 square miles (78,217 ha), at a density of more than 27,000 people per square mile (106/ha). The metropolitan area, which encompasses parts of three states, is the United States’ largest at 19.8 million people (U.S. Census Bureau 2013).

New York City's climate is characterized by cold winters and warm, humid summers. The average annual air temperature (1981-2010) is 12.7 °C, with average seasonal temperatures in summer (June 1 – August 31) of 24.2 °C and 1.7 °C in winter (December 1 – February 28). The average annual growing season is 200 days (NYS Climate Office 2013). Annual average precipitation is 127 cm, distributed evenly throughout the year, with an average annual snowfall of 65 cm (NOAA 2013a; NOAA 2013b). The city is subject to urban heat island climate effects, with increased average and nighttime temperatures and decreased wind velocity (Bornstein 1968; Childs and Raman 2005).

The original topography and soils of New York City were shaped by glaciation. Striated bedrock outcroppings serve as foundation for skyscrapers, and erratic boulders are found throughout the city. The terminal moraine of the Wisconsonian glaciation forms the spine of Long Island, on which the boroughs of Queens and Brooklyn are located. Knob-and-kettle glacial topography is typical of many upland areas, with outwash plains on the eastern side of the terminal moraine (Greller 1972; Kieran 1959).

New York City retains 57% of its native plant species; 779 native species persist of the 1357 ever recorded in the city. The borough of Staten Island contains the greatest number of plant species (921), and Brooklyn the fewest (621). Current species are 37.7% non-native. Since the mid-19th century, 46% of New York City's native herbaceous species and 23% of its native woody plant species have been extirpated, while non-native species have been extirpated at lower rates. These losses include species within parks and other areas protected from development (DeCandido, Muir, and Gargiullo 2004).

Forests are an important habitat type in the undeveloped areas of New York City Parks. The forests of the region have been classified as eastern mixed hardwood, mixed mesophytic oak, mixed mesophytic types in the oak-chestnut region's glaciated section of the eastern deciduous forest, and red oak forests (Lefkowitz and Greller 1973). Preceding European settlement and urbanization, the uplands of the New York City area were covered by a mixed hardwood forest dominated by oaks (*Quercus spp.*), hickories (*Carya spp.*), and chestnut (*Castanea dentata*), with maples (*Acer spp.*), ashes (*Fraxinus spp.*), cherries (*Prunus spp.*), sweetgum (Liquidambar styraciflua) and tulip trees (*Liriodendron tulipifera*) (Sisinni and Emmerich 1995; Greller 1972; Loeb 1987). Loss of chestnuts (*Castanea dentata*) following the introduction of a fungal pathogen (chestnut blight, *Cryphonectria parasitica*) affected forest composition throughout the area (Greller 1972; Lefkowitz and Greller 1973). Of the city's ca. 12,000 ha of park land, which comprises 14% of the city's total land area, ca. 4,500 ha is not in active recreation, roads, or other intensive use. Of this undeveloped land, 71% was forested in 1995 (Sisinni and Emmerich 1995). PLANYC reforestation initiatives aim to increase the proportional cover of urban forest at the municipal scale by 2030 (City of New York 2007).

New York City Parks

The New York City Department of Parks and Recreation (NYC Parks) manages more than 29,000 acres (11,700 ha) of land comprising 14% of the area of New York City.

Federal Gateway National Recreation area, and land owned by the New York State Department of Environmental Conservation and NYS Department of Parks add an additional 3683 ha of designated parkland, for a total of 17% of New York City's land area. Approximately 25% of this land is managed for the conservation of its flora and fauna (DeCandido, Muir, and Gargiullo 2004). Land managed by the NYC Parks Department includes more than 10,000 acres (4,000 ha) of forest, woodland, freshwater wetland and salt marsh ecosystems. In Manhattan and the Bronx, the parks with the largest areas of natural vegetation were established in the 1800s; in Queens, Brooklyn and Staten Island they were set aside in the 20th century. Significant portions many park natural areas were converted into landfills between the 1930s and the 1970s (DeCandido, Muir, and Gargiullo 2004). Following a period of fiscal crisis in the 1970s and 1980s that led to neglect of city parks, NYC Parks turned its attention to management of its wild areas (NRG 2013).

The NYC Parks Natural Resources Group (NRG) was established in 1984 to develop and implement management programs for the protection, acquisition and restoration of the City's natural resources. The Natural Resources Group has received recognition for its urban ecological work, including awards from the Society for Ecological Restoration, the Nature Conservancy and the U.S. Environmental Protection Agency. The Natural Resources Group conducts ecological restoration of forests, salt marshes, riparian zones, meadows and other habitat types in all five of the city's boroughs (NRG 2013).

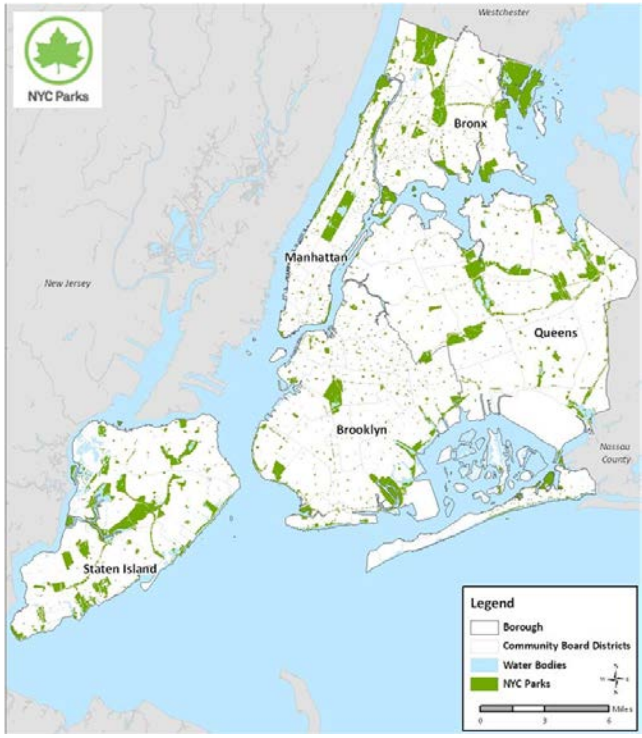

Figure 1.1: Lands managed by the New York City Department of Parks and Recreation. Map: Craig Mandel, NYC Parks Natural Resources Group.

Early Ecological Restoration of New York City Forests

Restored areas examined in this study were among the first forest restoration efforts made in New York City following a baseline vegetation inventory of parks with the largest areas of natural habitats. At the time, natural habitat areas were represented on maps as green polygons much like the map in Figure 1.1 above. Following a city-wide cover type analysis using aerial photography, detailed mapping was done of parks with more than 10 acres (4 ha) of natural vegetation. Visually distinct vegetation patterns in aerial photographs were delineated and numbered. Each demarcated area was then visited, and the dominant plants and site characteristics within each were described. The vegetative associations or formations described were called “entities,” and the technique called “entitation” (Hunt 1988; Sisinni and O’Hea Anderson 1993; Sisinni and Emmerich 1995). This technique, currently being repeated in several NYC parks, has subsequently been updated to include GIS mapping.

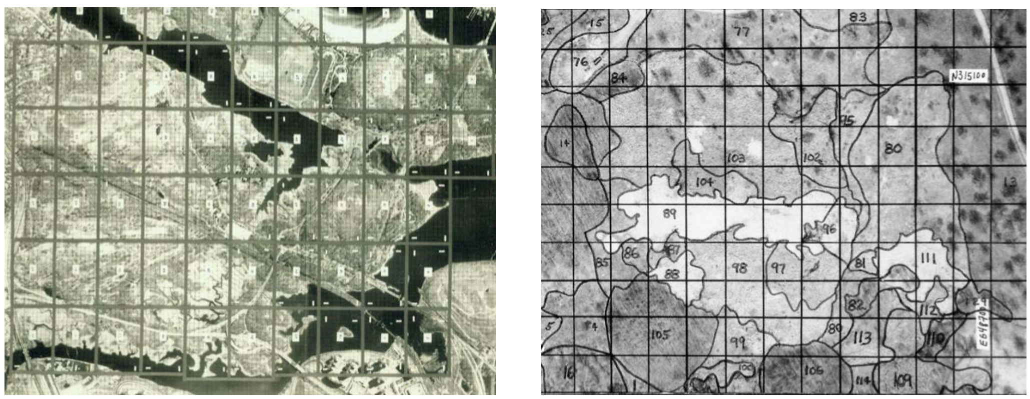

Figure 1.2: Initial vegetation mapping. Left: Pelham Bay Park, aerial photograph with grid. Right: vegetation associations mapped and numbered by visual assessment of finer-scale aerial photography.

Figure 1.3: Original invaded conditions. Top: *Ampelopsis brevipedunculata* and *Celastrus orbiculatus* dominated vinelands. Left: summer, Pelham Bay Park, 1992. Right: winter, Riverdale Park, 1991. Bottom: *Rosa multiflora* dominated understory, Pelham Bay Park. Left: winter; right: summer. *R. multiflora* occurred with and without invasive vine species. Photos: NYC Parks Natural Resources Group archives.

This initial vegetation mapping led to the discovery of high-quality habitat areas, many of which were subsequently designated Forever Wild Preserves (Feller 2012). It also alerted managers to the presence of large areas within park forests that had been invaded by non-native woody species, and to the ubiquity of abandoned, burned automobiles and related fire impacts in some areas. Many invaded areas were dominated by non-native, invasive shrubs and vines, with few standing trees (NRG 1991; Sisinni and Emmerich 1995).

Invasive plant species can fundamentally alter the structure, composition and dynamics of plant communities (Woods 1997). They prevent the persistence and establishment of many indigenous plants by outcompeting them, and dominant exotic species change the availability of resources in the environment (D'Antonio and Chambers 2006). They can form monocultures that tend to support a lower diversity of animal species, and long-term invasion can alter ecosystem properties (Vitousek 1990; Pimentel, Zuniga, and Morrison 2005). The effects of invasive species compound the impacts of habitat loss and fragmentation due to development and other land use change on remnant habitats in urban areas. Urban forest fragments are particularly vulnerable to exotic species invasion. Their extensive edges are often permeable to wind-dispersed seeds, are frequently disturbed by human activities, and are often bordered by horticultural areas where exotic species are introduced.

NRG began its first ecological forest restoration initiatives in 1985. The Urban Forestry and Education Program (UFEP), initiated in 1991, expanded this work. It addressed invasion of non-native woody plants in New York City Park forests, and long-term regeneration of native forests was a primary concern. These early restoration efforts focused on exotic invasive woody species porcelain berry (*Ampelopsis brevipedunculata*), oriental bittersweet (*Celastrus orbiculatus*), multiflora rose (*Rosa multiflora*), and Norway maple (*Acer platanoides*) (NRG 1991). Sites selected for restoration were dominated by invasive non-native woody plants and had little to no regeneration of native trees (Toth and Sauer 1994; NRG 1991). These sites included patches of native canopy trees with exotic invasive species understory (Figure 1.3, bottom row), patches with invasive species in both the canopy (e.g. *A. platanoides*) and understory, and many open vinelands (Figure 1.3, top row). In all sites, invasive exotic woody plants were the dominant species. These patches in the forested landscape ranged in size from 0.015 to 0.3 ha. NRG and Prospect Park Natural Resources staff removed woody invasive plants by manual, mechanical and chemical means, and native trees were planted in cleared areas (Figure 1.4). This study examines the long-term fate of these early forest restoration efforts, and compares them to invaded areas that were not restored.

Figure 1.4: Restoration treatments. Top: invasive species removal. Left: mechanical removal, Pelham Bay Park, 1992; right: foliar herbicide spray, Pelham Bay Park, 1992. Bottom: propagation and planting. Left: Van Cortlandt Park nursery, 1994; right: newly planted site, Pelham Bay Park, 1992. Photos: NYC Parks Natural Resources Group archives.

Study Sites

Pelham Bay Park

Pelham Bay is New York City's largest park. Located at the northeastern edge of the mainland, it borders suburban Westchester County, densely urban sections of the Bronx, and the Long Island Sound. The park contains remnant forest, fresh and saltwater marshes, 21 km of shoreline, and variety of uses including two golf courses, a landfill, a wildlife refuge, and a public beach. Native Americans used the site's coastal, fresh water, and upland forest resources. In the vicinity of Pelham Bay, as in the rest of the region, Native Americans managed forests using fire (Greller 1972). A 20,000 ha estate was established in 1654, and was later divided into smaller estates. Smaller properties were consolidated as a park in 1888 (NRG 1986). Common human disturbances over time have included agriculture and estate landscaping to landfilling and conversion for recreational use, followed by increased fire frequency due to arson in the 1970s and 1980s. These changes in social use of the area resulted in a patchwork of habitat types within unmanaged areas. Of the park's 840 ha of land, 360 ha has been developed for active recreation or highways. Of the remaining undeveloped 480 ha, 370 ha are forested, approximately two thirds. It was selected as the pilot site for the restoration efforts studied here for its size and its diversity of uses and habitats, which managers considered representative of City parks (Sisinni and Emmerich 1995). Initial vegetation mapping was initiated in 1986, and restoration activities began in 1987 (Sisinni and Emmerich 1995; NRG 1986). Both restored and invaded but not restored sites were sampled in Pelham Bay Park.

Prospect Park

Prospect Park, in central Brooklyn, straddles the top of the terminal moraine at its north end, and its south end is found on a glacial outwash plain. Bedrock is 100 m beneath gravel, till, loam and boulders deposited by glacial retreat. The forested central core of the park, Brooklyn's largest remnant forest, is located primarily on the hilly morainal uplands (Toth and Sauer 1994). The outwash plain was farmed by both Native Americans and later European settlers, while the uplands remained largely wooded. Widespread deforestation of Long Island during the Revolutionary War is likely to have impacted the park's forests, although the extent of clearing in the park area is unknown. In the mid-19th century when the park was established, the site was chosen for its mature woodlands. Frederick Law Olmsted and Calvert Vaux designed the park, and redesigned both hydrology and topography in many areas to achieve their vision, creating a flowing stream and lake system fed both by park runoff and by the New York City drinking water supply. Original park plantings included both native and non-native species, some of which (such as *Acer platanoides*) are now known to be invasive (Toth and Sauer 1994). Prospect Park has had a dedicated natural resource management staff since the 1980s. Evaluation and mapping of Prospect Park's natural areas in the mid-1980s found invasive species and widespread compaction and erosion due to trampling. Woodland restoration activities began in 1991.

Inwood Park

At the northern tip of the island of Manhattan, Inwood Hill Park's 79.5 ha contain salt marshes, forested uplands, and a glacial valley that contains many trees that exceed 200 years in age. The site of Inwood Hill Park was a site of Native American settlement, providing resources from both the Hudson and Harlem rivers. During the 17th century, European settlers established farms and a fort in the area that is now the park. In the 18th century, country estates, philanthropic institutions and a public library were found in what is now the park. Properties were bought by the Parks Department to form the park in 1916 (NRG 1989). During the Robert Moses period, an eight-lane highway was constructed through the park along the Hudson River. Salt marshes in the park have been drained, filled, and restored. To increase navigability for ships, the tidal waterway between Manhattan and the Bronx was widened. Vegetation mapping of Inwood Park was completed in 1989, and restoration activities were initiated in 1987 (NRG 1989).

Cunningham Park

Cunningham Park is located in central Queens, on western Long Island, on the Harbor Hill terminal moraine. Its 132 ha include knob-and-kettle areas that are seasonally wet, moist flatlands, and gently rolling uplands. The northern part of the park, where study sites were located, has well-drained rolling upland topography with a few dry kettles (Lefkowitz and Greller 1973). The shoreline resources of Little Neck and Flushing bays provided resources to Native Americans, who also farmed and hunted in the area. Dutch and then English colonists established settlements beginning in the 1600s, and most of the area was deforested during the Revolutionary War. Expansion of dense urban development into Queens came in the early 20th century. Cunningham Park was assembled from several parcels between 1928 and 1944. It contains the site of the nation's first automobile highway, which is now used as a pedestrian and bicycle pathway, and the park is connected via a former railway line turned parkland with three other large parks in Queens. In the1940s and 1950s, major highways were built within the park along its northern and southern borders and through the center (NYC Parks 2013). The vegetation of the park's natural areas was surveyed in 1988. Restoration was begun in Cunningham in 1994, but study sites were located in unrestored areas in the northeastern portion of the park, where mountain bike trails are the primary current use.

Van Cortlandt Park

Van Cortlandt Park is located in the central north Bronx, bordering Westchester county. European settlement of the area began in the 1630s, and through the 18th century the area of the park was used for farming, grain production, milling, and estates. In the 19 h century, two railroad lines were laid and the Croton Aqueduct was built through the park area to bring fresh water to New York City. Following the establishment of 440-hectare Van Cortlandt Park in 1888, development focused on the creation of recreational areas. In the 1940s and 1950s, three major highways were built through the center of the park, fragmenting it into distinct sections (NRG 1988). It currently contains a golf course and other recreational facilities, a stream and freshwater lake, a network of hiking trails, and 267 ha of woodlands, including designated high-quality habitat areas. Vegetation mapping of Van Cortlandt Park was completed in 1988, and forest restoration began in 1989. Sites sampled in this study were invaded but not restored.

Hypotheses

Can forest structure and native species dominance be restored in invaded urban forest patches by removing woody invasive species and planting native trees? If removal with planting is an effective restoration strategy, after 15-20 years the community composition and structure of restored forests should be different from invaded urban forest patches that were not restored in their species composition, structure, and amount of regeneration of native forest trees.

Species Composition

If removal and planting were successful, restored areas should not be dominated by the invasive species that were the target of the restoration, and should contain a higher proportion of native tree species. Reduction in the dominance of invasive exotic woody plants and establishment of native tree canopy were primary goals of the restoration project. After 15-20 years, lower abundance of targeted exotic species in restored sites than in unrestored invaded sites would indicate long-term effects of restoration. Greater native tree abundance in restored sites would likewise indicate that a primary species composition goal of the restoration had been met.

Forest Structure

In comparison to invaded forests that were not restored, restored forests should have greater canopy closure and a more complex forest physiognomy, with layers of herbaceous vegetation, understory shrubs, and trees (Barbour et al. 1998). Since restored sites had few mature trees immediately following restoration, due to either their location in open vinelands or due to removal of invasive trees, the development of a multi-layered canopy is not expected. Invaded sites that were not restored were expected to have fewer trees due to vine encroachment and resulting tree toppling. This restoration effort focused on increasing shade by increasing native tree canopy, on the premise that native forest plants would have increased competitive advantage in relation to shade-intolerant exotic invasive species under shade. Lower density of exotic vines and shrubs in the woody understory and greater native tree abundance in restored sites compared to unrestored sites would indicate effects of restoration on forest structure after 15-20 years.

Regeneration

Since ecological restoration aims to restore historic forest development trajectories with the eventual goal of a self-sustaining community of native plants (SERI 2004), restored forests should contain naturally-regenerating seedlings and saplings of native woody species. These represent the potential future of the forest, and would indicate at least partial restoration of historic trajectories. Greater prevalence of native seedlings and saplings in restored than in unrestored sites would also indicate attainment of a central goal of the restoration effort.

METHODS

In 2009, with the help of interns and volunteers, I sampled thirty sites in park forests, ten in each of three parks that had received restoration treatment in the 1990s. All of these sites were treated by removal of woody invasive species and addition of native tree seedlings. In 2010, I sampled park forests that were similarly invaded at the time of the original restoration, but were not restored. To capture forest strata, I sampled ground layer vegetation using line intercept transects; woody understory trees, shrubs and vines using stem counts; and the canopy by measuring DBH of all trees. I recorded a suite of site characteristics, including evidence of recent and historic disturbance, soil surface cover, and adjacent land uses. Data collection protocols were designed to expand existing long-term forest health monitoring (McDonnell, Rudnicky, and Koch 1989) by adding new site types and establishing additional locations for long-term monitoring. They were also designed to provide ecological data that would indicate successional trajectories (as indicated by community composition), effects of restoration practices, and the role of invasive species in restoration outcomes.

Site Selection and Establishment

Treated Restoration Sites

Records of restoration work completed in the 1990s by NRG and by Prospect Park Natural Resources staff were evaluated for data quality and completeness. Parks with the highest-quality extant data were selected as research sites: Pelham Bay Park in the Bronx, Prospect Park in Brooklyn, and Inwood Park in Manhattan. Paper records were converted to digital format. Within each park, ten plot locations were selected based on the quality and completeness of data describing conditions prior to restoration, restoration treatment, and post-treatment monitoring and management (Table 1.1). Where more than ten plots with sufficient data were available, plots were spaced as widely as possible within the restored area (Appendix 1D).

Table 1.1: Date of first restoration activity within the management unit, restored sites.

Park	Plot	First Restoration
Inwood	IN03	August 1991
	IN07	August 1991
	IN08	August 1991
	IN11	August 1991
	IN12	December 1993
	IN13	August 1991
	IN14	September 1991
	IN15	September 1991
	IN25	July 1988
	IN29	January 1991
Pelham Bay	PB03	April 1992
	PB06	June 1992
	PB07	June 1992
	PB08	April 1992
	PB09	April 1992
	PB10	April 1992
	PB13	July 1991
	PB93-1	April 1993
	PB93-2	October 1991
	PB93-3	October 1991
Prospect	PR0102	October 1993
	PR1102	September 1991
	PR1104	September 1991
	PR1105	September 1991
	PR1106	September 1991
	PR1603	September 1991
	PR1702	September 1991
	PR1803	October 1993
	PRRA01	October 1993
	PRRA02	October 1993

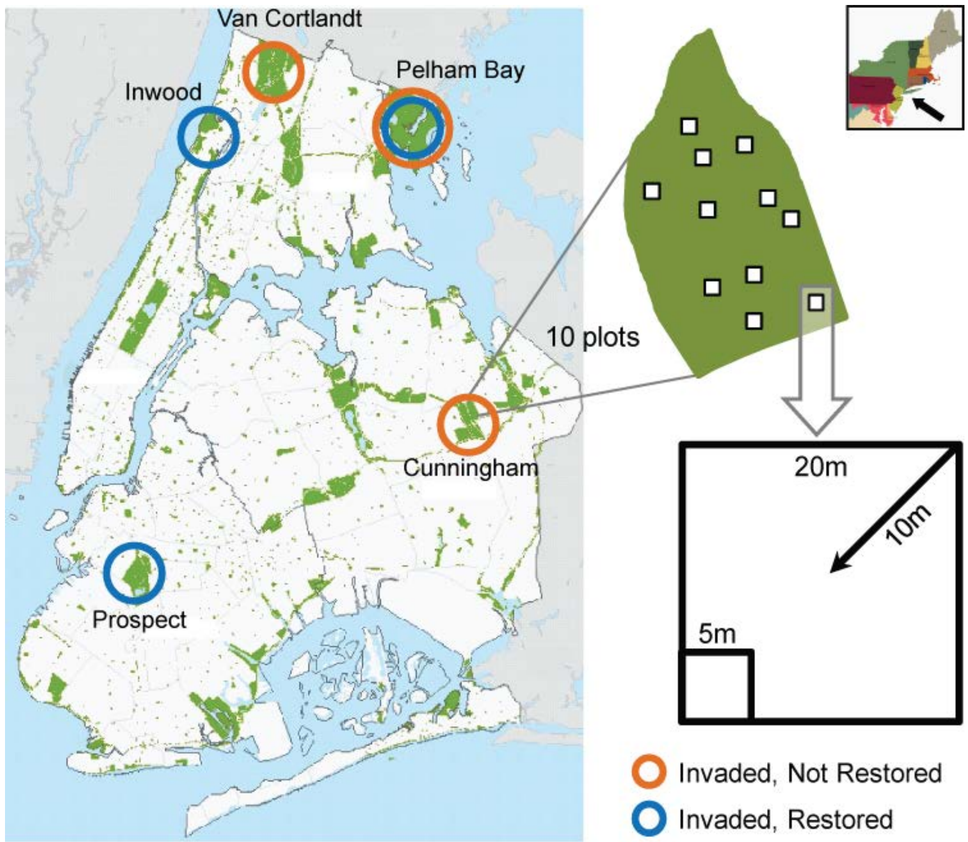

Figure 1.5: Study design. Ten restored or unrestored plots were sampled in each park. Pelham Bay, the city's largest park, had sufficient area of both treated and untreated forests to sample both (10 plots of each site type). Plot park locations: Inwood Park, Manhattan; Van Cortlandt Park, Bronx; Pelham Bay Park, Bronx; Cunningham Park, Queens, and Prospect Park, Brooklyn. Sampling design diagram after Cadenasso et al. (2007a). Lands managed by the New York City Department of Parks and Recreation are shown in green. Within each 20 m x 20 m plot, all trees were censused. Three 5 m x 5 m subplots were randomly located to sample shrubs, vines, and saplings. Four line transects were used to measure ground layer cover, extending 10 m from each plot corner toward the plot center.

Using restoration project records, historic and current Parks maps, topographic maps, and Brunton Safari/Nexus mirrored sighting compasses, plots were established in the location of original restoration treatment monitoring plots (Emmerich 1992; Toth and Sauer 1994; Handel and Kostel-Hughes 1999). In Prospect Park, plots had been monitored annually and plot corner markers replaced as needed, so all original plot corners were clearly marked in 2009. In Pelham Bay and Inwood parks, some plot corners had been marked with iron rebar or nails that had completely rusted, while others had surviving markers (aluminum cans that had been secured with iron nails, leaving behind a characteristic flattened soda can with a ~2 cm hole in the center). Where corner markers were not present, plots were established using surveying descriptions.

Original monitoring plots were 5 m x 5 m, 6 m x 6 m, or 5 m x 7 m in size (Emmerich 1992; Toth and Sauer 1994; Handel and Kostel-Hughes 1999); each 20 m x 20 m restored plot established for this study contained an original monitoring plot within its boundaries. Where corner markers were not present, plot perimeters were established in the cardinal directions. Where corner markers were present and the original plot was skewed to follow original plot boundaries, the skew was recorded. Plot locations were mapped and described, three or more landmarks were established for one anchor corner of each plot, and GPS coordinates of the anchor corner of each plot were recorded.

Four thirty-meter measuring tapes were stretched taut using compass bearings to establish the perimeter of the plot. In the case of large tree trunks in the perimeter line, diameter was measured and the line continued on the other side of the stem. Measuring tapes were stretched taut such that both ends of a tape were the same height above the ground at their termini; this height varied depending upon vegetation conditions (e.g. above a rose thicket, or below a shrub canopy). All corners were marked with a 30 cm length of 1 in (2.54 cm) diameter white PVC pipe that was hammered into the ground. Where it was not possible to put a corner stake into the ground due to a hard surface, the surface was marked with gold paint.

Untreated Comparison Sites

Unrestored, invaded comparison sites were selected and sampled in 2010. These plots were also established in New York City parks, in locations described as similarly invaded to areas restored under the UFEP program at the time of the original restoration work (McDonnell et al. 1997; McDonnell, Rudnicky, and Koch 1989), but which were not restored. Sites were selected in three parks: Pelham Bay Park and Van Cortlandt Park in the Bronx, and Cunningham Park in Queens.

Forested areas that had not received restoration treatment were first delineated by NRG managers with extensive knowledge of site histories (T. Wenskus and R. Love, pers. comm.). Vegetation mapping units within delineated non-restored areas were then compared with restoration treatment database records to exclude any restored units. Unrestored areas with no record of restoration treatment were then cross-referenced with the vegetation descriptions that were used to direct 1990s UFEP restoration activities (Natural Resources Group 1986; 1988; 1990). Mapping units in which UFEP target woody invasive species *A. brevipedunculata*, *C. orbiculatus*, or *R. multiflora* were listed among the dominant species were selected as locations for sampling. Where more than ten potential sites were available within a park, plots were spaced as widely as possible within the restored area. Plot perimeters were established in the cardinal directions (not adjusted for 12 degree 2010 declination) using the same protocol as the restored site plots. Where the shape of a mapping unit required the plot to be rotated from due north, skew was recorded.

Site Characterization

A suite of site characteristics was collected for each plot (Appendix 1B), including hydrologic features, soil surface cover, and human impacts. Cover of each feature was estimated in increments of 20%. Topography, animal activity, and adjacent land uses were described, and a narrative description of each plot was composed. Site history was compiled from maps, historical documents and interviews with land managers. Canopy closure was measured using a concave spherical densiometer, and slope and aspect of the plot were recorded using a clinometer and compass.

Vegetation Sampling

Vegetation sampling methods were designed for comparability with the Permanent Forest Reference Plot System established at the time of the original restoration work in intact New York City forests as part of a series of Urban-Rural Gradient Ecology studies (McDonnell et al. 1997; Pouyat, McDonnell, and Pickett 1997; Cadenasso et al. 2007a). Treated restoration plots were sampled in 2009, and comparison untreated and less disturbed sites were sampled in 2010. Ten plots were sampled in each of the three parks during the summer growing season, from June to August each year. All plots were revisited in early spring (April and May) and late fall (October) to record presence of spring ephemeral and late-flowering herbaceous species.

The basic unit of sampling was a 20 m x 20 m (400 m^2) square plot (Figure 1.5 and Appendix 1B). Sampling one plot required 3-8 hours of effort, depending upon the remoteness of the plot, density and height of woody vegetation, and number of field assistants (1 to 4). Establishing the plot perimeter with meter tapes was the most time-consuming aspect of the sampling. Where woody plant material had to be removed in order to access the plot, it was cut from outside the plot, creating a path around but not within the plot area. Within each 20 m x 20 m plot, vegetation was sampled by stratum to capture both community composition and structure.

Trees

All trees greater than 2.54 cm (1 in.) in diameter at breast height (DBH) and 1 m in height inside the 20 m plot were identified and their DBH was recorded. Multi-stemmed trees were measured below points of bifurcation and the height of the split was noted. Each tree was classified as canopy or sub-canopy.

Shrubs, saplings and woody vines

Three 5 m x 5 m subplots were randomly located inside each 20 m x 20 m plot. The species identity and number of stems of all shrubs, woody vines and tree saplings greater than 1 m in height within the subplots were recorded. Saplings were defined as individuals of tree species less than 2.54 cm DBH and more than 1m in height. Stems were considered to be inside the plot when the entire stem at ground level was within the delineated area.

Ground layer vegetation

Herbaceous plants and seedlings of woody species less than 1 m in height were sampled using four 1 cm wide, 10 m long line transects that were extended 45° from each corner of the plot toward the plot center. Centimeters of intercept of each species within each meter of the taut measuring tape were recorded in whole centimeters such that minimum intercept was 1 cm. Each species was counted separately; total cover for a meter with layered ground-layer vegetation could exceed 100 cm. Intercept of a species was considered continuous unless gaps between leaves exceeded 5 cm. Only live stems and leaves were counted. Where centimeters of a given meter were not vegetated, the character of the soil surface was recorded. Where woody vines and shrubs (usually targeted invasive species) covered the transect line greater than 1m in height, hand pruning was selectively used to cut away the top layer of leaves and stems so that ground-layer vegetation below 1m could be measured.

Many unrestored sites contained a sufficient density of woody and/or thorny stems (primarily of targeted invasive species) that it proved impossible to stretch measuring tapes in a straight line – or for a person to physically access the site – without moving or removing some of the vegetation. Hedge shears and hand pruning shears made sampling possible in these cases. Where vines, canes or branches of invasive species prevented establishment of the plot perimeter, they were selectively cut from outside the plot to provide access. When it was necessary to cut woody stems to make sampling possible within the plot (primarily *R. multiflora, C. orbiculatus, A. brevipedunculata* and occasionally *Rubus spp.*), care was taken not to cut stems that would be counted (e.g. perimeter shrubs more than 1m in height with stems originating in the plot).

To avoid influencing subsequent sampling within the plot by trampling, sampling of ground layer transects was done first, often while the perimeter was being established. Subplots were established next, and trees were sampled last. Effort was made to avoid unnecessary trampling and walking inside the plots. Where plots were covered with dense woody and thorny vegetation as described above, order of sampling was adjusted to ensure accurate counting of all stems before any stems were cut.

Immediately following sampling, each complete plot was inspected for additional species not recorded via one of the above methods. When previously unrecorded plants were found, they were identified and classified as common or uncommon within the plot.

Species Identification

Identification was done in the field and verified using at least two keys (Newcomb 1989; Clemants and Gracie 2006; Cope and Muenscher 2001; Martine 2002; Martine 2003; Graves 2011; Gleason and Cronquist 1991; Holmgren, Holmgren, and Gleason 1998). For plants that could not be identified in the field, specimens were collected of individuals outside the plot, and identification was verified by staff of the Brooklyn Botanic Garden, where voucher specimens are housed.

Data Analysis

I combined treatment databases maintained by NYC Parks NRG and Prospect Park Natural Resources staff into a relational MS Access database using Access 2007. These included a database of vegetation descriptions georeferenced to map units that were described prior to initial restoration, detailed NYC Parks Natural Resource Group restoration activity logs, and Prospect Park Natural Resources restoration activity log and long-term plot monitoring data. I gathered additional data describing initial restoration activities and early monitoring under the Urban Forestry and Education Program from hand-written and other sources in NYC Parks Department archives, and this data was entered in MS Excel and added to the database as tables. Field data collected for this study in 2009 and 2010 was also entered in Excel and added to the relational database.

Species were categorized by origin and management status for analysis. Species that were removed on more than 1,000 occasions during the period 1988-2009, as indicated by the number of dates on which removal was reported, were considered to be primary target species, and species that were removed on more than 100 occasions were considered secondary targets for removal. Species were considered native if their current or historic distribution includes the greater New York City metropolitan area (USDA 2013a; Weldy and Werier 2013; Burns and Honkala 2013), Species were categorized as potentially invasive if they were considered weedy or invasive in part of their U.S. introduced range (USDA 2013a) but were not primary or secondary target species. Species categorized as “other” included non-invasive exotic species not considered weedy or invasive in their U.S. introduced range (USDA 2013a), and plants that were not identifiable beyond the genus level, and/or that contained both native and non-native species within the genus (e.g. *Malus* sp.).

Differences between restored and unrestored plots were compared using nonparametric Wilcoxon (Mann-Whitney) signed ranks and Kruskal-Wallis tests, which use response rank scores to test similarity of means across groups where distributions are not equal, a common characteristic of ecological data. T-tests also assumed unequal variances. This analysis was done using JMP 10.0.0 (SAS Institute 2012).Outlier box plots representing data analyzed in this manner display the median (horizontal line in the box, the 50^{th} percentile), 25^{th} and 75^{th} percentiles (lower and upper box ends), and whiskers extend to the furthest data point within ±1.5x the interquartile range. Points outside the box and whiskers are considered outliers (SAS Institute 2012, Cary, NC).

Analysis of community composition was performed using CANOCO 4.5 (Ter Braak and Smilauer 2002), and resulting values for measures of diversity, richness, evenness, and variance were subjected to t-tests assuming unequal variances when restored and unrestored sites were compared. Wilcoxon/Kruskal-Wallis Rank Sums tests were used to examine trends among all site types (restored, unrestored, and when comparing all parks), with Each Pair Student's t tests or Tukey-Kramer HSD to compare means (α = 0.05 for all tests).

RESULTS

Forest Structure

Figure 1.6: Forest structure of unrestored and restored sites. Left, unrestored open condition dominated by *R. multiflora, A. brevipedunculata*, *C. orbiculatus* and native *Vitis spp.* in Cunningham Park, 2010. Right, restored forest with tree saplings in the understory, Prospect Park, 2010. These sites are typical of the most- and least-invaded conditions sampled.

Number of Trees

Restored sites contained a significantly higher number of tree stems per plot than sites that were invaded but not restored (p < 0.0001, Figure 1.7).

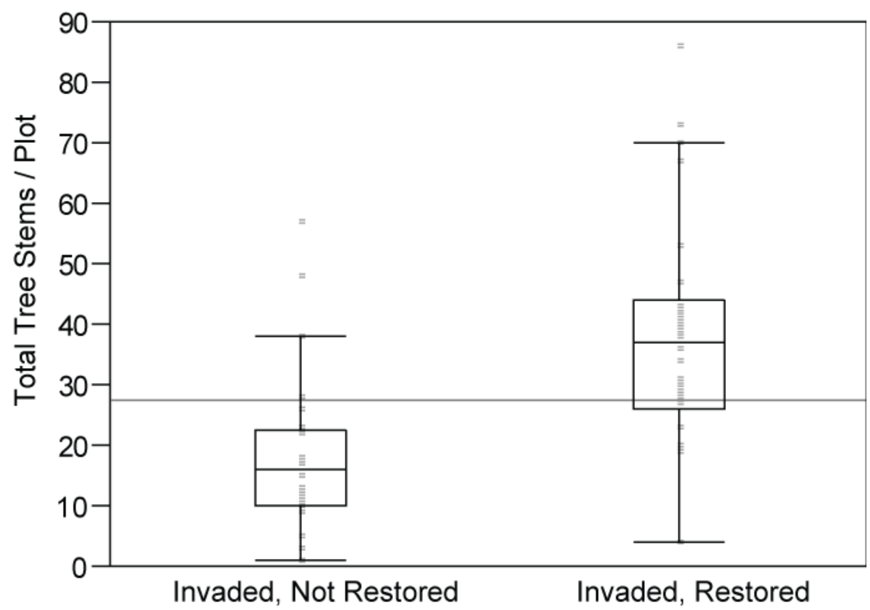

Figure 1.7: Total tree stems per plot in restored and unrestored sites. Restored plots had significantly more tree stems (p < 0.0001; Restored sites – mean: 37.9, range: 4-86; Unrestored sites – mean: 17.6, range: 1-57).

Structure of the Woody Understory

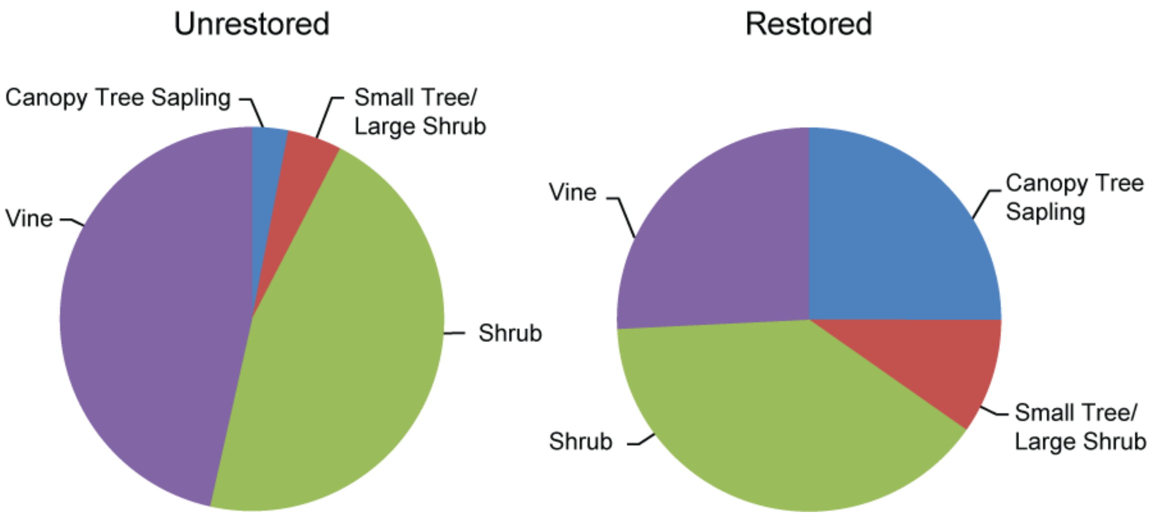

Figure 1.8: Proportion of woody understory stems by growth form in restored and unrestored sites.

Table 1.2: Total woody understory stems and proportion of all woody understory stems by growth form.

	Unrestored		**Restored**	
Growth Form	Total Stems	Proportion	Total Stems	Proportion
Canopy Tree	103	3%	405	25%
Understory Tree	154	5%	157	10%
Shrub	1555	46%	639	39%
Vine	1574	46%	417	26%
Total	3386	-	1618	-

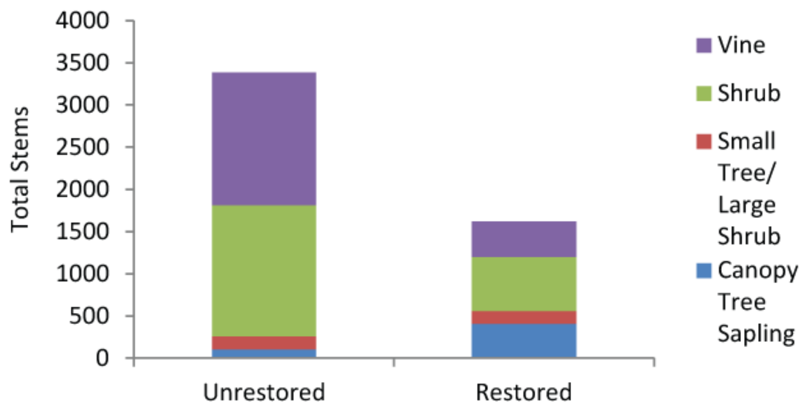

Figure 1.9: Total woody understory stems by growth form in restored and unrestored sites.

Restored and unrestored forests differed in the proportion of different growth forms in the woody understory (Figures 1.8 and 1.9, Table 1.2). Unrestored forest understories were dominated by vines and shrubs, while restored forest understories had a greater proportion of canopy tree seedlings and understory trees.

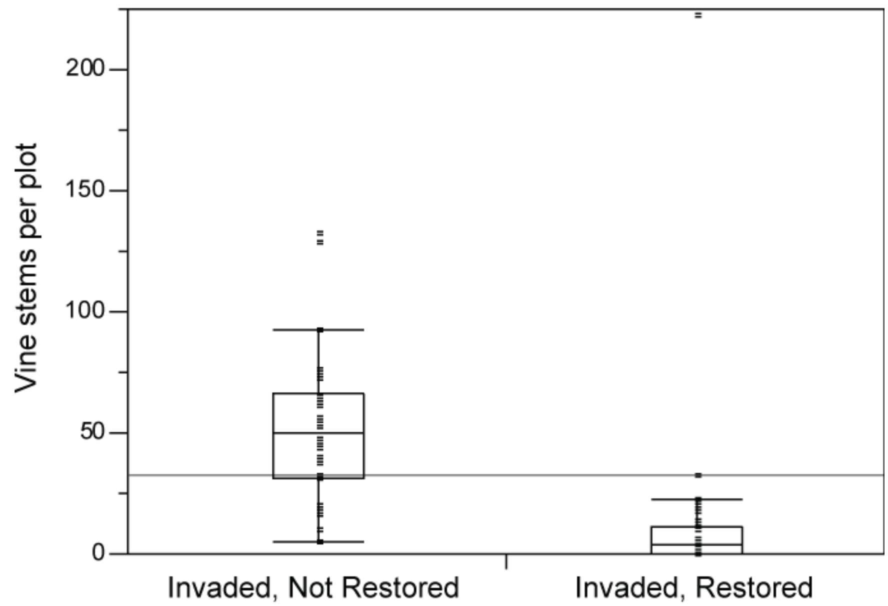

Figure 1.10: Woody understory vine stems per plot in restored and unrestored sites. Restored sites had significantly fewer vine stems ($p < 0.0001$; Restored sites – mean: 13.9, range: 0-223; Unrestored sites – mean: 52.5, range: 5-133). The site with the highest number of vine stems, restored site RA02 in Prospect Park, was predominantly covered by native *Toxicodendron radicans*.

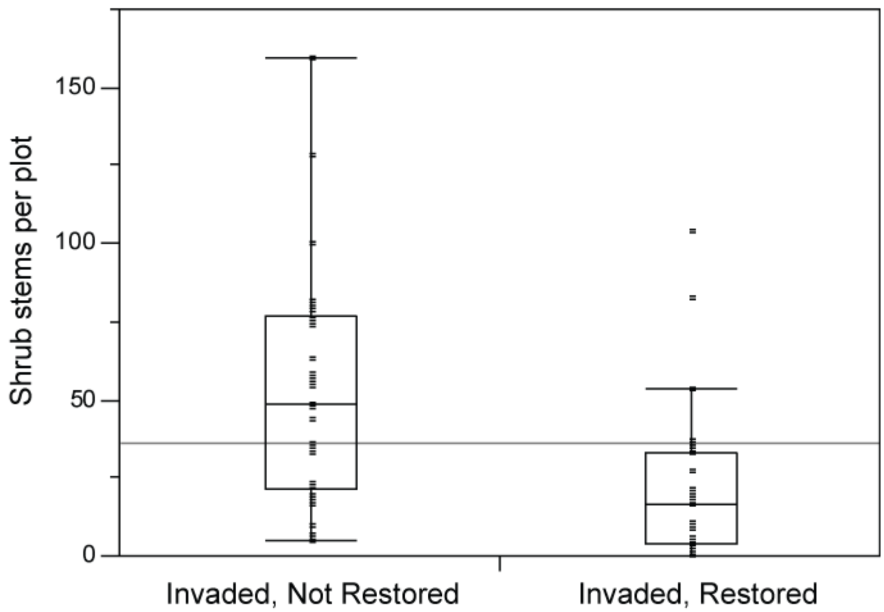

Figure 1.11: Woody understory shrub stems per plot in restored and unrestored sites. Restored sites had significantly fewer shrub stems ($p < 0.0001$; Restored sites – mean: 21.3, range: 0-104; Unrestored sites – mean: 51.8, range: 5-159). Restored sites with the highest number of shrub stems were Inwood Park plot 13, dominated by *Lindera benzoin*, and Pelham Bay 13, in which the majority of shrubs were *R. multiflora* and *Ligustrum* sp..

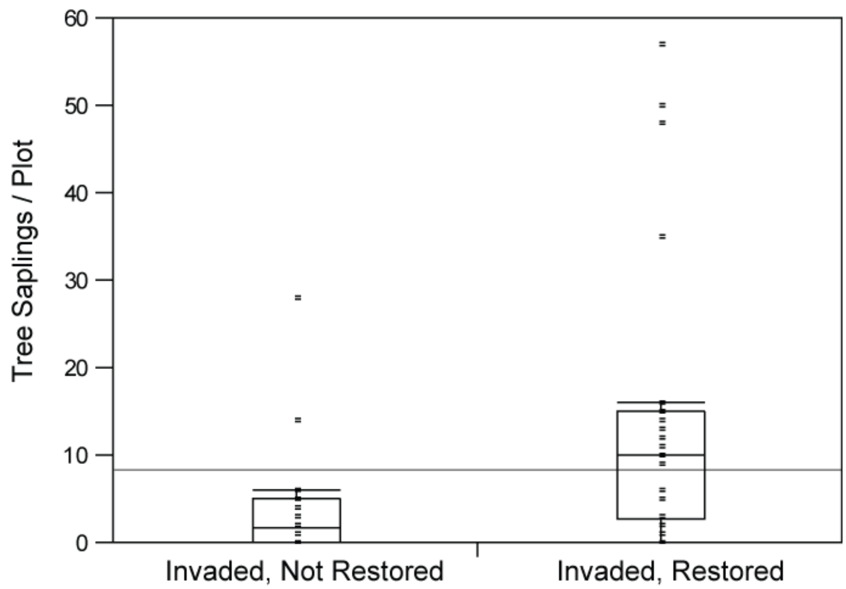

Figure 1.12: Woody understory tree sapling stems per plot in restored and unrestored sites. Unrestored sites had significantly fewer tree saplings (p < 0.0001; Restored sites – mean: 13.5, range: 0-57; Unrestored sites – mean: 3.4, range: 0-28).

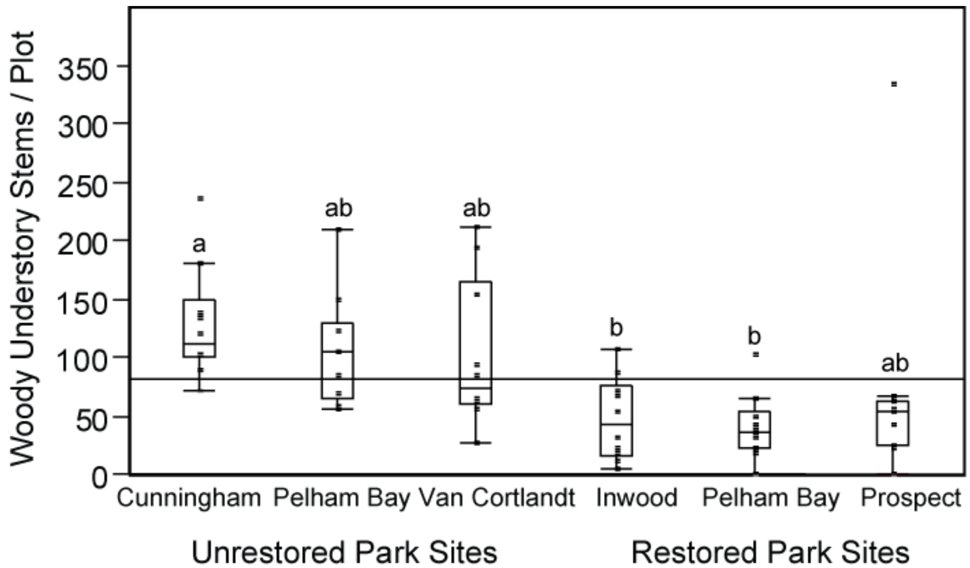

Figure 1.13: Total number of woody understory stems per plot by park. Restored parks combined had significantly fewer total woody understory stems per plot than sites that were not restored ($p < 0.0001$; Restored sites – mean: 53.9, range: 1-334; Unrestored sites – mean: 113.1, range: 27-237). Cunningham – mean: 128.7, range: 72-237; Pelham Bay unrestored – mean: 108.8, range: 56-211; Van Cortlandt – mean: 101.9, range: 27-231; Inwood – mean: 48.1, range: 6-108; Pelham Bay restored – mean: 41.6, range: 2-104; Prospect – mean: 72.1, range: 1-334. The site with the highest number of vine stems, restored site RA02 in Prospect Park, was predominantly covered by native *Toxicodendron radicans*.

Restored areas had fewer vine, shrub and total woody understory stems (Figures 1.10, 1.11 and 1.13). There were no significant differences in the total number of tree saplings > 1m in height per plot, but restored sites contained more canopy tree saplings than sites that were invaded and not restored; the sapling populations of restored sites had a higher proportion of native trees ($p < 0.0001$, Figure 1.11).

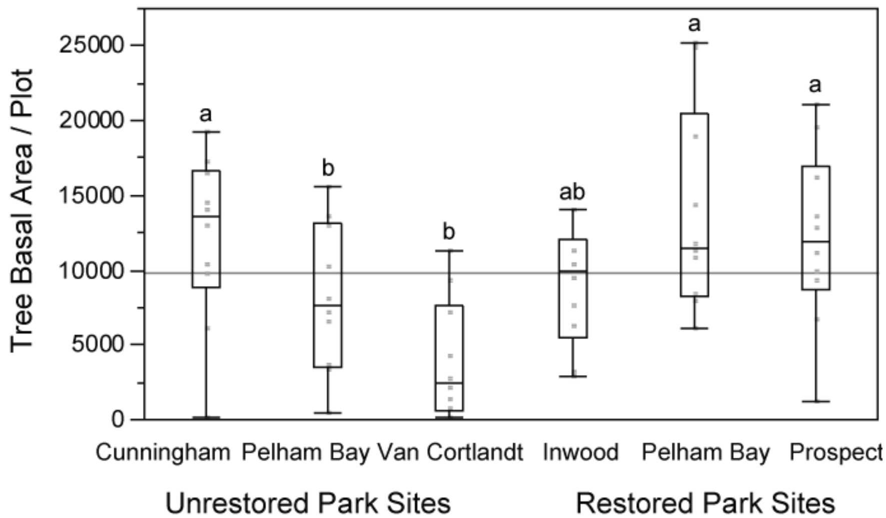

Figure 1.14: Total tree basal area (cm^2) per plot by park. Sites with different letters are significantly different (p = 0.0053). Cunningham – mean: 12103, range: 120-19227; Pelham Bay unrestored – mean: 8159, range: 398-15630; Van Cortlandt – mean: 3934, range: 189-11343; Inwood – mean: 9036, range: 2828-14007; Pelham Bay restored – mean: 13983, range: 6128-25138; Prospect – mean: 12132, range: 1192-21020.

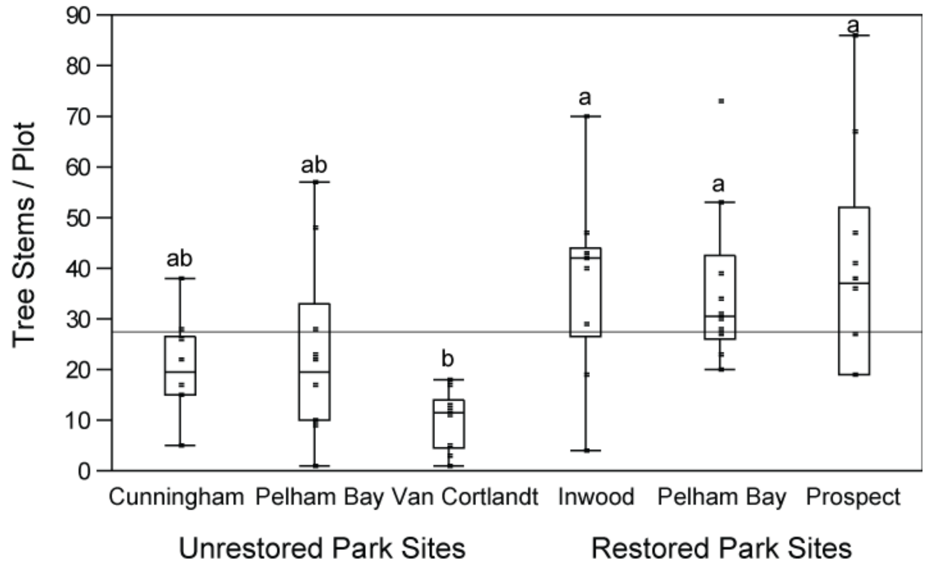

Figure 1.15: Number of tree stems per plot by park. Sites with different letters are significantly different ($p < 0.0001$). Cunningham – mean: 20.5, range: 5-38; Pelham Bay unrestored – mean: 22.5, range: 1-57; Van Cortlandt – mean: 9.7, range: 1-18; Inwood – mean: 37.9, range: 4-70; Pelham Bay restored – mean: 35.8, range: 20-73; Prospect – mean: 39.9, range: 19-86.

Ground Layer Cover

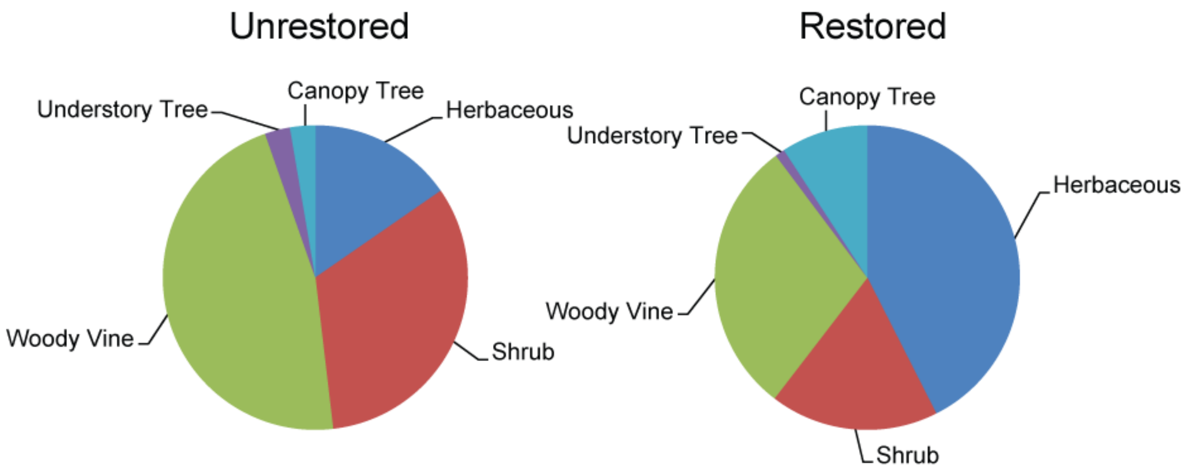

Figure 1.16: Proportion of ground cover per plot by growth form in restored and unrestored sites.

Woody vines and shrubs covered a greater proportion of the ground layer in unrestored sites, while herbaceous plants and canopy tree seedlings constituted a greater proportion of total ground cover in restored sites.

Table 1.3: Total ground layer cover and proportion of total ground layer cover by growth form.

	Unrestored		**Restored**	
Growth Form	Total cm	Proportion	Total cm	Proportion
Herbaceous	657	16.4%	1098	27.4%
Shrub	1404	35.1%	269	6.7%
Woody Vine	1992	49.8%	442	11.1%
Understory Tree	115	2.9%	15	0.4%
Canopy Tree	114	3%	138	3%
Total	4282	-	1963	-

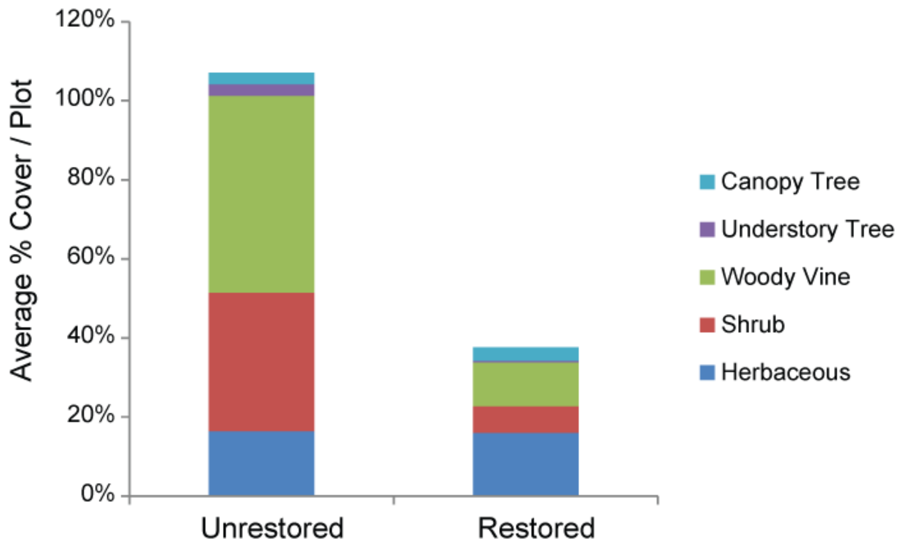

Figure 1.17: Average proportion by growth form of ground cover less than 1 m in height in restored and unrestored sites.

Unrestored sites had higher total ground layer cover (*p* < 0.0001). Differences between ground layer cover of restored and unrestored sites were primarily in the cover of woody vines and shrubs.

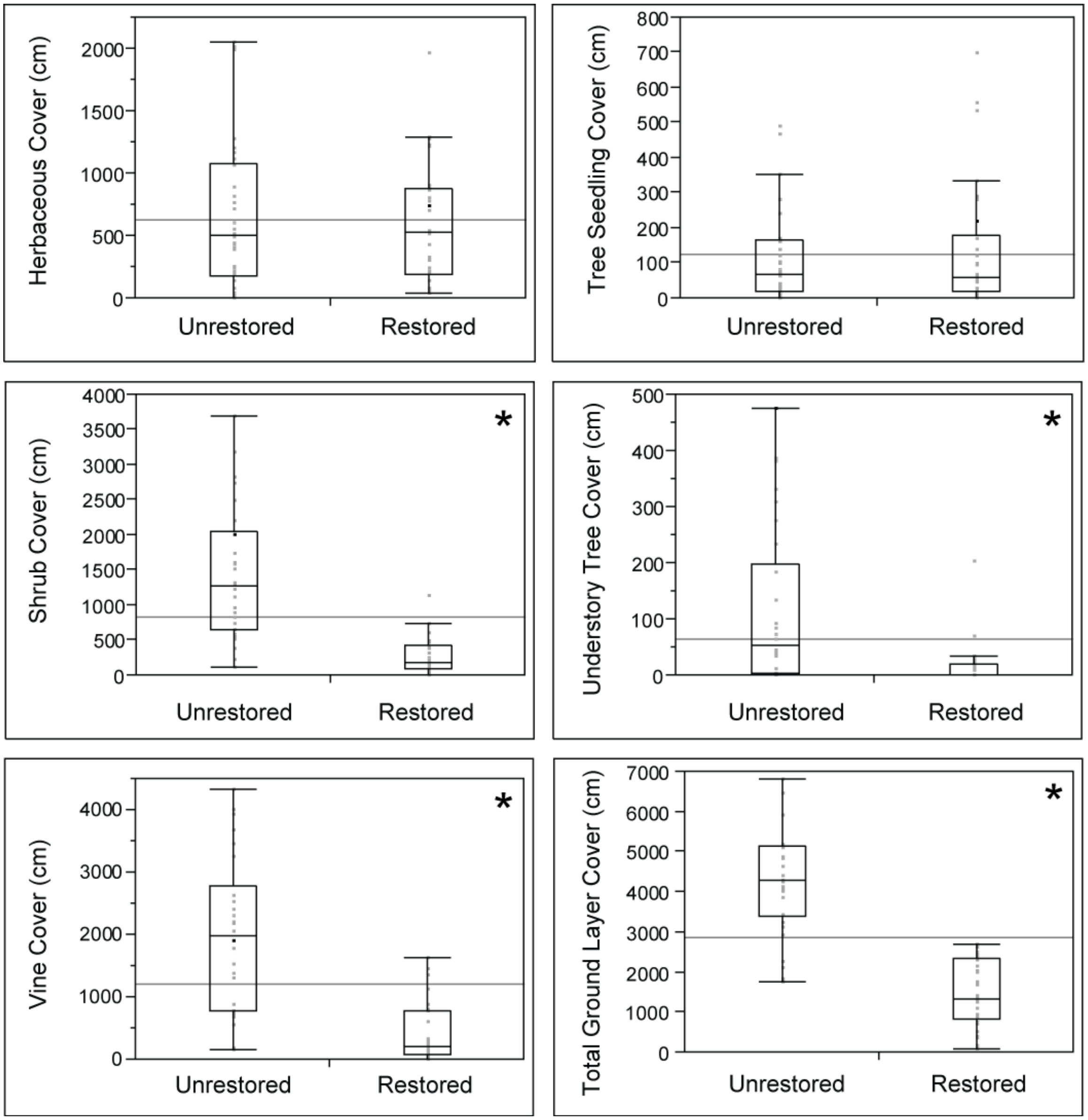

Figure 1.18: Differences in total plot cover (cm) by growth form among restored and unrestored sites. Cover by herbaceous plants and tree seedlings did not differ significantly between restored and unrestored plots. 4000 cm represents 100% cover of four 10m line transects; species overlap was permitted. Restored plots had lower cover by shrubs (p < 0.0001; Restored sites – mean: 269.1, range: 0-1144; Unrestored sites – mean: 1403.9, range: 116-3694), vines (p < 0.0001; Restored sites – mean: 442.1, range: 2-1635; Unrestored sites – mean: 1991.7, range: 157-4328), understory trees (p

< 0.0001; Restored sites – mean: 15.3, range: 0-203; Unrestored sites – mean: 114.6, range: 0-475), and total ground layer cover ($p < 0.0001$; Restored sites – mean: 1466.2, range: 80-2689; Unrestored sites – mean: 4281.8, range: 1746-6806).

Native, Non-Native, and Invasive Species

Targeted Invasive Species

Invasive species targeted for removal were less abundant in restored sites than in invaded sites that did not receive restoration treatment. Both primary target invasive species and secondary target species (removed on at least 100 occasions) were fewer in restored plots. Ground layer cover of both primary and secondary target invasive species was lower where restoration was done. For more detailed information on species in each site, see Appendix 1A.

Restoration activity records corroborate management priorities (Wenskus, pers. comm.). The four most frequently removed species were the same species reported to be the primary targets of restoration efforts at a city-wide scale: *A. brevipedunculata*, *C. orbiculatus*, *A. platanoides* and *R. multiflora*. These species were removed on more than 1,000 occasions during the period 1988-2009, as indicated by the number of dates on which removal was reported (Table 1.4).

Table 1.4: Non-native, invasive species removed on at least 100 different occasions, 1988-2009. (Mean for all targeted species: 103 removal dates.) Primary target species *C. orbiculatus, R. multiflora, A. brevipedunculata,* and *Acer pseudoplatanus* were removed on > 1000 occasions between 1998 and 2009. Secondary target species were removed on more than 100 occasions.

	Non-native Invasive Species	**Total Removals**
Primary target species	*Ampelopsis brevipedunculata*	1883
	Celastrus orbiculatus	1842
	Acer platanoides	1054
	Rosa multiflora	1050
Secondary target species	*Acer pseudoplatanus*	849
	Lonicera japonica	698
	Lonicera maackii	624
	Artemisia vulgaris	615
	Morus alba	451
	Alliaria petiolata	412
	Ailanthus altissima	251
	Lonicera tatarica	218
	Rubus phoenicolasius	209
	Phragmites australis	208
	Rhodotypos scandens	170
	Phellodendron amurense	157
	Fallopia japonica	136
	Broussonetia papyrifera	112
	Arctium minus	106
	Frangula alnus	103

Targeted Invasive Species in the Woody Understory

The woody understory consisted of woody vines, individual trees > 1m in height and < 2.5 cm in diameter, and shrubs. Restored sites had fewer total stems and proportion of both primary and secondary target invasive species.

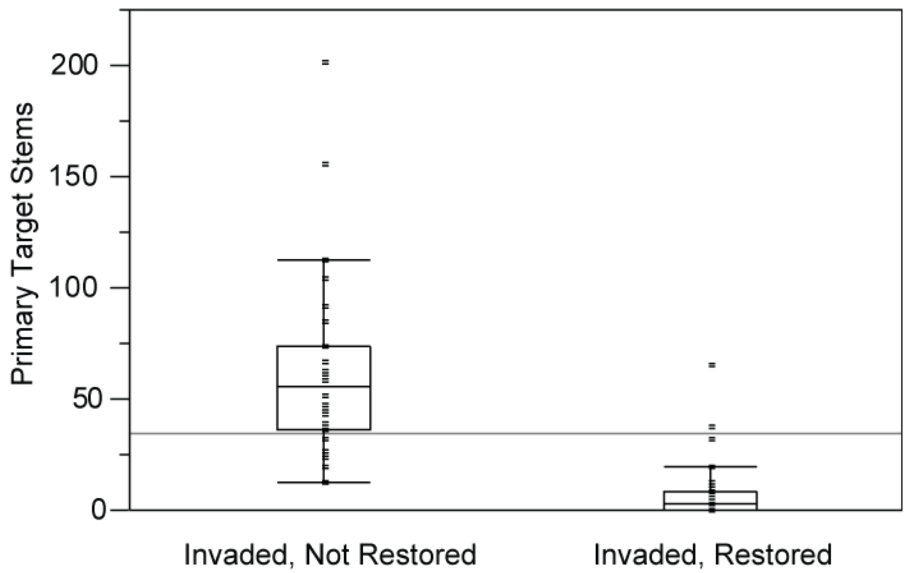

Figure 1.19: Number of woody understory stems per plot belonging to primary target species in restored and unrestored sites. Restored sites had significantly fewer primary target invasive species stems in the woody understory than unrestored sites ($p < 0.0001$; Restored sites – mean: 7.9, range: 0-65; Unrestored sites – mean: 62.3, range: 13-201).

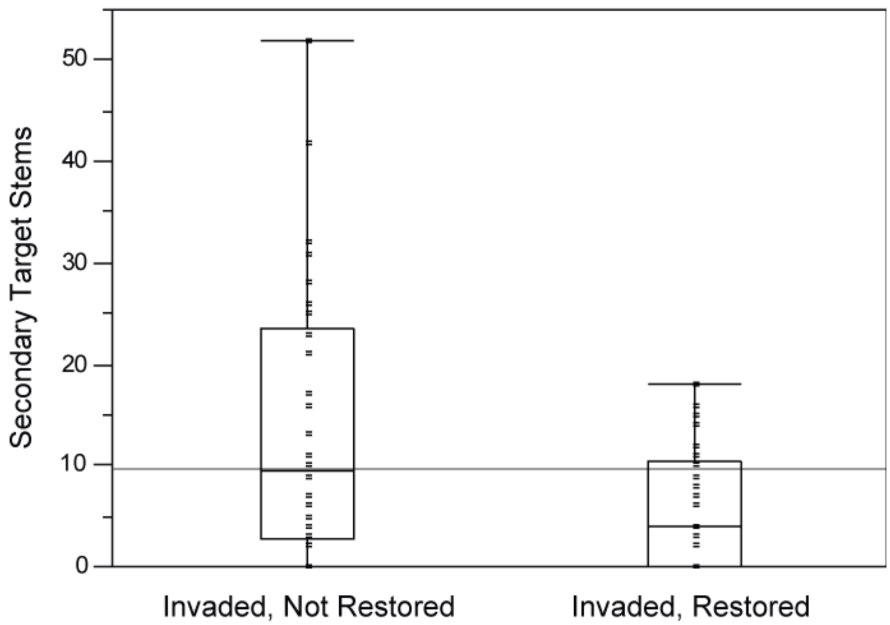

Figure 1.20: Number of stems of secondary target species per plot in the woody understory of restored and unrestored forest plots. Secondary target stems were significantly fewer in restored plots (p = 0.0027; Restored sites – mean: 5.9, range: 0-18; Unrestored sites – mean: 13.8, range: 0-52).

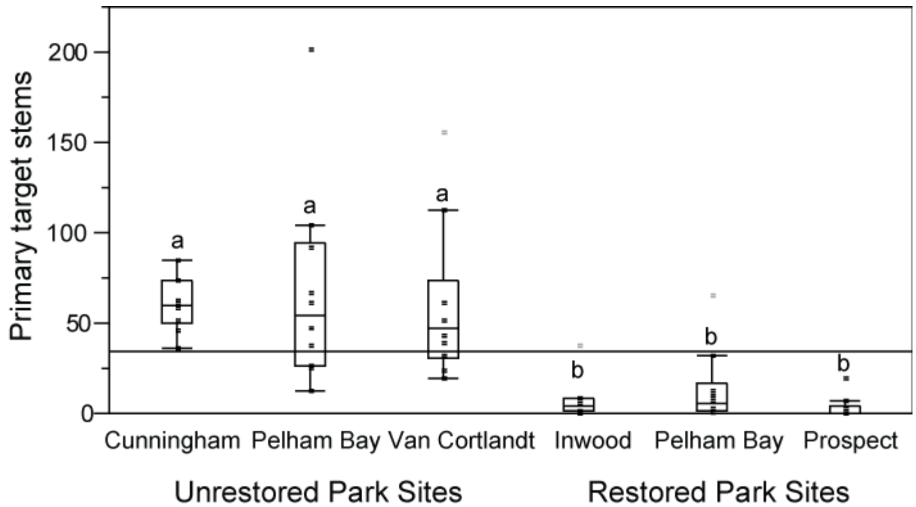

Figure 1.21: Number of primary target species stems per plot by park. The number of target invasive species stems per plot was significantly higher in unrestored than in restored parks Means for sites with different lower-case letters were significantly different (p < 0.0001, range: 1-334, mean: 83.5). Cunningham – mean: 59.8, range: 36-85; Pelham Bay unrestored – mean: 67.2, range: 13-201; Van Cortlandt – mean: 60.0, range: 20-156; Inwood – mean: 7.1, range: 0-37; Pelham Bay restored – mean: 13.4, range: 0-65; Prospect – mean: 3.2, range: 0-20.

Targeted Invasive Species in the Ground Layer

Sites where primary target invasive species were abundant in the late 1980s and early 1990s but were not restored had higher total and proportional cover by those species in 2010. Secondary target species were also more abundant in unrestored sites.

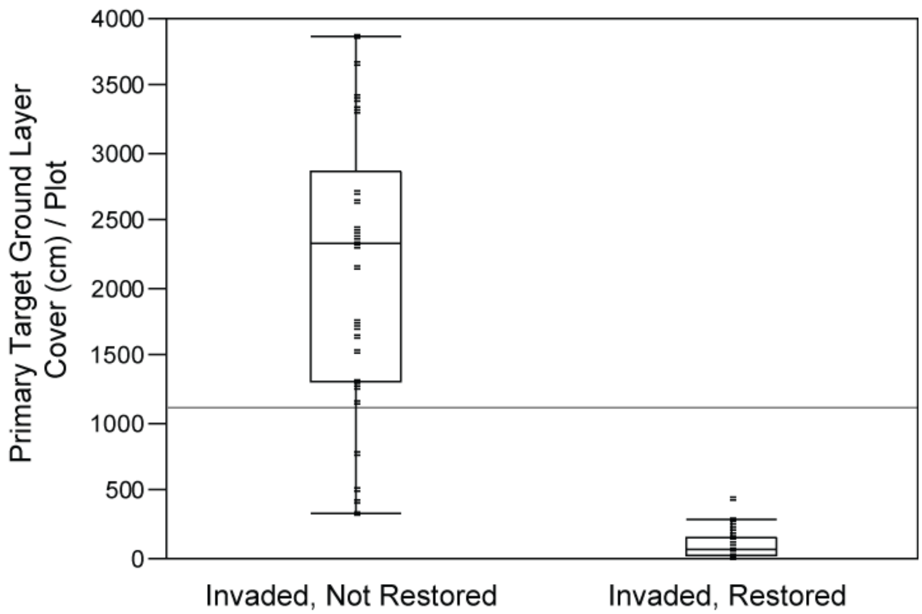

Figure 1.22: Total cover (cm) per plot of primary target species in the ground layer of restored and unrestored forest plots. Primary target stems were significantly fewer in restored plots ($p < 0.0001$; Restored sites – mean: 104.7, range: 0-447; Unrestored sites – mean: 2151.5, range: 324-3866).

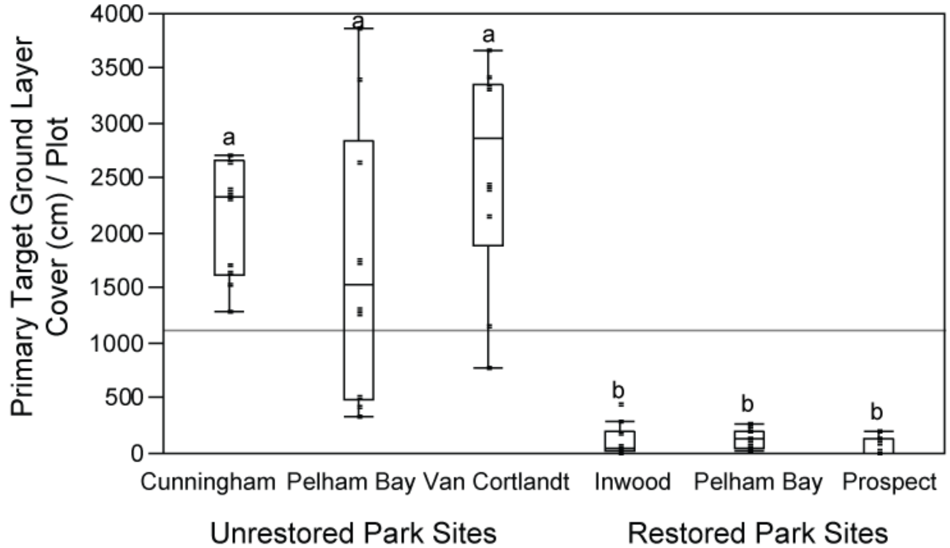

Figure 1.23: Ground layer cover (cm) of primary target species per plot by park. Cover of primary target invasive species was significantly higher in unrestored than in restored parks. Different letters indicate significant differences ($p < 0.0001$). 4000 cm represents 100% cover of four 10m transects per plot. Cunningham – mean: 2129, range: 1282-2707; Pelham Bay unrestored – mean: 1725, range: 324-3866; Van Cortlandt – mean: 2601, range: 773-3676; Inwood – mean: 125, range: 0-447; Pelham Bay restored – mean: 129, range: 13-275; Prospect – mean: 60, range: 0-202.

Figure 1.24: Total cover (cm) per plot of secondary target species in the ground layer of restored and unrestored forest plots. Stems of secondary target species were significantly fewer in restored plots ($p < 0.0001$; Restored sites – mean: 228.8, range: 0-1439; Unrestored sites – mean: 669.7, range: 46-1769).

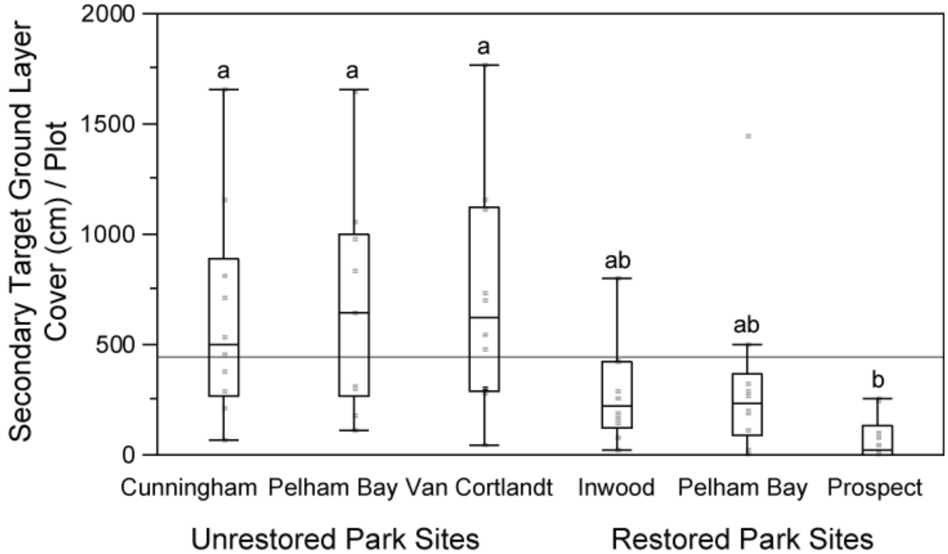

Figure 1.25: Ground layer cover (cm) of secondary target species per plot by park. Cover of secondary target invasive species was significantly higher in unrestored than in restored parks (*p* = 0.0002). Cunningham – mean: 625, range: 71-1652; Pelham Bay unrestored – mean: 671, range: 106-1650; Van Cortlandt – mean: 712, range: 46-1769; Inwood – mean: 280, range: 22-803; Pelham Bay restored – mean: 334, range: 0-1439; Prospect – mean: 73, range: 0-256.

Native Species

A higher proportion of the vegetation of restored sites was composed of native plants. Native plants composed a larger fraction of ground layer cover, woody understory stems, and trees.

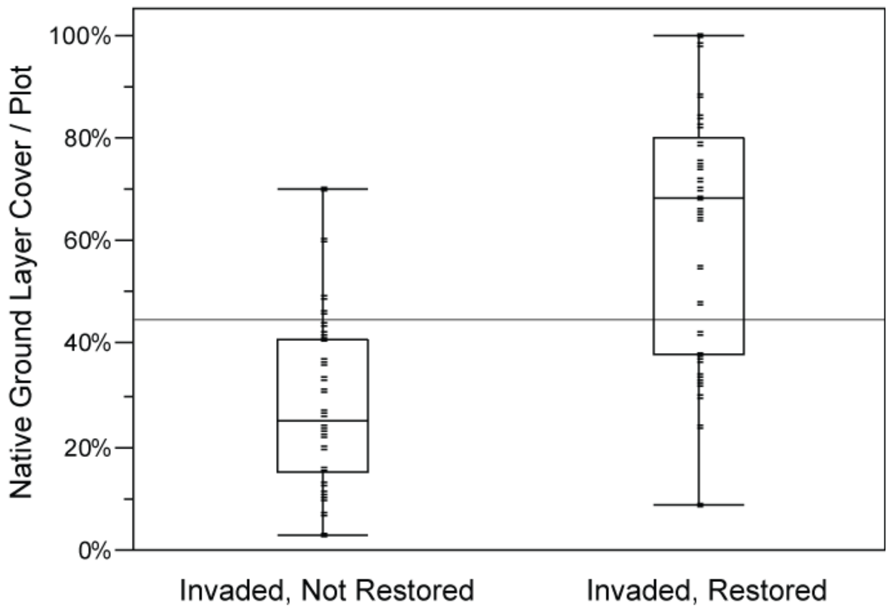

Figure 1.26: Proportion of ground layer cover per plot occupied by native plants. Native plants made up a larger proportion of ground layer cover in restored than in unrestored plots ($p < 0.0001$; Restored sites – mean: 61.3%, range: 9-100%; Unrestored sites – mean: 28.6%, range: 3-70%).

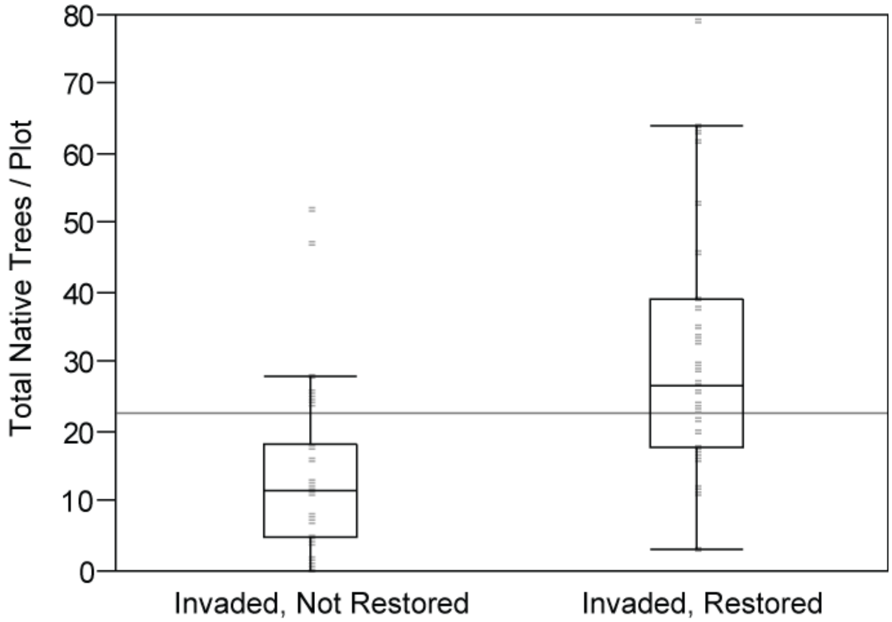

Figure 1.27: Total native tree stems per plot in restored and unrestored sites. Plots in restored sites had significantly more native tree stems than those in unrestored sites ($p < 0.0001$; Restored sites – mean: 31.4, range: 3-79; Unrestored sites – mean: 14.4, range: 0-52).

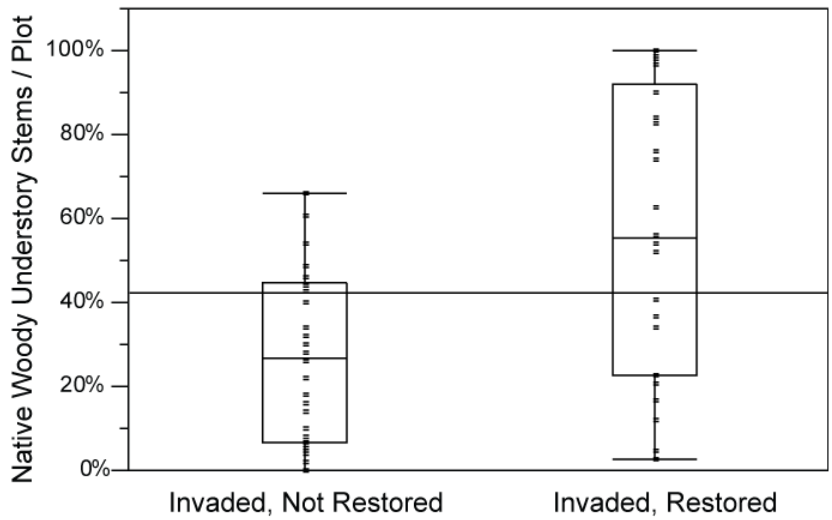

Figure 1.28: Proportion of native woody understory stems per plot in restored and unrestored sites. The proportion of understory sapling stems that were native species was significantly higher in restored sites (*p* = 0.0003; Restored sites – mean: 58%, range: 3-100%; Unrestored sites – mean: 27%, range: 0-66%).

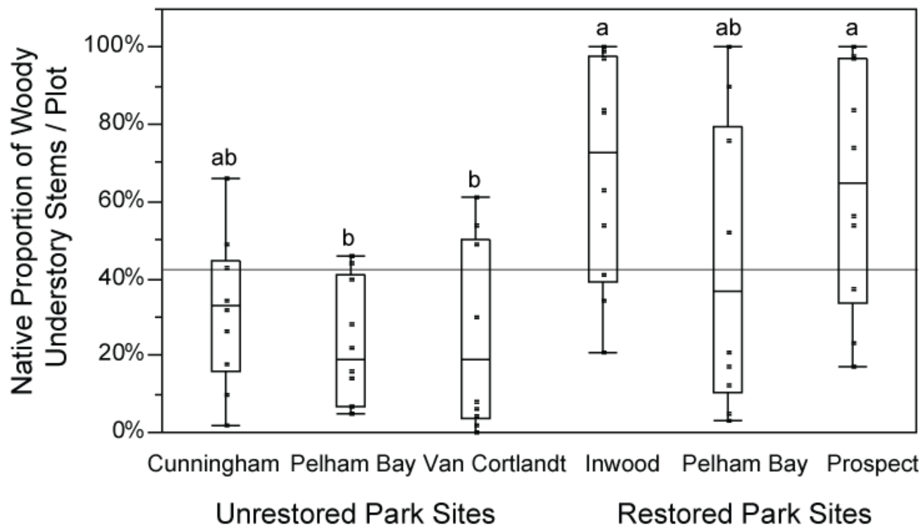

Figure 1.29: Proportion of woody understory stems per plot made up by native species in each park. Different letters indicate groups with significant differences ($p < 0.0035$). Cunningham – mean: 32%, range: 2-66%; Pelham Bay unrestored – mean: 23%, range: 5-46%; Van Cortlandt – mean: 26%, range: 0-61%; Inwood – mean: 68%, range: 21-99.9%; Pelham Bay restored – mean: 43%, range: 3-99%; Prospect – mean: 64%, range: 17-99.8%.

Relative Composition of Forest Strata

Relative proportions of native, non-native, invasive and other species differed between restored and unrestored sites. Here I consider the role of targeted species in relation to other native and non-native plants. I considered species that are considered weedy or invasive in part of their U.S. introduced range (USDA 2013a) but were not targeted by this restoration to be potential invasive species for the region. Other plants included exotic species not considered weedy or invasive in their U.S. introduced range (USDA

2013a), and plants that were not identifiable beyond the genus level in genera that contained both native and non-native species (e.g. *Malus* sp.).

Native, Non-Native and Invasive Tree Species

The total number and basal area of tree stems > 1m in height and 2.5 cm DBH were higher in restored sites. Trees in both types of sites were composed of ca. 80% native species, but restored and unrestored sites differed in their species composition.

Table 1.5: Shade tolerance, growth rate, and successional associations (Burns and Honkala 2013; USDA 2013b) of the most abundant tree species in restored and unrestored sites, in order of decreasing total basal area. Here, successional associations refer to forest stand age since disturbance where species have been observed. Non-native species are indicated with an asterisk.

Site Type	**Species**	**Shade Tolerance**	**Growth Rate**	**Successional Association**
Restored	*Robinia pseudoacacia*	intolerant	rapid	early
	Prunus serotina	intolerant	rapid	early, gap-phase
	Liriodendron tulipifera	intolerant	rapid	early
	Quercus rubra	intermediate	intermediate	intermediate
	Carya cordiformis	intermediate	intermediate	intermediate
	Quercus palustris	intolerant	intermediate	early - intermediate
	Fraxinus pennsylvanica	moderate	intermediate	multiple
	Acer saccharum	high	intermediate	early - late
	Acer rubrum	moderate	intermediate	intermediate
Unrestored	*Sassafras albidum*	intolerant	intermediate	early
	Robinia pseudoacacia	intolerant	rapid	early
	Prunus serotina	intolerant	rapid	early, gap-phase
	Liquidambar styraciflua	intolerant	intermediate	early
	Carya cordiformis	intermediate	intermediate	intermediate – late
	*Morus alba**	intolerant	intermediate	early
	Quercus palustris	intolerant	intermediate	early - intermediate
	Acer rubrum	moderate	intermediate	intermediate

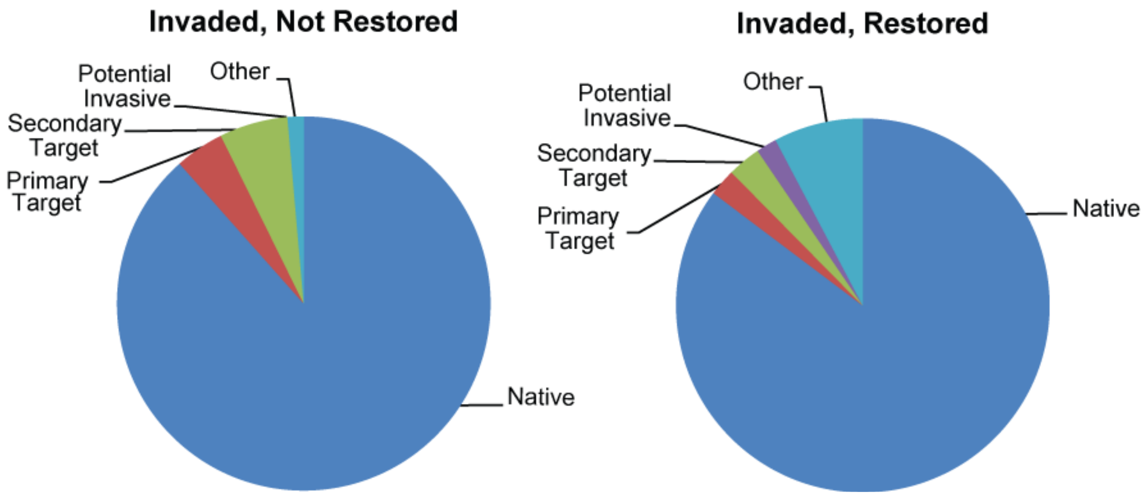

Figure 1.30: Average proportion of basal area per plot by species category in restored and unrestored sites. Restored and unrestored areas did not differ significantly in the proportion of tree species by category.

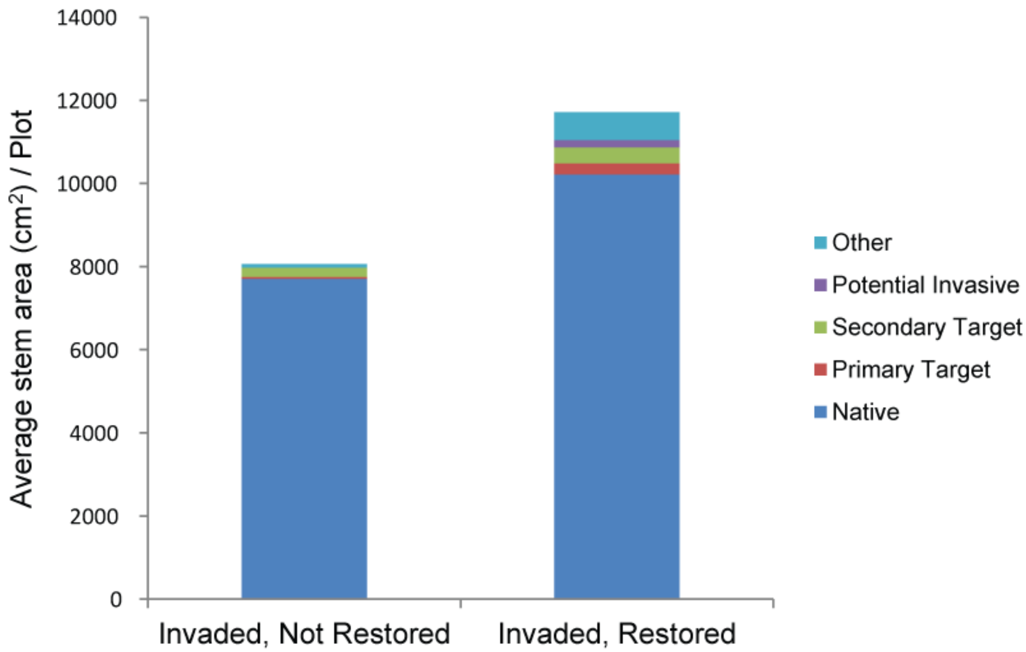

Figure 1.31: Average basal area per plot (cm^2) by species category in restored and unrestored sites. Restored sites had a higher total tree basal area per plot than sites that were not restored (p = 0.0413).

Table 1.6: Average total basal area by tree species and average proportion of basal area per plot in restored and unrestored invaded areas.

Site Type	Native		Primary Target		Secondary Target		Potential Invasive		Other	
	cm^2	%	cm^2	%	cm^2	%	cm^2	%	cm^2	%
Restored	404	83%	11	3%	16	3%	15	3%	26	6%
Not Restored	266	84%	5	5%	14	7%	0	-	10	3%

Native, Non-Native and Invasive Species in the Woody Understory

Although the total number of native woody stems was similar between restored and unrestored sites, the proportion of native and primary target stems differed significantly. Primary target invasive species made up a larger proportion of total woody understory stems in unrestored sites. Native plants made up a larger fraction of the woody stems in restored sites.

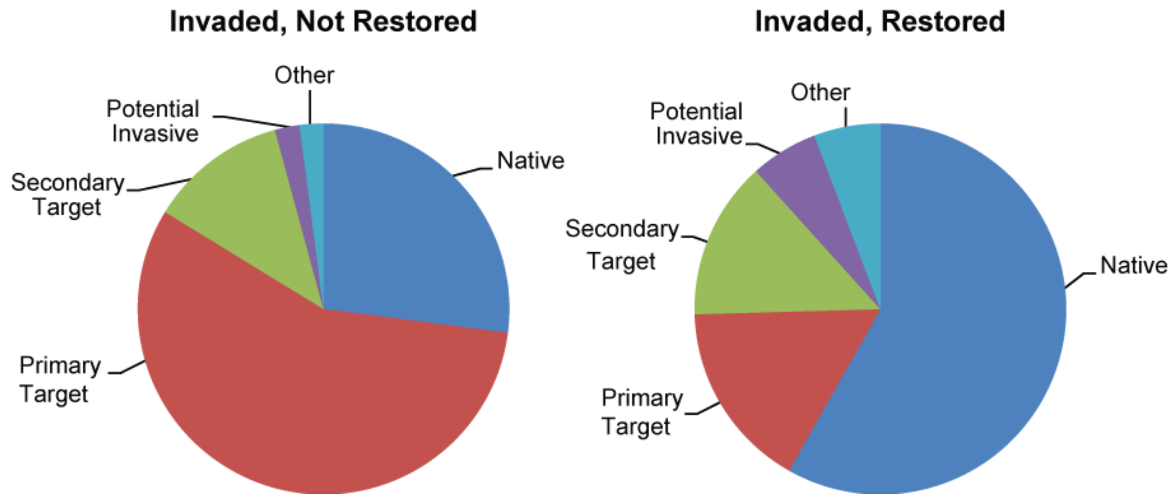

Figure 1.32: Average proportion of woody understory stems by species category in restored and unrestored invaded sites. Restored and unrestored sites differed significantly in the proportion of native woody understory stems ($p < 0.0001$), and unrestored sites had a significantly higher proportion of primary target stems ($p < 0.0001$).

Table 1.7: Average woody understory stems per plot by site type and species category.

Site Type	Native		Primary Target		Secondary Target		Potential Invasive		Other	
	Stems	%	Stems	%	Stems	%	Stems	%	Stems	%
Restored	33	58%	8	16%	6	14%	3	6%	4	6%
Unrestored	33	27%	62	57%	14	12%	2	2%	4	2%

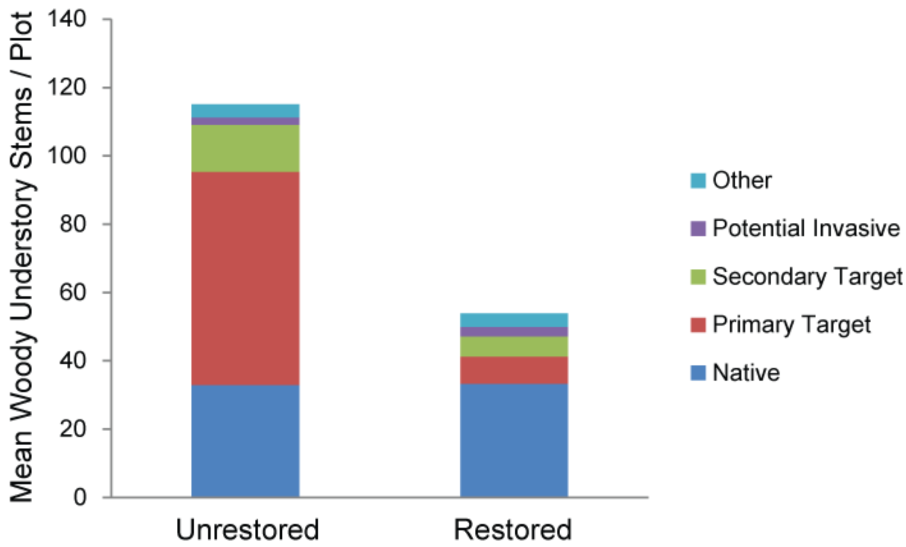

Figure 1.33: Average number of stems per subplot by species category. Restored and unrestored sites did not differ significantly in number of native woody understory stems, but unrestored sites had a significantly higher number of total stems ($p < 0.0001$), primary target stems ($p < 0.0001$), and secondary target stems ($p = 0.0366$).

Native, Non-Native and Invasive Species in the Ground Layer

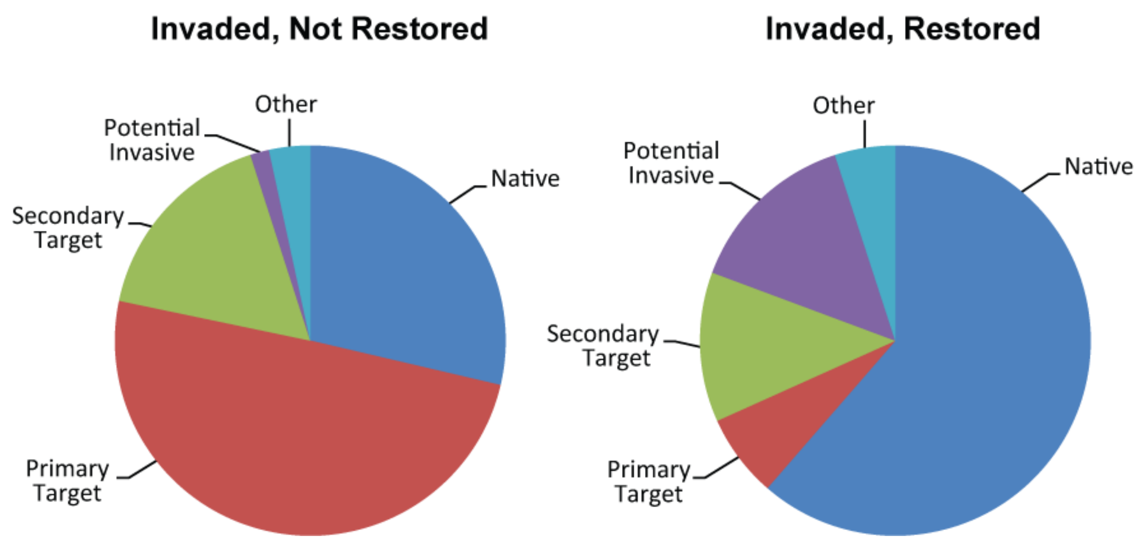

Figure 1.34: Proportion of total cm of ground layer cover occupied in restored and unrestored plots by species category.

In restored plots, an average of 7% of the ground layer was occupied by targeted invasive species. In sites reported to be invaded by the same species in 1990 that were not restored in 2010, these species occupied 50% of the ground layer (Figure 1.34).

Table 1.8: Average ground layer cover occupied per plot by species category in restored and unrestored invaded sites.

Site Type	**Native**		**Primary Target**		**Secondary Target**		**Potential Invasive**		**Other**	
	cm	**%**	**cm**	**%**	**cm**	**%**	**cm**	**%**	**cm**	**%**
Restored	874	61%	105	7%	229	12%	213	14%	55	5%
Not Restored	1249	29%	2152	50%	670	17%	61	2%	141	3%

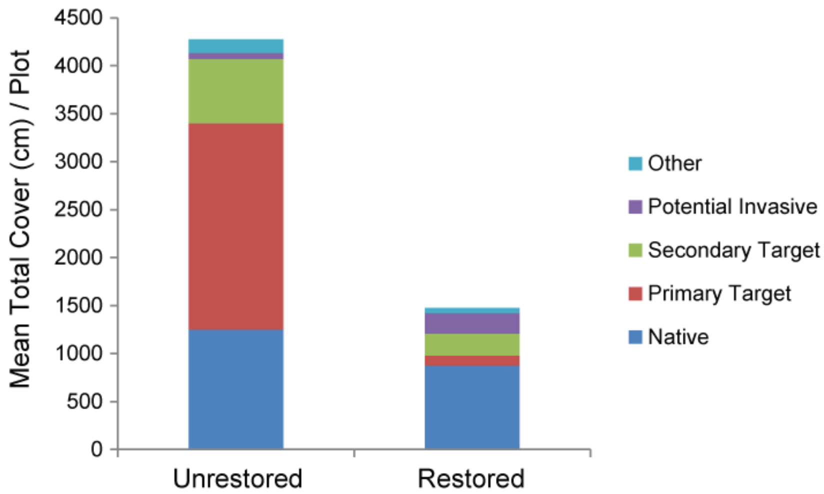

Figure 1.35: Average total ground layer cover by growth form. Unrestored sites had significantly greater total ground layer cover ($p < 0.0001$), cover by primary target species ($p < 0.0001$), and cover by secondary target species ($p = 0.0001$).

Plant Community Characteristics of Forest Strata

Plant community attributes differed among forest strata (Table 1.9). In the ground layer, sites that were not restored had more total cover and a greater range of cover values. Restored sites had greater evenness in the ground layer vegetation. There were no significant differences between restored and unrestored sites in ground layer diversity or species richness. Unrestored sites had significantly more total woody understory stems than restored sites. Woody understory stems did not differ significantly in diversity, richness, or evenness between restored and unrestored sites. Trees were more diverse, more species rich, and had greater basal area in restored sites. Evenness did not significantly differ between site types.

Table 1.9: Differences in community properties between restored and unrestored forest strata.

Forest Stratum	Attribute	Trend	Prob > |t|
Trees	Shannon Diversity	Treated > Untreated	.0011*
	Richness	Treated > Untreated	.0012*
	Evenness		0.3113
	Total Stem Area	Treated > Untreated	.0006*
Woody Understory	Shannon Diversity		0.2566
	Richness		0.3521
	Evenness		0.867
	Total Stems	Untreated > Treated	.0001*
Ground Layer	Shannon Diversity		0.4743
	Richness		0.4314
	Evenness	Treated > Untreated	.0097*
	Total Cover	Untreated > Treated	.0001*

Species Composition

Trees

Dominant tree species differed between restored and unrestored sites. Both site types shared a set of native trees including *Robinia pseudoacacia, Prunus serotina, Carya cordiformis, Quercus palustris* and *Acer rubrum*. In unrestored sites, *Sassafras albidum*, *Liquidambar styraciflua*, *Morus alba*, and *Malus sp*. were also among the most abundant trees. Abundant trees in restored sites included some of the most frequently-planted species of the restoration effort: *Liriodendron tulipifera*, *Quercus rubra*, *Fraxinus pennsylvanica* and *Acer saccharum*.

Table 1.10: Ten tree species with greatest total basal area in restored and unrestored sites. Non-native species are indicated with asterisks. Trees in common between the most abundant species of restored and unrestored sites are in bold.

Site Type	Species	Total Stem Area (cm^2)	Avg DBH	Avg Stems/ Plot	Total Stems All Plots	Canopy Stems All Plots	Subcanopy Stems All Plots
Restored	***Robinia pseudoacacia***	2377	28	7	84	47	13
	Prunus serotina	1760	12	6	146	10	15
	Liriodendron tulipifera	1321	10	8	126	18	14
	Quercus rubra	1129	36	2	31	15	75
	Carya cordiformis	1065	18	5	59	14	4
	Quercus palustris	743	50	2	15	13	91
	Fraxinus pennsylvanica	623	7	6	92	3	1
	Acer saccharum	547	6	6	93	0	1
	Acer rubrum	511	7	8	75	0	7
Unrestored	*Sassafras albidum*	1690	11	12	150	13	137
	Robinia pseudoacacia	1608	30	5	54	34	20
	Prunus serotina	1605	20	4	81	19	62
	Liquidambar styraciflua	620	44	2	14	9	5
	Carya cordiformis	380	13	3	29	5	24
	*Morus alba**	355	17	3	21	0	21
	Quercus palustris	265	44	1	6	4	2
	Acer rubrum	242	17	2	14	1	13
	Malus sp.	214	5	3	43	0	43

Woody Understory

Dominant species differed in the understories of restored and unrestored sites. Sites that were not restored were dominated by primary target species and *Lonicera japonica*. Native and non-native *Rubus* species and native woody *Vitis spp.* vines also frequently occurred in open vineland unrestored sites. Restored sites were dominated by the frequently-planted native shrub *Lindera benzoin*, and by the native (though undesirably toxic to humans) *Toxicodendron radicans*. Primary target species *R. multiflora* and *A. brevipedunculata* were present in lower total numbers and in lower proportions in restored sites. Saplings of *Fraxinus Americana*, *Carya cordifomis* and *Prunus serotina* made up a larger fraction of the woody understory in restored sites.

Table 1.11: Ten most abundant species in the woody understory in restored and unrestored sites. Non-native species are indicated with asterisks. Restored sites had a total of 69 species in the woody understory; unrestored sites had 54. See Appendix 1A for a full list of species.

Site Type	Growth Habit	Species	Total Stems	% of All Stems
Restored	Shrub	*Lindera benzoin*	244	11%
	Vine	*Toxicodendron radicans*	229	10%
	Shrub, Vine	*Rosa multiflora**	157	7%
	Tree, Shrub	*Viburnum dentatum*	94	4%
	Tree	*Fraxinus americana*	72	3%
	Tree	*Prunus sp.**	71	3%
	Vine	*Lonicera japonica**	57	3%
	Tree	*Carya cordiformis*	46	2%
	Tree	*Prunus serotina*	46	2%
	Vine	*Ampelopsis brevipedunculata**	45	2%
Unrestored	Shrub, Vine	*Rosa multiflora**	999	29%
	Vine	*Ampelopsis brevipedunculata**	559	16%
	Vine	*Lonicera japonica**	308	9%
	Vine	*Celastrus orbiculatus**	307	9%
	Shrub	*Rubus pensilvanicus*	305	9%
	Tree, Shrub	*Viburnum dentatum*	106	3%
	Vine	*Vitis aestivalis*	94	3%
	Vine	*Vitis labrusca*	80	2%
	Vine	*Parthenocissus quinquefolia*	73	2%
	Shrub	*Rubus phoenicolasius**	63	2%

Ground Layer

Dominant species differed in the ground layer cover of restored and unrestored sites. Both restored and unrestored sites contained invasive species that were not targeted by the restoration examined here. Total cover for all unrestored sites was 2.9 times greater than total ground layer cover for restored sites (Table 1.12). See Appendix 1A for a full list of ground layer species.

Table 1.12: Ten species with the greatest amount of total ground layer cover in restored and unrestored sites. Non-native species are indicated with an asterisk.

Site Type	Growth Habit	Species	Total Cover (cm)	% of All Cover
Restored	Herb	*Circaea lutetiana*	4395	10%
	Vine	*Toxicodendron radicans*	3758	8%
	Vine	*Lonicera japonica**	2988	7%
	Vine	*Parthenocissus quinquefolia*	2915	7%
	Herb	*Impatiens capensis*	2758	6%
	Herb	*Aegopodium podagraria**	2499	6%
	Herb	*Alliaria petiolata**	2167	5%
	Shrub	*Lindera benzoin*	1771	4%
	Shrub	*Rosa multiflora**	1527	3%
	Vine	*Hedera helix**	1310	3%
Unrestored	Shrub	*Rosa multiflora**	30408	24%
	Vine	*Ampelopsis brevipedunculata**	24214	19%
	Vine	*Celastrus orbiculatus**	9882	8%
	Vine	*Lonicera japonica**	8474	7%
	Herb	*Alliaria petiolata**	5995	5%
	Shrub	*Rubus pensilvanicus*	5406	4%
	Vine	*Parthenocissus quinquefolia*	4787	4%
	Herb	*Impatiens capensis*	4492	4%
	Vine	*Toxicodendron radicans*	4305	3%
	Vine	*Vitis aestivalis*	4038	3%

Regeneration

Tree Saplings in the Understory

The total number of saplings per plot did not significantly differ between restored and unrestored sites. However, the number of seedlings of native canopy trees per plot and the proportion of all seedlings that were canopy trees were both higher in restored plots than in unrestored plots. All but one restored site contained native tree saplings, while more than a quarter of unrestored sites had zero native tree saplings. Unrestored sites had fewer tree saplings in total, and a lower proportion of those saplings were native. A greater proportion of woody understory stems belonged to native canopy tree species in restored sites, and a higher proportion of the stems in unrestored sites were target invasive species.

Table 1.13: Average native sapling and native canopy tree saplings per plot in restored and unrestored sites.

	Restored	**Unrestored**
Saplings Native	80%	56%
Saplings Native Canopy Tree Spp.	70%	33%
% of All Stems Native Saplings	34%	7%

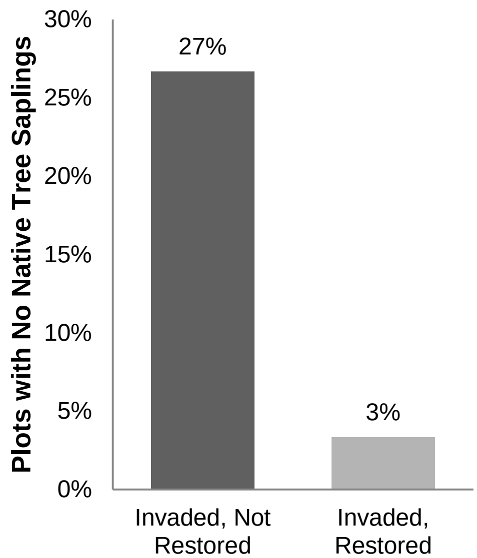

Figure 1.36: Proportion of plots lacking native sapling regeneration by site type.

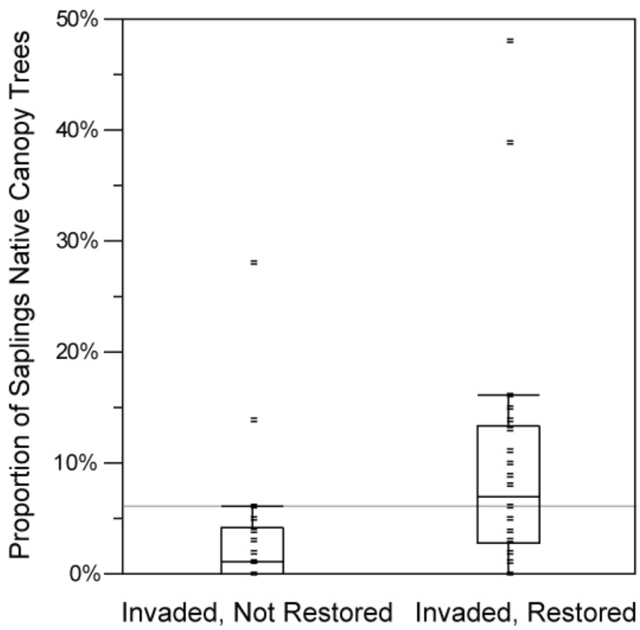

Figure 1.37: Proportion of understory saplings per plot that were native canopy tree species in all restored and unrestored sites ($p < 0.0001$; Restored sites – mean: 9.5, range: 0-48; Unrestored sites – mean: 3.0, range: 0-28).

A higher proportion of tree saplings were native in restored sites ($p = 0.0035$, mean: 43%). Total ground layer cover by tree seedlings did not differ significantly between restored and unrestored sites.

Table 1.14: Species of canopy and understory tree seedlings found in the ground layer of restored and unrestored plots. Non-native species are indicated with an asterisk.

Site Type	Species	% of Plots	Total Cover (cm)	% of Cover
Restored	Prunus serotina	57%	349	8.40%
	Carya cordiformis	37%	450	10.84%
	Fraxinus pennsylvanica	27%	829	19.96%
	Acer pseudoplatanus*	17%	472	11.37%
	Acer platanoides*	17%	242	5.83%
Unrestored	Sassafras albidum	50%	1190	34.73%
	Prunus serotina	37%	818	23.88%
	Carya cordiformis	23%	157	4.58%
	Morus alba	13%	211	6.16%
	Robinia pseudoacacia	13%	66	1.93%

Table 1.15: Species of tree saplings found in the greatest number of plots in restored and unrestored plots. Non-native species are indicated with an asterisk.

Site Type	Species	% of Plots	Total Stems	% of Stems
Restored	Fraxinus americana	47%	72	12.8%
	Prunus serotina	43%	46	8.2%
	Carya cordiformis	37%	46	8.2%
	Quercus rubra	23%	13	2.3%
	Acer platanoides*	20%	10	1.8%
Unrestored	Sassafras albidum	37%	44	16.7%
	Viburnum dentatum	33%	106	40.2%
	Malus sp.	27%	22	8.3%
	Carya cordiformis	17%	9	3.4%
	Ailanthus altissima*	13%	7	2.7%

DISCUSSION

Clear differences in plant community composition and structure between restored and unrestored urban forest patches indicate that restoration treatments had significant effects that persisted for more than 15 years. These forests are now on divergent trajectories of plant community development as a result of restoration.

Forest Structure

Restored forests exhibited a markedly different vertical structure compared to invaded forests that were not restored (Figures 1.7, 1.8 and 1.9, Table 1.2). They had a significantly greater mean number of total tree stems and greater mean tree basal area, lower mean abundance of woody understory stems, lower overall mean ground layer cover, and greater average proportion of herbaceous plants in their ground layer vegetation. Together, these findings indicate that restored forests differed significantly in their structure from invaded sites that were not restored. The direction of that difference is toward greater abundance of native canopy trees and away from a vine- and shrub-dominated state with little tree canopy. These differences in structure indicate that the restoration effort achieved a central goal of the restoration project.

The contrast in structure between restored and unrestored sites indicates that invasion by this suite of woody species is a long-term change with persistent effects community composition.

Native and Non-Native Species Composition

Clear differences in the species composition of restored and unrestored sites also indicate a strong effect of restoration. The ground layer and woody understory of unrestored sites remained dominated by woody invasive plants that were abundant in those locations 15-20 years before (Tables 1.6, 1.7 and 1.8; Figures 1.19, 1.20, and 1.22). In restored sites, mean abundance of the targeted species was less by an order of magnitude (Figure 1.22), and the mean abundance of other invasive species was also reduced in restored sites (Figure 1.24). Trees of restored and unrestored sites did not differ significantly in the proportion of their stems that were native (both ca. 80%), but unrestored sites had fewer trees, more non-native tree saplings, and fewer native tree seedlings (Figure 1.36). These differences in target species abundance and native tree regeneration indicate achievement of additional goals of the restoration effort.

Both restored and unrestored sites, however, contained invasive species that were either not present or not understood to be a problem when the restoration work began. The abundance of Japanese honeysuckle (*Lonicera japonica*), goutweed (*Aegopodium podagraria*), and garlic mustard (*Alliaria petiolata*) in many plots indicates that continuous species introduction is a variable that must be considered. Cities are loci of species introduction and invasion that provide a variety of habitat types and resource subsidies to intentionally and unintentionally introduced species (Klotz and Kühn 2010; Von Der Lippe and Kowarik 2007).

Tree Regeneration

Restored sites differed from invaded sites that were not restored in the composition of regenerating tree species in the woody understory (Figure 1.37). In sites that were not restored, few saplings attained a meter in height, and native tree seedlings and saplings were entirely absent from a quarter of unrestored plots (Figures 1.8, 1.12 and 1.38). Seedlings and saplings represent potential future canopy trees. The differences observed here between tree sapling abundance and seedling composition indicate that restored and unrestored forests are likely to have different future canopy compositions.

Where trees were present in unrestored sites, they were dominated by rapidly-growing, shade-intolerant species such as *S. albidum*, *R. pseudoacacia*, and *P. serotina*. *Robinia pseudoacacia* and *P. serotina* are both early colonizers of disturbed sites. The exact nature of disturbance that may have cleared the way for these species in each site is not known, though weakening of canopy trees by heavy invasive vines leading to tree death or wind throw in storms is likely. *Sassafras albidum* sprouts vigorously from roots following fire and other disturbances (USDA 2013b), and its high abundance in unrestored sites may also indicate a role of fire history in setting the conditions for invasion. Although precise records of fire locations were not available, fires initiated by the burning of stolen automobiles in New York City parks were a management concern in the 1980s and removal of abandoned and burned cars was conducted as part of the restoration project (NRG 1986).

Shade-intolerant native species *Robinia pseudoacacia* and *Prunus serotina* were also dominant in restored sites, and fast-growing *Liriodendron tulipifera* was the most successful planted species. Bare soil, removal of established vegetation, and availability of sunlight at the ground level all contribute to a hospitable environment for these pioneering species. Slower-growing, more shade-tolerant species *Quercus rubra* and *Acer saccharum* were abundant as well; these two trees were among the most frequently planted species, which is likely to have contributed to their abundance. Although it was less frequently planted, bitternut hickory (*Carya cordiformis*) was among the most abundant tree species in restored sites. With a large geographic range, this species is tolerant of a variety of soil and moisture conditions, and reproduces by abundant root sprouts as well as by seed (Burns and Honkala 2013). These characteristics may be adaptive in the variable, often compacted soils of urban parks.

Restored and unrestored sites shared a suite of shade-intolerant, quickly-growing trees that colonize disturbed sites. The most successful planted species shared these characteristics. Together, these findings indicate that restoration activities created sites that were hospitable for seedlings and saplings of species adapted to high light conditions and disturbed soils.

Variability and its Sources

Variability between sites and between plots was expected due to heterogeneity in urban land use history (Hope et al. 2003; Cadenasso, Pickett, and Schwartz 2007). This was certainly the case. For example, the likely cause of lack of native tree regeneration in the single restored site where they were absent turned out, upon further investigation, to have likely been due to historic large mammal compaction. It was the former site of the Elephant House of the Prospect Park Zoo (A. Wong, personal communication, Appendix

1D). In addition to soil variation associated with glacial deposition, sites varied in the degree to which they were subject to historic agricultural cultivation and development prior to becoming parks and in local-scale management history prior to restoration (see Appendix 1D for more information on individual site histories).

In addition to finer-scale variability between individual sites, it was clear that there was considerable variability in the degree to which restored sites were reinvaded by targeted invasive species. Potential causes of this variation may be related to differences in management frequency between first restoration and 2009, or to site characteristics such as soil factors, adjacent land use, and propagule sources.

Successional Trajectories

The differences in woody understory composition and structure discovered by this study indicate divergent successional trajectories between restored and unrestored sites, and a lasting effect of restoration treatment after 15-20 years. Present differences in cover, stem density, and species composition observed between restored and unrestored sites create very different environments for the germination, establishment and growth of native plants, setting the stage for different pathways of vegetation development in the future.

The abundance and distribution of tree species in the Northeastern United States have been substantially changed in the past 300 years by agriculture, urbanization and other intensive land uses (Fuller et al. 2006; Foster, Motzkin, and Slater 1998); in the long view, soils and forests of the region are relatively recently developed following the retreat of the glaciers of the last ice age. Current climate is no more likely to be stable than that of the past; urban forest patches have been proposed as model study sites for effects of global warming due to their exposure over decades to elevated temperature, carbon dioxide, nitrogen deposition and ozone (Carreiro and Tripler 2005). Like all plant communities, urban forest patches are in a constant state of change following and in response to disturbance. It is unknown, given the novel assemblages currently inhabiting urban forests like those described here, how long current communities will persist. In some forest systems, rapidly growing, short-lived species are replaced by shrubs of intermediate age, and then by long-lived trees (Chapin et al. 1994); without intervention in the long run, *R. multiflora*, *A. brevipedunculata* and *C. orbiculatus* may all be replaced by forest canopy trees (perhaps the global cosmopolitan *Ailanthus altissima*), or decreased in abundance by the introduction or evolution of pests, pathogens or predators. However, since these species have the potential to reduce both the extent of and native diversity in remnant patches of urban forest, and since these forest remnants are disproportionately important for the ecological and social benefits they offer in the urban environment (Ehrenfeld 2000), managing them to maximize current and near-term biodiversity and ecosystem function seems prudent.

The restored forest patches studied here were richer and more diverse in tree species than forests that were not restored. Fields and forest gaps developing from an open, disturbed state to closed-canopy forest tend to increase in tree richness and diversity over time, while decreasing in ground-layer cover and diversity as competition for light and nutrients increase (Tilman 1985). The greater mean tree basal area, mean number of tree stems and native tree saplings in restored sites (Figures 1.7-1.9, 1.12, 1.18, 1.36 and 1.37) suggest that restoration has accelerated the development of a multi-layered, closed forest canopy with native tree regeneration in areas formerly dominated by invasive woody species that had little to zero native tree regeneration. However, likelihood of these forests coming to resemble more pristine or historic conditions of composition and structure in the next 200-300 years is small. In the urban environment, where exotic species introductions are frequent and urban environmental conditions cause the conditions for plant germination, establishment and growth to differ from those of non-urban forests, long-term changes in species composition may be more difficult to achieve than structural changes like the establishment of multiple forest strata.

The invaded forest patches examined here are currently in a relatively early stage of development following disturbance, whether the most recent major disturbance was restoration, regrowth of a cut woodlot, the 1978 burning of a stolen automobile, abandonment of pastures, or a hurricane. The direction that successional processes take is in all cases likely to be influenced by effects of urban heat island warming, atmospheric deposition of pollutants, altered carbon and nitrogen dynamics, altered species pools of not only plants but other organisms that are their predators, dispersers, pollinators, mutualists, and pathogens (Pickett et al. 2011; Karpati et al. 2011; Matteson and Langellotto 2009; Williams 1911). Ecological succession is a long-term process, and the rate and direction of successional change depends upon differences in site and species availability, and on the differences in performance of those species (Pickett, Collins, and Armesto 1987; Pickett, Meiners, and Cadenasso 2011). In cities, regional and local environmental context combines with urban changes to biophysical patterns and processes, and with ongoing change in human activities. Social factors, from trends in harvest of wild plants by different ethnic groups to political priorities, strongly influence the conditions for plant community dynamics over time. The degree to which an urban park or green space is protected or restored is related to its social context (Pickett et al. 2008).

Urban Ecological Succession

The model of forest succession employed in the restoration work described in this study conceptualized invaded areas as forest gaps, and focused on shade and rapid canopy establishment as the primary means of effecting restoration. Understory plants were not part of the original plan. Three major bodies of knowledge about vegetation change over time are applicable to these forests: primary forest succession (e.g. Chapin et al. 1994), succession in old fields of the Eastern U.S (Egler 1954; Odum 1960; Bazzaz 1975; Pickett and McDonnell 1989), and forest gap succession, which describes processes in gaps created in intact forests (Catovsky et al. 2006). All three of these frameworks for understanding succession are applicable to urban remnant forests; current vegetation in the Northeastern United States from the terminal moraine of the Wisconsin Ice Sheet northward has undergone primary succession, and since cities generally expand from agricultural settlements, in many cities some fraction of land now covered by buildings or parks was previously put to agricultural use. Both biotic and edaphic legacies of agriculture (Foster and Aber 2006) are part of the history of present-day urban wild places. Forest gaps are created by wind, fire, and other causes in the urban environment. Competitive dynamics, seedling banks, priority effects, facilitation, inhibition and tolerance, and the importance of light and nutrient availability are essential to understanding forest change, and what has been learned about succession from these systems can and should be applied in the urban environment. The processes that influence vegetation development over time (Pickett, Meiners, and Cadenasso 2011) are the same in large forests, old fields and urban parks. However, the outcomes of vegetation development in the urban environment will be different from that of non-urban environments that once shared a regional set of species and environmental conditions.

While succession following agricultural abandonment or forest gap creation may follow similar patterns if undisturbed, the frequency, intensity, type and duration of disturbance events are of a different magnitude in urban environments (Alberti 2005). Sites are seldom left even relatively undisturbed by direct and indirect effects of shifting use patterns. Current site conditions are highly heterogeneous at the landscape scale (Cadenasso, Pickett, and Schwartz 2007; Cadenasso and Pickett 2008), and a number of different land uses may have followed each other in a given site, each with its own legacy of biotic and abiotic conditions. These legacies include frequent introduction of new species. Forest gap succession quite rightly presupposes a surrounding forest. However, the urban matrix subjects urban forest patches to different conditions in terms of propagule quantity and identity, buffering from other land uses and habitat types, and local climate and hydrology (Ehrenfeld 2000). Although park forests may experience less frequent direct disturbance than some other urban land types, pre-urbanization species composition and ecological processes may be altered or absent, and resulting communities may be stable and resilient (Suding, Gross, and Houseman 2004). These limitations should be kept in mind when using successional frameworks to guide urban ecological restoration.

Conceptual frameworks have been developed to describe urban ecosystems that incorporate the pervasive human impacts that are typical of cities (Grimm et al. 2000; Pickett et al. 2009). Forest patches in cities are subject to urban environmental conditions, and urban ecological restoration will benefit from expanding its focus to include both the social and the ecological dimensions of urban systems in order to better predict and manage long-term outcomes.

Implications for Urban Ecological Restoration

Environmental conditions change over time due to factors both intrinsic and extrinsic to the biota, and communities change in response. This model of ecological succession is central to restoration ecology. Many restoration plans focus on an end state, assuming that a brief disturbance that adds and removes individuals of species to resemble a community that often develops into the target state will lead to the desired end. However, changed conditions may mean that this never occurs, or is partially realized.

Urban environmental conditions affect the development of vegetation over time, and thus a different approach to setting goals and evaluating success for urban ecological restoration is required. Altered disturbance regimes, species pools, local climate, and changes in soil chemistry, structure and biota will differentially affect site conditions, species availability and species performance, affecting community composition and architecture (Pickett, Meiners, and Cadenasso 2011). The results will be novel assemblages, and the outcomes of interactions in these novel assemblages are currently difficult to predict (Hobbs, Higgs, and Harris 2009; Kowarik 2011). In the urban environment, vegetation development will include a large proportion of recently disturbed sites, and it is not likely to converge upon pre-urbanization habitat types or species compositions. Its future composition will be determined in part by the novel assemblages and interactions that draw upon a global species pool.

Managing succession for urban ecological restoration requires recognizing that a trajectory that differs from pre-urbanized conditions may not constitute failure. It also requires a species-specific approach to evaluating the effects of newly introduced organisms, combined with management of the frequency, intensity, and nature of disturbance, and with long-term management of long-term processes. New ways of understanding, valuing, restoring and preserving the biodiversity and ecological function of natural areas in cities are needed. Solutions to urban environmental problems must be both social and ecological in nature.

Adaptive Successional Phasing: Time as a Tool

To address the challenges of ecological restoration in the urban environment, I propose using adaptive successional phasing to plan to direct urban ecological restoration. Trajectories of development of novel assemblages are unknown. To preserve and enhance biodiversity in the urban environment, I suggest that identifying native species that are surviving and thriving under current conditions should be a first step toward restoration. These species can be used to understand limiting present and past conditions that may need to be ameliorated before other species can be re-introduced, and their common set of requirements and traits can be used to identify additional native species that may be dispersal-limited by the urban matrix. These species may be useful for initiating restoration under current conditions, as they will provide habitat value to native plants and animals in the short term.

Composition and structure should be expected to change over time, and should not be expected to arrive at a stable state. Vegetation development is a long-term process, and so restoration should be considered a long-term project. Processes within the habitat will also cause change, and may provide habitat conditions for species that would not thrive early in the restoration process. Additional species introductions should be planned for as their requirements are met by succession.

This process begins by understanding current conditions. Species well-adapted to disturbed sites differ in their niche requirements from those that occupy older, more complex, shaded environments. Species typical of less disturbed, older sites are often introduced to open, disturbed sites as a restoration treatment. It is perhaps unsurprising that these individuals often die or are outcompeted by others that are better adapted to current conditions. Current soil, temperature, and hydrologic conditions must be taken into account in urban environments, as these are often drastically changed. Native species that are currently surviving and thriving in cities provide both indicators of current conditions that may not be immediately obvious and have long-evolved relationships with other regional and local biota.

In this study, native species were found thriving in heavily invaded sites. Perhaps unsurprisingly, these were fast-growing, sun-loving species such as *Prunus serotina* and *Robinia pseudoacacia*. Native *Rubus* and *Vitis* species were common in the open vinelands created by invasive plants. When all sites were considered, additional regenerating native tree species were found to be abundant in urban forest patches: *Acer rubrum*, *Quercus palustris* and *Carya cordiformis* are swamp species that tolerate a wide range of soil conditions, including low oxygen. Whether thriving native species like these could be used effectively to compete with invasive plants as part of a restoration strategy remains to be investigated. Treating the existing urban species pool as a palette may help to identify species that may thrive in particular restoration sites, where initial conditions may include high light and soil conditions typical of urban sites. Additional analysis of species traits and tolerances in relation to conditions found in urban landscapes may yield further insight, increasing the ability to predict what species will succeed.

Failure of regeneration is common in urban and suburban forest patches due to reduced natural disturbances and increased human disturbances (Guldin, Smith, and Thompson 1990; Broshot 2011). In New York City, Rudnicky and McDonnell (1989) found that increased abundance of non-native trees in the ground layer and understory was due to arson, trampling, and vandalism in addition to non-human disturbances. Both sanctioned and unsanctioned uses of urban forest patches will affect outcomes of urban ecological restoration, and effects of human disturbance, such as soil compaction by trampling, may be compounded by natural disturbances like large storms. Anticipating disturbance must be part of planning and management in urban natural areas.

One of the major shortcomings of current efforts toward ecological restoration, both in the urban environment and elsewhere, is the idea that somehow long-term ecological processes can quickly be “set right” and that no further management will be needed. This perception is detrimental to the long-term effectiveness of ecological restoration. While the amount of effort required should decrease once disturbances and their legacies are removed or remediated, ensuring that the initial investment is not overwhelmed by unanticipated changes requires monitoring and adaptive management.

Target trajectories for restoration are often least-disturbed regional reference sites, which have had decades or centuries of soil and biotic interactions since they may last have experienced major disturbance. Restoration sites, in contrast, may be cleared of vegetation and their soils disturbed in the process of site preparation. In fragmented and frequently-disturbed urban locations, the idea that a restoration is finished once planted is especially short-sighted. For urban habitats to provide long-term ecological benefits to the residents of cities, they will need long-term protection, management and restoration. Successional change plays out over the lifetimes of multiple generations of organisms. Forests are structured by long-lived organisms; individual red oak trees may not produce seeds until they reach 50 years of age. Where the dominant organisms may live 200-300 years, ecological processes their life cycles control can necessarily be long-term in nature.

While the reality for many land managers is that funding comes in short bursts and does not cover maintenance beyond a brief initial phase, altering trajectories of ecological community development over time requires a long-term perspective and long-term engagement. If it is possible to embrace the long-term nature of vegetation development, time can become a tool. Using principles of ecological succession, restoration plans can be developed that incorporate and utilize change over time.

In this study, the success of native species that compete well in open, sunny environments suggests that when ecological restoration involves creating a bare, open soil condition where there is little shade, species that are well-adapted to those conditions should be used to establish the initial phase of the restoration. More shade-tolerant species that compete well under higher-nutrient and lower light conditions should be introduced in a subsequent phase to reduce the likelihood of reinvasion. The species planted most frequently in the restoration efforts examined here, slower-growing *Quercus rubra*, was not as abundant in restored plots as more shade-intolerant planted and spontaneous native species. Adaptive phasing over time would allow for more shade-tolerant species like red oak to be introduced later in forest development. In disturbed sites, shrubs have been shown to have facilitative effects in the restoration of trees (Gómez-Aparicio et al. 2004; Gómez-Aparicio 2009); in light of this, planting native shrubs in advance of tree planting may be another strategy to consider.

Larger-scale spatial and temporal changes, including climate and sea level change, must also be anticipated in urban restoration planning. Cities are already commonly several degrees warmer than their surroundings, and looking to the native plants of the nearest areas with warmer climates may be appropriate in planning for both urban heat island survival and a warming climate. Distribution shifts are predicted for many species coming decades (Woodall et al. 2008; USFS 2013) The most abundant trees observed in this study, including sassafras, black cherry, black locust, and tulip poplar, are species with distributions that extend into the southern U.S. (Burns and Honkala 2013).

Conclusion

The restoration effort described here has achieved its central goals. Invasive species removal followed by planting has resulted in forests that are on a markedly different trajectory of vegetation development from forests that were not restored, resulting in persistent structural and compositional shifts, greatly lower invasive species abundance, a more complex forest structure, and greater native tree recruitment after 15-20 years.

Neither of these divergent trajectories, however, is likely to result in a forest resembling New York forests of the pre-colonial period. Drastically changed urban climate, hydrology, soils, disturbance regimes, and species introductions affect the development of vegetation over time. A different approach to setting goals and evaluating success for urban ecological restoration is required. In the urban environment, there will continue to be many recently-disturbed sites, and future community composition will be determined in part by novel assemblages and interactions that draw upon a global species pool.

To preserve and enhance biodiversity and the survival of native species in one of the world's largest metropolises is a challenge that requires a particularly urban approach to ecological restoration. Managing urban wild areas for ecological benefits requires recognizing that a successional trajectory that differs from pre-urbanized conditions may not be failure. It requires a species-specific approach to evaluating the effects of newly introduced organisms, and management of the nature, frequency, intensity, and nature of disturbance. Disturbance that is characteristic of urban areas will require future restorations. Long-term management is necessary for long-term processes, and to understand the nature of the changes and to make informed decisions about management, long-term monitoring, research, and continuity of institutional memory are also important. Recognizing that the urban environment is different from non-urban sites, it is possible to work with current conditions to improve ecosystem function (Palmer et al. 2004).

Restoring ecological health and function in urban areas is both an urgent concern and a long-term strategy for the health and well-being of more than half of humanity. It is also important to the planetary biosphere, as the number and size of cities rises exponentially across the globe. The number of urban inhabitants is projected to double by 2050, and more than half of the land projected to be urban by 2030 has yet to be built (United Nations 2012). New ways of understanding, valuing, restoring and preserving the biodiversity and ecological function of natural areas in cities are needed.

An urban approach to ecological restoration must value existing habitats, and look to thriving native species as both indicators of environmental conditions and candidates for restoring sites exhibiting typically altered urban conditions. Time can be used as a tool for restoration with an adaptive successional phasing approach to restoration. By anticipating disturbance and ecological succession, management can be targeted as ecological processes unfold. Solutions to urban environmental problems must be both social and ecological in nature. An urban approach to directing succession for ecological restoration is needed to avoid selecting goals that are unattainable in cities. It will also improve our ability to preserve and enhance urban biodiversity, both in the near term and in the lifespan of oak trees.

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CHAPTER 2

Management effort and urban environment influence the long-term fate of ecological restoration in urban woodlands

ABSTRACT

Urbanization introduces a suite of stressors to ecological systems, transforming the biophysical landscape. More than half of the world's population now lives in cities. Urban populations are expected to double by 2030, and 60% of the area projected to be urban land in 2050 has yet to be built. Recognizing that some ecosystem services must be provided at the local level, municipalities are turning to ecological restoration of urban forests to provide essential ecosystem services such as air and water purification, heat island reduction, and health benefits. Restoration of remnant urban forest patches is an approach that has been recently embraced.

To test the idea that management effort and effects of urbanization are important to the long-term success of ecological restoration in urban forest remnants, I examined forests invaded by a suite of woody invasive plants 15-20 years after restoration. I compared these restored areas with forest patches that were similarly invaded but not restored during the same time period, and with a less-disturbed urban forest remnant that was not invaded at the time of the initial restoration. I examined relationships between management effort, soil surface characteristics, indicators of disturbance, adjacent land use, and plant community composition between all site types and among restored sites to examine factors influencing variability in restoration outcomes.

There were significant differences in community composition among restored, unrestored and less-disturbed sites in all forest strata, indicating that restoration treatment had significant and persistent effects on vegetation after 15-20 years. Differences between restored and unrestored plant communities were most strongly associated with whether or not a site was restored, and with soil surface characteristics related to plant invasion and human impacts. Among restored sites, differences in plant community composition were strongly associated with restoration effort, assessed by the number of dates on which sites were treated 1988-2009, and with soil surface characteristics related to both the urban environment and invasion. These findings indicate that ongoing management effort is important to long-term effects of ecological restoration. They also illustrate the importance of the urban environment to key factors influencing vegetation dynamics: soils, propagules and disturbance regimes. Models used for ecological restoration in urban environments should take these effects into account to improve restoration effectiveness.

KEYWORDS

adaptive management, community ecology, ecological restoration, long-term research, New York City, parks, plant ecology, urban forest

INTRODUCTION

Urbanization changes ecological systems, transforming the biophysical landscape (Williams et al. 2009; Gaston 2010; Niemelä et al. 2011). More than half of the world's population now lives in cities, and urban populations are expected to double by 2030; 60% of the area projected to be urban land in 2050 has yet to be built (United Nations 2012). Recognizing that some ecosystem services must be provided at the local level, municipalities are turning to ecological restoration to provide essential ecosystem services such as air and water purification, heat island reduction, and health benefits (e.g. City of New York 2007; Gobster 2007; Westphal 2010).

To test the idea that management effort and effects of urbanization are important to the long-term success of ecological restoration in urban forest remnants, I examined forests invaded by a suite of woody invasive plants 15-20 years after restoration. I compared these restored areas with forest patches that were similarly invaded but not restored during the same time period, and with a less-disturbed urban forest remnant that was not invaded at the time of the initial restoration. I examined relationships between management effort, soil surface characteristics, indicators of disturbance, adjacent land use, and plant community composition between all site types, and among restored sites to examine factors influencing variability in restoration outcomes.

Restoration Ecology in the Urban Environment

As cities and their proportion of global land cover expand, improving the ability of urban land to provide ecological benefits is increasingly critical. Urbanization presents an array of stressors to ecological systems, resulting in habitat transformation and fragmentation, altered climate, soils, and hydrology, and frequent disturbance (Zipperer and Pickett 2012; Niemelä et al. 2011). Cities are sites of frequent species introductions, with high proportions of non-native and invasive species (Sukopp, Hejný, and Kowarik 1990).

As a result of these factors, cities fail to provide many ecosystem services (United Nations 2012). While many of these services, such as food and building materials, can be imported, not all ecosystem services can be outsourced. Clean air, local climate, and other amenities of urban green space must be provided at the local level. Fragments of habitat in urban areas are therefore disproportionately important as islands of biodiversity, habitat, psychological wellbeing, climate amelioration, and other environmental benefits (Barton and Pretty 2010). Ecological restoration of urban forest patches is being adopted by municipalities as a strategy to provide improved air quality, heat island cooling, and carbon sequestration (City of New York 2007; City of Seattle 2011; Gobster 2007).

The practice of ecological restoration is an experimental one. Although restoration ecology has the potential to test ecological theories by manipulation of ecosystem properties (Bradshaw 1987), rigorous experimental design is not incorporated in the initial phases of many restoration projects, and long-term outcomes have thus far rarely been analyzed (Falk, Palmer, and Zedler 2006). Many restoration projects are funded over a short time period consisting of site preparation, planting, and monitoring of initial establishment. High-resolution data on prior conditions is often lacking, and even when long-term management consistent with restoration goals is possible, funding and personnel for the type of systematic long-term data collection and analysis that supports adaptive management may not be available. Research that examines the long-term outcomes of restoration activities is needed to increase understanding of how ecological interventions affect long-term processes like ecological succession (Falk, Palmer, and Zedler 2006). This is even more the case in urban environments, where additional factors resulting from urbanization may change the way that ecosystems respond to restoration. Models of ecosystem processes developed in more pristine environments may have limited applicability to cities (Ehrenfeld 2000).

Assessment of long-term outcomes of ecological restoration must consider the initial target state toward which the restoration was aimed. Many efforts toward ecological restoration envision targets in terms of a pristine reference site or climax state (SERI 2004), but a more dynamic, multi-dimensional approach is needed. In urban areas, it must also take into account spatial heterogeneity, novel disturbance patterns, and the interaction of the social with the ecological. Integrating contemporary succession theory with urban environmental conditions may improve the effectiveness of ecological restoration in cities.

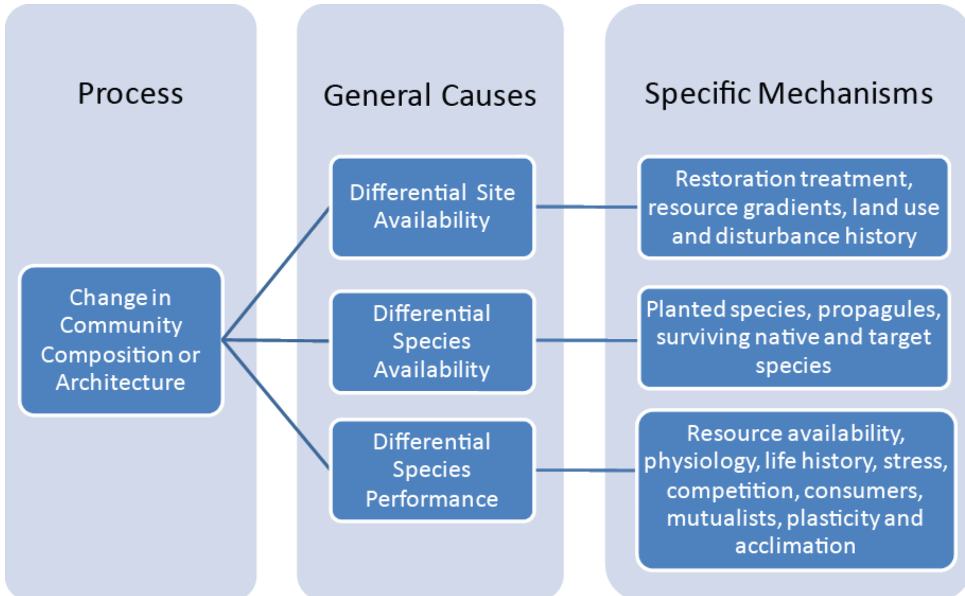

Figure 2.1: Factors influencing successional processes in ecological restoration, after Pickett et al. (2011) and Pickett and McDonnell (1989).

In ecological restoration, management can be considered a disturbance that is intentionally used to alter successional processes by changing site or species availability. In restoration efforts aimed at reducing the dominance of invasive plant species, the disturbance is often removal of undesired plants and/or addition of desired species. In management, as in other types of disturbance, timing and frequency matter (Luken 1990). The success of efforts to restore areas dominated by non-native plant species is highly variable, and factors influencing success rates are generally not well understood (Pluess et al. 2012). In urban environments, even less is known about restoration success.

Ecological Restoration of Forests in New York City Parks

In the Northeastern United States, extensive forest clearing in the early colonial period was followed by abandonment as land more favorable for agriculture became available with westward U.S. expansion, resulting in subsequent reforestation (Foster and Aber 2006; Cronon 1983). Plant communities of many of these abandoned fields and pastures have followed a similar successional pattern, passing through phases dominated by annual herbs, perennials, shade-intolerant and shade-tolerant trees in turn, eventually becoming shady, closed-canopy forests where undisturbed (Foster, Motzkin, and Slater 1998; Fuller et al. 2006). These re-grown forests also now include a set of introduced species, some of them invasive species that transform ecosystems, altering resource availability for many other organisms.

As New York City has expanded and agglomerated to become one of the United States' largest urban areas, small islands of habitat have been preserved or allowed to revegetate within the urban matrix by a combination of planning, accident, and philanthropic largesse. Within the City itself, the fraction of public land not now in ballfields and playgrounds is managed by a division of the New York City Department of Parks and Recreation (NYCDPR), the Natural Resources Group (NRG). NRG oversees more than 10,000 acres (4,000 ha) of forest, woodland, freshwater wetland and salt marsh ecosystems, and conducts ecological restoration of forests, salt marshes, riparian zones, meadows and other habitat types (NRG 2013).

NRG initiated its first science-based ecological restoration of urban woodlands in 1985 (NRG 1985). These early interventions, among the first of their kind to be undertaken, removed invasive woody plants and planted native trees in the resulting clearings (NRG 1991). Species targeted by this program included porcelain berry (*Ampelopsis brevipedunculata*), oriental bittersweet (*Celastrus orbiculatus*), multiflora rose (*Rosa multiflora*), and Norway maple (*Acer platanoides*). Invaded vinelands appeared to be expanding as trees on the edges of invaded areas fell under the weight and shade of exotic woody vines. Using a gap-succession model for forest regeneration, managers predicted that planted native tree seedlings would eventually change light and other resource availability such that competition would favor native species better adapted to the understory, creating an unsuitable environment for establishment invasive plants less tolerant of shade. This study examines the fate of these early forest restoration efforts after 15-20 years. Restoration sites were located in mixed oak-hickory upland forests within New York City Parks.

Hypotheses

Prior study showed invasive species removal followed by planting resulted in persistent structural and compositional shifts, significantly lower invasive species abundance, a more complex vertical forest structure, and greater native tree recruitment in restored sites compared to invaded sites that were not restored (Chapter 1). These findings indicated that successional trajectories of vegetation development had diverged between restored forests and invaded forests that were not restored.

Variability in outcomes among restored sites remained to be explored. To examine the sources of differences between restored sites, and to test the hypothesis that management effort and urban environmental effects are important to restoration outcomes in urban forest patches, I examined relationships between management effort, soil surface characteristics, indicators of disturbance, adjacent land use, and plant community composition between restored, unrestored and less disturbed forests, and among restored sites to examine factors influencing variability in restoration outcomes.

METHODS

Thirty sites were sampled in restored New York City Park forest remnants in summer 2009, ten in each of three parks (Chapter 1). All of these sites were treated in the 1990s by removal of woody invasive species (*A. brevipedunculata*, *C. orbiculatus, R. multiflora* and *A. platanoides*), and were planted with native tree seedlings. In 2010, another thirty plots were sampled (ten per park) in New York City park forests that were in a similarly degraded condition at the time of the original restoration (1988-1992), but which were not restored. In each 20 m x 20 m plot, the DBH of all tree stems was measured. All woody saplings, vines and shrubs were counted in three 5 m x 5 m subplots. Four 10m line transects were established, from each corner toward plot center, along which cm of intercept of all ground-layer vegetation were measured (see Chapter 1 for a full description of methods).

To compare these sites with a less-disturbed urban forest type, an additional ten sites were also sampled in 2010 at the New York Botanical Garden (NYBG), in old-growth forest sites where invasive species were not dominant in the 1990s. These less-disturbed sites were located in the Thain Family Forest, a 50-acre (20-ha) remnant of native hardwood forest dominated by oaks (*Quercus spp.*), tulip poplar (*Liriodendron* *tulipifera*), sweet gum (*Liquidambar styraciflua*), and maples (*Acer* spp.) (NYBG 2001). The plots sampled were in the location of Permanent Forest Reference Plots originally established as part of a series of Urban Rural Gradient Ecology studies (McDonnell et al. 1997). NYBG plots were sampled using the same protocol as the treated and untreated New York City Park sites (Chapter 1).

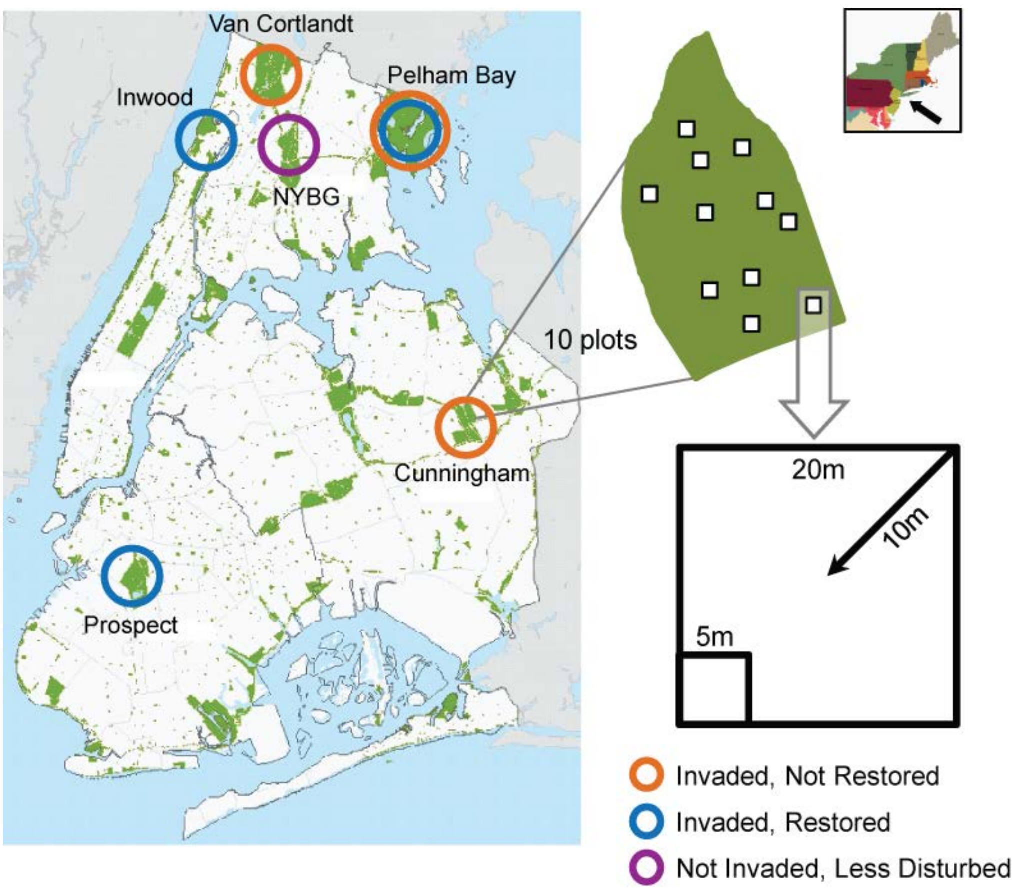

Figure 2.2: Study park locations and plot design. Restored: Inwood Park, Manhattan; Pelham Bay Park, Bronx; Prospect Park, Brooklyn. Not restored: Van Cortlandt Park, Bronx; Pelham Bay Park, Bronx; Cunningham Park, Queens. Not invaded in 1988, less disturbed: New York Botanical Garden, Bronx. Map: Craig Mandel, NRG.

Site characteristic data was collected for each plot, including hydrologic features, soil surface cover, and indicators of disturbance, and human impacts. Topography, animal activity, and adjacent land uses and land types were described. Site history information was compiled from maps, historical documents and communication with land managers. Canopy closure was measured using a spherical densiometer, and slope and aspect of the plot were recorded using a clinometer and compass.

Table 2.1: Site characteristics recorded in all plots.

Site Characteristic	**Features Recorded**
Hydrologic Features	Pond Stream Perennial surface water Ephemeral surface water Flood debris Gully Sheet erosion
Soil Surface Cover	Leaf litter Woody debris Rock outcrop Bare soil Built structure
Human Impacts	Garbage (on surface) Plant damage Built structure (ruined walls, wells, building foundations) Official trail Informal trail Tire track (bicycle, car) Campfire
Topography	Aspect Slope Surface texture
Animal Activity	Impacting whole plot
Adjacent Features (within 50 m)	Road Official trail Informal trail Parking lot Building Forest Lawn
Plot Description	Narrative characterization

Hydrologic features, soil surface cover types, and human impacts in each plot were estimated by visual assessment of proportional cover in quintiles. Where soil surface was bare, earthworm casting abundance was noted, and where woody debris were present their size was noted. Small woody debris was predominantly composed of stems of shrubs and vines; large woody debris was composed of tree limbs and stems. Aspect and percentage slope were recorded, and surface topography was described. Only animal activity affecting the whole plot, such as widespread herbivory or trampling, was recorded. Adjacent land uses were within 50 m of a plot edge. For roads and official trails, trail surface was noted and for roads, number of lanes was recorded.

Data Analysis

I created a relational database (MS Access 2007) by combining a NYCDPR Natural Resources Group database of pre-restoration vegetation descriptions, detailed NRG restoration activity logs, Prospect Park Natural Resources restoration activity, and Prospect Park Natural Resources long-term plot monitoring data. Field-collected data from 2009 and 2010 were added to this database, as were additional data describing initial restoration activities and early monitoring under the Urban Forestry and Education Program gathered from NYC Parks Natural Resources Group records.

Restoration treatment activities (> 6,000 records, 1988-2009) were categorized according to management type, including a) manual and mechanical removal, where invasive plants were removed by pulling, weeding, mowing, and other machine methods; b) herbicides, where invasive plants were removed using chemical means, such as foliar spray of large vines or cut-and-dab application to individual woody stems; and c) planting. Informative signage, education, planning, mapping and other restoration-related activities tied to individual sites were not systematically reported, so only three primary categories of restoration activity were included in the analysis. Miscategorized and incomplete entries were excluded. Target species were identified for all management treatments. Management effort was calculated from the number of days on which each treatment type was recorded in a management unit.

Statistical Analysis

Data describing the vegetation of each forest stratum by species and environmental variables were subjected to Canonical Correspondence Analysis (CCA) using CANOCO 4.53 (Ter Braak and Smilauer 2002a). Both species and environment data were included in a direct gradient analysis using environmental data to extract patterns from only the explained variation using CCA. Scaling was focused on inter-species differences, using biplot scaling with untransformed data. No samples, species, or environmental variables were deleted, weighted, or made supplementary, except outliers removed from ground layer analysis and environmental variables with no variability (zero values for all sites): perennially wet, pond, stream, gully, tire tracks, and campfire. Monte-Carlo permutation tests were used to evaluate both the significance of the first ordination axis and the significance of the canonical axes together, with 499 permutations under a reduced model. Permutations were unrestricted, that is, not restricted for spatial or temporal structure or for split-plot design. When restored and unrestored sites were compared, initial site type was used to describe restoration (restored / not restored / uninvaded); when comparing restored sites to one another, frequency of restoration type replaced site type in the analysis. In analysis of restoration effects on ground layer plant community composition, two plots with very poor drainage and one with non-restoration plantings were excluded from the analysis due to effects of these characteristics on ground layer community composition unrelated to restoration (Appendix 2).

Interpretation of Diagrams

The analysis described above produces ordination diagrams in CanoDraw (Ter Braak and Smilauer 2002a). The first two canonical axes are shown. Scale is relative, axes are composed of combined environmental variables, and distance between plot points approximates similarity of the plots' plant communities in composition and abundance of species (Ter Braak and Prentice 2004).

Distance between plot points in these diagrams approximates the dissimilarity of their species composition, measured by their Chi-squared distance. Site characteristic values can be approximated by projecting the location of a plot point onto the axis of an environmental variable's vector arrow; plot points are ordered by predicted increase of values for a particular environmental variable, in the direction of that factor's vector (Ter Braak and Smilauer 2002b).

In diagrams displaying both environmental variables and plots, plot points are arrayed in relationship to arrow vectors representing site characteristics. Arrows point in the expected direction of steepest increase of values of that variable. The length of each arrow indicates the proportion of the variability associated with that factor (value of eigenvector), and angles between arrows indicate correlation of environmental variables. The degree of correlation can be approximated by projecting the head of each arrow onto the axis of another variable's arrow. Longest vectors, indicating the environmental variables explaining the greatest proportion of the variation between plot plant communities, are shown in figures.

RESULTS

Ground Layer Vegetation

Ground layer plant communities of invaded sites that were restored differed from those of sites that were not restored. Restored sites were also more similar to less-disturbed forest sites than were invaded unrestored sites (Figure 2.3). Closer grouping of plot points indicates less difference between ground-layer communities in sites that were not restored than in both unrestored and less-disturbed forest sites.

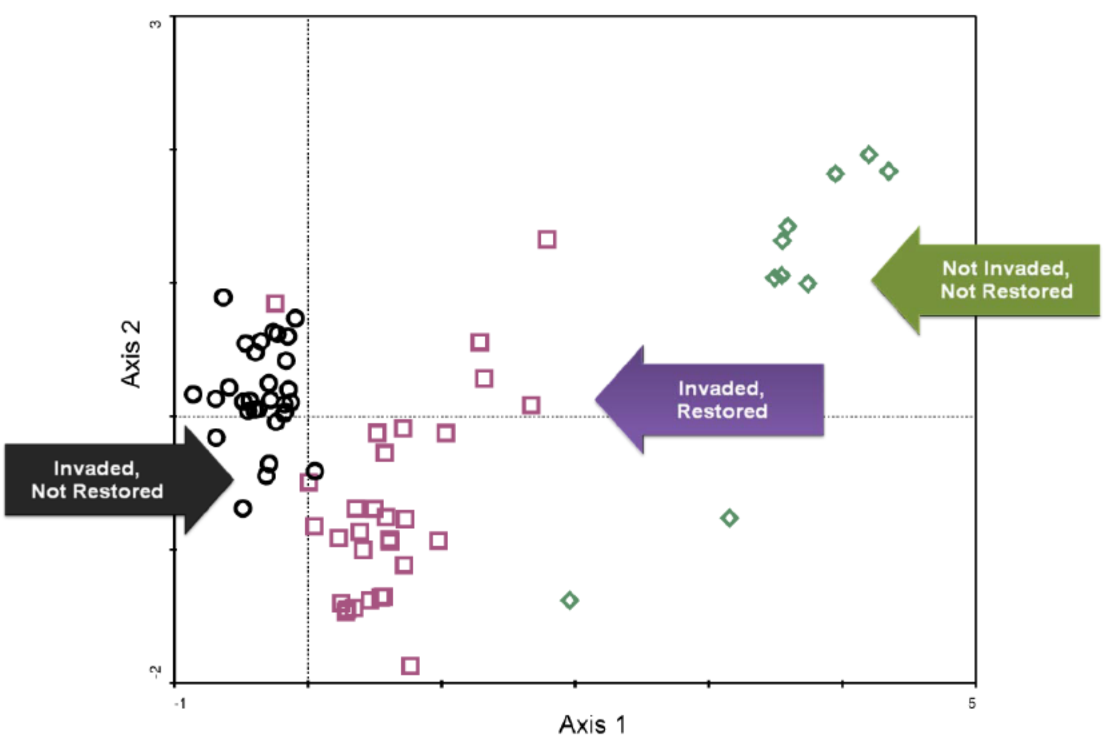

Figure 2.3: Restored (□) and unrestored (O) NYC Park plots, and less-disturbed forest (◇) plots at the New York Botanical Garden (NYBG), arrayed by ground layer plant community composition. NYBG plots outside the central group differ in species

composition from other NYBG sites (abundance of *Quercus* sp. and *Cyperus* sp. identified only to genus level).

Plant Community Characteristics of Forest Strata

Table 2.2: Plant community analysis of restored, unrestored and less-disturbed forest sites in New York City. ‡ Where Restored > Unrestored, the less-disturbed site was intermediate and not significantly different from either.

Forest Stratum	Attribute	Trend	Prob> ChiSq
Ground Layer	Shannon Diversity	(Restored & Unrestored) > Less Disturbed	.0309*
	Richness	(Restored & Unrestored) > Less Disturbed	.0309*
	Evenness	Less Disturbed > Restored > Unrestored	.0003*
	Total Cover	Unrestored > Restored > Less Disturbed	.0001*
Woody Understory	Shannon Diversity		.4965
	Richness		.4965
	Evenness		.9646
	Total # of Stems	Unrestored > Others	.0001*
Trees	Shannon Diversity	Restored > Unrestored‡	.0023*
	Richness	Restored > Unrestored‡	.0023*
	Evenness		.2675
	Total Stem Area	(Restored & Less Disturbed) > Unrestored	.0004*

Less-disturbed urban old-growth forests had less diverse, less species-rich ground layer plant communities with less total cover than both restored and unrestored sites. Their plant communities were more even in the distribution of cover by species. In the woody understory, sites that were not restored had a greater number of total stems than both restored and less-disturbed sites, primarily of woody vines and shrubs. Restored sites had more diverse and species-rich tree composition than sites that were not restored. Both restored and less-disturbed forests had greater total tree stem area than unrestored forests (Table 2.2).

Ground layer community composition: Restored and unrestored invaded sites

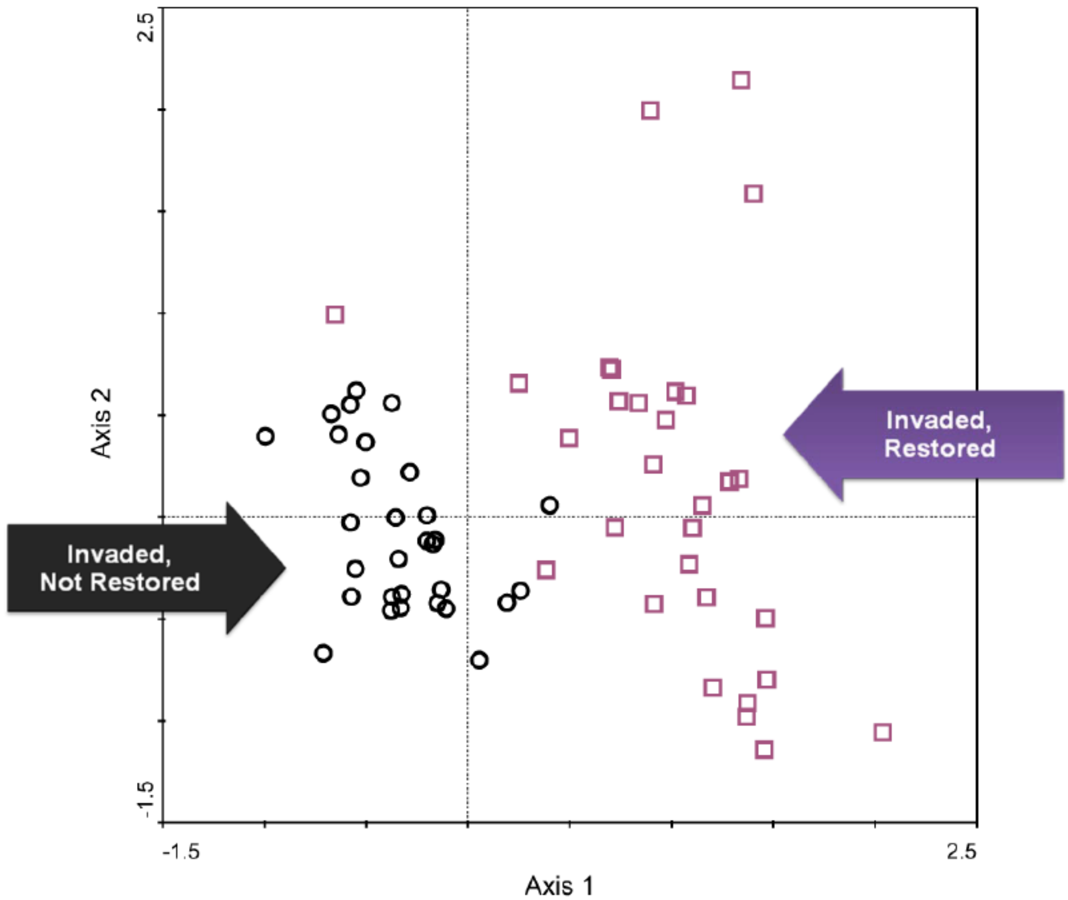

Figure 2.4: Restored (□) and unrestored (O) invaded sites arrayed by ground layer plant community composition.

When only invaded sites were considered, the separation of restored and unrestored ground layer communities was distinct (Figure 2.4). When invaded sites were arrayed in relationship to environmental variables (Figure 2.5), differences between restored and unrestored sites were most strongly associated with restoration treatment. Soil factors such as leaf litter and erosion were also important.

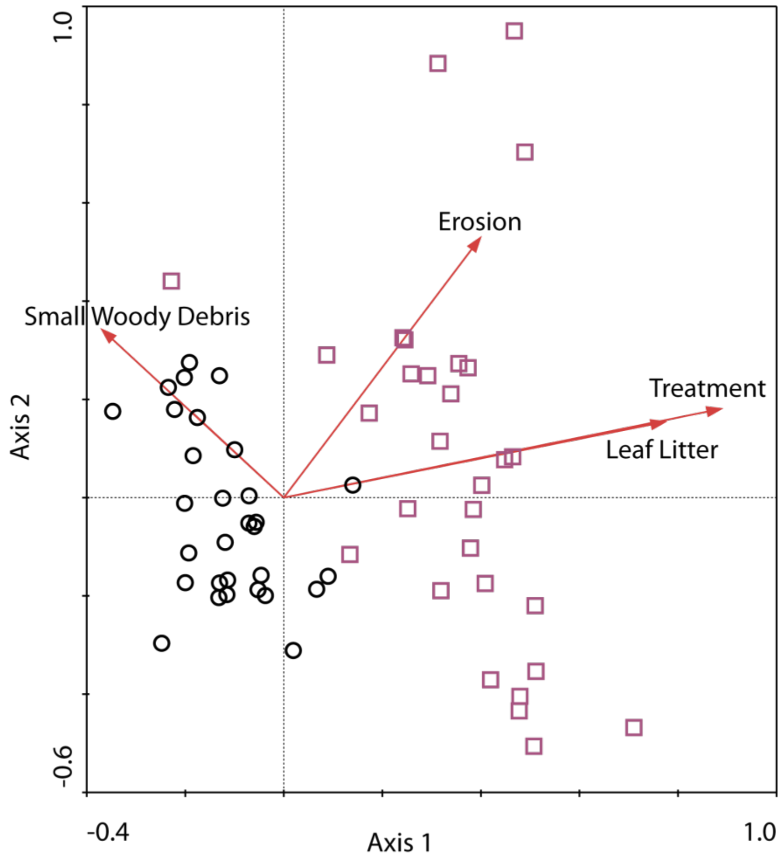

Figure 2.5: Site characteristics (→) associated with ground layer plant community composition in restored (□) and unrestored (O) sites. Site characteristics associated with ground layer plant community composition of invaded sites are displayed as arrow vectors, where arrow length indicates the proportion of the variability associated with that factor, angles between arrows indicate correlation of variables, and distance between plot symbols approximates the similarity of their ground layer plant communities. Longest vectors, indicating the environmental variables explaining the greatest proportion of the variation between plot plant communities, are shown.

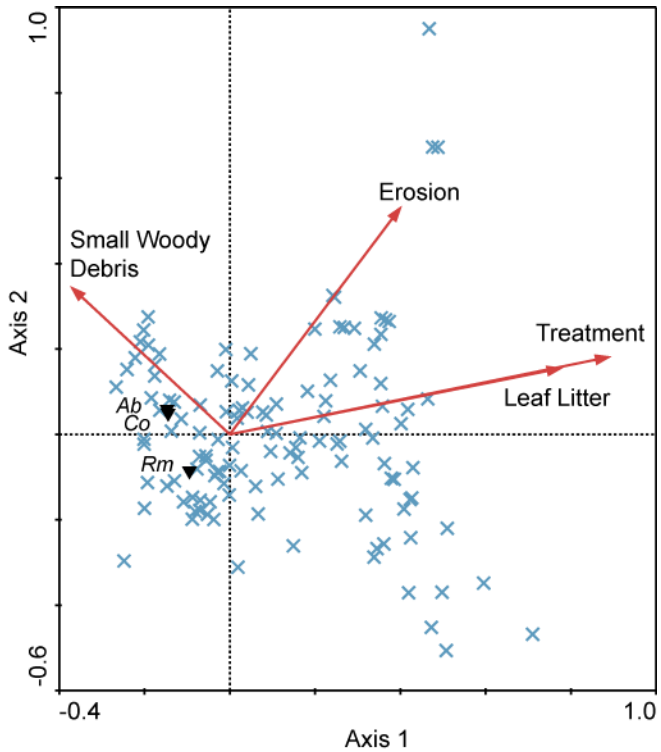

Figure 2.6: Ground layer species (✕) associated with environmental variables (→) in restored and unrestored invaded sites. Primary target species *Ampelopsis brevipedunculata* (Ab), *Rosa multiflora* (Rm), and *Celastrus orbiculatus* (Co) are identified with a triangle (▼).

When individual species were arrayed in relation to environmental variables, invasive species that were primary targets for removal (*Ampelopsis brevipedunculata*, *Rosa multiflora*, and *Celastrus orbiculatus*) were negatively associated with restoration treatment. They were also positively associated with soil cover by small woody debris generated by these species, rather than by leaf litter. Factors most strongly associated with difference between community assemblages are shown.

Ground Layer Vegetation and Management Effort in Restored Sites

The total number of days when restoration activities were carried out in the management unit was the factor most strongly associated with ground layer plant community composition. The association between restored plant community composition and restoration effort was especially strong for the total number of days spent planting, removing invasive species, and for and days spent on all restoration methods combined. Restoration treatments were negatively associated with erosion, small woody debris cover, bare soil cover, and steep slopes. Restoration treatment effects were positively associated with adjacent roads and official trails (Figure 2.7). Effects of the number of restoration treatment days in the first six months and first year following initial planting (not shown) were highly correlated with total restoration treatment.

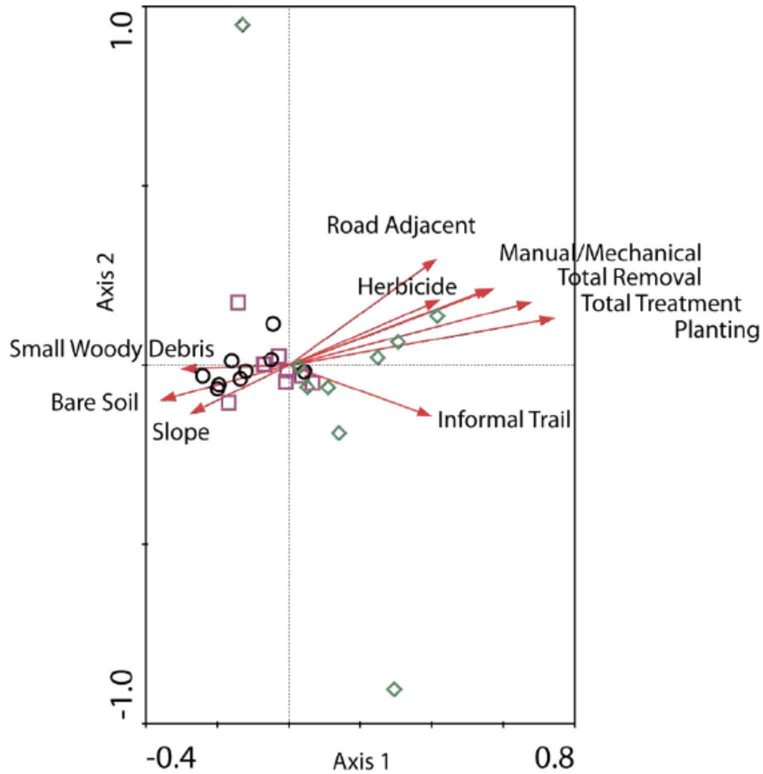

Figure 2.7: Site characteristics (→) in relationship to ground layer plant community composition of restored park sites: Pelham Bay Park (O), Inwood Park (□) and Prospect Park (◇). Restoration treatments were quantified by number of days recorded for each treatment action.

Distance between points representing plots from each site (Figure 2.7) indicates that variability of restored ground-layer plant community composition was greater in Prospect Park than in Inwood or Pelham Bay Park, as shown by distance between plots of each type.

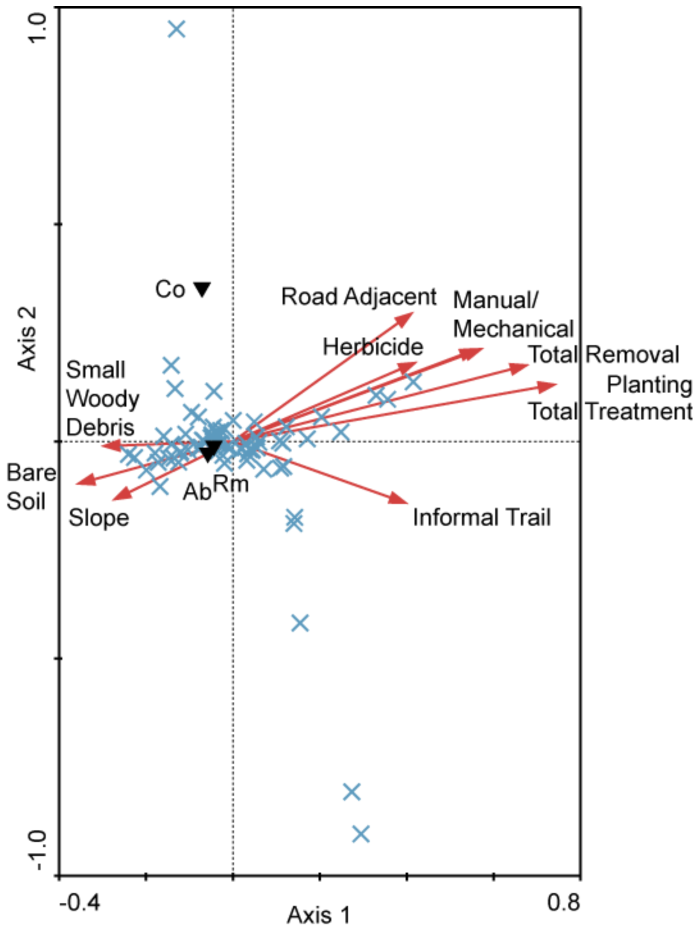

Figure 2.8: Species (✕) in the ground layer of restored sites arrayed in relation to site characteristics (→). Primary target species *Ampelopsis brevipedunculata* (Ab), *Rosa multiflora* (Rm), and *Celastrus orbiculatus* (Co) are identified with a triangle (▼).

Targeted invasive species in the ground layer (*Rosa multiflora*, *Ampelopsis brevipedunculata,* and *Celastrus orbiculatus*) were associated with a lower frequency of all types of post-restoration treatment, especially total planting. *Rosa multiflora* and *Ampelopsis brevipedunculata* were often associated with one another

(Figure 2.8). The greater variability in species composition of Prospect Park plots compared to Inwood and Pelham Bay was associated with the dominance of different species (Chapter 1 and Appendix 1A).

Woody Understory Community Composition

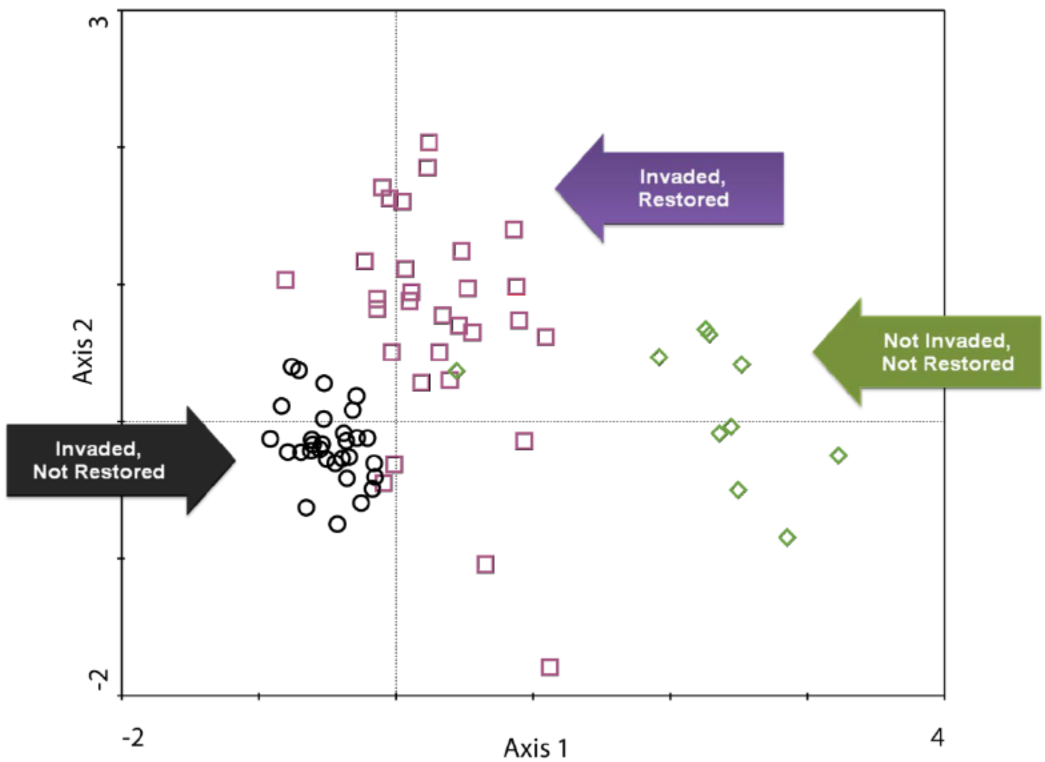

Figure 2.9: Restored (□) and unrestored (O) NYC Park plots, and less-disturbed forest (◇) plots at the New York Botanical Garden, arrayed by woody understory plant community composition.

Shrubs, vines and saplings in the woody understory showed similar trends to the ground layer. Restored, unrestored and less-disturbed forests grouped distinctly by site types based on plant community composition (Figure 2.9).

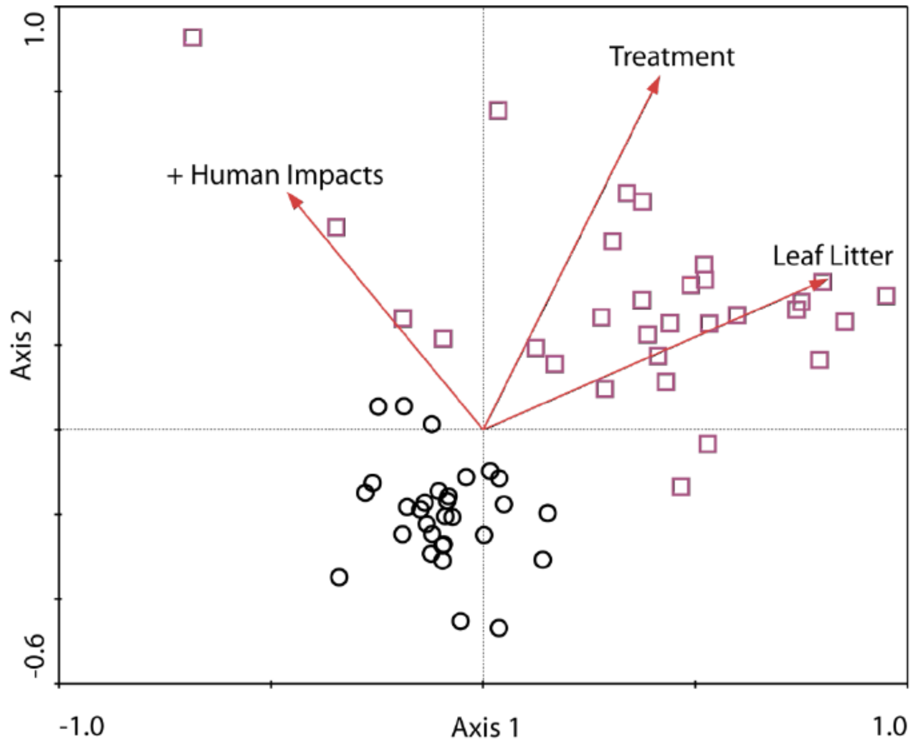

Figure 2.10: Site characteristics (→) associated with woody understory plant community composition in restored (□) and unrestored (O) invaded sites.

In invaded sites, the factors most strongly influencing community composition of the woody understory were similar to those of the ground layer: whether or not a site was restored, and soil factors such as leaf litter cover. Effects of intensive human impacts that created bare soil and soil compaction (such as camping and social gathering areas with associated trails) were associated with differences within site types, as were adjacent land uses (Figure 2.10).

Among restored sites, all categories of restoration treatment effort were associated with differences in woody understory community composition. Targeted invasive species were negatively associated with restoration effort. Soil surface factors such as leaf litter cover and small woody debris from woody invasive plants were also associated with plant community variability, as were adjacent mowed areas and disturbance factors such as informal trails and additional human impacts (Figure 2.11).

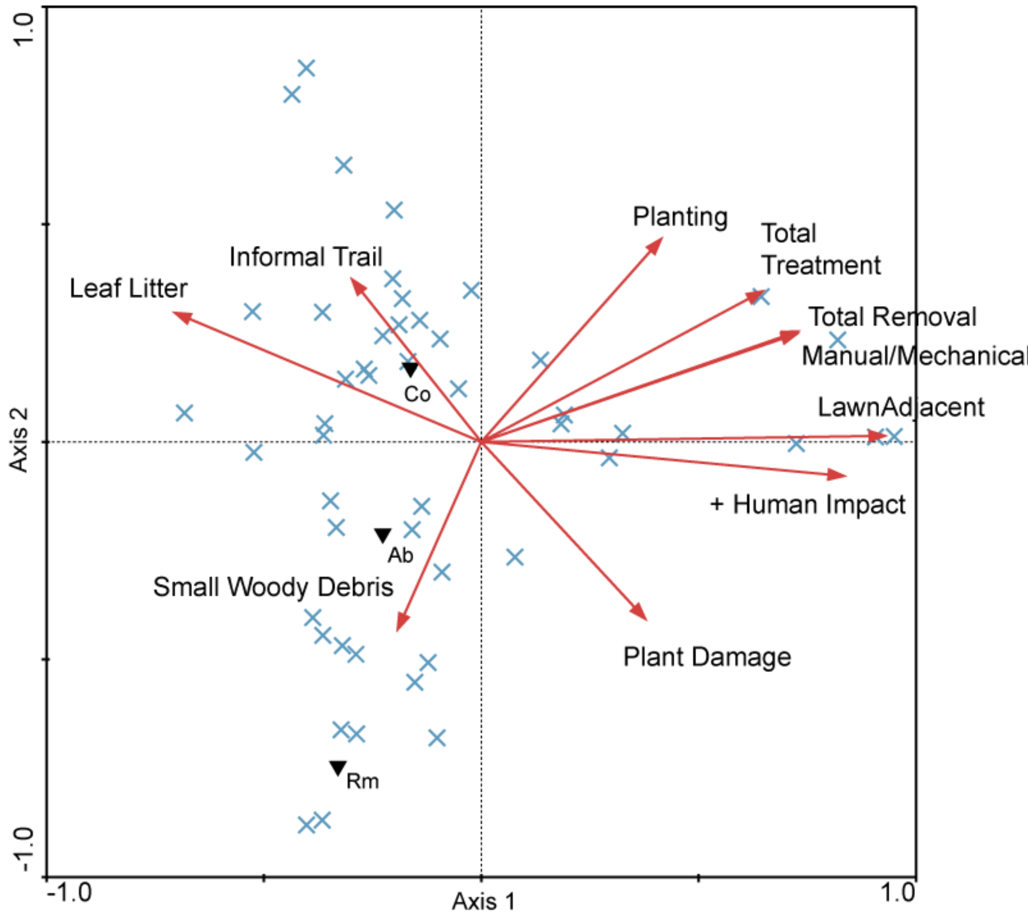

Figure 2.11: Woody understory species (✕) of restored sites arrayed in relation to site characteristics (→).Primary target species *Ampelopsis brevipedunculata* (Ab), *Rosa multiflora* (Rm), and *Celastrus orbiculatus* (Co) are identified with a triangle (▼).

Tree Composition

Tree community composition followed a similar trend to the ground layer and woody understory. Restored sites differed from unrestored ones, and were more similar to the uninvaded urban forest in their composition. Differences between site types in tree composition were less different than in the ground and woody understory strata (Figure 2.12).

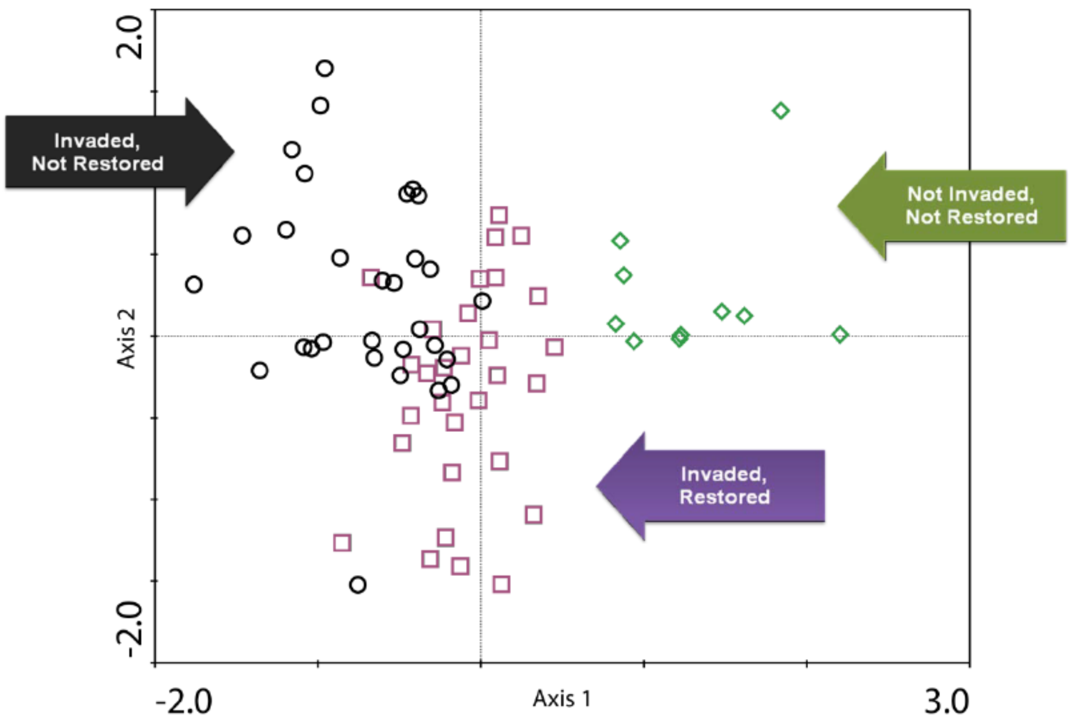

Figure 2.12: Restored (□) and unrestored (O) NYC Park plots, and less-disturbed forest (◇) plots at the New York Botanical Garden, arrayed by tree species composition.

Tree species composition: Restored and unrestored invaded sites

Among invaded sites, restoration treatment was associated with differences in tree composition. Soil surface characteristics and adjacent land use were also important to tree composition.

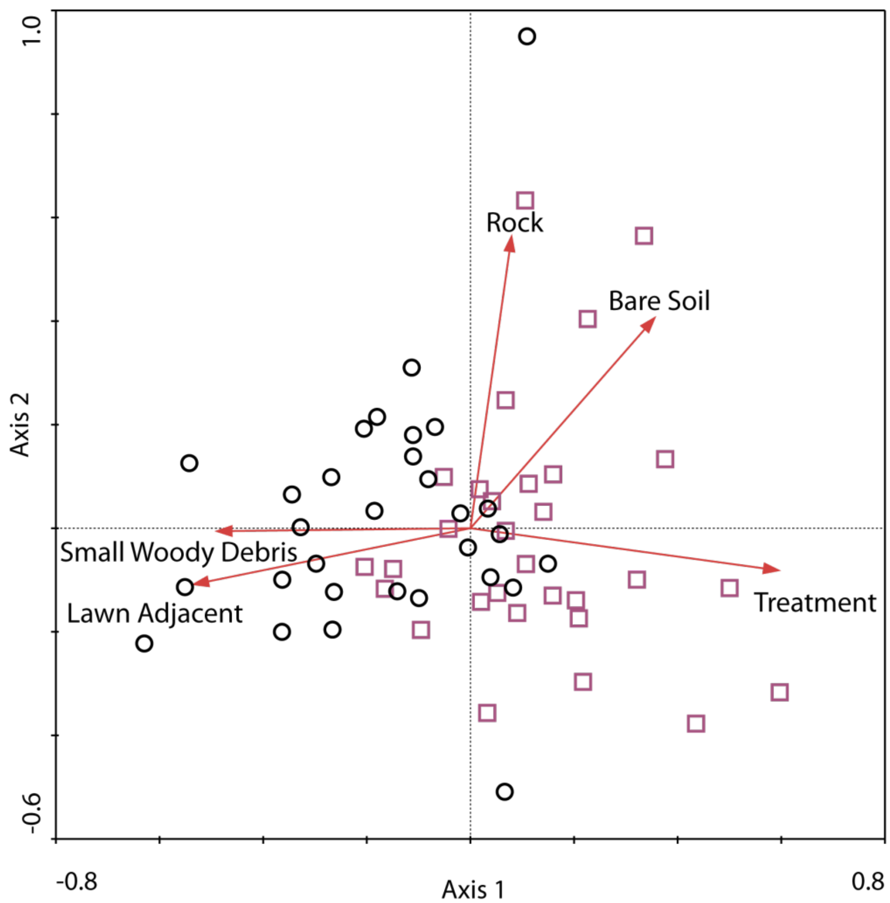

Figure 2.13: Site characteristics (→) associated with tree community composition in restored (☐) and unrestored (O) invaded sites.

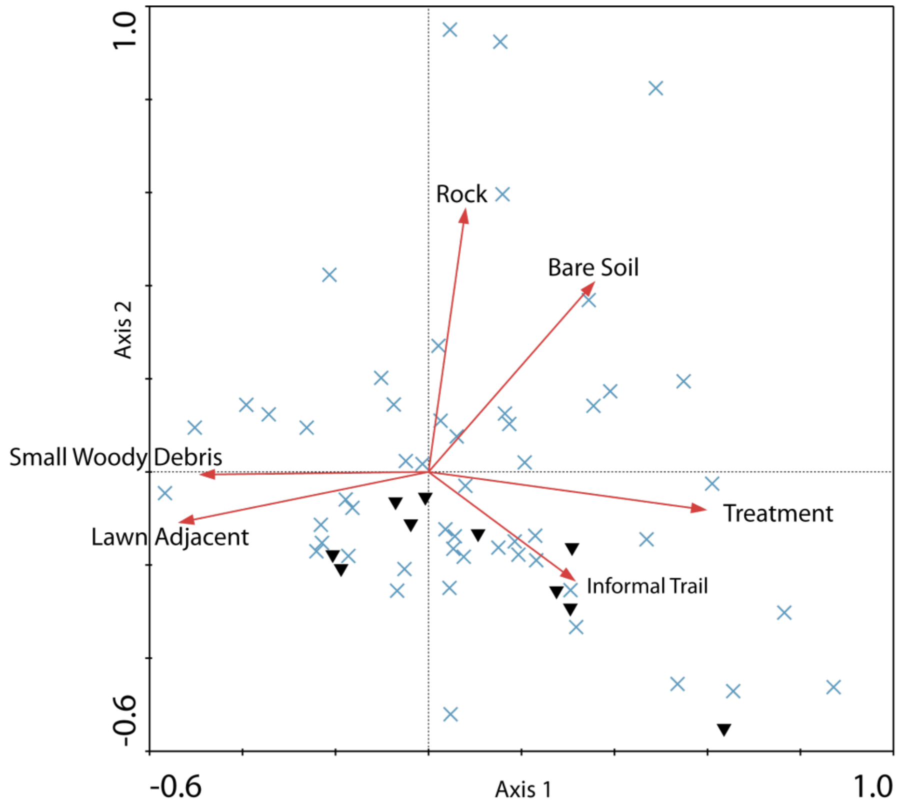

Figure 2.14: Tree species (✕) of restored sites arrayed in relation to site characteristics (→). Species planted on more than 400 occasions are shown in triangles (▼).

Planted tree species were strongly associated with restoration treatment, and negatively associated with bare soil and rock outcrops (Figure 2.14).

Tree species composition and management effort in restored sites

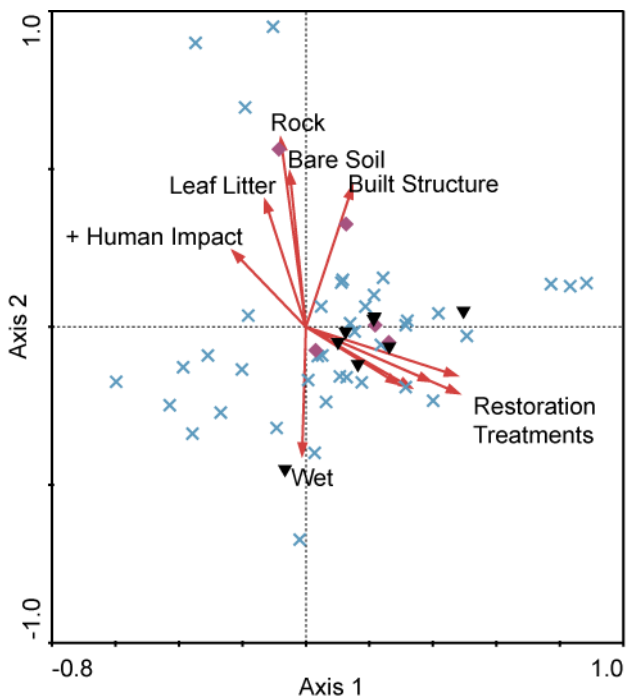

Figure 2.15: Tree species (✖) of restored sites arrayed in relation to site characteristics and restoration treatment. Species planted on more than 400 total days (▼) and targeted invasive species (◆) are shown. Targeted invasive species were removed on at least 100 occasions city-wide as part of the restoration effort, 1998-2009.

Among restored sites, total restoration effort, total removal, and total planting had important effects on tree community composition. Surface factors such as rock, bare soil, leaf litter cover, and soil moisture were also important to tree composition (Figure 2.15).

Presence of frequently planted tree species in 2010 was positively associated with more frequent restoration activity. Invasive species *Acer pseudoplatanus* and *Acer platanoides* were less abundant in sites that had greater restoration effort. These species were also associated with rockier sites, bare soil, and built structures (Figure 2.15). Frequently planted species *Acer rubrum*, *Quercus bicolor*, *Quercus palustris*, *Acer saccharum*, *Fraxinus pennsylvanica*, and *Liquidambar styraciflua* were associated with wet sites and with restoration effort.

DISCUSSION

There were significant differences in community composition among restored, unrestored and less-disturbed sites in all forest strata (Figures 2.3, 2.9 and 2.12). These differences indicate that restoration treatment had significant and persistent effects on vegetation after 15-20 years. Restored sites were more similar to a less disturbed forest site than were unrestored sites in their plant community composition. Differences between invaded site plant communities were most strongly associated with whether or not a site was restored, and with soil surface characteristics including erosion, small woody debris cover, and bare soil (Figures 2.5, 2.10 and 2.13). Among sites that were restored, soil factors were also important, and differences were strongly associated with the amount of restoration effort applied at the site over time, measured as the number of dates on which restoration treatments were applied (Figures 2.7, 2.8, 2.11, 2.14 and 2.15).

Differences in Composition of Forest Strata

Ground Layer Plant Composition

The forest floor is the environment in which seeds germinate and plants become established. High mortality is typical of these early phases of plant development, and changes in competitive interactions, light availability, soil properties, disturbance frequency and other factors can differentially affect germination, establishment and growth. In the restoration treatments examined in this study, removal of invasive vegetation altered ground layer conditions for planted native tree seedlings by reducing the abundance of invasive woody plants.

After 15-20 years, unrestored sites were associated with higher soil cover by small woody debris, which was also associated with presence of target invasive woody species *Ampelopsis brevipedunculata*, *Rosa multiflora*, and *Celastrus orbiculatus*, and negatively associated with restoration (Figures 2.4 and 2.5). Small woody debris in these sites consisted primarily of dead stems of these target woody invasive species, especially *Rosa multiflora*, which frequently formed dense thickets > 1 m in height. In heavily invaded sites, this type of woody debris covered 100% of the soil surface.

The effects of restoration on ground layer plant composition were positively associated with higher soil surface cover by leaf litter, while they were negatively associated with erosion, small woody debris cover, and bare soil (Figure 2.5). Among restored sites, restoration effort was negatively associated with small woody debris (invasive woody species litter) and bare soil (Figure 2.7). Associations between restoration and both higher leaf litter and lower cover by small woody debris reflects reduction in local abundance of invasive woody species. Bare soil was common underneath dense woody invasive species cover where woody debris was absent. This may reflect inhospitable germination and growth conditions for many species, and may also be a result of high rates of surface litter removal by invasive earthworms that were present in all parks (Kostel-Hughes, Young, and McDonnell 1998; Kostel-Hughes, Young, and Carreiro 1998). Introduced earthworms may affect restoration success due to influence on soil and hydrologic properties including litter depth, nutrient cycling rates, mixing depth of organic matter and soil porosity (Szlavecz et al. 2006; Baker et al. 2006; McDonnell et al. 1997; Pavao-Zuckerman 2008). The effects of these important soil organisms on vegetation dynamics and restoration outcomes deserve further attention (Bohlen et al. 2004).

Rosa multiflora and *Ampelopsis brevipedunculata* were also often associated with one another in the ground layer (Figures 2.6 and 2.8). Further investigation could shed light on the nature of this association; porcelain berry vines creates clearings by shading and weighting trees, and the rose may follow or increase in abundance in resulting clearings.

Unrestored sites were more similar to one another in their ground layer community composition than other types of sites. This decreased variability reflects homogeneity of plant communities dominated by woody invasive plants and the species that persist in their presence (McKinney 2006), and was a primary concern of the initial restoration effort. In contrast, differences among ground-layer plant communities in restored sites were associated with dominance by unique species. Greater variability in ground layer composition restored sites reflects change in the direction of a central goal of the restoration.

The amount of restoration effort employed in a location was important to differences in ground layer plant composition between restored sites. Effects of the number of restoration treatment days in the first six months and first year following initial planting were analyzed, but are not shown due to high correlation with total restoration treatment. Targeted invasive species *Rosa multiflora*, *Ampelopsis brevipedunculata,* and *Celastrus orbiculatus* in the ground layer were associated with less management after the initial restoration treatment, especially planting. This finding highlights the importance of ongoing management following initial restoration.

Woody Understory Composition

The factors most strongly influencing differences in community composition of the woody understory between restored and unrestored sites were also similar to those of the ground layer: whether or not a site was restored, and soil factors, particularly leaf litter cover. Effects of additional intensive human impacts such as camping and social gathering areas with their associated trails that resulted in bare soil and soil compaction were also associated with differences in woody understory composition. Among sites of each type, the degree of use of forested areas for a variety of unsanctioned uses that result in trampling and soil compaction was associated with species composition. This underscores the importance of planning for and managing human disturbance in ecological restoration in the urban environment.

Canopy and Sub-canopy Tree Composition

The current composition of the forest canopy integrates germination, establishment and growth conditions of the past. In this study, canopy trees pre-date restoration treatments, while sub-canopy trees in restored sites may include both spontaneous and surviving planted individuals. Mortality of planted trees was high in the early phases of the restoration due to herbivory (Wenskus, pers. comm.). Native trees planted as part of the restoration had a maximum of approximately 20 years of growth between initiation of restoration and this sampling, and planted seedlings were 1-2 years old at time of planting. Although some species may have reached the height of surrounding canopy (e.g. *Liriodendron tulipifera*, which was utilized specifically for its rapid growth and height), few surviving planted trees were likely to have reached sexual maturity in 20 years (Burns and Honkala 1990). Saplings of planted species large enough to be considered sub-canopy trees in this analysis are unlikely to be the result of reproduction by planted individuals, but rather the result of spontaneous recruitment from existing seed banks, dispersal of seeds into the restored area from the adjacent forest, and/or seedlings released by the removal of invasive plants. The influence of seed banks and adjacent sources of propagules are outside the scope of this study because they were not recorded when restoration was initiated, but information on these variables is valuable to understanding restoration outcomes and should be incorporated in early site analysis and restoration planning.

Current and past human disturbances such as abandoned built structures were associated with a smaller degree of the variation between tree communities. These reflect both the legacy effects of prior land ownership and use patterns; trees such as the flowering ornamental cherries and crabapples that were frequently planted near buildings before the creation of city parks, as were horticultural species including invasive species *Acer pseudoplatanus* and *Acer platanoides*. Both introduced maples were widely planted before their effects on native forests were widely known. These species were positively associated with rockier sites, bare soil, and built structures, and negatively associated with restoration effort.

In the 1970s and 1980s, fires due to the burning of stolen automobiles were a frequent occurrence (Wenskus pers. comm., Matsil and Feller 1996). The biological legacies of this disturbance are currently difficult to separate from those of other disturbances. The high incidence of *Sassafras albidum* in some sites may be related to its propensity to sprout vigorously following fire (USDA 2013). Existing records of fire location and intensity were not sufficient to incorporate this into the current analysis, but fire may be another contributing factor in restoration outcomes and successional trajectory direction. This and other unknown site history variables have been treated as part of the background noise of the heterogeneous urban environment. It is assumed here that some sites may have burned, and others disturbed by historic human land uses in other ways.

Implications for Restoration of Urban Woodlands

The results of this study emphasize three elements that need greater consideration in urban ecological restoration: heterogeneity, appropriate targets, and management effort. Models of forest restoration appropriate for less disturbed sites need to be modified to both predict and manage restoration of urban sites with a heterogeneous and intensive disturbance history.

Urban Soil Legacies

At a landscape scale, urban environments are characterized by a high degree of spatial heterogeneity (Effland and Pouyat 1997; Cadenasso, Pickett, and Schwartz 2007; Pickett and Cadenasso 2009). This heterogeneity is the result of both a tendency for cities to be located in sites of high biodiversity and heterogeneity (Kuhn, Brandl, and Klotz 2004), and by human cultural practices that add species over time both intentionally and unintentionally. The vegetation of cities like New York has been subject to a series of different phases of human influence, from forest clearing to agriculture to construction and sealed soils, resulting in heterogeneous patterns in the biophysical environment (Cadenasso, Pickett, and Schwartz 2007; Pickett and Cadenasso 2009; Cadenasso, Pickett, and Grove 2006; Alberti 2005; Pickett et al. 2011). Site-specific legacies of these landscape-level patterns may override restoration treatments where treatments are not tailored to particular site conditions (Pavao-Zuckerman 2008). The significant the effects of treatment demonstrated here despite this heterogeneity demonstrate the capacity of restoration treatment to change trajectories of vegetation development over time.

Targets and Goals in Urban Forest Restoration

Urban environments are subject to frequent human disturbance, so much so that a snapshot of any city today would contain only a handful of buildings that will still stand in two centuries. For many species of trees, that time period represents a life span or less. Forests in cities occur as remnant fragments surrounded by a variety of land uses or in places where land abandonment has permitted regeneration. Some of these are young forest patches on recently disturbed land, while in other locations where land was set aside early in urban development, forested patches may contain trees older than those found in surrounding areas that were deforested as the urban area expanded (Loeb 2011).

The question of appropriate targets in restoration is not a new one (e.g. Hobbs and Norton 1996; White and Walker 1997), and the static, pristine reference site has widely been rejected as a benchmark for restoration activities (Millar and Brubaker 2006). In cities, where disturbance regimes are altered, exotic species are frequently introduced, and remnant habitat fragments are small and surrounded by a matrix of paved and built land (Pickett et al. 2011; Kowarik 2008) the question of appropriate targets is an important one if land managers are to choose attainable goals.

Human disturbances can changes ecosystems rapidly, while recovery is slow in comparison (McLaughlin 2013); it is unrealistic to expect a habitat fragment to acquire in a matter of months or a few years properties that accumulate over centuries of complex interactions in natural systems. In the urban matrix, soils are disturbed and organic soil layers are often removed or sealed with concrete or asphalt. Soils not sealed, turned, moved or compacted are altered by atmospheric and other deposition of nutrients and pollutants (Pouyat et al. 2010). While habitat transformation is the primary cause of urban plant species loss, changes to soils alter plant communities even in remnant forests protected from development (Robinson, Yurlina, and Handel 1994; DeCandido, Muir, and Gargiullo 2004; Drayton and Primack 1996). Urban habitat fragments also often contain invasive species (Klotz and Kühn 2010; Sukopp 2004). Small remnant habitats can be overwhelmed by prolific growth, and edge effects can contribute to invasion (Cadenasso and Pickett 2001). Invasive plants are rarely completely eradicated once established (Pyšek and Richardson 2010), though much management focuses on eradication as a goal . Even less disturbed sites may be invaded, including the old-growth urban forest described here (NYBG 2001).

These findings indicate that ecological restoration targets for urban habitats must include urban variables. The boundaries of what is possible in a site may have changed irrevocably. Ecological restoration in such cases, then, must also flex its boundaries. The less-disturbed reference site remains an important baseline from which to understand historic potential and as a palette from which to draw species, but its utility as a target state is limited in cities where direct and indirect impacts of the urban environment preempt prior trajectories of vegetation and soil development. A shift in focus is needed from the static endpoint of a pristine climax to a multifaceted projection of what is possible given current conditions. Strategies based on a process-centered approach to directing succession in ecological restoration will be more successful.

Management Matters

While urbanization is one of the most long-lasting and intensive types of landscape transformation (Williams et al. 2009) , conflict between ecological function and economic pressure is hardly unique to urban environments. On government, protected, and private lands, compromises are made on a daily basis between social and ecological values. Land managers have limited funds, ownership and jurisdictional boundaries overlay ecological features, and legacies of past land use shape current conditions.

People in cities depend not only on the ecosystems that provide food, water and other services at a distance, but on the urban environment itself. Forested patches in urban landscapes can provide air quality and local cooling, as well as social and human health benefits (Gaston, Davies, and Edmonson 2010; Barton and Pretty 2010; TEEB 2011). Economic, social and biophysical attributes of an urban system affect the degree to which urban environments can provide essential ecosystem services. The findings of this study support the conclusion that that investment of time, materials and labor affect the long-term outcomes of urban ecological restoration. Level of management effort has influenced long-term outcomes in this urban ecological restoration effort, and the degree to which sites resembled desired conditions was associated with this effort. The effort expended in the most frequently managed restored forests observed here, however, is likely less than the equipment, time, and other inputs invested in comparable-sized manicured park areas.

These findings also point to another place in which ecological restoration must flex its boundaries to encompass urban environments. Where human disturbance is extensive, diverse, pervasive and frequent and habitats are fragmented into small patches, management requirements are likely to be higher than in less altered systems. The most straightforward and simple form of ecological restoration removes a single disturbance (for example a pipe draining chemical waste into a pond) and allows time for ecosystem processes to restore function (SERI 2004). Urban areas offer no simple solutions. Ongoing management will be required where urban impacts exceed the pace of ecosystem recovery. The challenge of restoring resilient and functional ecosystems to urban environments requires thinking about species and nature in ways that stray far from the model of the pristine.

A Possibility-Focused Approach to Urban Ecological Restoration

Ecological restoration seeks to alter trajectories of community development over time, with the goal of self-sustaining, functional ecosystems that resemble pre-disturbance or reference conditions (SERI 2004). Relatively pristine reference sites and historical records are important to understanding the historic potential of a site’s biophysical conditions. Functioning ecosystems resembling those formerly found where cities now stand contain species, interactions and processes that evolved over time in response to the local climate, topography, and soils. However, where baseline conditions have changed, reference states are no longer likely or possible outcomes. Adjustments to the framework are necessary (Palmer et al. 2004). New reference states are needed.

Degraded sites should not be viewed as a temporary problem to be surmounted. Rather, they are the starting point from which complexity, biodiversity and ecosystem function may be increased. They will change over time, but they may not follow trajectories that increase desired ecosystem properties and functions. What can be done to increase ecological function and environmental benefits in urban landscapes? I suggest a possibility-based assessment and phased approach for ecological restoration in urban environments.

These findings suggest that the effects of urbanization on three elements of a site should be paid more attention in planning urban ecological restoration: soil characteristics, propagules, and disturbance. These are understood to be important elements of vegetation dynamics (Pickett, Meiners, and Cadenasso 2011; Figure 2.1) and of ecological restoration (Falk, Palmer, and Zedler 2006), and should be incorporated into restoration planning. Urban ecological restoration based on models that do not include urbanization’s effects on these factors may fail to meet its goals.

Urban Soils

Geological, climatic, and biotic changes in soil over time are combined with patterns of human land use in urban environments. Cities are spatially heterogeneous, and legacies of past land use may not be immediately apparent. In this study, soil surface characteristics were shown to be important to plant community composition; legacies of compaction, erosion and species invasion affected the growth of plants in all forest strata. Invasive earthworms were present in all sites, and some sites had been used as dumps for household and other refuse. Site-specific knowledge, historical documents, maps and records should be consulted to map and plan for urban ecological interventions. Toxic, nutrient-poor, nutrient-loaded or other problematic soil conditions that can affect restoration outcomes may need to be mitigated to bring soil conditions into ranges that desired species can tolerate. Thorough understanding of current soil conditions will greatly increase the likelihood of successful plantings by matching species with conditions in which they can thrive. In the following chapter, I will further examine the role of soils in the urban environment on the outcomes of this restoration.

Urban Propagules

The focus of land management activities is usually restricted to parcels owned or operated by a single organization or agency. These boundaries may not encompass entire habitat types, limiting management effectiveness. In urban environments, where habitat patches are small and fragmented (Medley, McDonnell, and Pickett 1995), and land ownership is likewise heavily dissected, these effects are compounded. In the parks studied here, forests abut residential neighborhoods, golf courses, eight-lane roads, and industrial areas. A variety of social uses, from baseball to off-trail cycling, create further fragmentation and intrusions. In urban forest fragments, edge effects have important influences on habitat conditions. Among these are propagule introductions, especially of bird- and wind-dispersed plant species (Carreiro et al. 2009). Several new invasive species had become abundant in the 20 years following the initiation of restoration in New York City parks, including garlic mustard (*Alliaria petiolata*) and goutweed (*Aegopodium podagraria*). Species are constantly introduced to urban areas (Kowarik 2008), and managers of urban forests should expect the arrival of novel plants, animals and pests in the future. The need for long-term management should be anticipated and incorporated in planning.

Urban Disturbance Regimes

An understanding of past, current and likely future disturbance types and frequencies may increase the success of restoration efforts. By anticipating patterns resulting from prior disturbance (like fire effects on soils and plant composition) and from changes in disturbance type and frequency (like altered hydrology in urban stream channels), more effective responses can be developed. If future disturbance is likely to exceed the regenerative capacity of recovering habitats, management may need to be increased or preventative measures may need to be engaged to decrease disturbance.

Conclusion

The impetus to protect, expand, connect and restore biodiversity extends from the speed, extent and severity of human-driven transformation of Earth's surface. Urbanization presents a particularly harsh environment for many native species, while creating habitats for synanthropic ones (Shochat et al. 2006). It is probably impossible to maintain or re-create complete historic assemblages of plants and animals in small fragments of habitat surrounded by an urban matrix (Carreiro et al. 2009; Harper et al. 2005; Soulé 1991). However altered, these islands of biodiversity and ecological function can and do offer essential environmental benefits that cannot be outsourced (Kowarik 2011; TEEB 2011; Barton and Pretty 2010; Gaston, Davies, and Edmonson 2010).

The future of forested patches in urban environments is linked to human actions (Carreiro and Zipperer 2011). The findings of this study indicate that ecological restoration efforts in cities can be improved with greater attention to urban soils, urban propagule dynamics, and urban disturbance regimes. These results also show that ongoing restoration effort makes a difference in long-term outcomes. Ecological restoration in urban environments will benefit from incorporating contemporary succession theory (e.g. Pickett, Meiners, and Cadenasso 2011; Figure 2.1). Ecological restoration in the urban environment must respond to urban environmental variables that change site soil, climate, and hydrologic conditions, introduce a global species pool, and alter disturbance regimes. To improve the effectiveness of urban ecological restoration, a long-term view is needed, both toward site history and toward the future change. Urban restoration ecologists must cautiously look both ways.

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CHAPTER 3

Soil effects on long-term outcomes of ecological restoration in urban forest patches

ABSTRACT

Municipalities are increasingly turning to ecological restoration of urban habitats to address urban environmental problems. Ecological processes in urban habitat patches are dependent upon soil functions, and in urban environments soils are subjected to numerous transformations, from topsoil removal and waste disposal to warmer temperatures and non-native species introductions. Despite these alterations, urban soils can nonetheless be seen as the “brown infrastructure” that provides numerous ecological services.

To test the hypothesis that impacts of urbanization on soils affect long-term outcomes of ecological restoration, I compared physical and chemical characteristics of soils in restored, unrestored and less-disturbed forest patches with urban soil classification mapping and with plant community composition and structure. Urban landscapes are characterized by highly heterogeneous land use. I expected legacies of historic uses and management to affect restoration outcomes, and I predicted that common urban environmental effects on soil, especially soil compaction, pollution, and presence of soils derived from human-deposited materials, would affect restoration outcomes. I sampled 30 sites in restored New York City Park forests in 2009, 10 in each of 3 parks that been treated by removal of woody invasive species and planted with native tree seedlings in the 1990s. In 2010, I sampled an additional 30 plots in New York City Park forests that

were in a similarly degraded condition at the time of the original restoration vegetation survey (1988-1992), but which had not been restored, and 10 plots in a less-disturbed remnant of original forest vegetation.

All sites were impacted by anthropogenic factors known to reduce plant growth, change distributions of soil biota, and alter nutrient cycles. No one urban soil impact dominated effects on plant community composition. Instead, sites were similar in their degree of metal contamination and cover of exotic earthworm castings, while other impacts varied among sites, including which metals were found in plot soils. The success of urban restoration efforts can be increased by improving the fit between current soil conditions and survival of desired plants. This will require finer-scale mapping of urban soils and greater emphasis on site-specific evaluation of soil conditions in restoration planning. Current vegetation may provide information about past history and legacies of toxicity, informing assessment of restoration potential. Where soils have been transformed by effects of the urban environment, choice of species for ecological restoration should reflect current conditions.

KEYWORDS

Urban soils, New York City, urban ecology, plant community ecology, ecological restoration

INTRODUCTION

"The land is the finest for cultivation that I ever in my life set foot upon, and it also abounds in trees of every description."

Henry Hudson, 1609 (NYCSWCD 2005)

"Soil is destiny." Michael Feller, Chief Naturalist,
New York City Department of Parks and Recreation (2012)

Soil is the structural base, nutrient source, and living ecosystem that supports terrestrial life. This thin layer of the Earth's surface influences the distributions of communities of plants and animals, including humans. It is central to ecosystem processes and functions, from primary production to decomposition (Brady and Weil 2007).

In urban environments, soils are subjected to numerous transformations. Beginning with early settlement, changing land use patterns and practices affect soils in profound ways. Clearing of original vegetation, agricultural tillage, soil movement, road building, and residential and industrial development all leave their mark. Physical and chemical alterations resulting from social and cultural factors result in urban soils that are more spatially and temporally variable than non-urban soils (Cadenasso, Pickett, and Schwartz 2007).

Direct urban soil alterations include cutting and filling to provide level surfaces for building, sealing with impermeable surfaces, moving topsoil, and adding fertilizers and other plant growth media, and filling undesirable areas such as wetlands, depressions, and excavated areas with materials for disposal (NYCSWCD 2005; Craul 1999). Many urban soils have little organic layer and little vegetation. Added materials like garbage, bricks, coal ash, and dredge spoil can alter hydrologic and chemical properties of urban soils and take up physical space that would otherwise be available for plant roots, water, and nutrients. Concrete, road salts and plaster contribute to higher pH (NYCSWCD 2005).

Urban effects on soils may also be indirect, such as soil hydrophobicity, introduced plant and animal species, warming by urban heat island effects, and atmospheric deposition of pollutants (Pouyat et al. 2010; Effland and Pouyat 1997). Urban soils are typically warmer than non-urban soils, and have a high probability of being contaminated (Ziska, Bunce, and Goins 2004; Pavao-Zuckerman 2008). The availability of resources for growth of all types of organisms are altered in urban soils by changes in water availability, increased nitrogen inputs from atmospheric deposition, high levels of base cations from construction, demolition, dumping and road salting, and deposition of heavy metals from a variety of sources. Shifts in human cultural activity resulting from innovation and changing fashions are also part of the disturbance regime of urban environments, resulting in shifts in patterns of landscape alteration (Pickett and Cadenasso 2009). These combined changes in soil composition, physical structure, chemical makeup, hydrology, and resource availability can have major effects on plant growth (Bradshaw 2002). Specific effects on biological processes and fauna of soil are still largely unknown, but are likely to depend on interacting urban environmental factors (Pouyat et al. 2010).

Despite all of these alterations, urban soils can nonetheless be seen as “brown infrastructure” that supplies plant nutrients, serves as a habitat for plants and animals, stores carbon and mineral nutrients, functions as part of hydrological cycles by absorbing, storing and regulating water supply, and reduces the bioavailability of pollutants generated by human activity (Pouyat et al. 2010; Effland and Pouyat 1997). Urban soils are often viewed as sterile, monotonous “moon landscapes” (Craul 1999) but at a landscape scale, urbanization creates complex patterns of chemical, physical and biological conditions, leaving some largely undisturbed or highly fertile soils in an urban soil mosaic (Pouyat et al. 2010). In highly urbanized areas, original soils are often limited to patches within the urban matrix that were unsuitable for urban development, such as wet and frequently flooded areas, rocky ground, and steep or unstable slopes (Effland and Pouyat 1997). These areas have, in some cases, been preserved in parks. However, even physically unaltered soils can be affected by urbanization (Pouyat et al. 2008), and urban parks contain areas with a variety of histories, from relatively intact forest fragments to metal-processing dump sites regrown with spontaneous vegetation.

Municipalities are turning to ecological restoration of habitat fragments as a way to address multiple environmental problems faced by cities. Natural areas in urban parks are disproportionately important to both local biodiversity and the human communities that rely on them for social, aesthetic, health and other environmental benefits (Sadler et al. 2010). These habitat fragments may be in areas of relatively intact original soils, or in areas of human-deposited soil materials. Restoration efforts often focus on reestablishment of native plant communities in areas either devoid of plant cover or invaded by exotic species. In New York City, where the majority of the land area has been transformed by human activity and average temperatures are typically 2–3°C warmer than adjacent rural lands (McDonnell et al. 1997), ecological restoration of urban forests has been adopted as a measure to improve the ability of urban environments to provide basic ecosystem services that would otherwise be impossible or extremely costly to produce (City of New York 2007; Bolund and Hunhammar 1999).

Soils of remnant forests in New York City have been found to contain elevated levels of lead, copper, and nickel (Pouyat and McDonnell 1991); to be more hydrophobic than rural soils (McDonnell et al. 1997); and to have decreased depth, mass and density of leaf litter (Kostel-Hughes, Young, and Carreiro 1998) compared to forests at the rural end of an urban-rural gradient. These forests have also been found to have large populations of introduced earthworms, which are capable of changing important soil properties including litter decomposition dynamics, soil porosity, and depth of organic matter (Steinberg et al. 1997; McDonnell et al. 1997; Szlavecz et al. 2006).

The New York City Department of Parks and Recreation’s Natural Resources Group (NRG) has been engaged in ecological restoration of urban forest fragments since its formation in 1986 (NRG 1991; Sisinni and O’Hea Anderson 1993). Early NRG forest restoration projects focused on large areas invaded by a suite of woody exotic species: porcelain berry (*Ampelopsis brevipedunculata*), oriental bittersweet (*Celastrus orbiculatus*), and multiflora rose (*Rosa multiflora*). To evaluate the long-term effects of restoration on plant community composition and structure, I revisited woodland restoration sites 15-20 years after these species were removed and native trees were planted (Chapter 1).

Significant differences in vegetation composition and structure between restored and unrestored sites indicated that invasive species removal followed by planting resulted in persistent structural and compositional shifts. Restored sites had greatly lower invasive species abundance, an increase in vertical structural complexity, and greater native tree recruitment. Successional trajectories of vegetation development of restored and unrestored forests had diverged after 15-20 years (Chapter 1). Variability in outcomes among restored sites was associated with both amount of restoration management effort and with soil surface characteristics including proportional cover by leaf litter and woody debris (Chapter 2). To further explore relationships between soil characteristics and urban land use and land cover that affect outcomes of ecological restoration, in this study I examine the role of soil factors in plant community outcomes.

Hypotheses

In a previous study, variation in restoration outcomes was seen among urban forest patches that were restored (Chapter 1). Subsequent analysis showed that a large part of this variation was explained by amount of restoration effort, measured in days that restoration treatments were applied, and that soil surface cover was also associated with differences in trajectories of plant community change (Chapter 2). To test the hypothesis that impacts of urban site history on soils affect the long-term outcomes of ecological restoration, I compared physical and chemical soil characteristics and soil classification maps for New York City with plant community composition and structure of restored forests, unrestored forests, and less-disturbed remnant forests.

Urban environments have both direct and indirect effects on forest patches in urban environments, altering physical and chemical characteristics of urban soils (Carreiro and Tripler 2005; Steinberg et al. 1997; McDonnell et al. 1997; Pouyat and McDonnell 1991; Pouyat et al. 2010; Groffman et al. 2006; Pouyat et al. 2008). Direct impacts like cutting, filling, dumping and building may remove, bury or mix soil layers, create areas of high pollutant concentrations, add materials, and compact soil pore space, and in urban parks, compaction and erosion are increased by recreational use (Craul 1999; Pouyat et al. 2010). Urban forest fragments are impacted by the indirect effects of atmospheric pollution, urban heat island effects, changed ground wind speeds, altered disturbance regimes, changed water availability, and the introduction of exotic species (Walsh et al. 2005; Ziska, Bunce, and Goins 2004; Effland and Pouyat 1997; Alcoforado and Andrade 2008). Their small patch size exposes them to greater influence of external inputs of energy, matter and species from the highly contrasting paved, built and landscaped matrix surrounding them (Carreiro et al. 2009).

These impacts affect not only physical and chemical properties of soils but also germination and growth of plants, altering both the pool of available species and their competitive environment, with potential to reduce the success of ecological restoration in urban environments by creating conditions outside the ranges of tolerance of desired native species (Pavao-Zuckerman 2008; Bradshaw 2002; Pickett and Cadenasso 2009; McKinney 2008). I expected these effects of the urban environment to affect patterns of plant community composition in restored patches of urban forest. I examined several soil parameters that can be altered by the urban environment: parent material, soil compaction, surface litter, and chemical parameters including acidity, total organic carbon content, and trace metals.

While urban parks may contain areas with intact soil profiles, other areas have been subjected to a variety of alterations (Pouyat et al. 2010). Sites I sampled were located in soils derived from both natural parent materials and human-deposited materials. Fill soils, which differ from native soils in both profile and content, often contain calcium-rich construction debris and are low in organic matter (NYCSWCD 2009). I expected effects of soil disturbance and composition to result in differing plant community outcomes in restored forests located in these more recently-disturbed sites.

Soil compaction reduces soil pore space, reducing aeration, water availability, and infiltration, physically inhibiting root growth at high bulk densities (Brady and Weil 2007). Soil compaction is a common feature of highly used urban parks due to trampling, bicycling and other recreational uses (Jim 1998; Toth and Sauer 1994; Kissling et al. 2009). I expected degree of soil compaction to affect plant community composition among restored sites.

Compared to rural forests, urban forest patches in New York City have been found to have elevated metal concentrations (Pouyat and McDonnell 1991); to be more hydrophobic (McDonnell et al. 1997); and to have decreased leaf litter depth, mass and density (Kostel-Hughes, Young, and Carreiro 1998). These forest patches also have abundant introduced earthworms, which can change litter decomposition dynamics, soil porosity, and organic matter mixing in the soil profile (Steinberg et al. 1997; McDonnell et al. 1997; Szlavecz et al. 2006). I expected soil litter quantity and heavy metal concentrations to be associated with differences in plant communities among restored sites.

METHODS

Site Locations

I sampled 30 sites in restored New York City Park forests in summer 2009, 10 in each of 3 parks (Chapter 1 and Figure 3.1). All of these sites had been treated by removal of woody invasive species (*A. brevipedunculata*, *R. multiflora*, *C. orbiculatus* and *A. platanoides*) and planted with native tree seedlings in the 1990s. In 2010, I sampled an additional 30 plots (10 per park in 3 parks) in New York City park forests that were in a similarly degraded condition at the time of the original restoration vegetation survey (1988-1992), but which had not been restored.

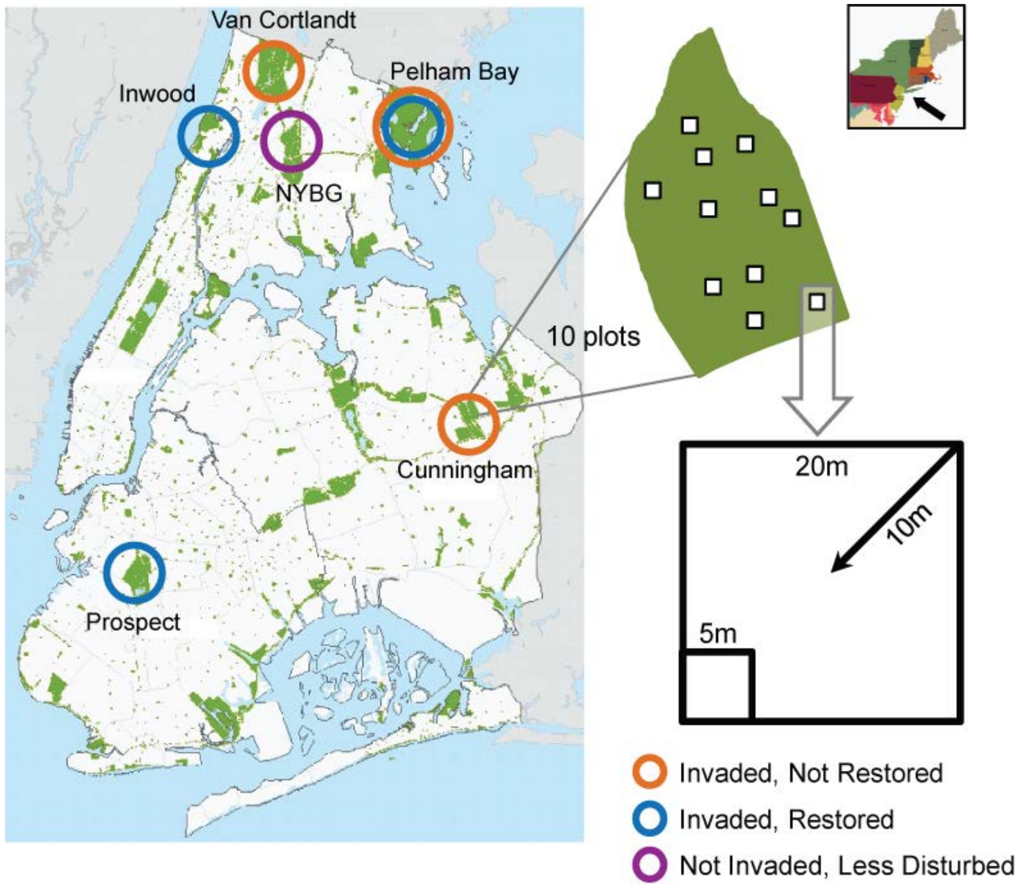

Figure 3.1: Study plot park locations. Restored: Inwood Park, Manhattan; Pelham Bay Park, Bronx; Prospect Park, Brooklyn. Not restored: Van Cortlandt Park, Bronx; Pelham Bay Park, Bronx; Cunningham Park, Queens. Not invaded in 1988, less disturbed: New York Botanical Garden, Bronx. Map: Craig Mandel, NRG.

Vegetation Sampling

In each 20 m x 20 m plot, I measured the diameter at breast height (DBH) of all tree stems (Figure 3.1). I counted all stems of tree saplings, woody vines and shrubs in three 5 m x 5 m subplots randomly located within each plot. I recorded intercept (cm) of all ground-layer vegetation along four 10 m line transects that extended from each corner of the plot toward the center (Chapter 1). To compare invaded sites (restored and unrestored) with a less-disturbed urban forest type, in 2010 I sampled 10 sites using the same methods at the New York Botanical Garden (NYBG), in less disturbed old-growth forest sites where invasive species were not dominant in the 1990s (Chapter 2).

Soil Characterization

Within each plot, surface litter was collected from 4 randomly selected 15 cm x 15 cm areas, one in each quadrant of the plot. Samples of the top 15 cm of soil from the 4 litter collection sites in each plot were bulked and mixed. Soil penetrometer readings were taken adjacent to each of the four subsample sites within each plots with a Dickey-John soil compaction tester with a ¾" cone in fall of 2011 prior to leaf drop, under moist conditions (12 readings per plot).

Both litter and bulked A-horizon samples were air dried in paper bags at room temperature. Total organic carbon was measured by loss on ignition, and pH was measured using a 1:1 mix of soil and de-ionized water using a pH meter accurate to 0.01 at the Brooklyn College soil laboratory in Brooklyn, NY. Trace metals were determined on air-dried samples with portable X-ray fluorescence (PXRF) spectrometry, using an Innov-X Delta Standard Model. The PXRF was standardized with a stainless steel '316' alloy clip, and soil samples analyzed at a 90 second scan time at the USDA Natural Resource Conservation Service laboratory in Somerset, NJ.

Data Analysis

I created a relational database (MS Access 2007) by combining NRG databases of pre-restoration vegetation descriptions (entitation mapping, Chapter 1), NRG restoration activity logs, Prospect Park Natural Resources restoration activity, Prospect Park Natural Resources long-term plot monitoring data, and field-collected data.

GIS plot locations were compared with New York City Soil Survey maps (NYCSWCD 2013). These maps provide 1:12,000 scale mapping of soil series phases and complexes using the USDA soil classification system. The minimum area for classification was 1.43 acres.

To determine whether soil classes were associated with different restoration outcomes, I compared soil classes and parent materials with plant community composition and soil surface physical and chemical parameters. To examine the role of soil surface characteristics in variability in restoration outcomes, I compared soil surface physical and chemical parameters with plant community composition in restored sites.

Statistical Analysis

Data describing the vegetation of each forest stratum by species and soil variables were analyzed using Canoco 4.53 (Ter Braak and Smilauer 2002a). Species and soil data were subjected to direct gradient analysis using environmental data to extract patterns from only the explained variation using Canonical Correspondence Analysis (CCA). Scaling was focused on inter-species differences, using biplot scaling with untransformed data. No sites, species, or soil variables were weighted or made supplementary. Monte-Carlo permutation tests were used to evaluate both the significance of the first ordination axis and the significance of the canonical axes together, with 499 permutations under a reduced model. Permutations were unrestricted, that is, not restricted for spatial or temporal structure or for split-plot design.

The CCA analysis described above produces ordination diagrams in CanoDraw (Ter Braak and Smilauer 2002a). The first two canonical axes are shown in figures. Scale is relative, axes are composed of combined environmental variables, and distance between plot points approximates similarity of the plots' plant communities in composition and abundance of species (Ter Braak and Prentice 2004). Distance between plot points in these diagrams approximates the dissimilarity of their species composition, measured by their Chi-squared distance. Site characteristic values can be approximated by projecting the location of a plot point onto the axis of an environmental variable's vector arrow; plot points are ordered by predicted increase of values for a particular environmental variable, in the direction of that factor's vector (Ter Braak and Smilauer 2002b). In diagrams displaying both environmental variables and plots, plot points are arrayed in relationship to arrow vectors representing site characteristics. Arrows point in the expected direction of steepest increase of values of that variable. The length of each arrow indicates the proportion of the variability associated with that factor (value of eigenvector), and angles between arrows indicate correlation of environmental variables. The degree of correlation can be approximated by projecting the head of each arrow onto the axis of another variable's arrow. Longest vectors, indicating the environmental variables explaining the greatest proportion of the variation between plot plant communities, are shown in figures.

Differences in plant community diversity, evenness and richness scores from the CCA analysis between soil parent materials were compared using nonparametric Wilcoxon (Mann-Whitney) signed ranks and Kruskal-Wallis tests and Tukey-Cramer HSD means comparisons (JMP 10.0.0, SAS Institute 2012). Outlier box plots representing data analyzed in this manner display the median (horizontal line in the box, the 50^{th} percentile), 25^{th} and 75^{th} percentiles (lower and upper box ends), and whiskers extend to the furthest data point within ±1.5x the interquartile range. Points outside the box and whiskers are considered outliers (SAS Institute 2012, Cary, NC).

RESULTS

Soil Mapping Units

Plots were located in 27 different New York City Soil Survey mapping units, 10 of which were derived from human-deposited materials. (NRCS 2013, Table 3.1). Soils derived from different parent materials did not differ significantly in the surface soil parameters sampled (n= 30, DF = 4, Wilcoxon/Kruskal Wallis Test with Tukey-Kramer HSD means comparison).

Table 3.1: New York City Soil Survey map units in which plots were found, with parent material and parks in which plots were found in each soil type. Map units are phases and complexes of soil series (NRCS 2013).

Map Unit	Soil Name	Parent Material	Sites
CCD	Chatfield-Charlton complex, 15 to 35 percent slopes, very rocky	Shallow till - high rock outcrop	Inwood, NYBG
CCHC	Charlton -Chatfield-Hollis complex, 0 to 15 percent slopes, very rocky	Shallow till - high rock outcrop	Inwood, NYBG, Pelham Bay
CCHRC	Chatfield-Charlton-Hollis-Rock outcrop complex, 0 to 15 percent slopes	Shallow till - high rock outcrop	NYBG
CGHRC	Chatfield-Greenbelt-Hollis-Rock outcrop complex, 0 to 15 percent slopes	High rock outcrop with fill	Pelham Bay
ChBs	Charlton fine sandy loam, sandy substratum, 3 to 8 percent slopes	Deep Till	Cunningham
ChBss	Charlton fine sandy loam, sandy substratum, 3 to 8 percent slopes, very stony	Deep Till	Prospect
ChD	Charlton loam, 15 to 25 percent slopes	Deep Till	Van Cortlandt
ChDs	Charlton fine sandy loam, sandy substratum, 15 to 25 percent slopes	Deep Till	Cunningham
ChDss	Charlton fine sandy loam, sandy substratum, 15 to 25 percent slopes, very stony	Deep Till	Prospect
CHRD	Chatfield-Hollis-Rock outcrop complex, 15 to 35 percent slopes	Shallow till - high rock outcrop	Van Cortlandt
CHRE	Chatfield-Hollis-Rock outcrop complex, 35 to 60 percent slopes	Shallow till - high rock outcrop	Inwood, NYBG

CwC	Charlton fine sandy loam, 8 to 15 percent slopes, eolian material	Deep Till	Inwood, Prospect
FFA	Fluventic Hapludolls-Fluvaquentic Endoaquolls complex, 0 to 3 percent slopes, frequently flooded	Alluvium	Van Cortlandt
FGB	Flatbush-Greenbelt complex, 0 to 3 percent slopes	Fill	Prospect
GbA	Greenbelt sandy loam, 0 to 3 percent slopes	Fill	Cunningham
GbB	Greenbelt sandy loam, 3 to 8 percent slopes	Fill	Cunningham, Van Cortlandt
GbD	Greenbelt sandy loam, 15 to 25 percent slopes	Fill	Van Cortlandt
MuA	Mosholu silt loam, 0 to 3 percent slopes	Fill	Van Cortlandt
NoA	North Meadow sandy loam, 0 to 3 percent slopes	Fill	Cunningham
PPA	Paxton complex, 0 to 3 percent slopes, truncated	Deep Till	Pelham Bay
PxBs	Paxton loam, 3 to 8 percent slopes, very stony	Deep Till	Pelham Bay
PxCs	Paxton loam, 8 to 15 percent slopes, very stony	Deep Till	Pelham Bay
RHCF	Rock outcrop-Hollis-Chatfield complex, 60 to 80 percent slopes	Shallow till - high rock outcrop	Inwood
ScB	Scio silt loam, till substratum, 3 to 8 percent slopes	Silty deposits over till	Pelham Bay
SiA	Siwanoy silt loam, 0 to 3 percent slopes	Deep Till	Pelham Bay
UGB	Urban land-Greenbelt complex, 3 to 8 percent slopes	70-90% Impervious	Pelham Bay
VzE	Verrazano sandy loam, 25 to 35 percent slopes	Fill	Prospect
WdB	Woodbridge loam, 3 to 8 percent slopes	Deep Till	Pelham Bay
WdBs	Woodbridge loam, 3 to 8 percent slopes, very stony	Deep Till	Pelham Bay

Soil Surface Characteristics

Soil Map Units

Table 3.2: Average, minimum and maximum values of soil surface characteristics of all plots by NRCS soil map unit: litter depth and weight; depth to root-growth inhibiting level of soil hardness; slope; pH, and total organic carbon (TOC), 2010. For soil types in which only one plot was located, values for one plot are shown. Soil litter data were not available for NYBG plots. *Soil type CCD was found in 3 NYBG plots and one NYC Parks plot; litter data for this type is from NYC Parks only. **Map unit CCHRC was found only in NYBG plots.

Soil Map Unit	Plots		Litter Weight	Litter Depth	Depth >300psi	Slope %	pH	TOC
CCD	4	**mean***	**23**	**3**	**> 45**	**22**	**5.3**	**9.4**
			-	-	-	8	4.4	5.5
			-	-	-	45	7.5	12.0
CCHC	7	**mean**	**16**	**2**	**30**	**8**	**5.6**	**9.9**
		min	14	2	6	0	4.3	5.6
		max	19	3	> 45	20	7.5	15.0
CCHRC	4	**mean****	-	-	-	**16**	**4.5**	**11.3**
		min	-	-	-	5	4.1	8.0
		max	-	-	-	37	5.1	16.0
CGHRC	1	n=1	**22**	**2**	**> 45**	**7**	**4.5**	**15.7**
ChBs	1	n=1	**12**	**2**	**15**	**0**	**5.5**	**12.7**
ChBss	1	n=1	**13**	**1**	**9**	**44**	**6.3**	**12.4**
ChD	5	**mean**	**17**	**2**	**18**	**17**	**5.1**	**7.0**
		min	13	2	6	0	4.8	4.5
		max	25	3	> 45	33	5.4	9.5
ChDs	2	**mean**	**15**	**2**	**31**	**5**	**5.0**	**10.0**
		min	8	2	12	0	4.3	8.9
		max	21	2	> 45	10	5.6	11.2
ChDss	5	**mean**	**30**	**3**	**18**	**22**	**5.8**	**6.9**
		min	17	2	9	10	5.1	6.0
		max	47	5	> 45	37	7.6	8.3
CHRD	1	n=1	**11**	**2**	**7**	**9**	**5.0**	**7.1**
CHRE	6	**mean**	**18**	**2**	**40**	**39**	**4.9**	**7.6**
		min	13	2	8	25	4.5	4.4

		max	23	3	> 45	55	5.3	13.0
FGB	4	**mean**	**23**	**2**	**19**	**4**	**5.9**	**7.8**
		min	20	2	6	0	5.2	6.8
		max	25	3	> 45	8	6.6	10.1
GbA	2	**mean**	**21**	**3**	**12**	**0**	**4.5**	**13.0**
		min	18	3	12	0	4.0	10.8
		max	24	3	12	0	5.0	15.2
GbB	5	**mean**	**12**	**2**	**19**	**0**	**5.4**	**10.0**
		min	9	2	8	0	4.6	7.5
		max	17	3	> 45	0	6.0	15.9
GbD	1	n=1	**12**	**1**	**6**	**17**	**5.3**	**6.0**
MuA	1	n=1	**12**	**1**	**6**	**0**	**6.9**	**9.6**
NoA	1	n=1	**18**	**3**	**8**	**5**	**5.6**	**9.1**
PPA	2	**mean**	**13**	**2**	**8**	**6**	**6.6**	**7.2**
		min	6	2	8	0	6.5	6.0
		max	19	2	8	11	6.7	8.4
PxBs	3	**mean**	**8**	**2**	**38**	**0**	**5.8**	**9.6**
		min	8	2	15	0	5.0	5.6
		max	10	3	> 45	0	6.2	12.8
PXCs	1	n=1	**9**	**1**	**6**	**0**	**7.7**	**8.4**
ScB	2	**mean**	**17**	**1**	**8**	**6**	**7.0**	**7.3**
		min	10	1	7	0	6.7	7.2
		max	24	1	8	12	7.2	7.3
SiA	1	n=1	**9**	**2**	**9**	**8**	**6.6**	**8.6**
WdB	1	n=1	**17**	**2**	**12**	**0**	**5.1**	**8.9**
WdBs	2	**mean**	**18**	**2**	**9**	**2**	**6.0**	**10.4**
		min	9	2	7	0	4.6	7.2
		max	27	2	11	3	7.3	13.6
All Map Units	63	**mean**	**15**	**2**	**15**	**9**	**5.5**	**8.9**
		min	6	1	6	0	4	4.4
		max	47	5	15	55	7.7	15.9

Restored, Unrestored and Less Disturbed Sites

Table 3.3: Soil litter and surface characteristics, all sites. Soil hardness in excess of 300 psi (2 MPa) is sufficient to inhibit plant root growth; all NYC Parks sites had mean soil hardness in excess of this threshold. Average depth to compaction includes only compacted sites. Earthworm castings were recorded as present or absent on the soil surface (1/0). All sampled plots had soil surface cover by earthworm castings. Litter, compaction and earthworm casting data were not available for NYBG plots.

	n		Litter Weight (g)	Litter Depth (cm)	Depth >300psi	Slope %	pH	TOC	Worm Castings
Cunningham	10	**Mean**	**15**	**2**	**20**	**2**	**5.2**	**11**	**1**
		Min	8	2	8	0	4.0	9	1
		Max	24	3	> 45	10	6.0	16	1
Pelham Bay Unrestored	10	**Mean**	**16**	**2**	**25**	**10**	**5.1**	**10**	**1**
		Min	8	1	3	0	4.5	5	1
		Max	27	4	> 45	33	6.2	16	1
Van Cortlandt	10	**Mean**	**15**	**2**	**13**	**14**	**5.4**	**7**	**1**
		Min	11	1	6	0	4.6	5	1
		Max	25	3	> 45	33	6.9	10	1
All Unrestored Sites	30	**Mean**	**15**	**2**	**19**	**9**	**5.2**	**9**	**1**
		Min	8	1	3	0	4	5	1
		Max	27	4	0	33	6.9	16	1
Inwood	10	**Mean**	**16**	**2**	**37**	**18**	**5.9**	**7**	**1**
		Min	8	2	6	0	4.8	4	1
		Max	23	3	> 45	45	7.5	11	1
Pelham Bay Restored	10	**Mean**	**12**	**2**	**9**	**3**	**6.7**	**8**	**1**
		Min	6	1	6	0	6.1	6	1
		Max	24	3	15	12	7.7	11	1
Prospect	10	**Mean**	**25**	**3**	**18**	**17**	**5.9**	**8**	**1**
		Min	13	1	6	0	5.1	6	1
		Max	47	5	> 45	44	7.6	12	1
All Restored Sites	30	**Mean**	**16**	**2**	**21**	**13**	**5.9**	**8**	**1**
		Min	6	1	6	0	4.8	4	1
		Max	47	5	15	45	7.7	12	1
NYBG	10	**Mean**	-	-	-	**24**	**4.5**	**11**	**-**
		Min	-	-	-	5	4.1	5	-
		Max	-	-	-	55	5.1	16	-
All Sites	60	**Mean**	**15**	**2**	**15**	**9**	**5.5**	**9**	**1**
		Min	6	1	6	0	4	4	1
		Max	47	5	15	55	7.7	16	1

All sites contained abundant earthworm castings. Compaction sufficient to impede root growth was encountered in all parks. Soil compaction was widespread and spatially variable.

Total Organic Carbon and pH

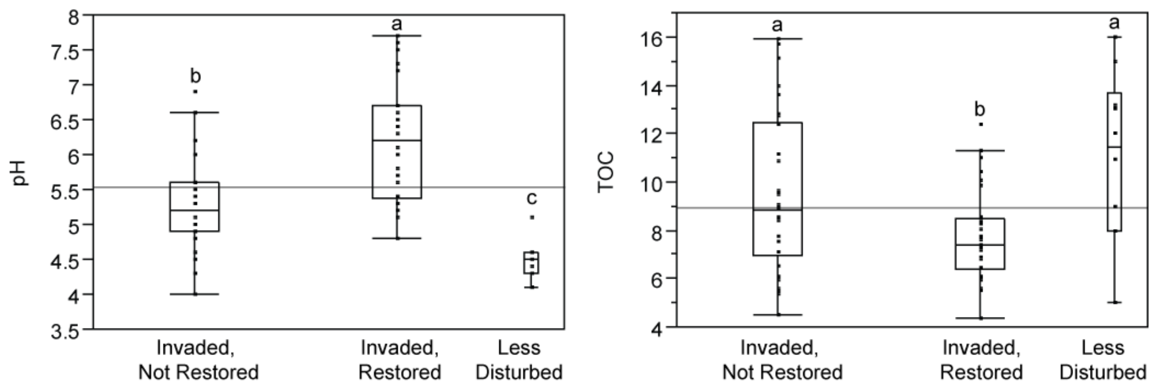

Figure 3.2: Total organic carbon (TOC) and pH in restored, unrestored and less-disturbed urban forests. Different lower-case letters indicate significant differences.

NYBG sites were steeper than unrestored sites (p = 0.0098); restored sites were not significantly different from either NYBG or unrestored sites. Litter weight, litter depth and depth to compaction did not differ between restored and unrestored sites. Restored sites had lower TOC and higher pH than other site types ($p < 0.04$ and $p < 0.0001$, respectively); unrestored sites also had higher pH than the less disturbed forest at NYBG (p = 0.0137).

Soil Metals

All parks had sites with metal levels exceeding the New York State thresholds for Protection of Ecological Resources (NYSDEC 2013; Table 3.4). These standards combine phytotoxicity (20% reduction in growth, following Efroymson et al. 1997) with toxicity to invertebrates and bioaccumulation effects (NYSDEC 2006). Results for mercury were not suitable for comparison to standards due to equipment malfunction.

Table 3.4: Soil metal levels by park. Mean, minimum and maximum values by park are shown in relation to New York State Department of Environmental Protection Standards for Cleanup Operations for Protection of Environmental Resources (NYSDEC 2006), metal levels in Eastern U.S. soils (Shacklette and Boerngen 2007), New Jersey background levels (Fields et al. 1993), and New Jersey Department of Environmental Protection levels recorded in fill soils (Asterisks indicate levels in excess of the NYS DEC standards for Protection of Environmental Resources.

Park		S	Cl	K	Ca	Ti	Cr	Mn	Fe	Co	Ni	Cu	Zn	As	Se	Rb	Sr	Zr	Mo	Sn	Sb	Ba	Pb
Cunningham	**Mean**	**942**	**0**	**9919**	**3970**	**4084**	**59***	**486**	**21921**	**344**	**0**	**39**	**84**	**13**	**1**	**56**	**97**	**499**	**0**	**9**	**0**	**435***	**193***
	Min	0	0	9282	2486	3800	52*	309	19897	311	0	31	57	8	0	46	82	396	0	0	0	392	102*
	Max	1359	0	10687	6495	4401	71*	663	23226	376	0	51*	181*	18	3	64	112	578	0	31	0	514*	316*
Pelham Bay Unrestored	**Mean**	**690**	**0**	**10968**	**6438**	**4612**	**82***	**659**	**29418**	**469**	**10**	**57***	**127***	**17***	**2**	**63**	**145**	**555**	**0**	**10**	**0**	**467***	**229***
	Min	0	0	9309	4196	3992	52*	356	22368	400	0	29	84	0	0	51	116	464	0	0	0	379	77*
	Max	1561	0	11987	9189	5867	110*	955	36709	564	57*	103*	242*	44*	3	73	166	691	0	43	0	607*	467*
Van Cortlandt	**Mean**	**1092**	**0**	**11760**	**7898**	**4867**	**104***	**547**	**30333**	**476**	**9**	**57***	**153***	**28***	**1**	**64**	**182**	**449**	**1**	**5**	**0**	**570***	**380***
	Min	0	0	10332	4461	3761	52*	323	22208	346	0	22	89	7	0	50	122	366	0	0	0	419	83*
	Max	2630	0	15338	12907	9078	230*	850	45348	706	94*	192*	294*	62*	5*	92	486	532	8	45	0	1207*	1823*
Inwood	**Mean**	**737**	**41**	**10167**	**5289**	**3789**	**51***	**586**	**22526**	**400**	**3**	**47**	**165***	**18***	**2**	**62**	**91**	**585**	**0**	**19**	**0**	**431***	**383***
	Min	0	0	9162	2242	3297	38*	321	20473	293	0	31	81	8	0	56	74	420	0	0	0	359	148*
	Max	2546	411	10910	12887	4441	64*	1315	25485	453	25	62*	383*	59*	3	66	134	857	0	44	0	549*	1603*
Pelham Bay Restored	**Mean**	**765**	**0**	**11062**	**5037**	**4827**	**77***	**496**	**29113**	**507**	**0**	**52***	**105***	**9**	**1**	**66**	**134**	**646**	**0**	**0**	**0**	**453***	**192***
	Min	0	0	9148	2840	4325	58*	225	23745	417	0	28	69	0	0	58	111	572	0	0	0	340	77*
	Max	1456	0	12806	7362	5514	104*	923	38881	646	0	135*	179*	15*	3	77	160	733	0	0	0	564*	596*
Prospect	**Mean**	**719**	**0**	**9510**	**3941**	**4112**	**60***	**402**	**19069**	**322**	**2**	**66***	**119***	**16***	**1**	**57**	**86**	**555**	**0**	**14**	**3**	**390**	**198***
	Min	0	0	8688	1371	3712	36	259	14077	217	0	41	72	7	0	41	68	356	0	0	0	291	42
	Max	1565	0	10782	6093	4496	84*	492	23409	422	24	107*	211*	27*	3	69	119	827	0	49	31	493*	428*
NYBG	**Mean**	**0**	-	**7**	**9**	-	**44***	**2**	**5**	**7**	**22**	**39**	**75**	**12**	-	-	-	-	-	-	-	-	**156***
	Min	0	-	4	5	-	33	0	2	5	16	18	49	4	-	-	-	-	-	-	-	-	51
	Max	0	-	12	16	-	50*	9	9	10	26	59*	106*	19*	-	-	-	-	-	-	-	-	270*
NYS DEC SCO PER	-	-	-	-	-	-	**41**	**1600**	-	-	**30**	**50**	**109**	**13**	**3.9**	-	-	-	-	-	-	**433**	**63**
Eastern U.S. Soils	**Mean**	-	-	-	-	-	**33**	**260**	-	**5.9**	**11**	**13**	**40**	**4.8**	**0**	-	-	-	-	-	-	**290**	**14**
	Min	-	-	-	-	-	1	<2	-	<0.3	<5	<1	<5	<0.1	<0.1	-	-	-	-	-	-	-	<10
	Max	-	-	-	-	-	1000	7000	-	70	700*	700*	2900*	73*	4	-	-	-	-	-	-	1500	300*
NJ Background	**Mean**	-	-	-	-	-	**8.6**	**287**	-		**9.7**	**10**	**47.7**	**3.01**	-	-	-	-	-	-	-	-	**17.6**
	Min	-	-	-	-	-	5.5	52	-		5.3	6 2	32.8	1.79	-	-	-	-	-	-	-	-	14.8
	Max	-	-	-	-	-	11.8	660	-		17	11	69.1	3.83	-	-	-	-	-	-	-	-	19.7
NJDEP Fill	**Mean**	-	-	-	-	-	-	-	-				**575***	**13.2***	-	-	-	-	-	-	-	-	**574***

Restored Sites

Soil Classification

Soil mapping units were not significantly related to plant community composition in restored sites (Table 3.5).

Table 3.5: Results of canonical correspondence analysis (CCA) of soil mapping units (soil series phases and complexes, NRCS 2013) in relation to plant community composition of the ground layer, woody understory and trees of restored sites.

	Soil Classification (NRCS)			
	p value Axis 1	F-Ratio Axis 1	*p* value all Axes	F-Ratio All Axes
Ground Layer	0.244	1.303	0.452	1.01
Woody Understory	0.164	1.483	0.164	1.118
Trees	0.266	1.464	0.588	0.961

Parent Materials

Wilcoxon/Kruskal-Wallis nonparametric signed rank tests with Tukey-Kramer HSD (n=30, DF=4) revealed no significant relationships in restored sites between parent materials (Table 3.1) and differences in soil surface characteristics (Table 3.2), or metals (Table 3.4).

Among restored sites, plots in sites classified with different soil parent materials also did not differ significantly in Shannon diversity, evenness, number of species in the ground layer or woody understory, or in tree evenness from soils with other parent materials. However, plots with fill-derived parent materials had greater tree diversity and tree species richness, but not evenness, than deep till soils (prob > ChiSq = 0.0087* and 0.0236*, respectively). Neither fill soils nor deep till soils differed significantly from sites with other parent materials (Figure 3.3).

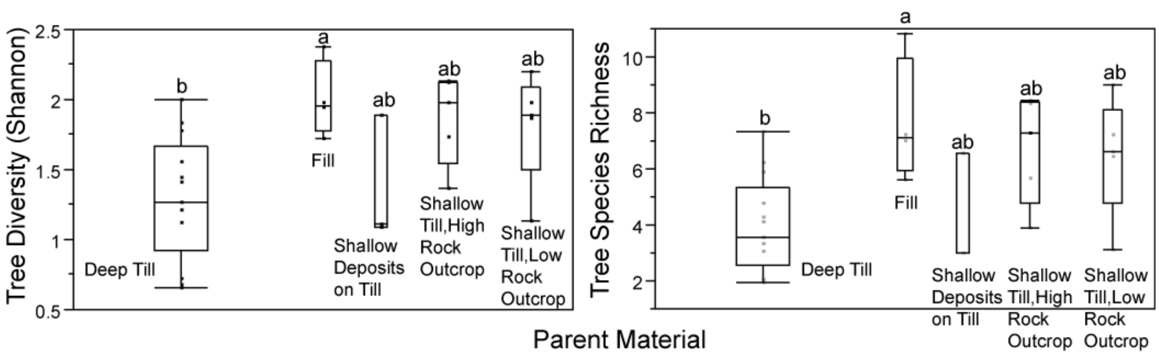

Figure 3.3: Diversity and richness of trees in restored sites. Tree diversity and tree species richness (prob > ChiSq = 0.0087* and 0.0236*, respectively) were higher in sites with fill soil parent materials than in sites with deep till parent materials, but neither was significantly different from sites with other soil parent materials.

Soil Metals, pH and Total Organic Carbon

Soil chemical properties were associated with differences in plant communities of restored sites, but none of the parameters tested dominated in its influence on restored plant communities.

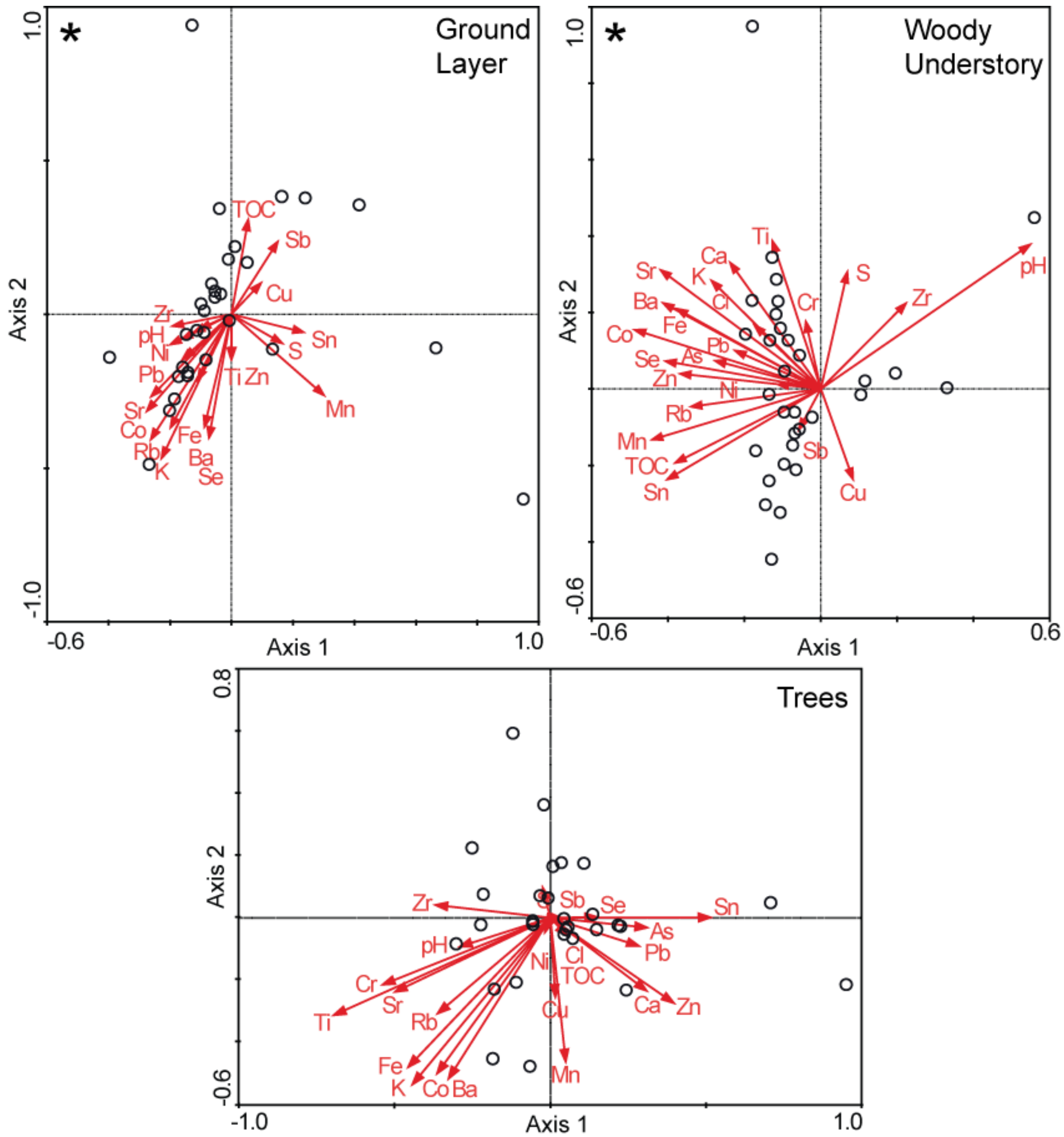

Figure 3.4: Canonical correspondence analysis (CCA) of soil pH, metals and total organic carbon (TOC) in relation to ground layer, woody understory and tree composition in restored sites. Soil chemical properties are displayed as arrow vectors, where arrow length indicates the proportion of the variability associated with that factor, angles between arrows indicate correlation of variables, and distance between plot symbols approximates the similarity of their ground layer plant communities. Longest vectors

indicate the environmental variables explaining the greatest proportion of the variation between plot plant communities.

Metal levels, pH and total organic carbon were associated with differences in plant community characteristics in the ground layer and woody understory of restored sites. The strength of this association was similar among the parameters tested (Figure 3.4, Table 3.6).

Table 3.6: Results of canonical correspondence analysis (CCA) of soil pH, metals and total organic carbon with ground layer, woody understory and tree composition in restored sites.

	Ground Layer		**Woody Understory**		**Trees**	
	F ratio	*p* value	F ratio	*p* value	F ratio	*p* value
Axis 1	0.4	0.026*	0.483	.0380*	0.571	0.48
All Axes	0.864	0.842	1.299	.0240*	1.087	0.232

Soil Surface Characteristics

Litter weight and depth were associated with differences in woody understory and tree composition, but not in the ground layer. Primary target species were not associated with compaction, slope, litter weight or litter depth in restored sites, or when all sites were combined.

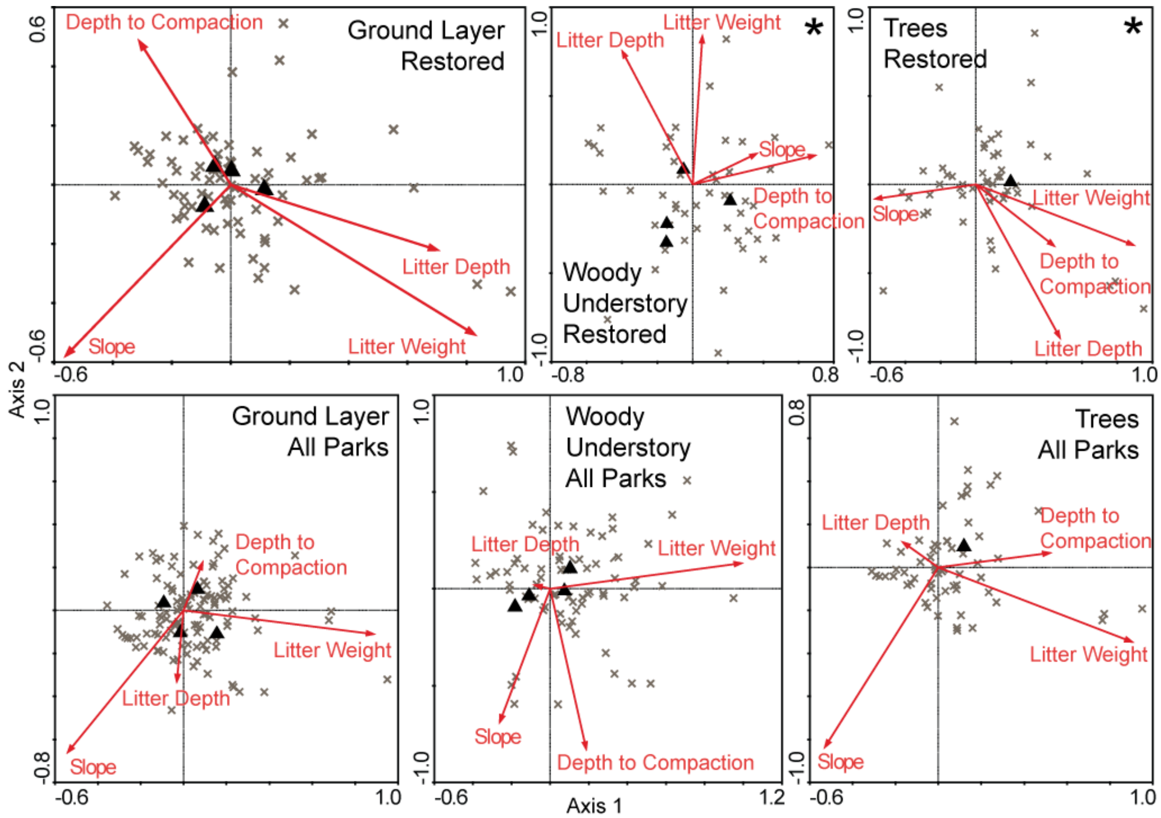

Figure 3.5: Species of restored and all invaded park sites arrayed in relation to surface soil physical characteristics. Targeted invasive species are shown as black filled triangles (▲); all other species are represented by an x.

Relationships between species composition and abundance were significant only in restored woody understory plants and trees (Table 3.7). Litter weight and depth were correlated with woody understory and tree community composition (Figure 3.5). Targeted invasive species were less strongly associated with the soil surface physical characteristics than many other species in all forest strata (Figure 3.5).

Table 3.7: Results of canonical correspondence analysis (CCA) of relationships between plant community composition and soil surface conditions in restored and all invaded sites. Variables included were depth to compaction >300 psi (2 MPa), litter depth, litter weight, and slope.

		Ground Layer		**Woody Understory**		**Trees**	
		F ratio	*p* value	F ratio	*p* value	F ratio	*p* value
Restored	Axis 1	1.178	0.5720	0.1873	0.0020*	1.801	0.0300*
	All Axes	0.921	0.8480	1.485	0.0020*	1.345	0.0100*
All Invaded	Axis 1	1.391	.4400	0.1890	0.1680	2.289	1.299
	All Axes	0.807	0.9300	1.074	0.2960	0.0780	0.1120

DISCUSSION

No single one of the soil characteristics examined in this study had a dominant effect on plant community composition. At the municipal scale, soils of invaded urban forest remnant sites exhibited a variety of impacts consistent with urban heterogeneity of land use history. Although some invaded forest sites were located in soils derived from human-deposited materials, restored plant communities on these soils did not differ significantly from other restored sites as a group. Urban forest fragment soils exhibited multiple indicators of human disturbance, including trampling and compacted layers, introduced earthworms, remains of built structures, and discarded building materials and household refuse. Many sites had high levels of heavy metals, which may be a result of both atmospheric deposition and, in some areas where multiple metals were high, dumping sites.

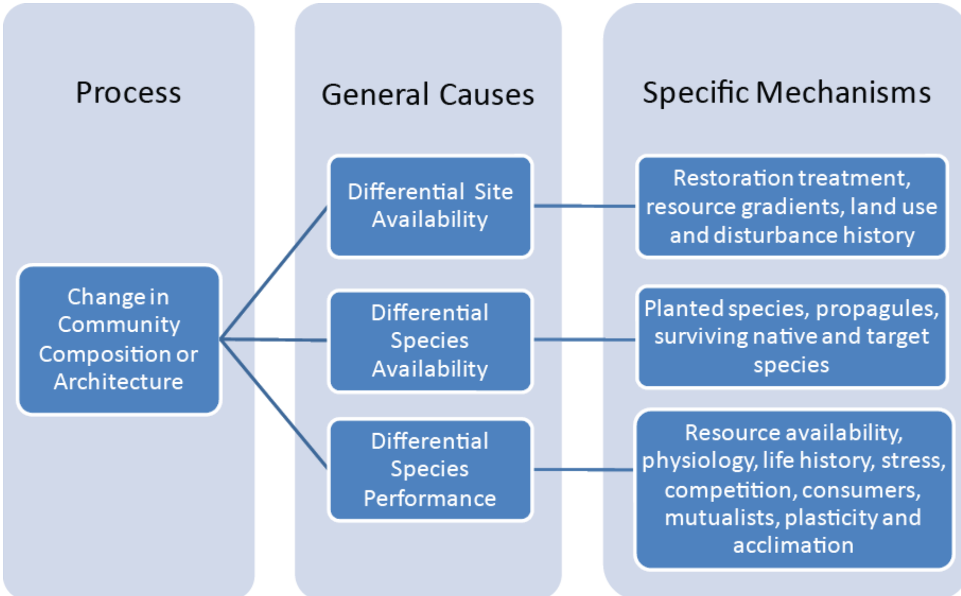

Figure 3.6: Factors influencing successional processes in ecological restoration, after Pickett et al. (2011) and Pickett and McDonnell (1989).

Effects of the urban environment on forest remnant soils interact with site history, disturbance, and species availability to influence the long-term outcomes of restoration (Figure 3.6). In urban environments, heat island effects, remnant species pools, exotic species introductions, and altered disturbance regimes combine to influence community composition. Effects of the urban environment on soils contribute to differential species performance, altering resource availability and stress in relation to plant species physiology and tolerances. In this study, multiple influences of the urban environment were found to change soil conditions of invaded forest patches. Each of these impacts has the potential to influence plant community composition. Combinations of these factors in sites across the city indicate that urban forest patches are subject to multiple impacts in the urban environment, and that these factors should be considered in decisions regarding site soil preparation and plant choice for ecological restoration.

Anthropogenic Parent Materials

The classification and mapping of soil has only recently begun to include soils changed by urban land use (Pouyat et al. 2010). New York City Soil Survey maps used in this study improve upon prior 1:62,500 scale mapping with a minimum delineation area of 40 acres (NYCSWCD 2009). This represents an important step toward mapping at a scale that approaches the scale of variation in urban soils. However, it does not reach the 1:6000 large scale level recommended by Effland and Pouyat (1997) for urban areas. In urban environments, the finer the scale of the observation, the more likely soil conditions can be related to human-caused change than to non-anthropogenic factors (Pouyat et al. 2010).

In the sites sampled here, factors other than parent material (Figure 3.6) were more important to plant community outcomes than parent material. Tree diversity and species richness in sites with fill-derived soils were significantly different only when compared to sites with soils derived from deep till (Figure 3.3), and no differences were seen in ground layer or woody understory richness and diversity. The lack of significant relationships between soil parent material seen here, despite the importance of parent materials to urban soils (Pouyat et al. 2007), likely reflects the patchiness of cities at fine to medium spatial scales as a result of urban climatic, topography, land cover, infrastructure, organisms and social effects (Pickett and Cadenasso 2009). Exotic plant invasions are often associated with anthropogenic disturbance (Lockwood, Hoopes, and Marchetti 2006), and the invaded sites examined in this study exhibited evidence of multiple disturbances, the nature of which varied among locations. This finding emphasizes the importance of site-specific site history and site condition data in planning for ecological restoration in urban environments. Larger-scale mapping should be combined with historic information that addresses the fine scale of human-caused changes in order to provide information useful for site-specific decision making.

This finding also suggests a systematic study of relationships between anthropogenic soils and plant communities as a direction for future research. Managers have observed relationships between plant community composition and soils containing dumped or dredged materials (K. Bounds and M. Feller, personal communication). These associations deserve further attention.

Surface Soil Characteristics

Compaction

Both short-term and long-term effects of trampling have been shown to result in reduced plant cover, plant height, species density, and leaf litter biomass while increasing soil density. Long-term and more intense trampling, common in urban parks, results in more pronounced effects (Kissling et al. 2009). Trampling and high density of footpaths have been associated with species loss in urban woodland fragments (Drayton and Primack 1996). Trampling is only one of several potential causes of soil hardness in urban parks, but it was an important concern of managers involved in the restoration efforts described here. Restoration activities included fencing and signage to decrease off-trail use of forested areas by park visitors.

Soils exhibiting resistances of 1MPa (145 psi) or greater can it reduce root growth, and at 5MPa (725 psi) root growth may be completely inhibited (Passioura 2002). The threshold of 300psi (2MPa) used here indicates soil hardness with potential to affect the growth of plant roots. The depth to this degree of soil hardness was not uniform, but presence of compacted layers in all sites and most plots indicates that soil compaction is likely to affect the outcomes of urban ecological restoration both by influencing rooting and growth conditions for planted species and by effects on moisture, hydrology, aeration and other conditions important to germination and establishment of all plants.

pH

Average pH values were within ranges suitable for plant growth, although some sites had low enough pH (< 5.0 - 5.5) that aluminum or manganese toxicity could occur (Bell 2002). Mean pH values for both restored and unrestored were higher than those found in rural mixed oak forests 80 km from New York City, which ranged from 3.65 to 4.55 (Schuster et al. 2008). Soil pH in the less-disturbed forest of the New York Botanical Garden was closest to this reference (Figure 3.2); soils in unrestored sites were significantly higher, and restored sites had the highest pH. Variation in soil acidity outside ranges of native plant species tolerance may impact plant survival, growth and vigor, influencing community composition.

Significantly higher pH in restored sites is may warrant further investigation. The modern disturbance history of the invaded sites examined in this study is not completely known by park managers and it is possible, based on the metals analysis discussed below, that disturbance involving dumping and filling with a variety of materials could have been related to initial invasion in some sites. Base cations are often increased in urban soils by dumping, filling, weathering and deposition of calcareous building materials like concrete and limestone (Pouyat et al. 2007). The cause of the relationship between restoration and soil acidity is beyond the scope of this study, but may deserve further attention.

Total Organic Carbon

Invaded sites that were not restored tended to be densely covered by woody shrubs and vines, while restored sites had lower ground-layer cover and more young trees, while less disturbed sites were forests with mature trees and multi-layered canopies. It is interesting that unrestored sites and the less disturbed forest at the New York Botanical Garden were not significantly different in total organic carbon content. This may indicate a site difference not captured by the variables sampled, or an effect related to species composition, earthworms, or restoration.

Large quantities of woody vegetation were removed from restored sites, so lower organic carbon may follow decreased litter inputs in restored sites compared to sites that were not restored, and to less-disturbed sites where carbon had a longer time to accumulate following the most recent disturbance. Differing species composition in restored and unrestored sites may also have contributed to observed differences in organic carbon. All sites had abundant earthworm castings on the soil surface, but differences may exist in density or species composition.

Litter

Average values for litter weight and depth are similar to those found by Kostel-Hughes and others (1998) in New York City forests, and are low compared to more rural deciduous forests. New York City forests have been shown to have lower leaf litter depth, mass and density, faster leaf litter decomposition, and decreased fungal activity relative to rural forests in the region, including colonization rates and species richness of mycorrhizal fungi (Pouyat, McDonnell, and Pickett 1997; Karpati et al. 2011). These trends have been associated with soil metal contamination, warmer soils resulting from urban heat island effects, and with abundance of non-native earthworms (McDonnell et al. 1997).

All restored and unrestored Park plots had abundant earthworm castings on the soil surface. Invasion of non-native earthworms has been associated with changes in soil structure, nutrient availability, burial of surface materials, incidence of root diseases, and plant growth (Baker et al. 2006). Non-native earthworms can increase both rates and depth of mixing of surface litter, with important consequences to soil moisture, germination sites, nutrient availability, and soil carbon and nitrogen cycling (Bohlen et al. 2004; Baker et al. 2006; Szlavecz et al. 2006). In New York, abundance of introduced earthworms in urban sites has been shown to reduce safe sites for germination for large-seeded species via rapid leaf litter processing. The same study associated more abundant exotic earthworms in urban sites with higher temperatures resulting from urban heat island effects, and with urban atmospheric nitrogen deposition (Kostel-Hughes, Young, and Carreiro 1998).

Changes to litter dynamics can influence the conditions for plant germination, establishment, and growth (Facelli and Pickett 1991), as well as the survival of planted individuals. Soils of urban forest patches in New York City have been found to have higher abundance of non-native seeds in their seed banks (Kostel-Hughes, Young, and McDonnell 1998). Altered litter dynamics can affect competitive dynamics between desired native plants and exotic species (Schramm and Ehrenfeld 2010), affecting the plant community trajectories that ecological restoration aims to direct. Effects of the urban environment on soil litter dynamics should be taken into account in urban ecological restoration. Changes in plant species composition, soil biota, temperature and disturbance regimes may result in conditions that fall outside of the range of tolerance for native species. Current conditions should be thoroughly characterized and species chosen based on current and predicted soil conditions.

Metal Concentrations

All sites sampled contained plots with metal concentrations high enough to potentially affect plant growth (Efroymson et al. 1997). This finding has important implications for ecological restoration in cities. Urban soils may have high levels of contamination by heavy metals including lead, copper, chromium and zinc derived from industrial processes, interior and exterior lead paint, widespread deposition of the combustion products of leaded gasoline, and the wear of tires, brake linings, building materials, galvanized fencing, and treated lumber (Pickett and Cadenasso 2009; Effland and Pouyat 1997; Pouyat and McDonnell 1991; Craul 1999). These materials can both pose threats to human health and alter ecosystem processes (Pickett and Cadenasso 2009).

Studies in the New York City metropolitan area have shown decreased abundance of soil macroinvertebrates, decreased density of soil microarthropods, and decreased soil fungal biomass in the presence of metal levels approaching or exceeding levels found to affect soil invertebrates and soil microbial processes (McDonnell et al. 1997). They have also shown relationships between total soil metal load and community composition of regenerating forests, with different developmental trajectories above and below critical metal thresholds (Gallagher et al. 2011). However, microbial processes have also been shown to be more rapid in urban soils (Pouyat et al. 2010), and the widespread abundance of worm castings in all sites (Table 3.3) indicates that toxicity is not currently limiting earthworm populations.

The type of analysis performed in this study did not provide information regarding the speciation of the metals detected, which limits its usefulness in understanding the bioavailability (and thus the toxicity) of these elements, which is mediated by both the compounds they form and other soil characteristics such as pH (Ye, Baker, and Wong 2002). However, mean metal levels exceeding NYS DEC standards based on plant toxicity indicate that metal levels may impact the plant communities in these sites. Disturbed soils that lose alkalinity and organic matter may have a decreased capacity to bind metals, resulting in increased availability and transport (Pouyat et al. 2010). Soil effects of disturbance and biomass removal associated with restoration should be considered among the potential effects of restoration activities. Soil amendment may be necessary prior to restoration in urban sites with high metal concentrations.

Conclusion

Changes in chemical, physical, and biotic components of urban soils can profoundly affect aboveground communities. Together, these findings emphasize the importance of effects of the urban environment on soils to long-term outcomes of urban ecological restoration. Among the restored sites examined here, no one urban soil impact stood out as an influence on vegetation dynamics. Restored sites were similar in their degree of invasion by exotic earthworms and presence of compacted soil layers. They exhibited heterogeneity in other impacts, including variation in the metals found in their soils. All sites were impacted by at least one anthropogenic factor known to reduce plant growth, change distributions of soil biota, and alter nutrient cycling, but type and degree of impact varied among sites.

High pH and low organic content in restored sites indicate that restoration itself is a management practice that affects soils. To better understand and predict outcomes of restoration, effects of restoration itself on soils should be considered.

Complex interactions between anthropogenic and natural processes of soil formation result in urban soil conditions that are difficult to predict from larger-scale geological information, and legacies of prior land use are often not known. A site-specific approach that anticipates common soil conditions of urban environments is needed in urban ecological restoration. Fragments of remnant forest in cities are disproportionately important for the social and ecological benefits they provide to urban residents. Their small size and degree of urban impact requires attention to urban environmental changes such as local microclimate, nearby sources of propagules, human use and visitation patterns, and legacies of past land use, all of which influence vegetation development (Figure 3.6).

To account for soil conditions resulting from shifting patterns of urban land use over time, finer-scale mapping of urban soils is needed. Thorough soil characterization will improve the success of urban restoration efforts by improving the fit between current conditions and survival of planted species. Soil conditions in urban forest remnants may exceed parameters of species tolerance, and so soils may need to be amended or species choices adapted to existing conditions. Some impacts of the urban environment on forest fragment soils are likely to be persistent, such as continued urban heat island warming, atmospheric deposition of pollutants, and high use and visitation by urban residents. These should be considered and incorporated in restoration planning. New tools for predicting urban soil conditions from urban plant community composition could be developed by treating naturally occurring and human-caused urban soil heterogeneity as a set of natural experiments. Current vegetation may give clues as to past history and legacies of toxicity, informing assessment of restoration potential.

Legacies of land use may override restoration treatments where those treatments are not tailored to site conditions. Thorough understanding of urban soil conditions combined with site preparation and planting plans that take these conditions into account may greatly increase the likelihood of survival for planted species, and of desired long-term changes in plant community trajectories.

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CONCLUSION

Targets and Goals in Urban Ecological Restoration

Ecological restoration seeks to alter trajectories of community development over time, with the goal of self-sustaining, functional ecosystems that resemble pre-disturbance or reference conditions (SERI 2004). Relatively pristine reference sites and historical records are important to understanding the historic potential of a site's biophysical conditions, and functioning ecosystems resembling those formerly found where cities now stand contain species assemblages, interactions and processes that developed over time in response to local climate, topography, biota, and soils. However, where baseline conditions have changed, reference states are no longer likely or possible outcomes. Adjustments to the framework are necessary. New reference states are needed.

Target states for restoration are often the least-disturbed regional reference sites, which have had decades or centuries of soil and biotic interactions since they may last have experienced major disturbance. Many restoration plans assume that adding and/or removing individuals of various species to resemble a community that often develops into the target state will lead to the desired end. However, changed conditions may mean that target states are only partially realized or never occur. The appropriateness of this approach to setting restoration goals has long been questioned in non-urban sites (Hobbs and Norton 1996; White and Walker 1997). It is based on a climax model of ecological succession that does not take into account current advances in ecological theory predicting multiple outcomes and alternate stable states (e.g., Pickett, Meiners, and Cadenasso 2011; Suding, Gross, and Houseman 2004). In urban areas, which are drastically transformed, goals set using this approach are unlikely to be achieved within a few human lifetimes. Current advances in successional theory predict that differences in site availability, species pools, and species performance will determine changes in community composition and architecture (Pickett, Meiners and Cadenasso 2011). In urban environments, mechanisms and constraints that influence differential site availability, species availability and species performance are influenced by urban environmental conditions; where disturbance regimes are altered and unlikely to return to prior conditions in the foreseeable future, there will continue to be many recently-disturbed sites and their future composition will be determined in part by novel assemblages and interactions that draw upon a global species pool. The performance of species will be influenced by alterations to resource availability, stress factors, and the abundance of mutualists and predators, all of which are influenced by the urban environment.

Neither of the divergent trajectories in vegetation dynamics observed in this study between restored and unrestored invaded forests is likely to result in a forest resembling New York forests of the pre-colonial period. A different approach to setting goals and evaluating success for urban ecological restoration is required if land managers are to choose attainable near-term – much less long-term – goals. Managing urban succession for ecological restoration requires recognizing that a trajectory that differs from pre-urbanized conditions may not be failure. It also requires a species-specific approach to evaluating the effects of newly introduced organisms, combined with management of the frequency, intensity, and nature of disturbance, and long-term management of long-term processes. New ways of understanding, valuing, restoring and preserving the biodiversity and ecological function of natural areas in cities are needed.

The findings of these studies indicate that ecological restoration targets for urban habitats must include variables that reflect the effects of urban landscapes on conditions that affect species pools, site availability, and species performance (Figure 2.1), using frameworks like those that have been developed for urban soils (Effland and Pouyat 1997; Pickett and Cadenasso 2009). The boundaries of what is possible in a site may have changed irrevocably. Ecological restoration in such cases, then, must also burst its former boundaries. There is fruitful ground for collaboration between the science of restoration ecology and the practice of design in the urban environment (Palmer et al. 2004; S. Pickett, Cadenasso, and McGrath 2013). The less-disturbed reference site remains an important baseline from which to understand historic potential and as a palette from which to draw communities, but its utility as a target state is limited in cities where disturbance preempts prior trajectories of vegetation and soil development. A shift in focus is needed from the static endpoint of a pristine climax to a multifaceted projection of what is possible given current conditions. Strategies based on such an approach will be more likely to succeed.

A Possibility-Centered Approach to Urban Ecological Restoration

Degraded sites should not be viewed as a temporary problem to be surmounted. Rather, they are the starting point from which complexity, biodiversity and ecosystem function may be increased. They will change over time, but they may not follow trajectories that increase desired ecosystem properties and functions without intervention. What can be done to increase ecological function and environmental benefits in urban landscapes? I suggest a possibility-based assessment and phased approach for ecological restoration in urban environments.

Such an approach must begin with a thorough understanding of current conditions. This requires knowledge – or inference – of the past. Context-specific knowledge, historical documents, maps and records should be consulted to map and plan for urban ecological interventions. These findings showed that urban soils, urban propagules, and urban disturbance regimes were particularly important to restoration outcomes in urban forest patches.

Urban Soils

Urban environments are characterized by a high degree of spatial heterogeneity (Cadenasso, Pickett, and Schwartz 2007; Pickett and Cadenasso 2009). This heterogeneity is the result of both a tendency for cities to be located in sites of high biodiversity and heterogeneity (Kuhn, Brandl, and Klotz 2004), and by human cultural practices that add species over time both intentionally and unintentionally. The vegetation of cities like New York has been subject to a series of different phases of human influence, from forest clearing to agriculture to construction and sealed soils, resulting in heterogeneous patterns in the biophysical environment.

Changes in chemical, physical, and biotic components of urban soils can profoundly affect aboveground communities. The introduced earthworms observed in all sites, for example, may affect restoration success due to influence on soil and hydrologic properties (Szlavecz et al. 2006; Baker et al. 2006; McDonnell et al. 1997). The effects of these important soil organisms on vegetation dynamics and restoration outcomes deserve further attention (Bohlen et al. 2004). Soils sampled in this study had soil hardness sufficient to inhibit root growth (cf. Passioura 2002), widespread metal contamination with potential to affect plant growth, and low depth and weight of surface litter consistent with past findings in New York City forests (Pouyat, McDonnell, and Pickett 1997; Kostel-Hughes, Young, and Carreiro 1998). Combined with evidence that urban environments alter soil temperatures, nutrient cycling and mycorrhizal communities (Carreiro et al. 2009, Karpati et al. 2011, Pavao-Zuckerman 2008), these findings emphasize the importance of understanding urban soils to long-term outcomes of urban ecological restoration. Among the restored sites examined here, no one urban soil impact stood out as an influence on vegetation dynamics. Instead, sites were similar in their degree of invasion by exotic soil biota and exhibited heterogeneity in other impacts, including variation in the metals found in their soils. All sites were impacted by anthropogenic factors known to reduce plant growth, change distributions of soil biota, and alter nutrient cycles.

Effects of the urban environment on soil biota and nutrient cycling are likely to influence the success of native species plantings, and conditions for germination, establishment and growth. They may also affect competitive dynamics between desired native plants and exotic species in the long term, affecting outcomes of urban ecological restoration. The significant effects of treatment on plant community outcomes observed in this study demonstrate the power of restoration treatment to change trajectories of vegetation development over time. However, soil conditions were important to differences between restored sites, and all sites were impacted by at least one effect of urban conditions on soils that can affect plant growth. Soil legacies may override restoration treatments where those treatments are not tailored to particular site conditions. Thorough understanding of current soil conditions may greatly increase the likelihood of survival of planted species.

In urban environments, the finer the scale of the observation, the more likely soil conditions can be related to human-caused change than to non-anthropogenic factors (Pouyat et al. 2010). The lack of significant relationships between soil classes and vegetation seen in this study, despite the importance of parent materials to urban soils (Pouyat et al. 2007) likely reflects the patchiness of cities at fine to medium spatial scales as a result of urban climatic, topography, land cover, infrastructure, organisms and social effects (Pickett and Cadenasso 2009). In the context of ecological restoration, finer-scale mapping combined with information that addresses the fine scale of human-caused changes is needed to provide information for site-specific decision making.

Urban Propagules

Urban habitat fragments also often contain invasive species. Small remnant habitats can be overwhelmed by prolific growth, and edge effects can contribute to invasion. Invasive plants are almost never completely eradicated once established, though much management often focuses on eradication as a goal. Even relatively less disturbed sites may be invaded, including the old-growth urban forest described here. In urban forest fragments, edge effects have important influences on habitat conditions. Among these are propagule introductions, especially of bird- and wind-dispersed plant species.

Urban environments are loci of species invasion that provide a wide variety of habitat types and resource subsidies to intentionally and unintentionally introduced species (Klotz and Kühn 2010; Von Der Lippe and Kowarik 2007). Several new invasive species had become abundant in the 20 years following the initiation of restoration in New York City parks, including garlic mustard (*Alliaria petiolata*) and goutweed (*Aegopodium podagraria*). Species are constantly introduced to urban areas (Kowarik 2008), and managers of urban forests should expect the arrival of novel plants, animals and pests in the future.

Urban Disturbance Regimes

Urban environments are subject to frequent human disturbance, so much so that a snapshot of any city today would contain only a handful of buildings that will still stand in two centuries. An understanding of past, current and likely future disturbance types and frequencies may increase the success of restoration efforts. For example, failure of tree regeneration is common in urban and suburban forests due to both reduced natural disturbances and increased human disturbances (Guldin, Smith, and Thompson 1990; Broshot 2011). In New York City, Rudnicky and McDonnell (1989) found that increased abundance of non-native trees in the ground layer and understory was due to arson, trampling, and vandalism in addition to non-human disturbances. Both sanctioned and unsanctioned uses will affect outcomes of urban ecological restoration, and human disturbances like soil compaction can be compounded by natural disturbances like large storms.

By anticipating patterns resulting from prior human disturbance (like the effects of former building sites on soils and plant composition) and from changes in disturbance type and frequency (like changes in fire regimes), more effective responses can be developed. If future disturbance is likely to exceed the regenerative capacity of recovering habitats, management may need to be increased or preventative measures may need to be engaged to decrease disturbance.

Disturbance that is characteristic of urban areas will require future restorations. Long-term management is necessary for long-term processes, and to understand the nature of these changes and to make informed decisions about management, long-term monitoring, research, and continuity of institutional memory are also important.

Adaptive Successional Phasing: Time as a Tool for Urban Ecological Restoration

In highly degraded systems where altered soils, habitat fragmentation and multiple exotic species introductions are the norm, long-term ecological processes are an important tool for restoration (Bradshaw 2002). By anticipating impacts of the urban environment and changes over time due to ecological succession, management can be targeted as ecological processes unfold. To address the challenges of ecological restoration in the urban environment, I propose using adaptive successional phasing to plan to direct urban ecological restoration.

Urban Influences on Ecological Succession

The processes that influence vegetation development over time (Pickett, Meiners, and Cadenasso 2011) are the same in large forests, old fields and urban parks. However, the outcomes of vegetation development in the urban environment will be different from that of non-urban environments that once shared a regional set of species and environmental conditions.

While succession following agricultural abandonment or forest gap creation may follow similar patterns if undisturbed, the frequency, intensity, type and duration of disturbance events are of a different magnitude in urban environments (Alberti 2005). Sites are seldom left even relatively undisturbed by direct and indirect effects of shifting use patterns. Urban site conditions are highly heterogeneous at the landscape scale (Cadenasso, Pickett, and Schwartz 2007; Cadenasso and Pickett 2008), and a number of different land uses may have followed agriculture in a given site, each with its own legacy of biotic and abiotic conditions. These legacies include frequent introduction of new species. Forest gap succession quite rightly presupposes a surrounding forest. However, urban forests are small patches in an urban matrix, which provides different conditions in terms of propagule quantity and identity, buffering from other land uses and habitat types, and local climate and hydrology (Ehrenfeld 2000). Pre-urbanization species composition and ecological processes may be altered or absent, and resulting communities may be stable and resilient (Suding, Gross, and Houseman 2004). These limitations should be kept in mind when using successional theory to guide urban ecological restoration.

Conceptual frameworks have been developed to describe urban ecosystems that incorporate the pervasive influence of anthropogenic change that is typical of cities, incorporating social variables (Grimm et al. 2000; Pickett et al. 2009; Pickett and McDonnell 1989; Pickett, Meiners and Cadenasso 2011). This study illustrates the influence of urban environmental conditions on site and species availability and differential species performance that determine long-term trajectories of change in community composition. The mechanisms and constraints governing site conditions and species pools are both biophysical and social in nature. Urban ecological restoration will benefit from expanding its focus to include both social and ecological dimensions of urban ecosystems to better predict and manage long-term outcomes.

Thriving Urban Natives

Trajectories of development in novel assemblages are unknown. To preserve and enhance biodiversity in the urban environment, I suggest that identifying native species that are surviving and thriving under current conditions as a first step in restoration. These species that persist despite the transformation of their habitats by urbanization deserve further study (Clemants and Moore 2003; Moore et al. 2002; Del Tredici 2010; Kowarik 2011). With knowledge of their requirements and tolerances, thriving native species may be used to understand limiting present and past conditions that may need to be ameliorated before other species can be re-introduced. The potential for species' requirements and traits to be used to identify soil conditions is a subject worthy of future study. Currently thriving native species have potential to be useful for initiating restoration under current conditions, as they will provide habitat value to native plants and animals in the short term, before additional native species that may be dispersal-limited by the urban matrix or requiring remediation of urban biophysical conditions could be reintroduced.

In the forests examined here, restored and unrestored forests shared a suite of spontaneous, shade-intolerant, quickly-growing trees that colonize disturbed sites, including *Prunus serotina* and *Robinia pseudoacacia*. The most successful planted species (e.g. *Liriodendron tulipifera*) also shared these characteristics (Appendix 1A). Native *Rubus* and *Vitis* species were common in open vinelands dominated by invasive plants. Additional abundant, regenerating native tree species were found when all sites were included, such as *Acer rubrum*, *Quercus palustris* and *Carya cordiformis,* swamp species that tolerate a wide range of soil conditions including low oxygen. Whether thriving native species like these could be used effectively to compete with invasive plants as part of a restoration strategy remains to be investigated. Treating the existing urban species pool as a palette may help to identify species that may thrive in particular restoration sites, where initial conditions may include high light and the diverse, altered soil conditions typical of urban sites. Additional analysis of species traits and tolerances may yield further insight, increasing our ability to predict what species will succeed.

These findings indicate that restoration sites were hospitable for germination and/or release of seedlings and saplings of species adapted to high light conditions and disturbed soils, also known as early successional conditions. When ecological restoration involves creating a bare, open soil condition where there is little shade, species that are well-adapted to those conditions should be used to establish the initial phase of the restoration. More shade-tolerant species that compete well under higher-nutrient and lower light conditions should be introduced in a subsequent phase to reduce the likelihood of reinvasion. The species most frequently planted in the restoration studied here, slower-growing *Quercus rubra*, was not as abundant in restored plots as more shade-intolerant planted and spontaneous native species. Adaptive phasing over time would allow for red oak and other more shade-tolerant species to be introduced later in forest development. In disturbed sites, shrubs have been shown to have facilitative effects in the restoration of trees (Gómez-Aparicio et al. 2004; Gómez-Aparicio 2009); in light of this, planting native shrubs in advance of tree planting may be worthy of consideration.

Composition and structure of restored urban forests should be expected to change over time, and should not be expected to arrive at a stable state (Pickett and McDonnell 1989). Vegetation development is a long-term process, and restoration should be considered a long-term project. Processes within the habitat will also cause change, and may provide habitat conditions for species that would not thrive early in the restoration process. Additional species introductions should be planned for as their requirements are met by changes in biophysical conditions, such as shading by canopy closure or development of soil organic matter.

The Role of Long-term Management in Urban Ecological Restoration

The restored forests examined here differed from unrestored forests in their vertical structure. They had more tree stems, greater tree stem area, fewer woody understory stems, less ground layer cover, and more of their ground layer vegetation was composed of herbaceous plants. They had more native canopy trees and were less dominated by exotic shrubs and vines, achieving a central goal of the restoration project. The prevalence of other, secondary target invasive species was also reduced where restoration took place, and native tree seedlings were more abundant. This contrast in structure suggests that, without intervention, invasion by this suite of woody species is a long-term change with persistent negative effects on the recruitment of canopy trees. The ground layer and woody understory of unrestored sites remained dominated by woody invasive plants that were abundant in those locations 15-20 years before.

The differences observed here in woody understory composition and vertical structure suggest divergent successional trajectories between restored and unrestored sites, and a lasting effect of restoration treatment after 15-20 years. These differences were most strongly associated with whether or not a site was restored, and with soil surface cover. Present differences in cover, stem density, and species composition observed between restored and unrestored sites present different site conditions for the germination, establishment and growth of plants. These differences in availability and condition of sites for plant growth interact with restoration-altered abundance of plant propagules, setting the stage for different pathways of vegetation development in the future.

Among sites that were restored, differences between sites were strongly associated with the amount of restoration effort applied at the site over time. Targeted invasive species *Rosa multiflora*, *Ampelopsis brevipedunculata,* and *Celastrus orbiculatus* in the ground layer were associated with less management after the initial restoration treatment, especially less frequent planting. Although the restoration methods examined here removed invasive plants, not all propagule sources for these species were removed from restoration sites. Soil seed banks, root fragments, and nearby individuals may all contribute new individuals of both desired and invasive plant species. Continued removal of invasive plants and introduction of native species impacted long-term plant community composition. This finding highlights the importance of ongoing management following initial restoration.

One of the major shortcomings of current efforts toward ecological restoration, both in the urban environment and elsewhere, is the idea that somehow long-term ecological processes can quickly be “set right” and that no further management will be needed. This perception is detrimental to the long-term effectiveness of ecological restoration. While the amount of effort required should decrease once disturbances and their legacies are removed or remediated, ensuring that the initial investment is not overwhelmed by unanticipated changes requires monitoring and adaptive management. Legacies of past land use may increase the amount of effort required to provide adequate conditions for the growth of desired species. In fragmented and frequently-disturbed urban locations, the idea that a restoration is finished once planted is especially short-sighted. For urban habitats to provide long-term ecological benefits to the residents of cities, they will need long-term protection, management and restoration. Successional change plays out over the lifetimes of multiple generations of organisms. Forests are structured by long-lived organisms; individual red oak trees may not produce seeds until they reach 50 years of age. Where dominant organisms may live 200-300 years, ecological processes their life cycles control can be long-term in nature.

While the reality for many land managers is that funding comes in short bursts and does not cover maintenance beyond a brief initial phase, altering trajectories of ecological community development over time requires a long-term perspective and long-term engagement. If it is possible to embrace the long-term nature of vegetation development, time can become a tool. Using principles of ecological succession, restoration plans can be developed that incorporate and utilize change over time.

The findings of this study support the conclusion that that investment of time, materials and labor affect the long-term outcomes of urban ecological restoration. The level of management effort expended in a site influenced long-term outcomes of urban ecological restoration, and the degree to which sites resembled desired conditions was associated with this effort.

These findings also point to another place in which ecological restoration must move beyond its former boundaries to encompass urban environments. Where human disturbance is extensive, diverse, pervasive and frequent, management requirements are likely to be higher than in relatively undisturbed systems. Urban environments provide anthropogenic pulse, press and ramp disturbances, such as fires, pollutants, and elevated temperatures (Lake 2000). The most straightforward and simple form of ecological restoration removes a single disturbance (for example a pipe draining chemical waste into a pond) and allows time for ecosystem processes to restore function (SERI 2004). Urban areas offer no simple solutions. Ongoing management will be required where urban disturbance exceeds the pace of ecosystem recovery. The challenge of restoring resilient and functional ecosystems to urban environments requires thinking about species and nature that strays far from the model of the pristine.

Fragments of habitat in cities can and must provide basic ecological services that cannot be outsourced. The success of ecological restoration in cities can be improved with greater attention to urban soils, urban propagule dynamics, and urban disturbance regimes. Goals and targets need to encompass both the social and ecological context of urban landscapes. To improve the effectiveness of urban ecological restoration, a long-term view is needed, both toward past history and toward the future.

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APPENDICES

Appendix 1A

Ranked Species Lists

Planted Species

Table A1A.1: Species planted on more than 100 occasions in all restored sites combined,1988-2009, by growth form.

Canopy Tree Species	Total Planting Frequency
Quercus rubra	1016
Quercus montana	543
Liriodendron tulipifera	520
Acer rubrum	474
Betula lenta	458
Quercus alba	445
Liquidambar styraciflua	404
Cornus florida	372
Quercus velutina	287
Quercus palustris	275
Pinus strobus	244
Acer saccharum	242
Carya cordiformis	202
Celtis occidentalis	194
Fraxinus americana	184
Nyssa sylvatica	144
Sassafras albidum	113

Understory Tree/Shrub Species	Total Planting Frequency
Viburnum dentatum	728
Hamamelis virginiana	482
Cornus racemosa	422
Aronia arbutifolia	339
Cornus amomum	328
Ilex verticillata	262
Carpinus caroliniana	120

Shrub Species	Total Planting Frequency
Lindera benzoin	574
Sambucus nigra	496
Clethra alnifolia	431
Viburnum prunifolium	248
Viburnum acerifolium	116

Herbaceous Species	Total Planting Frequency
Eurybia divaricata	478
Ageratina altissima	459
Solidago caesia	403
Symphyotrichum cordifolium	385

Vine Species	Total Planting Frequency
Parthenocissus quinquefolia	148

Trees

Table A1A.2: Tree species of unrestored park sites, ranked by total basal area (cm^2), all sites combined.

Cunningham		Pelham Bay		Van Cortlandt	
Species	Total Basal Area	Species	Total Basal Area	Species	Total Basal Area
Robinia pseudoacacia	962465	*Sassafras albidum*	1386159	*Robinia pseudoacacia*	174604
Prunus serotina	824515	*Prunus serotina*	67334	*Prunus serotina*	64918
Liquidambar styraciflua	180956	*Carya cordiformis*	44115	*Carya cordiformis*	15948
Sassafras albidum	51875	*Morus alba*	41910	*Acer saccharum*	12668
Acer rubrum	41007	*Fraxinus pennsylvanica*	33589	*Juglans nigra*	12668
Malus sp.	15948	*Quercus palustris*	23916	*Sassafras albidum*	8495
Ulmus americana	8012	*Carya ovata*	19731	*Morus alba*	2734
Quercus palustris	6433	*Quercus rubra*	16513	*Liquidambar styraciflua*	1257
Liriodendron tulipifera	6013	*Liquidambar styraciflua*	7854	*Malus sp.*	1134
Populus deltoides	4717	*Liriodendron tulipifera*	5281	*Acer platanoides*	990
Quercus velutina	4072	*Quercus alba*	3473	*Ulmus sp.*	707
Morus alba	3267	*Ulmus sp.*	3267	*Carya ovata*	511
Acer platanoides	3068	*Acer platanoides*	1104	*Ailanthus altissima*	452
Quercus rubra	1662	*Malus sp.*	881	*Tilia americana cf.*	314
Quercus alba	1046	*Fraxinus americana*	881	*Liriodendron tulipifera*	145
Cornus alternifolia	346	*Robinia pseudoacacia*	683	*Fraxinus americana*	113
Carya ovata	269	*Frangula alnus*	189	*Quercus velutina*	13
Cornus florida	201	*Quercus velutina*	165		
Juglans nigra	177	*Acer pseudoplatanus*	165		
Ailanthus altissima	177	*Acer rubrum*	143		
Morus rubra	38	*Juglans nigra*	44		
Sambucus nigra	28	*Ulmus americana cf.*	16		
Lindera benzoin	20	*Fraxinus americana cf.*	10		
Prunus sp.	13	*Ailanthus altissima*	7		

Table A1A.3: Tree species of all restored park sites combined, ranked by total basal area (cm^2).

Inwood		**Pelham Bay**		**Prospect**	
Species	**Total Basal Area**	**Species**	**Total Basal Area**	**Species**	**Total Basal Area**
Liriodendron tulipifera	275068	*Robinia pseudoacacia*	3284234	*Prunus serotina*	481027
Quercus rubra	177952	*Prunus serotina*	853412	*Quercus rubra*	279735
Prunus serotina	121736	*Carya cordiformis*	613893	*Liriodendron tulipifera*	276483
Acer saccharum	84085	*Acer rubrum*	134224	*Quercus rubra*	228683
Robinia pseudoacacia	81383	*Quercus palustris*	100829	*Fraxinus pennsylvanica*	142431
Sassafras albidum	59223	*Malus sp.*	35566	*Ulmus sp.*	130561
Celtis occidentalis	45201	*Fraxinus pennsylvanica*	18821	*Quercus palustris*	116053
Carya cordiformis	24773	*Liriodendron tulipifera*	14420	*Prunus serotina*	82423
Acer platanoides	24661	*Acer pseudoplatanus*	11100	*Liquidambar styraciflua*	48930
Morus alba	20587	*Liquidambar styraciflua*	7528	*Prunus sp. non-native*	26805
Carya alba	18869	*Acer saccharum*	7436	*Fraxinus americana*	12125
Gleditsia triacanthos	11747	*Crataegus monogyna*	5958	*Acer saccharum*	11713
Quercus alba	8742	*Quercus rubra*	2463	*Acer saccharinum*	10751
Malus sp.	5542	*Ulmus sp.*	951	*Acer platanoides*	10029
Prunus avium	4004	*Acer platanoides*	585	*Betula lenta*	9203
Acer rubrum	2091	*Frangula alnus*	254	*Acer pseudoplatanus*	7682
Aesculus hippocastanum	2083	*Quercus bicolor*	214	*Ulmus sp.*	7329
Fraxinus americana	1901	*Fraxinus sp. non-native*	206	*Fraxinus pennsylvanica*	4015
Rhus typhina	1825	*Pinus strobus*	161	*Viburnum sieboldii*	3739
Fraxinus pennsylvanica	1379	*Juglans nigra*	119	*Carya glabra*	3505
Ulmus glabra cf.	1075	*Acer saccharinum*	104	*Acer negundo*	2784
Ulmus sp.	1018	*Crataegus sp.*	71	*Celtis occidentalis*	1987
Ailanthus altissima	731	*Acer negundo*	69	*Morus alba*	1867
Fagus grandifolia	434	*Nyssa sylvatica cf.*	36	*Acer rubrum*	1676
Betula lenta	161	*Viburnum prunifolium cf.*	10	*Liquidambar styraciflua*	1392
Acer saccharinum	133	*Pinus strobus cf.*	7	*Tilia cordata*	1146
Amelanchier sp. arborea/canadensis	123			*Ailanthus altissima*	1029
Platanus occidentalis	87			*Prunus avium/padus cf.*	887
Quercus montana	75			*Ostrya virginiana*	683
Rhamnus cathartica	15			*Amelanchier sp.*	560
Physocarpus opulifolius	7			*Populus deltoides*	320
Tilia americana	7			*Acer platanoides*	320
				Rhamnus cathartica cf.	161
				Prunus avium cf.	156
				Frangula alnus	133
				Liriodendron tulipifera	88
				Robinia pseudoacacia	79
				Betula populifolia	75
				Platanus hybrida	68
				Quercus palustris	35
				Prunus avium	28

Ulmus pumila cf.	24
Quercus alba	23
Acer saccharinum	13
Betula nigra	10
Carya cordiformis	8
Broussonetia papyrifera	7
Malus sp.	6
Cornus florida/kousa cf.	6
Hamamelis virginiana	6
Quercus velutina	1
Cornus sp.	1
Viburnum prunifolium cf.	1
Rhamnus cathartica	1
Nyssa sylvatica	1

Woody Understory

Table A1A.4: Five most abundant species for each growth form in the woody understory of restored and unrestored sites, ranked by total number of stems, site types pooled.

Restored Sites			Unrestored Sites		
Growth Form	**Species**	**Total Stems**	**Growth Form**	**Species**	**Total Stems**
Canopy Tree Saplings			**Canopy Tree Saplings**		
	Fraxinus americana	72		*Sassafras albidum*	44
	Prunus sp.	71		*Prunus serotina*	25
	Carya cordiformis	46		*Carya cordiformis*	9
	Prunus serotina	46		*Ailanthus altissima*	7
	Acer negundo	31		*Acer platanoides*	5
Understory Small Trees / Large Shrubs			**Understory Small Trees / Large Shrubs**		
	Viburnum dentatum	94		*Viburnum dentatum*	106
	Cornus sp.	29		*Malus sp.*	22
	Aronia arbutifolia	11		*Corylus sp.*	11
	Frangula alnus	8		*Rhus glabra*	6
	Cornus racemosa cf.	7		*Cornus sp.*	5
Shrubs			**Shrubs**		
	Lindera benzoin	244		*Rosa multiflora*	999
	Rosa multiflora	157		*Rubus pensilvanicus*	305
	Lonicera maackii	44		*Rubus phoenicolasius*	63
	Rubus pensilvanicus	42		*Rubus occidentalis*	35
	Ligustrum sp.	41		*Lonicera maackii*	31
Vines			**Vines**		
	Toxicodendron radicans	229		*Ampelopsis brevipedunculata*	559
	Lonicera japonica	57		*Lonicera japonica*	308
	Ampelopsis brevipedunculata	45		*Celastrus orbiculatus*	307
	Vitis riparia	33		*Vitis aestivalis*	94
	Celastrus orbiculatus	25		*Vitis labrusca*	80

Table A1A.5: Tree sapling stems in the woody understory of restored and unrestored sites, ranked by number of stems.

Site Type	Species	% of Plots	Total Stems	% of Stems
Restored	Fraxinus americana	47%	72	12.8%
	Prunus serotina	43%	46	8.2%
	Carya cordiformis	37%	46	8.2%
	Quercus rubra	23%	13	2.3%
	Acer platanoides*	20%	10	1.8%
	Acer pseudoplatanus*	20%	20	3.6%
	Acer saccharum	17%	17	3.0%
	Liriodendron tulipifera	17%	6	1.1%
	Viburnum dentatum	17%	94	16.8%
	Acer negundo	13%	31	5.5%
	Acer rubrum	13%	7	1.2%
	Morus alba*	13%	6	1.1%
	Cornus sp.	10%	29	5.2%
	Prunus avium*	10%	4	0.7%
	Quercus alba	10%	5	0.9%
	Quercus montana	10%	5	0.9%
	Sassafras albidum	10%	10	1.8%
	Aronia arbutifolia	7%	11	2.0%
	Celtis occidentalis	7%	7	1.2%
	Fraxinus pennsylvanica	7%	10	1.8%
	Prunus sp.*	7%	71	12.7%
	Acer saccharinum	3%	1	0.2%
	Amelanchier arborea/canadensis cf.	3%	1	0.2%
	Broussonetia papyrifera*	3%	1	0.2%
	Cornus racemosa cf.	3%	7	1.2%
	Corylus americana	3%	5	0.9%
	Frangula alnus*	3%	8	1.4%
	Ilex opaca	3%	1	0.2%
	Nyssa sylvatica	3%	2	0.4%
	Ostrya virginiana	3%	3	0.5%
	Pinus strobus cf.	3%	1	0.2%
	Populus alba*	3%	2	0.4%
	Quercus velutina	3%	1	0.2%
	Rhamnus cathartica*	3%	1	0.2%
	Rhus typhina	3%	1	0.2%
	Ulmus pumila*	3%	4	0.7%
	Zanthoxylum simulans	3%	2	0.4%
Unrestored	Sassafras albidum	37%	44	16.7%
	Viburnum dentatum	33%	106	40.2%
	Malus sp.	27%	22	8.3%
	Carya cordiformis	17%	9	3.4%
	Ailanthus altissima*	13%	7	2.7%
	Acer rubrum	7%	2	0.8%
	Prunus serotina	7%	25	9.5%
	Acer platanoides*	3%	5	1.9%
	Acer saccharum	3%	4	1.5%

Cornus sp.	3%	5	1.9%
Corylus sp.*	3%	11	4.2%
Fraxinus americana	3%	2	0.8%
Fraxinus pennsylvanica	3%	2	0.8%
Humulus sp.	3%	7	2.7%
Ilex crenata*	3%	2	0.8%
Juglans nigra	3%	1	0.4%
Quercus velutina	3%	1	0.4%
Rhus glabra	3%	6	2.3%
Robinia pseudoacacia	3%	1	0.4%
Taxus sp.*	3%	1	0.4%
Viburnum sieboldii*	3%	1	0.4%

Table A1A.6: Species found in the woody understory of restored park sites, ranked by total number of stems.

Inwood		Pelham Bay		Prospect	
Species	**Total Stems**	**Species**	**Total Stems**	**Species**	**Total Stems**
Lindera benzoin	169	*Rosa multiflora*	111	*Toxicodendron radicans*	220
Rosa multiflora	45	*Lonicera japonica*	47	*Prunus sp. non-native*	71
Vitis riparia	33	*Viburnum dentatum*	36	*Lindera benzoin*	63
Lonicera maackii	31	*Ligustrum obtusifolium cf.*	24	*Viburnum dentatum*	58
Cornus sp.	28	*Carya cordiformis*	20	*Fraxinus americana*	51
Carya cordiformis	26	*Acer negundo*	20	*Fraxinus americana*	41
Ampelopsis brevipedunculata	19	*Fraxinus americana*	19	*Forsythia sp.*	30
Rubus pensilvanicus	15	*Ligustrum sp.*	17	*Rubus pensilvanicus*	27
Rubus phoenicolasius	14	*Prunus serotina*	17	*Prunus serotina*	24
Acer saccharum	10	*Ampelopsis brevipedunculata*	16	*Prunus serotina*	21
Lonicera japonica	10	*Lonicera maackii*	13	*Celastrus orbiculatus*	19
Sassafras albidum	10	*Lindera benzoin*	12	*Rubus phoenicolasius*	14
Vitis riparia cf.	7	*Frangula alnus*	8	*Acer pseudoplatanus*	13
Celtis occidentalis	7	*Rubus pensilvanicus/allegheniensis cf.*	8	*Acer negundo*	11
Quercus rubra	7	*Acer pseudoplatanus*	7	*Ampelopsis brevipedunculata*	10
Acer platanoides	6	*Parthenocissus quinquefolia*	7	*Fraxinus pennsylvanica*	10
Prunus serotina	5	*Toxicodendron radicans*	5	*Rhodotypos scandens*	9
Hedera helix	5	*Celastrus orbiculatus*	5	*Cornus racemosa cf.*	7
Quercus montana	5	*Acer rubrum*	4	*Forsythia intermedia cf.*	6
Toxicodendron radicans	4	*Rubus pensilvanicus cf.*	4	*Aronia arbutifolia*	6
Parthenocissus quinquefolia	3	*Sambucus nigra*	3	*Acer saccharum*	6
Liriodendron tulipifera	3	*Rubus phoenicolasius*	3	*Rubus occidentalis*	6
Morus alba	3	*Populus alba*	2	*Acer platanoides*	5
Prunus avium	3	*Acer platanoides*	2	*Aronia arbutifolia cf.*	5
Quercus alba	3	*Liriodendron tulipifera*	1	*Quercus rubra*	5
Fraxinus americana	2	*Ligustrum obtusifolium*	1	*Corylus americana*	5
Rubus occidentalis	2	*Rubus allegheniensis*	1	*Ulmus pumila cf.*	4
Philadelphus sp.	2	*Quercus rubra*	1	*Rhododendron sp. non-native*	4
Celastrus orbiculatus	1	*Acer saccharum*	1	*Morus alba*	3
Rhus typhina	1	*Pinus strobus cf.*	1	*Ostrya virginiana*	3
Ulmus sp.	1	*Cornus sp.*	0	*Vitis sp. non-native*	3
Acer rubrum	1			*Parthenocissus quinquefolia*	3
				Acer platanoides	2
				Prunus sp. non-native	2
				Acer rubrum	2

	Acer pseudoplatanus	2
	Zanthoxylum simulans	2
	Rubus odoratus	2
	Quercus alba	2
	Nyssa sylvatica	2
	Rosa multiflora	2
	Liriodendron tulipifera	2
	Acer saccharinum	1
	Prunus avium cf.	1
	Amelanchier arborea/canadensis cf.	1
	Clethra alnifolia	1
	Lonicera sp.	1
	Rosa multiflora	1
	Broussonetia papyrifera	1
	Prunus avium	1
	Ilex opaca	1
	Quercus velutina	1
	Cornus sp.	1
	Viburnum prunifolium cf.	1
	Rhamnus cathartica	1
	Nyssa sylvatica	1

TableA1A.7: Species in the woody understory of unrestored park sites, ranked by total number of stems.

Cunningham		Pelham Bay		Van Cortlandt	
Species	Total Stems	Species	Total Stems	Species	Total Stems
Rosa multiflora	501	*Rosa multiflora*	282	*Ampelopsis brevipedunculata*	270
Rubus pensilvanicus	156	*Ampelopsis brevipedunculata*	267	*Rosa multiflora*	216
Lonicera japonica	150	*Lonicera japonica*	130	*Rubus pensilvanicus*	131
Vitis aestivalis	75	*Celastrus orbiculatus*	123	*Celastrus orbiculatus*	114
Celastrus orbiculatus	70	*Viburnum dentatum*	46	*Rubus phoenicolasius*	61
Parthenocissus quinquefolia	42	*Vitis labrusca*	32	*Vitis labrusca*	48
Viburnum dentatum	41	*Parthenocissus quinquefolia*	31	*Vitis riparia*	44
Smilax rotundifolia	39	*Sassafras albidum*	29	*Lonicera japonica*	28
Toxicodendron radicans	33	*Lonicera maackii*	29	*Viburnum dentatum*	19
Rubus allegheniensis	27	*Prunus serotina*	24	*Vitis aestivalis*	17
Ampelopsis brevipedunculata	22	*Rubus pensilvanicus*	18	*Rosa sp. non-native*	16
Malus sp.	19	*Toxicodendron radicans*	12	*Rubus occidentalis*	13
Rubus occidentalis	17	*Lonicera morrowii*	8	*Philadelphus coronarius*	10
Sassafras albidum	12	*Smilax rotundifolia*	7	*Humulus sp.*	7
Corylus sp. non-native	11	*Ligustrum obtusifolium*	7	*Acer saccharum*	4
Wisteria sinensis	10	*Carya cordiformis*	6	*Lindera benzoin*	3
Sambucus nigra	9	*Rhus glabra*	6	*Sassafras albidum*	3
Rubus pensilvanicus/ allegheniensis cf.	9	*Rubus occidentalis*	5	*Carya cordiformis*	3
Lindera benzoin	6	*Cornus sp.*	5	*Ailanthus altissima*	3
Viburnum dilatatum	5	*Viburnum prunifolium*	4	*Lonicera maackii*	2
Acer platanoides	5	*Philadelphus tomentosus*	3	*Vitis sp.*	2
Philadelphus coronarius	4	*Lindera benzoin*	2	*Fraxinus americana*	2
Ailanthus altissima	4	*Rubus phoenicolasius*	2	*Malus sp.*	2
Vitis sp.	4	*Fraxinus pennsylvanica*	2	*Toxicodendron radicans*	1
Rhodotypos scandens	3	*Vitis aestivalis*	2		
Ilex crenata	2	*Vitis x novae-angliae cf.*	1		
Viburnum plicatum	2	*Malus sp.*	1		
Acer rubrum	2	*Robinia pseudoacacia*	1		
Lonicera morrowii	1	*Rubus allegheniensis*	1		
Prunus serotina	1	*Euonymus alatus*	1		
Juglans nigra	1	*Azalea sp. non-native*	1		
Taxus sp. non-native	1				
Alliaria petiolata	1				
Viburnum sieboldii	1				
Quercus velutina	1				

Ground Layer

Table A1A.8: Five most abundant species in each category of growth habit in the ground layer of restored and unrestored sites, ranked by total cover (cm).

Unrestored Sites		**Growth Habit**	**Restored Sites**	
Species	**Total cm**		**Species**	**Total cm**
Alliaria petiolata	5995	Herbaceous	*Circaea lutetiana*	4395
Impatiens capensis	4492	Herbaceous	*Impatiens capensis*	2758
Artemisia vulgaris	2819	Herbaceous	*Aegopodium podagraria*	2499
Circaea lutetiana	1775	Herbaceous	*Alliaria petiolata*	2167
Geum canadense	584	Herbaceous	*Hemerocallis fulva*	1097
Ampelopsis brevipedunculata	24214	Vine	*Toxicodendron radicans*	3758
Celastrus orbiculatus	9882	Vine	*Lonicera japonica*	2988
Lonicera japonica	8474	Vine	*Parthenocissus quinquefolia*	2915
Parthenocissus quinquefolia	4787	Vine	*Hedera helix*	1310
Toxicodendron radicans	4305	Vine	*Ampelopsis brevipedunculata*	912
Rosa multiflora	30408	Shrub	*Lindera benzoin*	1771
Rubus pensilvanicus	5406	Shrub	*Rosa multiflora*	1527
Rubus phoenicolasius	1940	Shrub	*Rubus pensilvanicus*	1107
Philadelphus coronarius	672	Shrub	*Rubus phoenicolasius*	872
Lonicera morrowii	551	Shrub	*Rubus allegheniensis*	553
Malus sp.	1385	Shrub/Tree	*Viburnum dentatum*	473
Viburnum dentatum	1026	Shrub/Tree	*Cornus sp.*	184
Cornus sp.	207	Shrub/Tree	*Frangula alnus*	37
Ilex crenata	119	Shrub/Tree	*Malus sp.*	34
Cornus sp. non-native	117	Shrub/Tree	*Crataegus sp.*	6
Sassafras albidum	1190	Tree	*Fraxinus pennsylvanica*	829
Prunus serotina	818	Tree	*Acer pseudoplatanus*	472
Morus alba	211	Tree	*Carya cordiformis*	450
Acer saccharum	195	Tree	*Sassafras albidum*	393
Acer rubrum	188	Tree	*Prunus serotina*	349

Table A1A.9: Seedlings of canopy tree and understory tree/shrub species in the ground layer of restored and unrestored sites, ranked by total cover (cm). Non-native species are denoted with an asterisk.

Site Type	Species	% of Plots	Total Cover (cm)	% of Cover
Restored	Prunus serotina	57%	349	8.40%
	Carya cordiformis	37%	450	10.84%
	Fraxinus pennsylvanica	27%	829	19.96%
	Acer pseudoplatanus*	17%	472	11.37%
	Acer platanoides*	17%	242	5.83%
	Sassafras albidum	10%	393	9.46%
	Acer negundo	10%	170	4.09%
	Acer saccharum	10%	155	3.73%
	Quercus rubra	10%	75	1.81%
	Prunus sp.*	7%	302	7.27%
	Liriodendron tulipifera	7%	104	2.50%
	Quercus alba	7%	59	1.42%
	Acer saccharinum	7%	13	0.31%
	Prunus avium*	3%	259	6.24%
	Rhus typhina	3%	138	3.32%
	Populus alba*	3%	73	1.76%
	Quercus montana	3%	40	0.96%
	Celtis occidentalis	3%	19	0.46%
	Broussonetia papyrifera	3%	10	0.24%
	Cornus florida	3%	1	0.02%
Unrestored	Sassafras albidum	50%	1190	34.73%
	Prunus serotina	37%	818	23.88%
	Carya cordiformis	23%	157	4.58%
	Morus alba	13%	211	6.16%
	Robinia pseudoacacia	13%	66	1.93%
	Juglans nigra	10%	109	3.18%
	Quercus velutina	10%	47	1.37%
	Quercus rubra	10%	43	1.26%
	Acer rubrum	7%	188	5.49%
	Fraxinus americana	7%	75	2.19%
	Liquidambar styraciflua	7%	50	1.46%
	Acer platanoides	7%	41	1.20%
	Ulmus sp.	7%	23	0.67%
	Ailanthus altissima	7%	16	0.47%
	Acer saccharum	3%	195	5.69%
	Prunus sp. non-native	3%	64	1.87%
	Tilia americana cf.	3%	50	1.46%
	Quercus palustris	3%	31	0.90%
	Acer pseudoplatanus	3%	25	0.73%
	Cornus florida	3%	15	0.44%
	Fraxinus pennsylvanica	3%	12	0.35%

Table A1A.10: Ground layer species of restored park sites, ranked by total cover (cm).

Inwood		Pelham Bay		Prospect	
Species	**Total cm**	***Species***	**Total cm**	***Species***	**Total cm**
Toxicodendron radicans	2292	*Lonicera japonica*	2422	*Aegopodium podagraria*	2499
Circaea lutetiana	2076	*Impatiens capensis*	1553	*Circaea lutetiana*	1746
Alliaria petiolata	1410	*Parthenocissus quinquefolia*	1471	*Lindera benzoin*	817
Parthenocissus quinquefolia	1206	*Toxicodendron radicans*	874	*Fraxinus pennsylvanica*	609
Impatiens capensis	1177	*Rosa multiflora*	837	*Toxicodendron radicans*	592
Hedera helix	1092	*Geum canadense*	587	*Rubus pensilvanicus*	390
Lindera benzoin	891	*Circaea lutetiana*	573	*Acer pseudoplatanus*	378
Ampelopsis brevipedunculata	680	*Alliaria petiolata*	486	*Ageratina altissima*	364
Hemerocallis fulva	633	*Hemerocallis fulva*	464	*Phytolacca americana*	361
Rubus phoenicolasius	618	*Rubus pensilvanicus*	404	*Prunus sp. non-native*	302
Lonicera japonica	566	*Duchesnea indica*	363	*Alliaria petiolata*	271
Rosa multiflora	530	*Persicaria virginiana*	357	*Eurybia divaricata*	260
Phytolacca americana	448	*Rubus sp.*	325	*Prunus avium*	259
Sassafras albidum	393	*Celastrus orbiculatus*	313	*Parthenocissus quinquefolia*	238
Rubus pensilvanicus	313	*Rubus pensilvanicus/ allegheniensis cf.*	300	*Hedera helix*	218
Laportea canadensis	289	*Rubus allegheniensis*	289	*Acer platanoides*	208
Rubus allegheniensis	264	*Viburnum dentatum*	250	*Rosa multiflora*	160
Vitis riparia	248	*Prunus serotina*	215	*Celastrus orbiculatus*	141
Amphicarpaea bracteata	217	*Carya cordiformis*	196	*Acer saccharum*	127
Carya cordiformis	214	*Rubus phoenicolasius*	184	*Persicaria maculosa*	125
Lonicera maackii	201	*Fraxinus pennsylvanica*	183	*Persicaria virginiana*	113
Cornus sp.	184	*Rubus pensilvanicus cf.*	155	*Acer negundo*	104
Rubus occidentalis	173	*Ampelopsis brevipedunculata*	143	*Clethra alnifolia*	100
Philadelphus coronarius	162	*Rubus flagellaris cf.*	138	*Ampelopsis brevipedunculata*	89
Viburnum dentatum	161	*Ligustrum obtusifolium cf.*	128	*Rubus occidentalis*	72
Eurybia divaricata	160	*Lonicera maackii*	107	*Prunus serotina*	71
Smilax rotundifolia	153	*Juncus sp.*	101	*Rubus phoenicolasius*	70
Rhus typhina	138	*Acer pseudoplatanus*	94	*Rubus odoratus*	67
Vinca sp.	132	*Ageratina altissima*	76	*Viburnum dentatum*	62
Aster sp. native	132	*Populus alba*	73	*Solanum dulcamara*	61
Oclemena sp.	123	*Persicaria maculosa*	72	*Quercus alba*	59
Hesperis matronalis	115	*Acer negundo*	66	*Viburnum prunifolium cf.*	55
Liriodendron tulipifera	104	*Aster sp.*	65	*Pteridophyta sp.*	47
Persicaria maculosa	95	*Lindera benzoin*	63	*Arctium minus*	47
Vitis riparia cf.	74	*Rubus flagellaris*	63	*Viola sp.*	41
Quercus rubra	73	*Maianthemum racemosum*	57	*Carya cordiformis*	40
Polygonatum biflorum	65	*Ligustrum sp.*	54	*Solidago caesia*	40
Prunus serotina	63	*Onoclea sensibilis*	49	*Malus sp.*	34
Wisteria sinensis	52	*Viola sp.*	40	*Sicyos angulatus*	33
Rubus sp.	48	*Frangula alnus*	37	*Rubus pensilvanicus/ allegheniensis cf.*	33
Echinocystis lobata	47	*Solidago sp.*	36	*Rhododendron sp. non-*	32

				native	
Quercus montana	40	*Gramineae sp.*	20	*Impatiens capensis*	28
Ageratina altissima	39	*Polygonum sp.*	19	*Ambrosia trifida*	22
Fraxinus pennsylvanica	37	*Persicaria longiseta*	14	*Broussonetia papyrifera*	10
Solidago caesia	35	*Carex sp.*	11	*Sanicula canadensis*	10
Acer platanoides	34	*Artemisia vulgaris*	10	*Scrophularia marilandica*	10
Pachysandra terminalis	32	*Geum canadense cf.*	8	*Oxalis stricta*	9
Rubus laciniatus	30	*Acer saccharinum*	7	*Epipactis helleborine*	5
Maianthemum racemosum	29	*Crataegus sp.*	6	*Pilea pumila*	1
Geranium maculatum	28	*Quercus rubra*	2	*Cornus florida*	1
Acer saccharum	28			*Ulmus sp.*	0
Narcissus pseudonarcissus	23				
Oxalis stricta	20				
Celtis occidentalis	19				
Geum canadense	15				
Symphyotrichum cordifolium cf.	9				
Liliaceous sp.	6				
Persicaria longiseta	6				
Sanicula canadensis	6				
Acer saccharinum	6				
Celastrus orbiculatus	5				
Vitis sp.	4				

TableA1A.11: Ground layer species of unrestored park sites, ranked by total cover (cm).

Cunningham		Pelham Bay		Van Cortlandt	
Species	Total cm	Species	Total cm	Species	Total cm
Rosa multiflora	19661	*Ampelopsis brevipedunculata*	10172	*Ampelopsis brevipedunculata*	13616
Lonicera japonica	4726	*Rosa multiflora*	4628	*Celastrus orbiculatus*	6274
Vitis aestivalis	3048	*Lonicera japonica*	3088	*Rosa multiflora*	6119
Rubus pensilvanicus	2838	*Parthenocissus quinquefolia*	2949	*Impatiens capensis*	3146
Alliaria petiolata	1393	*Toxicodendron radicans*	2857	*Artemisia vulgaris*	2319
Malus sp.	1313	*Alliaria petiolata*	2576	*Alliaria petiolata*	2026
Celastrus orbiculatus	1163	*Celastrus orbiculatus*	2445	*Vitis labrusca*	2001
Parthenocissus quinquefolia	1157	*Rubus pensilvanicus*	1226	*Rubus phoenicolasius*	1757
Toxicodendron radicans	1089	*Impatiens capensis*	1195	*Rubus pensilvanicus*	1342
Philadelphus coronarius	672	*Circaea lutetiana*	940	*Vitis riparia*	1260
Rubus pensilvanicus/ allegheniensis cf.	435	*Sassafras albidum*	850	*Circaea lutetiana*	813
Ampelopsis brevipedunculata	426	*Prunus serotina*	515	*Parthenocissus quinquefolia*	681
Viburnum dentatum	378	*Artemisia vulgaris*	500	*Lonicera japonica*	660
Wisteria sinensis	235	*Geum canadense*	444	*Vitis aestivalis*	564
Viburnum prunifolium	215	*Vitis aestivalis*	426	*Toxicodendron radicans*	359
Lindera benzoin	196	*Lonicera morrowii*	418	*Viburnum dentatum*	324
Acer rubrum	188	*Persicaria perfoliata*	402	*Maianthemum racemosum*	291
Rubus occidentalis	175	*Viburnum dentatum*	324	*Eurybia divaricata*	278
Smilax rotundifolia	159	*Aster sp.*	281	*Prunus serotina*	232
Impatiens capensis	151	*Lonicera maackii*	267	*Viburnum dilatatum*	228
Sassafras albidum	145	*Rubus flagellaris cf.*	192	*Sassafras albidum*	195
Sambucus nigra	135	*Rubus phoenicolasius*	183	*Acer saccharum*	195
Ilex crenata	119	*Ligustrum obtusifolium*	156	*Smilax rotundifolia*	194
Rubus allegheniensis	115	*Smilax rotundifolia*	150	*Helianthus sp.*	191
Juglans nigra	106	*Lindera benzoin*	140	*Morus alba*	179
Taxus sp. non-native	93	*Barbarea vulgaris*	134	*Rosa sp. non-native*	166
Rhodotypos scandens	85	*Persicaria maculosa*	130	*Geum canadense*	140
Viburnum opulus s.l.	83	*Duchesnea indica*	114	*Cornus sp.*	134
Carex sp.	74	*Aster sp. native*	113	*Lonicera morrowii*	133
Prunus serotina	71	*Carya cordiformis*	106	*Carex sp.*	109
Deutzia sp. non-native	66	*Dennstaedtia punctilobula*	104	*Lonicera maackii*	105
Corylus sp. non-native	47	*Carex vulpinoidea*	99	*Lysimachia ciliata*	100
Acer platanoides	41	*Viburnum prunifolium*	84	*Cuscuta gronovii*	99
Liquidambar styraciflua	38	*Rubus occidentalis*	71	*Onoclea sensibilis*	98
Maianthemum racemosum	38	*Polygonatum biflorum*	70	*Epilobium ciliatum/coloratum cf.*	98
Quercus velutina	37	*Malus sp.*	69	*Aster sp. native*	87
Quercus palustris	31	*Maianthemum racemosum*	68	*Philadelphus coronarius cf.*	87
Robinia pseudoacacia	25	*Cornus sp.*	64	*Cornus sp. non-native*	76
Liliaceaeous sp. non-	23	*Fraxinus americana*	62	*Cornus amomum cf.*	75

native					
Lonicera maackii	23	*Ageratina altissima*	56	*Phragmites australis*	75
Circaea lutetiana	22	*Phytolacca americana*	51	*Symphyotrichum sp.*	71
Rubus flagellaris	19	*Symphyotrichum sp.*	46	*Humulus sp.*	70
Taraxacum officinale	17	*Cornus sp. non-native*	41	*Persicaria longiseta*	65
Convallaria majalis	16	*Onoclea sensibilis*	40	*Prunus sp. non-native*	64
Frangula alnus	14	*Robinia pseudoacacia*	37	*Eutrochium purpureum*	56
Ailanthus altissima	13	*Carex annectens*	33	*Carya cordiformis*	51
Ilex verticillata	12	*Morus alba*	32	*Pilea pumila*	50
Cornus sp.	9	*Frangula alnus*	32	*Tilia americana cf.*	50
Quercus rubra	5	*Quercus rubra*	27	*Persicaria sagittata*	48
		Hedera helix	27	*Solanum dulcamara*	44
		Oxalis stricta	27	*Oxalis dillenii cf.*	39
		Acer pseudoplatanus	25	*Viburnum sieboldii*	36
		Rubus flagellaris	24	*Lindera benzoin*	35
		Rubus allegheniensis cf.	22	*Juncus tenuis*	32
		Vitis labrusca	19	*Thelypteris noveboracensis*	28
		Poa sp.	17	*Lactuca biennis cf.*	27
		Solanum dulcamara	16	*Humulus lupulus*	25
		Fraxinus pennsylvanica	12	*Ulmus sp.*	23
		Rhodotypos scandens	10	*Humulus japonicus*	20
		Quercus velutina	10	*Liliaceous sp.*	16
		Arisaema triphyllum	6	*Cornus florida*	15
		Persicaria virginiana cf.	5	*Phytolacca americana*	14
		Persicaria longiseta	4	*Apiaceous sp.*	14
		Rubus sp.	4	*Fraxinus americana*	13
		Smilax herbacea	3	*Liquidambar styraciflua*	12
				Quercus rubra	11
				Gramineae sp.	8
				Clematis virginiana cf.	8
				Sicyos angulatus	7
				Smilax herbacea	6
				Robinia pseudoacacia	4
				Solidago sp.	3
				Malus sp.	3
				Rubus occidentalis	3
				Juglans nigra	3
				Oxalis stricta	3
				Viola sp.	3
				Ailanthus altissima	3

Appendix 1B

Data Collection Sheets

Forest Restoration Study Soil Sampling

Date: / /	Personnel:
Park: Van Cortlandt Prospect Pelham Inwood Cunningham	Site # Restored Unrestored
Temperature (ºF):	GPS coordinates:
Current weather:	Cloud cover %: Recent rainfall:

Narrative description of soil sample (e.g., rocky, dry, wet, fill, with detail):

Soil tests to be done in laboratory: pH and Total Organic Matter

Soil Characterization

4 samples collected at 30cm depth, in each of 4 quadrants

Texture by feel:	O depth (cm):	**Litter depths** at each sample collection point
Color:	Litter description:	
Earthworm castings: present absent		
Aspect:	Slope:	

Litter depths at each sample collection point

Sample	1	2	3	4
cm				

Compaction
psi at 3in increments; if compact layer, circle

3				
6				
9				
12				
15				
18				

Notes:

Date:
Comments:

Park or Garden:
Personnel:

Unit or Area:
Site Number:

SITE LOCATION

GPS Coordinates:

Anchor Corner (mark with * on map below): NE NW SE SW

Plot type: Untreated UFEP Permanent Forest Reference

Landmark 1
Species or Description:

DBH (cm) or measurements if not a tree:

Bearing toward anchor corner:

Distance to anchor corner:

Landmark 2
Species or Description:

DBH (cm) or measurements if not a tree:

Bearing toward anchor corner:

Distance to anchor corner:

Landmark 3
Species or Description:

DBH (cm) or measurements if not a tree:

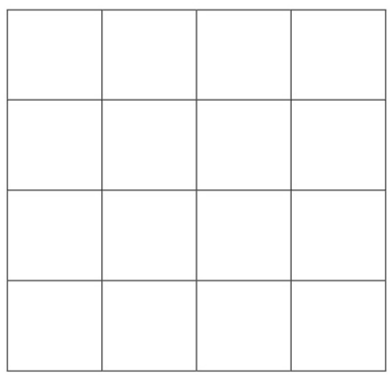
Bearing toward anchor corner:

Distance to anchor corner:

Label: Landmark title/number, azimuth, direction of bearing, any other features, trails/roads, degrees plot skew

North

West



East

South

Narrative Description of Location:

Date: Park or Garden: Unit or Area:
Comments: Personnel: Site Number:

SITE CHARACTERIZATION

Hydrologic Features Estimate percentage cover in 20% increments. Draw a diagonal line if not present.

Feature	Pond	Stream	Perennial	Ephemeral	Flood debris	Gully	Sheet erosion
% Cover	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100

If other, describe and estimate % cover as above:

Comments:

Soil Surface Cover Estimate percentage cover in 20% increments. Draw a diagonal line if not present.

Feature	Leaf litter	Woody debris	Rock outcrop	Bare soil	Built structure
% Cover	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100

If other, describe and estimate % cover as above:

Comments:

Human Impacts Estimate percentage cover in 20% increments. Draw a diagonal line if not present.

garbage	plant damage	built structure	official trail	informal trail	tire tracks	campfire
0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100	0-20 20-40 40-60 60-80 80-100

If other, describe and estimate percentage cover:

Comments:

Topography

Aspect (degrees)	Slope (%)	Surface texture (planar, mound/pit; describe)

Comments:

Animal Activity

If extreme animal herbivory, extreme insect herbivory, or other forest-changing animal impacts are present, describe and estimate percent cover:

Adjacencies Check all that occur within 50m. For roads, note # of lanes.

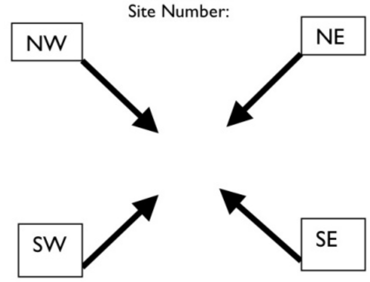
Road	Official trail	Informal trail	Parking lot	Building	Forest	Lawn	Other (describe):

Narrative: describe all adjacent land uses and their azimuth from the plot center:

Date:
Comments:

Park or Garden:
Personnel:

Unit or Area:
Site Number:



HERBACEOUS LAYER AND SEEDLINGS

Extend a line transect 10m from each plot corner toward the plot center. Record the number of centimeters occupied by plants <1m and nonliving features (e.g. leaf litter, rock, bare soil) within each meter along the 10m transect. Meter 0-1 is the meter closest to the center of the plot. When a species is not present in a meter, draw a diagonal line across the box.

Corner (NE, SW, SE, NW)	**Species** (Common or Scientific Name) and non-living features	**Centimeters Occupied in Each 1m Segment**									
		0-1	**1-2**	**2-3**	**3-4**	**4-5**	**5-6**	**6-7**	**7-8**	**8-9**	**9-10**

Date:
Comments:

Park or Garden:
Personnel:

Unit or Area:
Site Number:

North

A	**B**
C	**D**

TREES

All individuals of tree species >1m in height and >2.54 cm DBH present in the 20m x 20m plot area should be recorded below.

A B C D	**Species**	**DBH** (cm)	**Canopy/ Sub-canopy** (C/S)	**A B C D**	**Species**	**DBH** (cm)	**Canopy/ Sub-canopy** (C/S)

Date: Park or Garden: Unit or Area:
Comments: Personnel: Site Number:

1	**A** **B** / **C** **D**		**3**	**4**
5	**6**		**7**	**8**
9	**10**		**11**	**12**
13	**14**		**15**	**16**

SAPLINGS, VINES and SHRUBS

Randomly choose three 5x5m subplots using the numbering schema shown on the right. Subplot #1 is the NW corner of the plot.
Record the number of stems of all **tree saplings** >1m in height <u>and</u> <2.54 cm DBH, all **shrubs** >1m, and all **woody vines** >1m present. Shrub and vine **stems** should be counted where they come out of the ground. Multiple-stemmed trees should be with # of stems.

↓Species # of Stems↘	**Sub-plot:**	**Sub-plot:**	**Sub-plot:**		**↓Species # of Stems↘**	**Sub-plot:**	**Sub-plot:**	**Sub-plot:**

Date: | Park or Garden: | Unit or Area:
Comments: | Personnel: | Site Number:

ADDITIONAL SPECIES
Record all species present in the 20m x 20m plot area that were not recorded via other sampling.

HERBACEOUS SPECIES

Species	**Common/ Uncommon** (C or U)		**Species**	**Common/ Uncommon** (C or U)

SHRUBS, VINES and SAPLINGS

Species	**Common/ Uncommon** (C or U)		**Species**	**Common/ Uncommon** (C or U)

Date: Park or Garden: Unit or Area:
Comments: Personnel: Site Number:

SPECIMENS FOR IDENTIFICATION

Record specimens collected for later identification below. Assign each a letter that refers only to that species in the plot listed above, and include any working title used in the field. If no sample was collected on this date, indicate when/where a sample of the species was previously collected for identification.

Working Title (Genus, possible species identity, or descriptive title)	**Identifying Label** (Letter)	**Sample Collected** (✓) (Note date or plot if previously collected)	**Notes, Comments, Identifying Features**

Appendix 1C

Historic Site Descriptions

Table A1C.1: Historic characteristics based on entitation mapping in restored sites, and species removed. Initial entitation mapping was based on polygons delineated from aerial photography. Subsequent restoration used these maps with other sources of information to identify management concerns. Note that in some records, dominant species are not identified, and invasive species removed were not always reported.

Plot	Map Year	Map Unit	ha	Unit Description	Historic Features of Unit	Uses & Disturbance in Unit	Unit Comments	Abundant Species in Unit	Other Species in Unit	Species Removed from Unit 1998-2009
IN03	1989	IN81	0.4	Closed Forest, Deciduous, Hemicrypto-phytes, Slope, Moist	Well, Stone Foundation	Foot traffic, Fire, Trash, Defecation	Ranger trail runs through unit and leads to large tree stump, unit very diverse, north portion (near 2 large hemlocks) void of groundcover / understory while southeast portion has more diversity and changing topography. Remains of stone foundations. Soil is very moist.	Norway maple, sycamore maple, bitternut hickory	sassafras, white ash, eastern hemlock, black locust, osage orange, asters, poison ivy, white snakeroot, day lily, wild bean, porcelain berry, great ragweed, wild rose, goldenrod, Virginia knotweed, Asiatic dayflower	*Acer platanoides, Acer pseudoplatanus, Ailanthus altissima, Alliaria petiolata, Ampelopsis brevipedunculata, Celastrus orbiculatus, Lonicera maackii, Lonicera tatarica, Morus alba, Phellodendron amurense, Quercus rubra, Rosa multiflora, Rubus phoenicolasius, Vitis sp.*

IN07	1989	IN117	2.5	Closed Forest, Deciduous, Hemicrypto-phytes, Slope, Dry / Moist	Full-crown tree, Pipes	Foot traffic, Erosion, Dumping, Trash	Large forest unit along Dyckman St. extending north between units 53 & 54. Footpaths and erosion throughout. Ash dieback evident. Dense shade resulting in sparse groundcover, and major soil erosion and compaction at Payson Ave.	sycamore maple, ash spp., mulberry	black cherry, red oak, rose of Sharon, dogwood, *Vinca*, garlic mustard, poison ivy, asters, cool season grass, wild grape, porcelain berry, white snakeroot, enchanter's nightshade, Virginia knotweed, Virginia creeper, tulip tree, hackberry, white oak, mock orange, day lily, spicebush, ivy	*Acer platanoides, Acer pseudoplatanus, Ailanthus altissima, Ampelopsis brevipedunculata, Celastrus orbiculatus, Lonicera japonica, Quercus rubra, Rubus phoenicolasius*
IN08	1989	IN81	0.4	Closed Forest, Deciduous, Hemicrypto-phytes, Slope, Moist	Well, Stone Foundati on	Foot traffic, Fire, Trash, Defecation	Ranger trail runs through unit and leads to large tree stump, unit very diverse, north portion (near 2 large hemlocks) void of groundcover / understory while southeast portion has more diversity and changing topography. Remains of stone foundations. Soil is very moist.	Norway maple, sycamore maple, bitternut hickory	sassafras, white ash, eastern hemlock, black locust, osage orange, asters, poison ivy, white snakeroot, day lily, wild bean, porcelain berry, great ragweed, wild rose, goldenrod, Virginia knotweed, Asiatic dayflower	*Acer platanoides, Acer pseudoplatanus, Ailanthus altissima, Alliaria petiolata, Ampelopsis brevipedunculata, Celastrus orbiculatus, Lonicera maackii, Lonicera tatarica, Morus alba, Phellodendron amurense, Quercus rubra, Rosa multiflora, Rubus phoenicolasius, Vitis sp.*

IN11	1989	IN81	0.4	Closed Forest, Deciduous, Hemicrypto-phytes, Slope, Moist	Well, Stone Foundation	Foot traffic, Fire, Trash, Defecation	Ranger trail runs through unit and leads to large tree stump, unit very diverse, north portion (near 2 large hemlocks) void of groundcover / understory while southeast portion has more diversity and changing topography. Remains of stone foundations. Soil is very moist.	Norway maple, sycamore maple, bitternut hickory	sassafras, white ash, eastern hemlock, black locust, osage orange, asters, poison ivy, white snakeroot, day lily, wild bean, porcelain berry, great ragweed, wild rose, goldenrod, Virginia knotweed, Asiatic dayflower	*Acer platanoides, Acer pseudoplatanus, Ailanthus altissima, Alliaria petiolata, Ampelopsis brevipedunculata, Celastrus orbiculatus, Lonicera maackii, Lonicera tatarica, Morus alba, Phellodendron amurense, Quercus rubra, Rosa multiflora, Rubus phoenicolasius, Vitis sp.*
IN12	1989	IN66	0.4	Closed Forest, Deciduous, Geophytes, Slope, Moist	Foundation	Foot traffic , Dead ashes	Unit located east of Henry Hudson Parkway north and peculiar that no one tree species dominates. Wild celery is dominant understory, but some *Polygonum*. Unit appears to be transition. Interesting land features are 2 remnants of wooden structure (possibly former root cellar) and large hollow along footpath.	American elm, white ash, arrowwood, mulberry, sedge (celery)	eastern hemlock, horse chestnut, hickory spp., maple spp., *Ailanthus*, wild celery, polygonum	*Acer platanoides, Ampelopsis brevipedunculata, Lonicera japonica, Lonicera maackii, Morus alba, Robinia pseudoacacia, Rosa multiflora, Fallopia japonica*

IN13	1989	IN104	4.8	Closed Forest, Deciduous, Chamae-phytes, Slope, Moist		Foot traffic, Erosion, Trash, Fire	Unit located in a valley, soil is rich and moist, forest is made of large tulip trees (avg. 30" dbh.) Unit borders Indian caves. Overstory shade and lack of adequate groundcover, therefore much exposed topsoil that washes away into drainage ditches throughout unit.	Tulip tree, Oak, red, Chestnut oak, Red maple, Black birch, Spicebush	Sassafras, bitternut hickory, hackberry, mulberry, black cherry, Norway & sycamore maple, white ash, arrowwood, jewelweed, Virginia creeper & knotweed, Aster, goldenrod, Solomon's plume, Rubus, maple leaf viburnum, garlic mustard.	*Acer platanoides, Ailanthus altissima, Alliaria petiolata, Ampelopsis brevipedunculata, Berberis thunbergii, Celastrus orbiculatus, Juglans cinerea, Liriodendron tulipifera, Lonicera japonica, Lonicera maackii, Lonicera tatarica, Morus alba, Phellodendron amurense, Prunus serotina, Ranunculus ficaria, Rhodotypos scandens, Rosa multiflora, Rubus phoenicolasius, Wisteria sp., Acer pseudoplatanus, Aralia spinosa, Broussonetia papyrifera, Paulownia tomentosa*

IN14	1989	IN117	2.5	Closed Forest, Deciduous, Hemicrypto-phytes, Slope, Dry / Moist	Full-crown tree, Pipes	Foot traffic, Erosion, Dumping, Trash	Large forest unit along Dyckman St. extending north between units 53 & 54. Footpaths and erosion throughout. Ash dieback evident. Dense shade resulting in sparse groundcover, and major soil erosion and compaction at Payson Ave.	sycamore maple, ash spp., mulberry	black cherry, red oak, rose of Sharon, dogwood, vinca, garlic mustard, poison ivy, asters, cool season grass, wild grape, porcelain berry, white snakeroot, enchanter's nightshade, Virginia knotweed, Virginia creeper, tulip tree, hackberry, white oak, mock orange, day lily, spicebush, ivy	*Acer platanoides, Acer pseudoplatanus, Ailanthus altissima, Ampelopsis brevipedunculata, Celastrus orbiculatus, Lonicera japonica, Quercus rubra, Rubus phoenicolasius*
IN15	1989	IN117	2.5	Closed Forest, Deciduous, Hemicrypto-phytes, Slope, Dry / Moist	Full-crown tree, Pipes	Foot traffic, Erosion, Dumping, Trash	Large forest unit along Dyckman St. extending north between units 53 & 54. Footpaths and erosion throughout. Ash dieback evident. Dense shade resulting in sparse groundcover, and major soil erosion and compaction at Payson Ave.	sycamore maple, ash spp., mulberry	black cherry, red oak, rose of Sharon, dogwood, *Vinca*, garlic mustard, poison ivy, asters, cool season grass, wild grape, porcelain berry, white snakeroot, enchanter's nightshade, Virginia knotweed, Virginia creeper, tulip tree, hackberry, white oak, mock orange, day lily, spicebush, ivy	*Acer platanoides, Acer pseudoplatanus, Ailanthus altissima, Ampelopsis brevipedunculata, Celastrus orbiculatus, Lonicera japonica, Quercus rubra, Rubus phoenicolasius*

IN25	1989	IN117	2.5	Closed Forest, Deciduous, Hemicrypto-phytes, Slope, Dry / Moist	Full-crown tree, Pipes	Foot traffic, Erosion, Dumping, Trash	Large forest unit along Dyckman St. extending north between units 53 & 54. Footpaths and erosion throughout. Ash dieback evident. Dense shade resulting in sparse groundcover, and major soil erosion and compaction at Payson Ave.	sycamore maple, ash spp., mulberry	black cherry, red oak, rose of Sharon, dogwood, *Vinca*, garlic mustard, poison ivy, asters, cool season grass, wild grape, porcelain berry, white snakeroot, enchanter's nightshade, Virginia knotweed, Virginia creeper, tulip tree, hackberry, white oak, mock orange, day lily, spicebush, ivy	*Acer platanoides, Acer pseudoplatanus, Ailanthus altissima, Ampelopsis brevipedunculata, Celastrus orbiculatus, Lonicera japonica, Quercus rubra, Rubus phoenicolasius*

IN29	1989	IN61	1.6	Closed Forest, Deciduous, Lianas, Slope, Dry / Moist	None reported	Foot traffic, Voodoo	Large forest unit with great species variety, disturbed, vigor is questionable, many dead ash trees and patches of dense Rubus throughout, possible indication of long-term effect in changing of species composition and forest vigor. Variety of species with the absence of quality native species is alarming and confusing; generally the unit appears "trashy".	white ash, mulberry, American hackberry, poison ivy	red oak, *Ailanthus*, American elm, bitternut hickory, Norway maple, *Rubus*, day lily, aster, Virginia knotweed, garlic mustard, Japanese honeysuckle, pink lady's finger	*Acer platanoides, Acer pseudoplatanus, Ailanthus altissima, Alliaria petiolata, Ampelopsis brevipedunculata, Broussonetia papyrifera, Artemisia vulgaris, Celastrus orbiculatus, Hedera helix, Hibiscus syriacus, Lonicera japonica, Lonicera maackii, Lonicera tatarica, Morus alba, Prunus serotina, Rhodotypos scandens, Robinia pseudoacacia, Rosa multiflora, Rubus phoenicolasius, Smilax rotundifolia, Vinca minor, Wisteria sp.*
PB03	1986	PB582	0.4	Closed Forest, Deciduous, Therophytes , Level, Well drained	None reported	Foot traffic, Fireplace	Mid-aged cherry-hickory-ash stand. Understory dominated by jewelweed; also *Rubus*, privet, rose, poison ivy.	black cherry, bitternut hickory, white ash, sassafras, pin oak, *Ailanthus*	jewelweed, *Rubus*, privet, rose, poison ivy	*No record*

PB06	1986	PB01a	1.2	Terrestrial Herb., Hemicrypto-phytes, Level, Moist	Road	Foot traffic, Pollution	Scrubland dominated by perennial herb geophyte grass about two meters tall; abundant brambles and patches of sumac and scattered trees that are mature from planting.	goldenrod, *Rubus*, chives, perennial herbs	brambles, sumac, Scotch pine, pin oak, goldenrod, jewelweed, *Phragmites*, ash saplings	*Ampelopsis brevipedunculata, Artemisia vulgaris, Celastrus orbiculatus, Fallopia scandens, Frangula alnus, Lonicera japonica, Rosa multiflora, Toxicodendron radicans*
PB07	1986	PB01a	1.2	Terrestrial Herb., Hemicrypto-phytes, Level, Moist	Road	Foot traffic, Pollution	Scrubland dominated by perennial herb geophyte grass about two meters tall; abundant brambles and patches of sumac and scattered trees that are mature from planting.	goldenrod, *Rubus*, chives, perennial herbs	brambles, sumac, Scotch pine, pin oak, goldenrod, jewelweed, *Phragmites*, ash saplings	*Ampelopsis brevipedunculata, Artemisia vulgaris, Celastrus orbiculatus, Fallopia scandens, Frangula alnus, Lonicera japonica, Rosa multiflora, Toxicodendron radicans*
PB08	1986	PB06	1.4	Woodland, Deciduous, Hemicrypto-phytes, Level, Moist	Pasture	Fire	Woodland dominated by black locust, rare pin oak, Scotch pines, and elderberry present. No tree regeneration; tree mortality along old road and storm drain at border. Fires regularly burn understory; fire break would encourage regeneration.	black locust, goldenrod, *Rubus*, chives, perennial herbs, black cherry	black locust, pin oak, Scotch pines, elderberry	*Ampelopsis brevipedunculata, Artemisia vulgaris, Carya cordiformis, Celastrus orbiculatus, Fallopia scandens, Frangula alnus, Lonicera japonica, Phragmites australis, Rosa multiflora, Toxicodendron radicans, Vitis riparia, Vitis sp.*

PB09	1986	PB06	1.4	Woodland, Deciduous, Hemicrypto-phytes, Level, Moist	Pasture	Fire	Woodland dominated by black locust, rare pin oak, scotch pines, and elderberry present. No tree regeneration; tree mortality along old road and storm drain at border. Fires regularly burn understory; fire break would encourage regeneration.	black locust, goldenrod, *Rubus*, chives, perennial herbs, black cherry	black locust, pin oak, Scotch pines, elderberry	*Ampelopsis brevipedunculata, Artemisia vulgaris, Carya cordiformis, Celastrus orbiculatus, Fallopia scandens, Frangula alnus, Lonicera japonica, Phragmites australis, Rosa multiflora, Toxicodendron radicans, Vitis riparia, Vitis sp.*
PB10	1986	PB06	1.4	Woodland, Deciduous, Hemicrypto-phytes, Level, Moist	Pasture	Fire	Woodland dominated by black locust, rare pin oak, scotch pines, and elderberry present. No tree regeneration; tree mortality along old road and storm drain at border. Fires regularly burn understory; fire break would encourage regeneration.	black locust, goldenrod, *Rubus*, chives, perennial herbs, black cherry	black locust, pin oak, Scotch pines, elderberry	*Ampelopsis brevipedunculata, Artemisia vulgaris, Carya cordiformis, Celastrus orbiculatus, Fallopia scandens, Frangula alnus, Lonicera japonica, Phragmites australis, Rosa multiflora, Toxicodendron radicans, Vitis riparia, Vitis sp.*

PB13	1986	PB680	1.0	Terrestrial Herb., Hemicrypto-phytes, Level, Well drained	None reported	Foot traffic, Auto Access, Dumping, Embankment	Large meadow cut by several dirt paths. Area is disturbed; heavy dumping site. Herbaceous vegetation consists mainly of mugwort; meadow is bordered by Ailanthus on southeastern corner. Evidence of embankments and trails created for dirt biking.Two Austrian pines and a black oak interspersed in meadow.	mugwort, wild parsnip, goldenrod, thistle, black oak, Austrian pine	None reported	*Ailanthus altissima, Ampelopsis brevipedunculata, Celastrus orbiculatus, Ligustrum vulgare, Rosa multiflora*
PB93-1	1986	PB576	8.2	Closed Forest, Deciduous, Phanero-phytes, Level, Moist	Drainage ditch	Foot traffic, Trash	Old overgrown, paved footpath. Large diverse / dense unit. Really no order of dominance because of diversity.	red oak, black birch, tulip poplar, black tupelo, box elder, black walnut	dogwood, hickory, elm, Norway maple, black cherry, *Viburnum*, spicebush, Virginia creeper, aster, *Rubus,* false Solomon's seal, poison ivy, ferns, thistle, jewelweed, wild grape.	*Ampelopsis brevipedunculata, Celastrus orbiculatus, Lonicera maackii, Rosa multiflora*
PB93-2	1986	PB620	1.9	Scrub, Deciduous, Phanero-phytes, Level, Moist	None reported	None reported	Scrubland with some trees that was formerly a meadow.	black cherry, buckthorn, green ash, crab apple, pin oak, *Rubus*	hawthorn, sweet gum, red maple, Virginia creeper, Viburnum, goldenrod, cool season grass, Indian hemp, aster, pokeweed	*Acer platanoides, Ampelopsis brevipedunculata, Artemisia vulgaris, Celastrus orbiculatus, Frangula alnus, Lonicera japonica, Lonicera tatarica, Rosa multiflora, Rubus occidentalis*

PB93-3	1986	PB620	1.9	Scrub, Deciduous, Phanero-phytes, Level, Moist	None reported	None reported	Scrubland with some trees that was formerly a meadow.	black cherry, buckthorn, green ash, crab apple, pin oak, *Rubus*	hawthorn, sweet gum, red maple, Virginia creeper, Viburnum, goldenrod, cool season grass, Indian hemp, aster, pokeweed	*Acer platanoides, Ampelopsis brevipedunculata, Artemisia vulgaris, Celastrus orbiculatus, Frangula alnus, Lonicera japonica, Lonicera tatarica, Rosa multiflora, Rubus occidentalis*

Table A1C.2: Historic characteristics based on entitation mapping in invaded sites that were not restored. Initial entitation mapping was based on polygons delineated from aerial photography. Study sites were selected from areas that managers identified as having been invaded but not restored since initial vegetation mapping. Entitation mapping records were cross-referenced with management records to identify sites where targeted invasive species and conditions were present in initial mapping, and where no restoration had occurred. Original descriptions are provided here. Common names were used to identify species, and ranked abundances were not recorded in many locations. Primary targeted species were *R. multiflora* (multiflora rose or "rose"), *A. brevipedunculata* (porcelain berry, in some cases included in "wild grape"), *C. orbiculatus* (oriental bittersweet, "bittersweet"), and *A. platanoides* (Norway maple).

Plot	Year Mapped	Mapping Unit	Unit ha	Unit Description	Unit Historic Features	Unit Uses & Disturbance	Unit Comments	Unit Abundant Species	Unit Other Species
CHP01	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp, *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut

CHP02	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp., *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut
CHP03	1988	CHP243	0.4	Woodland, Deciduous, Lianas, Level		Foot traffic, Dumping, Compaction	Unit has a fairly open floor in most areas and there are patches of arrowwood and red maple. Mostly a black cherry overstory with a few sweet gum scattered in the unit. Also a patch of young black locust. Some black cherry and sassafras regeneration. Some dumping along 210th St. Large patch of black cherry regeneration.	black cherry, red maple, black locust, sweet gum	tulip tree, flowering dogwood, sweet cherry, Japanese honeysuckle, jewelweed, *Polygonum* sp., rose, Virginia creeper, *Rubus*, poison ivy, enchanter's nightshade, lily-of-the-valley, false Solomon's seal, garlic mustard

CHP04	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp, *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut
CHP05	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp, *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut

CHP06	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp, *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut
CHP07	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp, *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut

CHP08	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp, *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut
CHP09	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp, *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut

CHP10	1988	CHP233	7.1	Closed forest	Old stone foundation, large ridge (possibly a landfill), many pits.	Many footpaths	Huge emergent vineland. Trees mainly black cherry, black locust growing more on edge than in interior. Some red maple, oak spp., Ailanthus, Sassafras (on edges) and hickory. Rubus, poison ivy and Japanese honeysuckle also growing vigorously. Herbaceous cover includes dense blanket of enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, etc. Many footpaths. Old stone foundation, a large ridge (possibly a landfill), and many pits.	Virginia creeper	red maple, oak spp, *Ailanthus*, sassafras, hickory, *Rubus*, poison ivy, Japanese honeysuckle, enchanter's nightshade, jewelweed, garlic mustard, Virginia knotweed, maple leaf viburnum, arrowwood, flowering dogwood, American hazelnut
PU01	1986	PB384	0.1	Woodland, Deciduous, Lianas, Slope, Well drained	Road	Auto access, Bridle path, Dumping	Area is predominantly lianas that are growing into the crowns of the trees.	mulberry, ash, tulip poplar, black locust, bittersweet, wild grape	American elm, honeysuckle, rose, *Rubus*, sumac, jewelweed, thistle, goldenrod, mustard, milkweed
PU02	1986	PB388	0.4	Woodland, Deciduous, Lianas, Level, Well drained	House yard	None reported	This unit is directly behind the area in which the brick house was situated. This may account for the large amount of liana growth which limits the ability of the trees to proliferate.	wild grape, Virginia creeper, honeysuckle sp., black cherry, sassafras, bitternut hickory	*Ailanthus*, red oak, ash, crabapple, sycamore, tulip poplar, elderberry, *Rubus*, *Viburnum*, buckthorn

PU03	1986	PB825	0.3	Woodland, Deciduous, Lianas, Level, Well drained	Drain ditch	Foot traffic, Trash	Many trees species with some vines. Area is wet near the drainage ditch along the northern border.	Black cherry, Hickory, Black walnut, Wild grape, Bittersweet, Honeysuckle sp.	ash, pin oak, sweet gum, American elm, red maple, tulip poplar, black tupelo, rose, sumac, spicebush, Virginia creeper, poison ivy, catbrier, *Rubus*, goldenrod, white snakeroot, Virginia knotweed
PU04	1986	PB812	0.2	Woodland, Deciduous, Lianas, Slope, Well drained	None reported	Foot traffic, Trash	Woodland with lianas	sweet gum, black locust, wild grape, black cherry, ash, sassafras	honeysuckle, bittersweet, rose, *Viburnum*
PU05	1986	PB878	0.2	Closed Forest, Deciduous, Lianas, Slope, Excess drained	Fence, Exotic planting, Retaining wall	Foot traffic Dumping Trash	Built up area by Hutchinson River Parkway and Orchard Beach Road. A paved path is present with remains of a fence. Hawthorn and lianas create a layer that is almost impenetrable to light; therefore, there is little ground cover.	hawthorn, black cherry, bittersweet, *Ailanthus*, mulberry, wild grape	violet, goldenrod, white snakeroot, poison ivy, crab apple, shining sumac
PU06	1986	PB473	0.4	Woodland, Deciduous, Lianas, Undulating, Well drained	Landfill	Sports, Foot traffic, Auto Access, Compaction, Dumping, Trash	Woodland with a lot of lianas between road and high marsh. Chunks of concrete piled near road. Area possibly filled in during road construction.	red oak, white oak, black oak, bittersweet, wild grape, *Rubus*	jewelweed, cool season grass

PU07	1986	PB793	0.8	Closed Forest, Deciduous, Lianas, Slope, Well drained	Fence	None reported	Diverse unit along site of old Shore Road within Split Rock Golf Course.	*Ailanthus*, black cherry, ash, bittersweet, black walnut, eastern hemlock	basswood, black locust, American elm, swamp white oak, pin oak, mulberry, American sycamore, hickory, wild grape, white snakeroot, rose, pokeweed, jewelweed
PU08	1986	PB796	0.4	Closed Forest, Deciduous, Phanerophytes, Level, Well drained	Fence	Fire	Unit on northern edge of golf course extending beyond the fence. A great deal of bittersweet.	red oak, black oak, tulip poplar, sassafras, black cherry, black walnut	*Rubus*, Japanese honeysuckle
PU09	1986	PB857b	0.6	Woodland, Deciduous, Lianas, Slope, Well drained	Fence, Stream	Auto access Bridle path	Woodland with an abundance of bittersweet between golf course and Hutchinson River Parkway exit to I-95. Many birds noted.	pin oak, *Ailanthus*, mulberry, sassafras, black cherry, bittersweet	None reported
PU10	1986	PB802	0.1	Woodland, Deciduous, Lianas, Slope, Well drained	None reported	None reported	Unit adjacent to service road in Split Rock golf course.	*Ailanthus*,, black walnut, swamp white oak, wild grape, sassafras, bittersweet	tulip poplar, sweet gum, gray birch, hickory, jewelweed, *Rubus*

VC01	1988	VC415	0.4	Closed Forest, Deciduous, Chamaephytes, Undulating, Wet	Fence, Channel	Foot traffic	Black locust forest with thicket-like understory of rose. Some black willow, a few black cherry, sassafras, American elm, tulip tree and mulberry. Some spots with bittersweet and wild grape. Mixed in with all the rose is Virginia knotweed, garlic mustard, fringed loosestrife, mugwort and avens sp. Water supply sewer cap in unit.	black locust, rose	black willow, black cherry, sassafras, American elm, tulip tree, mulberry, bittersweet, wild grape, Virginia knotweed, garlic mustard, fringed loosestrife, mugwort, avens sp., moonseed, Tartarian honeysuckle, Aster, silky dogwood, jewelweed, goldenrod, arrowwood
VC02	1988	VC196	0.4	Vineland, Deciduous, Lianas, Undulating, Dry	Culvert	Foot traffic	Large bittersweet vineland rose thicket. These plants and wild grape, Japanese honeysuckle and Virginia creeper are covering the ground, shrubs and climbing into the trees (black cherry and sassafras). In most places the vegetation is impenetrable. There are several small paths--may be drug or cruising area?	bittersweet, rose, black cherry, Japanese honeysuckle	wild grape, Japanese honeysuckle, Virginia creeper, sassafras, hickory spp., poison ivy, white ash, white wood aster, American elm, *Rubus*, catbrier, garlic mustard, goldenrod, *Ailanthus*, Virginia knotweed
VC03	1988	VC529a	1.0	Closed Forest, Deciduous, Lianas, Undulating, Dry	Landfill, Exotics	Foot traffic, Bike/ ATV, Vehicle access, Compaction, Trash	Black cherry forest becoming a vineland, particularly on edges near meadow (unit 528). Many of the tree trunks are covered but canopy still relatively free of lianas and there's some regeneration near Van Cortlandt South. Black locust concentrated here.	black cherry, bittersweet, black locust	white pine, Sassafras, Virginia knotweed, poison ivy, rose, hickory spp., Norway maple, white ash, mulberry, mugwort, cool season grass, arbor vitae, chives, honeysuckle sp., pin oak, etc.

VC05	1988	VC525a	1.1	Closed Forest, Deciduous, Hemicryptophytes, Undulating, Dry	Exotics, Full-crown tree, Hydrant	Bike/ ATV, Foot traffic, Picnic, Compaction, Erosion, Dumping	Forest with lots of black cherry 5-30', older black locust (most >30') and planted white pines in the southern half. Lots of soil compaction and erosion from the many bike and foot trails. Groundcover varies, but is predominantly rose, Virginia knotweed, and some jewelweed. Rose concentrated in North half of unit. Some openings in canopy. Large pin oaks and American sycamore may have been planted. Soil compaction problem; could we manage for pines here or is there too much usage by the community?	Black cherry black locust, white pine	rose, Virginia knotweed, jewelweed, pin oak, American sycamore, garlic mustard
VC06	1988	VC662	1.7	Herbaceous, Deciduous, Hemicryptophytes, Undulating, Dry	Foundation, Culvert	Foot traffic, Trash	Large herbaceous community crossing Croton Aqueduct trail of mugwort and jewelweed mixed with Rubus and patches of bittersweet and wild grape. Two foundations and blocked up culverts.	Mugwort, Jewelweed, Rubus, Bittersweet, Walnut, black	wild grape, pin oak, black cherry, white ash, smooth sumac, crab apple, red oak, hickory spp., American elm, mulberry, Norway maple, American sycamore, Eastern cottonwood, bladdernut, maple leaf viburnum, Joe-Pye weed, Sassafras, Ailanthus, fringed loosestrife, clustered snakeroot, false Solomon's seal, Virginiaknotweed, white avens, rose

VC07	1988	VC662	1.7	Herbaceous, Deciduous, Hemicryptophytes, Undulating, Dry	Foundation, Culvert	Foot traffic, Trash	Large herbaceous community crossing Croton Aqueduct trail of mugwort and jewelweed mixed with Rubus and patches of bittersweet and wild grape. Two foundations and blocked up culverts.	Mugwort, Jewelweed, Rubus, Bittersweet, Walnut, black	wild grape, pin oak, black cherry, white ash, smooth sumac, crab apple, red oak, hickory spp., American elm, mulberry, Norway maple, American sycamore, Eastern cottonwood, bladdernut, maple leaf viburnum, Joe-Pye weed, Sassafras, Ailanthus, fringed loosestrife, clustered snakeroot, false Solomon's seal, Virginiaknotweed, white avens, rose
VC08	1988	VC662	1.7	Herbaceous, Deciduous, Hemicryptophytes, Undulating, Dry	Foundation, Culvert	Foot traffic, Trash	Large herbaceous community crossing Croton Aqueduct trail of mugwort and jewelweed mixed with Rubus and patches of bittersweet and wild grape. Two foundations and blocked up culverts.	Mugwort, Jewelweed, Rubus, Bittersweet, Walnut, black	wild grape, pin oak, black cherry, white ash, smooth sumac, crab apple, red oak, hickory spp., American elm, mulberry, Norway maple, American sycamore, Eastern cottonwood, bladdernut, maple leaf viburnum, Joe-Pye weed, Sassafras, Ailanthus, fringed loosestrife, clustered snakeroot, false Solomon's seal, Virginia knotweed, white avens, rose

VC09	1988	VC669	0.0	Vineland, Deciduous, Lianas, Slope, Moist		Trash, Erosion	Fairly open vineland. Wild grape and bittersweet covering canopy of black birch, black cherry, Ailanthus, flowering dogwood, and American basswood. Ground layer has some wild grape and bittersweet along with poison ivy, jewelweed, arrowwood, and rose.	Grape, wild, Bittersweet, Birch, black	black cherry, Ailanthus, flowering dogwood, American basswood, poison ivy, jewelweed, arrowwood, rose, Rubus, sensitive fern, false Solomon's seal, chives.
VC10	1988	VC430	0.5	Closed Forest, Deciduous, Lianas, Slope, Moist	Exotics	Vehicle access, Culvert, Dumping	A red maple and red oak forest with a lot of bittersweet adjacent to the Saw Mill River Parkway. Unit includes several planted flowering dogwood (diseased) along the Parkway border. A lot of trash from Parkway; recreation misuse.	Red maple, Oak, red, Bittersweet	flowering dogwood, silver maple, pin oak, smooth sumac,mulberry, white ash, silky dogwood, buckthorn, arrowwood, wild grape, porcelain berry, deadly nightshade, Virginia creeper, rose, goldenrod, chives, smooth dogwood, Catalpa, Norway maple, great ragweed, mugwort

VC11	1988	VC165	1.2	Closed Forest, Deciduous, Lianas, Undulating, Dry	Foot traffic, Horses	Very viney forest with good mix of pioneering trees. Lots of wineberry, rose, mugwort, and jewelweed in understory. Many open patches with more vines than trees.	Mulberry, Locust, black, Grape, wild, Bittersweet, Eastern cottonwood, Cherry, black	black willow, Ailanthus, Sassafras, American elm, red oak, pin oak, wineberry, rose, mugwort, jewelweed, staghorn sumac, red maple, American sycamore, dogwood sp., enchanter's nightshade, garlic mustard, Joe-Pye weed, Virginia creeper, white avens, cow parsnip, elderberry

Table A1C.3: Historic conditions of restored sites: Prospect Park. Prospect Park was initially mapped using a different system from other parks. This system created management areas based on ecological and use zoning, rather than aerial delineation of vegetation patches as was done in entitation mapping in other study parks.

				1992 Soil Disturbance			1992 Woodland Conditions				
Plot	Mgmt Unit	Woodland in 1865	1992 Cover Type	Desire Lines	Erosion	Bare Soil	Stable	Declining	Degraded	1992 Canopy Gap	1992 Proposed Cover Type
0102	1	0	Woodland	1	0	0	0	0	1	1	Forest Core
RA01	18	1	Woodland	1	1	0	1	0	0	0	Forest Core
RA02	18	1	Woodland	0	1	0	1	0	0	0	Forest Core
1803	18	1	Woodland	1	1	0	0	1	0	0	Forest Core
1702	17	1	Woodland	1	1	0	0	1	0	0	Forest Core
1603	16	1	Woodland	1	1	0	0	0	1	1	Forest Core
1102	11	1	Woodland	1	1	1	0	0	1	0	Forest Core
1106	11	1	Woodland	1	0	1	1	0	0	0	Forest Core
1105	11	1	Woodland	1	0	1	0	1	0	0	Forest Core
1104	11	1	Woodland	1	0	1	0	0	1	0	Forest Core

Table A1C.4: Additional site history, Prospect Park restored sites. Source: Anne Wong, Prospect Park Natural Resources, personal communication, 2012.

Plot	Site History
0102	Erosion control implemented prior to restoration
RA01	Trampled and bare prior to restoration
RA02	Trampled and bare prior to restoration
1803	Erosion control implemented prior to restoration
1702	Erosion control implemented prior to restoration
1603	Former site of Prospect Park Zoo elephant house. A two-story building with a basement was removed, sandy loam soil was brought in, and hill was re-established using heavy machinery, resulting in extreme soil compaction. Original restoration plant palette was xeric woodland, but due to poor drainage wetland plants are now being used.
1102	Former wood chip compost area that had been bulldozed flat; wood chips were dumped over the side of the hill. Compost was removed from the top, and soil added.
1106	Midwood area once had braided carriage paths that don't correspond to current path locations.
1105	Flat area; no erosion control
1104	Flat area; no erosion control

Appendix 1D

Site Location Coordinates

Table 1D.1: Site location coordinates in decimal degrees. For NYC Park locations, coordinates are for the anchor corner of the plot. For NYBG plots, coordinates are for the plot center.

Site Type	Location	Plot	West	North
Restored	Inwood	IN03	-73.92648927830	40.87111951760
		IN07	-73.92560857940	40.86965102230
		IN08	-73.92658110440	40.87176209350
		IN11	-73.92707788190	40.87182751840
		IN12	-73.92947910710	40.86974489010
		IN14	-73.92516213670	40.86898950470
		IN15	-73.92496046990	40.86903195420
		IN25	-73.92895593780	40.86805887730
		IN29	-73.92904694670	40.86875966950
	Pelham Bay	PB03	-73.80971381160	40.86408329920
		PB06	-73.80008861480	40.86044629060
		PB07	-73.79964652380	40.86028590240
		PB08	-73.80037931800	40.86346327410
		PB09	-73.79946922090	40.86343296220
		PB10	-73.79990901800	40.86312274170
		PB13	-73.80160424300	40.87339717910
		PB93-1	-73.81114068570	40.86556103290
		PB93-2	-73.80696703740	40.87240685160
		PB93-3	-73.80714349450	40.87297443300
	Prospect	PR0102	-73.97294500680	40.65880799120
		PR1102	-73.96733824660	40.66530079220
		PR1104	-73.96618134890	40.66290810080
		PR1105	-73.96631920700	40.66307172100
		PR1106	-73.96683374660	40.66490101600
		PR1603	-73.96756837820	40.66454981610
		PR1702	-73.96965560160	40.66370658130
		PR1803	-73.96958501280	40.66239374240
		PRRA01	-73.97140200810	40.66163428890
		PRRA02	-73.97242017850	40.66074312890
Unrestored	Pelham Bay	PU01	-73.81215770530	40.86912358200
		PU02	-73.81273433280	40.86828688100
		PU03	-73.79599509400	40.88125378260
		PU04	-73.79561690510	40.88231084860
		PU05	-73.81896538150	40.87304643130
		PU06	-73.81587981520	40.87190511940
		PU07	-73.81345840890	40.88405458980
		PU08	-73.81363480660	40.88637305690
		PU09	-73.81464073190	40.88543307710
		PU10	-73.81340729250	40.88441045300
	Van Cortlandt	VC01	-73.88651347230	40.90157596890
		VC02	-73.89597635500	40.90833616100
		VC05	-73.88785157620	40.88624568260
		VC05	-73.88821123700	40.88530282940
		VC06	-73.88357289430	40.89942333470
		VC07	-73.88385284090	40.89905252610

		VC08	-73.88415136970	40.89944461160
		VC09	-73.88299245550	40.90063682850
		VC10	-73.88631444150	40.90594015690
		VC11	-73.89385825680	40.90612154330
Less Disturbed	New York Botanical Garden	NYBG104-1	-73.87600000000	40.86300000000
		NYBG108-1	-73.87700000000	40.86000000000
		NYBG119-1	-73.87600000000	40.86200000000
		NYBG120-1	-73.87500000000	40.86200000000
		NYBG123-1	-73.87600000000	40.86500000000
		NYBG135-1	-73.87500000000	40.86400000000
		NYBG136-1	-73.87500000000	40.86300000000
		NYBG136-2	-73.87400000000	40.86300000000
		NYBG136-3	-73.87500000000	40.86300000000
		NYBG154-1	-73.87400000000	40.86400000000

Appendix 2

Ground Layer Outliers

These three plots were excluded from ground layer plant community analysis due to anomalous histories that led to differences in plant composition unrelated to restoration.

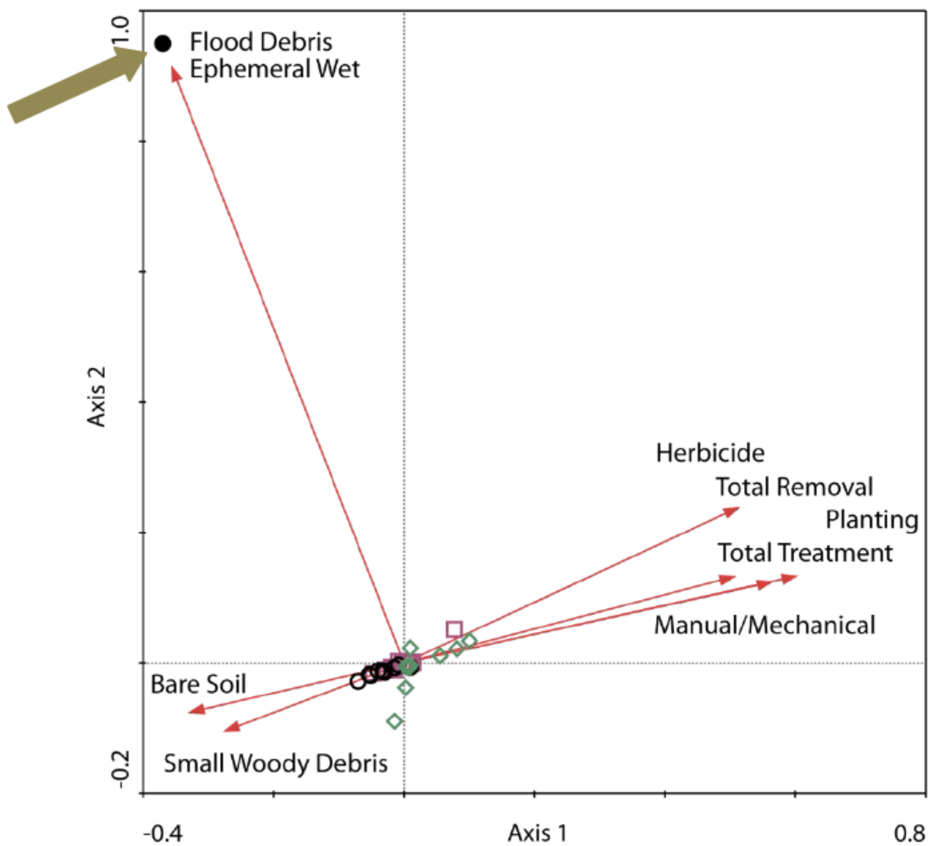


Figure A2.1: Outlier site Pelham Bay 93-3 (blue arrow), a wet site, with the most explanatory vectors. Ground layer plant community composition of restored park sites is arrayed in relationship to site characteristics (→) for Pelham Bay Park (O), Inwood Park (☐) and Prospect Park (◇).

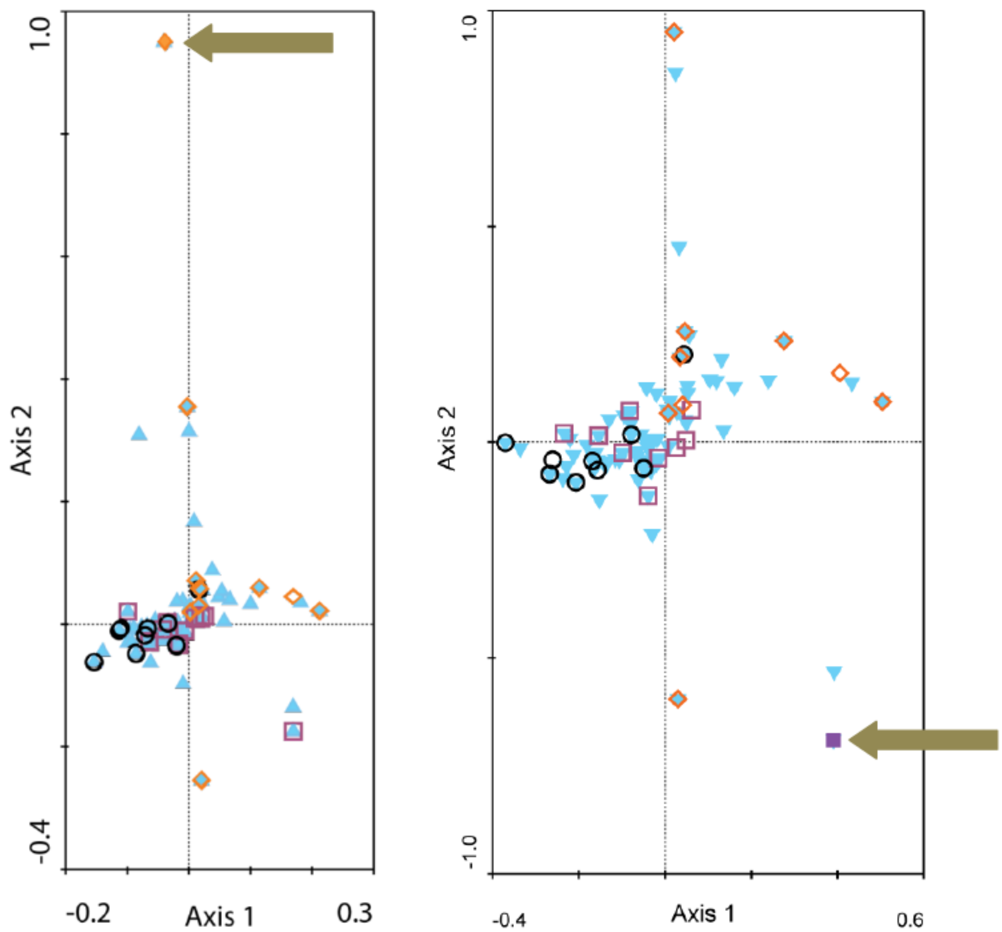


Figure A2.2: Outlier site Prospect 1603 (left), former site of the Prospect Park Zoo Elephant House, is a highly compacted site with little ground layer vegetation. Outlier site Inwood 29 (right), a former estate garden site in which the ground layer is dominated by persistent planted garden species including *Hedera helix*, *Hesperis matronalis*, and *Vinca minor*. Ground layer plant community composition of restored park sites is arrayed in relationship to species (▲) for Pelham Bay Park (O), Inwood Park (□) and Prospect Park (◇).

Appendix 3

Data Quality and Continuity in Long-Term Research

These results rely on data that were collected a minimum of 15 years prior to the start of this investigation. Finding, organizing and evaluating data were the central task of the first years of this study, and I encountered a number of common pitfalls in the handling of long-term data sets.

The best design for long-term study of ecological restoration is to characterize the site fully prior to initiation of restoration activity, fully document all restoration activities, to begin regular data collection immediately following treatment, and maintain data collection at regular intervals.

Record-keeping over a period of twenty years is likely to be variable in any institution. Factors that influence the quantity and quality of data collection over time include personnel turnover, consistency of institutional memory, priority shifts within organizations, and the “accordion effect” of budgets that result in wide staff size variation from year to year. Interpretation of the purpose and goals of data collection may also change over time, leading to differing emphasis on the importance of collecting different kinds of data and the thoroughness with which it is recorded and archived. Changes in data management over time, especially in a period of rapidly changing technology, can lead to data loss when information recorded in one format is not translated into subsequent frameworks for data management.

Data held in the NYC Department of Parks Natural Resources Group treatment databases has been subject to many of these factors, though NRG has had a high level of long employee tenure compared to similar land-management agencies, and personnel who conducted the original restoration work studied here were still involved with the organization at the time of this analysis 15-20 years later. Extant records of prior condition, initial treatment, initial establishment and mortality, and ongoing treatment were incomplete at the city-wide scale. Sites were selected based on completeness of data, but in all cases some records were missing, and site-specific treatment data (herbicide applications, mechanical removal techniques, species planted) were available only at the scale of the management unit, not the individual monitoring plot.

Increasing use of GIS to record management activities means that it will be possible to analyze current and future restoration activities at a finer scale. Useful information resulting from analysis of field-collected data may also increase emphasis on data collection on-site and at the time of treatment, decreasing the error inherent in delayed or forgotten information recording.

Unrestored sites described in these studies are likely to shortly become pre-restoration data sets from which long-term data can be collected. These sites were unrestored due to the pace of action of a small management agency with a limited staff and budget with a large amount of land to manage, and eventually all of them will likely be treated. This represents an important opportunity to understanding the long-term effects of current and future restoration efforts.