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Plant diversity increases in an urban wildland after four decades of unaided vegetation development in a post-industrial site

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Abstract

Spontaneous plant communities found in abandoned post-industrial landscapes develop in unique conditions which can create novel community assemblages. We examined changes in plant community composition and its relation to soil properties in an urban brownfield more than 40 years following site abandonment to better understand the community's long-term trajectory. A former railyard and industrial area built on fill material, the study site includes four primary habitat types: grasslands, perennial forb assemblages, shrubland, and early successional forest. Plant species cover was measured in permanent plots in 2008 and 2016/17. In 2008, soil samples were collected and analyzed for a suite of properties. Species richness and Shannon diversity increased across the site from 2008 to 2016 (40 and 48 years post-abandonment) though increases in these parameters were highest in plots with lower metal concentrations. Evenness changed little in this time period. Percent cover of woody species increased in grassland and forb plots while percent cover of vines increased across all habitat types. Forb species tended to be associated with higher nutrient concentrations while woody species distribution was correlated with higher concentrations of heavy metals. The mitigation of soil stress, limitations in propagule availability, and loss of tree canopy cover following disturbances may have all played a role in influencing increases in diversity in this time period. Considering difficulties in comparing post-industrial abandoned landscapes to other anthropogenic and natural habitats, long-term study is needed to refine our understanding of community assembly in urban brownfields and better guide management practices.

Keywords Brownfields · Anthropogenic soil · Heavy metal pollution · Community assembly · Vacant land

Introduction

Increasing urban land use is a global phenomenon which shows no signs of slowing during the next several decades and has many important implications for biodiversity on both global and local scales (Seto et al. 2012). As cities and their wider metropolitan regions are dynamic systems, over time properties may become abandoned and then colonized by plants with little or no human assistance (Del Tredici 2010). Globalization, economic crises, and changing human population patterns all contribute to shrinking cities, which among other things are characterized by the abandonment of land (Martinez-Fernandez et al. 2012). In some cases, such vacant land may have the potential to become green spaces that support biodiversity and associated ecosystem functions (Robinson and Lundholm 2012; Bonthoux et al. 2014).

Spontaneous urban plant communities can be found on unremediated post-industrial lands – abandoned urban sites often with contaminated soils which are considered derelict from lack of use and called brownfields in the United States. While commonly referred to as unwanted wastelands, such properties can develop into ecosystems that have been called 'new wilderness' (Kowarik 2011), 'informal urban greenspace' (Rupprecht and Byrne 2014) and nature-like 'urban wildlands' (Zhang et al. 2013). Urban wildlands are previously developed or disturbed sites in which plant communities colonize and develop naturally with no or little direct, continued human impact or management. Such post-industrial habitats can harbor biodiversity and provide ecosystem functioning that is comparable to many natural sites (Gallagher et al. 2018).

Brownfields or urban wildlands typically feature altered hydrologic and climate conditions, dispersal limitations, pollution and degraded soil conditions which can all conspire to

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develop novel community assemblages (Hobbs et al. 2009). As cities also tend to be hotspots for invasions, these plant communities may also feature greater numbers of introduced species creating novel communities, a mixture of both old and new world species, that have not coexisted prior to European colonization (Johnson and Handel 2016). Early stages of plant community development on anthropogenic sites such as post-mining or construction waste sites are often characterized by the colonization of ruderal species, though trajectories can be highly variable (Kompala-Baba and Baba 2013; Kopel et al. 2015). Soil properties such as water and nutrient availability, and pH can strongly influence community composition in disturbed, anthropogenic landscapes (Řehounková and Prach 2008; Prach et al. 2014). Community composition can also be directly and indirectly influenced by land use history (Johnson et al. 2015) as well as patch size and the proximity of seed sources (Matthies et al. 2017). Often brownfields and other post-industrial sites are also affected by soil contamination from heavy metals which can create high levels of abiotic stress and can limit diversity (Freedman and Hutchinson 1980; Salemaa et al. 2001).

Given the often unique and stressful combination of conditions found in brownfields and other human-impacted sites, their spontaneous plant communities may exhibit patterns and rates of community assembly that differ from communities experiencing less anthropogenic stress. Long term research is needed to better understand plant community assembly in abandoned and degraded urban sites on the timescale of decades, rather than only during the establishment phase. Ecological legacies and site specific conditions strongly influence ecosystem trajectory and additional knowledge of these patterns and processes can improve restoration and conservation practices in post-disturbance landscapes (Cramer et al. 2008; Christensen 2014).

We examined patterns of plant community composition and soil characteristics in an urban brownfield 40 to 48 years post-abandonment to examine two key questions about plant community trajectories in heavy metal contaminated, anthropogenic environments:

1) How have vascular plant diversity and community composition changed over the course of 8 years?
2) What is the relationship between soil properties and plant community composition?

To address these questions, we assessed plant species cover in 27 plots in an abandoned rail yard and industrial site and then again in the same plots eight years later. We expected to observe little change in diversity metrics during this time period since studies of abandoned agricultural land of similar ages, another type of post-anthropogenic landscape, have observed stable or declining patterns of vascular plant richness (Myster and Pickett 1994). We expected lower richness and diversity in areas of the site with higher metal concentrations assuming stress from the soil conditions would limit the type and number of species which could establish (Freedman and Hutchinson 1980; Salemaa et al. 2001).

Methods

Site description

Liberty State Park (LSP), located in Jersey City, New Jersey, USA (centered at 40.703889° N; 74.053889° W) was created at the site of a former large rail yard and industrial complex which was abandoned in 1969. Historically, the area was an estuary used as fishing grounds by the Lenape, indigenous inhabitants of this region (U.S. Army Corps of Engineers 2005). In the early 1800s, the railroad industry began placing fill material – construction debris, dredge spoils, and other waste materials – to create dry land for improved access to deeper waters in New York Harbor. LSP's subsequent industrial history contaminated its soils with heavy metals at levels which exceed residential and ecological screening criteria (Gallagher et al. 2008; Appendix 1 Table 2). While the outer perimeter (approx. 400 ha) was capped with clean soil and opened to the public, the interior 82 ha of the park (referred to as the LSP Interior) was left largely undisturbed. Within the LSP Interior, heavy metal concentrations ranged widely and were unevenly distributed (Gallagher et al. 2008). LSP's anthropogenic soils are classified as a sandy-skeletal over loamy, mixed, mesic Typic Udorthent (Soil Survey Staff 2010). Since the late 1960s, and earlier in some locations, volunteer plants colonized the LSP Interior, forming a mosaic of habitat types – wetlands, grasslands, shrubland, and early successional forest (Gallagher et al. 2011). Average annual precipitation is approximately 1140 mm, average monthly temperature can range from 3 to 26 °C with extremes ranging from −30 to 40 °C, and the growing season is approximately 174 days (U.S. Army Corps of Engineers 2005). Prior to intense urbanization in this region, upland plant communities in the vicinity of LSP were likely northern hardwood forests with mixed oak and sugar maple (Woods et al. 2007).

Plant community cover

Plant community cover data was collected from 27 permanent plots twice, once in the summer of 2008 and again in the summers of 2016 and 2017 (grouped as the 2016 data set) (Fig. 1). Plot locations were recorded with a handheld GPS unit (Ashtech ProMark2 GPS Survey System: Magellan, San Dimas, California, USA); this unit was then used to relocate plot markers for sampling in 2016. The majority of these plots had been the subject of previous studies and are representative of the LSP Interior's four upland vegetation assemblage types:

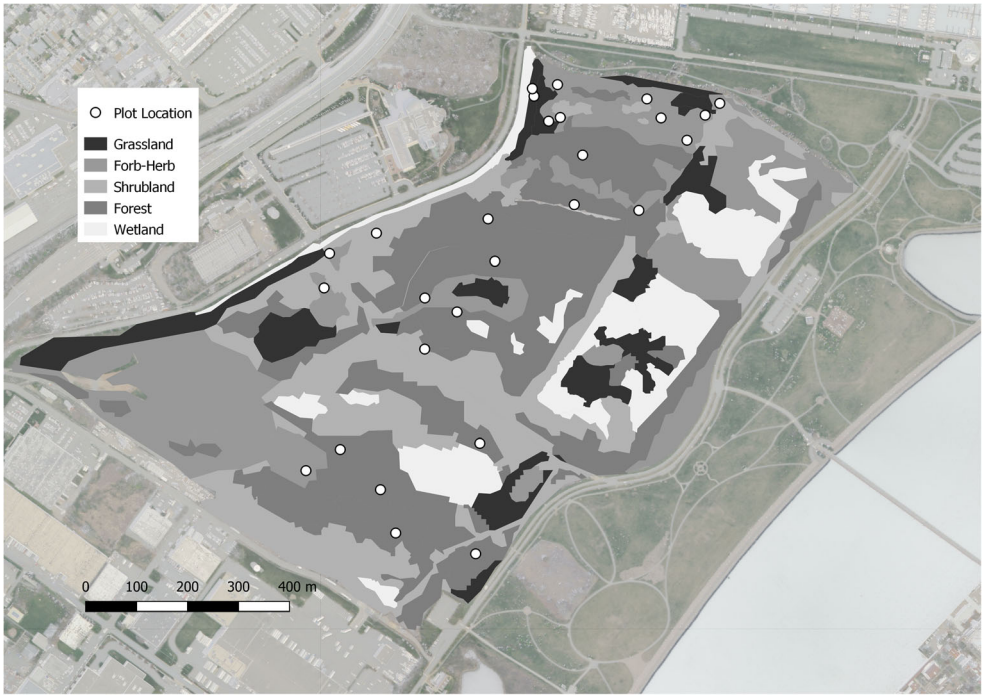

Fig. 1 Distribution of study plots and habitat types in the Liberty State Park Interior based on 2009 vegetation cover (Gallagher et al. 2011)

forest, shrubland, grassland, and forb-herb (Gallagher et al. 2011). Each plot was assigned an assemblage type based on which growth form (e.g. tree, shrub, etc.) covered more than 50% of the plot in 2008. While percent cover of growth forms did change within plots from 2008 to 2016, the 2008 habitat type assignments were maintained for the 2016 plots for the purpose of assessing changes within plots over time. Four of the permanent plots were classified as grassland, nine as forb-herb, four as shrubland, and ten as forest. The plot locations cover a range of heavy metal concentrations. LSP permanent plots are ranked according to their soil total metal load (TML) – a rank sum index of As, Cr, Cu, Pb, and Zn total soil concentrations measured in 2005 that is unique to the site and ranges from 1 to 5 (Gallagher et al. 2008). A higher TML score indicates a greater concentration of heavy metals. While samples used to generate the TML index were collected in 2005, comparison with other soil samples collected in 2015 show that the total metal concentrations in the site's forest plots have remained relatively stable over time (Salisbury et al. 2017).

At every plot, a specific percent cover for each species was visually estimated within a 5 m diameter circle. Both the total combined cover (ground plus canopy cover) and the total relativized percent cover (ground and canopy cover normalized to 100%) were calculated. Species were identified according to Gleason and Cronquist (1991) while plant growth form and native status in the state of New Jersey were based on classifications from the United States Department of Agriculture's Plants Database (USDA and NRCS 2018). Some samples were only identified to genus level (e.g., *Carex, Cyperus, Chamaecyparis, Crataegus, Fraxinus, Rubus* and in some cases *Eupatorium*) as often specimens were not in reproductive stage during the survey.

Soil sampling

One soil sample from each plot was collected from the top 10 cm of soil (approximating the A-horizon) using a hand trowel in the summer of 2008. Samples were air dried and separated by dry sieving into gravel, sand and silt-clay fractions. The A-horizon of LSP's interior soil consists of an unconsolidated, friable fine sandy loam overlying a C-horizon with very high gravel content (25–70% in C, compared to <2% in A; Soil Survey Staff 2010). Consequently, researchers pushed a 2 cm auger into the soil until it could no longer be readily pushed in to estimate the depth of the A-horizon. All chemical analyses were conducted by the UMass Amherst Soil and Plant Nutrient Testing Laboratory (Amherst, MA, USA; NECC 2011). Soil pH was measured in a 1:1 (*v*/v) ratio of soil to water. Soil samples were extracted with a Modified Morgan solution (0.62 *N* NH_4OH + 1.25 *N* CH_3COOH) and analyzed with an ICP-OES (Spectro Ciros, Spectro Analytical Instruments, Kleve, Germany) to determine plant extractable macronutrients (P, K, Ca, Mg), micronutrients (B, Mn, Zn, Cu, Fe, S), Al and heavy metals (Pb, Cd, Ni, Cr). Cation exchange capacity (CEC) was estimated from the base saturation of Ca, Mg, and K. NO_3-N was determined using a 2 *M* KCl extraction and an ion selective electrode. Organic matter (OM) was determined from loss on ignition in a muffle furnace at 450 °C.

Data analysis

Plant community cover data was normalized to 100% and used to determine the total species richness, Shannon Index of diversity (Magurran 1988), and evenness J index (Pielou 1966) for each plot in each year. Percent cover was used as the importance value for calculating the Shannon Index. Combined cover was calculated separately as the sum of all cover values. Combined cover may be greater than 100% when there are multiple layers of vegetation, such as tree canopy and forb cover within a plot. The three diversity metrics and combined cover for each plot in 2008 and 2016 were compared using a paired t-test. The change in percent cover and number of species from 2008 to 2016 within each habitat type were tested using a chi-square test for differences between 1) each growth form category (e.g. tree, shrub, etc.), and 2) native and introduced categories. Since several of the expected values for the chi-square contingency tables were less than 5, a permutation test was used (2000 permutations) to create a distribution for the test statistic (Hope 1968).

Since there was an uneven number of plots representing each habitat type (e.g. forest, shrubland, etc.) and species richness increases with sample size (Colwell and Coddington 1994), species accumulation curves were created for each habitat type in each year using the vegan package's 'specaccum' function in R (Oksanen et al. 2016). This function created the species accumulation curves by calculating the average species richness for each additional plot using a permutation method that adds plots in random order. In order to compare species richness between habitat type, the average estimated richness at four plots was extracted from each curve since four plots was the smallest sample size. A chi-square test was used to compare species richness estimated for a sample size of four plots to test if species richness varied between habitat type and year.

The relationship between total metal load (TML) and each diversity metric was analyzed using linear regression for 1) data within each year, and 2) the difference between the two years (2016 minus 2008). Normality and homogeneity of variance were assessed prior to analysis, no transformations were needed.

Non-metric multi-dimensional scaling (NMDS) was used to visually compare community composition between years. A permutation-based ANOVA (Anderson 2001) was used to test for differences in community composition between sample year and habitat type. Soil properties (with the exception of silt-clay which is complementary to sand and gravel content) were overlaid on the 2008 NMDS chart to visually assess the relationship between soil characteristics and community composition for the year the soil samples were collected. The three soil properties which produced the maximum correlation with the community dissimilarity matrix were determined using the `bioenv` function of vegan (Oksanen et al. 2016). Mantel tests were used to assess if there was a correlation between 2008 community composition and the distribution of soil properties using 1) the entire suite of soil properties, and 2) the three soil properties with the greatest correlation identified by `bioenv`. All analyses were performed in the R Statistics computing environment v. 3.6.2 using the vegan (Oksanen et al. 2016) and ggplot2 (Wickham 2016) packages.

Results

In total 72 taxa (species and genera) were observed over the course of the study period in the 27 plots, with a total of 48 species observed in 2008 and 62 in 2016. Ten species were only identified in 2008 while 24 were unique to 2016 (Appendix 2 Table 3). Eight of the unique 2016 species are forbs while the remaining new species are split between graminoids, shrubs, vines, and trees (4, 5, 3, and 4 unique species, respectively). For the species which were observed in both 2008 and 2016, on average those species were retained in about 40% of the plots where they were observed in 2008. In 2008, *Calamagrostis epigeios* L., *Artemisia vulgaris* L., and *Rhus copallinum* L. were the most frequently observed species (present in 17, 15, and 13 plots respectively). *C. epigeios* and *R. copallinum* were also the most frequently observed species in 2016 (in 19 and 13 plots, respectively), while *Celastrus orbiculatus* Thunb. and *Parthenocissus quinquefolia* L. were tied for the third most frequently observed species (found in 12 plots). *Betula populifolia* Marsh. and *Populus deltoides* Bartr. ex Marsh. were the two tree species with the most cover for the entire set of plots in 2008 (5.2 and 5.1% of total cover in sampled plots, respectively) and 2016 (7.1 and 5.8%). Four tree species were found only in 2016: *Acer rubrum* L., *Crataegus* sp., *Platanus occidentalis* L., and *Quercus rubra* L.. None of the species identified in the study were listed as endangered in New Jersey (NJDEP 2016).

Several soil properties measured in 2008 were quite variable between plots, with Al, Ca, Mg, NO_3-N, Mn, Zn, Fe, Pb, and Ni varying over two orders of magnitude (Table 1). Macronutrient variables (e.g. P, NO_3-N) tended to strongly correlated together as did the metal and micronutrients (e.g. Zn, Fe; Appendix 3 Table 4). A-horizon depth only correlated with Zn and organic matter (OM). Some OM values were unusually high for mineral soils. The presence of coal dust in LSP soils (Soil Survey Staff 2010) likely inflated the OM levels measured by loss on ignition, a phenomenon also observed at other post-industrial sites (Rawlins et al. 2008). Unfortunately measuring recent organic carbon pools in coal contaminated soil is challenging and typically requires utilizing isotopic analysis or a multi-step chemi-thermal process (Ussiri and Lal 2008). The maximum values for plant available Zn and Pb are higher than median concentrations observed in non-urban soils in the region (Sanders 2003).

Table 1 Summary of soil properties from samples collected in 2008. Number of plots = 27

Parameter	Mean	Median	Min	Max	Std. Dev.
pH	5.9	5.6	4.5	7.7	1
CEC[a] (meq/100 g)	18.4	15	3.1	53.4	12.7
OM (%)	13.9	13.7	4.3	23.3	6
NO_3-N (ppm)	5	2	0	25	6
P[b] (ppm)	5	3	2	11	3
Ca (ppm)	2052	1047	144	10,272	2642
K (ppm)	134	97	41	481	99
Mg (ppm)	180	105	18	1060	207
Al (ppm)	50	28	6	186	44
B (ppm)	0.4	0.3	0.1	1.1	0.3
Cd (ppm)	0.2	0.1	0	0.9	0.2
Cr (ppm)	0.1	0	0	1.5	0.3
Cu (ppm)	2.2	1.6	0.4	5.3	1.5
Fe (ppm)	35.8	12.7	1.2	178.9	49.6
Mn (ppm)	4.4	3.4	0.9	17.8	3.3
Ni (ppm)	1	0	0	11	2
Pb (ppm)	17.2	6	1.3	133.2	29.7
S (ppm)	40.4	29.1	11.2	148.8	33.5
Zn (ppm)	57.4	23.1	4.3	546.9	104.3
A horizon depth (cm)	10.8	10	4.6	24.4	4.6
Gravel (%)	16.44	14.38	4.43	38.74	8.56
Coarse sand (%)	13.79	13.3	6.04	32.03	5.61
Medium sand (%)	44.99	46.05	29.13	57.78	7.97
Fine sand (%)	17.64	17.2	9.69	24.83	4.76
Clay silt (%)	7.14	7.42	2.71	12.18	2.61

[a] CEC = Cation exchange capacity, calculated from base saturation of Ca, Mg, and K

[b] All elements are Modified Morgan plant extractable concentrations

Species richness and Shannon diversity significantly increased from 2008 to 2016 across the entire site (Fig. 2-a, 2-b; $t_{richness} = 3.90$, $df = 26$, $p < 0.001$; $t_{diversity} = 4.27$, $df = 26$, $p < 0.001$). The higher richness in 2016 for the entire site is mainly attributed to increases in the number of graminoid and vine species (Fig. 3-a). Combined cover, which reflects multiple layers of vegetation cover (i.e. tree or shrub cover on top of ground cover), was significantly lower in 2016 ($t_{combined} = -2.6$, $df = 26$, $p = 0.01$; Fig. 2-d). Evenness did not significantly change over time, though notably evenness increased over time within the grassland plots but showed no change or a decreasing pattern in other habitat types (Fig. 2-c). From 2008 to 2016, plots with low total metal load (TML; lower concentrations of heavy metals) accumulated more species compared to high TML plots (Fig. 4-b; $r^2 = 0.16$, $p = 0.04$). Within each year, no relationship was observed between TML and the community parameters (data not shown).

Estimated species richness for a sample size of four plots did not vary between habitat type or year ($\chi^2_{habitat\ type\ x\ year} = 2.63$, $p = 0.47$; Fig. 5). In 2016, forest plots gained the most forb species (6) while forb-herb plots lost 3 forb species (Fig. 3-a). Shrubland and forb-herb plots gained the greatest number of new tree species (5 and 4, respectively) during this time period. Proportions of the percent cover of the five growth forms were significantly different between 2008 and 2016 within grassland and forb-herb plots ($\chi^2_{grassland} = 14.3$, $p = 0.003$; $\chi^2_{forb\text{-}herb} = 17.2$, $p = 0.002$) as the percent cover of shrubs and trees increased in 2016 (Fig. 3-b). In the shrubland plots the proportion of cover by the different growth forms was also different between years ($\chi^2_{shrubland} = 38.2$, $p < 0.001$) having shifted to greater vine and tree percent cover in 2016 (Fig. 3-b). The percent cover of native species is highest in the forest plots and lowest in grassland plots both in 2008 and in 2016 (Fig. 6-a). The number and percent cover of native and introduced species did not significantly change from 2008 to 2016 (Fig. 6).

In 2008, forb species tended to be clustered together in NMDS space and associated with nutrient related soil parameters (e.g. P, NO_3-N, cation exchange capacity) while tree and shrub species were more strongly related to A-horizon depth and metal or micronutrient variables (Fig. 7-a and b). There was a weak correlation between the differences of plot

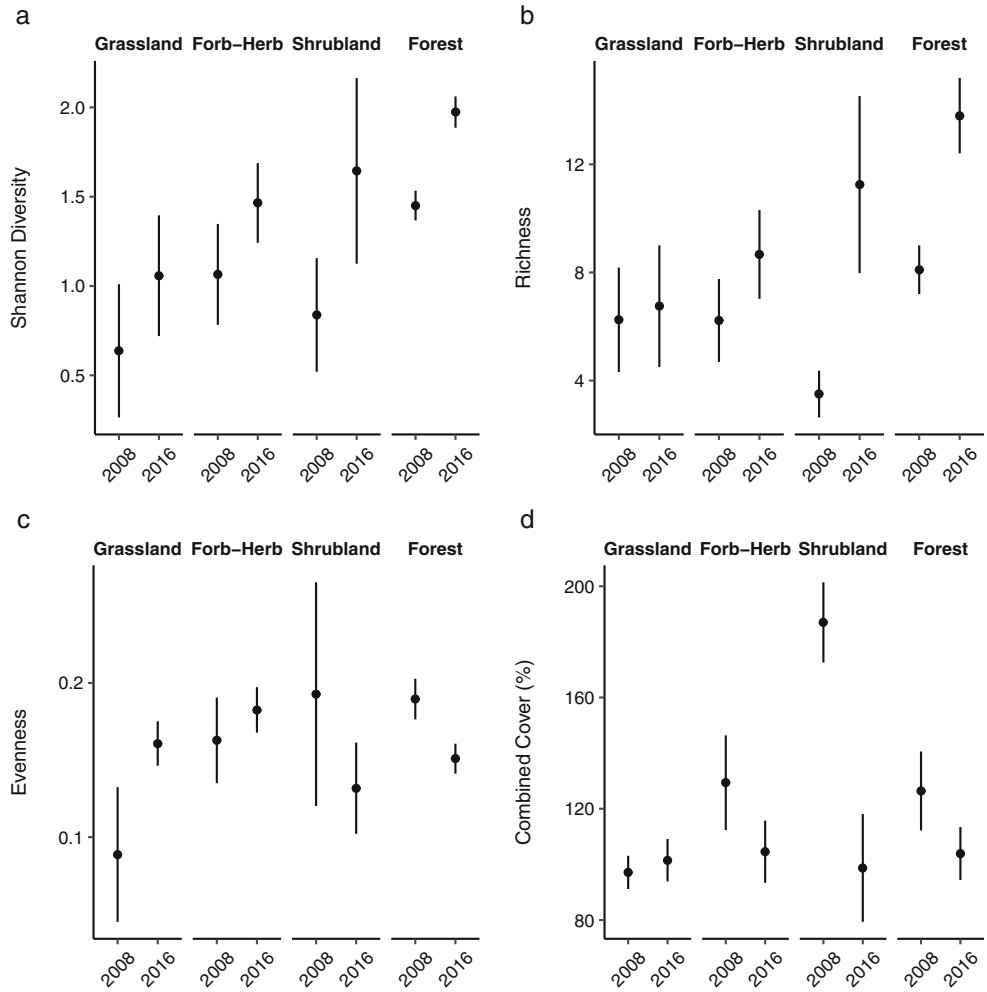

Fig. 2 Change in mean Shannon diversity index (**a**), species richness (**b**), evenness (**c**), and total combined cover (**d**) from 2008 to 2016 in grassland, forb-herb, shrubland, and forest plots ($n = 4$, 9, 4, and 10, respectively). Error bars indicate one standard error. Habitat type assignment was based on dominant vegetation of plots in 2008

community composition and soil properties of each plot in 2008 (Mantel $r = 0.19$, $p = 0.04$). Out of the 26 soil parameters, A-horizon depth, plant available Mg and Ni produced the closest correlation with 2008 community composition (Mantel $r = 0.30$, $p = 0.003$). In 2016 forbs were more broadly distributed in NMDS space, overlapping more with other species (Fig. 7-c). This reflects changes in the percent cover of shrubs and trees in plots which had been primarily grasses and forbs in 2008 (Fig. 3). In both 2008 and 2016, community composition was distinct among the different habitat types (e.g. grassland versus shrubland) according to the permutation-based ANOVA ($R^2_{2008} = 0.26$, $p = 0.002$; $R^2_{2016} = 0.24$, $p = 0.002$). However, overall composition and composition within each habitat type did not significantly change from 2008 to 2016.

Discussion

There have been at least seven surveys of the flora within the land that is now considered Liberty State Park over the past 100 years. In 1878, Brown attributed significant colonization of filled lands to the ballast of ships coming from Europe (Brown 1878). By the middle of the twentieth century however, the commerce of the site had transitioned to mainly commuters between New York City and Jersey City. The source of propagule migration at this time would have shifted from global to regional. Shortly after the closure of the railroad in 1969, a study characterized the vegetation of the site as dominated by early successional herbaceous species, many of which were non-native (Texas Instruments 1976). For example, at drier sites plant heights were described as "uniformly low and there was no dense, rank vegetative growth." Identified species included *A. vulgaris*, which had the greatest importance value as it occurred at every sampling site. *Phragmites australis* Cav. occurred in dense stands in the wetter sites and had the second greatest importance value. *Ambrosia artemisiifolia* L., and *Panicum sp*. were also frequently encountered. Since these early studies, the site has remained fallow with a growing mixture of both native and non-native plant species.

Shannon diversity and species richness continued to increase in the unmanaged interior of Liberty State Park

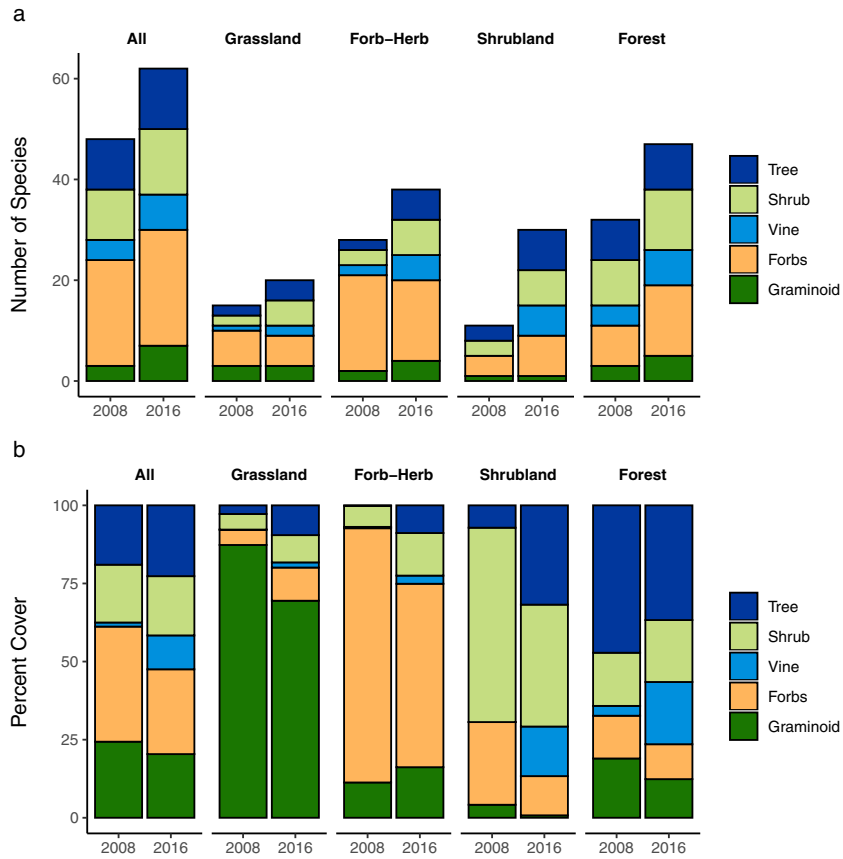

Fig. 3 The total number (**a**) and percent cover (**b**) of graminoid, forb, vine, shrub, and tree species in 2008 and 2016 in the entire site (27 plots) and within each habitat type (4, 9, 4, and 10 plots, respectively). Habitat type assignment was based on dominant vegetation of plots in 2008

(LSP) 40 years after abandonment. The addition of new species to the site and the increasing percent cover of trees and shrubs in plots previously dominated by grasses and forbs along with the increasing percent cover of vines in all plots, suggest that from 40 to 48 years post-abandonment this plant community has been in a state of transition rather than in a stable state. There are several factors which may explain the continued accumulation of species at the site: loss of canopy cover, mitigation of abiotic stress, and propagule availability.

The decrease in combined cover from 2008 to 2016 and the decrease in tree percent cover in forest plots corresponds with observations of reductions in leaf area index (LAI) in forest plots from 2013 to 2016 (Salisbury 2017). The decrease in LAI following 2013 corresponds to a series of severe weather events which included a storm surge following Hurricane Sandy in October 2012 (Salisbury 2017). Loss of tree canopy cover could also account for the increase in the number of vine species and their percent cover observed during this time period by the creation of forest gaps and edges. Edge effects created by forest fragmentation and disturbance have been attributed to observations of increased woody vine (liana) abundance observed in other hardwood forests in the Mid-Atlantic region (Matthews et al. 2016; Ward et al. 2020). Additionally loss of tree cover may reflect the accumulation of long term stress from pollutants experienced by the site's trees, which may lessen their lifespans (Dickinson et al. 1991) or increase their susceptibility to pathogens. Notably *B. populifolia*, a dominant tree species at the site is susceptible to birch leaf miner (*Fenusa pusilla*). The creation of canopy gaps could facilitate the increasing number of grass, forb, and vine species observed in the forest and shrub plots and the increase in species richness over time (Schumann et al. 2003). Additionally, since many of the plots were located near the edges of 2009 habitat types (Fig. 1), changing composition within plot likely also reflects expansion and contraction of different habitat types within the site.

Mitigation of soil stress through the accumulation of organic matter over time may have enabled the addition of new species at the site as a whole and within each habitat type. It is assumed that following abandonment of rail and industrial operations, the soil of the LSP interior was generally low in organic matter in addition to having elevated heavy metal concentrations. Such conditions could have created abiotic constraints on community trajectory similar to limitations observed at some primary succession sites (Anderson 2007). The accumulation of soil organic matter likely improved soil conditions through increasing water holding capacity and nutrient availability. Total concentrations of Cu, Pb, and Zn in

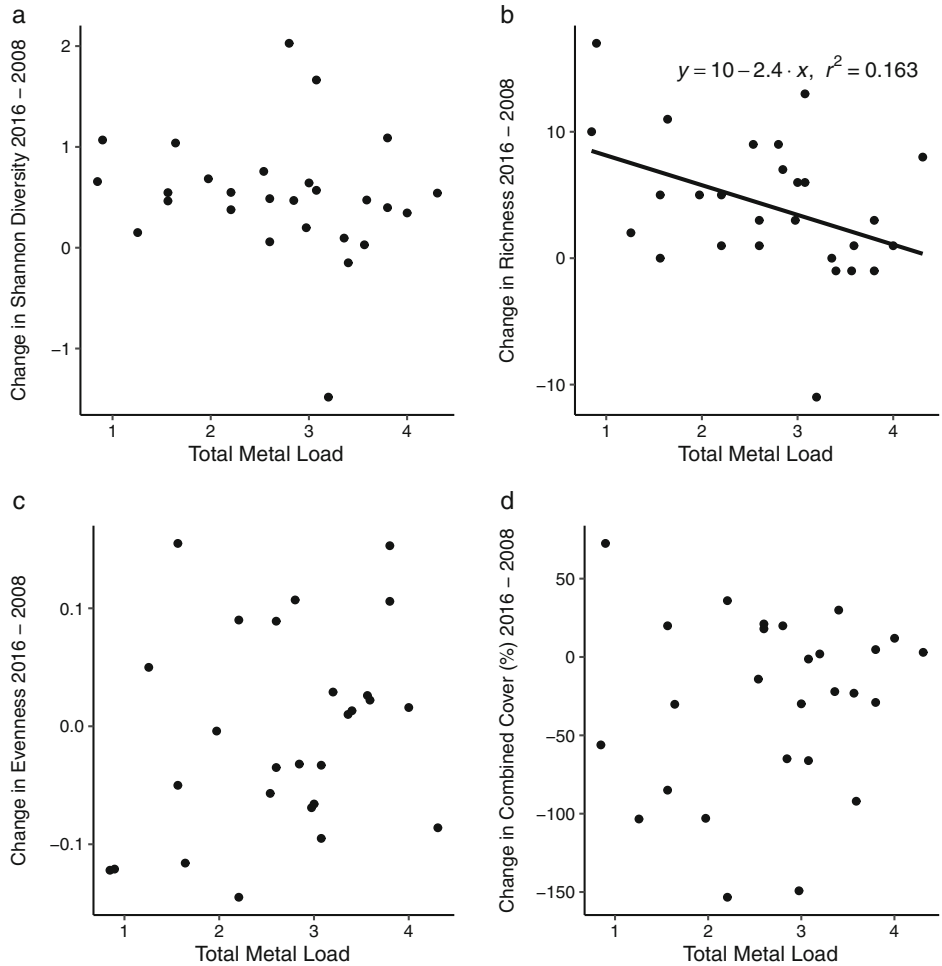

Fig. 4 Change in Shannon diversity (**a**), species richness (**b**), evenness (**c**), and combined cover (**d**) from 2016 to 2008 as a function of soil total metal load. (A higher total metal load indicates relatively higher concentrations of As, Cr, Cu, Pb, and Zn)

forest soils at LSP have remained fairly stable between 1995 and 2015 (Salisbury et al. 2017), but the addition of organic matter in the soil may have reduced heavy metal bioavailability and mitigated its role as a stressor (Brown et al. 2000; Ruttens et al. 2006; Bolan et al. 2014). Additionally facilitative relationships such as those between plants and mycorrhizae may also be particularly important in such metal contaminated environments (Krumins et al. 2015). While these relationships are not well understood, a survey of mycorrhizal communities between low and high total metal load (TML) sites at LSP indicated a significant difference in mycorrhizal community composition and plant productivity (Evans et al. 2015).

Previous work at LSP found that at a coarse scale the spatial distribution of forest assemblages followed the distribution of high TML areas within the site (Gallagher et al. 2008), similar to the associations between soils with higher metal concentrations and woody species observed in this study's 2008 datasets. Disturbed habitats in both post-agriculture and post-industrial landscapes with generally more fertile soils tend to support more ruderal plant communities while trees tend to establish on less fertile sites (Prach and Pysek 2001). Tree colonization has also been associated with sandy (Rebele 1992) and mesic (Prach et al. 2014) soil conditions in anthropogenic or disturbed habitats. Soil organic matter accumulation rates can vary among plant and community type and it is also possible that the correlation between the herbaceous plant communities and nutrients reflects greater organic matter accumulation (Post and Kwon 2000; Hernández et al. 2013). Taken together, these studies suggest the colonization of woody species on low fertility sites may be a common pattern in post-industrial environments, at least in temperate climates.

Propagule availability was likely limited at LSP since it is situated between dense urban development and Upper New York Bay, influencing its rate of species accumulation and overall trajectory (Anderson 2007; Cramer et al. 2008). Since LSP's soil is entirely anthropogenic fill and the site was never forested, there was presumably a very limited seed bank to provide source material. The

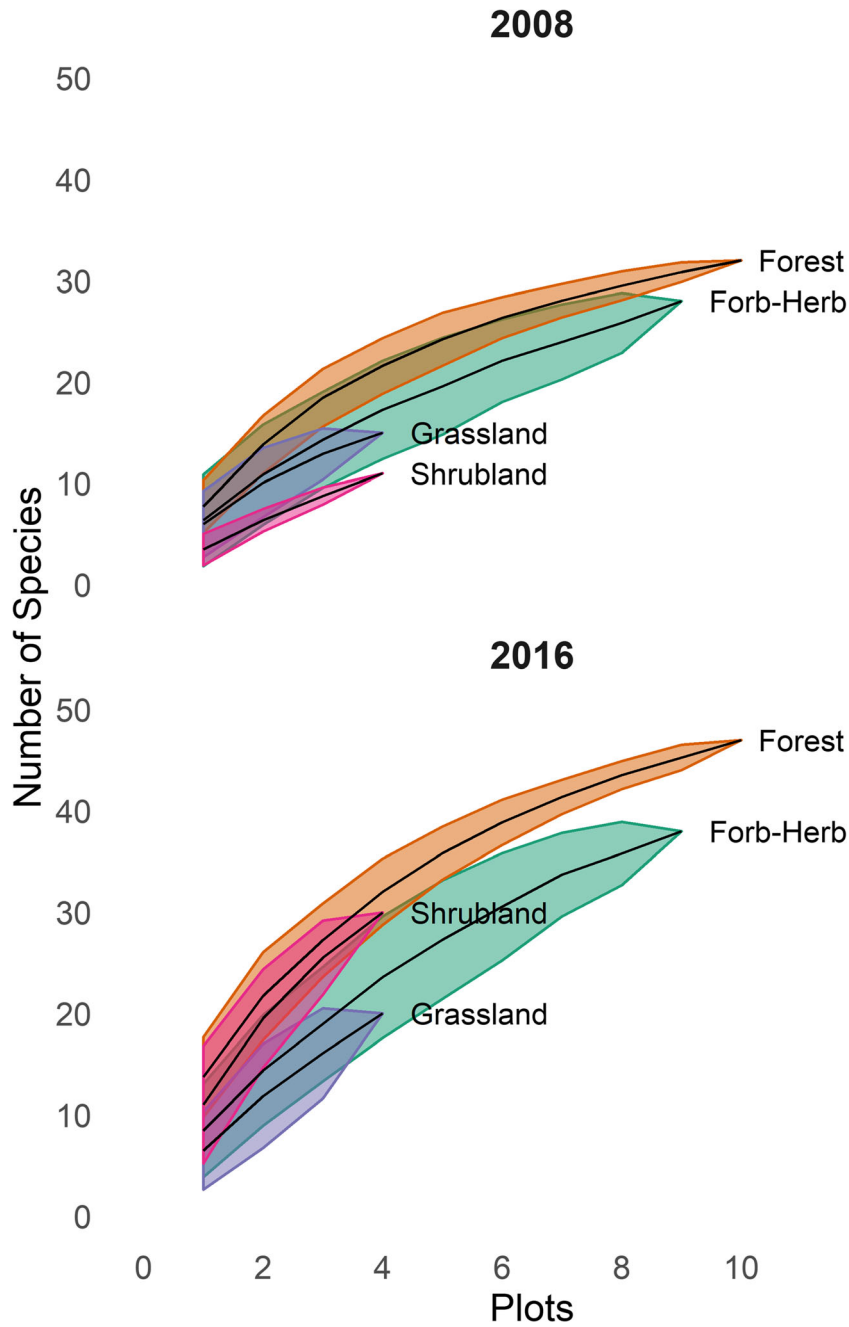

Fig. 5 Species accumulation curves for each habitat type (forest, forb-herb, grassland, shrubland) and year based on the number of sampled plots. The highlighted area around each curve represents one standard deviation about the mean number of species

development of landscaping for Liberty State Park's outer 243 ha waterfront and historic site may have been an important propagule source. Other potential sources of propagules include marginal ruderal areas along transportation corridors and managed greenspaces (parks, cemeteries, golf courses) within a 3 km radius of the park. Patches of hardwood forests which could be propagule sources are located farther away in a 9 to 13 km radius north and west the park. Seed dispersal can be an important driver of plant community assembly in urban environments, which are often highly fragmented (Bonthoux et al. 2014; Johnson et al. 2018). Urbanization tends to favor plant species with seeds with high dispersal capability, such as seeds dispersed by wind or by animals (Concepción et al. 2015). Further research is needed to understand the relative importance of canopy development and subsequent losses, mitigation of soil stress, and propagule availability on the trajectory of the LSP plant community.

It is difficult to assess whether the timeframe for community assembly at this site has been occurring at a relatively slow or rapid pace given the anthropogenic nature of this site. Additionally, since post-industrial landscapes are relatively new and therefore lack an "ecological

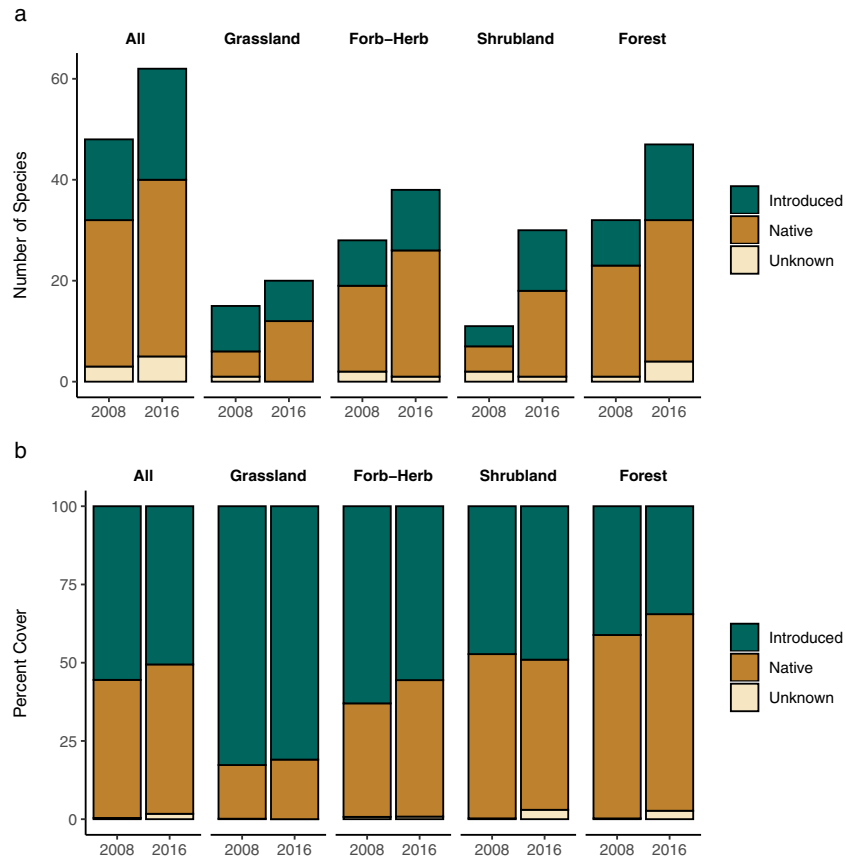

Fig. 6 The total number (**a**) and percent cover (**b**) of native and introduced species in 2008 and 2016 in the entire site (27 plots) and within each habitat type (4, 9, 4, and 10 plots, respectively). Habitat type assignment based on dominant vegetation of plots in 2008

reference," it is unclear what stable states or climax assemblages in an environment made entirely of coarse fill and contaminated with heavy metals should look like (Gallagher et al. 2011). In temperate climates, succession in abandoned agricultural landscapes (also referred to as old fields), which are another example of post-anthropogenic landscapes, can occur fairly quickly, with species richness reaching a plateau by 20 years in some cases (Myster and Pickett 1994). Though these succession rates are strongly influenced by land use history and severity of disturbance caused by previous use (Cramer et al. 2008). In comparison to such landscapes, changes in plant diversity observed around years 40 to 48 at Liberty State Park could be considered fairly slow for sites undergoing secondary succession. Granted, this result may not be surprising considering the large differences in initial site conditions such as soil properties, site history, and geographic context between the study site and most old fields. Perhaps a more appropriate analog for comparison are primary succession environments such as in volcanic landscapes and glacial forelands. Earlier work demonstrated that in 1969, 80% of the high metal load areas of site and 60% of the lower metal load areas were barren land (Gallagher et al. 2011). Primary succession environments are characterized by a lack of topsoil and long distances from seed sources (Jones and Del Moral 2005; Del Moral et al. 2010). These conditions at primary succession sites may result in species richness taking anywhere from decades to centuries to reach a peak, though this timeline can be highly variable and depend on interacting limitations from competition, dispersal and abiotic factors (Anderson 2007). In this context, Liberty State Park's changes in diversity appear to have proceeded rather rapidly. Clearly, additional long-term research in abandoned urban landscapes is needed to improve our understanding and expectations of changes in plant community composition over time in these environments.

This current transition phase of the Interior's plant community could produce several possible outcomes which range from a hardwood forest characterized by shade tolerant species typical of the northeastern U.S. or an alternate stable state(s) dominated by early successional species or a combination of both early and late successional species. Previous research at LSP posited that the site's history, anthropogenic soil and isolation from other natural areas, would favor long-term maintenance of the site's early successional community composition (Gallagher et al. 2011). In the seven vegetative surveys dating from 1878 through 2005 mentioned earlier, there was only one mention of an individual that could be considered a climax species. That

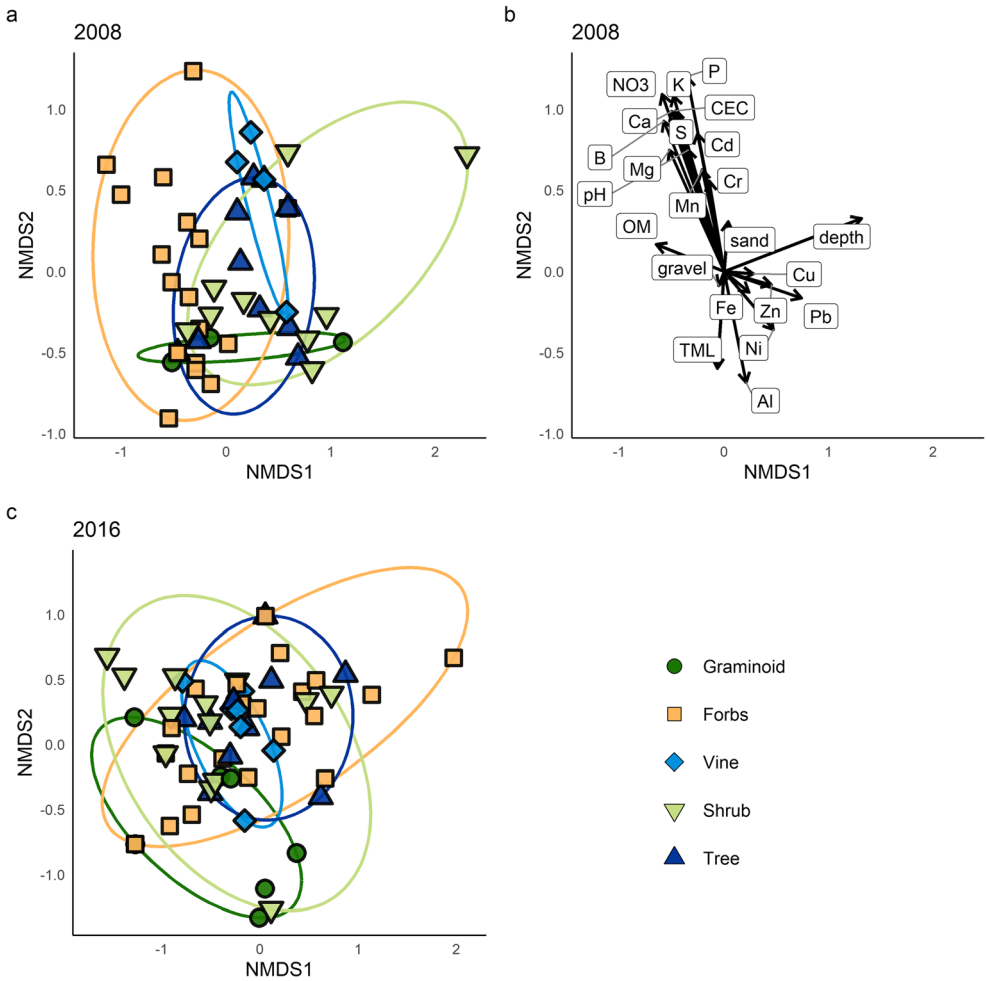

Fig. 7 Non-metric multidimensional scaling analysis of plant community composition in 2008 (**a**) and 2016 (**c**). Each point on graphs a and c correspond to a species, symbols and ellipses correspond to plant growth form. Soil properties (**b**) measured in 2008 were fit onto the 2008 NMDS graph

was an individual *Q. rubra,* that due to its location was probably planted. The four tree species which were only identified in the 2016 survey (*Q. rubra, Acer rubrum* L., *Platanus occidentalis* L., and *Crataegus* sp.; Appendix B), are known to be flexible in their successional status and shade tolerance, found in both disturbed sites and more mature forests (Tirmenstein 1991a, b; Sullivan 1994). It is unclear if the presence of these species indicates a transition to a different forest type.

Findings from this research highlight important management considerations for urban brownfields: should they be maintained in early successional states, how can they contribute to regional urban biodiversity, are these habitats actually ecological traps, and what is the role of non-native species? The long-term maintenance of an early successional ecosystem may be beneficial as this habitat type is in decline through the northeastern U.S. (Brooks 2003) and supports many early successional obligate species (King and Schlossberg 2014). If other urban brownfields and wastelands also follow a similar slow trajectory from herbaceous to woody habitats, then they may play an important role in offsetting the regional declines in early successional habitats.

Since the heterogeneity of soil conditions within the LSP Interior supported several different habitat types, it also facilitated greater plant diversity. Other research has also found that urban wildlands and vacant, spontaneously vegetated urban properties are capable of contributing to the biodiversity of the larger urban region (Tropek et al. 2013; Bonthoux et al. 2014; Matthies et al. 2017). However, the higher levels of pollution and increased proximity to human activity may mean such sites are actually ecological traps – habitat which appears valuable to organisms but actually reduces reproductive success (Leston and Rodewald 2006). Interestingly early work on the reproductive success of *Troglodytes aedon* (house wren) at LSP demonstrated no significant difference in physical metrics or clutch size when compared to an offsite control (Hofer et al. 2010). More research is needed to understand the potential benefits and costs of such sites to both plants and animals. While rare or threatened plant species have been observed in other anthropogenic and disturbed

habitats (e.g. Helldin et al. 2015), no regionally endangered plant species were observed in the two survey years. However, the site has been known to contain *Juncus torreyi* Coville (Torrey's Rush), an endangered plant species within the state of N.J. (Anderson 1989).

The disturbed and novel conditions found at sites such as the LSP Interior can favor the establishment of non-native species (Cramer et al. 2008). While study plots contained a higher number of native species compared to introduced, notably the percent covers of native and introduced species were approximately equal. This pattern of greater native species richness is similar to results from a survey of northern New Jersey which found that in areas with more than 50% urban land cover, non-native woody plant species accounted for approximately 20–40% of total species richness (Aronson et al. 2015). Since the percent cover of native species is highest in forest plots, it is possible that if forest cover increases across the site native species cover will increase as well. However, Johnson and Handel (2016) observed that restoration activities such as invasive species removal were necessary to increase native species abundance in urban forest fragments. While in traditional restoration approaches the removal of non-native species may be desirable, it is worth considering the role and value of such species in anthropogenic habitats which have been severely altered, creating novel environments and novel assemblages (Kowarik 2011). Brownfields have the potential to increase regional urban biodiversity and provide underrepresented habitat types, though care must be taken to minimize potential harm from non-native species and ecological exposure to pollution.

Conclusion

Almost five decades after railroad and industrial activities were discontinued at what would become the urban wildland of the interior of Liberty State Park, species diversity and richness continue to increase across the site. The percent cover of woody species increased in plots which had been dominated by herbaceous species while the percent cover of vines increased in all plots. A reduction in tree canopy cover from either weather-related disturbances and/or long-term stress, the mitigation of soil stress through the accumulation of organic matter, and limitation in propagule availability all have likely played a role in influencing the plant community trajectory. Correlations between forb species and nutrient related soil properties, as well as between woody species and metal related soil properties show that variability in the site's soil properties has enabled the establishment of multiple habitat types across the site, contributing to its overall diversity. The LSP Interior demonstrates that a degraded post-industrial brownfield is capable of supporting many different plant species and that after almost 50 years it continues to have the potential to increase its diversity. The continued increase in diversity during this time period suggests a trajectory that is occurring slower than would be expected for secondary succession on abandoned agricultural landscapes, perhaps driven by the site's unique land use history and isolation from other natural areas. While clearly abiotic features of the site have had significant impacts on its plant community, research is needed to better understand the impacts of biotic relationships such as competition and facilitation on community assembly of urban wildlands developing on brownfields. Finally, this study demonstrates that urban ecological systems may behave more dynamically and are best characterized through long-term studies.

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Appendix 1

Table 2 Percentage of sampling sites in 2005 within the LSP Interior which exceeded Ecological Soil Screening Criteria (ECO-SSL; Maximum Allowable Toxic Concentration, United States Environmental Protection Agency 2003) and Lowest Observed Effective Concentration (LOEC; Efroymson et al. 1997) for select total heavy metal concentrations, based on previous work at the site (Gallagher et al. 2011). $N = 35$ sample points

	As	Cr	Cu	Pb	V	Zn
ECO-SSL	76%	*	84%	88%	*	*
LOEC	20%	80%	48%	16%	50%	44%

*No applicable standard

Appendix 2

Table 3 Plant species observed in 2008 and 2016, the number of plots they were observed in, the number of plots which retained a species for both years, and the total percent cover for the 27 study plots. N = native, I = introduced, U = unknown native status

		Number of Plots			Percent Cover (%)	
Species	Native Status	2008	2016	Retained	2008	2016
Forbs						
Achillea millefolium L.	N	1	0	0	0.05	0
Alliaria petiolate M. Bieb.	I	1	1	0	0.07	0.02
Ambrosia artemisiifolia L.	N	1	0	0	0.14	0
Apocynum cannabinum L.	N	7	5	2	2.89	0.57
Artemisia vulgaris L.	I	15	11	11	21.93	14.82
Asclepias syriaca L.	N	1	1	0	0.23	0.05
Centaurea stoebe L. ssp. *micranthos* Hayek	I	5	2	1	1.5	0.4
Conyza canadensis L.	N	0	1	0	0	0.04
Coronilla varia L.	I	0	1	0	0	0.15
Daucus carota L.	I	1	2	0	0.1	0.07
Erigeron annuus L.	N	1	0	0	0	0
Eupatorium sp.	U	8	0	0	0.26	0
Eupatorium hyssopifolium L.	N	0	9	0	0	1.39
Eupatorium serotinum Michx.	N	0	7	0	0	0.42
Euthamia graminifolia L.	N	2	1	0	0.17	0.04
Hypericum gentianoides L.	N	0	1	0	0	0.14
Hypericum perforatum L.	I	2	3	0	0.38	0.09
Lactuca canadensis L.	N	0	2	0	0	0.11
Linaria vulgaris Mill.	I	3	1	0	0.1	0.03
Melilotus albus Thunb.	I	2	0	0	0.87	0
Nuttallanthus canadensis L.	N	0	1	0	0	0.06
Oenothera biennis L.	N	1	1	0	0.05	0.07
Penstemon digitalis Nutt. ex Sims	N	0	1	0	0	0.04
Plantago aristata Michx.	N	1	0	0	0.14	0
Solidago canadensis L.	N	6	11	4	1.94	4.66
Solidago juncea Aiton	N	5	6	3	0.68	1.56
Solidago rugosa Mill.	N	8	7	4	4.77	1.71
Trichostema dichotomum L.	N	1	2	0	0.14	0.27
Verbascum Thapsus L.	I	3	4	1	0.42	0.43
Graminoid						
Calamagrostis epigeios L.	I	17	19	14	15.09	15.3
Carex sp.	U	0	1	0	0	0.17
Cyperus sp.	U	0	1	0	0	0.05
Dichanthelium clandestinum L.	N	0	1	0	0	0.05
Festuca rubra L.	I	2	2	1	3.79	2.95
Panicum virgatum L.	N	8	5	2	5.46	1.72
Schizachyrium scoparium Michx.	N	0	2	0	0	0.15
Shrub						
Elaeagnus umbellata Thunb.	I	0	2	0	0	0.41
Euonymus alatus Thunb.	I	0	1	0	0	0.08
Fallopia japonica var japonica Siebold & Zucc.	I	4	7	3	4.26	4.55
Frangula alnus Mill.	I	0	1	0	0	0.05
Ilex opaca Aiton	N	1	0	0	0	0

Table 3 (continued)

Species	Native Status	Number of Plots			Percent Cover (%)	
		2008	2016	Retained	2008	2016
Myrica pensylvanica Mirb.	N	1	3	0	0.13	0.7
Phragmites australis Cav.	I	2	3	2	1.84	1.76
Elaeagnus umbellata Thunb.	N	13	13	7	8.44	5.38
Euonymus alatus Thunb.	N	4	8	3	1.61	2.86
Fallopia japonica var japonica Siebold & Zucc.	N	5	7	2	1.57	1.36
Frangula alnus Mill.	I	0	2	0	0	0.09
Ilex opaca Aiton	U	1	9	1	0.1	0.97
Myrica pensylvanica Mirb.	I	4	6	1	0.29	0.71
Phragmites australis Cav.	N	2	0	0	0.26	0
Elaeagnus umbellata Thunb.	N	0	1	0	0	0.04
Tree						
Acer rubrum L.	N	0	4	0	0	0.42
Ailanthus altissima Mill.	I	6	7	2	2.12	2.32
Betula populifolia Marsh.	N	5	7	3	5.22	7.11
Carya glabra Mill.	N	1	0	0	0	0
Chamaecyparis sp.	N	1	0	0	0.05	0
Crataegus sp.	U	0	2	0	0	0.09
Fraxinus sp.	U	1	1	1	0.03	0.44
Juniperus virginiana L.	N	1	1	0	0	0.08
Morus alba L.	I	2	1	0	2.53	0.44
Platanus occidentalis L.	N	0	1	0	0	0.36
Populus deltoides Bartr. ex Marsh.	N	4	7	2	5.13	5.83
Populus tremuloides Michx.	N	3	3	2	3.82	2.53
Prunus serotine Ehrh.	N	1	9	0	0.1	2.42
Quercus rubra L.	N	0	7	0	0	0.63
Vine						
Ampelopsis brevipedunculata Maxim.	I	0	9	0	0	0.99
Celastrus orbiculatus Thunb.	I	0	12	0	0	2.7
Fallopia scandens L.	N	0	1	0	0	0.08
Lonicera japonica Thunb.	I	3	9	0	0.19	2.19
Parthenocissus quinquefolia L.	N	4	12	2	0.51	1.57
Toxicodendron radicans L.	N	1	10	1	0.17	1.16
Vitis riparia Michx.	N	4	10	3	0.44	2.15

Appendiix 3

Table 4 Spearman correlation coefficients for soil properties measured in 27 sites in 2008. Values in bold indicate significant correlation at $p < 0.05$

	pH	CEC	OM	NO_3-N	P	Ca	K	Mg	Al	B	Cd	Cr	Cu	Fe	Mn	Ni	Pb	S	Zn
		meq/100 g	%								ppm								
CEC[a]	0.26																		
OM	0.07	**0.54**																	
NO_3-N	**0.65**	**0.49**	0.15																
P[b]	−0.03	**0.74**	0.1	0.28															
Ca	**0.84**	**0.59**	0.22	**0.76**	0.3														
K	**0.52**	**0.74**	0.19	**0.72**	**0.58**	**0.79**													
Mg	**0.70**	**0.55**	0.29	**0.75**	0.25	**0.91**	**0.82**												
Al	**−0.74**	−0.07	0.13	**−0.47**	−0.06	**−0.63**	−0.32	**−0.55**											
B	**0.44**	**0.87**	**0.40**	**0.63**	**0.62**	**0.69**	**0.84**	**0.70**	−0.25										
Cd	0.35	0.15	0.12	0.14	0.14	**0.42**	0.28	**0.41**	**−0.45**	0.18									
Cr	−0.38	0.37	0.14	−0.12	0.35	−0.13	0.1	−0.15	**0.49**	0.32	0.05								
Cu	−0.16	0.01	0.08	−0.14	0.03	−0.13	0	−0.07	**0.40**	−0.01	**0.41**	**0.55**							
Fe	**−0.86**	−0.12	0.02	**−0.51**	0.01	**−0.69**	−0.36	**−0.53**	**0.86**	−0.28	−0.24	**0.61**	**0.48**						
Mn	0.04	**0.58**	0.06	**0.28**	**0.66**	0.31	**0.51**	0.14	0.02	**0.53**	−0.09	0.23	−0.19	−0.15					
Ni	−0.33	0.29	0.13	−0.09	0.31	−0.08	0.17	−0.06	**0.50**	0.14	0.35	**0.64**	**0.68**	**0.56**	0.14				
Pb	−0.06	0.04	0.08	−0.11	0.12	−0.01	0.03	−0.05	0.01	0.02	**0.45**	0.12	**0.39**	0.11	0.11	0.37			
S	**0.45**	**0.77**	0.29	**0.60**	**0.55**	**0.70**	**0.82**	**0.70**	−0.16	**0.84**	0.08	0.26	0.06	−0.28	**0.47**	0.1	−0.18		
Zn	0.12	0.09	0	0.05	0.2	0.19	0.18	0.16	−0.11	0.07	**0.79**	0.27	**0.63**	0.08	−0.05	**0.62**	**0.65**	0.02	
A-horizon depth (cm)	0.05	−0.11	**−0.45**	−0.07	−0.01	−0.02	−0.13	−0.17	−0.19	−0.15	0.28	0.19	0.17	0.02	0.03	0.17	0.18	−0.21	**0.46**

[a] CEC = Cation exchange capacity, calculated from base saturation of Ca, Mg, and K

[b] All elements are plant extractable concentrations

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