Altered vegetative assemblage trajectories within an urban brownfield

Frank J. Gallagher a,*, Ildiko Pechmann b, Claus Holzapfel b, Jason Grabosky a

a Urban Forestry Program, Department of Ecology, Evolution and Natural Resources, Rutgers, The State University, 14 College Farm Road, New Brunswick, NJ 08901-8551, USA
b Department of Biological Sciences, Rutgers, The State University, 195 University Avenue, Newark, NJ 07102, USA

High concentrations of soil metals, impact the trajectory of vegetative assemblages in an urban brownfield leading to the speculation of an alternate stable state.

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ABSTRACT

Recognizing the growing importance of both structure (maintenance of biodiversity) and function (fostering natural cycles) of urban ecologies, we examine coarse scale (herbaceous, shrub and forest) beta guild trajectory in an urban brownfield. The distribution of the pioneer forest assemblage dominated by Betula populifolia Marsh. and Populus spp. could be correlated positively with total soil metal load (arsenic, cadmium, chromium, copper, lead, zinc, lead and vanadium), whereas herbaceous and shrub guilds were negatively correlated. Distinct assemblage development trajectories above and below a critical soil metal threshold are demonstrated. In addition, we postulate that the translocation of metals into the plant tissue of several dominant species may provide a positive feedback loop, maintaining relatively high concentrations of metals in the litter and soil. Therefore assembly theory, which allows for the development of alternate stable states, may provide a better model for the establishment of restoration objectives on degraded urban sites.

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1. Introduction

Many post-industrial urban landscapes present the opportunity to convert brownfields into future open green spaces. However, before such restoration initiatives can be undertaken and reasonable objectives established, a clear understanding of urban ecology and a corresponding philosophy of restoration must be developed. In addition to typical competitive and facilitative biological interactions, assemblage development on degraded sites is often subject to the following three strong filters: a) the ability of plant diaspores to reach these isolated urban areas is reduced; b) the soil resources needed to establish and sustain plant growth are often limited; and c) toxic substances within the soil often exceed the tolerance threshold of many plant species (Wagner, 2004). This paper focuses on the latter of these filters and the resulting implications for assemblage development and the potential for alternate stable states.

Brownfield soils often contain trace metals such as arsenic (As), cadmium (Cd), copper (Cu), zinc (Zn), lead (Pb) and others, in concentrations above regulatory screening criteria (Dudka et al., 1996). While these elements can be adsorbed or occluded by carbonates, organic matter, iron magnesium oxides and primary or secondary minerals their soluble fractions create an effective abiotic filter that “can limit the establishment, growth and/or reproduction” (Adriano, 1986; Ross, 1994), of sensitive plant species. Plants that respond passively to increasing soil metal concentrations tend to allow accumulation of metals in the plant proportionate to the concentration in the external environment until, some threshold is reached, after which homeostasis or failure of growth ensues (Baker, 1981). Other species exclude metal ions from particular organs via active processes by sequestering metal ions in metallothioneins or phytochelatins (Veski et al., 1999; MacFarlane and Burchett, 2000). In rare cases hyperaccumulation of metal ions is achieved through highly specialized physiological mechanisms that concentrate certain metals.

Under optimal growing conditions, when abiotic filters do not limit plant growth, it is probable that species interaction, specifically competition, contributes significantly to vegetative assemblage development (Lawton, 1987; Wilson and Gitay, 1995). However, the stress associated with polluted environments tends to favor species that exhibit resistance traits over species with strong competitive ability, relationships within these assemblages tend toward facilitation rather than competition (Wagner, 2004; Connell and Slatyer, 1977; Bertness and Callaway, 1994). Therefore under conditions of stress the corresponding vegetative assemblages may exhibit both structural and developmental differences.
Ecological restrictions resulting in the presence, absence, or abundance of species can produce repeatable guild developmental patterns or trajectories (Wilson and Gitay, 1995). For example, Sanger and Jetschke (2004) demonstrated that birch dominated the early colonization of a former uranium-mining dump. They argue that the early arrival of this metal-tolerant species alters the development sequence (i.e. birch – woodland vs. grass – herb assemblage) and that colonization by many other species may be delayed or totally inhibited. Such examples provide evidence for non-equilibrium or stochastic models of assemblage development where species transition is determined by both abiotic and biotic filters (Hobbs and Norton, 2004).

In this paper we describe the dynamics of coarse scale (i.e. beta guild) vegetation assemblages trajectory based on a sequence of aerial photographs of an urban brownfield known to have heterogeneous soil metal loading (Gallagher et al., 2008a). We hypothesized that assemblage trajectories had been influenced, if not directed, by soil metal contamination, which limited recruitment and or establishment of non-tolerant species from the regional species pool. We compared guild trajectories on soils with contrasting metal loads that had been isolated from human impacts for approximately 40 years. More specifically, we expected that beta guild trajectory would be distinctively different above and below a previously defined critical soil metal threshold (Gallagher et al., 2008b).

In addition, it may be possible for an ecological restriction to be reinforced through a feedback loop that is capable of operating on a large spatial scale (Belyea, 2004). Such feedback systems could result in an alternate stable steady state that is maintained for an unusually long period of time. In the Chihuahuan Desert, derelict agricultural fields exhibit soils with slow infiltration rates. The rainfall runs off rapidly and therefore the soils do not hydrate well. This results in drier conditions that produce sparse vegetation. In addition, it may be possible for an ecological restriction to be further decimating the soils, which results in increased evaporation. Rainfall runs off rapidly and therefore the soils do not hydrate well. This results in drier conditions that produce sparse vegetation.

2. Material and methods

2.1. Study area

The 251-acre (102 ha) study site is located within Liberty State Park on the west bank of Upper New York Bay at the Hudson Estuary (centered at 40°42’14” N, 74°03’14” W). Prior to its development by the Central Rail Road of New Jersey (CRBN) the site was an intertidal mud flat and salt marsh. The area was filled using debris from construction projects and refuse from New York City and the surface was stabilized with cinder and ash typically used by the rail road. Industrial use of the site for commodity transport and storage, including coal, resulted in relatively high levels of soil metals. In 1967, the CRBN discontinued operations leaving the site isolated and undisturbed. Today the park consists of approximately 1100 acres, of which 251 remain undeveloped, fenced with limited access, which serves as the study site.

2.2. Soil sampling and spatial analysis

To determine the spatial distribution of metal contamination, soil samples were collected in triplicate from 32 sites representing the various vegetative assemblage types during the summer of 2005. Soil samples were collected with a hand spade or soil corer from the depth of the greatest root concentration. Depending on the vegetation type the samples were collected between 10 and 25 cm from the surface. The sites were also examined for depth of soil above the original rail yard fill by measurement of visual soil layering in the cores as contamination was associated with rail yard activity. In general, the bottom of the root zone correlated well with the top of the rail yard fill. The samples were examined for concentrations of arsenic (As), chromium (Cr), copper (Cu), mercury (Hg), lead (Pb), vanadium (V), and zinc (Zn) using Atomic Absorption Spectroscopy (AAS). Since all the metal data were positively skewed, 16 additional sites were sampled in 2006. In addition, to better define the border areas, 16 additional sites were added to the eastern border area, to the north, and to the south of the study area. Finally, at the three sites with the highest metal load, four additional samples were taken (one in each compass direction at 20 m). Kriged maps with least mean error and root mean square error values were used to calculate area soil metal concentrations. The resulting contour maps were transformed into vector data and analyzed in the ArcGIS environment. (see Gallagher et al., 2008a for details on soil sample analysis).

Individual soil data were kriged to estimate the distributions of soil metals. Block kriging was used to accommodate the considerable standard deviation found in the soil metal data (Stein, 1999). Since the metal data was highly skewed it was transformed before performing kriging. Logarithmic (McGrath et al., 2004) and rank order (Jang et al., 2001) transformations were used to normalize the data distribution and provide more stable variograms. The results were back-transformed using the reverse function of the linear regression between the original metal data and the ranks (Wu et al., 2006). To assess the performance of kriging on the differently transformed data sets, the mean error (ME), root mean square error (RME) and the coefficient of determination ($r^2_p$) were calculated and yielded acceptable results (see Gallagher et al., 2008a for details on performance evaluation).

Since the concentration of individual soil metals did not produce significant relationships when compared to either plant productivity or assemblage diversity, the total soil metal load (TML) was determined by calculating the summation of the rank order transformation of the individual metal concentration as described by Jang et al. (2001). The spatial distribution of the data set was developed using the block kriging method and Surfer Surface Mapping Software (Release 8.0, Golden Software, Golden, CO, USA), which compensates for the high standard deviation (Lénaers et al., 1990).

The resulting index provided a way to evaluate the cumulative soil metal load using a scale from 0–5 in 0.1 unit increments, with 5 indicating the highest cumulative soil metal concentrations. This metric was then used as a relative index of soil contamination. A critical TML threshold value of 3.0 was used as a benchmark for areas of high metal load as our previous work indicated that soil metal loads above this level had a significant sublethal impact resulting in metal-induced stress on plant productivity, long-term growth and diversity (Gallagher et al., 2008b).

2.3. Soil metal distribution

The total soil metal load (TML) was unevenly distributed (Fig. 1a) and exhibited considerable local variation (Gallagher et al., 2008a). The concentration of As exceeded the Lowest Observed Effective Concentration (LOEC, Oak Ridge, 1997) at 20% and Pb at 16% of the sites. Cr exceeded the LOEC at most of the sites (80%). The other measured soil metals exceeded the LOEC at approximately half of the sites (Table 1).

2.4. Spatial vegetation classification

The variability in substrate materials combined with the undulating topography of the abandoned rail yard has created a patchy distribution of soils and cinders that has influenced plant colonization over the last 40 years. A unique mosaic of assemblage types exists within the study site (United States Army Corps of Engineers, 2004). The hardwood assemblage is composed primarily of Gray Birch (Betula populifolia Marsh.) (35% cover), Eastern Cottonwood (Populus deltoids M.) (all cover) and Quaking Aspen (Populus tremuloides Michx.) (14% cover) (Gallagher et al., in press). Shrubs are dominated by three species of the genus Rhus; Staghorn Sumac (R. typhina L.), Smooth Sumac (R. glabra L.), and Winged Sumac (R. copallinum L.). Herb and grass assemblages are dominated by a mix of several native and non-native species: Goldenrod (Solidago spp.), Mugwort (Artemisia vulgaris L.), Switchgrass (Panicum virgatum L.), Cheat Reedgrass (Calamagrostis epigeios L.) and Red Fescue (Festuca rubra L.) (all nomenclature after Gleason and Cronquist, 1991).

Historic true color aerial photographs from 1976, 1984, 1993, 1996, 2000, and a 1999 gray scale photograph, were scanned at 600 dpi and were geo-referenced in ArcGIS using a 2002 aerial photo (released by NJDEP, spatial resolution, 0.62 m) as a reference. The rectification resulted in less than 0.6 m RSM (root square mean) error for each of the photographs. In delineating the assemblages, three field-verified digital vegetation maps were used, the first from the 2003 United States Army Corps of Engineers (USACE) Liberty State Park Environmental Assessment (USACE, 2004) based on the 2002 NJDEP aerial photo, the second from a 1996 aerial photograph which was field verified during a park natural resource assessment (MacFarlane, 1996) and the third was the 2009 field survey. In addition, an early report (Texas Instruments, 1976) was also used to verify the 1976 digital vegetation map. The 2003 vegetation map differentiated 12 alpha guilds based on species composition (Table 2). The alpha guilds were classified according to the Ecological Classification of New York State (Edinger et al., 2002). The identified assemblages ranged from successional northern hardwood to grasslands. Linear regressions between assemblage distribution (area ratios) and total soil metal load, indicate significant relationships within the successional northern hardwood ($r^2 = 0.62$, $p < 0.01$) and semi-emergent marsh ($r^2 = 0.49$, $p < 0.01$) (Gallagher et al., 2008a).
Interestingly, the area ratio of the successional northern hardwood assemblage increases as the soil metal load increases, indicating a competitive advantage under increasing soil metal stress. Since wetlands would not be expected to follow the same development trajectory associated with dry grasslands, the approximately 22 acres of herbaceous wetlands were removed from the analysis.

Assemblages were then grouped into four-beta guilds, herbaceous, wetland, shrub and hardwood forest (Fig. 1b). Working backwards from 2009, and using the 1986 digital map and the 1976 report as references, the four guilds were on-screen digitized at 1:200 scale using ArcMap (ArcGIS 9.0, 2004). The beta guild distribution (percent cover) was then determined for each of the eight years. The results were grouped into two data sets to examine guild trajectory in areas above and below the critical TML of 3.0.

To compare the change in each beta guild over time both above and below the critical TML, we calculated the difference in percent cover for the three guilds at each consecutive time interval according to the following formulae:

Shrub guild \( s_{t1} = -(s_{0} - s_{t1}) \)

Herbaceous guild \( h_{t1} = -(h_{0} - h_{t1}) \)

Hardwood forest guild \( f_{t1} = -(f_{0} - f_{t1}) \)

where 0 = percent cover in the targeted year and 1 = percent cover in the latter year of photo sequence.

Pearson's correlations between beta guild distribution (percent cover 2003) and total soil metal load (2005) were calculated. Linear regressions were developed for time and beta guild distribution percentages both above and below the critical TML value of 3.0. In addition, paired t-tests were used to compare individual beta guild distribution above and below the critical TML over the duration of the study period. Statistical analyses were conducted using both SPSS (release 17, SPSS Inc.) and Minitab (Minitab release 14).

3. Results

3.1. Beta guild distribution relative to TML

When the area (% cover) of the beta guilds based on the 2003 vegetation map is plotted against the TML (2005) at intervals of 0.1, distinctly different relationships are detectable (Fig. 2) each yielding statistically significant correlations. The correlation between the forest and TML is strong and positive \( (r = 0.78, p < 0.01) \).


Beta guild trajectories both above and below the critical TML threshold of 3 were examined over a period of forty years. Below the critical TML threshold the dynamic transition between guilds, as measured by percent cover, exhibit a fairly traditional trajectory (Fig. 3a). The herbaceous guild colonized much of the rail yard within the first seven years (1969–76). Shrub and hardwood tree guilds gradually colonized the site, with the shrub guild at greater concentrations than the hardwood tree guild until the mid 1980s. By the late 1980s hardwood tree guild covers a greater area than the shrub guild and by the mid 1990s it is the dominant guild. Finally by 2009 the hardwood forest guild covers 69 percent of the site.

Above the critical TML threshold (Fig. 3b) the herbaceous guild dominates in the early years and covers over 70 percent of the site by 1976. The hardwood tree guild begins to colonize by the late 1970s and then expands rapidly in the early 1980s. The shrub guild is virtually absent in areas above the critical TML threshold until 1996. It never exceeds 10 percent of the area covered and is in decline between 2003 and 2009. By 2009 the hardwood forest guild covers 76 percent of the site, the forest guild overtakes (percent cover) the herbaceous guild (above and below the critical TML threshold) in approximately the 28th year and the shrub guild begins to decline between 2003 and 2009.

A paired T test between the temporal changes in the percent coverage for each beta guild above and below the TML threshold (Fig. 4a–c) indicates that the differences between the herbaceous and forest guild trajectories were not significant \( (p = 0.87, p = 0.79, 0.87) \).
Discussion

4.1 Soil metal induced stress

Under natural conditions, metals are present in soils as derivatives of parent geologic materials. Serving as a micronutrient, metals at low concentrations are required for normal metabolic function of both flora and fauna. At concentrations above functional ranges, the plant can no longer maintain homeostasis and metabolic functions are inhibited (Harrison, 1999). It follows then, that species intolerant of high metal concentrations would be excluded from the assemblage on the site with high soil metal loads even if a seed source existed within the regional pool. Therefore, vegetative assemblages growing on metaliferous soil should exhibit structural differences when compared to similar but uncontaminated regional environments. These differences have been shown to manifest themselves in arrested successional developments (Niering and Goodwin, 1974), such as those found on naturally heavy metal-rich rock outcrops (e.g. serpentine outcrops, Reynolds et al., 1997; Harrison, 1999). Kimmerer (1981) as well as Kalin and van Everdingen (1988) demonstrated that assemblage development on mine tailings proceeded slowly, often remaining in the herbaceous stage for decades or centuries. In addition, the colonization and distribution of plant species on mine tailings from five Pb/Zn mines in China were still dominated (69.4% of total) by herbaceous species after twenty years of development (Li et al., 2006).

Our study corroborates the concept that patterns of assemblage development, especially the advancement of tree species, can be impacted if not driven by soil metal concentration. Interestingly, in contrast to the previously mentioned mine tailing studies, it appears that the hardwood assemblage at Liberty State Park has developed preferentially on the soils with increased total metal load. Sangue and Jetschke (2004) reported similar results on a former uranium-mining disposal area (East Germany). They argue that the early arrival of tolerant woody species alters the development sequence (i.e. birch-woodland vs. grass -herb assemblage) hence the colonization by many other species may be delayed or totally inhibited.

This is apparently the case at Liberty State Park. An examination of historic photos indicates that assemblage development in areas where the TML was high (3 or above) favored colonization of Gray Birch (B. populifolia) and Cottonwood (Populus spp.) known to be metal tolerant species. The shrub guild dominated by three species of Sumac (Rhus spp.) not known to be metal tolerant does not appear until 1996 and never covers more than 10% of the area. The strong significant correlation between soil TML and forest distribution is probably the result of both metal tolerance and the

Table 1

Comparison of soil metal concentrations (µg g⁻¹, mean ± S.D.) with the Lowest Observed Effective Concentration (LOEC, OakRidge, 1997) and Maximum Acceptable Toxic Concentration (MATC, ECO-SSL, 2003). Triplicate samples were taken at one meter intervals at 35 sampling points. MDL = Minimum Detection Limits, % above – area of the study site above ECO-SSL or LOEC.

<table>
<thead>
<tr>
<th>As</th>
<th>Cr</th>
<th>Cu</th>
<th>Hg</th>
<th>Pb</th>
<th>V</th>
<th>Zn</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt;0.005</td>
<td>&lt;0.01</td>
<td>3.1</td>
<td>0.002</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>17.9</td>
</tr>
<tr>
<td>Min</td>
<td>0.005</td>
<td>9.7 ± 2.5</td>
<td>44.0 ± 2.5</td>
<td>&lt;0.005</td>
<td>860 ± 11.1</td>
<td>&lt;0.005</td>
</tr>
<tr>
<td>7.5%</td>
<td>16.7 ± 2.5</td>
<td>22.5 ± 4.8</td>
<td>74.0 ± 11.6</td>
<td>0.1 ± 0.1</td>
<td>18 ± 3.8</td>
<td>26.9 ± 15.2</td>
</tr>
<tr>
<td>Median</td>
<td>33.5 ± 6.8</td>
<td>38.8 ± 7.9</td>
<td>153 ± 27.7</td>
<td>0.3 ± 0.1</td>
<td>406.0 ± 73.6</td>
<td>44.0 ± 20.7</td>
</tr>
<tr>
<td>75%</td>
<td>121.9 ± 29.8</td>
<td>60.2 ± 25.7</td>
<td>253.0 ± 58.2</td>
<td>0.7 ± 0.3</td>
<td>520 ± 181.1</td>
<td>76.1 ± 33.2</td>
</tr>
<tr>
<td>Max</td>
<td>977.6 ± 44.3</td>
<td>208.8 ± 10.4</td>
<td>1870.0 ± 315.0</td>
<td>3.6 ± 0.6</td>
<td>4640 ± 1799</td>
<td>193.2 ± 112.6</td>
</tr>
<tr>
<td>% above ECO-SSL</td>
<td>76</td>
<td>76</td>
<td>84</td>
<td>88</td>
<td>**</td>
<td>**</td>
</tr>
<tr>
<td>% above LOEC</td>
<td>20</td>
<td>80</td>
<td>48</td>
<td>16</td>
<td>50</td>
<td>44</td>
</tr>
</tbody>
</table>

Table 2

The dominant species within each alpha guild as classified according to Ecological Communities of New York State (Edinger et al., 2002). Beta guilds grouped as follows: 1) forest, 2) shrub, 3) herb.

|-----------------|-------------------------|-----------------|-------------------------------------------|-------------------------------------------|------------------------------------------|-------------------------|--------------------------------|----------------------------------|----------------|--------------------------|--------------|--------------------------------|--------------------------------|-------------------------|---------------------------------|---------------------------------|---------------------------------|-----------------------------------|---------------------------------|---------------------------------|-----------------------------------|

Fig. 2. Guild distribution (percent cover) of the three beta guilds, Hardwood Forest (HF) Shrub and Herbaceous (Herb) and trend lines for 2003. Pearson correlations for each guild and total soil metal load (2005) are: hardwood forest assembly r = 0.78, p < 0.01, shrub assembly r = 0.46, p = 0.03, herbaceous assembly r = -0.54, p = 0.01.
nearest neighbor effect driven by the clonal nature of the dominant tree species. In spite of similar clonal attributes of Rhus spp. and the resulting ability to partition resources, these shrubs could not overcome the abiotic filter in the areas of high TML. In contrast, below a TML of 3 the shrub guild is well represented, covering 15% of the site by 1985 and 22% of the site by 2000.

4.2. Beta guild trajectory

Several basic ecological models are fundamental in understanding the debates surrounding guild trajectories. Succession theory, or the classic deterministic model (Clements, 1916), states that vegetative assemblages tend to follow predictable patterns of development in which a predetermined state or end point is known with a high degree of probability. The stochastic model (Gleason, 1926; Van der Maarel and Sykes, 1993) envisions a more random process that is dependent upon niche availability; regional species pool composition and order of arrival. This theory proposes a complex random process in which trajectories cannot be defined. More recently the argument known as Assembly Theory proposes that "community development is determined by random variation in species' colonization rates and the subsequent likelihood of their establishment and persistence in the community" (Young et al., 2001). While succession theory proposes a deterministic process with primarily one endpoint or climax community, assembly theory allows for the possibility of several steady states that may last for significant periods of time. Determinism in assembly theory is a function of the mechanism that results in altered steady states rather than the species composition (Suding et al., 2004).

At Liberty State Park, assemblage trajectories appear to be altered by the soil metal load. Below a critical TML threshold, herb/grass, shrub and early hardwood tree assemblages have developed in a pattern typical for the northeastern region of the United States. For these areas it would be reasonable to project a traditional endpoint provided that seed recruitment is not a major filter. Above the critical TML threshold, herbs/grasses were more quickly replaced with metal tolerant pioneer tree species apparently skipping the shrub stage (the linear regression between time and cover of the shrub guild above the critical TML threshold was not significant). Two distinct scenarios are envisioned when projecting a future trajectory for assemblage development in areas above the critical TML threshold. In the first scenario, the TML remains high, as metals which accumulate in the plant tissue of tolerant species continually cycled through decaying vegetative material, fostering the development of a positive feedback loop. This abiotic filter could continue to limit success of climax tree species resulting in an alternate stable state in which pioneer tree species tolerant of high metal loads dominate. Conversely, as the site ages, the continued addition of organic matter might eventually mitigate the high soil metal concentrations. Under this scenario the removal of the abiotic filter, could result in a trajectory that leads to a typical hardwood climax.

4.3. The potential for an alternate stable state

Alternative ecological states are defined as biotic assemblages and environmental conditions that persist at a particular spatial
extent and temporal scale (Sutherland, 1974; Petraitis and Dudgeon, 1999; Suding et al., 2004). They are widely being used to predict ecosystem collapse as the result of climatic changes or other large-scale environmental change (Scheffer et al., 2001) or to define assemblage structure in degraded and therefore unpredictable sites (Nystrom et al., 2000). Models of alternate states include feedback mechanisms that reinforce their persistence in space and time (Van de Koppel et al., 2001).

Rigorous tests to determine if these assemblages actually represent a stable equilibrium are generally beyond the scope of most restoration efforts or research initiatives as such sites are generally subjected to invasion or species turnover (Walker and Wilson, 2002).

Liberty State Park is a unique urban site that has the potential to address the question of equilibrium stability. The site has been isolated and basically inaccessible to the public for the past forty years. The site exhibits little surface water discharge and is bounded on three sides by capped parkland and on the fourth by the Hudson River Walkway constructed on a compacted levee. Groundwater movement toward the northeast is extremely slow (Mantey et al., 2009 and found to be between 5 and 7 (with a mean of 5.7), a level does not generally facilitate soil metal leaching. This is supported by samples taken from deep test pits that penetrated into the original sediment, which contained only background levels of metals (USACE, 2004). In addition, the translocation of several metals into the tissue of several dominant vegetative species (Table 3) was found to be considerably high (Gallagher et al., 2008a). In addition, fresh leaves and leaf litter was collected from the several sites within the project area in 2009 (Schafer K., unpublished data). The data suggest that the metals within the litter correlate well with that of the fresh leaf concentrations (Fig. 5). These metals will cycle within the leaf litter and upper soil horizons as the vegetation decomposes. Such a feedback loop has the potential to result in an abiotic filter inhibiting the establishment of typical but less tolerant climax hardwood species and deserves further study. This scenario supports the concept that models for assembly rules, at least those associated with the degraded environments of the urban context, must account for abiotic filters as argued by Keddy (1992) and Weih and Keddy (1995) rather than focusing primarily on the importance of biotic interactions such as competition or facilitation between species.

4.4. Natural attenuation leading to a traditional succession scenario

It is known that metals are often adsorbed or occluded by a variety of compounds such as carbonates, organic matter, Fe-Mn oxides, and primary or secondary minerals (Ross, 1994). Since there have been no human impacts at the site that would change the concentration of Fe or Mn between 1969 and 2009, we expect that change in soil metal concentrations will be related at least in part to translocation into plant tissue and adsorption to organic matter. We compared soil metal data from samples taken in 1996 (New Jersey Department of Environmental Protection) and 2005 (Fig. 6). While it must be acknowledged that the sampling protocols (split spoon vs. hand spade) were not equivalent and the 1996 samples were not taken in triplicate as were the 2005 samples a repeated measures ANOVA of the log transformed soil metal concentration for the common species does indicate a significant difference ($F = 6.373$, $p = 0.013$) between the samples. In all cases the 1996 samples had higher concentrations than the 2005 samples. It should also be noted that the metals which demonstrated the greatest change, Pb and Zn, were the metals which exhibited higher rates of translocation into plant tissue. However, since Pb does not translocate to the aerial section of the plant it is not contained in the leaf litter at significant concentrations. It has been demonstrated that leaf litter can provide a sink for metals. The metals bind passively to organic

<table>
<thead>
<tr>
<th>Table 3</th>
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<tbody>
<tr>
<td><strong>Table 3</strong> Average rate of heavy metal concentrations (µg g⁻¹) in leaves (l) stems (st) and roots (r) and soil (s) of several dominant plant species, Betula populifolia, Rhus copallinum, Artemisia vulgaris and Populus tremuloides.</td>
</tr>
<tr>
<td>l stdev r stdev st</td>
</tr>
<tr>
<td>As</td>
</tr>
<tr>
<td>B. populifolia</td>
</tr>
<tr>
<td>R. copallinum</td>
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<tr>
<td>A. vulgaris</td>
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<td>P. tremuloides</td>
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<td>B. populifolia</td>
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<td>R. copallinum</td>
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<td>A. vulgaris</td>
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<td>P. tremuloides</td>
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<tr>
<td>Zn</td>
</tr>
<tr>
<td>B. populifolia</td>
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<td>R. copallinum</td>
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<td>A. vulgaris</td>
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<td>P. tremuloides</td>
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surfaces or actively through the physiological activity of the microbial colonizers (Gadd, 1999; Ledin, 2000). Leaf litter can also act as a source when microbial activity mobilizes the metal (Gadd, 1999) or through the action of deposit feeders (Weis and Weis, 2004). Metal sequestration is dependent upon the rates of uptake and retention by the various tissue types, translocation to deposit feeders and release through decomposition.

In this young terrestrial system, the continued addition of organic material as a result of plant growth may be acting as a sink, which would reduce the pool of available soil metals. If this concept is valid, then the strength of TML as the deterministic abiotic factor may decrease over time leading to the establishment of species with lower soil metal tolerance. This suggests that the current B. populifolia dominated hardwood assemblage may not be the end point for assemblage development. To better understand the potential for litter to act as a sink that would mitigate the potential for metal induced plant stress the metal flux within heterogeneous mixture of soil compounds and assemblage conditions will need to be modeled. Such a model will have to consider the complex relationships between soil pH, soil moisture and redox potential (Bostick et al., 2001).

5. Conclusion

The restoration of degraded sites especially those associated with the urban context has become an area of increasing interest (Dobson et al., 1997; Young, 2000). Many of these sites lack an ecological legacy and often do not respond predictably to traditional management practices (Hobbs and Harris, 2001; Klotzli and Grootjans, 2001). While succession has provided the model on which most restoration efforts have been based, their success rates have been varied and are somewhat disappointing (Lockwood, 1997). Our study demonstrates that the anthropogenic legacy of an urban brownfield can produce an environmental stress gradient resulting in distinct vegetative assemblages with different guild trajectories. In addition, the eventual end point for assemblage development is uncertain as the continued presence of elevated soil metals has the potential to lead to the development of an alternative steady state.

It may therefore be more effective to define long-term restoration objectives with flexible endpoints that clearly articulate the potential for the site exhibit alternate trajectories or remain in an alternate steady state rather than proceed along the traditional fixed trajectory. Specifically, managers of sites within the urban context should consider the role that naturally assembled urban wildlands can play in contaminant stabilization before prescribing a traditional a species mix based upon traditional concepts of diversity and accepted trajectories.

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