THE ROLE OF SOIL METAL CONTAMINATION IN THE VEGETATIVE ASSEMBLAGE DEVELOPMENT OF AN URBAN BROWNFIELD

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Abstract of the Dissertation

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Anthropogenic sources of toxic elements have seriously compromised the ecological integrity of many green areas in urban landscapes. Analysis of soil samples from a brownfield within Liberty State Park, Jersey City, New Jersey, USA, shows that arsenic, chromium, lead, zinc and vanadium exist at concentrations above those considered ambient for the area. Accumulation and translocation features were characterized for the dominant plant species of four vegetative assemblages.

A comparison of soil metal maps and vegetative assemblage maps indicates that northern hardwoods dominated areas of increasing total soil metal load while semi-emergent marshes, consisting mostly of endemic species were restricted primarily to areas of low soil metal load.

Using both satellite imagery and field spectral measurement we examined plant productivity at the assemblage and individual specimen level. In addition, we studied longer-term growth trends via tree core data (basal area increase). Leaf chlorophyll content within the hardwood assemblage correlated with a threshold model for metal tolerance, decreasing significantly beyond a total soil metal index of 3.5. Biomass production in Betula populifolia (Marsh) (Gray Birch), the co-dominant tree species, demonstrated an inverse relationship with the concentration of Zn in the leaf tissue during the growing season. Incremental basal area growth in B. populifolia also exhibited a reciprocal relationship with soil metal load.

The results of this study also indicate that B. populifolia, employs a strategy whereby different wing loading rates result from variation in size and weight of the seed. The decrease in seed size correlated well with total soil metal load, while the correlation with seed weight was marginal.

All three areas of the study indicate that assemblage development is impacted if not driven by soil metal contamination. Hence, models for vegetative assembly development, at least those associated with contamination gradients within the urban context, should account for these abiotic factors rather than focusing primarily on competition or facilitation between species.

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Dedication

Wendell Berry said "you don't know who you are until you know where you are". I do not think he would mind a slight amendment: After over thirty-three years of marriage, you don't know who you are without your partner. Thank you Ellen.

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Acronyms and Abbreviations

CRRNJ	Central Railroad of New Jersey
dbh	Diameter at Brest Height
ERI	Environmental Resource Inventory
FR	Feasibility Report
GIS	Geographic Information System
GMP	General Management Plan
HRE	Hudson-Raritan Estuaries Program
IFS	Integrated Feasibility Study
NJDEP	New Jersey department of Environmental Protection
NJDPF	New Jersey Division of Parks and Forestry
NYD	New York District, United States Army Corps of Engineers
RST	Remote Sensing Technology
TI	Texas Instrument
TML	Total Soil Metal Load
USACE	United States Army Corps of Engineers
USDA	United States Department of Agriculture

Chapter I

Introduction

Problem Statement/Needs Assessment

Throughout the world, activities associated with the industrial revolution have led to the contamination of many urban landscapes. Where a direct threat to human health could be documented, mitigation is normally a priority, but brownfields where the risk of human exposure is not critical have proven more difficult to remediate. At any one time, the New Jersey Department of Environmental Protection (NJDEP) oversees some 23,000 contaminated sites. An estimated 10,000 of these are potential brownfield sites (Department of Environmental Protection, 2006). These sites exhibit metal contamination that prevents public access but do not warrant large scale remediation. Resulting from the land uses of the industrial revolution, brownfield soils typically contain higher levels of trace metals such as cadium, copper, lead and zinc (Dudka et al., 1996) and/or others. While these elements are often adsorbed or occluded by carbonates, organic matters, Fe - Mn oxides and primary or secondary minerals (Adriano, 1986; Ross, 1994) they have the potential for sub-lethal impact on the biota. At Liberty State Park, soil metals exist at concentrations above those considered ambient for the region (Saunders, 2002) and in several cases above the state's soil screening criteria. Due to its location and current ecological condition restoration via soil removal or capping have been eliminated as viable options. An alternative mitigation plan, one that examines the relationship between the endemic vegetation assemblages and the soil metal contaminants, is being explored.

Brownfield redevelopment is a relatively recent, complex and dynamic area of public policy. Government agencies at all levels - local, state, and federal - have been grappling with the liability, environmental, and cost issues posed by brownfield restoration. Liberty State Park (LSP) is one of the most successful restoration projects in the state, if not the country. Early restoration efforts focused on debris clean up, facilities development and capping hazardous soils. As a result, the former Central Railroad of New Jersey (CRRNJ) complex has been transformed into a vital historic, ecological and recreational resource.

In the center of the park, there remains approximately 251 acres, comprising the former railroad yard, that are undeveloped. Much of the area has been re-colonized by various plant communities. These communities represent unique associations of both endemic and non-native species that can be considered the by-product of the cultural events that have taken place during the past several centuries. A broad-based, goal-driven approach has been used to develop the General Management Plan (GMP) for the site (Appendix I).

The plan is predicated on two premises:

1. Various plant communities have re-colonized much of the site. Like the surrounding community of people, these assemblages are diverse and have origins throughout the world. This diversity is further enhanced by the rapid rate of natural succession. Hence, there is ecological and aesthetic value in most of the existing natural assemblage.

2. Soils present a potential challenge. They consist primarily of fill brought in by the railroad companies between 1860 and 1919 to stabilize the rail-yard surface. Much of it is non-consolidated material from construction projects in Manhattan, or refuse from throughout New York City and the surrounding area. The soils, classified as historic fill, have use limitations. Future development within the area will have to creatively combine soils mitigation and/or stabilization to eliminate potential risk to the visitor and reduce the possibility of bioaccumulation of metals within the system.

With an understanding of the risk presented by the historic fill and an appreciation for the ecological and aesthetic value of the existing vegetative assemblages, the Liberty State Park Interdisciplinary Planning Committee established the following objectives within the GMP:

1. Provide public access for interpretive programs, allowing visitors to touch the natural world.
2. Maintain as much of the site as possible, especially wetlands and special plant communities, under a conservation mandate, while providing public access.
3. The landscape of the interior should reflect the history of the park, as well as the connection to the harbor/estuary.
4. Provide public access to the perimeter of the site for multiple uses.
5. Improve topography, enhance wetlands and provide open water, and enhance aesthetic values and sight lines where possible. In those areas that are to be disturbed, new elevations will be established that enhance the existing

wetlands, possibly creating open water habitat and taking advantage of the spectacular views of the harbor and New York City skyline.

6. The planning effort will be conscious of other neighboring redevelopment efforts.

Planning Initiative

Recognizing the significance of the project to the overall ecological health of the harbor and its public benefit, the Unites States Army Corps of Engineers (USACE) included LSP as one of the top 13 restoration efforts to be included in the Hudson-Raritan Estuaries Program (HRE). The USACE planning process for the HRE was authorized under the Water Resources Development Act of 2000 (WRDA 2000), Section 905 (b) Preliminary Analysis, and advanced with the "Needs and Opportunities Report". These studies identified potential restoration sites for which there does exist public support and the potential for a cost share sponsor.

With the New Jersey Department of Environmental Protection (NJDEP) as a cost share sponsor and immense public support for the project, Liberty State Park LSP, is one of the first sites identified by the Needs and Opportunities Report for which an ecosystem restoration is planned. Under the HRE, the New York District of the USACE (the District), along with its co-sponsor, the Port Authority of New York and New Jersey (PANYNJ), have conducted an Integrated Feasibility Study and Environmental Impact Statement for the restoration of a 251-acre portion of LSP.

A combined Feasibility Study and Environmental Impact Statement was completed in July of 2004. The inventory includes a report of existing information, an ecological communities' description, and a discussion of the ecological communities' functional assessment based on a survey conducted within the LSP restoration site. A conceptual development plan was completed in 2005 (Appendix II). The design and development documentation is nearing completion (2007). The USACE has included the construction of this project in the WRDA 2005 authorization request. Both the United States Senate and House of Representatives have passed legislation authorizing the project. The Environmental Impact Statement does not, however, address the risk associated with, or the potential for remediation, or stabilization of contaminants within the soils.

To realize the proposed plan, two basic questions must be answered: The first deals with the viability of the soils. As a result of the site's historic land use, various soils studies of LSP have been conduvted to determine if an environmental or public health hazard exists. Extensive samples were taken between 1990 and 1995. These were analyzed for their physical character, volatile and non-volatile organic compounds and metal constituents. Several metals and some organic compounds were found to be above residential standards and the soil has been classified as "historic fill". Pursuant to state statute, P.L. 1997, c.278, the "presumptive remedy" for remediation of such sites is to use engineering and institutional controls for in-place containment and control of the fill material. However, as mentioned above, the GMP calls for much of the area to remain unmediated, as the current ecological and aesthetic values are assumed to be greater than the potential for bioaccumulation of metals. The identification and confirmation of metal movement within the system will determine whether bioremediation, natural attenuation, or simply stabilization are possible.

The second question concerns the development of viable ecological benchmarks, which can be used to measure the success of the restoration effort. Sound ecological restoration, resulting in the best possible outcome for a specific site, is based upon ecological knowledge and the interests of the stakeholders. Public involvement in the planning process has been encouraged in order to learn about the concerns, issues, expectations, and values of existing and potential visitors, park neighbors, people with traditional and cultural ties to lands within the park, concessionaires, cooperating associations, scientists and scholars, other government agencies, and partners. Now all must agree on the standards of a functional urban forest and created wetlands. Defining the expected ecological outcomes for this restoration effort is essential if an agreement is to be reached, an expectation that should be based on data derived from the site.

Literature Review

Defining the Restoration Effort

The documented interest in ecological restoration, at least in North America, may have begun with Aldo Leopold's focus on both the prairies of the Midwest, which were decimated by poor agricultural practices, or the mountains of the Southwest, overgrazed through negligent wildlife management. His philosophy concerning the rehabilitation of the land is expressed in essays such as "Thinking Like a Mountain" (Leopold 1949) or "The Land Ethic" (Leopold 1953) and are founded in a basic understanding of two principals. First, ecosystems are built on the dynamic interrelationships of production and consumption and second, we have, through our technology, gained the immense power to alter or even destroy these systems. "We remember his successful Sauk County, Wisconsin, restorations, the subject of his famous Almanac and his vigorous campaigns at the University of Wisconsin-Madison Arboretum restorations, what most regard as some of the earliest intentional, ecologically based restorations in North America" (Higgs 1977). His philosophy towards restoration was driven by utility. What species would provide the best wildlife habitat or yields the best timber production? However, he was a pragmatist with a soft heart. Leopold finishes a discussion concerning species selection with the following, "The only conclusion I have ever reached is that I love all trees, but am in love with pine" (Leopold 1949). In some ways our current approach to the practice of ecological restoration has not significantly changed, in that the benchmarks of a successful project are hard to define and often driven by the pragmatic.

In 1990, the Society for Ecological Restoration adopted and published the following definition: "Ecological restoration is the process of intentionally altering a site to establish a defined, indigenous, historic ecosystem. The goal of this process is to emulate the structure, function, diversity and dynamics of the specified ecosystem" (Society for Ecological Restoration 1990). The argument that followed this and subsequent similar definitions, focused on the description of the reference system. How literally should the reference be defined and at what point in time should the reference be taken? Even when reference sites exist our ability to replace indigenous dominant species cannot replicate the time period and the multitude of relationships inherent to the process of succession.

In 1995, the Society broadened this definition as follows: "Ecological restoration is the process of renewing and maintaining ecosystem health" (Higgs 1997). It is interesting that this broader definition defies the trend of reductionism, common to most sciences, and moves towards the more holistic benchmark of ecosystem health. However, the definition does little to provide concrete framework with which to measure success. "Restorationist often must encounter their system as terra incognita and must try to solve system specific problems by trial and error" (Hobbs, Nuttle and Halle 2004).

The rapid increase in ecological restoration initiatives, especially in areas where anthropogenic alteration of the site has eliminated virtually all remnants of the historic landscape, is forcing reconsideration of these definitions. As scenarios with poorly defined reference sites have become common in the practice of ecological restoration, the Board of the Society of Ecological Restoration again broadened the definition to “the process of assisting recovery and management of ecological integrity. Ecological integrity includes a critical range of variability in biodiversity, ecological process and structures, regional and historical context and sustainable cultural practices” (SER 2002). Hence, the importance of the historic reference appears to have been considerably diminished. In addition, most evaluative work has been based on criteria that measure technical performance (e.g., Briggs et al. 1994). However, restoration must also be evaluated in terms of the aesthetic, cultural, political, and ethical contexts that envelop each project.

The restoration project at LSP appears to stretch this definition even further. The historic reference for the site would be mudflat. While an attempt to create mudflat and salt marsh will be undertaken in approximately 40 acres where tidal flow can be reestablished, the majority of the site will be managed as uplands. A conscious decision has been made as part of a public planning process that the current ecological and aesthetic values of the extant system are greater than the need to reestablish mudflat and saltmarsh in that section of the New York Harbor. It is this focus on the ecology of the site, as demonstrated by the existing assemblage development and function that defines this project as a restoration initiative, rather than simply a reclamation effort in which stabilization and public safety would be the objectives.

Regardless of the changing targets provided by the evolution of the definition, sound ecological restoration fosters the best possible outcome for a specific site based on ecological knowledge and the diverse perspectives of the interested stakeholders, to this end, it is as much a process as product. Hence “the definition of sound ecological restoration will vary from site to site, but will always be rooted by ecological fidelity: the combination of structural replication, functional success and durability” (Higgs 1977).

Two basic ecological models are fundamental to understanding the debates surrounding the definition of ecological restoration. Succession theory, the standard presented in biology and ecology text for many decades, states that systems tend to follow predictable patterns of development, in which a predetermined state or end point is known with a high degree of probability. A second argument, known as assembly theory, recently gaining momentum states 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 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 a significant period of time. Assembly theory is more flexible allowing for random events and alternate steady states.

The fundamental difference between succession and assembly theory, the treatment of steady states, has significant implications for restoration ecology. Using traditional succession theory, the identification of a reference to be used as a measurable benchmark is relatively defined. On the other hand, if we consider the potential of alternate steady states, then the identification of a traditional reference becomes more problematic. Despite the paradigmatic differences between these models, they also hold much in common. “Both seek to explain community composition. Both suggest that there is a historical composition to this composition. Both acknowledge that communities develop through time toward relatively stable states. Both assume the importance of biotic interactions, especially competition” (Young 2001).

Succession has provided the model on which most restoration efforts have been based (Lockwood 1997). In a review of 566 projects, all but three were end-oriented attempts to emulate the composition of extant assemblages. The driving force behind this practice may be the realities of time, place, and funding source. The paradigm that a single or a few interventions which replace dominant species in a similar spatial pattern appears to be deeply imbedded in the decision making process. Requisite abiotic conditions and the sequence of species colonization are basically ignored. The practice of restoration ecology, often driven by fixed costs and implementation schedules, finds it difficult to accommodate flexibility in either the schedule or the end point.

Perhaps this is why many restoration efforts fail. In a review of 87 published studies (Lockwood, Pimm 1999), 20 percent were considered either successful or unsuccessful, while most were only partially successful. In addition, 61% of the studies which targeted functional goals (i.e., primary productivity, water quality, etc.) were considered successful, while 41% of those that targeted structural goals were considered successful. However, the majority of success structural restoration projects focus on the establishment of a dominant species as benchmark of success, rather than composition of the assemblage.

Given these results, perhaps assembly theory provides a more effective model for many restoration initiatives, provided several criteria can be met. First, the introduction of species when appropriate, must consider assemblage structure at various stages of development. Second, introduction should follow a logical sequence that allows species to persist long enough to build requisite relationships. Introduction in mass can lead to increased competition and a reduction of species richness, an outcome antithetical to most restoration initiatives. Lastly, the order and timing of introduction can be controlled, since order of establishment often affects competitive interactions (Lockwood 1997). The upland restoration initiative at LSP allows for the examination of such a model if introduction can be controlled, colonization can be monitored, and a long intervention period can be accommodated.

Assembly Paradigms

When considering which community assembly paradigm would best serve as the foundation for the restoration effort at LSP the following abiotic and biotic filters should be considered: The site was created by the CRRNJ and, therefore, lacks a long term natural history. The soils of the site are not related to the geology, they consist of “historic fill” as described above. Soil pH, moisture, nutrition, and toxicity play a role in assemblage development.

The site is isolated, with a relatively small and completely urban watershed. Propagule migration to the site is therefore limited. Wind will bring primarily highly mobile propagules of an urban origin. Animal vectors are limited primarily to avian species. Furthermore, the decision to separate the adjacent proposed saltmarsh and the freshwater wetlands eliminates the potential for propagule migration with the tides from the Upper New York Bay.

Several species of conifer and Quercus sp. have been planted on the site. The spread of these species appears limited, and successful examples of reproduction are rare. Given the nature of the site, it appears that applying an equilibrium model of assemblage development presents several challenges. Since a historic reference is not possible, and no “proxy records” exist for the site, the logical alternative would be to find the closest possible “naturally occurring” wooded area. While several potential sites exist in the area (i.e., Inwood Hill Park, New York , and Flatrock Brook, Englewood Cliffs, New Jersey), they have obvious connections to site specific geology and have a specific history of human use. Such histories are especially important in urban-forested wetlands, where anthropogenic alteration of the water cycle is extensive. In a study of 21 forested wetlands distributed throughout northeastern New Jersey, the hydrology of wetlands in urban regions reflect not only the hydrogeomorphic setting but also the unique set of disturbances resulting from the historical land use of each site (Ehrenfeld et al., 2003). Their study based upon the assumption that urban regions form distinctive landscapes for which reference sites should be available actually suggested that “the many sources of disturbance to wetlands in urban regions prevent the development of consistent classifications.” Hence, the restoration effort of the uplands and freshwater wetlands should be approached without strict reliance on a proxy or historic reference.

In addition, soil conditions are known to have a significant impact on the development of forests on mesic or xeric sites. In a comparative study of northern temperate deciduous forest after disturbance, the pattern of stand development was dependent upon soil type variations at a small scale within the study site (Liptzin, D. Ashton, P.M.S. 1999). Since the soils of LSP are extremely heterogeneous, the product of fill during several periods of the late nineteenth and early twentieth centuries, the development of the typical northeastern temperate deciduous assemblage on the historic fill would seem unlikely. Furthermore, the variation of the soils prevents the prescription of a single forest type as a proposed end point.

Assemblage Development

The process of assemblage development or redevelopment can be viewed as one that progresses through a series of filters, allowing for the selection of certain individuals from a regional pool (Diaz et. al. 1998, Weiher and Keddy, 1995). In developing management strategies that might direct future assemblage development, it is important to identify such filters, which include both abotic and biotic factors (Hobbs and Norton 2004).

Abiotic filters of significance at the site include:

1. While the site obviously has climate typical of the northeastern temperate region, the site includes divergent micro-climates that result from the gentle undulation in the former rail yard topography.
2. The absence of persistent fresh surface water favors species tolerant to periods of desiccation.
3. The fertility of historic fill is typically low, and its structure is highly compacted.
4. The site has experienced a changing frequency of disturbance. After being disturbed regularly for approximately 100 years, the site has remained fairly undisturbed for the past 40 years.
5. A 1995 soil survey revealed metals in excess of most NJDEP standards.

Biotic filters at the site include; competition between native and non-native species, as several invasive exotic species became well established early when disturbance was frequent (biological legacy) and selective grazing patterns of avian and small mammal populations. For example the population of <u>Sylvilagus</u> <u>floridamus</u> (eastern cottontail rabbit) has exhibited extreme density fluctuations.

1. Seasonal use by species typical of the Atlantic Flyway.
2. Selective grazing patterns of avian and small mammal populations.

3. Competition between native an non-native species, as several exotic species became well established early (biological legacy) when disturbance was frequent.

Recruitment to the site has changed significantly over time, as the site was transformed from an industrial area to a park. As a transportation hub during the later part of the nineteenth and early part of the twentieth centuries, the railroads and shipping traffic considerably extended the boundaries of the regional recruitment pool. The resulting assemblage contained a number of non-native species that today comprise approximately 131 acres, or one half of the site. After the railroad terminated operations, however, the area became fairly isolated, as it is surrounded by the bay and canals on three sides, and a rail line and the New Jersey Turnpike on the remaining side. Recruitment via the bay and canals is virtually eliminated as the bulkhead and armored promenade prevent immigration, while a well established salt-marsh provides an effective biological filter in the only area where elevations allow tidal flow. In addition, no freshwater streams enter the site. Hence current natural recruitment is limited to species whose seed or spore use either wind-born or avian-facilitated disbursement strategies. Avian facilitated strategies are also favored, as the park is located along the Atlantic Flyway and provides good cover within the urban context. In the Fresh Kills Landfill, less than 4k from LSP, approximately 71% of woody seedlings were bird dispersed species (Robinson, G.R., Handel, S.N. 1993). The transition from a site formally open to recruitment at a global scale, to one that is relatively isolated limits the process of assemblage development by reducing recruitment opportunities over time.

Soil Metal Contamination

Since metals in small concentrations are known to be essential for plant growth, the mechanisms for uptake are inherent, if somewhat species specific. Under normal conditions, metals are present in the soil as a derivative of the parent geologic material. Hence, plant tissue can be expected to contain some level of these metals, and in some cases at concentrations higher than in the surrounding soil. In addition to the large variation in uptake between species or genotypes of the same species, there is considerable difference in the accumulation with the various tissue types of the same plant.

Like any mineral, most plants will tolerate or have a requirement for metals at concentrations within a particular range (functional range). At concentrations below the functional range, plants will exhibit a deficiency in some aspect of their growth. At concentrations above the functional range, the plant can no longer maintain homeostasis, and metabolic functions are inhibited. For example in clones of Betula pendula taken from mine spoil sites, the internal concentration of zinc in the root tissue was maintained, regardless of increasing external concentrations. However, once a threshold value was exceeded, the plant failed to control metal uptake and was lethally inundated (Hilary and Wilkins, 1987).

Furthermore, “differential tolerance arises when different genotypes exhibit severe inhibition at different levels of the metal concerned” (Macnair 1993). For example, in a study of grasses at a copper mine site, populations of Calamagrostis epigejos from the uncontaminated site performed best on substrates of low and medium contamination level, whereas the copper smelter population was the best of three populations on the highly contaminated substrate” (Lehmann and Rebele, F. 2003). Metabolic inhibition is also exhibited when nutrient deficiency occurs as the result of increased metal concentration. Questions regarding the genetic mechanisms of tolerance, the contributions of major and minor genes, dominant and recessive allele combination activity, and tolerance representing a shift in, or an extension of, the functional range are interesting, but not particularly relevant to this study. What is important is that the species comprising the current assemblage, derived from the larger regional urban pool, exhibit metal tolerance at some level. Hence, one of the important filters during assemblage development is the tolerance to higher than normal soil metal concentrations.

In addition, several investigators have questioned whether tolerance to a particular metal infers tolerance to other similar metals? Very early work (Gregory and Bradshaw, 1965) seemed to indicate that such inferences were rare. In examining grasses found on mine spoils and seven metals found in high concentrations, their results indicated that except for a correlation between zinc and nickel, tolerance to different metals appeared to be independent. Later, Allen and Sheppard (1971), Cox and Hutcheson (1980) suggested similar patterns of metal tolerance. Their work demonstrated that populations of Agrostis tenuis (Colonial Bentgrass) collected from various mine spoil sites exhibited tolerance to copper, lead, nickel, and zinc at much higher concentrations than similar populations from pastureland. In addition, their study indicated that while a tolerance to nickel may infer a tolerance to zinc, it did not for copper or lead. They concluded that while tolerance occurs in relationship to a single metal, an individual can exhibit tolerance to several metals found in association. In addition, plants may exhibit tolerance to high concentrations of metals not found in the soil. For example, several populations of Silene vulgaris (maidenstears) exhibited a high level of tolerance to metals not found in high concentration at the collection site (Schat and ten Bookum, 1992). Species with such genetic plasticity would obviously have a competitive advantage early in the assemblage development process and are the type that probably colonized the study site; however, it should not be assumed that species will exhibit a broad range of metal tolerance because they have colonized a particular site.

Bioaccumulation of metals in plants appears to follow one of three strategies (Baker 1981). Passive strategies tend to allow accumulation of metals in the plant, proportionate to the external environment but then failure of growth or homeostasis ensues beyond a threshold. Other species are known to exclude metal ions through active processes in particular organs. For example, lead copper, and zinc metal translocation was restricted at the root endodermis (stele) of Auicennia marina (mangrove) as well as removing any metal ions in the xylem by means of storage in cell walls and vacuoles, or binding by metallothioneins or phytochelatins

(MacFarlane and Burchett, 2000 and Vesk et al., 1999). Such strategies appear to be fairly common, as metal concentration in root tissue is generally higher than in other parts of the plant. The concentrations of metals in the roots of both Spartina alterniflora (cord grass) and Phragmites australis (common reed) are known to be 4 to 1,000 times greater than in the leaf tissue (Windham, et. al., 2003). In Fagus sylvaticus (beech), concentrations of Cadium differed throughout the plant as described below (Bertels 1989):

Concentration mg/kg

Soil	0.05	1.00	3.00	7.00	21.00
Root	11.80	76.50	88.40	267.00	787.00
Cotyledons	0.09	95.00	2.45	2.29	1.93
Primary Leave	0.07	93.00	1.70	9.60	15.00
Stems	0.42	17.90	28.30	58.00	117.00

Finally, hyperaccumulation occurs when species develop highly specialized physiology that concentrates certain metals. Pteris vittata L. (ladder brake fern) is known as a hyperaccumulater of arsenic. In hydroponic experiments, as much as 744 mg/kg-1 was sequestered in the plant tissue (Ma et al., 2001). Such hyperaccumulators are the current subject of many phytoextraction projects on contaminated sites throughout the country. The ability to hyperaccumulate metals must be the result of phenotypic plasticity providing a selective advantage under extreme conditions (Turner, A.P., 1994). Since the concentration of metals in the historic fill of the site is not extreme, the presence of hyperaccumulators is possible, but not likely.

Metal Speciation

It is generally accepted that plant growth and metal uptake are related to free ion activities (McBride, 1994). In several experiments conducted using soil or nutrient solution, the free ion model was able to predict metal uptake by plants (Pavan et al., 1982; Bell et al., 1991). However, it has also been shown that free metal ion concentration is a function of soil acidity and total metal loading (Sauve et al., 1997) and, therefore, total metal concentration can provide useful information concerning plant uptake. In addition, free copper ion concentration was shown (Knight et al., 1998) to correlate well with total copper concentration, regardless of pH. Many plants are able to reduce metal ions in vicinity of their roots by changes in pH within the rhizosphere, or by exudation of reducing substances (Watsel, et al.,1991).

Experiments focusing on ion activity are generally laboratory simulations using a single plant species. They often intentionally focus on species that are known accumulators of metals or varieties that have been genetically selected for their tolerance to abnormally high metal concentrations. While little work has been done with endemic assemblages established on contaminated sites, the existing literature suggests that correlations between assemblage structure and total soil metal concentration are possible (Ge and Hendershot, 2002; Malawska and Wi£komirski, 2000).

Thesis Statement

The basic inquiry of this study concerns the role that soil metals, in concentrations typical for historic fill, have in assemblage development. We believe that vegetative assemblages tolerant of soil metal contamination, either through genetic variation and/or plasticity, have colonized the site. We hypothesize that of the three strategies previously described for growth under metal stress, assemblage development on historic fill will be dominated by species that expend less energy in maintaining stasis. Hence, metal translocation at some level would be apparent, but hyperaccumulation or total exclusion on the other hand, should not be common, at least for the dominant species. We also expected that soil metals at certain concentrations would have discernable impact on both assemblage productivity and diversity. Such impact would translate into a relationship between soil metal concentration and the distribution of the existing vegetative assemblages.

Study Objectives

In support of the GMP for the park and to test the above hypothesis the following study objectives were developed:

1. Existing vegetative assemblage structure, including species composition, density and diversity was characterized using a nested plot survey based upon historical vegetative maps and aerial photographs. (Appendix III). In addition,

the age distribution of the co-dominant tree species Betula. populifolia (gray birch), Populus deltoides (eastern cottonwood) and P. tremuloides (quaking aspen) helped to define the history of colonization and rates of growth.

2. The soil metal concentration at the root/soil interface was determined. To asses soil metal attenuation the differences between the 2005 and 1995 soil metal data are given for comparison. However, since the 1995 data contained only one measurement per site, means and variance could not be determined. These data were, therefore suspect, and not useful for analytical comparisons (Appendix IV).
3. Metal translocation to plant root, shoot/stem, and leaf tissue was examined to determine if a common mode of uptake was apparent.
4. To determine if the soil metal contaminates were impacting assemblage structure, correlations between the concentration of the soil metals and the indices of assemblage density and diversity were developed.
5. Patterns of assemblage development over the period that the rail-yard has been abandoned were determined through examination of historic photographs and field survey; these patterns were correlated with soil metal load.

Site Description

The 251-acre project site lies within LSP, in Jersey City, on the west bank of Upper New York Bay. The land which currently comprises the park was originally part of an intertidal mud flat and salt marsh that was filled for use as a railroad yard. The fill materials consist primarily of debris from construction projects and refuse from New York City, which were deposited to stabilize the surface between 1860 and 1919. Between 1864 and 1967, the CRRNJ used the site as a rail yard for both freight and passenger service. Industrial activities at the site resulted in localized hydrocarbon spills and pesticides, iron tailings, and coal ash. There is no known hydrocarbon free product within the study site, and the areas of high pesticide contamination have been identified and will be mitigated separately. Due to the use of the site for coal transport and storage, higher than normal mercury levels are unevenly distributed throughout. In 1967, CRRNJ discontinued operations at the site, and over the next few years the land was abandoned until the State of New Jersey New Jersey, Division of Parks and Forestry (NJDPF) acquired it in 1970.

The first ecological descriptions of the site were contracted soon after acquisition. The Texas Instrument (TI) Study conducted in 1976, provided several sets of data on the character of the vegetation, upland fauna and littoral benthos communities (Texas Instruments 1976).

The TI study characterized the vegetation of the site as that of very early succession, dominated by herbaceous species, many of which were exotic. At one sampling location, the plant heights were described as“uniformly low and there was no dense, rank vegetative growth” (Texas Instrument 1976). Of the species identified, Artemisia vulgaris (mugwart)had the greatest importance value, as it occurred at every sampling site. Phragmites australis had the second greatest importance value and occurred in dense stands. Ambrosia artemisiifolia (ragweed), and Panicum sp. (panic grass) were also frequently encountered. Woody vegetation was classified as sparse, with large trees entirely absent. A few small stands of P deltoids saplings existed at one of the sampling sites. Shrubs were dominated by Rhus copallinum (winged sumac) and Spirea latifolia (white meadowsweet).

During the summers of 1995 and 1996, David McFarlane, then a graduate student at Rutgers University, conducted a survey of the plants and animals of the site. The survey resulted in the first map that identified assemblage boundaries. The map was justified using aerial photographs of the site from the same time period. The assemblage structure was characterized by McFarlane as “The interior 225 acres of Liberty State Park are a strange and interesting urban wilderness. Succession, a natural process that changes the composition of biological communities in a geographic area over time, is occurring in this undeveloped site in many different locations and forms. The complex history of the site has created a number of different soil types and an interesting microtopography, and hydrologic regimes, which favor the adaptations of some organisms over others. Some portions of the site closely approximate patterns of succession observed in similar ecosystems within the region, while others have been colonized by rare and unusual species, some of which are unknown in other parts of the state. The water table is relatively high in many locations, although it is locally scarce at the surface of the soil where gravel and sand prevail. The composition of the surface soils on the site has strongly influenced the various rates of succession taking place and in many ways serve as templates for communal development" (McFarlane 1996, pg.14).

It is clear from McFarlane's survey that the assemblage structure was dominated by early colonial species. One grouping identified by McFarlane but absent in subsequent reports were the pioneer communities, areas where the gravel and mineral ash soils were clearly visible and the sparse vegetation was dominated by lichens or xerophytic grasses. Today, the remnants of these communities exist under a sparse canopy of trees, usually B. populifolia,or as part of a path that has been subject to continued disturbance.

While species richness has obviously increased over the years, direct correlations between the studies are difficult, as the methodologies and classification systems used differed. However, the following broad observation can be made from the past surveys: Overall plant species richness, measured simply as the number of species encountered, has increased by at least 71% over the past 30 years, despite the development loss of approximately 50% of the site to development. Forested areas have increased dramatically in size and density over the past thirty years, and by 2005 had occupied approximately 56 acres of the project site. Shrub assemblages also appear to have increased in both size and diversity and they currently occupy at least 32 acres. The difference between the McFarlane and USACE surveys in 2003 is difficult to estimate, as McFarlane did not differentiate between shrub and woodland, due to the very early stage of the woodland assemblages. Invasive exotic species have been prevalent throughout the site's history. However, the structure of these assemblages has changed. Assemblages dominated by P. australis apparently decreased in size within the study area between 1996 and 2003, perhaps by as much as 19%. Conversely, in the dredge spoil area that was graded in 1993 and planted with rye grass, P. australis has invaded. While direct comparisons between the surveys are difficult estimates of the shifts in vegetative guilds between the years of 1995 and 2003 are given in Figure 1.

Soils

Due to the Park's history as an industrial rail yard, an extensive investigation into soil contamination was conducted in 1995. The areas of the Park that were sampled include Middle Cove, the Freight Yard Area and the Waterfront Area. The Freight Yard Area has been divided into several sections: the Soil Staging Area, the Dredge Spoils Area, the Central Area, and the Sewer Line Area (Appendix V).

The existing upper soil horizon has developed from overgrowth-derived organic matter since incremental abandonment of the railroad yards. This horizon varies from five (5) to ten (10) cm. in depth, but in both herbaceous and woody plants, the root are concentrated in this horizon. Deeper soils are virtually all anthropogenic fill materials of varying texture. The geotechnical cores taken by the USACE (USACE EIS 2004) and by NJDEP (NJDEP 1995), indicate alternating lenses of sand and clay to a depth varying between 15 and 30 feet below the current surface. "At this depth, the pre-1860s (pre-fill) natural alluvial soils of the estuary are reached. Small amounts of peat, indicting former approximate depths of a relict salt marsh are found in some locations. The US Department of Agriculture (USDA) Soil Conservation Service (SCS) has not prepared the Soil Survey Report of Hudson County. However, a preliminary mapping based on field observation was published in 1952. In general, natural soils (sediments) in the vicinity of the project area were derived from stratified drift and wash from glacial till. More specifically, the project area would have been inundated alluvium and muck prior to landfill creation" (USACE EIS 2004).

Hydrologic Regime

Since hydrology is a major filter determining assemblage distribution, two measured conditions of the site can be considered. First, as part of the geotechnical analysis for the USACE FS, twelve permanent piezometers in existing drill holes were installed for long-term groundwater level monitoring purposes. An additional eight temporary piezometers were also installed in selected existing drill holes. Groundwater depths ranged from about .4m below existing grade to about 1.7m below existing grade, as measured from the twenty piezometers installed throughout the site. Maximum fluctuations in groundwater levels up to 0.95m were observed in the piezometers. Groundwater gradients between piezometers were typically very flat. Groundwater flow direction and average gradient calculations using LMNO Groundwater Flow Direction and Gradient Calculator software suggests that groundwater flow is oriented approximately N35^{o} and discharges towards the adjacent Morris Canal (Figure 2).

In addition, the N.J. State regulated wetlands of the site have been mapped and the topography of the site has been archived (USACE 2004). Each regulated wetland is labeled according to NJDEP wetland regulations. These wetlands are typically perched and seasonally flooded Palusturine systems, dominated by both herbaceous and woody assemblages. A summary of the description of these types and their respective wetland indicators is provided as Appendix VI.

General Methodology

Study Design

The focus of this study is on the section of the site, approximately 110 acres, which according to the GMP will not be disturbed. In these areas the intent of the DPF is to use mostly non-invasive practices to monitor/direct assemblage development to achieve the project's stated objectives. Within these boundaries, the study can be divided into several integrated sections: Vegetative surveys conducted in both 1996 and 2003 yield information on assemblage boundaries. Density, as measured by stem count or basal area and diversity, measured by the Shannon Index based on percent composition, were defined using a nested plot survey. In addition, differences in growth based upon basal area and core samples for dominant tree species were determined. Productivity was also measured at the assemblage level, through the development of a Normalized Vegetative Difference Index (NDVI) using satellite imagery (Appendix VII). The resulting assemblage boundary maps and characterizations established the context for the study.

The soils of the area had been extensively examined (New Jersey Department of Environmental Protection, 1995). These data were used to develop a profile of the targeted soil metals and examine any obvious relationships between soil metals and vegetative density and diversity. The results are included as Appendix VIII.

Samples of plant tissue from the dominant species of the assemblages were tested for concentration of the targeted metals. This information was used to determine if the dominant species used a common metabolic strategy, passive uptake, exclusion or hyperaccumulation in areas of abnormally high soil metal load. If one strategy was used by most dominant plants a uniform approach to vegetative management could have been recommended. Finally, productivity at the specimen and assemblage levels and assemblage diversity were then compared to soil metal load to determine the impact of the soil metal contaminants on vegetative assemblage development.

Experimental Protocol

Assemblage Definition

The vegetation assemblages within the LSP restoration site were identified and categorized based on the descriptions and associations provided in Ecological Communities of New York State (Edinger et al. 2002). The assemblage descriptions were slightly modified to characterize the disturbed and urban conditions of the site more accurately. The identified upland assemblages include, Successional Northern Hardwood, Successional Shurbland, Successional Old Field, Maritime Shrubland, Maritime Grasslands, Common Reed/Mugwart, Lawn, Floodplain Forested Wetlands, Shrub Swamp Wetland, Shallow Emergant Marsh, and Common Reed Dominated Wetland.

Using the Geographic Information System (GIS) program Arc-View, a 10m X 10m grid system was applied to the site, oriented so that the axis is parallel to transects used in the 2003 ERI performed by the USACE. Sampling sites were then determined by comparing assemblage boundary maps, aerial photography, and soil test pit location to achieve adequate representation of the several targeted assemblages and to ensure tight referencing to previous soils information (Figure 3).

In several cases, sites TP-10,TP-16, and TP-40, the predetermined location was no longer representative of the description and the location was adjusted to gain a better representation of the desired assemblage. In these cases, the sample site may be up to 10m away from the original 1995 soils test pit. In one case, site 48, the soil test pit had been covered and the area mitigated with several feet of clean fill. The closest possible sample location, approximately 26m south, was used. Tenth (1/10) hectare sample plots (20mx50m) were delineated at each site. In cases where tenth-hectare sample plots were not possible, as the target assemblage did not encompass that much area, adjacent ten square meter quadrates were delineated within the boundaries and added together to get the best possible representation of the assemblage. In all cases species were identified until at least 95% repetition was achieved. In herbaceous assemblages, the representative 10m2 quadrate was sampled using five meter two plots placed at the four corners and one in the center. All sampling sites were located using a Corvalilis MicroTechnology MC-GPS and identified by their soil test pit number.

The following field data were collected at each site, and a field log book has been maintained: Site Reference Number, NAD84 Coordinates, Community Description, and General Comments, Aspect of Slope, Position on Slope, Species, Stem Count or Tree Diameter.

At sites dominated by herbaceous communities, stems were counted manually. In tree dominated assemblages, standard d-tape was used to determine tree diameter at breast height (dbh approximately 1.4m from the ground). D-tape is graduated into diameter equivalents using the relationship circumference = pi x diameter. “Thus, the diameter inch equivalents are marked on the tape at intervals of 3.1416 in. to a precision of 0.1 in. Deviations from "true" circumference of the tree result from rough bark and surface irregularities. The magnitude depends on bark characteristics of the tree and species and the extent of surface irregularities. When irregular trees are measured with d-tapes, convex deficits occur where the tape passes over areas of the tree surface that have depressions. The resulting diameter measurement will produce a basal area that is larger than the true basal area because the d-tape assumes the circumference of the tree measured is a circle" (Morgan and Williams, 2002). The use of calipers reduces this bias, however the difference between the two methods of measurement is not statistically significant (Morgan and Williams, 2002). In addition, the trunk of B. populifolia, one of the sites dominant species, often assumes a triangular shape making the use of calipers or the Biltmore Stick difficult and less reliable.

Tree Assemblage Age

As the shift in the vegetative assemblage is dominated by the transition from herbaceous to woody species, an examination of tree age distribution was undertaken. Assemblages dominated by P. deltoids and B. popalifolia were chosen for age determination based upon their overall size, as it was assumed that the larger assemblages were established earlier. The soil metal concentration, using the 1995 soils data, was also considered to provide a range of conditions. Within these assemblages, samples from various size classes were taken for age determination. Conventional diameter class, (k) is defined in an absolute scale (e.g., 0-1 cm for k=1, > 1-2 cm for k =2, etc.) (Lorimer, 1980). Individual trees at each sampling site were chosen using directional transects. The initial direction was determined randomly with a coin toss and the sequence of North, East, South, and West was then followed. Trees that were cored at 1.4 meters were permanently labeled and their geographic positions recorded for future reference. The smallest size classes were destructively sampled to yield a complete cross section.

The core samples and cross sections were aged using a dissecting microscope. Samples were stained and examined as described in chapter four. Annual growth increments were measured to the nearest 0.01 cm and recorded. After cross dating using signature years (Fritts 1976 or 1979), annual increments were averaged to determine a mean growth chronology for the target species. Diameter measurements were then correlated with ring measurement to develop a representative size/age model.

Productivity

While stem counts and basal area measurements provide a method of comparison for similar assemblages, in order to compare productive across all of the sampled assemblages a universal measurement of productivity is required. The application of GIS and Remote Sensing Technology (RST) offers the potential to characterize spatial and temporal heterogeneity across the entire site. The principle of this assessment is based on the fact that changes in the growing conditions of vegetation induce modifications in biochemical composition (e.g. chlorophyll concentration), physiology, and canopy architecture. These modifications influence vegetation reflectance and can, thus be used indirectly as an indicator of productivity.

Reliable, high-resolution vegetation reflectance spectra in the visible to near-infrared (400-1350 nm) were obtained from existing airborne or satellite imaging. Surfer 8.0 software was used in combination with the Kriging and Block Kriging method of analysis to provide a reliable estimate of the spatial distribution of chlorophyll. A regression analysis comparing the distribution of chlorophyll against the concentration of metals in both the soils and plant tissue was then performed.

Targeted Metals

While the 1995 soil survey included twenty-three metals, among other contaminants, it was appropriate to develop a more focused list of target metals. Hence, the following two criteria were used to determine appropriate targeted metals: 1) Are there known risks of bioaccumulation, and 2) is the concentration high enough, above DEP Soil Screening Criteria, to be of concern? Those metals that satisfied both criteria were chosen for examination. In addition, sodium was chosen as its impact on plant metabolism is known to be considerable and we suspect that the ground water may be influenced by the Upper New York Bay. Aluminum was used as a benchmark for metal analysis and chromium and mercury, due to the intense public concern, were also chosen.

In addition, to the above criteria, the metal concentrations were examined to see whether they were significantly above the published means (Saunders 2002) for the State of New Jersey. Published means for the Urban Costal Plain was used rather than the Urban Piedmont as the soils, sand gravel, and grit more closely resemble a costal plains complex. An extremely liberal r^2 value of 0.6 was used to be inclusive of even weak correlation. Finally, a comparison between the individual soil metals and diversity or density of the sample sites was done to determine if there was an obvious significant relationship (also Appendix VII). The only metal to satisfy both of these criteria, and had not already been selected, was Vanadium. While it's .083 regression may be the result of a nexus with either another metal or physical parameter, the high correlation deserves consideration.

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Figure 1: Shifts in Assemblage Distribution (Acres Cover) between the yearsn1995 and 2003.

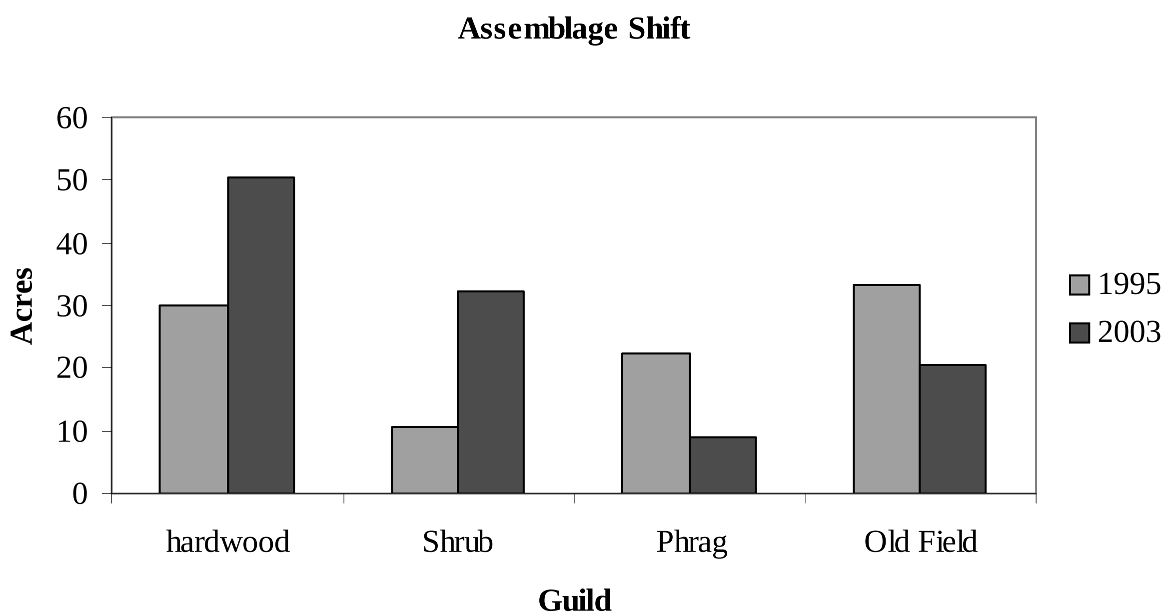
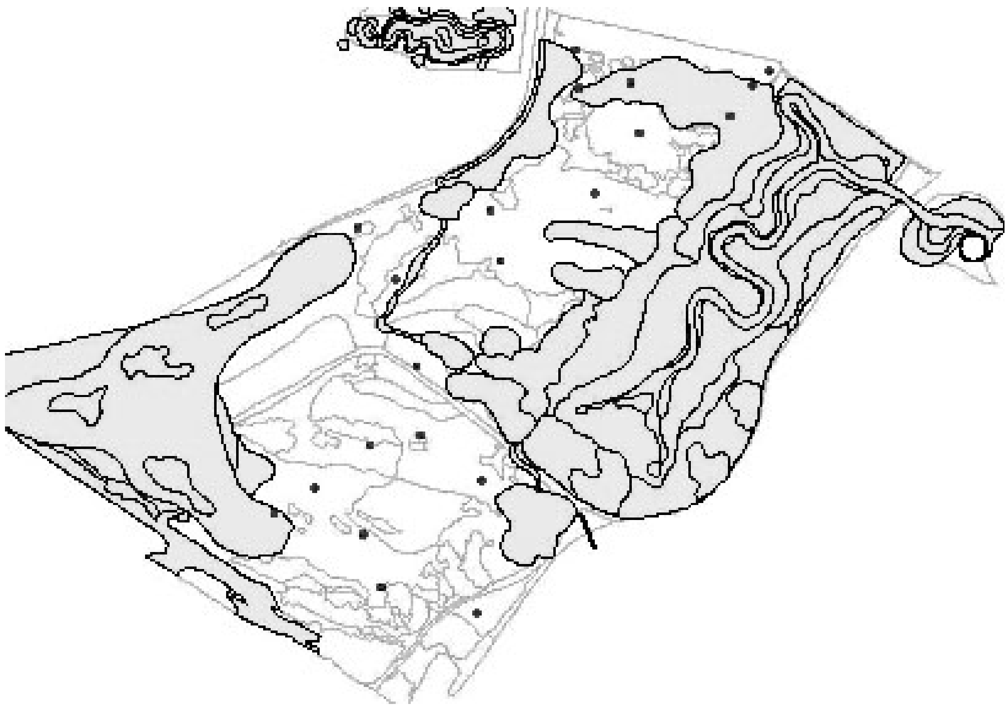


Figure 2:Groundwater Contour Map of the 15 November 2003 Groundwater Readings and regulated wetlands in white (USACE FS 2004).

Figure 3: Sample Locations: each dot represents a sampling site. Lines represent assemblage boundaries Shaded areas represent those areas to be mitigated with clean fill and were not sampled with the exception of three sites on the northern boundary where the proposed plan is questionable.



Chapter II

Soil Metal Concentrations and Vegetative Assemblage Structure in an Urban Brownfield[1]

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Abstract

Anthropogenic sources of toxic elements have had serious ecological and human health impacts. Analysis of soil samples from a brownfield within Liberty State Park, Jersey City, New Jersey, USA, showed that arsenic, chromium, lead, zinc, and vanadium exist at concentrations above those considered ambient for the area. Accumulation and translocation features were characterized for the dominant plant species of four vegetative assemblages. The trees Betula populifolia and Populus deltoides were found to be accumulating Zn in leaf tissue at extremely high levels. Betula populifolia, P. deltoides and Rhus copallinum accumulated Cr primarily in the root tissue. A comparison of soil metal maps and vegetative assemblage maps indicates that areas of increasing total soil metal load were dominated by successional northern hardwoods while semi-emergent marshes consisting mostly of endemic species were restricted primarily to areas of low soil metal load.

Key words: Translocation Assemblage Structure; Metal Accumulation; Brownfield; Railyard; Historic fill.

Capsule: This study yields insight into the impact of metal contaminated soils on vegetative assemblage structure and development.

Introduction

In many areas throughout the world, industrial activities have resulted in contaminated urban landscapes. While many severely contaminated sites have been identified and mitigated, less contaminated sites ("brownfields") have proven more difficult to remediate. As a result of industrial land uses, brownfield soils typically contain high concentrations of trace metals such as cadmium, copper, zinc, lead (Dudka et al., 1996) and others. These elements are often adsorbed or occluded by carbonates, organic matter, iron-magnesium oxides and primary or secondary minerals (Adriano, 1986; Ross, 1994). However, despite considerable advances in contaminated land assessment in recent years, there are still obvious difficulties in identifying whether or not, and under what conditions, soil contamination should be a matter of concern (French et al., 2006). Many studies addressing metal contamination focus on hot spots (Ramsey and Argyraki, 1997; Sastre et al., 2001) offering engineering solutions that can be relatively expensive. Phytoextraction or phytostabilization methods offer low-cost and effective, albeit long-term, solutions (Canell, 1999), while the developing vegetated assemblage provides aesthetic improvement and economic benefits (Paulson et al., 2003). However, if these green methods are to be employed, an understanding of the ecological risk associated with contaminant assimilation is critical.

Bioaccumulation of metals in plants appears to involve one of three strategies (Baker, 1981). Passive strategies 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. A second strategy is the exclusion of metal ions from particular organs via active processes. For example, Cu, Pb, and Zn translocation is restricted at the root endodermis (stele) of Avicenna marina (mangrove), while metal ion removal occurs in the xylem of Eichhornia crassipes (water hyacinth) by means of storage in cell walls and vacuoles or by binding to metallothioneins or phytochelatins (Vesk, et.al., 1999; MacFarlane and Burchett, 2000). Such strategies appear to be fairly common, as metal concentration in root tissue is generally higher than in other parts of the plant. Metal concentrations of metals four to 1,000 times greater in roots than in leaf tissue have been found in both Spartina alterniflora (salt marsh cordgrass) and Phragmites australis (common reed) (Windham, et al., 2003). Finally, hyperaccumulation occurs when species develop highly specialized physiological mechanisms that concentrate certain metals. For example, in hydroponic experiments using the fern Pteris vittata (Ladder Brake Fern), concentrations of As as high as 744 mg/kg were sequestered in the plant tissue (Ma et al., 2001). Thus, many studies discuss plant physiology and the ecology of metal uptake and accumulation under controlled conditions or in plantations. However, there is a lack of information concerning these phenomena in endemic assemblage development.

Liberty State Park (LSP) is the site of an abandoned rail yard with As, Cr, Cu, Pb, V, and Zn concentrations above those considered ambient for New Jersey’s soils (Saunders, 2002), and in several cases above the New Jersey Department of

Environmental Protection (NJDEP) soil cleanup criteria (NJDEP, 1999). Variability in substrate material deposition has created a patchy distribution of soils and cinders, which has influenced colonization by plants since the closing of the rail-yard over the last 36 years. This has produced a unique mosaic of community types within the restoration site (United States Army Corps Engineers, 2004).

The objectives of the study were to determine the translocation characteristics of soil metal contaminants in a naturally re-vegetated urban brownfield and to determine whether vegetative assemblage distribution is related to soil metal load. We had hypothesized that 36 years of assemblage development on the historic fill of the rail yard would favor species that expend less energy (passive uptake model) to maintain homeostasis. Hence, metal translocation in the dominant species may be apparent at some level and neither hyperaccumulation nor total exclusion would be common. We also hypothesized that there would be a relationship between soil metal concentration and the distribution of the emerging hardwood assemblage.

Materials and Methods

Study Area

The 251 acre (102 ha) project site lies within LSP, in Jersey City, NJ, on the west bank of Upper New York Bay (center of site = 40º 42’16” N x 74º 03’17” W). The land was originally an intertidal mud flat and salt marsh that was filled between the years of 1860 and 1919 for use as a railroad yard for the Central Railroad of New

Jersey. The fill materials consisted primarily of debris from construction projects and refuse from New York City. After 1967, when the railroad discontinued operations, the site remained isolated and undisturbed. The site was transferred to the New Jersey Division of Parks and Forestry in 1970. Due to the use of the site for transport and storage of many goods, soil metal concentration was and remains relatively high and unevenly distributed.

The vegetative assemblages within the study site were identified and categorized by the United States Army Corps of Engineers (USACE, 2003), based on the descriptions and associations provided in Ecological Communities of New York State (Edinger et al. 2002). The identified upland assemblages included successional northern hardwood (SNH), successional shrubland (SSB), successional old field (SOF), maritime shrubland (MS), maritime grasslands (MG), common reed/mugwort (CRM), floodplain forested wetlands (FFW), shrub swamp wetland (SSW), shallow emergent marsh (SEM) and common-reed-dominated wetland (CRW) (USACE, 2003). Figure 1 shows the vegetation assemblage map, created with ArcGIS, ArcMAP® v. 9.1.

Using the Geographic Information System (GIS) software Arc-View 3.2®, a 10 m X 10 m grid system was applied to the site, oriented so that the axes were parallel to transects used in the 2003 natural resources inventory (NRI) performed by the USACE. Sampling sites were then determined by comparing assemblage boundary maps and aerial photography achieving adequate representation of several targeted assemblages. Twenty-two sampling sites were selected within the dominant assemblages: two sites from the SOF, four each within the CRM and the MS communities, and twelve within the SNH (Figure 1).

Sampling

During the summer of 2005, soil samples were collected in triplicate at one meter spacing and their GPS (Corvallis Microtecnology MC-GPS, accuracy 1m) coordinates were recorded. Soil sampling depth corresponded with the depth of greatest root concentration. Gravel and visible plant material (roots and leaves) were removed from soil samples placed in clean, new polypropylene containers, and stored at $4^{o}C$. The sites were also examined for depth of soil above the original rail yard fill by measurement of visual soil layering in soil cores. A LaMotte field soil pH meter was used for analysis of soil pH.

To ensure good correlation between soil and tissue samples, all biological samples were collected from the dominant species at each sampling site. Therefore, plant tissues were taken from three woody species: Betula populifolia (gray birch) (N=8), Rhus copallinum (winged sumac) (N=5), Populus deltoides (eastern cottonwood) (N=3), and one herbaceous species, Artemisia vulgaris (mugwort) (N=4). Roots were visually traced from the bole and excavated using a hand spade. Root fibers were collected, loose soil was removed with distilled water, and 10-15 g (wet weight) samples were stored in clean polypropylene containers. Woody tissue was collected by cutting a cross-sectional ‘cookie’ at approximately one meter height, from which a wedge representing all growth years and of at least 25 g (wet weight) was taken from the specimen. Representative leaf samples were also collected from each specimen. With woody species, five grams (wet weight) leaf samples were collected from the upper, middle, and lower sections of the plant to ensure that the samples were representative of the entire plant. Herbaceous plants were clipped above the ground and the entire plant was collected. All samples were given archive identification numbers and delivered for analysis to the University of Medicine and Dentistry of New Jersey (UMDNJ) with accompanying Chain-of-Custody sheets.

Analysis

Each soil sample was mixed thoroughly and sieved to <125 µm. Sub-samples (1 g) were dried at 60 °C for 48 hours (constant weight). Aliquots, weighed on a calibrated analytical balance to the nearest milligram (mg), were ashed in a crucible in a muffle furnace with temperature ramping overnight to 450 °C, then re-weighed to estimate organic carbon (‘loss on ignition’). Additional dry aliquots of ~0.5 g were weighed to the nearest milligram, treated with 10 ml trace-metal grade HNO^3, and acid extracted in Teflon bombs in a MARS-5 (CEM Corp.) programmed microwave instrument at >170 ° C for 30 min. These acid extracts were reduced to a minimum volume on hot-plates and re-diluted with 1% HNO^3 for analysis by Atomic Absorption Spectroscopy (AAS). A method blank and a National Institute of Standards and Technology (NIST) Standard Reference Material (SRM) 1944 (“New York – New Jersey Waterway Sediment”) sample were run simultaneously with each set of 12 soil samples. These samples were analyzed for Cr, Cu, Pb, V, and Zn by flame Atomic

Absorption Spectrometry AAS in a Perkin-Elmer Model # 603. The cold-vapor AAS method of Hatch and Ott (1968) was used for Hg analysis with a MAS-50D mercury analyzer (Bacharach, Inc.). Arsenic was determined in the presence of a $Mg(NO^3)^2/Pd(NO^3)^2$ matrix modifier by graphite furnace AAS in a Perkin-Elmer Z5100 with Zeeman background correction.

Biological samples were dried for 48 hours at 60 °C to a constant weight, weighed to the nearest mg and distributed as 0.3 g sample of each triplicate. They were then first treated with 30% H^2O^2 to mineralize cellulose, then prepared similarly to soil samples. A method blank and a standard reference material (NIST SRM1573a – tomato leaves) were run simultaneously with each set of 12 plant samples. Metals and metalloids in biological samples were analyzed similarly to the soil samples. However, when there was <1% absorption by flame AAS, a graphite furnace AAS with Zeeman effect was employed for increased sensitivity (Perkin-Elmer Z5100 instrument). Minimum detection levels (MDLs) were calculated for each metal by taking three times the standard deviation of the values measured for the blanks (Table 2). For statistical purposes, values <MDL were expressed as one-half the MDL, e.g., when calculating a mean.

Data treatment

Raw datasets were analyzed using several software packages. The descriptive statistical parameters were calculated with SPSS (release 11.5, SPSS Inc.) and Minitab® (Minitab release 12.23). Vegetation-metal distribution maps and the sampling plan were performed with ArcView® (version 3.2 ArcGIS, ESRI Co.) and ArcMap® (Version 9.1, ArcGIS, ESRI).

To estimate distributions of soil metals, the individual soil data were kriged. The considerable standard deviation for each metal data set was accommodated for by using block kriging, which estimates the average value of the rectangular blocks centered on the grid nodes (Stein, 1999). The block kriged maps were created using Surfer Surface Mapping Software® (release 8.0., Golden Software Inc.). Since all the metal data were highly skewed (Table 2.), data were transformed before performing kriging. In order to normalize the data distribution and provide more stable variograms, we selected logarithmic and rank order transformations. Logarithmic transformation was performed as described by McGrath et al. (2004). Since we wished to provide a prediction for each metal in its original concentration, back-transformation of kriging results was necessary and therefore carried out by exponentiation of the logarithmic formula.

The rank order transformation was performed as described by Juang et al. (2001) and the results were also back-transformed, using the reverse function of the linear regression, performed between the original metal data and the ranks as published by Wu et al. (2005). To evaluate the performance of the kriging on differentially transformed data sets, the mean error (ME), root-mean square error (RMSE) and the coefficient of determination (r^2p) were calculated as described by Isaaks and Srivastava (1989) (Table 6). In addition to developing the kriged map for each soil metal, a total soil metal load index, summarizing the rank-ordered values, was calculated and kriged. The index provided a way to assess the impact of total soil metal load on vegetative assemblage development.

Kriged maps with the smallest ME and RMSE values were entered in the area calculation procedure. The contour maps obtained from Surfer were transformed into vector data and analyzed in ArcGIS environment. Based on the 2003 vegetation map, areas of assemblages were compared with calculated areas of different cumulative metal loads and graphed against the summarized rank-ordered values (Figure 3). Considering that all the metal data were positively skewed, sixteen additional sites were sampled. Besides avoiding the misclassification due to the over- or under estimation from kriging, the new sites were defined as extensions of the sites of high soil metal concentrations. Therefore, around these sites, i.e., TP-25, TP-14/16, and TP-16, four new sites were selected in a 20 m diameter circle. We also add five new sites (TP-17, TP-42, TP-42/36, TP-41/42, and TP-47) in order to close the eastern border of the study area.

During the kriging procedure, the sites surrounding the hot spots (twelve sites) were left out of the analysis and the values of the remaining 27 sites were kriged. The additional twelve sites were used to calculate the ME and RMSE for the kriging results.

Results

General Findings

We compared soil depths, assemblages, pH and organic content. The average soil depth was greatest in CRM communities with values between 10 and 20 cm. The SNH and MS communities had a mean of about 10 cm depth, while the SOF communities showed greater variation with generally shallow soils (Table 1). Tukey analysis, however, indicate no statistical difference in soil depth between assemblage types (F = 0.49). At each location, the root systems exhibited greater development, more lateral than vertical, just above the original densely packed rail fill.

The pH varied considerably, between 5.0 and 6.8 at the SNH sites and between 5.4 and 7.4 at the MS sites, but remained relatively consistent for the herbaceous assemblages (CRM, CRW and SOF), between 6.0 and 6.8 (Table 1).

Soil organic contents ranged from 8.81% (at TP-1) to 52.63% (at TP-14) with means of between %20 and 32%. A comparison of assemblage types for soil organic content indicated a significant (P = 0.043) difference between only the CRM and SNH assemblages. The difference appears to be the result of the undulating topography created by the rail beds and the accumulation of organic material in the lower elevations.

Soil Metal Concentrations

The statistically representative concentrations for each metal are presented in Table 2. In order to assess potential ecological risks, soil metal concentrations were compared with the threshold values found in the Soil Cleanup Criteria (SCC) recommended by the New Jersey Department of Environmental Protection (NJDEP, 1999) and to a new list, the Generic Soil Remediation Standards (GSRS), proposed in 2004 (NJDEP, 2004), and now (July 2007) in the final review (both listed in Table 3). Of these two sets of criteria, the GSRS is much more rigorous for As, but threshold values for the other metals have increased. The GSRS for Zn is more than ten times higher than the SCC for Zn. The distribution of soil metal concentrations and total metal load (rank sum index) are presented in Figure 2.

The concentration of As exceeded the GSRS in every soil sample. Cr exceeded criteria at only two locations (4.5 %). Pb exceeded the standard in 68% of the samples, but not in the assemblages dominated by A. vulgaris. Cu, Hg, V, and Zn levels remained below the critical value at each site. The highest concentrations of As, Cu, Pb, and Zn were found at one location, TP-25 (Figure 2), a site within the SNH assemblage with a shallow root zone (0-10 cm) and relatively high organic content (31%). V was the only metal to exhibit any correlation with soil pH (Table 4), indicating that enhanced metal leaching of As, Cr, Cu, Pb, and Zn was not associated with pH. This observation is also supported by samples taken from deep test pits that penetrated through the historic fill into the original sediment, which had only background levels of metal (data not shown, NJDEP 1995).

Metal Translocation

While metal translocation differed considerably among the dominant plant species, there were some recognizable trends (Table 2). Results indicate that As is the most stable of the metals studied, exhibiting little translocation into plant tissue. Cr exhibited a similar trend, with slightly higher rates of accumulation in the root tissue of all four dominant species. Extremely high concentrations of Zn were observed in leaf tissue of P. deltoides and B. populifolia, except at sites TP-18 and TP-28 (Figure 3). In several cases, the bioaccumulation of Zn in these leaf samples exceeded soil concentrations by orders of magnitude. B. populifolia, P. deltoides, A. vulgaris, and R. copallinum all accumulated significant concentrations of Cu and Pb in root tissue and Zn in leaf tissue (Table 2). Also, soil and root Cr levels had a strong linear correlation in both B. populifolia (N = 8, $r^2 = 0.99$, p < 0.01) and R. copallinum (N = 5, $r^2 = 0.62$, $p < 0.05$). No other statistically significant relationships between soil and root metal concentration were observed.

Assemblage Development:

To assess the relationship between assemblage structure and soil metal concentrations, the area ratios of each assemblage type (percent of area covered) was compared with total soil metal load (rank sum index) (Figure 3). These data indicate that there was a significant relationship between the total soil metal load and percent cover for both the SNH ($r^2 = 0.62$, $p < 0.01$) and SEM ($r^2 = 0.49$, $p < 0.01$). Interestingly, the percent cover of the SNH assemblage increased as the metal load increased, while the opposite was true for the SEM assemblage. To assess trends in successional development, the percent cover for each assemblage type was calculated using geo-referenced data from 1995 and 2003. The results indicate that the SNH assemblage dominated by B. populifolia or P. deltoides now occupy nearly one-third of the study area, a 23% increase over eight years.

Discussion

Metal Translocation

Many plants have developed mechanisms to limit the translocation of metals across the root endodermis by storing them in cell walls and vacuoles and/or binding by metallothioneins or phytochelatins (MacFarlane and Burchett, 2000). Therefore, metal concentrations tend to be greater in root tissue than in other parts of the plant. For example, concentrations of metals 4-1,000 times greater in roots than in leaves have been found in both Spartina alterniflora (saltmarsh cordgrass) and Phragmites australis (common reed) (Windham et al, 2003). Our data support such trends for the metals measured with the exception of Zn. Root metal concentrations of Cr, Cu, and Pb were lower than in the associated soil. In addition, these metals were translocated to the aerial parts of the plants, but the concentration was at least an order of magnitude less than in the root tissue. Only B. populifolia demonstrated the ability to translocate Pb to leaf tissue. However, B. populifolia and P. deltoides translocated Zn at high levels to leaf tissue, in some cases to concentrations that were an order of magnitude greater than soil concentrations (Figure 4), thereby disproving the null hypothesis (for Zn only) that metal translocation would be apparent at some level (passive uptake model) and that neither hyperaccumulation nor total exclusion would be common.

A model for As uptake has been developed for Brassica juncea (Indian mustard) (Pickering et al, 2000), involving thiolate donors (e.g., glutathione or phytochelatins). Using a tolerant variety of this species, they demonstrated the ability for uptake and storage in the root, with minimal translocation to the aerial sections of the plant. Similar results were obtained in this study, as all four species had concentrations of As in root tissue that were significantly less than soil concentrations. In addition, B. populifolia and P. deltoides demonstrated minimal As uptake to leaf tissue at most sites. Interestingly, at two sites with relatively high As soil concentration, both dominated by P. deltoides, the patterns of uptake were different. At TP-10, it appears that the relatively high As soil concentrations were reflected in roots of P. deltoides, (soil = 197 μg/g, root = 20.2 μg/g). However, the root tissue of P. deltoides at TP-25 (soil = 270 μg/g, root = 3.3 μg/g) did not exhibit proportionately high concentrations in the root tissue. Perhaps the high concentration of Zn in the root tissue at TP-25, more than twice that at TP-10 (270 μg/g at TP-10 vs. 604 μg/g at TP-25), and the greater (than most transitional metals) ability of Zn+2 to form complex ions (Bohn et al, 1985) due to its high binding constant had a competitive influence, reduced the potential for As uptake at site TP-25. Such antagonisms between are known to occur in Pseudotsuga menziesii (Douglas fir) and Thuja plicata (Western Red Cedar) (Zasoskia et al, 1990).

Cr soil concentrations were below the GSRS and SCC criteria throughout the site. The correlation between root and soil concentrations regardless of species was strong (N = 21, $r^2 = 0.73$, $p < 0.01$), and as mentioned previously, was strongest in B. populifolia (N = 8, $r^2 = 0.99$, $p < 0.01$). It is generally accepted that plant growth and metal uptake are related to free ion activities (Hough et al, 2005; McBride, 1994). In several experiments conducted using soil or nutrient solution, the free ion model was able to predict metal uptake by plants (Pavan et al., 1982; Bell et al., 1991). However, it has also been shown that free metal ion concentrations are a function of soil acidity and total metal loading (Sauvé et al., 1996). In this study, we demonstrate that total soil Cr concentration could be used as an effective predictor of root Cr concentration.

Soil Cr concentrations were generally at least an order of magnitude greater than root concentrations. Similar results have been obtained in S. alterniflora and P. australis (Windham et al, 2003). In addition, it has been reported that Cr does not typically translocate well between roots and the aerial parts of the plants, as there appears to be preferential storage of Cr in root cortex cell vacuoles (Kabata-Pendias and Pendias, 2001). On the other hand, recent studies with Populus x euramericana clone I-214 found Cr(III) translocation to the stem that was seven times greater than that in the soil concentration; however, the concentration of Cr(VI) was always below the analytical threshold (Giorgio and Sebastiani, 2006). Our data support the earlier findings, since Cr was generally excluded from aerial plant sections.

Cu soil concentration was generally below New Jersey soil screening criteria. However, within the range of 65-125 μg/g, it has been found to be phytotoxic (Ross, 1994) (Table 3). Bioaccumulation of Cu has been well documented in both aquatic (Catriona et al, 2004; Mehta et al, 1999, and others) and terrestrial plant species (Lin and Wu, 1994; Gardea-Torresdey et al, 2004). These studies indicate higher levels of accumulation in the root tissue. Similar results were obtained in this study: Cu concentrations in roots tend to be about an order of magnitude lower than that of the soil and approximately an order of magnitude greater than that of leaf or stem tissue.

Soil Pb concentrations regularly exceeded both New Jersey screening criteria. The spatial distribution and translocation rates for Pb were similar to those for Cu. Pb uptake is generally low in soils with pH over 5.5 (Blaylock et al 1997). In this study, with soil pH values between 5.2 and 6.6, concentrations of Pb in roots tended to decrease by at least an order of magnitude in all four dominant species, as compared to the soil samples. Stem tissue had concentrations that were more than an order of magnitude less than root tissue, which is supported by studies of planted plots of B. pendula and P. deltoides grown on contaminated land in the United Kingdom. There, foliar and stem Pb were found to be similarly low (French et al, 2006).

Only two sites, TP-43/14 and TP-48, both of which were dominated by B. populifolia, exhibited detectable levels of Pb in the leaf tissue. Pb is known to accumulate in the salt marsh macrophytes S. alterniflora and P. australis and exhibit higher concentrations in roots than in aerial sections of the plants (Windham et al, 2001). Several terrestrial plants, including agricultural crops, have been studied for their ability to accumulate Pb and their possible use in phytoextraction mitigation (Haung and Cunningham, 1996). The aforementioned studies demonstrated significant Pb accumulations in root tissue of several species, with the highest rates of accumulation, 24,000 µg/g, observed in Ambrosia artemisiifolia (ragweed). While uptake of Pb in aerial sections of plants could be facilitated by using synthetic chelates (Huang and Cunningham, 1996), untreated samples yielded results similar to those obtained in our study.

Zn accumulated at the greatest concentrations within the sampled plant tissue. Collectively, the mean concentration of Zn in roots, 301 µg/g, was greater than the mean soil concentration of 228 µg/g. Concentrations in stem tissue were slightly lower. The greatest concentrations, however, were found in the leaves of B. populifolia and P. deltoides. The mean concentration of Zn in leaves of B. populifolia exceeded soil concentration by more than an order of magnitude (leaf = 905 µg/g, soil = 87 µg/g). The Zn concentration in the leaf tissue of P. deltoides exceeded the soil concentration by a factor of three (leaf =1553 µg/g, soil = 556 µg/g). The general mechanics of Zn uptake in roots and translocation to shoots have been described (Lasat, 1996). Translocation of Zn has been well studied in Avicennia marina (mangrove) (MacFarlane and Burchett, 2000), which can translocate Zn to leaves at levels well above nutritional requirements. Birch species (B. papyrifera, B. pendula, B. pubescens) have been previously reported as being tolerant to and accumulators of metals, particularly Zn and Pb (Prasad, 1999; Margui et al., 2007).

Clones of B. pendula taken from mine spoil sites have high concentrations of zinc in the root tissue that are maintained, regardless of increasing external concentrations (Hillary and Wilkins, 1987), indicating a high threshold level and an ability to actively control Zn uptake. Also, B. populifolia has been used as an indicator of metal contamination via the measurement of leaf phytochelatin (Gawel and Hemond, 2004). Our study presents evidence for the first time that both B. populifolia and P. deltoides have the ability to bioaccumulate Zn to levels far above ambient soil conditions. Utriainen et al (1997) demonstrated a differential tolerance to both Cu and Zn in propagated specimens of B. pendula and that such tolerance was the product of genetic plasticity. The abundance of B. populifolia at LSP, where the soils exhibit a broad range of metal concentrations, suggests that B. populifolia possesses a similar genetic plasticity.

The mechanics of metal translocation in P. deltoides have also been studied. Using hybrids (P. deltoides x P. nigra), Di Baccio et al (2005) found that shoot biomass was reduced with increased Zn concentrations. While our results cannot be used to determine individual growth rates, it is clear that both B. populifolia and P. deltoides are dominant at LSP, and that they are driving the development of the SNH assemblages. Their ability to tolerate metals appears to yield a selective advantage under these soil conditions.

Assemblage Development

Under normal conditions, metals are present in soils, as derivatives of parent geologic materials. Plant tissue can therefore be expected to contain these metals, in some cases at concentrations higher than in the surrounding soil. At concentrations above functional ranges, however, the plant can no longer maintain homeostasis and metabolic functions are inhibited (Hilary and Wilkins, 1987). It follows, then, that a species intolerant of high metal concentrations would be excluded from the assemblage on the site, even if a seed source existed within the regional pool. Therefore, vegetative assemblages growing on metalliferious soil should exhibit structural differences, when compared to similar but uncontaminated regional environs. Kimmerer (1981) and Kalin and van Everdingen (1988) demonstrated that succession may proceed slowly, often remaining in the herbaceous stage for decades or centuries. 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). This study corroborates the concept that patterns of succession associated with 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 tailings study, it appears that the SNH assemblage at the study site have developed preferentially on the soils with increased total metal load (Figure 3), perhaps as a function of reduced competition. In addition, the rate of change appears to be relatively rapid in recent years. A comparison of the 1995 and 2003 vegetation maps shows that the assemblages dominated by B. populifolia or P. deltoides have increased 23% in 8 years and now occupy nearly one-third of the study area. Hence, the mode of successional development at LSP appears to have been a relatively long period of stasis, perhaps twenty to thirty years, when the site was dominated by herbaceous species, followed by the rapid advancement of metal-tolerant woody species. One possible explanation for delayed establishment and subsequent rapid advance of trees on the site could be a shift in the dominant mode of propagation. Early colonizing plants were undoubtedly the result of chance recruitment to suitable micro-sites via long-range seed dispersal. Once established, vegetative reproduction via rhizomes or a combination of rhizome and locally produced seed would allow for rapid invasion into the herbaceous assemblages.

Also of interest was the relationship between the distribution of wetland communities and soil metal load. There are 20 mapped herbaceous wetland assemblages on the site, covering approximately 23 acres (~ 10 ha) (USACE FR/ERI, 2004). Nine of these are classified as semi-emergent marsh dominated by endemic species while 11 are classified as dominated by P. australis (USACE ERI, 2004). As mentioned earlier, the distribution of the semi-emergent marsh communities exhibited a statistically significant ($r = -0.70$, $p < 0.01$) inverse relationship (Figure 3) with total soil metal load (rank sum index). Hence, it appears that where total metal load is low, SEM assemblages that include a greater number of endemic species are able to compete more effectively. On the other hand, our study indicated assemblages dominated by P. australis (CRW and CRM) demonstrated the ability to tolerate a broad range of soil metal loads. Metal uptake (Keller et al. 1998) and distribution in P. australis has been described (Windham, et al. 2003). Much of the metal load is sequestered in the roots, and production is not greatly inhibited (Peverly et al., 1995), provided that threshold values for deleterious effects are not exceeded.

Conclusions

The null hypothesis that a passive model of metal uptake would be common in the dominant species of the various assemblages was rejected, as all three models of metal bioaccumulation were evident. The most dominant species of the site, B. populifolia and P. deltoides, demonstrated the ability to translocate Zn to high levels in leaf tissue. It appears that this ability to translocate metals at high rates conveys some type of adaptive advantage, and should be investigated further.

The preferential occupation of the successional northern hardwood assemblages on sites with elevated total metal loads results in a directional (beginning with areas of high soil metal load) pattern of succession on the heterogeneous soils associated with historic fill. Conversely, the areas of lower soil metal load appear to facilitate semi-emergent assemblages (SEM) that are not dominated by the invasive P. australis. Continued observation of the sem-emergant wetlands is needed to determine if these wetlands are stable. These two patterns provide some support for our second hypothesis by demonstrating that soil metal load correlates with assemblage distribution. However, that correlation was only significant for two of the ten identified assemblages and, in fact, the distributions of the CRM, CRW, and MS assemblages, dominated by aggressive metal tolerant species, were indifferent to total soil metal load.

This study provides several insights for land managers of urban brownfields. Differential tolerance to and methods of assimilating metals can either lead to or eliminate the ecological risk associated with the translocation of metal contaminants. For example, at LSP, the risk associated with soil Zn, at concentrations that are for the most part below New Jersey's soil screening criteria, is increased as the leaves concentrated this metal, and consequently the leaf litter would yield elevated levels of Zn. Conversely, soil As levels, which exceeded the referenced screening levels, posed little environmental risk as the dominant species were sequestering As at the root zone with almost no translocation to above-ground tissues.

Finally, differential assemblage development based on soil metal load further complicates the already atypical assemblage structure that develops in the urban context. The eventual impacts that these abiotic filters will have on long term species composition is an interesting question and deserves further investigation. At this point, however, there appears to be little value in assuming that the traditional species trajectories associated with woodland succession will apply.

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Table 1: Physical and chemical characteristics of the soil samples, the sites are arranged by assemblage type.

Sites & Community types	pH	TOC%	Depth
TP-21 CRM	6.6	28.4	13.2
TP-21/40 CRM	6.2	32.7	18.7
TP40C CRM	6.8	14.9	12.8
TP-16CRM	6.2	13.6	15.2
TP-1 MS	6.0	8.8	13.5
TP-3 MS	5.8	28.5	9.8
TP-40 B MS	7.4	31.0	5.3
TP7/8 MS	5.4	51.0	7.3
TP-10 SNH	5.4	32.3	8.7
TP-14 SNH	5.2	52.6	8.2
TP-14/16 SNH	5.2	34.9	8.8
TP-18 SNH	6.0	23.7	12.5
TP-24 SNH	6.6	29.8	11.7
TP-25 SNH	6.2	30.7	10.2
TP-28 SNH	5.0	45.5	10.2
TP-28/17 SNH	5.4	9.7	4.3
TP-41 SNH	6.0	10.0	16.5
TP-43 SNH	5.4	28.3	4.3
TP-43/14 SNH	5.0	21.1	6.2
TP-48 SNH	6.1	19.8	6.8
TP-8 SOF	6.0	47.4	13.2
TP-40 SOF	6.6	17.3	4.7

Table 2: **Table 5**: Average concentrations of metals in the soil (s) and leaves (l), stem (st) and root (r) of the dominant plant species. All concentrations are represented as μg/g, MD = minimum detection limit, stdev = standard deviation.

	As								**Cr**						
	l	**stdev**	**r**	**stdev**	**st**	**stdev**	**s**	**stdev**	**l**	**stdev**	**r**	**stdev**	**st**	**stdev**	**s**
B. populifolia	0.02	0.031	2.32	2.1	<MD	0	31.1	25.1	0.261	0.197	13.1	16.6	0.261	0.483	48.8
R. copallinun	<MD	0	1.2	0.805	<MD	0	27.3	20	0.412	0.131	6.57	4.62	0.026	0.036	47.5
A. vulgaris	<MD	0	1.36	0.658	<MD	0	20.3	10.8	0.305	0.239	5.53	1.96	0.14	0.139	32.7
P. tremuloide	0.013	0.022	8.13	10.5	<MD	0	162	126	0.232	0.088	5.03	3.58	0	0	48.7
	Cu								**Pb**						
	l	**stdev**	**r**	**stdev**	**st**	**stdev**	**s**	**stdev**	**l**	**stdev**	**r**	**stdev**	**st**	**stdev**	**s**
B. populifolia	5.62	0.783	46.1	31.3	1.74	0.675	124	76.5	2.25	4.19	129	91.5	11.3	9.025	266
R. copallinun	4.52	0.937	52	42.4	2.82	2.15	176	126	<MD	0	52.4	39	1.44	1.11	382
A. vulgaris	17.3	8.19	64.8	17.7	7.12	2.94	220	75.6	<MD	0	56.2	25.5	0.516	0.291	411
P. tremuloide	8.55	0.469	78.4	34.8	2.92	0.394	678	736	<MD	0	128	61.8	4.81	3.82	1875
	Zn								**V**						
	l	**stdev**	**r**	**stdev**	**st**	**stdev**	**s**	**stdev**	**l**	**stdev**	**r**	**stdev**	**st**	**stdev**	**s**
B. populifolia	905	503	221	127	123	76.4	157	165	9.08	1.98	10.8	8.14	0.036	0.084	53.8
R. copallinun	37.3	40.5	275	238	26	19.1	190	79.6	6.96	2.27	9.5	8	0.015	0.034	18.3
A. vulgaris	76.2	48.9	240	170	99.6	70.6	452	382	9.55	2.73	8.64	3.67	0.015	0.034	77.2
P. tremuloide	1553	212	454	169	120	35.4	812	732	6.15	1.82	9.09	2.48	0.018	0.031	35.3

Table 3: Soil Remediation standards in μg /g (SCC, GSRS) in residential areas (NJDEP, 1999, 2004.). SCC=New Jersey Department of Environmental Protection Soil Cleanup Criteria, GSRS = Generic Soil Remediation Standards.

Metal	As	Cr	Cu	Hg	Pb	V	Zn
SCC(1999)	20	240	600	14	400	370	1500
GSRS (2004)	0.4	300	3100	23	400	550	23000

Table 4: Pearson correlations between soil metal concentrations, organic content (TOC%), pH, and soil depth (cm) in 2005 (Correlation *r* value (upper) and *p* values (lower) (2-tailed), with significance in bold)

	depth	TOC	pH	As	Cr	Cu	Hg	Pb	Zn	V
Depth	1	-0.337	0.341	-0.105	-0.136	0.013	-0.002	-0.012	0.262	-0.13
	.	0.125	0.121	0.642	0.547	0.953	0.994	0.957	0.238	0.564
TOC		1	-0.318	0.362	**0.435***	0.183	0.202	0.176	0.108	**0.425***
		.	0.149	0.098	0.043	0.414	0.368	0.434	0.631	0.049
pH			1	-0.174	-0.353	0.177	-0.077	0.1	0.38	**-0.457***
			.	0.439	0.107	0.43	0.734	0.657	0.081	0.032
As				1	0.15	**0.774****	0.138	**0.794****	**0.518***	0.099
				.	0.505	0	0.541	0	0.014	0.661
Cr					1	-0.018	0.014	0.033	-0.073	**0.904****
					.	0.936	0.95	0.885	0.748	0
Cu						1	0.03	**0.973****	**0.849****	-0.033
						.	0.893	0	0	0.886
Hg							1	-0.025	0.092	0.106
							.	0.913	0.685	0.64
Pb								1	**0.801****	0
								.	0	0.998
Zn									1	-0.152
									.	0.5
V										1
										.

* Correlation is significant at the 0.05 level (2 tailed).
** Correlation is significant at the 0.01 level (2 tailed).

Table 5: Summary table of the kriging results from the transformed data set showing also the prediction errors. ME – mean error, RMSE – root-mean square error, $r^2{}_p$ – the coefficient of determination.

Metals	Transformation	Skewness Coefficient	Anisotropy Angle	Mean	ME	RMSE	r2p
As	Lognormal	0.323	55	1.76	-0.03	0.11	0.83
Cr	Rank ordered	0.211	0	0.61	-0.01	0.04	0.84
Cu	Rank ordered	0.251	75	0.51	0.01	0.03	0.81
Pb	Lognormal	-0.305	75	2.81	0.05	0.17	0.52
Zn	Rank ordered	-0.202	55	0.62	0.04	0.14	0.74
Sum Metal	N/A	0.194	90	3.14	0.22	0.78	0.65

Figure 1: 2005 and 2006 sampling points within each assemblage type.

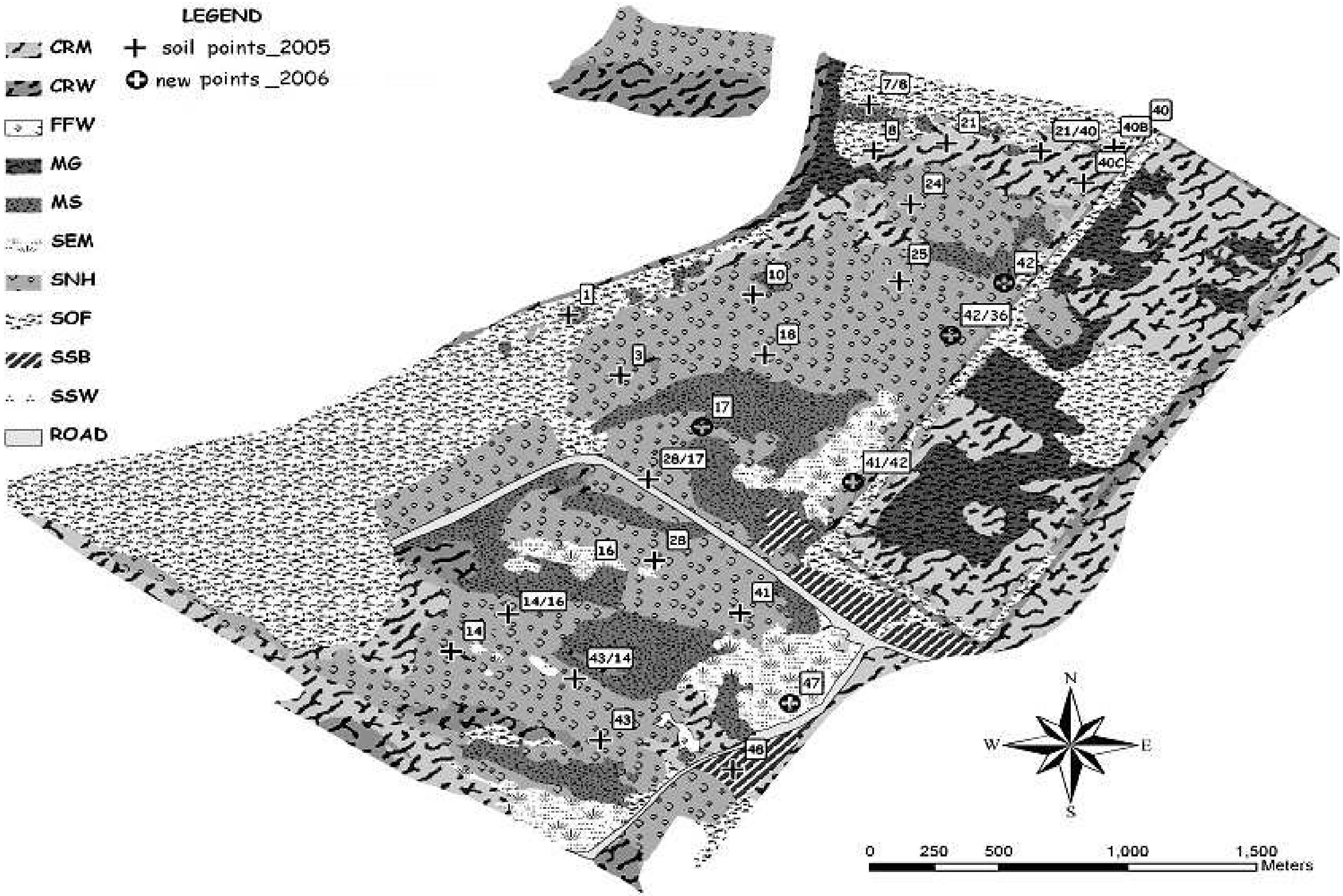


Figure 2: Contour maps of each metal and the summarized metal load showing the sampling locations. A: arsenic; B: chromium; C: copper; D: lead; E: zinc and F: sum metal

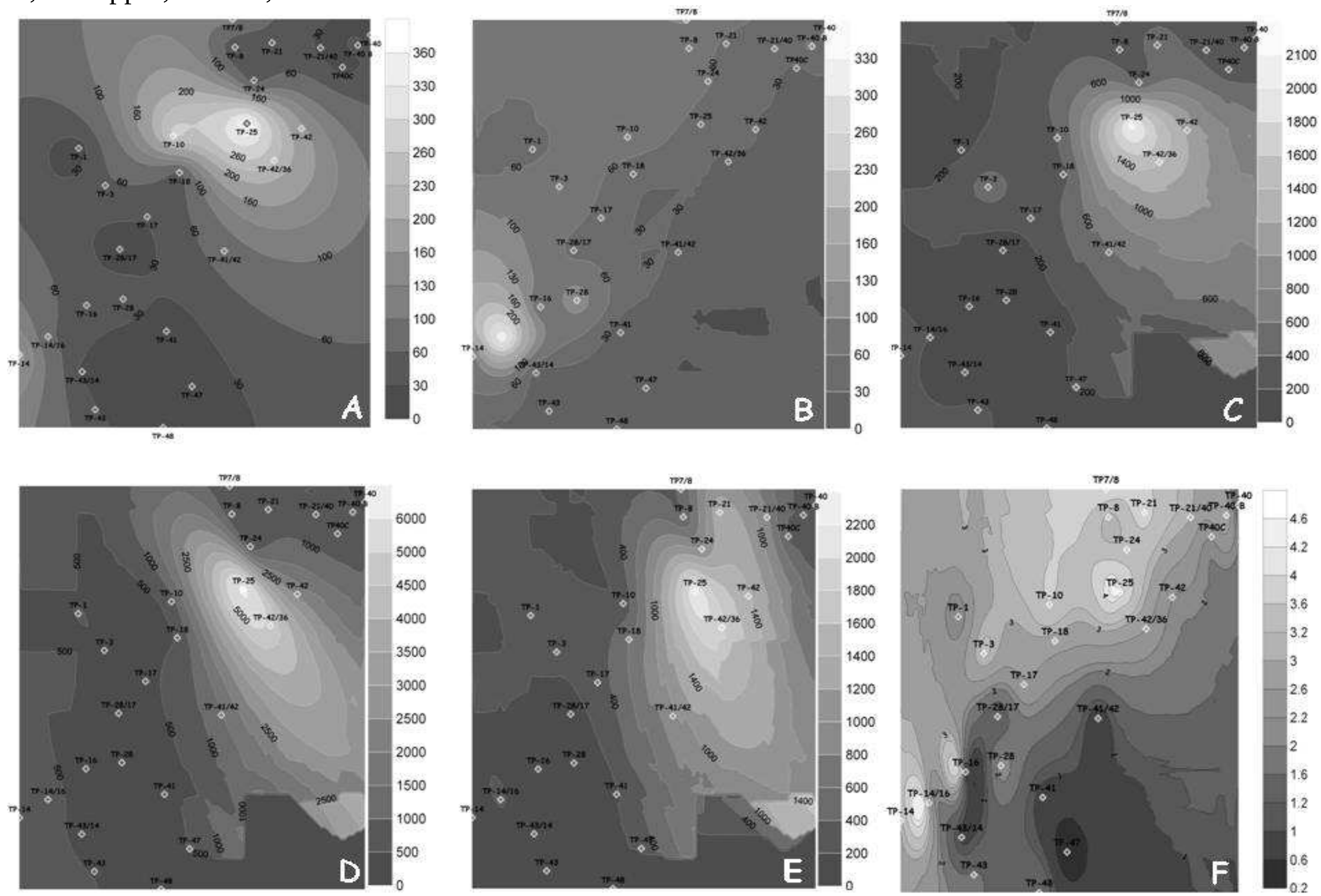


Figure 3. Area ratios of each assemblage type compared to total soil metal load (rank sum index).

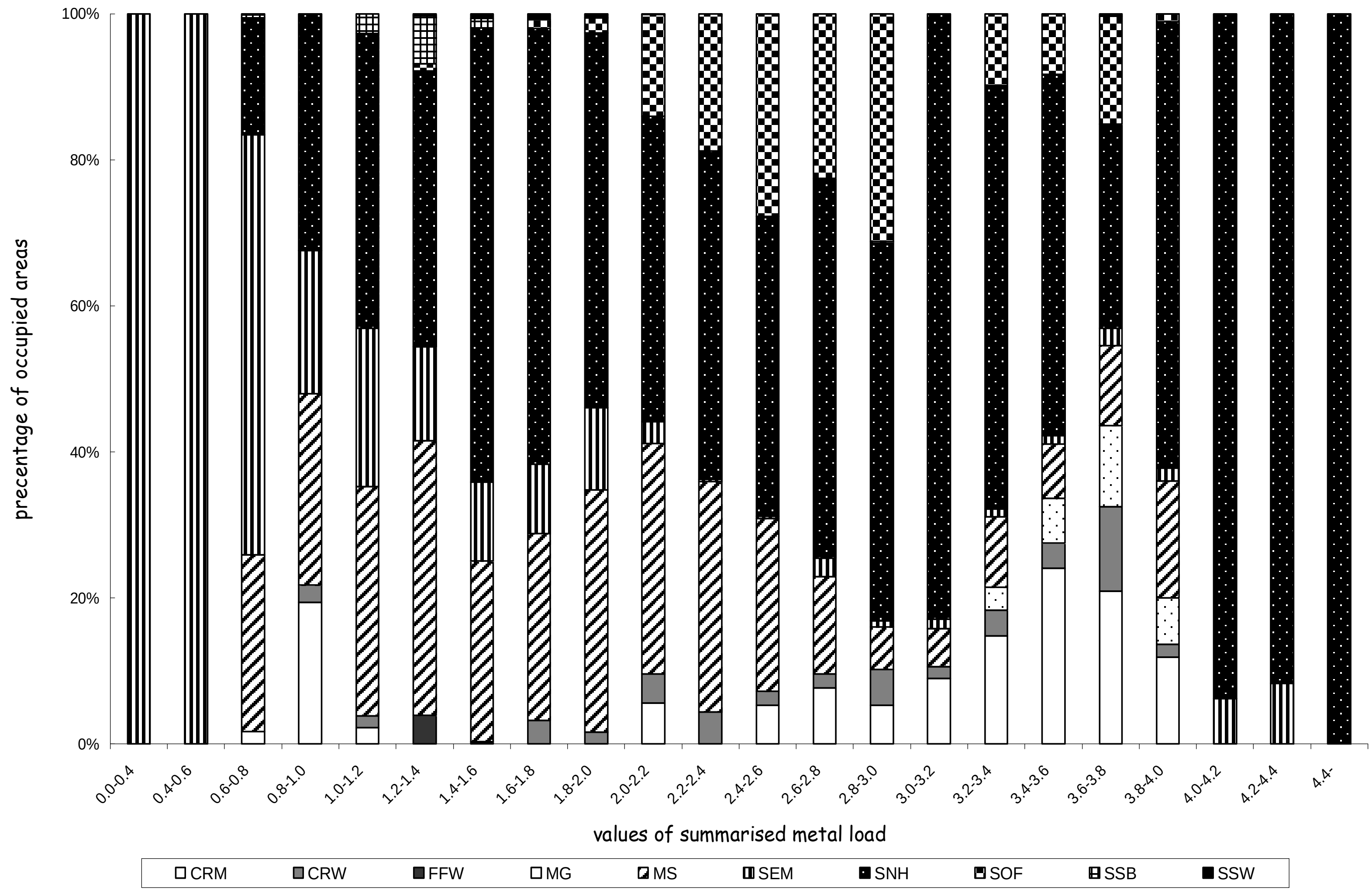
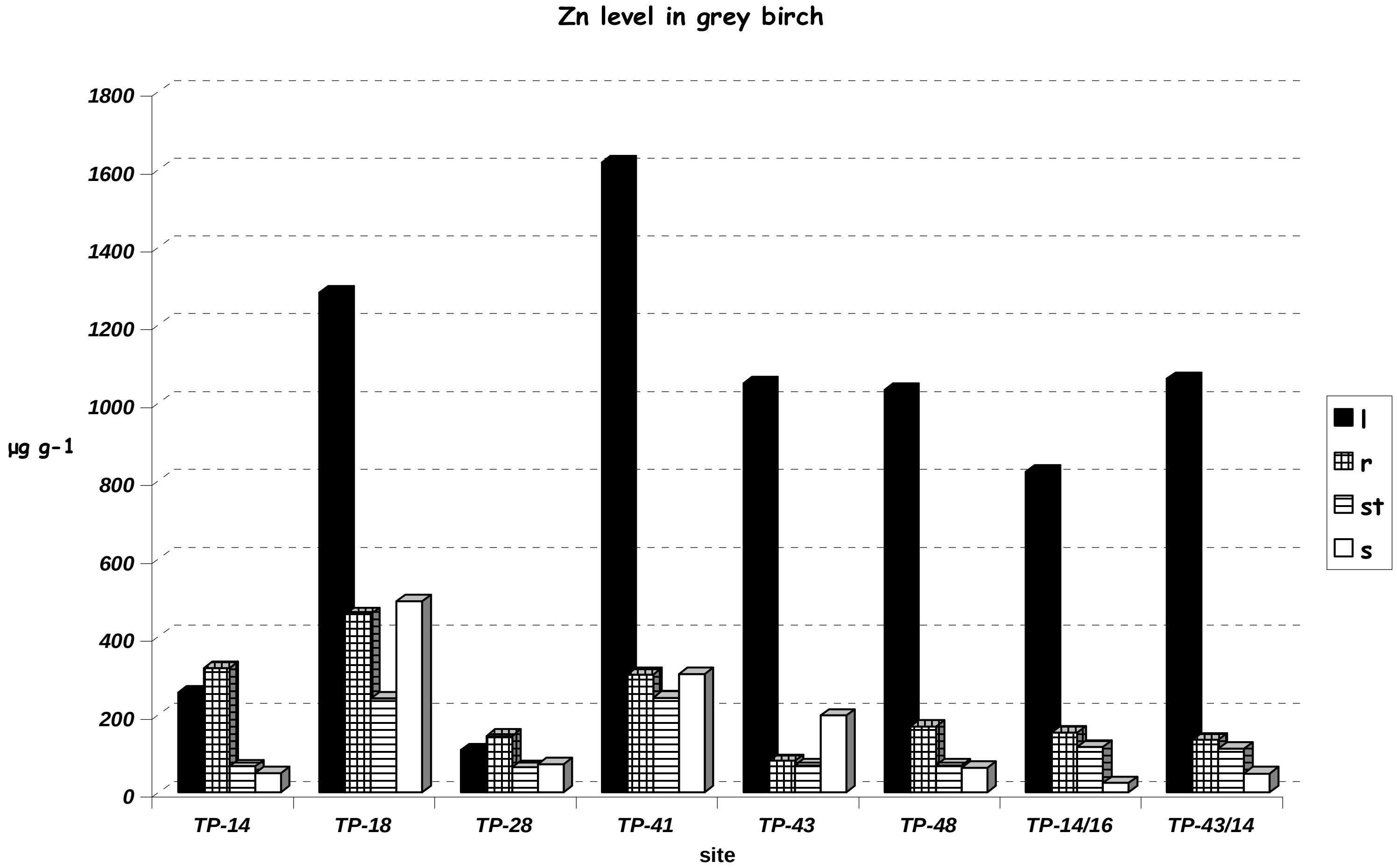


Figure. 4 Zinc concentration in the leaf tissue of B. populifolia.



Chapter III

Soil Metal Concentrations and Productivity of Betula populifolia (gray birch) as Measured by Field Spectrometry and Incremental Annual Growth in an Abandoned Urban Brownfield in New Jersey.[2]

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Abstract

Analysis of soil samples from a forested brownfield within Liberty State Park, Jersey City, New Jersey, USA, shows that arsenic, chromium, lead, zinc and vanadium exist at concentrations above those considered ambient for the area. Using both satellite imagery, field spectral measurement this study examines plant productivity at the assemblage and individual specimen level. In addition, longer term growth trends via tree core data (basal area increase) were studied. Leaf chlorophyll content within the hardwood assemblage correlated well with a threshold model for metal tolerance, decreasing significantly beyond a total soil metal load (TML) of 3.0. Biomass production (calculated with RG – Red/Green ratio index) in Betula populifolia (gray birch), the co-dominant tree species, demonstrated an inverse relationship with the concentration of Zn in the leaf tissue during the growing season. Incremental basal area growth in B. populifolia exhibited a statistically significant relationship to total soil metal load. Ecosystem function as measured by plant production is impaired at a critical soil metal load.

Key words: Brownfield, Plant Productivity, Remote sensing techniques, Metal Tolerance, Betula populifolia

Introduction

With the restoration of many industrial waterfronts throughout the United States there are many questions concerning brownfield remediation, and specifically the ecological character of these contaminated sites. Existing redevelopment paradigms tend to exclude or limit the option of open space in favor of hardscapes to eliminate the potential of contaminant transfer. Recognizing the growing importance of both structure (maintenance of biodiversity) and function (fostering natural cycles) of urban ecologies, we have examined primary productivity and long term growth on a well known urban brownfield.

Many urban soils contain trace metals, typically cadmium (Cd), copper (Cu), zinc (Zn), lead (Pb), and others above established screening criteria (Dudka et al., 1996). While these elements are often adsorbed or occluded by carbonates, organic matter, iron-magnesium oxides and primary or secondary minerals (Adriano, 1986; Ross, 1994), their eventual fate and impact on vegetative assemblage development is not well understood.

However, metal-induced metabolic stress in plants is well documented. In general, plant stress differs depending on the metal being studied, its concentration and the aspect of growth examined. For example, Convolvulus arvensis (field bindweed) did not tolerate soil Cd above 20 mg/l whereas the threshold biomass reduction levels for soil Cu(II) and Cr(VI) were not reached at 80 mg/l (Gardea-Torresdey, et. al. 2004).

Metal-induced photosynthetic inhibition can result from the reduction of the enzymes involved in chlorophyll biosynthesis (De Filipps and Pallaghy, 1994), substitution of metal ions within the chlorophyll molecule, reduction in the concentration of chlorophyll pigment within the leaf (Kastori et al., 1998), and membrane disruption (Droppa and Horvath, 1990). Such studies demonstrate the specificity with which the growth responds to varying concentrations of metal contaminants. With such variation in the response of plants to metal-induced stress, it is understandable why there is far less data available at the assemblage level.

In-situ studies that examine plant response to metal contaminated soil have generally focused on abandoned mines, rail yards or smelters. For example, Malawska and Wilkomirski (2000) examined soil metal and aromatic petroleum hydrocarbon distribution and uptake in <u>Taraxacum</u> <u>officinale</u> (common dandelion) within the different areas of the railway junction in Iława Główna, Poland. They demonstrated a considerable heterogeneity in both contaminant distribution and translocation within the rail junction. Trace level metal speciation and translocation was examined within three abandoned rail yards in Montreal, Quebec, (Murray and Hendershot, 2000). Free, dissolved and total metal concentration demonstrated limited ability to predict uptake within plant tissue, in this study. Mercury (Hg) tolerance, translocation characteristics, and a reduction of plant frequency with increasing Hg concentrations was observed in several species of grasses and scrubs growing in an abandoned mine near Atlanta, Idaho (Ellis and Eslick, 1997). In a related study of afforested arable land in northern Europe, tree growth and litter accumulation resulted in lower soil pH and increased metal mobility (Anderson et al, 2002). Despite these considerable advances in the assessment of contaminated soil, there are still obvious difficulties in identifying whether or not and under what conditions soil metal contamination should be a matter of concern (French et al., 2006).

At Liberty State Park (LSP) in Jersey City, NJ, the site of an abandoned rail yard, As, Cr, Cu, Pb, and Zn occur at concentrations above those considered ambient for New Jersey's soils (Saunders, 2002) and in several cases above New Jersey Department of Environmental Protection's soil cleanup criteria (New Jersey Department of Environmental Protection, 1999). variability in the distribution of soils and cinders has resulted in a unique mosaic of assemblage types (defined as discrete plant communities) (United States Army Corps Engineers, 2004). Due to the various hydrologic conditions and soil metal load there has been a preferential development and distribution of the northern hardwood (NH) assemblages on the site (Gallagher et al. 2007). That study demonstrated that the NH assemblages, dominated by <u>Betula populifolia</u> (gray birch; 37.5% cover) and <u>Populus</u> <u>deltoides</u> (eastern cottonwood; 13.7 % cover), developed in areas of higher total soil metal load, while herbaceous assemblages remained more competitive in areas of lower total soil metal load. The objectives of the present study were to investigate (1) the relationship between Zn load in the leaf and primary productivity in the dominant tree species and (2) soil TML and primary productivity in the emerging hardwood assemblage. We hypothesized that stress-induced loss of primary production would be detectable only when TML exceeded critical threshold values, and that such loss would be detectable using vegetation indices derived from field reflectance measurement and hyperspectral imagery. We also hypothesized that the long term impact of high TML would be discernible in the growth rates of individual trees despite their success during assemblage establishment.

Material and Methods

Study Area

The 251 acre (102 ha) project site includes both hardwood and herbaceous assemblages and is located on the west bank of Upper New York Bay (centered at 40°42’14” N x 74°03’14”W). Approximately 40 acres of dredge spoil will be capped and 20 acres is being used as soil stockpile area and are therefore not included in the study. Of the remaining 191 acres 50% is covered by NH. Originally an intertidal mud flat and salt marsh, it was filled between the years of 1860 and 1919 for use as a railroad yard owned by the Central Railroad of New Jersey (CRRNJ). The fill materials consisted of debris from construction projects and municipal waste from New York City. The subsequent use of the site for rail transport and commodity storage, including coal, resulted in relatively high and unevenly distributed soil metal concentrations. In 1967, the CRRNJ discontinued operations leaving the site isolated and undisturbed. From 1970 through 1984 the State of New Jersey purchased the land for use as a park. Currently salt water intrusion into the site does not impact the study area (United States Army Corps Engineers, 2004). Today the park consists of approximately 1100 acres, of which approximately 251 acres remain undeveloped.

The current plan calls for approximately 100 acres (40 ha) of hardwood assemblage to be left undisturbed. Therefore, an understanding of the impact associated with the soil metal load on the development of the site is critical.

Sampling

In 2005, a 10 m X 10 m grid system was applied to the site, using the Geographic Information System (GIS) software Arc-View 3.2®. The grid was oriented so that the axes were parallel to transects used in the 2003 natural resources inventory (NRI) performed by the United States Army Corps of Engineers (USACE) to ensure good correlation to the defined assemblage types. Sampling sites were then determined by comparing assemblage boundary maps and aerial photography to achieve adequate representation of the targeted assemblages and ensure good correlation between soil metal load and assemblage type.

Thirty-two sites were sampled, three soil samples were collected at one meter spacing at each site and their GPS (Corvallis Microtechnology MC-GPS, accuracy 1m) coordinates were recorded. The samples were collected from the areas of greatest root concentration between 10 and 25 cm below 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. A LaMotte field soil pH meter was used for analysis of soil pH. Figure 1 shows the locations of the sampling points, the boundaries of the northern hardwood assemblages and the total soil metal loads.

Multispectral imagery

IKONOS multispectral imagery was purchased from Space Imaging$^{TM/SM}$. The image was taken on June 24, 2004. The four bands cover the visible and a small portion of the near-infrared range (450 nm – 900 nm) with a 4m (pixel size=$16m^2$) spatial resolution. The image was georeferenced and atmospherically corrected by the company. From the image, Normalized Difference Vegetation Index (NDVI) (Tucker, 1979) was calculated for each pixel using ENVI 4.2 (Environment for Visualisation, ITT Visual Information Solutions), utilizing the third (Red: 640nm – 720nm), and the fourth (NIR: 770 nm – 880nm) bands (NDVI = NIR-RED/NIR+RED).

Field Spectrometry

Satellite imagery provides data from only a single moment in time and therefore does not indicate if production is stable throughout the growing season. Thus, to determine if the seasonal variation in productivity (as a measure of the normalized difference vegetation index - NDVI) correlated with metal uptake, leaf samples were taken monthly from May through September of 2006 for both reflectance and Zn concentration analysis. At seven sites, birch trees with diameters at breast heights (DBHs) between 8.8 and 11.2 cm were chosen from areas where the total soil metal load ranged between 0.84 and 3.5.

Both mature and fresh leaves were collected for field spectral measurements. A handheld FieldSpec Pro Full Range spectrometer (manufactured by Analytical Spectral Devices) was used to measure light intensity reflected by the upper side of birch leaves. The device measures reflectance in the visible and short-wave infrared range (350 – 2500 nm). In all cases the spectrometer was configured with an 8^{o} angle of view, giving a 2.5cm diameter reading area at a height of 15cm. Only healthy leaves were taken off the branches and arranged on black background. The measurements were repeated twice for each sampling sites using three different, randomly picked set of leaves (12 leaves per set). All measurements were calculated as averages of 16 readings in order to reduce the noise. The measurements were also referenced to a Spectralon® white reference panel both before and after each sampling period to ensure calibration accuracy. The spectral measurements were recorded as reflectance values in ASCII format. After spectral measurements were recorded the leaves were saved and carried to the UMDNJ lab for metal analysis.

Separation of Collected Spectra

The median spectral profiles were used for analysis. To further reduce noise, the reflectance values were imported into Excel and smoothed using the Savitzky–Goley algorithm (Tsai and Philpot, 1997). After the smoothing procedure, the data were tested for significant differences between medians using the non-parametric Mann-Whitney U-test.

Long Term Growth

Increment core samples were collected from six of the above specimens on May 15, 2006. Transverse longitudinal sections of 30 µm were cut from each core using a sliding microtome with the blade set at 10^o. Four sections, each covering at least six identifiable growth years, were collected from each core sample. Each sample was stained with a solution of Safranin (1/3) and Alcian Blue (2/3). The samples were then permanently mounted on a slide using Permount resin. The samples were examined using a dissecting microscope and the distances between annual growth rings from the years 1999-2005 was determined. Each core sample was represented by four sections, from which the mean and standard deviation was calculated and used for comparison. Incremental growth rates (basal area added per year, mm^2) were determined by calculating the total basal area ($BA=\Pi r^2$) for each year class and subtracting the basal area of the previous year classes. The slope of the line created by the basal area added per year for each tree was used for comparison as well as cumulative (total basal area added) growth.

Laboratory analysis

Each soil sample was treated as reported previously (Gallagher et. al., 2007). Briefly, aliquots of dried, sieved samples were weighed, ashed at 450°C (to calculate organic C as 'loss on ignition'). Additional dry aliquots of ~0.5 g, weighed to the nearest milligram, were treated with 10 ml trace-metal grade HNO_3, and acid-extracted in Teflon bombs in a MARS-5 (CEM Corp.) programmed microwave instrument. These acid extracts were analyzed by flame Atomic Absorption Spectroscopy (AAS) for Cr, Cu, Pb, V, and Zn in a Perkin-Elmer 603 atomic absorption spectrophotometer. A method blank and a National Institute of Standards and Technology (NIST) Standard Reference Material (SRM) #1944 ("New York – New Jersey Waterway Sediment") sample were run simultaneously with each set of 12 soil samples to eliminate the potential error caused by contamination during sample preparation. The cold-vapor AAS method was used for Hg analysis with a MAS-50D mercury analyzer (Bacharach, Inc.). Arsenic was determined in the presence of a $Mg(NO_3)_2/Pd(NO_3)_2$ matrix modifier by graphite furnace AAS in a Perkin-Elmer Z5100 with Zeeman background correction.

Since Zn represented 99% of the metal load translocated to the leaf tissue of B. populifolia during our previous work (Gallagher et. al., 2007), it was the tissue analyzed during this study. Leaf samples were dried for 48 hours at 60°C to a constant weight, weighed to the nearest mg and distributed as 0.3 g sample of each triplicate. These were treated first with 30% H_2O_2 to mineralize cellulose, and then prepared similarly to soil samples. A method blank and a standard reference material (NIST SRM1573a – tomato leaves) were run simultaneously with each set of 12 plant samples. Leaf samples were analyzed for Zn similar to the soil samples. However, when there was <1% absorption by flame AAS, a graphite furnace AAS with Zeeman effect was employed for increased sensitivity (Perkin-Elmer Z5100 instrument). Minimum detection levels (MDLs) was calculated by taking three times the standard deviation of the values measured for the analytical blanks.

To study the relationship between plant productivity and metal distribution in the soil, the previously published Total Soil Metal Load (TML) map was reutilized (Gallagher et al., 2007). The concentration of each metal that exceeded the Soil Cleanup Criteria (SCC) recommended by the New Jersey Department of Environmental Protection

(New Jersey Department Environmental Protection, 1999), namely As, Cu, Cr, Pb, Zn were rank-ordered as described by Juang et all. (2001). The rank order values were then summarized for each site as TML values. Finally, the data was block kriged, to account for the high standard deviation in the original datasets using Surfer Surface Mapping Software® (Release 8.0., Golden Software Inc.). The contour maps were exported in AutoCad format (.dxf) and imported in ArcMap (ESRI, ArcGIS 9.1) for further analysis.

Imagery and Signatures

Since the Mann-Whitney U-test showed significant differences in spectral profiles among sites at each sampling time (data not shown), the field-collected reflectance values were used to investigate differences in plant productivity. NDVI and Red/Green Ratio Index (RG) (Gamon & Surfus, 1999) were calculated as indicators of chlorophyll concentration in the leaves. For NDVI the RED reflectance values measured between 640nm and 720nm were averaged. To determine the NIR, mean values between 721nm and 880nm were entered into the equation. To average the Green range, the mean of reflectance values measured between 550 nm and 640nm was calculated. The resulting indices were correlated with the Zn concentration in leaves as well as in soil, and also with the total metal load in soil.

To determine biomass production at the assemblage level, the NDVI transformed pixels of the 2004 June IKONOS image were overlaid with the 2003 vegetation map. The hardwood assemblages were clipped based on the vegetation map. Then, only the areas of NH community were overlaid on the contours of the total metal load distribution map, resulting in different sets of pixels with pixel values referring to productivity (NDVI) and grouped by total metal load. Within each group, the pixel values were averaged and the mean NDVI values were then correlated with both the TML and soil Zn concentration.

Data Treatment

The raw datasets were analyzed with different software packages. The descriptive statistical parameters were calculated with SPSS (release 11.5, SPSS Inc.) and Minitab (Minitab release 12.23). Vegetation-metal distribution maps and the sampling plan were created with ArcView® (version 3.2 ArcGIS, ESRI Co.) and ArcMap® (Version 9.1, ArcGIS, ESRI).

Results

Soil Metal Concentration

The statistically representative concentrations for each metal within the NH assemblages are presented in Table 1. An assessment of the soil metal concentrations was accomplished through comparison to the threshold values found in the Soil Cleanup Criteria (SCC) recommended by the New Jersey Department of Environmental Protection (New Jersey Department Environmental Protection, 1999) (listed in Table 2). In addition, the results were compared to the EPA's Ecological Soil Screening Level (ECO-ESSL, US EPA, 2003) for Heavy Metals regarding their effects on terrestrial plants (Table 1). Where there were no ECO-SSL values available, critical values of a screening benchmark released by the Oak Ridge National Laboratory (Efroymson et all., 1997) were used (listed in Table 2).

Soil concentrations of Cr, Hg, and V in the hardwood assemblage did not exceed the SCC. The concentration of As whenever it was above the minimum detection level (MDL), exceeded the SCC. Cu exceeded the SCC in 16% of the samples, while Zn exceeded these standards in only five percent of the samples. Pb exceeded the criteria in 52% of the samples. The highest concentrations of As, Cu, Pb, and Zn were found at one location, TP-25. V was the only metal to exhibit any significant (0.05 level) correlation (r = -0.45) with soil pH, indicating that enhanced metal leaching of As, Cr, Cu, Pb, and Zn with lower soil pH (Max=7.4, min=5.0, mean 5.9) has not been significant. This observation is also supported by samples taken from deep test pits that penetrated through the historic fill into the original sediment (between 1 and 2 meters bellow the current surface). These samples, which reached the depth of nine meters had only background levels of metal (data not shown, NJDEP 1995) indicating that significant leaching had not occurred.

While soil metal concentration in excess of the residential standards was limited, Maximum Acceptable Toxic Concentration (MATC, ECO-SSL, 2003) screening levels were exceeded at more than three-quarters of the sites for As, Cu and Pb. In addition, the lowest observed effective concentration (LOEC, OakRidge, 1997) criteria were also exceeded for all metals (Table 1). These data indicate that while soil As and Pb concentration may be relevant to residential soil removal criteria, three-quarters of the samples taken demonstrated the potential for metal-induced phytotoxicology.

Zn and Seasonal Productivity

The concentrations of Zn in the leaf tissue of B. populifolia demonstrate a significant relationship ($r^2 = 0.41$, $p < 0.01$) with soil Zn concentrations (2005 data). The concentrations of Zn in leaf tissue exhibited a significant spike (Figure 3a) during June and then decreased throughout the summer. The June peak also correlated ($r^2 = 0.41$, $p < 0.01$) with a decrease in productivity (RG ratio index) (Figure 3b). However, when broken down by site, both TP-14 and TP-14/16, sites with the highest total metal load (TML>3), yielded much stronger regressions (TP-14 $r^2 = 0.81$, $p = 0.04$; TP-14/16 $r^2 = 0.81$, $p = 0.04$). TP-18, with a TML of 2.8, exhibited a statistically borderline relationship ($r^2 = 0.71$, $p = 0.08$) between leaf Zn concentrations and productivity. At the remaining three sites, where the TML was below 2, the relationship between leaf Zn concentration and productivity (NDVI) was statistically insignificant (Table 3).

Assemblage Productivity

NDVI minimum and maximum values within the hardwood assemblage from June 2004 yielded a fairly broad range, from a low of 0.290 to a high of 0.998. However, NDVI values for each sampling site (read by pixels in which the coordinate of the sampling site had fallen, pixel size = 16 m^2) were more consistent, ranging from

0.486 to 0.522. When ranked according to TML at intervals of 0.5, mean NDVI values yielded a significant negative correlation (r = -0.78, $p < 0.01$). When fit to a second order polynomial curve, the relationship between TML and NDVI yielded a highly significant relationship ($r^2 = 0.85$, $p < 0.01$). These data clearly indicate that a tolerance threshold, the level at which sub-lethal impact is discernable, was reached at a TML of approximately 3.0 (Figure 2)

Long Term Productivity

Of the seven trees examined for leaf productivity, six were chosen for long term productivity analysis. Site TP-18 was considered an outlier as its height to diameter at breast height (dbh) ratio was 28 % less than the mean of all samples. This difference was the result of a discrepancy in the site index (relationship between soil properties, tree spacing and growth habit; Hironaka et al., 1991), specifically wider spacing between individual trees, which has been shown to result in lower height to diameter ratio (Mäkinen, 2002).

Cumulative growth in B. populifolia, as a measure of increased basal area over a six year period from 1999-2005, exhibited a broad range from 2093 mm^2 to 3732 mm^2. Statistically similar annual growth rates were exhibited between sites TP-48 and 14, TP-48 and 14/16, and TP-48 and 43 (Table 4 shows only the significant regressions). Total basal area growth at sites TP-14 and 14/16, where total soil metal load was the greatest (TML > 3.0), decreased as compared to the other samples. As in the imagery data, significant rectilinear relationship ($r^2 = 0.85$, $p < 0.01$) was demonstrated between cumulative growth and TML. In addition, the incremental growth rate for TP-14 and 14/16 exhibited a negative slope (Table 5). The relationship between the slope of the incremental growth at all sites and TML was highly significant ($r^2 = 0.93$, $p < 0.01$).

Discussion

The broad range in TML (0.01-4.4) exhibited at the study site is typical of urban brownfields where past industrial use has created heterogeneous soil metal distribution, with localized areas having high loadings. For example, a study of three abandoned rail yards in Quebec, Canada, (Ge et al., 2000) yielded maximum soil concentrations of Cu (653 μg/g), Pb (1180 μg/g), and Zn (1350 μg/g) with standard deviations ranging from less than 1 to 97, similar to but somewhat lower than those of LSP.

Early studies (Allen and Sheppard, 1971; Cox and Hutcheson, 1980) have shown that while metal tolerance occurs in relationship to a single metal, an individual species can exhibit tolerance to several metals found in association Also, Hanzlka (2004) demonstrated that metal adsorption capacity of organic matter can change with different metal mixtures, further complicating plant-metal interactions. Since our study site exhibited a heterogeneous mix of soil metals and a broad range (9-51%) of soil organic matter content, we chose to examine the relationship between short term plant growth (NDVI and RG ratio), long term plant growth (tree basal area)TML.

Past studies have used reflectance data to determine metal-induced stress in vegetation (Kemper and Sommers, 2002; Kooistra, et. al., 2004). These focus on grass assemblages and produce a decrease in various reflectance indices with increasing soil metal load. Our study suggests that both the NDVI and RG Ratio indices can also be useful in detecting metal-induced stress in hardwood assemblages.

Assemblage Productivity

The strong negative correlation and significant curvilinear regression between primary productivity and TML in this study (Figure 2) offers support for a threshold model of metal tolerance at the assemblage level. Productivity decreases rapidly at a TML of approximately 3.0. Such a growth response appears to be fairly common at the species level and may be linked to an ability to maintain a constant internal contaminant concentration regardless of external concentrations For example, in bench studies using metal tolerant clones of Betula spp., concentrations of Zn in the root and shoot tissue were maintained as external soil concentrations increased until a threshold value was reached (Hillary and Wilkins 1987). Interestingly, the non-tolerant variety of the same plant reached a threshold at which the cells became inundated with zinc at much lower external soil concentrations than the tolerant varieties (Hillary and Wilkins 1987). Our data suggest that a similar model of metal tolerance is functioning in at least the dominant species of the NH at LSP.

Seasonal Productivity

The measured loss in productivity may be temporal in nature. There was an approximate 65% decrease in primary productivity at the time of June sampling (Figure 3a). This decrease has a statistically significant negative relationship ($r^2 = 0.40$, $p < 0.01$) to the 76% increase in the concentration of Zn in the leaf tissue (Figure 3b). As the concentration of Zn in the leaf tissue decreases during July, August and September, primary production increases. The data also indicate that this relationship was most significant in areas of high TML and became insignificant as TML fell below 3 (Table 6). Temporal relationships have been demonstrated between Zn leaf concentration, chlorophyll a/b and peroxidase activity in Avicennia marina (Gray Mangrove) (MacFarlane G.R. , Burchett 2001) and Populus spp. (Laureysens et. al. 2004). In both cases, however, leaf metal load increased throughout the growing season**.** A seasonal reduction in leaf metal load, similar to that which was observed in our study was documented in Spartina alterniflora (Windham et al 2003) while examining plants from the contaminated sediments of the Hackensack Meadowlands, New Jersey. The proposed mechanism for the reduction however, was the secretion of metal cations via salt glands (Windham et al, 2001). Since B. populifolia does not excrete salts, a different mechanism must either actively shift the metal load to other (than leaf) tissue or Zn transport discontinues as leaves continue to grow.

Long Term Productivity

The long term impact of TML on hardwood productivity (measured as an increase in basal area) in B. populifolia, exhibited trends similar to those observed above. Total growth over the six year period demonstrated a statistically significant ($r^2 = 0.85$, $p < 0.01$) curvilinear relationship with TML with a critical TML of approximately 3. These results indicate that over the six years, the threshold model of metal-induced stress was applicable. In addition, since annual growth should increase each year for trees of this age class, we compared the slope of the line created by incremental/annual growth (basal area added per year) against TML. Once again, a highly significant ($r^2 = 0.93$, $p < 0.01$) curvilinear relationship was discovered. In addition, the negative slope at sites TP-14 and TP-14/16 (highest TML) indicates that their growth rate had actually decreased over the study period (Table 4). This study obviously does not address the cause of the observed decrease in growth. However as it has been noted, the growing conditions at all sites were relatively constant (i.e. equivalent site index), and the trees at higher TML level were growing in stands of low diversity. This suggests that intraspecific competition, in addition to the metal-induced stress, may also be contributing to decrease in growth.

Conclusion

This study demonstrates that vegetation indices such as NDVI and RG ratio can be predictive of metal-induced stress in hardwood assemblages on urban brownfields. In addition, it appears that a curvilinear relationship between NDVI and TML is reflective of the sub-lethal impact of metal-induced stress, in spite of the dominance of the metal-tolerant species, *B. populifolia* or at least metal-tolerant varieties of this species at the site.

Plant production, as indicated by leaf greenness index, was most inhibited in the co-dominant species B. populifolia, during the early part of the growing season. The negative correlation between plant productivity and Zn concentration in the leaf tissue was also significant, exhibiting the greatest decrease in productivity for the longest period of time at sites where the total metal load was greatest (TML at or above 3). Why there is a higher concentration of Zn in the leaf tissue and a decrease in productivity at this time is interesting and deserves further investigation. Perhaps the greater solubility of Zn, as compared to the other metals studied, especially during the spring when soil moisture is greater, facilitates Zn transport to the leaf thereby inhibiting chlorophyll production.

Even though yearly growth rates may vary considerably, the cumulative growth over time (six years) is significantly impacted at TML of 3 or above. In addition, when the TML is compared to the slope of the line created by the incremental growth, a strong curvilinear relationship was evident. Furthermore, at sites where TML was above 3 annual growth rates were actually decreasing.

Since this study has indicated that ecosystem function as measured by plant production is impaired at a critical soil metal load (above 3), site managers in such cases should consider mitigation. An organic soil amendment or a chelating agent that could reduce mobile fraction of the TML may improve productivity.

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Table 1. Comparison of soil metal concentrations (μg g^{-1}, mean ± S.D.) and with Soil Cleanup Criteria (SCC, 1999), the Lowest Observed Effective Concentration (LOEC, OakRidge, 1997) and Maximum Acceptable Toxic Concentration (MATC, ECO-SSL, 2003). All ecological criteria except for Hg were exceeded at most of the sites.

	As	**Cr**	**Cu**	**Hg**	**Pb**	**V**	**Zn**
MDL	0.005	<0.01	3.1	0.002	<0.01	<0.01	17.9
Min	<MDL	9.7±2.5	44.0±2.5	<MDL	86.0±11.1	<MDL	80.0±12.9
25%	16.7±2.5	22.5±4.8	74.0±11.6	0.1±0.1	185±38.8	26.9±15.2	93.1±21.5
Median	33.5±6.8	38.8±7.9	153±27.7	0.3±0.1	406.0±73.6	44.0±20.7	159.0±48.4
75%	121.9±29.8	60.2± 25.7	253.0±58.2	0.7±0.3	520±181.1	76.1±33.2	547.0±221.8
Max	977.6±44.3	208.8±10.4	1870.0±315.2	3.6±6.0	4640±1799	193.2±112.6	6501.0±1491
% above SCC (1999)	68	N/E	16	N/E	52	N/E	5
% above ECO-SSL	76	80**	84	N/E	88	50**	44**

* according to the EPA's Integrated Risk Information System (1998)
** % above the critical value of Lowest Ecological Effect .
N/E – none of the sites exceeded the given critical values
MDL: Minimum Detection Level

Table2. Soil Cleanup Criteria, recommended Generic Soil Remediation Standards (2004), the Low Ecological and Sever Ecological Levels (1998) screening criteria (μg/g).

Metal	**As**	**Cr**	**Cu**	**Hg**	**Pb**	**V**	**Zn**
SCC (1999)	20	240	600	14	400	370	1500
ECO-SSL	18 (MATC)	15 LOEC)	70 (MATC&EC_{10})	32.15 (LOEC)	120 (MATC)	26.25 (LOEC)	231 (LOEC)

*Cr standard was defined by EPA (Integrated Risk Information System, EPA, 1998)
MATC: Maximum Acceptable Toxical Concentration
$EC_{10:}$ Effect Concentration for 10% of test population
LOEC: Lowest Observed Effect Concentration

Table 3. Linear regressions between leaf Zn concentration and RG ratio index values at each site and the composite. These data indicate a significant correlation at sites with the highest TML and weaker correlations at sites with lower TML. (n=5)

Site	r^2	p	**TML**
TP 14/16	0.81	0.04	3.6
TP 14	0.81	0.04	3.1
TP 18	0.71	0.08	2.8
TP 43	0.31	0.33	1.6
TP 43-14	0.6	0.12	0.9
TP 41	0.46	0.21	0.85
Composite	0.41	>0.01	

Table 4. Linear Regressions between growth rates mm tree core. (n=7), only significant results are shown.

Site Comparisons	r^2	p
48-41	0.58	0.05
48 - 14/16	0.75	0.01
48 - 43	0.76	0.01

Table 5. Slope for incremental (yearly) growth as a measure of basal area added. (n=5).

Site	**Slope**
TP 14/16	-22.43
TP 14	-15.68
TP 43	47.52
TP 43/14	51.95
TP 41	71.77
TP 48	65.86

Table 6. Linear regression Values: Normalized Difference Vegetation Index (NDVI) vs. Total Soil Metal Load (TML). (n=5)

Site	**TML**	r^2	p
TP 14/16	3.6	81.3	0.04
TP 14	3.1	81	0.04
TP 18	2.8	70.7	0.08
TP 43	1.6	30.8	0.33
TP 43-14	0.90	60.2	0.12
TP 41	0.85	45.6	0.21
All Sites		40.5	>0.01

Figure 1. Total Soil Metal Load (TML) throughout the study area and the distribution of Hardwood Assemblage (hatched areas)

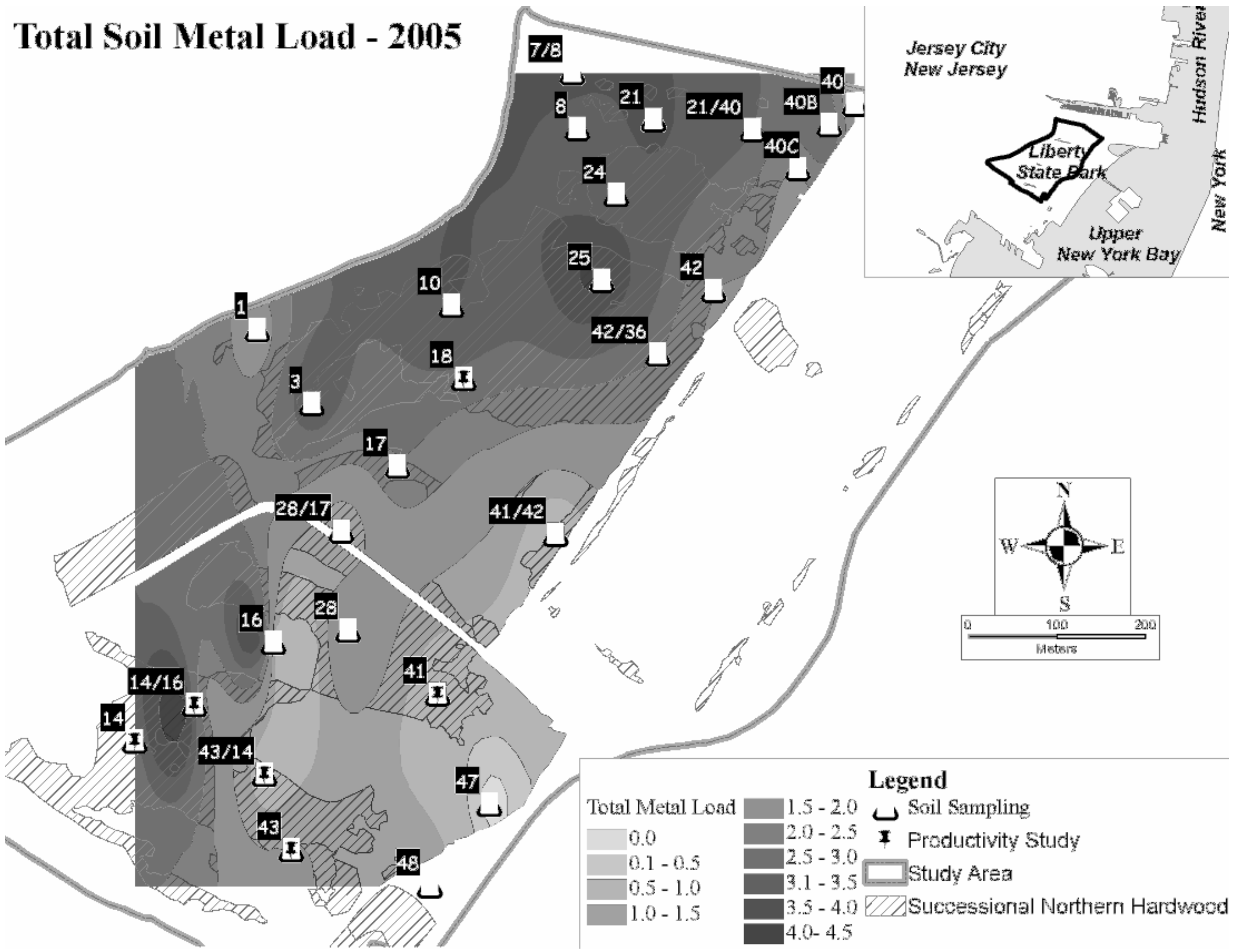


Figure 2: Curvilinear relationship between TML and the mean NDVI in the Hardwood Assemblage and TML. These data indicate that a threshold model of metal tolerance explains the distribution of the hardwood assemblage.

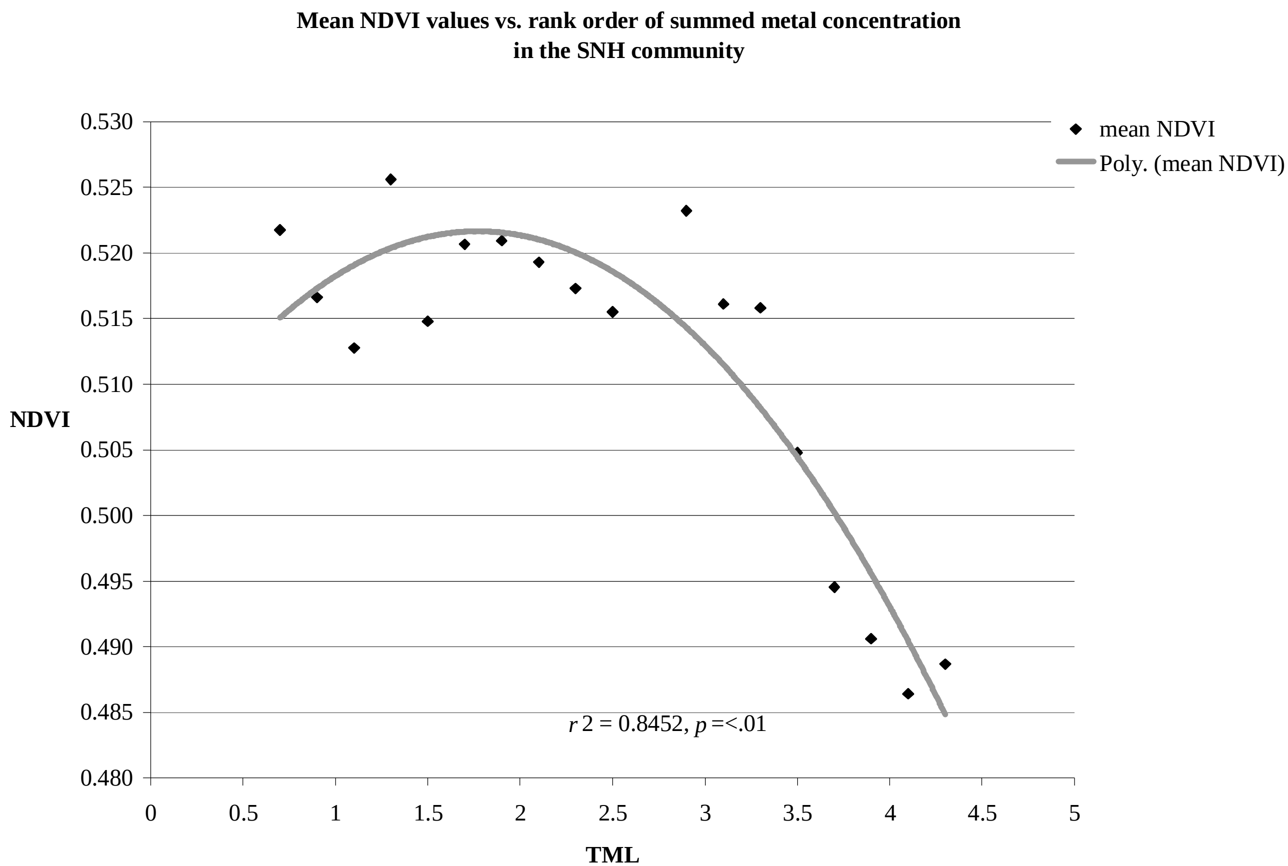


Figure 3a: The concentration of Zn ($\mu g\ g^{-1}$) in the leaf tissue of B. populifolia demonstrated a significance increase during the spring. The sites are ranked from left to right by increasing Total Soil Metal Load.

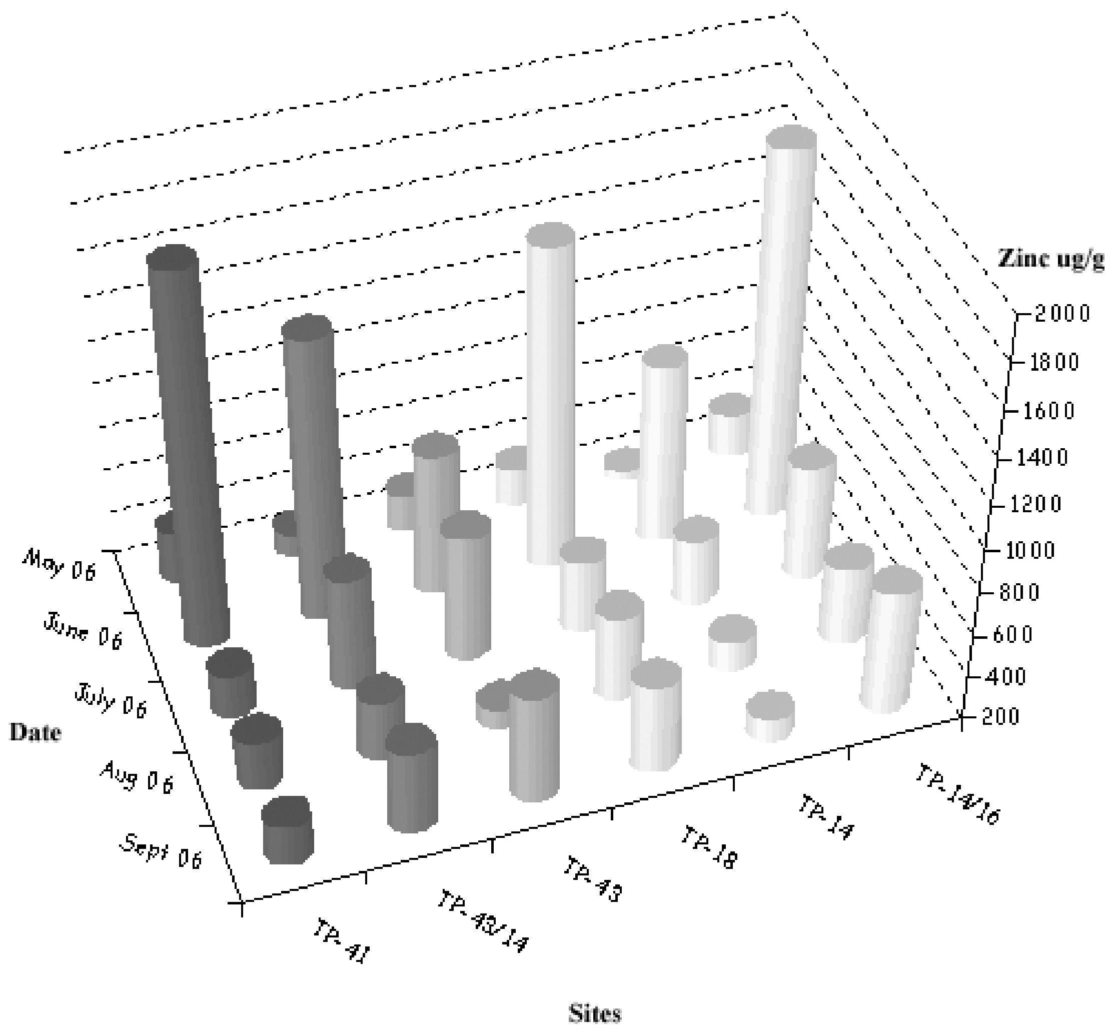


Figure 3b: The Red/Green Ratio Index calculated from leaf reflectance of B. populifolia demonstrates a strong correlation with the increase in leaf Zn (Figure 3a). The sites are ranked from left to right by increasing Total Soil Metal Load.

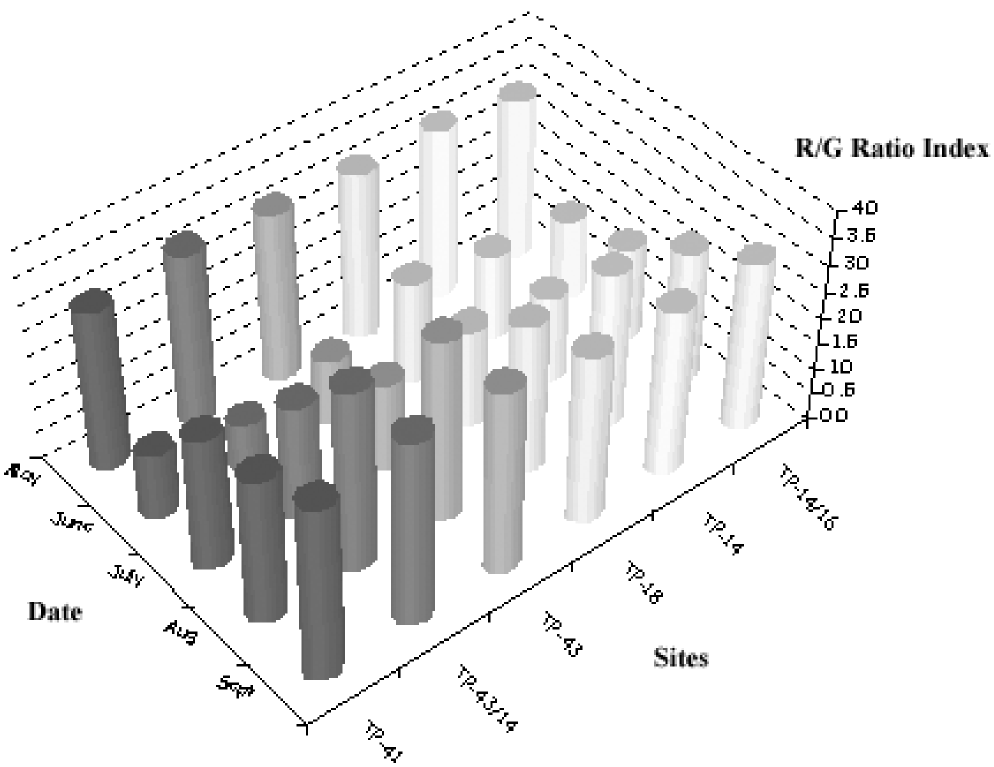
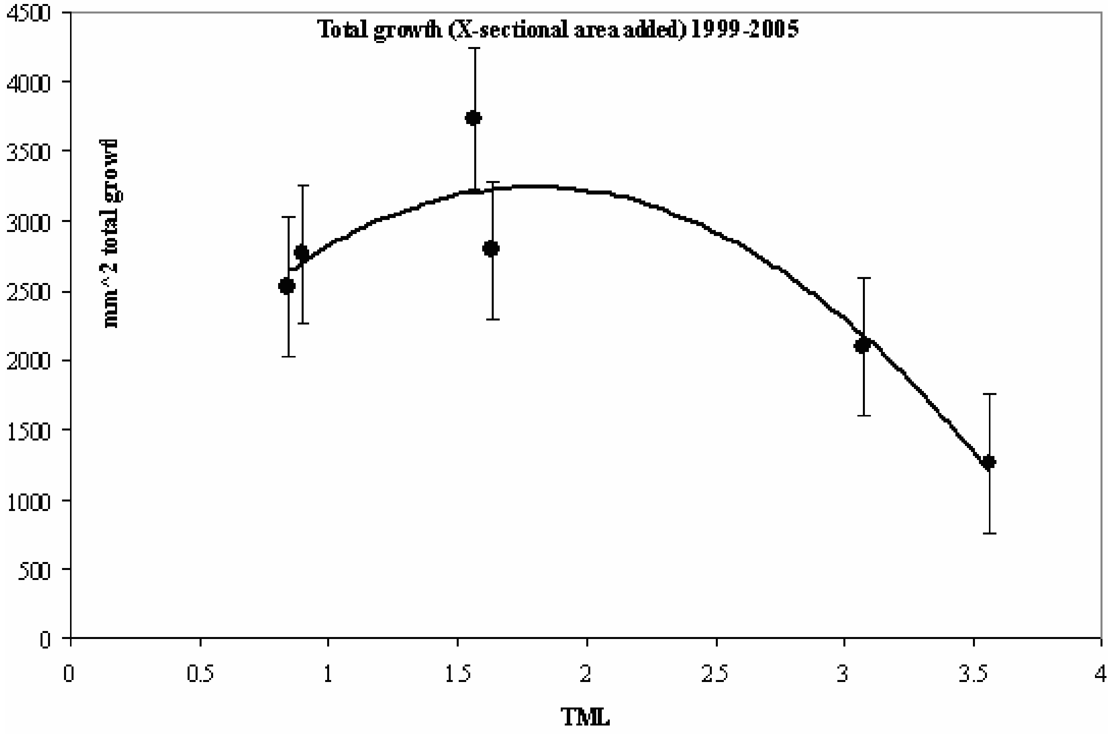


Figure 4: Cumulative cross sectional area added, (at dbh) for the six year period of 1999 through 2005 in mm^2 vs. TML. These data are reflective of the NDVI results and indicate a significant (r^2=0.85, p<.01) long term growth inhibition above a TML of 3.



Chapter IV

Morphological Variation in the Seed of *Betula populifolia* Marsh.:

The Effects of Soil Metal Contamination.

Abstract

Seed dispersal effectiveness is an important factor that contributes to the success of plant migration and early assemblage development. Variation in seed architecture determines the dispersal strategy employed and the distance the seed travels. In addition, few species of trees are known to colonize disturbed sites that exhibit higher than normal concentrations of soil metals typical of urban brownfields. The patchy distribution and soil contamination of brownfields result in strong abiotic filters, which limit recruitment from regional species pools. In the northeastern part of the United States, one of the most successful species in these situations is Betula populifolia Marsh (gray birch). Our previous work demonstrated that B. populifolia had dominated the study site and its distribution could be positively correlated with increasing soil metal loads. Hence, we questioned whether the size and weight of B. populifolia seeds collected from areas of various soil metal loads within an urban brownfield would differ, thereby impacting their dispersal effectiveness. If seeds with a specific architecture were being selected for in these situations, it would rapidly alter genetic variation within the urban populations of this species. The results of this study indicate that B. populifolia employs a strategy whereby different wing loading rates result from varying the size and weight of its seeds. The decrease in seed size correlated well with total soil metal load, while the correlation with seed weight was marginal. Wing loading rates (seed weight/seed surface area) however, exhibited no significant relationship with soil metal load, indicating that potential for seed distribution from areas of high soil metal load had not been impaired.

Key words - seed dispersal; rate of descent; metalliferous soil; assemblage structure; pappus; wing load.

Introduction

In 1900, approximately 20% of the people in the United States lived in cites (U.S. Census Bureau 1900). Today more than 80% are urban dwellers (U.S. Census Bureau 2000). This fundamental change in where we live has undoubtedly changed who we are and how we came to understand natural systems. Increasingly then, urban green-space will provide the experiential framework required for the development of ecological identity, so critical to concepts of sustainability and the development of a land ethic. Much of the future new urban green space can be found in today's brownfields.

Assemblage development or redevelopment can be viewed as a process whereby a series of filters allow for the selection of certain species from a regional pool (Weiher & Keddy, 1995; Diaz et al., 1998). Typical filters include both abiotic and biotic limiting factors (Hobbs & Norton, 2004). While many biotic filters, such as germination success, competition, and seed predation are important, seed dispersal is also a primary concern. This is especially true in urban areas where open space is fragmented, making seed recruitment more difficult. In addition, seed morphology generally correlates well with distribution strategies. Of the three vectors associated with seed dispersal, animal, water, and wind, wind-dispersed species tend to produce functionally similar structures that increase buoyancy in relationship to weight. While many species across a broad taxonomic range exhibit such structure, it has been suggested (Matlack, 1987) that diaspore architecture is the result of functional and phyletic constraints.

Abiotic factors associated with assemblage development typically include climate, availability of water, soil chemistry, and disturbance regime. Of these, soil chemistry is known to have a significant impact on the development of woody species, especially when growing on mesic or xeric sites. In a study of soil types in a northern temperate deciduous forest, the pattern of stand development after disturbance was dependent upon soil type variations at a very small scale (Liptzin & Ashton, 1999).

Furthermore, in urban landscapes most soils have been compromised by human activity, especially those activitiesassociated with the industrial revolution. Such soils typically contain trace metals such as cadmium, copper, zinc, lead and/or others (Dudka et al., 1996). These elements are often adsorbed or occluded by carbonates, organic matter, Fe-Mn oxides, and primary or secondary minerals (Adriano, 1986; Ross, 1994). Since metals in small concentrations are known to be essential for plant growth, the mechanisms for uptake are inherent, if somewhat species specific. Most plants will tolerate or have a requirement for metals at concentrations within a particular range (functional range). At concentrations above the functional range plants can no longer maintain homeostasis, and metabolic functions are inhibited

(Hilary & Wilkins, 1987). Hence, soil metal contamination can impact plant metabolism and therefore contribute to assemblage development.

B. populifolia is a pioneer species that commonly invades old fields and burnt-over or cleared land, and generally competes/grows well in soils of little nutritional value (Elias, 1980). In addition, it often colonizes urban green space and is the co-dominant tree species (35% cover) of the study area (Gallagher, et. al., 2007). The tree produces small, winged seeds that are wind-dispersed. It has also been demonstrated that seed impingement (attachment to a substrate) and germination is greater in mineral soils with little leaf litter (Houle, 1992). The dispersal range of the seed is a function of its small size and relatively light wing loading rates. If the metal contaminated soil was altering the architecture and hence wing loading rate of seeds from contaminated sites, it could result in a strong selective pressure rapidly alter the genetic composition of these isolated populations within urban environments.

Since plants grown in metalliferous soils often express growth abnormalities (Gardea-Torresdey et al., 2004), we questioned whether B. populifolia colonizing the study site exhibit modification in seed morphology, especially seed size and weight. Our previous work had demonstrated that B. populifolia dominated the study site and its distribution could be positively correlated with increasing soil metal loads (Gallagher et al., 2007). In addition, we demonstrated that soil metal loads above a critical threshold concentration negatively impacted primary productivity and yearly growth of trees in the study area (Chapter III). The purpose of this study was to determine whether seeds, from areas of high soil metal load exhibited morphological differences that correlated with total soil metal load (TML).

Materials and Methods

Study Area

Liberty State Park (LSP) is located (40^{o} 42' 16, 74^{o} 03' 06) on the west bank of Upper New York Bay in Jersey City, N.J., and is one of the most successful urban brownfield restoration initiatives in the state. The land that comprises the park was originally part of an intertidal mud flat and salt marsh that was filled for use as a rail yard. The fill materials consisted primarily of debris from construction projects and refuse from New York City, which were deposited to stabilize the surface between 1860 and 1919. Between 1864 and 1967, the Central Railroad of New Jersey (CRRNJ) used the site as a rail yard for both freight and passenger service. Due to the industrial land use, higher than normal levels of soil metals are unevenly distributed throughout the site (Gallagher et al. 2007). In the center of the park there remain approximately 251 acres that are undeveloped. In spite of the contamination the area has been colonized by various plant assemblages. These assemblages represent unique associations of both endemic and non-native species that can be considered the by-product of the cultural events that have taken place during the past 150 years. The hardwood assemblage is dominated by B. populifolia (35% cover) Populus deltoides (16% cover) and P. tremuloides (14% cover).

Procedure

The total soil metal load (TML) for the study area has previously been described (Gallagher et al., 2007). Soil samples were collected from 32 sites over an area of approximately 40 ha and examined for concentrations of arsenic (As), chromium (Cr), copper (Cu), mercury (Hg), lead (Pb), vanadium (Va), and zinc (Zn). TML was determined as a rank order transformation of the mean soil metal concentration as described by Juang et al. (2001). The results were back-transformed, using the reverse function of the linear regression, performed between the original metal data and the ranks, as published by Wu et al. (2005). The resulting index provides a means to evaluate the cumulative TML at any one site using a scale from 0-5, with 5 indicating the highest concentrations (Figure 1).

Several catkins were collected from six similar-sized trees in different areas from within the study area on October 8, 2007. The six sites were chosen because they had a similar site index, were dominated by B. populifolia and exhibited a broad range in TML. Sites chosen were numbered 14, 14-16, 18, 41, 43, and 48. The seeds from each site were stripped from the catkins and thoroughly mixed. Approximately two-hundred seeds from each sampled tree were randomly separated for examination. Each seed was measured across the entire seed, including both pappus and germ as the entire surface area contributes to wing loading rates (Figure 2a). The seeds were segregated according to size at the 0.1mm level. Since the rate of vertical descent and resulting lateral transport of wind-dispersed seed is a function of both size and weight, the seeds were then weighed using Sartorius F1800 balance. As seed weight was often below 0.1 mg, they were weighed in groups (usually between 3 and 5). These results were then divided by the number of seeds weighed to determine individual seed weight.

The surface area of the seed was calculated using the formula SA = πr^2C, where C is a ratio representing the difference between the surface area of several typical seeds, measured using ArcMap (ArcGIS, 2004) and the surface area of a circle that encompassed those seeds (Figure 2b). The entire seed surface area was used to calculate wing load as the germ size varied considerably (32-46%) at each site. Ten seeds from each site were photographed and their surface areas calculated. The mean of these ten was used as the constant (C) for each site. Wing loading rates were then calculated by dividing the weight by the surface area.

Standard ANOVA analysis was used to determine if the relationship between seed size and weight were significant. Tukey grouping was used to determine similarity between data sets and a two-tailed t-test was used to determine if the difference between seed size and weight above and below the TML threshold were significantly different. The descriptive statistical parameters and regressions were calculated with Minitab (Minitab version 14.2, 2005) and SAS System version 9.1 (SAS, 2003), with levels of significance set at α =0.05. The Anderson–Darling normality test was used to assess seed size distribution within each sample.

Results

Seed size at each site varied by as much as 220% and was normally distributed with far greater numbers in the mid-size classes (Table 1). Tukey analysis indicates that two sets of data (sites 41 and 18, 14 and 14-16) produced seeds of statistically similar size. Interestingly, the TML at sites 41 and 18 (0.85 and 2.85, respectfully) were below the previously defined threshold for soil metal tolerance of 3.0, whereas sites 14 and 14-16, which exhibited the smallest mean seed size, were above the threshold. Site 48 exhibited the largest mean seed size; however, the mean seed size at sites 41, 48, 43 and 18 varied by less than 17%. A two-tailed t-test between the sites above and below a TML of 3.0 was significant ($F = 0.76$, $p<.01$), indicting that seed size at sites 14 and 14-16 were significantly smaller than the rest of the samples.

With two exceptions, site 48 and site 14-16, seed weight did not necessarily increase with seed size. Correlations between seed size and weight for each site were inconsistent (Table 2). The correlation at site 48 was marginal ($r = 0.48$, $p = 0.04$) but statistically significant. At site 14-16, which had the highest TML, the relationship was strong ($r = 0.82$, $p < 0.01$). In addition, four of the correlations were positive and two were negative. Separation of data into subsets of sites above and below the soil metal tolerance threshold value of 3.0 indicate a weak but significant relationship ($F = 0.26$, $p < 0.01$).

The wing loading rate (weight to surface area ratio) generally decreased with increasing seed size. Correlations between wing loading rates and seed size within each site yielded significant relationships (Table 3). Wing loading rates of seeds above the TML threshold of 3 were significantly lower than those below 3 ($F=.11$, $p<.01$, Figure 3). The mean wing loading rate was negatively correlated with TML, however, the correlation was not statistically significant ($r = -0.31$, $p = 0.55$). While both sites above the TML threshold exhibited the lowest wing loading rate, the trend of the quadratic polynomial regression between mean wing loading rate and TML marginally indicates a threshold model of metal tolerance, it is not statistically significant ($r^2 =.15$, $p = 0.55$) (Figure 4).

Discussion

Minor differences in seed structure have been correlated to significant differences in dispersal performance (Dolan,1984; Matlack,1987; Peroni,1994). Higher wing loads produce faster rates of descent and hence decrease the potential for dispersion. On the other hand, lighter wing loading rates, which can be produced either by decreasing mass or increasing pappus size, can increase dispersal effectiveness. Our study indicates that seed size and weight in B. populifolia scale independently, producing a variety of wing loading rates, thereby increasing dispersal effectiveness.

In addition to variation in size and weight as described above there are several other mechanisms that help to explain the effectiveness of B. populifolia as an early colonizer of disturbed areas. In these fragmented landscapes, especially those isolated within urban settings, it is long distance distribution of seed that maintains gene flow within the metapopulation (Cain et al., 2000). Wind-borne seed dispersal in plants tends to be biphasic: it behaves in accordance with the aerodynamic nature of seed when winds are under a specific velocity, but once that critical wind velocity is exceeded, the size and shape become less important, as the force of the wind is strong enough to carry the seed above the tree canopy. The wind velocity required to lift the seed above the canopy is not great with extremely small and light seed, such as those of B. populifolia. Seeds pushed above the canopy tend to have far greater travel distances than seeds that are released and remain within or below the canopy (Horn et al., 2001), thereby enhancing dispersal effectiveness.

Also, catkins of B. populifolia tend to be persistent, lasting well into the winter. The scales and seed tend to separate first at the terminus of the catkin, proceeding gradually backward. The staged release of seeds correlates well with an increase in the prevailing westerly wind and increasing lift that result after the leaves are shed from the tree as sparse canopies, such as those with little or no leaf mass, tend to have a more organized vertical eddy motion (Nathan & Gabriel, 2005).

Wing loading rates within the sites ranged considerably; from 360% at site 14-16 to a high of 600% at site 18, indicating the potential for varied distribution exists at all sites. Wing loading rates from sites with higher TML were lower than the other sites; however this significance can be attributed to the fact that these sites produced little seed with high wing loading rates (Figure 3). Mean wing loading rates were more consistent ranging from 0.03 at site 14/16 where TML was greatest to 0.05 mg/mm^2 at site 18, where our previous studies indicated greatest growth rates. At sites 14 and 14-16 which exhibit the highest TML, the mean wing loading rates were slightly lower (Figure 4). Hence, seed from the area of high TML would respond to the biphasic wind velocity principals similar to seed from areas of lower TML. .

Seed size and weight or mass are involved in distinct, food storage vs. flight, but related ecological functions (Matlack, 1987). While seed with lower wing loading rates generally achieve greater dispersal distance, diaspores with higher wing loading rates generally have heavier seed that produce more vigorous seedlings (Dolan, 1984). These competing selective pressures have resulted in a broad spectrum of seed dispersal strategies. Using B. papyrifera (paper birch), Borkbum et al. (1965) suggested that seed weight and germination success were positively correlated. If such relationships between seed size, germination, and vigor are applied to the results of this study they could indicate that many of the lighter seeds may not be successful. This has strong implications for sites 14 & 14-16 where TML concentrations are higher, which exhibited the lightest mean weights (0.8 mg. and .13 respectfully). The relatively strong positive regression (r = 0.48, p = 0.04) between seed size and weight at site 14-16, may indicate that the tree could not partition resources to favor germ weight. Further research is needed to determine if these seeds are less viable than those from sites below the TML threshold.

Conclusion

This study confirms earlier work, which indicates that B. populifolia, like other birch species, produces small wind-borne seed with considerable variation in size and weight. These data also suggest that wing loading rates are altered by changes in both the size of the pappus and the weight of the seed. Variability is further enhanced by the fact that seed size and weight do not scale evenly. This variability has the potential to produce broad range of wing loading rates, well suited for colonization in the patchy habitat of urban areas.

Mean seed size and weight were smaller at sites where TML was greater than 3.0, resulting in seed with slightly lower wing loading rates. These results correlate well with patterns of productivity measured at these sites in previous studies. The results also indicate that wing loading rates and hence the potential for dispersion are not negatively impacted by higher TML. Further research is needed to determine if the seeds from area of higher TML are viable. However if the seeds are viable then the slightly enhanced dispersal effectiveness created by the lower wing loading rates may provide a selective advantage for colonization of the patchy, metal contaminated open space in urban environments. If on the other hand the seeds are less viable, then the success of B. populifolia on metalliferous soils must be due to a genetic tolerance that is maintained in the metapopulation.

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Table 1: Seed Size Distribution (mm), n=number of individuals, StEr = standard error, Min=minimum value, Max=maximum value, subscript letters indicate Tukey Groupings.

Site	N	TML	Mean	St Er	Min	Max
14_a	205	3.08	2.06	13.34	1.4	2.6
$14/16_a$	202	3.56	2.15	12.56	1.4	3.1
41_b	200	0.85	2.60	7.96	2	3.4
18_b	201	2.85	2.63	13.53	1.9	3.2
43_c	199	1.64	2.90	7.33	2	3.4
48_d	215	1.56	3.09	7.62	2	4.2

Table 2: Pearson correlations between seed size and weight at each sample site. In general it appears that size and weight do not scale evenly. The one exception, site 14-16 is the site with the heaviest TML.

Site	Correlations
41	$r = .02$, $p = 0.13$
48	$r = .48$, $p = 0.04$
43	$r = -.07$, $p = 0.85$
18	$r = -.09$, $p=.76$
14	$r = .21$, $p=.45$
14/16	$r = .82$, $p>.01$

Table 3 Pearson correlations comparing wing loading rates and seed size indicate a reciprocal relationship. The relationship proved statistically significant at all sites.

Site	Correlations
41	r = -.88, p<.01
48	r = -.82, p<.01
43	r = –.69, p<.01
18	r = -.66, p=.01
14	r = -.76, p<.01
14/16	r = -.70, p<.01

Figure 1: Total Soil Metal Load (TML). All sampling sites were used to block-krig the original soils metal data used to create the TML map.

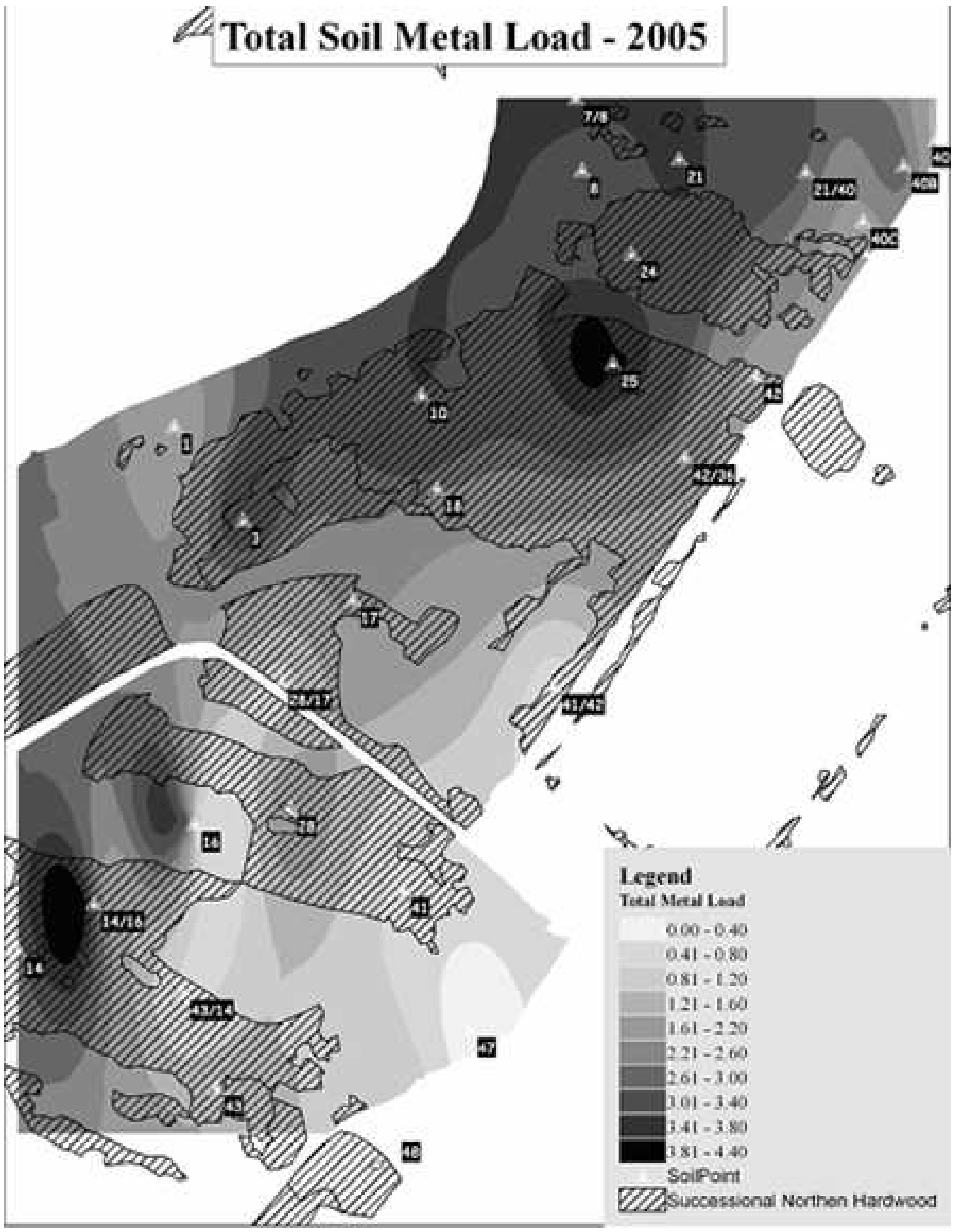

Figure 2: Each of the approximately 200 seeds from each site was measured across the entire width, including both pappus and germ, to determine a radius. The surface area of the seed was calculated using the formula $SA=\pi r^2 C$. C is the mean ratio representing the difference between the surface area of 10 typical seeds, measured using ArcMap and the surface area of a circle that encompassed those seeds.

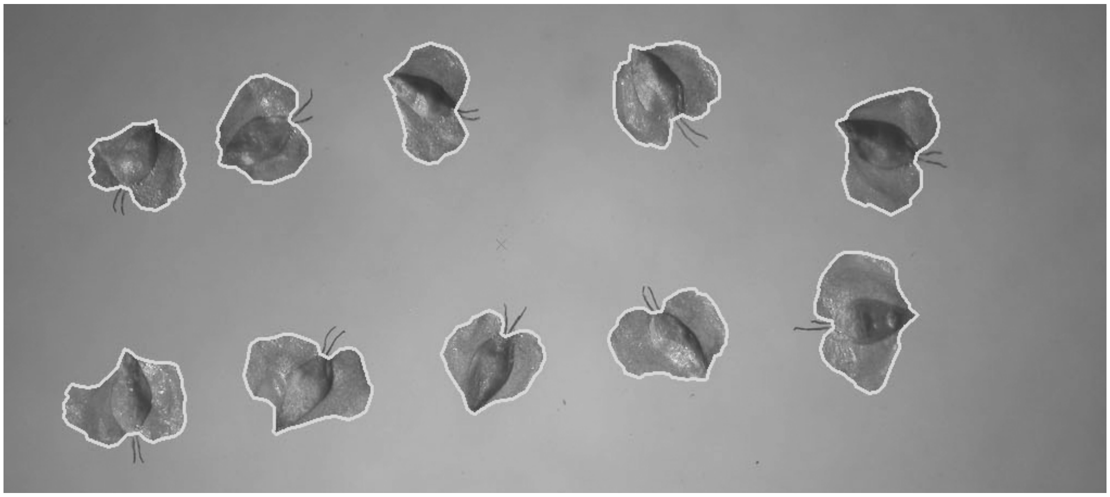

Figure 3: Wing loading rates by seed size. Those below the TML threshold of 3 (sites 14 & 14-16) were lower than those above 3 (F=.11, *p*<.01). The major difference is that these sites produce little seed with high loading rates.

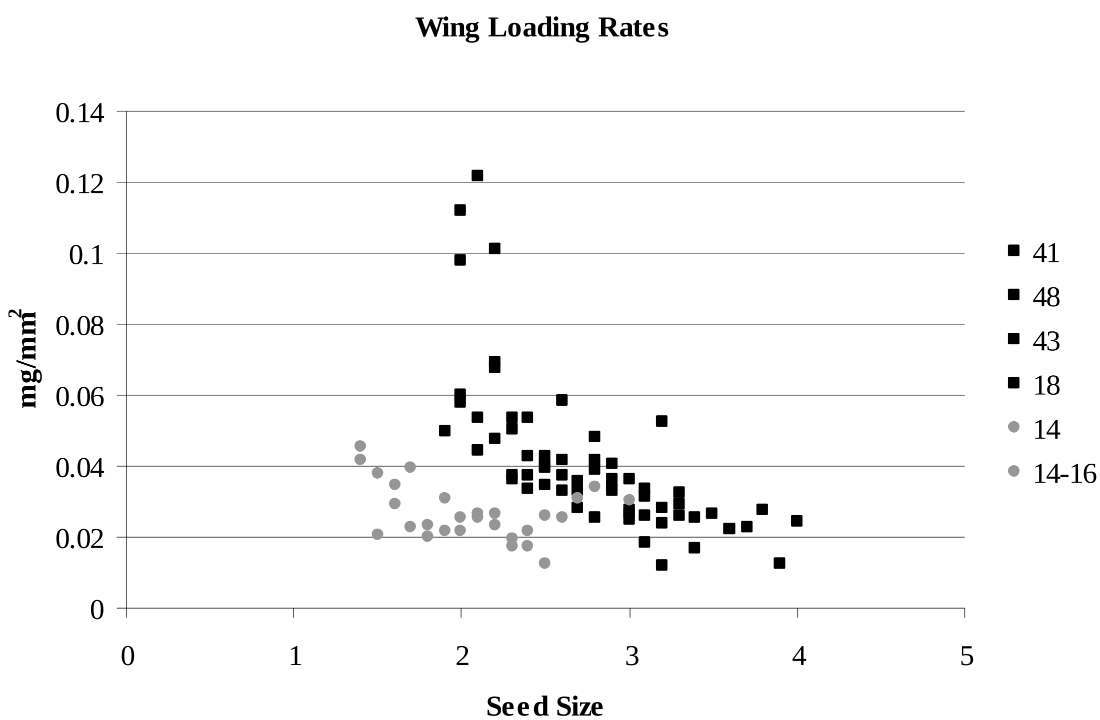

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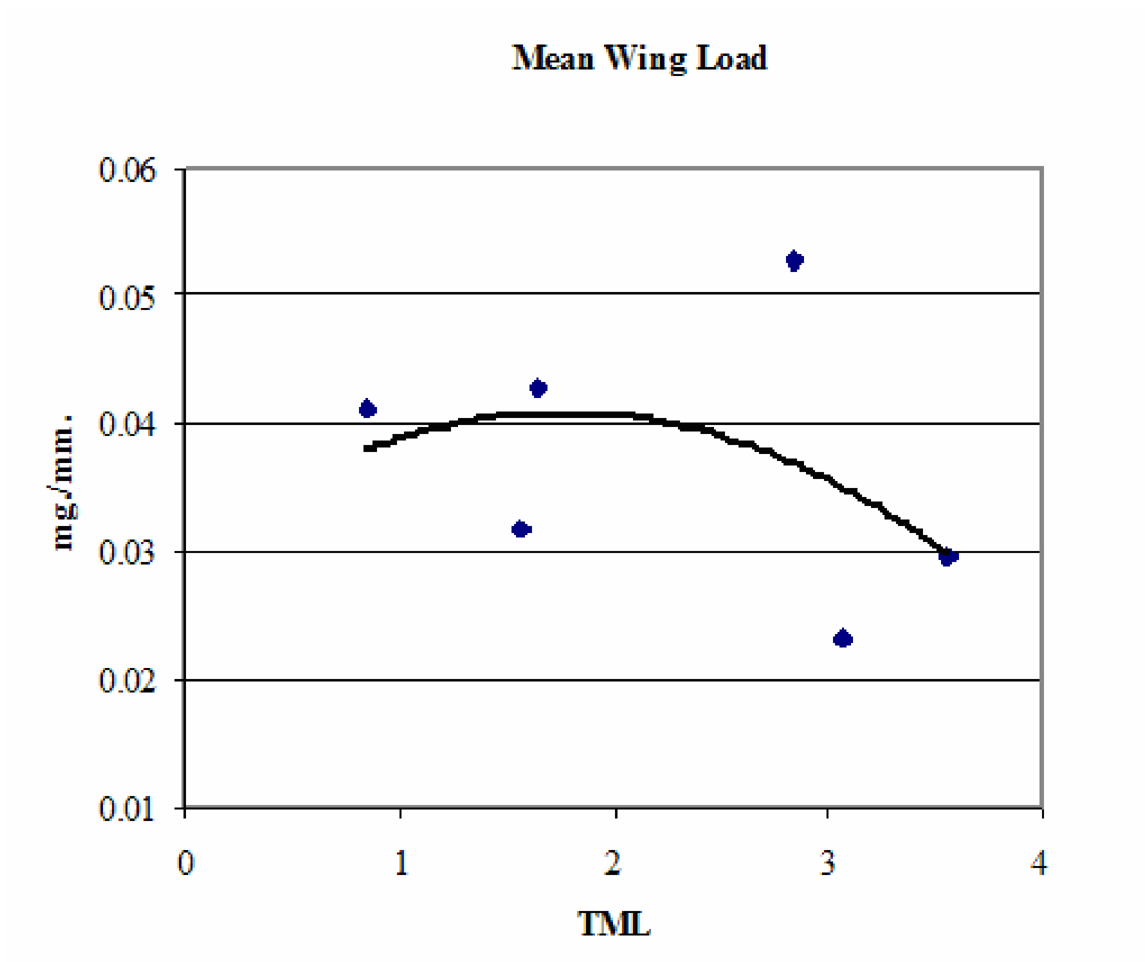
Figure 4: The relationship between the mean wing loading rates and TML. The polynomial regression indicates that a threshold model may be exhibited, however the polynomial regression was not statistically significant (r^2 =.15, p = 0.55).



Chapter V

Vegetative Assemblage Development in an Urban Brownfield:

Summary and Synthesis

Importance of Urban Ecosystems

During this year (2008) an unprecedented global demographic transition will occur, the majority of us will live in cities (Flavin, 2007). This fundamental change in where we live will undoubtedly change who we are and how we come to understand natural systems. There are now over 20 urban centers that house at least 10 million people; in 1950 there were only two. While they are generally considered the home of the best creative and artistic talent, the pioneers of ground breaking public policy and considerable economic drivers they are often simultaneously the sites of abject poverty and extreme environmental degradation. If sustainability is truly a human goal then the development of a functional land ethic must be possible within the paradoxical context of the urban environment. Increasingly then, urban green-space, will provide the experiential framework required for the development of ecological identity, so critical to the establishment to the concept of a land ethic and the practice of sustainability. While the proceeding papers do not pretend to address these global environmental and social issues, they do help us better understand urban ecologies, which in turn help to foster stronger ecological identities.

Recognizing the growing importance of both structure (maintenance of biodiversity) and function (fostering natural cycles) of urban ecologies, we have examined assemblage composition, distribution, productivity and finally seed distribution, at a site where the soil metal load is significantly above concentrations considered normal for the region. The results indicate that soil metals at higher concentrations impacted the distribution of certain vegetative assemblages and plant productivity especially at or above threshold concentrations. However, the relationship was not always obvious nor was it always linear. In addition, the concentration of individual soil metals appears to be less important than the total soil metal load (TML). Hence, plant productivity in the hardwood assemblage exhibited a threshold response to increasing soil metals, significantly decreasing when the total soil metal load was above a critical concentration. Finally, the potential for seed dispersal in the co-dominant tree species, B. populifolia, appears not to be negatively impacted by the soil metal load. A synthesis of these results along with additional corroborating evidence, not presented in the preceding chapters, and the possible implications for plant assemblage development is offered below.

Metal Translocation at the Species Level

We first question whether the dominant plant species were favoring one of the three known physiological pathways of metal assimilation. If the dominant species were either excluding, passively accumulating or hyperaccumulating soil metals it would have indicated the presence of a physiological advantage under this type of metal induced stress. We analyzed plant tissue from the roots, stems and leafs for the presence of metals. While the results from the four most dominant species were presented in chapter two, the results from all of the eight species that were tested are presented below (Table 1). Metal translocation differed considerably from species to species; however, there were some recognizable trends. All data indicate that As is the most stable of the metals studied, exhibiting little translocation into the plant tissue. Both Cr and Cu exhibited trends similar to As, with slightly higher rates of accumulation in the root tissue. Also, soil and root Cr exhibited a strong linear correlation in both B. populifolia (N = 8, $r^2 = 0.99$, $p < 0.01$) and R. copallinum (N = 5, $r^2 = 0.62$, $p < 0.05$). No other statistically significant relationships between soil and root metal concentration were observed. Tissue concentration of Cr, and Cu exhibited a log normal distribution, often exhibited in terrestrial plants (U.S. Department of Energy 1998) and also wetland plant (Peverly, et. al. 1995), where the soil concentration is generally an order of magnitude higher that the concentration in the roots, which is also an order of magnitude greater than that found in aerial sections of the plant. Pb was generally sequestered in the root tissue of the species examined. Translocation of Pb to the aerial section of the plant particularly the leaves, was only detectable in two specimens of B populifolia. A notable exception was the generally high concentrations of Zn observed in leaf tissue of P. deltoides and B. populifolia. In one case, the bioaccumulation of Zn in the leaf tissue of B. populifolia exceeded soil concentrations by an order of magnitude.

Translocation of Zn has been studied in several species of birch, for example B. papyrifera, B. pendula, and B. pubescens are known to be tolerant of and have the ability to accumulate metals, particularly Zn and Pb (Prasad, 1999; Margui et al., 2007). To our knowledge however, this study was the first to demonstrate Zn accumulation in the leaf tissue of B. populifolia at concentrations higher than that of the soil.

In summary, the data indicate that all three methods (i.e.: exclusion, passive uptake and a form of hyperaccumulation) of metal translocation occurred in the species examined. Hence there was no specific physiological rule that could be applied to the establishment of species on the site.

Metal Attenuation

The study site is relatively isolated within an urban community, exhibiting no surface water discharge and bounded on one side by the Hudson River. In addition, soil pH was generally, between 5 and 7 with a mean of 5.7. Groundwater movement towards the northeast is extremely slow and the known saltwater intrusion along the eastern boundary reaches approximately 10 meters. Hence we also questioned whether the difference in the soil metal concentration measured in 1995 and 2005 could be correlated with rates of biological uptake.

It is known that metals are often adsorbed or occluded by carbonates, organic matter, Fe-Mn oxides, and primary or secondary minerals (Adriano, 1986; Ross, 1994), and since there have been no human impacts at the site that would change the concentration of Fe or Mn between 1995 and 2005, we expected that the change in soil metal concentrations at the site could be correlated to the translocation rates in the various plant assemblages. In addition, it has been demonstrated that leaf litter can provide a sink for metals, which bind passively to organic surfaces or actively through the physiological activity of the microbial colonizers (Gadd, 1993; Ledin, 2000). Leaf litter can also act as a source when microbial activity mobilizes the metal (Gadd, 1993) or through the action of deposit feeders (Weis and Weis 2004). Hence, metal sequestration or adsorption is dependent upon the rates of uptake and retention by the various tissue types, translocation to deposit feeders and release through decomposition.

Comparing soil metal data from 1995 to that of 2005, there was a significant decrease in several of the soil metal concentrations (Appendix IV). These data suggest that as the rate at which metals were metabolized (and presumably bound) increased, (i.e. As<Cr<Cu<Pb<Zn) the rate of attenuation, the difference between soil metal concentration of 1995 and 2005, also increased (r^2=.83, p=.03) (Figure 1). Hence, in this young terrestrial system, the continued addition of organic material as a result of plant growth appears to be acting as a sink for the examined soil metals. If true then the impact the abiotic filter imposed by the higher soil metals should decrease over time, leading to an enhanced ability of species with lower soil metal load tolerance to be successful at the site, suggesting that the current *B. populifolia* dominated hardwood assemblage is may not be the logical end point for assemblage development at this site.

On the other hand, Zn did translocate to the aerial sections of the dominant tree species and especially into the leaf tissue. Hence, the potential for higher levels of Zn to be incorporated into and cycling within the leaf litter and upper soil horizons exists. Whether or not this feedback loop will result in a biological filter strong enough to prevent the establishment of climax hardwood species (i.e. Quercus sp, Fagus sp., and Carya sp. etc.) typical for the region deserves further consideration. If so the assemblage composition trajectory could terminate with the current B. populifolia- and P. deltoids- dominated mix, resulting in an alternative steady state, contrary to what has been proposed above under the continued soil metal attenuation postulate.

While these data are interesting and suggest that natural attenuation of metal contamination is possible at the study site, there are several concerns. The 1995 soil samples were not taken in triplicate; therefore the variation in the data could not be analyzed, whereas the 2005 soils data represents the mean of triplicate samples. Hence their comparison is suspect. In addition, to accurately assess sequestration of soil metals the equation must be mass balanced for the various tissue types (i.e. root, steam and leaf). Future soil samples using the same protocols and enhanced plant metric measurements will make this possible.

Assemblage Distribution and Total Soil Metal Load

Under normal conditions, metals are present in soils as derivatives of parent geologic materials. Plant tissue can therefore be expected to contain these metals, and as mentioned above, in some cases at concentrations higher than in the surrounding soil. At concentrations above functional ranges, however, the plant can no longer maintain homeostasis and metabolic functions are inhibited (Hilary and Wilkins, 1987). It follows, then, that a species intolerant of high metal concentrations would be excluded from the assemblage on the site even if a seed source existed within the regional pool. Therefore, vegetative assemblages growing on metalliferious soil should exhibit structural differences when compared to similar but uncontaminated regional environs. For example, Kimmerer (1981) and Kalin and van Everdingen (1988) demonstrated that succession 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 succession associated with 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 tailings studies, it appears that the hardwood assemblage at LSP has developed preferentially on the soils with increased total metal load (TML vs. % cover, $r = 0.78$, $p < 0.01$) (Figure 2), perhaps as a function of reduced. In contrast the emergent marsh assemblage, those not dominated by invasive species, appear to be better established in areas of low TML (TML vs. % cover, $r = -0.70$, $p < 0.01$). Hence these two guilds appear to be reacting to the soil metal load in reciprocal fashion (also Figure 2).

The findings of this study more closely resemble the work of Sanguer and Jetschke (2004) where birch was an earlier colonizer of a former uranium-mining dump. They argue that the early arrival of tolerant 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. Historic aerial photographs were digitized and the four guilds, hardwood forest, wetlands, shrub and herbaceous plants, were mapped using ArcMap (ArcGIS, 2004). An examination of the results indicates that species trajectories in areas where the TML was high (3 or above) favored rapid development of B. populifolia- or Populus sp.- dominated assemblages (Figure 3 and 4). The rapid development of the hardwood assemblage in these areas appears to inhibit the development of shrubs at the higher TML.. Interestingly in both scenarios the hardwood community overtakes the herbaceous communities in the 28th year. In addition guild trajectories between year 31 and 34 are relatively flat. This could be an early indication that the current assemblages are entering a period of stasis. Hence, 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 (Keddy 1992, Weiher and Keddy 1999) rather than focusing primarily on competition or facilitation between species.

Primary Productivity and Soil Metal Load

In chapter III we report on the relationship between primary productivity and TML, in the hardwood assemblage, measured at the assemblage level and the individual tree level. We also report on the seasonal variation of leaf productivity and leaf Zn concentration within those trees. In each case negative correlations were observed between plant production and metal load.

The negative correlation and curvilinear relationship between primary productivity within the hardwood assemblage and TML in this study (Figure 5) offers strong support for a threshold model of metal tolerance. Such growth response appears to be fairly common at the species level and it has been suggested (Hillary and Wilkins 1987) that it might be due to the plants' ability to maintain internal concentrations of metals at certain levels regardless of external concentrations. It appears that the curvilinear relationship between NDVI and TML is reflective of the sub-lethal impact of metal-induced stress, which becomes significant at a TML of approximately 3 (on a scale of 1-5), in spite of the dominance of metal-tolerant tree species or at least metal-tolerant varieties at the site.

Not reported in chapter III was the fact that the productivity in the successional old field assemblage also yielded significant correlations (r=-.63, p=.02) with TML (Figure 6) however, this relationship was linear. It should also be remembered that successional old field was dominated by native grasses and herbs. Herbaceous assemblages dominated by either P. australis or Artemisia vulgaris did not exhibit significant correlations between productivity and TML (r^2=.07, p=.34), indicating that within these assemblages primary productivity and the resulting competitive effectiveness is not impaired by increasing soil TML. Furthermore, the distribution of these assemblage types yielded a significant positive and linear relationship (r=.47, *p*=.03) with increasing TML, thereby indicating a competitive advantage over the successional old field in areas of higher TML. Finally, production within the maritime shrub assemblage also did not correlate significantly with increasing TML (*r*=.20, *p*=.86). However, the correlation between its distribution (% area covered) and TML was significant (*r*=-.76, *p*<.01). In this case productivity is not inhibited while, as mentioned earlier, while distribution exhibits a pattern that suggests a strong relationship. Hence, it appears that the distribution of the maritime shrub assemblage may have more to do with competition from the hardwood assemblage, which had colonized areas of high metal load early in areas of high metal load.

Seed Dispersal in Betula populifolia

As urban ecology plays an increasingly important role in the re-gentrification of former industrial landscapes (Gamborg and Larson 2003) an understanding of forest assemblage development or redevelopment in these areas is critical. However in the urban context the persistent soil contamination and the patchy distribution of available open space, present strong abiotic filters. Hence, chapter three examines seed dispersal in one of the most successful species known to colonize urban green space, B. populifolia.

Wing loading rates within the sites ranged considerably; from 360% at site 14-16 to a high of 600% at site 18, indicating that the potential for varied distribution exists at all sites. Such variation is consistent with similar work on other species of birch (Matlack, 1987, 1989). Mean wing loading rates were more consistent ranging from

0.03 at site 14/16 where TML was greatest to 0.05 mg/mm^2 at site 18 (Figure 7). Hence, seed from the area of high TML would respond to the biphasic wind velocity principals (behaving in accordance with the aerodynamic nature of seed when winds are under a specific velocity, but once that critical wind velocity is exceeded, the size and shape become less important) similar to seed from areas of lower TML.

While seed with lower wing loading rates generally achieve greater dispersal distance, diaspores with higher wing loading rates generally have heavier seed that produce more vigorous seedlings (Dolan, 1984). If such relationships between seed size, germination, and vigor are applicable to B. populifolia the results of this study could indicate that many of the lighter seeds from the sites with high TML may not be viable. Low viability of seed from the patchy areas of high TML would indicate that the dominance of this species in these areas was dependant on external seed sources.

Synthesis

The growth and distribution of the species encountered and the resulting vegetative assemblages have been impacted as described above by TML. Furthermore at least within the hardwood assemblage species diversity (Shannon Index) also exhibits a significant negative curvilinear relationship with TML (Figure 8). Hence, the response in both structure and function to the environmental stress gradient resulting from the increasing TML has resulted in alternative assemblage trajectories above the critical TML threshold. The difference in these ecological histories and the continued presence of higher than normal soil metal concentrations has the potential to lead to the development of alternative endpoints above and below the critical TML threshold.

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Table 1: Translocation of metals into various (l=leaves, st=stem, r=roots) plant tissue. µg g^{-1}.

	As				Cr				Cu				Pb				Va				Zn			
Species	**l**	**r**	**st**	**s**	**l**	**r**	**st**	**s**	**l**	**r**	**st**	**s**	**l**	**r**	**st**	**s**	**l**	**r**	**st**	**s**	**l**	**r**	**st**	**s**
Artemisia vulgaris	0.00	1.68	0.0	37.4	0.38	5.93	0.37	38.2	19.0	81.9	8	345	0	67	1	563	11.51	8.94	0.46	58.0	95.4	377	172	1058
Artemisia vulgaris	0.00	0.55	0.0	14.2	0.62	2.06	0.15	20.2	21.4	57.0	7	224	0	22	1	421	11.47	2.57	0.00	39.0	107	374	118	603
Artemisia vulgaris	0.00	2.29	0.0	11.9	0.40	6.63	0.07	38.4	20.0	85.4	7	168	0	91	1	349	11.61	11.3	0.00	29.0	122	339	154	207
Artemisia vulgaris	0.00	1.08	0.0	13.2	0.09	6.38	0.00	20.3	23.3	46.3	11	153	0	47	1	303	6.98	8.60	0.00	193	54.8	59.8	50.7	159
Artemisia vulgaris	0.00	0.52	0.0	15.7	0.28	2.67	0.07	32.1	3.13	40.6	2	129	0	43	1	552	4.94	3.94	0.00	0.00	1.81	239	21.5	157
Onoclea sensibilis	0.00	2.91	0.0	35.1	0.72	13.4	0.51	53.4	4.62	48.0	4	67.0	0	47	0	168	14.82	21.8	0.13	72.3	2.69	37.5	25.5	38.7
Phragmites australis	0.00		0.0	35.1	0.27		0.00	53.4	5.41		4	67.0	0		1	168	2.06		0.00	72.3	2.09		30.2	38.7
Polygonum cuspidatum	0.00	2.56	0.0	12.5	0.51	12.7	0.06	43.4	4.29	40.3	7	48.3	0	43	1	156	9.40	22.5	0.00	18.1	98	131	59.1	96
Solidago virgaurea	0.00		0.0	11.9	0.47		0.12	38.4	9.30		10	168	0		1	349	8.76		0.11	29.0	69.5		396	207
Rhus copallinum.	0.00	0.75	0.0	12.5	0.37	4.92	0.00	41.3	5.14	126	1	124	0	121	1	453	8.71	6.12	0.00	19.6	54.7	694	14.8	309
Rhus copallinum.	0.00	0.92	0.0	38.9	0.58	10.1	0.00	70.1	5.58	35.7	2	379	0	27	1	341	7.49	11.6	0.08	44.2	29.1	147	11.3	168
Rhus copallinum.	0.00	1.24	0.0	57.1	0.31	2.42	0.00	50.6	4.44	17.3	3	199	0	28	3	409	4.28	3.25	0.00	9.31	2.51	165	23.5	218
Rhus copallinum.	0.00	1.21	0.0	24.8	0.04	6.66	0.12	46.6	2.98	53.5	3	212	0	54	0	421	6.19	11.8	0.00	66.9	1.06	49.2	4.19	232
Betula populifolia	0.00	0.90	0.0	87.4	0.25	9.34	0.00	37.8	4.32	33.0	1	103	0	94	4	383	9.63	3.31	0.00	20.7	256	320	67	49.9
Betula populifolia	0.05	3.34	0.0	42.8	0.57	52.8	0.00	209	5.57	51.4	2	76.4	0	332	16	196	11.11	28.0	0.24	7.37	824	153	118	24.9
Betula populifolia	0.00	1.77	0.0	33.3	0.06	10.7	0.00	27.6	6.93	120	1	238	0	74	4	393	9.08	10.6	0.04	118	1284	458	241	491
Betula populifolia	0.01	1.64	0.0	22.5	0.03	13.7	0.00	68.5	5.45	20.1	1	44.3	0	56	4	173	5.96	16.5	0.00	37.8	109	145	64.2	72.5
Betula populifolia	0.08	0.91	0.0	13.3	0.13	1.83	0.00	9.7	6.11	28.6	2	229	0	60	3	449	8.09	5.78	0.00	40.6	1617	303	243	304
Betula populifolia	0.02	1.56	0.0	21.0	0.37	3.86	1.32	10.2	5.29	40.6	2	68	0	112	26	86	9.38	7.92	0.00	118	1051	81.3	70.0	198
Betula populifolia	0.00	7.16	0.0	17.6	0.19	2.83	0.64	11.4	6.15	36.0	3	69.9	8	181	22	117	7.30	10.6	0.00	8.28	1062	135	112	49.0
Betula populifolia	0.00	1.26	0.0	10.7	0.48	9.44	0.12	16.7	5.14	38.7	1	166	10	125	10	333	12.09	3.98	0.00	79.8	1034	169	69.6	64
Populus tremuloides	0.00	0.95	0.0	23.6	0.20	3.18	0.00	44.0	8.62	43.6	3	249	0	92	1	486	5.35	6.73	0.00	24.9	1693	487	127	718
Populus tremuloides	0.04	3.25	0.0	270	0.16	2.76	0.00	40.4	8.05	113	3	1527	0	199	9	4640	4.86	8.87	0.05	54.0	1657	604	152	1586
Populus tremuloides	0.00	20.20	0.0	193	0.33	9.16	0.00	61.6	8.99	78.4	2	257	0	93	4	500	8.24	11.7	0.00	27.0	1309	270	81.8	131

Figure 1: The difference between soil metal concentrations in 1995 and 2005 (Delta Soil) compared to the summation of the tissue concentration (Sum(LRS)) of for each metals.

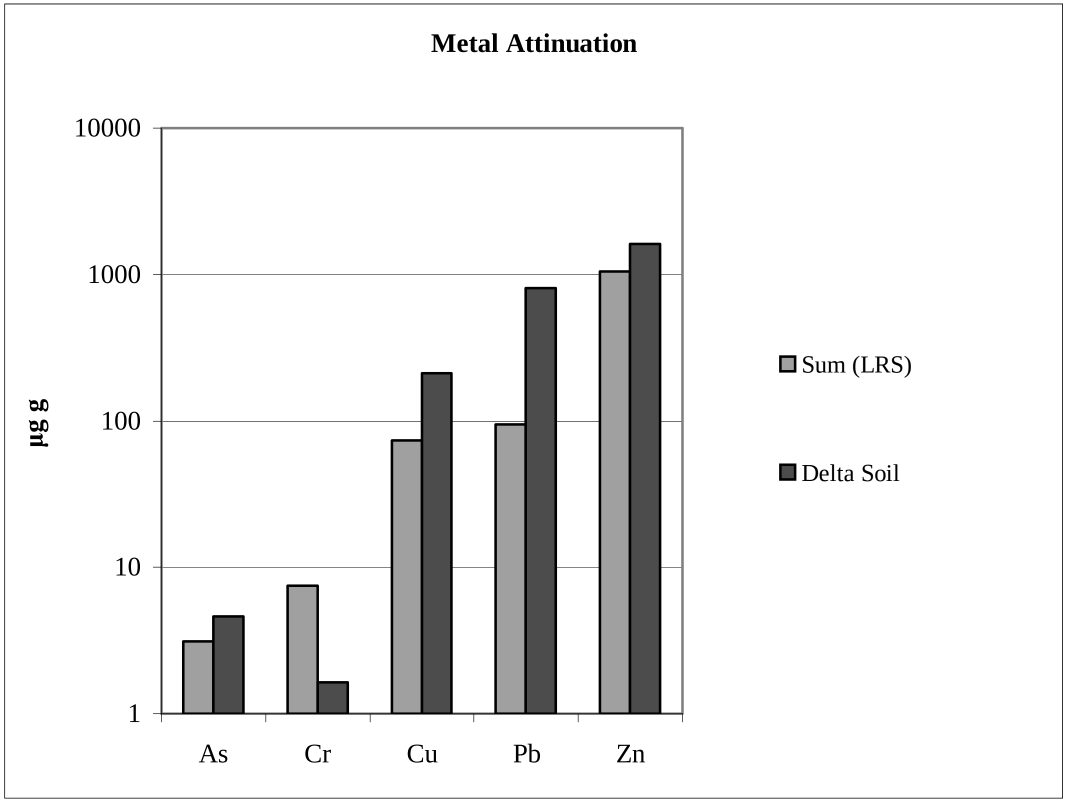


Figure 2: The relationship between total soil metal load and percent cover of the hardwood assemblage (SNH, $r^2 = 0.62$, $p < 0.01$) and the emergent marsh assemblage (SEM, $r^2 = 0.49$, $p < 0.01$), which were not dominated by invasive species.

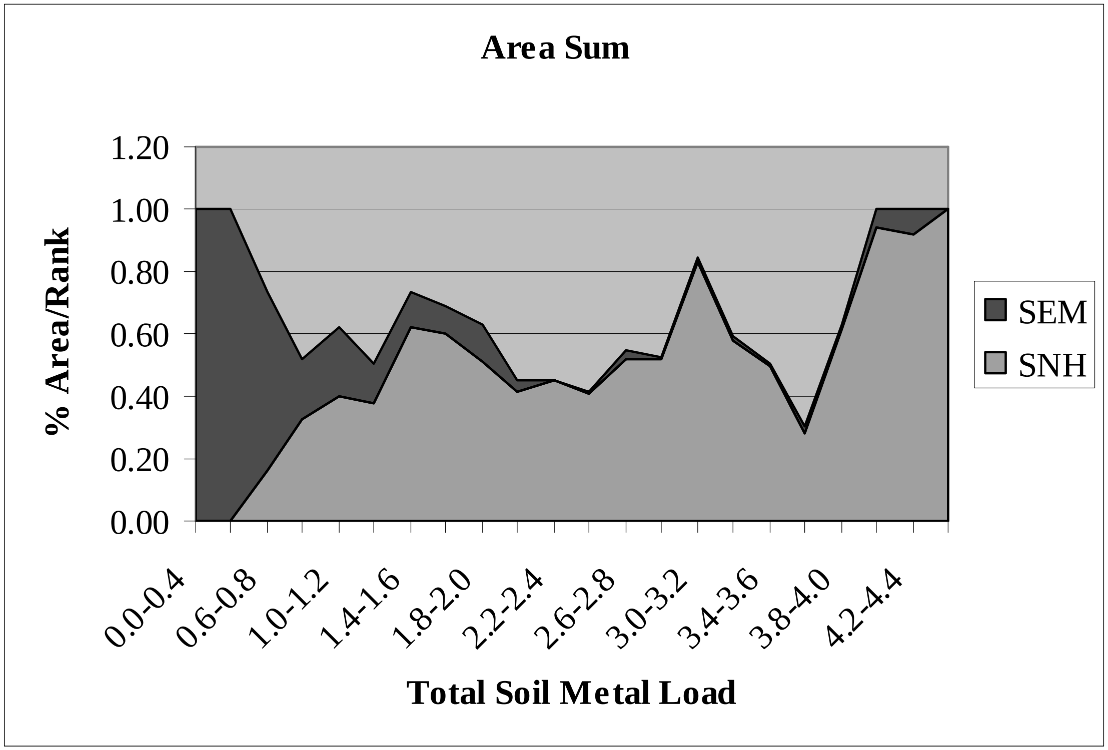


Figure 3: Assemblage Development Trajectory, TML < 3. The relative similarity (% covered) between shrub and hardwood is maintained between years 6 and 23.

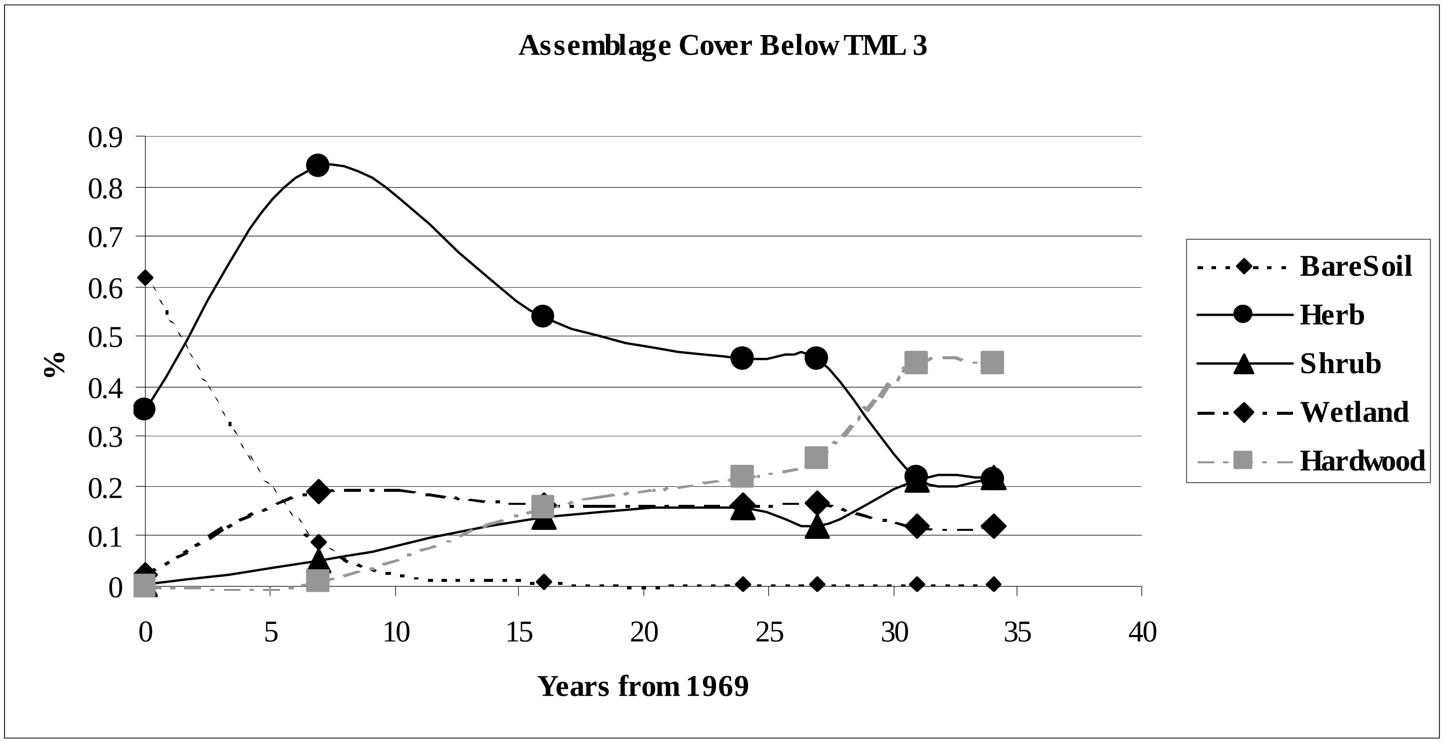


Figure 4: Assemblage Development Trajectory, TML > 3. The Tree assemblage increases in size rapidly beginning at year 6, while the shrub assemblage remains minimal until year 26.

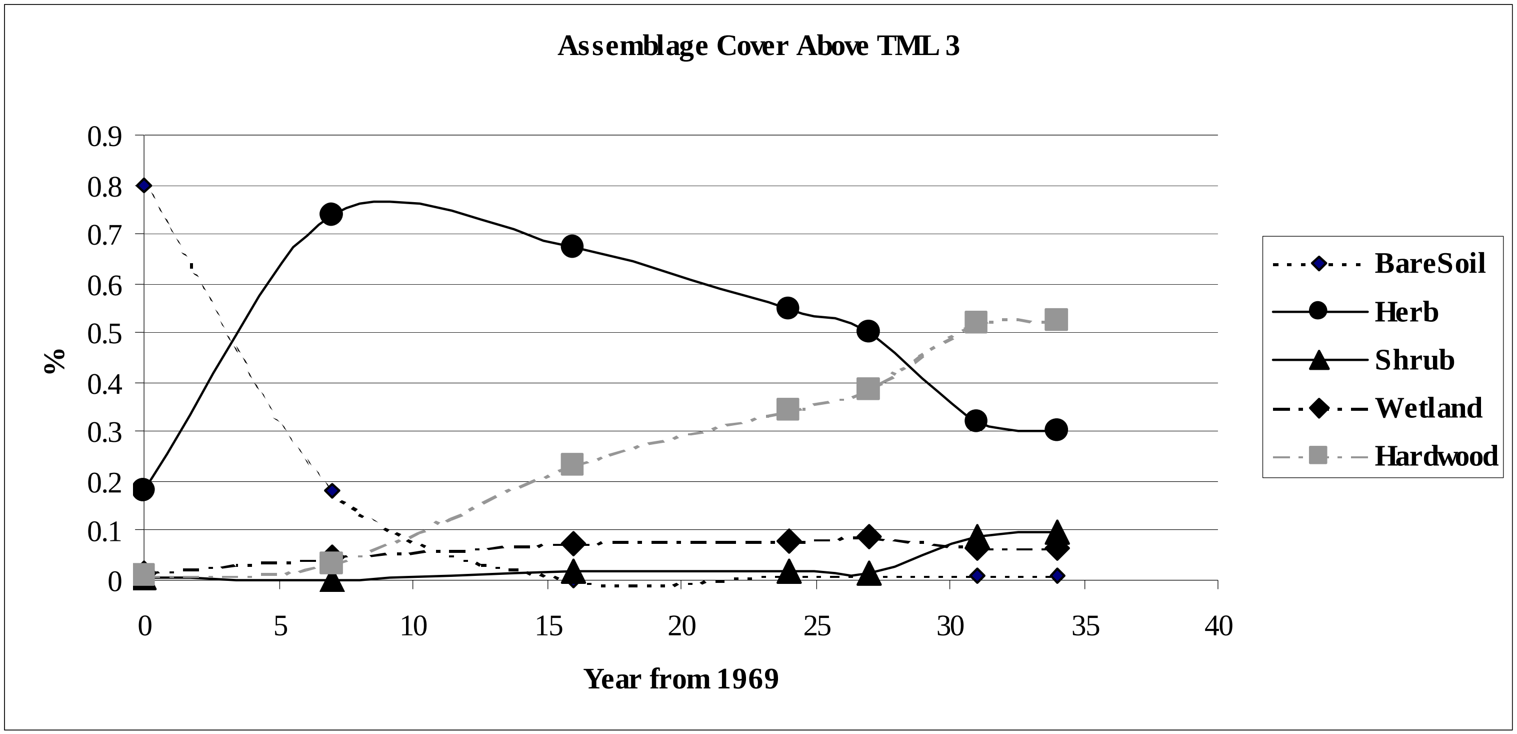


Figure 5: The relationship (quadratic polynomial) between TML and NDVI (primary productivity) in the Hardwood Assemblage.

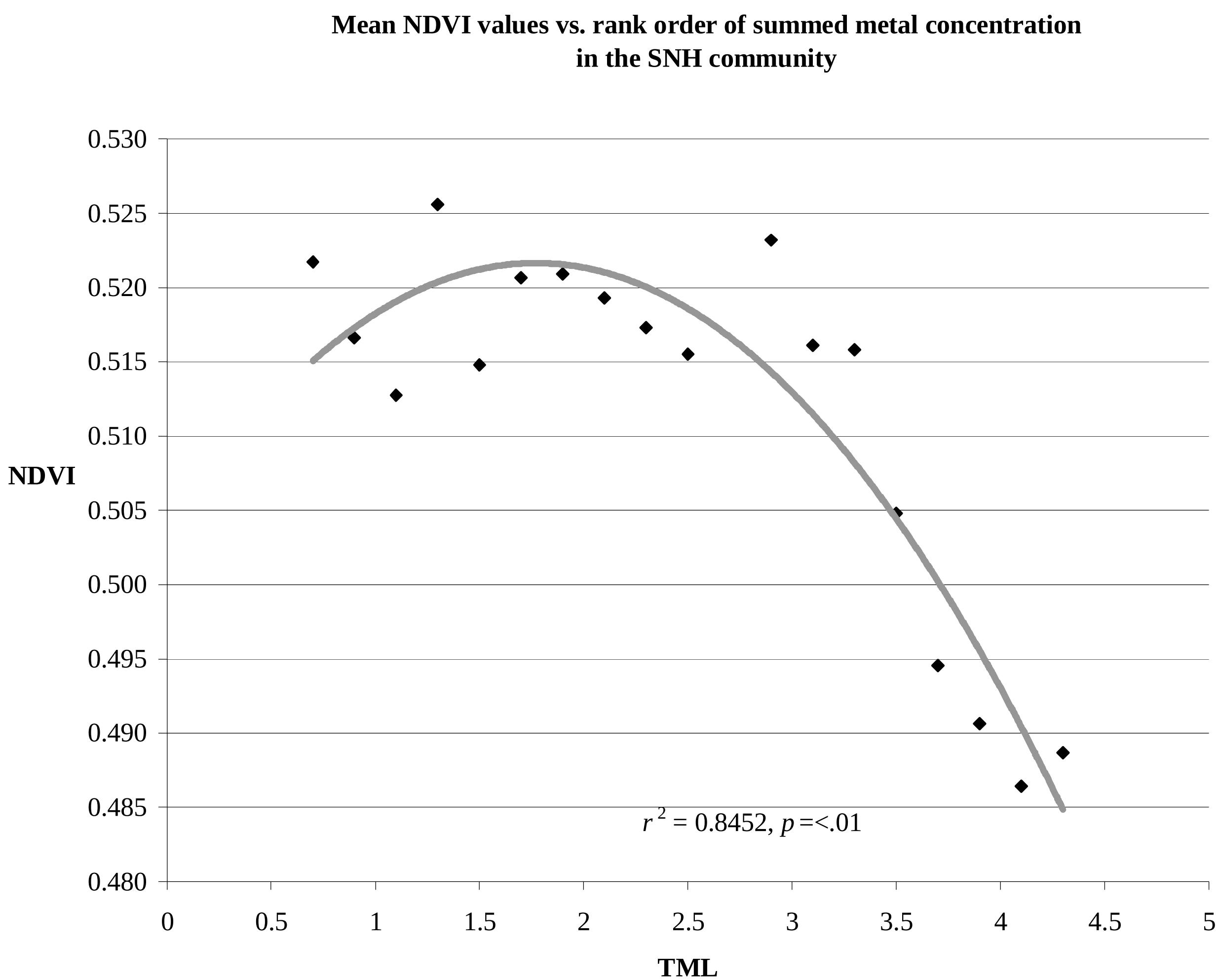


Figure 6: The correlation (*r*=-.63, *p*=.02) between TML and NDVI (primary productivity) in the successional old field assemblage

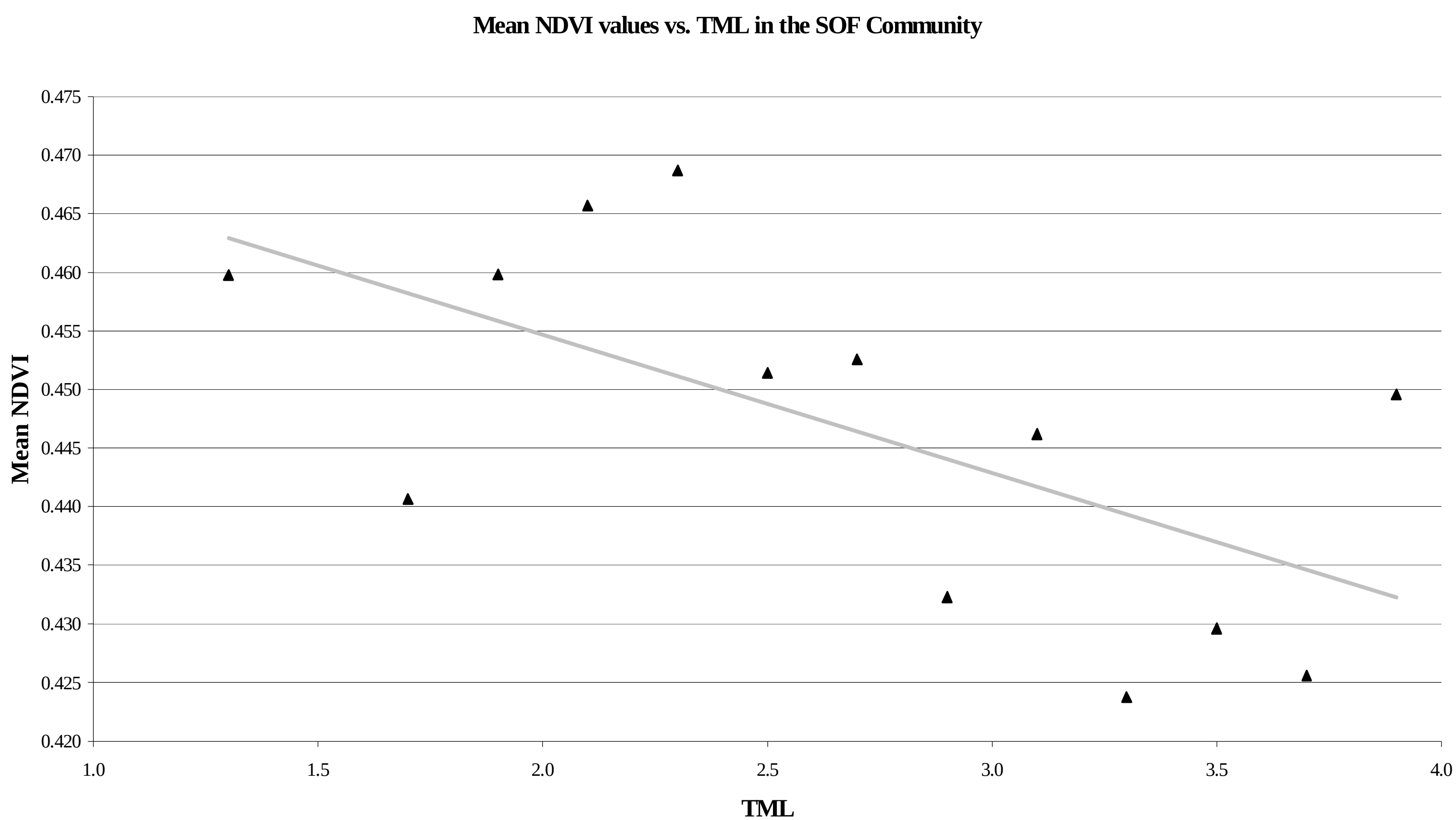


Figure 7: The relationship between the mean wing loading rates and TML. The quadratic polynomial regression indicates that a threshold model may be exhibited, however the polynomial regression was not statistically significant (r^2 =.15, p = 0.55).

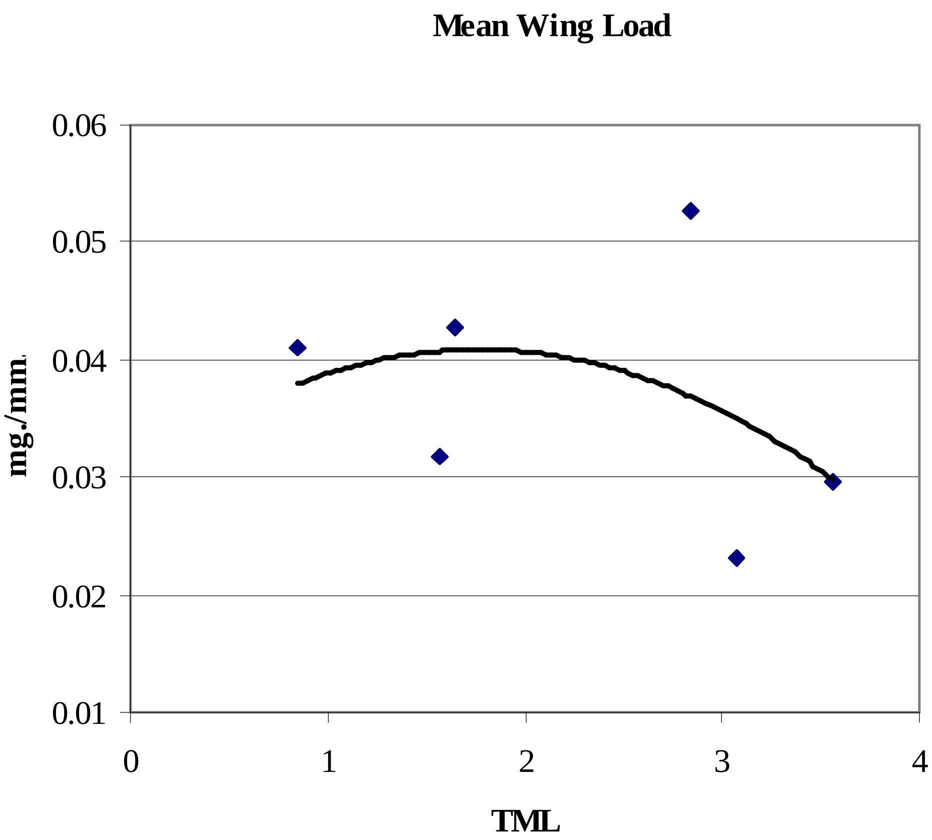
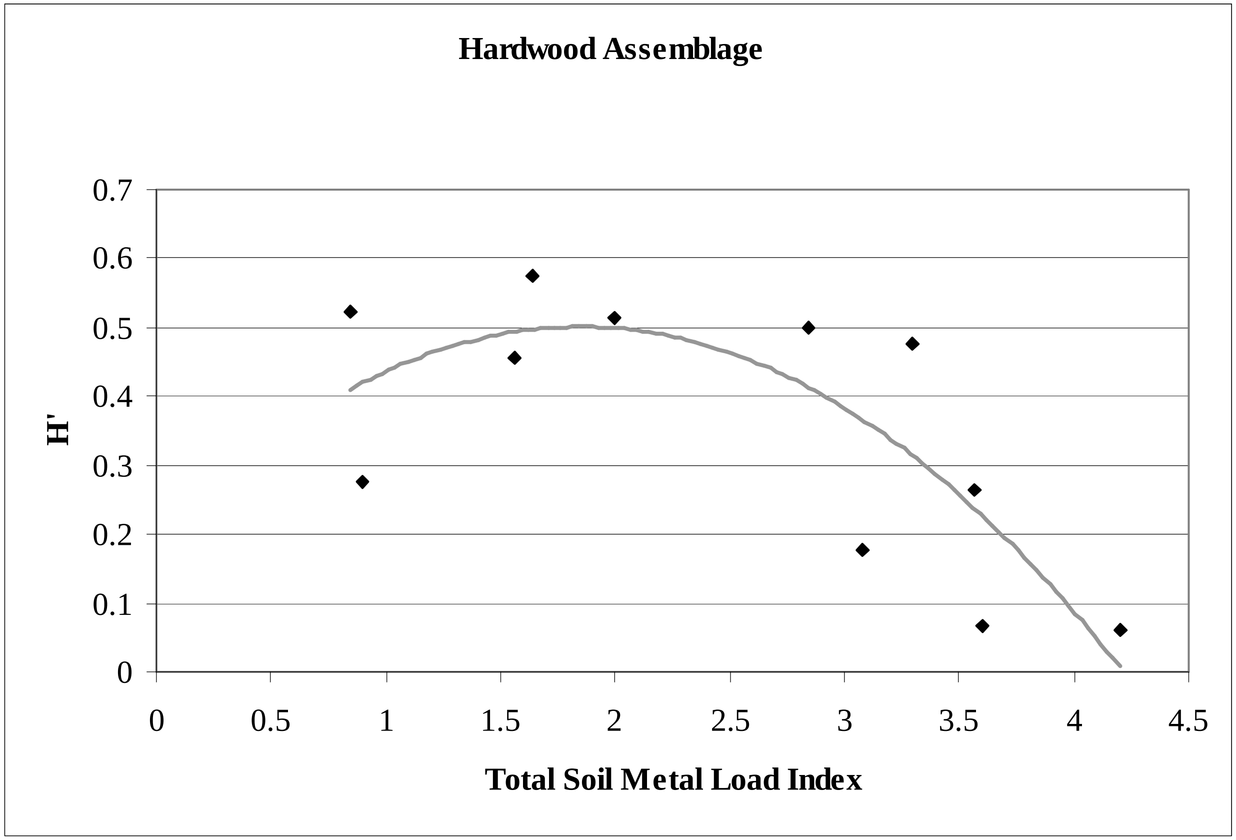


Figure 8: The relationship between species diversity (Shannon Index H') and TML. The polynomial regression is significant (r^2=.62, p=.02), indicating a threshold response.

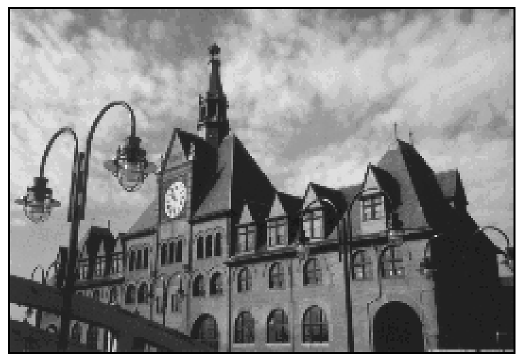


Appendix I

The Future of Liberty State Park

Summary of the Interdisciplinary Planning Committee

Background



Liberty State Park is an extraordinary and unique public resource. With the Manhattan skyline, the Statue of Liberty and Ellis Island as a spectacular backdrop, it is also one of New Jersey's most dramatic parks.

Central Railroad of New Jersey

The historic Central Railroad of New Jersey Terminal (CRRNJ), a grand setting for much of New Jersey's transportation history in the northeast, sits prominently at the north end of Liberty State Park.

A two-mile promenade, Liberty Walk links the picnic area, Interpretive Center and the CRRNJ Terminal while presenting visitors with a sweeping view of the Hudson

River. Liberty Science Center, a popular attraction for students and families, is located in the park's western section.

Liberty State Park was once an urban industrial area. As a result of its historical land use, the Division of Parks and Forestry has spent the past 25 years since its acquisition planning and building a park infrastructure that is suitable for public recreation. With more than four million visitors annually, the park's development continues to be an extraordinary success.

As part of the Division's Waterfront Improvement Plan for Liberty State Park, development of an 88-acre Green Park was completed in 1999. Bordered on the west by Freedom Way and on the east by Liberty Walk, the Green Park is comprised of magnificent crescent lawns, trails and landscaping improvements, including newly planted trees, shrubs and wildflower meadows. Approximately 4 miles of paved walkway have been added, as well as seven plaza areas located along Liberty Walk, which provides views of Ellis Island and the Statue of Liberty.

The Green Park is just one example of the development potential of Liberty State Park as a premiere recreational site and urban green space. The next stage of the park development will focus on providing public access to the 251-acre interior section and enhancing the park's historic resources.

In addition, Liberty Science Center is an innovative learning resource for lifelong exploration of nature, humanity and technology, supporting the growth of our diverse region and promoting informed stewardship of the world.

Activities in the park include boating, canoeing, picnicking, fishing, hiking, biking, and numerous special events. The park features the historic Central Railroad of New Jersey Terminal building, an Interpretive Center, Liberty Landing Marina, the Liberty Science Center and ferry service to the Statue of Liberty and Ellis Island. Currently 4.3 million annual visitors are accommodated within the 598 upland acres, 523 tidal acres, 25 structures and 5.3 miles of roads. The true tourism and economic impact potential, both locally and regionally, at Liberty State Park will only be realized upon the complete restoration of the CRRNJ Terminal and the expansion of Liberty Science Center. The Division, through its partnership with the Save Ellis Island! Foundation and the National Park Service will also play a critical role in the continued restoration of Ellis Island and in providing increased access to this landmark. Improved interpretive facilities, including exhibits and programming, will help attract both a regional and national audience, presenting a broader picture of the area's pivotal role in America's immigrant and industrial history.

As the National Monuments reveals the story of the peopling of the United State of America and Liberty Science looks into the possibilities of the future it is mission of Liberty State Park is to provide the public with access to the harbor’s resources, a sense of its history and the charge of responsibility for its continued improvement.

These various interests must produce cooperative efforts that enrich the lives of the people in the surrounding community and the experience of all that visit Liberty State Park.

Critical Future Issues - The Division of Parks and Forestry looks forward to working with the many partners of the park to accomplish the following tasks:

1. Providing for a fully accessible park that will accommodate 6+ million visitors annually via mass transit, vehicular traffic and pedestrian walkways.
2. Day use improvements to accommodate anticipated open space noncommercial recreational use.
3. Completion of interior portion of park to enhance wetlands provide for interpretive space and public access.
4. Remediation of the remainder of soil contaminated areas.
5. Restoration of CRRNJ Terminal to accommodate public use and special events, including the completion of train shed areas to provide program, exhibit, special event and interpretive uses.
6. Redevelopment of southern portion of park to complete Liberty Walk.
7. Restoration of Caven Point Pier and the other piers, located along the southern boundary of the park.

Capital Improvement Project Needs – In order to accomplish the task outlined above the following capital improvement projects will have to be undertaken. The cost for these projects is speculative at this time; however, preliminary estimates call for between $68 and $150 million:

1. Terminal Interior, 1st and 2nd Floor
2. Train Sheds Stabilization
3. Administration Building
4. Liberty Walk Access Improvements
5. Site work/Remediation
6. Day Use Improvements
7. Southern Waterfront Improvements
8. Interior Park Plan Implementation
9. Southern Embankment Improvements
10. Southern Jetty Redevelopment
11. Access Improvements
12. Interpretive Center Expansion
13. Train Shed Redemptive Use
14. Restoration of Caven Point Pier

The Surrounding Community

Jersey City, the state's oldest metropolitan area, is enjoying a renaissance of construction as we enter the new millennium. A beacon to more than 10 million immigrants who landed on nearby Ellis Island in the great migrations of the 1800s and early 1900s, Jersey City is still the golden door to opportunity on the west bank of the Hudson River. It is becoming Wall Street west and Silicon Valley east, as scores of Wall Street firms move their facilities from Manhattan to New Jersey and as the new communications firms and internet providers flock to take advantage of the city's fiber optic cable network.

The land adjoining Liberty State Park has been an integral part of this revitalization. Along the parks southern boarder Port Liberte', a 1290 unit and golf course, now houses a permanent residential population and is entering into a new phase of expanded development. A proposed sports complex along the parks western boundary proposes will bring activity to the area of the park which currently houses the Liberty Science Center. The northern boundary, along the Morris Canal has experienced increase residential development and will be further impacted by the expansion of the Jersey City Medical Center. Finally, the National Park Service and Save Ellis Island! Inc., a non-profit organization, have entered into a partnership to restore the remaining 29 buildings on the island. The agreement recognizes the Save Ellis Island! as the primary fund raising entity for the estimated $300 million project.

Recently the completion of the Light Rail Park and Ride, and the addition of ferry service to the Liberty Landing Marina are beginning to address the problems of access to the park. Connecting to mass transit and the ability to enhance pedestrian flow into and around the park are critical to the future success of Liberty State Park.

Basic Charge to the Interdisciplinary Planning Committee

In the center of the park there remains approximately 251acres, the former railroad yard, which is undeveloped. Much of the area has been re-colonized by various plant communities. These communities represent unique associations of both endemic and non-native species that can be considered the by-product of the cultural events that have taken place during the past several centuries. A broad-based, goal-driven approach is being used to develop the General Management Plan (GMP) for the site. The planning process stresses the fundamental relationship between resource significance and visitor experience. The planning process encourages feedback from the local community. Most importantly, the planning process results in documentation of planning efforts that build a consensus among participants, assure logic and consistency in the proposals, and provide a valid rationale in decision-making. The members of the interdisciplinary planning committee, who represent various public and private interest groups, have agreed to participate in the development of the GMP.

Premise I: Inherent Ecological Value

Various plant communities have re-colonized much of the site. Like the surrounding community of people these assemblages are diverse and have origins throughout the world. This diversity is further enhanced by the rapid rate of natural succession (change inherent within any ecosystem). Hence, there is ecological and aesthetic value in some of the existing natural association.

Premise II: Soils Condition

The soils of the area consist of fill brought in by the railroad companies between 1860 and 1919 to stabilize the surface. Much of it is non-consolidated material resulting from construction projects in Manhattan, or refuse from throughout New York City and the surrounding area. It is classified as historic fill and has some limitations. Allowing public access via the creation of a trail system will have to creatively combine soils mitigation boardwalk construction, plantings and some fencing to ensure the safety of pedestrians through the site.

Agreement I, Planning Objectives:

1. Provide public access for interpretive programs allowing visitors to touch the natural world.

2. Maintain as much of the site as possible, especially wetlands and special plant communities, under a conservation mandate while providing public access.

3. The landscape of the interior should reflect the history of the park as well as the connection to the harbor/estuary. The history of the area now known as Liberty State Provide public access to the perimeter of the site for multiple uses.

4. Improve topography, enhance wetlands and provide open water, and enhance aesthetic values and sight lines where possible. In those areas that are to be disturbed, new elevations will be established that enhance the existing wetlands, possibly creating open water habitat and taking advantage of the spectacular views of the harbor and New York City skylines.

The planning effort will be conscious of other neighboring redevelopment efforts.

<u>Agreement III, Proposed Protection Strategies</u>

Key to accomplishing the committee's stated objectives will be the integrated use of the protection of critical areas; the conservation of woodland and field areas; and the restoration of wetland habitats and provisions for visitor services according to the following:

1. The existing wetland areas, which are protected under the Freshwater Wetland Act, will be enhanced where possible.

2. There is a unique plant community atypical for this area that has been identified as the moss mat community. Due to its unique association of species, which is characteristic of communities at northern latitudes, this critical area will also be protected.

3. Most of the area will be maintained under a conservation mandate, which allows for the management of invasive species and enhancement with species that would increase biologic diversity.

4. Saltwater wetland restoration may also be possible.

<u>Agreement IV, Methodologies</u>

Preservation- The preservation of the wet-thicket and sedge ponds, which currently occupy approximately 3 acres, is critical. A buffer of 100 feet must also be included, and the area could be expanded to approximately 10 acres. In addition, the half-acre moss mat community is found in association with 2 acres of "common reed", a half-acre of sumac and 1.3 acres of sedge ponds, which act as a buffer. This complex should remain intact and allow the "moss mat community" to expand.

Conservation - The pioneer forest comprises a significant section of the interior. Management strategies in these areas will consist of removing invasive species and a limited amount of wildlife enhancement plantings. In addition, those field areas existing between the wooded areas should be allowed to succeed, creating a more contiguous forest. Field areas include herbs and grasses that inhabit the dry, gravel soils of the old railroad beds. Many grass fields are gradually giving way to early succession woody species. While this transition will be encouraged in those areas between the forested stands, thereby creating a more contiguous forest, some grasses will be maintained. The primary management practice will be to mow the area on an annual/biannual basis or more frequently depending upon the desired use.

Restoration - Grasses and species of trees typical of early succession currently dominate the dredge spoil site. It may be possible within these areas to create a cross section of the vegetative communities, which existed prior to the development of the area. The creation of an area representative of the transition from salt marsh to upland forest would provide for an exceptional study of habitat restoration in the urban environment while enriching the visitor experience.

The "common reed" (Phragmites australlis) dominates the marsh meadow. The "Common Reed" grows as a nearly impenetrable; dense stands are 10-12 feet in height. While some of these stands will be involved in the restoration efforts, which may include the reintroduction of freshwater habitat into the park, several acres should be left for its inherent wildlife values.

The restoration of both freshwater and marine wetlands could re-introduce aquatic habitats to the center of the park and create a network of interconnected wetland/waterways.

Interpretive Enhancements - Interpretive trails will be developed in areas already disturbed by former roads to minimize disturbance. However, several connecting trails will have to be developed. These should follow the vegetative communities to allow for viewing wildlife while minimizing disturbance.

Interpretive and Recreational Enhancements - In order to provide open space non-commercial recreational enhancements, connection between existing facilities and access for interpretation, several trails and perimeter green spaces will have to be established. While the perimeter must serve as a buffer to the more ecologically sensitive areas, its width can vary greatly to create interesting areas capable of supporting trails, which explore interpretive themes, picnic areas or other forms of passive recreation. The amount of area dedicated to the various types of activities would be determined during the future design phase but should loosely follow the wood lot edge.

The 18-acre soil stockpile area, recently disturbed by the storage of soil used for landscaping purposes throughout the park, will provide for a range of non-commercial passive recreational activities. Its location in the extreme southwest corner of the site, next to the industrial complex and across from the proposed sports complex may lend itself to such use and is critical as a buffer to the more ecologically sensitive areas within the site.

By using fill to increase the elevation of certain areas within the perimeter visual and noise barriers can be created. They can be used to obscure the view of the industrial area or decrease the noise from traffic along Phillips Dr. They can also be used to create interesting lines of site between the Science Center and the Interpretive Center or to direct storm water into the wetland areas.

Agreement V, Feasibility

The conceptual plan presented above has been developed with the intention of providing the best land use given exiting conditions. At this point engineering studies that focus on mitigation of historic fill, hydrology for freshwater wetlands and reintroduction of salt marsh habitat must be undertaken. It has been determined that the area is of such significance to the eventual success of other park amenities, and also to the quality of life for surrounding residents, that a professional international competition for the actual design should be conducted. Such competitions tend to increase the visibility of the project and attract more creative designers, giving the project the attention it deserves.

Conclusion

Liberty State Park, the cornerstone of the Gold Coast, is already a successful rehabilitation story. However, with the completion of the interior section the park has the potential to be an international showcase for the restoration of a former urban brown field. The tremendous interest in the remaining undeveloped section of the park is symbolic of a broader struggle that often occurs within New Jersey, and throughout much of the nation, to balance the protection of natural resources with the need for continued economic development and recreational opportunities. This General Management Plan strives to complete the park in a manor that honors its history while at the same time provides for the residents of the surrounding community, state, national and international visitors. The plan must be consistent with the Division's stewardship principle that "activities must be within the physical and biological capabilities of the natural/historic resource."

Appendix II

Conceptual development Plan for the Interior of Liberty State Park

Appendix III

Site Description

The 251-acre project site lies within LSP, in Jersey City, on the west bank of Upper New York Bay. The land, which currently comprises the park was originally part of an intertidal mud flat and salt marsh that was filled for use as a railroad yard. The fill materials consist primarily of debris from construction projects and refuse from New York City, which were deposited to stabilize the surface between 1860 and 1919. Between 1864 and 1967 the Central Railroad of New Jersey (CRRNJ) used the site as a rail yard for both freight and passenger service. Industrial activities at the site resulted in localized hydrocarbon spills and pesticides, iron tailings and coal ash. There is no known hydrocarbon free product within the study site and the areas of high pesticide contamination have been identified and will be mitigated separately. Due to the use of the site for coal transport and storage, higher than normal mercury levels are unevenly distributed throughout. In 1967 CRRNJ discontinued operations at the site, and over the next few years the land was abandoned until it was acquired by the New Jersey Division of Parks and Forestry (NJDPF).

The first ecological descriptions of the site were contracted soon after acquisition. The Texas Instrument (TI) Study conducted in 1976 provided several sets of data in order to characterize the vegetative, upland fauna and littoral benthos communities (Texas Instruments 1976).

The TI study characterizes the vegetation of the site as that of very early succession dominated by herbaceous species, many of which were exotic. At one sampling location the plant heights were described as “uniformly low and there was no dense, rank vegetative growth” (Texas Instrument 1976). Of the species identified Artemisia vulgaris, had the greatest importance value as it occurred at every sampling site. Phragmites austalis had the second greatest importance value and occurred in dense stands. Ambrosa artemisiifolia, and Panicum sp. were also frequently encountered.

Woody vegetation was classified as sparse with large trees entirely absent. A few small stands of Populus deltoids saplings existed at one of the sampling sites. Shrubs were dominated by Rhus copallinum and Spirea latifolia.

During the summers of 1995 and 1996 David McFarlane conducted a survey of the plants and animals of the site. The survey resulted in the first map which identified assemblage boundaries. The map was justified using aerial photographs of the site from the same time period. The assemblage structure was characterized by McFarlane as follows “The interior 225 acres of Liberty State Park are a strange and interesting urban wilderness. Succession, a natural process which changes the composition of biological communities in a geographic area over time, is occurring in this undeveloped site in many different locations and forms. The complex history of the site has created a number of different soil types and an interesting microtopography, which favors the adaptations of some organisms over others. Some portions of the site closely approximate patterns of succession observed in similar ecosystems within the region, while others have been colonized by rare and unusual species, some of which are unknown in other parts of the state. The water table is relatively high in many locations, although it is locally scarce at the surface of the soil where gravel and sand prevail. The composition of the surface soils on the site has strongly influenced the various rates of succession taking place and in many ways serve as templates for communal development." (McFarlane 1996)

It is clear from McFarlane's survey that the assemblage structure was dominated by early colonial species. One grouping identified by McFarlane but absent in subsequent reports were the pioneer communities, areas where the gravel and mineral ash soils were clearly visible and the sparse vegetation was dominated by lichens or xerophytic grasses. Today the remnants of these communities exist under a sparse canopy of trees, usually Betula populifolia or as part of a path that has been subject to continued disturbance.

During this survey, areas classified as marsh-meadow, comprising 225 (later determined to be 251) acres, were dominated by Phragmites australis. Such assemblages existed in the depressions created by old railroad beds or in more recently disturbed areas. In much of its distribution Phragmites australis dominated assemblages forming homogenous dense stands. However McFarlne also recognized mixed or fragmented stands of Phragmites australis where some herbaceous and woody species were established. Interestingly the USACE study of 2003 identified approximately 19 acres of wetlands dominated by Phragmites australis. However, this survey also considers those wetlands located in the Dredge Spoils area, which were not part of McFarlanes survey. Removing these wetlands from the total yield approximately 10 acres of Phragmites australis dominated wetlands a significant decrease from 1996 to 2003.

Woodland thickets dominated by Betula populifolia, Populus deltoidies, P. grandidentata Myrica, pensylvanica, Rubus laciniata, accounted for approximately 51 acres. These assemblages appear to be expanding as the subsequent survey identified at least 57 acres of successional northern hardwood forest. While the difference is not large and some error is expected as the surveying techniques were different, an overlay of the GIS maps and arial photographs indicate a gradual expansion of successional northern hardwood. In addition this habitat type was completely absent from the TI survey.

The USACE, NRI characterized the site as “covered by vegetation communities that reflect decades of human alteration. The substrate is composed of fill material that was deposited on the site during the 19th and 20th centuries, when the site functioned as a rail yard. When the rail yard was closed in 1967, no site restoration was performed. Through natural succession and in response to an assortment of differing physical and chemical conditions, a variety of habitats have developed over a period of 36 years. Past construction has created compacted depressions with poor drainage, allowing for the formation of small isolated wetlands. Variability in substrate materials has created patchy distribution of soils and cinders, which has controlled the colonization of the substrate by plants. Vegetation grows sparsely in areas with cinder substrate, while trees and shrubs grow in dense stands over soil substrates. As a result, vegetation communities have developed on the fill material at different rates due to the varying soil and hydrologic conditions, and the timing of fill placement. This created a mosaic of community types within the LSP Restoration site. The community types present range from bare soil or lichens on coal ash to pioneer tree communities growing on soil deposits. Thus, the site displays variable stages of successional colonization, and provides habitat for a variety of terrestrial animal species."

The USACE 2003 survey identified 12 assemblages within the site consisting of eight terrestrial and four wetland types. Successional Northern Hardwood consisted of Populus tremuloidies, Populus deltoids, Betula populifolia, Rhus copallina and Spiraea tomentosa. Successional Shrubland consist of Rubus laciniata, Rhua glabra, Myrica pensylvanica, Solidago canadensis, Phragmites australis and Polygonum cuspidatum. Successional Old Field, Calamagrostis epigeios, Verbascum thapsus, Centauria maculosa, Linaria vulgaris, Solidago canadensis, Rhus typhina, Populus tremuloidies *and* Populus deltoids. Maritime Shrubland consist of Rhus copallina, Rhus glabra, Rhus typhina, Solidago canadensis, Phragmites australis, Artemisia vulgaris, Betula populifolia *and* Populus tremuloidies. Maritime Grassland consist of Spartina patens, Linaria vulgaris, Phragmites australis, Lythrum salicaria, Baccharis halimifolia, Iva frutescens, Rhus copallina, Rhus typhina, Populus tremuloidies *and* Populus deltoidies. Common Reed/Mugwort consist of Artemisia vulgaris, Phragmites australis, Apocyanum cannibinum, Verbascum thapsus, Lythrum salicaria, Rhus copallinaand Populus tremuloidies. Mowed Lawn of Poa annua. Unpaved/Paved Road consist of Artemisia vulgaris, Melilotus alba, Verbascum thapsus, Daucus carota *and* Centauria maculosa.

Floodplain Forest Wetland consists of Betula populifolia, Populus deltoids and Onoclea sensibilis. Shrub Swamp Wetlands consist of Rubus laciniata, Rhus copallina, Rhus typhina, Phragmites australis, Lythrum salicaria, Onoclea sensibilis and Populus tremuloidies. Shallow Emergent Marshs consist of Lythrum salicaria, Phragmites australis, Scirpus cyperinus, Spiraea tomentosa and Betula populifolia. Common Reed-dominated Wetland consists of Phragmites australis, Lythrum salicaria and Onoclea sensibilis.

The USACE survey summarized the forested and wetland assemblages as follows: “forest communities will be dominated by native poplars and birches, with further growth of slow-growing maple and oak saplings under the tree canopy. Forest boundaries will continue to expand slowly into areas that are now covered by shrublands or grasslands. Maritime and successional shrubland communities are likely to expand over the successional old fields. Maritime shrubs are located throughout the old field communities, and the grasses acreage is likely to decrease as the shrubs become dominant. Existing native maritime grassland communities are located adjacent to monocultures of the invasive species Phragmites australis and Artemisia vulgaris. The invasive species will likely encroach upon the grasslands and eventually outcompete them for resources, and the maritime grasslands will likely be rare or non-existent within the LSP Restoration site in as little as ten years.

When assessing the current conditions and projecting the likely future for freshwater wetlands within the site the USACE concluded with the following: “Wetlands within the

LSP Restoration site may increase in area due to the accumulation of organic matter, but will likely decrease in value over the next ten years without the proposed restoration project. Common reed and/or purple loosestrife are common in most of the freshwater wetlands. It is likely that, with the exception of the floodplain forested wetland, existing wetlands will develop into monocultures of these invasive species." Their data, however, show an increase in Phragmites australis occurring in the more recently disturbed dredge spoil containment area. Those areas which have not been disturbed since the rail road ceased operation showed an overall decrease in areas dominated by Phargmites australis.

A comparison of the vegetative species encountered during each study is included below. While species richness has obviously increased over the years, direct correlations between the studies are difficult as the methodologies and classification systems used differed. However, the following broad observation can be made from the past surveys:

1. Overall plant species richness, measured simply as the number of species encountered, has increased by at least 71% over the past thirty years, despite the development loss of approximately 50% of the site.

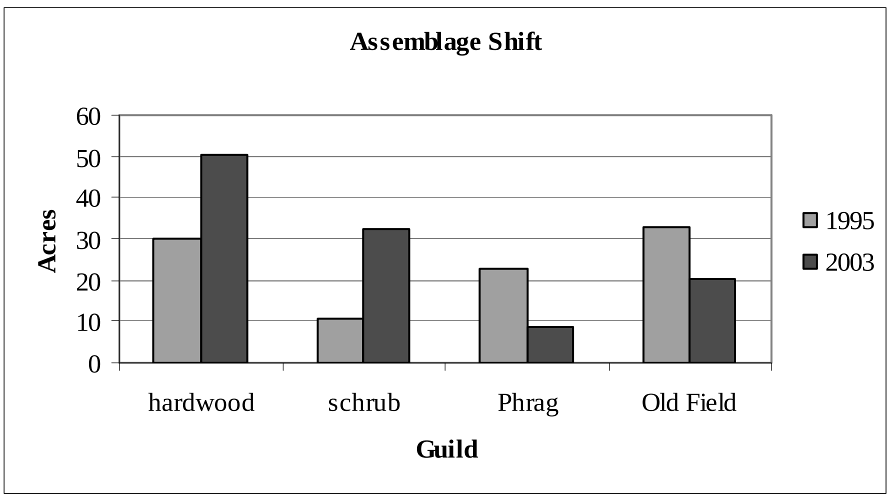
2. Forested areas have increased dramatically in size and density over the past thirty years and know occupy approximately at least 56 acres, of the project site.

3. Shrub assemblages appear also to have increased in both size and diversity and currently occupy at least 32 acres. The difference between the McFarlane and USACE surveys is difficult to estimate as McFarlane did not differentiate between shrub and woodland, due to the very early stage of the woodland assemblages.

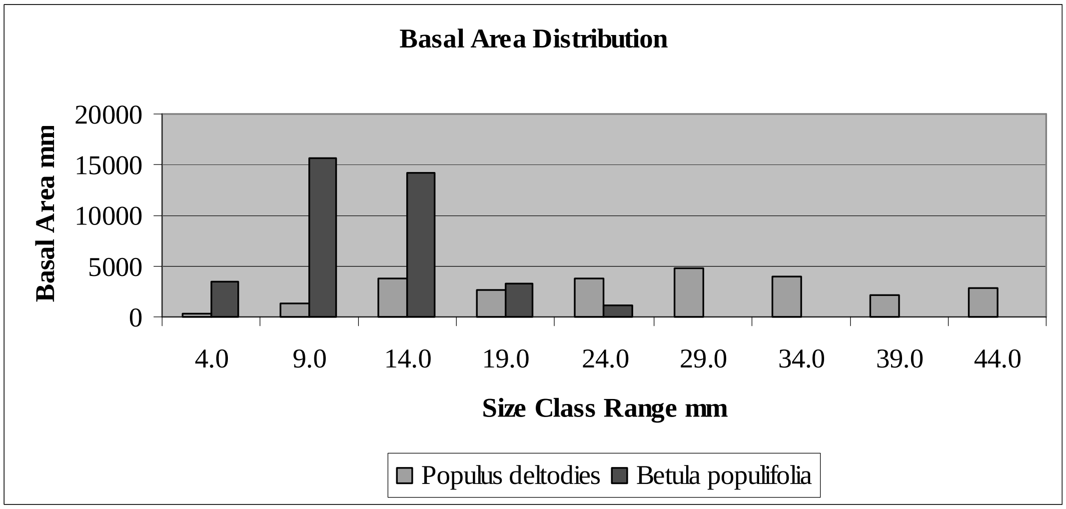
4. Invasive exotic species have been prevalent throughout the sites history. However, the structure of these assemblages has changed. Assemblages dominated by Phragmites australis apparently decreased in size, within the study area between 1996 and 2003, perhaps by as much as 19%. Conversely in the dredge spoil area that was graded in 1993 and planted with rye grass, Phragmites australis has invaded.

The following table demonstrates the overall change in assemblage structure and is keyed to the above maps.

Change in Area Covered by Guild



Total basal area encountered in the sample sites for the two dominate tree species of the study area:



Appendix IV

Soil Results and Comparisons 1995 vs. 2005

1995 Soil Metal Concentrations (µg/g)

	TP--1	TP--2	TP--3	TP--5	TP--7	TP--8	TP--10	TP--14	TP--16
Al	7820	1960	7570	13900	7800	4550	3490	2770	2940
Sb	6.5	6.4	7.1	6.4	53.3	25.9	17.8	6.5	6.7
As	4	120	14.7	5.8	29.8	24.8	72.7	9.1	15.6
Ba	394	110	167	314	492	117	200	93.3	134
Be	0.48	0.18	0.57	0.64	0.38	0.42	0.61	0.2	0.28
Cd	1.3	2.6	1.1	1.3	0.38	0.82	0.63	0.42	0.43
Ca	2960	3460	2940	3340	7170	3620	2320	2620	631
Cr	43.1	44.1	34.9	47.1	157	48.4	56	11.1	30.3
Co	10.9	8.4	8.2	10.1	154	8.3	8.4	2.8	5.4
Cu	117	900	99.9	138	778	238	190	30.1	78.8
Fe	19400	37700	19700	21900	87300	43200	37500	18200	15000
Pb	493	421	753	325	789	436	512	37	202
Mn	2180	731	2530	3020	3150	1460	481	855	568
Mg	244	802	288	270	231	277	209	68.2	63.7
Hg	0.53	0.37	0.92	0.44	0.75	0.61	0.74	0.97	0.34
Ni	17.3	46.7	29.6	18.8	49.5	28.4	31.4	9.3	25.1
K	732	215	990	737	1200	545	464	772	673
Se	0.38	0.97	1.2	0.4	3.5	2.1	3.1	0.6	1.1
Ag	1.1	1.1	1.2	1.1	0.97	1.1	1.1	1.1	1.1
Na	103	110	119	154	286	205	202	276	70.9
Ti	0.24	0.36	0.77	0.26	0.37	0.54	0.56	0.41	0.42
V	21.2	49.9	59.9	25	31.1	30.3	54.2	19.8	36.2
Zn	322	731	523	299	1060	520	601	27.4	59.4
Ph	6.6	7.3	6.6	6.9			5.4		

Soil Results and Comparisons 1995 vs. 2005

1995 Soil Metal Concentrations (µg/g) (cont.)

	TP--17	TP--18	TP--21	TP--24	TP--25	TP--26	TP--27	TP--28	TP--31
Al	12100	1970	3470	2900	10000	3540	4110	2790	1420
Sb	30.2	9.1	7.5	7.5	357	6.5	14.8	11.8	6.6
As	29.8	27	9.6	31.2	545	26.1	30.7	16	15.1
Ba	690	93.5	103	112	648	118	136	167	111
Be	0.58	0.34	0.35	0.32	3.1	0.41	0.55	0.23	0.39
Cd	17.5	0.41	2.8	4.3	2.3	0.42	1.2	0.95	0.42
Ca	6890	574	5560	5300	5080	1520	2650	5740	308
Cr	122	57.1	47.9	78	55.1	28.2	48.5	13.8	29.1
Co	28.5	6.6	11.1	17.7	337	5.4	8.3	3.7	5.7
Cu	2900	171	415	778	3790	65.6	206	53.2	117
Fe	144000	32500	40400	82000	99500	23000	29500	13300	64500
Pb	2440	338	724	511	11800	195	474	3240	163
Mn	3020	575	2780	1900	2730	356	1190	845	164
Mg	1120	231	455	796	475	60.8	485	70.2	208
Hg	0.65	0.72	0.62	0.5	1	0.43	0.5	0.33	0.33
Ni	163	25.5	44.1	86.3	112	12.7	30.1	20.1	19.1
K	1110	277	416	401	1460	555	602	316	399
Se	0.88	1.7	1.2	0.77	5.2	2.6	2.4	2.2	1.2
Ag	3.1	1.1	1.1	1.2	1.8	1.1	1.2	1	1.1
Na	301	75.2	162	142	1780	68.5	282	109	20.8
Ti	0.5	0.4	0.43	0.47	0.59	0.6	0.44	0.42	0.41
V	101	68.3	48	48.1	60.5	52.8	78.5	40.2	27.9
Zn	7660	82	786	1180	16000	34.4	564	622	24
Ph	6.9	4.5	7.4	6.9	5.2	4.9	5.7	7.8	

Soil Results and Comparisons 1995 vs. 2005

1995 Soil Metal Concentrations (μg/g) (cont.)

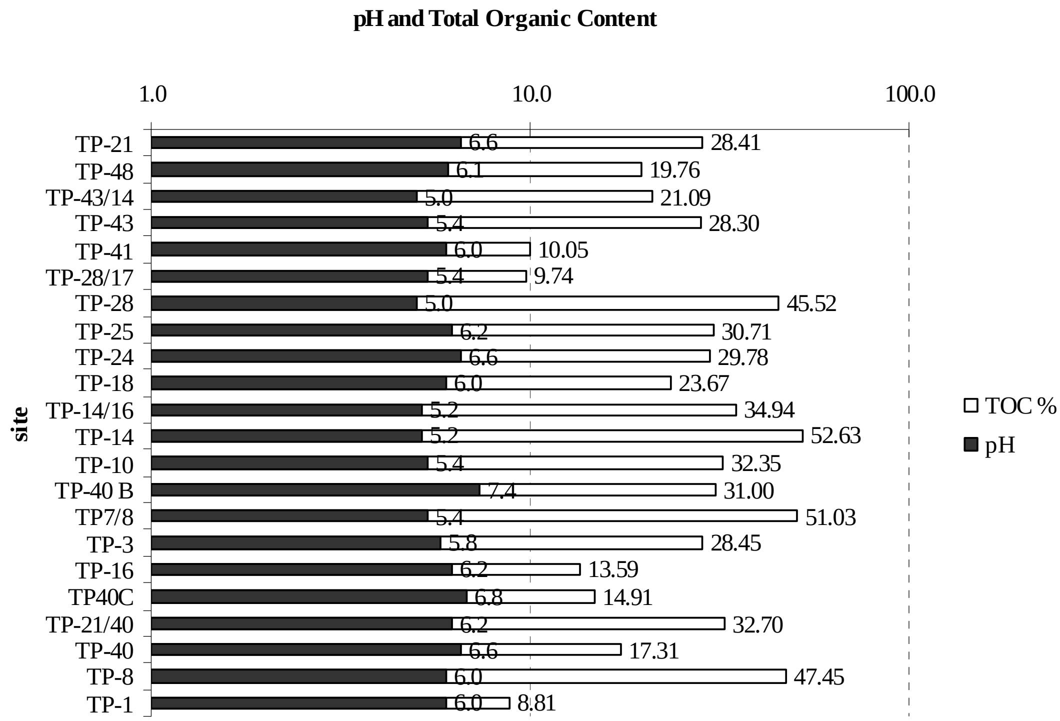
	TP--38	TP--40	TP--41	TP--42	TP--43	TP--44	TP--45	TP--47	TP--48
Al	4790	3360	2470	2230	7760	7630	6360	6730	3660
Sb	6.7	25.6	7.3	6.6	19.8	5.9	15	10.9	6.5
As	309	21.6	27.7	14.5	5.8	9.8	9.8	4.3	8.6
Ba	149	122	4130	56.6	158	106	119	41.4	145
Be	0.55	0.31	0.23	0.33	0.51	0.42	0.41	0.26	0.39
Cd	0.63	0.37	0.47	0.42	0.39	0.6	1.5	0.66	0.69
Ca	5910	909	4300	4200	2640	8190	4650	1420	707
Cr	32.6	60.7	20.1	11.7	34.8	30.3	25.6	15.9	26.4
Co	6.7	7	3.3	6	16	11	7.6	6.4	3.9
Cu	153	239	57	80.4	145	185	64.2	31.3	72.9
Fe	25000	51700	16900	18900	25400	30900	25900	14800	20300
Pb	172	687	556	80	106	564	143	46.5	76.7
Mn	1630	449	1130	324	3280	6170	1490	2500	610
Mg	388	227	105	219	312	489	285	120	53.3
Hg	0.35	0.73	0.47	0.06	0.26	1.2	0.19	0.09	0.17
Ni	18.4	34.6	13.4	14.5	30.8	30.3	25.2	10.2	11.5
K	506	545	245	232	1340	3140	758	876	1270
Se	2	2.1	0.86	0.78	0.36	0.4	0.38	0.35	1.4
Ag	1.1	0.95	1.2	1.1	1	0.98	1.1	0.98	1.1
Na	268	84.5	56	113	90.6	648	99.7	125	97.2
Ti	0.42	0.36	0.46	0.41	0.39	0.37	0.21	0.18	0.2
V	27	87.3	25.2	22.8	22	22.1	39	29.5	26
Zn	362	119	2180	60.9	345	441	125	41.8	19.9
Ph	7.7			8		7.8			

2005 Soil Metal Concentrations (µg/g)

Site	As	As stdev	Cr	Cr stdev	Cu	Cu stdev	Hg	Hg stdev
TP-1	13.70	7.91	45.01	20.33	135.24	54.87	0.06	0.08
TP-10	282.57	155.46	91.58	59.50	379.18	32.36	1.42	2.18
TP-14	181.18	57.02	84.98	73.89	223.82	64.33	0.29	0.13
TP-14/16	68.13	24.15	334.56	141.83	202.58	46.42	0.34	0.09
TP-16	41.00	18.84	62.01	10.61	78.28	34.72	0.21	0.21
TP-18	44.40	9.56	34.90	17.23	314.70	33.01	4.23	6.87
TP-21	51.92	7.84	53.48	3.79	482.37	56.49	0.51	0.29
TP-21/40	23.31	2.08	47.73	4.64	331.91	59.83	0.43	0.08
TP-24	33.63	2.52	62.69	3.66	354.66	45.89	0.56	0.54
TP-25	384.41	253.15	50.54	39.71	2200.38	265.02	0.26	0.29
TP-28	42.52	9.32	128.53	22.14	81.11	2.75	0.00	0.00
TP-28/17	13.83	5.24	48.20	13.47	53.65	14.05	0.66	0.34
TP-3	55.17	20.02	98.85	22.35	524.82	486.07	0.04	0.03
TP-40	16.08	3.14	24.34	8.48	185.04	52.78	0.61	0.43
TP-40 B	36.28	9.09	67.76	8.31	309.00	59.98	0.31	0.06
TP40C	16.75	7.84	24.15	16.06	270.75	106.47	0.08	0.08
TP-41	14.88	4.79	10.90	5.43	76.86	29.64	0.00	0.00
TP-43	29.29	2.26	14.08	3.55	229.79	94.71	0.08	0.07
TP-43/14	22.30	5.16	14.50	3.29	88.64	15.92	0.00	0.00
TP-48	13.33	7.44	20.87	24.95	95.15	22.07	0.21	0.22
TP7/8	116.81	36.05	103.38	8.56	406.44	50.45	3.22	1.58
TP-8	22.90	2.88	73.88	12.66	320.53	30.87	0.29	0.19
Min	13.33	2.08	10.90	3.29	53.65	2.75	0.00	0.00
5%	13.71	2.27	14.11	3.55	76.93	14.14	0.00	0.00
25%	18.14	4.88	26.98	6.15	105.18	31.24	0.08	0.07
Median	34.95	7.88	52.01	13.07	250.27	48.43	0.29	0.16
75%	54.36	19.72	82.20	22.30	348.98	59.94	0.55	0.33
95%	277.50	150.54	127.27	73.17	522.70	257.09	3.13	2.15
Max	384.41	253.15	334.56	141.83	2200.38	486.07	4.23	6.87

2005 Soil Metal Concentrations (μg/g)

Site	Na	Na stdev	Pb	Pb stdev	Zn	Zn stdev	Va	Va stdev
TP-1	0.88	0.82	492.12	278.73	338.12	134.26	21.26	18.42
TP-10	1.73	0.77	737.95	166.50	192.65	130.53	80.79	50.73
TP-14	4.97	1.56	926.17	583.50	36.71	19.46	72.59	74.18
TP-14/16	2.44	0.48	857.53	143.85	237.97	42.95	316.80	207.33
TP-16	1.73	0.09	196.32	77.39	45.30	25.23	84.18	27.28
TP-18	1.89	1.23	517.65	138.88	603.87	517.58	88.29	28.69
TP-21	3.48	0.30	786.17	8.08	1462.35	362.66	12.37	21.42
TP-21/40	5.70	0.87	626.76	80.53	896.73	32.82	40.05	29.98
TP-24	1.77	0.09	693.40	68.63	1019.71	290.99	55.30	42.91
TP-25	9.68	3.77	6673.22	1603.66	2326.77	2046.85	57.51	38.93
TP-28	9.78	2.97	320.82	25.65	62.83	10.81	226.19	71.86
TP-28/17	3.32	1.51	173.43	51.02	106.77	15.96	88.52	17.14
TP-3	1.47	0.60	476.53	224.86	233.80	228.08	164.41	77.10
TP-40	2.31	0.07	368.05	92.12	192.15	53.94	34.66	16.45
TP-40 B	3.40	0.41	615.94	149.80	337.94	72.61	35.79	30.43
TP40C	0.94	0.36	524.08	262.01	350.80	353.02	24.82	18.32
TP-41	0.75	0.41	96.58	34.02	156.75	116.81	10.79	18.70
TP-43	2.50	0.32	459.62	164.43	88.60	20.32	9.84	17.05
TP-43/14	2.62	0.62	148.45	51.34	10.88	18.84	0.00	0.00
TP-48	1.77	0.61	244.57	103.24	22.08	15.36	22.54	19.71
TP7/8	4.74	0.43	834.97	46.92	444.05	145.24	118.62	15.83
TP-8	5.86	1.45	669.33	121.81	389.58	77.34	85.29	18.82
Min	0.75	0.07	96.58	8.08	10.88	10.81	0.00	0.00
5%	0.88	0.09	149.70	26.07	22.81	15.39	9.89	15.86
25%	1.74	0.37	332.63	55.66	93.14	21.54	23.11	18.34
Median	2.47	0.60	520.87	112.52	235.88	74.98	56.40	24.35
75%	4.43	1.14	726.81	165.99	430.43	207.37	87.54	41.92
95%	9.49	2.90	922.74	568.26	1440.21	509.83	223.10	76.95
Max	9.78	3.77	6673.22	1603.66	2326.77	2046.85	316.80	207.33



pH and Total Organic Content
1.0
10.0
100.0
site
TP-21
6.6
28.41
TP-48
6.1
19.76
TP-43/14
5.0
21.09
TP-43
5.4
28.30
TP-41
6.0
10.05
TP-28/17
5.4
9.74
TP-28
5.0
45.52
TP-25
6.2
30.71
TP-24
6.6
29.78
TP-18
6.0
23.67
TP-14/16
5.2
34.94
TP-14
5.2
52.63
TP-10
5.4
32.35
TP-40 B
7.4
31.00
TP7/8
5.4
51.03
TP-3
5.8
28.45
TP-16
6.2
13.59
TP40C
6.8
14.91
TP-21/40
6.2
32.70
TP-40
6.6
17.31
TP-8
6.0
47.45
TP-1
6.0
8.81
TOC %
pH

Correlations - metals 1995

		AS1995	CR1995	CU1995	PB1995	HG1995	VA1995	ZN1995
As1995	Correlation	1	.254	.978(**)	.960(**)	.462	.291	.988(**)
	Sig. (2-tailed)	.	.361	.000	.000	.083	.292	.000
	N	15	15	15	15	15	15	15
Cr1995	Correlation	.254	1	.355	.137	.247	.150	.204
	Sig. (2-tailed)	.361	.	.194	.627	.374	.593	.466
	N	15	15	15	15	15	15	15
CU1995	Correlation	**.978****	.355	1	**.943****	.434	.251	**.978****
	Sig. (2-tailed)	.000	.194	.	.000	.106	.367	.000
	N	15	15	15	15	15	15	15
PB1995	Correlation	**.960****	.137	**.943****	1	.388	.260	**.964****
	Sig. (2-tailed)	.000	.627	.000	.	.153	.349	.000
	N	15	15	15	15	15	15	15
HG1995	Correlation	.462	.247	.434	.388	1	.264	.422
	Sig. (2-tailed)	.083	.374	.106	.153	.	.341	.117
	N	15	15	15	15	15	15	15
VA1995	Correlation	.291	.150	.251	.260	.264	1	.331
	Sig. (2-tailed)	.292	.593	.367	.349	.341	.	.228
	N	15	15	15	15	15	15	15
ZN1995	Correlation	**.988****	.204	**.978****	**.964****	.422	.331	1
	Sig. (2-tailed)	.000	.466	.000	.000	.117	.228	.
	N	15	15	15	15	15	15	15

** Correlation is significant at the 0.01 level (2-tailed).
* Correlation is significant at the 0.05 level (2-tailed).

Appendix V

A Summary of the Soil Contamination at Liberty State Park (Bell 1995)

Due to the Parks history as an industrial rail yard an extensive investigation into soil contamination was conducted in 1995. The areas of the Park which were sampled include Middle Cove, the Freight Yard Area and the Waterfront Area. The Freight Yard Area has been divided into several sections: the Soil Staging Area; the Dredge Spoils Area; the Central Area; and the Sewer Line Area. All sampling was conducted in accordance with the NJDEP guidelines as defined in the Field Sampling Procedures Manual (May 1992 ed.). The results of the survey and information on the quality assurance/quality control of the samples can be found in the Liberty State Park Field Log a copy of which is kept in the Interpretive Center's Library.

Residential Soil Cleanup Criteria are used as a reference for the degree of contamination at Liberty State Park. These numbers are a very conservative estimate of "risk" to human health. The Residential Soil Cleanup Criteria are based on a residential land use where the potential for human exposure is continual, 24 hours a day, 350 days per year over 30 years. Because the length of time an individual uses the Park is much less than the time used for residential criteria, the use of this standard will be even more protective of human health. By using the Residential Soil Cleanup Criteria to base any necessary remedial activities at Liberty State Park, the NJDEP determined that any remedial alternative analyses and remedial actions identified would be conservative to prevent a threat to human health and the environment.

Middle Cove

Background

The Middle Cove area is a former tidewater basin located along the Park's eastern waterfront and is comprised of approximately 9 acres of land. When the Liberty State Park sea wall was constructed, soil from the area surrounding the McAllister Tug and Barge property was excavated and deposited in the Middle Cove area. In 1987; then were concerns that the deposited material contained chromate waste contamination. Previous sampling events in Middle Cove did not identify chromate waste contamination there.

Sampling Activity

In January 1995, the NJDEP conducted sail borings in Middle Cove. Six initial exploratory borings were conducted to identify the depth of fill material by advancing split-spoon coring devices from the surface to the bottom of the fill material. Historical borings and the initial borings indicated that the maximum depth of the fill in Middle Cove was approximately 27 feet deep. A well-sorted, coarse sand cap ranging in depth from three to eight feet covered the entire Middle Cove area. In some areas, such as boring Al, the entire boring from surface to 25 feet was coarse sand with no indication of other fill material.

General field observations of stained soil indicated that some areas of the Middle Cove are contaminated with petroleum hydrocarbon material. This finding was supported by field equipment results, such as the Organic Vapor Analyzer (OVA) and Photoionization Detector PD), and was confined by laboratory analysis of the soil. The stained soil had a distinct petroleum-like odor and was present in finer grained soils under the coarse sand cap. The petroleum stained soil also occurred significantly below the water table surface in a few of the borings (MC6 at 16 - 22 feet) (Appendix ill. Map II] and Appendix IX part 2 Boring Logs Middle Cove).

Analysis

The soil boring program demonstrated that the Middle Cove area has been filled primarily with coarse sand with some evidence of other fill material. The two eastern-most borings, Cl and C2.5, contain the largest amount of non-sand fill material. Analytical results indicate that boring Cl contained concentrations of total petroleum hydrocarbons (TPHC), at 6400 pans per million (ppm). Test boring Cl also contained elevated levels of lead at 479 ppm, which is above the Residential Soil Cleanup Criteria of 400 ppm.

A floating product layer has not been documented on the water table in any of the borings or the well point located at the down gradient portion of the Middle Cove area. Analysis for evidence of chromate waste was also negative.

Contaminated fill material is present in the Western portion of Middle Cove. There is visual evidence of petroleum hydrocarbon product in Middle Cove, but no significantly elevated concentrations were identified in the soil samples. The extent of the fill material has been defined by visual observations and/or with analytical data. The fill material is largely present only below the water table and does not appear to have impacted ground water in down gradient portions of the Middle Cove area. There is six to eight feet of clean sand overlying the contaminated fill material thus human exposure to the contamination is highly unlikely. Based on the observations and the analytical data, there is no threat to human health due to exposure to the contaminated fill material, and therefore there is no further action necessary in Middle Cove area at this time. Based on the analysis of the data collected in the area, any development of this area which would not disturb the clean sand would be unrestricted. If' in the future there is a documented impact to ground water related to petroleum hydrocarbons, it may be necessary to conduct further investigations and remedial alternative analyses of this area as a potential source of contamination.

Freight Yard

Background

The Freight Yard Area is an undeveloped area in the central portion of the Park comprised of approximately 205 acres. The Freight Yard boundaries are defined as Phillips Drive to the west, Freedom Way to the east, James Hamill Drive to the south and Audrey Zapp Drive to the north. Included with the Freight Yard Area soil sampling event was the Waterfront Area, a 59 acre portion located between Freedom Way to the east, Liberty Walk to the west, the Rail Terminal to the north and Middle Cove to the south.

The Freight Yard Area was created between 1860 and 1919 with fill material consisting of trash, excavation spoils and dredge spoils from the Elliss Island area and New York City. The area was used by the Central Railroad of New Jersey for coal transport, stockyards and the main freight switching yard of the railroad in Jersey City. Small spills and sloppy housekeeping may have contributed to the contamination in this area.

Freight Yard - Soil Staging Area

Background

The Soil Staging Area is a 33 acre section located in the south-west corner of the Freight Yard Area bordered by Phillips Drive to the west, and James Hamill Drive to the south. This area has been designated as a storage area to stockpile an estimated. 800,000 cubic yards of clean soil being transported from a Department of Transportation wetlands restoration project in Wayne, New Jersey. Prior to the first soil shipment in December 1994, the NJDEP completed a site characterization of this area.

Sampling Activity

In October 1994, the NJDEP conducted soil borings in the Soil Staging Area. A total of 34 test pits were excavated to a maximum depth of seven feet or to the depth of ground water, whichever was less, and 11 soil samples were collected for analysis.

Analysis

The data results from the Soil Staging Area showed low levels of polycyclic aromatic hydrocarbons (PAR) and inorganic contamination. The PAR contaminants are considered byproducts of incomplete coal combustion. The only contaminant of concern was arsenic. Arsenic was identified at test pit 04B above Residential Soil Cleanup Criteria. The level detected was 429 ppm and the Residential Soil Cleanup Criteria for Arsenic is 20 ppm. In subsequent sampling to delineate the extent of this elevated arsenic contamination, four soil borings were taken around this test pit. All four samples had concentrations below 20 ppm.

Most of the contaminants identified in the Soil Staging Area are similar to those identified in historic fill. The only contaminant not typical of historic fill, and in need of further evaluation, is arsenic. It will be necessary to conduct additional limited investigation of the arsenic and further remedial alternative analysis to determine the remedial action necessary for this area. Areas of historic fill would be amenable to a protective cover to prevent human exposure such as a one foot layer of clean soil.

Freight Yard – Central Area

Background

The Freight Yard - Central Area is a 125 acre section of the 205 acres known as the entire Freight Yard Area. This section excludes the Soil Staging Area and the Dredge Spoils Area. As previously stated the Waterfront Area was included in this sampling event and the results are included in this section.

Sampling Activity

The sampling protocol and parameters for the Central Area of the Freight Yard were based on the sampling results of the Soil Staging Area, visual observations of the site and a review of aerial photographs of previous land uses. In February 1995 the NJDEP installed 82 test pits and collected 52 samples for laboratory analysis. Visual observations and field equipment (OVA and P11)) readings were also used to detect the presence of petroleum hydrocarbons, which were also confirmed by laboratory analysis.

Analysis

A wide range of contaminants, primarily semi-volatile organics and metals, were identified above the Residential Soil Cleanup Criteria in the Central Area of the Freight Yard. Most of the semi-volatile organics present above the Residential Soil Cleanup Criteria are by-products of incomplete coal combustion. These include, but are not limited to, compounds such asBenzo(b)Fluoranthene, Pyrene, Chrysene, and Phenanthrene.

Arsenic and lead were detected at levels exceeding the Residential Soil Cleanup Criteria which are 20 ppm for arsenic and 400 ppm for lead. Arsenic was detected above the Residential Soil Cleanup Criteria at multiple locations, with a maximum concentration of 545 ppm at test pit 25. The maximum concentration of lead, also at test pit 25, was 12,800 ppm and exceeded the Residential Soil Cleanup Criteria in numerous other samples.

Dieldrin, a pesticide, was detected above Residential Soil Cleanup Criteria of 0.042 ppm at testpit 69 at 0.28 ppm. Also, one PCB sample had a concentration of 4.3 ppm which is above Residential Soil Cleanup Criteria of 0.49 ppm, located at test pit 72.

<u>Summary</u>

Contaminants detected in the Central Area of the Freight Yard are similar to those found in historic fill, With a few exceptions,, the contaminants which exceed the Residential Soil Cleanup Criteria are isolated and contaminant levels are consistent with those identified in historic fill.

It is considered acceptable for sites with areas of historic fill to be capped, such as a one foot cover of clean soil, and to restrict use of those areas winch would create any disturbance or removal of the soil cover. This would be a suitable remedy to prevent exposure in most sections of the Freight Yard Area.

It may be necessary to evaluate separately the area where test pits 53 and 54 are located. The area of petroleum stained soils at these locations may need further investigation based on the future use of this area.

The fill material at test pits 69 and 72 where the elevated levels of pesticide and PCB's were detected is not characteristic of historic till. Contamination at these two locations in the Waterfront Area will require additional limited investigation and remedial alternative analysis which will incorporate any future use of this area of the Park.

Freight Yard Area – Dredge Spoils

<u>Background</u>

The Dredge Spoils Area is a 47 acre section located in the north-east corner of the Freight Yard Area, bordered by Audrey Zapp Drive and Freedom Way. It was a surface impoundment constructed in 1980's for the purpose of storing dredged material generated by construction of the sea wall along the eastern edge of the Park for Liberty Walk and allowing these materials to dewater. Roughly 335,000 cubic yards of dredged material was placed in the impoundment between 1981 and 1987. In 1983 the eight foot high earthen berms that formed the impoundment for the dredge spoils were excavated and spread over the entire site. Previous sampling in 1987 showed elevated metals and high sulfide gases. These results were typical of harbor sediment in industrialized areas.

Sampling Activity

This sampling event was conducted to confirm previous analytical results.

In February 1995 the NJDEP conducted test soil borings based on a uniform gird pattern in the Dredge Spoils Area. A total of 46 samples, taken at approximately six-foot depths, were obtained for laboratory analysis.

Analysis

Initial analytical results for the Dredge 5poi1s area indicate the presence of low levels of semi- volatile organics similar to those detected at the Freight Yard Area. Low levels of PCB's above Residential Soil Cleanup Criteria were identified at five of the six samples which were analyzed for them. The maximum concentration was 1.31 ppm, slightly elevated above the Residential Soil Cleanup Criteria of 0.49 ppm at test pit QI8DS, which was collected at a soil depth between 0-2 feet. Dieldrin, a pesticide. was detected above the Residential Soil Cleanup Criteria of 0.0.42 ppm at two locations within the Dredge Spoils Area. One sample had a concentration at 0.72 ppm and a second sample had a concentration at 0.57 ppm. Arsenic, lead and cadmium were found in concentrations above the Residential Soil Cleanup Criteria. In 31 of the 46 samples, Arsenic exceeded the Residential Soil Cleanup Criteria of 20 ppm and Cadmium exceeded the Residential Soil Cleanup Criteria of 1 ppm in 26 of the 46 samples. Lead only exceeded the Residential Soil Cleanup Criteria of 400 ppm in 6 of the 46 samples, with the maximum concentration at 3380 ppm.

Most of the contaminants identified in the Dredge Spoils Area were similar to those found in other areas of the Park although the levels tended to be much lower. However, PCB’s and pesticides are not typical historic fill for the Park. One area of potential concern is the location of test pit Q1SDS, where the P0 contamination was detected. The sample was collected within the first two feet of the surface soil, thereby increasing the potential of human contact. At this time a chain-link fence surrounds the entire Dredge Spoil Area prohibiting public access and therefore minimizing a direct contact threat. There is a well-established vegetative cover which virtually eliminates any risk of contaminated dust particles becoming air-borne minimizing any potential of an inhalation threat. Given the current use of this area, it would be appropriate to maintain a vegetative cover with a one foot cover of clean soil as a remedial measure. Additional investigation to delineate the extent of the pesticide and PCB contamination and remedial alternative analyses may be necessary to evaluate any future use of this area.

Freight Yard – Sewer Line

<u>Background</u>

An sewer line runs through the Freight Yard Area beginning at a sewer out-fall pipe on Phillips Street which is located on the west side of the Freight Yard and terminates at a diffuser box in Lower New York Bay. Due to the historical practice of utilizing chromite ore processing residue or chromate waste residue as fill material in Hudson County, it was decided to sample along the sewer pipeline to determine if chromate waste was present.

Sampling Activity

The initial sampling plan required that test pits were to be dug every 100 feet along the sewer line. However, when the first test pit was installed, it was determined that field conditions were not conducive for the use of a backhoe to dig the test pits. The depth of the pipeline ranged from approximately eight to 13 feet deep. Ground water along the pipeline was just over six feet deep. During the first attempt to dig through the water table, the test pit repeatedly collapsed due to ground water in the pit. The sampling plan was then modified to conduct borings with a drill rig, which allowed samples to be successfully obtained.

To determine where the 80 inch sewer out-fall pipeline was located to obtain samples, a geophysical subsurface mapping procedure was used. Utility maps were used as a basis for estimating the pipeline position with reasonable accuracy. The western-most section of the pipeline was approximately 25 feet further north than expected. Additionally, the geophysical investigation also indicated and plotted the location of two buried utility pipelines which were further south than expected. It is expected that they were formerly used as a potable water supply line to Ellis Island and an eight inch sanitary sewer force main from Ellis Island. These pipelines ran parallel to the sewer out fall pipe, approximately 100 feet north. The successful mapping technique permitted accurate positioning of the drill rig to obtain samples next to the pipeline.

Analysis

The contaminants tested for in this area were limited to total and hexavalent chromium. Three samples, 5WL24, SWL25, 5WL26, all exceed the current Soil Cleanup Criteria for both total and hexavalent chromium. This represents an area of greater than 200 linear feet of chromium contamination. The maximum concentration of total chromium was 26,800 ppm at SWL25. Hexavalent chromium was detected as well in these three locations exceeding the 10 ppm Residential Soil Cleanup Criteria, with the maximum concentration of 42 ppm (Appendix XII part 5, Sewer Line Area Boring Logs). These samples were particularly wet and may warrant further sampling to confirm and better delineate the extent of contamination.

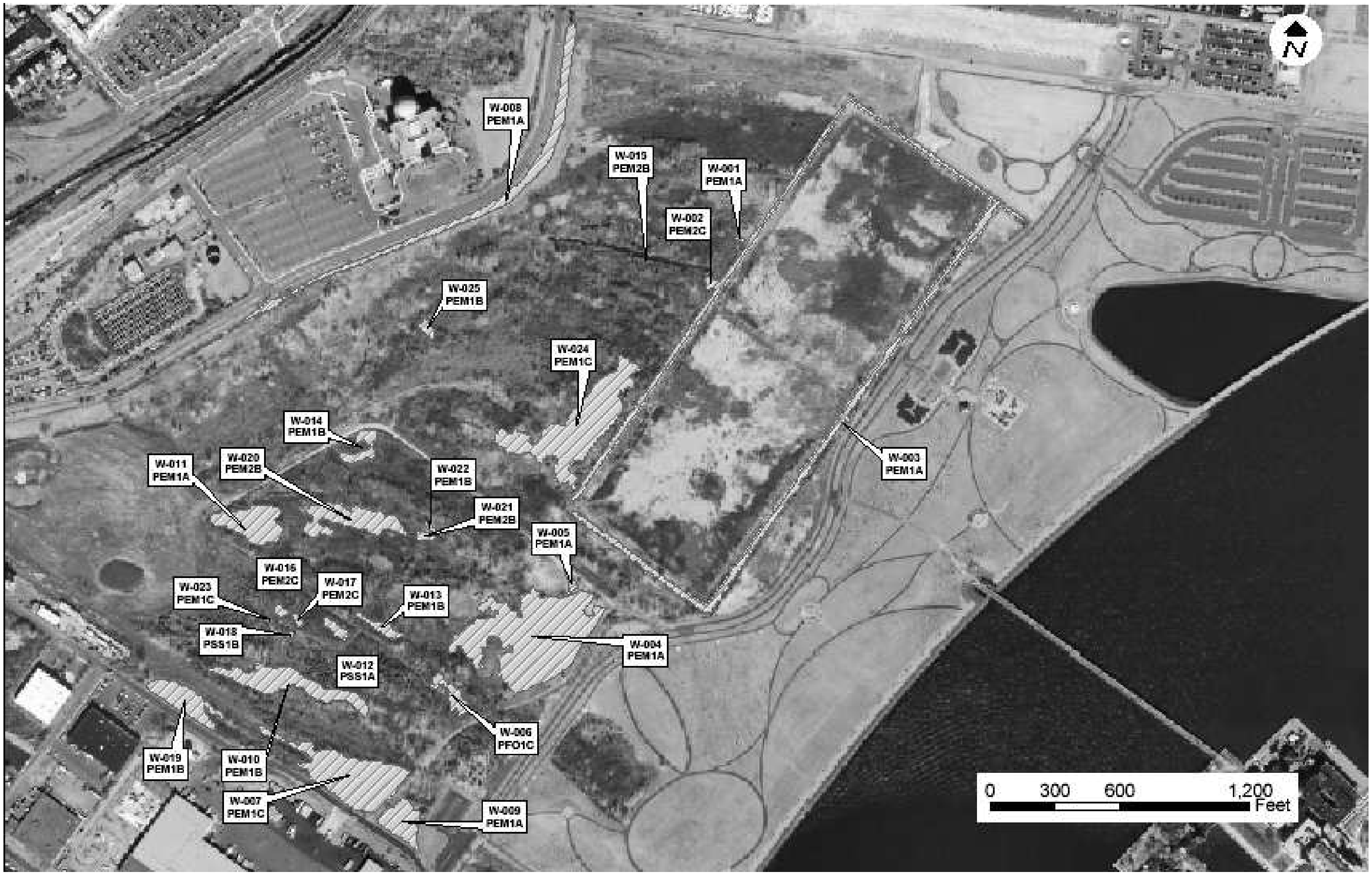
Due to the depth at which the chromium contamination is present, the risk to human health and the environment is minimal. However, it will be necessary to conduct a further investigation of the contamination to delineate the extent and a remedial alternative analysis be developed to determine the appropriate remedial action based on the additional information. Since this site is located within the Soil Staging Area. it will be necessary to isolate this area from the placement of soil from the Department of Transportation wetlands restoration project until the subsequent investigation has been completed.

Conclusion

The soil at Liberty State Park consist primarily of cinder and ash fill, mixed rubble and debris and dredge spoil. The contamination levels are well established and are typical of historic fill. Using the current NJDEP Soil Cleanup Criteria and remedial policies established by the NJDEP, it is recommended that exposure to surface soils in the rail yard and dredge spoil areas be eliminated through institutional and other engineering controls, such as a one foot cover of clean soil.

Appendix VI

Mapped Freshwater Wetlands Description



Wetland Number	Type	Ecological Community	Size (Acres)	Substrate	Wetland Indicators	Dominant Vegetation
W-5	PEM1A	Common Reed-dominated Wetland	0.06		Saturated soils at surface	Common reed: FACW; Reed canary grass: FACW+
W-6	PFO1C	Floodplain Forest Wetland	0.26		Standing water; surface water depth 2"; Water table 6" below surface Buttressed tree trunks; Low chroma soils	Eastern cottonwood (*Populus deltoides*): FAC; Gray birch (*Betula populifolia*): FAC; Purple loosestrife: FACW+; Poison ivy (*Toxicodendron radicans*): FAC
W-7	PEM1C	Shallow Emergent Marsh	2.10	Disturbed sandy soils	Surface soil saturated; Water table 6" to 16" below surface Low chroma soils; Water stained leaves	Common reed: FACW; Purple loosestrife: FACW+; Steeplebush: FACW; Eastern cottonwood: FAC
W-8	PEM1A	Common Reed-dominated Wetland	1.56	Gravel	Standing water; Surface water depth from 0" to 12"	Common reed: FACW; Purple loosestrife: FACW+; Swamp rosemallow (*Hibiscus moschuetos*): OBL; Curlytop knotweed (*Polygonum lapathifolium*): FACW+

Wetland Number	Type	Ecological Community	Size (Acres)	Substrate	Wetland Indicators	Dominant Vegetation
W-9	PEM1A	Common Reed-dominated Wetland	0.59		Surface soils saturated; Water table 6" below surface; Low chroma soils with mottling	Common reed: FACW; Purple loosestrife: FACW+; Eastern cottonwood: FAC
W-10	PEM1B	Common Reed-dominated Wetland	1.20	Disturbed sand and clay soils	Surface soils saturated; Water table 5" below surface; Low chroma soils with mottling	Common reed: FACW; Purple loosestrife: FACW+; Eastern cottonwood: FAC
W-11	PEM1A	Common Reed-dominated Wetland	0.90		Standing water; Surface water depth 2"; Water table 4" below surface; Low chroma soils	Common reed: FACW; Sensitive fern: FACW; Marsh fern (*Thelypteris palustris*): FACW+; Purple loosestrife: FACW+
W-12	PSS1A	Shrub Swamp	0.12	Gravel; sand; shell	Standing water; Surface water depth 1" in some areas; Water table 4" below soil surface; Water stained leaves, shallow root systems and buttressed trees	Red maple (*Acer rubrum*): FAC; purple loosestrife: FACW+; Steeplebush: FACW; Sensitive fern: FACW; Marsh fern: FACW+; Swamp azalea (*Rhododendon viscosum*): OBL; Gray birch (*Betula populifolia*): FAC

Wetland Number	Type	Ecological Community	Size (Acres)	Substrate	Wetland Indicators	Dominant Vegetation
W-13	PEM1B	Common Reed-dominated Wetland	0.18	Coarse sand	Saturated soils; Water table 2" below soil surface; Low chroma soils	Common reed: FACW; Marsh fern: FACW+; Poison ivy: FAC; Gray birch: FAC
W-14	PEM1B	Common Reed-dominated Wetland	0.29	Debris and soil	Low chroma soils with mottling	Common reed: FACW; Canadian clearweed (*Pilea pumila*): FACW
W-15	PEM2B	Common Reed-dominated Wetland	0.01	Cinders	Standing water; Surface water depth 2"	Common reed: FACW; Sensitive fern: FACW; Gray birch: FAC
W-16	PEM2C	Shallow Emergent Marsh	0.06		Standing water; Surface water depth 12"+; water stained leaves	Woolgrass (*Scirpus cyperinus*): OBL; Common reed: FACW; Northern bayberry (*Myrica pensylvanica*): FAC; Gray birch: FAC; Red maple: FAC
W-17	PEM2C	Shallow Emergent Marsh	0.04		Standing water; Surface water depth 8"	Purple loosestrife: FACW+; Woolgrass: OBL; Northern bayberry: FAC; Gray birch: FAC; Red maple: FAC

Wetland Number	Type	Ecological Community	Size (Acres)	Substrate	Wetland Indicators	Dominant Vegetation
W-18	PSS1B	Shrub Swamp	0.02		Saturated soils; Water table 16" below surface; Low chroma soils with mottling	Silky dogwood (*Cornus amomum*): FACW; Purple loosestrife: FACW+; Yellow nutsedge (*Cyperus esculentus*): FACW; Gray birch: FAC; Northern bayberry: FAC
W-19	PEM1B	Common Reed-dominated Wetland	0.72		Saturated soils; Low chroma soils with mottling; Water stained leaves	Common reed: FACW; Purple loosestrife: FACW+; Marsh fern: FACW+
W-20	PEM2B	Shallow Emergent Marsh	0.99		Saturated soils; Water table 16" below surface; Low chroma soils with mottling	Marsh fern: FACW+; Sensitive fern: FACW; Royal fern (*Osmunda regalis*): OBL; Narrowleaf cattail (*Typha angustifolia*): OBL; Steeplebush: FACW; Purple loosestrife: FACW+; Common reed: FACW; Canadian clearweed: FACW
W-21	PEM2B	Shallow Emergent Marsh	0.08		Saturated soils; Water stained leaves and water marks	Sensitive fern: FACW; Steeplebush: FACW; Purple loosestrife: FACW+; Common reed: FACW

Wetland Number	Type	Ecological Community	Size (Acres)	Substrate	Wetland Indicators	Dominant Vegetation
W-22	PEM1B	Shallow Emergent Marsh	0.03		Saturated soils; Water stained leaves and water marks; Low chroma soils	Reed canary grass: FACW; Steeplebush: FACW; Eastern cottonwood: FAC; Gray birch: FAC
W-23	PEM1C	Common Reed-dominated Wetland	0.02		Standing water observed on several occasions.	Phragmites australis, FACW; Onoclea sensibilis, FACW.
W-24	PEM1A	Shallow Emergent Marsh	2.68		Saturated soils at surface;	Wool grass (*Scirpus cyperinus*): OBL; Soft rush: FACW+; Red maple (*Acer rubrum*): FAC; Common reed: FACW; purple loosestrife: FACW+
W-25	PEM1B	Common Reed-dominated Wetland	0.08		Frost in soil – soil saturated.	Phragmites australis, FACW; Onoclea sensibilis, FACW.

Appendix VII
Normalized Vegetative Difference Index (NDVI)

Appendix VIII

Correlation between 1995 Soil Metals and Initial Ecological Characteristics

Metals (1995) and Bio (2004-5)

		Type	Chromium	Cobalt	Copper	Iron	Lead	Magnesium
TP--8	**SOF**	SC	48.4	8.3	238	43200	436	1460
TP--40	**SOF**	SC	60.7	7	239	51700	687	449
TP--21	**DMM**		47.9	11.1	415	40400	724	2780
TP-40C	**DMM**	SC						
TP--16	**DMW**		30.3	5.4	78.8	15000	202	568
TP--3	**MS**		34.9	8.2	99.9	19700	753	2530
TP--7	**MS**	SC	157	154	778	87300	789	3150
TP-17	**MS**		122	28.5	2900	144000	2440	3020
TP-40B	**MS**	SC						
TP--14/17	**MS/DMM**	BA	122	28.5	2900	144000	2440	3020
TP--10	**SNH**	BA	56	8.4	190	37500	512	481
TP--14	**SNH**	BA	11.1	2.8	30.1	18200	37	855
TP14/16	**SNH**	BA						
TP--18	**SNH**	BA	57.1	6.6	171	32500	338	575
TP--24	**SNH**	BA	78	17.7	778	82000	511	1900
TP--25	**SNH**		55.1	337	3790	99500	11800	2730
TP--28	**SNH**	BA	13.8	3.7	53.2	13300	3240	845
TP--41	**SNH**	BA	20.1	3.3	57	16900	556	1130
TP--43	**SNH**	BA	34.8	16	145	25400	106	3280
TP-43/14	**SNH**	BA						
TP--48	**SNH/W**	BA	26.4	3.9	72.9	20300	76.7	610
Mean			57	38	761	52406	1509	1728
Meadium			48.15	8.3	181	35000	534	1295
Median Urban Peidmont			18.5	6.3	29.5	14600	111	2190
NJ Residential Standard			240		600		400	
R sq.	**D BA**		32.5	29.4	55.1	55.6	5.9	7.3
R sq.	**D SC**		32.8	43.6	42.7	25.6	0.2	78.8
R sq.	**H'**		9.6	18.9	3.1	3.5	0.4	3.6
Correlations (Pearson) H'			-0.144	0.03	0	-0.15	0.18	0.33

Metals (1995) and Bio (2004-5)

		Type	Manganese	Mercury	Nickel	Potassium	Selenium	Silver	Sodium
TP--8	**SOF**	SC	277	0.61	28.4	545	2.1	1.1	205
TP--40	**SOF**	SC	227	0.73	34.6	545	2.1	0.95	84.5
TP--21	**DMM**		455	0.62	44.1	416	1.2	1.1	162
TP-40C	**DMM**	SC							
TP--16	**DMW**		63.7	0.34	25.1	673	1.1	1.1	70.9
TP--3	**MS**		288	0.92	29.6	990	1.2	1.2	119
TP--7	**MS**	SC	231	0.75	49.5	1200	3.5	0.97	286
TP-17	**MS**		1120	0.65	163	1110	0.88	3.1	301
TP-40B	**MS**	SC							
TP--14/17	**MS/DMM**	BA	1120	0.65	163	1110	0.88	3.1	301
TP--10	**SNH**	BA	209	0.74	31.4	464	3.1	1.1	202
TP--14	**SNH**	BA	68.2	0.97	9.3	772	0.6	1.1	276
TP14/16	**SNH**	BA							
TP--18	**SNH**	BA	231	0.72	25.5	277	1.7	1.1	75.2
TP--24	**SNH**	BA	796	0.5	86.3	401	0.77	1.2	142
TP--25	**SNH**		475	1	112	1460	5.2	1.8	1780
TP--28	**SNH**	BA	70.2	0.33	20.1	316	2.2	1	109
TP--41	**SNH**	BA	105	0.47	13.4	245	0.86	1.2	56
TP--43	**SNH**	BA	312	0.26	30.8	1340	0.36	1	90.6
TP-43/14	**SNH**	BA							
TP--48	**SNH/W**	BA	53.3	0.17	11.5	1270	1.4	1.1	97.2
Mean			359	1	52	773	2	1	256
Meadium			231	0.65	30.2	609	1.3	1.1	130.5
Median Urban Peidmont			311	0.5	12.4	693	0.41	<DL	90.1
NJ Residential Standard				14	250			110	
R sq.	**D BA**		43.6	7	51.2	7.3	4.5	53.3	60.9
R sq.	**D SC**		15.6	4	16.9	42.8	42.8	19.5	95.2
R sq.	**H'**		0.3	14.3	1.4	3.7	6.6	1.7	47
Correlations (Pearson) H'			-0.033	-0.747	-0.07	0.079	-0.372	-0.107	-0.737

Metals (1995) and Bio (2004-5)

		Type	Tallium	Vanadium	Zinc	pH level	Density	H
TP--8	**SOF**	SC	0.54	30.3	520		23.4	0.558071
TP--40	**SOF**	SC	0.36	87.3	119		116.3	0.889741
TP--21	**DMM**		0.43	48	786	7.4		
TP-40C	**DMM**	SC					67.07	0.358724
TP--16	**DMW**		0.42	36.2	59.4			
TP--3	**MS**		0.77	59.9	523	6.6		
TP--7	**MS**	SC	0.37	31.1	1060		0.22	0.13027
TP-17	**MS**		0.5	101	7660	6.9		
TP-40B	**MS**	SC					24.7	0.113131
TP--14/17	**MS/DMM**	BA	0.5	101	7660	6.9	0.097	0.361334
TP--10	**SNH**	BA	0.56	54.2	601	5.4	1.3232	0.103694
TP--14	**SNH**	BA	0.41	19.8	27.4		0.92	0.171337
TP14/16	**SNH**	BA					0.909	0.265617
TP--18	**SNH**	BA	0.4	68.3	82	4.5	2.19	0.483094
TP--24	**SNH**	BA	0.47	48.1	1180	6.9	0.806	0.406217
TP--25	**SNH**		0.59	60.5	16000	5.2		
TP--28	**SNH**	BA	0.42	40.2	622	7.8	1.692	0.521976
TP--41	**SNH**	BA	0.46	25.2	2180		1.52697	0.514121
TP--43	**SNH**	BA	0.39	22	345		2.036	0.574807
TP-43/14	**SNH**	BA					0.818	0.274992
TP--48	**SNH/W**	BA	0.2	26	19.9		1.14	0.455902

Mean	0.4	51	2282.688
Meadium	0.445	44.1	526
Median Urban Peidmont	<DL	29.6	75.3
NJ Residential Standard			1500
R sq. D BA	4.4	17.7	47.5
R sq. D SC	14	96	83.5
R sq. H'	5.6	6.5	1.7
Correlations (Pearson) H'	-0.437	-0.15	-0.02

Curriculum Vita

Frank Gallagher

2004–present, Administrator: Division of Parks and Forestry, Office of the Director. Responsible for the future development of Liberty State Park, review and analysis of plans, policy, and projects concerning the current brownfield restoration and redevelopment issues.

1995–present, Adjunct Professor (PTL): Rutgers the State University, the Department of Ecology, Evolution and Natural Resources. Instructor of Environmental Issues, a survey course examining recent trends of the major issues at the global and local levels.

2007-Columbia University: Visiting Critic: A6853 Design Studio Organized a multi-disciplinary collaborative MAAD graduate design studio with Kate Orff

2004 Acting Assistant Director: Division of Parks and Forestry, State Park Service, Oversight of New Jersey's 380,000+ acres of parks, forests, recreation areas.

1994-2004 Administrator: Division of Parks and Forestry; 1994 - 2004. As an administrator within the Office of the Director, I am responsible for the development of site specific general management plans, and the oversight of the division's interpretive and educational programs.

1996–2004 Commissioner: New Jersey Commission on Environmental Education. The commission was established by the Environmental Education Act in January of 1996. Served as the commission's chair from 1999-2003.

1983-1985 Adjunct Professor of Biology: Upsala College. Responsibilities included instruction in the following courses: Introduction to Biology, Field Biology, Environmental Science and Evolution.

1984-1994 Chief of Interpretive Service: Liberty State Park, DEP. Responsibilities included the design and implementation of a comprehensive interpretive program, including educational programs, teacher training and exhibit development.

1987-1988 Senior Biologist: CDM Inc. CDM is an environmental consulting firm that deals with a wide spectrum of issues.

1986-1987 Senior Biologist: Aware Inc.; 1986-87. Aware Inc. is an environmental consulting firm that deals primarily with toxic and hazardous waste.

1980-1984 Naturalist: High Point State Park; 1980-84. Prepared and presented interpretive and educational programs.

1978-1984 Natural Resource Assistant: Sussex County Soil Conservation District.

Education

2008 Ph.D., Rutgers the State University, Ecology Evolution and Natural Resources.
1989 Post Graduate Work, Rutgers the State University, Graduate School of Education.
1983 M.A., Biology, Montclair State College.
1978 B.A., Botany, Rutgers the State University.

Publications:

2008 Gallagher F.J., Pechmann I., Bogden J., Grabosky J., and Weis P. Soil Metal Concentrations and Plant Productivity in an Urban Brownfield, Environmental Pollution. In Press.
1998 Gallagher, F.J. *Branching Out, Risk and Forestry*. American Forest Foundation.
1998 Iozzi, L. and Gallagher, F. *Environmental Issues Focus on Risk*., American Forest Foundation..
1997 Gallagher, F.J. *The Presidents Message*., The Daily Planet. Vol. 10 No. 3.
1995 Gallagher, F.J. *HEP Hits the Road*., The Tidal Exchange Vol. 5. No. 1.
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1993 Gallagher, F.J. *Communication?*, The Daily Planet Vol. 8 No. 2.
1990 Gallagher, F.J. *Environmental Education, the Status of the Master Plan.*, The Alliance. Vol. 5. No. 1. .
1989 Gallagher, F.J., Kindervatter, D. *Cent' Anni, A Railroad Reborn*. New Jersey Outdoors. Vol. 16. No.5.
1989 Gallagher, F.J. *Efforts Towards a Master Plan for Environmental Education*. The Alliance. Vol. 4. No. 3.
1989 Gallagher, F.J. *Professionalism in Environmental Education.*, The Alliance. Vol. 4. No. 2.
1989 Gallagher, F.J. *Should the Three R's End in EE*. The Alliance. Vol. 4. No. 1.
1987 Gallagher, F.J. *Wildlife Photograph*s. Sussex County Voice.
1979 Gallagher, F.J. *Management and Restoration Guide for Sussex County Lakes*. Sussex County Water Quality Management Program. Sussex County S.C.D. & Planning Dept.

Select List of Invited Lectures:

2007 “The Myths Associated with Sustainable Forestry”, at a meeting of the American Forest Foundation. Idaho City, Idaho.

2007 “The Relationship between Soil Metal Load and Vegetative Assemblage Structure, Colombia University, School of Architecture.

2007 “Soil Metal Concentrations and Vegetative Assemblage Structure in an Urban Brownfield” poster presentation at the Meadowlands Symposia, Hackensack, New Jersey.

2007 “Soil Metal Concentrations and Vegetative Assemblage Structure in an Urban Brownfield”, International Society Arboriculture, Conference Hawaii.

2006 “Demographic Transition and the Corresponding Need for Natural Resource Based Educational Materials. New Jersey Shade Tree Federation’s annual conference in Cherry Hill, NJ.

2005 “Demographic Transition and its Impact on Regional Forest Resources”. The American Forest Foundation’s Project Learning Tree Conference, International Conference. Portland Or.

2001 “Connecting Forestry to People” Amman Jordan, invitation through the American Forest Foundation by Princess Abdullah of Jordan.

Awards
1999 Outstanding Service, USDA, Forest Service Conservation Education Implementation Team.
1990 Excellence in Environmental Education., New Jersey Audubon.
1989 Excellence in Environmental Education, 1989, the Alliance for New Jersey. Environmental Education.

Certifications and Licenses
1992 Brain-Based Instructional Strategies, Academy for Professional Development, Department of Education.
1991 4 MAT Learning Styles and Modalities, Introductory and Intermediate, Department of Education.
1985 Federal Fish Game and Wildlife Service, Special Purpose, Rehabilitation.
1983 Federal Fish Game and Wildlife Service, Master Bird Banding Permit.