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Characterization of urban wetland vegetation and management practices

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CHARACTERIZATION OF URBAN WETLAND VEGETATION
AND MANAGEMENT PRACTICES

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Abstract

Urban wetlands are important ecosystems to moderate flooding risks and improve water quality. Vegetation is a key component of urban wetlands, as plants promote sedimentation, play key roles in biogeochemical cycling, and provide food and habitats for other organisms; however, little is known about the standing vegetation and seed banks of urban wetland plant communities. Understanding variables that can impact the establishment and growth of wetland plants can increase the success of urban wetland management and rehabilitation projects. This research investigates the standing vegetation and seed banks of urban wetlands in Broome County, New York, with the ultimate goal of identifying plant species that we would recommend for urban wetland restoration or creation projects.

Standing vegetation and soil characteristics were sampled in eight urban wetlands in south-central New York to characterize the vegetation and soil parameters and to compare these features to those of previously sampled natural wetlands. Urban sites had a higher percent cover of invasive plants and significantly lower species richness. However, native species were also common in urban flora. Urban wetland vegetation and soil characteristics are different than those in nearby natural wetlands, and our increased knowledge of these urban ecosystems allowed us to identify native species that can be used in urban wetland restoration projects.

Urban wetland seed banks were profiled by exposing sediment cores from four wetlands to flooded and drawdown treatments in the Research Greenhouse at

Binghamton University. We found high spatial variation in species richness and seedling density among the sites. Invasive species comprised a high percentage of seedlings for three wetlands, but not for the fourth site. Our findings illustrate that urban wetland seed banks may be viable and can contribute to the revegetation of disturbed sites, but supplemental planting of native species should be considered to reduce the establishment of invasive species.

We evaluated the effects of a complete regrade and expansion of an urban retention wetland on its seed bank and standing vegetation. The density and species composition of seedlings that emerged from the seed bank were determined under drawdown and flooded conditions from sediment cores collected before (2011) and after (2014) the regrade. The standing vegetation composition was recorded just prior to the regrading, and twice in each growing season (2012-2014) after the regrade. Seedling densities were nearly three-fold greater than those after regrading, and seedling density significantly decreased in the drawdown treatment. Species richness in the standing vegetation decreased immediately after the regrade and rebounded over three years. This study indicates that a regrading project can substantially reduce seedling density of an urban wetland seed bank, but standing vegetation may show signs of recovery within a short time span, perhaps due to the presence of a prolific bud bank.

To determine if certain plant species may be more tolerant of urban wetland characteristics, we conducted two experiments to distinguish between sediment and flooding effects: 1) the growth responses of five plant species to the sediment from three different urban wetlands, both *in situ* and at a common garden site, and 2) a flooding regime study which assessed the growth responses of three wetland plant species to four different flooding regimes. Species that were commonly found in urban wetlands generally had higher mean relative growth rates than those not commonly found in urban wetlands. We observed that plants had higher relative growth rates at the common garden site than in the wetlands. Thus, we expect that hydrological variables may have more of an impact on native species establishment and growth in urban wetlands than sediment characteristics. Our results indicate that different species may vary in their responses to flooding regimes.

This work shows that urban wetlands are fundamentally different from natural wetlands in south-central upstate New York, and that these ecosystems need to be managed appropriately. Although invasive species are common in urban wetlands, some native species can establish and survive under urban conditions, and these species should strongly be considered in planting schemes of creation or rehabilitation projects in urban landscapes.

This dissertation is dedicated to my father, Arthur Larson.
Your love of nature sparked my love of science.

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Chapter 1: General Introduction

Wetlands are a critically important group of habitats that globally cover an area 33% larger than that of the United States (Millennium Ecosystem Assessment 2005). These ecosystems provide a number of services, including food for humans and wildlife, water supply, erosion control, nutrient cycling, waste treatment, and climate regulation (Costanza et al. 2014; Mitsch and Gosselink 2015). Wetlands retain storm runoff, and are therefore important for flood-prone areas. Water inputs into wetlands often have a high turbidity, which can be harmful for human consumption. As the water flow slows in wetlands, these particles can settle, and the result is a higher quality of water output (Farrell and Scheckenberger 2003; Mitsch and Gosselink 2015). Surface water in wetlands infiltrates into groundwater storage, a source of clean water that benefits between 1.5 and 3 billion people (Millennium Ecosystem Assessment 2005). Although the economic values of ecosystem services are hard to quantify (Boyer and Polasky 2004), wetlands are estimated to be worth over $140,000/ha/year as of 2011 (Costanza et al. 2014).

Despite these ecosystem services, more than 50% of the wetlands in the United States have been lost in the past 200 years (Dahl 1990), and more than half of the remaining wetlands have been altered due to agriculture and urbanization (Mitsch et al. 1998; Mitsch and Gosselink 2015). Increased appreciation of wetland functions and services has led to the current “no net loss” policy, which declares that while a wetland can be destroyed, another wetland of equal or greater size must be created as a replacement (Boyer and Polasky 2004; Mitsch and Gosselink 2015). The costs of preserving wetlands are likely to be high, particularly in urban landscapes as undeveloped land is a valuable commodity (Boyer and Polasky 2004). Yet these ecosystems are critical for improving the quality of life for urban residents.

The epicenter of human influences can be seen in cities, which hold the greatest density of humans. Urban wetlands are generally defined as wetlands located in urban landscapes with high anthropogenic influences, such as high inputs of pollutants and increased presence of exotic species. With 82% of the total U.S. population residing in urban landscapes, urbanization is a significant cause of coastal and freshwater wetland losses (Ravit et al. 2017; World Bank 2017). Unlike natural wetlands, urban sites experience altered sediment chemistry and flooding regimes due to human activities (Forman 2003; Faulkner 2004; Zhu et al. 2008; Pickett et al. 2011) including increased input of pollutants into the aquatic ecosystems (Pankratz et al. 2007; Göbel et al. 2007; Zhu et al. 2008; Gasperi et al. 2012) and a "flashy hydrology" as a result of increased impervious surface cover (Forman 2003; Ehrenfeld et al. 2003; Pickett et al. 2011). Elevated levels of nutrients and metals from anthropogenic sources may be reduced through sedimentation, uptake by plants, or other biogeochemical processes, thus improving water quality (Gale et al. 1993; Bachand and Horne 1999; Nairn and Mitsch 1999; Harrison et al. 2011). Urban wetlands are particularly important for urban residents in flood-prone landscapes, as these habitats may reduce flooding (Woodcock et al. 2010) and yet they continue to be threatened and neglected (Hettiarachchi et al. 2015). Thus, a high priority for urban managers should be to appropriately restore or rehabilitate sites so as to enhance urban wetland ecosystem functions (Ravit et al. 2017).

While numerous studies have examined urban wetland water quality (Ehrenfeld 2000; Malaviya and Singh 2012), soil quality (Ehrenfeld 2000; Stander and Ehrenfeld 2009a; Stander and Ehrenfeld 2009b) and hydrologic features (Ewing 1996; Moscrip and Montgomery 1997; Kaye et al. 2006; Stander and Ehrenfeld 2009a; Stander and Ehrenfeld 2009b; Pickett et al. 2011), few have examined the plant community and growth responses to sediment and hydrological variables within these systems. Vegetation is a key component of urban wetlands: plants promote sedimentation and improve water quality (Mitsch and Gosselink 2015), provide surface area for colonization by microbial communities (Arshad and Frankenberger 1997), and play key roles in the biogeochemical cycling of nutrients (Faulwetter et al. 2009; Laanbroek 2010). Different species of plants vary in their ecological functions, including nutrient accumulation and retention of nutrients in different tissues (Kao et al. 2003). Understanding variables that can impact the establishment and growth of wetland plants can improve the success rate of urban wetland management and rehabilitation projects.

Additionally, little is known about the composition of urban wetland seed banks and their relationship with the standing vegetation. Seed banks are a potential pool for standing vegetation, and can give us insight into what may naturally germinate in the field (van der Valk and Davis 1978; Leck 2003; Hopfensperger 2007). The seed bank and its compositional similarities to standing vegetation in non-urban landscapes have been studied for various reasons, including revegetation and restoration efforts (Leck 2003; Cobbaert et al. 2004), vegetation dynamics (Zedler 2000; Amiaud and Touzard 2004;

Bossuyt and Hermy 2004; Grandin 2008), and invasive species management (Zedler 2000; Hausman et al. 2007). Seed banks may play a central role in the re-establishment of vegetation after a major habitat alteration (Brown and Bedford 1997; Brown 1998; Cobbaert et al. 2004; Kaplan et al. 2014; Osunkoya et al. 2014). On the other hand, influences of urbanization on environmental quality and hydrology may limit seedling establishment; thus, insight into urban wetland seed banks could have important management and restoration implications.

My doctoral research has focused on plant communities and species' responses to anthropogenic influences experienced by urban wetland ecosystems. In order to understand the plant communities in urban wetlands, we surveyed eight urban wetlands (Appendix A) to document species richness, common invasive and native species, and sediment characteristics (Larson et al. 2016, Chapter 2). The objectives of this study were to characterize the vegetation of urban wetlands and selected soil parameters in south-central New York. By comparing the vegetation and soil characteristics of these urban wetlands to previously sampled natural wetlands, we were able to distinguish key environmental characteristics that will increase the success of urban wetland restoration and rehabilitation projects.

Restoration of urban wetlands may rely on seed banks for revegetation, but because little is known about urban wetland seed banks, we examined their viability by comparing seedling density and species composition both within and among urban wetlands (Larson and Titus 2018, Chapter 3). The main goal of this chapter was to evaluate the profiles of seed banks of four urban wetlands in the vicinity of Binghamton, New York, including species richness, dominant taxa, relative importance of invasive and native species, and the dominant wetland indicator status. Additionally, we compared the species assemblage of seed banks to their respective standing vegetation to discuss potential plant community dynamics in these urban wetlands, allowing us to make recommendations on the potential use of urban sediments for revegetation projects.

Major habitat alterations, like full-site regrading projects, will inevitably impact both the standing vegetation and seed banks of urban wetlands, and understanding these impacts will help ecologists and managers evaluate potential planting or seeding schemes in urban wetland restoration projects. Lieberman is an urban stormwater retention pond located on the Binghamton University campus in Vestal, New York, that underwent a complete regrade and expansion to accommodate increased runoff from new infrastructure. Our main goal was to understand the effects of this regrade on the urban wetland plant community by recording changes in both the seed bank and the standing vegetation (Larson et al. under review, Chapter 4). This study increased our understanding of the roles that seed banks and standing vegetation can play in passive revegetation after a major habitat alteration, and whether urban wetlands are able to recover after such projects.

We observed that some species were commonly found in urban wetlands, while others were surprisingly uncommon, perhaps because certain species may be more tolerant of urban wetland characteristics; therefore, these common species may be more desirable to use in urban wetland planting projects (Chapter 5). This chapter aims to distinguish between the effects of impacted urban sediment and flashy urban flooding regimes on the growth rates of five wetland plant species by conducting two experiments: 1) an urban wetland sediment study and 2) a flooding regime study. The first study examined growth responses of five plant species to the sediment from three different urban wetlands, both *in situ* and at a common garden site. We hypothesized that plants would have higher relative growth rates at our common garden site because plants grown *in situ* would experience a harsher environment: periods of drought, flashy flooding regimes, and potentially more herbivory. We also hypothesized that, based on the sediment characteristics discussed in Chapter 2, plants would have higher relative growth rates in sediment with a higher availability of ammonium. Plants would have the lowest growth rates in sediment with a relatively lower amount of available nitrogen and high soil electrical conductivity. The second experiment assessed the growth responses of three wetland plant species to different flooding regimes. We expected that plants would generally favor drawdown conditions and natural flooding regimes, as opposed to constantly flooded conditions and urban flooding regimes. We also suspected that species that were commonly found in urban wetlands (*Typha* x *glauca*, *Juncus effusus,* and *Leersia oryzoides*) would have higher growth rates than species that were uncommon (*Carex stricta* and *Sparganium americanum*) for both experiments.

The final chapter summarizes our findings about urban wetland vegetation characteristics and the implications of our work for future urban wetland restoration projects.

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Chapter 2: Urban Wetland Characterization in South-central New York State

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Abstract:

Urban wetlands can serve to reduce flooding and improve water quality, yet we know little about their plant communities. Our study aims to characterize the vegetation and soil parameters of these important ecosystems, and to compare these features to those of previously sampled natural wetlands in south-central New York. Vegetation and soil characteristics were sampled in eight urban wetlands and compared to six forested wetlands, five scrub-shrub wetlands, and seven emergent wetlands. Urban sites had significantly lower species richness and a higher percent cover of invasives, including *Typha* x *glauca, Phalaris arundinacea,* and *Lythrum salicaria*. However, non-invasive species were also common in urban flora, including *Leersia oryzoides, Ludwigia palustris,* and *Sagittaria latifolia.* Urban wetlands had a high percentage of obligate wetland species, and most closely resembled emergent wetlands in their vegetation composition. Soil pH and soil electrical conductivity were significantly higher in urban sites, but potential net N-mineralization rates were significantly lower. Urban wetland vegetation and soil characteristics are different than those in nearby natural wetlands, and our increased knowledge of these urban ecosystems will lead to more successful restoration and creation projects.

Key Words: Urban wetland flora; Species richness; Invasive species; Soil electrical conductivity; Floristic Quality Assessment

1. Introduction

Urban wetlands are ecosystems in urban landscapes with high anthropogenic influences, such as high inputs of pollutants and increased presence of exotic species (Ewing 1996; Magee et al. 1999). These wetlands are important ecosystems (Mitsch and Gosselink 2000; Savard et al. 2000) because they may reduce urban flooding (Woodcock et al. 2010), remove pollutants and improve water quality (Gale et al. 1993; Bachand and Horne 2000; Nairn and Mitsch 2000; Harrison et al. 2011), and yet they continue to be threatened and neglected (Hettiarachchi et al. 2015). Thus, wetland restoration in urban areas should become a high priority, just as wetland restoration projects are being implemented across the United States and elsewhere (Middleton 1999; Bakker et al. 2002; Baldwin 2004).

Urban wetlands experience increased runoff and “flashy” hydrology due to the high percentage of impervious surfaces in the surrounding landscape, as well as increased sedimentation (e.g., see Ewing 1996). While numerous studies have examined urban wetland water quality (Ehrenfeld 2000; Malaviya and Singh 2012), soil quality (Ehrenfeld 2000; Lopez and Fennessy 2002; Stander and Ehrenfeld 2009a, 2009b), and hydrologic features (Moscrip and Montgomery 1997; Kaye et al. 2006; Stander and Ehrenfeld 2009a, 2009b), few have examined the plant communities within these systems (but see Doherty and Zedler 2014). It is vital to understand species composition and vegetation structure in urban wetlands to serve as a basis for future wetland restoration and construction efforts.

If urbanization is increasing nutrient inputs and altering the hydrology, we expect urban wetlands to differ in plant composition compared to their counterparts, including changes in species richness and diversity (Zedler 2000, 2005; Chen et al. 2014). Although some argue that species richness will increase in urban areas due to an influx of non-native species (Baldwin 2004; Chu and Molano-Flores 2013), others suggest that species richness will decrease in urban wetlands, potentially as a result of lower water quality and the presence of dominant invasive species (Ehrenfeld 2000). For example, species richness of southeastern Ontario wetlands has been shown to decrease with an increase in the density of nearby paved roads (Findlay and Houlahan 1997). Species richness of urban ponds was also lower than what was expected of pristine ponds in northern England; the authors attributed this pattern to management techniques or other habitat qualities (Noble and Hassall 2015).

Species richness may be lower in urban wetlands as a result of a greater presence of invasive species (Zedler and Kercher 2004). Wetlands surrounded by agriculture and urban land cover were found to have significantly more non-native species than wetlands in undeveloped landscapes (Magee et al. 1999). Nitrate enrichment to wetlands decreased the biomass of native species in prairie potholes in the presence of the invasive graminoid *Phalaris arundinacea*, suggesting that increased nutrient concentrations favor invasive species (Green and Galatowitsch 2002), especially since urban areas may be a source of non-native species (Taylor and Irwin 2004; Qian and Ricklefs 2006). However, the plant composition and structure of urban wetlands in New Jersey was similar to undisturbed sites, suggesting that forested urban wetlands may not universally have a greater presence of exotic species (Ehrenfeld 2005). The relationship between urbanization and the importance of invasive plant species will become clearer as more urban wetland sites are examined.

The goals of this study were to characterize the vegetation of urban wetlands and selected soil parameters in south-central New York. We also aimed to compare these urban wetlands to previously sampled natural wetlands with respect to vegetation and soil characteristics. Our data provided the opportunity to relate species richness to soil traits to test for correlations that will increase the success of urban wetland restoration and rehabilitation projects.

2. Methods

2.1 Study sites and design

Our study took place in the summer of 2011. The urban sites are in the Southern Tier region of south-central New York, found in the northern headwaters of the Chesapeake Bay watershed. We focused on urban wetlands that are in the vicinity of Binghamton in Broome County, NY, which lies within a metropolitan area of ca. 200,000 people. The city is surrounded by suburban residential areas, although most of the county is rural (Vink et al. 2013).

We sampled eight urban wetlands (0.2 ha-6.5 ha) that are surrounded by residential or commercial areas and receive runoff from impervious surfaces (Table 1). Wetlands were chosen based on the presence of potential pollution sources, as well as having a clearly defined inlet and outlet. Despite these common features, a few sites stood out from the group. For example, Site 8 (Cutler Pond) is a wetland bordering a natural kettle hole with open water. Site 4 is a former riverbed that lies adjacent to a controlled access highway. The others show clear human impacts. For example, Site 1 has long been an inundated area, although the site has undergone multiple construction projects to transform the wetland into a stormwater retention pond. Site 2 is heavily managed as a stormwater wetland, with portions that are regularly mowed to ensure that water from the Susquehanna River immediately downstream can backflow into the site.

Hydrology also varied among wetland sites. Site 1 had a small channel as the main inlet, which emptied into a large pool spanning from the middle of the wetland to the outlet. Site 2 is a mosaic of small channels and hummocks with unclear waterflow patterns. Sites 3 and 4 both have a main channel that runs through the wetland, although Site 3 had no standing water during the survey. Site 5 surrounds a deep channel that consistently has flowing water. Sites 6-8 are all wetlands that border standing water. We noted considerable variation in water depths within study sites. For example, we observed abrupt water level rises during storm events in 6 of the 8 wetlands. This suggests that water depth may not be an accurate parameter to broadly characterize urban wetlands.

We compared these urban wetlands to 18 previously sampled natural wetlands (Heintzman et al. unpublished data): seven emergent, five scrub-shrub, and six forested sites. All natural wetlands occurred on state lands and fell within five New York counties: Broome, Chenango, Cortland, Tioga, and Tompkins. Sites were randomly selected using the National Wetland Inventory database and ranged in area from 0.26-2.64 ha. All but 4 sites were located more than 15 km from urban centers with a population of at least 10,000. Vegetation and soil chemistry data were collected for all 26 sites using the same methodology.

2.2 Vegetation

Vegetation sampling locations at each urban wetland site were chosen by randomly selecting transects perpendicular to a baseline bordering one side of each wetland. The number of sampling points varied with site size (urban wetlands: 15-52 sampling points comprised of 35-121 nested plots). At each point, nested plots were used to sample herbaceous cover (1 m^2 quadrats) and shrub cover (10 m^2 quadrats). In each quadrat, percent cover estimates were recorded for each species (Mueller-Dumbois and Ellenberg 1974). A circular 100 m^2 plot was established at every third position to sample trees when present. Species and circumference at breast height were recorded. Taxa were identified to the species level using Gleason and Cronquist (1991), with nomenclature updated according to the NY Flora Atlas (Weldy et al. 2015). Taxa that could not be identified to the species level were identified to the genus level if possible or recorded as an unknown species. We ultimately identified seven species of *Galium*, five species of *Eleocharis*, two species of *Potamogeton*, and four species of *Ranunculus*, although we did not consistently identify them to species in the field. Vegetation data for each wetland were summarized as relative percent cover, defined as the percent cover of a species divided by the total cover of all species in that same wetland.

Plant information was found using the USDA (United States Department of Agriculture, National Resources Conservation Service 2012) plant database for the Northeast region and the NY Flora Atlas (Weldy et al. 2015). All species were assigned a wetland indicator status, from obligate (OBL) to facultative upland (FACU), based on The National Wetland Plant List (Tiner 2005; Lichvar 2014). For our purposes, we equate “invasives” with non-native species, although there is some ambiguity on the status of *Phalaris arundinacea* (Galatowitsch et al. 1999; Weldy et al. 2015). Information regarding invasive taxa was found using the DEC (Department of Environmental Conservation) list of invasive species for New York State. Floristic Quality Assessment Index (FQAI), first developed by Swink and Wilhelm (1979, 1994), was used to estimate the habitat quality of all the sites and was calculated using the FQAI calculator from the Mid-Atlantic Wetlands Work Group (Penn State Riparia Floristic Quality Assessment Calculator 2016). Adjusted FQAI (I') values, which include the presence of invasive species in the index calculation as described by Miller and Wardrop (2006), are an effective tool to assess ecosystem health in urban areas and should be considered in floristic quality assessments (Lopez and Fennessy 2002; Rooney and Rogers 2002; Miller and Wardrop 2006). Comparing I' values may be a valuable tool to quickly assess both natural and urban wetlands, and to determine systems in need of rehabilitation and restoration.

2.3 Soil characteristics

Three soil samples (top 5 cm of the sediment) were collected at each wetland site, approximately marking the main inlet, middle of the wetland, and outlet. Samples were immediately transported back to the lab, stored in a cold room (5 °C), and processed within 24 h using standard methods (Zhu and Ehrenfeld 1999). Soil was sieved to remove roots and large organic debris, such as leaves and twigs. Soil characteristics included pH, electrical conductivity, soil organic matter (SOM), and extractable inorganic nitrogen (N). Soil pH and electrical conductivity were measured using a 1:4 soil (g) to water (mL) slurry. Soil organic matter was determined from samples dried at 105 °C as loss on ignition after ashing in a 550 °C muffle furnace. Inorganic nitrogen was extracted from 20 g fresh soil samples using 50 mL 1 M KCl. Samples were shaken using a reciprocating shaker for an hour, then allowed to settle overnight in cold storage. Settled samples were gravity-filtered through Whatman #40 ashless filter papers. Filtrate was acidified with 0.2 mL 6 M HCl and placed in cold storage until analysis. We also incubated soil samples (20 g fresh soil) for 28 days to estimate the net nitrification and net N mineralization rates under dark conditions and 22 °C (standard lab conditions), followed by the same extraction method described above. Ammonium nitrogen (NH_4-N) and nitrate nitrogen (NO_3-N) concentrations for the original and post incubation extractions were determined using a Lachat QuickChem Flow-Injection Autoanalyzer 8000 series and then expressed as mg N kg^{-1} dry soil. The method for ammonium analysis is based on the Berthelot reaction (Lachat QuikChem Method: 10-107-06-1-C) and the method for nitrate analysis uses a copperized cadmium column to reduce nitrate to nitrite (Lachat QuikChem Method: 10-107-04-1-C). Net nitrification rates were then calculated based on the changes in nitrate concentrations over the 28-day incubation period, and expressed as mg NO_3 -N kg^{-1} dry soil day^{-1}. Net mineralization rates were calculated as the sum of the change of ammonium and nitrate concentrations over the 28-day incubation period and expressed as mg N kg^{-1} dry soil day^{-1}.

2.4 Statistical analysis

Data were summarized in Excel, with mean pH based on hydrogen ion concentrations, and analyzed using either SPSS or SAS Proc GLM. To compare vegetation and biogeochemistry among wetland categories, data were analyzed using a single-factor Analysis of Variance (unbalanced, one-way ANOVA). Significant results from the ANOVA tests were further analyzed with Tukey's HSD test to determine which groups were different from each other with a $p<0.05$. The departure from normality for soil electrical conductivity was high so we used a non-parametric Kruskal-Wallis H test of significance to assess the differences among wetland habitats (Kruskal and Wallis 1952). We employed a Non-metric Multidimensional Scaling (NMS) ordination in PC-ORD to portray the differences in species composition among the wetland habitat types, using presence/absence data (McCune and Mefford 1999). For the NMS ordination, autopilot mode was used with the Sørensen distance measure, 0.0005 stability criterion, random starting configurations, and a maximum of 500 iterations. The NMS ordination utilized 10 runs with real data and 50 runs with randomized data. The best solution was selected based on the following: a $p<0.05$ for the Monte Carlo test comparing stress for the real data to a randomized data set, and final solutions with stress <20. Linear regressions were used to test for correlations between species richness and all soil parameters.

3. Results

3.1 Vegetation

We distinguished 135 species in the urban wetland survey. Nineteen herbaceous taxa and one shrub species were most important based on relative percent cover (Table 2). Two non-invasive species were common in urban wetlands: rice cutgrass (*Leersia oryzoides),* which was found in seven sites, and water purslane (*Ludwigia palustris),* which occurred in six. *Cornus sericea* appeared in three of the urban wetlands. *Sagittaria* *latifolia* occurred in four sites and was the fifth most abundant species in Site 1. *Carex stricta* was the second most abundant species in Site 2, but was absent from all other urban wetland sites. Site 8 was dominated by two non-invasive species that were only found in this wetland, *Decodon verticillatus* (68.1% relative percent cover) and *Nuphar variegata* (5.8% relative percent cover). We recorded eight shrub species and a single tree (*Fraxinus pennsylvanica*) for the eight sites.

Urban wetland flora included invasive species, such as reed canary grass (*Phalaris arundinacea),* cattail (*Typha* x *glauca)*, and purple loosestrife (*Lythrum salicaria)*. *Phalaris arundinacea* was one of the top three dominant taxa, based on relative percent cover, in five out of the eight urban sites, but not present at the other three sites. *Typha* x *glauca* was also dominant in five urban wetlands, and present in all urban sites but one. *Lythrum salicaria* was dominant in three urban wetlands and found in six sites. *Typha* x *glauca* and *Lythrum salicaria* were absent in the 18 natural wetlands (Heintzman et al. unpublished data). *Phragmites australis* was also present in one urban site (Site 7), but not in the natural wetlands. As a result, urban wetlands had a substantially higher relative percent cover of invasive species than native wetland categories (urban wetland average, 25.5%; natural wetland average, 11.7%).

Species richness was significantly lower in urban wetlands than in natural wetland categories (Fig. 1; ANOVA, $F_{3,22} = 6.37$, $p = 0.003$). Urban wetlands had a mean of 31.8 species ($n = 8$), while natural wetlands averaged 55.8 ($n = 18$). As a consequence of both low species richness and a high presence of invasive species, the adjusted FQAI (I') of urban wetlands was significantly lower (Fig. 1; ANOVA, $F_{3,22} = 6.10$, $p = 0.004$). The average I' of urban wetlands was 27.7 ($n = 8$), while that for natural wetlands was 39.4 (*n*

= 18). We found the same significant trend with traditional FQAI values, but actual values were 53-63% lower than I' values.

Further analysis revealed that urban wetlands had different plant communities than the natural wetland categories with respect to wetland indicator species (Fig. 2). Forested wetlands had a significantly lower proportion of obligate wetland species (ANOVA, $F_{3,22} = 6.96$, $p = 0.002$, Tukey HSD) and a significantly higher proportion of facultative upland species (ANOVA, $F_{3,22} = 3.84$, $p = 0.024$, Tukey HSD) than urban wetlands. Proportions of facultative wetland and facultative species did not differ among any of the wetland habitats.

Based on the results presented above, we found that the plant communities of urban wetlands were clustered separately from natural wetlands (Fig. 3, $r = 0.135$ for Axes 1 and 3). The Non-metric Multidimensional Scaling ordination revealed that the plant communities of emergent, scrub-shrub, and forested communities overlap, whereas there is a distinct cluster of urban wetlands. The NMS ordination concluded that a 3-dimensional solution is the best fit for species presence/absence data, with a final stress of 12.12, final instability of 0.00036, and 239 iterations. Axes 1, 2, and 3 explained 78.9% of the variation among the 26 wetlands, with Axis 3 accounting for 50.0% of the variation.

3.2 Soil characteristics

Analysis of soil characteristics revealed that urban wetlands had significantly higher soil electrical conductivity than the natural wetland categories (Table 3; urban wetland median = 150 $\mu S\ cm^{-1}$, natural wetland median = 33 $\mu S\ cm^{-1}$; Kruskal-Wallis H

= 14.6, p = 0.002). Urban wetland soil electrical conductivity ranged from 123 to 6380 $\mu S\ cm^{-1}$, while the range for natural wetlands was 23-243 $\mu S\ cm^{-1}$. Soil pH was significantly higher in urban wetlands (mean = 6.9) compared to natural wetland categories (Table 3; means 4.8-5.7 for natural wetland categories; $p < 0.001$, Tukey HSD). There were no significant differences in SOM among the wetland habitats, perhaps because of the high variation in SOM values (urban range: 7.2-28.4%, natural range: 4.6-64.1%).

We found that the concentrations of extractable inorganic nitrogen were not significantly different among wetland habitat types (Table 3). Extractable NH_4-N ranged from 4.9-27.5 mg NH_4-N kg^{-1} for urban wetlands, and the range for natural wetlands was 0.7-128.6 mg NH_4-N kg^{-1}. Extractable NO_3-N was low in urban wetlands, with a range of 0.1-0.5 mg NO_3-N kg^{-1}. The extractable NO_3-N concentrations were more variable for the natural wetlands, with a range of 0.1-20.1 mg NO_3-N kg^{-1}. Potential net nitrification rates were also not significantly different among habitat types (study range: 0.0-2.0 mg NO_3-N $kg^{-1}day^{-1}$). However, urban wetlands had significantly lower potential net N-mineralization rates than the natural wetlands with a range of -0.7-1.8 mg N $kg^{-1}day^{-1}$ (Table 3; $p < 0.001$, Tukey HSD). Urban wetlands had a mean net N-mineralization rate of -0.2 mg N $kg^{-1}day^{-1}$ (range: -0.7-0.1 mg N $kg^{-1}day^{-1}$), and the corresponding value for natural wetlands was 0.7 mg N $kg^{-1}day^{-1}$ (range: -0.2-1.8 mg N $kg^{-1}day^{-1}$).

3.3 Post-hoc species richness comparisons with soil traits

On the basis of linear regressions, species richness was negatively correlated with soil pH (r = -0.48, p = 0.014) and soil electrical conductivity (r = -0.49, p = 0.014), but positively correlated with potential net N-mineralization rate ($r = 0.49$, $p = 0.011$). Correlations between species richness and the other four soil parameters were non-significant.

4. Discussion

Our urban wetlands had a lower species richness and a greater presence of invasive species compared to natural wetlands, which is similar to the findings of other studies (Ehrenfeld 2000; Zedler and Kercher 2004; Noble and Hassall 2015). Urban sites had a vegetation structure similar to that of natural emergent wetlands, specifically as a result of a high presence of obligate wetland species. This may be a consequence of the similarities in hydrology between emergent sites and urban sites. We observed standing water in many of the urban wetlands, as seen in natural emergent wetlands and in contrast to forested and scrub-shrub wetlands. While urban wetland vegetation in our area reflects some features of natural emergent wetlands, swamps (Zhu and Ehrenfeld 1999; Ehrenfeld 2005), wet meadows (Magee et al. 1999), and ponds (Noble and Hassall 2015) can all be found in urban ecosystems. Understanding more about the hydrology of urban wetlands, specifically focusing on the relationship between water depth and plant communities, may provide further insight into the plant community structures and the ecosystem functions of these habitats.

Our study also provides insight into the variation of urban wetland vegetation. While we can certainly describe trends in the plant communities, we found that sites vary in their species composition. Site 8 was dominated by non-invasive species (*Decodon verticillatus* and *Nuphar variegata*) that were not observed in any other urban wetland. Interestingly, this is also the only site that has yet to be invaded by *Typha* species. Site 8 is always inundated, and was certainly wetter than any of the other urban wetlands in this survey. We suspect that the hydrology of this wetland has resulted in a distinctive assemblage of plant species. Moreover, Site 8 serves as an example that not all urban wetlands are dominated by invasive species.

We found that *Carex stricta* occurred in only one urban wetland. This native sedge species was only found in the mowed sections of Site 2. It appeared that the mowing kept *Typha* x *glauca* from spreading into the area, thus allowing *Carex stricta* to maintain itself. Our results are supported by Hall and Zedler (2010), who found that native *Carex* spp. were able to expand vegetatively once *Typha* x *glauca* rhizomes were removed.

The presence of invasive species and their influence on native plant populations may have important implications for urban wetland management. *Typha* x *glauca* may tolerate the frequent flooding of an urban wetland, in contrast to *Carex* spp. (Hall and Zedler 2010), potentially giving *Typha* x *glauca* a competitive advantage (Wilcox et al. 1985; Wilcox et al. 2008). High nutrient levels generally increase plant biomass, and invasive species may outcompete native species under these circumstances. For example, the biomass of the native *Typha latifolia* and *Carex stricta* decreased when grown with *Phalaris arundinacea*, possibly because of *P. arundinacea*'s rapid growth rate and canopy cover (Wetzel and van der Valk 1998). Given that many of the urban wetlands are dominated by invasive species, future work should focus on identifying variables that may influence non-invasive plant growth and success in urban wetlands, including soil quality, water quality, and hydrology.

Despite the fact that most of the urban wetlands were dominated by invasive species, we were surprised that so many species (135) occurred in urban wetlands. Many of these species were non-invasive, including dominant species like *Carex stricta, Leersia oryzoides, Sagittaria latifolia,* and *Sparganium americanum.* These species can clearly tolerate conditions in at least some urban wetlands, and future urban restoration/construction projects should consider including planting or seeding of such species in their project plans.

Our soil chemistry data may indicate that urban wetlands are receiving a substantial amount of pollutants, as reflected in high electrical conductivity and higher pH levels. Soil organic matter was highly variable and did not differ significantly across all 26 sites, further reflecting the variation of soil traits among these wetlands. We were surprised that urban wetlands had low concentrations of extractable inorganic nitrogen (NH_4^+ and NO_3^-), as well as low potential net nitrification and net N-mineralization rates, although rates this low have been previously reported (Stander and Ehrenfeld 2009a, 2009b). Others have found net nitrification and net N-mineralization rates to be higher than what we found in our urban settings (Zhu and Ehrenfeld 1999). Considering that we found no significant difference in soil organic matter among wetlands, it is unclear why urban wetlands have significantly lower potential net N-mineralization rates than natural wetlands. However, these rates can vary over the course of the growing season, and so more data are needed to adequately describe spatial and temporal variation of soil characteristics of both urban and natural wetlands.

Further analysis revealed that species richness was negatively correlated with soil electrical conductivity and pH. This may be a consequence of plant intolerance to pollutants in the soil. Municipalities in northeastern United States often combat ice and snow on roadways by applying liberal amounts of road salt, and accumulation of road salt may be one reason that we see an increase in soil electrical conductivity in urban wetlands. Higher salt concentrations may reduce species richness (Richburg et al. 2001). Roadway contaminants may enter wetland systems and alter the pH of surrounding soils (Angold 1997); we believe that the higher pH in the urban wetlands may reflect the presence of roadside pollutants and that these pollutants could reduce species richness. It is unclear why species richness is correlated with an increase in potential N-mineralization rates or why our potential N-mineralization rates are so low.

The floristic quality assessment index has been recommended for management assessment and monitoring programs (Miller and Wardrop 2006). Although there are some criticisms regarding the use of FQAI and other biological index assessment tools (Green 1979), Lopez and Fennessy (2002) found that FQAI was negatively correlated with disturbance, which included sites that were located in urban regions. Adjusted FQAI (I') values, as described here, were highly correlated with anthropogenic disturbance (Miller and Wardrop 2006). Our urban I' values are similar to other heavily disturbed sites (Lopez and Fennessy 2002; Miller and Wardrop 2006; Wilson et al. 2013). Adjusted FQAI values (I') may not always best represent the habitat quality of sites, so DeBerry and Perry (2015) cautioned managers to look at both FQAI and I' before creating management plans; however, our I' data showed the same pattern as FQAI values. Based on our results, FQAI values may be a useful tool to define reference sites in an area, as well as to determine sites in need of rehabilitation or restoration.

5. Conclusion

As urban areas expand globally, human populations will increasingly rely on these ecosystems. We found that many non-invasive species can be found in urban wetlands, and that these wetland sites are highly variable in their plant composition and soil characteristics. It is important for managers to view urban wetlands differently than natural wetlands, especially in terms of plant communities. Existing urban wetlands may serve as a guide for future urban restoration or creation projects, and these wetlands and their plant communities could provide valuable information to create high diversity ecosystems within urban areas.

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Table 1 Urban wetland site information, including latitude and longitude, area, and known sources of runoff.

Site	Lat., Long.	Area (ha)	Sources of runoff
1	42.088, -75.962	0.2	Impervious surfaces on Binghamton University campus
2	42.122, -75.982	2.0	Residential area
3	42.110, -76.010	0.7	Highways and a parking lot
4	42.135, -75.904	1.8	Highway, high traffic main road, parking lots
5	42.099, -76.003	0.6	Residential area, shopping plaza
6	42.100, -75.834	0.6	High traffic roads in an industrial complex
7	42.100, -75.837	1.0	High traffic roads in an industrial complex
8	42.128, -75.909	6.5	Residential area and parking lots

Table 2 Plant taxa and relative percent cover (%) in the eight urban wetlands. These were the species found to represent at least 5% of the vegetation in at least one wetland site. A dash indicates that the species was not seen at that site.

Species	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8
Carex stricta Lam.	-	27.4	-	-	-	-	-	-
Cornus sericea L.	1.5	-	-	-	0.5	9.4	-	-
Decodon verticillatus (L.) Ell.	-	-	-	-	-	-	-	68.1
Dipsacus fullonum L.	0.3	-	5.0	-	-	-	7.3	-
Eleocharis sp.	-	-	7.3	-	-	-	-	-
Galium spp.	-	-	-	5.0	-	-	-	-
Glechoma hederacea L.	-	-	-	12.1	-	-	-	-
Leersia oryzoides (L.) Sw.	13.6	0.1	0.1	2.8	32.6	0.7	0.1	-
Lythrum salicaria L.	-	-	3.2	26.9	3.9	9.6	2.6	2.7
Myosotis scorpioides L.	18.0	-	-	2.8	-	-	-	-
Nuphar variegata Engelm. ex Durand	-	-	-	-	-	-	-	5.8
Phalaris arundinacea L.	-	42.6	-	8.9	-	26.7	19.3	5.2
Phragmites australis (Cav.) Trin. ex Steud.	-	-	-	-	-	-	6.5	-
Potamogeton sp.	10.0	-	-	-	-	-	-	-
Ranunculus sp.	8.2	-	-	-	-	-	-	-
Sagittaria latifolia Willd.	7.9	-	-	0.3	2.1	-	1.5	-
Solidago rugosa Mill.	-	-	-	-	2.3	9.1	2.4	-
Sparganium americanum Nutt.	-	-	-	-	24.2	-	-	5.1
Typha x *glauca* Godr.	17.7	19.7	67.6	6.0	1.0	11.7	31.9	-
Poaceae	-	-	-	-	-	-	13.7	-

Table 3 Soil characteristics for each habitat type expressed as median values with ranges in parentheses. All extractable –N data and potential net N- rates are for dry soil. F and H values refer to means for all data sets. Note: ** indicates $p < 0.01$, *** indicates $p < 0.001$, NS indicates no significant difference among wetland habitat types; all statistics reflect df = 3, 22. Superscripts are not displayed for nonsignificant results.

	Urban	Emergent	Scrub-shrub	Forested	F-ratio
Weighted pH	6.8^a (6.3-7.5)	4.8^b (4.4-6.4)	5.4^b (4.6-5.9)	5.7^b (5.3-6.6)	13.11***
Conductivity ($\mu S\ cm^{-1}$)	150^a (123-6380)	35^b (26-61)	26^b (23-243)	33^b (25-55)	H=14.6**
SOM (%)	10.2 (7.2-28.4)	12.3 (4.6-17.2)	10.4 (5.1-64.1)	15.2 (6.4-48.7)	NS
Extractable NH_4-N (mg NH_4-N kg^{-1})	13.5 (4.9-27.5)	25.0 (1.9-128.6)	8.0 (2.9-12.8)	2.3 (0.7-17.5)	NS
Extractable NO_3-N (mg NO_3-N kg^{-1})	0.2 (0.1-0.5)	0.6 (0.1-20.1)	3.5 (0.4-7.3)	8.5 (1.4-13.3)	NS
Net Nitrification (mg NO_3 -N kg^{-1} day^{-1})	0.1 (0.0-0.3)	0.7 (0.4-1.2)	0.6 (0.2-1.1)	0.3 (0.0-2.0)	NS
Net N-Mineralization (mg N kg^{-1} day^{-1})	-0.2^a (-0.7- 0.1)	0.6^b (0.1-0.9)	0.6^b (-0.2-1.8)	0.8^b (0.1-1.5)	7.32***

Fig. 1 Mean species richness and mean adjusted FQAI (I') for each wetland habitat type (Urban (n = 8), Emergent (n = 7), Scrub-shrub (n = 5), and Forested (n = 6)), ± 1 SE. Means not sharing a common letter as a result of one-way ANOVA tests differ significantly at p = 0.05 according to Tukey means comparison.

Fig. 2 Mean proportions of USDA wetland indicator categories for each wetland habitat type, ± 1 SE (OBL = obligate wetland, FACW = facultative wetland, FAC = Facultative, FACU = Facultative upland). Means not sharing a common letter as a result of one-way ANOVA tests differ significantly at p = 0.05 according to the Tukey means comparison.

Fig. 3: NMS ordination depicting the similarity among wetland sites (n = 26) based on species composition (presence/absence). U = Urban, E = Emergent, S = Scrub-shrub, and F = Forested.

Fig. 1

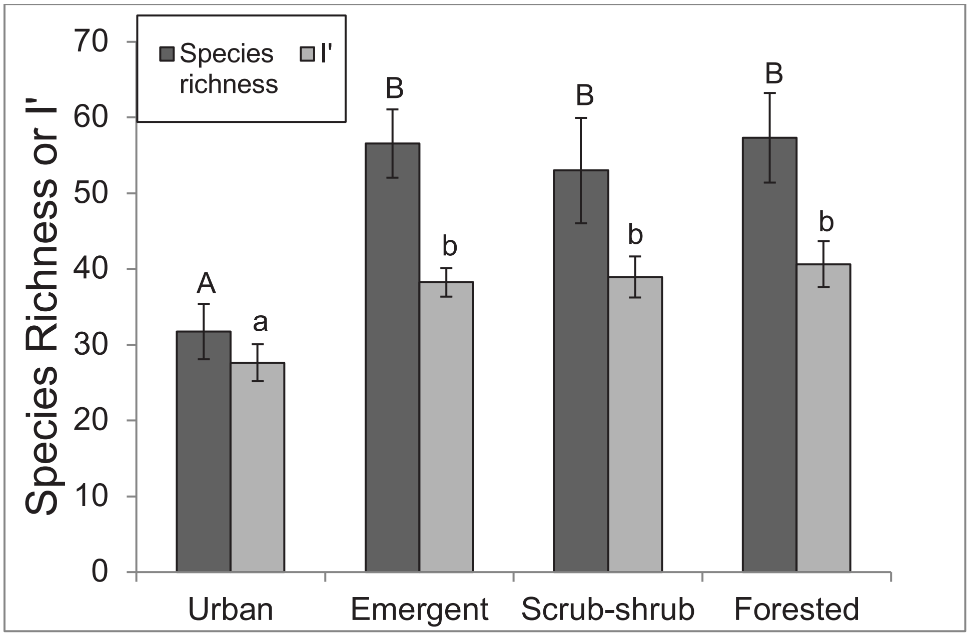

Species richness
I'
Species Richness or I'
70
60
50
40
30
20
10
0
A
a
B
b
B
b
B
b
Urban
Emergent
Scrub-shrub
Forested

Fig. 2

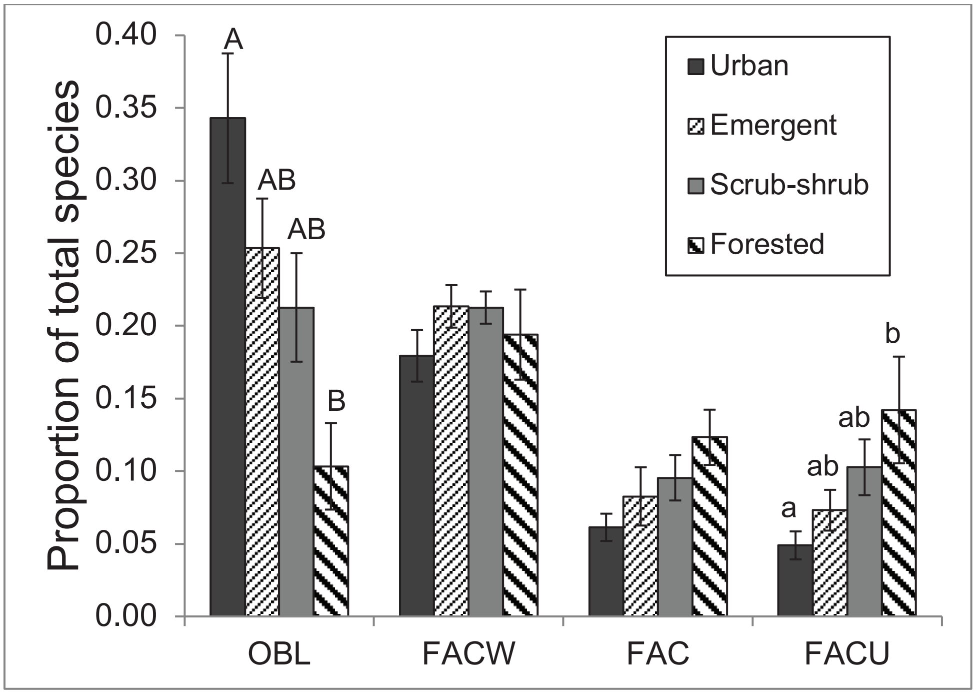
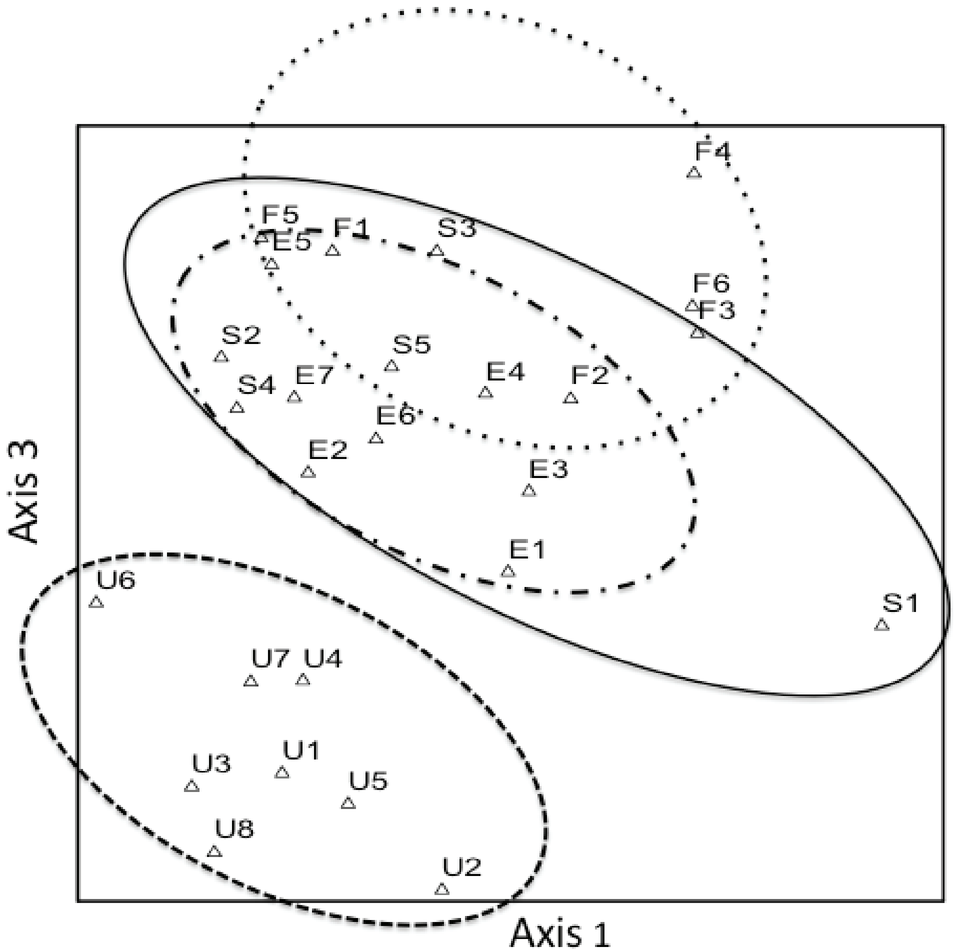

Proportion of total species
0.40
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
A
AB
AB
B
b
ab
ab
a
OBL
FACW
FAC
FACU
Urban
Emergent
Scrub-shrub
Forested

Fig. 3


F4
F5
E5
F1
S3
F6
F3
S2
S5
E4
F2
S4
E7
E6
E2
E3
E1
U6
S1
U7
U4
U3
U1
U5
U8
U2
Axis 3
Axis 1

Chapter 3: Urban wetland seed bank profiles in south-central New York State

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Abstract.

Wetlands are important habitats in urban landscapes that reduce flooding and improve water quality, yet urban wetland seed banks are rarely studied. Our main objective was to profile urban wetland seed banks in south-central New York. We exposed sediment cores from four wetlands in Broome County, New York, to flooded and drawdown treatments for 16 mo, and recorded community composition and seedling density. We found high spatial variation in species richness and seedling density among the four sites. Species richness ranged from 28 to 56 species, with Sample Based Extrapolation (S_{est}), Jackknife1, and Chao1 analyses estimating similar expected species richness values (S_{est} projected 37.9 – 77.0 species, Jackknife1 analysis estimated 40.5 – 77.8 species, and Chao1 projected 32.2 – 79.1 species). Mean seedling density ranged from 3,367 seedlings/m^2 to 19,132 seedlings/m^2. These seed banks were dominated by obligate wetland species (75.8 – 93.3%). Invasive species comprised a high percentage of seedlings for three wetlands (40.8 – 80.9%), but not for the fourth site (4.2%). *Lythrum salicaria*, *Typha* sp., and *Ludwigia palustris* were common species based on relative seedling density for three seed banks, while *Leersia oryzoides, Schoenoplectus tabernaemontani,* and *Alisma triviale* were common species in the fourth site. Similarity indices between the standing vegetation and their respective seed banks, based on presence/absence data, were low (13 – 34%). Species richness and seedling densities were within the ranges of natural wetland seed bank studies.

Key words: Urbanization; Invasive species; *Lythrum salicaria; Typha* x *glauca;* Species richness

Introduction.

Urban wetlands are important ecosystems, reducing urban flooding (Woodcock, Monaghan, and Alexander 2010), removing pollutants and improving water quality (Gale, Reddy, and Graetz 1993; Bachand and Horne 2000; Nairn and Mitsch 2000; Harrison *et al.* 2011). Compared to natural wetlands, these ecosystems are influenced by increased sedimentation and higher levels of nitrogen, phosphorus, metals, and salt from runoff of impervious surfaces (Findlay and Houlahan 1997; Ehrenfeld 2000; Larson *et al.* 2016). Urban wetlands are also characterized by having lower species richness and a greater presence of invasive species (Galatowitsch, Anderson, and Ascher 1999; Ehrenfeld 2000; Zedler and Kercher 2004; Noble and Hassall 2015; Larson *et al.* 2016). We expect to see impacts in urban wetland vegetation because of these anthropogenic influences.

Little is known about the impact of urbanization on wetland seed banks. Seed banks are an integral component of the plant community as a potential source for standing vegetation, and provide insight into what may naturally germinate and establish in the field (Warr, Thompson, and Kent 1993; DeBerry and Perry 2000; Hopfensperger 2007). Altered environmental factors, as a result of urbanization, may limit seedling establishment. For example, increased sedimentation can alter both standing vegetation and seed banks (van der Valk, Swanson, and Nuss 1983; Lee *et al.* 2014; Wang *et al.* 2014). Increased total nitrogen concentrations in the sediment of a polluted urban riverbed were correlated with a decline of seed bank species richness and diversity (Cui *et al*. 2013). Elevated salt concentrations from road salt may influence seedling establishment; for example, Miklovic and Galatowitsch (2005) found that species richness, species diversity, and the total aboveground biomass of emerged seedlings decreased with increased concentrations of NaCl.

The presence of invasive species in the standing vegetation may also impact urban wetland seed banks, altering both species richness and species diversity (Miklovic and Galatowitsch 2005; Yakimowski, Hager, and Eckert 2005; Hager *et al.* 2015). Wetland invaders may dominate urban wetland seed banks due to their reproductive strategy; for example, a single *Lythrum salicaria* L. plant can produce thousands of viable seeds, and form large seed banks (Welling and Becker 1990; Ågren 1996). Road corridors may provide efficient dispersal routes for invasive species to enter urban wetlands (Zedler and Kercher 2004). *Typha angustifolia* L. and *Phragmites australis* (Cav.) Trin. ex Steud., for example, are commonly found along roadsides and may benefit from vehicle dispersal and construction disturbances, as well as reduced competition from salt-intolerant native species (Galatowitsch, Anderson, and Ascher 1999; Zedler and Kercher 2004). These changes to seed banks can, in turn, lead to changes in the standing vegetation (Leck and Leck 2005; Frieswyk and Zedler 2006; Wilcox 2012).

Collectively, these conditions may impair germination and establishment of seedlings in urban wetlands. We were therefore interested in whether urban wetland seed banks in upstate New York contained viable seeds, and whether these soils could support the germination and establishment of seedlings. Seed banks are often heterogeneous, varying at the landscape level as a consequence of varying habitats, plant communities, and hydrology (Parker and Leck 1985; Middleton 2000; Peterson and Baldwin 2004; Liu *et al*. 2006; James *et al*. 2007). Seed banks also vary at the site level, creating a heterogeneous patchwork of species (Brock, Theodore, and O'Donnell 1994; Bonis, Lepart, and Grillas 1995; Blood and Titus 2010). Thus, we examined the viability of urban wetland seed banks by comparing seedling density and species composition both within and among urban wetlands.

The main goals of this research were to evaluate the profiles of seed banks in the vicinity of Binghamton, New York, including species richness, dominant taxa, relative importance of invasive and native species, and the dominant wetland indicator status of four urban wetlands. Additionally, we compared the species assemblage of seed banks to their respective standing vegetation to discuss potential plant community dynamics in these urban wetlands.

Materials and Methods.

SITE SELECTION AND DESCRIPTION.

The study took place in the Binghamton metropolitan area, located in Broome County, New York. The county, although largely rural, includes suburban residential areas that surround the small city of Binghamton. The city has a population of approximately 47,000, and there are about 320,000 residents within a 48-km range of the city (U.S. Census Bureau 2010). Although Binghamton is considered a small city, the United Nations estimates that half of the world's urban residents live in cities with less than 500,000 people (United Nations 2014); thus, Binghamton is an appropriate location to study urban ecology.

We chose four sites based on a survey completed in 2011: Sites 1, 4, 6, and 7 of Larson *et al*. (2016). The urban wetlands had a high percentage of invasive species, and a lower species richness than natural wetlands in the area. The vegetation of these urban wetlands was dominated by obligate wetland species and most closely resembled emergent wetland plant communities, with few woody species present. These wetlands are all located in areas with high impervious surface cover, including roads and highways, sidewalks, parking lots, and buildings.

The four wetlands were chosen from the original survey due to their impacted soils, accessibility, and site history. The mean soil electrical conductivity of these urban wetlands (range: 145 – 6,380 $\mu S\ cm^{-1}$) was significantly higher than natural wetlands (Larson *et al*. 2016). All sites receive runoff from impervious surfaces in high traffic areas and have clearly defined inlets and outlets. There was little variation in the distance from the edge of each wetland to the nearest impervious surface (5 – 30 m), and distance to the nearest road (5 – 150 m). Wetlands ranged in area from 0.2 ha to 1.8 ha.

Site 1 is of interest due to its history of disturbance. This wetland, located on the Binghamton University campus in Vestal, New York, underwent a major reconstruction in 2011. The area of the retention wetland was doubled to accommodate increased runoff on campus. The reconstruction did not include seeding or successful supplemental planting. Site 1 is also further from major roadways than the other three wetlands: Sites 4, 6, and 7 are within 20 m of highways and heavily trafficked roads. Site 4 is a former riverbed between Front St. and Interstate 81 in Binghamton, New York. Sites 6 and 7 may be hydrologically connected (42.099° N, -75.836° W); they are separated by Conklin Kirkwood Road in Kirkwood, New York. Site 6 is adjacent to Industrial Park Drive and directly across the street from a truck wash. Site 7 is bordered by Colesville Road and US Highway 11.

SEED BANK COLLECTION.

We collected sediment cores from all four sites in early May 2014, before seedling germination in the field. Twelve sediment cores (15.2 cm diameter, 5 cm depth) were collected from each wetland, two from each of six plots. The total volume of sediment collected from each plot was approximately 912 cm^3. Sampling locations at each site were selected along transects perpendicular to a baseline bordering one side of each wetland. Locations were determined using a stratified random approach, with distances between transects varying depending on the size of the wetland. Only three cores were collected from plots with standing water, one at Site 1 and two at Site 7. We did not collect cores from areas with deep channels or open water. Sediment was stored in a cold room at 4.4 °C for less than 2 wk before the experiment was initiated.

EXPERIMENTAL SET-UP FOR SEED BANKS.

Large debris was removed from each sediment core. The two samples from each plot were then homogenized and divided into two treatments, flooded and drawdown, to account for seedlings that may only germinate and grow in flooded or drawdown conditions (van der Valk and Davis, 1978). Water levels in the flooded treatment were 5.5 cm above the sediment surface, whereas drawdown treatments experienced water that was 5 cm below the sediment surface. The sediment, approximately 1 cm thick, was evenly spread on top of sterilized play sand in plastic germination trays (20.6 cm x 10.2 cm x 2.4 cm). These trays were randomly assigned to positions in eight 1,200-L fiberglass tanks filled with reverse osmosis water in the Binghamton University Research Greenhouse. Each of these tanks contained sediment from one wetland. Tanks were exposed to natural light for the duration of the studies. Water temperatures were maintained at 23 °C by refrigerated circulators (CFF-500, Remcor, Franklin Park, IL). We collected data on seedling density and community composition from June 2014 until September 2015.

STANDING VEGETATION SAMPLING.

To compare the composition of the seed banks with their corresponding vegetation, we focused on the herbaceous plant data because only one woody seedling emerged from the 48 study cores. All standing vegetation data were collected in July 2014. Vegetation sampling locations at each urban wetland site were chosen by randomly selecting transects, within intervals, perpendicular to a baseline bordering one side of each wetland. The number of sampling locations varied with site size. We recorded vegetation data from 44 sampling plots located on 11 transects in Site 1, 32 plots along 10 transects in Site 4, 35 sampling plots on 10 transects in Site 6, and 35 sampling plots along 10 transects in Site 7. Estimates of the percentage of cover were recorded for each herbaceous species within 1-m^2 quadrats, to the nearest 5% (Mueller-Dombois and Ellenberg 1974).

SPECIES IDENTIFICATION.

Taxa were identified to the species level using Gleason and Cronquist (1991), with nomenclature updated according to the *New York Flora Atlas* (Weldy, Werier, and Nelson 2017). Taxa that could not be identified to the species level were identified to the genus or family level if possible, or recorded as unknown species. Seedlings that could not be identified in the trays were excavated and planted in separate pots until they could be identified. We identified 93.3% of all seedlings; 80.9% of all seedlings were identified to species and 12.4% to the genus level.

The native status and wetland indicator status of each species were found using the United States Department of Agriculture plant database for the Northeast region (USDA NRCS 2012) and the *New York Flora Atlas* (Weldy, Werier, and Nelson 2017). For our purposes, we equate "invasives" with nonnative species, although there is some ambiguity on the status of *Phalaris arundinacea* L. (Galatowitsch, Anderson, and Nelson 1999; Weldy, Werier, and Nelson 2017). All cattails in the standing vegetation were identified as the invasive *Typha* x *glauca* Godr. as a result of high variability in the gap size between male and female flowers on the inflorescence, as well as leaf width (see Selbo and Snow 2004). *Typha* seedlings were identified as *Typha* sp. because these morphological traits did not exist in seedlings.

DATA ANALYSES.

We compared the seed banks among wetlands by calculating seedling density, defined as the number of seedlings per square meter, based on the wetland surface area collected. A one-way ANOVA was conducted for mean seedling density, using SAS Proc GLM (version 9.4, SAS, Cary, NC). Significant results from the ANOVA tests were further analyzed with Tukey's HSD test to determine which groups were different from each other with a $P < 0.05$. We measured species importance as relative seedling density, expressed as the number of seedlings for a species divided by the total number of seedlings in that same wetland. This was calculated separately for drawdown and flooded treatments to avoid bias due to the inability of some species to germinate in both treatments.

A common problem in ecological studies is adequately sampling an area to capture all, or most, of the ecosystem's species (Hubbell 2001; Moro, de Sousa, and

Matias 2014). We used nonparametric statistical estimators, based on seed bank abundance data, to estimate the total species richness for all four seed banks (Colwell 2013). We report sample-based extrapolation (S_{est}), with extrapolation from 6 to 12 samples, to evaluate if we adequately sampled the wetland. We also report estimated species richness values for each seed bank using JackKnife1 and Chao1 extrapolations, calculated using EstimateS (Gotelli and Colwell 2011; Colwell *et al*. 2012). These species richness estimations were calculated using abundance data and 1,000 runs. We used Classic Chao1 instead of the biased-Corrected option because our CV values were > 0.5 for all sites.

We assessed the variation within wetlands by comparing species richness and seedling density at the plot level. Again, seedling density was calculated separately for drawdown and flooded treatments. Species for each seed bank were considered "common" if the relative seedling densities were greater than 5.0% in either treatment.

We used Sørenson's similarity index to compare the seed bank of one site to the seed bank of each other site based on relative seedling density, and each seed bank to its respective standing vegetation based on presence/absence data (Sørenson 1948). For these calculations, we included the presence of shrubs and trees in the standing vegetation. Vegetation data for each wetland were summarized as relative cover, defined as the percent cover of a species divided by the total cover of all species in that same wetland.

Results.

SEED BANK VARIATION AMONG SITES.

We observed a total of 10,941 seedlings and 93 distinct species. Total observed species richness ranged from 28 to 56 species per wetland site (Table 1). We found that S_{est} estimated similar total species richness values as the Jackknife1 and Chao1 analyses; S_{est} projected 37.9 – 77.0 species for 12 samples, the Jackknife1 analysis estimated 40.5 – 77.8 species, and Chao1 species richness projected from 32.2 – 79.1 species (Table 1). In every extrapolation, Site 7 had the lowest species richness of all four sites. Site 4 had the highest species richness for S_{est} and Jackknife1 analyses, but Site 6 had the highest species richness for the Chao1 extrapolation (Fig. 1).

Mean seedling density ranged from 3,367 seedlings/m^2 – 19,132 seedlings/m^2 (Table 1). Seedling density was significantly higher in Sites 4 and 6 than in Site 1 ($F_{3,20}$ = 4.15, P = 0.019, Tukey HSD). All four seed banks were dominated by obligate wetland species (Fig. 2, range: 75.8 – 93.3%). In each site and treatment combination, the three most common taxa cumulatively comprised over 65% of observed seedlings (Table 2; 67.8 – 81.5% for the drawdown treatment; 70.4 – 96.2% for the flooded treatment).

Invasive species abundance was substantially lower in Site 1 (Fig 2, 4.2% based on relative seedling density) than in the other three wetlands (40.8 – 80.9%). *Lythrum salicaria* was a common species in every wetland except for Site 1 (Table 2). In fact, only two *L. salicaria* seedlings emerged from Site 1. *Phalaris arundinacea* was also found in all four seed banks, although only one seedling was observed from Site 1. *Typha* sp. was found in all four wetland seed banks, and was common in Sites 4 and 7, as well as in the flooded treatment for Site 6 (Table 2). *Typha* sp. was not common, however, in Site 1 (seven seedlings total). We also observed species that are not normally considered wetland species, most frequently *Plantago major* L.

Native species were also common in our urban wetland seed banks. *Juncus* was an important genus: *Juncus effusus* L., *Juncus articulatus* L., and *Juncus tenuis* Willd. were found in all four sites. In fact, *Juncus* spp. were common in all four wetlands (Table 2). *Ludwigia palustris* (L.) Elliot was a common species that emerged from the seed bank of Site 6, and was the most common species in the flooded treatment for Site 4 (Table 2). Sedges, including *Carex*, *Cyperus*, and *Eleocharis* species, were relatively common in Site 6 (158.4 seedlings/m^2) and Site 4 (40.3 seedlings/m^2), but not in Site 7 (13.1 seedlings/m^2) or Site 1 (6.8 seedlings/m^2). We found only one woody seedling in this study (*Acer rubrum* L., Site 4). Relative seedling densities for all species in each seed bank can be found in Appendix B (Table B1).

Similarity indices ranged from 46% to 68% for comparisons among Sites 4, 6, and 7 (Table 3). All similarity indices with Site 1 were far lower (range: 7% – 21%). This is likely because the seed bank of Site 1 did not include *Lythrum salicaria* or *Ludwigia palustris*, and only seven *Typha* sp. seedlings.

SEED BANK VARIATION WITHIN SITES.

We observed substantial spatial variation in species richness and seedling density within each site (Table 4). The two larger sites (Sites 4 and 6) have more species than the smaller sites (Sites 1 and 7). Overall, Site 4 has the greatest intrasite variation in terms of species richness (9 – 39 species) and seedling density (1,206 – 33,495 seedlings/m^2 for the drawdown treatment, and 768 – 10,772 seedlings/m^2 for the flooded treatment). We observed the least but still considerable spatial variation in Site 1 (Table 4; 384 – 7,291 seedlings/m^2 and 55 – 1,918 seedlings/m^2 for the drawdown and flooded treatments, respectively).

In general, common species were found in most plots at each site. At Site 1, *Alisma triviale* Pursh., *Leersia oryzoides* (L.) Sw., and *Veronica serpyllifolia* L. were found in all plots but one. *Schoenoplectus tabernaemontani* (C.C. Gmel.) Palla, *Juncus effusus,* and *J. tenuis* were found in all plots but two, while *J. acuminatus* was found in half the plots. *Lythrum salicaria* and *Typha* sp. emerged in every plot at Site 4. *Ludwigia palustris*, another common species in the flooded treatment, was found in four of the six plots. In Site 6, *Ludwigia palustris*, *Lythrum salicaria*, *Solidago canadensis* L., and *Typha* sp. were common in every plot. *Phalaris arundinacea* was also found in every plot, although relative seedling density was low (3.4% in the drawdown treatment, 0% in the flooded treatment). In Site 7, *J. effusus* and *P. arundinacea* were found in all plots but one, while *Typha* sp. and *Lythrum salicaria* emerged from all plots. *Solidago canadensis* was found in four of the six plots. Taxa with relatively low seedling densities were generally not found in multiple plots.

STANDING VEGETATION AND SIMILARITY TO SEED BANKS.

Species richness of the standing vegetation ranged from 24 species to 54 species (Table 3). *Sagittaria latifolia* Willd. (32.7% relative cover), *Potamogeton* sp. (24.6%), and *Schoenoplectus tabernaemontani* (11.4%) were common in Site 1, and no woody species were present. The herbaceous community in Site 4 was dominated by *Lythrum salicaria* (30.9%), *Myosotis scorpioides* L. (9.1%), and *Glechoma hederacea* L. (9.0%), followed closely by *Galium* sp. (8.5%) and *Typha* x *glauca* (8.3%). The herbaceous layer of Site 6 was dominated by *Phalaris arundinacea* (26.0%), *L. salicaria* (15.3%), and *T.* x *glauca* (13.4%). Site 7 was dominated by *T.* x *glauca* (31.5%), *Phragmites australis* (22.5%), and *Phalaris arundinacea* (20.1%). Relative percent cover for all herbaceous species in each wetland can be found in Appendix B (Table B2). All woody species are listed in Appendix B (Table B3).

Similarity indices between the standing vegetation and their respective seed banks were generally low (Table 3, range: 13 – 34%). The seed bank that was most different from the standing vegetation was at Site 6 (13%), at least partially because there were only herbaceous species in the seed bank.

Discussion.

URBAN WETLAND SEED BANK COMPOSITION AND VARIATION.

Urban wetland seed banks in the vicinity of the small city of Binghamton, New York, like their corresponding standing vegetation, have a high percentage of invasive obligate wetland species. Of these, only one individual was woody (*Acer rubrum*). Other studies have found that woody species are uncommon in seed banks (Leck, Parker, and Simpson 1989; Osunkoya *et al*. 2014; however, see Blood, Pitoniak, and Titus 2010).

Our data indicate that there is substantial variation among and within urban wetland seed banks. Species composition and seedling density of common species vary among wetlands; the similarity indices between the seed banks of our sites illustrate these variations. Wetlands that are dominated by the same species, namely Sites 4, 6, and 7, all have higher similarity indices with each other than with Site 1, which has a vastly different species assemblage. Variation within the seed banks is also consistent with other studies (Peterson and Baldwin 2004).

The observed and estimated (S_{est}, Jackknife1, and Chao1) species richness of these seed banks was within the range of other wetland studies (Leck and Simpson 1987; ter Heerdt and Drost 1994; Collins and Wein 1995; Leck 2003; Leck and Leck 2005; Kenow and Lyon 2009; Blood, Pitoniak, and Titus 2010; Farrel *et al*. 2010; Stroh *et al*. 2012; Cui *et al*. 2013; Middleton 2016). The three non-parametric statistical estimators (S_{est}, Jackknife1, and Chao1) for species richness were similar, highlighting certain trends in the variation of species richness among our sites, namely that Site 1 is distinct from the other three wetlands. For all wetland sites, species richness was higher for the drawdown treatment, with few species emerging from the inundation treatment. This is consistent with other findings (van der Valk and Davis 1978; Collins and Wein 1995; Leck 2003; Farrel *et al*. 2010; Middleton 2016).

Given the common species in the standing vegetation of all the sites, it is not surprising that the seed banks of most of our urban wetlands were dominated by invasive species. Overall, the species with the highest seedling density was *Lythrum salicaria*. Although *L. salicaria* was present in the standing vegetation in Site 7, it certainly was not a common species, so we were surprised to see that it was the most common species in the site's seed bank. This is evidence for the reliance of *L. salicaria* on seed bank presence for dispersal and maintaining populations and could give insight into the ability of *L. salicaria* to invade wetlands (Yakimowski, Hager, and Eckert 2005; Frieswyk and Zedler 2006). Although *Phalaris arundinacea* and *Phragmites australis* were common in the standing vegetation in three sites, their abundances in the seed banks were low or nonexistent, respectively. This may provide evidence that these species propagate asexually as opposed to producing large amounts of viable seeds (however, see Albert *et al*. 2015 for both the sexual and asexual reproductive success of *Phragmites australis*).

Site 1 appears to be an exception among our sites; this seed bank has a low presence of invasive species, in particular *L. salicaria* and *Typha* species. Unlike the other wetlands in this study, Site 1 is more distant from major roadways, perhaps delaying establishment of *L. salicaria* and its further invasion into the wetland. Given that we only found two *L. salicaria* seedlings in the seed bank and trace amounts in the standing vegetation, *L. salicaria* may be in the early stages of invasion into Site 1. It is also important to note that the low seedling density observed at Site 1, for both drawdown and flooded treatments, is likely a result of recent construction (M. Larson, Binghamton University, unpublished manuscript).

A striking find was that *Typha* sp. was a common species in the seed banks of Sites 4 and 7 and common in Site 6, but only seven *Typha* sp. seedlings were found in Site 1, again suggesting that Site 1 is an exceptional urban wetland. This is especially surprising given that *T.* x *glauca* was the second most dominant species in the standing vegetation preconstruction (Larson *et al*. 2016). This wetland's seed bank seems to be a bit of an anomaly; *Typha* spp. (*T. latifolia* L., *T. angustifolia,* and *T.* x *glauca*) were present in other seed bank studies (van der Valk and Davis 1978; Leck and Simpson 1987; Collins and Wein 1995; Leck 2003; Blood, Pitoniak, and Titus 2010), the dominant seedling component under saturated conditions (Farrel *et al*. 2010), and the most abundant species present in a marsh seed bank (ter Heerdt and Drost 1994). It is possible that different genotypes of *T.* x *glauca* have established in the local area, or that mixed populations coexist within wetland sites.

Although *Typha* x *glauca* is invasive in many urban wetland systems, *Typha* species may rely on vegetative propagules to establish persistent populations rather than establishing a persistent seed bank. While seedling recruitment may be important, *T.* x *glauca* stands are initially dominated by a few rapidly growing F1 clones (Travis *et al*. 2011). Recent studies have shown, amid some controversy (Selbo and Snow 2004), that *T. latifolia* and *T. angustifolia* do hybridize, with *T. angustifolia* as the maternal parent (Travis *et al*. 2010; Ball and Freeland 2013). *Typha* x *glauca* is assumed to be highly sterile, rarely producing fertile pollen (Dugle and Copps 1972; Smith 1987); thus, sterile F_1 hybrids in Site 1 would result in a lack of cattail seedlings in the seed bank. However, there is much evidence that *T.* x *glauca* can backcross with its parent species (Kuehn, Minor, and White 1999; Travis *et al.* 2010, 2011). Molecular data from cattails in Site 1 could help determine the potential lack of viable seeds in the seed bank.

URBAN WETLAND PLANT COMMUNITY DYNAMICS AND MANAGEMENT IMPLICATIONS.

As expected, the seed banks generally have more species than the standing vegetation, potentially resulting in the low similarity indices (Hopfensperger 2007). Low similarity values between seed banks and the standing vegetation were observed at all sites. A meta-analysis found that the Sørenson's similarity index between seed banks and standing vegetation ranged from 25% to 79% (Hopfensperger, 2007). Our study is at the lower end of these data, with similarity values ranging from 19% to 34%. Environmental conditions present during the seed bank studies could result in species emerging under different conditions than those that are present in the wetlands. Many of the species that we observed in the seed bank are perennial species; wetlands that are dominated by annual plant species often have a higher similarity index between the seed bank and standing vegetation (Ungar and Woodell 1993; Jutila 2003).

Our findings illustrate that urban wetland seed banks in a small city may be viable and can contribute to the revegetation of disturbed sites, potentially affecting future plant communities. Given that invasive species, specifically *Lythrum salicaria* and *Typha* sp., are common species in our urban wetland seed banks, supplemental planting of native species should be considered for wetland reconstructions. High variation in both species richness and seedling density indicates that some patches may be slow to recover if solely reliant on a seed bank. Management practices should consider supplemental planting and seeding to increase the successful establishment of native plant species.

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Conservation District, it has not gone through the Agency's publications review process and, therefore, may not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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Table 1. Seedling density, observed species richness, and total species estimates based on sample-based extrapolation (S_{est}), Jackknife1, and Chao1 extrapolations for each wetland site (seedlings/m^2 of wetland surface area). Means for seedling density shown with standard errors (n = 6 for each site). Seedling density means not sharing a lowercase letter following as a result of two-way ANOVA tests differ significantly at P = 0.05 according to Tukey means comparisons. Species extrapolations shown with standard deviations.

Site	Seedling density	Observed Species Richness	S_{est}	Jackknife1	Chao1
1	3367 [a] ± 1225	37	56.3 ± 8.5	56.2 ± 5.4	49.1 ± 8.8
4	18762 [b] ± 3966	56	70.7 ± 6.5	75.4 ± 9.4	79.1 ± 14.8
6	19132 [b] ± 4186	52	77.0 ± 9.5	77.8 ± 8.2	77.6 ± 17.9
7	8721 [ab] ± 3703	28	37.9 ± 5.6	40.5 ± 5.3	32.2 ± 4.9

Table 2. Relative seedling densities for top three taxa (ranked 1-3), and their sum, in each treatment (DD = drawdown, FL = flooded) for each of the four wetland sites. Invasive species are in bold.
Species key: J = *Juncus* spp., LO = *Leersia oryzoides,* AT = *Alisma triviale,* ST = *Schoenoplectus tabernaemontani,* **LS = *Lythrum salicaria,*** LP = *Ludwigia palustris,* **T = *Typha sp*.,** E = *Eleocharis* spp., SC = *Solidago canadensis*

	Site 1		Site 4		Site 6		Site 7	
Rank	DD	FL	DD	FL	DD	FL	DD	FL
1	J (57.0)	J (34.8)	**LS (75.1)**	LP (48.3)	LP (31.3)	E (65.8)	**LS (50.6)**	**LS (47.7)**
2	LO (17.2)	ST (18.9)	**T (4.0)**	**T (25.8)**	**LS (24.1)**	**T (22.4)**	J (20.7)	**T (23.5)**
3	ST (7.3)	AT (16.7)	J (3.9)	**LS (22.1)**	J (12.4)	LP (5.9)	**T (13.2)**	SC (18.5)
Σ	81.5	70.4	83	96.2	67.8	94.1	84.6	89.7

Table 3. Percent similarity for comparisons between seed banks, and between seed banks and the standing vegetation at each site, as well as the species richness of standing vegetation. Similarity indices were calculated based on relative seedling density for seed bank comparisons and are in bold. The index between a seed bank and the standing vegetation was calculated using species presence/absence data. Species richness for the standing vegetation of each site is in italics.

Site	1	4	6	7
1	-	-	-	-
4	**7**	-	-	-
6	**16**	**48**	-	-
7	**21**	**68**	**46**	-
Standing vegetation within site	33	22	19	22
Species richness of standing vegetation	*29*	*41*	*54*	*24*

Table 4. Median values of species richness and seedling density for the two treatments (DD = drawdown treatment, FL = flooded treatment) for all four wetlands. Seedling density is expressed as seedlings/m^2 of wetland surface area. Species richness and seedling density ranges of six plots in each site are in parentheses.

	Site 1	Site 4	Site 6	Site 7
Species Richness	13 (5 - 20)	17.5 (9 - 39)	19.5 (12 - 29)	11 (3 - 18)
DD Seedling Density	1932 (384 - 7291)	16254 (1206 - 33495)	17638 (4358 - 28781)	4070 (110 - 24614)
FL Seedling Density	493 (55-1918)	1124 (768 - 10772)	535 (247 - 5866)	384 (110 - 5482)

Fig. 1 Species richness rarefaction and extrapolation curves based on A) sample-based extrapolation (S_{est}), B) Jackknife1, and C) Chao1 using EstimateS software (Colwell 2012). Sample-based extrapolation rarefaction is extrapolated to 12 samples, whereas Jackknife1 and Chao1 are based on six samples. Error bars represent standard deviations. Triangle = Site 6, open circle = Site 4, square = Site 1, closed circle = Site 7.

Fig. 2 Stacked bars show percentage of seedlings for each wetland indicator status (WIS) category for identified seedlings in each seed bank. Gray bars indicate the percentage of invasive seedlings for identified seedlings in each seed bank. FACU = Facultative upland, FAC = Facultative, FACW = facultative wetland, OBL = obligate wetland.

Fig.1

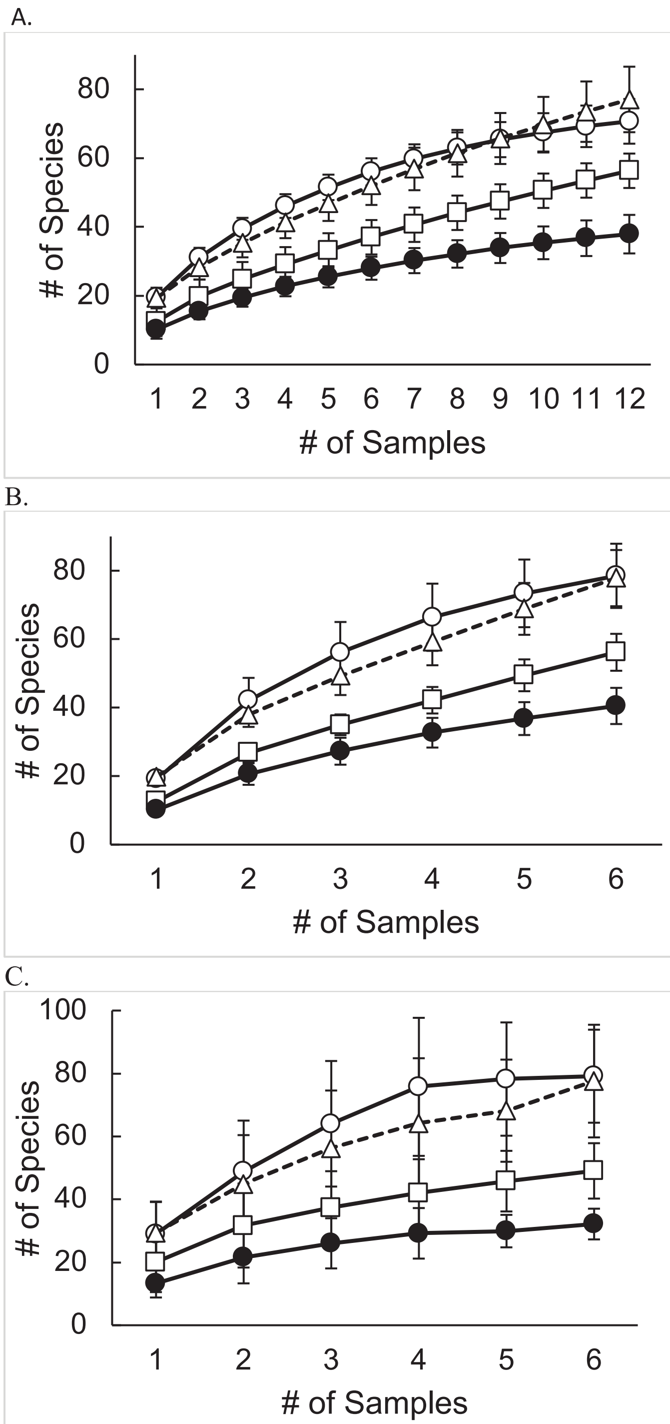
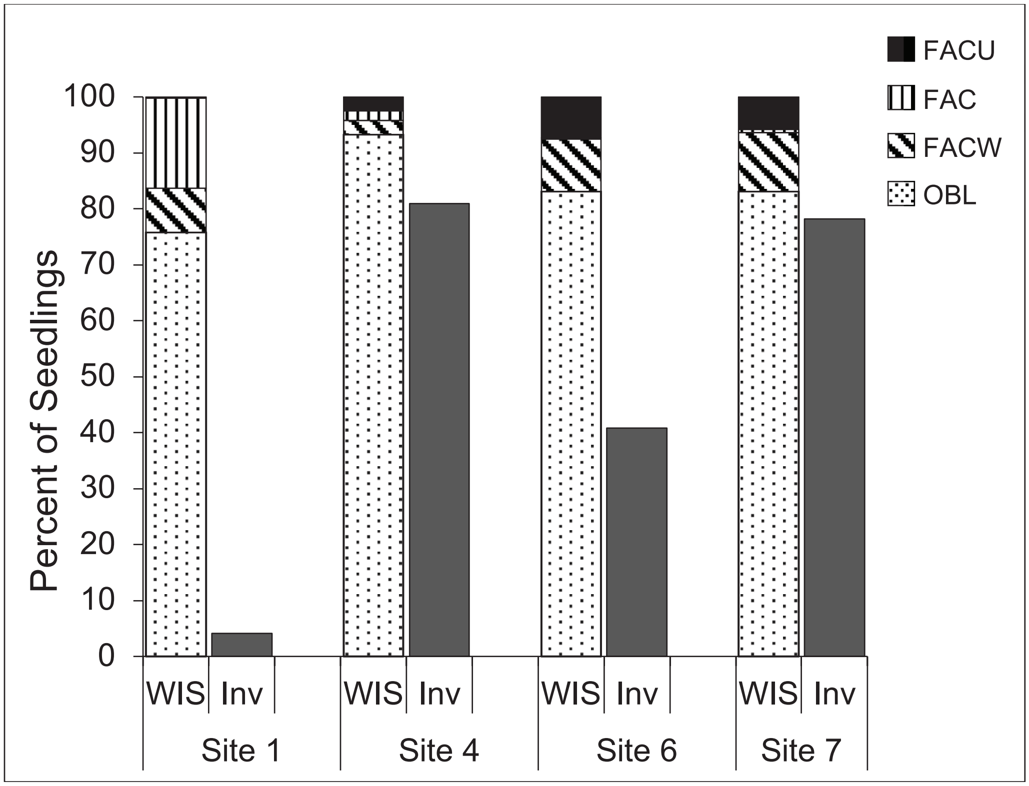

A.
# of Species
0
20
40
60
80
1 2 3 4 5 6 7 8 9 10 11 12
# of Samples
B.
# of Species
0
20
40
60
80
1 2 3 4 5 6
# of Samples
C.
# of Species
0
20
40
60
80
100
1 2 3 4 5 6
# of Samples

Fig.2


FACU
FAC
FACW
OBL
Percent of Seedlings
100
90
80
70
60
50
40
30
20
10
0
WIS
Inv
WIS
Inv
WIS
Inv
WIS
Inv
Site 1
Site 4
Site 6
Site 7

Chapter 4: Impact of habitat alteration on the seed bank and standing vegetation of an urban retention wetland

The following chapter is formatted for and submitted to *Wetlands Ecology and Management*.

Larson MA, Shepherd J, Titus JE (under revision) Impact of habitat alteration on the seed bank and standing vegetation of an urban retention wetland. *Wetlands Ecology and Management*.

Abstract:

The impacts of major habitat alterations on plant communities in urban wetlands are poorly understood, despite the importance of these ecosystems. Regrading of wetlands can disrupt seed banks and standing vegetation, thus limiting potential revegetation and increasing the likelihood of invasive species establishment. We evaluated the effects of regrading an urban retention wetland on its seed bank and standing vegetation. Sediment cores for the seed bank study were collected in April 2011 (before) and in May 2014 (after). The density and species composition of seedlings that emerged from the seed bank were determined under drawdown and flooded conditions. The standing vegetation composition was recorded in June 2011 just prior to the regrading, and twice in each growing season (July and August, 2012-2014). Seedling densities were nearly three-fold greater than those after regrading, and seedling density significantly decreased in the drawdown treatment. Species richness in the standing vegetation decreased immediately after the regrade and rebounded over three years. Relative cover of invasive species decreased after regrading, primarily due to a decrease in *Myosotis scorpioides* and *Typha* x *glauca*. Information about the seed bank composition and 2011 standing vegetation was not sufficient to make predictions about the recovering vegetation, likely because we did not include asexual propagules in our assessment. This study indicates that a regrading project can substantially reduce seedling density of an urban wetland seed bank, but standing vegetation may show signs of recovery within a short time span due to the presence of a prolific bud bank.

Keywords: Urbanization, wetland reconstruction, invasive species, *Sagittaria latifolia*, *Potamogeton*, bud bank

Introduction

Urban areas are projected to increase three-fold by 2030 (Batty 2008; Seto et al. 2012; Nilon et al. 2017), thus impacting the wetlands within these landscapes. These ecosystems provide unique opportunities for studying community responses to changes in the environment. Urban wetlands, for example, are subject to changes in hydrology (Ewing 1996; Moscrip and Montgomery 1997; Kaye et al. 2006; Stander and Ehrenfeld 2009a, b), high inputs of pollutants (Findlay and Houlahan 1997; Ehrenfeld 2000; Larson et al. 2016), and increased presence of exotic species (Galatowitsch et al. 1999; Ehrenfeld 2000; Zedler and Kercher 2004; Bowman Cutway and Ehrenfeld 2010; Noble and Hassall 2015; Larson et al. 2016). Urban wetlands may also experience frequent disturbances (Grayson et al. 1999) from erosion due to altered hydrology (Ravit et al. 2017), removal of aboveground biomass (e.g., mowing) for crop harvest (Vécrin et al. 2007), invasive species management (Lawrence et al. 2016), stormwater control (Blecken et al. 2017), or regrading to accommodate increased input. Constructed wetlands provide an opportunity to study the resilience of urban wetlands to major disturbances. These projects involve reestablishing wetland hydrology and restoring vegetation by planting, seeding, or adding donor wetland soil (Brown and Bedford 1997; Middleton 1999; Zedler 2000; Craft et al. 2003; Hopple and Craft 2013). Because wetland construction projects commonly fail to monitor plant community establishment in these heavily disturbed wetlands, little is known about the resilience of urban wetland plant communities to disturbances (Zedler 2000), and even less is known about the role that urban wetland seed banks can play in recovery.

Sediment that contains a viable seed bank can be important to wetland revegetation (Vécrin and Muller 2003; Nishihiro et al. 2006; Muller et al. 2013; Kaplan et al. 2014). Given that urban wetlands have lower species richness than natural wetlands (Galatowitsch et al. 1999; Ehrenfeld 2000; Zedler and Kercher 2004; Noble and Hassall 2015; Larson et al. 2016), standing vegetation may take longer to recover from major habitat alterations, like reconstruction projects where the wetland is completely regraded. However, a high presence of invasive species in degraded wetlands may yield an undesirable plant community in reconstructed wetlands (D'Antonio and Meyerson 2002; Robertson and James 2007; Ficken and Menges 2013; Landis and Leopold 2014; Shang et al. 2016; Larson and Titus 2018). The prolific seed production and longevity of some invasive species in seed banks (Welling and Becker 1990; Ågren 1996; Neff et al. 2009; Passos et al. 2017) may promote invasive species establishment in newly opened substrate (D'Antonio and Meyerson 2002; Meyer et al. 2013). Understanding the relationship between standing vegetation and the seed bank may be critical to effectively meet restoration or mitigation goals (Ficken and Menges 2013; Wall and Stevens 2015; Cui et al. 2016) and increase the resilience of these wetland ecosystems (Hopfensperger 2007).

While seed banks can aid in the recovery of a wetland, the characteristics of a seed bank itself may change after a large-scale habitat alteration; the seed banks of recently restored wetlands often differ from those of reference wetlands (Neff et al. 2009; Beas et al. 2013). For example, changes in hydrology and legacy effects from agricultural land use can lead to an increase in seedling density of generalist and invasive species in both seed banks and standing vegetation (Bissels et al. 2005; Greet et al. 2013; Bart and Davenport 2015). Restored wetland seed banks may recover quickly from alteration events if there is sufficient seed dispersal into the new site, as seen in flood plains (Osunkoya et al. 2014) or tidal freshwater wetlands (Leck 2003; Leck and Leck 2005; Neff et al. 2009). We expected that a large-scale reconstruction and regrade of a wetland would change the seedling density and species richness of the seed bank; however, isolated wetland sites would have low seed dispersal from nearby seed sources. Thus, we predicted that a major regrading project would reduce the seedling density and species richness in the seed bank of a relatively isolated wetland, and that the standing vegetation may be slow to recover.

Lieberman is an urban retention wetland located on the Binghamton University campus in Vestal, New York, that underwent a complete regrade and expansion to accommodate increased runoff from new infrastructure. We investigated the effects of this major habitat alteration on the urban wetland plant community by recording changes in both the seed bank and the standing vegetation. We hypothesized that there were two strong influences on the recovering standing vegetation: 1) the seed bank and 2) the vegetation before the regrade as a propagule source. If the seed bank is important, then common species in the seed bank before the regrade would also be common in the new vegetation. Alternatively, species that were common in the standing vegetation prior to the regrade (Larson et al. 2016) should be common in the revegetated wetland.

Methods

Study Site

Lieberman, referred to as "Site 1" in Larson et al. (2016) and Larson and Titus (2018), is located on the Binghamton University campus in Vestal, NY. The campus supports a high volume of traffic; as of 2016, approximately 48% of the 13,000 full-time undergraduate students commute to campus, in addition to administrators, faculty, staff members, and ca. 4000 graduate students (Annual Survey of Colleges 2016). The site is fairly isolated and does not receive hydrological input from other wetlands, with the nearest wetland complex located approximately 0.6 km away. The drainage wetland receives runoff from 0.56 km^2 portion of campus, including parking lots, paved roadways and sidewalks, and buildings (Kearney et al. 2013). The main inlet, via a number of culverts and drainage ditches, is located in the southwest corner of the site (Fig. 1). Groundwater seepage may also be a source of water. The wetland drains through a culvert into Fuller Hollow Creek which discharges into the Susquehanna River (Zhu et al. 2008), the largest tributary of the Chesapeake Bay.

In 2004, the wetland was drained and a berm was built along the east side of the pond; the wetland has since served as a retention pond to accommodate campus runoff (Fig. 1a). This 0.15 ha stormwater retention wetland had a small channel near the inlet that opened to a larger, inundated marsh dominated by *Sagittaria latifolia* and *Alisma triviale*. *Myosotis scorpioides*, *Typha* x *glauca*, and *Leersia oryzoides* were found near the inlet and around the perimeter of the wetland.

Due to increased infrastructure, the wetland was regraded to accommodate increases in impervious surface runoff. Construction of Lieberman began in spring of

2011, when targeted dormitories were demolished. Regrading of the wetland and the surrounding area began in July 2011, and resulted in the destruction of existing vegetation and both upheaval and spreading of wetland sediment. Sediment traps were added near the main inlet and outlet, connected by a snaking channel (Fig. 1b). Overall, the area of the wetland doubled to approximately 0.34 ha. Landscape was created using sediment from the wetland, and the wetland is notably more inundated than before. No widespread seeding or planting occurred. Regrading was completed in the early spring of 2012.

Seed bank collection

In April 2011, we randomly selected 15 plots, with five points on each of three transects perpendicular to the main axis of the wetland before regrading. These transects were randomly selected along 25m intervals on the edge of the retention pond. We collected a total of 30 sediment cores (15.2 cm diameter, 5 cm depth), two from each plot. Sediment was stored in a cold room at 4.4 °C for one month. In early May 2014, we collected 12 sediment cores from six plots, before seedling germination in the field, as described in Larson and Titus (2018).

Experimental set–ups for both seed banks were described in Larson and Titus (2018), based on van der Valk and Davis (1978). These studies were conducted in temperature-controlled fiber glass tanks in the Research Greenhouse at Binghamton University. Large debris, including rhizomes and tubers, were removed from the sediment. Sediment cores were subjected to two treatments: a simulated drawdown treatment with water levels 5 cm below the sediment surface, and a simulated flooded treatment with water levels 5.5 cm above the sediment surface. Seedling data were collected from June 2011 until seedling emergence ceased in January 2012 for the Lieberman seed bank before regrading, and from June 2014 until September 2015 for the samples collected after the regrade. Seedlings that could not be identified in the trays were excavated and planted in separate pots until they could be identified.

Vegetation sampling

Standing vegetation was sampled in June 2011 from 15 sampling points on 11 transects just prior to regrading (Larson et al. 2016). We observed the progression of revegetation by sampling the standing vegetation in July and August every growing season after regrading, from 2012-2014. Percent cover estimates were recorded for each herbaceous species within 1 m^2 quadrats, to the nearest 5% (Mueller-Dombois and Ellenberg 1974). We recorded vegetation data from 44 sampling points located on 11 transects in the three growing seasons after regrading. Flooding during the 2012 sampling periods reduced the number of sampling points to 39 (for both July and August).

Species identification

Taxa were identified to the species level using Gleason and Cronquist (1991), with nomenclature updated according to the NY Flora Atlas (Weldy et al. 2017) to the species, genus, or family level; those that could not be identified were recorded as unknown species. We identified 92.2% and 97.8% of seedlings to the species or genus level for the 2011 and 2014 seed banks, respectively. The native status, wetland indicator status, growth habit (graminoid, forb, or shrub/tree), and duration (annual, biannual, or perennial) of each species were found using the USDA (United States Department of Agriculture) plant database for the Northeast region and the NY Flora Atlas (Weldy et al. 2017). For the purposes of this paper, we equate “invasives” with non-native species.

Cattails in the standing vegetation were identified as the invasive *Typha* x *glauca* due to substantial variation in the gap size between male and female flowers, as well as leaf width (Selbo and Snow 2004). *Typha* seedlings were identified as *Typha* sp. due to a lack of these morphological traits.

Data analyses

For the purposes of this paper, we assigned codes for all comparisons analyzed: SB11.14 (seed bank before regrading in 2011 versus three years after regrading in 2014), SV11.12 (standing vegetation before regrading in 2011 versus the first growing season after regrading in 2012), SV12.14 (standing vegetation in the first growing season after regrading in 2012 through 2014), and SV11.14 (standing vegetation before regrading in 2011 through 2014).

We compared the seed banks between sampling times (SB11.14) by calculating seedling density, defined as the number of seedlings/m^2, based on the wetland surface area sampled. We used a two sample t-test to assess for differences in seedling densities before and after regrading (VassarStats February 27 2017). We measured species importance as relative seedling density, expressed as the number of seedlings for a species divided by the total number of seedlings in that same survey. This was calculated separately for drawdown and flooded treatments to avoid a bias to species that could only grow in one of the treatments. We compared seed bank compositions (SB11.14) using Sørenson's similarity index (Sørenson, 1948).

Vegetation data for each sampling period were summarized as relative cover, defined as the cover of a species divided by the total cover of all species in that same wetland, as well as mean percent cover, or the total cover of a species divided by the number of sampling quadrats that species was found in. Changes in standing vegetation composition were summarized by calculating species richness and species diversity (SV11.12, SV12.14, SV11.14) using Shannon-Weaver Diversity indices (H').

Results:

Changes in the seed bank (SB11.14)

We observed several substantial changes in the seed bank following the regrading project (SB11.14), both in quantitative and qualitative terms. We found 53 species before the regrade, but only 37 species afterwards. Total seedling densities before regrading were nearly three-fold greater than those in 2014 (Table 1; $t = 2.47$, $df = 19$, $p = 0.023$). Seedling density significantly decreased in the drawdown treatment ($t = 2.15$, $df = 19$, $p = 0.045$), but not in the flooded treatment (Table 1). We also observed a shift in the proportions of wetland indicator status categories (Table 2); the 2011 seed bank was comprised of 57.5% obligate wetland species, while we observed far more (75.8%) in 2014. In contrast, we found a higher percentage of FACW species before the regrade (20.6%) than in 2014 (7.8%). Perennial species comprised most of both seed banks (Table 2; 99.5% before and 84.0% after). We observed a shift in plant growth habits (Table 2); the 2011 seed bank was split between forb/herbaceous and graminoid species (40.4% forbs, 59.6% graminoids), but largely comprised of graminoid species after the regrade (12.8% forbs, 87.2% graminoids). We found low percentages of invasive species seedlings for both seed bank surveys (Table 1; 4.3% before and 4.2% after, respectively).

Both seed bank surveys were dominated by *Juncus* spp. (42.3% drawdown before the regrade; 57.0% and 34.9% for drawdown and flooded treatments after the regrade, respectively) and *Leersia oryzoides* (22.8% drawdown before the regrade and 17.2% and 14.4% for drawdown and flooded treatments after the regrade, respectively). *Alisma triviale* was common under flooded conditions before the regrade (57.4%) but not after (16.7%, flooded treatment); the species was not common under drawdown conditions for either survey. *Schoenoplectus tabernaemontani* was common after the regrade (7.3% and 18.9% for drawdown and flooded treatments, respectively), but not before. The similarity index between the two seed bank surveys was 49% due to similar dominant species. Despite obvious differences in total species richness, the changes in species composition were additions or losses of species with low seedling density (< 10 seedlings). *Sagittaria latifolia*, *Lemna minor, Stachys palustris, Myosotis scorpioides* and *Epilobium* species (*E. ciliatum, E. coloratum, E. hirsutum,* and *E. palustre*) failed to establish in samples collected after regrading. Two *Eleocharis* species, *Juncus acuminatus, J. bufonius*, and *Schoenoplectus tabernaemontani* were all found in 2014, but not before. We did not observe *Typha* sp. seedlings before the regrade, and only seven emerged afterwards.

In addition to a shift in species composition, we observed striking differences in the density of certain species (SB11.14; Table 3). *Leersia oryzoides*, for example, densely populated the seed bank with 944 seedlings/m^2 before the regrade, but only 562 seedlings/m^2 emerged after. Similarly, we found 894 seedlings/m^2 of *Juncus effusus* in 2011, but 443 seedlings/m^2 in 2014. *Alisma triviale* was also less common in the 2014 seed bank (160 seedlings/m^2), while 736 seedlings/m^2 were found in the 2011 seed bank. Relative seedling density data for all taxa can be found in Appendix C (Table C1).

Changes in the standing vegetation

We observed several changes in the standing vegetation immediately after regrading (SV11.12), and a still different aboveground plant community by the end of the 2014 growing season (SV11.14). Species richness in the standing vegetation decreased slightly (SV11.12) and then rebounded (SV12.14), exceeding the species richness before the regrade (Fig. 2; 24 species in June 2011, 33 species in August 2014). We found that species diversity (H') sharply declined immediately after regrading (SV11.12), but steadily increased by 2014 (SV12.14; Fig. 2). Relative percent cover of invasive species decreased after regrading (SV11.12), due largely to decreases in *Myosotis scorpioides* and *Typha* x *glauca* cover (Fig. 2 and 3). We did not observe a net change in invasive species cover in the growing seasons after regrading (SV12.14); the highest percentage of invasive species cover was 6.9% the first survey after construction in July 2012 (Fig. 2). Although the wetland was dominated by obligate wetland species (88.5% relative cover), the cover of obligate wetland species increased (SV11.14; Table 4, > 96% relative cover for all sampling dates after regrading). There was no change in relative cover of plant duration (SV11.14), with over 97% relative cover of perennial species for all sampling dates. After regrading, the increase in relative cover of graminoid species nearly doubled (SV12.14; Table 4).

Despite few changes in general vegetation characteristics, we observed several shifts in the species composition of the standing vegetation (Table 5). The mean cover of many common species decreased immediately after regrading (SV11.12), with the exception of *Potamogeton* sp., which increased from 8.1% mean relative cover to 30.4% in July 2012 (SV11.12; Fig. 3). In fact, *Potamogeton* sp. was the only common species that considerably decreased (14.8%) by August 2014 (SV12.14). *Myosotis scorpioides* did not recover over the three years after regrading, although this was the most common species in 2011 (SV11.14; 10.7% mean cover in 2011 , < 0.04% for all surveys after the regrade). Similarly, *Typha* x *glauca* cover substantially decreased immediately following regrading (SV11.12; 10.4% mean cover in 2011, 0.03% mean cover in July 2012), but showed signs of recovery (SV12.14; 2.2% in August 2014). *Potamogeton* sp. and *Sagittaria latifolia* were consistently the most common species in the wetland after regrading (SV12.14). *Alisma triviale* exhibited an overall increase in cover (SV11.14), becoming more common in August 2014 (9.5%) than in 2011 (2.2%). *Leersia oryzoides* cover steadily increased (SV12.14), with a net increase in mean percent cover after regrading (SV11.14; 8.1% in June 2011, 14.8% in August 2014). *Schoenoplectus tabernaemontani* and *Eleocharis palustris* were common after regrading, but were not present in the June 2011 survey (SV11.14). Like *L. oryzoides,* the mean percent cover of *S. tabernaemontani* increased over the 2012 and 2013 growing seasons (SV12.14). Relative percent cover data for all taxa can be found in Appendix C (Table C2).

Discussion:

Habitat alteration reflected in seed bank and standing vegetation

The regrading of Lieberman was a large scale habitat alteration that substantially impacted the seed bank and the standing vegetation. After regrading, seed bank species richness and seedling density remained lower than the 2011 seed bank survey (SB11.14). The total seedling density before the regrade is within the range of other urban wetlands in the Greater Binghamton area, but the seedling density in 2014 was significantly lower

(Larson and Titus 2018). The overall seed bank depletion may be a consequence of the dredging and leveling work needed to expand the wetland area. Other studies have shown that seed banks can initially be negatively affected by major habitat alterations; for example, an extreme flooding event increased seedling density but reduced species richness and diversity, yet the riparian seed bank itself recovered quickly and was considered resilient (Osunkoya et al. 2014). Neff et al. (2009) found that the seedling density of a recently restored tidal marsh significantly increased by more than 40-fold within a year, and species richness was significantly higher than any other reference site. The seedling density and species richness of these studies is likely a result of prolific seed dispersal, allowing the seed banks to recover quickly. The seed bank of Lieberman may require more time to recover due to a low seed dispersal into the wetland, possibly due to its isolated position in an urban, fragmented landscape.

The dominant species in the seed bank did not drastically change, with *Juncus* species (including *J. effusus, J. tenuis,* and *J. articulatus*) and *Leersia oryzoides* remaining common. However, we observed an overall shift in species composition, including a depletion of some species that were common in the seed bank before the regrade, namely *Sagittaria latifolia* (SB11.14). This is particularly interesting because *S. latifolia* was one of the most common species in the standing vegetation after the regrade, indicating that *S. latifolia* may rely on asexual reproduction as opposed to seed dispersal. *Schoenoplectus tabernaemontani* was present in the 2014 seed bank, but not before (SB11.14); this species was unexpectedly dominant in the new standing vegetation. *Schoenoplectus tabernaemontani* may be effectively producing seeds to colonize the new habitat (Neff et al. 2009).

We observed a shift in the wetland indicator status proportions; both the 2014 seed bank and the standing vegetation (2012-2014) had higher percentages of obligate species and decreases in upland and facultative wetland species (SB11.14, SV11.14). The reconstructed wetland is conspicuously more inundated (M Larson, personal observation). These changes in hydrology may have shifted the seed bank to favor obligate wetland species. Landscape architects and engineers need to pay particular attention to rehabilitating urban wetland hydrologies to favor wetland plant establishment (Wang et al. 2016; Schwab and Kiehl 2017).

The regrading project initially decimated the standing vegetation (SV11.12), but plant cover steadily increased (SV12.14). Unlike the seed bank, we saw a shift in common species after regrading. *Myosotis scorpioides* and *Typha* x *glauca,* two of the common species in the 2011 standing vegetation, did not rapidly establish compared to native species, like *Sagittaria latifolia, Leersia oryzoides,* and *Schoenoplectus tabernaemontani.* Although both seed banks contained few invasive species (4.2% and 4.3%, respectively), we were surprised to see that invasive species cover in the new standing vegetation was lower, as many invasive species rapidly colonize disturbed sites (D'Antonio and Meyerson 2002; Ehrenfeld 2008; Bowman Cutway and Ehrenfeld 2010; Meyer et al. 2013), although not all urban habitats have a high presence of exotics (e.g, Ehrenfeld 2005). While we only observed six *Lythrum salicaria* seedlings before the regrade and two after (SB11.14), we may be seeing the beginnings of *L. salicaria*'s invasion into the site. Purple loosestrife has been observed in the standing vegetation since the conclusion of this study. Continued dispersal of purple loosestrife from outside the wetland and seed rain from established plants may increase the presence of *L.*

salicaria. Although *Typha* x *glauca* had surprisingly low seedling densities in both seed bank surveys, as discussed in (Larson and Titus 2018), we predict that *Typha* will continue to spread vegetatively in the newly altered habitat. Future invasive species management may need to include *L. salicaria* and *Typha* x *glauca* removal; for example, Ho and Richardson (2013) recommend the removal of invasive species for 5-7 years to ensure native plant establishment and limit invasive species dominance. Continued monitoring of the standing vegetation will provide important information regarding invasive species management in urban wetlands.

Using seed bank and 2011 standing vegetation to make predictions about vegetation recovery

We originally expected that the seed bank surveys could serve as a guide to predicting species that would establish after the regrading project. The most common species in both seed banks, based on seedling density, were *Juncus* spp., *Leersia oryzoides*, *Alisma triviale* (2011 only) and *Schoenoplectus tabernaemontani* (2014 only). While these species were found in the standing vegetation after regrading (SV12.14), they were not the most common. Information about the seed bank composition was not sufficient to allow us to make predictions about the species composition of the reconstructed wetland.

We also hypothesized that the standing vegetation prior to regrading would allow us to predict what species would be present in the standing vegetation (SV11.14). However, two of the most important species (*Myosotis scorpioides* and *Typha* x *glauca*), based on relative cover from the 2011 survey, were not common after at the end of the study (Larson et al. 2016). *Leersia oryzoides*, however, recovered to similar mean percent cover values after three years (SV11.14).

Based on our seed bank surveys and 2011 standing vegetation data, we did not predict that *Sagittaria latifolia* and *Potamogeton* sp. would be the two most important species in the standing vegetation after regrading. The establishment of these two species may be a consequence of asexual propagation, namely through the production of corms or tubers (Gleason and Cronquist 1991; Dorken and Barrett 2003, 2004; Van Drunen and Dorken 2012), and rhizomes (Gleason and Cronquist 1991; Wiegleb and Brux 1991), respectively. Many *Potamogeton* species spread vegetatively from turions and the fragmentation of stolons and rhizomes (Wiegleb and Brux 1991; Lundholm and Len Simser 1999; Combroux and Bornette 2004; Vári 2013; Kaplan et al. 2014). Asexual propagules of *S. latifolia* may be important to restore vegetation (Williams et al. 2008), perhaps because tubers or corms increase the likelihood of survival in disturbed habitats (Dorken and Barrett 2003) and dispersal rates within sites (Dorken and Barrett 2004). Similarly, asexual reproduction strategies of *Potamogeton* may be more successful in habitats with frequent disturbances (Wiegleb and Brux 1991), although other studies have shown that *Potamogeton* species can readily germinate under flooded conditions (Wang et al. 2016). Meyer et al. (2013) also observed that *Potamogeton* can rapidly colonize newly restored side-channels along the Rhine River. The recovery of standing vegetation in a riverine wetland after restoration was attributed to an increased recruitment from rhizomes and other vegetative fragments, suggesting that bud banks can be important for wetland recovery from major habitat alterations (Combroux et al. 2002; Combroux and Bornette 2004).

Management implications

This study indicates that a complete regrading project can drastically reduce the seed bank community. We observed a significant decrease of emerging seedlings, indicating a potential dilution of seeds in the seed bank. This may have led to shifts in both the standing vegetation and seed bank communities. Although most of the species that were lost after the regrading project were originally observed in small numbers, even common species were drastically reduced in their seedling density. Despite having substantially lower seedling densities in our seed bank study (SB11.14), the standing vegetation recovered after three years (SV12.14). This may be a consequence of the existing bud bank, as opposed to the seed bank. Bud banks should be considered in restoration management plans, as these likely play an important role in vegetation recovery (Lundholm and Len Simser 1999; Combroux et al. 2002).

It is encouraging that the urban wetland recovered within three growing seasons without an increase in invasive species cover. While vegetation cover rapidly establishes in some systems (Meyer et al. 2013), other studies estimate that the time required for an ecosystem to recover after wetland restoration or creation may be several decades, or even centuries (Jones and Schmitz 2009; Moreno-Mateos et al. 2012, 2015; Stefanik and Mitsch 2012; Curran et al. 2014; Johansen et al. 2017). For example, while species richness of restored wetlands was similar to natural wetlands 10 years after restoration, the plant community of restored wetlands contained species of lower Coefficient of Conservatism (C of C) and a lower percentage of obligate wetland species (Hopple and Craft 2013). The vegetation of passively restored wetlands were not similar to reference sites after as many as 25 years (González et al. 2016). Moreover, urban environmental conditions require modified restoration and rehabilitation designs to ensure restoration project success (Ravit et al. 2017). Shifts in plant communities and ecosystem functions may need to be documented over longer periods of time (Stefanik and Mitsch 2012), and future research should monitor long-term changes in restored ecosystems.

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Table 1: Total seedling density, seedling density for drawdown (DD) and flooded (FL) treatments, species richness, invasive species (%) before and after regrading (2011 and 2014, respectively). Seedling density is expressed as seedlings/m^2. Significant results, with a $p < 0.05$ (*) were determined using a t-test.

	Total Seedling Density*	DD Seedling Density *	FL Seedling Density	Species richness	Invasive Species
2011	9880 ± 1509	8086 ± 1433	1794 ± 534	53	4.3
2014	3367 ± 1225	2764 ± 995	603 ± 256	37	4.2

Table 2: Proportions of wetland indicator status, duration, and growth habit of the seed bank before and after regrading (2011 and 2014, respectively). OBL = obligate wetland, FACW = facultative wetland, FAC = Facultative, FACU = Facultative upland, U = Upland

		2011	2014
Wetland Indicator Status	OBL	57.5	75.8
	FACW	20.6	7.8
	FAC	20.9	16.2
	FACU	0.6	0.2
	U	0.3	-
Duration	Perennial	99.5	84.0
	Annual	0.4	16.0
	Biennial	<0.1	-
Growth Habit	Forb	40.4	12.8
	Graminoid	59.6	87.2
	Tree/Shrubs	1.76	-

Table 3: Relative seedling density for species with a relative seedling density greater than 5% in at least one treatment; drawdown (DD) and flooded (FL). Invasive species are in bold.

	2011		2014	
Species	DD	FL	DD	FL
Alisma triviale Pursh	5.5	57.4	2.2	16.7
Juncus acuminatus Michx.	-	-	6.5	-
Juncus articulatus L.	7.1	-	0.7	12.1
Juncus bufonius L.	-	-	5.6	-
Juncus effusus L.	22.1	0.1	15.5	2.3
Juncus sp.	-	-	22.2	19.7
Juncus tenuis Willd.	13.1	-	6.6	0.8
Leersia oryzoides (L.) Sw.	22.8	2.3	17.2	14.4
Lemna minor L.	0.00	23.7	-	-
Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	-	-	7.3	18.9
***Veronica serpyllifolia* L.**	**8.5**	**1.6**	**4.6**	**5.3**

Table 4: Relative % cover of wetland indicator status, duration, and growth habit of standing vegetation before (2011) and after regrading (2012-2014). OBL = obligate wetland, FACW = facultative wetland, FAC = Facultative, FACU = Facultative upland

		2011	2012		2013		2014	
		Jun	Jul	Aug	Jul	Aug	Jul	Aug
Wetland Indicator Status	OBL	88.5	98.4	99.3	99.4	96.4	96.3	97.1
	FACW	4.6	1.3	0.5	0.1	1.2	1.7	0.3
	FAC	5.0	0.2	0.2	0.5	2.4	1.7	2.5
	FACU	1.9	0.0	0.0	0.0	0.1	0.2	0.1
Duration	Perennial	97.9	98.0	99.5	99.2	95.9	98.4	97.1
	Annual	1.6	1.1	0.5	0.8	4.1	1.1	2.8
	Biennial	0.5	0.9	-	-	-	0.4	-
Growth Habit	Forb	77.1	80.4	65.7	66.7	63.1	59.2	61.0
	Graminoid	21.1	19.6	34.3	33.3	36.9	40.8	39.0
	Tree/Shrubs	1.8	-	-	-	<0.1	-	-

Table 5: Relative percent cover for species in the standing vegetation with cover greater than 5% before (2011) and after regrading (2012-2014). Invasive species are in bold.

	2011	2012		2013		2014	
	June	July	August	July	August	July	August
Alisma triviale Pursh.	3.7	0.8	1.8	3.1	1.7	5.7	8.6
Eleocharis palustris (L.) Roem. & Schult.	-	2.0	10.1	4.7	6.4	6.0	-
Leersia oryzoides (L.) Sw.	13.6	0.5	3.4	6.9	12.5	8.5	13.3
***Myosotis scorpioides* L.**	**18.0**	**0.8**	**0.5**	**-**	**-**	**-**	**0.1**
Potamogeton sp.	10.0	68.3	43.3	25.9	20.4	24.6	13.3
Ranunculus sp.	8.2	0.1	-	-	-	-	-
Sagittaria latifolia Willd.	7.9	22.9	33.7	44.0	44.3	32.7	37.9
Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	-	1.0	3.3	8.3	7.2	11.4	13.4
***Typha* x *glauca* Godr.**	**17.7**	**0.1**	**0.7**	**0.6**	**0.9**	**1.0**	**1.9**

Fig. 1: Aerial photographs of Lieberman; a) before regrading and b) after regrading. The white dotted line indicates the wetland border. Images are from Google Earth 2006 and 2014, respectively. Images captured 19 February 2018.

Fig. 2: Species diversity (H', diamond), species richness (square), and relative % cover of invasive species (triangle) for the standing vegetation before and after regrading.

Fig. 3: Mean % cover for selected species in the standing vegetation before and after regrading. SL = *Sagittaria latifolia* (open diamond), P = *Potamogeton* spp. (closed diamond), ST = *Schoenoplectus tabernaemontani* (closed circle), LO = *Leersia oryzoides* (closed square), AT = *Alisma triviale* (closed triangle), EP = *Eleocharis palustris* (open circle), TxG = *Typha* x *glauca* (open triangle), MS = *Myosotis scorpioides* (open square).

Fig. 1

a.

b.

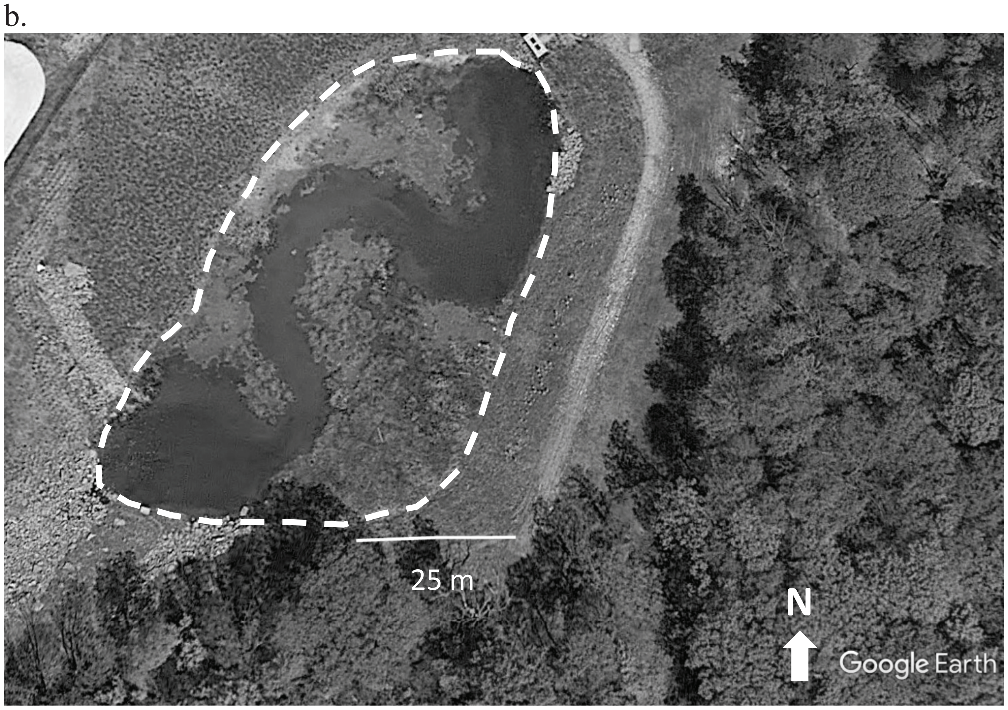


Fig 2.

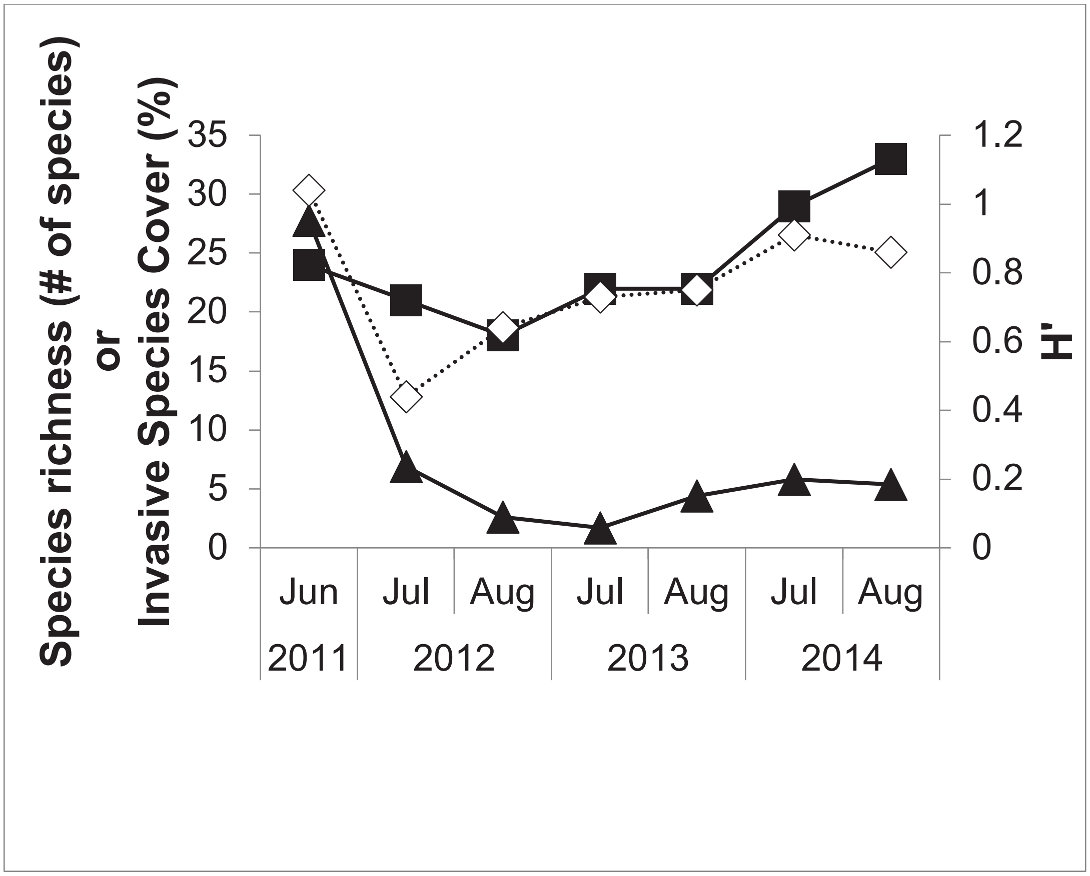
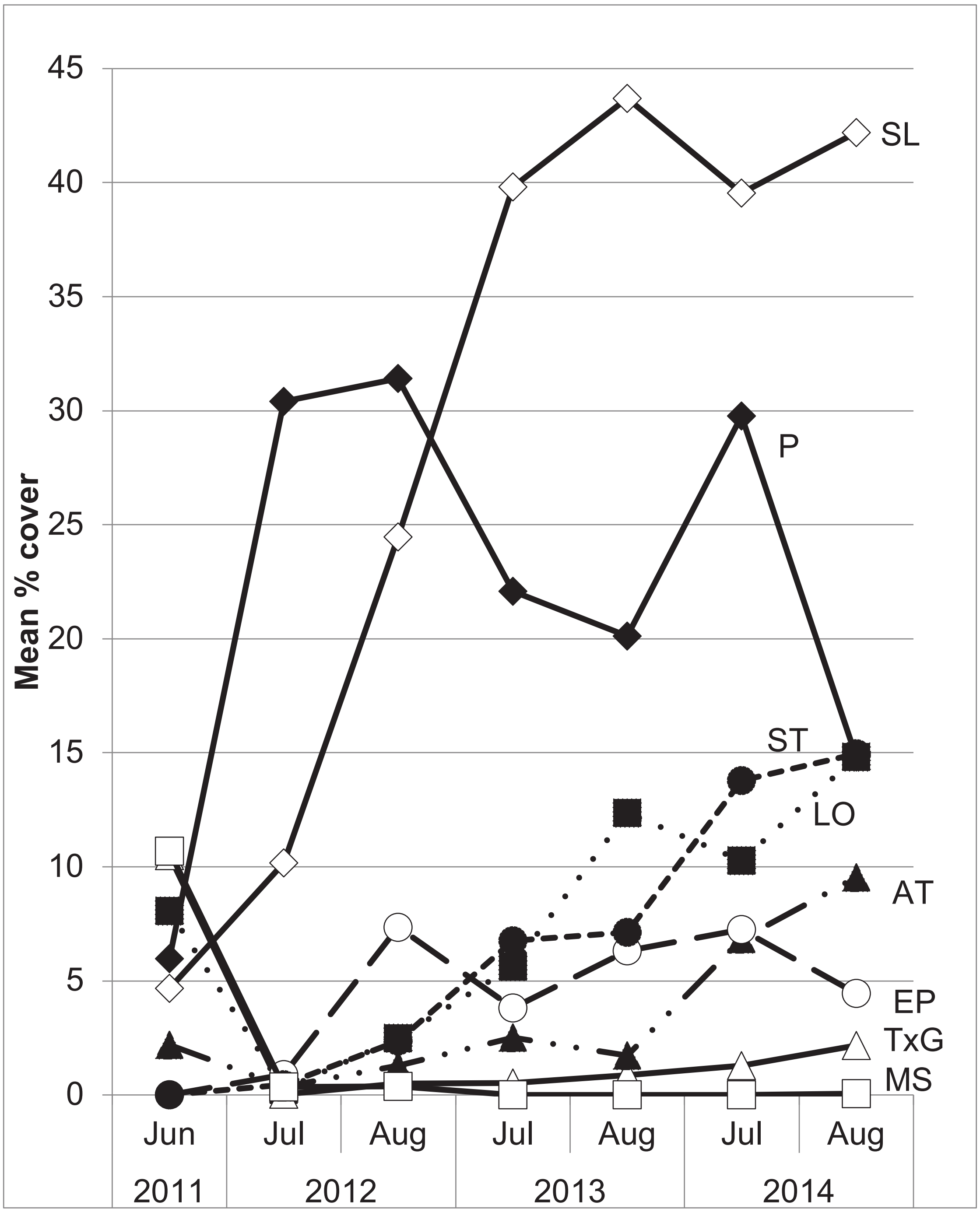

Species richness (# of species)
or
Invasive Species Cover (%)
35
30
25
20
15
10
5
0
1.2
1
0.8
0.6
0.4
0.2
0
H'
Jun
Jul
Aug
Jul
Aug
Jul
Aug
2011
2012
2013
2014

Fig. 3



Mean % cover
45
40
35
30
25
20
15
10
5
0
SL
P
ST
LO
AT
EP
TxG
MS
Jun
Jul
Aug
Jul
Aug
Jul
Aug
2011
2012
2013
2014

Chapter 5: Impacts of urban wetland sediment and flooding regime on relative growth rates of five wetland plant species

Abstract:

Hydrology and sediment characteristics are major determinants of vegetation composition, a key component of urban wetlands. We conducted two experiments to distinguish between sediment and flooding effects: 1) the growth responses of five plant species to the sediment from three different urban wetlands, both *in situ* and at a common garden site (ERF), and 2) a flooding regime study which assessed the growth responses of three wetland plant species to four different flooding regimes: constant drawdown conditions, constant flooded conditions, a treatment mimicking "natural" wetland flooding duration (flooded conditions for 3 days a week), and a treatment mimicking a flashy "urban" wetland flooding duration (flooded conditions for 2 days a week). As predicted, species that were commonly found in urban wetlands (*Juncus effusus, Leersia oryzoides* and *Typha* x *glauca*) generally had higher mean relative growth rates than *Sparganium americanum* and *Carex stricta,* which were not commonly found. We observed that plants had higher relative growth rates at the common garden site than in the wetlands, providing indirect evidence that hydrological variables may have more of an impact on native species establishment and growth in urban wetlands. Our results indicate that different species may vary in their responses to flooding regimes. *Carex stricta* had the highest relative growth rates under drawdown conditions, while the growth rates of *Juncus effusus* were similar regardless of flooding regimes. *Leersia*

oryzoides had the highest growth rates under flooded conditions at ERF, but the highest growth rates were observed in the drawdown treatment in the Research Greenhouse. We would recommend using native species like *Juncus effusus,* and perhaps *Leersia oryzoides*, for urban wetland management projects as both seem to tolerate urban wetland sediment and some flooding conditions.

Introduction:

Urban wetlands experience altered sediment chemistry and flooding regimes due to anthropogenic influences (Forman 2003; Faulkner 2004; Zhu et al. 2008; Pickett et al. 2011). Urbanization has resulted in an increased input of pollutants into aquatic ecosystems (Pankratz et al. 2007; Göbel et al. 2007; Zhu et al. 2008; Gasperi et al. 2012), such as nitrogen (Faulkner 2004; Kasper and Jenkins 2007; Kearney et al. 2013), phosphorus (Kasper and Jenkins 2007; Malaviya and Singh 2012) and metals (Malaviya and Singh 2012). These wetlands also experience a "flashy hydrology" as a result of increased impervious surface cover in urbanized areas (Forman 2003; Ehrenfeld et al. 2003; Pickett et al. 2011), and many are designed to collect and control impervious surface runoff, sediments, and pollution in urban areas (Pankratz et al. 2007; Woodcock et al. 2010). Thus, understanding the roles that sediment and hydrology play in urban wetland ecosystems is crucial.

Vegetation is a key component of these impacted wetlands; plants stabilize sediment and reduce erosion, promote sedimentation and improve water quality (Mitsch and Gosselink 2015), and provide surface area for colonization by microbial communities (Arshad and Frankenberger 1997), which in turn play key roles in the biogeochemical processing of pollutants in wetlands (Faulwetter et al. 2009; Laanbroek 2010). Different species of plants vary in their ecological functions; for example, *Sparganium americanum* and *Juncus effusus* differed substantially in their rates of nutrient accumulation, and retained nutrients in different tissues (Kao et al. 2003). Understanding variables that can impact the establishment and growth of wetland plants can impact the success of urban wetland management and rehabilitation projects.

Flooding regimes are major determinants of vegetation composition in both natural and urban wetlands (Casanova and Brock 2000; Zedler 2000; Keddy 2010; Webb et al. 2012; Mitsch and Gosselink 2015; Campbell et al. 2016). Plant zonation is influenced by depth, duration, and frequency of flooding, and periodic flooding can impact plant community structure (Casanova and Brock 2000). Certain species (e.g. *Sparganium americanum*) may be found in areas with more standing water and tolerate more inundated conditions, while others (e.g. *Leersia oryzoides*) are found under drawdown conditions (Roznere and Titus 2017). *Typha* x *glauca* tolerated the varying hydroperiod and frequent flooding of an urban wetland in Wisconsin, while the seedlings of *Carex* spp., including *C. stricta,* often died (Hall and Zedler 2010). Species richness, plant cover, and aboveground biomass were the highest under shorter periods of flooding, indicating that duration may be an important factor in determining wetland plant composition (Campbell et al. 2016). Overall, flooding may reduce plant biodiversity (Peterson and Baldwin 2004; De Jager et al. 2012) and plant growth (Lenssen et al. 1999; Webb et al. 2012; Campbell et al. 2016), thus flooding regimes may need to be appropriately managed in terms of flooding depth and duration to positively affect plant communities.

Hydrological factors and sediment characteristics are often linked. For example, flooding can change a phosphorus-limited system to one that is nitrogen-limited, but this relationship depends on the sediment type (Saaltink et al. 2018). Accumulation of organic matter (Lenssen et al. 1999), nitrogen levels (Kearney and Zhu 2012), redox conditions (Pezeshki 2001; Mossman et al. 2012), and the presence of pollutants (Deng et al. 2006; Kearney and Zhu 2012) can impact wetland plant growth. For example, *Typha* x *glauca* responds to nutrient (N and P) enrichment with increased ramet density, height, and biomass, outcompeting native *Carex* species (Woo and Zedler 2002). Invasive plants, like *Typha* x *glauca* and *Phragmites australis*, may be more tolerant of elevated salt levels in urban environments than native plants (Zedler and Kercher 2004; Vasquez et al. 2005). Understanding plant tolerances to urban wetland sediment and hydrological conditions is important for wetland rehabilitation or creation projects in urban landscapes, as certain species may increase the likelihood of successful native vegetation establishment.

This chapter aims to differentiate between the effects of impacted urban sediment and flashy urban flooding regimes on the growth rates of five wetland plant species. We conducted two experiments to distinguish between sediment and flooding effects on plant growth rates: 1) an urban wetland sediment study and 2) a flooding regime study. The first study examined growth responses of four plant species to the sediment from three different urban wetlands. The plants were grown *in situ* and at a common garden site. We hypothesized that plants would have higher relative growth rates at our common garden site because plants grown *in situ* would experience a harsher environment: periods of drought, flashy flooding regimes, and potentially more herbivory. We also hypothesized that, based on the sediment characteristics discussed in Larson et al. (2016), plants would have higher relative growth rates in urban sediment with the highest availability of ammonium. Plants would have the lowest growth rates in sediment with a relatively lower amount of available nitrogen and the highest soil electrical conductivity. The second experiment assessed the growth responses of three wetland plant species to different flooding regimes. We expected that plants would generally favor drawdown conditions, as opposed to constantly flooded conditions. We also predicted that species that were commonly found in urban wetlands (*Typha* x *glauca*, *Juncus effusus,* and *Leersia oryzoides*) would have higher growth rates than species that were uncommon (*Carex stricta* and *Sparganium americanum*) for both experiments.

Methods:

Study species and common garden site description

We selected five species based on our results from an urban wetland plant survey in the Southern Tier of upstate New York (Larson et al. 2016) to test the effects of impacted urban sediment and altered urban hydrologies on plant growth rates: *Carex stricta, Juncus effusus, Leersia oryzoides, Sparganium americanum*, and the invasive *Typha* x *glauca*. These are a mix of species that were common in our survey (*T.* x *glauca, J. effusus,* and *L. oryzoides*) and uncommon (*C. stricta* and *Sparganium americanum*). *Typha* x *glauca, J. effusus,* and *L. orzyzoides,* were all found in more than four of the eight urban wetlands, whereas *C. stricta* and *Sparganium americanum* occurred at one site. Plants were purchased from the Southern Tier Consulting, Inc. in West Clarksville, NY, with the exception of *Typha* x *glauca*. All purchased plants were planted as bare roots, except for *Carex stricta* in the sediment experiment, which were planted as plugs.

We collected rhizomes of *Typha* x *glauca* with single ramets in late May 2012 from the Binghamton University Nature Preserve in Vestal, NY.

All common garden experiments were conducted at the Binghamton University Ecological Research Center (ERF), located in the Binghamton University Nature Preserve. The facility is enclosed by a 3m fence. All plants were watered with tap water. *In situ* plots were cleared of standing vegetation and regularly weeded to reduce shading and competition from other plants.

Sediment experimental set-up:

We collected sediment from three wetlands: Sites 4, 6, and 7 (Larson et al. 2016). We chose these wetlands so that we had a range in available nitrogen and soil electrical conductivity values. Site 6 had the highest average extractable ammonium nitrogen (27.5 mg NH_4-N kg^{-1}), while Site 4 had the lowest (5.9 mg NH_4-N kg^{-1}). Site 7 ammonium values were between these extremes (17.5 mg NH_4-N kg^{-1}), but the site had the highest soil electrical conductivity (6380 $\mu S\ cm^{-1}$). Sites 4 and 6 had much lower soil electrical conductivity values: 173.3 $\mu S\ cm^{-1}$ and 144.7 $\mu S\ cm^{-1}$, respectively. For the purposes of this chapter, we will refer to Site 4 as "sediment with low ammonium availability," Site 6 as "sediment with high ammonium availability," and Site 7 as "sediment with high soil electrical conductivity."

We planted four species at the end of May 2012: *Typha* x *glauca, Juncus effusus, Sparganium americanum,* and *Carex stricta*, both *in situ* at each wetland site and at ERF. Each site had six replicate blocks, consisting of one of each plant species randomly positioned in a row. Plants were grown in 5.3 L pots lined with plastic bags to contain sediment and roots. Pots at the common garden site were regularly watered with tap water. Plants in the wetlands were left to natural watering events (rain and flooding). All plants were monitored and weeded once a week over the growing season. Plants were harvested at the end of August 2012.

Flooding regimes experimental set-up

To test for growth responses to various flooding regimes, we subjected plants to four flooding duration treatments:

• Drawdown (never flooded) – water level 5cm below sediment level

• Flooded (always flooded) – water level 15cm above sediment level

• "Natural" treatment, where plants were flooded once a week (15cm above sediment level) for 3 days, and kept under drawdown conditions for the remainder of the week

• "Urban" treatment, where plants were flooded once a week (15cm above sediment level) for 2 days, and kept under drawdown conditions for the remainder of the week

This study was conducted at two locations: 1) ERF (five replicated blocks), and 2) the Research Greenhouse at the Binghamton University (eight replicated blocks). The Research Greenhouse allowed us to control for temperature and reduce herbivory. For this experiment, we focused on three native wetland plant species: *Carex stricta, Juncus effusus,* and *Leersia oryzoides*. Plants were grown in sediment collected from Lake Lieberman, an urban retention pond located on Binghamton University campus. Plants were grown in 5.3 L pots and lined with plastic bags to contain sediment and roots. Pots at the common garden site were stored in 19 L plastic buckets and watered, based on their treatment, with tap water. Pots in the Research Greenhouse were stored in eight 1200 L fiberglass tanks; the RO water was kept at a constant height within each tank, and pots were placed on pavers to allow for the drawdown conditions. Flooded plants were kept under the water level and on the bottom of the tanks for the duration of the experiment, while plants treated with natural and urban flooding regimes were moved from pavers to the bottom of the tanks throughout on their cycles. Water temperatures were maintained at 23°C by refrigerated circulators (CFF-500, Remcor, Franklin Park, IL., U.S.A.). Plants grown at ERF were planted in May 2013 and harvested in August 2013, while those grown in the Research Greenhouse were planted in July 2013 and harvested in September 2013.

Data Analyses

We analyzed plant growth by recording root and shoot biomass, then calculating relative growth rates (RGR, Equation 1) for each species. After harvest, plants were divided into aboveground and belowground tissue, and dried at 60° C to a constant weight.

Eq. 1: RGR= (ln FW – ln IW)/# growing days
ln- natural logarithm
FW- Dry weight of experimental plant at the end of the study
IW- Average dry weight of "initial" plants randomly selected at the beginning of the study

We analyzed RGR data for the sediment experiment (*Typha* x *glauca, Juncus effusus,* and *Sparganium americanum*) using t-tests for each species to test for the effect of plants grown at ERF versus those grown *in situ*. One-way ANOVAS were used to test the effects of sediment type on mean relative growth rates for *Juncus effusus, Typha* x *glauca,* and *Sparganium americanum*. We analyzed final biomass differences of *Carex stricta* between growth sites using t-tests because we could not obtain accurate initial biomass measurements with plugs. Flooding regime experimental data were analyzed using a single-factor Analysis of Variance, separately for each species. Flooding regime data are presented separately for plants grown at ERF versus those grown in the Research Greenhouse. Significant results from the ANOVA tests were further analyzed with Tukey's HSD test to determine which groups were different from each other with a $p < 0.05$. Statistical tests were run using VassarStats (6 July 2018).

Results

Urban wetland sediment experiment

Plants generally had higher relative growth rates at the common garden site (ERF) than in the corresponding wetlands (Figure 1). *Juncus effusus* consistently had the highest mean relative growth rates than the other two species, regardless of sediment type, while *Sparganium americanum* typically had the lowest. There was no significant difference in mean relative growth rates for *J. effusus* among sediment types (Table 1). We found significant differences in *S. americanum* growth responses to sediment types, with a low mean relative growth rate when grown in sediment with high soil electrical conductivity (Table 2; one-way ANOVA $F_{2,13} = 4.98$, $p = 0.0248$). According to VassarStats, a Tukey HSD test revealed that sediment treatments were not significantly different from one another. Similarly, *Typha* x *glauca* exhibited significantly different growth responses to the three sediment types, although the lowest mean relative growth rate was observed in the sediment with high ammonium availability (Table 3; one-way ANOVA $F_{2,12} = 6.94$, $p = 0.0099$).

All three species had significantly higher mean relative growth rates in the sediment with low ammonium availability when planted at ERF versus *in situ* (Figure 1a;

J. effusus p = 0.009, *S. americanum* p = 0.003, *T.* x *glauca* p < 0.0001). *Juncus effusus* and *Sparganium americanum,* when grown in sediment with high ammonium availability, had significantly higher relative growth rates when grown at ERF than in the corresponding wetlands (p = 0.023 and p = 0.002, respectively); however, the mean relative growth rates for *Typha* x *glauca* were not significantly different between ERF and *in situ* (Figure 1b). Plants grown in sediment with high soil electrical conductivity did not show a significant difference in mean relative growth rates between planting sites (Figure 1c). *Sparganium americanum* treated with sediment with high soil electrical conductivity had very low relative growth rates at ERF, while those planted *in situ* died.

The biomass for *Carex stricta* was not significantly different between the common garden site and *in situ* sites, regardless of sediment type. The highest mean biomass for *C. stricta* was in the sediment with low ammonium availability (5.9 g dry weight and 5.3 g dry weight when grown at ERF and *in situ*, respectively). Mean biomass for sediment with high ammonium availability and sediment with high soil electrical conductivity ranged from 3.6 g dry weight (n = 6) to 1.5 g dry weight (n = 2).

Flooding regime experiment: ERF

According to VassarStats, we did not observe any significant trends in growth rates among treatments for any species, although we can infer patterns of plant growth responses to flooding durations based on mean relative growth rates. *Carex stricta* had the highest mean relative growth rate in the drawdown treatment, but negligible (urban) or negative (flooded and natural regimes) growth rates in the other treatments (Figure 2a). Mean relative growth rates for *Juncus effusus* were not influenced by treatments (Figure

2b). *Leersia oryzoides* had the highest growth rates under flooded conditions (Figure 2c), with the lowest mean relative growth rates exhibited in the drawdown treatment.

Flooding regime experiment: Research Greenhouse

Carex stricta had a significantly higher mean relative growth rate in the drawdown treatment, but experienced a net biomass loss in all treatments with periods of flooding (Table 4; one-way ANOVA $F_{3,27} = 15.44$, $p < 0.0001$). According to VassarStats, a Tukey HSD test revealed that all treatments were significantly different from one another (Figure 3a). As in the experiment at ERF, *Juncus effusus* had similar mean relative growth rates for all treatments (Figure 3b). Although the mean relative growth rates were not statistically significant, *Leersia oryzoides* had the highest growth rates under drawdown conditions, which is a stark contrast to our results from those grown at ERF (Figure 3c).

Discussion:

As expected, species that were commonly found in urban wetlands (*Juncus effusus, Leersia oryzoides* and *Typha* x *glauca*) generally had higher mean relative growth rates than *Sparganium americanum* and *Carex stricta,* which were not commonly found. We originally predicted that plants grown in the sediment with the highest extractable ammonium nitrogen would have the highest growth rates because of a relatively high availability of ammonium, however, no species exhibited significantly higher growth rates in this treatment. *Sparganium americanum* was the only species that seemed to be sensitive to sediment with a high sediment electrical conductivity (Site 7). Little is known about the tolerance of *S. americanum* to salt contamination, but this species may not be found in urban wetlands due to the accumulation of road salt from impervious surface runoff (Miklovic and Galatowitsch 2005). Because wetland plants had significantly higher growth rates at the common garden site than those grown *in situ*, other hydrological variables may have more of an impact on native species establishment and growth in urban wetlands.

Carex stricta consistently exhibited the highest relative growth rates under drawdown conditions. As a tussock sedge predominantly found in emergent wetlands, this pattern may reflect *C. stricta*'s limited tolerance for flooded conditions. *Carex* species are often uncommon in urban wetlands, occasionally establishing on higher microtopography that is less likely to be flooded by impervious surface runoff. This observation has been observed in *C. schidtii* (Yan et al. 2015) and *C. stipata* (Magee and Kentula 2005), while increased sedimentation from flooding treatments decreased the mean biomass for *C. stipata* and *C. rostrata* (Ewing 1996). Kercher and Zedler (2004) found that *C. stricta, C. granularis,* and *C. canadensis* were sensitive to various flooding regimes. Other research indicates that *C. stricta* seedlings may be sensitive to flooding conditions, but the plants are tolerant of varying flooding regimes once established (Budelsky and Galatowitsch 2004).

Juncus effusus had similar mean relative growth rates for every hydrological treatment, indicating that this species may be tolerant of a variety of flooding conditions. Roznere and Titus (2017) found that *J. effusus* exhibited random distributions in relation to water depth, suggesting that their dominance may be at least partially attributed to their tolerance of varying water levels, and Magee and Kentula (2005) found that *J. effusus* occupied habitats with high water level variability. However, Magee and Kentula (2005) also observed *J. effusus* most commonly in habitats with lower flooding durations, so more information is needed to understand the growth responses of *J. effusus* to urban flooding regimes. Additionally, other studies indicate that some *Juncus* species are efficient at removing pollutants from impacted wetlands, and thus may be tolerant of impacted urban wetland sediment (Syranidou et al. 2017). This may explain why *J. effusus* consistently had the highest mean relative growth rates for all urban wetland sediment types.

We found that *Leersia oryzoides* was inconsistent in its response to flooded treatments. When grown at ERF, *L. oryzoides* had highest relative growth rates under flooded conditions, with the lowest mean relative growth observed in the drawdown treatment. However, we observed the opposite pattern in the Research Greenhouse, with lowest mean relative growth rates in the flooded treatment. This inconsistent pattern makes it difficult to distinguish a growth response for *L. oryzoides*. Magee and Kentula (2005) observed that *L. oryzoides* was most commonly found in conditions similar to *J. effusus*: saturated soils with high variation in water levels, while Roznere and Titus (2017) observed that *L. oryzoides* was found on substrate not far above the water level. In contrast, Pierce et al. (2009) found that overall productivity of *L. oryzoides* was unaffected by flooding treatments, with highest aboveground biomass in saturated (flooded) conditions, while further experiments indicated that *L. oryzoides* may accumulate the most biomass under intermittent flooding regimes (Koontz and Pezeshki 2011). More information is needed to understand the growth responses of *L. oryzoides* to flooding regimes.

Future research

Based on our data, it is hard to distinguish a pattern between natural and urban treatments, perhaps because the treatments are so similar. Future research should focus on treatments that more accurately represent and distinguish hydrological regimes of natural and urban wetlands. Ideally, we would determine three hydrological parameters of multiple urban wetlands: flooding duration, stage (water level) height, and frequency. Because urban wetlands receive runoff from impervious surfaces, we expect that urban stormwater retention wetlands may have higher water levels during peak storm events, and longer flooding durations due to more water entering the system and held within stormwater retention wetlands. These systems may also experience more frequent flooding, as even small storm events could result in a large amount of impervious surface runoff, while these smaller storms may not impact natural wetlands with such intensity. With these data, we can better model flooding regimes that reflect urban wetlands, and distinguish them from those of natural wetlands.

Conclusions:

We found that although native plants can tolerate impacted urban wetland sediment, different species varied in their responses to flooding regimes. We would recommend planting species like *Juncus effusus* and perhaps *Leersia oryzoides*, as both seem to tolerate urban wetland sediment and some flooding conditions. We also caution restoration ecologists against using *Carex stricta* or *Sparganium americanum* in urban wetland planting schemes, as these may be sensitive to urban wetland sediment and hydrology. Managers should consider planting species that align with a project's hydrological attributes.

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Table 1: One-way ANOVA results for the mean relative growth rates for *Juncus effusus,* grown at ERF, among the three sediment types.

	df	SS	MS	F-value	P-value
Treatment	2	0.000119	0.000059	1.66	0.2255
Residuals	14	0.0005	0.000036		

Table 2: One-way ANOVA results for the mean relative growth rates for *Sparganium americanum*, grown at ERF, among the three sediment types.

	df	SS	MS	F-value	P-value
Treatment	2	0.00228	0.000114	4.98	0.0248
Residuals	13	0.000298	0.000023		

Table 3: One-way ANOVA results for the mean relative growth rates for *Typha* x *glauca,* grown at ERF, among the three sediment types.

	df	SS	MS	F-value	P-value
Treatment	2	0.000158	0.000079	6.94	0.0099
Residuals	12	0.000136	0.000011		

Table 4: One-way ANOVA results for the mean relative growth rates for *Carex stricta* among flooding treatments when grown in the Binghamton University Research Greenhouse.

	df	SS	MS	F-value	P-value
Treatment	3	0.000151	0.00005	15.44	< 0.0001
Residuals	27	0.000088	0.000003		

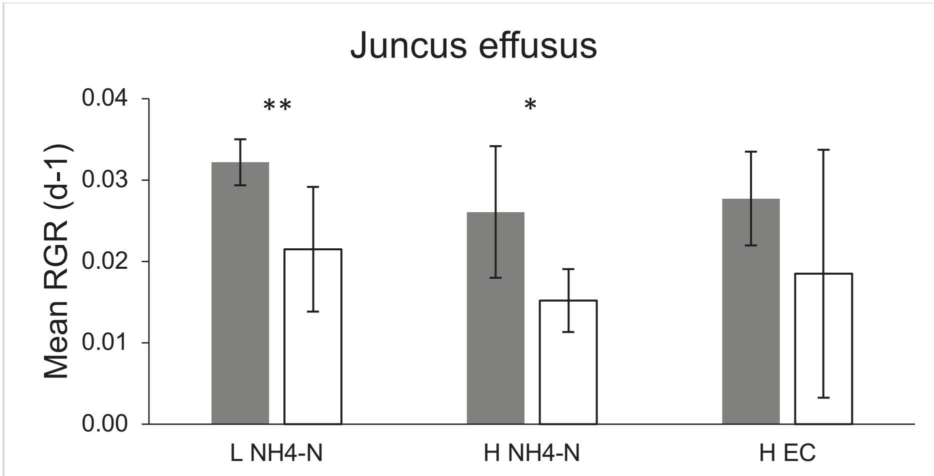
Figure 1: Relative growth rates (day $^{-1}$) for three wetland species: a) *Juncus effusus,* b) *Sparganium americanum,* and c) *Typha* x *glauca* planted at the common garden site (ERF, gray bars) and *in situ* (white bars) for sediment collected from three wetlands: low ammonium availability (L NH_4-N), high ammonium availability (H NH_4-N), and high soil electrical conductivity (H EC). Means show the standard deviations (n = 3-6). All *S. americanum* died *in situ* at Site 7 (high soil electrical conductivity). Significant t-test results designated by *($p < 0.05$) or ** ($p < 0.01$). Error bars indicate standard deviations.

Figure 2: Mean relative growth rates for a) *Carex stricta*, b) *Juncus effusus*, and c) *Leersia oryzoides* for the flooding regime experiment at ERF. Error bars indicate standard errors. DD = drawdown treatment, FL = flooded treatment, N = "natural" wetland flooding duration (flooded conditions for 3 days), and U = "urban" wetland flooding duration (flooded conditions for 2 days).

Figure 3: Mean relative growth rates for a) *Carex stricta*, b) *Juncus effusus*, and c) *Leersia oryzoides* for the flooding regime experiment in the Research Greenhouse. Error bars indicate standard errors. DD = drawdown treatment, FL = flooded treatment, N = "natural" wetland flooding duration (flooded conditions for 3 days), and U = "urban" wetland flooding duration (flooded conditions for 2 days).

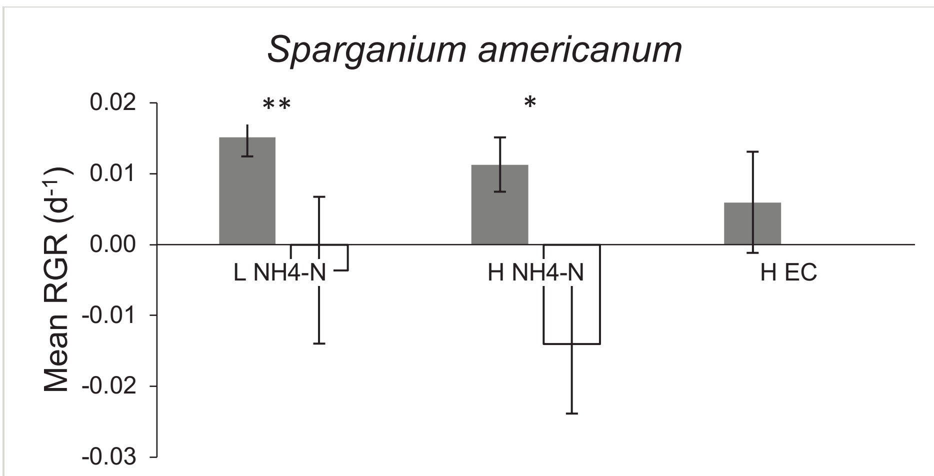
Figure 1:

a.



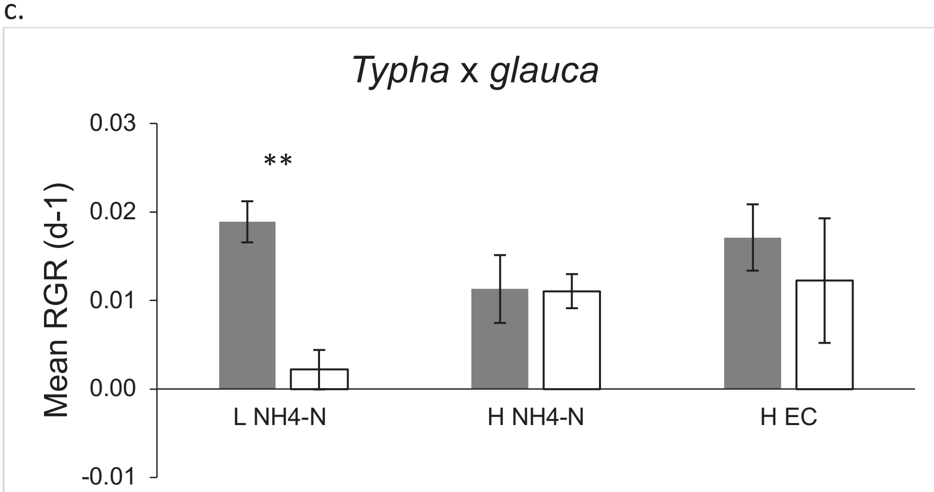
Juncus effusus
Mean RGR (d-1)
0.04
0.03
0.02
0.01
0.00
**
*
L NH4-N
H NH4-N
H EC

b.



Sparganium americanum
Mean RGR (d-1)
0.02
0.01
0.00
-0.01
-0.02
-0.03
**
*
L NH4-N
H NH4-N
H EC

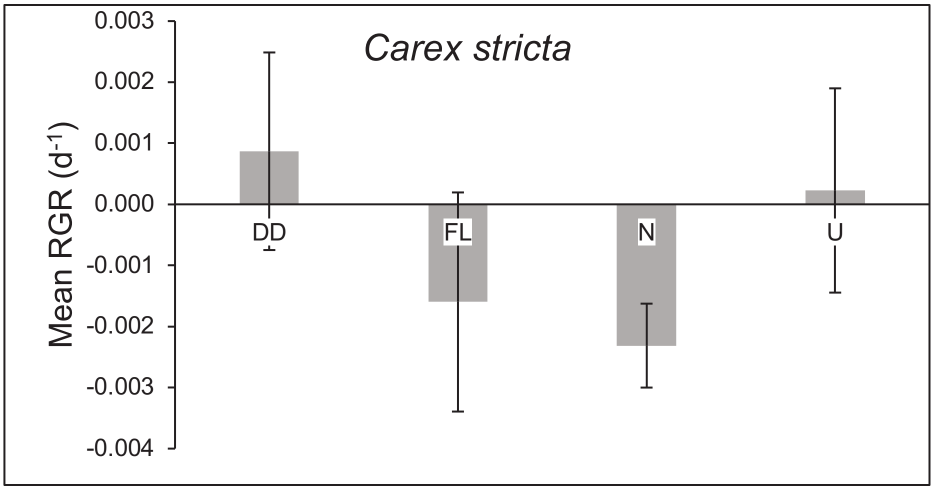
c.



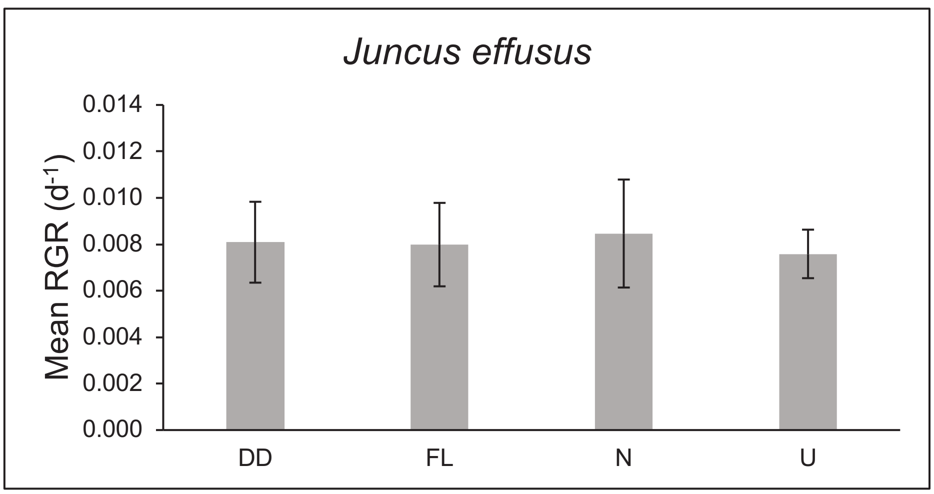
Typha x glauca
Mean RGR (d-1)
0.03
0.02
0.01
0.00
-0.01
**
L NH4-N
H NH4-N
H EC

Figure 2:

a.


Carex stricta
Mean RGR (d^{-1})
0.003
0.002
0.001
0.000
-0.001
-0.002
-0.003
-0.004
DD
FL
N
U

b.


Juncus effusus
Mean RGR (d^{-1})
0.014
0.012
0.010
0.008
0.006
0.004
0.002
0.000
DD
FL
N
U

c.

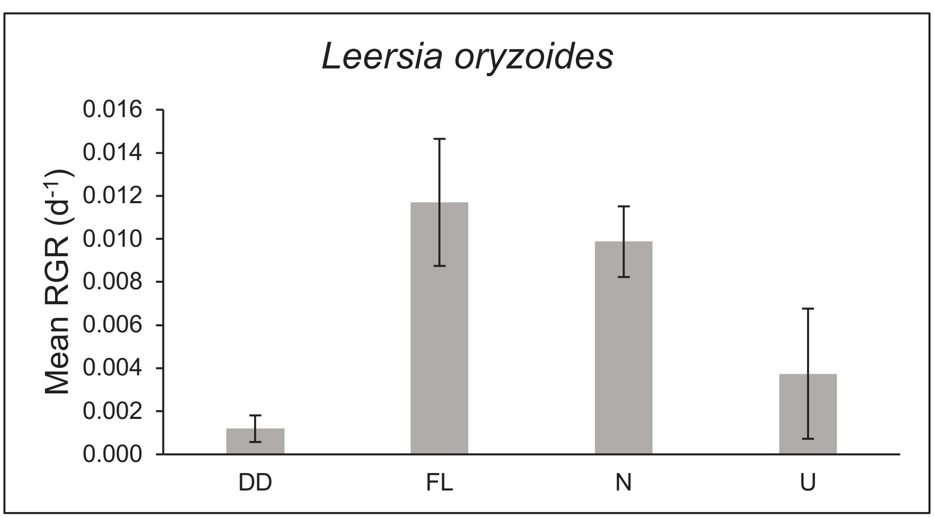
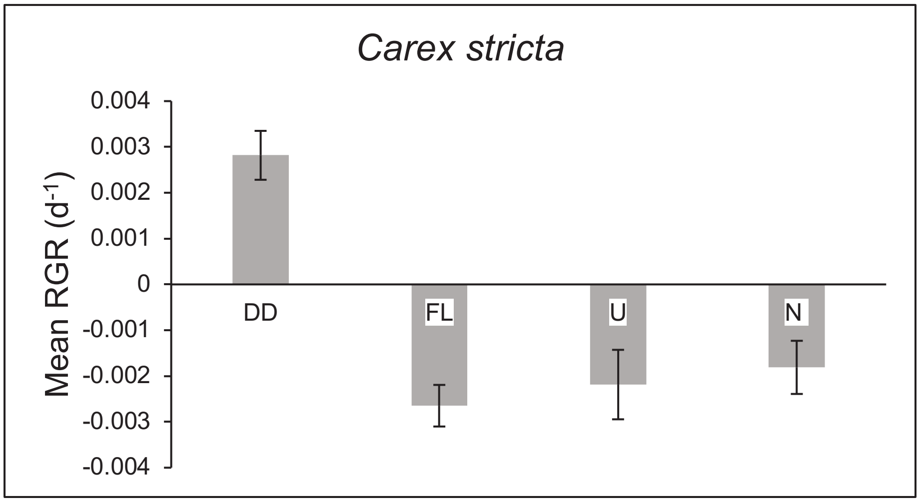
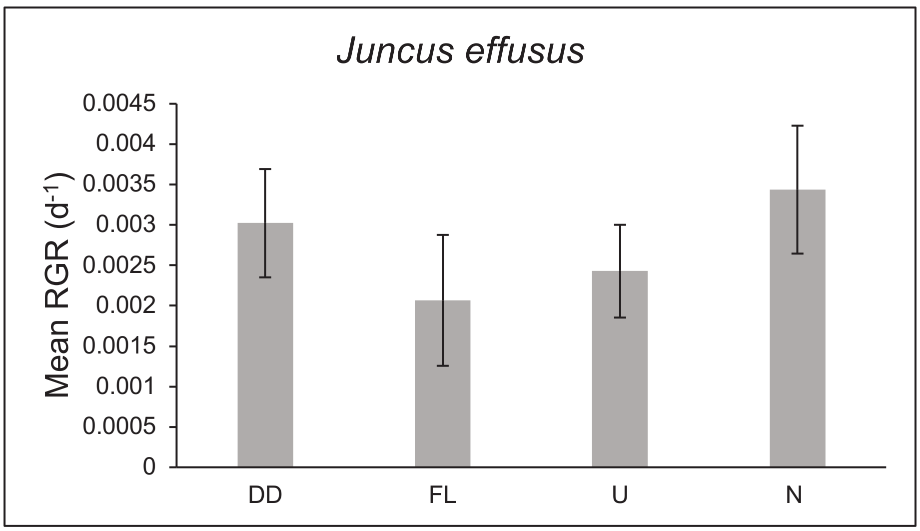

Leersia oryzoides
Mean RGR (d^{-1})
0.016
0.014
0.012
0.010
0.008
0.006
0.004
0.002
0.000
DD
FL
N
U

Figure 3:

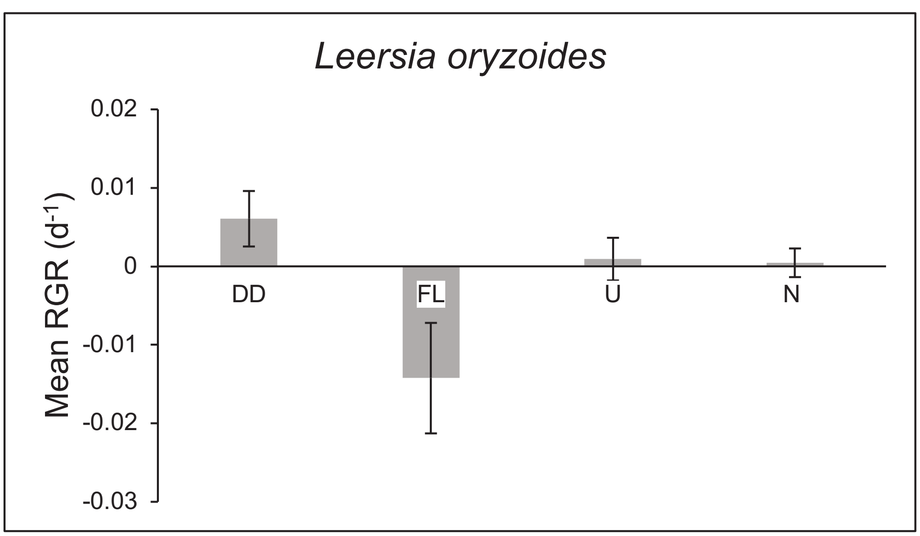
a.


Carex stricta
Mean RGR (d-1)
0.004
0.003
0.002
0.001
0
-0.001
-0.002
-0.003
-0.004
DD
FL
U
N

b.


Juncus effusus
Mean RGR (d-1)
0.0045
0.004
0.0035
0.003
0.0025
0.002
0.0015
0.001
0.0005
0
DD
FL
U
N

c.


Leersia oryzoides
Mean RGR (d-1)
0.02
0.01
0
-0.01
-0.02
-0.03
DD
FL
U
N

Chapter 6: Conclusions

Urban wetlands differ from natural wetlands

This dissertation sheds light on the plant communities of urban wetlands and their importance to wetland rehabilitation success. Our work indicates that urban wetlands have different plant communities than natural wetlands; thus, urban wetland rehabilitation may need to be managed differently than traditional restoration. Urban wetland vegetation and soil characteristics are different from those in nearby natural wetlands, and our increased knowledge of these urban ecosystems will lead to more successful urban restoration and creation projects.

Urban wetlands had significantly lower species richness and a higher percent cover of invasives, including *Typha* x *glauca, Phalaris arundinacea,* and *Lythrum salicaria,* than natural wetlands (Larson et al. 2016, Chapter 2). Native species, including *Leersia oryzoides, Ludwigia palustris,* and *Sagittaria latifolia*, were also common. These urban wetlands most closely resembled emergent wetlands in their vegetation composition, likely due to a high percentage of herbaceous obligate wetland species. Soil pH and soil electrical conductivity were significantly higher in urban sites, but potential net N-mineralization rates were significantly lower. Urban wetland construction projects need to be especially mindful of invasive species, as improper management practices could lead to dominance of a few species (Zedler 2000). However, this chapter illustrates that native species can establish and thrive in urban wetlands, and these species should be included in planting or seeding plans to reduce the likelihood of invasive species dominance.

Urban wetland seed bank characteristics

Understanding urban wetland seed bank characteristics can also increase the success of urban wetland rehabilitation projects. We found high spatial variation in species richness and seedling density among the four sites (Larson and Titus 2018, Chapter 3). These seed banks were dominated by obligate wetland species. Like the standing vegetation in most urban wetlands (Larson et al. 2016, Chapter 2), invasive species comprised a high percentage of seedlings for three wetlands (40.8% – 80.9%), but not for Site 1 (4.2%). *Lythrum salicaria*, *Typha* sp., and the native *Ludwigia palustris* were common species based on relative seedling density for three seed banks, while *Leersia oryzoides, Schoenoplectus tabernaemontani,* and *Alisma triviale* were common species in Lieberman. Because these seed banks were often dominated by invasive species, managers may need to consider supplemental plantings to reduce early establishment of invasive species. Our findings illustrate that seed banks may be viable and can contribute to the revegetation of disturbed urban sites. Given that invasive species, specifically *Lythrum salicaria* and *Typha* sp*.,* are common species in our urban wetland seed banks, supplemental planting of native species should be considered for wetland construction projects.

A major habitat alteration impacted the plant community of an urban wetland

We were able to evaluate the influence of seed banks and standing vegetation by documenting the effects of a major habitat alteration on recovering vegetation (Larson et al., under revision). Regrading disrupted both the seed banks and standing vegetation in Lieberman, theoretically limiting potential revegetation success. Seedling densities before the regrade were nearly three-fold greater than those after regrading, and seedling density significantly decreased in the drawdown treatment. Species richness in the standing vegetation decreased immediately after the regrade, but rebounded three years after the regrade. Information about the seed bank composition and standing vegetation before the regrade was not sufficient to make predictions about the recovering vegetation, likely because we did not include asexual propagules in our assessment. This study indicates that a regrading project can substantially reduce seedling density of an urban wetland seed bank, but standing vegetation may show signs of recovery within a short time span due to the presence of a prolific bud bank. In other words, in order to make predictions about recovering vegetation, managers should evaluate the seed bank and the bud bank as potential propagule sources.

Recommended species for urban wetland rehabilitation projects

Our understanding of urban wetland plant communities indicates that certain native species may be more tolerant of urban wetland conditions, such as contaminated sediment and flashy hydrologies. We also noted that some native species were common in urban wetlands (*Juncus effusus* and *Leersia oryzoides*), while others were relatively uncommon (*Carex stricta* and *Sparganium americanum*). We found that although native plants can tolerate impacted urban wetland sediment, different species vary in their growth responses to flooding regimes (Chapter 5), as observed in other studies (e.g., Magee and Kentula 2005; Roznere and Titus 2017). We would recommend planting species like *Juncus effusus* and perhaps *Leersia oryzoides*, as both seem to tolerate urban wetland sediment and some flooding conditions. We also caution restoration ecologists against using *Carex stricta* or *Sparganium americanum* in urban wetland planting schemes, as these may not establish or survive under urban wetland conditions. More broadly, managers should consider planting species that align with a project's hydrological attributes, as hydrology likely plays a fundamental role in plant establishment and survival in urban wetlands.

Applied management implications for urban wetland construction projects

There are two approaches used in restoration projects: 1) a "self-designed" approach and 2) a heavily engineered or "designed" project plan (Galatowitsch and van der Valk 1996; Mitsch and Wilson 1996; Mitsch et al. 1998; Zedler 2000; Mitsch and Gosselink 2015). A self-designed approach emphasizes a "build it and they will come" mentality. These projects involve restoring or constructing the appropriate wetland hydrology and sediment components, but allow the plant community to establish passively. Designed restoration projects, however, require not only careful planning of the site's hydrology and sediment components, but also the vegetation; these plans include supplemental planting or seeding (e.g., Galatowitsch 2006). The regrading and subsequent passive revegetation of Lieberman supports the theory that self-designed wetlands can quickly establish a plant community within three growing seasons. Based on our data, no supplemental planting or seeding was required to successfully revegetate the wetland (Larson et al., under review).

However, our work also highlights the presence of invasive species in both the standing vegetation and seed banks, and other urban rehabilitation projects or habitat alterations could be impacted by the presence of invasive species. Managers who are concerned about invasive species establishment may consider a more designed approach that includes supplemental planting or seeding of native species. Our research demonstrates that native species can establish and thrive in urban wetlands, and these species should be considered in the planting schemes of designed wetlands (Larson et al. 2016; Larson and Titus 2018).

We can recommend native plant species to include in restoration projects to increase the likelihood of successful plant establishment and reduce the risk of invasive species dominance by understanding the hydrology of potential rehabilitation sites. Our research indicates that proper hydrology is critical to support desired plant communities. For example, we learned that Lieberman was noticeably more inundated after the regrading project, and this new feature may have altered the seed bank and the standing vegetation; in particular, we noticed an increase in obligate wetland species in the seed bank after the regrade, as well as the establishment of unexpected common species (e.g., *Sagittaria latifolia*, *Potamogeton* sp., and *Schoenoplectus tabernaemontani*) that prefer inundated conditions in the standing vegetation (Larson et al., under review). Changes in the seed bank and standing vegetation after the regrading project may be a consequence of altered environmental variables after the reconstruction. Our experimental data also indicate that hydrology likely played a significant role in these changes.

When designing an urban wetland, I would first recommend the use of seeding mixes that meet the goals of the new habitat. These seed mixes would include a variety of species and various functional groups to maximize the likelihood of a diverse plant community. We found, over the course of many experiments, that planting bare root or plugs often resulted in transplant shock and plant mortality. For example, supplemental planting in Lake Lieberman after the regrading in 2012 ended in the death of all 250 plants (Larson, unpublished data). Urban wetlands can be harsh environments due to their impacted sediment and altered hydrology, and direct seeding may allow for an "environmental sieve" to select for plants to establish that can handle urban conditions.

Variation among urban wetlands

Finally, we observed substantial variation in the plant communities of urban wetlands in Broome County, NY. While invasive species were common in most of our urban wetlands, native species were dominant in one of our sites. Site 8 (Cutler Pond), the wetland bordering a natural kettle hole, was dominated by native species (*Decodon verticillatus* and *Nuphar variegata*) that were not observed in any other urban wetland, and *Typha* species were noticeably absent. We suspect that hydrologic characteristics of this wetland resulted in a distinctive assemblage of plant species. Moreover, Site 8 serves as an example that not all urban wetlands are dominated by invasive species (Larson et al. 2016). We also observed that the seed bank of Lieberman was mostly comprised of native species, a stark contrast to the other urban wetland seed banks (Larson and Titus 2018). Restoration ecologists and managers of urban wetland construction projects need to evaluate the specific vegetation characteristics and environmental variables of a site to determine an appropriate planting scheme.

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Zedler JB (2000) Progress in wetland restoration ecology. *Trends Ecol Evolut* 15(10): 402–407.

Appendix A: Maps for eight urban wetland sites.

Figure A1: Aerial photograph of Site 1 (Lieberman). Imagery date: 31 March 2006 and copyright 2018 New York GIS. Image obtained using Google Earth 9 August 2018.

Figure A2: Aerial photograph of Site 2. Image obtained using Google Earth 9 August 2018.

Figure A3: Aerial photograph of Site 3. Image obtained using Google Earth 9 August 2018.

Figure A4: Aerial photograph of Site 4. Image obtained using Google Earth 9 August 2018.

Figure A5: Aerial photograph of Site 5. Image obtained using Google Earth 9 August 2018.

Figure A6: Aerial photograph of Site 6. Image obtained using Google Earth 9 August 2018.

Figure A7: Aerial photograph of Site 7. Image obtained using Google Earth 9 August 2018.

Figure A8: Aerial photograph of Site 8. Image obtained using Google Earth 9 August 2018.

Figure A1:

Figure A2:


Cyndy St
Jerry St
Harry L Dr
Valley Plaza
Wilkins Way
Watson Blvd
142 m
N
Google Earth

Figure A3:

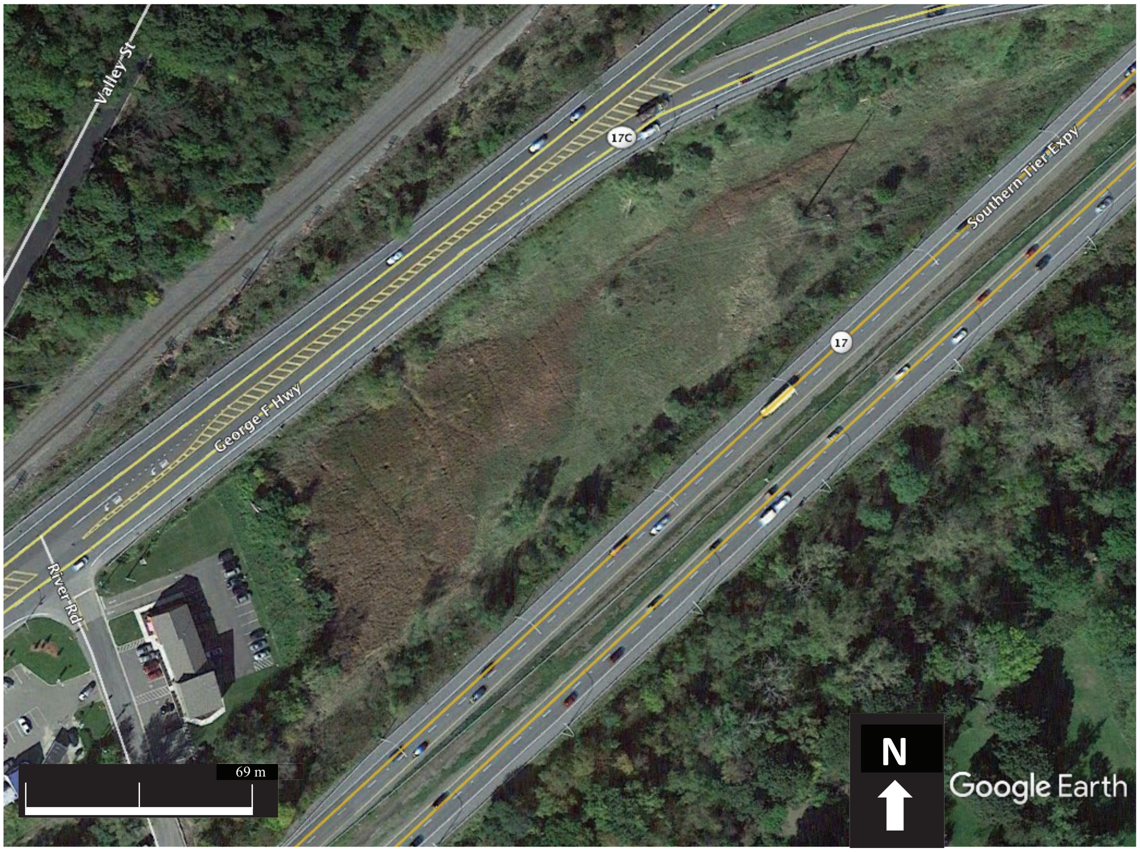

Valley St
17C
Southern Tier Expy
17
George F Hwy
River Rd
69 m
N
Google Earth

Figure A4:

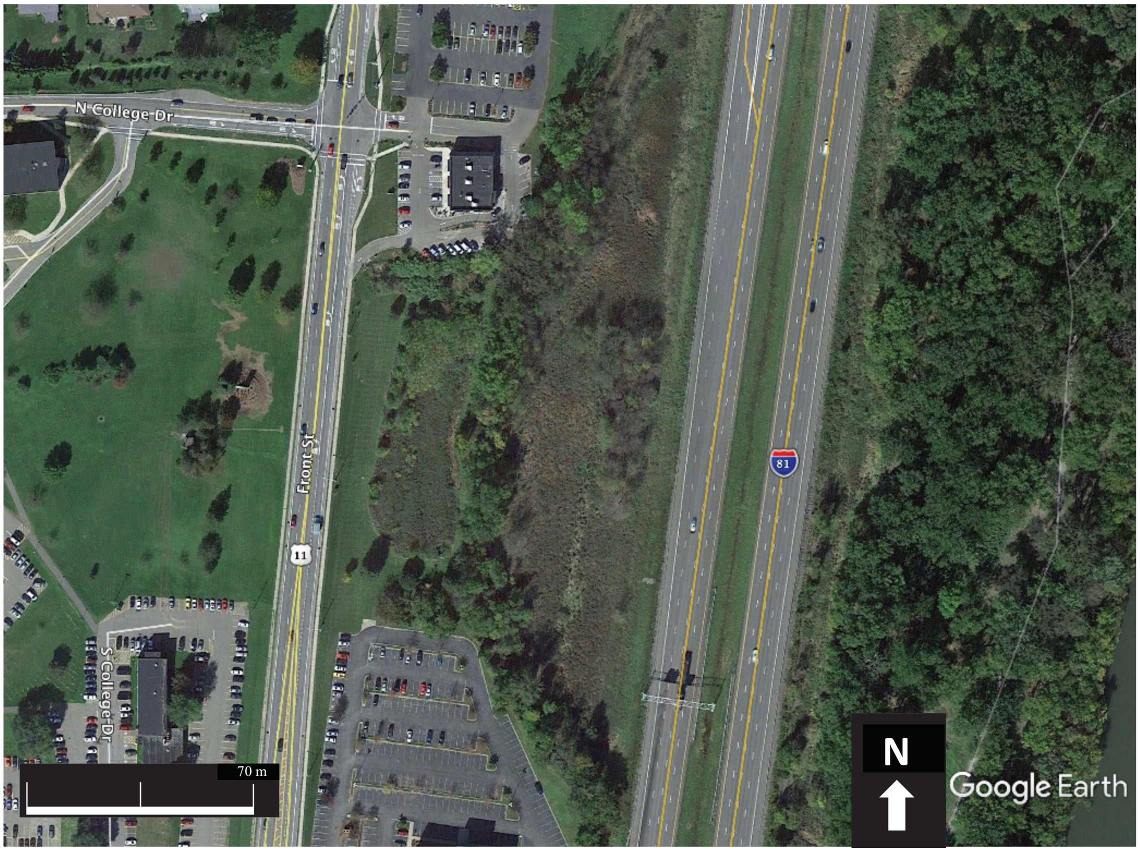
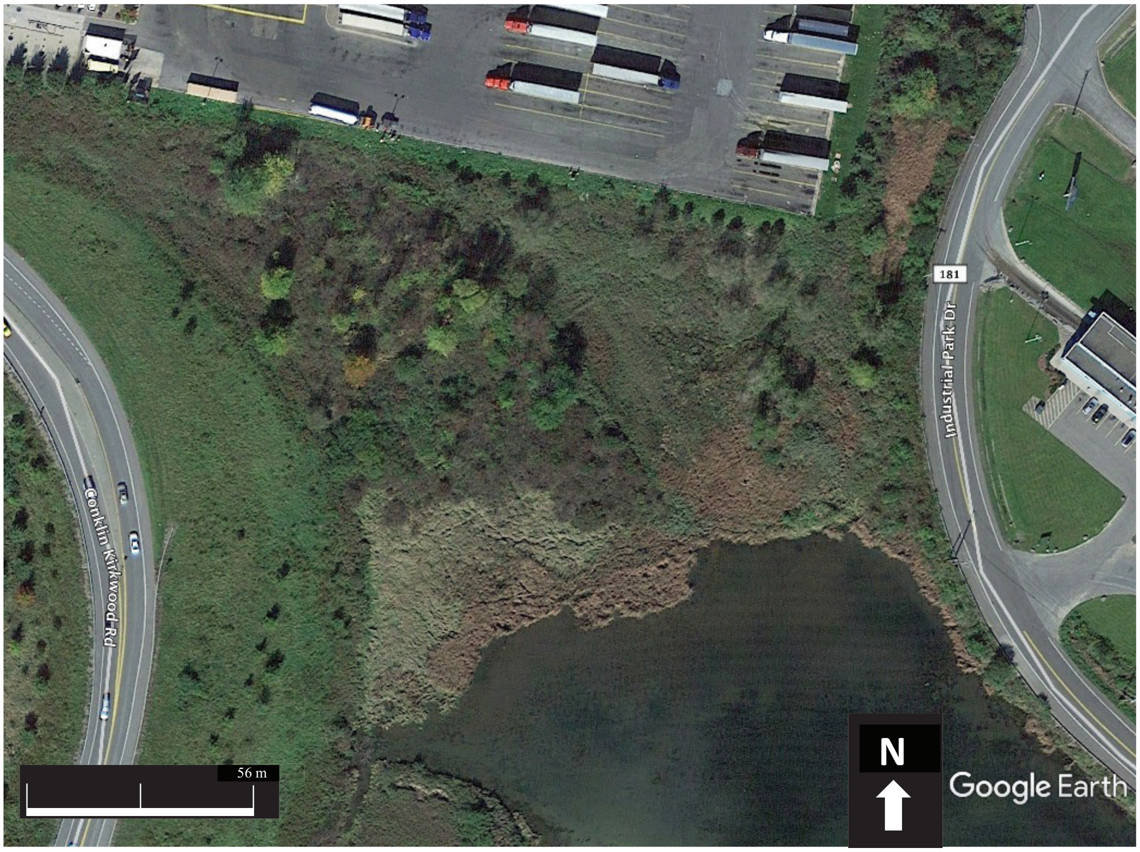


Figure A5:

Figure A6:



Conklin Kirkwood Rd
Industrial Park Dr
181
56 m
N
Google Earth

Figure A7:


New York State Bicycle Route 17
Colesville Rd
Court St
11
Rte 11
Kirkwood Rd
56 m
N
Google Earth

Figure A8:


Manor Dr
Front St
Ferndale Dr
11
Elm Dr
Elaine Dr
Iris Dr
Adams Dr
Old Front St
Rte 11
81
169 m
N
Google Earth

Appendix B: Supplemental material for Chapter 3.

Table B1: Relative seedling densities for all taxa in drawdown (DD) and flooded (FL) treatments for each of the four wetland sites. Invasive species are in bold. Unidentified seedlings were combined into one category, with the number of unidentified species in parentheses. Multiple taxa that were identified to the genus or family level, but could not be identified to the species level, are distinguished using superscripts.

	Site 1		Site 4		Site 6		Site 7	
Scientific name	DD	FL	DD	FL	DD	FL	DD	FL
Acalypha rhomboidea Raf.	-	-	-	-	0.03	-	-	-
Acer rubrum L.	-	-	0.03	-	-	-	-	-
Alisma triviale Pursh.	2.15	16.67	-	-	0.18	0.99	-	0.34
Bidens cernua L.	0.17	-	-	-	-	-	-	-
Bidens frondosa L.	-	-	-	-	-	-	0.12	-
Brassicaceae[A]	-	-	0.42	-	0.03	-	0.06	-
Brassicaceae[B]	-	-	-	-	-	-	0.19	-
Carex sp.	-	-	0.76	-	1.65	-	1.68	-
Cyperus sp.	-	-	-	-	0.05	-	-	-
Cyperus strigosus L.	0.33	-	0.12	-	0.18	-	-	-
***Digitaria ischaemum* (Schreb.) Muhl.**	**-**	**-**	**0.06**	**-**	**0.03**	**-**	**-**	**-**
***Digitaria sanguinalis* (L.) Scop.**	**-**	**-**	**0.15**	**-**	**-**	**-**	**-**	**-**
***Dipsacus fullonum* L.**	**-**	**-**	**-**	**-**	**-**	**-**	**0.37**	**-**
***Echinochloa crus-galli* (L.) P. Beauv.**	**0.99**	**1.52**	**-**	**-**	**0.03**	**-**	**-**	**-**
Eleocharis ovata (Roth) Desv.	-	-	-	-	0.03	-	-	-
Eleocharis palustris (L.) Roem. & Schult.	-	-	0.06	0.37	0.08	0.99	-	-
Eleocharis sp.[A]	-	0.76	-	-	-	-	-	-
Eleocharis sp.[B]	3.64	-	0.64	-	1.44	0.33	0.37	-
Eleocharis sp.[C]	-	-	0.03	0.12	-	64.47	-	-

Epilobium ciliatum Raf.	0.17	-	0.21	-	-	-	-	-
Epilobium coloratum Muhl. Ex. Willd.	0.33	-	0.21	-	0.64	-	-	-
***Epilobium hirsutum* L.**	**0.17**	**-**	**0.03**	**-**	**0.13**	**-**	**-**	**-**
Equisetum arvense L.	-	-	-	-	-	-	0.12	-
Erechtites hieraciifolius (L.) Raf ex. DC	0.66	-	-	-	-	-	-	-
Erigeron canadensis L.	-	-	-	-	0.03	-	0.06	-
Eupatorium perfoliatum L.	-	-	0.49	-	3.35	-	-	-
Eutrochium maculatum (L.) E.E. Lamont	-	-	0.03	-	-	-	-	-
Galium sp.	-	-	2.03	-	0.44	-	-	-
Hypericum mutilum L.	-	-	0.06	-	0.05	0.33	-	-
Iris sp.	-	-	0.06	-	-	-	-	-
Juncus acuminatus Michx.	6.45	-	-	-	0.08	-	-	-
Juncus articulatus L.	0.66	12.12	0.36	0.25	0.26	1.32	1.24	2.68
Juncus bufonius L.	5.62	-	0.03	-	-	-	4.10	-
Juncus effusus L.	15.54	2.27	2.00	0.25	2.81	-	9.25	-
Juncus pelocarpus E. Mey.	-	-	0.06	-	-	-	-	-
Juncus sp.	22.15	19.70	1.27	-	9.22	0.66	5.77	-
Juncus tenuis Willd.	6.61	0.76	0.18	-	0.03	-	0.31	-
Leersia oryzoides (L.) Sw.	17.19	14.39	0.58	2.22	0.03	-	0.37	-
Lemna minor L.	-	-	-	-	-	-	-	2.68
Ludwigia palustris (L.) Elliott	0.17	-	0.18	48.34	31.33	5.92	0.12	2.68

Lycopus americanus Muhl. ex W.P.C. Barton	-	-	0.30	-	0.08	-	-	-
***Lythrum salicaria* L.**	**0.33**	**-**	**75.06**	**22.07**	**24.07**	**2.63**	**50.59**	**47.65**
Myosotis laxa Lehm.	-	-	-	-	0.03	-	-	-
Myosotis sp.	-	-	0.03	-	0.03	-	-	-
Oxalis sp.	-	-	0.12	-	-	-	-	-
Panicum capillare L.	-	-	0.06	-	0.03	-	-	-
Panicum dichotomiflorum Michx.	0.17	-	0.15	-	0.03	-	-	-
Penthorum sedoides L.	-	-	0.09	-	0.44	-	-	-
Persicaria amphibia (L.) Delarbre	-	-	-	-	0.05	-	-	-
Persicaria lapathifolium (L.) Delarbre	0.17	-	0.03	-	-	-	-	-
Persicaria pensylvanica (L.) M. Gómez	-	-	0.03	-	0.21	-	-	-
Persicaria sagittata (L.) H. Gross	-	-	-	-	0.03	-	-	-
***Phalaris arundinacea* L.**	**0.17**	**-**	**1.73**	**-**	**3.40**	**-**	**7.14**	**-**
***Plantago major* L.**	**0.17**	**-**	**0.36**	**-**	**0.28**	**-**	**0.12**	**-**
Poaceae	0.99	-	0.03	-	0.08	-	-	-
Polygonaceae	-	-	-	-	-	-	0.06	-
Polygonum bellardii All.	-	-	0.03	-	-	-	-	-
Polygonum sp.[A]	0.33	-	-	-	-	-	-	-
Polygonum sp.[B]	-	-	0.03	-	0.05	-	-	-
Potamogeton sp.	-	1.52	-	-	-	-	-	-
Ranunculus repens L.	0.17	-	1.73	-	0.05	-	-	-
Rumex verticillatus L.	-	-	0.03	-	-	-	-	-

Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	7.27	18.94	0.03	-	0.05	-	0.37	-
Solidago canadensis L.	-	-	2.25	-	5.36	-	2.36	18.46
***Trifolium pratense* L.**	**-**	**-**	**0.03**	**-**	**-**	**-**	**-**	**-**
***Typha* sp.**	**0.33**	**3.79**	**3.97**	**25.77**	**1.96**	**22.37**	**13.16**	**23.49**
***Veronica serpyllifolia* L.**	**4.63**	**5.30**	**-**	**-**	**-**	**-**	**0.19**	**-**
Unknown (# of species)	2.31 (6)	2.27 (2)	3.82 (12)	0.62 (1)	11.71 (11)	-	1.86 (4)	2.01 (1)

Table B2: Relative percent cover for all herbaceous taxa in the standing vegetation for each of the four wetland sites. Invasive species are in bold. Unidentified taxa were combined into one category, with the number of unidentified species in parentheses. Multiple taxa that were identified to the genus or family level, but could not be identified to the species level, are distinguished using superscripts.

Species	Site 1	Site 4	Site 6	Site 7
Acer sp.	-	-	0.27	
Alisma triviale Pursh.	5.65	-	-	-
Alnus serrulata (Aiton) Willd.	-	-	-	0.02
Bidens sp.^{A}	0.15	-	-	-
Bidens sp.^{B}	1.22	-	-	-
Boehmeria cylindrica (L.) Sw.	-	0.22	-	-
***Butomus umbellatus* L.**	**-**	**4.72**	**-**	**-**
Carex crinita Lam.	-	0.16	-	-
Carex sp.	1.31	-	0.11	-
Carex sparganioides Muhl. ex Willd.	-	-	-	0.24
Carex vulpinoidea Michx.	-	-	0.36	0.24
Cicuta maculata L.	-	0.16	0.14	-
Cyperaceae	-	-	0.03	0.72
Cyperus sp.	-	-	0.03	-
***Daucus carota* L.**	**-**	**-**	**0.03**	**0.02**
Dianthus sp.	-	-	0.03	-
***Dipsacus fullonum* L.**	-	-	**0.03**	**2.98**
***Echinochloa crus-galli* (L.) P. Beauv.**	**0.32**	-	-	-
Eleocharis sp.	5.99	-	-	0.12
Equisetum arvense L.	-	0.03	1.10	0.60
Eupatorium perfoliatum L.	-	-	0.44	-
Euthamia graminifolia (L.) Nutt.	-	-	0.08	-
Galium sp.	0.58	8.53	2.94	-
Galium trifidum L.	-	-	-	1.22
Geum sp.	-	0.56	1.57	-
***Glechoma hederacea* L.**	-	**8.99**	-	-
Glyceria melicaria (Michx.) F.T. Hubb.	-	-	-	0.36
Glyceria sp.^{A}	-	-	0.14	-
Glyceria sp.^{B}	-	1.87	0.19	-
Impatiens capensis Meerb.	-	3.44	-	0.33
Iris versicolor L.	-	-	0.03	-
Juncus acuminatus Michx.	-	-	1.65	-
Juncus articulatus L.	0.86	-	-	-

Juncus effusus L.	-	-	1.73	-
Juncus tenuis Willd.	-	0.16	-	-
Leersia oryzoides (L.) Sw.	8.49	5.56	-	-
Lemna minor L.	-	-	5.80	4.84
***Lotus corniculatus* L.**	-	-	**1.92**	-
Ludwigia palustris (L.) Elliott	-	0.06	-	-
Lycopus americanus Muhl. ex W.P.C. Barton	0.04	-	-	-
***Lysimachia nummularia* L.**	-	**1.25**	-	-
***Lythrum salicaria* L.**	**0.04**	**30.92**	**15.30**	**2.86**
Myosotis laxa Lehm.	0.21	0.12	-	-
***Myosotis scorpioides* L.**	-	**9.09**	-	-
Onoclea sensibilis L.	-	-	1.37	-
Oxalis sp.[A]	-	0.03	-	-
Oxalis sp.[B]	-	0.16	0.16	-
Parthenocissus quinquefolia (L.) Planch.	-	0.16	-	-
***Persicaria hydropiper* (L.) Delarbre**	**-**	**-**	**-**	**0.24**
Persicaria hydropiperoides (Michx.) Small	0.02	-	-	-
Persicaria pensylvanica (L.) M. Gómez	0.49	0.72	-	-
Persicaria sagittata (L.) H. Gross	-	1.28	0.58	-
***Phalaris arundinacea* L.**	**1.13**	**4.72**	**26.02**	**20.05**
***Phragmites australis* (Cav.) Trin. ex Steud.**	-	-	**7.99**	**22.53**
***Plantago major* L.**	**0.21**	**0.03**	**-**	-
Poaceae	0.11	0.03	0.14	1.34
Polygonum sp.	0.34	0.06	-	-
Populus tremuloides Michx.	-	-	0.03	-
Potamogeton sp.	24.60	-	-	-
Ranunculus sp.	-	5.93	-	0.02
Rumex sp.	0.02	-	-	-
Sagittaria latifolia Willd.	32.68	-	0.82	-
Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	11.40	-	-	-
Scirpus atrovirens Willd.	0.75	0.31	-	-
Scripus sp.	-	-	1.15	-
Securigera varia (L.) Lassen	-	-	2.20	0.24
***Solanum dulcamara* L.**	-	-	**0.55**	-
Solidago rugosa Mill.	-	-	0.33	-
Solidago sp.	0.09	1.72	7.83	1.43
Sparganium americanum Nutt.	0.28	-	-	-
Spiraea alba Du Roi	-	-	0.27	-
Spiraea tomentosa L.	-	-	0.27	-

***Taraxacum officinale* F. H. Wigg.**	**-**	**0.16**	-	-
***Typha* x *glauca* Godr.**	**1.05**	**8.28**	**13.38**	**31.52**
***Veronica serpyllifolia* L.**	**1.35**	**-**	-	-
Veronica sp.	0.11	-	-	-
Vicia americana Muhl. ex Willd.	-	-	0.03	-
Vitis sp.	-	-	1.24	1.19
Seedlings	0.09	0.41	0.27	6.58
Unknown Taxa	0.41 (1)	0.19 (2)	1.46 (4)	0.31 (1)

Table B3: Presence of woody taxa in the standing vegetation for each of the four wetland sites. A "P" indicates that the species was present in the survey. Invasive species are in bold.

Species	Site 1	Site 4	Site 6	Site 7
Acer negundo L.	-	P	-	-
Acer saccharinum L.	-	P	P	-
Cornus amomum Mill.	-	-	P	-
Cornus sericea L.	-	P	P	P
Cornus sp.	-	P	-	-
Ulmus americana L.	-	P	-	-
***Elaeagnus umbellata* Thunb.**	**-**	**-**	**P**	**-**
Fraxinus americana L.	-	P	-	-
Lonicera sp.	-	P	P	-
Populus deltoides W. Bartram ex Marshall	-	-	P	-
Populus tremuloides Michx.	-	-	P	-
Salix sp.	-	-	P	-
Viburnum dentatum L.	-	-	P	-
Viburnum lentago L.	-	-	P	-

Appendix C: Supplemental material for Chapter 4.

Table C1: Relative seedling density for drawdown (DD) and flooded (FL) treatments before (2011) and after (2012) regrading in Lieberman. Invasive species are in bold. Unidentified seedlings were combined into one category, with the number of unidentified species in parentheses. Multiple taxa that were identified to the genus or family level, but could not be identified to the species level, are distinguished using superscripts.

	2011		2014	
Species	DD	FL	DD	FL
Alisma triviale Pursh	5.45	57.43	2.15	16.67
Bidens cernua L.	0.34	-	0.17	-
*Cyperu*s sp.	0.14	-	-	-
Cyperus strigosus L.	-	-	0.33	-
Digitaria sp.	0.11	-	-	-
***Echinochloa crus-galli* (L.) P. Beauv.**	**0.02**	**-**	**0.99**	**1.52**
Eleocharis sp.[A]	-	-	-	0.76
Eleocharis sp.[B]	-	-	3.64	-
Epilobium ciliatum Raf.	1.11	0.10	0.17	-
Epilobium coloratum Muhl. Ex. Willd.	-	-	0.33	-
***Epilobium hirsutum* L.**	**1.56**	**-**	**0.17**	**-**
Epilobium palustre L.	0.72	-	-	-
Equisetum arvense L.	0.16	-	-	-
Erechtites hieraciifolius (L.) Raf ex. DC	0.07	-	0.66	-
Galium sp.	0.84	-	-	-
Hypericum mutilum L.	0.07	-	-	-
Juncus acuminatus Michx.	-	-	6.45	-
Juncus articulatus L.	7.12	-	0.66	12.12
Juncus bufonius L.	-	-	5.62	-
Juncus effusus L.	22.08	0.10	15.54	2.27
Juncus sp.	-	-	22.15	19.70
Juncus tenuis Willd.	13.11	-	6.61	0.76
Leersia oryzoides (L.) Sw.	22.82	2.34	17.19	14.39
Leersia virginica Willd.	0.68	-	-	-
Lemna minor L.	0.00	23.73	-	-
***Linaria vulgaris* Mill.**	**0.34**	**-**	**-**	**-**
Ludwigia palustris (L.) Elliott	-	-	0.17	-

Lycopus americanus Muhl. ex W.P.C. Barton	-	0.10	-	-
***Lythrum salicaria* L.**	**0.14**	**-**	**0.33**	**-**
***Myosotis scorpioides* L.**	**3.95**	**0.10**	**-**	**-**
***Nasturtium officinale* W.T. Aiton**	**0.02**	**-**	**-**	**-**
Oxalis sp.	0.02	-	-	-
Panicum dichotomiflorum Michx.	-	-	0.17	-
Persicaria hydropiperoides (Michx.) Small	0.02	-	-	-
Persicaria lapathifolium (L.) Delarbre	-	-	0.17	-
Persicaria pensylvanica (L.) M. Gómez	0.02	-	-	-
***Phalaris arundinacea* L.**	**-**	**-**	**0.17**	**-**
***Plantago major* L.**	**0.68**	**-**	**0.17**	**-**
Poaceae	0.05	-	0.99	-
Polygonum sp.[A]	-	-	0.33	-
Potamogeton sp.	-	3.67	-	1.52
Ranunculus hispidus Michx.	0.05	-	-	-
Ranunculus repens L.	-	-	0.17	-
Sagittaria latifolia Willd.	0.09	0.61	-	-
Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	-	-	7.27	18.94
***Stachys palustris* L.**	**2.42**	**-**	**-**	**-**
***Trifolium pratense* L.**	**0.02**	**-**	**-**	**-**
***Typha* sp.**	**-**	**-**	**0.33**	**3.79**
***Veronica serpyllifolia* L.**	**8.47**	**1.63**	**4.63**	**5.30**
Vicia tetrasperma (L.) Schreb.	0.00	0.20	-	-
Unknown Taxa (# species)	7.32 (17)	9.98 (3)	2.32 (6)	2.27 (2)

Table C2: Relative percent cover for standing vegetation before (2011) and after (2012-2014) regrading in Lieberman. Invasive species are in bold. Unidentified seedlings were combined into one category, with the number of unidentified species in parentheses. Multiple taxa that were identified to the genus or family level, but could not be identified to the species level, are distinguished using superscripts.

	2011	2012		2013		2014	
	Jun	Jul	Aug	Jul	Aug	Jul	Aug
Alisma triviale Pursh.	3.70	0.81	1.77	3.10	1.74	5.65	8.56
Bidens sp.[A]	-	-	-	-	-	0.15	0.92
Bidens sp.[B]	-	-	-	-	-	1.22	0.04
Carex lurida Wahlenb.	0.06	-	-	-	-	-	-
Carex sp.	-	-	1.52	2.74	0.24	1.31	0.65
Cirsium sp.	1.18	-	-	-	-	-	-
Cornus sericea L.	1.46	-	-	-	-	-	-
Cyperus sp.	-	0.06	-	-	-	-	-
***Dipsacus fullonum* L.**	**0.34**	-	-	-	-	-	-
***Echinochloa crus-galli* (L.) P. Beauv.**	**-**	**0.29**	**-**	**0.42**	**2.03**	**0.32**	**1.96**
Eleocharis palustris (L.) Roem. & Schult.	-	2.02	10.14	4.70	6.41	5.99	-
Eleocharis sp.	-	-	-	-	-	-	4.00
Equisetum arvense L.	1.46	-	-	-	-	-	-
Eupatorium perfoliatum L.	0.34	-	-	-	-	-	-
Euthamia graminifolia (L.) Nutt.	0.39	-	-	-	-	-	0.10
Galium sp.	3.42	-	-	-	-	0.58	0.12
Impatiens capensis Meerb.	1.12	-	-	-	-	-	-
Juncus articulatus L.	-	0.81	0.74	0.28	0.09	0.86	0.04
Juncus effusus L.	0.79	-	-	-	-	-	-
Juncus sp.	-	-	-	0.03	-	-	-
Juncus tenuis Willd.	2.58	-	-	-	-	-	-
Leersia oryzoides (L.) Sw.	13.58	0.52	3.43	6.94	12.52	8.49	13.29
Lemna minor L.	0.06	-	-	-	-	-	-
Ludwigia palustris (L.) Elliott	-	-	0.11	0.03	-	-	0.02
Lycopus americanus Muhl. ex W.P.C. Barton	-	0.06	-	-	-	0.04	0.04
***Lythrum salicaria* L.**	-	-	**0.04**	**0.08**	**0.14**	**0.04**	-
Myosotis laxa Lehm.	-	-	-	-	-	0.21	-
***Myosotis scorpioides* L.**	**18.01**	**0.81**	**0.53**	**-**	**-**	**-**	**0.06**
Myosotis sp.	-	-	-	0.03	0.09	-	-

Panicum virgatum L.	-	0.17	0.18	-	-	-	-
***Persicaria hydropiper* (L.) Delarbre**	-	-	-	-	-	-	**0.02**
Persicaria hydropiperoides (Michx.) Small	-	0.17	-	-	-	0.02	-
Persicaria pensylvanica (L.) M. Gómez	-	0.06	0.28	0.11	1.13	0.49	0.31
Persicaria sagittata (L.) H. Gross	-	-	-	-	-	-	0.02
***Phalaris arundinacea* L.**	-	**1.27**	**0.18**	**-**	**-**	**1.13**	**0.02**
***Plantago major* L.**	**-**	**-**	**-**	**-**	**0.07**	**0.21**	**0.12**
Poaceae A	-	-	-	-	-	0.11	-
Poaceae B	-	-	-	-	-	-	0.04
Poaceae C	0.45	-	-	0.70	0.31	-	-
Polygonum sp. A	-	-	-	0.56	-	0.34	0.06
Polygonum sp. B	-	-	-	-	-	-	0.10
Polygonum sp. C	-	-	-	0.50	0.35	-	0.26
Populus tremuloides Michx.	-	-	-	-	0.02	-	-
Potamogeton sp.	10.04	68.32	43.29	25.93	20.40	24.60	13.29
Ranunculus hispidus Michx.	-	0.06	-	-	-	-	-
Ranunculus sp.	8.19	0.06	-	-	-	-	-
Rumex sp.	-	-	-	-	-	0.02	0.02
Sagittaria latifolia Willd.	7.86	22.87	33.71	43.97	44.31	32.68	37.85
Schoenoplectus tabernaemontani (C.C. Gmel.) Palla	-	1.04	3.25	8.34	7.24	11.40	13.41
Scirpus atrovirens Willd.	-	-	-	-	-	0.75	0.20
Scripus sp.	-	-	-	-	-	-	0.02
Solidago sp.	3.70	-	-	-	-	0.09	-
Sparganium americanum Nutt.	-	-	-	0.36	0.68	0.28	0.71
***Trifolium pratense* L.**	**-**	**-**	**-**	**-**	**-**	**-**	**0.02**
Trifolium sp.	-	0.17	-	-	-	-	-
***Typha* x *glauca* Godr.**	**17.68**	**0.06**	**0.67**	**0.64**	**0.87**	**1.05**	**1.94**
Veronica anagallis-aquatica L.	-	0.29	-	-	-	-	-
***Veronica serpyllifolia* L.**	**0.11**	**-**	**0.04**	**0.11**	**0.31**	**1.35**	**1.73**
Veronica sp.	-	-	-	-	-	0.11	-
Seedlings	-	0.06	0.04	0.06	0.02	0.09	0.04
Unknown Taxa (# species)	3.48 (2)	0.06 (1)	0.11 (2)	0.36 (1)	1.01 (2)	0.41 (1)	-