

# Urban Wetland Reconstruction Impacts on Vegetation: A Case Study

Megan A. Larson, Julian Shepherd,  
and John E. Titus



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**Cover Photograph:** A view of an urban retention wetland (Lieberman) in Vestal, NY, USA, shortly after reconstruction. Photograph © Megan Larson.

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# Urban Wetland Reconstruction Impacts on Vegetation: A Case Study

Megan A. Larson<sup>1,2,\*</sup>, Julian Shepherd<sup>1</sup>, and John E. Titus<sup>1</sup>

**Abstract** - This study documents the effects on the seed bank and standing vegetation of an urban wetland after its expansion to accommodate increased runoff. While total wetland area doubled, densities of seedlings emerging from soil cores declined by 66%. Significant species compositional changes occurred in the seed bank, as did changes among functional groups: non-natives unexpectedly declined, while annuals and graminoids increased. The standing vegetation consisted overwhelmingly of herbaceous taxa before and after the reconstruction, and plant cover, after complete denudation of the vegetation, recovered within 3 years. Compositional changes included unexpected declines in the relative cover of non-native species and increases in the relative cover of graminoids and obligate wetland species—the last in keeping with observed hydrologic change. Seeding and planting may not be necessary, even to reach short-term goals (i.e., less than 5 years) of high plant cover, in reconstructed wetlands.

## Introduction

The projected global increase in urban areas (Nilon et al. 2017, Seto et al. 2012) will surely encroach upon countless more wetlands. Furthermore, land use managers are likely to promote the creation of new wetlands and the expansion of existing wetlands to accommodate increases in urban stormwater runoff. The increased runoff that accompanies urbanization leads many cities to look toward green infrastructure, such as retention ponds and treatment wetlands, to mitigate runoff impacts (Balderas Guzman et al. 2018). In this case study, we take advantage of an unusual opportunity to evaluate the impacts of a major reconstruction project on the vegetation of a wetland that will receive increased runoff. This project, which completely regraded the entire surface of a previously constructed wetland and doubled its size, serves as an example of what may be expected in similar ecosystems in the future.

Vegetation is a key component of wetlands, which promote sedimentation, reduce flooding downstream, and improve water quality through reductions in inorganic nitrogen and phosphorus (Mitsch and Gosselink 2015). Different species of plants may vary in their ecological functions (e.g., Kao et al. 2003), and this potential variation motivates our focus on changes in plant community composition subsequent to the reconstruction project.

Urban wetlands may experience frequent disturbances (Grayson et al. 1999) from erosion due to altered hydrology (Ravit et al. 2017), removal of aboveground biomass for crop harvest (Vécrin et al. 2007), invasive species management (Lawrence et al. 2016), and stormwater control (Blecken et al. 2017). Because wetland construction projects commonly fail to monitor plant community establishment, little is known about the response of urban wetland vegetation to disturbances (Zedler 2000). Emergence of seedlings from soil seed banks (Alderton et al. 2017, Kaplan et al. 2014, Muller et al. 2013, Nishihiro et al. 2006) and regrowth from surviving plant parts (Combroux and Bornette 2004, Combroux et al. 2002) have great potential to recolonize soil surfaces denuded of vegetation. Disturbance may favor subsequent colonization by invasive species (D'Antonio and Meyerson 2002, Matthews et al. 2009) as well as annuals (e.g., van der Valk 1981). The seed bank

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itself may also be altered by major disturbance (Neff et al. 2009, Osunkoya et al. 2014). The extensive mixing and spreading of the sediment in our expanded wetland led to 3 hypotheses regarding the influence of disturbance on the plant community: (1) the soil seed bank would be diluted, resulting in reduced density of seedlings emerging from surface sediment samples, and there would be (2) an increase in non-native plants and (3) an increase in annual plants in the standing vegetation subsequent to the regrading.

In addition to disturbance per se, changes to wetland topography may alter the plant community in terms of dominant wetland indicator status (WIS), as wetland plants are distributed within wetlands by their responses to water level (e.g., Keddy and Ellis 1985, Roznere and Titus 2017, Tiner 1991). In this case study, engineering plans called for greater topographic diversity in the reconstructed wetland, both by creation of a deeper channel through the wetland and by raising mounds elsewhere at the site. This planned topographic variation led to our fourth hypothesis: there would be a broadening in the spectrum of plant species toward greater diversity of WIS categories in the standing vegetation.

Furthermore, our data allowed us to compare the species composition and growth habits of the seedlings that emerged from the seed bank before and after the regrade as well as examine successional changes in the standing vegetation for 3 years after the construction project.

## Study Site and Methods

### Study Site

Lieberman is a small (0.15 ha) urban retention wetland located on the Binghamton University campus in Vestal, NY, USA (Lat 42.087°, Long -75.962°; altitude 308 m). This site is referred to as “Site 1” in Larson et al. (2016) and Larson and Titus (2018). The site is relatively isolated and does not receive hydrological input from other wetlands, with the nearest wetland complex ~0.6 km away. Lieberman receives runoff from 0.56 km<sup>2</sup> of campus, including parking lots, paved roadways and sidewalks, and buildings (Kearney et al. 2013). The main inlet, via culverts and drainage ditches, is located in the southwest corner of the site (Fig. 1). Groundwater also seeps into the wetland, which drains through a culvert into Fuller Hollow Creek, which in turn discharges into the Susquehanna River (Zhu et al. 2008), the largest tributary of Chesapeake Bay. Binghamton lies in a deciduous forest biome with a humid continental climate. Mean annual precipitation is 99.8 cm and mean monthly low and high temperatures are 3.4 and 12.5°C, respectively (Wikipedia 2020). The campus supports a relatively high volume of traffic: ~48% of the 13,000 full-time undergraduate students commute to campus, in addition to administrators, faculty, staff, and ~4000 graduate students (College Board 2016).

In 2004, the pre-existing pond was drained and a berm was built along the east side of the pond; the resulting wetland supported wetland vegetation throughout and served as a retention pond to accommodate campus runoff (Fig. 1A). This stormwater retention wetland included a small channel near the inlet that opened to a larger, inundated marsh dominated by *Sagittaria latifolia* Willd. (Broadleaf Arrowhead) and *Alisma triviale* Pursh (Northern Water Plantain). *Myosotis scorpioides* L. (Water Forget-me-not), *Typha x glauca* Godr. (Hybrid Cattail), and *Leersia oryzoides* (L.) Sw. (Rice Cutgrass) were common near the inlet and around the perimeter of the wetland (Larson et al. 2016). There have been occasional sightings of herbivores, including *Odocoileus virginianus* (Zimmerman) (White-tailed Deer), *Ondatra zibethicus* (L.) (Muskrats), and *Branta canadensis* (L.) (Canada Geese), although the impacts of these species on the vegetation are unknown.

The wetland was regraded to accommodate increases in runoff from newly constructed impervious surfaces on campus. Construction of the expanded Lieberman wetland with heavy construction equipment began in July 2011 and resulted in haphazard earth moving and burial of



Figure 1. Aerial photographs of Lieberman (A) before regrading and (B) after regrading. The white dotted lines indicate the wetland border. Google Earth Pro 7.3.3.7786 (April 2006 and 13 May 2015, respectively). Vestal, New York, USA. Google, New York GIS 2021 [28 March 2021].

existing vegetation (Fig. 2) as well as the upheaval and spreading of wetland sediment over ~0.34 ha—more than doubling the wetland area. Sediment accumulation ponds, initially too deep to support vegetation, were added near the main inlet and outlet and connected by a meandering channel (Fig. 1B). Landscape was created using sediment from the wetland. Regrading was completed in the early spring of 2012, at which point topographic variation was not realized, because of the slumping of relatively fluid sediments. No seeding occurred. Experimental plant propagules were added to the wetland in July 2012, but all died within the first growing season. These experimental plants were not planted in survey locations.

### Seed Bank Collection

Sampling locations were selected randomly in April 2011 (prior to regrading) at 5 points along each of 3 randomly selected transects perpendicular to a baseline bordering the east side of the wetland. At each point, 2 sediment cores (15.2 cm diameter, 5 cm deep) were collected, for a total of 30 cores. Standing water was present at only 1 point. Sediment was stored in a cold room at 4.4 °C for 1 month. In early May 2014 (>2 years after regrading was completed), we collected 2 randomly selected cores from 6 random transects, for a total of 12 sediment cores.

Experimental setups 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 1200-L fiberglass tanks in the Research Greenhouse at Binghamton University. Large debris, including rhizomes and tubers, was removed from the sediment. The 2 samples from each point were combined, homogenized, spread in a 1-cm-thick layer over sand in germination trays, and subjected to 2 treatments to maximize seedling yield: a permanent drawdown treatment with water levels 5 cm below the sediment surface and a permanent flooded treatment with water levels 5.5 cm above the sediment surface. The germination trays 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). Seedling data were collected weekly until seedling emergence ceased. Seedlings that could not be identified in the germination trays were transplanted into separate pots until they could be identified. Seedling densities for the 2 treatments combined were expressed as counts per m<sup>2</sup> of original sediment core surface. Data were also summarized as relative seedling density for each species, i.e., the percent of all seedlings attributed to that species. For each seed bank, a species was considered “common” if its relative seedling density exceeded 5% in either water level treatment. Seedling counts were also summarized for the following functional groups: native status (native vs. non-native), longevity (annual, biennial, or perennial), WIS (obligate wetland = OBL; facultative wetland = FACW; facultative = FAC; facultative upland = FACU; and upland = U), and growth habit (graminoid, forb, and vine).

### Vegetation Sampling

For every sampling period, quadrats were chosen by randomly selecting transects perpendicular to a baseline bordering the eastern edge of the wetland. Standing vegetation was sampled in June 2011 from 3 sampling points on each of 5 transects just prior to regrading (Larson et al. 2016). For 3 growing seasons after the regrade (2012–2014), we recorded the progression of revegetation by sampling the standing vegetation in early July and early August each year. Absolute cover estimates were recorded for each herbaceous species within 1-m<sup>2</sup> quadrats, to the nearest 5% (Mueller-Dombois and Ellenberg 1974). We recorded vegetation data from 4 quadrats along each of 11 transects in the 3 growing seasons after regrading. Thus, 44 plots were sampled on each occasion, except in 2012 when only 39 plots were sampled because flooding limited access. For each species, cover was also summarized as relative cover, i.e., the percent of the total cover recorded for all species, and similarly summarized for each category within the functional groups. Species in the standing

vegetation were considered “common” if the relative percent cover was greater than 5%. For some analyses, data were summarized as mean percent cover, or the total cover of a species divided by the number of sampling quadrats, and, for other analyses, data were converted to a presence–absence basis to address the concern that vegetation samples were taken in different months. The presence vs. absence of a species is likely to change much less on a seasonal basis than cover. Functional group categories for species in the standing vegetation were the same as for seedlings, with two exceptions: there were only four WIS categories because there were no upland (U) plants in the standing vegetation, and the growth habit categories were graminoid, forb, and tree/shrub.

### Plant Identification

Taxa were identified to the species level when possible using Gleason and Cronquist (1991), with nomenclature updated according to the New York Flora Atlas (Weldy et al. 2017). We identified 92.2% (2011) and 96.9% (2014) of seedlings at least to the genus level for the seed bank assessments. The native status, longevity, WIS, and growth habit of each species were found using the USDA plant database (USDA NRCS 2012) for the northeast region and the New York Flora Atlas (Weldy et al. 2017). Cattails in the standing vegetation were identified as the invasive and non-native *Typha x glauca* because of 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. because of a lack of these morphological traits. Five species of *Juncus* (*Juncus acuminatus* Michx. [Tapertip Rush]; *Juncus articulatus* L. [Jointleaf Rush]; *Juncus bufonius* L. [Toad Rush]; *Juncus effusus* L. [Common Rush]; and *Juncus tenuis* Willd. [Poverty Rush]) were identified but could not be reliably identified to species. Therefore, for the purposes of our taxonomic, longevity, and WIS analyses on seedlings, all *Juncus* species were combined into a single *Juncus* spp. category.

### Data Analyses

To test our seed bank dilution hypothesis, we applied a Kruskal–Wallis test (Social Science Statistics 2020) to compare mean seedling densities (sum of both water level treatments) from the 2011 and 2014 cores sampled. We used the same nonparametric test to compare species richness of these 2 seedling communities. Compositional differences between the communities, on the basis of species and functional groups, were assessed with chi-square tests based on seedling counts.

Figure 2. Photograph of the re-grading process in Lieberman, depicting the complete denudation of the wetland. Picture taken by Megan Larson, used with permission.



For these chi-square tests, the null hypothesis tested was that the distribution of seedlings among categories (species or each type of functional group) was the same for both years. Species and categories were combined as necessary to ensure that all expected values were at least 5. We also compared seed bank species compositions using Sørensen's similarity index (Sørensen 1948) based on seedling relative densities.

Changes in standing vegetation composition were summarized by calculating species richness and Shannon–Wiener diversity ( $H'$ ). To compare the pre-disturbance vegetation with its development after regrading, the Mahalanobis distance (MD) between the 2011 vegetation sample and the 6 July and August 2012–2014 samples was used to evaluate the distinctness of the 2011 pre-disturbance vegetation for the 4 functional group categorizations. In each case, the data consisted of the number of seedlings present in each category (2 categories each for native status and growth habit, 3 for longevity, and 5 for WIS), and a variance–covariance matrix for 2012–2014 data was constructed in Excel. The inverse of this matrix was determined manually following the procedure of van Biezen (2013), and the MD was calculated in Excel according to McCaffrey (2017). Manual calculation of the inverse matrix was feasible for these functional groups but not for the large number of species in the dataset. The resulting MD values were compared to critical values for chi-square with the degrees of freedom equaling the number of categories (McCaffrey 2017).

The related concept of outlier analysis was applied to evaluate differences in species composition, expressed as presence-absence rather than cover data, between the 2011 sample and separate samplings for July and August for 2012–2014. PC-ORD software (McCune and Mefford, 1999) compared the Sørensen's distances among the 7 samples, with an outlier defined as a sample for which the mean distance to the other samples exceeded 2 standard deviations away from the overall mean distances among the 7 samples.

Analyses of variances (ANOVAs) were performed to test for among-year differences in cover and richness for all vegetation plots sampled in 2011–2014, and separately for 2012–2014. For each year after the regrade, July and August data were averaged for each plot, and the same plots were used throughout this period. Significant overall results were further analyzed at the  $P = 0.05$  level with Tukey's honestly significant difference (HSD) tests.

## Results

### Seed Bank: Before vs. After the Regrade

We observed several substantial changes in the seed bank following the regrading project. Seedling densities before regrading were nearly 3-fold greater than those in 2014 (Table 1;  $H = 6.01$ ,  $df = 19$ ,  $P = 0.014$ ). Although we observed a decrease in the overall number of species (53 species before the regrade, 37 species afterwards), the decrease in species richness per sampling point (2 cores) was not statistically significant (Table 1;  $H = 1.27$ ,  $df = 19$ ,  $P = 0.259$ ).

The similarity index between the 2 seed bank surveys was 42% because of similar common species, including *Juncus* spp. and *L. oryzoides* (Table 2). *Alisma triviale* was relatively more abundant under flooded conditions before the regrade, while *Schoenoplectus tabernaemontani* (C.C. Gmel.) Palla (Softstem Bulrush) was common after the regrade but not before. Despite some similarities in common species, we observed an overall shift in species composition ( $c^2 = 694.5$ ,  $df = 10$ ,  $P < 0.001$ ), likely due to the increase in *S. tabernaemontani* and *Eleocharis* sp. (Spikerush) as well as a decrease in *A. triviale*, *Lemna minor* L. (Common Duckweed), and *M. scorpioides* in 2014. We did not observe *Typha* sp. seedlings before the regrade, and only 7 emerged afterwards.

Table 3 presents the results of the  $c^2$  tests, which showed a significant change in native status, as the percent of non-natives declined from 14.1% of seedlings before the regrade to 7.8% after. Proportions of seedlings in longevity categories also changed significantly, as the percent of

Table 1. Mean density (seedlings/m<sup>2</sup>) and species richness for seedlings emergent from the soil seed bank before (2011; 15 sampling points) and after (2014; 6 points) regrading. Means ± SE. Probability (P) is for t-tests.

	Density	Species
2011	9880 ± 1509	17.5 ± 1.7
2014	3367 ± 1225	13.3 ± 2.1
P	0.014	0.259

Table 2. Relative seedling density for species with values >5% in at least 1 treatment; drawdown (DD) and flooded (FL).

Species	2011		2014	
	DD	FL	DD	FLr
<i>Alisma triviale</i> Pursh (Northern Water Plantain)	5.5	57.4	2.2	16.7
<i>Juncus</i> spp. (Rushes)	42.3	0.1	57.0	34.9
<i>Leersia oryzoides</i> (L.) Sw. (Rice Cutgrass)	22.8	2.3	17.2	14.4
<i>Lemna minor</i> L. (Common Duckweed)	0.0	23.7	-	-
<i>Schoenoplectus tabernaemontani</i> (C.C. Gmel.) Palla (Softstem Bulrush)	-	-	7.3	18.9
<i>Veronica serpyllifolia</i> L. (Thymeleaf Speedwell)	8.5	1.6	4.6	5.3

annuals rose more than 6-fold from 0.7% to 4.7%. There was no significant change among WIS categories, but growth habit profiles changed significantly, as graminoids rose from 58.9% to 86.0% of seedlings, while forbs fell correspondingly.

### Standing Vegetation: Before vs. After the Regrade

Mean absolute cover for all species per plot initially declined from 70.1% in 2011 to 50.6% in 2012; thereafter, it increased to 88.8% in 2013 and 116.3% in 2014 (ANOVA,  $F_{3,138} = 19.7$ ,  $P < 0.0001$ ). Table 4 summarizes the species with the highest relative cover to compare the standing vegetation prior to the regrade (2011) with the 3 subsequent years, for which relative cover was averaged over the July and August samplings. The most notable changes from 2011 to 2012 were the precipitous declines in relative cover for the non-natives *M. scorpioides* and *Typha x glauca* and the sharp increases in *Potamogeton* sp. (Pondweed) and *S. latifolia*. During the 3 growing seasons after the regrade, cover rose monotonically for *L. oryzoides* and *S. tabernaemontani*, remained high for *S. latifolia*, and declined for *Potamogeton* sp. Outlier analysis showed that the species composition in 2011 differed substantially from the post-regrade sampling periods by virtue of its Sørensen's distance being 2.11 standard deviations from the mean of all 7 vegetation samplings. Species richness per plot in the standing vegetation significantly decreased immediately after the regrade, then rebounded (Fig. 3A; ANOVA,  $F_{3,138} = 11.2$ ,  $P < 0.0001$ ), while  $H'$  for the vegetation overall sharply declined immediately after regrading, then steadily increased by 2014 (Fig. 3b).

Changes in the distribution of plant cover among the 4 types of functional groups were substantial in some cases (Table 5). The proportion of plant cover of non-native species first declined abruptly from 2011 to 2012 (Fig. 3C), due largely to the aforementioned decreases in relative cover of *M. scorpioides* and *Typha x glauca* (Table 4), then remained low. For longevity, perennial species accounted for >97% of the plant cover for all sampling years. The relative cover of graminoid species nearly doubled by 2014 and that of obligate wetland species increased from 89% prior to the regrade to >96% afterwards. In contrast to our fourth hypothesis, the diversity of WIS categories actually declined.

Our findings via Mahalanobis distance values (Table 6) were based on presence–absence data, and, thus, differ somewhat from the percent cover data. Of the 4 types of functional groups, only WIS showed a significant change following the regrade, concomitant with the increase in obligate wetland species.

Table 3. Percentages of seedlings in different functional group categories for native status, longevity, wetland indicator status (WIS), and growth habit that emerged from the seed bank before (2011) and after (2014) regrading. Seedling counts shown in parentheses. Chi-square values shown with degrees of freedom are all significant at the 0.001 level (\*\*\*), except for the non-significant (n.s.) value for WIS. OBL = obligate wetland; FACW = facultative wetland; FAC = Facultative; FACU = facultative upland; U = upland

		2011	2014	Chi-square (df, P)
Native status	Native	85.9 (4206)	92.2 (654)	21.8 (1, ***)
	Non-native	14.1 (692)	7.8 (55)	
Longevity	Annual	0.7 (22)	4.7 (14)	40.5 (1, ***)
	Biennial	<0.1 (1)	0 (0)	
	Perennial	99.2 (3002)	95.3 (281)	
WIS	OBL	80.3 (2456)	82.3 (241)	7.5 (3, n.s.)
	FACW	5.0 (153)	2.4 (7)	
	FAC	13.1 (981)	15.0 (44)	
	FACU	1.0 (32)	0.3 (1)	
	U	0.6 (17)	0 (0)	
Growth habit	Forb	41.1 (2046)	14.0 (101)	195.3 (1, ***)
	Graminoid	58.9 (2937)	86.0 (618)	
	Vine	<0.1 (2)	-	

Table 4. Relative percent cover for species in the standing vegetation with cover >5% before (2011) and/or after (2012–2014) the regrade. Non-native species are in bold.

Species	2011	2012	2013	2014
<i>Alisma triviale</i> Pursh. (Northern Water Plantain)	3.7	1.3	2.4	7.1
<i>Eleocharis palustris</i> (L.) Roem. and Schult. (Common Spikerush)	-	6.1	5.6	5.0
<i>Leersia oryzoides</i> (L.) Sw. (Rice Cutgrass)	13.6	2.0	9.7	10.9
<b><i>Myosotis scorpioides</i> L. (Water Forget-me-not)</b>	18.0	0.7	-	-
<i>Potamogeton</i> sp. (Pondweed)	10.0	55.8	23.2	19.0
<i>Ranunculus</i> sp. (Buttercup)	8.2	<0.1	-	-
<i>Sagittaria latifolia</i> Willd. (Broadleaf Arrowhead)	7.9	28.3	44.1	35.3
<i>Schoenoplectus tabernaemontani</i> (C.C. Gmel.) Palla (Softstem Bulrush)	-	2.1	7.8	12.4
<b><i>Typha x glauca</i> Godr. (Hybrid Cattail)</b>	17.7	0.4	0.8	1.5

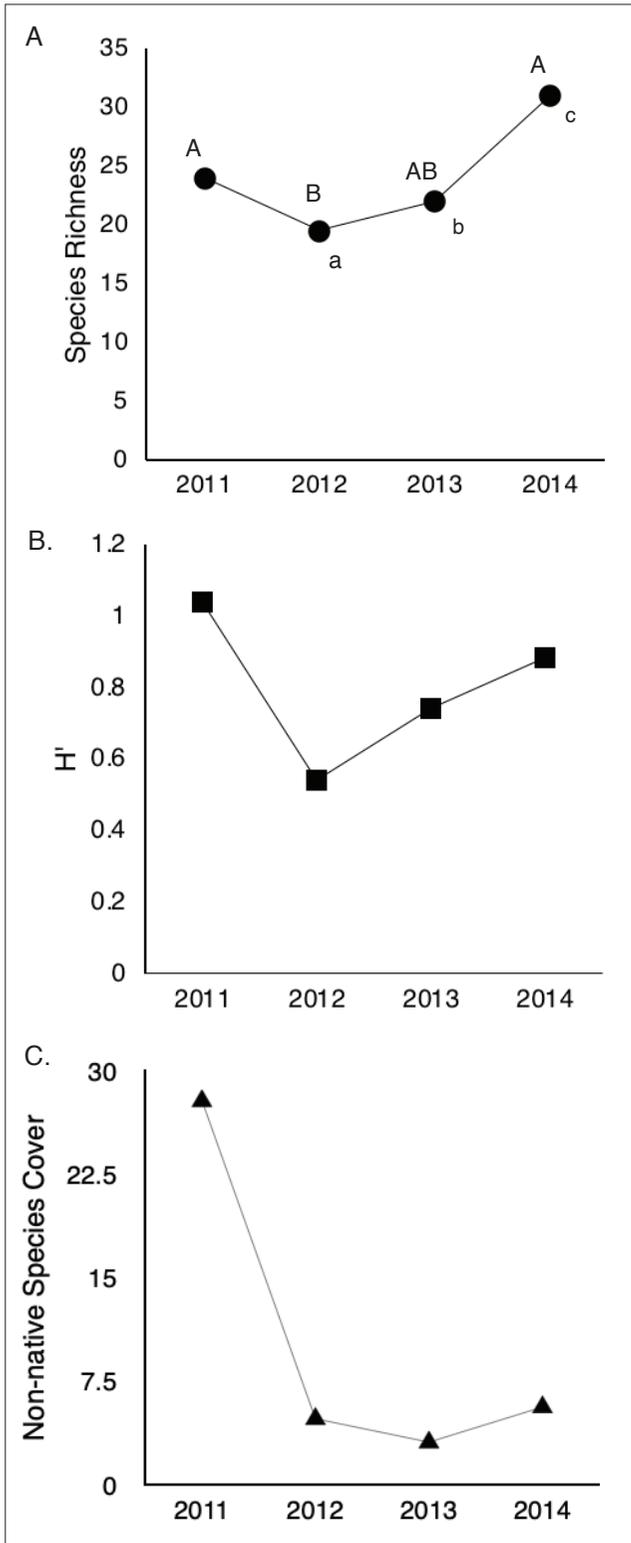


Figure 3. Values of (A) species richness, (B) Shannon-Weiner diversity ( $H'$ ), and (C) relative % cover of non-native species for the standing vegetation before and after regrading. Data for 2012–2014 are each the mean of 2 samples. Means not sharing a common letter differ significantly at  $P = 0.05$ , according to Tukey’s honestly significant differences tests.

Despite few changes in general vegetation characteristics, in addition to the cover changes cited above, we observed several shifts in the species composition of the standing vegetation from 2012 through 2014 (Table 4).

## Discussion

### Habitat Alteration Reflected in Vegetation

The regrading of Lieberman was a major habitat alteration that substantially impacted the seed bank and the standing vegetation. As predicted in our first hypothesis, we observed a reduced density of seedlings emerging from soil cores, indicating a dilution of the soil seed bank. The total seedling density before the regrade is within the range of other urban wetlands in the Binghamton area, but the seedling density in 2014 was significantly lower than other urban wetland seedling communities (Larson and Titus 2018). The overall seed bank dilution is likely 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 but then recover; for example, an extreme flooding event increased seedling density but reduced species richness, yet the riparian seed bank itself recovered quickly and was considered resilient (Osunkoya et al. 2014). Neff et al. (2009) reported that the density of seedlings emerging from sediment samples collected from a recently restored tidal marsh significantly increased by more than 40-fold within a year, and species richness was significantly higher than any reference site. The seedling density and species richness of these studies are likely a result of prolific seed production, allowing the seed banks to recover quickly. The lower seedling density from Lieberman sediment samples may be a consequence of low seed dispersal into the wetland, possibly due to its isolated position in an urban, fragmented landscape.

Although *Juncus* spp. and *L. oryzoides* remained important components of the seed bank before and after the regrade, we observed an overall shift in species composition, including a

Table 5. Relative % cover of functional types by category for native status, longevity, growth habit, and wetland indicator status (WIS) of standing vegetation before (2011) and after (2014) the regrade. WIS-H' is diversity based on presence-absence data for WIS categories (as in Table 3).

Type	Category	2011	2012	2013	2014
Native status	Native	72.2	95.2	97.0	94.4
	Non-native	27.8	4.8	3.0	5.6
Longevity	Annual	1.6	0.8	2.4	2.0
	Biennial	0.5	0.5	-	0.2
	Perennial	97.9	98.7	97.6	97.8
Growth habit	Forb	77.1	73.0	64.9	60.1
	Graminoid	21.1	27.0	35.1	39.9
	Tree/shrub	1.8	-	-	-
WIS	OBL	88.5	98.8	97.9	96.7
	FACW	4.6	0.9	0.6	1.0
	FAC	5.0	0.2	1.4	2.1
	FACU	1.9	0.0	<0.1	0.2
WIS-H'		1.29	0.90	0.93	0.99

depletion of some species that were common in the seed bank before the regrade. Other species emerged that were not observed in 2011. For example, *S. tabernaemontani* was present after the regrade but not before; this species was unexpectedly common in the new standing vegetation.

The regrading project initially decimated the standing vegetation, but plant cover steadily increased. Like the seed bank, we observed a shift in species composition after regrading. Counter to our second hypothesis, the non-native *M. scorpioides* and *Typha x glauca*, 2 of the common species in the 2011 standing vegetation, did not rapidly establish compared to native species, like *S. latifolia*, *L. oryzoides*, and *S. tabernaemontani*. Although both seed banks contained few non-native species, we were surprised that non-native species cover in the new standing vegetation was lower, as many invasive species rapidly colonize disturbed sites (Bansal et al. 2019, Bowman-Cutway and Ehrenfeld 2010, D'Antonio and Meyerson 2002, Ehrenfeld 2008, Meyer et al. 2013); however, not all urban habitats have a high presence of exotics (e.g., Ehrenfeld 2005).

Although initially low, the presence of non-native species may increase, resulting in the need for management of these species. We observed only 6 seedlings of the non-native *Lythrum salicaria* L. (Purple Loosestrife) before the regrade and 2 after, perhaps signaling the beginning of *L. salicaria*'s invasion into the site. Continued dispersal of *L. salicaria* from outside the wetland and seed rain from established plants may increase the presence of *L. salicaria*. Although *Typha* had low seedling densities in both seed bank surveys, we predict that *Typha* will continue to spread vegetatively in the newly altered site, as *Typha* readily colonizes and forms monodominant stands in disturbed habitats (Bansal et al. 2019). 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 to 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.

After the regrade, we were surprised that *S. latifolia* and *Potamogeton* sp. were the 2 most common species in the standing vegetation because neither taxon was common in the seed bank surveys or in the 2011 standing vegetation (Table 4). We were surprised that *S. latifolia* and *Potamogeton* sp. were the 2 most common species in the standing vegetation after regrading, because neither taxon was common in the seed bank surveys or in the 2011 vegetation. The establishment of these 2 species may have resulted from hydrologic change coupled with asexual propagation, namely through the production of *S. latifolia* corms or tubers (Dorken and Barrett 2003, 2004; Van Drunen and Dorken 2012) and *Potamogeton* sp. rhizomes (Gleason and Cronquist 1991, Wiegleb and Brux 1991), respectively. Many *Potamogeton* species spread vegetatively from turions and the fragmentation of stolons and rhizomes (Combroux and Bornette 2004, Kaplan et al. 2014, Vári 2013). Asexual propagules of *S. latifolia* may be important for restoring vegetation, perhaps because they 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

Table 6. Mahalanobis D values based on presence-absence data for functional group types in the standing vegetation before vs. after the regrade.

Type	# categories	D	P
Native status	2	1.90	>0.05
Longevity	3	2.19	>0.05
Wetland Indicator Status	4	17.42	0.0016
Growth habit	3	4.75	>0.05

and Brux 1991), and 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 and Bornette 2004, Combroux et al. 2002).

In contrast to habitats with established plant communities, heavily disturbed urban wetlands have early successional, non-equilibrium communities that may exhibit substantial changes to the standing vegetation composition. A Gleasonian approach would predict that we would see changes in plant functional groups, with annual plant species rapidly establishing after a disturbance, followed by the colonization of clonal perennial species (Odland and del Moral 2002, van der Valk 1981). Although this approach suggests that clonal perennial species would eventually establish, the standing vegetation at Lieberman was dominated by perennial species in less than 3 growing seasons; many of these species were clonal graminoids (e.g., *Eleocharis palustris* [L.] Roem. and Schult. [Common Spikerush], *L. oryzoides*, and *S. tabernaemontani*). We also observed an increase in graminoids emerging from the seed bank, suggesting that something is favoring the establishment of graminoids in our system. We originally predicted that we would observe a compositional shift toward an increase in annual species as well as a greater diversity among WIS categories. Although we saw a slight increase in the relative percent cover of annuals, the vegetation after the regrade remained largely composed of perennial species; thus, our third hypothesis was incorrect. We also observed a greater percentage of OBL species (Table 5), rather than an increase in the diversity of WIS. This trend was not observed in the seed bank, where the percentages of OBL species remained constant. Changes in hydrology may have resulted in an environment that favored obligate wetland species; the wetland is conspicuously more inundated since the regrade (M Larson, pers. observ.), and much of the originally planned topography was not successfully constructed. Landscape architects and engineers need to pay particular attention to rehabilitating urban wetland hydrologies to favor wetland plant establishment of targeted functional groups (Schwab and Kiehl 2017, Wang et al. 2016).

### Management Implications

The complete regrading and expansion of this urban wetland effected a sharp decline in the number of seeds in the seed bank. This dilution may have contributed to shifts in both the standing vegetation and the 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, the standing vegetation recovered after 3 years, indicating a resilient urban wetland ecosystem.

While the current case study focuses on a single, relatively small wetland, we are not aware of any other sites in our region subjected to the same degree of disturbance. Yet we expect that created and constructed urban wetlands will become more common. The complete regrade and expansion of our urban wetland provided a unique opportunity to document changes in species composition of the seedlings that emerged from the seed bank as well as successional changes in the standing vegetation, and it is encouraging that the urban wetland recovered within 3 growing seasons without an increase in non-native species cover. It appears that seeding and planting may not always be necessary to promote the re-establishment of vegetation after an existing wetland is reconstructed. Allowing vegetation to reestablish from existing seed and bud banks may increase the likelihood of rapid recolonization of a diverse aquatic plant community (Alderton et al. 2017). While vegetation cover rapidly establishes in some systems (Meyer et al. 2013, Mitsch et al. 2005),

other studies estimate that the time required for an ecosystem to recover after wetland restoration or creation may be several decades, or even centuries (Curran et al. 2014; Johansen et al. 2017; Jones and Schmitz 2009; Moreno-Mateos et al. 2012, 2015; Stefanik and Mitsch 2012). Moreover, urban environmental conditions require modified restoration and rehabilitation designs to ensure restoration project success (Ravit et al. 2017). Future case studies like ours will increase our understanding of vegetation recovery of wetlands in urban landscapes.

### Acknowledgements

The authors thank Craig Reynolds, Evan Schulz, Anthony Paolini, and James Liebner for their assistance in the Research Greenhouse and field. We also thank Dr. Doug Wilcox for contributing to the design of the germination trays, Dr. Mark Blumler for assistance in plant identification, and Dr. Weixing Zhu for advice on statistical analyses. Anonymous reviewers provided constructive comments on earlier drafts of the manuscript. This study was partially supported by the Center for Integrated Watershed Studies (CIWS) of Binghamton University, State University of New York; a Wallace Research Foundation grant to Binghamton University; and a United States Environmental Protection Agency Region 2 Wetland Development Grant (EPA-R2-09WPDG) awarded to the Upper Susquehanna Coalition (USC), administrated by the Tioga County SWCD, and subcontracted to Binghamton University. Although the information in this document has been funded in part by the United States Environmental Protection Agency under assistance agreement CD-972253-09-0 to the Tioga County Soil and Water 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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