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ORIGINAL RESEARCH

# 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.

**Keywords** Urban wetland flora · Species richness · Invasive species · Soil electrical conductivity · Floristic quality assessment

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## 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.

## Methods

### 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.

### 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

**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 |

(1 $m^2$ quadrats) and shrub cover (10 $m^2$ quadrats). In each quadrat, percent cover estimates were recorded for each species (Mueller-Dombois 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.

## 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 $day^{-1}$.

### 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.

## Results

### 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.

### Soil Characteristics

Analysis of soil characteristics revealed that urban wetlands had significantly higher soil electrical conductivity than the

**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 | - |

natural wetland categories (Table 3; urban wetland median = 150 μS $cm^{-1}$, natural wetland median = 33 μS $cm^{-1}$; Kruskal-Wallis H = 14.6, $p$ = 0 .002). Urban wetland soil electrical conductivity ranged from 123 to 6380 μS $cm^{-1}$, while the range for natural wetlands was 23–243 μS $cm^{-1}$. 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–

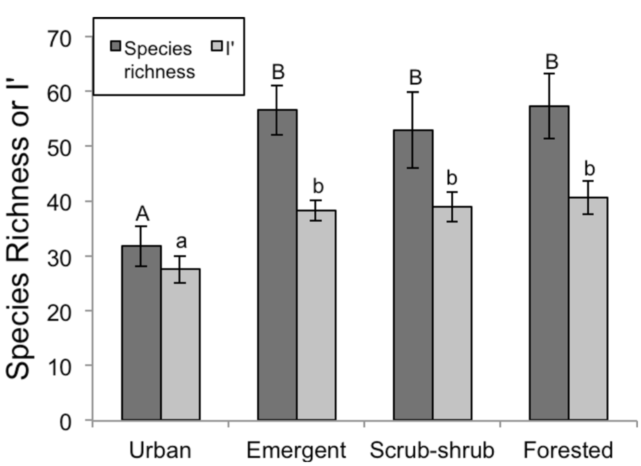


**Fig. 1** Mean species richness and mean adjusted FQAI (I') for each wetland habitat type (Urban ($n$ = 8), Emergent ($n$ = 7), Scrub-shrub ($n$ = 5), and Forested ($n$ = 6)), ± 1 SE. Means not sharing a common letter as a result of one-way ANOVA tests differ significantly at $p$ = 0.05 according to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

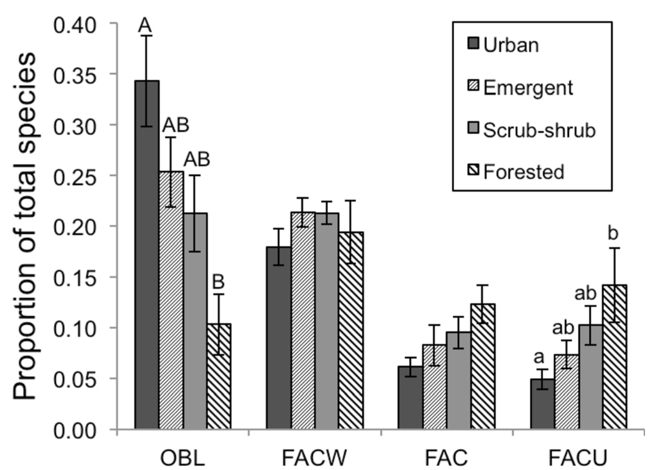


**Fig. 2** Mean proportions of USDA wetland indicator categories for each wetland habitat type, ± 1 SE (OBL = obligate wetland, FACW = facultative wetland, FAC = Facultative, FACU = Facultative upland). Means not sharing a common letter as a result of one-way ANOVA tests differ significantly at $p$ =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

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 study 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 $kg^{-1}$ $day^{-1}$).

### 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.

## 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.

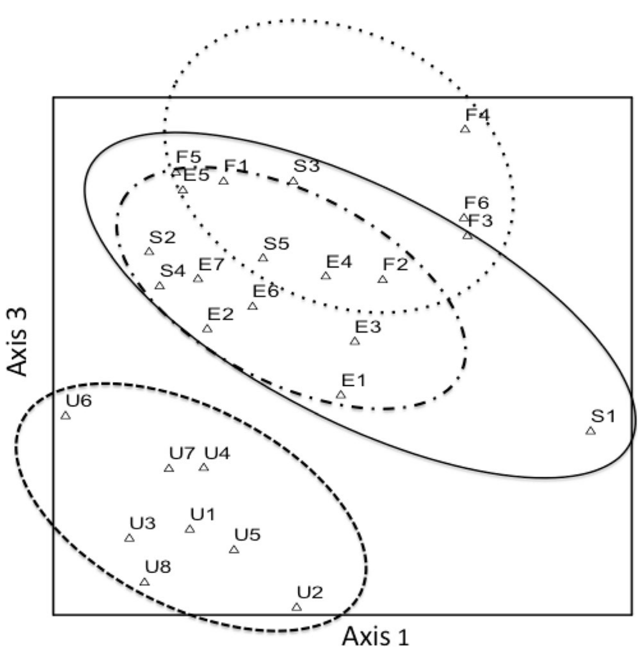


**Fig. 3** NMS ordination depicting the similarity among wetland sites ($n = 26$) based on species composition (presence/absence). U = urban, E = Emergent, S = Scrub-shrub, and F = Forested

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 only in 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

**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

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

**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

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.

## 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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