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Are stormwater detention ponds protecting urban aquatic ecosystems? a case study using depressional wetlands

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Abstract
Stormwater wet detention ponds (hereafter "detention ponds") are implemented to mitigate impacts of urban stormwater runoff on downstream waterbodies. We evaluated the effectiveness of detention ponds in providing this protection by quantifying hydrological, chemical, and biological responses in urban depressional wetlands with and without detention ponds draining into them and comparing these responses to non-urban reference depressional wetlands. We predicted if detention ponds protect waterbodies, then the hydrology, water and soil chemistry, and plant communities of urban depressional wetlands receiving detention pond drainage should be more similar to non-urban depressional wetlands than urban depressional wetlands not receiving detention pond drainage. We found wetlands receiving detention pond drainage had post-storm water level recession rates that were slower than wetlands not receiving detention pond drainage, but faster than non-urban wetlands. We also found wetlands with and without detention ponds draining into them (i.e., urban wetlands) to be more similar to each other regarding water and soil chemistry and vegetation than to non-urban wetlands. Compared to non-urban wetlands, both urban wetland types had elevated pH, inorganic nitrogen, and total phosphorus in their soils and waters, and greater coverage and species richness of disturbance-adapted native and nonnative plant species of lower conservation value. Differences in plant communities were related to changes in hydrology and water and soil chemistry, suggesting detention ponds need to better mitigate the effects of urbanization on these factors. Our findings reveal the need to improve detention pond efficiency and/or identify alternative strategies for protecting waterbodies from the effects of urbanization.

Keywords Hydrology · Stormwater management · Stormwater ponds · Urbanization · Water quality · Wetlands

Introduction

Global urban landcover is projected to double between 2010 and 2050 (Angel et al. 2011). This expansion raises concerns regarding the progression of habitat fragmentation and loss (Liu et al. 2016; Swenson and Franklin 2000), as well as the protection of already fragmented habitats. Fragmented habitats embedded within urban areas provide for a variety of organisms and can serve as hotspots for biodiversity (Cornelis and Hermy 2004; Tommasi et al. 2004). Understanding the factors that affect biodiversity within these embedded ecosystems can therefore contribute to efforts in conserving and enhancing urban biodiversity.

Aquatic ecosystems are particularly susceptible to the impacts of urbanization due to associated alterations in hydrological and chemical dynamics (Azous and Horner 2000). Aquatic ecosystems in urban landscapes often exhibit more rapid increases and decreases in water levels (i.e., flashier hydrology) than those in more natural landscapes due to increased surface runoff caused by more impervious surfaces (Chithra et al. 2015; Walsh et al. 2005). Flashier hydrology could result in increased soil erosion (Park et al. 2012) and alterations in plant communities of downstream aquatic ecosystems (Reinelt 1998). Furthermore,

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runoff from urban landscapes interacts with various sources of nitrogen and phosphorus, such as atmospheric deposition, fertilizers, lawn and garden waste, and pet waste, potentially increasing nutrient loadings in the waters and soils of receiving aquatic ecosystems (Hobbie et al. 2017; Law et al. 2010; Yang and Toor 2017). Additionally, minerals (e.g., calcium and sodium bicarbonate) can leach out of impervious surfaces, increasing pH and conductivity in receiving aquatic ecosystems (Wright et al. 2011; Kaushal et al. 2020). Over time, changes in water chemistry can lead to subsequent changes in soil/sediment chemistry (Reddy and DeLaune 2008). The subsequent increase in hydrological flashiness as well as soil and water nitrogen, phosphorus, conductivity, and pH in urban landscapes can affect plant community composition by potentially facilitating the establishment of more disturbance-tolerant plants, such as exotic and native facultative species (e.g., Ehrenfeld and Schneider 1993; Leishman et al. 2004).

Engineered ecosystems, such as stormwater ponds, green roofs, and constructed wetlands, are important tools utilized worldwide as 'Best Management Practices' (BMPs) to lessen urban impacts on aquatic ecosystems (Marsalek and Chocat 2002). Stormwater wet detention ponds (hereafter "detention ponds"), according to the United States Environmental Protection Agency (US EPA), are permanent wet basins primarily designed for downstream flow control (US EPA 1990). This BMP type has become widely used (Collins et al. 2010) due to both their utility in flood control (Lawrence et al. 2010) and ease in permitting (Filshill and Martin 2011). This widespread usage has led to the construction of tens of thousands of detention ponds in certain regions, with the rate of pond construction paralleling that of urban expansion (Beckingham et al. 2019; Sinclair et al. 2020).

Detention ponds are designed to detain urban stormwater runoff (Chen et al. 2007), capture nutrients contained within that runoff through sedimentation and assimilation (Hogan et al. 2007; Mallin et al. 2002), and steadily release runoff into downstream aquatic ecosystems via horizontal flow, decreasing potential downstream soil erosion and flooding (Park et al. 2012). Across Florida, USA, the minimum peak discharge requirement should not exceed pre-development peak discharge from the 25-year, 24-h return period storm event to protect downstream aquatic ecosystems from potential soil erosion (F.A.C. 62§330.010b (1–5) 2020). Excluding a few instances (e.g., Hancock et al. 2010), detention ponds meet their goal of flood control (e.g., Harper and Baker 2007). In contrast, detention ponds rarely meet presumed nutrient removal expectations (Beckingham et al. 2019). In Florida, USA, state regulations assume that detention ponds remove 80% of total nitrogen and total phosphorus that runs off urban landscapes. However, only 60%-65% of total phosphorus and less than 50% of total nitrogen is typically removed by detention ponds (Harper and Baker 2007). In addition, inorganic nitrogen entering detention ponds that is assimilated by plants can be released as both inorganic and organic forms when vegetation senesces (Reddy and DeLaune 2008). Unless denitrification occurs, nitrogen entering detention ponds can be transformed into other forms of nitrogen, eventually being transported downstream (Gold et al. 2019). Thus, detention ponds can act as net sources of nitrogen into downstream aquatic ecosystems (e.g., Lusk and Toor 2016). The continued reliance on detention ponds as a BMP, despite evidence of not meeting water quality goals, necessitates the critical evaluation of their effectiveness in mitigating urban impacts on downstream aquatic ecosystems into which they drain through horizontal flow.

Depressional wetlands embedded within urban areas are an ideal ecosystem to assess the effectiveness of detention ponds in mitigating the effects of urbanization on aquatic ecosystems. These wetlands are geographically isolated from other waterbodies, making their water sources runoff from upland habitats, groundwater seepage, and precipitation. Pathways by which water leaves are evapotranspiration and groundwater seepage (Tiner 2003). This hydrologic isolation means that elevated nutrients in depressional wetlands can be more directly connected to upland ecosystems rather than overland flow from other waterbodies (Hopkinson 1992). Furthermore, depressional wetlands exhibit more rapid changes in water levels than other waterbodies due to being surrounded by upland ecosystems (Ehrenfeld et al. 2003). One way to assess the hydrologic flashiness induced by urbanization is the wetland water level increase and decrease rates in response to storm events. Depressional wetlands also tend to have anaerobic, acidic, and low-nutrient soils (Reddy and DeLaune 2008), limiting the plant species that can reside in these conditions to more obligate (Lichvar 2016) and nutrient-conservative native species (Houlahan et al. 2006). Thus, comparing urban depressional wetlands with and without inputs from detention ponds provides an opportunity to directly assess the degree to which detention ponds help to protect aquatic ecosystems.

Depressional wetland plant communities, in particular, can assist in determining whether detention ponds buffer the effects of urbanization. This is because aquatic plant communities can act as indicators to hydrological (Hudon 2004; Magee and Kentula 2005) and nutrient dynamic alterations (Ehrenfeld and Schneider 1993). For instance, obligate species tend to decrease, and facultative species tend to increase, in abundance with increases in hydrologic flashiness and elevated nutrient conditions (Ehrenfeld and Schneider 1993). This response is because obligate species are typically adapted to a smaller hydrological niche of acidic, saturated/anaerobic soil conditions, while facultative species are adapted to a wide range of soil pH and saturation (Lichvar 2012). For native plant species, they tend to be less tolerant to disturbances, such as hydrologic and nutrient alterations associated with urbanization, leading to decreased native abundance, whereas exotic species tend to increase, especially in areas of high nutrient loads, due to their tolerance to disturbances (Ehrenfeld and Schneider 1993; Gavier-Pizarro et al. 2010).

The objectives of this project were to: (1) quantify the effects of detention ponds on hydrological dynamics and water and soil chemistry in depressional wetlands; (2) evaluate the effects of detention ponds on wetland plant communities (cover and species richness of obligate, facultative, native, and exotic plants); and (3) estimate the degree to which changes in plant communities are related to changes in hydrology and/or soil and water chemistry. To meet these objectives, we compared hydrology, soil and water chemistry, and plant communities of urban depressional wetlands having detention ponds draining into them to urban depressional wetlands not having detention ponds draining into them, as well as to non-urban, reference depressional wetlands. Comparing non-urban wetlands to urban wetlands with and without detention ponds draining into them allowed us to separate the general effects of urbanization from the direct effects of detention ponds. We hypothesized that if detention ponds are lessening urban impacts on receiving aquatic ecosystems, then wetlands receiving detention pond drainage would more closely resemble non-urban wetlands regarding hydrological dynamics, soil and water chemistry, and plant communities than urban wetlands not receiving detention pond drainage.

Methods

Study region and design

This study was conducted in Alachua County, FL, USA. Alachua County is 226,624 ha in size with approximately 270,000 residents living within nine cities/towns (www.alachuacounty.us; www.census.gov), and contains a mosaic of agricultural, natural, and urban/residential land covers. The climate is subtropical with wet seasons (May–September) averaging 32 °C and dry seasons (October–April) averaging 23 °C (www.usclimatedata.com).

To meet our objectives, we compared hydrological dynamics, water and soil chemistry, and plant communities among three different wetland types ($N=19$): urban wetlands receiving drainage from detention ponds (hereafter "stormwater wetlands"; $n=7$), urban wetlands not receiving drainage from detention ponds (hereafter "non-stormwater wetlands"; $n=6$), and non-urban reference wetlands (hereafter "non-urban wetlands"; $n=6$). See Fig. 1a for wetland locations. Both non-stormwater wetlands and stormwater wetlands were located adjacent to detention ponds. However, non-stormwater wetlands receive urban runoff via overland flow, while the majority of urban stormwater runoff entering stormwater wetlands does so only after passing through a detention pond. All urban wetlands were surrounded by similar population and housing densities. Each urban wetland was listed as a drainage easement and occurred on private land where no management was occurring. Non-urban wetlands tended to be surrounded by pine flatwoods and were sampled to control for overall effects of urbanization.

To locate candidate wetlands adjacent to detention ponds, we used current maps of Florida land cover provided by the Florida Department of Environmental Protection Geospatial Open Data (https://geodata.dep.state.fl.us/datasets/statewide-land-use-land-cover) and the City of Gainesville's (COG's) 2018 stormwater network maps (COG personal communication). We then conducted in-person visits to verify the isolation of candidate wetlands from other waterbodies. All surveyed wetlands were forested, seasonally flooded (i.e., they may dry out during Florida's dry season), and contained sandy, acidic soils (Supplement A). The plant communities comprised a mix of cypress and hardwood trees, understory shrubs, and a variety of herbaceous plants. Prior to the 1900's, the dominant land cover type in Alachua County was longleaf pine forests, floodplains, and wetlands (Envision Alachua 2013; Volk et al. 2017). Therefore, pre-existing conditions of the urban wetland sites were likely similar to present non-urban wetlands.

All detention ponds adjacent to either a stormwater or non-stormwater wetland ranged in age from 1980–2007, with similar age distributions between both wetland types (Appendix A). We did not choose wetlands adjacent to detention ponds that were built after 2007 because of the lack in time for urban effects to accumulate. Preliminary univariate and multivariate analyses revealed no evidence of detention pond age affecting water levels, soil and water chemistry, or plant communities, allowing us to exclude pond age from our analyses ($P \geq 0.16$).

For each wetland, we collected hydrological, chemical, and vegetation data from within six 3×3 m plots, and tree basal area data from 50 m^2 circular plots surrounding each of these six 3×3 m plots. Each 3×3 m plot was placed along two transects (three plots per transect) extending from the center to the edge of each wetland with one transect extending in a random direction on the northern and the other extending in a random direction on the southern half of each wetland to prevent transects from overlapping (Fig. 1b-c). The center plots of each transect were also placed at least 4 m away from the wetland center to prevent plot overlap.

Hydrological dynamics

We quantified rates of post-storm water level increase and decrease using water level data loggers (HOBO™ U20-001–04; ONSET Corporation, MA). Data loggers were

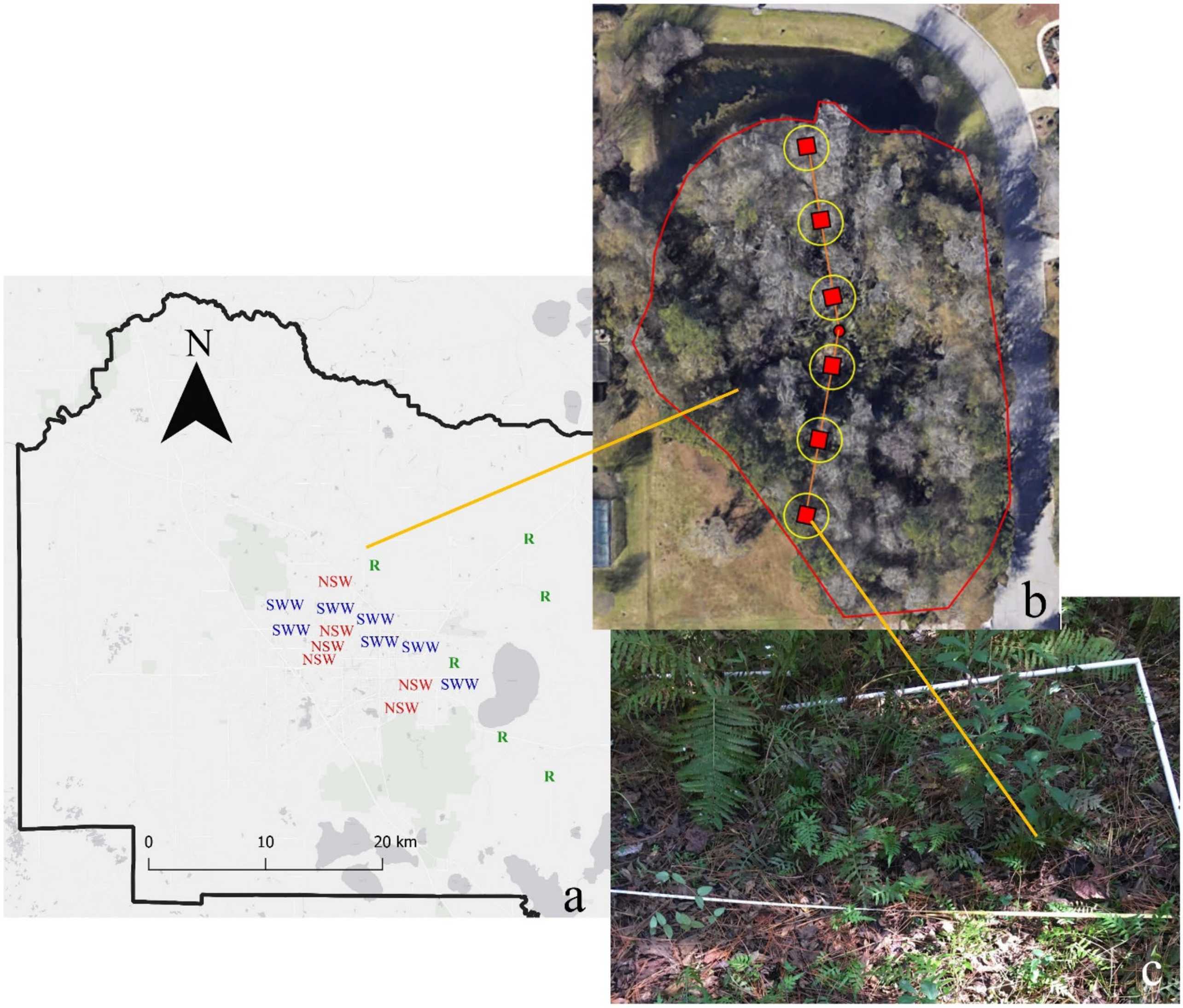

Fig. 1 Study design showing (**a**) locations of stormwater wetlands (SWW) in blue, non-stormwater wetlands (NSW) in red, and non-urban reference wetlands (R) in green; (**b**) an example of transect and plot layout, including six 3 × 3 m plots where vegetation cover, soil, and water was sampled and six 50 m^2 circular areas where tree basal was sampled (this wetland was an NSW with an adjacent detention pond located above the wetland. Both the wetland and detention ponds are surrounded by residential development); and (**c**) a view of an individual 3 × 3 m plot occurring at a wetland's edge

deployed during three separate two-week deployments occurring from 27-Aug-2018 to 27-Oct-2018 to capture water level fluctuations in the wetlands when our study region typically experiences daily thunderstorms with large pulses of water triggering urban stormwater runoff. Water levels were recorded at the wetland centers every two hours with each of the three wetland types being represented during each deployment. Two stormwater and two non-stormwater wetlands were excluded, as their water levels were too low to deploy loggers (≤ 5 cm). We used data from three weather stations located across Alachua County, FL to identify when storm events occurred, and the amount of precipitation produced by each event (range in precipitation per storm event = 0.03–2.80 cm across all stations). We identified storm events as those precipitation events where weather stations detected precipitation accumulation and by the rapid rise in water levels detected by our data loggers.

To quantify rates of water level increase, we subtracted pre-storm baseline water levels from post-storm peak water levels and divided this value by the amount of time to reach peak level. To quantify the rates of decrease, we subtracted post-storm baseline water levels (i.e., when water levels returned to pre-storm levels) from peak levels and divided this value by the amount of time to reach baseline level. We then averaged rates of increase and decrease across storm events (range = two to seven storm events per deployment).

Using Spearman correlations, we found no evidence of either post-storm rates of water increase or decrease being related to the number of storm events (rho = -0.28 and 0.07, respectively; $P \geq 0.20$). However, we did find weak evidence that the amount of precipitation was positively related to post-storm rates of water level increase ($r = 0.48$; $P = 0.08$), but not rates of decrease ($r = -0.09$; $P = 0.72$). We therefore included precipitation amount as a covariate in our statistical analyses (described below).

Soil and water chemistry

We collected water and soil samples, and quantified water pH and conductivity using a Hannah Instrument HI9829 multiparameter meter (Smithfield, RI, US), from three locations in each wetland between 10-Sept-2018 to 25-Sept-2018. One sampling point occurred in the center and the other two occurred at the end points of each transect. When standing water was absent at wetland edges, we walked along each transect towards the center until water was present. Water samples were not collected from two sites (one stormwater and one non-stormwater wetland) due to these wetlands drying out faster than anticipated prior. Water samples were filtered (Fisherbrand Glass Fiber Filter Circles with a 2.5 µm nominal pore size) the day of collection and stored at 4 °C until analyzed. Soil samples were collected using a 2.54 cm diameter soil auger, at the center and end point of each transect, then was air-dried at room temperature, and passed through a 2 mm sieve in the lab.

Water and soil samples were then sent to the University of Florida Institute of Food and Agricultural Science Analytical Research Laboratory (arl.ifas.ufl.edu) where they were processed using standard EPA methods to quantify ammonium-nitrogen (NH_4^+-N), nitrate-nitrogen (NO_3^--N), total Kjeldahl nitrogen (TKN), total phosphorus (TP), orthophosphate (Ortho-PO_4^{-3}; water only), and pH (soil only). We then calculated organic nitrogen (ON) as the difference between TKN and $NH4^+$-N, and total nitrogen (TN) as the sum of NO_3^--N and TKN. We quantified inorganic, organic, and total forms of N and P because all are constituents of urban stormwater runoff, and all are either available for plant or microbial uptake or can become available via microbial processes (Lusk and Toor 2016). There were no obvious differences in water chemistry values between locations within a wetland. Therefore, for analyses, we averaged the water chemistry values from samples collected at the inner and outer portions of the wetlands, whereas soil chemistry data from the wetland center and edges were analyzed separately due to differences in soil saturation.

Plant community

We estimated the areal cover of each individual plant species occurring within each of the six 3 × 3 m plots sampled per wetland using the following ordinal cover classes: 1: ≤ 1%; 2: 1–5%; 3: 5–25%; 4: 25–50%; 5: 50–75%; 6: 75–95%; 7: 95–100%. Species cover was estimated in two different vertical strata: the "field" strata (i.e., all plants < 1 m in height and all herbaceous species of any height) and the "shrub" strata (i.e., all woody plants and vines between 1–5 m). Species cover in an additional "tree" stratum was quantified as the basal area (m^2/ha) of all small and mature tree species > 5 m in height and with a diameter at breast height ≥ 2 cm within $50m^2$ circular plots surrounding the 3 × 3 m plots. The observed plants in each strata (Appendix B) were then classified as obligate, facultative, native and/or exotic (specified to the Atlantic Gulf Coastal Plain regional list) using Lichvar et al. (2016), United States Department of Agriculture Natural Resources Conservation Services' Plant Database (https://plants.sc.egov.usda.gov/java), and/or the Florida Native Plant Society's website (www.fnps.org).

From the cover estimates, we calculated wetland-level values for species richness and abundance of obligate, facultative, native, and exotic species. Species richness was calculated as the number of species identified in each plant category. To estimate abundance of each plant category, we first estimated the abundance for each individual species separately for each stratum (i.e., field, shrub, and tree). For the shrub and field strata, we averaged the midpoint values of each species' cover class (e.g., midpoint of cover class 4 = 37.5%) across all plots (i.e., six within each wetland). For trees, we averaged basal area for each tree species across all plots. From these values, we estimated abundance of obligate, facultative, native, and exotic species by summing cover or basal area estimates for all species belonging to that category separately for each vegetation stratum.

In addition to cover and abundance, we estimated overall plant community quality using the Floristic Quality metric (FQ) (Cohen et al. 2004). This metric simply averages the conservation coefficient (CC) values developed by Reiss et al. (2005) for plant species of depressional wetlands in northcentral Florida across all of the plant species occurring in a given wetland. Values range from zero to 10, with plant species having lower CC values being more tolerant to anthropogenic disturbance than those with higher CC values. Species having no CC value were assigned a value of zero, as they were indicative of disturbed conditions.

Statistical analysis

We used ANOVA to detect differences among wetland types in post-storm rates of water increase and decrease, as well as floristic quality. For the rates of water level increase and decrease, we included the average precipitation amounts as a covariate. Rates of water increase were log-transformed [$\log_{10}$ (X)] prior to analysis to meet assumptions for normality.

We used non-parametric permutational analysis of variance (PERMANOVA; Anderson 2001) and homogeneity of multivariate dispersion tests (PERMDISP; Anderson et al. 2006) to detect differences among wetland types in (a) soil chemistry, (b) water chemistry, (c) obligate and facultative species richness and cover, and (d) native and exotic species richness and cover. PERMANOVA was used to detect overall differences among wetland types, while PERMDISP was used to tests for differences among wetland types in dispersion, i.e., variability in multidimensional space. Multivariate tests were used to detect effects of detention ponds on overall soil and water chemistry and plant community composition and to limit Type I errors associated with repeated univariate analyses of covarying response variables (Legendre and Legendre 2012). The soil data response matrix used for these analyses contained chemical parameters for center and edge soils, the water data matrix contained estimates for overall water chemistry, and both vegetation data matrices contained cover/basal area estimates for each vegetation stratum (i.e., field, shrub, and tree), as well as overall species richness as vectors. Prior to analyses, multiple soil and water response variables were log transformed [$\log_{10}$ (X)] and vegetation response variables were Hellinger's transformed due to being right-skewed (reported in tables). All variables were then standardized ($\overline{X}$/ 2*SD) to control for different measurement units (Gelman 2008). Both the PERMANOVA and PERMDISP tests were conducted on Euclidean distance matrices using 10,000 permutations. We estimated pseudo P-values as the proportion of permutation tests that exhibited differences among wetland types as strong as or stronger than detected in our actual datasets.

We used Redundancy Analyses (RDAs) to visually explore the overall differences in the response data matrices detected by PERMANOVA and PERMDISP. Wetland type was the constraining, categorical explanatory variable in these analyses. To assess the degree to which each variable in the response matrix related to separation among wetland types in the resulting RDA, we graphed variables with the highest loadings (i.e., represents the relationship between variable and axis) relative to the constraining RDA axes where separation among wetland types occurred using vector overlays. High loadings were determined by the clear bimodal separation between high and low scores exhibited in each response matrix.

To assess the degree to which differences among wetland types in vegetation were related to differences in water level fluctuations and/or soil and water chemistry, we calculated Pearson correlation values (r) for relationships between wetland level values of soil and water parameters and wetland scores along the RDA axes where differences among wetland types in vegetation occurred. We did so only for soil and water parameters that contributed to separation among wetlands as revealed by higher levels of loading in initial RDAs. We reported and visualized these relationships by graphing vector overlays for those variables related at the $P \leq 0.10$ level.

All statistical analyses were performed using R version 3.5.1 (R Development Core Team 2018). PERMANOVA, PERMDISP, and RDAs were conducted using the 'vegan' package (Okansen et al. 2018). All data for our analyses can be found in [Data] Iannone et al. (In prep.)

Results

Effects on water levels

We found no differences among wetland types in post-storm rates of water level increase (ANOVA; $F_{2, 10} = 0.88$, $P = 0.44$). In contrast, stormwater wetland water levels decreased 11% slower than non-stormwater wetlands and 13% faster than non-urban wetlands (ANOVA; $F_{2,10} = 4.51$, $P = 0.04$). The water level rates of increase in stormwater, non-stormwater, and non-urban wetlands had medium (min, max) values that were, respectively, 0.19 (0.11, 0.24) cm/h, 0.13 (0.12, 0.89) cm/h, and 0.16 (0.005, 0.28). The water level decrease in stormwater, non-stormwater, and non-urban wetlands had median (min, max) values that were, respectively, -0.11 (-0.21, -0.084) cm/h, -0.17 (-0.29, -0.10) cm/h, and -0.076 (-0.11, -0.009) cm/h. Medians, min and max are reported rather than mean and SD due to data not being normally distributed.

Effects on soil and water chemistry

PERMANOVAs revealed statistically significant differences among wetland types in both soil and water chemistry, although soil differences were marginal (soils: pseudo-$P_{2, 18} = 0.08$; water: pseudo-$P_{2, 16} = 0.001$). PERMDISPs revealed soil and water chemistry to vary similarly within each wetland type (pseudo-$P_{2, 18} \geq 0.19$). Both urban wetland types were similar in soil and water chemistry and differed significantly from non-urban reference wetlands in water chemistry (Tables 1 and 2). However, stormwater wetlands did not differ significantly in soil chemistry from non-urban wetlands, whereas non-stormwater wetlands did (Table 1).

Subsequent RDAs revealed separation between both urban wetland types and non-urban wetlands in both soil and water chemistry (with weaker separation in soil compared to water chemistry), and little separation between urban stormwater and non-stormwater wetlands (Fig. 2a, b). Separation between urban and non-urban wetlands primarily occurred along RDA1 for both datasets, which accounted for 12% and 28% total variability in soil and water chemistry, respectively (Fig. 2a, b). For soil chemistry, this separation was related, illustrated by higher loading scores, to increases in pH and TP both in the center and outer soils of urban versus non-urban wetlands (RDA1 loadings = 0.25–0.57; Fig. 2a). For water chemistry, this separation was related (i.e., higher loading scores) to increases in pH, Ortho-P, TP, conductivity, NO_3^--N, and NH_4^+-N in urban versus non-urban wetlands (RDA1 loadings = 0.38–0.74; Fig. 2b). No other water or soil chemical parameters were strongly related to the separation detected between urban and non-urban wetlands (RDA1 loadings ≤ 0.17). Median (min, max) values for all soil and water parameters are listed in Tables 1 and 2, respectively.

Effects on plant community

PERMANOVAs revealed significant differences among wetland types in the cover and species richness of obligate and facultative species (PERMANOVA; pseudo-$P_{2, 18}$ = 0.0007), and the cover and species richness of native and exotic species (PERMANOVA; pseudo-$P_{2, 18}$ = 0.0003). PERMDISP analyses revealed greater within wetland-type variability in obligate and facultative plant cover and species richness for urban versus non-urban wetlands (PERMDISP; pseudo-$P_{2, 18}$ = 0.06). No differences within wetland type variability were detected for native and exotic cover and species richness (PERMDISP; pseudo-$P_{2, 18}$ = 0.23).

Follow-up RDAs revealed that the separation among urban and non-urban wetland types for obligate and facultative species, and for native and exotic species primarily occurred along RDA1, which explained 25% and 23% of total variability, respectively (Fig. 3a, b). RDA analyses also revealed greater variability among urban than non-urban wetlands in obligate and facultative plant cover and species richness (Fig. 3a). Regarding differences between urban and non-urban wetlands in obligate and facultative species, these differences were related (i.e., higher loading scores) to decreased obligate tree cover in urban versus non-urban wetlands, and increased facultative species richness, facultative tree cover, and obligate cover in the field and shrub layers in urban versus non-urban wetlands (RDA1 loadings = 0.39–0.72; Fig. 3a). Regarding differences between urban and non-urban wetlands in native and exotic species, these differences were related (i.e., higher loading scores) to decreased native shrub cover in urban versus non-urban wetlands, and increased exotic field cover, and increased native and exotic species richness in urban versus non-urban wetlands (RDA1 loadings = 0.43–0.59; Fig. 3b). Differences between urban and non-urban wetlands were not related to other plant community characteristics (RDA1 loadings ≤ 0.31). Median (min, max) values for species richness and cover of obligate/facultative and exotic/native species are reported in Tables 3 and 4, respectively.

ANOVA and post-hoc comparisons revealed that wetland-level floristic quality (FQ) was 31% lower in urban compared to non-urban wetlands (ANOVA; $F_{2, 18}$ = 7.17, P = 0.006), and FQ did not differ between stormwater and non-stormwater wetlands. The median (min, max) FQ values were 3.6 (3.2, 4.1) and 3.5 (2.0, 4.4) in stormwater and non-stormwater wetlands, respectively, while that for non-urban wetlands was 4.8 (3.8, 5.7).

Potential drivers of changes in the wetland plant communities

There was a range of factors that potentially influenced differences in plant communities between urban and non-urban wetlands. Pearson correlations revealed the increase in facultative species richness and tree cover, as well as cover of obligate plants in the field and shrub layer of urban vs non-urban wetlands were strongly related to increased water TP, Ortho-P, NH_4^+-N, pH, and NO_3^--N and soil pH (both inner and outer wetland sections) found in urban relative to non-urban wetlands (r = -0.43 to -0.70). Greater obligatory tree cover in non-urban vs. urban wetlands

Table 1 The median (min, max) of the soil chemical parameters at the center and edges of the wetlands for the different wetland types. NSW = non-stormwater wetlands; SWW = stormwater wetlands; and R = non-urban wetlands. Wetland types having different letter subscript were significantly different as revealed by PERMANOVA and post hoc comparisons. Medians (min, max) are reported rather than mean and SD due to data not being normally distributed.

Response Variables	NSW [a]	SWW [a,b]	R [b]
Soil Center			
NH_4^+-N (mg/kg) *	1.24 (0.56, 3.99)	1.01 (0.72, 9.87)	1.13 (0.77, 5.23)
pH	5.16 (4.31, 6.48)	5.55 (3.99, 6.06)	3.98 (3.68, 4.23)
TP (mg/kg) *	279.10 (63.03, 545.23)	589.84 (92.06, 3394.20)	137.95 (20.97, 594.04)
NO_3^--N (mg/kg) *	1.07 (0.073, 25.44)	0.94 (0.021, 52.25)	0.37 (0.17, 0.54)
Organic N (mg/kg) *	1440 (376.34, 9944)	1353 (754.74, 14408)	2047 (467.39, 9928)
TN (mg/kg) *	1445 (376.97, 9973.72)	1353 (755.53, 14470)	2049 (468.33, 9934)
Soil Edge			
NH_4^+-N (mg/kg) *	1.38 (0.68, 2.0)	1.77 (1.17, 3.55)	1.16 (0.82, 5.45)
pH	5.31 (4.59, 5.85)	4.56 (4.23, 5.85)	3.95 (3.77, 4.41)
TP (mg/kg) *	110.98 (89.86, 293.28)	343.21 (51.69, 544.82)	72.38 (18.20, 487.21)
NO_3^--N (mg/kg) *	0.20 (0.005, 0.53)	0.29 (0.017, 8.55)	0.24 (0.16, 2.38)
Organic N (mg/kg) *	754.14 (376.64, 1139.02)	1471.04 (533.42, 7506.99)	1261.32 (356.48, 8262.59)
TN (mg/kg) *	761.93 (377.86, 1140)	1475.34 (534.73, 7519)	1263 (357.46, 8270)

* Variable was log-transformed (Log_{10}[X]) prior to analysis.

Table 2 The median (min, max) of the water chemical parameters for the different wetland types. NSW = non-stormwater wetlands; SWW = stormwater wetlands; and R = non-urban wetlands. Wetland types having different letter subscript were significantly different as revealed by PERMANOVA and post hoc comparisons. Medians (min, max) are reported rather than mean and SD due to data not being normally distributed.

Response Variables	NSW [a]	SWW [a]	R [b]
NH_4^+-N (mg/L)	0.32 (0.13, 0.70)	0.24 (0.10, 0.43)	0.10 (0.07, 0.31)
Ortho-P (ug/L) *	173.21 (116.58, 330.32)	163.49 (8.48, 696.53)	23.24 (1.86, 288.41)
TP (ug/L) *	457.42 (247.57, 1236.14)	313.83 (109.39, 890.38)	128.32 (55.10, 369.23)
Organic N (mg/L)	1.74 (0.74, 6.66)	1.23 (0.95, 2.75)	1.77 (1.39, 5.11)
NO_3^--N (mg/L) *	0.02 (0.01, 0.07)	0.01 (0.01, 0.10)	0.01 (0.01, 0.01)
TN (mg/L) *	2.0 (1.07, 7.02)	1.43 (1.15, 3.07)	1.87 (1.47, 5.18)
pH	6.03 (4.54, 6.37)	5.97 (5.08, 6.61)	3.89 (3.63, 4.36)
Conductivity (µS/cm)	112.50 (66.50, 193.0)	163.25 (58.0, 234.75)	87.88 (55.25, 115.25)

* Variable was log-transformed (Log_{10}[X]) prior to analysis.

was strongly related to slower rates of post-storm water level decline detected in non-urban wetlands (r = 0.59; Fig. 4a).

Higher values for native and exotic species richness and exotic cover in the field layer in urban compared to non-urban wetlands along RDA1 were similarly related to increased soil pH (both inner and outer sections), water pH, TP, and Ortho-P (r = -0.54 to -0.72) in urban wetlands vs. non-urban wetlands, while greater native shrub cover was strongly related to slower rates of post-storm water level decline (r = 0.54) detected in non-urban wetlands (Fig. 4b). We found no evidence of other aspects of soil or water chemistry affecting the differences between urban and non-urban wetland plant communities ($P \geq 0.34$).

Discussion

Our study revealed little differences between urban wetlands with and without detention ponds draining into them regarding hydrology, soil and water chemistry, and plant communities, and that both urban wetland types differed from non-urban reference wetlands. These findings suggest

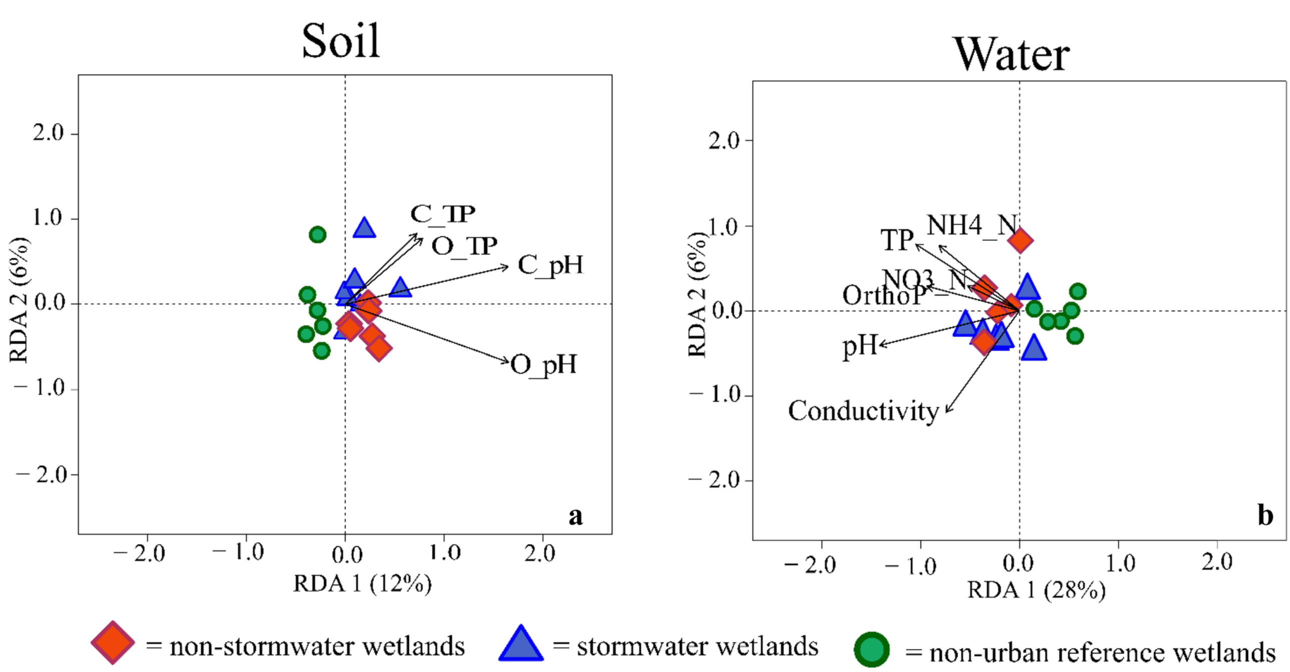

Fig. 2 RDA graph showing differences between urban wetlands and non-urban wetland types in (**a**) soil and (**b**) water chemistry. Values for soils collected in the wetland edge and center signified by an "O" and "C", respectively, prior to parameter name. Vectors show the variables with the greatest loading, i.e., those most related to separation among wetland types along RDA1

Table 3 The median (min, max) of obligate (OBL) and facultative (FAC) species richness and cover in the different vegetation strata (field, shrub, and tree) for the different wetland types. NSW = non-stormwater wetlands; SWW = stormwater wetlands; and R = non-urban wetlands Wetland types having different letter subscript were significantly different as revealed by PERMANOVA and post hoc comparisons. Medians (min, max) are reported rather than mean and SD due to data not being normally distributed.

Response Variables	NSW [a]	SWW [a]	R [b]
OBL Field Cover (%)	4.5 (0.0, 30.0)	3.1 (0.1, 21.5)	0.3 (0.1, 3.2)
FAC Field Cover (%)	1.5 (0.3, 3.8)	2.2 (0.5, 11.3)	0.4 (0.1, 4.12)
OBL Shrub Cover (%)	0.0 (0.0, 0.6)	0.0 (0.0, 0.2)	0.0 (0.0, 0.0)
FAC Shrub Cover (%)	0.8 (0.6, 1.7)	0.8 (0.0, 11.0)	0.5 (0.0, 1.2)
OBL Tree Cover (m^2/ha)	0.0 (0.0, 0.4)	0.0 (0.0, 56.7)	31.2 (16.3, 72.7)
FAC Tree Cover (m^2/ha)	20.0 (16.0, 52.1)	45.5 (3.3, 115.2)	7.1 (2.9, 16.6)
OBL Richness	4.0 (0.0, 9.0)	4.0 (1.0, 5.0)	2.0 (2.0, 6.0)
FAC Richness	8.0 (7.0, 14.0)	9.0 (4.0, 14.0)	5.0 (2.0, 7.0)

that detention ponds are not meeting their intended function of lessening urban impacts on downstream aquatic ecosystems into which they drain (echoing similar conclusions from Booth and Jackson 1997; Lusk and Toor 2016; Mallin 2002). For instance, despite detention ponds slowing post-storm rates of water decline, they did not do so sufficiently to mimic rates of non-urban wetlands. Both urban wetland types also had elevated pH in their soils and waters, as well as elevated conductivity and nutrient levels in their water relative to non-urban wetlands. Regarding plant communities, urban wetlands contained more facultative native and exotic plant species indicative of disturbance. Furthermore, these differences in vegetation were related to differences in hydrology and soil and water chemistry, suggesting detention ponds need to better mitigate these impacts of urbanization if they are to meet their intended goal of protecting downstream aquatic ecosystems for which they are often accredited.

Despite stormwater wetlands having post-storm rates of water level decline 11% slower than non-stormwater wetlands, these rates were still 13% faster than those of non-urban wetlands. This finding suggests that although detention ponds are likely providing some benefits regarding mitigating soil erosion by slowing water level decline (Fiener et al. 2005), they are not recreating pre-development hydrological dynamics of non-urban wetlands as intended. Therefore, detention ponds may not be supplying urban wetlands with water in ways that mimic the baseflow and overland flow experienced in non-urban wetlands, causing water levels to decline more rapidly producing flashier hydrology. Nevertheless, our hydrological data were collected over a short period during Florida's wet season. During the wet season, Florida storms are frequent with rapid rates of rainfall, which could promote flashier hydrology relative to winter/dry season storms, which are less frequent with steadier rates of rainfall. Collection of hydrological data during dry season is needed to determine if the inability of stormwater ponds to mimic non-urban hydrology extends beyond Florida's wet season.

Unlike hydrology, both stormwater and non-stormwater wetlands did not differ from one another regarding soil and water chemistry, suggesting that detention ponds are not mitigating the impacts of urban stormwater runoff on downstream soil and water chemistry, thereby, corroborating findings from other studies (e.g., Beckingham et al. 2019; Harper and Baker 2007; Lusk and Toor 2016). We

Table 4 The median (min, max) of native and exotic species richness and cover in the different vegetation strata (field, shrub, and tree) for the different wetland types. NSW = non-stormwater wetlands; SWW = stormwater wetlands; and R = non-urban wetlands. Wetland types having different letter subscript were significantly different as revealed by PERMANOVA and post hoc comparisons. Medians (min, max) are reported rather than mean and SD due to data not being normally distributed.

Response Variables	NSW [a]	SWW [a]	R [b]
Native Field Cover (%)	7.9 (5.2, 31.3)	14.0 (3.3, 28.8)	7.2 (1.0, 19.3)
Exotic Field Cover (%)	0.4 (0.0, 2.9)	0.2 (0.0, 6.2)	0.0 (0.0, 0.0)
Native Shrub Cover (%)	1.4 (0.7, 4.3)	1.3 (0.1, 11.1)	16.0 (1.7, 27.0)
Exotic Shrub Cover (%)	0.0 (0.0, 0.5)	0.0 (0.0, 0.1)	0.0 (0.0, 0.0)
Native Tree Cover (m^2/ha)	25.4 (18.8, 67.3)	60.3 (39.0, 117.6)	41.5 (37.0, 83.1)
Exotic Tree Cover (m^2/ha)	0.0 (0.0, 0.0)	0.0 (0.0, 0.8)	0.0 (0.0, 0.0)
Native Richness	24.0 (19.0, 28.0)	26.0 (19.0, 29.0)	15.5 (12.0, 23.0)
Exotic Richness	1.50 (0.0, 4.0)	2.0 (0.0, 4.0)	0.0 (0.0, 0.0)

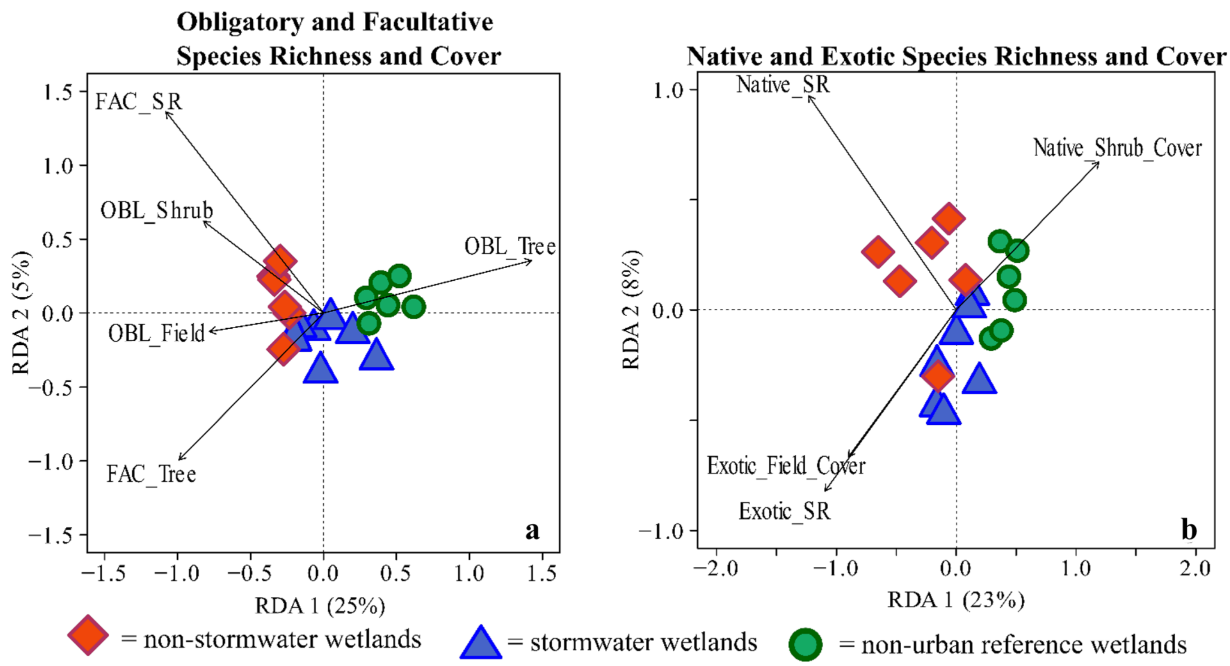

Fig. 3 RDA graphs showing differences between urban wetlands and non-urban wetland types in (**a**) obligate and facultative cover in the different strata and species richness and (**b**) native and exotic cover in the different strata and species richness. Vectors show the variables with the greatest degree of loading, i.e., those most related to separation among wetland types along RDA1. FAC and OBL = facultative and obligate, respectively. SR = species richness

found urban wetlands, regardless of whether they were protected by detention ponds, have higher TP and pH levels in their soils, and higher NH_4^+-N, NO_3^--N, ortho-P, TP, conductivity, and pH levels in their water, than non-urban

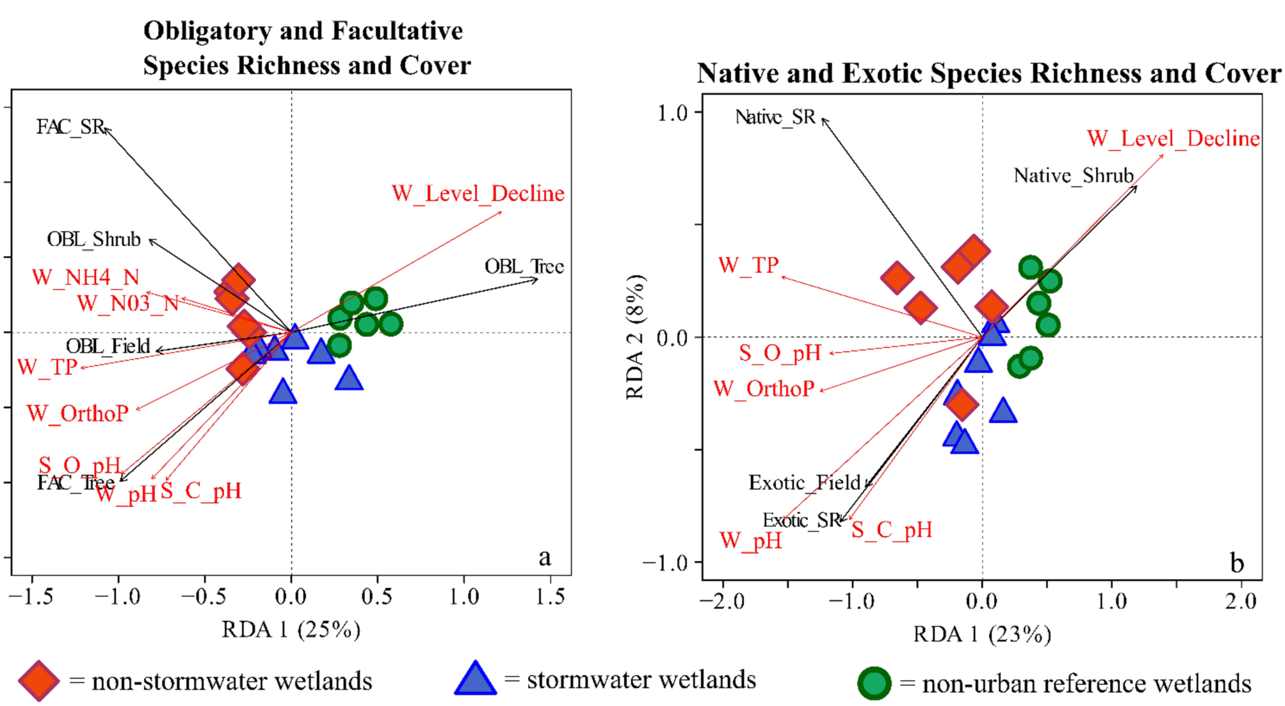

Fig. 4 Potential abiotic drivers of differences between urban and non-urban wetlands in (**a**) obligate and facultative cover in certain strata and species richness, (**b**) native and exotic cover in certain strata and species richness. Red vector overlays show the degree to which water (W) and soil (S) chemical parameters, including both soils on the outer portions (S_O) and center (S_C) portions of the wetlands are correlated to variation among wetlands occurring across RDA1 ($P \leq 0.10$). FAC and OBL = facultative and obligate, respectively. SR = species richness

wetlands. Sources of these differences from urban landscapes include pet waste (Fissore et al. 2012), plant material such as grass clippings and yard waste (Lusk and Toor 2016; Yang and Toor 2017), disturbed nutrient-rich parent soil material (Bachmann et al. 2012), fertilizer (Yang and Toor 2017), minerals leaching from impervious surfaces, and other construction materials (Wright et al. 2011; Kida and Kawahigashi 2015). Therefore, there is a need to either improve detention ponds ability to protect downstream aquatic ecosystems from these contaminants or for alternative stormwater management strategies to achieve these goals. We discuss potential alternatives below.

Stormwater and non-stormwater wetlands were also similar regarding plant communities. Both had higher facultative cover in the tree strata, higher exotic cover in the field strata, and higher overall facultative and exotic species richness relative to non-urban wetlands. These findings likely reflect the general tolerance of facultative and exotic species to disturbed and nutrient-rich conditions typical of urban wetlands (Ehrenfeld and Schneider 1993; Gavier-Pizarro et al. 2010; Magee and Kentula 2005). However, in contrast to other studies (e.g., Ehrenfeld and Schneider 1993; Miller and Zedler 2003), we found higher obligate cover within the field and shrub strata, and higher overall native species richness, in urban vs. non-urban wetlands. Further investigation of our data revealed that these contradictory findings might reflect the commonality of two obligate native plant species well-adapted for urban environments, *Spirodela polyrhriza* (Giant Duckweed) and *Cephalanthus occidentalis* (Buttonbush), in our urban wetlands. The lower floristic quality values found in the urban wetlands vs. the non-urban wetlands also suggest that the greater obligate cover and native species richness found in urban wetlands is due to increases in species adapted to disturbed, nutrient-rich conditions (Lopez and Fennessy 2002). Nevertheless, the contradictory increases in obligate and native species that we found relative to other urban wetland studies (Ehrenfeld and Schneider 1993; Miller and Zedler 2003) reinforces that the effects of urbanization on aquatic vegetation can differ geographically. Thus, assessing the efficacy of detention ponds at protecting aquatic plant communities in other urban locations is needed.

We found evidence that differences in the plant communities between urban and non-urban wetlands were driven both by rates of water level decline and differences in soil and water chemistry. Regarding water levels, we found that increases in obligate tree and native shrub coverage were positively related to slower rates of water level decline within non-urban wetlands. Other studies have also found greater abundance in obligate wetland species within wetlands with less dynamic water levels (Ehrenfeld and Schneider 1993; Magee and Kentula 2005), as these plant types have a narrower hydrological niche (Lichvar 2012). Increased cover of native shrubs found in non-urban wetlands also suggests these species are less adapted to more dynamic hydrological conditions found in urban wetlands. Collectively, these findings suggest that enhancing the ability of detention ponds to slow rates of post-storm water level declines via lowering discharge rates/extending discharge duration may help mitigate the effects of urbanization on the plant communities. We discuss some strategies below.

We also found evidence that differences between urban and non-urban wetlands in plant communities were driven by impacts of urbanization on soil and water chemistry. Differences between urban and non-urban wetlands in native and exotic species richness and cover were strongly related to increased soil and water pH, water TP and Ortho-P in urban wetlands. These findings agree with those of other studies (Ehrenfeld et al. 2001; King and Buckney 2001; Leishman et al. 2004) and may reflect increased mineral leaching from impervious surfaces and higher usage, and runoff, of fertilizers in urban landscapes (Park et al. 2012; Weidenhamer and Calloway 2010). However, phosphorus is typically not included in fertilizers in our study region due to naturally high levels of soil phosphorus, suggesting alternative sources of phosphorus such as pet waste and/or increased soil erosion (Hobbie et al. 2017).

The increased obligate and facultative species richness and cover found in urban wetlands were also strongly related to increased soil and water pH, and water TP and Ortho-P, found in urban versus non-urban wetlands. These differences were also related to increased water NO_3^--N and NH_4^+-N found in urban wetlands, which can result from fertilizer runoff or atmospheric deposition (Hobbie et al. 2017). Despite the relative closeness of urban vs. non-urban wetlands in our study, atmospheric deposition in urban wetlands can be higher due to localized sources such as motor vehicle emissions (Bettez and Groffman 2013). The positive relationship between phosphorus and obligate plant cover may reflect the commonality of *S. polyrhriza* in the urban wetlands, a species adapted to phosphorus-rich soils and waters (Perniel et al. 1998).

Differences between urban and non-urban wetlands in propagule pressure could have contributed to differences in plant communities but was likely not the only contributing factor given the degree to which differences in plant communities were related to differences in hydrology and soil and water chemistry. Nevertheless, longer-term studies are needed to determine the relative degree to which urbanization changes plant communities in depressional wetlands via changes in hydrology, soil and water chemistry, and/or propagule pressure. Collectively, relationships between differences in plant communities and soil and water chemistry suggest that efforts to protect urban aquatic plant communities from stormwater runoff need to better mitigate the effects of urbanization on soil and water chemistry.

Since our results suggest that increases in pH, phosphorus, and nitrogen are the strongest drivers of changes in the urban wetland plant communities, strategies to mitigate these changes are needed. Building larger, or chains of, detention ponds to increase water residence times and subsequent nutrient sedimentation, assimilation, and/or transformation may enhance mitigation (Mallin et al. 2002). Detention ponds can also be designed to have multiple sections, each serving different functions, such as forebays to trap phosphorus and nitrogen-rich sediment, and planted sections for nutrient uptake (e.g., Scarborough and Mensinger 2005; Mallin et al. 2002). Additionally, incorporating other stormwater control measures into the watershed, upstream of a detention pond, such as bioretention, swales, and other infiltration practices, can help decrease nutrient loads to downstream aquatic ecosystems (Cizek and Hunt 2013). However, these strategies are costly and require more land (Booth and Jackson 1997). Therefore, simple implementations, such as trash racks and/or debris screens installed at inflows and outflows, accompanied by regular maintenance, can prevent nutrient-rich materials, such as lawn clippings, from entering or leaving detention ponds (Strynchuk et al. 2000).

Altering detention pond plantings may also help mitigate changes in soil and water chemistry. Vegetated buffer strips around receiving aquatic ecosystems can decrease erosion and sediment input (Correll 2005), while internal vegetation, such as floating wetlands, can enhance nutrient uptake through assimilation (Chang et al. 2013) and may also promote permanent nitrogen removal via denitrification (Hohman et al. 2021). Emergent aquatic plants tend to decrease phosphorous (Borne 2014), while submerged aquatic plants tend to decrease minerals, potentially decreasing pH, in the water (Gu and Dreschel 2008). However, nutrient assimilation by vegetation is only temporary until the vegetation senesces. Therefore, future research on the long-term fate of vegetative assimilated nutrients is needed.

In addition to those research needs already stated, other factors need consideration when assessing the degree to which detention ponds protect the aquatic ecosystems into which they drain. First, determining the degree to which detention ponds protect aquatic ecosystems other than depressional wetlands (e.g., streams, lakes) would be beneficial. More information on how pond management strategies, such as algaecide use or bank and littoral shelf plantings, affect downstream aquatic ecosystems is also needed. Studies have found that establishing vegetation in detention ponds can improve the water quality within the ponds (Chang et al. 2013; Mallin et al. 2002). However, the efficacy of these plantings for protecting downstream aquatic ecosystems is unclear. Finally, despite floristic quality being lower in urban wetlands, these wetlands had higher obligate cover and native species richness than non-urban wetlands, suggesting these urban wetlands do have conservation value. Therefore, studies aimed at understanding this conservation value, and how to protect, restore, and enhance it are needed.

Conclusion

Aquatic ecosystems embedded within and adjacent to urban landscapes are susceptible to the impacts of altered hydrological and nutrient dynamics (Azous and Horner 2000), necessitating strategies to mitigate these impacts. Detention ponds are implemented in this regard with the goals of reducing peak storm flows (Chen et al. 2007) and removing pollutants through sedimentation and assimilation (Hogan et al. 2007; Mallin et al. 2002). However, our study shows that the degree to which detention ponds meet these goals is insufficient to maintain the hydrological, chemical, and biological characteristics of the wetlands into which they drain. Therefore, more work is needed to either enhance the efficacy of detention ponds and/or utilize alternative strategies to mitigate urban impacts on aquatic ecosystems. Detention ponds are used widely, therefore enhancing their ability to improve water quality through management strategies (e.g., plantings) or design (e.g., longer detention times) is globally relevant for addressing urban impacts on aquatic ecosystems. Finally, by revealing that a heavily relied on BMP does not meet one of its primary goals, our investigation suggests the need to confirm the degree of environmental protection provided by other highly adopted BMPs.

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Author contribution All authors contributed to the study design and interpretation of results. Kayla M. Hess and Basil V. Iannone III led data collection. Kayla M. Hess, Basil V. Iannone III, and James S. Sinclair led data analyses. Kayla M. Hess wrote the first draft of this manuscript with inputs and edits from all other authors. All authors read and approved the final manuscript.

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Data availability The data can be found in the Institutional Repository at the University of Florida (see Iannone et al., 2022).

Declarations

Ethics approval Not Applicable.

Consent to participate Not Applicable.

Consent for publication All authors have contributed to and approved the manuscript's content. The materials within this manuscript are neither published nor being considered for publication elsewhere.

Conflict of interests The authors declare no conflicts of interest.

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