



Nutrient retention via sedimentation in a created urban stormwater treatment wetland

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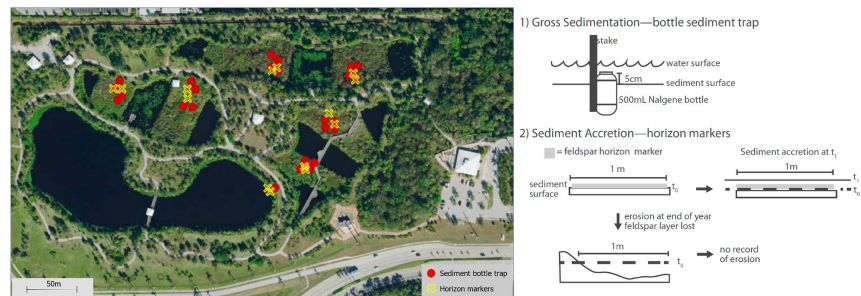
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HIGHLIGHTS

- A created stormwater treatment wetland may lose its nutrient retention capacity.
- The lithosphere sedimentation study aims to remove nutrients from the hydrosphere.
- Sediment bottle traps and horizon markers estimate gross and net sedimentation.
- Wetlands must be designed to minimize resuspension to maximize nutrient retention.
- More wetlands should be placed in the watershed to minimize algal blooms.

GRAPHICAL ABSTRACT



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ABSTRACT

Nutrient removal by a 4.6-ha urban stormwater treatment wetland system in a 20-ha water/nature park in southwest Florida has been investigated for several years, suggesting that the wetlands are significant sinks of both phosphorus and nitrogen although with a slightly decreased total phosphorus retention in recent years. This study investigates the role of sedimentation on changes in nutrient concentrations and fluxes through these wetlands. Sedimentation bottles along with sediment nutrient analyses every six months allowed us to estimate gross sedimentation rates of $9.9 \pm 0.1 \text{ cm yr}^{-1}$ and nutrient sedimentation rates of approximately $7.8 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $81.7 \text{ g-N m}^{-2} \text{ yr}^{-1}$. Using a horizon marker method to account for lack of resuspension in the sedimentation bottles suggested that net nutrient retention by sedimentation may be closer to $1.5 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $33.2 \text{ g-N m}^{-2} \text{ yr}^{-1}$. Annual nutrient retention of the wetland system determined from water quality measurements at the inflow and outflow averaged $4.23 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $11.91 \text{ g-N m}^{-2} \text{ yr}^{-1}$, suggesting that sedimentation is a significant pathway for nutrient retention in these urban wetlands and that resuspension is playing a significant role in reintroducing nutrients, especially phosphorus, to the water column. These results also suggest that additional sources of nitrogen not in our current nutrient budgets may be affecting overall nutrient retention.

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1. Introduction

Nutrient pollution and eutrophication continue to afflict many aquatic ecosystems in urbanized communities (Hobbie et al., 2017; Janke et al., 2017). Excessive nutrient loads into waterways, in addition to increased temperatures and intense storm frequencies, can lead to

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downstream eutrophication and expansion of algal blooms (Paerl, 1997; Paerl et al., 2016, 2018). Despite knowledge of the negative effects of nutrient enhancement to the landscape, human activities have increased the flux of nitrogen and phosphorus into the ocean by over 3-fold in just a few decades (Howarth et al., 2002; Glibert, 2017; Glibert and Burford, 2017). South Florida has been at the center of many local and national news stories because of large algal blooms caused by excess nutrients entering the waterways (Ballard, 2018; Bojorquez, 2018; Sweedler, 2018).

Algal blooms, although sometimes natural phenomena, have been increasing in size, frequency, and public attention in the past decades (Anderson et al., 2002; Brand and Compton, 2007; Wells et al., 2015). A lot of attention has been focused on Florida, as algae blooms in both fresh and saltwater environments cause anoxic condition, human health concerns, and fish kills (Gannon et al., 2009; Hu et al., 2006; Kirkpatrick et al., 2004; Weisberg et al., 2019). Marine and freshwater phytoplankton productivity has been linked to nitrogen and phosphorus inputs, respectively (D. M. Anderson et al., 2002; Conley et al., 2009). There is a strong correlation between nitrate loading and phytoplankton abundance (Heisler et al., 2008), yet globally, nitrogen and phosphorus are being added to the landscape at increasing rates (FAO, 2015). These issues continue to plague the world's aquatic systems despite the documentation that wetlands can remove excess nutrients from domestic wastewater, farm runoff, and urban stormwater runoff (Kovacic et al., 2000; Fink and Mitsch, 2004; Mitsch et al., 2014b; Land et al., 2016; Griffiths and Mitsch, 2017).

Natural processes in treatment wetlands make them a useful tool to help mitigate the effects of added nutrients to the landscape (Mitsch and Gosselink, 2015). Retention of nitrogen and phosphorus in the sediments of a wetland can be one effective pathway for removing them from the water column (Mitsch and Gosselink, 2015; Griffiths and Mitsch, 2017). Created treatment wetlands can be used to reduce nutrient levels from agricultural runoff, domestic wastewater, and urban stormwater (Mitsch et al., 2014b; Land et al., 2016; Griffiths and Mitsch, 2017). The main processes of nitrogen retention are microbial denitrification, algal uptake, sedimentation, and vegetative uptake, while phosphorus retention is often controlled by algal uptake, co-precipitation, sedimentation, sorption, and vegetative uptake (Saunders and Kalff, 2001; Vymazal, 2007). By identifying the primary processes responsible for nutrient removal in a given system, wetlands can be better created and managed for improved nutrient retention.

Sedimentation contributes significantly to nutrient and contamination removal from the water column (Nahlik and Mitsch, 2008; Mitsch et al., 2014b). Phosphates have a negative ionic charge, making them very susceptible to bind to positively charged minerals such as calcium and iron and fall as sediments to the bottom of the water column (Nahlik and Mitsch, 2008). Particulate organic nitrogen is removed from the water column via sedimentation in constructed wetlands, where the nitrogen either becomes trapped within the sediments or denitrifying bacteria transform nitrogen at the sediment-water interface into other inert forms (Taylor et al., 2005; Lee et al., 2009). By understanding how nutrients are retained within the sediments of a created wetland, we can better understand how to maximize nutrient retention within the system.

1.1. Goals and objectives

The goal of this study is to quantify the rate of sedimentation and mass of nitrogen and phosphorus removed via this process to determine the significance of sedimentation in the nutrient retention of the wetlands. By understanding this process, treatment wetlands can be better designed to enhance nutrient retention.

1.2. Hypotheses

The hypotheses tested are:

1. Sedimentation plays a significant role in nutrient retention.
2. Resuspension of sediments is the main process responsible for the decreasing phosphorus retention efficiency within the wetlands because it reintroduces phosphorus to the water column, sometimes making sediments a source, rather than a sink, of phosphorus.
3. Nutrient saturation within the soils are causing the wetlands to less effectively retain nutrients.

2. Materials and methods

2.1. Study site

This study was conducted at Freedom Park in Naples, Florida, USA (26°10'28"N, 81°47'22"W). Freedom Park is a 20-ha stormwater treatment wetland and restored bottomland system constructed from 2007 to 2009. The source of Freedom Park's water comes from storm runoff from about 3100 ha of residential, industrial, commercial, and recreational land (Fig. 1; Bishop et al., 2014; Nesbit and Mitsch, 2018). The wetland complex at Freedom Park is composed of a 1.9 ha settling pond three constructed wetlands totaling 2.7 ha that, in sequence, naturally remove nutrients from the urban runoff before it reaches the Gordon River (Bishop et al., 2014). The Gordon River discharges into Naples

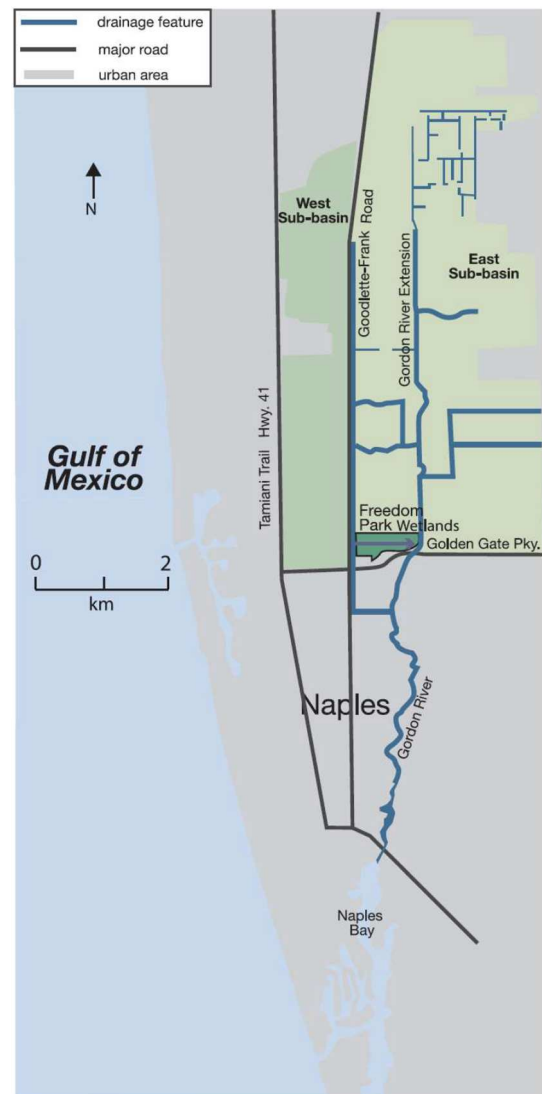


Fig. 1. Freedom Park watershed in urban Naples, Florida. From Nesbit and Mitsch (2018).

Bay which flows into the Gulf of Mexico. The inflow to the wetlands is pumped from a stormwater culvert on the northeast side of the wetland park into the settling pond and the treatment wetlands outflow is managed by a weir. Water between wetlands flows via gravity through three wetlands that are made up of alternating deep- and shallow-water sections. Each wetland has a total of five cells beginning with a deep-water, ponded cell which averages 1.3 m in depth, followed by a shallow-water, vegetated cell which averages 15–20 cm in depth; within each wetland there are three deep-water cells and two shallow-water cells. The main purpose of creating the wetlands was to remove nutrients and other pollutants from the water before it reached the downstream ecosystems. Recent studies of the wetlands found nitrogen and phosphorus retention are similar to other urban stormwater wetlands on average, but during large storm events, nutrients are not as efficiently removed (Griffiths and Mitsch, 2017; Nesbit and Mitsch, 2018). Annually, phosphorus retention is about 48% and nitrogen retention is about 26% (Griffiths and Mitsch, 2017).

2.2. Sediment sampling and analyses

In May 2016, bottle sediment traps were placed in Freedom Park to collect suspended sediments that fall out of the water column. Four 500-mL wide-mouth Nalgene bottles were placed in each vegetated section of Freedom Park 9 m apart from one another as well as in the pond (Fig. 2). Each of the Nalgene bottles were capped and attached to a reinforcing steel bar. Every six months, the rebar and bottles were placed at each site by pressing the rebar into the ground, burying the bottle into the soil so that the opening was approximately 5 cm above the soil surface. This ensures that only sediments falling from the water column are trapped within the bottle and prevents surrounding sediments from sliding into the bottle and distorting the results. The bottles were capped when placed and the caps were removed approximately 1 h later once the sediments had settled from the water column to prevent human disturbed sediments from being collected. The sediment bottles were placed every six months in May (the beginning of the wet season

when water depth ranged from 0 to 10 cm) and in November (the beginning of the dry season when water depth ranged from 0 to 30 cm). Every 6 months, the sediment bottles were capped and removed from the wetlands and replaced with clean, acid washed bottles. The collected bottles were returned to the lab where sediment accumulation depth was measured, dried at 30 °C for 48–72 h, or until constant weight is achieved, and weighed. These dry weights were used to determine sediment accumulation rates and, with the original sediment volume, bulk density. Additionally, samples were ground using a mortar and pestle so that sediments could pass through a 2 mm sieve and sent to The Ohio State University STAR lab for nutrient analysis.

In November 2017, feldspar horizon markers were placed in the wetlands to help determine net sedimentation for the previously collected bottle samples. A total of fourteen horizon markers were placed using methods adapted from Baumann et al. (1984) and Harter and Mitsch (2003). Two horizon markers were placed in each vegetation section of each wetland and two in the pond (Fig. 2). Horizon markers were placed at the beginning of the dry season as water levels were dropping. Water level at each site ranged from 0 to 25 cm during application. Each site was marked with a tall PVC pipe prior to feldspar application. Approximately 1 cm of feldspar was applied to a 1.3 m² area in a circle at each site by placing a large trash can with the bottom removed into the soil at the site, pouring feldspar into the top of the trash can, and waiting for the feldspar to settle out of the water column (where water was present). A constructed, movable wooden boardwalk was used to place and remove the trash can at each site to prevent disturbance of the sediments during horizon marker placement and subsequent sampling. Horizon markers were in place for approximately one year before a cryogenic core was taken (Knaus and Cahoon, 1990; Harter and Mitsch, 2003; Mitsch et al., 2014a), and depth of sediment accumulated above the white horizon marker over the year was measured.

Because the sediment bottles did not allow resuspension, they only provided a gross estimate of sedimentation. The horizon markers, on the other hand, provided an estimate of net sedimentation since sediments could be resuspended. Comparing the net sedimentation rate

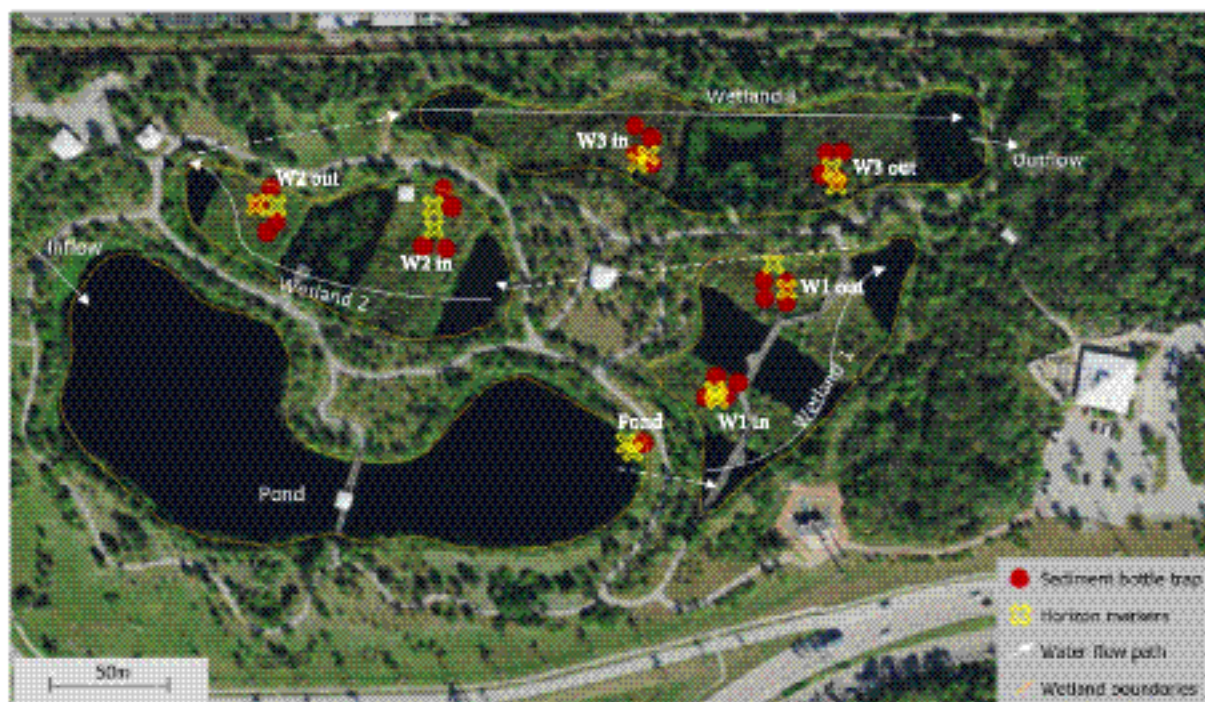


Fig. 2. Freedom Park wetlands with locations of sedimentation bottle traps (dots) and horizon marker placement (X's) with the site names adjacent to the locations. Also shown are the inflow where stormwater is pumped into the pond, flow paths including underground pipes controlled by gravity between wetlands (dotted arrows) and flow within the wetlands (solid arrows). The boundaries of the pond and wetlands are denoted by the solid line.

with gross sedimentation measured by the bottle traps for the same time period was used with soil nutrient concentrations to estimate annual nutrient retention via sedimentation.

2.3. Water sampling and hydrology

Water quality was collected as described by Griffiths and Mitsch (2017) every two weeks beginning March 2, 2016 through June 13, 2018. Hydrologic outflow of the wetland system was determined using a continuous water stage monitor placed at the outflow weir and inflow was determined with a combination of inflow readings and flow meters at inflow pumps by Nesbit and Mitsch (2018) from August 2016 through June 2018. The combination of these data is used to determine nutrient loading into the wetlands and retention within the wetlands at Freedom Park from August 2016–June 2018.

2.4. Statistical analysis

Data were analyzed using JMP Pro 14 (SAS Institute Inc., Cary, NC, USA). Normality of all data was tested. Sedimentation and nutrient retention data failed to meet the criteria for normal distribution. Non-parametric Kruskal-Wallis analyses were used to determine the relationship between season and both sedimentation and nutrient; significance was determined at a level of $p < 0.05$. A Wilcoxon-signed rank test was used to determine the difference between the sedimentation rates using the gross bottle method and the horizon marker method.

Water quality and mass nutrient retention data met constraints for normality. Regressions and t -tests were used to determine the effect of season on nutrient retention within Freedom Park. Sedimentation rates and mass net nutrient retention via sedimentation were transformed by taking the log of the data in order to meet the normality constraints of the tests. An ANOVA was used on the transformed data to determine the difference of net sedimentation between each wetland cell. A Tukey test was used to determine which wetland cells were statistically different from each other.

3. Results

3.1. Sedimentation rates

Throughout the study, the highest gross sedimentation rates measured by the sedimentation bottles were recorded in the wet season 2016 with an average of $15.3 \pm 2.6 \text{ cm yr}^{-1}$. The subsequent dry season (2016–17) had the lowest recorded sedimentation rates with an average of $0.62 \pm 0.07 \text{ cm yr}^{-1}$. The average gross sedimentation rate over the entire study period was $9.9 \pm 0.1 \text{ cm yr}^{-1}$ or $9.7 \pm 3.7 \text{ kg-dry m}^{-2} \text{ yr}^{-1}$. There is a significant difference between the sedimentation rates based on the season that they were collected (Kruskal-Wallis, $p < 0.01$). The 2016 and 2018 wet seasons had significantly greater sedimentation rates than the 2016–17 and 2017–18 dry seasons (Fig. 3). The wet season 2017 did not have statistically different sedimentation rates from the other seasons. There is no statistical difference in sedimentation rates between the wetland cells (Kruskal-Wallis, $p = 0.43$).

From November 2017 to November 2018, gross sedimentation rates using the bottle trap method average $11.9 \pm 1.5 \text{ cm yr}^{-1}$ or $3.2 \pm 0.3 \text{ kg-dry m}^{-2} \text{ yr}^{-1}$ (Table 1). During the same time, net sedimentation rates using the horizon marker method averaged $3.87 \pm 1.10 \text{ cm yr}^{-1}$ or approximately $1.0 \text{ kg-dry m}^{-2} \text{ yr}^{-1}$. The horizon marker method resulted in a 67.6% lower sedimentation rate than sedimentation measured by the bottle trap method, a statistically significant difference (Wilcoxon, $p < 0.01$).

3.2. Nutrient loading and retention

Average nutrient retention within the wetlands was $3.24 \pm 1.20 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $10.2 \pm 3.2 \text{ g-N m}^{-2} \text{ yr}^{-1}$ during the entire sample period

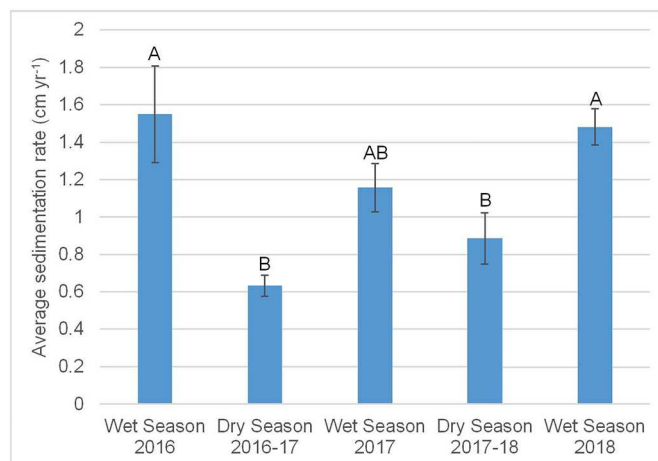


Fig. 3. Average sedimentation rate during wet and dry seasons in study period. Error bars represent standard error. Similarity letters represent the results of Kruskal-Wallis test performed on sedimentation rate for each season.

(August 2016–March 2018; Fig. 4). From August 2016 through July 2017, annual nutrient retention was $4.23 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $11.9 \text{ g-N m}^{-2} \text{ yr}^{-1}$. Both phosphorus and nitrogen retention were greater during the wet season (May–October) than the dry season (November–April; t -test, $p < 0.01$ and $p = 0.03$ respectively).

When only analyzing dry season data, phosphorus removal increases at an average rate of $0.048 \text{ g-P m}^{-2} \text{ yr}^{-1}$ from November 2017 to April 2018 (regression, $p = 0.03$) whereas in the wet seasons, the phosphorus retention rate is negative at an average of $-0.708 \text{ g-P m}^{-2} \text{ yr}^{-1}$ from May 2016 to October 2017 (regression, $p = 0.06$; Fig. 5a, b) suggesting decreased ability of the wetlands to retain phosphorus as they export phosphorus from the system. Nitrogen retention follows a similar trend with nitrogen retention during throughout the dry seasons of $0.54 \text{ g-N m}^{-2} \text{ yr}^{-1}$ (regression, $p = 0.01$) and net nitrogen retention of $-2.15 \text{ g-N m}^{-2} \text{ yr}^{-1}$ through the wet seasons, suggesting an export of nitrogen (regression, $p = 0.03$; Fig. 5c, d).

3.3. Sediment nutrient retention

On average, gross sediment nutrient retention rates captured by the sedimentation bottles were $8.42 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $89.49 \text{ g-N m}^{-2} \text{ yr}^{-1}$ for the first year of study and $7.10 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $71.81 \text{ g-N m}^{-2} \text{ yr}^{-1}$ for the second study year. There is no statistical difference in gross phosphorus or nitrogen retention rates by season (Wilcoxon, $p = 0.89$ and $p = 0.55$, respectively) or wetland cell (Wilcoxon, $p = 0.25$ and $p = 0.28$, respectively). Gross nutrient retention was statistically similar throughout the entire sample period with average rates of $7.76 \pm 1.72 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $80.65 \pm 16.45 \text{ g-N m}^{-2} \text{ yr}^{-1}$.

Table 1

Sedimentation rate estimates from November 2017–November 2018 using the bottle trap method (gross sedimentation) and horizon marker method (net sedimentation). Percent error indicates the percent of sediments that are resuspended after initially falling out of the water column. Sedimentation rates are annual averages \pm standard error.

Wetland cell	Sedimentation rate using bottle trap method (cm yr ⁻¹)	Sedimentation rate using horizon marker method (cm yr ⁻¹)	Percent difference
Pond	7.74 \pm 1.64	4.64 \pm 2.48	40.1
W1 in	12.92 \pm 0.77	8.21	36.3
W1 out	12.70 \pm 3.29	2.56 \pm 0.22	79.8
W2 in	15.84 \pm 2.34	4.89 \pm 2.12	69.2
W2 out	11.64 \pm 1.13	4.78 \pm 0.51	58.9
W3 in	10.40 \pm 1.02	0.29 \pm 0.15	97.2
W3 out	12.08 \pm 0.26	1.57	86.9
Average	11.90 \pm 1.50	3.87 \pm 1.10	67.6

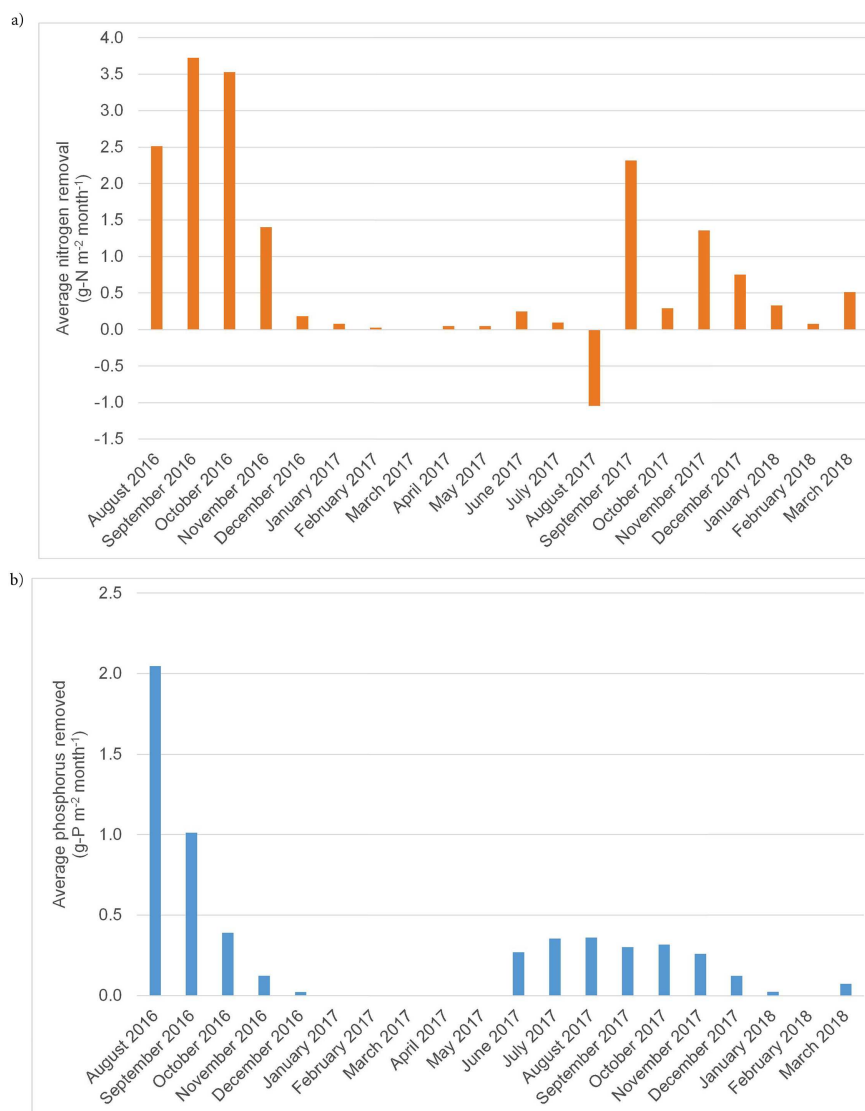


Fig. 4. Average a) nitrogen and b) phosphorus removal from the water column from August 2016 through March 2018.

Net annual nutrient sedimentation of phosphorus and nitrogen as measured by the horizon markers were $1.46 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $33.22 \text{ g-N m}^{-2} \text{ yr}^{-1}$ respectively and were statistically lower than gross nutrient retention rates (t-test, $p < 0.01$). Net nitrogen and phosphorus retention were an average of 67.6% lower than the estimated gross nutrient retention. There is no statistical change in net or gross nutrient retention via sedimentation with time in either phosphorus (ANOVA, $p = 0.99$ and $p = 0.89$, respectively) or nitrogen (ANOVA, $p = 0.88$ and $p = 0.30$, respectively). Net sediment nutrient retention was different based on the wetland cell (ANOVA, $p < 0.01$; Fig. 6). The pond and wetland 1 inflow have the greatest amount of net sediment nutrient accumulation whereas wetland 3 has the lowest.

4. Discussion

4.1. Sedimentation rates

Sedimentation rates in the Freedom Park wetlands are comparable to other wetlands in South Florida (Table 2; Reddy et al., 1993; Bhomia et al., 2015). The average sedimentation rate of stormwater treatment wetlands (STAs) designed to treat agricultural stormwater before it reaches the Florida Everglades is $1.0\text{--}1.7 \text{ cm year}^{-1}$ and the rate within the Everglades is $0.1\text{--}1.2 \text{ cm year}^{-1}$ (Reddy et al., 1993;

Bhomia et al., 2015). Freedom Park gross sedimentation is approximately $11.9 \text{ cm year}^{-1}$ and when resuspension is taken into account, sedimentation is approximately 3.9 cm year^{-1} , much greater than the STAs in the Everglades and falls above the average range of $0.3\text{--}1.9 \text{ cm year}^{-1}$ that Bhomia et al. (2015) observed by comparing various techniques used in approximately 38 studies of wetlands around the world. Freedom Park sedimentation rates of $1.0 \text{ kg m}^{-2} \text{ yr}^{-1}$ of dry sediments are much lower, however, than rates of $4.5\text{--}4.9 \text{ kg m}^{-2} \text{ yr}^{-1}$ observed at the Olentangy River Wetlands Research Park in clay-dominated waters in Ohio (Anderson and Mitsch, 2006). This low net sedimentation rate by dry weight estimates at Freedom Park is comparable to areas in the Everglades where there are low nutrient concentrations (Reddy et al., 1993; Bhomia et al., 2015).

Gross sedimentation rates are high which suggests that potential sedimentation and nutrient retention could be much higher if resuspension could be minimized. On average, 67.6% of the sediment that falls out of the water column is resuspended at the Freedom Park wetlands. Resuspension is the greatest in the third and final wetland; thus a majority of this sediment is likely being exported from the wetlands rather than being contained within the wetlands. This is a resuspension rate of approximately 8.03 cm yr^{-1} .

Also of note, sedimentation rates are greatest in the initial pond and first wetland. This is important since the purpose of the pond was to

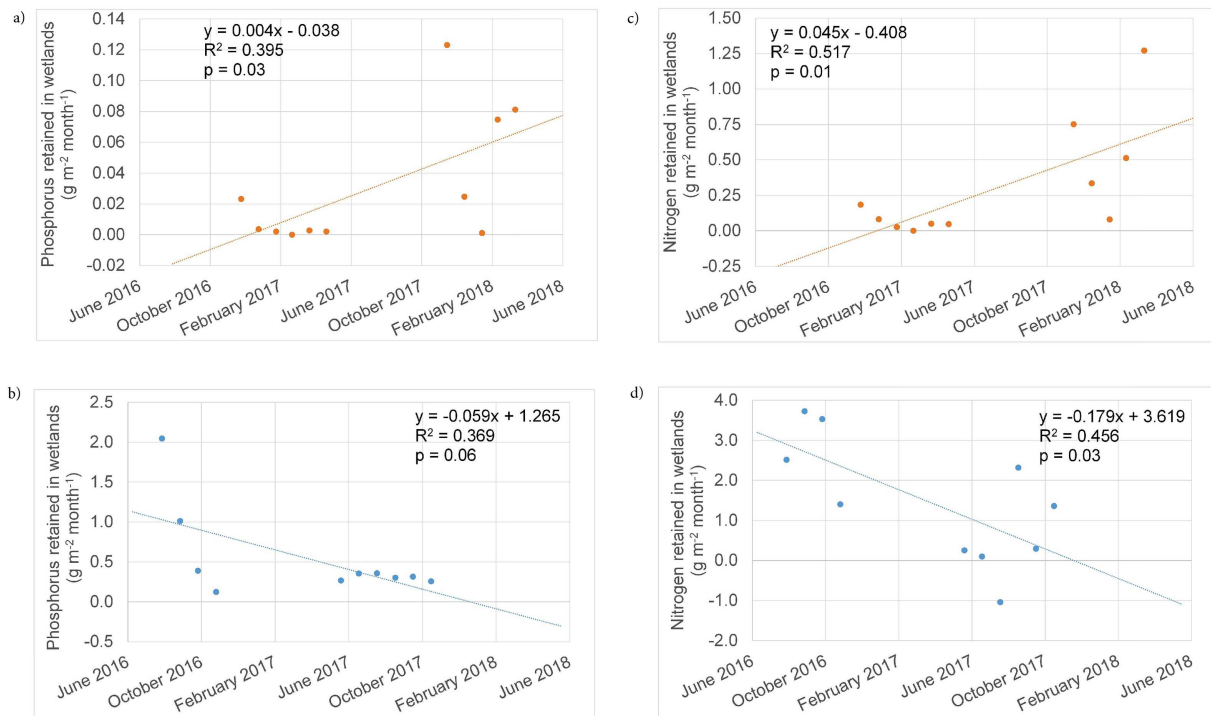


Fig. 5. Regressions of a) phosphorus retention during the dry season, b) phosphorus retention during the wet season, c) nitrogen retention during the dry season, and d) nitrogen retention during the wet season. Data are the mass of nutrients retained as calculated by the difference between inflow and outflow concentrations on each sampling date.

provide sediment reductions before the water reaches the wetlands (Bishop et al., 2014). Settling ponds are a common addition to urban stormwater treatment wetlands because it slows water as it enters the

wetlands (Mitsch and Gosselink, 2015). The results of this study reinforce this wetland design.

4.2. Nutrient retention in wetlands by sedimentation: a comparison with other studies

Total net nutrient removal from the water column within Freedom Park averages $4.23 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $11.91 \text{ g-N m}^{-2} \text{ yr}^{-1}$. Net nutrient retention via sedimentation within the wetlands averages $1.46 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $33.2 \text{ g-N m}^{-2} \text{ yr}^{-1}$ as determined by the horizon marker sedimentation rate and sediment nutrient concentrations. This means that sedimentation is responsible for approximately 34.5% of the total phosphorus retention within the wetlands. Since sedimentation is responsible for $33.2 \text{ g-N m}^{-2} \text{ yr}^{-1}$ being removed from the water column and nutrient retention within the wetlands as a whole is calculated as $11.9 \text{ g-N m}^{-2} \text{ yr}^{-1}$ based off of inflow and outflow loading rates, sedimentation retention of nitrogen is almost 3 times the nitrogen retention of the wetlands estimated from water quality differences between inflows and outflows. As such, there must be an additional source of nitrogen to make up for the $21.3 \text{ g-N m}^{-2} \text{ yr}^{-1}$ discrepancy between pumped inflow nitrogen loading and the mass of nitrogen being retained in the sediments. This suggests that there is a significant additional inflow of nitrogen into the wetlands that is unaccounted for. Further studies need to investigate other potential sources of nitrogen, such as other stormwater inflows or the daily use of reclaimed grey water to irrigate the park land that surrounds the wetland basins. These wetlands may be better at retaining nitrogen than the 26% retention estimated in our previous studies (Griffiths and Mitsch, 2017).

Resuspension accounts for $5.95 \text{ g-P m}^{-2} \text{ yr}^{-1}$ and $43.35 \text{ g-N m}^{-2} \text{ yr}^{-1}$ being reintroduced to the water column of the potential nutrients that could be retained as determined by the difference between the net horizon marker estimate of nutrient retention and the gross bottle trap estimate. This phosphorous resuspension is a high number compared to phosphorous resuspension estimated for 4 independent floodplain wetlands by a simulation model at the Des Plaines River Wetlands in NE Illinois of 1 to $5 \text{ g-P m}^{-2} \text{ yr}^{-1}$ (Wang and Mitsch, 2000). A

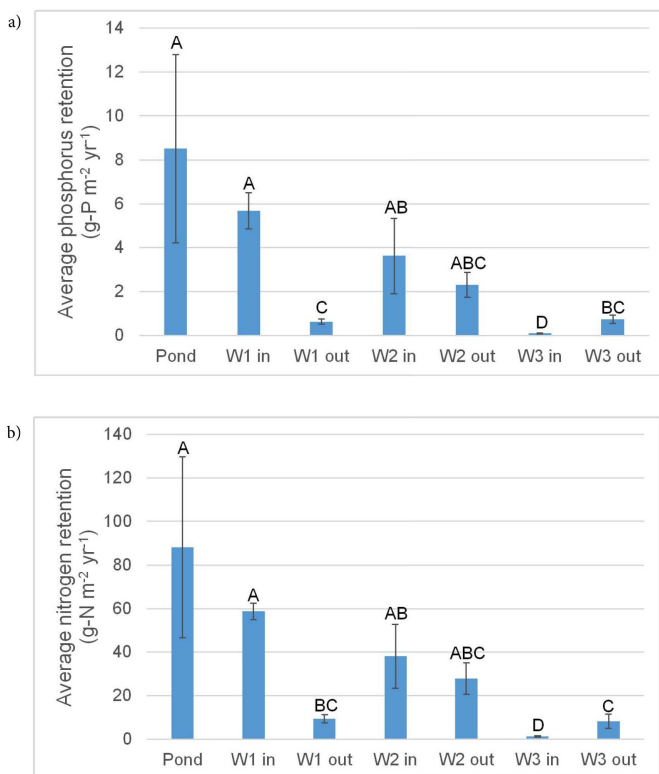


Fig. 6. Average net nutrient retention by sedimentation based on the wetland location for a) phosphorus and b) nitrogen. Wetland locations are defined by the wetland cell indicated in Fig. 2. Error bars represent standard error. Similarity letters represent the results of the ANOVA analysis.

Table 2

Sedimentation rates and nutrient retention via sedimentation as estimated from this study compared to other studies throughout the United States.

Location	Sediment accretion rate (cm yr ⁻¹)	Dry sediment accretion rate (kg-dry m ⁻² yr ⁻¹)	Nitrogen and phosphorus retention in sediments	Citation
Freedom Park, Florida	3.9	1.0	33.2 g-N m ⁻² yr ⁻¹ 1.46 g-P m ⁻² yr ⁻¹	This study
STAs, Florida	1.0–1.7		1.3–3.3 g-P m ⁻² yr ⁻¹	Bhomia et al. (2015)
Ever0067lades, Florida	0.1–1.2		5.4–24.3 g-N m ⁻² yr ⁻¹ 0.1–1.2 g-P m ⁻² yr ⁻¹	Reddy et al. (1993)
Olentangy River Wetlands Research Park, Ohio	4.9	4.5–4.9		Harter and Mitsch (2003); Anderson and Mitsch (2006)
Des Plaines River Wetlands, Illinois	0.5–1	1.2–4.2	1–5 g-P m ⁻² yr ⁻¹	Fennesy et al. (1994); Wang and Mitsch (2000)
Alluvial cypress swamp, Illinois		5.6	3.4 g-P m ⁻² yr ⁻¹	Mitsch et al. (1979)
Chesapeake Bay floodplain wetlands	0.2	0.98	5.5 g-N m ⁻² yr ⁻¹ 8 g-P m ⁻² yr ⁻¹	Noe and Hupp (2009); Noe et al. (2013)
Georgia floodplain	0.2	0.12 (past 30 yrs) 1.04 (past 100 yrs)	8.0 g-N m ⁻² yr ⁻¹ 0.75 g-P m ⁻² yr ⁻¹	Craft and Casey (2000)
Florida cypress swamp	0.6–2.9		3.2 g-P m ⁻² yr ⁻¹	Brown (1978)
North Carolina riparian buffer			4.3 g-P m ⁻² yr ⁻¹	Cooper et al. (1987)

sedimentation study in created riverine wetlands at the Olentangy River Wetland Park in central Ohio reported sediment retention rates of 40–45% (Harter and Mitsch, 2003; Mitsch, et al., 2014); these reports were low compared to the Des Plaines River Wetlands in northeast Illinois which had 76–99% sediment retention (Hey et al., 1994). This study found resuspension to be approximately 67.6% of the total sediments that fall out of the water column, thus sediment retention is approximately 32.4%, even lower than the low rates found by Harter and Mitsch (2003) in central Ohio. If methods are implemented to reduce resuspension such as increased vegetation or decreased hydraulic loading (Braskerud, 2002; Carleton et al., 2001), phosphorus and nitrogen retention could be significantly higher.

4.3. Riverine wetland sedimentation

The Freedom Park wetlands in this study were created on floodplains and lowlands in the Gordon River watershed. Floodplain wetlands accumulate a significant amount of sediment nutrients compared to the river load of nutrients (Noe and Hupp, 2009). Net sediment accumulation at Freedom Park is approximately 1029 g m⁻² yr⁻¹ which is slightly above the geometric mean of 976 g m⁻² yr⁻¹ estimated for Chesapeake Bay floodplain wetlands (Noe and Hupp, 2009). Freedom Park sediment accretion is 40.1 mm yr⁻¹, significantly higher than the Chesapeake Bay wetlands average of 1.8 mm yr⁻¹ (Noe and Hupp, 2009). This difference is likely due to the differences in created and natural wetlands since it has been found that channelization and interference with natural flows can effect overall sedimentation (Kroes and Hupp, 2010). Uncharacteristically high sedimentation rates of 45 mm yr⁻¹ were recorded in the Chesapeake Bay area as a result of beaver activity (Gellis et al., 2009). This suggests that unnatural hydrology at Freedom Park as cited by Nesbit and Mitsch (2018), may be causing irregular sedimentation rates.

The Freedom Park wetlands retain approximately 33.2 g-N m⁻² yr⁻¹ in the sediments through sedimentation. Compared to natural riverine and floodplain wetlands in more temperate climates, this rate is extremely high (Johnston et al., 1984; Craft and Casey, 2000; Noe and Hupp, 2009). The Chesapeake Bay floodplain wetlands averaged 5.5 g m⁻² yr⁻¹ (Noe and Hupp, 2009), a Wisconsin floodplain wetland averaged 12.8 g-N m⁻² yr⁻¹ retention via sedimentation (Johnston et al., 1984), and a Georgia floodplain wetland averaged 8.0 g-N m⁻² yr⁻¹ (Craft and Casey, 2000). Phosphorus retention within the sediments at Freedom Park averages 1.46 g-P m⁻² yr⁻¹. This retention is lower than other riverine wetlands such as Difficult Run in the Chesapeake Bay floodplains with 8 g-P m⁻² yr⁻¹ (Noe et al., 2013), a Florida cypress swamp with 3.2 g-P m⁻² yr⁻¹ (Brown, 1978), a North Carolina riparian buffer with 4.3 g-P m⁻² yr⁻¹ (Cooper et al., 1987), and an alluvial cypress swamp in Illinois with 3.4 g-P m⁻² yr⁻¹ (Mitsch et al.,

1979). Although nitrogen retention by sedimentation at Freedom Park is successful, phosphorus retention can be maximized by focusing on minimizing resuspension, creating a more natural hydrologic pattern, and increasing overall phosphorus sedimentation and sorption to the soils.

4.4. Effects of climate change delivery and retention of nutrients by sedimentation

As algal blooms continue to intensify and become more frequent in the face of climate change, it is more important than ever to ensure wetlands continue to act as nutrient sinks. Increased global atmospheric and water temperatures in combination with changes in seasonal patterns increase the time that algae can bloom, thus increasing the likelihood of blooms occurring (Wells et al., 2015). Additionally, evidence suggests that tropical cyclone activity may increase by 5–10% and precipitation rates might increase 20–30% (Goudie, 2006). With more storms occurring and added nutrients to the terrestrial landscape, larger loads of nutrients are entering aquatic ecosystems (Aumann et al., 2018; Howarth et al., 2002; Wells et al., 2015), making it even more important to make sure that wetlands such as those at Freedom Park are effective at retaining nutrients.

Alternatively, climate change may cause a decrease in precipitation (Havens and Steinman, 2013). If that is the case, the subsequent changes in water level will increase the potential of resuspension of sediments and their nutrient stocks and increased oxidation of exposed sediments may cause nutrients to be released from the soils (Havens and Steinman, 2013). It is also likely that a combination of these effects will be felt as wet seasons become wetter and dry seasons become drier (Mallakpour et al., 2018), decreasing nutrient retention by wetland sediments during all seasons of the year.

As wetlands face these potential issues with nutrient retention, it is important to enhance nutrient storage in sediments by maximizing sedimentation while minimizing resuspension. If nutrient retention can be increased in existing wetlands, this can help minimize harmful algal bloom effects downstream.

5. Conclusions

This study investigated the dynamics and role of sedimentation in overall nutrient retention within a created stormwater treatment wetland park in southwest Florida. We conclude the following:

- To better understand the efficiency of the wetlands, scientists and wetland managers need to have knowledge of all possible sources and sinks of nutrients and water into the wetlands.
- Resuspension in these urban treatment wetlands appears to be high.

Redesign of the morphology and hydraulic loading rates of the wetland basin should be investigated to reduce resuspension and increase nutrient retention.

- These wetlands, if properly managed, have the potential to provide significantly better nutrient retention and hence protection of downstream coastal aquatic ecosystems on Florida's southwest coast.
- Climate change effects on the efficiency of wetlands to remove nutrients from the water column needs to be closely monitored in order to minimize downstream effects such as harmful algal blooms.
- Additional wetlands should be placed within this urban watershed to minimize the possibilities of downstream harmful algal blooms, both freshwater blooms often limited by phosphorus and salt water blooms often limited by nitrogen.

CRedit author statement

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

- Anderson, C.J., Mitsch, W.J., 2006. Sediment, carbon, and nutrient accumulation at two 10-year-old created riverine marshes. *Wetlands* 26 (3), 779–792 doi: 10.1672/0277-5212(2006)26[779:SCANA]2.0.CO;2.
- Anderson, D.M., Glibert, P.M., Burkholder, J.M., 2002. Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. *Estuaries* 25 (4b), 704–726.
- Aumann, H.H., Behrangi, A., Wang, Y., 2018. Increased frequency of extreme tropical deep convection: AIRS observations and climate model predictions. *Geophys. Res. Lett.* 45 (24), 13,530–13,537. <https://doi.org/10.1029/2018GL079423>.
- Ballard, V., 2018, July 9. Gov. Scott declares state of emergency over algae bloom - South Florida Sun-sentinel. SunSentinel Retrieved from: <https://www.sun-sentinel.com/news/florida/fl-reg-scott-algae-emergency-20180709-story.html>.
- Baumann, R.H., Day, J.W., Miller, C.A., 1984. Mississippi deltaic wetland survival: sedimentation versus coastal submergence. *Science* 224 (4653), 1093–1095. <https://doi.org/10.1126/science.224.4653.1093>.
- Bhomia, R.K., Inglett, P.W., Reddy, K.R., 2015. Soil and phosphorus accretion rates in subtropical wetlands: Everglades stormwater treatment areas as a case example. *Sci. Total Environ.* 533, 297–306. <https://doi.org/10.1016/j.scitotenv.2015.06.115>.
- Bishop, M., Bays, J., Griffin, M., Gramer, W., 2014. More than a pretty space: stormwater treatment wetlands with multiple benefits at Freedom Park, Naples FL. *Proc. Water Environ. Fed.* 2012 (6), 8094–8115. <https://doi.org/10.2175/193864712811704620>.
- Bojorquez, M., 2018, July 9. Toxic algae a slimy mess for Florida's Lake Okeechobee - CBS News. CBS News Retrieved from: <https://www.cbsnews.com/news/florida-lake-okeechobee-toxic-algae-problem-2018-07-06/>.
- Brand, L.E., Compton, A., 2007. Long-term increase in *Karenia brevis* abundance along the Southwest Florida Coast. *Harmful Algae* 6 (2), 232–252. <https://doi.org/10.1016/j.hal.2006.08.005>.
- Braskerud, B.C., 2002. Factors affecting phosphorus retention in small constructed wetlands treating agricultural non-point source pollution. *Ecol. Eng.* 19 (1), 41–61. [https://doi.org/10.1016/S0925-8574\(02\)00014-9](https://doi.org/10.1016/S0925-8574(02)00014-9).
- Brown, S.L., 1978. A Comparison of Cypress Ecosystems in the Landscape of Florida. University of Florida, Gainesville, FL, USA.
- Carleton, J.N., Grizzard, T.J., Godrej, A.N., Post, H.E., 2001. Factors affecting the performance of stormwater treatment wetlands. *Water Res.* 35 (6), 1552–1562. [https://doi.org/10.1016/S0043-1354\(00\)00416-4](https://doi.org/10.1016/S0043-1354(00)00416-4).
- Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., ... Likens, G.E., 2009. Controlling Eutrophication: Nitrogen and Phosphorus. *Science* 323 (5917), 1014–1015. <https://doi.org/10.1126/science.1167755>.
- Cooper, J.R., Gilliam, J.W., Daniels, R.B., Robarge, W.P., 1987. Riparian areas as filters for agricultural sediment. *Soil Sci. Soc. Am. J.* 51 (2), 416–420. <https://doi.org/10.2136/sssaj1987.03615995005100020029x>.
- Craft, C.B., Casey, W.P., 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. *Wetlands* 20 (2), 323–332.
- Fennedy, M.S., Brueske, C., Mitsch, W.J., 1994. Sediment deposition patterns in restored freshwater wetlands using sediment traps. *Ecol. Eng.* 3 (4), 409–428. [https://doi.org/10.1016/0925-8574\(94\)00010-7](https://doi.org/10.1016/0925-8574(94)00010-7).
- Fink, D.F., Mitsch, W.J., 2004. Seasonal and storm event nutrient removal by a created wetland in an agricultural watershed. *Ecol. Eng.* 23 (4–5), 313–325. <https://doi.org/10.1016/j.ecoleng.2004.11.004>.
- Food and Agriculture Organization of the United Nations [FAO], 2015. World Fertilizer Trends and Outlook to 2018. Retrieved from FAO website. <http://www.fao.org/3/a-i4324e.pdf>.
- Gannon, D., Berens McCabe, E., Camilleri, S., Gannon, J., Brueggem, M., Barleycorn, A., Wells, R., 2009. Effects of *Karenia brevis* harmful algal blooms on nearshore fish communities in southwest Florida. *Mar. Ecol. Prog. Ser.* 378, 171–186. <https://doi.org/10.3354/meps07853>.
- Gellis, A.C., Hupp, C.R., Pavich, M.J., Landwehr, J.M., Banks, W.S.L., Hubbard, B.E., ... Reuter, J.M., 2009. Sources, Transport, and Storage of Sediment at Selected Sites in the Chesapeake Bay Watershed. Retrieved from: <https://pubs.usgs.gov/sir/2008/5186/>.
- Glibert, P.M., 2017. Eutrophication, harmful algae and biodiversity – challenging paradigms in a world of complex nutrient changes. *Mar. Pollut. Bull.* 124 (2), 591–606. <https://doi.org/10.1016/j.marpolbul.2017.04.027>.
- Glibert, P.M., Burford, M.A., 2017. Globally changing nutrient loads and harmful algal blooms: recent advances, new paradigms, and continuing challenges. *Oceanography* 30, 58–69. <https://doi.org/10.2307/24897842>.
- Goudie, A.S., 2006. Global warming and fluvial geomorphology. *Geomorphology* 79 (3–4), 384–394. <https://doi.org/10.1016/j.geomorph.2006.06.023>.
- Griffiths, L.N., Mitsch, W.J., 2017. Removal of nutrients from urban stormwater runoff by storm-pulsed and seasonally pulsed created wetlands in the subtropics. *Ecol. Eng.* 108, 414–424. <https://doi.org/10.1016/j.ecoleng.2017.06.053>.
- Harter, S.K., Mitsch, W.J., 2003. Patterns of short-term sedimentation in a freshwater created marsh. *J. Environ. Qual.* 32 (1), 325–334.
- Havens, K.E., Steinman, A.D., 2013. Ecological responses of a large shallow lake (Okeechobee, Florida) to climate change and potential future hydrologic regimes. *Environ. Manag.* 55 (4), 763–775. <https://doi.org/10.1007/s00267-013-0189-3>.
- Heisler, J., Glibert, P.M., Burkholder, J.M., Anderson, D.M., Cochlan, W., Dennison, W.C., ... Suddleson, M., 2008. Eutrophication and harmful algal blooms: A scientific consensus. *Harmful Algae* 8 (1), 3–13. <https://doi.org/10.1016/j.hal.2008.08.006>.
- Hey, D.L., Kenimer, A.L., Barrett, K.R., 1994. Water quality improvement by four experimental wetlands. *Ecol. Eng.* 3 (4), 381–397. [https://doi.org/10.1016/0925-8574\(94\)00008-5](https://doi.org/10.1016/0925-8574(94)00008-5).
- Hobbie, S.E., Finlay, J.C., Janke, B.D., Nidzgorski, D.A., Millet, D.B., Baker, L.A., 2017. Contrasting nitrogen and phosphorus budgets in urban watersheds and implications for managing urban water pollution. *Proc. Natl. Acad. Sci. U. S. A.* 114 (16), 4177–4182. <https://doi.org/10.1073/pnas.1618536114>.
- Howarth, R.W., Sharpley, A., Walker, D., 2002. Sources of nutrient pollution to coastal waters in the United States: implications for achieving coastal water quality goals. *Estuaries* 25 (4), 656–676. <https://doi.org/10.1007/BF02804898>.
- Hu, C., Muller-Karger, F.E., Swarzenski, P.W., 2006. Hurricanes, submarine groundwater discharge, and Florida's red tides. *Geophys. Res. Lett.* 33 (11), L11601. <https://doi.org/10.1029/2005GL025449>.
- Janke, B.D., Finlay, J.C., Hobbie, S.E., 2017. Trees and streets as drivers of urban stormwater nutrient pollution. *Environmental Science & Technology* 51 (17), 9569–9579. <https://doi.org/10.1021/acs.est.7b02225>.
- Johnston, C.A., Bubenzer, G.D., Lee, G.B., 1984. Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. *J. Environ. Qual.* 13 (2), 283–290. <https://doi.org/10.2134/jeq1984.00472425001300020022x>.
- Kirkpatrick, B., Fleming, L.E., Squicciarini, D., Backer, L.C., Clark, R., Abraham, W., ... Baden, D.G., 2004. Literature review of Florida red tide: implications for human health effects. *Harmful Algae* 3 (2), 99–115. <https://doi.org/10.1016/j.hal.2003.08.005>.
- Knaus, R.M., Cahoon, D.R., 1990. Improved cryogenic coring device for measuring soil accretion and bulk density. *J. Sediment. Petrol.* 60 (4), 622–623.
- Kovacic, D.A., David, M.B., Gentry, L.E., Starks, K.M., Cooke, R.A., 2000. Effectiveness of constructed wetlands in reducing nitrogen and phosphorus export from agricultural tile drainage. *Journal of Environment Quality* 29 (4), 1262. <https://doi.org/10.2134/jeq2000.00472425002900040033x>.
- Kroes, D.E., Hupp, C.R., 2010. The effect of channelization on floodplain sediment deposition and subsidence along the Potomac River, Maryland. *J. Am. Water Resour. Assoc.* 46 (4), 686–699. <https://doi.org/10.1111/j.1752-1688.2010.00440.x>.
- Land, M., Granéli, W., Grimvall, A., Hoffmann, C.C., Mitsch, W.J., Tonderski, K.S., Verhoeven, J.T.A., 2016. How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. *Environmental Evidence* 5 (1), 9. <https://doi.org/10.1186/s13750-016-0060-0>.
- Lee, C., Fletcher, T.D., Sun, G., 2009. Nitrogen removal in constructed wetland systems. *Engineering in Life Sciences* 9 (1), 11–22. <https://doi.org/10.1002/elsc.200800809>.
- Mallakpour, I., Sadeh, M., AghaKouchak, A., 2018. A new normal for streamflow in California in a warming climate: wetter wet seasons and drier dry seasons. *J. Hydrol.* 567 (September), 203–211. <https://doi.org/10.1016/j.jhydrol.2018.10.023>.
- Mitsch, W.J., Gosselink, J.G., 2015. *Wetlands*. 5th ed. John Wiley & Sons, Inc, Hoboken, NJ.
- Mitsch, W.J., Dorage, C.L., Wiemhoff, J.R., 1979. Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in Southern Illinois. *Ecology* 60 (6), 1116–1124. <https://doi.org/10.1126/science.151.3712.864-c>.
- Mitsch, W.J., Nedrich, S.M., Harter, S.K., Anderson, C., Nahlik, A.M., Bernal, B., 2014a. Sedimentation in created freshwater riverine wetlands: 15 years of succession and contrast of methods. *Ecol. Eng.* 72, 25–34. <https://doi.org/10.1016/j.ecoleng.2014.09.116>.

- Mitsch, W.J., Zhang, L., Waletzko, E., Bernal, B., 2014b. Validation of the ecosystem services of created wetlands: two decades of plant succession, nutrient retention, and carbon sequestration in experimental riverine marshes. *Ecol. Eng.* 72, 11–24. <https://doi.org/10.1016/J.ECOLENG.2014.09.108>.
- Nahlik, A.M., Mitsch, W.J., 2008. The effect of river pulsing on sedimentation and nutrients in created riparian wetlands. *Journal of Environment Quality* 37 (4), 1634. <https://doi.org/10.2134/jeq2007.0116>.
- Nesbit, T.A., Mitsch, W.J., 2018. Hurricane and seasonal effects on hydrology and water quality of a subtropical urban stormwater wetland. *Ecol. Eng.* 120 (March), 134–145. <https://doi.org/10.1016/j.ecoleng.2018.05.041>.
- Noe, G.B., Hupp, C.R., 2009. Retention of riverine sediment and nutrient loads by coastal plain floodplains. *Ecosystems* 12 (5), 728–746. <https://doi.org/10.1007/s10021-009-9253-5>.
- Noe, G.B., Hupp, C.R., Schenk, E.R., Rybicki, N.R., 2013. *Science Summary – Sediment and Nutrient Trapping in the Flood Plain of Difficult Run, Virginia, and Implications for the Restoration of Chesapeake Bay*.
- Paerl, H.W., 1997. Coastal eutrophication and harmful algal blooms: importance of atmospheric deposition and groundwater as “new” nitrogen and other nutrient sources. *Limnol. Oceanogr.* 42 (5part2), 1154–1165. https://doi.org/10.4319/lo.1997.42.5_part_2.1154.
- Paerl, H.W., Gardner, W.S., Havens, K.E., Joyner, A.R., McCarthy, M.J., Newell, S.E., ... Scott, J.T., 2016, April 1. Mitigating cyanobacterial harmful algal blooms in aquatic ecosystems impacted by climate change and anthropogenic nutrients. *Harmful Algae* 54, 213–222. <https://doi.org/10.1016/j.hal.2015.09.009>.
- Paerl, H.W., Otten, T.G., Kudela, R., 2018. Mitigating the expansion of harmful algal blooms across the freshwater-to-marine continuum. *Environ. Sci. Technol.* 52 (10), 5519–5529. <https://doi.org/10.1021/acs.est.7b05950>.
- Reddy, K.R., Delaune, R.D., Debusk, W.F., Koch, M.S., 1993. Long-term nutrient accumulation rates in the Everglades. *Soil Sci. Soc. Am. J.* 57 (4), 1147–1155.
- Saunders, D.L., Kalf, J., 2001. Nitrogen retention in wetlands, lakes and rivers. *Hydrobiologia* 443 (1/3), 205–212. <https://doi.org/10.1023/A:1017506914063>.
- Sweedler, M., 2018, July 16. Florida fights giant algal bloom in Lake Okeechobee - WSJ. Wall Street J. Retrieved from. <https://www.wsj.com/articles/florida-fights-giant-algal-bloom-in-lake-okeechobee-1531746000>.
- Taylor, G.D., Fletcher, T.D., Wong, T.H.F., Breen, P.F., Duncan, H.P., 2005. Nitrogen composition in urban runoff—implications for stormwater management. *Water Res.* 39 (10), 1982–1989. <https://doi.org/10.1016/J.WATRES.2005.03.022>.
- Vymazal, J., 2007. Removal of nutrients in various types of constructed wetlands. *Sci. Total Environ.* 380 (1–3), 48–65. <https://doi.org/10.1016/j.scitotenv.2006.09.014>.
- Wang, N., Mitsch, W.J., 2000. A detailed ecosystem model of phosphorus dynamics in created riparian wetlands. *Ecol. Model.* 126 (2–3), 101–130. [https://doi.org/10.1016/S0304-3800\(00\)00260-X](https://doi.org/10.1016/S0304-3800(00)00260-X).
- Weisberg, R.H., Liu, Y., Lembke, C., Hu, C., Hubbard, K., Garrett, M., 2019. The coastal ocean circulation influence on the 2018 West Florida Shelf K. brevis Red Tide Bloom. *Journal of Geophysical Research: Oceans* 124 (4), 2501–2512. <https://doi.org/10.1029/2018JC014887>.
- Wells, M.L., Trainer, V.L., Smayda, T.J., Karlson, B.S.O., Trick, C.G., Kudela, R.M., ... Cochlan, W.P., 2015. Harmful algal blooms and climate change: Learning from the past and present to forecast the future. *Harmful Algae* 49, 68–93. <https://doi.org/10.1016/j.hal.2015.07.009>.