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Evaluating the hydrologic and water quality performance of novel infiltrating wet retention ponds

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Abstract

Wet retention ponds temporarily store and slowly release stormwater to mitigate peak flow rates and remove particulate-bound pollutants. However, with sandy underlying soils, wet retention ponds may provide additional benefits through infiltration, thereby recharging groundwater and supporting baseflow in streams. Current design guidance often requires lining wet ponds to prevent infiltration; however, modern stormwater management strategies recommend maximizing runoff volume reduction through infiltration. Two infiltrating wet retention ponds in Fayetteville, NC, USA, were monitored for one year to assess volume reduction, peak flow mitigation, and water quality. In some months, 100% of stormwater runoff infiltrated and evaporated, with cumulative annual volume reductions of 60 and 51% for the two ponds. For events up to 76 mm (equivalent to the local 1-yr, 24-hr storm), measured peak flow reductions were similar to those of typical (non-infiltrating) wet ponds (median 99% reduction). Dissolved nitrogen species, total and dissolved phosphorus, and total suspended solids (TSS) concentrations were significantly reduced in both ponds; mean percent reductions were greater than 30% for each of these pollutants. Effluent concentrations were on par with typical (non-infiltrating) wet ponds previously monitored in North Carolina. Due to the aforementioned runoff reduction, nutrient and TSS loads were reduced by (at minimum) 35 and 67%, respectively. Infiltrating wet ponds were able to meet both peak flow and volume mitigation goals, suggesting that they could be a common tool in regions with sandy soils.

Key words: infiltration, low impact development, stormwater, stormwater control measure, wet pond, wet retention basin

Highlights

- A novel, hybrid, stormwater treatment practice, called infiltrating wet ponds was tested.
- Two ponds signficantly and substantially reduced runoff volumes and peak flows over year-long monitoring periods.
- Nutrient and TSS concentrations were generally reduced as well, with pollutant load reductions observed for all pollutants.
- Design guidance is provided for those who wish to employ infiltrating wet ponds locally.

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INTRODUCTION

Urbanization augments stormwater runoff volumes and peak flow rates, concomitantly increasing transport of anthropogenic pollutants to receiving water bodies (Leopold 1968; Bannerman *et al.* 1993; Jennings & Jarnagin 2002; Line & White 2007). Stormwater runoff often impairs lakes, rivers, and coastal waters, leading to fish kills, algal blooms, beach and shellfish water closures, lost tourism revenue, and drinking water source closures (Dyson & Huppert 2010; Qin *et al.* 2010). Stormwater Control Measures (SCMs) are employed to mitigate some of these deleterious impacts.

One such SCM is the wet retention pond (Figure 1), which has a permanent pool of water to promote sedimentation and provides additional storage capacity above the permanent pool to detain and slowly release (i.e., draw down) stormwater runoff, thereby mitigating peak flow rates (Hossain *et al.* 2005; Smolek *et al.* 2015). A drawdown orifice in the outlet structure (Figure 1) controls the release rate of the water quality volume (2–5 days in North Carolina) (NCDEQ 2017). The outlet structure also includes a bypass for events larger than the design storm return period. Wet ponds mitigate peak flows, but typically provide little volume reduction, one of the key tenets of Low Impact Development (LID) strategies (Dietz 2007; Dietz & Clausen 2008; Wilson *et al.* 2014).

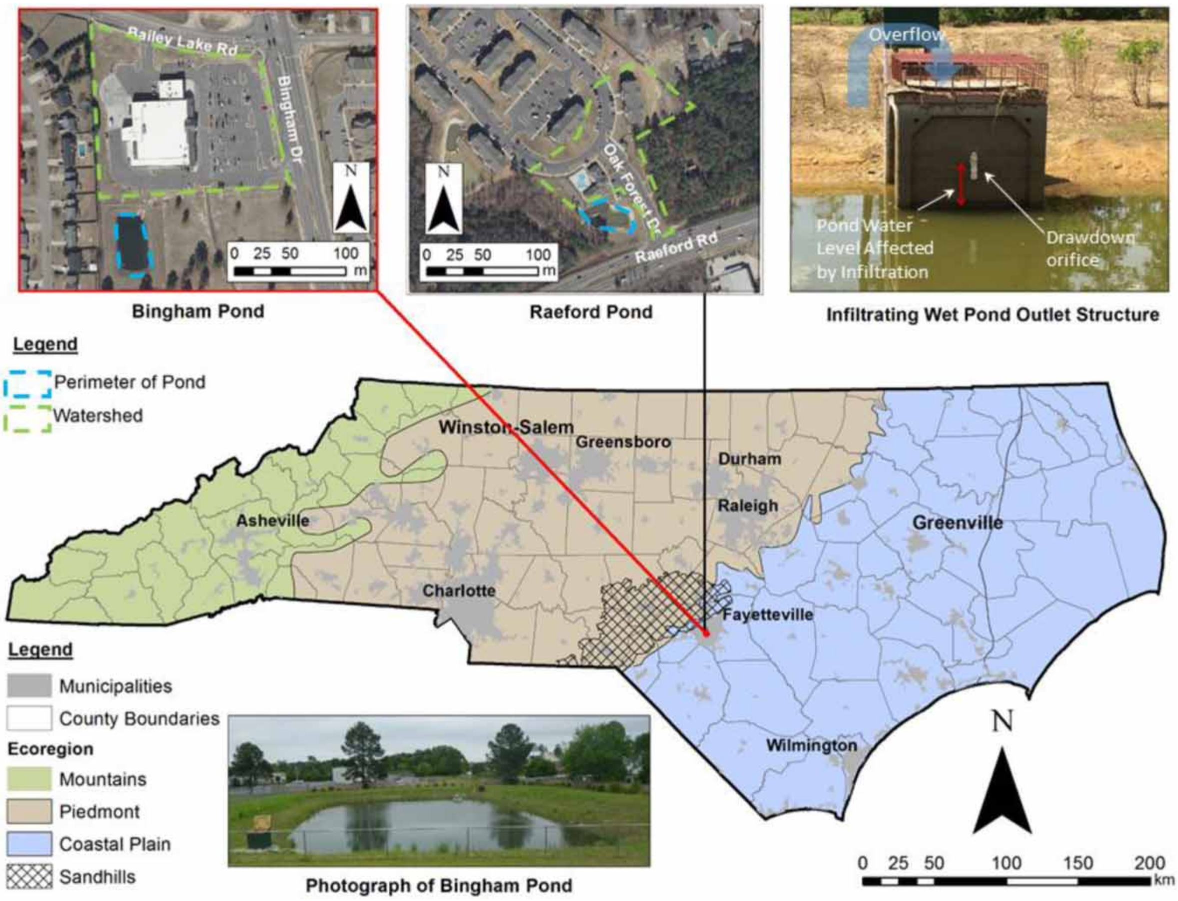

Figure 1 | Location and plan view of the Raeford and Bingham ponds, an example of an infiltrating wet pond outlet structure after a dry period, and a photograph of the Bingham pond.

Originally designed to control flooding by extending flow duration (Hancock *et al.* 2010), wet ponds were later adapted to improve stormwater quality through the implementation of drawdown orifices

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(Hvitved-Jacobsen *et al.* 1990; Mallin *et al.* 2002). Energy reduction of stormwater runoff allows total suspended solids (TSS), and sediment-borne pollutants to settle out (Pettersson 1998; Comings *et al.* 2000; Erickson *et al.* 2012; Egemose *et al.* 2015; Winston *et al.* 2017b). Sedimentation effectiveness is dependent on influent particle size distribution (Ferrara & Witkowski 1983; Greb & Bannerman 1997; Winston *et al.* 2017a); sand is trapped, while clay and fine silts often remain suspended in the water column. Adsorption, biodegradation, denitrification, and plant uptake are secondary treatment processes (Nix *et al.* 1988; McPhillips & Walter 2015), though dissolved nutrients often pass through ponds without substantial treatment (Istenič *et al.* 2012; Winston *et al.* 2013; Sønderup *et al.* 2016; Gold *et al.* 2017).

Design factors affecting wet pond performance include length-to-width ratio, which Mallin *et al.* (2002) found was directly related to pollutant removal. Surface area to drainage area percentage can predict wet pond performance (Wu *et al.* 1996), with 1–2% recommended to meet water quality goals (e.g., NCDEQ 2017). The presence of a water quality drawdown orifice also consistently improves water quality performance (Comings *et al.* 2000). Sønderup *et al.* (2015) found that young ponds remove more pollutants than old ponds, suggesting that maintenance (i.e., dredging) may be needed to ensure long-term functionality (Borne *et al.* 2015; Blecken *et al.* 2017).

Given the relative abundance of existing wet ponds in the USA (e.g., 76,000 of them in Florida (Sinclair *et al.* 2019)), designers and researchers seek to optimize their hydrologic and/or water quality performance through simple and inexpensive retrofits. Adding vegetation to SCMs generally improves water quality performance (Mallin *et al.* 2002; Greenway 2004). Winston *et al.* (2013), Borne (2014), Lynch *et al.* (2015), and Maxwell *et al.* (2020) showed that floating treatment wetlands (FTW) retrofitted in existing wet ponds can limit discharge concentrations for many pollutants. Middleton & Barrett (2008) found that simple valve and electronic actuator control of the drawdown orifice can improve pond performance such that it is similar to sand filters; Carpenter *et al.* (2014) had similar success with the addition of a sluice gate to the pond outlet to lengthen hydraulic retention time. Others have suggested the use of littoral sand filters (Erickson *et al.* 2012; Istenič *et al.* 2012), the addition of filters adjacent to the pond (Sønderup *et al.* 2015), or upflow filtration on the drawdown orifice (Winston *et al.* 2017b) as methods to improve pollutant capture.

Because LID strategies focus on reducing runoff volume, one simple wet pond modification is to promote infiltration into the underlying soil. Benefits of such a condition were noted (but not thoroughly investigated) by Line *et al.* (2012). Currently, many state agencies in the USA encourage or require clay liners or soil compaction in regions with sandy soils (WIDNR 2007; MPCA 2016; NCDEQ 2017), effectively reducing potential runoff reduction through infiltration and subsequent pollutant load reductions. However, compacted clay liners are difficult to construct correctly and sometimes fail (Silva & Almanza 2009).

Infiltration is an accepted and promoted mechanism of runoff reduction in other SCMs such as bioretention, permeable pavement, and infiltration basins (Wardynski *et al.* 2012; Hunt *et al.* 2012; Natarajan & Davis 2016). However, wet ponds often 'fail' in the Sandhills and coastal regions of North Carolina (Figure 1), where sandy soils predominate, because they do not maintain a permanent pool of water due to infiltration. Since these SCMs retain water for more than 48 hours after rainfall, they likewise are not infiltration basins (NCDEQ 2017). These 'infiltrating wet ponds' lack performance assessment and design guidance, but may function similarly to failed infiltration basins, which hold water and develop wet pond/wetland-like conditions but still encourage infiltration (Natarajan & Davis 2015, 2016).

An infiltrating wet pond is a potentially beneficial SCM in regions with sandy soils, yet currently is in an SCM's 'no man's land' with respect to permitting. We assess their hydrologic and water quality performance herein. Preliminary design guidance for infiltrating wet ponds is presented.

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SITE DESCRIPTIONS

Two wet retention ponds located 3 km apart in sandy underlying soils were monitored in Fayetteville, North Carolina, USA (Figure 1). Normal mean temperatures range from 6 °C (January) to 27 °C (July); the average annual rainfall is 1,186 mm (SCONC 2014) and is relatively well-distributed seasonally. Fayetteville is in Köppen-Geiger climate zone Cfa.

The Bingham pond drained a 2.37-ha, 77% impervious, commercial land use watershed (Table 1). The pond surface area at permanent pool elevation was 0.12 ha, with a forebay comprising 20% of this surface area. The outlet structure was a 3-m diameter circular riser with a 50-mm diameter drawdown orifice set 450 mm below the overflow elevation. The mapped underlying soils were Autryville loamy sand (Soil Survey Staff 2015), and two pre-construction infiltration tests showed infiltration rates of 356-mm/h and 399-mm/h in these soils. A 250-mm clay liner (1,089 kg of bentonite) was installed, but the clay liner was faulty and infiltration occurred. The pond was fringed by a vegetated shelf planted with native grasses.

Table 1 | Summary of Bingham and Raeford site characteristics

	Bingham pond	Raeford pond
Location (Latitude and longitude)	35°0′58″N, 78°59′11″W	35°2′30″N, 78°59′43″W
Date constructed	March 2011	June 2012
Permanent pool surface area (ha)	0.12	0.08
Forebay area as a percentage of permanent pool area	20	20
Mapped underlying soil/HSG*	Autryville loamy sand/A	Candor sand/A
Drainage area (ha)	2.37	1.94
Drainage area imperviousness (%)	77	44
Depth to SHWT (m)	>1	<1
Pre-construction infiltration rate (mm/hr)	356/399	Not measured
Outlet structure	3 m diameter round riser	1.5 × 1.5 m square riser
Drawdown orifice diameter (cm)	5	3.8
Temporary pool depth (m)	0.45	0.9

HSG, Hydrologic Soil Group.
SHWT, Seasonal High Water Table.
*Soil survey staff (2015).

The Raeford pond drained a 1.94-ha, 44% impervious, multi-family residential land use watershed (Table 1). The pond surface area at permanent pool elevation was 0.08 ha with the forebay representing 20% of the pond's surface area. The outlet structure was a 1.5-m square riser with a 38-mm diameter drawdown orifice located 0.9 m below the overflow. Underlying soils were Candor sand which have estimated infiltration rates of 150–500 mm/h (Soil Survey Staff 2015); pre-construction infiltration tests were not performed. The Raeford pond also had a clay liner added, but the liner failed to prevent infiltration. Thus, this pond is considered an infiltrating wet pond.

METHODS

Hydrologic and water quality monitoring

Monitoring equipment was installed at each pond in May 2013. At the Bingham pond, inflow entered via a 0.9-m reinforced concrete pipe (RCP). An ISCO 750 area velocity meter (AVM; Teledyne Isco,

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Lincoln, NE) measured velocity and depth of flow. At the Bingham outlet, an ISCO 6712 automated sampler was installed with an ISCO 730 bubbler module to measure depth of flow over a compound weir with a 152-mm tall, 45° v-notch lower portion and a 610-mm wide rectangular upper portion fitted inside the outlet structure.

Inflow entered the Raeford pond via a 0.60-m RCP (hereafter inlet 1) and a 0.38-m RCP (hereafter inlet 2). ISCO 6712 automated samplers, ISCO 730 bubbler modules, and compound weirs were installed at inlet 1 (102-mm tall 60° v-notch lower portion and 1.2-m wide rectangular upper portion) and the outlet (152-mm tall 30° v-notch lower portion and 610-mm wide rectangular upper portion). Velocity and flow depth at inlet 2 were measured using an ISCO 750 AVM.

Two methods were used to calculate flow rate and subsequently trigger sample aliquots. For AVMs, the known dimensions of the pipe and measured flow depth and velocity were used by the ISCO 6712 samplers to determine flow rate and cumulative runoff volume. For bubbler modules, measured depth over the weir was correlated to flow rate using developed stage-discharge relationships from tabulated weir equations (Walkowiak 2011). Flow rates were integrated with time to determine cumulative flow volume, which was used to trigger runoff volume-proportional composite samples at each monitoring site. All sample intake strainers were located in areas of well-mixed flow.

HOBO U20 water level loggers (Onset Computer Corporation, Bourne, MA) were placed in 80-mm PVC stilling wells attached to both outlet structures to record pond stage. Measured pressure in each stilling well was corrected with barometric pressure measured onsite. Topographic surveys of both ponds were obtained using a total station, and data were imported into AutoCAD to develop pond stage-storage relationships (Baird 2015).

Rainfall data were collected at each site using a manual rain gauge and a 0.254-mm increment tipping bucket rain gauge (Davis Instruments, Hayward, CA). The rain gauges were mounted on wooden posts in areas free of overhead obstructions. All hydrologic and rainfall data were collected on 2-minute intervals.

Water surface evaporation was estimated on 0.254-mm intervals at both sites using an atmometer (ETgage Model E, Loveland, CO) with a #30 turfgrass reference evapotranspiration (ET) cover. The atmometer was placed as close as possible to the water surface while avoiding inundation during high flows. The turfgrass reference ET was multiplied by a coefficient of 1.05 to calculate surface water evaporation (Allen *et al.* 1998).

Sampling methods and data analysis

Storm events were characterized by a minimum antecedent dry period of 6 hours and total rainfall depth of at least 2.5 mm. The number of aliquots varied per storm event and were usually different between inflow and outflow. The minimum number of aliquots needed for sample analysis was 5. Samples were only obtained and processed in the lab when they represented at least 80% of the measured runoff volume (Geosyntec Consultants and Wright Water Engineers 2009). Water levels in each pond were determined relative to the normal pool elevation. Calculation of water levels was based on stage-storage equations presented in Appendix B of Baird (2015).

Water quality samples were collected within 24 hours of the cessation of flow. Composite samples were agitated to re-suspend all particulates. A 1-L plastic bottle (for TSS) and a pre-acidified 125 mL plastic bottle (for nutrients) were filled from the composited sample. Approximately 20 mL of the composited sample was filtered through a 0.45-μm Whatman puradisc filter into a glass bottle for orthophosphate analysis. Samples were placed on ice and transported to the laboratory for analysis using US EPA (1983) and APHA *et al.* (2012) methods (Table 2). Laboratory-reported data represented the event mean concentration (EMC). Organic nitrogen, total nitrogen, and particulate-bound phosphorus concentrations were calculated (Table 2). At the Raeford pond, the reported influent concentration was the flow-weighted average of both measured inlet EMCs.

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Table 2 | Laboratory analytical methods and reporting limits

Pollutant	Pollutant name	Analytical method	Reporting limit (mg/L)
$NO_{2,3}$-N	Nitrate – Nitrite Nitrogen	SM 4500 NO3 F[a]	0.0056
TKN	Total Kjeldahl Nitrogen	EPA Method 351.2[b]	0.28
TAN	Total Ammonia Nitrogen	SM 4500 NH3 G[a]	0.007
ON	Organic Nitrogen	= TKN – TAN	NA
TN	Total Nitrogen	= TKN + $NO_{2,3}$-N	NA
OP	Orthophosphate	SM 4500 P F[a]	0.006
TP	Total Phosphorus	SM 4500 P F[a]	0.01
PBP	Particle Bound Phosphorus	= TP – OP	NA
TSS	Total Suspended Solids	SM 2540 D[a]	1

[a]APHA *et al.* (2012).
[b]US EPA (1983).

Because the ponds did not reliably return to normal pool elevation before the start of the next event, storm-by-storm analysis was limited. Volume reduction and peak flow mitigation was determined across monthly and annual time frames. The following parameters comprised the water balance:

$$V_f = V_i + \sum_{i=1}^{n} (V_{in} + P - V_{out} - E - F)_i \tag{1}$$

where n is the number of storm events, V_f is the final volume (m^3), V_i is the initial volume (m^3), V_{in} is the influent volume (m^3), P is the precipitation volume (m^3), V_{out} is the effluent volume (m^3), E is the evaporated volume (m^3), and F is the infiltrated volume (m^3) for the i^{th} storm event. All water budget components were directly measured except infiltration, which was calculated using Equation (1). Linear regression was utilized to model monthly runoff reduction as a function of monthly rainfall depth.

While events with paired inlet and outlet pollutant concentrations were typically tested, four storm events at each pond were sampled at the inlet only, since no outflow occurred during these events. These four inlet samples were used to calculate mean and median influent concentrations but were not used for the paired statistical tests. Inlet and outlet mean ($\bar{x}$) and median ($\tilde{x}$) concentrations as well as mean concentration reductions were presented for each pond as performance evaluation metrics. Further, boxplots of inlet and outlet concentrations were developed to visually represent the variability in the data sets. Finally, the performance of these infiltrating wet ponds was compared to that of standard wet retention ponds in the literature.

Pollutant load (g or kg) was calculated for each pollutant, monitoring location, and rainfall event as the product of concentration and runoff volume. A summation of loads was then determined as the total of all event-based loads. An annual pollutant load (L, kg) was calculated using Equation (2):

$$L = \frac{\sum_{i=1}^{n} (EMC_{loc} \times V_{loc})_i}{V_{loc,samp}} \times \frac{V_{loc,\,tot}}{1000} \tag{2}$$

where V_{loc} is the measured runoff volume at a particular monitoring location for the i^{th} event, EMC_{loc} is the event mean concentration (in mg/L) at a particular monitoring location, $V_{loc,samp}$ and $V_{loc,tot}$ are the total runoff volume sampled (in m^3) and the total observed runoff volume (in m^3) over the monitoring period, respectively.

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Statistical analysis

Paired inflow and outflow data were compared for peak discharge, water quality concentrations, and sampled storm event pollutant loading. Paired data sets were first tested for normality through visual inspection of quantile-quantile plots and three goodness-of-fit tests: Shapiro-Wilk, Anderson-Darling, and Lilliefors. Normal or log-normal data were tested for significance using the student's t-test. Non-parametric statistics (Wilcoxon signed rank test) were utilized for the remainder of the data sets. The effects of rainfall, average monthly water level, and antecedent dry period on monthly volume reduction were analyzed using simple and multiple linear regression. The effects of 5-min peak intensity, total rainfall, and antecedent dry period on each water quality parameter was also analyzed using simple and multiple linear regression. A criterion of 95% confidence was used for this research ($\alpha = 0.05$). R version 3.4.1 (R Core Team 2017) was used for data and statistical analysis.

RESULTS AND DISCUSSION

Rainfall

The Bingham pond was monitored for hydrology and water quality from May 15, 2013, through May 31, 2014. During this period, 77 hydrologic events were recorded ranging from 3 to 102 mm. The Raeford pond was monitored for hydrology and water quality from July 29, 2013, through August 9, 2014. During this period, 66 hydrologic events were recorded ranging from 3 to 102 mm. Many events exceeded the water quality event (25 mm) for which these ponds were designed to capture and slowly release. Mean and median event depths at both sites were 18 and 11 mm, respectively. One complete year of rainfall at the Bingham site and Raeford site totaled 1,330 mm and 1,067 mm, respectively. The 30-year normal precipitation for the Fayetteville area is 1186 mm (State Climate Office of North Carolina 2014). The months of June and July at each pond were not from the same year. The June and July precipitation totals for the Bingham site were from 2013, whereas those at the Raeford site were from 2014. This explains a large portion of the variation in rainfall totals between the two sites despite their close geographic proximity. A detailed description of the rainfall events examined at both sites is found in Baird (2015).

Hydrology

The water budgets on a monthly basis at the Bingham and Raeford ponds, respectively, are shown in Figures 2 and 3. The water levels at the first and last days of each month were used for calculations. The rainfall onto the pond, initial volume (function of initial water level), and final water level (and therefore final volume) were used to calculate the volume of water that infiltrated. The sum of outflow, evaporation, and infiltration was equal to the total inflow for each month. At both ponds, evaporation from the surface of the water was minimal, the majority of the runoff either infiltrated or became outflow. The monthly outflow volume had the most variation and was influenced by rainfall depth. A significant relationship was found at Bingham (regression slope p-value $= 0.007$) and Raeford (regression slope p-value $= 0.027$) between the total monthly precipitation and the volume reduction. De Macedo *et al.* (2019) have previously observed that rainfall intensity had a stronger effect on SCM performance than rainfall depth.

The largest monthly volume reductions at both ponds were 100% (i.e., no water was discharged from the outlet structure). The median monthly volume reductions were 67.4% and 50.5% at the Bingham and Raeford ponds, respectively. The lowest monthly volume reductions at the Bingham and Raeford ponds were 10.3% and −14.9%, respectively.

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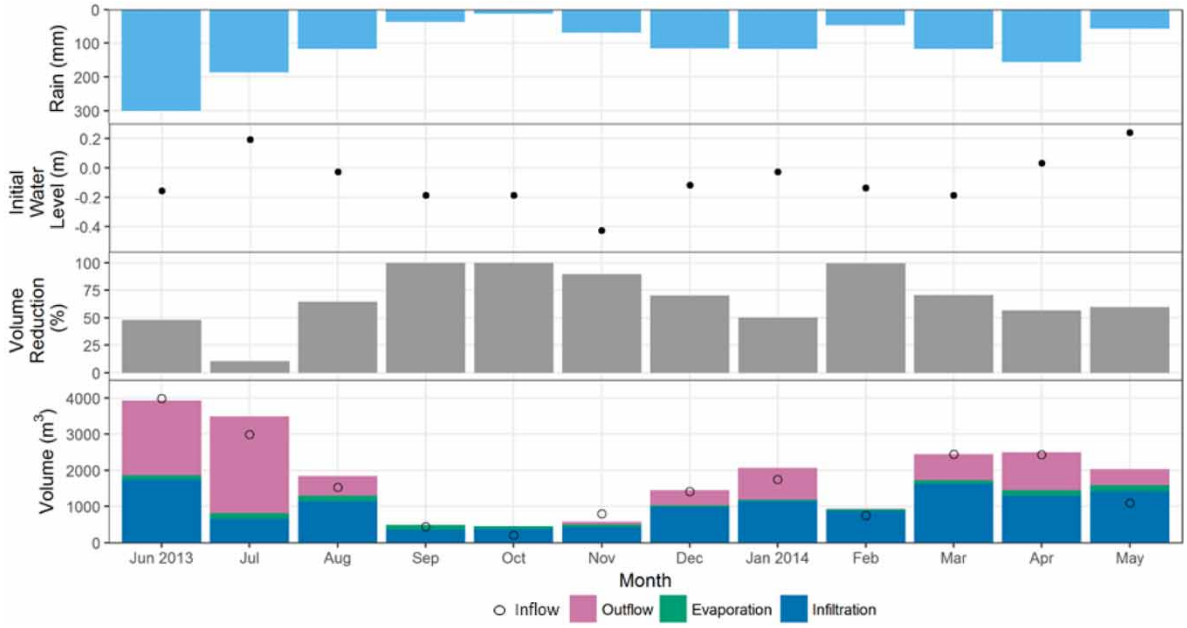

Figure 2 | Bingham pond rainfall and hydrologic data. Initial water level is the pond stage at the beginning of the month. The sum of outflow, evaporation, and infiltration is equivalent to the total inflow.

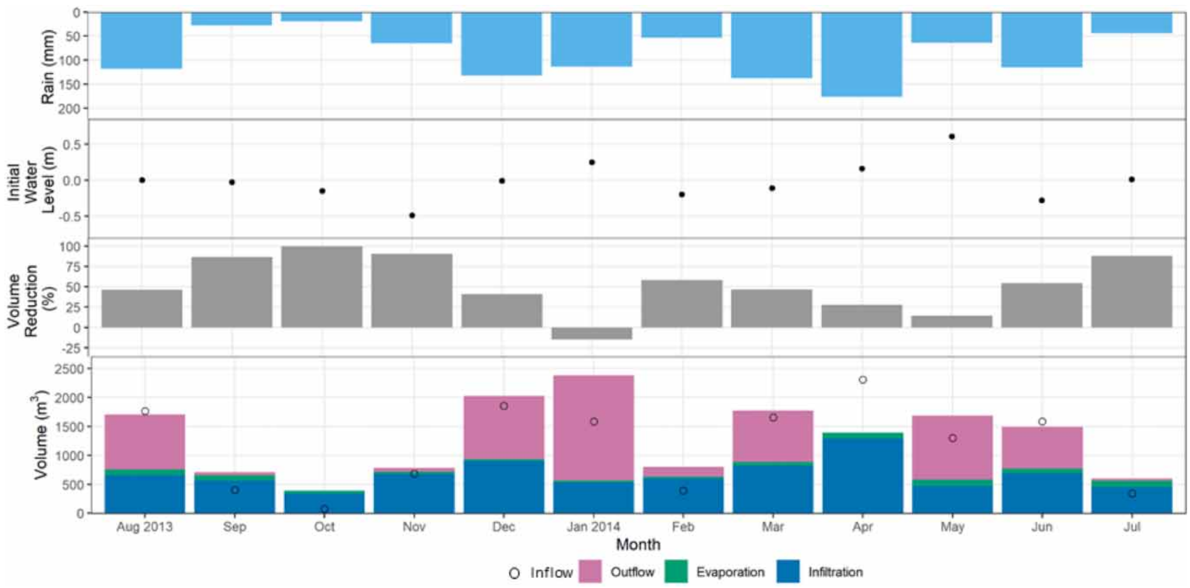

Figure 3 | Raeford pond rainfall and hydrologic data. Initial water level is the pond stage at the beginning of the month. The sum of outflow, evaporation, and infiltration is equivalent to the total inflow.

One anomaly is the Raeford pond's January 2014 runoff volume reduction, which was −14.9%. This result is an artifact of a large storm's inflow from late December remaining in the pond and being discharged in January 2014. The inflow was thus observed in December 2013, but much of the outflow occurred in January 2014. This occurs to a lesser extent in April and May (2014) as well. At the Bingham pond, the volume reductions for July and May are lower than average, while the volume reductions for June and April are higher than average for the same reason.

Runoff volume reduction was strongly dependent on rainfall depth (Figures 3 and 4); in months with little rainfall (e.g., September and October), the infiltrating wet ponds were able to completely or nearly eliminate outflow. These results are similar to infiltration-based LID practices, such as

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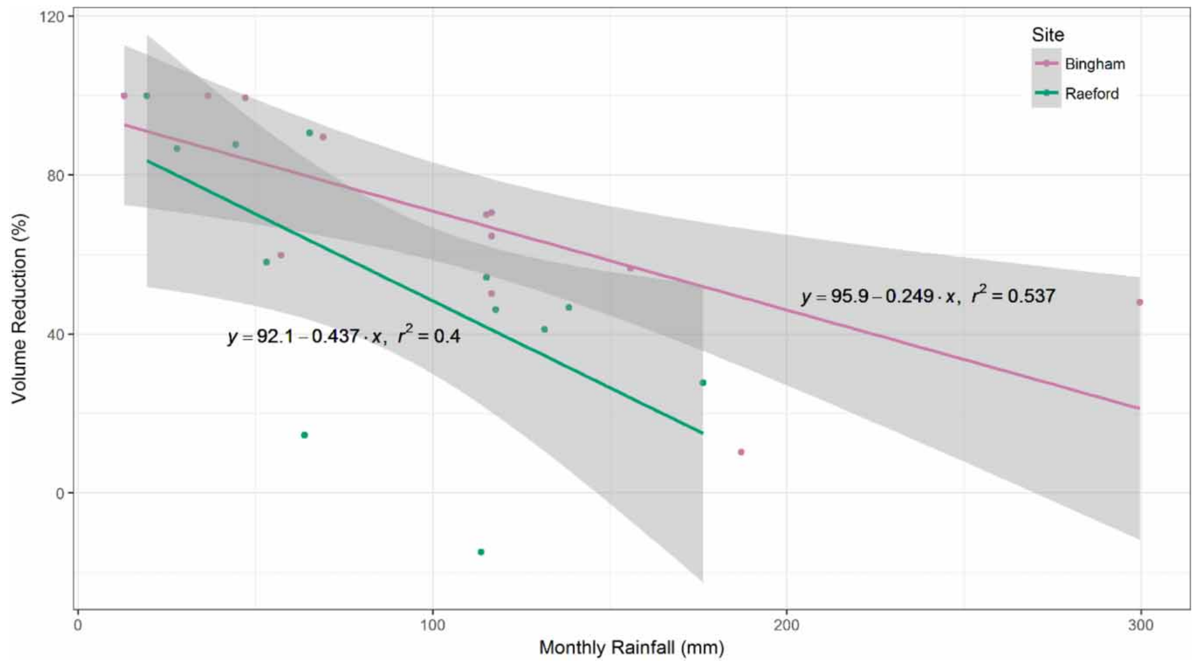

Figure 4 | Monthly volume reduction versus total monthly rainfall and fitted lines (Bingham $r^2 = 0.54$, Raeford $r^2 = 0.40$).

bioretention (Davis 2008; Brown & Hunt 2011) and permeable pavement (Wardynski *et al.* 2012; Drake *et al.* 2014) located over sandy soils. Inversely proportional relationships between monthly volume reduction and rainfall were significant at the Bingham (p-value = 0.007) and Raeford (p-value = 0.027) sites (Figure 4).

At the Bingham pond, 40% of the water left via outflow, 54% infiltrated, and 6% evaporated over the year-long monitoring period. At the Raeford pond, 49% of the water became outflow, 46% infiltrated, and 5% evaporated. The modest annual differences between the two wet ponds could be attributed to the different surface areas, loading ratios, monitoring windows and/or underlying soils. Because of infiltration, the volume reductions were an order of magnitude higher in these ponds than for ponds of similar design relying solely on evaporation for runoff reduction.

Despite a 25-mm clay liner and 1,089 kg (0.9 kg/m^2) of bentonite overlying the liner, significant volume reductions still occurred at the Bingham pond. If no liner had been installed, it is likely the observed volume reduction would have been appreciably higher; the SCM would probably have functioned as an infiltration basin due to the high infiltration rates of the underlying soil. The volume reductions monitored herein are attributed to an improperly installed, or damaged, liner.

Thirty-one events at Bingham and 16 at Raeford produced no outflow (100% volume reduction). Both wet ponds were able to mitigate the peak flows for large storm events. At the Bingham pond, storm events with depths of 42 mm, 42 mm, 49 mm, 57 mm, and 76 mm had peak flow reductions of 100%, 99.2%, 97.9%, 98.6 and 100%, respectively. Bingham was able to completely capture a 76 mm event. At the Raeford pond, storm events with depths of 34 mm, 40 mm, 46 mm, and 50 mm had peak flow rate reductions of 98.5%, 98.0%, 93.1% and 70.1%.

WATER QUALITY

Pollutant concentrations

Twenty-three storm events were sampled for water quality at the Bingham pond. A total of 19 paired nutrient samples and 17 paired TSS samples were collected. Rainfall depths for sampled storms

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ranged from 4 to 76 mm ($\bar{x} = 24$ mm, $\tilde{x} = 16$ mm). Five-minute peak rainfall intensity ranged from 3 to 135 mm/h ($\bar{x} = 36$ mm/h, $\tilde{x} = 21$ mm/h). At the Raeford pond, 20 storm events were sampled for water quality, with 16 paired inlet/outlet samples. Sampled event rainfall depth ranged from 5 to 102 mm ($\bar{x} = 28$ mm, $\tilde{x} = 21$ mm). Five-minute peak rainfall intensity varied from 7 to 133 mm/h ($\bar{x} = 37$ mm/h, $\tilde{x} = 31$ mm/h).

Statistical analysis showed significant differences in the influent and effluent concentrations at the Bingham pond for all pollutants except TKN, TN, and PBP (Table 3 and Figure 5). Similarly, at Raeford all but three constituent concentrations (TKN, ON, PBP) were significantly reduced by the pond (Table 3 and Figure 5).

Table 3 | Bingham and Raeford mean and median pollutant concentrations and mean EMC percent concentration reduction

Pollutant	$\tilde{x}$ inlet EMC (mg/L)	$\tilde{x}$ outlet EMC (mg/L)	Median conc. reduction (%)	$\bar{x}$ inlet EMC (mg/L)	$\bar{x}$ outlet EMC (mg/L)	Mean conc. reduction (%)
Bingham						
TKN	0.55	0.55	−1	0.74	0.63	15
$NO_{2,3}$	0.19	0.05	71	0.19	0.06	**66***
TN	0.74	0.57	24	0.93	0.69	26
TAN	0.16	0.04	74	0.2	0.05	**73***
ON	0.42	0.54	−27	0.54	0.57	**−6***
OP	0.008	0.003	65	0.013	0.004	**71***
PBP	0.049	0.051	−5	0.07	0.05	32^
TP	0.06	0.05	9	0.08	0.05	**38***
TSS	44	12	74	55	13	**77***
Raeford						
TKN	1	0.72	28	1.06	0.85	20^
$NO_{2,3}$	0.24	0.1	58	0.25	0.11	**57***
TN	1.26	0.79	37	1.31	0.96	**27***
TAN	0.23	0.08	65	0.25	0.09	**64***
ON	0.74	0.63	15	0.81	0.76	6
OP	0.064	0.027	58	0.072	0.033	**55***
PBP	0.12	0.1	15	0.13	0.11	11
TP	0.23	0.13	42	0.2	0.15	**28***
TSS	50	14	73	56	24	**58***

^Test of Significance was Wilcoxon Signed Rank.
Bolded values with a '*' in the 'Mean conc. Reduction' column are significant at the $\alpha = 0.05$ level. All statistical tests of significance were Student's t, unless otherwise noted.

Sediment

Mean effluent concentrations of TSS from the Bingham and Raeford ponds were 13 and 24 mg/L, respectively. Both of these effluent concentrations were less than the good water quality target of 25 mg/L suggested by Barrett *et al.* (2004). Significant removal of TSS occurred at both ponds, with 58–77% removal. At Raeford, 6 of 15 TSS samples met or exceeded the Barrett *et al.* (2004) target; while at Bingham, only 2 of 18 TSS samples crossed the threshold. One potential concern with infiltrating wet ponds is that the water level would become too shallow to dissipate the influent stormwater's energy, resulting in re-suspension of sediment and higher TSS concentrations released in the ponds' outflow. This was not observed at either pond. However, a significant positive relationship

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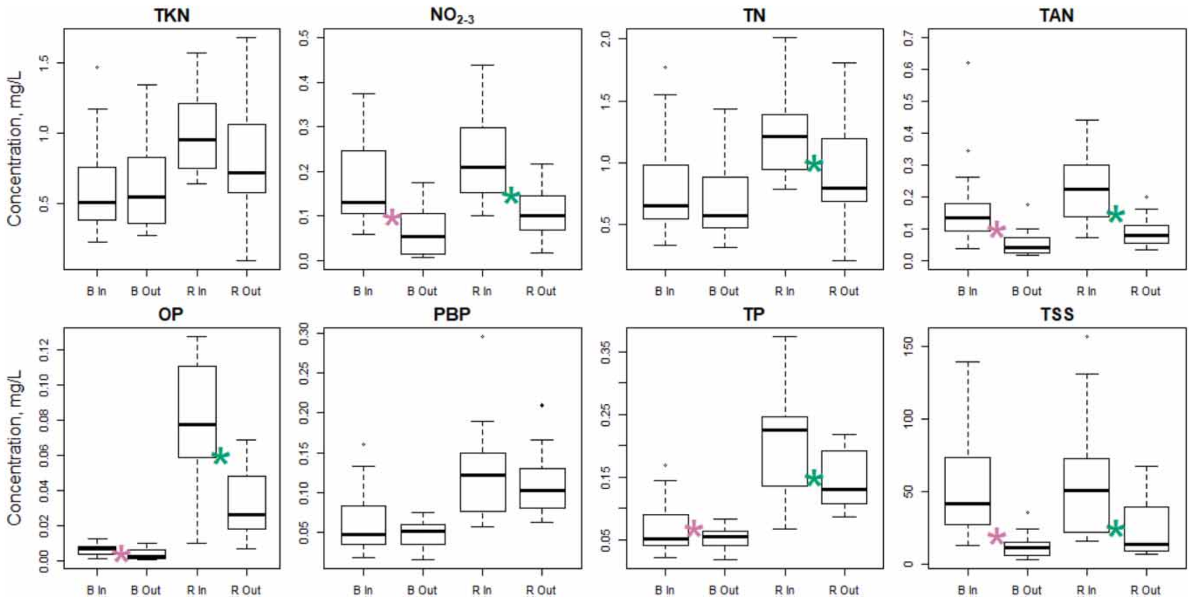

Figure 5 | Boxplots of nutrient and TSS concentrations at each monitoring station. Bingham and Raeford were abbreviated as B and R in this graphic. Significant differences between inlet and outlet concentrations are denoted by *.

did exist between the 5-min peak rainfall intensity and the influent TSS concentration at both ponds, likely due to higher intensity rainfall events eroding more sediment. These higher TSS concentrations appeared to be well mitigated by the ponds.

Phosphorus

Both ponds significantly reduced influent TP concentrations, with median concentration reductions of 9 and 42% at the Bingham and Raeford ponds, respectively. Bingham influent TP concentrations were lower than those of the Raeford pond (Figure 5). This difference is likely due to the extensive landscaping at the Raeford site (a residential complex) with phosphorus leaching expected from the fertilized turf (Soldat & Petrovic 2008). Providing further support for this theory were the substantially lower influent OP concentrations at the Bingham pond than those at the Raeford pond. Both ponds significantly reduced OP despite wet ponds possessing relatively few mechanisms for removing dissolved constituents (Istenič *et al.* 2012). Potential mechanisms for OP removal include the uptake of this pollutant by plants along the shelves or banks of each pond or dissolved phosphorus adsorption to sediment followed by sedimentation in the pond (Vymazal 2007). The extent to which each of these mechanisms occurred is unknown. Particle-bound phosphorus concentrations at the inlets and outlets were not significantly different at either pond.

At the Raeford pond (where influent TP concentrations were markedly higher than those at the Bingham pond), a significant positive relationship existed between the influent TP and the influent TSS (regression slope p-value < 0.0001), indicating that phosphorus was likely sediment-borne. However, the relationship at the Bingham pond was not significant. Similar to TSS, a significant positive relationship existed between the 5-min peak rainfall intensity and influent TP concentration at both locations.

McNett *et al.* (2010) considered the health of streams (e.g., excellent, good, fair, poor) in North Carolina as assessed by benthic macroinvertebrate tolerance to pollution and statistically related these to pollutant concentrations in-stream. This established benchmark effluent concentration goals. The effluent TP concentrations of the Bingham pond were consistently lower than the effluent target of 0.11 mg/L suggested by McNett *et al.* (2010) for good water quality. The Raeford pond's median effluent TP concentrations did not meet the same target.

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Nitrogen

Only the Raeford pond significantly reduced influent TN concentrations, but both ponds significantly reduced the NO_x-N concentrations with median concentration reductions exceeding 58%. The reduction in NO_x-N is suspected to be due to denitrification in anaerobic zones in the pond sediments where nitrate and nitrite would be converted to N_2(g) (Knowles 1982). It is also possible that some of the NO_x-N was taken up by plants along the shelves and banks (Vymazal 2007; Lenhart *et al.* 2012).

Inflow Total Ammonia Nitrogen (TAN) concentrations were significantly reduced in both ponds by at least 64%. The concentration reductions from inlet to outlet were likely due to one of three mechanisms: (1) adsorption of NH_4^+ ions to the negatively charged sediment particles in the stormwater; (2) NH_4^+ oxidation to nitrate via nitrification in the upper aerobic zone of the ponds, followed by subsequent reduction through denitrification processes; and (3) sequestration of NH_4^+ through plant uptake (Knowles 1982; Bannerman *et al.* 1993).

Organic nitrogen concentrations increased modestly as they passed through the Bingham pond, while those at the Raeford pond remained unchanged; neither change was statistically significant. At the Bingham site, grass clippings, leaf litter, and excrement from ducks and geese surrounding the pond were evident. Moreover, the landscaping crews cut the grasses to the edge of the pond, with mowers discharging the clippings onto the pond itself. Thus, it is likely that our inflow samples did not capture an important source of organic nitrogen at the Bingham pond, since sample strainers did not allow for collection of grass clippings. Grass clippings and leaf litter have been shown to contribute substantially to organic nitrogen in urban stormwater (Selbig 2016). Similar landscaping practices were not observed at the Raeford pond.

Because TAN was significantly reduced, the insignificant change in TKN and TN at both ponds were likely due to lack of significant reduction in ON. Both the Bingham and the Raeford ponds met the target good TN effluent concentration of 0.99 mg/L suggested by McNett *et al.* (2010). The concentrations of nutrients leaching from the basins into the groundwater were sufficiently low to not pose any problems for human health (Knobeloch *et al.* 2000).

Comparisons to other wet ponds

The mean effluent nutrient and TSS concentrations were not substantially different from those of non-infiltrating wet ponds previously monitored in North Carolina (Table 4). The mean TN effluent concentrations from non-infiltrating wet ponds ranged from 0.41 to 1.05 mg/L, while the mean effluent concentrations from the infiltrating wet ponds herein were 0.69 and 0.96 mg/L. The mean TP effluent

Table 4 | Mean effluent concentrations (in mg/L) from monitored wet ponds

Location	Reference	TKN	NO_x	TN	TAN	OP	TP	TSS
Bingham	Herein	0.63	0.06	0.69	0.05	0.004	0.05	13
Raeford	Herein	0.85	0.11	0.96	0.09	0.03	0.15	24
DOT (pre-retrofit)	Winston *et al.* (2013)	0.97	0.08	1.05	0.11	0.12	0.17	30
Museum (pre-retrofit)	Winston *et al.* (2013)	0.35	0.06	0.41	0.05	0.07	0.11	24
Ann McCrary	Mallin *et al.* (2002)	NA	NA	0.65	0.06	0.03	0.05	4
Silver Stream	Mallin *et al.* (2002)	NA	NA	0.51	0.04	0.02	0.06	6
Echo Farms Golf	Mallin *et al.* (2002)	NA	NA	0.62	0.08	0.04	0.07	4
Lakeside	Wu *et al.* (1996)	0.59	NA	NA	NA	NA	0.08	7
Waterford	Wu *et al.* (1996)	0.73	NA	NA	NA	NA	0.11	44
Runaway Bay	Wu *et al.* (1996)	0.63	NA	NA	NA	NA	0.08	22

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concentrations from non-infiltrating wet ponds varied from 0.05 to 0.17 mg/L, while those herein were 0.05 and 0.15 mg/L. The mean effluent TSS concentrations from these infiltrating wet ponds (13 and 24 mg/L) were less than those of a few non-infiltrating wet ponds, alleviating concerns about re-suspension when the water level was lower than the intended normal pool elevation.

Pollutant loading

Table 5 presents the summation of loads and percent load reduction for the sampled storm events at the Bingham and Raeford ponds. This analysis only includes events that had outflow and is therefore a conservative estimate of load reduction. The ON load at Bingham was reduced the least (37%) while the highest reduction was observed for TSS (85%). The ponds' abilities to reduce volumes through infiltration and evaporation are critical to their load reduction performance. For example, despite the Bingham pond discharging higher effluent ON concentrations, it still reduced ON loads (Table 5).

Table 5 | Summary of load reductions at the Bingham and Raeford ponds

Pollutant	Total inlet load (kg)	Total outlet load (kg)	Percent reduction total load (%)	Annual influent load (kg/yr)	Annual effluent load (kg/yr)	Annual influent load (kg/ha/yr)	Annual effluent load (kg/ha/yr)	Percent reduction annual load (%)	Annual load from undeveloped land in NC* (kg/ha/yr)
Bingham									
TKN	4.1	2.2	45	11.1	4.8	4.7	2	57	5.3
NO_x	1.1	0.4	68	2.8	0.8	1.2	0.3	74	1
TN	5.2	2.6	50	13.9	5.5	5.9	2.3	60	6.3
TAN	1.1	0.3	70	2.9	0.7	1.2	0.3	77	0.2
ON	3	1.9	37	8.2	4.1	3.5	1.7	50	–
Ortho-P	0.05	0.02	62	0.2	0.04	0.07	0.02	74	–
PBP	0.5	0.2	60	1.3	0.4	0.5	0.2	66	–
TP	0.6	0.2	61	1.5	0.5	0.6	0.2	68	0.5
TSS	451	66	85	1,109	141	468	59.5	87	349
Raeford									
TKN	7.3	4	45	14.5	7.9	7.5	4.1	46	5.3
NOx	1.7	0.7	57	3.3	1.4	1.7	0.7	58	1
TN	9	4.8	47	17.8	9.3	9.2	4.8	48	6.3
TAN	1.6	0.6	62	3.3	1.2	1.7	0.6	62	0.2
ON	5.7	3.4	40	11.4	6.7	5.9	3.4	42	–
Ortho-P	0.7	0.2	68	1.3	0.4	0.7	0.2	68	–
PBP	1	0.6	41	2	1.1	1	0.6	41	–
TP	1.6	0.8	49	3.1	1.6	1.6	0.8	49	0.5
TSS	504	167	67	981	325	506	168	67	349

Note: Only events with paired inlet and outlet samples were included in this analysis.
*Line & White (2007).

Percent load reductions at Raeford were generally lower than those at Bingham, reflecting that a lower fraction of water infiltrated at Raeford that at Bingham. Statistically significant reductions of loads were observed for all pollutants across both ponds. Annual loads of nitrogen, phosphorus, and sediment were reduced by at minimum 42, 41, and 67% by the two ponds, respectively. Effluent loads were in all cases lower than those reported for undeveloped land in the Piedmont of North

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Carolina (Line & White 2007), suggesting that these infiltrating wet ponds were able to abate the impacts of pollutant loads from their respective watersheds. In short, from a discharge volume and a nutrient and TSS load perspective, a development with an infiltrating wet pond functions as if it were a Low Impact Development (Dietz 2007).

DESIGN CONSIDERATIONS

Infiltrating ponds are not found in SCM design guidance documents (e.g., MPCA 2016; NCDEQ 2017). Yet, perhaps stormwater managers and designers should consider them as a hybrid practice of wet retention ponds and infiltration basins? For an infiltrating wet pond to be designed for peak flow attenuation, water quality treatment, and volume reduction, certain design elements will differ from those of a standard wet retention pond. An infiltrating wet pond will not have a liner across its entire footprint; rather, any impermeable membrane/layer would be restricted to forebays and perhaps other deep pools. Small pools should retain water year-round, serving (1) as a habitat for mosquito predators (Greenway *et al.* 2003; Hunt *et al.* 2006) and (2) to prevent re-suspension of sediment. Limiting liners to small fractions of the pond allows infiltration to occur across the majority of the pond's footprint, yet still maintains some water in critical areas for energy dissipation and mosquito predator habitat. Exactly how much infiltration may occur from one of these ponds is variable due to underlying soils. However, if *in situ* soils allow at least 5 mm/h of infiltration post-construction, that location might be a good candidate for infiltrating pond implementation in humid subtropical regions.

Because of highly variable water elevations within the pond, vegetation selection differs from that of traditional wet ponds with aquatic shelves and constructed stormwater wetlands. Any species selected must be both highly drought and water inundation tolerant. An initial recommendation for vegetation selection for infiltrating wet ponds was made by Hunt *et al.* (2020).

Finally, designers should consider contexts when, even if, underlying soils allow ponds to infiltrate, lining to prevent infiltration is still a best practice. While the measured nutrient concentrations herein were not alarming from a human-health consideration, there are times when runoff can carry toxic pollutants that should be kept from reaching groundwater. In these cases, a liner should be employed when ponds are sited over permeable soils.

Further research opportunities exist for infiltrating wet pond performance in less humid climates, especially in those where all temporarily stored water could evaporate. How does an 'empty' wet pond reduce pollution in incoming stormwater? Does a risk exist for pollutant resuspension? Also, this study focused on sediment and nutrients, but wet ponds are expected to treat many other pollutants as well (e.g., microplastics (Olesen *et al.* 2019) and Polycyclic Aromatic Hydrocarbons (Stephansen *et al.* 2020)). How might infiltrating wet ponds retain these pollutants?

SUMMARY AND CONCLUSIONS

Standard, non-infiltrating, wet ponds are designed specifically for peak flow attenuation and water quality treatment. They are not intended to reduce runoff volume, and are often designed, at high cost, to *purposefully* restrict infiltration. Infiltrating wet ponds offer a unique opportunity for volume reduction by dewatering below the normal pool prior to a storm event. Infiltrating wet ponds, when underlying soils allow, appear to be a viable practice for both mitigation of urban hydrology and improvement of runoff quality. The authors conclude that infiltrating wet ponds be a recommended type of SCM, when conditions allow, based on the following summary of results:

1. Substantial volume reductions were found when wet ponds were unlined, or partially lined, and located over primarily sandy, HSG A soils. One of the two infiltrating wet ponds studied infiltrated 56% of the annual influent runoff volume. Approximately 5–6% of the annual influent volume evaporated.
2. In months with low rainfall (<37 mm) these infiltrating wet ponds captured and infiltrated 100% of the influent volume. Volume reductions were lower for months with higher rainfall but still statistically significant.
3. The infiltrating wet ponds significantly reduced the peak flow rates for large storm events as is characteristic of standard wet ponds. The median peak flow reduction was 99% at both the Bingham and Raeford ponds.
4. Both ponds significantly reduced the NOx, TAN, OP, TP and TSS concentrations from inlet to outlet. The effluent concentrations appear to be on par with other non-infiltrating wet ponds.
5. Due to low effluent concentrations and substantial runoff volume reductions, the infiltrating wet ponds were able to significantly and substantially reduce the nutrient and TSS loadings to levels consistent with undeveloped land in North Carolina.

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DATA AVAILABILITY STATEMENT

All relevant data are available from an online repository or repositories (https://catalog.lib.ncsu.edu/catalog/NCSU3222699).

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