hydrology

MDPI

Article

Nitrogen Treatment by a Dry Detention Basin with Stormwater Wetland Characteristics

Charles P. Humphrey, Jr. *, Guy Iverson and Melissa Nolan

Environmental Health Sciences Program, Department of Health Education & Promotion, East Carolina University, Greenville, NC 27858, USA; iversong18@ecu.edu (G.I.); nolanme20@students.ecu.edu (M.N.)
* Correspondence: humphreyc@ecu.edu

Abstract: Dry detention basins (DB) are commonly used to reduce the rate of runoff in urban areas and may provide open space for recreation between storms. However, most are not effective at nitrogen removal in comparison to other measures, such as constructed wetlands. The study goal was to assess the nitrogen treatment efficiency of a DB that exhibited some wetland characteristics, including saturated soil near the inlet and wetland vegetation that covered 40% of the surface area. Influent and effluent samples were collected during multiple stages of eight storm events for nitrogen concentration analyses. High-frequency water stage, pH, dissolved oxygen (DO), and temperature loggers were deployed at the inlet and outlet prior to anticipated rain. As stormwater passed through the DB, the event mean concentrations (EMCs) and masses of TN declined by 20.7% and 52.3%, respectively, while the DO and pH dropped by 62% and 20.5%, respectively. Load reductions of TN exceeding 93% were observed during two small storms with rain depths of less than 0.16 cm and when the outflow volumes were reduced by greater than 82%. Temperature was significantly correlated ($p < 0.001$; $r = 0.964$) with volume reductions (via infiltration and evapotranspiration), and, thus, the treatment was better during warmer periods. The DB was effective at removing inorganic nitrogen, likely via nitrification, denitrification, and immobilization, but frequently exported higher EMCs of organic nitrogen. Overall, the DB exceeded the 10% TN removal expectation for dry basins. The findings from this study suggest that the TN treatment efficiency of DBs may be improved by incorporating wetland characteristics.

Keywords: constructed stormwater wetland; dry detention basin; nitrogen; stormwater; urban runoff

check for updates

Citation: Humphrey, C.P., Jr.; Iverson, G.; Nolan, M. Nitrogen Treatment by a Dry Detention Basin with Stormwater Wetland Characteristics. *Hydrology* **2022**, *9*, 85. https://doi.org/10.3390/hydrology9050085

Academic Editor: Shirley Gato-Trinidad

Received: 2 March 2022
Accepted: 6 May 2022
Published: 12 May 2022

1. Introduction

Nitrogen runoff to surface waters has been shown to cause a proliferation of algal blooms, some of which produce toxins that are deadly to humans, animals, and aquatic organisms [1,2]. Algae mats can cover the water surface, shading submerged aquatic vegetation (SAV) and causing their demise and a loss of habitat [3]. Decomposition of algae and SAV may result in a depletion of dissolved oxygen (DO) and impairment of water resources [4,5]. For example, nutrient runoff stimulated harmful algal blooms in Lake Erie that produced toxins, and the water treatment plant for Toledo, OH, USA was unable to remove them, causing more than 500,000 people to be without their primary water supply in 2014 [2]. The global increase in N loading to water resources is associated with an increase in human population. As the population grows, so too does agricultural and urban runoff stemming from increased fertilization of croplands [6] and more impervious surfaces associated with housing, road, and parking lot construction [7], which prevent rainwater infiltration into the soil.

Stormwater runoff is recognized as one of the most common non-point sources of water pollution [8]. When natural landscapes are covered with asphalt and other hard surfaces, rainwater is unable to infiltrate and percolate through the soil, resulting in an increase in overland flow and runoff that is transported directly to streams, lakes, and other waterways [9]. The volume and energy of the runoff water may cause erosion of streambanks and beds, increasing sediment transport and harming the aquatic habitat [10]. Urban runoff also contributes to flash floods, property damage, and creates a hazard for people living in areas adjacent to the drainageways [7,11]. Runoff water from parking lots during warm seasons may increase the water temperatures in nearby streams, which can have detrimental effects on aquatic organisms [7,12]. Many freshwater aquatic species cannot survive when water temperatures exceed 32 °C [13]. Increases in stream temperature result in decreased concentrations of DO [12], which may also negatively influence the health of aquatic organisms [4]. Stormwater control measures (SCMs) are practices used in urban areas to slow and treat stormwater runoff before it reaches surface waters. Dry detention basins (DBs) were one of the first SCMs commonly used to reduce the peak rate of runoff [14,15]. DBs typically include a depression-shaped grassed basin that receives runoff via an inlet pipe, and water exits via a smaller-diameter outlet pipe. During a rain event, runoff enters the basin but cannot exit at the same rate due to the smaller outlet pipe [16]. Thus, water ponds temporarily in the basin as drawdown occurs, thereby decreasing the peak flow rates and discharge to nearby receiving waters during the storm events [17]. While DBs can be effective at reducing peak flows, their ability to treat common pollutants associated with stormwater, such as nitrogen, is highly variable. For example, a review [18] of the influent and effluent characteristics of 25 dry detention basins showed that most did not provide any nitrogen removal, and many were a source of and exported nitrogen to the storm drainage system. A more recent study showed that the event mean concentrations (EMCs) of dissolved nitrogen for inflow and outflow from a basin were identical, also indicating no significant decrease in the nitrogen concentrations as the stormwater passed through the DB [19]. Other studies have shown that DBs can reduce nitrogen loads by greater than 60%, mostly via volume reduction from infiltration of stormwater [17,20,21]. The fate of nitrogen that infiltrates the soil of the DBs is often not accounted for [22,23], and studies of other non-point sources show that, if mass removal mechanisms, such as denitrification, do not occur in the subsurface, nitrogen may ultimately be delivered to adjacent surface waters via groundwater transport [24,25]. SCMs, such as constructed wetlands, have also shown high variability with regard to nitrogen treatment, but they are typically considered more efficient than DBs due to enhanced biochemical processes, such as denitrification and immobilization [15,22,23].

Unlike dry detention basins, stormwater wetlands often have permanent pools of water and hydrophytic vegetation that provide carbon to saturated soils. The water-logged and organic-matter-rich soils allow for an anaerobic environment where alternative (to oxygen (O_2)) electron acceptors are used by soil microorganisms for energy transfer [26]. When O_2 is absent, nitrate (NO_3^-) is the preferred electron acceptor for microorganisms to oxidize organic matter [27]. Denitrification of NO_3^- results in gaseous loss of nitrogen as dinitrogen (N^2) or nitrous oxide (N_2O), and effectively lowers the concentration and mass of nitrogen in soil and/or water [22,23]. Prior studies have shown that the biochemical processes that are active in wetlands are typically much more conducive to denitrification relative to dry detention basins, thus resulting in superior nitrogen removal in wetlands. For example, in a review of 19 stormwater wetlands, Collins and others [18] reported a median nitrogen treatment efficiency of 48%, greatly exceeding the 0% reduction in nitrogen exports from 25 dry basins reviewed in the same publication. Koch et al. [28], in a review of 19 wetland and 7 DBs, reported a mean TN efficiency for wetlands that was 34% greater relative to DBs. McPhillips and Walter [15] showed that denitrification potential was higher for wet basins in comparison to dry basins. The State of North Carolina assigns a nitrogen removal credit of 44% for wetlands and only 10% for dry basins that are constructed to treat urban runoff [29]. While constructed wetlands are often more efficient at nitrogen treatment, they are often more expensive to design, install, and maintain and require more land relative to dry basins [30,31]. Moreover, the open space of the dry basins between storm events (when dry) may provide for recreational activities (e.g., volleyball, soccer), while wetlands typically do not provide those benefits. There may be trade-offs for land use planners when selecting between traditional constructed wetlands and dry basins for controlling stormwater runoff. Thus, an SCM that couples the denitrification potential of a wetland with the recreational opportunities of a dry basin may be enticing to resource managers and developers. Recent research [21] has shown that DBs with overgrown wetland tree and shrub species may provide nitrogen removal services that exceed the 10% credit assigned to dry basins by the North Carolina Department of Environmental Quality. However, more work is needed to determine if dry basins with extensive areal coverage of herbaceous vegetation may also be effective at nitrogen removal. The goal of this study was to assess the nitrogen treatment efficiency of a DB that had volunteered wetland plants colonize 40% of the surface area of the SCM. The basin has both wetland and dry basin plant communities and characteristics.

2. Materials and Methods

2.1. Site Selection

The DB evaluated in this study is located on the campus of East Carolina University (ECU) in Greenville, North Carolina, USA. The DB was installed in 2010 to coincide with construction of an athletic complex for the university. An agreement between the city of Greenville and ECU allowed stormflow from ECU's new facility to be discharged to the city's stormwater infrastructure. However, ECU had to install a control measure ensuring stormwater flows delivered to the city's storm conveyance network were like pre-construction conditions. The DB was installed for that purpose. The DB receives drainage from a watershed that covers approximately 15.8 ha of mostly residential development (Figure 1). The impervious surface for the drainage area is 30%, and Greenville receives an average of 126 cm of rainfall [32]; thus, significant runoff for the watershed occurs during most years. The DB includes an inlet pipe (121.9 cm diameter) and two outlets. The primary outlet is a 20.3-cm-diameter metal pipe with a secondary outlet pipe (45.7 cm diameter) located 60 cm above the primary. Surface area of the DB is approximately 0.2 ha, and the basin vegetative cover originally included perennial grass. Landscaping crews for the university mowed the grass in the basin approximately once per week during the growing season. However, after a few years, soil wetness increased and *Typha* spp. (cattails) began growing near the basin inlet, which made mowing that portion difficult. The *Typha* were cut and removed once or twice per season for a few years. Due to budget constraints and loss of personnel, efforts to contain the *Typha* were reduced, and, by 2018, the *Typha* extended from the inlet to just north of the outlet pipe. *Typha* currently cover approximately 40% of the surface area, and grass 60%. The dense vegetation provided sufficient groundcover such that no evidence of erosion or gullies between the inlet and outlet were observed. Images of the DB are shown in Figure 2. The soils in the DB are mapped as Goldsboro sandy loam near the inlet and extending about halfway to the outlet (Supplemental). Goldsboro series soils typically have a seasonal highwater table that is 0.6 to 0.9 m below the surface [33]. However, stormwater infiltration near the inlet of the DB likely caused the soil moisture content and water table height to increase, facilitating the growth of the obligate wetland plants. Build-up of organic debris, common in wetland soils, was observed between the inlet and outlet. The soils north of basin outlet are mapped as Wagram loamy sand, which are better drained with a water table 1.5 to > 2 m deep [33], and organic debris accumulation was not observed on the surface. Outflow from the DB is piped to the storm drainage system for the City of Greenville and discharges into Greens Mill Run, a tributary of the Tar River. Both Greens Mill Run and the Tar River are on the 303(d) list of impaired waters due to poor aquatic habitat and excess nutrient content, respectively [34]. Efforts to improve water quality via better runoff management and nutrient load reductions have been encouraged by state and federal environmental regulatory agencies.

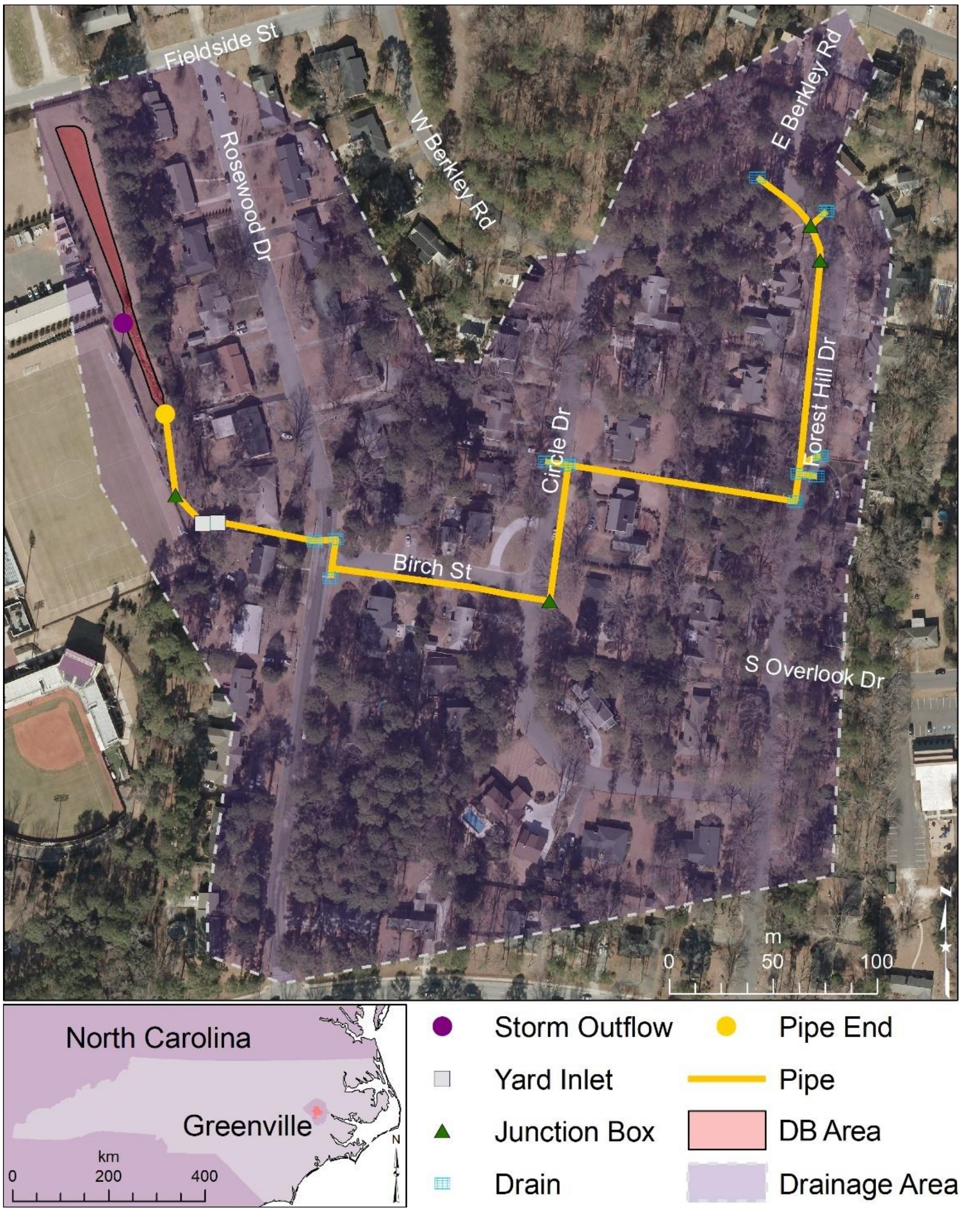

Figure 1. A dry detention basin (DB) with wetland characteristics receives drainage from a 15.7 ha watershed in Greenville, NC, USA.

(**a**)

(**b**)

Figure 2. *Cont*.

(**c**)

Figure 2. The hybrid dry basin/wetland is covered with 40% wetland plants, such as cattails (*Typha* sp.), near the inlet (**a**), while the rest of the surface area is mostly grass (**b**). The boundary between wetland plants and upland plants is shown in (**c**).

2.2. Hydrological Monitoring

Stilling wells were installed near the inlet and outlet pipes of the DB. The stilling wells were constructed by cementing a slip cap on the bottom of a 1.5-m section of 5-cm-diameter PVC well screen. The capped end was placed on the ground surface and the stilling well was attached via zip ties to an iron stake that was driven into the soil. A stage plate was also attached to the stilling well and stakes to enable researchers to quickly observe water levels at the inlet and outlet during a storm. Water level loggers (*Solinst Levelogger 3001*) were programmed to record pressure and temperature every 10 min and lowered to the bottom of the wells at the inlet and outlet. An additional logger was tied to a nearby tree to record pressure also at 10 min intervals and to allow for atmospheric pressure compensation of the stilling well readings. The loggers were deployed in July 2020 and recorded until June 2021. Velocity of water entering and exiting the dry basin was measured using a *Global Water FP101* flow probe, and inflow and outflow (L s^{-1}) was calculated by multiplying water velocity by cross-sectional area of water in the inlet and outlet pipes. During flow measurements, the stage readings near the stilling wells were recorded. Rating curves were developed for the inlet and outlet to estimate discharge rates. Regression analyses were used to calculate the drawdown rate for the rain events. Total inflow into the dry basin for monitored storms was estimated using the simple method [35], along with watershed characteristics (e.g., drainage area, impervious surface) and rainfall data from a USGS gauge in Greenville, NC [36]. Outflow volumes were estimated using the outlet rating curve. Volumes of inflow and outflow for the monitored storms were compared.

2.3. Field Readings and Sample Collection

A *Hanna Instruments* (HI) 9828 multiparameter was used to program autonomous logging probes that recorded temperature, pH, and dissolved oxygen concentrations every 10 min. The probes were placed at the inlet and outlet of the basin prior to 8 sampling events. These data were used to assess biochemical processes that may be occurring in the basin with regard to nitrogen transformations as stormflow moved through the basin. Samples of inflow and outflow were manually collected during 8 storm events in 2020 using HDPE bottles. Samples were typically collected when the stage increased or decreased by 4 to 5 cm. Sample bottles with inlet and outlet water samples were placed in ice-filled coolers and transported to the lab for filtration and analyses.

2.4. Laboratory Analyses of Samples

Water samples were filtered using *Whatman GF/F 0.7*-µm-pore-size filters. Filtrate was analyzed for total dissolved nitrogen (TDN) and dissolved organic carbon (DOC) using a *Shimadzu* TNN-L analyzer with catalytic thermal decomposition/chemiluminescence [37]. A Unity *SmartChem 200* discrete analyzer (Unity Scientific) was used to determine concentrations of ammonium–nitrogen (NH_4^+-N) and NO_3^--N in the filtrate using standard methods [38–41]. The filters were digested, and the digested samples were placed in the SmartChem for analyses of particulate nitrogen (PN) using the same methods. Concentrations of dissolved organic nitrogen (DON) were estimated by subtracting the concentrations of NH_4^+-N and NO_3^--N from TDN. Total nitrogen (TN) concentrations were estimated by summing TDN and PN concentrations.

2.5. Statistical Analyses

Individual samples of TN, PN, TDN, DON, NO_3^--N, NH_4-N, and DOC were flow-weight-composited to determine an event mean concentration (EMC) for the inlet and outlet for each storm. The EMCs were multiplied by inflow and outflow volumes to determine nutrient loads entering and exiting the DB (through the discharge pipe) during each storm. Treatment efficiency with regard to concentrations of nutrients and mass of nutrients for the DB was calculated using Equation (1).

$$Treatment\ Efficiency = \left[\frac{Influent - Effluent}{Influent}\right] * 100\% \quad (1)$$

Physicochemical properties of inflow and outflow (pH, temperature, DO) were also compared to assess potential nitrogen removal mechanisms. Mann–Whitney non-parametric tests or paired *t*-tests were used to determine if the differences in EMCs or nutrient masses entering and exiting the DB were statistically significant ($p < 0.05$). Mann–Whitney non-parametric tests were used when the data did not follow a normal or log normal distribution, and paired *t*-tests were used when normal or log normal distributions of the data were observed. Spearman correlation analyses were used to determine if significant relationships between temperature, rain amount, inflow volumes, outflow volumes, and volume reductions were observed for storms that generated measurable runoff. Multiple linear regression analyses using the hydrological variables were also performed. Statistical analyses were conducted with *R* statistical software and *Minitab 18*. The frequencies that EMCs and masses of nutrients for inflow samples exceeded outflow were also reported.

3. Results and Discussion

3.1. Flow Characterization

During the 1-year monitoring period (July 2020–June 2021), the watershed received 136.2 cm of precipitation. There were 127 days during the 365-day study period when the DB received at least 0.01 cm of rainfall (Supplementary Material). The average rainfall depth was 1.07 cm, with a daily range of 0.01 to 7.70 cm for days when some precipitation fell. The monthly rainfall totals ranged from 2.41 cm in April 2021 to 24.16 cm in June 2021 (Figure 3). Overall, during the study period, the watershed received about 8% more precipitation than the average year. Regression analyses revealed that the log10 of discharge was significantly associated with the log10 of the inlet ($r^2 = 0.590$; $p = 0.004$) and outlet ($r^2 = 0.612$; $p = 0.003$) water stages. These data indicate that, when the discharge increased, so too did the stage. The discharge data were variable, especially when the inlet and outlet stages were relatively low (under 5 cm), possibly because of differences in vegetative thickness and sediment accumulation that influenced the velocity measurements. Moreover, during some storms when the stage exceeded the top of the primary outlet pipe elevation, backwater influenced the inlet stage readings and flow estimates; thus, the simple method was used to estimate the inflow volumes, while the outlet stage and rating curve were used to estimate the outflow volumes. The maximum inlet and outlet stage readings of 32.8 cm and 62.8 cm, respectively, occurred on 15 August 2020 when 7.7 cm of rain was received within a 24-h period. The water levels returned to pre-storm stages within a day following the peak. The overall average drawdown of water in the DB following storms was 4.3 ± 3.1 cm h^{-1} (Figure 4). Differences in drawdown rates were observed when comparing the warm growing season (May–October) and the cool dormant months (November–April). The mean drawdown during the warm months (5.6 cm h^{-1}) was significantly faster ($p = 0.04$) relative to the drawdown during the cooler months (3.2 cm h^{-1}). Greater transpiration and evaporation due to actively growing plants, longer day light hours, and higher temperatures likely contributed to quicker drawdowns in the warm months [42]. The mean air temperature during the warm months of the study period was 20.8 °C, while the mean air temperature during the cool months was 10.1 °C. It is also possible that there was more infiltration during the warm months due to warmer waters and quicker infiltration [43], along with deeper antecedent groundwater levels and thicker vadose zones typically observed during the growing season in North Carolina [44].

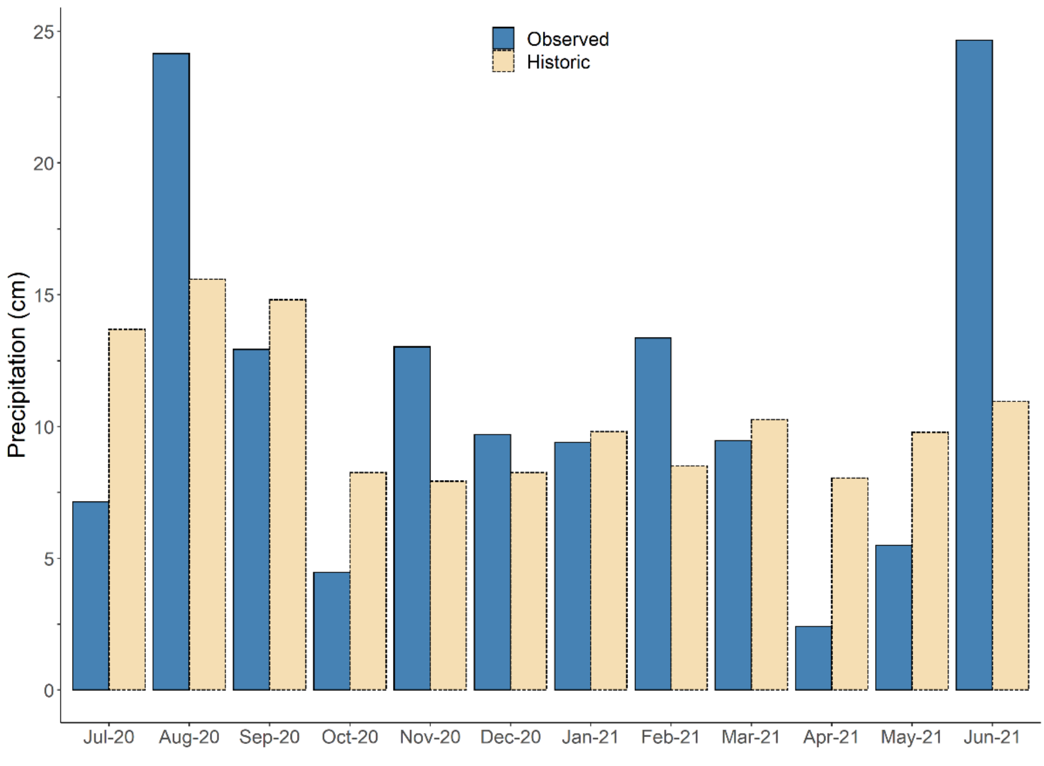

Figure 3. Average and observed monthly rainfall for Greenville, NC between the months of July 2020 and June 2021. Rainfall during the study was similar to the long-term historic mean, with 5 months exceeding the mean average and 7 months below the mean average. Overall, there was 8% more rainfall during the study than the long-term mean.

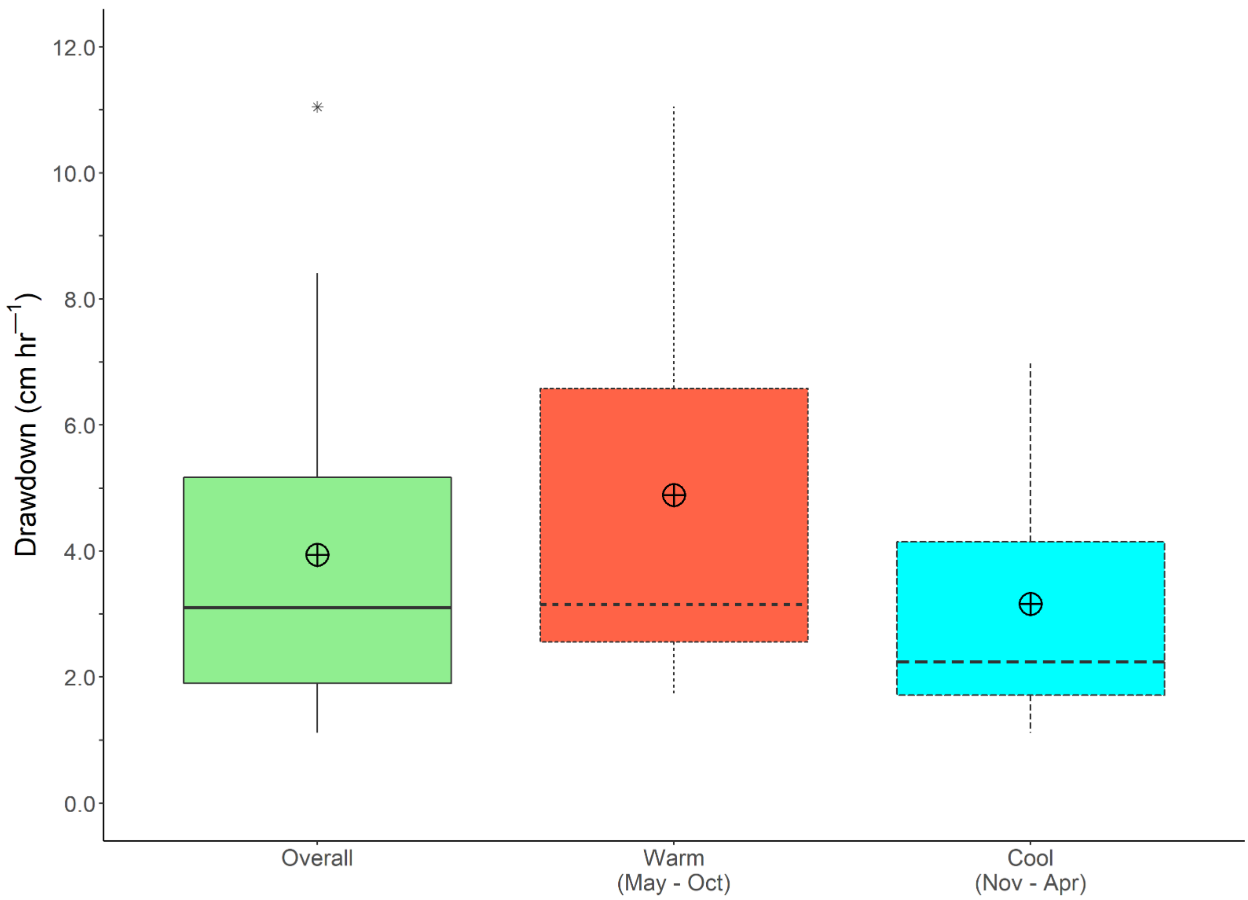

Figure 4. Water level drawdown in the dry detention basin during the warm season, cool season, and overall. Statistical outliers are shown as (*).

One of the main functions of SCMs is to reduce the rate of runoff delivered to receiving waterways during storms, and the DB was successful in that regard. For example, a rain that causes a 26-cm rise in the water level at the outlet stage of the DB would slow runoff by more than 6 h on average. During the study, the outlet stage exceeded 26 cm on 16 different occasions. Flow reductions, likely via infiltration and evapotranspiration, were common during the warm growing season (Table 1). For example, between May and October of the study period, the volume of effluent discharged through the outlet pipe was 33% less than the basin inflow volume. Volume reductions were highly variable though and ranged from −4% to 95% during the warm season. During the cool season between November and April, outflow exceeded inflow by 18%, likely because of groundwater inputs (baseflow) into the DB and less evapotranspiration. Water table depths in North Carolina are typically closer to the surface during cooler months, when evapotranspiration rates are lower and, thus, infiltration and groundwater recharge are more likely [44]. The outflow volume reduction percentage was significantly correlated to temperature ($r = 0.761$, $p < 0.001$), indicating that, as temperatures increased, the volume of outflow typically decreased. The volumes of inflow and outflow were also significantly correlated ($r = 0.964$, $p < 0.001$), so more outflow was expected during larger storms and increased inflow. Overall, there was a 5% decrease in outflow volumes during the entire study (Table 1). Multiple regression analyses using the variables in Table 1 revealed a significant ($p < 0.0001$) relationship $r^2 = 0.88$ with resultant Equation (2). However, the only significant coefficient was temperature. To refine the analysis and improve the fit of the model, a step-wise regression using the full model (Equation (2) was conducted to identify the ideal model based on the Akaike information criterion. The reduced model removed all variables from Table 1 except temperature and inflow volume (Equation (3) and revealed a stronger $r^2 = 0.89$, p-value < 0.0001, and had significant coefficients, including temperature ($p < 0.0001$; negative correlation) and inflow volume ($p < 0.0001$). These data suggest that the most important influencing factors related to outflow following rain events in DB were temperature and inflow volumes.

$$Outflow\ Vol. = 858.78 + 252.69(rainfall\ depth) + 9.63(days\ since\ rain) - 56.56(Temp.) + 0.52(inflow\ volume) - 148.76(rainfall\ intensity) \quad (2)$$

$$Outflow\ Vol. = 870.45 - 57.39(Temp.) + 1.01(inflow\ vol.) \quad (3)$$

Table 1. Storm and hydrology characteristics during the study period. Overall is total rainfall, total inflow, total outflow, mean volume reduction, mean temperature, mean rain intensity, and mean days since last rain.

Dates	Rain (cm)	Inflow (m^3)	Outflow (m^3)	Vol. Red. (%)	Temp (°C)	Rain Intensity (cm h^{-1})	Days Since Last Rain
23–24 July 2020	3.58	1806	1198	34	25.6	0.15	12.67
4 August 2020	5.16	2600	1611	38	25.4	0.34	1.8
7–11 August 2020	7.29	3676	3459	6	25.3	0.08	3.3
15 August 20	7.7	3881	2089	46	26.1	1.18	1.4
19 August 2020	2.29	1153	468	59	23.8	1.83	2.7
24 August 2020	0.33	167	45	73	24.9	0.33	2.6
9 September 2020	1.73	871	444	49	26.1	0.26	8.4
11 September 2020	1.02	512	198	61	26.6	0.58	1.8
17–18 September 2020	2.79	1409	959	32	24.1	0.08	2.6
29 September 2020	3.2	1614	1121	31	21.9	0.16	2.9
11 October 2020	0.38	192	34	82	21.5	0.03	10.5
16 October 2020	0.28	141	7	95	18.3	0.02	4.6
25 October 2020	3.53	1780	1226	31	19.4	0.43	8.8
1 November 2020	1.35	679	498	27	19.3	1.35	2.7
11–12 November 2020	9.22	4650	4677	−1	19.5	0.41	10.1
14 December 2020	1.57	794	595	25	14.8	0.18	6.6
16 December 2020	2.59	1307	1346	−3	9.3	0.37	1.8
20 December 2020	1.3	653	722	−11	7.2	0.23	3.7
24 December 2020	1.6	807	1180	−46	15	0.22	4.1
31 December 2020–3 January 2021	5.11	2575	3396	−32	11.9	0.08	7.4
8 January 2021	0.3	154	625	−306	4.7	0.03	5.1
26–28 January 2021	2.44	1230	1954	−59	7.7	0.49	10.3
31 January 2021	2.82	1422	1539	−8	6.3	0.19	3.3
5 February 2021	0.79	397	528	−33	7.5	0.11	3.75
7 February 2021	0.41	205	502	−145	7	0.04	1.4
11–16 February 2021	5.79	2920	4586	−57	6.9	0.83	4.2
18–19 February 2021	4.32	2178	3213	−48	5.2	0.13	1.9
22 February 2021	0.61	355	355	−16	14.7	0.08	3
26 February 2021	1.27	895	895	−40	8.7	0.18	3.8
16 March 2021	4.14	2058	2058	1	8.8	0.22	14.2
19 March 2021	0.71	792	792	−121	11.6	0.14	2.9
27 March 2021	2.08	345	345	67	17.6	1.66	3.5
31 March 2021	2.24	703	703	38	16.9	0.32	3.8
11 April 2021	1.24	628	301	52	18.1	0.83	1.1
3–5 May 2021	2.9	1460	1515	−4	22.5	0.05	7.8
29 May 2021	0.97	487	179	63	21.6	0.26	1.6
30 May 2021	0.76	384	362	6	18.3	0.34	0.2
Overall	95.8	47880	45725	5	16.5	0.38	4.7
Warm	43.9	22134	14913	33	23.2	0.4	4.6
Cool	51.9	26169	30812	−18	11.4	0.4	4.7

The discharge through the outlet pipe was lower than the inflow volumes during seven of the eight storm events monitored for water quality (Table 2). The volume reductions ranged from −19% to 95%, with a median reduction of 34% for those eight storms. A volume reduction of 38%, similar to the median of all the storms monitored for water quality, was observed for storm 1. The highest flow reduction (95%) of the events sampled for water quality was observed during storm 6 in response to a small rain (0.26 cm) that occurred over a 1-h period and when antecedent conditions were relatively dry (only 0.56 cm in previous week). While the inlet stage increased by approximately 7 cm, the outlet stage rose less than 0.6 cm during storm 6. Thus, most of the runoff entering the basin likely infiltrated. In contrast, the outflow from the basin exceeded the inflow by 19% for storm 8, which delivered 7.8 cm of rain over a 19-h period. Storm 8 had the highest peak rainfall intensity of the monitored storms (1.83 cm h^{-1}) and received 1.4 cm of rain the prior week. The increase in basin outflow may have been associated with baseflow (groundwater) inputs into the basin and less evapotranspiration. Figure 5 shows hydrographs for storm 1 (38% volume reduction, close to the median for storms sampled), storm 6 (95% volume reduction), and storm 8 (−19% volume reduction).

Table 2. Hydrological characteristics of 8 storm events sampled for water quality.

Storm	Rain (cm)	Overall Int. (cm h^{-1})	Peak Int. (cm h^{-1})	Rain (cm) Prior 7 days	Inflow (m^3)	Outflow (m^3)	Volume Red. (%)	Temp (°C)	Inlet TN (mg L^{-1})	Outlet TN (mg L^{-1})
1	5.16	0.34	1.3	0.61	2600	1611	38	25.4	5.20	2.64
2	2.74	0.08	0.4	1.14	1383	972	30	24.1	2.47	3.53
3	3.73	0.16	1.8	0.53	1883	1153	39	20.4	2.48	1.76
4	3.23	0.16	1.8	3.99	1627	1594	2	21.9	2.66	1.70
5	0.38	0.03	0.1	0.02	192	34	82	21.5	5.92	2.17
6	0.28	0.28	0.3	0.56	141	7	95	18.3	2.25	3.07
7	1.35	1.35	1.3	3.63	679	498	27	17.6	3.36	2.02
8	7.77	0.41	1.8	1.78	3920	4655	−19	19.5	2.63	1.91

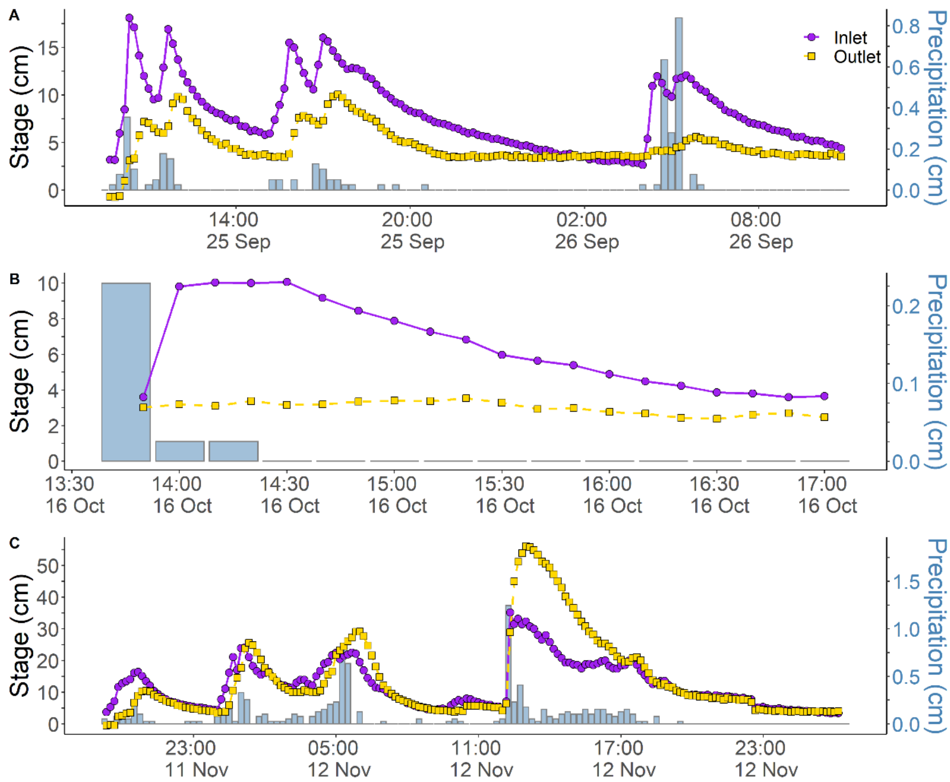

Figure 5. Hydrographs showing inlet- and outlet stage responses to rain events. (**A**) shows a storm during which a 38% reduction in outflow was observed, (**B**) shows a 95% reduction in outflow, and (**C**) shows a 19% increase in outflow, likely due to groundwater inputs.

3.2. Total Nitrogen Treatment

The median EMC of TN entering the DB (2.65 mg L^{-1}) was significantly greater (p = 0.04) relative to the median outflow EMC (2.10 mg L^{-1}) (Figure 6A). During 75% of the storms sampled, the outflow concentrations of TN were lower relative to inflow (Figure 6C). Overall, a 20.7% reduction in the EMC of TN was observed as stormflow entered and exited the DB through the outlet pipe. The concentrations of TN entering and exiting the DB were variable, with ranges in concentrations of 2.95 mg L^{-1} and 1.37 mg L^{-1}, respectively. The median concentration of TN entering the DB (2.65 mg L^{-1}) was within the spectrum of urban runoff concentrations (1.3–3.2 mg L^{-1}) summarized in a literature review of more than 3750 sites by Collins et al. [20], similar to the influent concentrations (2.9 mg L^{-1} reported in an SCM study in Maryland [35], and similar to the influent concentrations reported for 66 SCMs in Denmark by Sonderup et al. [45]. The median effluent concentration of TN for the BD (2.1 mg L^{-1}) was also within the range of TN concentrations for dry detention basins reported by Wissler et al. [21] (0.64–3.18 mg L^{-1}), Collins et al. [18] (1.81–3.63 mg L^{-1}), and Mazer [46] (1.25–2.85 mg L^{-1}). The two highest EMCs of inflow TN were observed during storms 5 (5.92 mg L^{-1}) and 1 (5.20 mg L^{-1}), when the antecedent conditions were relatively dry for both (0.02 and 0.61 cm of rain the prior 7 days before the storm events) (Table 2). The elevated TN concentrations may be related to build-up of nutrients on impervious surfaces during dry, antecedent periods and wash-off during rain [47]. The outflow EMCs of TN were elevated relative to the inflow during storms 2 and 6 due to increases in the EMCs of DON of 175% and 85%, respectively, between the inlet and outlet of the DB. Immobilization of inorganic nitrogen by wetland plants and microorganisms is a mechanism for lowering the EMCs of nitrogen [18], but that biomass may contribute to later exports of DON. For example, a recent study [19] evaluating the nitrogen removal efficiency of a DB that was converted into a stormwater wetland also showed that DON was commonly exported from the wetland, while inorganic nitrogen was removed. A multiple regression analysis using variables in Table 2 was performed, but the model did not produce statistically significant ($r^2 = -1.279$; p = 0.9281) results, and, thus, relationships between outflow TN concentrations and other parameters could not be confirmed.

The TN loads leaving the DB through the outlet pipe were significantly (p = 0.021) lower relative to the inflow masses (Figure 6B). During seven of the eight sampling events, the influent TN loads exceeded the outflow loads. The load reductions ranged from −0.3% to 93.5% for the eight storms (Figure 6D). The mean TN load reduction for the eight storms was 52.3% based on differences in the effluent masses of TN discharged from the outlet pipe and influent TN loads. However, it should be noted that there was a 33% reduction in the outflow volume relative to the inflow volume for the eight storms, and the ultimate fate of TN that infiltrated soil and percolated to groundwater is unknown. Groundwater transport of nitrogen from other non-point sources, such as septic systems, has been shown to be a significant contributor of nutrients to surface waters in the Coastal Plain of NC [24,25]. Those studies [24,25] included instrumentation of research sites with groundwater monitoring wells between the septic systems and creeks. If nitrogen in the groundwater beneath the DB is not attenuated along the flow-path toward Greens Mill Run, then those infiltrated nitrogen loads may be contributed to the creek after the storm. The highest load reduction occurred during storm 5, the storm that had an 82% volume reduction (Table 2). The second highest load reduction occurred during storm 6, where there was a 95% reduction in outflow volume (Table 2). Volume reductions of 39% and 51% have been reported for two dry basins in Raleigh, NC [21] and 80% for a dry basin in Ithaca, New York [22]; thus, prior research has shown that volume reductions can be significant in DBs but are also highly variable between sites.

The observed nitrogen treatment efficiency for the DB with regard to reduction in EMC (21%) and nitrogen masses (up to 52.5%) exceeded the NC DEQ credit of 10% for dry detention basins [29]. Therefore, the DB with wetland characteristics was performing better than expected by the state of NC for a dry basin.

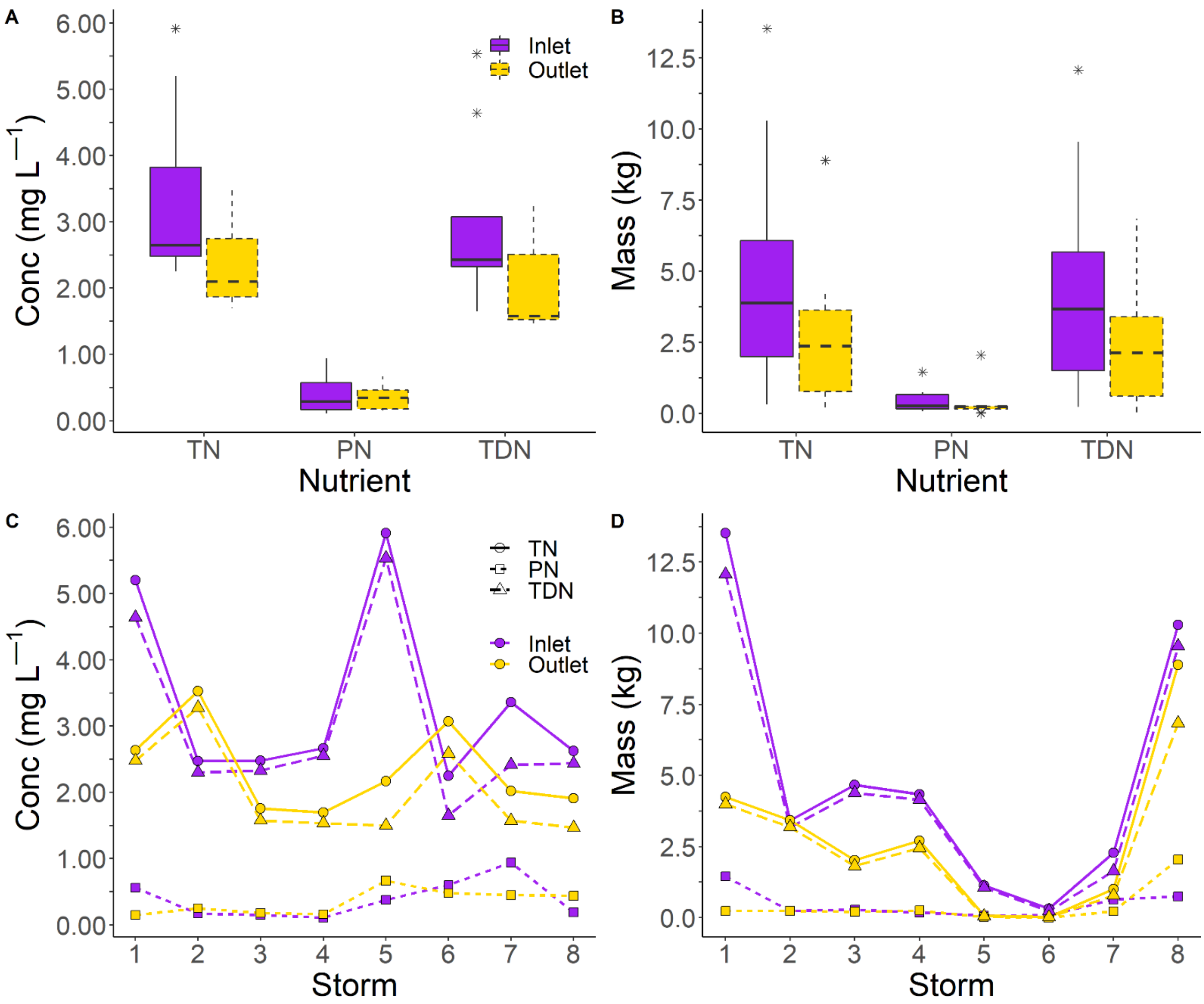

Figure 6. Box plots of event mean concentrations (**A**) and mass loadings (**B**) of total nitrogen (TN), particulate nitrogen (PN), and total dissolved nitrogen (TDN) (**A**,**C**). Time series of event mean concentrations (**C**) and mass loadings (**D**) of TN, PN, and TDN. Statistical outliers are shown as (*).

3.3. *Dissolved and Particulate Nitrogen Treatment*

Differences in the EMCs of TDN and PN between inflow and outflow were not statistically significant ($p = 0.06$, and $p = 0.542$, respectively). Most of the data, though, indicate that the DB was, in general, more efficient at reducing EMCs of dissolved nitrogen (median 35% efficiency) in comparison to particulate nitrogen (median −21% efficiency) (Figure 7). The EMCs of TDN leaving the DB were lower relative to the inflow EMCs during 75% of the storm events sampled, while the EMCs of PN in outflow exceeded inflow 62.5% of the times sampled (Figure 7). Influent and effluent data for individual storms were highly variable. The influent EMCs of TDN and PN ranged from 1.65 to 5.54 mg L^{-1} and 0.11 to 0.94 mg L^{-1}, respectively, while the effluent EMCs were between 1.47 mg L^{-1} and 3.28 mg L^{-1} for TDN and 0.15 mg L^{-1} and 0.67 mg L^{-1} for PN. The PN concentrations for influent may have been influenced by deciduous vegetation within the watershed and season during which the sampling occurred. For example, the highest EMC of PN in DB inflow (0.94 mg L^{-1}) was observed during storm 7, sampled in November during the middle of the fall season and with antecedent rain of 3.63 cm during the prior week. The antecedent rain may have contributed to leaf fall and mobilization of particulates and organic debris to the storm drainage network. The lowest EMCs of PN in DB influent were observed during storm 4 (0.11 mg L^{-1}), sampled in September. This sampling event also occurred with significant antecedent rainfall (3.99 cm) but was prior to when deciduous trees typically lose their leaves. Dissolved nitrogen concentrations were higher relative to particulate concentrations for both the influent and effluent during all eight storms.

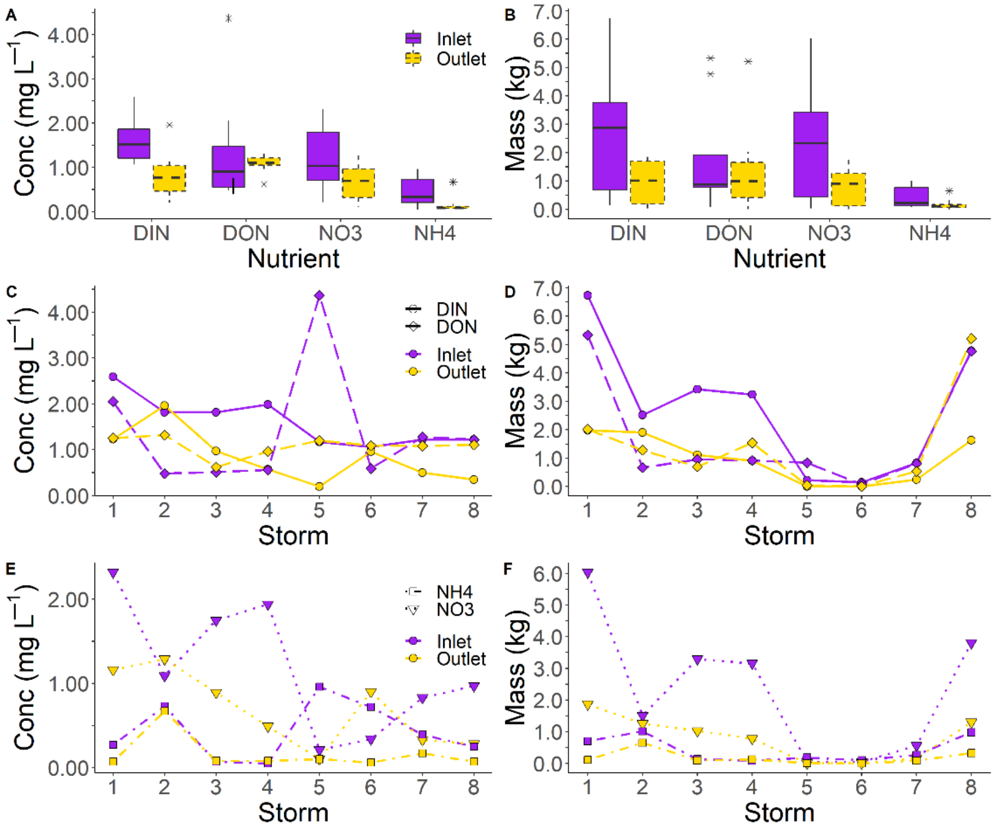

Figure 7. Box plots (**A**,**B**) and time series (**C**–**F**) of event mean concentrations and mass loadings of dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON), NO_3^-, and NH_4^+ at the inlet and outlet of the dry detention basin. Statistical outliers are shown as (*).

Significant differences were observed when comparing the inflow and outflow loadings of TDN (p = 0.020) but not PN (p = 0.564). The mean reduction in mass of TDN and PN for the eight storms was 55.4% and 14.8%, respectively. The mass reductions for individual storms ranged from −0.1% (storm 2) to 95.2% (storm 5) for TDN and −175% (storm 8) to 96% (storm 6) for PN (Figure 6).

TDN is the sum of DON and dissolved inorganic (DIN), which includes the species NO_3^- and NH_4^+. The median EMC of DIN in DB outflow (0.77 mg L^{-1}) was 49.7% lower relative to inflow (1.52 mg L^{-1}) (Figure 7A), and differences in the EMCs were statistically significant (p = 0.012). A 21.6% increase in the EMC of DON was observed when comparing median outflow (1.10 mg L^{-1}) to inflow (0.91 mg L^{-1}) (Figure 6A), but no significant differences (p = 0.643) were observed between the EMCs. These data reveal that the DB was efficient at lowering the EMCs of DIN but not DON. The EMCs of NO_3^- and NH_4+ were reduced, on average, by 21.8% and 39.7%, respectively, when comparing effluent concentrations to influent concentrations (Figure 7A), but significant differences between inflow and outflow EMCs were not observed for either species (NO_3^- p = 0.077; NH_4^+ p = 0.126) when compared individually. The reductions in EMCs varied greatly for individual storms (Figure 7E) and ranged from −165% to 74.7% for NO_3^-, and from −60% to 91.7% for NH_4^+. The reductions in EMCs of NH_4^+ and NO_3^- during storm 6 were 91.7% and −164.7%, respectively, the highest and lowest efficiencies reported for each species. The EMC of NH_4^+ decreased by 0.66 mg L^{-1} between the inlet and outlet, while the EMC of

NO_3^- increased by 0.56 mg L^{-1} during storm 6, possibly due to nitrification. Nitrification is an important process that allows for later gaseous loss of NO_3^- via denitrification if the environmental conditions are conducive to the transformation. Both NO_3^- and NH_4^+ are plant- and microorganism-useable forms of nitrogen and thus may be immobilized by plants and microbes and converted to ON [47]. Immobilization would result in an initial reduction in DIN concentrations but may contribute to the later export of dissolved and particulate forms of organic nitrogen (ON) as plant and microbe materials expire and decay [47]. These processes may have occurred to some degree in the DB as decreases in the EMCs of DIN were observed during seven of the eight storms (Figure 7C), and increases in the EMCs of PN and DON were observed during four of the eight storms (Figures 6C and 7C).

The influent loading of DIN exceeded the exports during each of the eight storm events (Figure 7D). Significant ($p = 0.020$) differences were observed between the influent and effluent loads of DIN. The mean treatment efficiency with regard to DIN mass reductions was 70.3%, with a range of 24.3% to 97%. The masses of NO_3^- and NH_4^+ leaving the basin were lower relative to the inflow masses during 100% and 87.5% of the sampling events, respectively. The mean load reduction efficiencies for NO_3^- and NH_4^+ were 68.1% and 53.2%, respectively (Figure 7B), and the differences in loading were also statistically significant (NO_3^- $p = 0.028$; NH_4^+ $p = 0.024$). The masses of PN leaving the basin were lower than influent masses during five of the eight storms (Figure 6D). The mean treatment efficiency with regard to reduction in PN mass was 14.8% (Figure 6B), but the differences between the influent and effluent loads of PN were not significant ($p = 0.564$). The masses of DON entering the DB were greater relative to masses exiting the basin during five of the eight storms (Figure 7D). The DON loads were reduced, on average, by 16.9% within the basin (Figure 7B), but the differences in influent and effluent loads were not statistically significant ($p = 0.713$). The outflow from the DB had a higher percentage of the TN as particulate relative to the inflow (Figure 8A,B). Additionally, the outflow had a higher percentage of the TDN as ON relative to the inflow (Figure 8C,D). These data indicate that the DB was not effective at reducing EMCs or masses of organic (dissolved and particulate) forms of nitrogen.

The median EMCs of DOC leaving the DB were 24% lower relative to the inflow concentrations, but the differences were not statistically significant ($p = 0.7984$) (Figure 9A). During 62.5% of the sampling events, the inflow EMCs of DOC exceeded outflow (Figure 9C). DOC is important as an energy source for denitrification and gaseous loss of N [19,20,48]. The median concentrations of DOC (14.65 mg L^{-1}) and NO_3^- (0.69 mg L^{-1}) leaving the DB and the resulting DOC to NO_3^- ratio of 21:1 indicate that carbon would not have limited denitrification [20]. However, DOC exports may contribute to water quality degradation via increasing the biological oxygen demand of receiving waters, resulting in a depletion of dissolved oxygen [2,4,48]. Water quality impairment related to depletion of dissolved oxygen has been a problem in the nutrient sensitive coastal waters of North Carolina [5]. Prior research [19] has shown that constructed stormwater wetlands often export higher EMCs of DOC relative to dry basins, so that too may be a consideration when selecting an SCM. The mass loadings of DOC exiting the DB were not significantly different ($p = 0.564$) in comparison to the inflow loadings (Figure 9B). The loads of DOC entering and exiting the DB were variable, but, during seven of the eight storm events, DOC load reductions were observed (Figure 9D). The mean load reduction was 34.6% (Figure 9B).

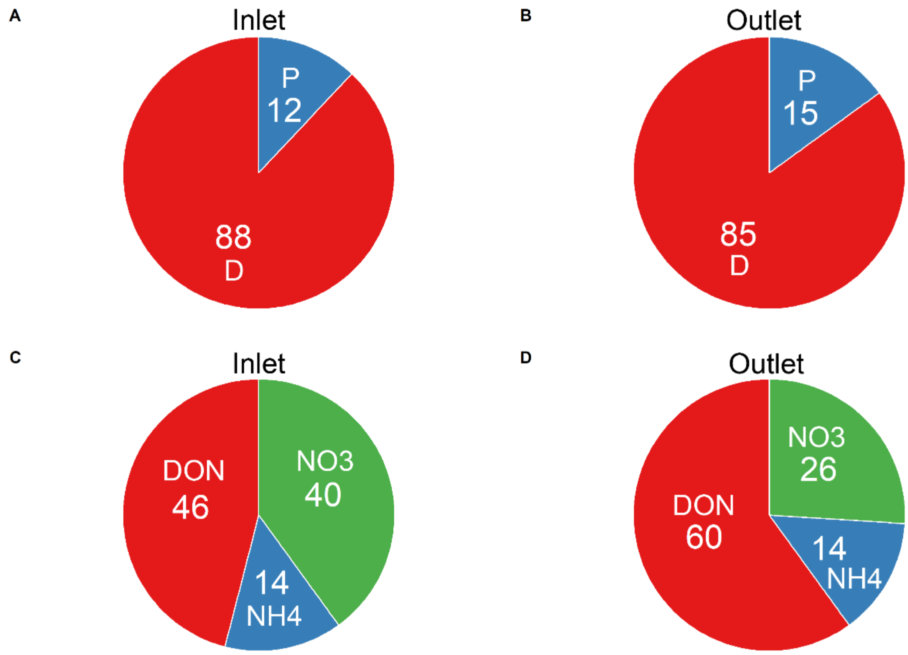

Figure 8. Speciation of nitrogen including percentage of total nitrogen in dry basin inflow (**A**) and outflow (**B**) that was particulate (P) and dissolved (D) nitrogen. The percentage of dissolved nitrogen for inflow (**C**) and outflow (**D**) that was organic (DON), nitrate (NO_3^-), and ammonium (NH_4^+) is also shown.

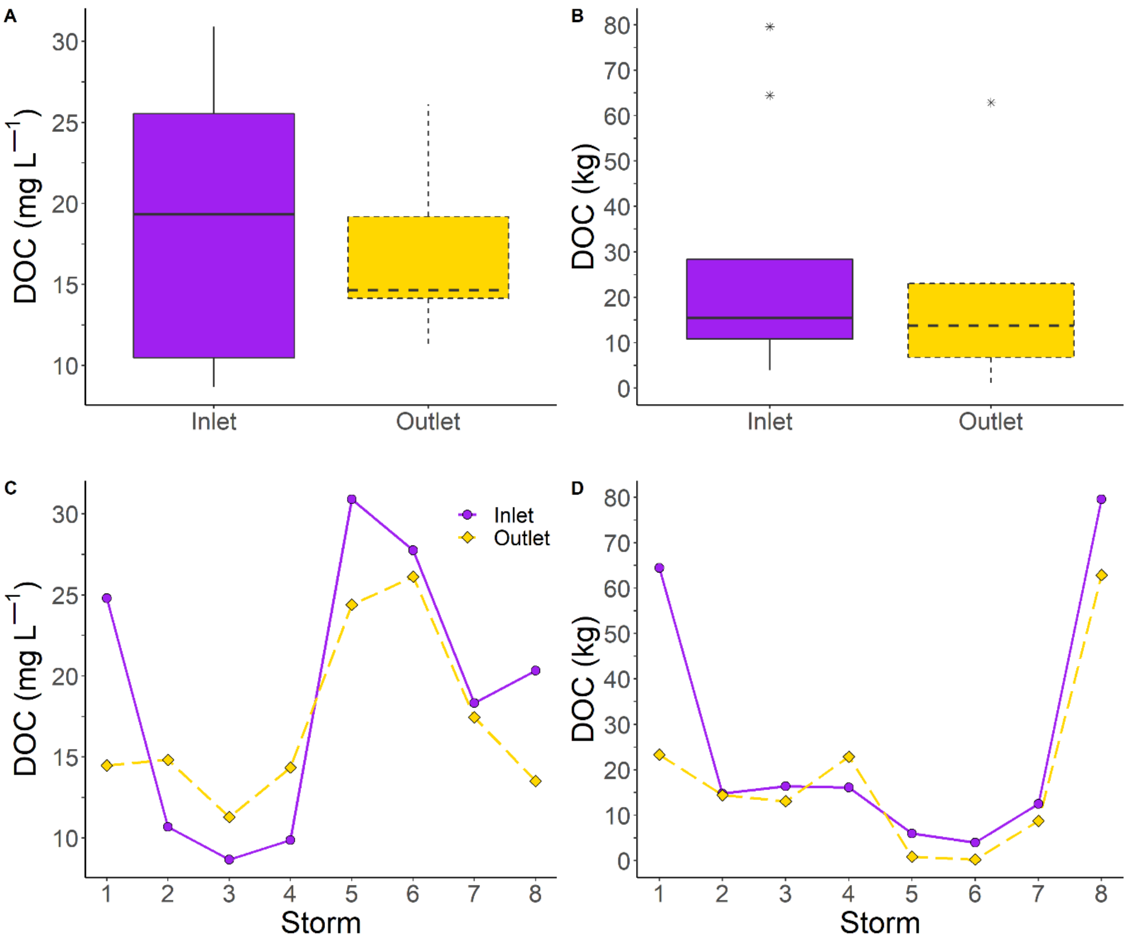

Figure 9. Event mean concentrations (**A**,**C**) and masses (**B**,**D**) of dissolved organic carbon (DOC) at the inlet and outlet of the dry detention basin. Statistical outliers are shown as (*).

3.4. Nitrogen Transformations

While the DB was not efficient at lowering EMCs or masses of PN and DON, the overall TN efficiency still exceeded expectations with regard to the reduction credit for dry basins assumed by NC DEQ. This was possible because other (than immobilization) nitrogen transformations, such as nitrification and denitrification, may have occurred in the DB to lower the mass. Nitrification is the conversion of NH_4^+ to NO_3^-, and the process takes place when aerobic microorganisms oxidize ammonium and reduce oxygen [15,20,49]. Denitrification is the conversion of NO_3^- to N_2 or N_2O gas, and the process is facilitated by microorganisms that live in anaerobic environments that contain labile carbon as an energy source [15,19,22]. Thus, if the DB has microenvironments between the inlet and outlet where alternating aerobic and anaerobic conditions exist, then concentrations of both NH_4^+ via nitrification and NO_3^- via denitrification may be reduced [15,20,22]. High frequency (10-min intervals) data show that, when runoff entered the DB and submersed the sensors, DO concentrations at the inlet increased, while DO concentrations at the outlet decreased (Figure 10A,B). During submergence of the sensors, the DO concentrations at the inlet were mostly greater than 4 mg L^{-1}, indicative of oxic conditions, while the DO concentrations near the outlet were often anoxic and dropped below 1.0 mg L^{-1} (Figure 10). Overall, the stormflow entering the DB during the eight sampled storms had a median DO concentration of 5.8 mg L^{-1}, indicating oxic conditions near the inlet. However, the median concentration of DO leaving the basin at the outlet (2.2 mg L^{-1}) was 62% lower relative to the inflow concentration, and the differences in DO were statistically significant ($p < 0.001$). Thus, DO was being consumed within the DB. The median pH of the stormwater inflow was 6.8, but the outflow was more acidic (5.4), and the differences were statistically significant ($p < 0.001$). High frequency pH data during a storm in September show an increase in pH near the inlet but a drop in pH near the outlet when the stage rises (Figure 10C). When NH_4^+ is converted to NO_3^-, hydrogen is released and may cause a drop in pH [19]. Nitrification may account for the decrease in NH_4^+ concentrations and pH between the inlet and outlet of the DB [18,19]. The mean pH of DB outflow was within the range of pH conditions (3.5–6.0) for nine wetlands monitored in the coastal plain of North Carolina [50].

The DB was ultimately effective at reducing the EMCs of TN, possibly because the conditions within the SCM were conducive to nitrogen transformations, including nitrification and denitrification that allowed for gaseous loss of nitrogen to the atmosphere. Because DIN concentration reductions offset the frequent DON and PN concentration increases, there was a net reduction in the EMC of dissolved nitrogen and TN as stormwater passed through the DB. The DB influent was comprised of 88% dissolved and 12% particulate nitrogen species. DIN accounted for 54% of the TDN, or 48% of the TN, while organic forms of nitrogen, including particulate and dissolved, accounted for 52% of the TN. The percentage of influent TN that was ON in our study (52%) was similar to the percentage (48%) reported in a literature review [18] of stormwater projects, but lower relative to the percentage (72%) from a study [49] conducted in Maryland, USA. Had more of the inflow been PN or DON, the treatment efficiency regarding lowering the EMC of TN may have been lower.

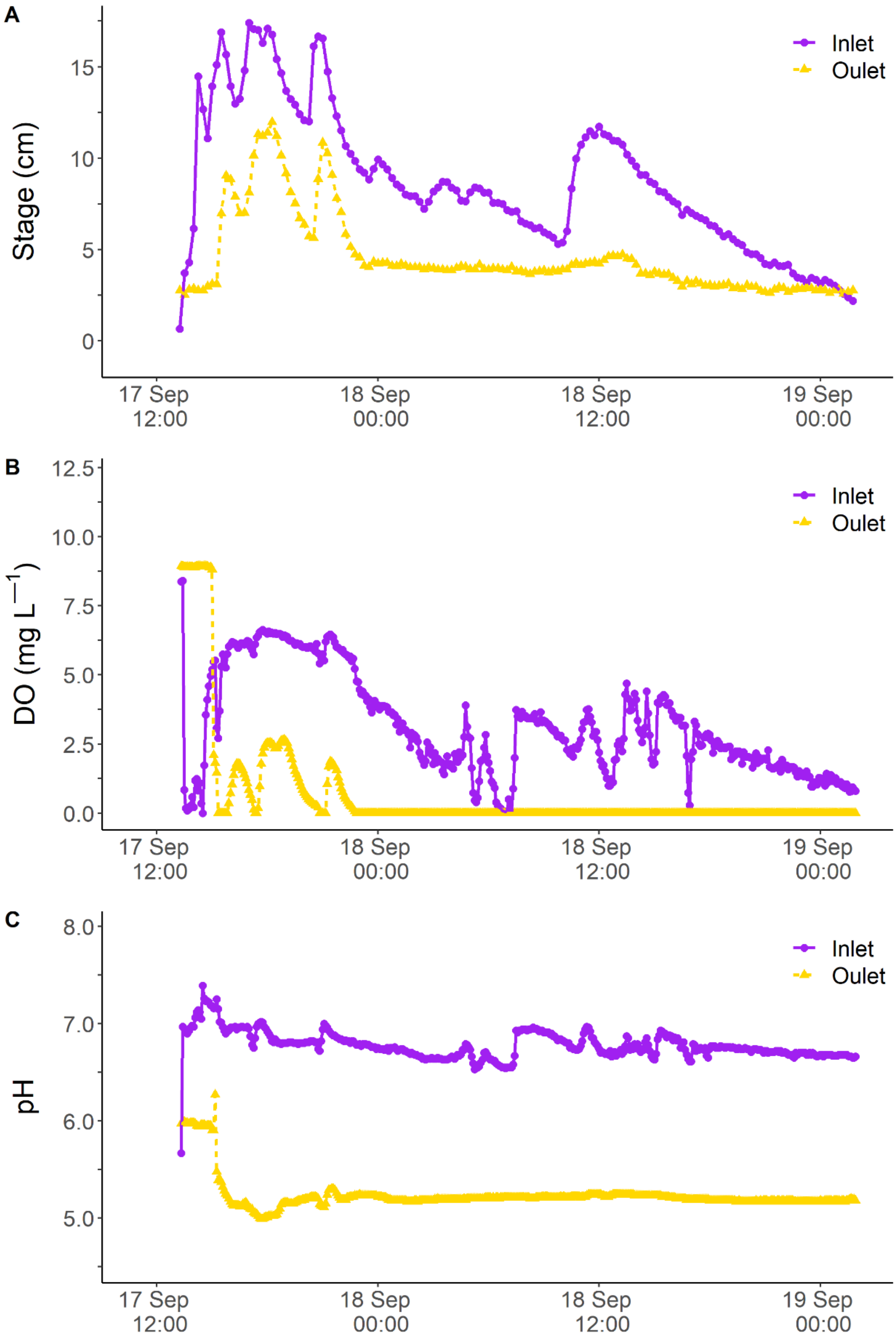

Figure 10. High-frequency water stage (**A**), dissolved oxygen concentration (**B**), and pH (**C**) of water at the inlet and outlet of the dry detention basin during a storm event in September 2020.

4. Conclusions

The goal of this study was to assess the nitrogen treatment efficiency of a DB that had saturated soils near the inlet and was partially (40%) covered with obligate wetland vegetation, and, thus, could be described as a hybrid between a DB and a constructed stormwater wetland. The outflow from the DB had a median EMC and mass of TN that were 21% and 52.3% lower, respectively, than the inflow. The biochemical characteristics of the water changed as it passed through the DB. More specifically, the EMCs of DIN declined and the EMCs of ON frequently increased, while the pH and DO also declined within the DB. The inlet portion of the DB was densely vegetated with obligate wetland plants, and the physicochemical properties of water near the inlet and outlet suggest that immobilization, nitrification, and denitrification were possible mass removal mechanisms in the DB. The DB was not effective at treating particulate and organic forms of nitrogen, but it was effective at treating dissolved inorganic forms, which comprised almost half of the influent TN. Mean outflow volume reductions of 33% were observed during the warm months between May and October, contributing to significant load reductions of TN within the DB. Temperature was inversely correlated with outflow volumes; thus, during the warmer months, there was relatively less outflow through the DB discharge pipe compared to the colder months for similar-size storms. The inflow and outflow volumes showed direct and significant correlations, so increased outflow can be expected during larger storm events with higher rainfall totals and more inflow. More research is suggested to better quantify the fate of nitrogen in stormwater that infiltrates the bottom of SCMs. Overall, the DB with wetland characteristics more than doubled the 10% TN efficiency credit expected for a DB for which it was originally designed and labeled. Because the grassed open space of the DB is not used for recreation, there are plans to convert the DB into a full stormwater wetland with a more diverse community of wetland plants. The *Typha* that have colonized the DB have provided sufficient ground cover to prevent erosion and gullies and have likely contributed to nitrogen uptake and transformations. However, some research has suggested that monocultures of *Typha* may provide a good mosquito breeding habitat, and that incorporating a diversity of wetland plants may encourage mosquito predators [51]. There are many other DBs in the Greens Mill Run watershed and other regions that could be retrofitted with wetland characteristics to enhance the TN treatment while still accommodating recreation. These modifications would likely be less expensive than a complete conversion to a stormwater wetland or wet detention basin and should be considered by resource managers.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/hydrology9050085/s1, Figure S1: Stage responses at the inlet and outlet of the dry basin in response to rain events; Figure S2: Rating curves for the inlet and outlet; Figure S3: Soil map, including dry basin with Goldsboro (GoA) soils near the inlet and Wagram (WaB) covering the rest.

Author Contributions: Co-authors of this article contributed in the following ways. Conceptualization, C.P.H.J. and G.I.; methodology, C.P.H.J. and G.I.; software, G.I. and C.P.H.J.; validation, C.P.H.J., G.I. and M.N.; formal analysis, C.P.H.J., G.I. and M.N.; investigation, M.N., C.P.H.J. and G.I.; resources, C.P.H.J. and G.I.; data curation, M.N., C.P.H.J. and G.I.; writing C.P.H.J. and G.I.; writing—review and editing, G.I. and C.P.H.J.; visualization, G.I. and C.P.H.J.; supervision, G.I. and C.P.H.J.; project administration, C.P.H.J.; funding acquisition, C.P.H.J. and G.I. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by the NC Department of Environmental Quality, grant number 8060. The APC was waived.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Summarized data are contained within the article. Raw data may be requested from corresponding author.

Acknowledgments: The authors would like to acknowledge John Gill, the City of Greenville, NC, Sound Rivers, and others for supporting this project.

Conflicts of Interest: The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript, or in the decision to publish the results.

References

1. Conley, D.J.; Paerl, H.W.; Howarth, R.W.; Boesch, D.F.; Seitzinger, S.P.; Havens, K.E.; Lancelot, C.; Likens, G.E. Controlling eutrophication: Nitrogen and phosphorus. *Science* **2009**, *323*, 1014–1015. [CrossRef] [PubMed]
2. Brooks, B.W.; Lazorchak, J.M.; Howard, M.D.; Johnson, M.-V.V.; Morton, S.L.; Perkins, D.A.; Reavie, E.D.; Scott, G.I.; Smith, S.A.; Steevens, J. Are harmful algal blooms becoming the greatest inland water quality threat to public health and aquatic ecosystems? *Environ. Toxicol. Chem.* **2016**, *35*, 6–13. [CrossRef] [PubMed]
3. Szabo, S.; Scheffer, M.; Roijackers, R.; Waluto, B.; Mihaly, B.; Nagy, P.T.; Borics, G.; Zambrano, L. Strong growth limitation of a floating plant (*Lemna gibba*) by the submerged macrophyte (*Elodea nuttallii*) under laboratory conditions. *Freshw. Biol.* **2010**, *55*, 681–690. [CrossRef]
4. Coffin, M.R.S.; Courtenay, S.C.; Pater, C.C.; Heuvel, M.R.V.D. An empirical model using dissolved oxygen as an indicator for eutrophication at a regional scale. *Mar. Pollut. Bull.* **2018**, *133*, 261–270. [CrossRef]
5. Paerl, H.W. Controlling Eutrophication along the Freshwater-Marine Continuum: Dual Nutrient (N and P) Reductions are Essential. *Estuaries Coasts* **2009**, *32*, 593–601. [CrossRef]
6. McLellan, E.L.; Cassman, K.G.; Eagle, A.J.; Woodbury, P.B.; Sela, S.; Tonitto, C.; Marjerison, R.D.; Es, H.M.V. The nitrogen balancing act: Tracking the environmental performance of food production. *BioScience* **2018**, *68*, 194–203. [CrossRef]
7. LeBleu, C.; Dougherty, M.; Rahn, K.; Wright, A.; Bowen, R.; Wang, R.; Orjuela, J.A.; Britton, K. Quantifying thermal characteristics of stormwater through low impact development systems. *Hydrology* **2019**, *6*, 16. [CrossRef]
8. US EPA. Source Water Protection (SWP): Common Considerations. 2022. Available online: https://www.epa.gov/sourcewaterprotection/common-considerations (accessed on 10 February 2022).
9. Bastia, J.; Mishra, B.K.; Kumar, P. Integrative assessment of stormwater infiltration practices in rapidly urbanizing cities: A case of Lucknow City, India. *Hydrology* **2021**, *8*, 93. [CrossRef]
10. Hardison, E.C.; O'Driscoll, M.A.; DeLoatch, J.P.; Howard, R.J.; Brinson, M.M. Urban land use, channel incision, and water table decline along coastal plain streams, North Carolina. *J. Am. Water Resour. Assoc.* **2009**, *45*, 1032–1046. [CrossRef]
11. Ballinas-Gonzalez, H.A.; Alcocer-Yamanaka, V.H.; Canto-Rios, J.J.; Simuta-Champo, R. Sensitivity analysis of the rainfall-runoff modeling parameters in data-scarce urban catchment. *Hydrology* **2020**, *7*, 73. [CrossRef]
12. Zeiger, S.J.; Hubbart, J.A. Urban stormwater temperature surges: A central US watershed study. *Hydrology* **2015**, *2*, 193–209. [CrossRef]
13. Beitinger, T.L.; Bennett, W.A.; McCauley, R.W. Temperature tolerances of North American freshwater fishes exposed to dynamic changes in temperature. *Environ. Biol. Fish.* **2000**, *58*, 237–275. [CrossRef]
14. Gaborit, E.; Muschalla, D.; Vallet, B.; Vanrolleghem, P.A.; Anctil, F. Improving the performance of stormwater detention basins by real-time control using rainfall forecasts. *Urban Water J.* **2013**, *10*, 230–246. [CrossRef]
15. McPhillips, L.; Walter, M.T. Hydrologic conditions drive denitrification and greenhouse gas emissions in stormwater detention basins. *Ecol. Eng.* **2015**, *85*, 67–75. [CrossRef]
16. Middleton, J.R.; Barrett, M.E. Water quality performance of a batch-type stormwater detention basin. *Water Environ. Res.* **2008**, *80*, 172–178. [CrossRef] [PubMed]
17. Pezzanti, D.; Beecham, S.; Kandasamy, J. Stormwater detention basin for improving road-runoff quality. *Water Manag.* **2012**, *165*, 461–471. [CrossRef]
18. Collins, K.A.; Lawrence, T.J.; Stander, E.K.; Jontos, R.J.; Kaushal, S.S.; Newcomer, T.A.; Grimm, N.B.; Ekberg, M.L.C. Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis. *Ecol. Eng.* **2010**, *36*, 1507–1519. [CrossRef]
19. Humphrey, C.P., Jr.; Iverson, G. Reduction in nitrogen exports form stormflow after conversion of a dry detention basin to a stormwater wetland. *Appl. Sci.* **2020**, *10*, 9024. [CrossRef]
20. Bettez, N.D.; Groffman, P.M. Denitrification potential in stormwater control structures and natural riparian zones in an urban landscape. *Environ. Sci. Technol.* **2012**, *46*, 10909–10917. [CrossRef]
21. Wissler, A.D.; Hunt, W.F.; McLaughlin, R.A. Hydrologic and water quality performance of two aging and unmaintained dry detention basins receiving highway stormwater runoff. *J. Environ. Manag.* **2020**, *255*, 109853. [CrossRef]
22. Morse, N.R.; McPhillips, L.E.; Shapleigh, J.P.; Walter, M.T. The role of denitrification in stormwater detention basin treatment of nitrogen. *Environ. Sci. Technol.* **2017**, *51*, 7928–7935. [CrossRef] [PubMed]
23. Gold, A.G.; Thompson, S.P.; Piehler, M.F. Seasonal variation in nitrate removal mechanisms in coastal stormwater ponds. *Water Resour. Res.* **2021**, *57*, e2021WR029718. [CrossRef]
24. Iverson, G.; O'Driscoll, M.; Humphrey, C., Jr.; Manda, A.; Anderson-Evans, E. Wastewater nitrogen contributions to coastal plain watersheds, NC, USA. *Water Air Soil Pollut.* **2015**, *226*, 325. [CrossRef]
25. O'Driscoll, M.; DeWalle, D.; Humphrey, C.; Iverson, G. Groundwater seeps: Portholes to evaluate groundwater's in fluence on stream water quality. *J. Contemp. Water Res. Educ.* **2019**, *166*, 55–78. [CrossRef]
26. Vepraskus, M.J.; Faulkner, S.P. Redox Chemistry of Hydric Soils. In *Wetland Soils Genesis, Hydrology, Landscapes, and Classification*, 1st ed.; Richardson, J.L., Veprasksus, M.J., Eds.; Lewis Publishers: Boca Raton, FL, USA, 2001; pp. 85–105.

27. Schlesinger, W.H. Biogeochemistry in freshwater wetlands and lakes. In *Biogeochemistry an Analyses of Global Change*, 2nd ed.; Academic Press: San Diego, CA, USA, 1997; pp. 224–260.
28. Koch, B.J.; Febria, C.M.; Gevrey, M.; Wainger, L.A.; Palmer, M.A. Nitrogen removal by stormwater management structures: A data synthesis. *J. Am. Water Resour. Assoc.* **2014**, *50*, 1594–1607. [CrossRef]
29. North Carolina Department of Environmental Quality. Stormwater Control Measure Credit Document. Available online: https://files.nc.gov/ncdeq/Energy%20Mineral%20and%20Land%20Resources/Stormwater/BMP%20Manual/SSW-SCM-Credit-Doc-20170807.pdf (accessed on 15 January 2022).
30. Hunt, W.F. *Urban Waterways, Urban Stormwater Structural Best Management Practices (BMPs)*; North Carolina Cooperative Extension Service Publication: Raleigh, NC, USA, 1999.
31. Weiss, P.T.; Gulliver, J.S.; Erickson, A.J. Cost and pollutant removal of storm-water treatment practices. *J. Water Resour. Plan. Manag.* **2007**, *133*, 218–229. [CrossRef]
32. US Climate Data. Climate Greenville—North Carolina. Available online: https://www.usclimatedata.com/climate/greenville/north-carolina/united-states/usnc0281 (accessed on 24 February 2022).
33. Natural Resources Conservation Service. Soils. Official Soil Series Description. Available online: https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/geo/?cid=nrcs142p2_053587 (accessed on 18 April 2022).
34. North Carolina Department of Environmental Quality. Integrated Report Files. Available online: https://deq.nc.gov/about/divisions/water-resources/water-planning/modeling-assessment/water-quality-data-assessment/integrated-report-files (accessed on 18 April 2022).
35. North Carolina Department of Environmental Quality, Stormwater Calculations. Available online: https://files.nc.gov/ncdeq/Energy%20Mineral%20and%20Land%20Resources/Stormwater/BMP%20Manual/B%20%20Stormwater%20Calculations.pdf (accessed on 24 February 2022).
36. USGS National Water Information System: Web Interface. USGS 02084000 Tar River at Greenville, NC. Available online: https://waterdata.usgs.gov/nwis/uv?site_no=02084000 (accessed on 24 February 2022).
37. Shimadzu Corporation. Total Organic Carbon Analyzer User's Manual. TOC-L TOC Analyzer. Available online: https://www.shimadzu.com/an/products/total-organic-carbon-analysis/toc-analysis/toc-l-series/index.html (accessed on 28 April 2022).
38. Solorzano, L. Determination of ammonia in natural waters by the phenol-hypochlorite method. *Limnol. Oceanogr.* **1969**, *14*, 799–801.
39. Westco Scientific Instruments, Inc. *SmartChem 200 Method 375-100E-1*; Westco Scientific Instruments, Inc.: Brookfield, WI, USA, 2008.
40. Westco Scientific Instruments, Inc. *SmartChem 200 Method 390-200E*; Westco Scientific Instruments, Inc.: Brookfield, WI, USA, 2008.
41. Clesceri, L.; Greenberg, A.; Eaton, A. *Standard Methods for the Examination of Water and Wastewater*, 20th ed.; American Public Health Association/American Water Works Association/Water Environment Federation: Washington, DC, USA, 1998.
42. Brooks, K.N.; Ffolliott, P.F.; Gregersen, H.M.; DeBano, L.F. *Hydrology and the Management of Watersheds*, 3rd ed.; Iowa State University Press/Ames: Ames, IA, USA, 2003; pp. 47–74.
43. Samanta, S.; Sheng, Z.; Munster, C.L.; Houtte, E.V. Seasonal variation of infiltration rates through pond bed in a managed aquifer recharge system in St-Andre, Belgium. *Hydrol. Processes* **2020**, *34*, 3807–3823. [CrossRef]
44. Humphrey, C.P., Jr.; O'Driscoll, M.A. Evaluation of soil colors as indicators of the seasonal high water table in coastal North Carolina. *Int. J. Soil Sci.* **2011**, *6*, 103–113. [CrossRef]
45. Sonderup, M.J.; Egemose, S.; Hansen, A.S.; Grudinina, A.; Madsen, M.H.; Flindt, M.R. Factors affecting retention of nutrients and organic matter in stormwater ponds. *Ecohydrology* **2016**, *9*, 796–806. [CrossRef]
46. Mazer, K.E. Converting a Dry Pond to a Constructed Stormwater Wetland to Enhance Water Quality Treatment. A Thesis Submitted to the Graduate Faculty of North Carolina State University. 2018. Available online: https://repository.lib.ncsu.edu/bitstream/handle/1840.20/35788/etd.pdf?sequence=1 (accessed on 28 April 2022).
47. Gorgoglione, A.; Gioia, A.; Iacobellis, V. A framework for assessing modeling performance and effects of rainfall-catchment drainage characteristics on nutrient urban runoff in poorly gauged watersheds. *Sustainability* **2019**, *11*, 4933. [CrossRef]
48. Lopez-Ponnada, E.V.; Lynn, T.J.; Ergas, S.J.; Mihelcic, J.R. Long-term field performance of a conventional and modified bioretention system for removing dissolved nitrogen species in stormwater runoff. *Water Res.* **2020**, *170*, 115336. [CrossRef] [PubMed]
49. Li, L.; Davis, A.P. Urban stormwater runoff nitrogen composition and fate in bioretention systems. *Environ. Sci. Technol.* **2014**, *48*, 3403–3410. [CrossRef] [PubMed]
50. Hartman, W.H.; Richardson, C.J.; Vilgalys, R.; Bruland, G.L. Environmental and anthropogenic controls over bacterial communities in wetland soils. *PNAS* **2008**, *105*, 17842–17847. [CrossRef]
51. Hunt III, W.F.; Aperson, C.S.; Kennedy, S.G.; Barrison, B.A.; Lord, W.G. Occurrence and relative abundance of moquitoes in stormwater retention facilities in North Carolina, USA. *Water Sci. Technol.* **2006**, *54*, 315–321. [CrossRef]