

# Effects of watershed-scale green infrastructure retrofits on urban stormwater quality: A paired watershed study to quantify nutrient and sediment removal

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## ABSTRACT

Urban stormwater represents a substantial source of nutrients and sediment to aquatic ecosystems. Green infrastructure (GI), including bioretention and permeable pavement, is an increasingly utilized method to treat stormwater pollutants. Using soil and plants as natural filters, these systems are effective at the site scale, but little evidence exists regarding their performance at the watershed-scale. Blueprint Columbus is an effort by the City of Columbus, Ohio, USA to retrofit GI to eliminate sanitary sewer overflows and remove 20% of total suspended solids (TSS) from existing developed areas. Changes in water quality resulting from the combined effects of many GI practices installed in 11.5 and 47.8 ha treatment watersheds were quantified using a paired-watershed approach. Based on water quality data collected by automated samplers over a 3.5-year period, significant reductions in particulate and dissolved nutrients as well as sediment were observed following the installation of GI in both treatment watersheds compared to the control. Total nitrogen (TN), phosphorus (TP), and TSS concentrations decreased by 13.7–24.1%, 20.9–47.4%, and 61.6–67.7%, respectively. Runoff attenuation by GI contributed to pollutant load reductions of 24.0–25.4% (TN), 27.8–32.6% (TP), and 59.5–78.3% (TSS). Orthophosphate concentrations and loads increased in the watershed with bioretention only but significantly decreased in the treatment watershed with bioretention and permeable pavement. Reductions in TSS concentration were similar (within a margin of 5%) to the percent of the watershed imperviousness treated by GI. Results demonstrate that GI was effective in reducing runoff event mean concentrations and loads at the watershed-scale.

## 1. Introduction

Urbanization results in the construction of impermeable surfaces, leading to higher volumes of impaired runoff (Goonetilleke et al., 2005) which negatively affect biodiversity and ecosystem functions (Wu et al., 2011), public health (Hathaway et al., 2015; Lim et al., 2015), the economy (Hellman et al., 2018), and ecosystem services (Marsalek and Rochfort, 2004; Stepenuck et al., 2002). Urban stormwater runoff is a substantial conveyance for anthropogenic and natural pollutants, both dissolved and particulate, to waterways. Fertilizers, vehicle emissions, and human and animal waste are sources of dissolved nutrients, namely

nitrogen (N) and phosphorus (P) compounds, in urban areas (Brinkmann, 1985; Yang and Toor, 2018). Construction sites, roads, and erosion are sources of sediment (i.e., particulates) (Ellis et al., 1987). Nutrients and sediment from urban runoff fuel eutrophication in receiving water bodies, resulting in rapid algal growth and hypoxia (Browman et al., 1979; Silva et al., 2019).

Low impact development (LID; Ahlblade et al., 2012; Dietz, 2007) is a land development strategy that mitigates the negative impacts of urbanization by employing stormwater control measures (SCM) to mimic pre-development hydrology. Green infrastructure (GI) SCMs, which passively improve stormwater quality and reduce stormwater

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runoff through infiltration and evapotranspiration (Chen et al., 2019), are often part of LID strategies. Bioretention cells, perhaps the most commonly used GI SCM, rely on an engineered sandy soil media and various plant species to provide at-source treatment of urban stormwater runoff (DeBusk et al., 2011; Hsieh and Davis, 2005; Wang et al., 2017). Nitrogen removal in bioretention cells occurs through sedimentation of organic, particulate-bound N (Lusk et al., 2020), denitrification in the anoxic internal water storage (IWS; i.e., an upturned elbow in the underdrain providing storage for inter-event exfiltration) zone (if employed; Collins et al., 2010a, 2010b; Kim et al., 2003; Lopez-Ponnada et al., 2020) and through plant uptake (Muerdter et al., 2019; Shrestha et al., 2018). Aerobic bioretention soil media facilitates nitrification, wherein ammonia is converted to nitrate (Fan et al., 2019). Bioretention cells reduce particulate P through sedimentation and filtration, and dissolved P through either plant uptake or sorption to iron or aluminum oxides on the surface of clays and silt (Hunt et al., 2012; Lijklema, 1980; Muerdter et al., 2019; Song and Song, 2019). Total suspended solids (TSS) in urban stormwater runoff are removed by bioretention through filtration and sedimentation (Trowsdale and Simcock, 2011). Bioretention effectively mitigates pollutant load by not only reducing pollutant concentrations but also attenuating stormwater runoff volumes via exfiltration to in situ soils and evapotranspiration (Davis et al., 2009; Winston et al., 2016b).

Permeable pavement, which consists of a porous surface course underlain by layers of open-graded aggregate, is another form of GI that reduces the deleterious effects of roads and parking lots on the urban water cycle (Brattebo and Booth, 2003). Unlike traditional pavement, permeable pavement permits runoff to infiltrate into the subsurface, providing opportunities for runoff volume reduction, groundwater recharge, and water quality improvement (Braswell et al., 2018; Roseen et al., 2012; Scholz and Grabowiecki, 2007; Tirpak et al., 2020; Winston et al., 2018). Through filtration and sedimentation, permeable pavement traps organic and inorganic particulates in stormwater (Kamali et al., 2017). Nitrification occurs in the aerobic aggregate voids and denitrification has been observed if IWS is employed (Bean et al., 2007; Braswell et al., 2018; Brown et al., 2009; Winston et al., 2016a). Particulate P removal occurs through sedimentation, while adsorption and transformation by microorganisms drive organic P removal (Sansalone et al., 2008; Tota-Maharaj and Scholz, 2010a, 2010b).

GI design differs across the world because local communities have unique expectations, local governments have various standards and funding abilities, and differences in regional climatic and historic factors, pollutant runoff rates, geologic and soil conditions, and receiving waterways (IPWEA, 2017). In the USA alone, standards that guide GI design vary from state to state (U.S. EPA, 2016). In the state of Ohio, the 2016 state-wide standard for newly developed and redeveloped sites was for GI to treat the 19-mm water quality volume for development sites of 0.4 ha or greater. By treating the water quality volume, it is assumed that bioretention treats 80% of total suspended solids. This design standard was increased from 19-mm to 23-mm in 2019.

Chinese sponge cities, Australia's Little Stringybark Creek, and select studies in cities in the USA are the only studies we have found that attempt to quantify the water quality changes resulting from GI implementation at watershed-scales (Yin et al., 2022; Walsh et al., 2021). This is evidence that installing GI retrofits at the watershed-scale is rare but becoming more widespread. In 2013 China launched its sponge city program, which uses LID for urban water management at large scales, with 30 pilot cities underway by 2016 (Nguyen et al., 2019). Past studies have shown sponge cities to improve water quality at the site scale. Bioretention demonstrated average pollutant removal rates of 73.9%, 72.0%, and 79.2% for total N (TN), total P (TP), and TSS, respectively; permeable pavement demonstrated 53.0%, 57.0%, and 34.9% pollutant removal rates for those same pollutants (Xu, 2020; Yin et al., 2022).

Among the limited studies that measure the water quality changes from GI retrofits at the watershed-scale, even fewer use nearby control watersheds to account for annual and seasonal variability. To

summarize those, nearly two decades of water quality monitoring in six catchments retrofitted with SCMs throughout Australia's Little Stringybark Creek revealed significant reductions in N and P, but not TSS to receiving streams (Walsh et al., 2021). Concentrations of TN, TP, and TSS were reduced by 58%, 38%, and 82%, respectively, in runoff from a 0.53 ha, residential catchment with sandy underlying soils in North Carolina following the installation of an in-street bioretention cell, four permeable pavement parking stalls, and a tree filter treating 91% of the catchment (Page et al., 2015). Pollutant loads for TN, TP, and TSS decreased by 79%, 72%, and 91%, respectively. However, a 1.7 ha residential LID watershed in Connecticut with permeable pavement and twelve bioretention cells demonstrated a reduction in N, while TSS and TP concentrations and loads increased due to stormwater flow through and fertilization of grass swales (Bedan and Clausen, 2009). As GI is designed for local conditions, the study presented herein builds upon these past studies to quantify the efficacy of retrofits at the watershed-scale using a control watershed. Findings will provide guidance to municipalities seeking to adopt GI practices and understand the totality of their benefits.

The City of Columbus devised a 40-year wet weather management plan in 2005 as a response to two different consent decrees from the Ohio Environmental Protection Agency (EPA): one for combined sewer overflows (CSOs) and one for sanitary sewer overflow (SSOs). Gray infrastructure was chosen to address the CSOs. In 2015, the Ohio EPA approved Blueprint Columbus, the City of Columbus's updated plan to address SSOs using GI retrofits. SSOs result from infiltration and inflow of stormwater into sanitary sewer systems and degrade water quality in urban areas (Field and O'Connor, 1997). With the goal of reducing SSOs, the City of Columbus, Ohio, has retrofitted hundreds of GI practices into a single neighborhood as part of Blueprint Columbus (Pawlowski et al., 2014). This study assesses changes in stormwater quality following GI implementation across two watersheds in Columbus, Ohio, compared to a nearby control watershed. The research objectives are to 1) to quantify changes in nutrients and sediment resulting from the implementation of GI at the watershed-scale, 2) discuss fundamental processes responsible for these water quality changes, and 3) provide recommendations on the efficacy of GI retrofits and future work to improve the outcomes and monitoring of projects at this scale.

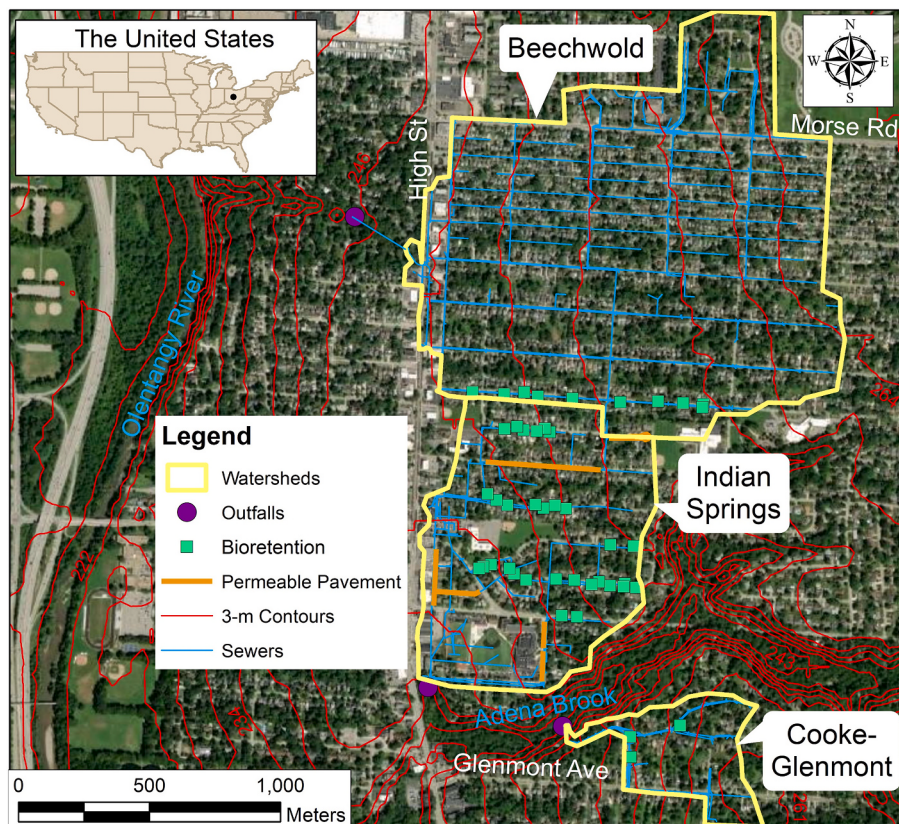
## 2. Materials and methods

### 2.1. Site description

Three watersheds, Cooke-Glenmont, Indian Springs, and Beechwold, in the Clintonville neighborhood of Columbus, Ohio, USA were monitored for stormwater hydrology and water quality at their respective storm sewer outfalls (Fig. 1). The proportion and types of land use in each watershed are presented in Table 1. Residential areas in the watersheds consisted of small, single-lot, single family parcels constructed between 1910 and 1950. Soils in the neighborhood were mapped as silt loam in the Cardington and Bennington soil series (Table 1; NRCS, 2019). The region experiences four distinct seasons, with a minimum mean average temperature normal in January of  $-5.5^{\circ}\text{C}$ , and a maximum mean average temperature normal in June of  $24^{\circ}\text{C}$  (based on the 30-year historical record in central Ohio; NOAA, 2020). Annual precipitation averages 1000 mm, including an annual average of 70 mm of snowfall.

The control and treatment watersheds were located within a 1 km radius (Fig. 1). Since the vast majority of the 111.5 ha Beechwold watershed was not retrofitted with GI (GI treated 2.1% of the watershed area), it served as the control for the study (Table 1). Statistical testing was performed to show this GI did not significantly affect water quality in Beechwold (see Section 3.1) and confirmed the appropriateness for its use as an experimental control.

Residential land use dominated the watersheds, with interspersed commercial and institutional land uses (Table 1). Imperviousness ranged



**Fig. 1.** Control (Beechwold) and treatment (Indian Springs and Cooke-Glenmont) watersheds, storm sewers, and locations of GI features in the Clintonville neighborhood of Columbus, Ohio. Watershed delineations were completed using the sewer network; however, 3-m contour lines show the flow path from the watersheds, through tributaries (i.e., Adena Brook), and into the receiving waterway (i.e., Olentangy River).

from 30.9% at Cooke-Glenmont (which also contained 21.6% forested area) to 40.3% at Indian Springs. Indian Springs had the highest percentage of institutional land use (17.4%) which represented substantial directly connected impervious areas (i.e., large roofs and parking lots draining to the storm sewer) in the watershed. The three largest sources of imperviousness in the watersheds were roofs (12.5–15.7%), roads (8.6–11.0%), and driveways (6.4–8.3%). Indian Springs had curbed roads, while, aside from a single road, roads in the Cooke-Glenmont watershed lacked curbs.

Different types and densities of GI were retrofitted into the watersheds. Thirty-two bioretention cells and four permeable pavement roads/alleys were constructed in the Indian Springs watershed over approximately a 1-year period. GI practices made up 1.73% of the watershed surface area, treating 69.7% of the watershed imperviousness. Bioretention treated 23.6% of the watershed area, adding 0.19 mm of watershed storage, while the remaining treatment was provided by permeable pavement (Table 1). Cooke-Glenmont, the smallest watershed (11.5 ha), had two large bioretention cells and a third, smaller bioretention cell retrofitted within its boundaries. Together, this GI accounted for 0.38% of the watershed area, treating 30.1% of its surface area and 66.5% of its impervious area. These three cells added 1.63 mm of watershed storage. Beyond GI, additional stormwater treatment in the watershed may have occurred as a portion of stormwater runoff from roads likely discharged through the forested area in the Cooke-Glenmont watershed.

## 2.2. GI Retrofits

Bioretention was designed to treat the 2016 water quality volume in Ohio of 19 mm. Among the 32 bioretention cells retrofitted into Indian

Springs, 11 extended into the street to intercept runoff along the curb. The remaining 21 cells were located behind the curb in front of houses. All bioretention cells in this watershed contained approximately 60 cm of 2–5% organic matter (by mass) loamy sand media installed above underdrains located at the bottom of the cross-section (i.e., no IWS) that were surrounded by 30 cm of aggregate. The median surface and drainage areas of the bioretention cells at Indian Springs were 8.74 m<sup>2</sup> and 0.26 ha, respectively. Bioretention cells were planted with (1) grasses, especially *Calamagrostis acutiflora*, *Panicum virgatum*, and *Penisetum alopecuroides*, (2) perennials including *Asclepias tuberosa*, *Iris versicolor*, and *Rudbeckia fulgida*, and (3) shrubs such as *Hypericum frondosum*. Four roads/alleys were retrofitted with curb-to-curb permeable interlocking concrete pavement with a total surface area of 6000 m<sup>2</sup>, representing 1.26% of the watershed. The permeable pavements contained 65–150 cm of aggregate with underdrains located at the bottom of the cross-section (i.e., no IWS).

The large bioretention cells in the Cooke-Glenmont watershed were installed in series. The upstream cell had a surface area of 152.1 m<sup>2</sup> (approximately 0.13% of the watershed area) and treated 15.8% of the watershed. A second, larger (255.4 m<sup>2</sup>) bioretention cell was located downstream of this cell and treated effluent from the upstream cell. The surface area and drainage area of the downstream cell represented 0.22% and 24.7% of the Cooke-Glenmont watershed, respectively. The smallest (28.1 m<sup>2</sup>) bioretention cell at Cooke-Glenmont occupied 0.02% and treated 5.4% of the watershed. Each of the Cooke-Glenmont bioretention cells included 60 cm of media underlain by 30 cm of aggregate surrounding an underdrain located at the bottom of the cross-section (i.e., no IWS) and had 30 cm of bowl storage. Bioretention cells were planted with (1) grasses and sedges, such as *Panicum virgatum* and *Carex morrowii*, and (2) perennials, especially *Polygonatum odoratum*,

**Table 1**  
Watershed characteristics.

| Watershed      | Area (ha) | Land Use (%) |            |               | Imperviousness (% of Watershed Area) | Imperviousness Treated by GI (% of Imperviousness) | % Watershed Area Draining to Bioretention | Added Storage Provided by Bioretention (mm) |
|----------------|-----------|--------------|------------|---------------|--------------------------------------|----------------------------------------------------|-------------------------------------------|---------------------------------------------|
|                |           | Residential  | Commercial | Institutional |                                      |                                                    |                                           |                                             |
| Beechwold      | 111.5     | 95.7         | 3.6        | 0.7           | 38.2                                 | 2.4                                                | 2.1                                       | 0.0075                                      |
| Indian Springs | 47.8      | 75           | 7.6        | 17.4          | 40.3                                 | 69.7                                               | 23.6                                      | 0.19*                                       |
| Cooke-Glenmont | 11.5      | 100          | 0          | 0             | 30.9                                 | 66.5                                               | 30.1                                      | 1.63                                        |

Note: Land surveying performed using RTK GPS to measure added storage from bioretention in the watershed in late 2019 through early 2020.

\* Although permeable pavement in this watershed contained an underdrain without internal water storage, it still provided temporary storage, which is not accounted for in this value.

**Table 2**

Construction and monitoring timeline.

| Watershed           | Project Phase        |                     |
|---------------------|----------------------|---------------------|
|                     | Pre-GI               | Post-GI             |
| Cooke-Glenmont*     | 9/29/2016-8/31/2017  | 9/11/2017-8/14/2019 |
| Indian Springs**    | 9/30/2016-11/31/2017 | 10/1/2018-8/31/2019 |
| Beechwold (Control) | Monitoring Period    |                     |
|                     | 9/29/2016            | 12/9/2019           |

\*Pre-GI phase at Cooke-Glenmont includes construction phase.

\*\*Secondary construction projects were happening at Indian Springs during post-GI.

*Rudbeckia fulgida*, and *Symphyotrichum ericoides*. Bioretention in both treatment watersheds utilized the same media which met Ohio's bioretention media guidance (OEPA, 2006).

### 2.3. Construction and monitoring timeline

GI construction occurred on different timelines in each treatment watershed (Table 2). Monitoring was divided into three phases: 1) pre-GI,

2) construction (i.e., the period in which GI was built), and 3) post-GI. The pre-GI phase refers to the monitoring period before the construction of GI, while the post-GI phase corresponds to the period after GI construction was finalized and treatment systems were operational. Additional infrastructure improvements associated with Blueprint Columbus (i.e., redirecting downspouts, implementing sump pumps, and lining sanitary sewer laterals) occurred at Indian Springs during the post-GI phase, as detailed in Boening-Ulman et al. (2022).

As the goal of this work was to assess differences in water quality between the pre- and post-GI project phases, data from the construction phase were not analyzed for Indian Springs. However, the construction phase was combined with the pre-GI phase at Cooke-Glenmont for the following reasons: The first water quality sample at Cooke-Glenmont was not collected until September 29, 2016, and the last sample before construction commenced was collected on December 6, 2016. Combining these project phases supplemented the roughly two months of pre-GI data with 7.5 months of additional data, allowing for improved statistical comparisons of changes in water quality from pre- to post-GI retrofit. This larger pre-GI data set also allows the impacts of seasonal changes in water quality to be captured (Smith et al., 2020). Including the construction phase with the pre-GI phase for Cooke-Glenmont was further justified using the Kruskal-Wallis test (Section 3.1).

### 2.4. Experimental design and data collection

This study utilized a “before-after, control-impact” paired watershed design to determine if water quality changes were due to the implementation of GI or other external factors, such as changes in climate or pollutant generation in the watersheds (Green, 1993; Page et al., 2015; Shuster and Rhea, 2013). “Before-after” refers to the time periods before and after GI was installed; “control-impact” refers to the control and GI-treatment watersheds. To support this analysis, data were collected during the periods listed in Table 2, except for winter months (mid-December to mid-March) when monitoring was discontinued to prevent damage to equipment.

Smaller storm events of <5.1 mm were not sampled since they yielded insufficient runoff for laboratory analysis; furthermore, instances of equipment malfunction precluded sampling; otherwise, all storm events were sampled. At Beechwold, Cooke-Glenmont, and Indian Springs, 102, 67, and 40 storm events, respectively, were sampled for water quality during the project phases (Table 2). Storm events sampled for water quality represented 39.1–67.5% of the total observed rainfall. The median event depth for sampled storms ranged from 14.0 to 16.8 mm across the watersheds, slightly greater than the median event depth

**Table 3**

Laboratory Methods, Preservation Procedures, and Method Detection Limits.

| Parameter                 | Abbreviation    | Laboratory Method                                     | Preservation                                  | MDL (mg/L) |
|---------------------------|-----------------|-------------------------------------------------------|-----------------------------------------------|------------|
| Total Kjeldahl Nitrogen   | TKN             | EPA Method 351.2 <sup>a</sup>                         | H <sub>2</sub> SO <sub>4</sub> (<2 pH), <4 °C | 0.078      |
| Nitrite                   | NO <sub>2</sub> | EPA Method 353.2                                      | H <sub>2</sub> SO <sub>4</sub> (<2 pH), <4 °C | 0.018      |
| Nitrate                   | NO <sub>3</sub> | EPA Method 353.2                                      | <4 °C                                         | 0.043      |
| Total Nitrogen            | TN              | Calculated<br>TKN + NO <sub>2</sub> + NO <sub>3</sub> | NA                                            | NA         |
| Total Ammoniacal Nitrogen | TAN             | EPA Method 350.1                                      | H <sub>2</sub> SO <sub>4</sub> (<2 pH), <4 °C | 0.0031     |
| Organic Nitrogen          | ON              | Calculated as<br>TKN-TAN                              | NA                                            | NA         |
| Orthophosphate            | OP              | EPA Method 365.2                                      | <4 °C                                         | 0.01       |
| Particle-bound Phosphorus | PBP             | Calculated as<br>TP-OP                                | NA                                            | NA         |
| Total Phosphorus          | TP              | EPA Method 365.2                                      | <4 °C                                         | 0.1        |
| Total Suspended Solids    | TSS             | Standard<br>Methods<br>2540D <sup>b</sup>             | <4 °C                                         | 2          |

<sup>a</sup> U.S. EPA (1983).<sup>b</sup> APHA et al. (2012).

for observed storms of 9.7–10.4 mm. This can be attributed to minimum sample volumes required for laboratory analyses, which prevented the analysis of smaller storms. The median 5-min peak rainfall intensity for sampled storms, which ranged between 13.7 and 18.3 mm/h across the watersheds, was similar to the median peak intensity recorded for all observed storms (13.7–15.2 mm/h). The median ADP for sampled events (2.7–3.7 days) was similar to the median ADPs from all observed events in the watersheds (3.1–3.3 days).

A 0.254 mm resolution tipping bucket (Davis Rain Collector) and a manual rain gauge were deployed within (or adjacent to) the control and treatment watersheds. The rain gauges were attached to 2-m tall wooden posts and installed in locations free from overhead obstructions. Rainfall data from the tipping bucket rain gages were stored on a 1-min interval on Hobo Pendant data loggers (Onset Computer Corporation, Bourne, Massachusetts).

All storm sewer outfalls were equipped with an area velocity meter (AVM; Teledyne Isco, Lincoln, Nebraska) to continuously monitor velocity and depth of flow. These data, along with pipe cross-sectional geometry, were utilized by an area velocity module to determine flow rate on 1-min (Cooke-Glenmont and Beechwood) or 2-min intervals (Indian Springs). Manual measurement of water level in the outfall was used to calibrate the AVM approximately weekly.

Flow rate was integrated with time to determine runoff volume; sample aliquots were triggered based upon runoff volume and subsequently collected by automated samplers (Teledyne Isco, Lincoln, Nebraska). Using rainfall depth measured at the manual rain gauge, automated sampler pacing was re-calibrated following each sampling event to ensure representative, runoff volume proportional samples. Samples were composited and consisted of five to 50, 350 mL aliquots which described >80% of the pollutograph (U.S. EPA, 2002).

Because of the short antecedent dry period (ADP; six hours) used to separate rainfall events, one to three samples from each watershed represented the water quality of multiple hydrologic events. This occurred when the flow had not returned to baseflow before the onset of the next rainfall event, causing the samplers to combine two storms within the composite bottle. In these cases, the separate hydrologic events were combined for pollutant concentration and load analysis.

## 2.5. Laboratory techniques

After fully suspending particulates by vigorously shaking the composite bottle to ensure a representative event mean concentration (EMC), composite samples were divided among a 500 mL pre-acidified bottle for total ammoniacal nitrogen (TAN), total Kjeldahl nitrogen (TKN), and nitrite (NO<sub>2</sub>) analysis, a 500 mL bottle for nitrate (NO<sub>3</sub>) and TSS analysis, and a 60 mL bottle (following field filtration through a 0.45 µm filter) for orthophosphate (OP) analysis. Samples were collected within 24 h of the cessation of rainfall, placed on ice (<4 °C), and transported to the laboratory.

Organic nitrogen (ON), TN and particle-bound phosphorus (PBP) concentrations were calculated using methods in Table 3. Nitrate-nitrate (NO<sub>2-3</sub>) concentrations were calculated as the sum of nitrate and nitrite concentrations for each sampled event. Samples were analyzed using either U.S. EPA (1983) or American Public Health Association (APHA et al., 2012) methods.

## 2.6. Data analysis

Summary statistics for pollutant concentrations were determined using laboratory-reported EMCs. A value of one-half the method detection limit (MDL) was substituted for concentrations below MDL (Table 3; Antweiler and Taylor, 2008); all concentrations above MDL were analyzed without transformation. Orthophosphate at Indian Springs (16.7% below MDL); TP at and Indian Springs (14.6% below MDL); and TSS at all three watersheds (Beechwood 11.4% below MDL; Cooke-Glenmont 18.4% below MDL; Indian Springs 29.7% below MDL) had >10% of concentrations below MDL. For all other analytes, concentrations below MDL occurred in <10% of sampled storm events.

Two substantial outliers were removed from the nitrate (and subsequently TN) data sets. On November 1, 2017 at Beechwood and May 21, 2017 at Cooke-Glenmont, nitrate concentrations of 850 and 780 mg/L, respectively, were reported by the laboratory; these were 2 orders of magnitude higher than the next highest nitrate concentration (9.5 mg/L).

Summary statistics derived from the data set included the number of observed events, median pollutant concentrations, and pollutant loads during each phase (i.e., pre-GI, or post-GI) for control and treatment watersheds. Observed changes in water quality due to GI implementation were compared to similar studies, including those in the International Stormwater BMP Database (ISBMPD; Clary and Jones, 2017).

Pollutant loads from each watershed were determined as the product of pollutant EMC and runoff volume on a storm-by-storm basis. Pollutant loads were reported on a watershed area-normalized basis and were calculated using the following equation:

$$L_{i,j} = \frac{EMC_{i,j} \times V_j}{1000 \times A_{WS}} \quad (1)$$

where  $L_{i,j}$  is the load of pollutant  $i$  (g/ha) for storm event  $j$ ,  $EMC_{i,j}$  is the event mean concentration of pollutant  $i$  (mg/L) for storm event  $j$ ,  $V_j$  is the measured runoff volume for storm event  $j$  (L), and  $A_{WS}$  is the watershed area (ha).

Annual loading ( $L_a$ , kg/ha/year) was estimated by accounting for storms not sampled for water quality. The ratio of long-term (i.e., 30 year) average annual rainfall depth for Columbus, Ohio ( $RF_{LTA}$ ; mm/yr) to total rainfall depth sampled for water quality ( $RF_{SAMP}$ ; mm) was utilized to scale the annual loading (Eq. 2).

$$L_a = \frac{\sum_{j=1}^n (EMC_{i,j} \times V_j) \times RF_{LTA}}{1000 \times A_{WS} \times RF_{SAMP}} \quad (2)$$

where  $n$  is the number of sampled storm events. In this calculation, it is assumed that the sampled storm events are representative of the overall population of runoff volume and pollutant concentration. To determine the effects of different event depths on annual load, bins of event depth were created and the load within each bin was summed. This load was then scaled by the ratio of the total rainfall depth to the total sampled rainfall depth within each bin to estimate the total load by event depth bin. Annual loading was calculated separately for each project phase to allow for comparison between pre-GI and post-GI phases.

All data analysis was completed using R statistical software version 3.4.2 (R Core Team, 2020). Except where noted, a criterion of 95% confidence ( $\alpha = 0.05$ ) was used. The Wilcoxon rank sum test was used to determine if sampled event characteristics varied from those of all observed events. Comparisons of rainfall characteristics across the watersheds and project phases were performed using the Kruskal-Wallis test.

Water quality data were log transformed, after which the Shapiro-Wilk test was used to check for normality of model residuals. When model residuals were normally distributed, demonstrated homoscedasticity, and showed no multicollinearity, analysis of covariance (ANCOVA) was used to compare treatment to control data. ANCOVA analysis was utilized to uncover significant differences in the slopes and intercepts of concentrations and loads (Page et al., 2015). If the residuals were not normally distributed, yet the sample size was large ( $>30$ ), parametric statistical analysis methods were still used (Ghasemi and Zahediasl, 2012) since sample sizes were considered large enough to approximate the population. When ANCOVA slopes were significantly different, the pollutant was related to storm characteristics using linear regression. When a significant linear relationship did not exist for water quality parameters during the pre-GI phase, but was present for the post-GI phase, a paired  $t$ -test was used to compare control and treatment watersheds during the post-GI phase. No statistical analyses were performed when data lacked either an adequate sample size or a significant linear relationship. Percent changes in pollutant concentration and storm event load were calculated and reported using least squares mean (LSM) analysis (Page et al., 2015):

$$\text{Change (\%)} = \left( \frac{10^{\bar{Y}_{\text{Post}}}}{10^{\bar{Y}_{\text{Pre}}}} - 1 \right) \times 100 \quad (3)$$

where  $\bar{Y}_{\text{Post}}$  is the treatment watershed LSM during the post-GI phase, and  $\bar{Y}_{\text{Pre}}$  is the treatment watershed LSM during the pre-GI phase. Herein, concentration and storm event load percent differences observed after the adoption of GI refer to the LSM percent difference. Conversely, the percent difference between annual load by project phase is simply reported as percent change (since annual load is a single value, not a distribution that can be statistically tested).

### 3. Results and discussion

#### 3.1. Justification of experimental design

Statistical tests were conducted to determine if Beechwold was an appropriate control for Cooke-Glenmont and Indian Springs. Using data collected in the Beechwold watershed during the time frames corresponding with pre- and post-GI phases at Cooke-Glenmont (Table 2), the only observed significant difference was a decrease in TAN concentration; all other pollutant concentrations were not significantly different. Watershed-scale TAN concentrations have been shown to vary based on anthropogenic activities and microbes in lawns (Parkin, 1987; Raciti et al., 2011). Given that GI only covered 0.0074% of the Beechwold watershed, this result was likely not attributable to GI installation.

During the time frames corresponding with the pre- and post-GI phases at Indian Springs, a significant decrease in TSS concentration was observed at Beechwold. This difference was due to differences in

median 5-min peak rainfall intensity observed across project phases (pre-GI = 22.1 mm/h, post-GI = 13.0 mm/h; see Section 3.2) driving differences in sediment transport (Sharma et al., 2016; Gong et al., 2016). Since sparse GI installed at Beechwold only treated 2.1% of the watershed surface area and no other water quality parameters exhibited significant differences with project phase, it was concluded that the use of the Beechwold watershed as the experimental control was appropriate.

Combining the pre-GI and construction phases at Cooke-Glenmont increased the data set from seven to 25 storms, which resulted in a more robust dataset to compare to post-GI. Only two pollutants exhibited a significant difference in pollutant concentrations between the pre-GI and construction phases, namely TAN and OP. TAN and OP are primarily dissolved pollutants; previous studies attributed seasonal changes in these pollutants to the breakdown of organic material and seasonal mineralization (Hathaway et al., 2012; Yang and Toor, 2018). Construction of SCMs would not be expected to release TAN or OP, but rather TSS and associated particulate-bound pollutants due to exposure of bare soils (Alsharif, 2010; Atasoy et al., 2006; Müller et al., 2020; Smith et al., 2020). For all other pollutants, no significant difference in concentration was observed between pre-GI and construction project phases at Cooke-Glenmont; thus, these phases were combined for the analysis that follows.

#### 3.2. Rainfall characteristics of sampled events

The Wilcoxon rank sum test showed a significant difference existed between the rainfall depths of observed and sampled storms ( $p < 0.001$ ), but no significant differences existed for other rainfall characteristics (i.e., average intensity, peak 5-min intensity, ADP, and rainfall duration); thus, sampled storm events were representative of the distribution of observed storms. As found in Boening-Ulman et al. (2022), significant differences in peak 5-min rainfall intensity were observed at all watersheds (including at the control when the project time frames corresponded with those at the treatment watersheds), where greater rainfall intensities were observed during the pre-GI phases than the post-GI phases. Otherwise, no significant differences in the rainfall characteristics were observed between phases. Using the Kruskal-Wallis  $k$ -sample tests showed that rainfall characteristics did not significantly differ among the three watersheds ( $p > 0.68$ ; Kruskal and Wallis, 1952). This result was expected, as the watersheds were within 1 km of one another.

#### 3.3. Nitrogen

The median TAN concentration at Cooke-Glenmont was reduced by 64.0% with the installation of GI ( $p < 0.001$ ; Table 4). A 67.7% reduction in TAN load was also observed at Cooke-Glenmont ( $p < 0.001$ ). Runoff attenuation with the installation of GI (Boening-Ulman et al., 2022) and decreased TAN concentrations with transformations occurring in GI accounted for this decrease in storm event load. Aerobic environments within bioretention media promote nitrification, where TAN is biologically oxidized to  $\text{NO}_2$  and further oxidized to  $\text{NO}_3$  (Hunt et al., 2012; Wang et al., 2017; Osman et al., 2019). The significant reduction in TAN concentrations and insignificant increase in  $\text{NO}_3$  concentrations (by 5.9%) supports the occurrence of nitrification within bioretention at Cooke-Glenmont. Data from a number of studies summarized in the ISBMPD support the occurrence of nitrification in individual bioretention cells, with the median  $\text{NO}_3$  concentration increasing from 0.35 mg/L (influent) to 0.48 mg/L (effluent) in bioretention cells (Clary and Jones, 2017). Other watershed-scale studies also demonstrated nitrification in GI, with TAN concentrations and storm event loads decreasing 19–71%, and concurrent  $\text{NO}_3$  concentrations and storm event loads either not changing significantly or increasing up to 100% (Bedan and Clausen, 2009; Page et al., 2015).

Similar to Cooke-Glenmont, 60.5% and 47.1% reductions in TAN concentrations and storm event loads, respectively, were observed at

**Table 4**

Summary statistics for concentrations and storm event loads for pollutants by project phase at Cooke-Glenmont (CG) and Indian Springs (IS). Interpretations related to project phase were made using ANCOVA. Comparisons to the control were done using a *t*-test on the post-GI data only.

|                      | Pollutant | Site | Pre-GI Phase |                |                  | Post-GI Phase |                |                  | Statistical Results |          |                   |
|----------------------|-----------|------|--------------|----------------|------------------|---------------|----------------|------------------|---------------------|----------|-------------------|
|                      |           |      | n            | Control Median | Treatment Median | n             | Control Median | Treatment Median | LSM % Difference    | p-value  | Interpretation    |
| Concentration (mg/L) | TAN       | CG   | 20           | 0.110          | 0.122            | 47            | 0.078          | 0.062            | −64                 | 1.55E-04 | Pre-GI > Post-GI  |
|                      |           | IS   | 22           | 0.085          | 0.190            | 18            | 0.048          | 0.068            | −60.5               | 0.305    | NSD btw. Phases   |
|                      | TKN       | CG   | 18           | 1.25           | 1.54             | 44            | 0.99           | 1.20             | −14.1               | 0.002    | Pre-GI > Post-GI  |
|                      |           | IS   | 21           | 0.94           | 1.10             | 16            | 1.06           | 1.00             | −16.5               | 0.192    | NSD btw. Phases   |
|                      | ON        | CG   | 15           | 1.11           | 1.44             | 41            | 0.94           | 1.18             | −19.6               | 0.003    | Pre-GI > Post-GI  |
|                      |           | IS   | 18           | 1.09           | 0.89             | 16            | 1.00           | 0.82             | 3.1                 | 0.352    | NSD btw. Phases   |
|                      | Nitrate   | CG   | 10           | 0.63           | 0.58             | 32            | 0.78           | 0.63             | 5.9                 | –        | –                 |
|                      |           | IS   | 13           | 0.56           | 0.99             | 16            | 0.81           | 0.82             | −22.7               | –        | –                 |
|                      | TN        | CG   | 17           | 1.7            | 2.35             | 44            | 1.8            | 1.88             | −13.7               | 0.002    | Pre-GI > Post-GI  |
|                      |           | IS   | 21           | 1.74           | 1.91             | 18            | 1.94           | 1.81             | −24.1               | –        | –                 |
|                      | OP        | CG   | 11           | 0.12           | 0.14             | 30            | 0.11           | 0.13             | 23.7                | 0.123    | NSD with control  |
|                      |           | IS   | 14           | 0.11           | 0.09             | 15            | 0.11           | 0.07             | −25.2               | 5.68E-06 | Pre-GI > Post-GI  |
|                      | PBP       | CG   | 19           | 0.21           | 0.23             | 39            | 0.16           | 0.21             | −30.9               | 0.007    | CG > Control      |
|                      |           | IS   | 19           | 0.15           | 0.2              | 14            | 0.11           | 0.06             | −66.5               | –        | –                 |
|                      | TP        | CG   | 20           | 0.26           | 0.32             | 47            | 0.19           | 0.25             | −20.9               | 0.002    | CG > Control      |
|                      |           | IS   | 22           | 0.24           | 0.2              | 18            | 0.23           | 0.11             | −47.4               | –        | –                 |
| Load (g/ha)          | TSS       | CG   | 19           | 74             | 160              | 46            | 49             | 49               | −61.59              | 8.59E-05 | Pre-GI > Post-GI  |
|                      |           | IS   | 18           | 51             | 83               | 18            | 30             | 20               | −67.66              | –        | –                 |
|                      | TAN       | CG   | 16           | 5.55           | 5.62             | 47            | 2.3            | 2.38             | −67.7               | 2.67E-05 | Pre-GI > Post-GI  |
|                      |           | IS   | 19           | 3.12           | 8.7              | 16            | 2.33           | 5.44             | −47.1               | 0.118    | NSD btw. Phases   |
|                      | TKN       | CG   | 15           | 56             | 42.6             | 44            | 49.3           | 52.7             | −20.9               | 0.004    | Pre-GI > Post-GI  |
|                      |           | IS   | 19           | 54.9           | 81.8             | 14            | 50.4           | 62.8             | −19.9               | –        | –                 |
|                      | ON        | CG   | 13           | 49.3           | 46               | 41            | 46             | 48.2             | −25.5               | 0.006    | Pre-GI > Post-GI  |
|                      |           | IS   | 17           | 46             | 37               | 14            | 46             | 54.9             | −2.4                | –        | –                 |
|                      | Nitrate   | CG   | 9            | 41.5           | 16.8             | 32            | 23.5           | 20.2             | 7.7                 | 0.292    | NSD with control  |
|                      |           | IS   | 13           | 13.5           | 19.1             | 14            | 20.2           | 32.5             | −3                  | –        | –                 |
|                      | TN        | CG   | 14           | 84.1           | 77.3             | 44            | 65             | 63.9             | −24                 | 0.004    | Pre-GI > Post-GI  |
|                      |           | IS   | 18           | 60.5           | 113.2            | 16            | 65             | 103.1            | −25.4               | –        | –                 |
|                      | OP        | CG   | 10           | 5.01           | 2.08             | 29            | 3.47           | 2.93             | 18.1                | 0.776    | NSD with control  |
|                      |           | IS   | 13           | 2.45           | 4.15             | 13            | 2.06           | 2.02             | −46.9               | –        | –                 |
|                      | PBP       | CG   | 14           | 6.36           | 7.85             | 37            | 5.53           | 7.13             | −40.3               | 0.086    | Pre-GI > Post-GI★ |
|                      |           | IS   | 14           | 5.04           | 9.76             | 12            | 4.74           | 3.77             | −49.2               | –        | –                 |
| (kg/ha)              | TP        | CG   | 15           | 7.76           | 9.24             | 47            | 6.66           | 8.34             | −32.6               | 0.039    | Pre-GI > Post-GI  |
|                      |           | IS   | 18           | 6.56           | 8.36             | 16            | 7.78           | 9.47             | −27.8               | –        | –                 |
|                      | TSS       | CG   | 14           | 3.23           | 6.47             | 46            | 1.69           | 1.68             | −78.29              | 0.926    | NSD btw. Phase    |
|                      |           | IS   | 15           | 2.07           | 3.74             | 16            | 1.33           | 1.43             | −59.52              | –        | –                 |

Note: Star (★) implies significance at  $p < 0.10$ , NSD implies no significant difference, dash (–) indicates that statistical analyses were not performed due to insufficient sample size or data failing to meet model assumptions, while negative LSM % differences imply reduction. TAN: total ammoniacal nitrogen, TKN: total Kjeldahl nitrogen, ON: organic nitrogen, TN: total nitrogen, OP: orthophosphate, PBP: particle-bound phosphorus, TP: total phosphorus, and TSS: total suspended solids.

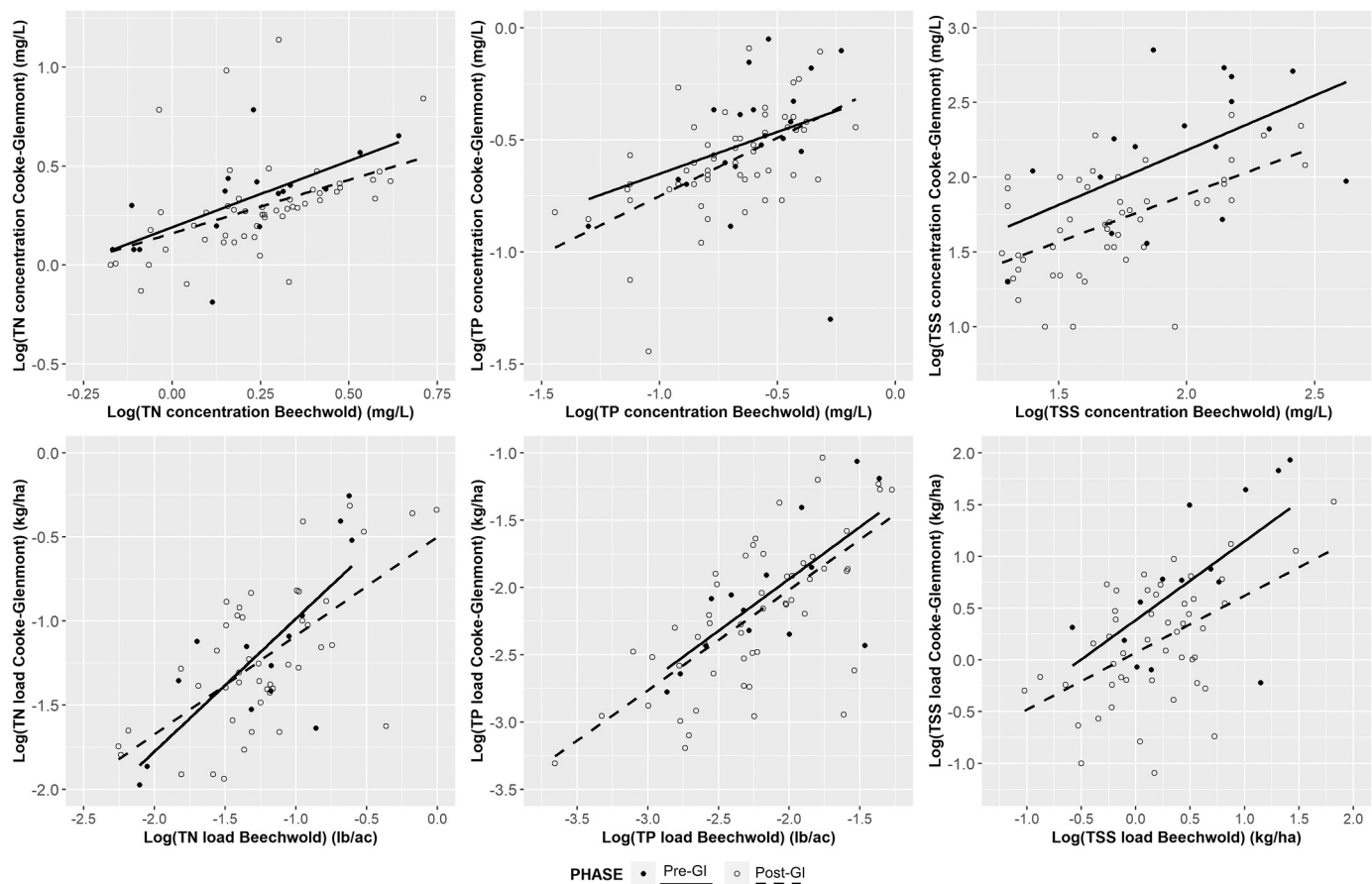
Indian Springs following the installation of GI. Along with bioretention, permeable pavement practices in the Indian Springs watershed allowed for nitrification due to the aerobic environment in the pavement sub-surface (Collins et al., 2010a, 2010b; Tota-Maharaj and Scholz, 2010a, 2010b). Data presented in the ISBMPD also supports the occurrence of nitrification in permeable pavement, as median  $\text{NO}_3$  concentrations increased from 0.59 to 1.36 mg/L from influent to effluent (Clary and Jones, 2017).

$\text{NO}_3$  concentrations and storm event loads did not meet statistical model assumptions, so statistical analyses could not be performed for either treatment watershed. At Cooke-Glenmont and Indian Springs, a 5.9% increase and 22.7% reduction in median  $\text{NO}_3$  concentration, respectively, were observed following the installation of GI (Table 4). Other LID studies found no or insignificant reductions in  $\text{NO}_3$  concentrations and storm event loads (Bedan and Clausen, 2009; Page et al., 2015). Dissolved nutrients, including  $\text{NO}_3$ , are often ineffectively removed by bioretention cells with traditional media; thus,  $\text{NO}_3$  removal may have been bolstered were initial media mixtures amended with materials such as coconut coir or biochar (Tirpak et al., 2020). Further

reductions may be possible were IWS zones or other restrictions to GI drainage, which have been shown to promote anaerobic conditions and subsequent denitrification (Hsieh et al., 2007; Page et al., 2015), retrofitted in Cooke-Glenmont and Indian Springs to remove  $\text{NO}_3$ . Despite this performance, it is anticipated that  $\text{NO}_3$  uptake by the bioretention cells in the treatment watersheds will improve as the plant roots and microbial communities become more established in the systems (discussed further in Section 3.7; Hopkinson and Giblin, 2008).

TKN concentrations and storm event loads decreased by 14.1% (to 1.20 mg/L) and 20.9%, respectively, at Cooke-Glenmont ( $p < 0.005$ ), and 16.5% (to 1.00 mg/L) and 19.9%, respectively, at Indian Springs ( $p > 0.05$ ) with the installation of GI (Table 4). Bedan and Clausen (2009) also saw a significant 76% reduction in TKN concentration to 1.0 mg/L. With median TKN effluent concentrations of 1.39 and 1.00 mg/L for single bioretention cells and permeable pavement reported in the ISBMPD, respectively (Clary and Jones, 2017), further TKN reduction beyond that observed herein is unlikely.

Since ON encompassed 92.8% of TKN at Cooke-Glenmont, ON reductions were similar to those of TKN. ON concentration decreased



**Fig. 2.** ANCOVA models for total nitrogen (TN), total phosphorus (TP), and total suspended solids (TSS) concentrations and storm event loads at Cooke-Glenmont (CG). Pre-GI TP concentrations did not fit a linear model. For all cases except TP concentration and TSS storm event load, significant differences were observed in the intercepts. Significant differences in slopes were observed for TSS concentration and storm event load ( $p < 0.005$ ).

19.6% ( $p < 0.005$ ) and storm event load decreased by 25.5% ( $p < 0.01$ ) between the pre- and post-GI phases at Cooke-Glenmont (Table 4). Particulate ON removal has been shown to occur in bioretention primarily through sedimentation and filtration processes (Li and Davis, 2014). Page et al. (2015) also saw significant decreases of 62% and 79% in TKN concentrations and loads, respectively, with the installation of GI, attributing this change to ON retention by bioretention. At Indian Springs, ON concentrations and storm event loads changed by 3.2 and –2.4%, respectively. ON and TAN made up 86% and 22%, respectively, of TKN at Indian Springs.

This difference in ON mitigation between treatment watersheds may be related to watershed characteristics. Cooke-Glenmont had the least imperviousness and most forested areas; Indian Springs was the opposite (Table 1). The greater impervious coverage and less vegetative cover at Indian Springs may have decreased the availability of ON in the watershed. This is supported by lower ON concentrations at Indian Springs, 0.89 and 0.82 mg/L for pre- and post-GI, respectively, than at Cooke-Glenmont, 1.44 and 1.18 mg/L for pre- and post-GI, respectively (Table 4). Concentrations of ON at Indian Springs may have reached an irreducible level (Hathaway and Hunt, 2010).

TN concentrations were significantly ( $p < 0.005$ ) reduced by 13.7% between the pre- and post-GI phases at Cooke-Glenmont (Table 4; Fig. 2). Similar to concentrations, TN storm event loads were significantly reduced (by 24.0%) at Cooke-Glenmont ( $p < 0.005$ ). Stormwater runoff attenuation provided by bioretention and N conversions within GI are responsible for this observed TN removal. No significant difference was observed in the slopes for TN concentration or storm event load at Cooke-Glenmont (Fig. 2), indicating that bioretention exhibited similar TN removal for both small and large storm events.

Reductions in TN concentrations and storm event loads between the pre- and post-GI phases at Indian Springs (24.1% and 25.4%, respectively) were similar to those at Cooke-Glenmont. In addition to bioretention treatment, permeable pavement provided for filtration and sedimentation to contribute to TN reductions. A study of watershed-scale GI installed in a residential neighborhood found TN concentrations and loads were reduced by 58 and 79%, respectively, with the adoption of GI (Page et al., 2015). TN reduction has been observed in bioretention cells summarized in the ISBMPD, where median TN concentrations were reduced from 1.24 to 1.04 mg/L (Clary and Jones, 2017). Since median TN concentrations during the post-GI phase (1.88 and 1.81 mg/L for Cooke-Glenmont and Indian Springs, respectively) were higher than those reported in the ISBMPD, further TN reduction is possible, perhaps through the use of IWS zones, media amendments, or through additional bioretention implementation to treat a greater fraction of the watersheds.

Based on past column studies, nitrification, sedimentation, and filtration were the main mechanisms of nitrogen conversion and reduction at Cooke-Glenmont and Indian Springs (Hsieh and Davis, 2005; Hunt et al., 2012; Li and Davis, 2014). These studies imply TAN was converted to  $\text{NO}_3$  in the aerobic soil media and in the aerobic pore spaces in Indian Springs' permeable pavement. There was no evidence of denitrification as GI did not have an IWS (Table 4). TKN and ON were removed as particulates settled or were filtered by GI. When combined, the conversion and reductions of individual N species resulted in significant TN reductions with the installation of GI (Fig. 2).

### 3.4. Phosphorus

Orthophosphate, the most biologically available form of phosphorus that causes eutrophication in lakes and rivers (Correll, 1998), accounted for 44–59% of TP at each watershed. Although not significant, median OP concentrations and storm event loads increased by 23.7 and 18.1%, respectively, from pre- to post-GI at Cooke-Glenmont (Table 4). The ISBMPD also showed an increase in OP concentrations in bioretention, with influent and effluent concentrations of 0.02 and 0.27 mg/L, respectively (Clary and Jones, 2017). Phosphorus export has been observed in bioretention column studies (Bratieres et al., 2008; Palmer et al., 2013) and field-scale studies of street-side bioretention (Chapman and Horner, 2010) in urban areas and has been tied to leaching from organic matter, typically compost, in the bioretention media (Hurley et al., 2017; Tirpak et al., 2020). OP concentration increases have also been attributed to lawn clippings, leaf litter, and decomposition of organic matter in bioretention (Passeport et al., 2009). OP storm event loads at Cooke-Glenmont during the post-GI phase were not significantly different from Beechwold.

OP concentrations significantly decreased by 25.2% at Indian Springs with the installation of GI ( $p < 0.001$ ; Table 4). This OP reduction may be caused by sorption to silt, clay, and organic matter and vegetative uptake within the bioretention practices (Davis et al., 2006; Roy-Poirier et al., 2010a, 2010b). Since the same bioretention media was utilized in Cooke-Glenmont and Indian Springs and phosphorus-containing fertilizers are prohibited in Ohio, the OP increases at the former and significant decreases at the latter may be related to types of GI implemented and bioretention hydrology. Even though the IWS design feature was not used in the permeable pavement at Indian Springs, past studies suggest the limestone aggregate and native clay subsoils perhaps allowed for the attachment of OP to iron and aluminum oxides on clay particles (Lijklema, 1980; Tirpak et al., 2020). Drake et al. (2014) found that OP concentrations were reduced in a permeable pavement practice by 26–35%. Braswell et al. (2018) also observed effective treatment of OP, with significant reductions in concentration of 71%.

PBP concentrations were significantly greater at Cooke-Glenmont than Beechwold ( $p < 0.01$ ; Table 4). This is again due to the

watershed characteristics, where Cooke-Glenmont represented a more forested watershed than Beechwold (Fig. 1). Wooded watersheds deliver elevated phosphorus concentrations and loads because of leaf detritus and soil erosion (Gulis and Suberkropp, 2003; Lusk et al., 2020; Smith et al., 2020). PBP concentrations and storm event loads ( $p < 0.1$ ) were reduced by 30.9% and 40.3%, respectively, between the pre- and post-GI phases at Cooke-Glenmont. Observed PBP reductions were attributed to sedimentation, filtration, and volume reduction within the bioretention cells (Roy-Poirier et al., 2010a, 2010b). PBP concentrations and storm event loads were reduced by 66.5% and 49.2%, respectively, in the Indian Springs watershed following the installation of GI. Similar to processes contributing to PBP removal in bioretention, permeable pavement has been shown to effectively remove particulate phosphorus through sedimentation and filtration (Winston et al., 2016a; Tirpak et al., 2020).

TP concentrations were significantly higher at Cooke-Glenmont than Beechwold ( $p < 0.005$ ), likely due to the substantial forested cover in the watershed (Fig. 1). TP concentrations and storm event loads significantly decreased ( $p < 0.05$ ) by 20.9% and 32.6% at Cooke-Glenmont and by 47.4% and 27.8% at Indian Springs, respectively, with the adoption of GI (Table 4). Since a significant reduction in TP storm event load was observed at Cooke-Glenmont in the post-GI project phase, PBP reduction offset the marginal OP increases in the watershed. No significant difference existed in the ANCOVA slopes for TP storm event load at Cooke-Glenmont (Fig. 2), indicating that bioretention exhibited similar TP removal for both small and large storm events. Previous residential and commercial GI studies demonstrated TP concentration and load reductions in the range of 29–72% (Line et al., 2012; Page et al., 2015). Studies on P removal in bioretention suggest these reductions were due to a combination of adsorption of dissolved phosphorus and sedimentation of particulate phosphorus (Hsieh and Davis, 2005; Roy-Poirier et al., 2010a).

### 3.5. Total suspended solids

TSS concentrations and loads decreased by 61.6% and 78.3%, respectively, at Cooke-Glenmont with the installation of GI ( $p < 0.001$ ) which treated 66.5% of the imperviousness area in the watershed

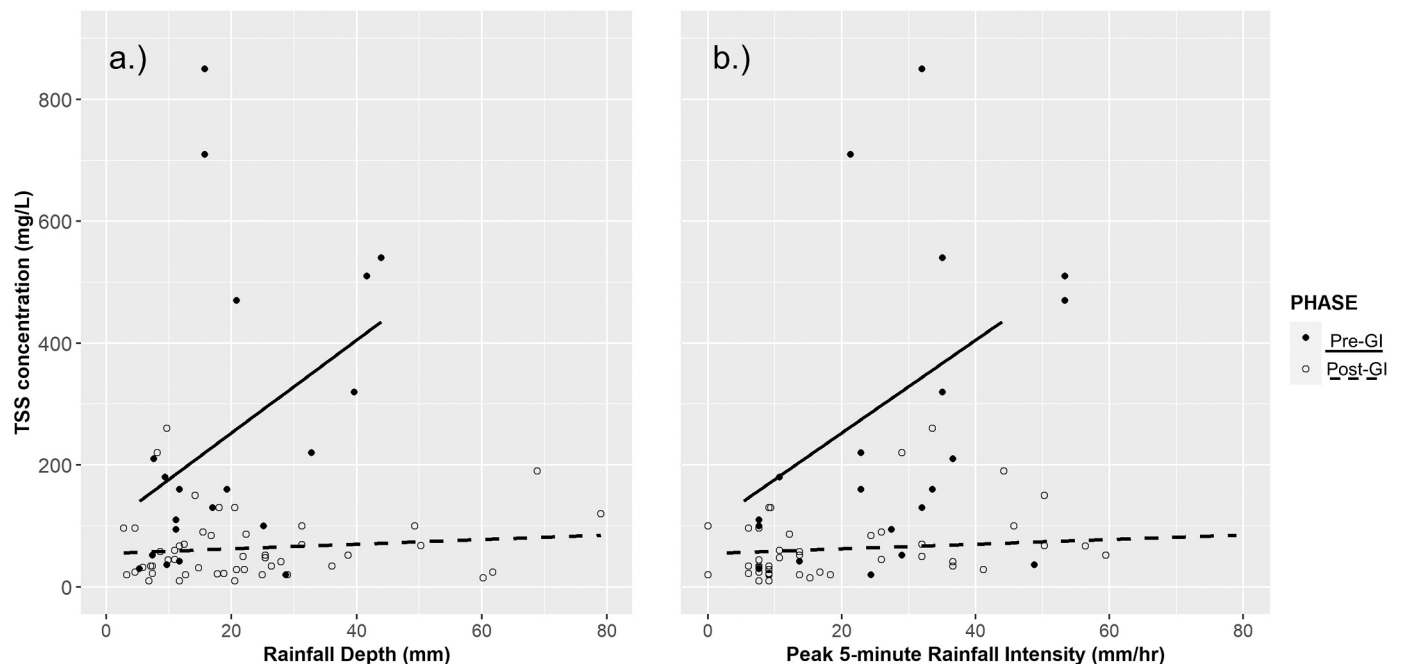


Fig. 3. a and b: Total suspended solids (TSS) versus rainfall depth (Fig. 3a) and versus peak 5-minute rainfall intensity (Fig. 3b) for pre- and post-GI phases at Cooke-Glenmont.

(Table 1). ANCOVA analysis for this watershed revealed that while the difference in intercepts were not significant ( $p > 0.5$ ), slopes were significantly different between the pre- and post-GI periods for TSS storm event loads ( $p < 0.005$ ; Fig. 2). This indicates that differences in TSS at Cooke-Glenmont could be related to variations in sediment export from the watershed for different rainfall characteristics (Fig. 3). Similar TSS generation was observed for smaller depth, low peak-intensity storm events during the pre- and post-GI phases. Storm events with these characteristics may mobilize less sediment, so further TSS reduction through GI is unlikely. Meanwhile, GI substantially reduced TSS concentrations for larger depth, higher peak-intensity storm events when influent concentrations were higher.

TSS concentrations decreased 67.7% between the pre- and post-GI phases at Indian Springs, where 69.7% of the impervious area in the watershed was treated by GI (Table 1). At both Cooke-Glenmont and Indian Springs, reductions in TSS concentrations closely paralleled the watershed imperviousness treated by GI. Storm event TSS loads decreased by 59.5% with the installation of GI (Table 4). Median TSS concentrations during the post-GI phase at both treatment sites (49 and 20 mg/L for Cooke-Glenmont and Indian Springs, respectively) were larger than those reported by the ISBMPD for effluent from single bioretention cells and permeable pavements (10.0 and 26.0, respectively; Clary and Jones, 2017). This implies further TSS reduction is possible through greater GI implementation targeting treatment of a greater percentage of the watersheds.

Bioretention cells and permeable pavements effectively decrease TSS concentrations through sedimentation and filtration, at times by several orders of magnitude (Brown et al., 2009; Trowsdale and Simcock, 2011). Another residential study reported significant reductions in TSS concentration similar to the percentage of the watershed impervious area treated by GI; Page et al. (2015) reported an 82% decrease in TSS, from 54 to 7 mg/L when 94% of the watershed imperviousness was treated by GI. Coupled with previous research, the data presented herein support the conclusion that the percent TSS concentration reduction in residential watersheds is similar to the percent of the watershed imperviousness treated by GI (within a margin of 5%). This information is valuable to engineers and regulators as they attempt to meet local regulations for TSS control.

### 3.6. Annual loading

Annual TAN loads decreased by 71.9% and 68.7% post-GI at Cooke-Glenmont and Indian Springs, respectively, due to the additional nitrification opportunities provided by the GI (Table 5). These results were similar to the 80–87% TAN annual load reductions reported by other residential and commercial GI studies (Line et al., 2012; Page et al., 2015). Pre-GI annual TAN loads of 1.47–2.46 kg/ha/yr were similar to the 1.54 kg/ha/yr reported for residential stormwater runoff with no LID treatment (Line et al., 2012). During the post-GI phase, annual loads

of 0.41–0.77 kg/ha/yr were similar to the post-GI phase of other LID studies (0–0.23 kg/ha/yr; Line et al., 2012; Page et al., 2015).

NO<sub>3</sub> annual loads decreased by 27.6% and 79.1% post-GI at Cooke-Glenmont and Indian Springs, respectively (Table 5). Such disparity between the two sites brings to question if nitrogen transformations were occurring in storm drains or elsewhere in the watershed (Kaushal et al., 2011). Previous studies on residential GI implementation also reported variable NO<sub>3</sub> annual load reductions. Whereas Page et al. (2015) reported a 60% decrease when treating 91% of the residential watershed drainage area with GI, Line et al. (2012) reported a 176% increase in NO<sub>2,3</sub> annual load. Further research into how hydraulic retention time, availability of organic carbon in storm sewers and GI, inter-event periods, denitrifying sites near plant roots, and biological activity in GI impact denitrification will allow for better categorization of NO<sub>3</sub> fluxes at the watershed-scale (Kaushal et al., 2011).

TKN annual loading decreased by 29.8% and 60.1% post-GI at Cooke-Glenmont and Indian Springs, respectively (Table 5). Although GI at Indian Springs treated a smaller percentage of the watershed, annual TKN load was reduced to a greater extent than in Cooke-Glenmont, again suggesting watershed characteristics contribute to treatment performance. Since a substantial fraction of the runoff at Cooke-Glenmont passes through a heavily forested area, more ON may be produced from this portion of the watershed. Therefore, TKN annual loads at Cooke-Glenmont could be sustained by organic matter (i.e., untreated by GI) from this wooded area. Other residential GI studies which implemented both bioretention and permeable pavement reported TKN annual loading reductions similar to that of Indian Springs (74–81%; Line et al., 2012; Page et al., 2015).

Annual TN loading decreased by 37.2% at Cooke-Glenmont and 60.5% at Indian Springs following GI implementation (Table 5). Decreased annual TN loading likely occurred at Cooke-Glenmont because organic matter from areas not treated by GI sustained TKN mass export. Further, varying rates of nitrogen conversion and removal may have impacted TN reductions, but further research is needed to investigate the mass balance of nitrogen in watershed-scale GI.

Annual OP load increased by 22.1% at Cooke-Glenmont but decreased by 69.7% at Indian Springs (Table 5). Differences in OP may be related to OP leaching from leaves and organic matter in Cooke-Glenmont, and OP binding to the limestone aggregate and iron- and aluminum-oxides in native clay soils in Indian Springs' permeable pavement. PBP annual loads were reduced to a similar extent at Cooke-Glenmont and Indian Springs, decreasing by 59.7% and 66.4%, respectively (Table 5). Greater TP annual load reduction occurred at Indian Springs (73.3%) than Cooke-Glenmont (51.9%). PBP and TP annual load reductions were similar to TSS concentration reductions in that they were within a margin of 10% of the percentage of the imperviousness treated by GI at Indian Springs (69.5%) and Cooke-Glenmont (66.5%). Line et al. (2012) reported a similar 54% TP reduction after the installation of GI. Pre-GI TP annual loads of 2.62 and 4.19 kg/ha/yr at

**Table 5**  
Annual nutrient and sediment loads (kg/ha/year) for watersheds by project phase.

| Pollutant          | Beechwald | Cooke-Glenmont |         |        | Indian Springs |         |        | Page et al. (2015) |               |        | Line et al. (2012) |       |        | Line et al. (2002) |
|--------------------|-----------|----------------|---------|--------|----------------|---------|--------|--------------------|---------------|--------|--------------------|-------|--------|--------------------|
|                    | Control   | Pre-GI         | Post-GI | % Diff | Pre-GI         | Post-GI | % Diff | SCM Calibration    | SCM Treatment | % Diff | NoTreat            | LID   | % Diff | Developed          |
| TAN                | 0.51      | 1.47           | 0.41    | -71.9  | 2.46           | 0.77    | -68.7  | 0.2                | 0.0           | -80    | 1.54*              | 0.23* | -87    | 3.2*               |
| TKN                | 5.12      | 10.01          | 7.03    | -29.8  | 19.85          | 7.92    | -60.1  | 2.6                | 0.5           | -81    | 5.46               | 1.51  | -74    | 25.4               |
| NO <sub>3</sub> -N | 1.06      | 1.35           | 0.98    | -27.6  | 6.26           | 1.31    | -79.1  | 0.3                | 0.1           | -60    | 1.41               | 2.72  | 176    | 5.1                |
| TN                 | 7.89      | 13.89          | 8.72    | -37.2  | 29.07          | 11.49   | -60.5  | 2.9                | 0.6           | -79    | 6.87               | 4.29  | -42    | 30.5               |
| OP                 | 0.72      | 0.97           | 1.19    | 22.1   | 1.48           | 0.45    | -69.7  | 0.3                | 0.1           | -55    | 0.05               | 0.05  | -      | -                  |
| PBP                | 0.65      | 2.17           | 0.88    | -59.7  | 4.07           | 1.37    | -66.4  | -                  | -             | -      | -                  | -     | -      | -                  |
| TP                 | 0.83      | 2.62           | 1.26    | -51.9  | 4.19           | 1.12    | -73.3  | 0.7                | 0.6           | -11    | 0.42               | 0.24  | -54    | 3.0                |
| TSS                | 525       | 2114           | 343     | -83.8  | 5198           | 242     | -95.3  | 157                | 12            | -92    | 244                | 8     | -97    | 2535               |

Note: % Diff refers to the percent difference in annual pollutant load between pre-GI and post-GI phases. This difference is an arithmetic difference, not least squares mean.

\* As NH<sub>3</sub>-N.

Cooke-Glenmont and Indian Springs, respectively, were similar to the 3.0 kg/ha/yr reported for a developed area (Line et al., 2002). Meanwhile, post-GI TP annual loads of 1.26 and 1.12 kg/ha/yr were less than that reported for a residential neighborhood without GI (Line et al., 2002). Coupling results of PBP and TSS identified in this study, future research should investigate possible relationships between percentage of watershed imperviousness treated by GI and the percentage of particulates removed.

TSS annual load reductions of 83.8 and 95.3% were observed at Cooke-Glenmont and Indian Springs, respectively (Table 5). However, a significant difference existed in 5-min peak rainfall intensity, with pre-GI having a higher peak intensity than post-GI. This difference was not accounted for using the simple arithmetic mean, so annual loading reductions likely represent overestimates. Other residential and commercial GI studies reported similar annual TSS load reductions of 92–97% (Page et al., 2015; Line et al., 2012), but they treated a larger (upwards of 90%) area of the watershed. Pre-GI annual TSS loads of 2114 and 5198 kg/ha/yr for Cooke-Glenmont and Indian Springs, respectively, exceeded the 1958 kg/ha/yr annual TSS load reported for a residential neighborhood by Line et al. (2012). Post-GI annual loads of 343 and 242 kg/ha/yr for Cooke-Glenmont and Indian Springs, respectively were substantially less than the typical TSS annual load for a developed area (Line et al., 2002).

### 3.7. Limitations

While having Beechwood as a control allowed for robust statistics and water quality findings, it has a large watershed area in comparison to the treatment watersheds and experienced low levels of GI implementation. Although the latter showed insignificant changes in water quality (statistically verified), both aspects of Beechwood may have had hydrologic effects (e.g., altered first flush volumes, changes in peak rainfall intensity). These effects can lead to differences in water quality which are not captured in the measurement campaign undertaken.

Spraakman and Drake (2021) define a mature bioretention cell as being three years or more post-construction. According to this definition, the bioretention cells in this study were not yet mature. Immature bioretention cells do not have fully established plants. As the plant roots and microbial communities become more established in bioretention as it matures, we hypothesize that pollutant uptake will be enhanced in these watersheds until clogging from sediment wash-off inhibits functionality. Longer-term studies are needed to test this type of hypothesis.

## 4. Summary and conclusions

The efficacy of retrofitting GI into large residential areas was investigated using a paired watershed approach following a 3.5-year monitoring study based in Columbus, Ohio, USA. GI treated particulates across a range of rainfall depths and intensities, and the majority of the pollutograph was well-treated, including nitrogenous compounds, phosphorus compounds, and sediments. Watershed-scale GI provided an aerobic environment for nitrification and biological conversion, and opportunities for plant uptake. Enhancements to GI that lengthen the contact time between soil media and stormwater and promote anaerobic conditions could have improved the ability of GI to abate nitrogenous pollutants. Watershed-scale GI also enhanced the removal of phosphorus species. OP concentrations and storm event loads increased in the treatment watershed with just bioretention but decreased in the treatment watershed with bioretention and permeable pavement, meaning OP sorbs to permeable pavement aggregates and aluminum and iron oxides in the native clay soil. Interestingly, the percent TSS concentration reduction closely mimicked (within a margin of 5%) the percentage of watershed imperviousness treated by GI. This information is vital to estimate how watershed-scale GI performs for TSS removal. Further studies are needed to determine the optimum density, placement, and type of GI; to enhance GI media, hydraulic design, and planting to

encourage pollutant uptake; and to uncover pollutant uptake by mature GI at the watershed-scale and necessary maintenance as GI ages. Such research will provide essential guidance to designers, regulators, and municipalities seeking to manage stormwater runoff through the widespread implementation of GI city-wide.

## CRedit authorship contribution statement

**Joseph S. Smith:** Conceptualization, Writing – original draft, Visualization, Investigation, Formal analysis. **Ryan J. Winston:** Methodology, Formal analysis, Writing – review & editing, Supervision, Conceptualization. **David M. Wituszynski:** Investigation, Writing - review & editing. **R. Andrew Tirpak:** Writing – review & editing. **Kathryn M. Boening-Ulman:** Investigation, Writing - review & editing. **Jay F. Martin:** Methodology, Resources, Writing – review & editing, Supervision, Project administration, Funding acquisition.

## Declaration of Competing Interest

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

## Data availability

Data will be made available on request.

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## References

- Ahiablame, L.M., Engel, B.A., Chaubey, I., 2012. Effectiveness of low impact development practices: Literature review and suggestions for future research. *Water Air Soil Pollut* 223 (7), 4253–4273. <https://doi.org/10.1007/s11270-012-1189-2>.
- Alsharif, K., 2010. Construction and stormwater pollution: Policy, violations, and penalties. *Land Use Policy* 27 (2), 612–616. <https://doi.org/10.1016/j.landusepol.2009.08.002>.
- American Public Health Association (APHA), American Water Works Association (AWWA), and Water Environment Federation (WEF), 2012. In: Bridgewater, Laura (Ed.), *Standard Methods for the Examination of Water and Wastewater*, 22nd ed. (Washington, DC).
- Antweiler, R.C., Taylor, H.E., 2008. Evaluation of statistical treatments of left-censored environmental data using coincident uncensored data sets: 1. Summary statistics. *Environ Sci Technol* 42 (10), 3732–3738.
- Atasoy, M., Palmquist, R.B., Phaneuf, D.J., 2006. Estimating the effects of urban residential development on water quality using microdata. *J Environ Manag* 79 (4), 399–408. <https://doi.org/10.1016/j.jenvman.2005.07.012>.
- Bean, E.Z., Hunt, W.F., Bidelsbach, A.B., 2007. Evaluation of four Permeable Pavement Sites in Eastern North Carolina for Runoff Reduction and Water Quality Impacts Eban. *J Irrig Drain Eng* 133 (6), 583–592. [https://doi.org/10.1061/\(ASCE\)0733-9437\(2007\)133](https://doi.org/10.1061/(ASCE)0733-9437(2007)133).
- Bedan, E.S., Clausen, J.C., 2009. Stormwater runoff quality and quantity from traditional and low impact development watersheds. *J Am Water Resour Assoc* 45 (4), 998–1008. <https://doi.org/10.1111/j.1752-1688.2009.00342.x>.
- Boening-Ulman, K.M., Winston, R.J., Wituszynski, D.M., Smith, J.S., Tirpak, R.A., Martin, J.F., 2022. Hydrologic impacts of sewershed-scale green infrastructure retrofits: Outcomes of a four-year paired watershed monitoring study. *Journal of Hydrology* 128014.
- Braswell, A.S., Winston, R.J., Hunt, W.F., 2018. Hydrologic and water quality performance of permeable pavement with internal water storage over a clay soil in Durham, North Carolina. *Journal of Environmental Management* 224 (July), 277–287. <https://doi.org/10.1016/j.jenvman.2018.07.040>.
- Bratieres, K., Fletcher, T.D., Deletic, A., Zinger, Y., 2008. Nutrient and sediment removal by stormwater biofilters: a large-scale design optimisation study. *Water Res* 42 (14), 3930–3940. <https://doi.org/10.1016/j.watres.2008.06.009>.

- Brattebo, B.O., Booth, D.B., 2003. Long-term stormwater quantity and quality performance of permeable pavement systems. *Water Res* 37 (18), 4369–4376. [https://doi.org/10.1016/S0043-1354\(03\)00410-X](https://doi.org/10.1016/S0043-1354(03)00410-X).
- Brinkmann, W.L.F., 1985. Urban stormwater pollutants: sources and loadings. *GeoJournal* 11 (3), 277–283. <https://doi.org/10.1007/BF00186341>.
- Browman, M.G., Harris, R.F., Ryden, J.C., Syers, J.K., 1979. Phosphorus loading from urban stormwater runoff as a factor in lake eutrophication: I. Theoretical considerations and qualitative aspects. *J Environ Qual* 8 (4), 561–566. <https://doi.org/10.2134/jeq1979.00472425000800040024x>.
- Brown, C., Chu, A., van Duin, B., Valeo, C., 2009. Characteristics of sediment removal in two types of permeable pavement. *Water Qual Res J Can* 44 (1), 59–70. <https://doi.org/10.2166/wqrj.2009.007>.
- Chapman, C., Horner, R.R., 2010. Performance Assessment of a Street-Drainage Bioretention System. *Water Environment Research* 82 (2), 109–119. <https://doi.org/10.2175/106143009x426112>.
- Chen, J., Liu, Y., Gitau, M.W., Engel, B.A., Flanagan, D.C., Harbor, J.M., 2019. Evaluation of the effectiveness of green infrastructure on hydrology and water quality in a combined sewer overflow community. *Sci Total Environ* 665, 69–79. <https://doi.org/10.1016/j.scitotenv.2019.01.416>.
- Clary, J., Jones, J., 2017. International Stormwater BMP Database (ISBMPD) - 2016 Summary Statistics. Werf. [https://static1.squarespace.com/static/5f8dbde10268ab224c895ad7/t/5fbd3c237ad3fe66120f69ea/1606237239545/2016\\_BMPDBSummaryStatistics\\_03-SW-1C0ch.pdf](https://static1.squarespace.com/static/5f8dbde10268ab224c895ad7/t/5fbd3c237ad3fe66120f69ea/1606237239545/2016_BMPDBSummaryStatistics_03-SW-1C0ch.pdf).
- Collins, K.A., Hunt, W.F., Hathaway, J.M., 2010a. Side-by-side comparison of nitrogen species removal for four types of permeable pavement and standard asphalt in eastern North Carolina. *J Hydrol Eng* 15 (6), 512–521. [https://doi.org/10.1061/\(ASCE\)HE.1943-5584.0000139](https://doi.org/10.1061/(ASCE)HE.1943-5584.0000139).
- Collins, K.A., Lawrence, T.J., Stander, E.K., Jontos, R.J., Kaushal, S.S., Newcomer, T.A., Cole Ekberg, M.L., 2010b. Opportunities and challenges for managing nitrogen in urban stormwater: a review and synthesis. *Ecol Eng* 36 (11), 1507–1519. <https://doi.org/10.1016/j.ecoleng.2010.03.015>.
- Correll, D.L., 1998. The Role of Phosphorus in the Eutrophication of Receiving Waters: a Review. *J Environ Qual* 27 (2), 261–266. <https://doi.org/10.2134/jeq1998.00472425002700020004x>.
- Davis, A.P., Shokouhian, M., Sharma, H., Minami, C., 2006. Water Quality Improvement through Bioretention Media: Nitrogen and Phosphorus Removal. *Water Environment Research* 78 (3), 284–293. <https://doi.org/10.2175/106143005x94376>.
- Davis, A.P., Hunt, W.F., Traver, R.G., Clar, M., 2009. Bioretention Technology: Overview of Current Practice and Future needs. *J Environ Eng* 135 (3), 109–117. [https://doi.org/10.1061/\(ASCE\)0733-9372\(2009\)135](https://doi.org/10.1061/(ASCE)0733-9372(2009)135).
- DeBusk, K.M., Hunt, W.F., Line, D.E., 2011. Bioretention Outflow: does it Mimic Nonurban Watershed Shallow Interflow? *J Hydrol Eng* 16 (3), 274–279. [https://doi.org/10.1061/\(ASCE\)HE.1943-5584.0000315](https://doi.org/10.1061/(ASCE)HE.1943-5584.0000315).
- Dietz, M.E., 2007. Low impact development practices: a review of current research and recommendations for future directions. *Water Air Soil Pollut* 186 (1–4), 351–363. <https://doi.org/10.1007/s11270-007-9484-z>.
- Drake, J., Bradford, A., Van Seters, T., 2014. Stormwater quality of spring-summer-fall effluent from three partial-infiltration permeable pavement systems and conventional asphalt pavement. *J. Environ. Manage.* 139, 69–79. <https://doi.org/10.1016/j.jenvman.2013.11.056>.
- Ellis, J.B., Revitt, D.J., Harrop, D.O., Beckwith, P.R., 1987. The contribution of highway surfaces to urban stormwater sediments and metal loadings. *Science of the Total Environment*, The 59 (C), 339–349. [https://doi.org/10.1016/0048-9697\(87\)90457-8](https://doi.org/10.1016/0048-9697(87)90457-8).
- Fan, G., Li, Z., Wang, S., Huang, K., Luo, J., 2019. Migration and transformation of nitrogen in bioretention system during rainfall runoff. *Chemosphere* 232, 54–62. <https://doi.org/10.1016/j.chemosphere.2019.05.177>.
- Field, R., O'Connor, T.P., 1997. Control Strategy for Storm-Generated Sanitary-Sewer Overflows, pp. 41–46.
- Ghasemi, A., Zahediasl, S., 2012. Normality tests for statistical analysis: a guide for non-statisticians. *International Journal of Endocrinology and Metabolism* 10 (2), 486–489. <https://doi.org/10.5812/ijem.3505>.
- Gong, Y., Liang, X., Li, X., Li, J., Fang, X., Song, R., 2016. Influence of rainfall characteristics on total suspended solids in urban runoff: a case study in Beijing. *China Water* 8 (7), 278.
- Goonetilleke, A., Thomas, E., Ginn, S., Gilbert, D., 2005. Understanding the role of land use in urban stormwater quality management. *J Environ Manag* 74 (1), 31–42. <https://doi.org/10.1016/j.jenvman.2004.08.006>.
- Green, R.H., 1993. Application of repeated measures designs in environmental impact and monitoring studies. *Aust J Ecol* 18 (1), 81–98. <https://doi.org/10.1111/j.1442-9993.1993.tb00436.x>.
- Gulis, V., Suberkropp, K., 2003. Leaf litter decomposition and microbial activity in nutrient-enriched and unaltered reaches of a headwater stream. *Freshw Biol* 48 (1), 123–134. <https://doi.org/10.1046/j.1365-2427.2003.00985.x>.
- Hathaway, J.M., Hunt, W.F., 2010. Evaluation of storm-water wetlands in series in Piedmont North Carolina. *J Environ Eng* 136 (1), 140–146.
- Hathaway, J.M., Moore, T.L.C., Burkholder, J.A.M., Hunt, W.F., 2012. Temporal analysis of stormwater control measure effluent based on windows of harmful algal bloom (HAB) sensitivity: are annual nutrient EMCs appropriate during HAB-sensitive seasons? *Ecol Eng* 49, 41–47. <https://doi.org/10.1016/j.ecoleng.2012.08.014>.
- Hathaway, J.M., Hunt, W.F., McCarthy, D.T., 2015. Variability of intra-event statistics for multiple fecal indicator bacteria in urban stormwater. *Water Resour Manag* 29 (10), 3635–3649. <https://doi.org/10.1007/s11269-015-1020-0>.
- Hellman, K., Wagner, K., Lass, D., Korfmacher, K., Gleeson Hanna, B., 2018. Estimating the Economic Impact of Stormwater Runoff in the Allen Creek Watershed. *Ecol Econ* 145 (September 2017), 420–429. <https://doi.org/10.1016/j.ecolecon.2017.11.022>.
- Hopkinson, C.S., Giblin, A.E., 2008. Nitrogen Dynamics of Coastal Salt Marshes. In *Nitrogen in the Marine Environment*. <https://doi.org/10.1016/B978-0-12-372522-6.00022-0>.
- Hsieh, C., Davis, A.P., 2005. Evaluation and Optimization of Bioretention Media for Treatment of Urban storm Water Runoff. *J Environ Eng* 131 (11), 1521–1531. [https://doi.org/10.1061/\(ASCE\)0733-9372\(2005\)131](https://doi.org/10.1061/(ASCE)0733-9372(2005)131).
- Hsieh, C., Davis, A.P., Needelman, B.A., 2007. Nitrogen Removal from Urban Stormwater Runoff through Layered Bioretention Columns. *Water Environment Research* 79 (12), 2404–2411. <https://doi.org/10.2175/106143007x183844>.
- Hunt, W.F., Davis, A.P., Traver, R.G., 2012. Meeting hydrologic and water quality goals through targeted bioretention design. *Journal of Environmental Engineering (United States)* 138 (6), 698–707. [https://doi.org/10.1061/\(ASCE\)EE.1943-7870.0000504](https://doi.org/10.1061/(ASCE)EE.1943-7870.0000504).
- Hurley, S., Shrestha, P., Cording, A., 2017. Nutrient leaching from compost: Implications for bioretention and other green stormwater infrastructure. *Journal of sustainable water in the built environment* 3 (3), 04017006.
- IPWEA, Institute of Public Works Engineering Australasia, 2017. Queensland Urban Drainage Manual, Fourth edition. Accessed online at: <https://ipweaq.intersearch.com.au/ipweaqspui/bitstream/1/4983/1/2042%20QUDM%20FINAL%2018%20August%202017%20%28%29.pdf>.
- Kamali, M., Delkash, M., Tajrishy, M., 2017. Evaluation of permeable pavement responses to urban surface runoff. *J Environ Manag* 187, 43–53. <https://doi.org/10.1016/j.jenvman.2016.11.027>.
- Kaushal, S.S., Groffman, P.M., Band, L.E., Elliott, E.M., Shields, C.A., Kendall, C., 2011. Tracking nonpoint source nitrogen pollution in human-impacted watersheds. *Environ Sci Technol* 45 (19), 8225–8232.
- Kim, H., Seagren, E.A., Davis, A.P., 2003. Engineered Bioretention for Removal of Nitrate from Stormwater Runoff. *Water Environment Research* 75 (4), 355–367. <https://doi.org/10.2175/106143003x141169>.
- Kruskal, W., Wallis, W., 1952. Use of ranks in one-criterion variance analysis. *J. Am. Statist. Assoc.* <https://doi.org/10.1080/01621459.1952.10483441>.
- Li, L., Davis, A.P., 2014. Urban stormwater runoff nitrogen composition and fate in bioretention systems. *Environ Sci Technol* 48 (6), 3403–3410. <https://doi.org/10.1021/es4055302>.
- Lijklema, L., 1980. Interaction of Orthophosphate with Iron(III) and Aluminum Hydroxides. *Environ Sci Technol* 14 (5), 537–541. <https://doi.org/10.1021/es60165a013>.
- Lim, K.Y., Hamilton, A.J., Jiang, S.C., 2015. Assessment of public health risk associated with viral contamination in harvested urban stormwater for domestic applications. *Sci Total Environ* 523, 95–108. <https://doi.org/10.1016/j.scitotenv.2015.03.077>.
- Line, Daniel E., White, N.M., Osmond, D.L., Jennings, G.D., Mojonner, C.B., 2002. Pollutant Export from various Land Uses in the Upper Neuse River Basin. *Water Environment Research* 74 (1), 100–108. <https://doi.org/10.2175/106143002x139794>.
- Line, D.E., Brown, R.A., Hunt, W.F., Lord, W.G., 2012. Effectiveness of LID for commercial development in North Carolina. *Journal of Environmental Engineering (United States)* 138 (6), 680–688. [https://doi.org/10.1061/\(ASCE\)EE.1943-7870.0000515](https://doi.org/10.1061/(ASCE)EE.1943-7870.0000515).
- Lopez-Ponnada, E.V., Lynn, T.J., Ergas, S.J., Mihelcic, J.R., 2020. Long-term field performance of a conventional and modified bioretention system for removing dissolved nitrogen species in stormwater runoff. *Water Res* 170, 115336. <https://doi.org/10.1016/j.watres.2019.115336>.
- Lusk, M.G., Toor, G.S., Inglett, P.W., 2020. Organic nitrogen in residential stormwater runoff: Implications for stormwater management in urban watersheds. *Sci Total Environ* 707, 135962. <https://doi.org/10.1016/j.scitotenv.2019.135962>.
- Marsalek, J., Rochfort, Q., 2004. Urban wet-weather flows: sources of fecal contamination impacting on recreational waters and threatening drinking-water sources. *Journal of Toxicology and Environmental Health - Part A* 67 (20–22), 1765–1777. <https://doi.org/10.1080/15287390490492430>.
- Muerdter, C.P., Smith, D.J., Davis, A.P., 2019. Impact of vegetation selection on nitrogen and phosphorus processing in bioretention containers. *Water Environment Research* 1–9. <https://doi.org/10.1002/wer.1195>.
- Müller, A., Österlund, H., Marsalek, J., Viklander, M., 2020. The pollution conveyed by urban runoff: a review of sources. *Sci Total Environ* 709, 136125. <https://doi.org/10.1016/j.scitotenv.2019.136125>.
- Nguyen, T.T., Ngo, H.H., Guo, W., Wang, X.C., Ren, N., Li, G., Liang, H., 2019. Implementation of a specific urban water management-Sponge City. *Sci Total Environ* 652, 147–162. <https://doi.org/10.1016/j.scitotenv.2018.10.168>.
- NOAA, 2020. Columbus Climate Graphs. accessed at: [https://www.weather.gov/iln/climate\\_graphs\\_cmh](https://www.weather.gov/iln/climate_graphs_cmh).
- NRCS, Natural Resources Conservation Service, United States Department of Agriculture, 2019. Web Soil Survey. Available online at the following link: <http://websoilsurvey.sc.egov.usda.gov/> (Accessed 2019).
- OEPA, 2006. Rainwater Land Development Manual. accessed at: <https://www.epa.ohio.gov/dsw/storm/rainwater>.
- Osman, M., Yusof, K.W., Takaijudin, H., Goh, H.W., Malek, M.A., Azizan, N.A., Abdurraheed, A.S., id., 2019. A review of nitrogen removal for urban stormwater runoff in bioretention system. *Sustainability (Switzerland)* 11 (19), 1–22. <https://doi.org/10.3390/su11195415>.
- Page, J.L., Winston, R.J., Mayes, D.B., Perrin, C.A., Hunt, W.F., 2015. Retrofitting residential streets with stormwater control measures over sandy soils for water quality improvement at the catchment scale. *Journal of Environmental Engineering (United States)* 141 (4), 1–11. [https://doi.org/10.1061/\(ASCE\)EE.1943-7870.0000898](https://doi.org/10.1061/(ASCE)EE.1943-7870.0000898).
- Palmer, E.T., Poor, C.J., Hinman, C., Stark, J.D., 2013. Nitrate and Phosphate Removal through Enhanced Bioretention Media: Mesocosm Study. *Water Environment Research* 85 (9), 823–832. <https://doi.org/10.2175/106143013x13736496908997>.

- Parkin, T.B., 1987. Soil Microsites as a source of Denitrification Variability. *Soil Sci Soc Am J* 51 (5), 1194–1199. <https://doi.org/10.2136/sssaj1987.03615995005100050019x>.
- Passport, E., Hunt, W.F., Line, D.E., Smith, R.A., Brown, R.A., 2009. Field study of the ability of two grassed bioretention cells to reduce storm-water runoff pollution. *J Irrig Drain Eng* 135 (4), 505–510. [https://doi.org/10.1061/\(ASCE\)IR.1943-4774.0000006](https://doi.org/10.1061/(ASCE)IR.1943-4774.0000006).
- Pawlowski, C.W., Rhea, L., Shuster, W.D., Barden, G., 2014. Some factors affecting inflow and infiltration from residential sources in a core urban area: Case study in a Columbus, Ohio, neighborhood. *J Hydraul Eng* 140 (1), 105–114. [https://doi.org/10.1061/\(ASCE\)HY.1943-7900.0000799](https://doi.org/10.1061/(ASCE)HY.1943-7900.0000799).
- R Core Team, 2020. A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
- Raciti, S.M., Burgin, A.J., Groffman, P.M., Lewis, D.N., Fahey, T.J., 2011. Denitrification in Suburban Lawn Soils. *J Environ Qual* 40 (6), 1932–1940. <https://doi.org/10.2134/jeq2011.0107>.
- Roseen, R.M., Ballesterro, T.P., Houle, J.J., Briggs, J.F., Houle, K.M., 2012. Water quality and hydrologic performance of a porous asphalt pavement as a storm-water treatment strategy in a cold climate. *Journal of Environmental Engineering (United States)* 138 (1), 81–89. [https://doi.org/10.1061/\(ASCE\)EE.1943-7870.0000459](https://doi.org/10.1061/(ASCE)EE.1943-7870.0000459).
- Roy-Poirier, A., Champagne, P., Filion, Y., 2010a. Bioretention processes for phosphorus pollution control. *Environ Rev* 18 (1), 159–173. <https://doi.org/10.1139/A10-006>.
- Roy-Poirier, A., Champagne, P., Filion, Y., 2010b. Review of bioretention system research and design: past, present, and future. *J Environ Eng* 136 (9), 878–889. [https://doi.org/10.1061/\(ASCE\)EE.1943-7870.0000227](https://doi.org/10.1061/(ASCE)EE.1943-7870.0000227).
- Sansalone, J., Kuang, X., Ranien, V., 2008. Time of Concentration estimated using Watershed. *J Irrig Drain Eng* 134 (5), 666–674. [https://doi.org/10.1061/\(ASCE\)0733-9437\(2008\)134](https://doi.org/10.1061/(ASCE)0733-9437(2008)134).
- Scholz, M., Grabowiecki, P., 2007. Review of permeable pavement systems. *Build Environ* 42 (11), 3830–3836. <https://doi.org/10.1016/j.buildenv.2006.11.016>.
- Sharma, A.K., Vezzaro, L., Birch, H., Arnbjerg-Nielsen, K., Mikkelsen, P.S., 2016. Effect of climate change on stormwater runoff characteristics and treatment efficiencies of stormwater retention ponds: a case study from Denmark using TSS and Cu as indicator pollutants. *SpringerPlus* 5 (1). <https://doi.org/10.1186/s40064-016-3103-7>.
- Shrestha, P., Hurley, S.E., Wemple, B.C., 2018. Effects of different soil media, vegetation, and hydrologic treatments on nutrient and sediment removal in roadside bioretention systems. *Ecol Eng* 112 (January), 116–131. <https://doi.org/10.1016/j.ecoleng.2017.12.004>.
- Shuster, W., Rhea, L., 2013. Catchment-scale hydrologic implications of parcel-level stormwater management (Ohio USA). *J Hydrol* 485, 177–187. <https://doi.org/10.1016/j.jhydrol.2012.10.043>.
- Silva, T.F.G., Vinçon-Leite, B., Lemaire, B.J., Petrucci, G., Giani, A., Figueredo, C.C., de O. Nascimento, N., 2019. Impact of urban stormwater runoff on cyanobacteria dynamics in a Tropical Urban Lake. *Water* 11 (946), 1–28. Retrieved from. <https://doi.org/10.3390/w11050946>.
- Smith, J.S., Winston, R.J., Tirpak, R.A., Wituszynski, D.M., Boening, K.M., Martin, J.F., 2020. The seasonality of nutrients and sediment in residential stormwater runoff: Implications for nutrient-sensitive waters. *J Environ Manag* 276, 111248.
- Song, Y., Song, S., 2019. Migration and transformation of different phosphorus forms in rainfall runoff in bioretention system. *Environ Sci Pollut Res* 26, 30633–30640. <https://doi.org/10.1016/j.chemosphere.2019.05.177>.
- Spraakman, S., Drake, J.A.P., 2021. Hydrologic and soil properties of mature bioretention cells in Ontario, Canada. *Water Sci. Technol.* 84 (12), 3541–3560. <https://doi.org/10.2166/wst.2021.464>.
- Stepenuck, K.F., Crunkilton, R.L., Wang, L., 2002. Impacts of urban landuse on macroinvertebrate communities in southeastern Wisconsin streams. *J Am Water Resour Assoc* 38 (4), 1041–1051. <https://doi.org/10.1111/j.1752-1688.2002.tb05544.x>.
- Tirpak, R.A., Winston, R.J., Feliciano, M., Dorsey, J.D., 2020. Stormwater quality performance of permeable interlocking concrete pavement receiving run-on from an asphalt traffic lane in a cold climate. *Environ Sci Pollut Res* 27 (17), 21716–21732.
- Tota-Maharaj, K., Scholz, M., 2010a. Effect of Hydrothermal Carbonization Reaction Parameters on. *Environ Prog Sustain Energy* 29 (3), 358–369. <https://doi.org/10.1002/ep>.
- Tota-Maharaj, K., Scholz, M., 2010b. Efficiency of Permeable Pavement Systems for the Removal of Urban Runoff Pollutants under varying Environmental Conditions. *Environ Prog Sustain Energy* 29 (3), 358–369. <https://doi.org/10.1002/ep>.
- Trowsdale, S.A., Simcock, R., 2011. Urban stormwater treatment using bioretention. *J Hydrol* 397 (3–4), 167–174. <https://doi.org/10.1016/j.jhydrol.2010.11.023>.
- U.S. Environmental Protection Agency, 2016. Summar of State Post Construction Stormwater Standards. Accessed at: [https://www.epa.gov/sites/default/files/2016-08/documents/swstdsummary\\_7-13-16\\_508.pdf](https://www.epa.gov/sites/default/files/2016-08/documents/swstdsummary_7-13-16_508.pdf).
- U.S. EPA, U.S. Environmental Protection Agency, 1983. Methods of Chemical Analysis of Water and Waste. EPA-600/4-79-020, Cincinnati, Ohio.
- U.S. EPA, U.S. Environmental Protection Agency, 2002. Urban Stormwater BMP Performance Monitoring: a Guidance Manual for Meeting the National Storm Water BMP Database Requirements.. [http://refhub.elsevier.com/S0301-4797\(20\)31172-5/sref64](http://refhub.elsevier.com/S0301-4797(20)31172-5/sref64).
- Walsh, C.J., Imberger, S., Burns, M.J., Bos, D., Fletcher, T., 2021. Dispersed urban-stormwater control improved stream water quality in a catchment-scale experiment. *OSF Preprints*. <https://doi.org/10.31219/osf.io/4j6vk>.
- Wang, S., Lin, X., Yu, H., Wang, Z., Xia, H., An, J., Fan, G., 2017. Nitrogen removal from urban stormwater runoff by stepped bioretention systems. *Ecol Eng* 106, 340–348. <https://doi.org/10.1016/j.ecoleng.2017.05.055>.
- Winston, R.J., Davidson-Bennett, K.M., Buccier, K.M., Hunt, W.F., 2016a. Seasonal Variability in Stormwater Quality Treatment of Permeable Pavements Situated over Heavy Clay and in a Cold climate. *Water Air Soil Pollut* 227 (5). <https://doi.org/10.1007/s11270-016-2839-6>.
- Winston, R.J., Dorsey, J.D., Hunt, W.F., 2016b. Quantifying volume reduction and peak flow mitigation for three bioretention cells in clay soils in Northeast Ohio. *Sci Total Environ* 553, 83–95. <https://doi.org/10.1016/j.scitotenv.2016.02.081>.
- Winston, R.J., Dorsey, J.D., Smolek, A.P., Hunt, W.F., 2018. Hydrologic performance of four permeable pavement systems constructed over low-permeability soils in Northeast Ohio. *J Hydrol Eng* 23 (4), 1–13. [https://doi.org/10.1061/\(asce\)jhe.1943-5584.0001627](https://doi.org/10.1061/(asce)jhe.1943-5584.0001627).
- Wu, J., Jenerette, G.D., Buyantuyev, A., Redman, C.L., 2011. Quantifying spatiotemporal patterns of urbanization: the case of the two fastest growing metropolitan regions in the United States. *Ecol Complex* 8 (1), 1–8. <https://doi.org/10.1016/j.ecocom.2010.03.002>.
- Xu, C., 2020. Research on Life-Cycle Environmental and Economic Benefits Assessment of Sponge City Source Control Facilities. Tsinghua University, Beijing, China.
- Yang, Y.Y., Toor, G.S., 2018. Stormwater runoff driven phosphorus transport in an urban residential catchment: Implications for protecting water quality in urban watersheds. *Sci Rep* 8 (1), 1–10. <https://doi.org/10.1038/s41598-018-29857-x>.
- Yin, D., Xu, C., Jia, H., Yang, Y., Sun, C., Wang, Q., Liu, S., 2022. Sponge city practices in China: from pilot exploration to systemic demonstration. *Water* 14 (10), 1531.