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Nutrients in Urban Stormwater Runoff: Current State of the Science and Potential Mitigation Options

Yun-Ya Yang[1] · Mary G. Lusk[1]

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Abstract

Purpose of Review Stormwater runoff of nutrients from developed landscapes is recognized as a major threat to water quality degradation through cultural eutrophication, which can lead to ecosystem imbalances and harmful algal growth. This review summarizes the current state-of-knowledge on the occurrence, sources, and transport processes of nitrogen (N) and phosphorus (P) in urban stormwater runoff and describes strategies for nutrient management of urban stormwater runoff. Future research needs identified from this review are provided as well.

Recent Findings Stormwater runoff of nutrients from urban environments to fresh water is controlled by multiple factors, including type of inputs, land use, development patterns, and management strategies. Recent research on stormwater management strategies has focused on internal nutrient cycling processes, such as microbial transformations of N in conventional wet ponds or bioretention cells, leading to a better understanding of the mechanisms that control the efficacy of stormwater management practices.

Summary Mitigating nutrient exports from urban environments will require controlling both quantities and sources of nutrient inputs into water systems, as well as new mechanistic understanding of the biogeochemical processes controlling nutrient treatment in stormwater ponds and low impact design (LID) structures. We need more research on source tracking of P from stormwater runoff as information is still relatively scarce. There is also a need to obtain better understanding of the dynamic interactions among multiple factors (e.g., sources, land use, characteristic of catchment and climate, management strategies) that control fate and transport of nutrients in urban stormwater runoff.

Keywords Nitrogen · Phosphorus · Stormwater runoff · Urban water quality · Stormwater management

Introduction

Urbanization leads to land use changes that alter the hydrological cycle, water chemistry, and ecosystem functioning in developed environments [1–3]. Urban stormwater runoff contains pollutants and is an important non-point source pollutant that causes water quality impairment [4]. With the replacement of vegetation and soil with impervious surface, stormwater runoff flows overland and picks up materials and pollutants in its path, including sediments, bacteria, and chemical constituents. Runoff water then flows into storm drains, treatment structures such as ponds or swales, and is eventually discharged to groundwater, streams, rivers, and estuaries (Fig. 1). Degradation of urban water quality is often associated with increasing impervious area and surface runoff [4, 5]. Stream quality degradation becomes noticeably at a tipping point at approximately 10% impervious surface cover [6]. In particular, the transport of nutrients, such as nitrogen (N) and phosphorus (P), via stormwater flow represents an important urban water quality consideration, because excess nutrients are associated with ecosystem impacts such as eutrophication, harmful algal blooms, and fish kills [7, 8]. Characterizing the sources and biochemical transformation processes of nutrients in urban stormwater runoff is critical because it can help determine where management should focus resources to achieve nutrient reduction in the environment.

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✉ Yun-Ya Yang
yunyayang@gmail.com

Mary G. Lusk
mary.lusk@ufl.edu

[1] Soil and Water Quality Laboratory, Gulf Coast Research and Education Center, Institute of Food and Agricultural Sciences, University of Florida, 14625 CR 672, Wimauma, FL 33598, USA

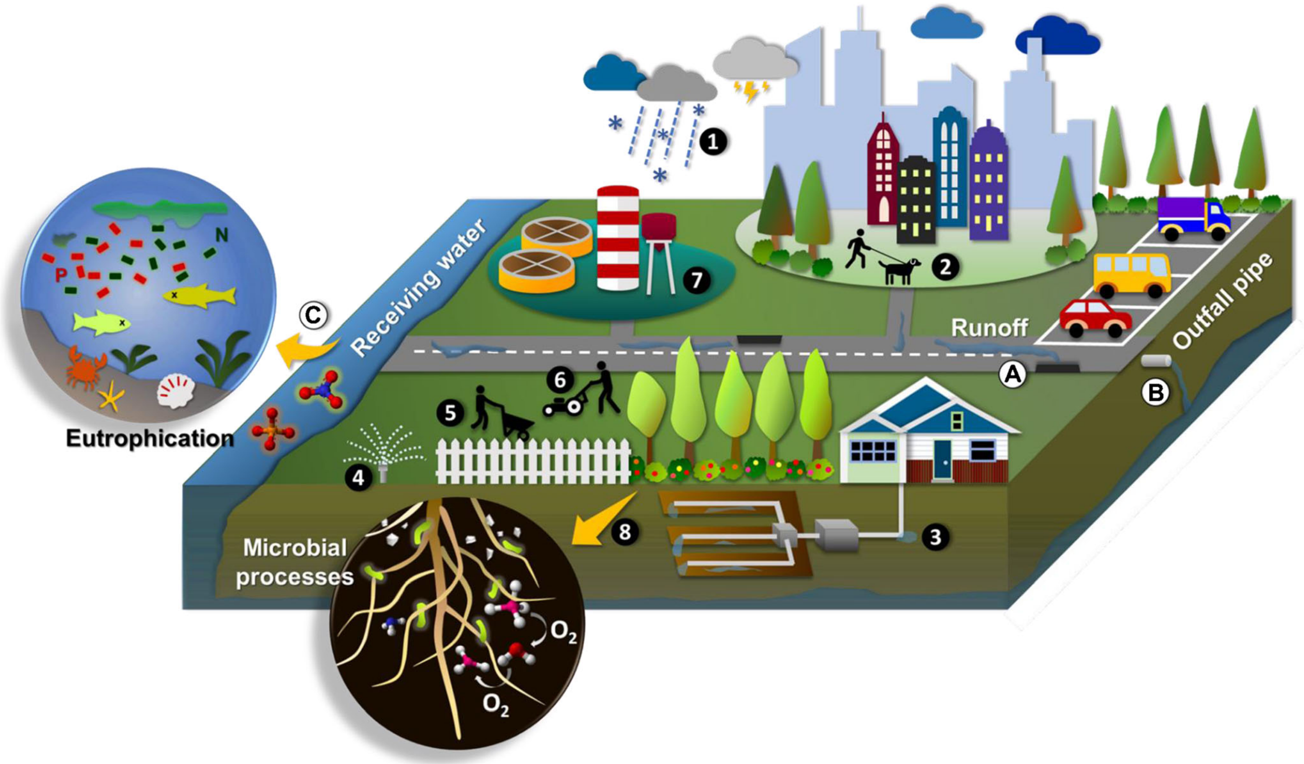

Fig. 1 Overview of pathways and sources of nutrients in urban environment. (A) Urban stormwater runoff is generated when precipitation from rain/snowmelt events over impervious surfaces. (B) Runoff water then makes its way into storm drains and discharges into streams, rivers, and estuaries untreated. (C) Excessive amounts of nutrients in water bodies can cause eutrophication, often leading to fish kills. The potential nutrient sources in urban stormwater runoff include (1) atmospheric deposition, (2) pet waste, (3) improperly functioning septic systems, (4) landscape irrigation, (5) use of chemical fertilizers on lawns, (6) soil and decomposition plant materials, (7) leaking sanitary sewers, and (8) microbial sources

Sources of N and P in urban landscapes include natural (e.g., atmospheric deposition), anthropogenic (e.g., chemical fertilizers, pet waste, leaking sanitary sewers), and biogenic (e.g., leaf litter, grass clippings) materials (Fig. 1). Knowledge of potential N and P sources in watershed and the transport processes that may remove nutrients will aid in developing strategies to control the impact of N and P in downstream aquatic ecosystems. To improve water quality, we will also need increased focus on methodological approaches for mitigating nutrient transport via stormwater runoff, either from structural or non-structural stormwater best management practices (BMPs) [9, 10]. For example, retention ponds, common in many urban areas, are often ineffective in removing dissolved N and P [11]. Alternative green infrastructure or low impact development BMPs, such as rain gardens and bioswales, attempt to do better to remove nutrients by mimicking the natural processes that promote stormwater infiltration at the site where it is generated, but we still need more studies to verify the efficacy of these practices under a wide range of environmental conditions.

In this review, we explore the following questions: (a) What has been learned about the occurrence, sources, and mechanisms that affect transport processes of nutrients (i.e., N and P) in urban stormwater runoff? (b) What mitigation strategies are currently used for managing urban stormwater runoff and what are removal mechanisms for nutrients? and (c) What further research is needed to protect urban ecosystems and improve water quality? This review can serve as a guide for developing a sustainable path for urban stormwater management.

Data Sources and Compilation

A comprehensive literature search was performed via Web of Science (http://apps.webfknowledge.com/), PubMed (https://www.ncbi.nlm.nih.gov/pubmed), and Google Scholar (scholar.google.com), with combinations of search terms: "urban stormwater runoff," "residential runoff," "urban watersheds," "stormwater quality," "sources and transport of nitrogen," "sources and transport of phosphorus," "nutrients management," "stormwater management," "low impact development," and "green infrastructure." We limited our search to peer-reviewed articles that were published between 2000 to September 2017. Monitoring data collected by US government agencies were also included in this review [12–14]. In the end, 117 relevant papers were selected for this review. The following information for articles within the scope was recorded: information about land use, study region, study approach, sample type (i.e., runoff, river, stream), N and/or P concentration, and sources of N and/or P. For the "nutrient sources in urban environments" section, we identified 29 relevant studies [15, 16, 17••, 18–29, 30••, 31–43] that were conducted in various conditions (e.g., baseflow vs wet season) across different regions and quantified by different techniques. Specifically, these 29 articles provide quantitative data for nutrients and show at least one potential nutrient source in urban environments. For the "stormwater management to mitigate nutrient pollution" section, we selected 57 articles [17••, 42, 44–67, 68••, 69–98] and compiled information about nutrient mitigation strategy and removal mechanism.

Nutrient Concentrations and Forms in Urban Stormwater Runoff

Nutrient concentrations/loads in urban waters across multiple regions in the USA are shown in Table 1. Historically, one of the most comprehensive data sets on nutrient concentrations in stormwater was collected between 1992 and 2002, when the US EPA conducted the national stormwater quality program that monitored over 3700 storm events from 360 sites across 16 states [13]. In that study, typical event mean concentrations (EMC) of total Kjeldahl nitrogen (TKN, the sum of NH_4–N plus organically bound N) and TP in urban stormwater runoff from the National Stormwater Quality Database (NSQD) were 1.74 and 0.37 mg/L, respectively. Similar EMC of TKN (1.67–1.73 mg/L) and TP (0.32–0.34 mg/L) in urban stormwater runoff were observed from other national studies in the USA (i.e., nationwide urban runoff program (between 1978 and 1983) [12] and the national stormwater database [14].

Both N and P are found in inorganic or organic forms as well as dissolved or particulate forms in stormwater runoff. The relative proportion of inorganic versus organic nutrient forms in stormwater runoff varies with geology, land use, and hydrologic conditions [99]. Inorganic forms of N include nitrate-N (NO_3–N), ammonium-N (NH_4–N), and nitrite-N (NO_2–N) while organic forms of N include dissolved organic N (DON) and particulate organic N (PON) [100]. In stormwater runoff, P is distributed between dissolved forms (e.g., organic P and orthophosphate, PO_4–P) and P primarily associated with fine particles (e.g., P sorbed to soil particles and organic matter) [15, 101–103]. For example, Vaze and Chiew [101] determined N and P loads are associated with different sediment sizes in urban stormwater runoff and observed that 20–50% of particulate total N (TN) and 40–50% of particulate total P (TP) in stormwater runoff was associated with sediment between 11 and 150 μm. In some cases, DON can be the majority of TN in urban runoff samples. Studies have observed high fractions of organic N (50–80%) in the winter snowmelt runoff in Saint Paul, Minnesota [16] and urban stormwater runoff in Tampa, Florida [17••]. Recent studies have found that residential stormwater runoff, water outflow from a stormwater retention pond, and urban streams were sources of biodegradable DON [17••, 18]. As differences in nutrient forms may affect microbial transformations and nutrient retention in the environment, it is essential to identify species of N and P in stormwater runoff.

Nutrient Sources in Urban Stormwater Runoff

Various biogeochemical and hydrologic tracers have been used widely to distinguish point and non-point nutrient sources in urban stormwater runoff. For example, stable isotope techniques were used to determine NO_3–N sources in urban residential runoff [19, 20], urbanized watersheds [21, 22], streams [23], and river basins [24–27]. More recently, high-resolution mass spectrometry has become a useful tool for characterizing the composition of organic molecules to better understand N sources associated with organic materials and microbial processes [17••, 18]. Other approaches such as comparison of nutrient mass balances across different land uses have been used to assess potential nutrient sources and to inform landscape nutrient management in urban watersheds [28, 29, 30••]. A summary of reported natural and anthropogenic nutrient sources in US urban waters across multiple regions can be found in Table 1. In this section, we briefly described the potential sources of N and P in urban environments and provide mechanistic understanding of processes driving transport of N and P in urban watersheds in the later section.

Based on collated information from the 29 relevant peer-reviewed literature [15, 16, 17••, 18–29, 30••, 31–43], 17 studies [16, 19–23, 26–29, 30••, 31–34, 41, 43] indicated atmospheric deposition (i.e., wet form such as rain and snow or dry form such as gases and aerosols) as a source of N and P in urban environments. For example, atmospheric deposition accounted for 19 to > 50% of TN loads in urban watersheds [22, 30••], and it contributed < 10 to > 90% of NO_3–N in urbanized water systems during storm events [19, 30••, 32, 33] (Table 1). Only one study stated that atmospheric deposition was the single most important P input in urban watersheds, contributing 13–33% of TP loads to watersheds in Minnesota [30••].

Numerous studies have identified chemical fertilizers as NO_3–N sources in water systems, such as river basins in Germany [25], river networks in China [26], and urban watersheds in the USA [28, 30••, 34]. Fifteen studies [19–22, 24–29, 30••, 33, 34, 39, 42] show lawn fertilizers can contribute N and P inputs in urban environments. For example, lawn fertilizers (range: 1596–3407 kg/km^2/year) were 37 to 59% of TN inputs in urban watershed in Minnesota [30••] and

Table 1 The concentrations/loads and potential sources of nutrients in urban waters across multiple regions in the USA

Land use/location	Approach/sample type	Watershed area	Experimental design	Nutrient concentration	Nutrient sources
Humid subtropical					
Urban residential Tampa, FL [20]	Dual isotopes of NO_3–N and water along with isotope mixing models Stormwater runoff	3.6–50 ha 42–81% ISA 6 sites	5 min intervals 21 events 2 months	Mean (mg/L): TN: 0.42 NO_3–N: 0.1 TP: 0.43 PO_4–P: 0.25	N: atmospheric deposition (35–64% of NO_3–N)[a], chemical fertilizers (1–39%)[a], soil and organic materials (7–33%), microbial sources (nitrification) P: soil[a] and mineralization from organic materials
Urban residential Tampa, FL [19]	Dual isotopes of NO_3–N and water along with isotope mixing models Stormwater runoff	11 ha 37% ISA 1 site	5 min intervals 25 events 3 months	Mean (mg/L): TN: 0.96 NO_3–N: 0.24	N: atmospheric deposition (43–71% of NO_3–N)[a], chemical fertilizers (< 1–49%)[a], soil and organic materials (< 1–8%), microbial sources (nitrification)
Urban residential Tampa, FL [17••]	FT-ICR-MS Stormwater runoff	11 ha 37% ISA 1 site	4 months	Mean (mg/L): TDN: 1.48 DON: 1.09 DIN: 0.39	N: leaf litter, grass clippings, soil
Urban watersheds Baltimore, MD [41]	Dual isotopes of NO_3–N and water along with isotope mixing models, mass balance calculations Stream samples	530–1910 ha 21–46% ISA 4 sites	Routinely sampling 3 years	Exports (kg/ha/year): NO_3–N: 3.8–7.0 TN: 4.5–9.9 PO_3–P: 61–181 TP:161–363	N: atmospheric deposition (8–15% of NO_3–N), leaking sanitary sewers (51–56%)[a], microbial sources (33–41%; nitrification)
Urban river-estuarine continuum Chesapeake Bay, MD [24]	Dual isotopes of NO_3–N along with isotope and salinity mixing models, mass balance calculations Stream samples	NA	Routinely sampling 2 years	Mean (mg/L): NO_3–N: 0.013	N: chemical fertilizers, wastewater, microbial sources (68% of TN; denitrification, assimilation)[a]
Urban watersheds Baltimore, MD [22]	Dual isotopes of NO_3–N along with isotope mixing model, mass balance calculations Stream samples	17,150 ha < 5–> 40% ISA 6 sites	Biweekly 7 months	NA	N: atmospheric deposition (5–94% of NO_3–N)[a], chemical fertilizers, wastewater (7–95%)[a], microbial sources (denitrification)
Humid subtropical					
Urban watersheds Baltimore, MD [28]	Mass balance calculations Stream samples	81–16,278 ha 4 sites	3 years	Yields (kg N/ha/year): TN: 4.9–11.4 NO_3–N: 3.5–6.8	N: atmospheric deposition, chemical fertilizers
Urban streams Austin, TX [95]	Dual isotopes of NO_3–N Stream samples	10–> 90% ISA 4 sites	3 and 4 occasions during baseflow and streamflow conditions, respectively	Mean (mg/L): NO_3–N: 0.02–1.9	N: atmospheric deposition, chemical fertilizers[a], sewage
Semi-arid					
Urban residential Aliso Creek Watershed, CA [36]	Concentration measurements Lawn irrigation driven runoff	28 ha 56% ISA 1 site	56 events 3-h intervals 1 week	Mean (mg/L): TN: 10.85 NO_3–N: 5.42 TP: 1.27 PO_3–P: 0.82	N: irrigation P: irrigation
Urban watersheds Salt Lake City, UT [21]	Dual isotopes of NO_3–N and water along with isotope mixing model Stream samples	NA	Weekly to monthly 1 year	Mean (mg/L): NO_3–N: 0.04–1.96 NH_4–N: < 0.01	N: atmospheric deposition, chemical fertilizers, leaking sewers[a]

Table 1 (continued)

Land use/location	Approach/sample type	Watershed area	Experimental design	Nutrient concentration	Nutrient sources
				DON: 0.11	
Urban watersheds Tucson Basin, AZ [33]	Triple isotopes of NO_3–N along with isotope mixing models Stormwater runoff	33–2698 ha 3–91% ISA 6 sits	3 summer seasons	Concentration (mg/L): NO_3–N: 0.3–1.1 NH_4–N: 0.02–0.6	N: atmospheric deposition (0–82%)[a], chemical fertilizers, microbial sources (nitrification)
Arid					
Urban watersheds Phoenix, AZ [34]	Triple isotopes of NO_3–N along with isotope mixing models Stormwater runoff	6–20,247 ha 42–69% ISA 10 sites	Biweekly 20 months	EMC (mg/L): NO_3–N: 0.16–1.65 NH_4–N: 0.006–10.13	N: atmospheric deposition, chemical fertilizer (6–65%)[a], microbial sources (nitrification)
Continental					
Urban residential subwatersheds Saint Paul, MN [29]	Extensive stormwater monitoring data coupled with statistical analysis Stormwater runoff	4–3170 ha 20–80% ISA 19 sites	Over 2300 events 10 years	EMC (mg/L): TN: 2.36 NO_x–N: 0.44 NH_4–N: 0.26 TP: 0.32 TDP: 0.99	N & P: atmospheric deposition, chemical fertilizers, leaf litter[a], vegetation litter[a], pet waste, soil
Mixed residential, commercial, and industrial urban subwatersheds, Saint Paul, MN [30••]	Mass balance calculations Stormwater runoff	16–3171 ha 35–56% ISA 7 sites	NA	Mean concentration for flow weighted storm flows ($kg/km^2/year$): TN: 230–634 TP: 28–92	N: atmospheric deposition (19–34% of TN)[a], chemical fertilizers (37–59%)[a], pet waste (28%), leaf litter, grass clippings, microbial sources (denitrification) P: atmospheric deposition (13–33% of TP)[a], pet waste (76%)[a], leaf litter, grass clippings
Urban residential watershed, Saint Paul, MN [16]	Mass balance calculations Snowmelt runoff	17 ha 51% ISA 1 site	2 years	Mean (mg/L): TN: 4.3 TP: 0.47	N: atmospheric deposition, leaf litter[a], pet waste, microbial sources (denitrification) P: leaf litter[a], lawns, soil
Urban residential Madison, WI [42]	Concentration measurements Stormwater runoff	1.21/6.47 ha 45% ISA 2 sites	8 months	Mean (mg/L): TN: 1.9–3.6 TP: 0.38–2.17	N: chemical fertilizers, leaf litter P: leaf litter[a]
Urban stream Pittsburgh, PA [23]	Dual isotopes of NO_3–N along with isotope mixing model Stormwater runoff	1570 ha 38% ISA 3 sites	2 years	Mean (mg/L): NO_3–N: 0.6–1.0	N: atmospheric deposition (6–34% of NO_3–N), sewage (72–94%)[a], microbial sources (7–22%; denitrification)
Mixed land use watersheds (21 to 70% developed) Central Connecticut, CT [32]	Dual isotopes of NO_3–N River samples	2 rivers	Routinely sampling 3 months	Concentration (mg/L): NO_3–N: ~ 0.1–2.7	N: atmospheric deposition (0–53% of NO_3–N), sewage (0–100%), microbial sources (nitrification)
Urban residential, commercial Madison, WI [35]	Concentration measurements along with urban-runoff model Stormwater runoff	17–94 ha 34–38% ISA 2 sites	25 events 1 year	Mean (mg/L): TP: 0.09–2.34 dissolved P: 0.03–1.54	P: lawns[a], streets

ISA impervious surface area, *EMC* event mean concentration, *NA* not available, *TDN* total dissolved nitrogen, *DON* dissolved organic nitrogen, *DIN* dissolved inorganic nitrogen, *NO_x–N* NO_3–N + NO_2–N, *TDP* total dissolved phosphorus

[a] Major sources as indicated in the literature

contributed on average 33–44% of NO_3–N in urban aquatic environments [19, 20, 34].

A combination of soil and organic materials ($n = 12$ of 29 reviewed paper) [15, 16, 17••, 18–20, 25, 29, 30••, 31, 35, 42] is considered as a significant contributor of nutrients to urban runoff in areas with high tree canopy and vegetation cover [17••, 18, 29]. Bratt et al. [16] estimated that during winter melt events, leaf litter contributed approximately 50% of annual export of N and P to an urban residential watershed in Saint Paul, Minnesota. Further, a study conducted by the US Geological Survey [35] reported that a large proportion of TP (range 44–67%) and dissolved P (45–71%) was contributed by lawns, whereas suspended solids (73–81%) was largely contributed by streets in two urban residential basins in Madison, Wisconsin.

Studies have also highlighted that human waste such as leaking sanitary sewers/septic systems ($n = 12$ of 29 reviewed papers) [21–27, 32, 36–38, 41] can be critical to nutrient exports in urban environments. For example, a recent study reported that reclaimed water (treated wastewater) used for lawn irrigation drove high concentrations of N and P in surface runoff from a residential catchment in southern California [36]. Significant sewage contributions of nutrients (> 23 to 96% of N inputs) to urban rivers [32], urban streams [23], and an urbanizing estuary [24] were found in many N source studies. In addition, reviews of nutrients in urban environments by Carey et al. [37] and Badruzzaman et al. [38] pointed out that improperly functioning septic systems can be a major source of nutrients in urban watersheds. Similarly, heterogeneity of human behaviors such as pet ownership and associated pet waste deposition in the landscape can also impact on nutrient in stormwater runoff ($n = 5$ of 29 reviewed paper) [16, 29, 30••, 39, 40]. For example, two studies conducted in Saint Paul, Minnesota, indicated that pet waste contributed more than 70% of TP inputs in urban watersheds [30••, 39]. Further, pet waste accounted for 2.3×10^6 N/kg/year to arid urban and residential landscapes in Phoenix region, Arizona [40].

Mechanisms That Affect Nutrient Transport in Urban Stormwater

Surrounding Land Use and Impervious Surfaces

The transport and mobilization of N and P in stormwater runoff are influenced by surrounding land use and the area and connectivity of urban impervious surfaces [3, 5, 104, 105]. A seminal paper by Walsh et al. [5] reported a clear relationship between watershed imperviousness and urban water quality. Heterogeneity of catchment effects on urban water quantity and quality have been observed in a national study [13], studies conducted in semi-arid urban catchments with impervious surfaces varying from 22 to > 90% [105], and in human-impacted watersheds with different land uses [106]. These studies suggested that nutrient export during storm events can vary significantly across a broad range of land use and impervious surface [41]. For example, according to NSQD, the median EMC for TKN in various urban land uses (e.g., residential, commercial, and industrial) ranged from 0.74 mg/L in open space to 2.0 mg/L in freeways, whereas TP ranged from 0.18 mg/L in institutional areas to 0.31 mg/L in open space and residential areas [13]. Higher watershed N export was found in agricultural (37 kg/N/ha/year), followed by suburban (15 kg/ha/year), and lower in forest (6 kg/N/ha/year) watersheds in Maryland [106]. Likewise, a 3-year stormwater runoff study conducted in Dongguan City of China found that EMC of TN and TP in stormwater runoff were greater in mixed commercial and residential catchments (87% impervious area; TN 16.7 mg/L; TP 3.2 mg/L) than for industrial areas (74% impervious area; TN 9.0 mg/L; TP 2.1 mg/L) and parking lots (37% impervious area; TN 1.1 mg/L; TP 1.2 mg/L) [104], likely due to the differences in impervious surface areas of the catchments. Lewis and Grimm [107] also observed the concentration of NH_4–N in stormwater runoff was significantly correlated with impervious area.

While the aforementioned studies indicate a significant positive relationship between impervious surface and nutrient transport, a number of studies in recent years have shown that percent impervious cover may not be as important as "urban form," or specific land cover patterns that relate features such as tree canopy coverage, connectivity of impervious surfaces, and type of stormwater infrastructure [108]. For example, in an investigation of water quality in urban catchments with variable urban form, Beck et al. [108] observed that impervious surface area alone was not sufficient to explain variations in water quality. Instead, the connectivity of land cover patches and the size and shape of lawns and buildings were the most predictive metrics for predicting water quality outcomes. Likewise, Goonetilleke et al. [109] observed that features such as the number of single family versus multifamily homes played a role in nutrient export, with areas dominated by detached single family homes having higher nutrient exports than those dominated by multifamily homes.

Hydrologic Characteristics

Climate variables such as the frequency and intensity of storms is another important factor that affects the nutrient transport in stormwater runoff. Increasing seasons of high flow often lead to greater contributions of nutrients to urban stormwater runoff [104]. For example, NO_3–N exports can be four- to fivefold greater during the wet years than the dry years in urban watersheds [110]. A review by Kaushal et al. [106] also pointed out the interannual variability of nutrient loads from urban watersheds, with peak exports of N and P during the wet years. Further, a positive correlation between rainfall intensity and loads of NH_4–N, NO_3–N, organic N, and TN in stormwater runoff from urban catchments in Arizona was reported by Lewis and Grimm [107]. Similarly, greater flow rates during the winter and spring rainy seasons resulted in greater inputs of TN and NO_3–N than in summer and fall dry seasons in an urban river-estuarine continuum in Maryland [24].

Other hydrological factors such as antecedent rainfall conditions and flow connectivity between nutrient sources and receiving waters can also affect nutrient transport in urban environments. For example, greater concentration of NH_4–N in stormwater runoff was observed after a longer antecedent dry period in arid urban catchments [107]. A change in NO_3–N sources as a function of stormwater runoff volume (June–December), mainly influenced by the residence time and antecedent moisture conditions, was observed in an urban watershed in Maryland [106]. Further, Hale et al. [111•] reported that spatial and temporal variations of N and P exports in stormwater runoff from arid urban watersheds in Arizona were mainly associated with changes of watershed hydrology, with precipitation volume and impervious surface having greater direct influence on inorganic N retention after a storm than catchment features such as grass cover.

During storm events, a first flush effect (i.e., rapid changes in pollutant load in the initial surface runoff of a storm) can influence the species of nutrients in stormwater runoff [104, 112, 113]. Li et al. [104] investigated the relationships between the first flush effect and various pollutants such as TN and TP among three different urban catchments in China and concluded that the first flush effect was influenced by antecedent conditions, rainfall intensity and depth. A positive relationship between first flush of PO_4–P and rainfall depth and runoff volume was observed by Hathaway et al. [114].

Nutrient buildup refers to the accumulation of nutrient sources on urban surfaces such as roadways and roofs. After a period of buildup, nutrients may be subject to washoff into the stormwater flow by the kinetic energy of raindrops [115]. The extent to which built up nutrients are washed off depends on nutrient form, antecedent dry days, and particle size. In stormwater runoff, a large fraction of P is often associated with soil and sediment particles [15, 101], while N is largely present in the dissolved or organic form [116]. This has implications for how each nutrient builds up on impervious surfaces and washes off into stormwater. Miguntanna et al. [116] observed that because N is so frequently present in dissolved forms, it is easily washed off by low-intensity rain events, such that its loading to stormwater is source limited and not transport limited. On the other hand, P was found to be transport limited, with PO_4^{3}–P constituting a significant portion of total P only in particles < 75 μm diameter [116]. The relative abundance of fine particles that would be expected to be most likely to transport PO_4^{3}–P was found to increase with antecedent dry days [117]; thus, the longer the antecedent dry period, the greater the P buildup.

Stormwater Management to Mitigate Nutrient Pollution

Many treatment strategies have been developed to manage nutrient pollution carried by urban stormwater. Burns et al. [44] and Fletcher et al. [45] provide a summary of the nomenclature that is commonly used to describe stormwater management strategies: BMPs; stormwater control measures (SCM); green infrastructure (GI); low impact design (LID) in the USA and New Zealand; sustainable urban drainage systems (SUDS) in the UK; and water sensitive urban design (WSUD) in Australia. While having different names ascribed to them depending on the country and/or the specific field of practice of the user, they are a collection of similar strategies and are all based on managing stormwater for the goal of pollutant treatment as well as flow volume reduction. However, since the terminology used to categorize stormwater management is region-specific and reflects a locally shared understanding, we consider here that stormwater nutrients are managed in the most general sense by both non-structural and structural practices, and we provide our discussion of treatment strategies based on that broad categorization (Table 2).

Non-structural Strategies to Manage Nutrients in Stormwater

Non-structural stormwater treatment strategies are pollution prevention practices that rely on education and institutional behaviors to limit the transport of nutrients from landscapes in the first place. These non-structural strategies include educating homeowners on how to fertilize lawns appropriately, urban design that leaves open spaces with pervious surfaces, and policies for builders/developers that require soil amendments during new construction to improve the infiltrative and nutrient-holding capacity of urban soils. Taylor et al. [46] provide an early synthesis of studies to quantify the efficacy of non-structural stormwater management strategies. Though their synthesis paper called for greater research investment in understanding the costs and benefits of non-structural practices, a decade later the data to quantify how well these

Table 2 Mitigation strategies of nutrients in stormwater runoff

	Strategy	Mechanism or example	Comments
Non-structural[a]	Urban planning [96, 97]	e.g., regulatory actions by a government entity to require LID practices for all new construction	Often requires coordination between stormwater departments, water utilities, and city planners
	Institutional controls [98]	e.g., a regional stormwater quality management plan and the commitment to fund the plan	
	Source control and pollution prevention [42, 51, 52]	e.g., street sweeping to remove fallen leaves	Most commonly studied non-structural practice
	Public education [57]	e.g., an education program that teaches homeowners to avoid fertilizer application just before a storm	Several studies exist on methods for public education but few exist on the impacts of education on eventual water quality
	Regulatory controls [56]	e.g., a county-wide ban on N fertilizers during the rainy season	Common in Florida, USA, where the use of N fertilizers is banned during the summer rainy season in many counties/cities
Conventional structural	Dry ponds [64]	Store water for 24–72 h, presumably allowing pollutants time to settle out, after which water is slowly released	Also called detention ponds; are normally dry in between storm events
	Wet ponds [17••, 59]	Store water for longer periods of time and rely on settling as well as biological processes to remove pollutants	Also called retention ponds; have a permanent pool of water
Alternative structural	Bioretention and rain gardens [67, 68••, 69]	Vegetated depressions in the landscape that capture stormwater flows and allow more natural infiltration	Work primarily by reducing runoff flow volumes, filtering and plant uptake of stormwater pollutants
	Swales [73, 74]	Vegetated channels that convey stormwater, usually to another downstream BMP	Work primarily by allowing sedimentation of particle-bound nutrients; often a preliminary step in a BMP treatment train
	Wetlands [77–79]	Natural or constructed wetlands used to treat stormwater runoff	Rely on filtering, biogeochemical transformations and plant uptake; often a final step in a BMP treatment train
	Permeable pavers [81, 84–86]	Replace conventional impervious surfaces and allow increased infiltration of stormwater; may be constructed of porous asphalt, concrete, or interlocking concrete pavers	May need to be periodically vacuumed or cleaned if clogging becomes a problem; typical life span of 20–30 years
	Green roofs [88–90]	Rooftops covered with lightweight growing media and plants to capture rainfall and reduce runoff volume	Shown to increase N and P concentrations, especially if fertilizers are used during plant establishment

[a] Descriptions of non-structural strategies are adapted from Taylor and Fletcher [46]

strategies function is still sparse. Much of the peer-reviewed studies on non-structural stormwater management have focused not on how well specific practices reduce pollutant loads but instead on the perceptions and preferences of homeowners and other stakeholders when they need to select a non-structural stormwater management option [47–49].

Among non-structural practices, source control and public education activities have been the most documented in peer-reviewed literature. One of the most studied non-structural practices is street sweeping, a form of source control for N and P in organic debris such as leaf litter and grass clippings. Baker et al. [50] collected street debris and measured the mass of N and P in swept material from streets with varying tree canopy cover in Minnesota. They observed that coarse organic matter (> 2 mm) in the collected street debris accounted for 72% of TN and 33% of TP on a dry mass basis. They also noted that nutrient recovery via street sweeping was seasonally variable, with the highest nutrient load reductions occurring from autumn street sweeping timed to coincide with widespread leaf fall. Selbig [42] measured N and P concentrations in stormwater from basins with and without fall (October–November) leaf removal in Madison, Wisconsin, and observed TN and TP load reductions of 74 and 84%, respectively, when leaves were actively removed from streets. In that study, leaves and other street debris were removed before each precipitation event for a 2-month period, a practice undoubtedly above and beyond the scope of practice that most municipal entities are able to implement. Thus, the Selbig [42] study represents a “best case” scenario but does underscore the potential for nutrient removal through street sweeping.

Nutrient removal via street sweeping is accomplished not only through physical removal of particulate N and P but also by preventing the leaching of dissolved nutrients from street solids. Hobbie et al. [51] measured leaching of P from leaf litter in urban gutters and reported 27 to 88% of initial P was leached within 24 h of leaf wetting by stormwater. In a laboratory study, Duan et al. [52] reported that P leaches rapidly from leaves once they come in contact with stormwater and that leaching continues for upwards of 100 h. One important implication of this is that structural controls such as grates over storm drains may prevent bulk organic materials (e.g., leaves, grass, tree bracts) from entering downstream stormwater flows, but those captured bulk materials may still leach dissolved nutrients into the stormwater flow. Thus, complete removal of these materials before they interact with stormwater is optimal [42].

Public education and outreach programs are another commonly studied practice for non-structural stormwater management, though most reports in the literature focus on knowledge gain and behavior changes as the outcomes and not on associated nutrient load reductions to receiving water bodies [53–55]. Fore [53] reported fertilizer use reductions of approximately 30 to 70% over a period of 3 to 6 years following fertilizer education campaigns in Washington. Closely related to education campaigns are regulatory limits on fertilizer use. In Florida, for example, more than 50 counties and municipalities have enacted a local fertilizer ordinance that bans the use of N fertilizers during the summer rainy season (June–September) [56]. A peer-reviewed scientific study of the impacts of the Florida bans on water quality has not yet been produced. However, Persaud et al. [57] report that 3 years after such a ban was enacted in Manatee County, Florida, 69% of residents in a study neighborhood of 6000 homes lacked awareness of the ordinance and why it was enacted, potentially leading to little actual compliance with the ordinance.

Persaud et al. [57] also concluded that public education is a necessary addition for obtaining desired behavior change results from fertilizer bans. These authors observed that homeowners were more willing to adopt positive behavior changes when stormwater education includes “proof” that their behavior changes would impact local water quality; in other words, it was important to homeowners in the study that they see scientific data linking homeowner fertilizer practices to nutrient and algal levels in neighborhood water bodies. Unfortunately, such data is difficult to gather, and we are aware of no peer-reviewed studies on the matter. One of the difficulties in obtaining this kind of data is in the fact that regulatory actions and education campaigns, like many other non-structural stormwater management practices, involve a dimension of human behavior, which is complex, sometimes unpredictable, and oftentimes involves conflicting viewpoints [47]. Another difficulty in linking non-structural management practices to actual water quality impacts is that many of these types of practices target one specific behavior, such as fertilizer application laws, while nutrient sources to a receiving water body are numerous [38], making it difficult to parse out the effects of changes in just one source.

Structural Strategies to Manage Nutrients in Stormwater

Structural practices to manage stormwater nutrients rely on constructed features such as detention ponds, rain gardens, or permeable pavers. Structural practices may capture stormwater for later reuse, but they most often involve engineered structures that detain stormwater for some period of time and then slowly release it into a receiving water body or underlying soils—with the presumption that detaining the stormwater or infiltrating it through underlying soils allows for a degree of pollutant attenuation before release to surface or groundwater.

Structural stormwater management has traditionally focused on practices to mitigate peak flows and prevent flooding in urban areas [58]. Hence, conventional—and the most common—structural practices for managing stormwater are dry ponds and wet ponds, both of which are constructed landscape features meant to control floods and manage large volumes of stormwater flow (Table 2). As such, the primary design criteria of these features is not focused on nutrient removal, and their ability to attenuate stormwater nutrient loads is limited [17••, 59], and in some cases, they may even be net nutrient exporters [60]. Though they have been in design since the 1970s, alternative stormwater management practices such as those found in LID and GI principles have gained considerable usage in the last decade [45]. Alternative practices include green roofs, bioretention, and vegetated open channels, among others. Their goals are often to achieve natural hydrology and improved water quality by managing urban stormwater with site design features that are thought be functionally equivalent to the natural landscape. Both conventional and alternative stormwater management practices often rely on physical, chemical, and/or biological processes to sequester or transform N and P before they reach surface or groundwater bodies [58].

Stormwater Ponds

A review by Koch et al. [61] found that TN removal by dry ponds and wet ponds was 27 ± 23 and 40 ± 31%, respectively, showing that these structures are not only limited in their N removal capacity but they also demonstrate great variability. Similarly, P removal in a wet pond in Washington was 19 to 50% [62], indicating that P removal is also limited in conventional stormwater treatment ponds. Wet ponds have been credited with slightly better N and P removal capacity than dry ponds [58], with increased water residence time for settling of sediment-bound P and interaction with anaerobic zones that promote denitrification being necessary factors contributing to their improved performance [62, 63].

Low levels of nutrient removal by stormwater ponds is largely due to the fact that filtering and sedimentation are ineffective for removing dissolved N and P from the system [62, 64]. The primary means of N removal is denitrification. The denitrification process requires a carbon source and anaerobic conditions and has been shown to be particularly limited in dry ponds because of limited interaction of NO_3–N with anaerobic areas [64]. In the Morse et al. [64] study, only 1% of incoming NO_3–Nwas denitrified in a dry stormwater basin in which there was seldom standing water after rain events; in contrast, a dry basin that had begun to function as a wetland because of construction error or clogging and that had pools of standing water throughout the year was able to denitrify 58% of incoming NO_3–N. Concerning dissolved P, the most important removal mechanisms are plant uptake and adsorption to pond sediments. For example, Comings et al. [62] observed greater removal of dissolved P in newly constructed ponds and those with fresh sediments, presumably because the fresh sediments were not yet P-saturated. In their comparison of wet ponds, a newly constructed pond with new sediment removed 62% of dissolved P versus only 3% removal in a pond in which the stormwater had limited contact with fresh sediment.

In recent years, a growing body of literature is giving more attention to internal stormwater pond processes that cycle nutrients in these ponds [17••, 59, 63, 65]. These studies are shifting the focus from viewing ponds in terms of just inputs versus outputs to a better mechanistic understanding of the biogeochemical processes affecting their performance for nutrient removal. Williams et al. [59] observed a distinct autochthonous dissolved organic matter (DOM) signature in stormwater ponds, indicating that discharge from the ponds could affect downstream nutrient cycles by altering the qualitative properties of organic molecules containing N and/or P. Lusk and Toor [17••] observed a fourfold increase in DON biodegradability in stormwater pond outflow versus pond inflow, suggesting intense in-pond processing of DON and/or production of novel DON within the stormwater pond.

Conventional wet and dry ponds are found globally in almost all urban landscape types, but given their limited efficacy, alternative practices are increasingly considered as supplemental or replacement strategies. A summary of these, which include LID and GI practices, is provided in Table 2. While statistics on the inflow nutrient concentrations versus outflow concentrations for many of these practices are summarized in the 2016 report of the International Stormwater BMP Database [66], it should be noted that treatment efficiency in percent removal numbers are not reported. The absence of reporting on removal percentages is due to the fact that different BMPs attenuate different volumes of flow, resulting in variable pollutant *loadings* associated simply with differences in flow volumes, while studies of BMPs often rely on *concentration* data, making comparisons between BMPs and between sites difficult.

Bioretention Cells

Among the LID practices are bioretention cells (sometimes called rain gardens), which are landscaped depressions that capture stormwater runoff and allow it to infiltrate soils. Stormwater nutrients are treated by biorentention cells through plant uptake, incorporation into soil organic matter, adsorption to soil particles, and denitrification. While comparisons of N and P inflow versus outflow concentrations for bioretention often indicate net increases in export concentrations, it should be noted that bioretention cells can achieve upwards of 90% reductions in stormwater volume, thus effecting an overall nutrient mass load reduction [68••]. For example, a bioretention cell in Maryland achieved only 9% concentration reduction in TN but a 41% reduction in TN loading due largely to flow volume reduction [67].

One key factor in the efficacy of bioretention to treat stormwater nutrients is the type of soil media used [68••, 69]. Field and mesocosm studies have demonstrated that bioretention cells perform best for nutrient treatment when constructed with sand to sandy loam soils with clay content kept to 5–8%. Having at least that much clay is important for P removal, since clay can provide the aluminum and iron oxide binding sites needed for P sorption processes [70, 71]. A saturated zone for denitrification is also important in biorention media [67, 69, 71]. For example, Brown et al. [69] observed that the majority of N flowing in to a North Carolina bioretention cell was in the organic form. This organic N was quickly converted to NO_3–N through mineralization and nitrification, and without a saturated soil zone to promote denitrification, the NO_3–N was exported from the system. In this way the nutrient forms exported from a conventional bioretention cell may not align with the nutrient forms entering the cell, but instead be indicative of internal nutrient cycling. Modified bioretention cells may include an internal water storage zone via an elevated underdrain outlet to promote increased N removal through denitrification [72].

Grass Swales

Grass swales are vegetated open channels that convey stormwater but that can also be used to filter nutrients and promote stormwater infiltration. Summary statistics from the 2016 International Stormwater BMP Database show that the outflows of grass swales generally have the same or higher dissolved N and dissolved P concentrations compared to the inflows [66]. However, opportunities for nutrient removal in swales can be enhanced by dense vegetation with a well-developed root system [73, 74], but this may be limited in the winter when plants are dormant and not taking up nutrients. Most recent research on grass swales has focused on their utility as part of a treatment train, in which they are a preliminary treatment step by allowing for sedimentation of particle-bound nutrients before stormwater enters another BMP type [75, 76]. Kachchu et al. [75] observed that 50 to 75% of stormwater sediments were removed in the first 10 m of a swale, allowing not only reduced sediment loading to a downstream BMP but also a reduced loading of nutrients associated with the sediment.

Wetlands

Wetlands are another type of alternative stormwater treatment that removes nutrients through plant uptake and biogeochemical transformations such as denitrification. Wetlands used for stormwater treatment may be constructed, natural, or incidental (resulting from previous development). Because wetlands typically contain both aerated and anaerobic sites, they are thought to be well-suited for N removal via nitrification and denitrification, as long as there is adequate carbon (C/N ratio of at least 5:1) and water temperatures are between 5 and 60 °C [58]. Though stormwater wetlands have been used and studied since the 1980s, in the last 5 years, a growing body of literature has focused on the use of floating treatment wetlands (FTW) to retrofit wet ponds. This emerging tool for stormwater treatment uses a floating surface that provides a growing substrate for hydrophilic plants that then take up nutrients from the pond. The systems are grown hydroponically with roots submerged in the water but plant stems above the water's surface [77–79]. While N and P removal rates of approximately 50% are reported for FTW [79], White and Cousins [78] observed that most N and P were removed by biomass below the growing mat; thus, whole plant removal—not just shoot removal—may be necessary to permanently remove nutrients from the pond. These authors add that FTW may be particularly well-suited as a downstream BMP to polish water to extremely low P levels.

Permeable Pavers

Permeable pavers can be used as alternatives to conventional impervious surfaces in urban areas and are especially useful when available land is limited because they do not require an increase in land area. Because they allow infiltration of stormwater onsite, they allow for more natural hydrological functioning. While permeable pavers have been shown to be highly effective in reducing runoff volumes, they have mixed performance efficiencies for nutrient removal [80]. A parking lot study with permeable pavers in Ontario, Canada, showed significantly reduced concentrations of NH_4–N and DON but increased NO_3–N concentrations, indicating that nitrification was occurring in the permeable paver system. Since these systems are designed to be free-draining, opportunities for denitrification in anaerobic zones are limited [81]. It was also noted in Mullaney and Lucke [82] study that the permeable pavers trapped PON, which may later mineralize and remobilize. Pretreatment of stormwater to allow removal of particulates before interaction with permeable pavers can prevent clogging and trapping of particulate nutrients [83]. Alternatively, periodic vacuuming of pavers to remove trapped sediments can increase the pavers' longevity and treatment effectiveness [84, 85]. Increased N and P removal can be achieved in permeable paver systems by using partial infiltration to detain water for over 24 h and encourage denitrification or through the use of iron oxide materials that promote P sorption under the paver base [86].

Green Roofs

A final LID practice considered here are green roofs, which are vegetated rooftops designed to capture rainfall and prevent runoff in the first place. Made with special lightweight growing mediums, they may be grown only with low-growing plants or may be more intensive with shrubs and even trees. Green roofs capture 20–100% of incoming rainfall, but this number has been shown to decrease as rainfall amount increases [87]. Green roofs have been shown to have little to no effect on reducing nutrient concentrations in runoff, primarily because they usually do have fertilizer inputs of N and P [87]. Fertilization during plant establishment can contribute nutrients in green roof runoff [88, 89]. For example, in the study conducted in Connecticut by Gregoire and Clausen [89], greater mean concentrations of TP (range 0.018–0.096 mg/L) and PO_4–P (range 0.003–0.079 mg/L) were observed in green roof runoff, suggesting that sources of P in runoff was presumably attributed to the growing media and fertilizer. Bliss et al. [88] and Berndtsson [90] also concluded that the use of P fertilizers likely causes elevated levels of P in green roof runoff.

The effectiveness of green roofs for stormwater mitigation can be affected by vegetation type, amount of vegetative cover, plant health, climate, and the water use efficiency of the plants [91–93]. Sims et al. [91] observed that a green roof in a semi-arid climate could retain more stormwater volume than green roofs in humid climates, likely because of lower antecedent soil moisture in the drier climate. Likewise, Volder and Dvorak [94] observed that the substrate volumetric water content was an important regulating factor in rainfall retention, with drier soils holding more water and producing less runoff. While succulents are the most common type of green roof vegetation because they are well adapted to dry conditions, almost any plant type can be used for green roofs. If food crops are used, the benefits of food production must be weighed against the use of N and P fertilizers that may increase the nutrient loading of green roof runoff [92].

Conclusions and Research Needs

Based on our review on the occurrence, sources, and transport processes of N and P in urban stormwater runoff and strategies to manage nutrients in stormwater, the following conclusions can be drawn: (1) There are numerous sources and pathways of nutrients in urban stormwater, underlying the fact that runoff likely contains nutrients from a mixture of potential sources and that management strategies must account for multiple pollutant sources. (2) There are multiple hydrological and biogeochemical processes that cause variability in stormwater nutrient concentrations temporally and spatially. (3) Current research trends are focusing on elucidating the mechanisms that control nutrient cycling in BMP structures such as ponds and bioretention cells; this research helps us go beyond simple input versus output characterizations and instead characterize internal biogeochemical processing.

We suggest the following recommendations for further research: Relatively few studies have identified P sources in urban stormwater runoff which requires fully developed methods for source identification. Information on different species of N and P in stormwater runoff is also needed to provide better understanding of nutrient cycling and fate of N and P in urban environments. We especially need to continue recent research on biogeochemical processes internal to BMP systems, such as treatment ponds and bioretention cells. This will enable a better mechanistic understanding of the controls on nutrient exports from BMPs to downstream waters. Along these lines, there is also a need to verify and compare BMP efficacies across BMP types and across environments. One thing that will aid in doing this is to investigate and report the flow weighted fluxes of nutrients in BMP inflows and outflows, enabling a more accurate comparison tool than concentrations alone, which do not account for differences in runoff volumes among BMP types. Studies on the interactions between hydrological and biogeochemical processes over long periods of time are also called for, as this would allow understanding of if and how BMP performance changes with time in response to climate and land use changes.

We also recommend research on the role of plant uptake of nutrients in BMP systems. In particular, we need information on how the littoral zones of conventional ponds can be planted to increase nutrient removal in the ponds. This will require work on the types of plants that are most suitable and understandings of how to maintain pond plantings over the long term. Finally, we recommend efforts be made to better understand the impacts of non-structural stormwater management strategies on water quality and to determine when and how efforts such as fertilizer bans and outreach campaigns deliver actual improvements on water.

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Compliance with Ethical Standards

Conflict of Interest The authors declare no conflict of interest.

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•• Of major importance

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