American Ecological Engineering Society

Journal of Ecological Engineering Design
journals.uvm.edu/jeed

Research Paper

Coastal Stormwater Pond Age and Phosphorus Cycling within Water and Sediment

Morolake M. Fatunmbi[1], Debabrata Sahoo[2], Amy E. Scaroni[3], Calvin B. Sawyer[4], Erik Smith[5], Sara A. White[1]

[1]Department of Plant and Environmental Science, Clemson University, Clemson, South Carolina, United States

[2]Department of Agricultural Sciences, Clemson University, South Carolina Water Resources Center, Pendleton, South Carolina, United States

[3]Department of Forestry and Environmental Conservation, Clemson University, Baruch Institute of Coastal Ecology and Forest Science, Georgetown, South Carolina, United States

[4]Department of Agricultural Sciences, Clemson University, Clemson, South Carolina, United States

[5]School of the Earth, Ocean and Environment, University of South Carolina, Baruch Field Laboratory, Georgetown, South Carolina, United States

Correspondence
Debabrata Sahoo
Department of Agricultural Sciences
Clemson University
509 Westinghouse Road
Pendleton, SC 29670, USA

Email: dsahoo@clemson.edu

Received
March 3, 2025

Accepted
August 20, 2025

Published
December 23, 2025

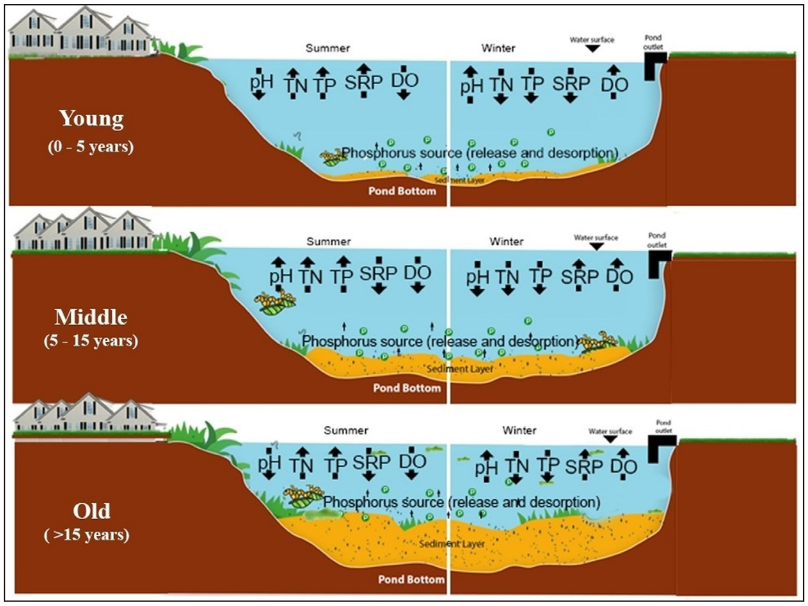
Editors
Sara W. McMillan, Editor in Chief
Mauricio E. Arias, Associate Editor

Effective management of coastal stormwater ponds (SWPs) requires an understanding of how pond age affects nutrient dynamics, particularly in reducing eutrophication and regulating nutrient loading to downstream waterbodies. The objectives of our study were to (1) assess the impact of pond age on water quality and (2) evaluate whether pond sediments behave as a phosphorus (P) source or sink. Nine SWPs in residential areas of Charleston, South Carolina, United States, were studied and categorized into 3 distinct age groups: Young (0 years to 5 years), Middle-age (5 years to 15 years), and Old (>15 years). Water samples and sediment cores were collected from 3 different sections of each SWP in summer 2023 and winter 2024 to assess temporal variation in water quality. Water quality and sediment characteristics were analyzed, with a focus on the estimation of equilibrium P concentration (EPC_0) to determine if sediment behaved as a P source or a sink. Water quality varies across pond ages. Old SWPs had higher total P and nitrogen levels, indicating accumulations of nutrients over time, while lower nutrient levels measured in Middle-age SWPs showed indications of enhanced nutrient sediment processing. EPC_0 measurements showed sediments in all ponds served as a source of P, releasing P back into the water column, contributing to internal P loading. These findings highlight the need for targeted management strategies to address P release from sediment to maintain water quality in coastal stormwater ponds.

Keywords Equilibrium phosphorus concentration; pond age; sediment; stormwater ponds; water quality

Fatunmbi MM, Sahoo D, Scaroni AE, Sawyer CB, Smith E, White SA. 2025. Coastal stormwater pond age and phosphorus cycling within water and sediment. J Ecol Eng Des. https://doi.org/10.70793/jeed.26

Graphic Abstract



1. Introduction

The global population is currently on the rise and is projected to reach approximately 9.5 billion by 2050, accompanied by increased and expanding urbanization in both developed and developing countries (Sun et al 2020). The construction of paved roads, driveways, and rooftops, commonly associated with urbanization, alters the natural landscape and affects the quantity and quality of freshwater (Gaffield et al 2003). Modification of land cover through the removal of vegetation and increased impervious surface areas leads to reduced infiltration, resulting in larger runoff volumes and contaminant loads in urban waterbodies, as well as decreased groundwater recharge (Beckingham et al 2019; McKercher et al). Urban expansion also compresses the time to peak flow, increases flashiness, and increases peak flow volumes in urban water bodies, making them more susceptible to flooding (Miller et al 2014).

Although the installation of parks, lawns, and gardens can provide relief from flooding and water quality benefits by reducing impervious surfaces and allowing infiltration of stormwater, the excessive use of fertilizers, pesticides, and other chemicals in these green spaces can potentially negate some benefits and contribute to water pollution downstream (Law et al 2004; Garn 2002). Generally, the quality of stormwater runoff is greatly influenced by land use characteristics and various hydrological factors (Sønderup et al 2016). Surface runoff can transport contaminants from various sources into adjacent coastal waterbodies, contributing to water quality degradation, including high bacterial levels and chemical and nutrient contamination (Serrano and DeLorenzo 2008; Yang et al 2020).

Stormwater ponds (SWPs) are a type of stormwater control measure commonly constructed to mitigate the impacts of impervious landscapes on surrounding water bodies (Sinclair et al 2020). SWPs are constructed as either dry ponds, which capture and temporarily store stormwater, slowly releasing it downstream, or as wet ponds that maintain a permanent pool of stormwater and gradually release it. In addition to retaining water, wet ponds also capture nutrients and contaminants carried by stormwater, serving as a buffer for other downstream water bodies such as lakes, wetlands, and estuaries (Tixier et al 2011; Scaroni et al 2021; McKercher et al 2024). SWPs are constructed to mimic some of the ecological services that natural wetlands provide, such as flood risk reduction and water filtration, but do not necessarily offer the same ecological benefits as natural wetlands (Perron and Pick 2020; Sinclair et al 2020). In coastal environments, where land use developments constantly change and intense storm events are common, SWPs are installed to help delay the time to peak flow and reduce nutrient loading to downstream aquatic systems by capturing and temporarily storing stormwater (Lusk et al 2025; Schroer et al 2018). These SWPs facilitate the settling of suspended solids and promote biogeochemical processes that reduce pollutant concentrations (Yang and Lusk 2018). In South Carolina, United States, SWPs are constructed not only to control flooding but are also mandated to comply with regulatory requirements for land alteration and development activities. These engineered structures are designed to (1) ensure that post-development runoff matches pre-development levels and (2) capture and retain initial runoff within the site, which provides water quality benefits primarily through sedimentation (Dickes et al 2016).

One of the major challenges to SWP function is the accumulation of sediment organic matter and high nutrient loads, such as nitrogen (N) and phosphorus (P) from runoff, resulting in eutrophication that can cause increased primary productivity, potentially leading to a harmful algae bloom (HAB), also known as harmful cyanobacterial bloom, within the SWP (Schroer et al 2018; Song et al 2017). Algal blooms impair environmental sustainability, diminish the aesthetic appeal of the area, and negatively impact property values (Sanseverino et al 2016).

Fatunmbi MM, Sahoo D, Scaroni AE, Sawyer CB, Smith E, White SA. 2025. Coastal stormwater pond age and phosphorus cycling within water and sediment. J Ecol Eng Des. https://doi.org/10.70793/jeed.26

Highlight
Older stormwater ponds exhibited higher total nitrogen and phosphorus concentrations, and sediment served as a source of phosphorus, depending upon sediment particle size and soluble reactive phosphorus.

P is a key nutrient of concern in SWPs and is primarily classified into organic and inorganic forms, which are further categorized into particulate and dissolved fractions (Reddy et al 1999). Particulate P is typically bound to sediments or organic matter and is mostly unavailable for plant uptake, while the dissolved P is the form readily available for plant and algal uptakes (Lin et al 2018; Tipping et al 2014). Although P can exist in forms and fractions that are initially unavailable for use by plants or aquatic organisms, changes in environmental conditions such as redox changes, pH shifts and microbial activities, can cause transformation in P form into bioavailable P (Hupfer and Lewandowski 2008; Lusk and Chapman 2021; Zhou et al 2005), which can be remobilized and re-released back into water bodies leading to internal P load in SWP (Frost et al 2019). Phosphate (PO_4-P) is often considered a significant concern for surface runoff due to its binding affinity with soil iron and calcium, which limits its leaching potential into groundwater (Hartshorn et al 2016). However, it can be transported in particulate form, attached to eroded soil, and released into water bodies under changes in environmental conditions that break its binding. Sediment within SWPs can serve as both a source and a sink of P to downstream water bodies. Duan et al (2016) suggested that SWP sediment exhibited a high rate of N retention, but with less efficiency in removing P from stormwater, due to differences in their biogeochemical cycling. In contrast, Schroer et al (2018) suggested that sediment in SWPs can efficiently trap P carried in stormwater. While N can be permanently removed via denitrification, soluble reactive P (SRP), a form of dissolved inorganic P, can be retained through biological uptake, mineral precipitation, or sorption onto surfaces in suspended particles and bed sediments that can be buried and retained within water systems (Duan et al 2016). However, during certain hydrologic conditions, the retained SRP may be remobilized with particulate phosphorus or desorbed from particles under anaerobic conditions.

Effective management of SWPs to remove total P (TP) depends on several factors (Janke et al 2021). For instance, soil texture can affect P binding capacity, with finer particle sizes offering a greater surface area for sorption than coarser sandy sediments (Zhou et al 2005). SWP management involves maintaining a long hydraulic residence time to allow particles to settle, regularly removing deposited sediments to maintain storage capacity, and implementing designs that prevent sediment resuspension caused by incoming runoff (Schroer et al 2018). The inconsistent nutrient cycling dynamics within SWPs are shifting research towards understanding the internal nutrient cycling of SWPs and focusing efforts on quantifying how biogeochemical processes regulate nutrient retention and release.

Identifying the role, connection, and impact of residential land use on P dynamics in SWPs is essential for understanding algae bloom dynamics (Frost et al 2019). Earlier studies conducted on SWPs showed that sediment regulates P dynamics and SWPs may

Study/Project Photographs. Sampling activity showing the process of using a cup sampler (left) and a sediment borer (right) for water and sediment sample collection.

buffer increasing P loads (Frost et al 2019; Taguchi et al 2020; Janke et al 2021). However, as SWPs age, their physical structure, biotic community, and biogeochemical processes will begin to change, potentially altering sediment capacity to retain or release P (Taguchi et al 2020). Despite its importance, very little is known about how pond age influences sediment characteristics, water quality, or P-cycling dynamics within SWPs, particularly in coastal environments (Kaye et al 2006; Mallin et al 2002; Schroer et al 2018; Song et al 2013). Coastal SWPs are particularly vulnerable to nutrient enrichment due to their shallow groundwater tables, tidal influence, and proximity to developed areas (Beckingham et al 2019). The objectives of this study were to (1) assess how the age of coastal residential SWP influences internal sediment P dynamics and (2) determine whether benthic sediment acts as a source or sink for P in the water column. We hypothesize that, over time, sediment P sorption capacity is likely to decrease due to increased P loading and reduced sediment input from the watershed (e.g., a neighborhood or housing community), leading to a diminished capacity for P binding in the sediment and potential for sediment to become a net source of P.

2. Study Area

The study was conducted in Charleston County, South Carolina (SC), United States (Figure 1). Sediment and water samples were collected from 9 coastal SWPs, each with varying dates of construction, situated within ~21 miles of the SC coastline. While most of the SWPs fell under the jurisdiction of Charleston County, one SWP fell under the jurisdiction of the City of Charleston. All SWPs were located within residential communities and owned by the local Homeowners Association, where the primary land use is developed areas. Data provided by Charleston County stormwater management, along with ArcGIS software, were used to identify the inlet and outlet structures as well as to confirm that the 9 SWPs are

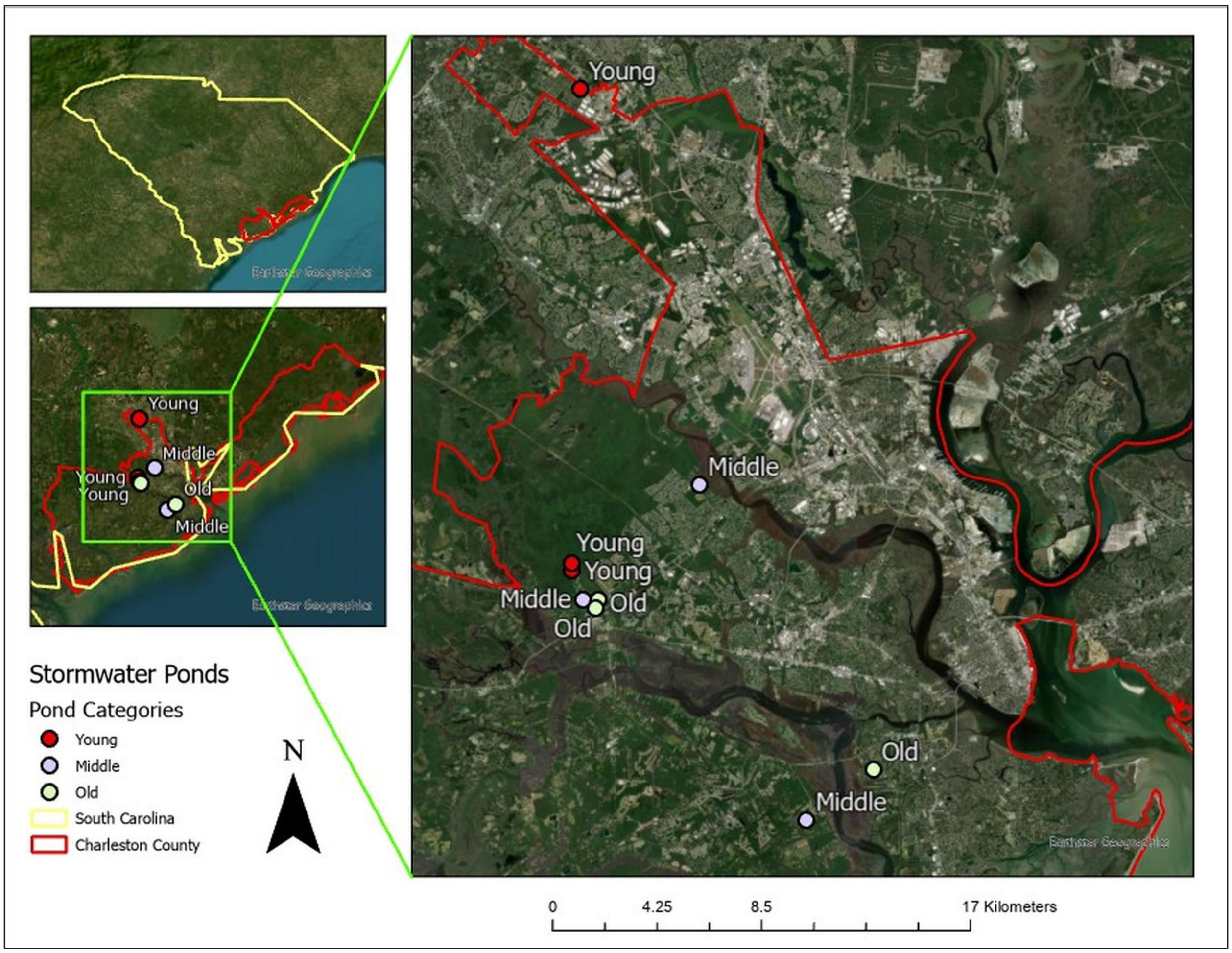


Fig. 1. Charleston map showing the selected 9 coastal stormwater ponds, between 2 years of age and 15 years of age, all located within residential areas. (Created using ArcGIS software, ESRI Earthstar geographics basemap)

"first order ponds," meaning SWPs that directly receive stormwater runoff from the surrounding watershed.

Based on the study's objective, with a base year of 2021, the selected SWPs were categorized into 3 age categories: Young (0-5 years), Middle-age (5-15 years), and Old (>15 years) (Table 1). The earliest construction year of SWPs selected was 2021. No SWP older than 1990 was selected as most SWPs relevant to the Municipal Separate Storm Sewer System (MS4) were brought online after the implementation of the National Pollution Discharge Elimination System (NPDES) by the United States Environmental Protection Agency (U.S. EPA) in 1984 (SCDHEC 2003).

3. Methodology

3.1. Water Sampling and Analysis

Field samplings were carried out in summer between 24 July and 26 July, 2023, and in winter between 3 January and 4 January, 2024, to capture the temporal variation in water quality and sediment-P dynamics. Each SWP was divided into 3 sections (inlet, midpoint, and outlet) where samplings were carried out to account for spatial variability (Kermorvant et al 2019). Physicochemical parameters such as pH, dissolved oxygen (DO), temperature, turbidity, chlorophyll a, phycocyanin, and specific conductivity were measured in situ from each section using a Xylem digital proDSS YSI water quality meter (YSI Inc, Yellow Springs, Ohio, United States).

A jon boat was used to access the sites within each SWP. From each pond section, subsurface water samples were collected from the top 10 cm into 3, 250 mL HDPE sample bottles, using the grab sampling method for analysis of water quality indicators and nutrient levels following the protocols prescribed by the Arkansas Water Resources Center. All the samples were stored on ice in an ice chest until returned to the lab. Some samples were shipped to the Arkansas Water Resources Center for water quality analysis (Daniels et al 2018). Filtered (0.45 µm pore size EZflow glass fiber filters) and acidified (sulfuric acid, pH <2) water samples were analyzed for ammonia-N (NH_4-N) (EPA 351.2 method [O'Dell 1996]), nitrate and nitrite (NO_3+NO_2-N) (EPA 353.2 method [EPA 1993a]), and SRP (EPA 365.1 method [EPA 1993b]) using a Skalar San++® continuous flow analyzer (Skalar Analytical, The Netherlands). Unfiltered water samples were analyzed for TP and TN following EPA 365.1 and 353.2 standard methods using a Skalar San++ continuous flow analyzer. Chloride (Cl^-), fluoride (F^-), and sulfate (SO_4^{2-}) were anlyzed using filtered unacidified water sampling following EPA 300.0 (Pfaff 1996) metods on Dionex ICS-1600 Ion Chromatography System (Thermo Fisher Scientific Inc, Waltham, Massachusetts, United States). An additional 2 L of water sample was collected from the same depth to carry out laboratory EPC_0 analysis.

Table 1 Description and characteristics of selected coastal stormwater ponds

Label	Year constructed	Age (year)	Inlets (#)	Depth (m)	Drainage area (m^2)	Pond surface area (m^2)	Drainage area: Pond area	Average lot size (m^2)	Impervious area (%)	Pervious area (%)	Residence type
Y1	2018	5	1	0.9	21830	535	40.8	875	38	62	SFR
Y2	2018	5	2	0.7	32945	500	65.9	1105	35	65	SFR
Y3	2019	4	2	1.2	25730	1840	14.0	8450	53	47	AC
M1	2010	13	2	0.9	12780	640	20.0	435	30	70	SFR
M2	2015	8	5	0.8	32645	3580	9.1	375	46	54	SFR
M3	2014	9	4	0.8	34840	2370	14.7	760	42	58	SFR
O1	2003	20	2	1.0	30035	475	63.2	780	35	65	SFR
O2	2001	22	2	1.1	34050	1675	20.3	1095	36	64	SFR
O3	2003	20	1	0.6	22090	595	37.1	1030	27	73	SFR
Median of each variable by pond age											
Y	**-**	**5**	**2**	**0.9**	**25730**	**535**	**40.8**	**1105**	**38**	**62**	**-**
M	**-**	**9**	**4**	**0.8**	**32645**	**2370**	**14.7**	**435**	**42**	**58**	**-**
O	**-**	**20**	**2**	**1.0**	**30035**	**595**	**37.1**	**1030**	**35**	**65**	**-**

Y – Young, M – Middle-age, O – Old
SFR – Single Family Residence, AC – Apartment Complex

3.2. Sediment Sampling and Laboratory Procedures

A universal sediment corer (Aquatic Research Instrument, Hope, Idaho, United States) was deployed from a jon boat at each section of the pond to collect the top 5 cm benthic sediment sample. Each sample was then placed into a Ziploc bag and stored in an ice chest alongside water samples collected. Sediment samples were refrigerated at 4 °C until they were analyzed.

To prepare phosphate stock solution for sediment P interaction, 2, 1-liter water samples were also collected from each SWP section, filtered through a United Scientific FG5340-1000 vacuum assembly using a Pall corporation 1 µm pore size 37 mm diameter A/E glass fiber filter and a Rocker 300 vacuum filtration pump, and stored in a clean pair of 1-liter bottles. Sediment samples from each pond section were sieved using a 4.75 mm sieve into a pre-labeled container. Wet sediment (25 g) was sieved, weighed, and transferred into a pre-labeled 250 mL Erlenmeyer flask. The EPC_0 method, as outlined by Haggard et al (2004), was used to estimate EPC_0 at concentrations of 0 mg, 0.05 mg, 0.2 mg, 0.5 mg, 1 mg, and 2 mg PO_4-P/L. EPC_0 is described as the concentration of dissolved P in the water at which there is no net exchange of P between the sediment and overlaying water column, which means the point at which sediments neither adsorb nor release P (Erickson et al 2004).

A prepared stock solution (0 mL, 2.5 mL, 5 mL, 25 mL, 50 mL, and 100 mL) was added to each flask already containing 25 g of sediment, respectively. Then, 100 mL of filtered pond water sample was added to the wet sediment. For proper mixing, the sediment slurry was placed on an analog orbital shaker table (VWR, Radnor, Pennsylvania, United States) and continuously shaken for an hour at 250 rpm.

After proper shaking, the shaker table was stopped, and the sediment slurry was allowed to settle for one hour. The water sample was then filtered and collected in a 60 mL HDPE bottle. A filtered water sample (60 mL) was also collected in an HDPE bottle to estimate ambient PO_4-P concentration. All samples were stored in an ice chest (packed with ice) and transported to the AWRC lab to estimate SRP and TP. The remaining sediment solution in the flask was poured into a pre-labeled, pre-weighed aluminum pan and placed in the oven to dry at 80 °C for 48 hours. The weight of the dried sediment was measured and used to estimate EPC_0. A subsample of dried sediment was further used to determine the organic carbon content using the loss-on-ignition method (Heiri et al 1999) and to perform particle size analysis following the standard test method for particle size distribution of soil using sieve analysis (Hossain et al 2022).

3.3. Equilibrium Phosphorus Concentrations Analysis

Sediment EPC_0 concentration was estimated as the x-intercept of the regression line with initial PO_4-P concentrations of the filtered pond water (Figure 2). The ambient PO_4-P concentration plus additional 0 mg, 0.05 mg, 0.2 mg, 0.5 mg, 1 mg, and 2 mg PO_4-P/L was the x-variable, and the concentration of P sorbed to the sediment (mg P/kg dry sediment) was the y-variable (Haggard et al 2004).

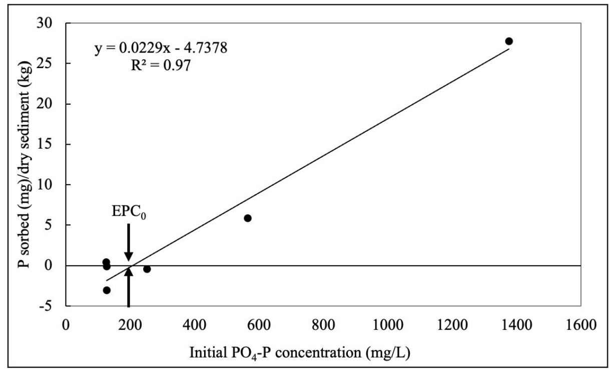


Fig. 2. An example of linear regression used to estimate equilibrium phosphorus concentration (EPC_0) in coastal South Carolina stormwater pond sediments.

P sorbed (P_s) to sediment was calculated as follows:

$$\frac{(AC)+(TLC(mg/L))}{TV} = TC\ (mg/L) \qquad \text{Equation 1}$$

$$TC(mg/L) - WC\ (mg/L) = P_s\ (mg/L) \qquad \text{Equation 2}$$

$$\frac{P_s\ (mg/L)\times 0.1L}{1000} = P_s(\mu g/L) \qquad \text{Equation 3}$$

EPC_0 was estimated as follows:

$$\frac{y\ intercept}{X\ variable} = \frac{P_s(\mu g/L)/DW_s\ (g)}{IC\ (\mu g/L)} \qquad \text{Equation 4}$$

Defined as:

- AC – water sample P ambient concentration (i.e, background concentration of the pond water sample)
- TLC – PO_4-P varying concentrations (treatment levels, e.g, 0.05 mg, 0.2 mg, 0.5 mg PO_4-P/L)
- TC – total concentration in sediment slurry
- WC – slurry water column P concentrations
- IC – initial P concentration
- P_s – P sorbed to sediment
- DW_s – dry weight of sediment

3.4. Data Analysis

Statistical analyses were performed to examine spatial and temporal variations in water column physicochemical properties across pond age categories. Each SWP was sampled at 3 locations (inlet, Middle, and outlet), providing a total of 9 measurements per pond age per season. The influence of pond age and season was determined using a Wilcoxon/Kruskal-Wallis test for measured parameters ($\alpha = 0.05$). The Redfield ratio, a predictive tool for identifying nutrient limitations that may lead to eutrophication in aquatic environments (Redfield 1958), was calculated based on molar concentration to determine the limiting TN and TP in each pond age category. To evaluate sediment P dynamics in SWPs, the mean EPC_0 in each sampled pond was compared with the mean SRP in the water column. The Kruskal-Wallis test was applied to identify differences in EPC_0 across pond age categories ($\alpha = 0.05$). Further, correlation analyses were completed among water chemistry characteristics, sediment particle size, organic carbon, and EPC_0. A linear regression model was constructed to assess the effects of sediment characteristics and SRP on EPC_0 for each age category of ponds. All statistical analyses were performed using JMP Pro 17.0 (SAS Institute Inc, Cary, North Carolina, United States).

4. Results and Discussion

4.1. Water Column Physicochemical Characteristics

Conductivity measurements taken at the third Old SWP (O3) were higher than measurements at all other SWPs sampled, with readings over 10000 µs/cm during winter sampling and over 5000µs/cm during the summer. These high conductivity values indicated the likelihood of saltwater intrusion from tidal sources. Similarly, the Cl^- concentration in O3 was significantly higher (1442 mg/L – 5823 mg/L) than for all other SWP categories ($p = 0.0067$ and 0.0072, respectively). As such, O3 values were considered outliers and excluded from the water quality assessment to avoid skewing the results and ensure an accurate representation of conditions measured in the Old SWP category (n = 2 going forward).

Physicochemical parameters were similar ($p > 0.05$) across all pond sections in all pond categories (age), indicating that sampling any section of the SWP will likely provide a representative measure of overall water quality. Further analysis of the physicochemical parameters with respect to pond age and between seasons indicated differences in some measured physicochemical parameters (Figure 3). Water temperature was similar in all SWP categories ($p = 0.8$), but differed by sampling season, with temperatures ranging from 30 ºC to 32 ºC in summer and 9 ºC to 11 ºC in winter, as expected (Figure 3a). Past studies suggest that pond depth, surrounding landscape, and especially seasonal differences are factors that influence water temperature (Nelson and Palmer 2007). Pond depth particularly affects how atmospheric conditions influence the water body. Shallow ponds tend to be warmer and cool more rapidly than deeper ponds, which are less affected by daily temperature fluctuations due to the surface area-to-volume ratio (Nelson and Palmer 2007). However, deeper ponds may develop thermal stratification with warmer water at the surface and cooler layers below (Song et al 2013). P cycling in SWP can be influenced by variation in temperature. Warmer surface temperatures in shallow ponds can increase microbial activity, mineralization of organic matter, and aid in the release of P from sediments (Wu et al 2014). In stratified ponds, the bottom layers of the water column may become anoxic, which can enhance the dissolution of iron-bound P, leading to internal P loading from sediments (McEnroe et al 2013; Wang et al 2024).

DO differed by age ($p = 0.006$) and season ($p < 0.0001$) (Figure 3b). Several interrelated factors influence seasonal changes in DO; the most pertinent is the inverse relationship between DO and temperature, as colder water holds higher DO concentrations than warm water. Photosynthetic activity by algae can also alter DO, with longer days and higher light intensity in summer promoting increased DO in the water column (He et al 2011), which could further be depleted and assimilated by microorganisms present in water (Gold et al 2017). The average pH was within the standard freshwater recommendation for the SC coastal region (6.5 - 8.5; SCDHEC 2003) across both sampling periods. pH did not vary between summer and winter ($p = 0.5$), but did differ by pond age ($p = 0.0001$), with Middle-age ponds having the highest pH-values reported in summer but not winter (Figure 3c), potentially indicating increased biological activities with changes in buffering capacities in SWPs during warmer months (Wu et al 2014). Variation in pH can influence P dynamics in SWPs by affecting the forms, mobility, and bioavailability of P. When the pH is near neutral, P often remains bound to iron, aluminum, or calcium compounds in sediments. However, under alkaline conditions, the binding strength of P to these compounds decreases, subsequently increasing the chances of P desorption into the water column. Extremely low pH (acidic conditions) also increases the chances of P mobility into the water column, as these conditions dissolve the bonds between P and binding compounds in the sediments (Jin et al 2006; Wu et al 2014; Zhao et al 2022).

Specific conductivity did not vary by season but did vary by SWP age (p = 0.01), with Middle-age SWP specific conductivity lower than that of Young or Old SWPs (Figure 3d). The accumulation of dissolved solids from various sources, evaporative concentration, and reduced flushing and renewal can contribute to higher specific conductivity in older ponds compared to younger ones (Hasibuan et al 2023). In contrast, Middle-age SWP may represent a transitional phase where flushing processes and vegetation uptake help regulate and improve concentrations. Turbidity was similar across pond age (p = 0.06) and between sampling periods (p = 0.4) (Figure 3e). This suggests that turbidity in the SWPs may be influenced by short-term disturbances, such as storm events and specific biological activities, rather than long-term factors including pond age and seasonal variations. Other factors that could most likely influence turbidity in coastal SWPs may include site-specific variables such as land use changes, sediment loads, and organic matter in runoff, nutrient levels in SWPs that promote algal growth, and activities around the pond banks, such as the presence of surrounding vegetation (Hoess and Geist 2021).

Chlorophyll-a concentrations were similar across SWP age (p = 0.8) but varied by season (p < 0.0001). Chlorophyll-a concentrations in summer were higher (Figure 3f), likely driven by higher photosynthetic activity (Carneiro et al 2024). Similarly, phycocyanin concentration, an indicator of cyanobacterial abundance, varied seasonally (p <0.0001), with higher concentrations observed during the summer. Gold et al (2021) noted that higher phycocyanin concentrations in the summer

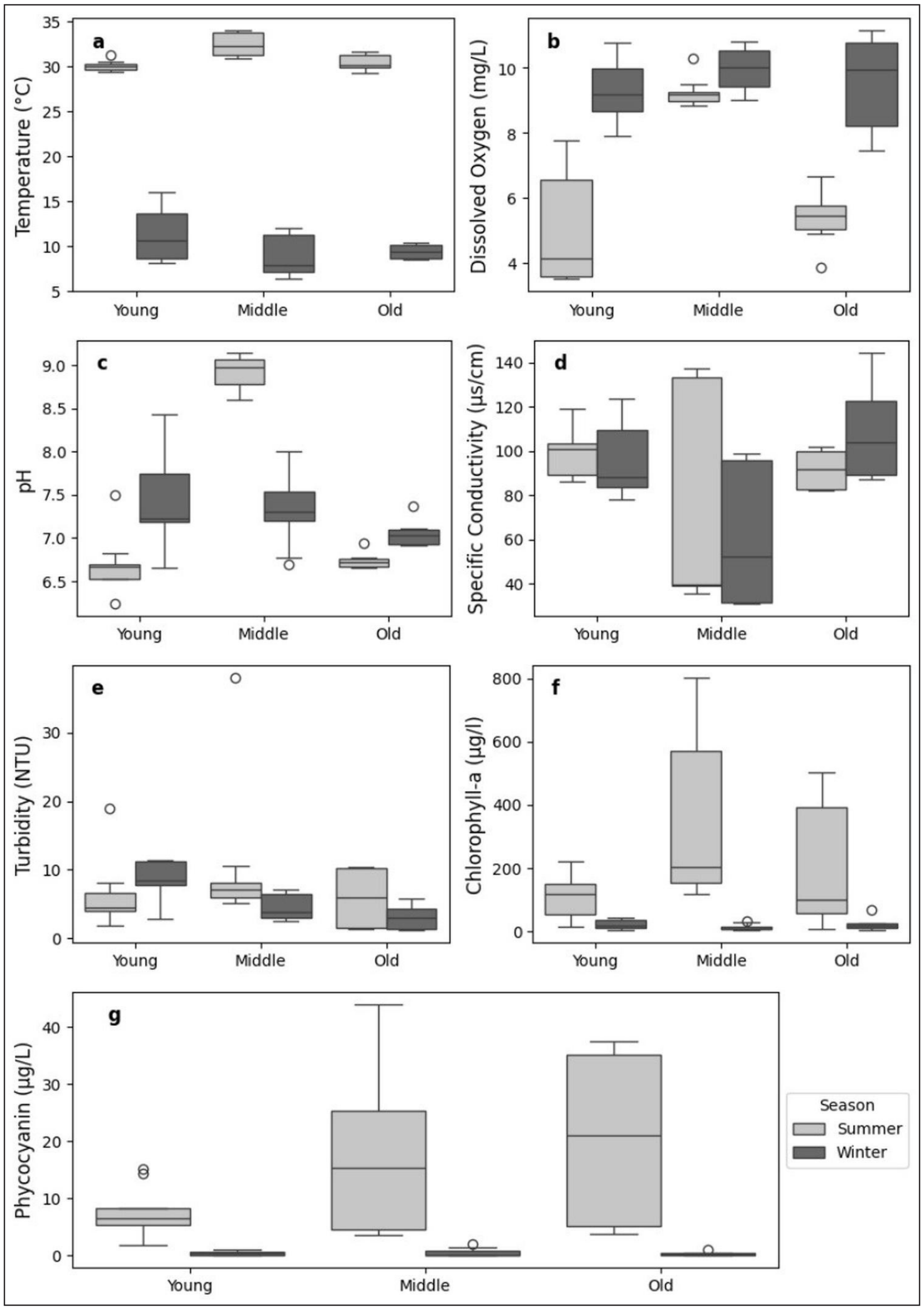


Fig. 3. Box plots of physicochemical measurements in stormwater pond water columns, categorized by pond age and sampling season. For each box plot in the Young and Middle-age category, n = 9, representing 3 ponds * 3 measurements (inlet, middle, and outlet). For the Old category box plots, n = 6, representing 2 ponds * 3 measurements (inlet, middle, and outlet).

could be attributed to warmer water temperatures and increased light intensity, which promoted the growth of cyanobacteria, particularly in Old SWPs, which typically have higher accumulations of organic matter, potentially supporting cyanobacterial growth (Gold et al 2021).

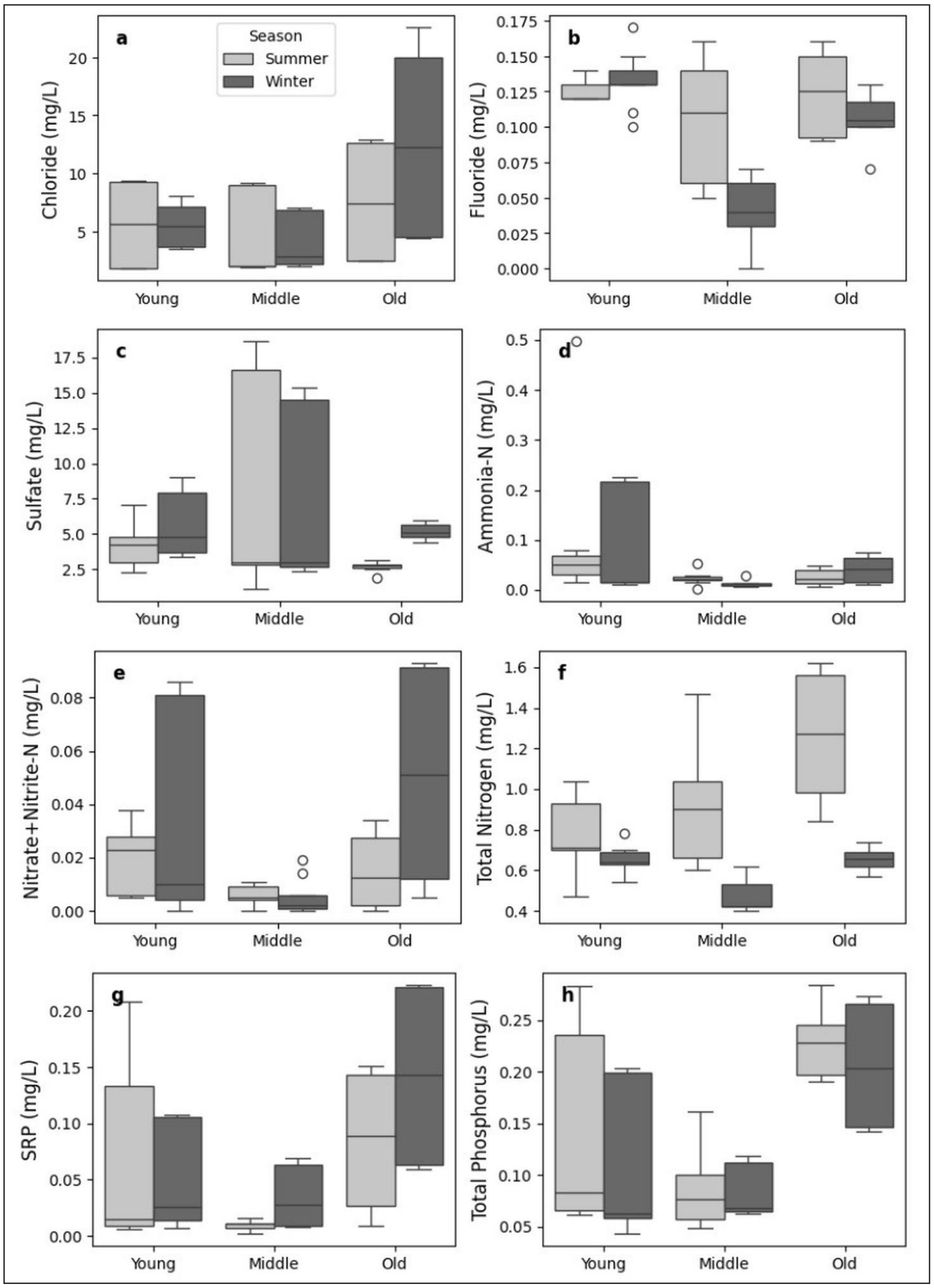


Fig. 4. Box plots showing water quality data of (a) Chloride, (b) Fluoride, (c) Sulfate, (d) Ammonia-N, (e) Nitrate + Nitrite – N, (f) Total Nitrogen (g) SRP (soluble reactive phosphorus), and (h) Total Phosphorus in water column for all pond categories between sampling seasons. For each box plot in the Young and Middle-age category, n = 9, representing 3 ponds * 3 measurements (inlet, Middle and outlet). For the Old category box plots, n = 6, representing 2 ponds * 3 measurements (inlet, middle and outlet).

4.2. Temporal Variations in SWP Chemical Characteristics

Pond age (p = 0.03), but not season, influenced Cl^- concentration. Specifically, Cl^- in Old ponds was higher than that in Young and Middle-age ponds (Figure 4a), but still within freshwater limits. Elevated Cl^- concentrations in SWPs are commonly attributed to road salts used for deicing (Corsi et al 2015; Lam et al 2020). However, in coastal SC, where snow and ice events are rare, this would be an unlikely source of Cl^- concentration. Instead, the higher Cl^- levels observed in the older ponds may be attributed to long-term accumulation and reduced dilution by freshwater inputs over time (Kaushal et al 2005), which could affect freshwater biota sensitive to increased ionic strength (Del Rosario and Resh 2000).

F^- concentrations varied by pond age (p = 0.001), but not by season (p = 0.07). F^- concentrations serve as conservative tracers indicating groundwater contribution or anthropogenic inputs (Xu et al 2022). The relatively low concentrations suggest limited influence from groundwater discharge. Pond age and season did not influence SO_4^{2-} concentration (p = 0.5 and p = 0.05, respectively), suggesting that the dynamics of SO_4^{2-} may be influenced by stable external inputs such as atmospheric deposition (Berner and Berner 2012), or that SO_4^{2-} transport into SWPs is minimal.

NH_4-N differed across all SWPs ages (p = 0.02), with the highest NH_4-N concentrations in Young ponds during both seasons (Figure 4d). NH_4-N did not vary by season (p = 0.07). This may reflect a greater inflow of NH_4-N and fresh runoff in newer ponds that have not yet developed sufficient microbial communities or sediment conditions for efficient nitrification (Hohman et al 2021). Similarly, pond age influenced NO_3+NO_2-N concentrations (p = 0.006), however, this effect did not differ seasonally (p = 0.9). Nevertheless, the lowest NO_3+NO_2-N concentrations were in the Middle-age SWP (Figure 4d), potentially indicating increased denitrification or plant uptake in the SWP. This result suggests that Middle-age ponds may have attained equilibrium between external inflow and internal loading, where microbial organisms and established vegetation may facilitate N removal (Hohman et al 2021). Additionally, our results suggest that there may be a potential functional significance of a peak in N-processing capacity during this stage of SWPs. TN concentration increased as SWPs aged (p = 0.03) and were higher in summer than winter (p <0.0001) (Figure 4f). The accumulation of TN in older ponds may be due to the buildup of organic matter in sediments and a reduced removal efficiency as SWPs ages (Gold et al 2017). Increased biological activity, stratification, and decreased outflow during dry seasons could explain the increased concentration of TN during the summer season (Janke et al 2022).

In terms of P dynamics, both SRP and TP concentrations were lowest in the Middle-age SWPs (p = 0.002 and p = 0.0002, respectively, Figures 4g and 4h) and were similar, regardless of season (p = 0.08 and p = 0.5, respectively). This pattern suggests a possible P retention period in the Middle-age SWPs, where sediments are still capable of effectively binding P and internal loading is reduced. Over time, P retained in sediments may begin to release back into the water column.

Table 2 Average total nitrogen: total phosphorus (TN:TP) ratio in stormwater pond sites from different sampling seasons

	TN:TP	
	Summer	Winter
Young pond	13:1	16:1
Middle-age pond	20:1	10:1
Old pond	16:1	5:1

Our findings showed that pond age influences the chemical properties in SWPs more than seasonal variables, suggesting that, altogether, the effect of aging on pond structure and functioning is more influential than short-term climatic variation during our sampling periods.

While Redfield ratios in some SWP age classes were present in appropriate ratios (16:1) based on molar concentration, the limiting nutrient in most ponds was likely to be TN (Table 2). This suggests that N removal processes, such as assimilation by aquatic plants and microbial denitrification, are effective in the SWPs (Tang et al 2020).

Nutrient limitation was assessed using the standard threshold N:P <10 indicates N limitation, N:P between 10 and 20 indicates balanced nutrients, and N:P >20 indicates P limitation.

4.3. Sediment-P and Water Column Interaction

To understand how age of SWP influenced P sediment behavior, estimated EPC_0, water column characteristics, and sediment characteristics were compared. The percentage sediment size distribution at each SWP age was measured across all 3 SWP sections in all the SWPs. SRP varied across SWP age classes (p = 0.002). However, EPC_0 did not differ either by SWP age or by sampling season (p = 0.4). Estimated EPC_0 values for each SWP were compared with water column SRP to determine the likelihood of sediment serving as a source or sink for P (Table 3). Most SWP sediment was P-saturated, implying that the sediment served as a P source. Outliers in our data were observed in SWP Y1 (both seasons) and M2 (winter), where the calculated EPC_0 was negative. A negative EPC_0 implies the sediment in those SWPs was not saturated with P at concentrations in our experiment, equilibrium was not attained, and the sediment served as a sink for P.

Water quality indicators measured for the Y1 pond showed higher nutrient concentrations compared to those of Y2 and Y3. The mean SRP concentration accounted for 15% of the TP, indicating SRP was available for plant uptake (Table 4). With a chlorophyll-a concentration of 75.6 µg/L, conditions were favorable for phytoplankton activities, including algal growth.

In SWPs Y2 and Y3, sediments acted as a source of P, and SRP was <3% of TP in the water column. M1 and M3, as well as all Old SWPs, exhibited desorption of sediment-associated P into the water column. P likely desorbed from sediment into the water column in any pond labeled as "Source" in Table 4. The higher the EPC_0 value, the more P is released from sediment into the water column (Haggard et al 2004). This internal P release from sediments, especially in older ponds, has been shown in earlier studies to contribute significantly to the higher chances of eutrophication, particularly in warm and deoxygenated conditions (Søndergaard et al

Table 3 Sediment equilibrium phosphorus concentrations (EPC_0) vs. water column soluble reactive phosphorus (SRP) across stormwater pond sections by season. Potential for sediment within each pond to serve as a source of phosphorus is noted in the condition column.

Pond age	Pond #	Season	EPC_0 (mg/L)	SRP (mg/L)	Condition
Young	Y1	Summer	-0.61*	0.16	
		Winter	-0.63*	0.11	
	Y2	Summer	2.19	0.02	Source
		Winter	2.23	0.01	Source
	Y3	Summer	0.82	0.01	Source
		Winter	6.17	0.02	Source
Middle-age	M1	Summer	30.9	0.01	Source
		Winter	0.95	0.01	Source
	M2	Summer	-1.44*	0.01	
		Winter	3.52	0.07	Source
	M3	Summer	0.91	0.01	Source
		Winter	0.50	0.03	Source
Old	O1	Summer	0.63	0.15	Source
		Winter	2.39	0.15	Source
	O2	Summer	0.49	0.02	Source
		Winter	0.85	0.02	Source
	O3	Summer	0.30	0.16	Source
		Winter	1.47	0.16	Source

*Data were excluded from analysis.
EPC_0 – equilibrium phosphorus concentrations
SRP – soluble reactive phosphorus

2003; Taguchi et al 2020). Chlorophyll-a concentrations in Middle-age SWPs suggested uptake of P from the water column. SRP was also less than 6% of TP in the water column, further supporting the possible rapid uptake and assimilation of desorbed P. A previous study by Janke et al (2022) similarly noted that biological assimilations can temporarily mask the effect of internal P loading in SWPs. Low SRP concentrations in the water column concurrent with the resuspension of P from the sediment are sufficient to support algal biomass (Søndergaard et al 2003). Disturbance of sediment leading to resuspension of P in Middle-age SWPs could be caused by other factors such as organisms' activities or wind action, particularly in shallow ponds (Nayeb Yazdi et al 2021; Søndergaard et al 2003). All Old SWPs behaved as a source of P to the water column, similar to Sønderup et al (2016), who reported that older ponds exhibited higher TP concentrations. SRP concentrations were more than 20% of TP in Old SWPs, suggesting SRP availability in the water column. Readily available P in the water column supports the growth of algal and phytoplankton communities (as measured by the presence of chlorophyll-a and phycocyanin).

It is essential to recognize that each pond is a unique ecosystem, and other factors contributing to sediment P behavior will include physical and biological characteristics surrounding the SWP. Multiple linear regression models were used to determine how sediment characteristics influenced EPC_0 in each SWP category (Table 5). Sediment characteristics in Young and Middle-age SWPs did not influence EPC_0 (p = 0.6). However, in Old SWPs, the reduced model (without organic carbon) predicted 40% of the responses, a better fit than for other pond ages, but still a poor predictor of EPC_0 (p = 0.03). In short, as percentages of sand and fine

Table 4 Water quality measurements in individual stormwater ponds (mean and standard error) throughout the study

Pond age	Pond #	DO (mg/L)	Chl-a (µg/L)	PC (µg/L)	TP (mg/L)	SRP (mg/L)	% SRP to TP
Young	Y1	6.60 ± 2.3	75.9 ± 63.8	3.97 ± 3.9	0.85 ± 0.1	0.13 ± 0.0	15
	Y2	7.02 ± 3.7	52.7 ± 29.8	4.37 ± 5.6	0.67 ± 0.1	0.01 ± 0.0	1
	Y3	7.89 ± 1.1	67.7 ± 95.4	3.65 ± 5.7	0.60 ± 0.1	0.02 ± 0.0	3
Middle-Age	M1	9.44 ± 0.5	372 ± 388	18.1 ± 19.3	0.76 ± 0.2	0.01 ± 0.0	1
	M2	9.50 ± 0.6	105 ± 106	8.19 ± 9.1	0.76 ± 0.5	0.04 ± 0.0	5
	M3	9.87 ± 0.9	71.1 ± 74.1	2.00 ± 2.2	0.53 ± 0.1	0.02 ± 0.0	4
Old	O1	6.47 ± 2.1	45.8 ± 44.7	2.63 ± 3.1	0.82 ± 0.2	0.18 ± 0.0	22
	O2	8.42 ± 2.7	184 ± 240	18.2 ± 19.4	1.09 ± 0.5	0.04 ± 0.0	4
	O3	2.54 ± 2.7	204 ± 185	47.1 ± 77.8	3.94 ± 1.5	0.69 ± 0.1	18

DO – dissolved oxygen, Chl-a – chlorophyll-a, PC – phycocyanin, TP – total phosphorus, SRP – soluble reactive phosphorus

Table 5 Multiple linear regression models of equilibrium phosphorus concentration (EPC_0) in different stormwater pond age

Pond age	Model fit	Prediction equation
Young	$R^2 = 0.1$, $p = 0.6$	1912 - 19.1(%Sand) - 19.3(%Fine) + 0.16(%OC)
Middle-age	$R^2 = 0.1$, $p = 0.6$	−91.6 + 0.94(%Sand) + 2.45(%Fine) − 1.93(%OC)
Old	$R^2 = 0.4$, $p = 0.08$ (full)	−4033 + 40.4(%Sand) + 40.3(%Fine) – (%OC)
	$R^2 = 0.4$, $p = 0.03$ (reduced)	−4034 + 40.4(%Sand) + 40.3(%Fine)
SRP interaction with sediment characteristics		
Young	$R^2 = 0.1$, $p = 0.9$	1923 − 11.9(SRP) − 19.2(%Sand) + 0.83(SRP*%Sand) − 19.3(%Fine) + 0.14(%OC) − 2.65(SRP*%OC)
Middle-age	$R^2 = 0.2$, $p = 0.8$	120328 + 14169(SRP) − 1206(%Sand) − 87352(SRP*%Sand) − 1203(%Fine) − 87265(SRP*%Fine) − 2.24(%OC) − 18.9(SRP*%OC)
Old	$R^2 = 0.6$, $p = 0.07$ (full)	−4365 + 5.63(SRP) + 43.6(%Sand) − 0.89(SRP*%Sand) + 43.8(%Fine) − 0.08(%OC) − 0.45(SRP*%OC)
	$R^2 = 0.6$, $p = 0.02$ (reduced)	− 4376 + 43.8(%Sand) − 0.59(SRP*%Sand) + 43.8(%Fine) + 4.64(SRP)

particles increased, EPC_0 increased, reflecting reduced P binding capacity with coarser sediments and enhanced P desorption. Interestingly, the interaction between sediment characteristics (without organic carbon) and SRP influenced EPC_0 in Old SWPs, but not in Young or Middle-age SWPs ($p = 0.02$, $p = 0.9$, and $p = 0.8$, respectively). Our results showed that in Old ponds, where organic matter is likely accumulated and sediments are more chemically saturated, both sediments and external P loads can have a combined effect that drives P release. The model suggests that increasing SRP, % sand, and % fine particles will result in higher EPC_0 values.

Although pond age was the main factor that was examined in our study, it is worth noting that several characteristics, such as the ratio of impervious to pervious area surrounding the pond or the size of the watershed compared to the pond, might also influence nutrient accumulation and movements through sediment P behavior.

5. Conclusion

This study provides insights into P sorption to sediment and how water quality in SWPs varies by pond age and season, including the various environmental characteristics that can influence P sorption dynamics. SWP age impacted water quality and influenced sediment P dynamics. Nutrient concentrations differed by SWP age class. Young SWPs had higher NH_4-N concentrations across both seasons in comparison with Middle-age and Old SWPs. Old ponds had the highest Cl^- concentrations, potentially indicating a greater accumulation of salt over time.

Monitoring salt accumulation in SWPs is crucial, as pond management practices may need to be adjusted based on the salt source. Planting vegetation around SWPs may help mitigate the impacts of landscape-derived salts on biological and chemical cycling in the coastal pond ecosystem. Total nutrient concentrations, including TN and TP, differed across pond ages, with higher concentrations in older SWPs. TN was identified as the limiting nutrient in most ponds, suggesting the presence of effective N removal processes, such as plant assimilation and microbial denitrification. In nearly all SWPs, regardless of age, our models indicated that sediment served as a source of P to the water column. Quantifying the chemical composition of sediments and their organic matter content is crucial for improving the reliability of model predictions. The regression models developed explained 60% or less of the EPC_0 dynamics, indicating that sediment particle size alone is insufficient to fully account for the internal P loading potential. To increase model prediction accuracy, future studies should include larger sample sizes, more SWPs, and should integrate various land use and environmental conditions. To develop reliable management strategies for controlling P release, we need a more in-depth understanding of the factors influencing EPC_0.

Our findings emphasize the importance of pond age as a key factor to consider for SWP system management, as it influences nutrient retention, accumulation, and potential releases.

While seasonal changes had nearly no effect on most of the measured parameters, nutrients and ion concentrations were significantly influenced by the structural and ecological maturity of SWPs. The observed variations in the physicochemical properties across SWP categories suggest that these SWPs undergo distinct functional transitions as they age. Young SWPs will likely benefit from early management interventions, such as vegetative planting and floating treatment wetlands, to enhance their nutrient uptake capacity. Middle-age SWPs appear to function optimally in terms of nutrient retention and reduction, suggesting a possible lifespan target for peak performance. Old ponds may require occasional dredging to maintain nutrient removal efficiency.

It is important to recognize that management strategies, particularly those involving sediment removal from ponds (dredging), may disrupt or alter natural sediment-water interactions and biotic communities (Cooke et al 2016; Lürling and Faassen 2012; Søndergaard et al 2003). As a result, the SWP's operational state might not necessarily align with its construction date. The need to differentiate between SWPs' ecological and chronological ages when assessing their performance is critical. Generally, future assessment and monitoring initiatives should consider SWPs' maintenance history, in addition to construction dates, to more accurately characterize the ecological performance of SWPs. Additionally, management practices should be adopted for SWPs based on their ecological age and should be supported by external management practices, including source control measures to help sustain water quality over time.

Supplementary Material

The online version of this manuscript contains a link to supplementary material that includes: **Table S1** Mean equilibrium phosphorus concentration values (EPC_0) and R^2 values from linear regression analyses used to estimate EPC_0 in all stormwater ponds sampled across two seasons (summer and winter), and **Table S2** Mean particle size distribution and organic carbon (OC) across stormwater pond sections (% fraction).

Acknowledgments

The authors acknowledge Charleston County and The City of Charleston, South Carolina, for providing assistance with site selection in this study. We also acknowledge Arkansas Water Resources Center (AWRC) for conducting water quality analysis. This material is based upon work supported by NIFA/USDA, under project number SC-1700584; Technical Contribution No. 7331 of the Clemson University Experiment Station; and SC Sea Grant Award No. NA220AR4170114. The authors also wish to acknowledge Ibrahim Busari, George Strickland, Morgan Chaudry, Jack Cuneo, and Allison Curl for their invaluable support during both the field and lab work.

Author Contributions Statement

Conceptualization: MF, DS, AES, SAW, CS, ES; methodology: MF, DS, AES, SAW, CS, ES; data analysis: MF, DS, AES, SAW, CS, ES; laboratory analyses: MF, DS; writing original draft: MF, DS; review/editing original draft: MF, DS, AES, SAW, CS, ES; investigation: MF, DS, AES, SAW, CS, ES; resources: DS, SAW, CS, ES; data curation: MF, DS; supervision: DS, SAW; project administration: DS; funding acquisition: DS, AES, ES. All authors have read and agreed to the published version of the manuscript.

Conflict of Interest Statement

The authors have no conflict of interest to report.

Data Availability Statement

Data requests can be directed to the corresponding author with access granted only upon approval from the funding agency.

References

Beckingham B, Callahan T, Vulava V. 2019. Stormwater ponds in the southeastern U.S. coastal plain: hydrogeology, contaminant fate, and the need for a social-ecological framework. Front Environ Sci. 7: 1–14. https://doi.org/10.3389/fenvs.2019.00117

Berner EK, Berner RA. 2012. Global environment: water, air, and geochemical cycles. 2nd ed. Princeton (NJ): Princeton University Press. https://www.jstor.org/stable/j.ctv30pnvjd

Carneiro FM, Santos AMC, Medina NG, Júnior PDM, Hortal J. 2024. Drivers of phytoplankton diversity in tropical artificial ponds. Perspect Ecol Conserv. https://doi.org/10.1016/j.pecon.2024.03.001

Cooke GD, Welch EB, Peterson S, Nichols SA. 2005. Restoration and management of lakes and reservoirs. 3rd ed. Boca Raton (FL): CRC Press. https://doi.org/10.1201/9781420032109

Corsi SR, De Cicco LA, Lutz MA, Hirsch RM. 2015. River chloride trends in snow-affected urban watersheds: Increasing concentrations outpace urban growth rate and are common among all seasons. Sci Total Environ. 508:488–497. https://doi.org/10.1016/j.scitotenv.2014.12.012

Daniels M, Austin BJ, Haggard BE. 2018. How to sample: collecting water samples is so easy, anyone can do it! https://scholarworks.uark.edu/awrcfs/8

Del Rosario RB, Resh VH. 2000. Invertebrates in intermittent and perennial streams: is the hyporheic zone a refuge from drying? J North Am Benthol Soc. 19(4):680–696. https://doi.org/10.2307/1468126

Dickes LA, Allen J, Jalowiecka M, Buckley K. 2016. A policy lens of South Carolina coastal stormwater management. J South Carolina Water Resour. 3:31–41. https://doi.org/10.34068/jscwr.03.04

Duan S, Newcomer-Johnson T, Mayer P, Kaushal S. 2016. Phosphorus retention in stormwater control structures across streamflow in urban and suburban watersheds. Water (Switzerland). 8(9). https://doi.org/10.3390/w8090390

[EPA] Environmental Protection Agency. 1993a. Method 353.2, revision 2.0: determination of nitrate-nitrite nitrogen by automated colorimetry. O'Dell JW, ed. Environmental Monitoring Systems Laboratory. EPA/600/R-93/100. https://www.epa.gov/sites/default/files/2015-08/documents/method_353-2_1993.pdf

[EPA] Environmental Protection Agency. 1993b. Method 365.1, revision 2.0: determination of phosphorus by semi-automated colorimetry. O'Dell JW, ed. Environmental Monitoring Systems Laboratory. EPA/600/R-93/100. https://www.epa.gov/sites/default/files/2015-08/documents/method_365-1_1993.pdf

Erickson AL, Williamson SM, Haggard BE. 2004. Method analysis of laboratory measures of stream sediment and water phosphorus equilibrium. Discovery Student J Dale Bumpers Coll Agric Food Life Sci. 5(1):10-15. https://scholarworks.uark.edu/discoverymag/vol5/iss1/5

Frost PC, Prater C, Scott AB, Song K, Xenopoulos MA. 2019. Mobility and bioavailability of sediment phosphorus in urban stormwater ponds. Water Resour Res. 55(5):3680–3688. https://doi.org/10.1029/2018WR023419

Gaffield SJ, Goo RL, Richards LA, Jackson RJ. 2003. Public health effects of inadequately managed stormwater runoff. Am J Public Health. 93(9):1527–1533. https://doi.org/10.2105/AJPH.93.9.1527

Garn HS. 2002. Effects of lawn fertilizer on nutrient concentration in runoff from lakeshore lawns, Lauderdale Lakes, Wisconsin. USGS Water Resources Investigation report 02-4130. https://consensus.fsu.edu/Fertilizer-Task-Force/industry/Effects%20of%20lawn%20fertilizer%20on%20nutrient%20concentration%20in%20Wisc.pdf

Gold AC, Thompson SP, Piehler MF. 2017. Coastal stormwater wet pond sediment nitrogen dynamics. Sci Total Environ. 609:672–681. https://doi.org/10.1016/j.scitotenv.2017.07.213

Gold AC, Thompson SP, Piehler MF. 2021. Seasonal variation in nitrate removal mechanisms in coastal stormwater ponds. Water Resour Res. 57(10):1–16. https://doi.org/10.1029/2021WR029718

Haggard BE, Ekka SA, Matlock MD, Chaubey I. 2004. Phosphate equilibrium between stream sediments and water: potential effect of chemical amendments. Trans Am Soc Agric Eng. 47(4):1113–1118. https://doi.org/10.13031/2013.16584

Hartshorn N, Marimon Z, Xuan Z, Chang N-B, Wanielista MP. 2016. Effect of floating treatment wetlands on control of nutrients in three stormwater wet detention ponds. J Hydrol Eng. 21(8):1–16. https://doi.org/10.1061/(asce)he.1943-5584.0001387

Hasibuan S, Syafriadiman S, Aryani N, Fadhli M, Hasibuan M. 2023. The age and quality of pond bottom soil affect water quality and production of Pangasius hypophthalmus in the tropical environment. Aquac Fish. 8(3):296–304. https://doi.org/10.1016/j.aaf.2021.11.006

He J, Chu A, Ryan MC, Valeo C, Zaitlin B. 2011. Abiotic influences on dissolved oxygen in a riverine environment. Ecol Eng. 37(11):1804–1814. https://doi.org/10.1016/j.ecoleng.2011.06.022

Heiri O, Lotter FA, Lemcke G. 1999. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. J Paleolimnol. 103(25):101–110. https://doi.org/10.1023/A:1008119611481

Hoess R, Geist J. 2021. Effect of fish pond drainage on turbidity, suspended solids, fine sediment deposition and nutrient concentration in receiving pearl mussel streams. Environ Pollut. 274:116520. https://doi.org/10.1016/j.envpol.2021.116520

Hohman SP, Smyth AR, Bean EZ, Reisinger AJ. 2021. Internal nitrogen dynamics in stormwater pond sediments are influenced by pond age and inorganic nitrogen availability. Biogeochemistry, 156(2):255–278. https://doi.org/10.1007/s10533-021-00843-2

Hossain MS, Islam MA, Badhon FF, Imtiaz T. 2022. Properties and behavior of soil - online lab manual. Mavs Open Press Open Educational Resources. 32. https://mavmatrix.uta.edu/oer_mavsopenpress/32

Hupfer M, Lewandowski J. 2008. Oxygen controls the phosphorus release from lake sediments — a long-lasting paradigm in limnology. Int Rev Hydrobiol. 93(4–5):415–432. https://doi.org/10.1002/iroh.200711054

Janke BD, Finlay JC, Taguchi VJ, Gulliver JS. 2022. Hydrologic processes regulate nutrient retention in stormwater detention ponds. Sci Total Environ. 823:153722. https://doi.org/10.1016/j.scitotenv.2022.153722

Janke BD, Natarajan P, Shrestha P; Taguchi VT, Finlay JC; Gulliver JS. 2021. Detecting phosphorus release from stormwater ponds to guide management and design. University Digital Conservancy. Report No. 597. https://hdl.handle.net/11299/218852

Jin X, Wang S, Pang Y, Chang Wu F. 2006. Phosphorus fractions and the effect of pH on the phosphorus release of the sediments from different trophic areas in Taihu Lake, China. Environ Pollut 139(2):288–295. https://doi.org/10.1016/j.envpol.2005.05.010

Kaushal SS, Groffman PM, Likens GE, Belt KT, Stack WP, Kelly VR, Band LE, Fisher GT. 2005. Increased salinization of fresh water in the Northeastern United States. Proceedings of the National Academy of Sciences of the United States of America. 102(38):13517–13520. https://doi.org/10.1073/pnas.0506414102

Kaye JP, Groffman PM, Grimm NB, Baker LA, Pouyat RV. 2006. A distinct urban biogeochemistry? Trends Ecol Evol. 21(4):192–199. https://doi.org/10.1016/j.tree.2005.12.006

Kermorvant C, D'Amico F, Bru N, Caill-Milly N, Robertson B. 2019. Spatially balanced sampling designs for environmental surveys. Environ Monit Assess. 191(524). https://doi.org/10.1007/s10661-019-7666-y

Lam WY, Lembcke D, Oswald C. 2020. Quantifying chloride retention and release in urban stormwater management ponds using a mass balance approach. Hydrol Process. 34(23):4459–4472. https://doi.org/10.1002/hyp.13893

Law N, Band L, Grove M. 2004. Nitrogen input from residential lawn care practices in suburban watersheds in Baltimore county, MD. J Environ Plan Manag. 47(5):737-755. https://doi.org/10.1080/0964056042000274452

Lin P, Klump JV, Guo L. 2018. Variations in chemical speciation and reactivity of phosphorus between suspended-particles and surface-sediment in seasonal hypoxia-influenced Green Bay. J Great Lakes Res. 44(5):864–874. https://doi.org/10.1016/j.jglr.2018.05.022

Lürling M, Faassen EJ. 2012. Controlling toxic cyanobacteria: effects of dredging and phosphorus-binding clay on cyanobacteria and microcystins. Water Res. 46(5):1447–1459. https://doi.org/10.1016/j.watres.2011.11.008

Lusk MG, Bean EZ, Iannone BV, Reisinger AJ. 2025. Stormwater ponds: Unaccounted environmental challenges of a widely-adopted best management practice in urban landscapes. J Environ Manage. 374:124170. https://doi.org/10.1016/j.jenvman.2025.124170

Lusk MG, Chapman K. 2021. Chemical fractionation of sediment phosphorus in residential urban stormwater ponds in Florida, USA. Urban Sci. 5(4):81. https://doi.org/10.3390/urbansci5040081

Mallin MA, Ensign SH, Wheeler TL, Mayes DB. 2002. Pollutant Removal Efficacy of Three Wet Detention Ponds. J Environmental Quality, 31(2), 654–660. https://doi.org/10.2134/jeq2002.6540

McEnroe NA, Buttle JM, Marsalek J, Pick FR, Xenopoulos MA, Frost PC. 2013. Thermal and chemical stratification of urban ponds: Are they "completely mixed reactors"? Urban Ecosyst. 16(2):327–339. https://doi.org/10.1007/s11252-012-0258-z

McKercher LJ, Kimball ME, Scaroni AE, White SA, Strosnider WHJ. 2024. Stormwater ponds serve as variable quality habitat for diverse taxa. Wetl. Ecol. Manag. 32(1):109–131. https://doi.org/10.1007/s11273-023-09964-x

Miller JD et al. 2014. Assessing the impact of urbanization on storm runoff in a peri-urban catchment using historical change in impervious cover. J Hydrol. 515:59–70. https://doi.org/10.1016/j.jhydrol.2014.04.011

Nayeb Yazdi M, Scott D, Sample DJ, Wang X. 2021. Efficacy of a retention pond in treating stormwater nutrients and sediment. J Clean. Prod. 290:125787. https://doi.org/10.1016/j.jclepro.2021.125787

Nelson KC, Palmer MA. 2007. Stream temperature surges under urbanization and climate change: data, models, and responses. J Am Water Resour Assoc. 43(2):440–452. https://doi.org/10.1111/j.1752-1688.2007.00034.x

O'Dell, JW. 1996. Method 351.2 - determination of total Kjeldahl nitrogen by semi-automated colorimetry. Environmental Monitoring Systems Laboratory, ed. In Methods for the Determination of Metals in Environmental Samples. William Andrew Publishing. p 449-463. https://doi.org/10.1016/b978-0-8155-1398-8.50025-2

Perron MAC, Pick FR. 2020. Stormwater ponds as habitat for Odonata in urban areas: the importance of obligate wetland plant species. Biodivers Conserv. 29(3):913–931. https://doi.org/10.1007/s10531-019-01917-2

Pfaff, JD. 1996. Method 300.0 - determination of inorganic anions by ion chromatography. Environmental Monitoring Systems Laboratory, ed. In Methods for the Determination of Metals in Environmental Samples. William Andrew Publishing. p 338-417. https://doi.org/10.1016/b978-0-8155-1398-8.50022-7

Reddy KR, Kadlec RH, Flaig E, Gale PM. 1999. Phosphorus retention in streams and wetlands: A review. Crit Rev Environ Sci Technol. 29(1):83–146. https://doi.org/10.1080/10643389991259182

Sanseverino I, Conduto D, Pozzoli L, Dobricic S, Lettieri T. 2016. Algal bloom and its economic impact. EUR JRC Tech Rep. https://doi.org/10.2788/660478

Scaroni AE, Sahoo D, Wallover CG. 2021. An introduction to stormwater ponds in South Carolina. Clemson (SC): Clemson Cooperative Extension, Land-Grant Press by Clemson Extension; 2021 Aug. LGP 1119. https://lgpress.clemson.edu/publication/an-introduction-to-stormwater-ponds-in-south-carolina/

Schroer WF, Benitez-Nelson CR, Smith EM, Ziolkowski LA. 2018. Drivers of sediment accumulation and nutrient burial in coastal stormwater detention ponds, South Carolina, USA. Ecosystems. 21(6):1118–1138. https://doi.org/10.1007/s10021-017-0207-z

Serrano L, DeLorenzo ME. 2008. Water quality and restoration in a coastal subdivision stormwater pond. J Environ Manage. 88(1):43–52. https://doi.org/10.1016/j.jenvman.2007.01.025

Sinclair JS et al. 2020. Stormwater ponds: an overlooked but plentiful urban designer ecosystem provides invasive plant habitat in a subtropical region (Florida, USA). Sci Total Environ. 711:135133. https://doi.org/10.1016/j.scitotenv.2019.135133

Søndergaard M, Jensen JP, Jeppesen E. 2003. Role of sediment and internal loading of phosphorus in shallow lakes. Hydrobiologia. 506(1):135–145. https://doi.org/10.1023/B:HYDR.0000008611.12704.dd

Sønderup MJ, Egemose S, Hansen AS, Grudinina A, Madsen MH, Flindt MR. 2016. Factors affecting retention of nutrients and organic matter in stormwater ponds. Ecohydrology. 9(5):796–806. https://doi.org/10.1002/eco.1683

Song K, Winters C, Xenopoulos MA, Marsalek J, Frost PC. 2017. Phosphorus cycling in urban aquatic ecosystems: connecting biological processes and water chemistry to sediment P fractions in urban stormwater management ponds. Biogeochemistry. 132(1–2):203–212. https://doi.org/10.1007/s10533-017-0293-1

Song K, Xenopoulos MA, Buttle JM, Marsalek J, Wagner ND, Pick FR, Frost PC. 2013. Thermal stratification patterns in urban ponds and their relationships with vertical nutrient gradients. J Environ Manage. 127:317–323. https://doi.org/10.1016/j.jenvman.2013.05.052

[SCDHEC] South Carolina Department of Health and Environmental Control. 2003. South Carolina stormwater management and sediment control handbook for land disturbance activities. August 2003. South Carolina Department of Health and Environmental Control, Environmental Quality Control and Bureau of Water, Office of Ocean and Coastal Resource Management. https://www.cicacenter.org/pdf/scsedswchandbook.pdf

Sun L, Chen J, Li Q, Huang D. 2020. Dramatic uneven urbanization of large cities throughout the world in recent decades. Nat Commun. 11(1):1–9. https://doi.org/10.1038/s41467-020-19158-1

Taguchi VJ, Olsen TA, Natarajan P, Janke BD, Gulliver JS, Finlay JC, Stefan HG. 2020. Internal loading in stormwater ponds as a phosphorus source to downstream waters. Limnol Oceanogr Lett. 5(4):322–330. https://doi.org/10.1002/lol2.10155

Tang S, Liao Y, Xu Y, Dang Z, Zhu X, Ji G. 2020. Microbial coupling mechanisms of nitrogen removal in constructed wetlands: A review. Bioresour Technol. 314:123759. https://doi.org/10.1016/j.biortech.2020.123759

Tipping E, Benham S, Boyle JF, Crow P, Davies J, Fischer U, Guyatt H, Helliwell R et al. 2014. Atmospheric deposition of phosphorus to land and freshwater. Environ Sci Process. Impacts. 16(7):1608–1617. https://doi.org/10.1039/c3em00641g

Tixier G, Lafont M, Grapentine L, Rochfort Q, Marsalek J. 2011. Ecological risk assessment of urban stormwater ponds: Literature review and proposal of a new conceptual approach providing ecological quality goals and the associated bioassessment tools. Ecol Indic. 11(6):1497–1506. https://doi.org/10.1016/j.ecolind.2011.03.027

Wang F et al. 2024. Seasonal hypoxia enhances sediment iron-bound phosphorus release in a subtropical river reservoir. Sci Total Environ. 936:173261. https://doi.org/10.1016/j.scitotenv.2024.173261

Wu Y, Wen Y, Zhou J, Wu Y. 2014. Phosphorus release from lake sediments: effects of pH, temperature and dissolved oxygen. KSCE J Civ Eng. 18:323–329. https://doi.org/10.1007/s12205-014-0192-0

Xu P et al. 2022. Characteristics of fluoride migration and enrichment in groundwater under the influence of natural background and anthropogenic activities. Environ Pollut. 314:120208. https://doi.org/10.1016/j.envpol.2022.120208

Yang K, Luo Y, Chen K, Yang Y, Shang C, Yu Z, Xu J, Zhao Y. 2020. Spatial–temporal variations in urbanization in Kunming and their impact on urban lake water quality. Land Degrad Dev. 31(11):1392–1407. https://doi.org/10.1002/ldr.3543

Yang YY, Lusk MG. 2018. Nutrients in urban stormwater runoff: current state of the science and potential mitigation options. Curr Pollut Rep. 4(2):112–127. https://doi.org/10.1007/s40726-018-0087-7

Zhao S, Shi X, Sun B, Liu Y, Tian Z, Huotari J. 2022. Effects of pH on phosphorus form transformation in lake sediments. Water Supply. 22(2):1231–1243. https://doi.org/10.2166/ws.2021.356

Zhou A, Tang H, Wang D. 2005. Phosphorus adsorption on natural sediments: modeling and effects of pH and sediment composition. Water Res. 39(7):1245–1254. https://doi.org/10.1016/j.watres.2005.01.026