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Nutrient dynamics in restored and unrestored urban streams in the Piedmont ecoregion of South Carolina

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The United States invests billions of dollars annually to perform stream restoration projects, yet few studies have investigated the effects this ecosystem manipulation has on nutrient cycling and associated water quality. Water quality improvement remains a substantial motivation for mitigating catchment-scale disturbances, especially in urban streams. Various urban land use practices impact the transfer and transport of nutrients such as soluble reactive phosphorus, ammonium, and nitrate plus nitrite from land into the streams and rivers. The uptake length (S_W), or the distance a dissolved nutrient travels downstream within a stream reach, can be measured using short-term nutrient injections, where shorter uptake lengths suggest greater nutrient retention. This study evaluated the efficacy of using nutrient injection experiments as a monitoring tool to assess nutrient retention efficiency in first-order urban restored (RES) and urban unrestored (URE) stream reaches within the Piedmont ecoregion of South Carolina during the winter and summer seasons of 2022. Results suggested that the lack of fine sediment, such as silt and clay, may affect the nutrient cycling of phosphorus. The total nitrogen:total phosphorus ratio indicated the stream was phosphorus-limited during the experiments. The mean soluble reactive phosphorus uptake length throughout the study was shorter in RES than URE, suggesting that the restored reach was more efficient in retaining dissolved phosphorus. During summer injection, RES observed the shortest soluble reactive phosphorus uptake length of 77 m, while URE marked the longest uptake length of 3059 m during the same period. However, during winter injections, the URE segment had both the shortest and longest uptake lengths. In summer, RES exhibited noticeably shorter ammonium uptake lengths, while ammonium uptake lengths could not be calculated in URE. The presence of engineered pools in RES assisted in nutrient dynamics and helped trap nutrients, particularly soluble reactive phosphorus and ammonium, and could be prioritized in stream restoration efforts. Preliminary results from this study could provide helpful insights into the effectiveness of stream restoration and in-stream structures on nutrient dynamics, although further research is needed.

KEYWORDS

nutrient cycling, nitrogen, phosphorus, uptake length, stream restoration, trapping efficiency

1 Introduction

Globally, stream ecosystems are experiencing a decline in water quality and biodiversity, primarily due to an excess of nutrients, such as nitrogen (N) and phosphorus (P), which have become more readily available due to significant land use changes and practices (Miserendino et al., 2011; Liao et al., 2018). In the United States, N and P are the most widespread and studied chemical stressors assessed in lotic ecosystems (EPA, 2022b). Nutrient contributions from both agriculture and urbanization exceed the intrinsic ability of freshwater ecosystems to process N and P enrichment (Nina and Jonathan, 1999; Cole et al., 1993; Manning et al., 2020; Caraco and Cole, 1999), causing poor water quality and reducing the quality of habitat for biota.

Depending on the watershed size and location, urban development generally encompasses a much smaller fraction of the total catchment area than agricultural development (Allan, 2004). However small these urban areas, their influence on watersheds is more pronounced because of impervious surfaces (Bledsoe and Watson, 2001; Paul and Meyer, 2001; Tranmer et al., 2022). The contrast between the geomorphology of riffles and pools, baseflows and stormflows, and hillslopes and in-stream dynamics is exacerbated by urbanization (Blaszczak et al., 2019). The urban land developmental process overwhelms the buffering capacity of streams through loss of riparian vegetation, increased stormwater drainage pipes, impervious surfaces, and runoff carrying various contaminants (Paul and Meyer, 2001). Together, these manipulations result in channel and geomorphic degradation from intense storm flows (Bledsoe and Watson, 2001; Russell et al., 2020), substantial loss of native biodiversity (Stranko et al., 2012), and increased nutrient export to downstream watersheds (Klocker et al., 2009).

The set of physical, biological, and chemical changes consistently observed within streams draining urban land is known as the urban stream syndrome (Walsh et al., 2005). The urban stream syndrome is rooted in rapidly urbanizing catchments that interrupt the natural organizational ability of streams to erode and deposit bed surface material in equilibrium (Wolman, 1967). While there are several explanations for channel modifications [e.g., channelization] due to changes in hydrology, on many occasions, the frequent hydraulic disturbance from storm flows leads to an unbalanced removal of sediment from the stream banks, causing channel enlargement and simplification (Russell et al., 2020). Advanced municipal infrastructure, like sanitary sewers and wastewater treatment plants, effectively reduces nitrogen waste from cities while potentially increasing diffuse nitrogen pollution across watersheds (Bernhardt et al., 2008; Howarth et al., 1996). It is well known that excess release of key contaminants such as N, P, and sediment/solids into stream water from stormwater runoff associated with watershed development deteriorates the physical environment and ecological processes within the stream (Brown et al., 2009; Kaushal et al., 2017; Kriech and Osborn, 2022; Paul and Meyer, 2001; Williams and Filoso, 2023).

The pressure from urban development has placed attention on reducing in-stream nutrient concentrations through a vast number of stream restoration projects (Lammers and Bledsoe, 2017). Stream restoration is a popular, multibillion-dollar investment practice implemented by state and federal agencies, non-government organizations (NGOs), and consultants to improve stream health, structure, and function (Bernhardt et al., 2005). Engineering design practices in the Southeastern United States traditionally follow the Rosgen's Natural Channel Design to restore the natural pattern, profile, and dimensions of a disturbed stream by emulating a stable channel within the same watershed (Rosgen, 2007). However, these restoration approaches do not address water quality, nutrient dynamics, and their fate and transport. One of the main drivers behind the widespread adoption of stream restoration initiatives in the United States can be explained by mitigation credits issued by regulatory agencies as a financial incentive for developers to invest in stream restoration to meet a waterbody's Total Maximum Daily Load requirement (Thompson et al., 2018; Williams et al., 2017).

Typically, urban stream restoration is an attempt to reduce the magnitude of flashiness produced by urbanization and reconnect the stream to the floodplain. Restoration involves installing native plant materials and constructing geomorphic complexity and channel stability with boulders, wood, and rock deflectors with the intention of dissipating water flow and increasing hydrologic residence time (Palmer et al., 2014). Previous studies of restored streams assume that once geomorphic complexity and hydrologic residence time are increased, then nutrient processing will be restored because water has more time to exchange back and forth between the surface water column and sediment (Ensign and Doyle, 2006; Craig et al., 2008; Bukaveckas, 2007; Roberts, Mulholland, and Houser, 2007; McMillan et al., 2014). Despite decades of stream restoration practice, factors that govern the responsiveness of in-stream nutrient dynamics to the restoration process are merely conceptual. Efforts to quantify the implications of N and P removal through stream monitoring research need more attention.

Current monitoring practices have focused on testing water quality improvement by analyzing nutrient cycling in post-restoration surface waters of streams (McMillan et al., 2014; Newcomer Johnson et al., 2016; Reisinger et al., 2016). Different stream features in a restored stream, such as riffle-pool sinuosity, change the interaction of the nutrients traveling downstream and potentially increase the retention of nutrient molecules within the stream by enhancing the timing and magnitude of downstream travel (Figure 1). Downstream transport of N and P in flowing water

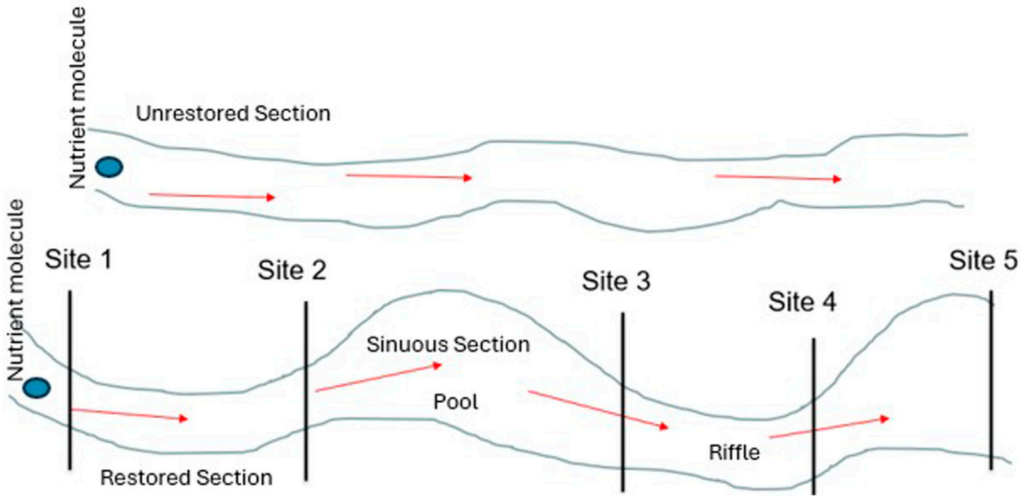

FIGURE 1
Conceptual diagram illustrating nutrient retention in a restored and unrestored first-order stream.

has been studied through short-term injections, where nutrient retention efficiency was measured as the spiraling length or downstream distance traveled before assimilation (Haggard et al., 2001; Chaubey et al., 2007). The rate at which aquatic ecosystems cycle nutrients from the dissolved state within the water column to the particulate state within the sediment or biota is defined as nutrient retention efficiency (Chaubey et al., 2007). This capture and release process influences the timing, magnitude, and form of nutrients that are transported downstream (Meyer et al., 1988).

The spiraling length of a nutrient molecule is the sum of the distance traveled in the particulate form (turnover length, S_p) and dissolved form (uptake length, S_w) (Stream Solute Workshop, 1990). Under baseflow conditions, S_w calculates the distance a dissolved nutrient travels downstream before it is removed from the water column, and nutrient S_w dominates the total spiraling length (Newbold et al., 1983). This transport (S_w) represents retention efficiency and varies by land use, anthropogenic disturbances (e.g., urbanization), and restoration design (Beechie et al., 2010; Haggard et al., 2005; Klocker et al., 2009; McMillan et al., 2014). Assessment of nutrient uptake ability can be performed using short-term nutrient injections within the restored reach (Stream Solute Workshop, 1990).

The overarching goal of this research was to analyze how nutrient injections could aid in monitoring in-stream nutrient dynamics in low-order urban restored (RES) and unrestored (URE) streams in the Piedmont ecoregion of South Carolina. The research question for this study focused on the controls of soluble reactive phosphorus (SRP), ammonium (NH_4-N), and nitrate plus nitrite (NO_3-N + NO_2-N) in two contrasting reaches of the same stream. The specific research questions of this study were to assess the influence of restoration design (e.g., riffle and pool geomorphology), reach length, season, discharge, and other background measurements on performance via estimated nutrient uptake lengths and nutrient trapping efficiencies. We hypothesized that unique spatial and temporal differences in nutrient retention would occur in RES and URE reaches.

To address these research questions, short-term nutrient injection techniques were used in restored and unrestored urban stream reaches to evaluate the spatial and temporal variations in N and P dynamics and understand nutrient retention, whole-stream nutrient trapping efficiency, and nutrient trapping efficiency in the engineered pools.

2 Materials and methods

2.1 Site description

The study was conducted in Richland Creek, a first-order urban stream in the southern inner piedmont ecoregion of South Carolina. For this study, reaches refer to the restored and unrestored locations, sites refer to sampling locations, and restored stream features refer to riffles and pools in RES and URE streams. Sampling occurred in winter (January–March) and summer (June–July) months of 2022. Two stream reaches were selected within Richland Creek in Greenville, South Carolina. The first stream reach was the RES reach in McPherson Park, and the second was the URE reach downstream, parallel to the E Park Ave highway (Figure 2). Richland Creek drains into the Saluda River basin (hydrological unit code 03050109) and is a tributary of the Reedy River with both reaches having approximately 1.8 km^2 of the watershed area that consists of less than 1% of pasture, mixed forest, and deciduous forest and approximately 99% urban land use (United States Geographical Survey, 2019).

Based on the information provided by the City of Greenville (*personal communication*), stream restoration was completed in December 2018 using Rosgen's Natural Channel Design that begins directly after a culvert in ~150 m of stream. The objective of this restoration effort was to stabilize the bank and reduce nutrient loading [e.g., total phosphorus (TP) and total nitrogen (TN)] generated by urban land use within the Richland Creek watershed. The stream restoration design plan can be obtained from the City of Greenville, South Carolina. Based on the design

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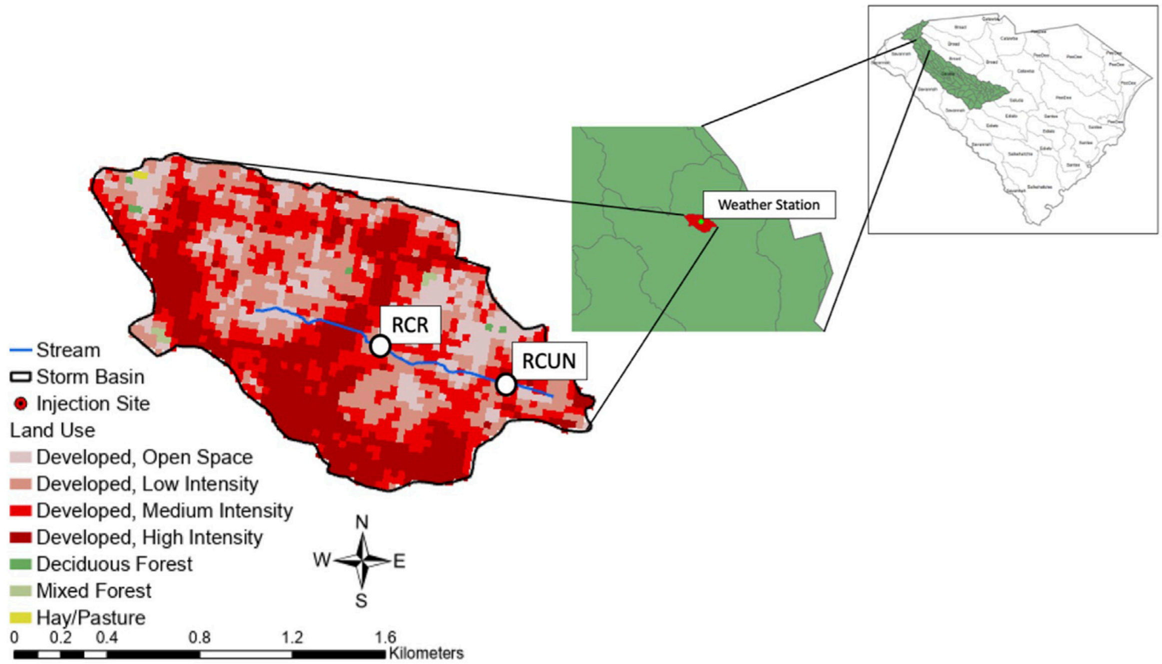

FIGURE 2
Richland Creek watershed and respective sites, Richland Creek—Restored (RES) and Richland Creek—Unrestored (URE), reside in the upper subwatershed of the Saluda River Basin in Greenville, South Carolina. The weather station is indicated by a green marker.

TABLE 1 Length (m) of each pool within the restored (RES) reach.

Site	Location	Pool length (m)
RES	POOL 1	8
Step pool begins	POOL A	6
	POOL B	4
	POOL C	5
	POOL D	5
Step pool ends	POOL E	5
	POOL 2	14
	POOL 3	12

plan, it was noticed that regenerative stormwater conveyances, native plants, and biodegradable materials were installed, and over-steepened bank sediment was removed. Detailed information on the installed pools, species of plants, type of plants, where they were planted, and type of biodegradable materials used can be referenced in the design plan. There are two types of stream features, one created step pool that is 25 m long, composed of five sub-pools ranging in length from 4 m to 6 m, and three constructed pools ranging in length from 8 m to 14 m (Table 1). The sequence of constructed step pools has a gradual decline in slope that dissipates energy from high stream discharge, controls erosion, enhances oxygenation, and improves downstream water quality; while the constructed pools provide diversity in stream discharge with a single, deeply excavated stream bed that impounds flow and captures sediment. The sequence of step pools and constructed pools was created during the stream restoration project. The study area focused on a restored reach of ~70 m length and an unrestored reach of ~80 m length, with injection experiments occurring at the top of the restored and unrestored reaches. The restored reach includes 11 sampling sites that were consistently sampled for each experiment. These sites contained three riffle and pool features and a step pool conveyance that was included in the sampling (Table 1). The study reach length was cut short due to an outlet pipe at the downstream boundary of the restored reach that continuously drains into the stream.

Approximately 0.5 km downstream of the RES (separated by a road culvert) section is the URE reach of Richland Creek (Figure 2). This reach is impacted by land uses similar to those of the RES reach. Anecdotal evidence and conversation with local residents within the project area indicated that the land was purchased around the 1930s, and the stream was channelized around the mid-1940s with a stone wall that currently runs parallel to the main road (East Park Ave.) along the entire stream reach. The stream bed is mainly composed of bedrock with remnant riffle-pool structures. Five sampling sites were consistently sampled for each experiment near these remnant riffle-pool structures. URE receives continuous discharge of the outlet pipe at the downstream boundary of the restored reach throughout the entire sampling period. Stormwater outlets recur throughout the reach but were not discharging during the sampling period.

The average width varied spatially (28%) between the RES and URE reaches where the RES reach (range: 2.4–2.7 m, average = 2.62 m ± 0.14) was wider than the URE reach (range: 1.8–2.6 m, average = 1.98 m ± 0.30) throughout the study period, likely due to stream features incorporated during restoration such as sinuosity from riffle and pool structures ($t = 4.35$, $p < 0.05$, $n = 6$) (Table 2).

TABLE 2 Depth, width, velocity, and discharge of sites within restored (RES) and unrestored (URE) reaches of Richland Creek during injections.

	Depth (cm)	Width (m)	Velocity (m/s)	Discharge (m^3/s)
Winter				
January				
RES	16	2.7	0.024	0.013
URE	21	1.8	0.053	0.013
February				
RES	19	2.7	0.018	0.003
URE	12	1.8	0.064	0.012
March				
RES	20	2.7	0.054	0.002
URE	23	1.8	0.030	0.003
Summer				
June				
RES	19	2.7	0.040	0.003
URE	23	1.8	0.039	0.005
Early July				
RES	19	2.4	0.034	0.002
URE	8	2.0	0.122	0.003
Late July				
RES	19	2.4	0.056	0.002
URE	8	2.6	0.090	0.003

TABLE 3 Average monthly precipitation (mm) recorded in Greenville County from 2022 to 2023 (AccuWeather, Inc., 2023) and by historical Greenville County weather stations over the last 30 years (National Oceanic and Atmospheric Administration, 2020).

	Winter precipitation (mm)				Summer precipitation (mm)		
County	January	February	March	Mean	June	July	Mean
Greenville data	96	119	142	119 ± 23	17	55	36 ± 27
Historic Greenville data	112	90	109	104 ± 12	112	117	115 ± 3.0

The average velocity and discharge between the RES and URE reaches were also similar and did not vary by season. The average discharge of both the RES and URE reaches was expected to increase during the winter season and decrease during the summer season (Table 2) (Dyer et al., 2022). The flow regime was affected by higher temperatures and lower precipitation in the summer season (Table 3). Summer flows are generally reduced due to water losses from evapotranspiration from the mixed deciduous forest vegetation (present in the RES and URE reaches) and infiltration along the stream channel (Lundquist and Cayan, 2002). The low-flow period in both sections persisted throughout the study period and may have been more affected by other processes because the stream carries such a low volume of water.

The impervious area in the watershed includes roads, bridges, and access areas in the park that potentially restrict the opportunity for organic matter to decompose into various particle sizes, consequently reducing the possibility of sediment transport into the stream. The dominant particle size of the creek was gravel. The creek contains minimal amounts of silt and clay particles, likely because they are typically found in pools where velocities decrease and allow them to settle (EPA, 2023). In general, the RES reach had 71% gravel, 29% sand, and less than 1% silt and clay, while the URE reach had 73% gravel, 27% sand, and less than 1% silt and clay as estimated during the sampling. [Advancing Standards Transforming Markets (2017)]

The average temperatures for the injection dates in winter were 10°C in January, 12°C in February, and 23°C in March. January and February average temperatures from the injection dates were within the 30-year minimum and maximum averages for Greenville County, where the study sites are located. However, March was 6°C higher than the historic average maximum temperature in the Greenville County area (National Oceanic and Atmospheric Administration, 2020). During the summer, the sampled sites collectively had an average temperature of 30°C in June and 28°C in July during the injection dates (Table 2), which was also within the 30-year minimum and maximum average temperature in Greenville County (National Oceanic and Atmospheric Administration, 2020). Given the overall weather data (Table 4), there were no atypical trends when compared with the long-range data. Therefore, there is a fair degree of confidence that the empirical findings from this study are representative of the seasonal conditions. The average monthly

TABLE 4 Date of each injection along with the weather station data and background total nitrogen and total phosphorus ratio (TN:TP) recorded on each day of an injection for restored (RES) and unrestored (URE) reaches of Richland Creek.

	Winter			Summer		
	January	February	March	June	Early July	Late July
RES	1/14/2022	2/26/2022	3/6/2022	6/21/2022	7/12/2022	7/25/2022
Time	2:29 p.m.	3:14 p.m.	12:49 p.m.	11:34 a.m.	11:24 a.m.	10:49 a.m.
Atm. temp (°C)	13	13	22	29	26	30
Dew point	−1	3	15	16	21	23
Humidity (%)	39	51	67	45	73	71
Wind speed (km/h)	2.1	0.6	2.9	1	0.5	1.4
Pressure (Pa)	14	15	15	15	15	15
Precipitation (mm)	0	0	0	0	0	0
Condition	Fair	Fair	Cloudy	Fair	Fair	Fair
TN:TP	72:1	74:1	43:1	34:1	24:1	27:1
URE	1/28/2022	2/20/2022	3/2/2022	6/22/2022	7/15/2022	7/21/2022
Time	11:49 a.m	5:39 p.m.	2:09 p.m.	11:09 a.m.	10:19 a.m.	11:53 a.m.
Atm. temp (°C)	6	12	24	31	29	27
Dew point	−2	−8	2	16	19	21
Humidity (%)	55	25	23	40	56	70
Wind speed (km/h)	2.1	1.3	4.2	2.9	0.5	2.6
Pressure (Pa)	15	15	15	15	15	14
Precipitation (mm)	0	0	0	0	0	0
Condition	Cloudy	Fair	Fair	Fair	Fair	Fair
TN:TP	79:1	67:1	39:1	35:1	28:1	27:1

precipitation from a Greenville weather station was compared with the 30-year average precipitation in Greenville County to understand how rainfall may have influenced the watersheds during the study period. Based on weather station data, higher than the 30-year average precipitation occurred in the winter, while 3× lower-than-normal precipitation occurred during the summer (Table 3) (National Oceanic and Atmospheric Administration, 2020). June was the driest month, with the lowest precipitation amounts that were below the historical precipitation levels for Greenville County (Table 3). This dry summer period may have been a result of ongoing climate warming in the rapidly growing state of South Carolina (Sanchez et al., 2020).

2.2 Nutrient injection experiments

All experiments were conducted under baseflow conditions between sunrise and sunset (Table 4). Field experiments avoided periods of storm events. Stream velocity measurements (Marsh-McBirney Inc., 1990) were conducted and discharge (Turnipseed and Sauer, 2010) was calculated during baseflow conditions, the day before each experiment, near the most upstream injection site of each study reach using a Flo-Mate 2000 flow meter (Hach Company, Frederick, Maryland, United States). Stream velocity measurements were taken using the midsection method (Young, 1950). Velocity measurements were taken at every 0.15 m within the transect at the standard 0.6 water depth using the top-setting wading rod near the injection site. Discharge was then calculated to estimate the amount of salt to be added to the Marriott bottle. One modified Marriott bottle (20 L polypropylene bottle) with a constant effusion velocity was used for injection. The target solute injection concentration was determined by the discharge of the stream, spike concentration, emitter rate, volume of the Mariotte bottle, and molecular weight of each solute for each experiment.

To quantify the in-stream uptake of N as ammonium (NH_4-N), nitrate plus nitrate (NO_3-N + NO_2-N), and soluble reactive phosphorus (SRP), 12 separate short-term solute addition experiments (six separate experiments per reach) were conducted during the winter season, January through March 2022, and the summer season, June through July 2022 (Table 4). Conductivity measurements were collected as part of the injection studies, and the measurements were also used to understand the spatial and temporal heterogeneity of stream flow and water quality. The duration of each injection depended on the discharge rate, but the plateau was generally reached over 1 h for the two selected reaches on all occasions. The plateau is the point in time when the downstream concentration of salt reaches a steady state condition, which is observed through the conductivity measurements using YSI ProDSS. The solute solution varied for each experiment date because the concentration of each solute depends on the discharge of the stream reach. The solute solution, containing sodium phosphate monobasic monohydrate ($NaH_2PO_4{\cdot}H_2O$, as a phosphate source), ammonium nitrate (NH_4NO_3, as an ammonium and nitrate source), and sodium chloride (NaCl, as a conservative tracer source), was added at constant flow rate using a Mariotte bottle at the injection site (beginning of the reach) on each separate experiment. This was conducted according to the standard procedures outlined in the Stream Solute Workshop (1990). NaCl was used to correct for dilution in NH_4-N, NO_3-N + NO_2-N, and SRP concentrations within the reach because chloride concentration remains relatively constant as water moves downstream. Spike concentrations of SRP (0.03 mg/L), NH_4-N (0.1 mg/L), NO_3-N + NO_2-N (0.1 mg/L), and NaCl (3 mg/L) solutes were set on each sampling date based on previous studies [e.g., Stream Solute Workshop, 1990; Haggard et al., 2001; Chaubey et al., 2007; McMillan et al., 2014; where a small spike concentration is set according to the demand for the nutrient. Excess nutrient concentration is not prescribed to avoid oversaturation of nutrients in the stream reach.].

The discharge of the stream was again calculated at the injection site on the injection day. If the discharge was within a 10% difference from the previous reading taken the day before injection, no change was made to the Mariotte bottle's emitter rate. Weather conditions were recorded for reference.

The Mariotte bottle was placed ~1.5 m upstream of the first sampling site inside the tripod consistently for all the experiments. Before injection started, background water samples were collected using acid-washed 250-mL HDPE wide-mouth bottles at each sampling site. Samples were taken in constricted, well-mixed areas of the stream, or riffles. At the RES sites, sampling was conducted at distinguished riffles, specifically at the head and the tail, where there was free-flowing, mixing water.

Water samples were taken at each site using three 250-mL bottles at the right, middle, and left portions of the stream channel. All three samples were mixed, and one composite sample was collected and analyzed. Sampling at each site was conducted with minimal disturbance by sampling from downstream to upstream. Water samples were filtered with 0.45-μm pore size filters before the injection started. The conductivity sensors were completely submerged in the nearest rifle next to the Mariotte bottle and at the most downstream sampling site. The Mariotte bottle was turned on once the sensors were in place. The conductivity readings were recorded at the most upstream site and the downstream site. Surface water plateau samples were collected in a manner similar to that of background samples when the conductivity reached a plateau at the downstream site. Once the samples were collected, injection was stopped, and conductivity measurements were recorded until conductivity readings returned to background conditions.

All water samples were filtered and acidified, if needed, then stored at < 4°C until delivery to the Arkansas Water Resources Center's certified water quality lab (https://awrc.uada.edu/water-quality-lab/). Filtered water samples were analyzed for nitrate, sulfate, fluoride, and chloride using an ion chromatograph (EPA Method 300.0, Dionex System 1600). Filtered, acidified (pH < 2) water samples were analyzed for dissolved nutrients on a wet chemistry autoanalyzer (Skalar Sans++ System), including SRP (EPA Method 365.1, method detection limit of 0.004 mg/L), NH_4-N (EPA Method 351.2, method detection limit of 0.020 mg/L), and NO_3-N + NO_2-N (EPA Method 353.2, method detection limit of 0.017 mg/L) and used for analysis described in Section 2.4. Unfiltered water samples were digested using the persulfate, autoclave method (APHA 4500–P–J), and then the digested samples were analyzed on the wet chemistry autoanalyzer for SRP and NO_3-N + NO_2-N following previously described methods to get TP and TN concentrations in the unfiltered samples. These measurements were used to obtain the concentrations of the three nutrient variables listed. The SRP values obtained from the filtered, acidified samples were used for the current analysis.

2.3 Calculations for nutrient uptake variables

Nutrient uptake length, a measure to study nutrient dynamics in a stream, is the distance a nutrient molecule travels in dissolved form before being taken up from the water column. It is calculated using the nutrient spiraling approach (Stream Solute Workshop, 1990). The nutrient uptake length, S_w, is derived from k, which is the uptake rate constant (m^{-1}). A simple first-order rate equation was used to determine the length (m) a nutrient travels in the water column because the proportion of added nutrients generally decreases exponentially with distance and is expressed as follows:

$$C_x = C_o{*}e^{-kx}, \tag{1}$$

where C_x is the diluted corrected concentration at each of the sampling locations, x is the distance downstream from the injection point, C_o is the diluted corrected concentration at the most upstream sampling location below the injection point, and k is the uptake rate constant that is determined as the slope of a line representing the proportion of nutrients remaining in the water column versus distance downstream (Stream Solute Workshop, 1990). S_w is calculated as the inverse of k for each injection as follows:

$$S_w = -1/k. \tag{2}$$

S_w is corrected to the average of the background nutrient and tracer concentrations and the change in nutrient concentration with distance downstream. Variations in hydrological properties such as stream discharge affect the retention efficiency of a stream, which impacts S_w. When velocity is considered with S_w, the uptake rate coefficient, K_c (L/s), is used to account for such variation (Stream Solute Workshop, 1990). The greater the K_c, the shorter the S_w, resulting in increased nutrient processing within the stream. In cases where nutrient concentrations did not decrease after applying dilution correction, these data were not included in the analysis.

The nutrient trapping efficiency (TE%), which represents the percentage of nutrients trapped within a stream reach during the nutrient injection period, is calculated by the following

$$TE\% = \left[(US_C - DS_C)/US_C\right]{*}100, \tag{3}$$

TABLE 5 Nutrient uptake length, S_w (m), across all sampling sites for select nutrients, followed by the nutrient uptake rate, K_c (1/s), mass transfer coefficient, V_f (m/s), and *p*-value for the uptake length of each nutrient.

	SRP				NH_4-N				NO_3-N + NO_2-N			
	S_w	K_c	V_f	*p*-value	S_w	K_c	V_f	*p*-value	S_w	K_c	V_f	*p*-value
January												
RES	248	0.00009	0.00002	0.0009	310	0.0001	0.00002	0.006	1951	0.00001	0.000003	0.76
URE	32	0.001	0.0003	0.004	47	0.0009	0.0002	0.008	a	a	a	a
February												
RES	a	a	a	a	a	a	a	a	a	a	a	a
URE	a	a	a	a	429	0.0001	0.00002	0.393	89	0.0007	0.0001	0.09
March												
RES	a	a	a	a	779	0.0001	0.00001	0.733	a	a	a	a
URE	1140	0.00003	0.00001	0.72	218	0.0001	0.00003	0.656	a	a	a	a
June												
RES	77	0.0004	0.00005	0.000026	75	0.0004	0.0005	0.15	a	a	a	a
URE	3059	0.00001	0.000003	0.74	a	a	a	a	a	a	a	a
Early July												
RES	a	a	a	a	102	0.0003	0.00003	0.12	a	a	a	a
URE	253	0.0004	0.00008	0.09	a	a	a	a	a	a	a	a
Late July												
RES	a	a	a	a	a	a	a	a	a	a	a	a
URE	a	a	a	a	a	a	a	a	a	a	a	a

a Value not reported and excluded from analyses because a decrease in nutrient concentration was not observed.

where US_c is the plateau-corrected upstream concentration of the nutrient and DS_c is the plateau-corrected downstream concentration of the nutrient, meaning the plateau concentrations of the upstream and downstream locations are corrected by the average of all background concentration samples. The estimation of TE% can also be used as an indicator of nutrient dynamics in streams. A similar calculation method can be used to estimate the TE% of restored stream features [e.g., engineered pool] by using the data collected from the nutrient injection experiments. The water samples taken from the upstream and downstream sections of the pools in the RES reach during the plateau were collected, corrected to the background concentration of the sampling site, and used to calculate the TE% of the pools.

2.4 Statistical analysis

The same reaches, sites, and pools were repeatedly measured throughout the study, resulting in pseudoreplication within the dataset for statistical analysis. Nutrient uptake length was calculated based on Equation 1, which used $\alpha = 0.05$ for a regression analysis between background corrected nutrient concentration and distance from the injection site, with the associated *p*-value reported for significance. Section 3.1, In-stream nutrient retention, analyzes the difference between nutrient uptake length, discharge, and water temperature using Spearman's ρ test. Nutrient uptake lengths were compared across reaches ($n = 2$) using a t-test assuming unequal variances and seasons assuming equal variances. Reported statistics were determined to be significant at $\alpha = 0.05$. Section 3.2, Nutrient trapping efficiency, reports the means of measured values with corresponding standard deviations to represent error and uses a regression analysis to model the relationship between TE% (dependent variable) and pool length (independent variable) in the RES reach. Section 3.3, Background water chemistry measurements, reports the means of measured values with corresponding standard deviations to represent error. It uses t-tests to measure spatial and seasonal variation for reported variables and a regression analysis to examine the relationship between background conductivity for each reach (dependent variable) and water temperature (independent variable). Section 3.4, Conductivity analysis and anomalies, reports the means of measured values with corresponding standard deviations to represent error. It uses t-tests to measure seasonal variance of URE mean conductivity, assuming equal variance, and a regression analysis to examine the relationship between background conductivity (dependent variable), water temperature (independent variable), and discharge (independent variable). Box plots were used to understand the distribution of conductivity measurements during the injection. JMP Pro version 16.0 software was used for statistical data analysis (SAS Institute, Cary, North Carolina).

3 Results

3.1 In-stream nutrient retention

SRP uptake length was reported two of six times in the RES reach and three of six times in the URE reach (Table 5). The RES reach

TABLE 6 Whole-stream trapping efficiency (TE%) of plateau soluble reactive phosphorus (SRP), ammonia (NH_4), and nitrate plus nitrite nitrogen (NO_3-N + NO_2-N) concentrations at each site in restored (RES) and unrestored (URE) reaches of Richland Creek during injections.

	January	February	March	June	Early July	Late July
SRP						
RES	31	a	a	36	a	a
URE	55	a	a	a	29	a
NH_4-N						
RES	25	a	24	43	34	a
URE	23	a	a	0.4	a	a
NO_3-N + NO_2-N						
RES	7	a	a	a	a	a
URE	a	38	a	a	a	a

a Value not reported and excluded from analyses because a decrease in nutrient concentration was not observed.

showed shorter SRP uptake lengths and higher uptake rate and mass transfer coefficient in June (77 m), when compared to the URE reach, which showed shorter uptake lengths and higher uptake rate and mass transfer coefficient in January (32 m) and early July (253 m) injections (Table 5). Together, there was no strong correlation between the SRP uptake length and discharge for the RES and URE reaches ($\rho = 0.51$, $p > 0.05$, n = 6) nor was there a significant relationship between SRP uptake length and water temperature ($\rho = 0.21$, $p > 0.05$, n = 6) throughout the study period. The mean SRP uptake lengths of the RES (163 m) and URE (1121 m) reaches did not differ using a *t*-test assuming unequal variances (t = 1.38, $p > 0.05$, $n_{RES} = 2$, $n_{URE} = 4$). No seasonal variation was found between the combined SRP uptake lengths of the RES and URE reaches throughout the study, using a *t*-test assuming unequal variances (t = 0.64, $p > 0.05$, n = 3).

The NH_4-N uptake occurred more in the RES than in the URE reach. NH_4-N uptake occurred four of six times within the RES reach, and it occurred three of six times in the URE reach (Table 5). The RES reach had the shortest NH_4-N uptake length on two of six occasions in June (75 m), and early July (102 m) injections, while the URE reach had the shortest in January (47 m), February (429 m), and March (218 m) injections (Table 5). Together, the NH_4-N uptake lengths for the RES and URE reaches did not correlate with water temperature ($\rho = 1.0$, $p > 0.05$, n = 7) throughout the study, nor was there a correlation between NH_4-N uptake and discharge ($\rho = 0.57$, $p > 0.05$, n = 7).

The low number of samples limited the strength of these comparisons. Sample sizes were limited because results were excluded where a decrease in nutrients was not observed (Table 5) due to illicit pollutant discharge on the planned experimental date. The shortest NH_4-N uptake length at the URE reach in January (47 m) may reflect premature sampling (before the plateau was observed).

The mean NH_4-N uptake lengths between the RES (317 m) and URE (231 m) reaches had no statistical difference using a *t*-test assuming unequal variances (t = 0.84, $p > 0.05$, $n_{RES} = 3$, $n_{URE} = 4$). When the NH_4-N uptake lengths were combined from the RES and URE reaches for each season, the winter NH_4-N uptake lengths were longer than the summer, but the difference was not statistically significant using a *t*-test assuming unequal variances (t = 2.17, $p > 0.05$, $n_{winter} = 5$, $n_{summer} = 2$).

There were only two instances of NO_3-N + NO_2-N uptake with subsequent uptake rate and mass transfer coefficients calculations in January (1,951 m) in the RES reach and February (89 m) in the URE reach (Table 5). The Spearman's ρ test could not be used to assess the relationship between nutrient uptake length, discharge, and water temperature, and a *t*-test could not be used to differentiate nutrient uptake length between sites and seasons, due to a lack of NO_3-N + NO_2-N uptake observations.

3.2 Nutrient trapping efficiency

There were two occasions of whole-stream TE% of SRP in the RES (34% ± 4%) and URE (42% ± 18%) reaches. From these two occasions, the RES TE% was higher in the summer while the URE TE% was higher in the winter (Table 6). The RES reach performed well with four of six observations of whole-stream TE% of NH_4-N (32% ± 9%), with higher TE% in the summer. The TE% for NH_4-N in the URE reach was observed only two of six times (12% ± 16%), with higher TE% in the winter (Table 6). The whole-stream TE% of NO_3-N + NO_2-N in the RES and URE reaches both showed one occasion in the winter (Table 6). The average TE% of SRP in the step pool of the RES reach was similar in the winter (6% ± 4%) and summer (5% ± 5%). The average SRP TE% in the constructed pools in the winter (11% ± 6%) and summer (13% ± 14%) was similar (Table 7). The TE% of NH_4-N in the step pool of the RES reach was more than three times higher in the summer (45% ± 37%) than the winter season (13% ± 13%), and it was also higher in the constructed pools in the summer (52% ± 18%) than the winter season (16% ± 8%) (Table 7). The RES reach had a TE% for NO_3-N + NO_2-N in step pools (32% ± 32%) and constructed pools (22% ± 16%) in the winter, while the summer only had one instance of uptake (Table 7).

Trapping efficiency and pool length (i.e., the longitudinal distance of the pool) were compared to determine if a higher trapping efficiency could be associated with a longer pool length. The step pool and constructed pools in the RES reach showed a positive exponential response and a significant relationship between

TABLE 7 Trapping efficiency (TE%) in pools for the RES reach during injections.

		January	February	March	June	Early July	Late July
				SRP			
	POOL 1	b	b	b	a	9	a
Step pool begins	POOL A	a	1	a	a	a	a
	POOL B	a	3	a	15	a	a
	POOL C	a	6	11	a	3	2
	POOL D	a	1	a	a	2	3
Step pool ends	POOL E	6	9	10	4	a	a
	POOL 2	18	a	a	9	a	a
	POOL 3	7	a	8	1	a	33
				NH_4-N			
	POOL 1	b	b	b	25	66	a
Step pool begins	POOL A	a	a	a	a	a	92
	POOL B	a	9	31	a	a	a
	POOL C	a	a	36	67	67	a
	POOL D	a	12	4	10	a	16
Step pool ends	POOL E	2	4	4	a	a	19
	POOL 2	21	12	a	65	67	a
	POOL 3	6	a	23	35	52	a
				NO_3-N + NO_2-N			
	POOL 1	b	b	b	a	52	a
Step pool begins	POOL A	a	a	a	a	a	a
	POOL B	a	67	a	a	a	a
	POOL C	a	a	a	a	a	a
	POOL D	a	a	a	a	a	a
Step pool ends	POOL E	6	a	22	a	a	a
	POOL 2	32	39	a	a	a	a
	POOL 3	3	15	a	a	a	a

a Value not reported and excluded from analyses because a decrease in nutrient concentration was not observed.
b Eliminated the first site because the samples were too close to the injection site and caused dilution.

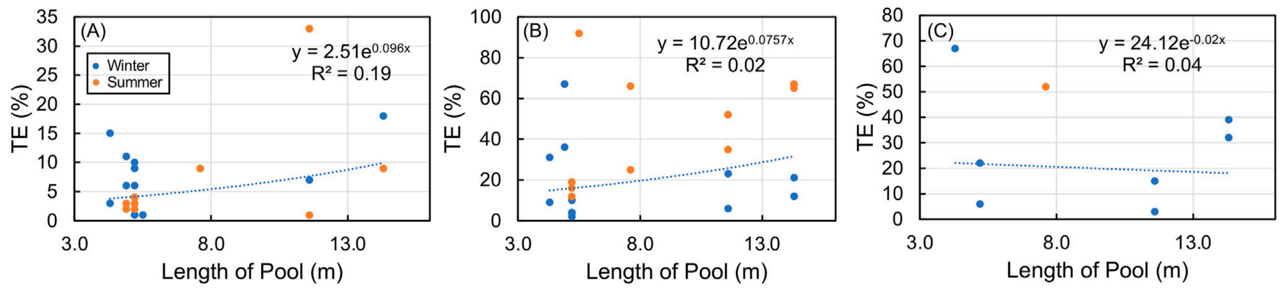

FIGURE 3
Whole-stream trapping efficiency (TE) in RES pools compared with each pool length for **(A)** soluble reactive phosphorus SRP, **(B)** NH_4-N, and **(C)** NO_3-N + NO_2-N in the summer (orange dot) and winter (blue dot).

the TE% of SRP and pool length (r = 0.44, $p < 0.05$, n = 21), while no relationship existed between the TE% of NH_4-N (r = 0.14, $p > 0.05$, n = 24) throughout both the winter and summer seasons (Figures 3A, B). The TE% of NO_3-N + NO_2-N also showed no relationship with pool length (r = 0.20, $p > 0.05$, n = 8) (Figure 3C). When considering all the pools in the RES reach, the highest average TE% for SRP and NH_4-N was in the summer, while the highest average TE% for NO_3-N + NO_2-N was in the winter.

3.3 Background water chemistry measurements

As expected, the water temperature increased from winter (average = 13 ± 3°C) to summer (average = 22 ± 1°C) in both sites (Table 5). The conductivity in the RES and URE reaches varied spatially between sites, likely because sampling was conducted on different dates (t = 2.68, $p < 0.05$, n = 6) (Table 2). The RES reach

TABLE 8 Event average of background water chemistry characteristics of restored (RES) and unrestored (URE) reaches of Richland Creek. Parameters include NH_4-N, NO_3-N + NO_2-N, DIN, SRP, TN, TP, F, Cl, SO_4, and DO in mg/L, water temperature (°C), conductivity (μS/cm), and pH.

	NH_4-N	NO_3-N + NO_2-N	DIN	SRP	TN	TP	F	Cl	SO_4	DO	Water Temp.	Conductivity	pH
Winter													
January													
RES	0.028	1.88	1.90	0.028	1.98	0.027	0.16	11.95	11.17	12	11.9	111	6.0
URE	0.026	1.96	1.98	0.022	2.07	0.026	0.17	14.19	10.14	12	9.3	100	5.9
February													
RES	0.025	1.80	1.83	0.028	1.86	0.025	0.16	12.69	13.34	12	14.4	122	6.3
URE	0.020	1.80	1.82	0.029	1.92	0.029	0.17	12.48	10.6	12	11.8	98	6.6
March													
RES	0.026	1.85	1.88	0.031	1.94	0.045	0.18	13.06	12.04	12	16.7	124	6.6
URE	0.022	1.90	1.93	0.036	1.98	0.051	0.19	12.25	10.21	11	15.9	110	6.6
Summer													
June													
RES	0.034	1.78	1.81	0.046	1.97	0.059	0.21	11.47	10.32	10	21.5	118	6.6
URE	0.032	2.07	2.10	0.041	2.10	0.060	0.18	11.79	8.93	10	21.7	114	6.7
Early July													
RES	0.033	1.66	1.69	0.051	1.87	0.079	0.25	11.09	10.95	9	22.4	131	6.5
URE	0.032	1.97	2.00	0.035	1.90	0.069	0.19	11.76	8.43	9	22.2	116	6.6
Late July													
RES	0.036	1.76	1.80	0.050	1.88	0.071	0.18	10.89	8.12	9	23.5	123	6.9
URE	0.042	1.81	1.86	0.055	2.05	0.076	0.24	11.66	9.73	9	22.8	120	6.7

lacked seasonal variation in background conductivity, while background conductivity in the URE reach increased in the summer (t = 3.40, $p < 0.05$, n = 3). Regression analysis confirmed that background conductivity increased as water temperature increased from winter to summer months for the URE reach (r = 0.95), but temperature and conductivity correlated poorly for the RES reach (r = 0.61).

Monthly averages of F, SO_4, and Cl in both sites did not exceed the EPA National Secondary Drinking Water Regulations (NSDWRs) recommendations (F: 2.0 mg/L, SO_4 and Cl: 250 mg/L) (Table 8) (Environmental Protection Agency, 2022a). Spatial differences in background concentration of NH_4-N were not observed between the RES and URE reaches (Table 8). The average background concentration of NH_4-N varied temporally for the RES (t = 6.41, $p < 0.05$, n = 3) and URE (t = 3.36, $p < 0.05$, n = 3) reaches, with the lowest concentrations occurring in the winter months in the RES (0.027 ± 0.001 mg/L) and URE (0.02 ± 0.003 mg/L) reaches. Higher average NH_4-N concentrations were measured in summer for the RES (0.035 ± 0.001 mg/L) and URE (0.04 ± 0.006 mg/L) reaches (Table 8). The average background concentration of NO_3-N + NO_2-N varied spatially between the RES and URE reaches throughout the study period (t = 2.47, $p < 0.05$, n = 6), with no difference associated with season (Table 8). The RES and URE reaches differed (t = 2.16, $p < 0.05$, n = 6) in the average background TN concentration, but TN was similar within each reach, regardless of season (Table 8). The background TN concentration is higher in the URE than in the RES reach.

TP concentration did not vary between the RES and URE reaches (t = 0.07, p > 0.05, n = 6) but did vary by season (both reaches reported t = 2.13, $p < 0.05$, n = 3), with higher background TP concentration in the RES reach in the summer (0.07 ± 0.01 mg/L) than the winter (0.03 ± 0.01 mg/L) and in the URE reach in the summer (0.07 ± 0.008 mg/L) than the winter (0.04 ± 0.01 mg/L) (Table 8). When comparing average background concentrations of SRP, both the RES and URE reaches concentration ranges were similar, from 0.02 mg/L to 0.05 mg/L (average = 0.04 ± 0.011) (Table 8). The SRP was significantly higher in the summer months than in winter months for the RES reach (t = 10.95, $p < 0.05$, n = 3), while the URE reach showed no significant seasonal difference (Table 8). In all the experiments, the TN:TP ratio exceeded the 16:1 baseline, making TN abundant and TP deficient (Table 4). The TN:TP ratio was similar between the RES and URE sites and seasons, with the RES reach ranging from 24:1 to 74:1 (46 ± 22) and the URE reach ranging from 27:1 to 79:1 (46 ± 22) throughout the study period (Table 4).

3.4 Conductivity analysis and anomalies

Conductivity readings in the RES reach took approximately 42 min (± 11) to observe a rise in conductivity, while the URE reach took an average of 30 min (± 7) throughout the experiments (Jordan, 2023). The average plateau time between the RES and URE reaches varied spatially by 60% on average throughout the sampled months. The RES reach plateaued after an average of 86 min (± 4), and the URE reach plateaued after an average of 47 min (± 6) during the experiments (Jordan, 2023). There was a distinct contrast in the mean conductivity between the winter months in the URE reach (Figure 4). The mean conductivity in March was 7% higher than in the other winter months. The summer months showed a significant

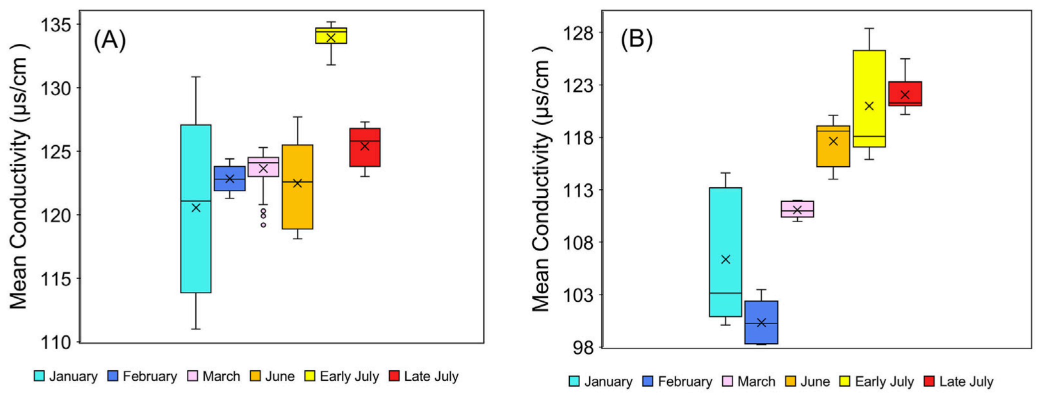

FIGURE 4
Boxplot of stream conductivity at **(A)** RES and **(B)** URE reaches of Richland Creek in the winter and summer seasons. The box plot shows conductivity between the upper and lower quantiles for monthly experiments. The line in the box represents the median of conductivity. The error bars show the maximum and minimum non-outliers of conductivity.

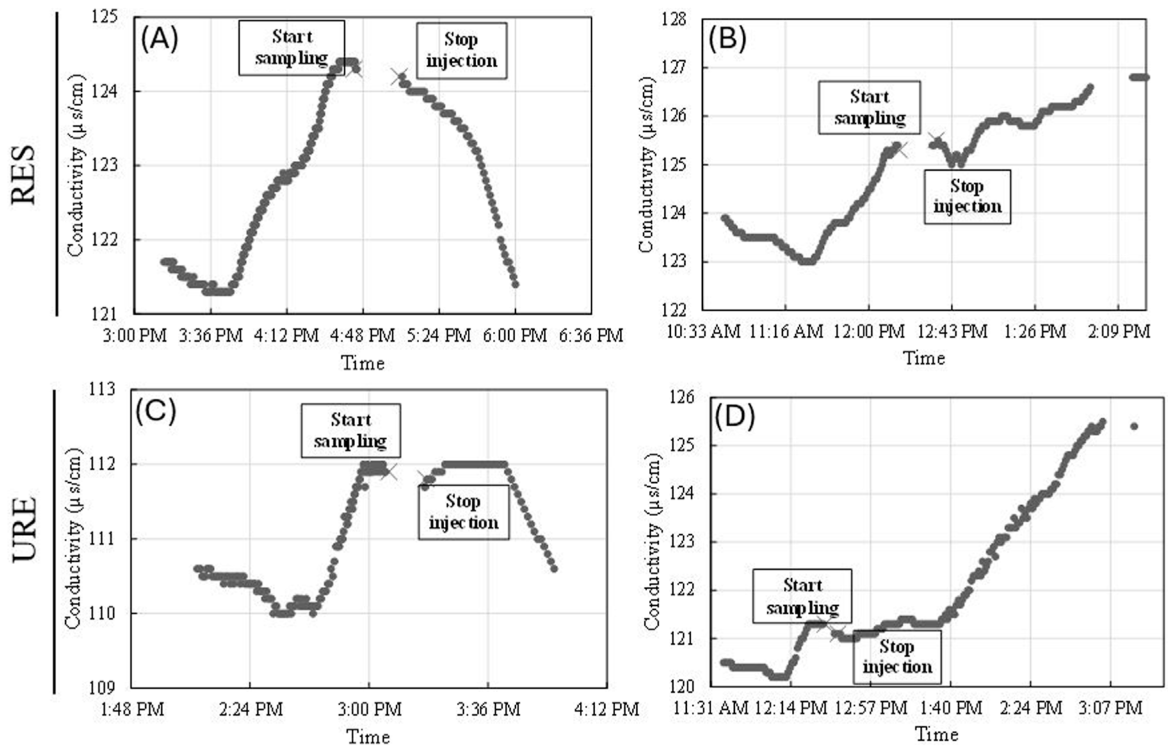

FIGURE 5
Conductivity curves representing **(A)** restored February injection, **(B)** restored late July injection, **(C)** unrestored March injection, and **(D)** unrestored late July injection.

increase ($t = 3.40$, $p < 0.05$, $n = 3$) in mean conductivity in the URE reach (Figure 4). The seasonal difference of 13% between winter and summer mean conductivity in the URE reach was likely due to the strong correlation of background conductivity with water temperature ($r = 0.96$, $p < 0.05$, $n = 6$) and discharge ($r = 0.92$, $p < 0.05$, $n = 6$).

Once samples were collected after the EC plateaued, the injection was terminated, and, in general, the conductivity levels gradually returned to the initial background levels. While conductivity measurement returned to background readings during the winter experiments in the RES and URE reaches (Figure 5), conductivity behaved differently for the RES and URE reaches in the summer season (Figure 5). The conductivity measurements began to rise after terminating the injection in the RES and URE reaches in the summer months (Figure 5). In late July, the RES reach had a gradual rise (slope = 0.012) in conductivity after presumed plateau of 125 µS/cm, which reached a final conductivity of 127 µS/cm; the URE reach had a steeper rise in conductivity (slope = 0.27) after reaching a plateau of 121 μS/cm, which reached a final background conductivity of 125 μS/cm after the injection was terminated. Normally, conductivity should have returned to initial readings.

4 Discussion

4.1 In-stream nutrient retention

The SRP uptake in the RES and URE reaches was responsive during some of the sampled months, even though these sites consisted of predominantly gravel material. The lack of fine sediment due to rip rap in the RES reach and bedrock in the URE reach affects P cycling and reduces the amount of P sorption possible in the stream. More than 70% of the stream sediment at both sites was gravel, while less than 30% was sand. Restoration efforts typically focus on the removal of fine sediments from the natural gravel substrate (Morgan et al., 2019). The predominant small cobble and gravel bed in the RES and URE reaches may have resulted in fewer observations of SRP uptake. The lack of reported nutrient uptake lengths among the 12 experiments was possibly due to either significantly higher background nutrient concentration, likely from additional inputs from urban land use or not injecting a sufficiently high injection concentration of nutrients compared to the background concentration during the injection experiment (Table 5, explained under "a"). The RES reach maintained the shortest SRP uptake length in June; this could be attributed to higher rates of primary production during periods of higher light availability that exert a higher demand for P (Mulholland, 2004). The URE reach may have had the shortest SRP uptake in January because the TN:TP ratio was the highest (79: 1, Table 4) in this sampled month, and therefore, the system was presumed to be phosphorus-limited with P in demand. The SRP uptake lengths in the RES reach were within the same range as five low-order restored urban streams in the Piedmont region of North Carolina (McMillan et al., 2014). Other streams had similar SRP uptake length; however, they were in low-order agricultural systems with a mixture of forest and pasture land uses (Marti and Sabater, 1996; Haggard et al., 2001; Chaubey et al., 2007).

Variation in discharge, previous rainfall events, weather conditions, and export of nutrients from the watershed could all assist in the complex interaction of nutrient availability in the water column and may influence the differences in NH_4-N uptake lengths for each site. The NH_4-N uptake in the RES reach was marked on more occasions than in the URE reach (Table 5), likely due to increased travel time attributable to transient storage and pools over the stream length. NH_4-N uptake may have been limited in the URE reach due to a lack of transient storage area, stream channelization, or disconnection to in-channel and riparian vegetation, all of which likely contribute to a decline in NH_4-N retention in small drainage streams (Bukaveckas, 2007; Le et al., 2018). These uptake lengths were within the range of other reported NH_4-N uptake length values in agricultural systems [e.g., forested and pasture land use] (Haggard et al., 2001; Simon and Benfield, 2002; Chaubey et al., 2007).

NO_3-N + NO_2-N was the dominant form of nitrogen for both sites (Table 5) and contributed to the high TN:TP ratio in the winter season (Table 2). When considering this high ratio, nitrogen was not limited during the winter sampling period. The discharge was high in January, which may have contributed to the detectable transport of this nutrient. A shorter uptake length is expected in this relatively slow-moving urban stream because other studies have documented longer nutrient uptake lengths with increasing discharge (Lautz and Siegel, 2007; Valett et al., 1996). However, the data were not consistent with expectations. Results from a previous study showed no retention of NO_3-N + NO_2-N and increased NO_3-N + NO_2-N concentration downstream of injection because of nitrification of injected NH_4-N (Chaubey et al., 2007).

4.2 Nutrient trapping efficiency

The overall mean TE% for all pools for SRP and NH_4-N was greater in the summer than in the winter, while the mean TE% for all pools for NO_3-N + NO_2-N was greater in the winter. Biotic activities are generally dominant during summer months. Pools play a crucial role in nutrient retention, with biotic activities in these areas aiding in the trapping and retention of nutrients. Higher TE% for NH_4-N may have been a result of restoration efforts in the RES reach. These TE% values could potentially be used for designing stream restoration projects and for nutrient crediting in such initiatives. Future research should investigate whether pool lengths exceeding 25 m are necessary to enhance the uptake of NH_4^+-N and NO_3^--N + NO_2^--N in urban headwater streams. To the best of the authors' knowledge, no previous studies have assessed nutrient TE% in different features within a restored site, particularly those with features like pools and step pools. The methodology and results obtained for pool length can also be applied when designing pools for restoration purposes.

4.3 Background water chemistry measurements

Observed discharges from stormwater pipes from winter to summer sampling dates in the RES reach (Table 2) may indicate that ecological processes that influence stream conductivity are less influenced by temperature. Inputs of ions through these stormwater pipes in the highly urban area of the RES and URE reaches may have impacted stream conductivity, which is a good indicator of human activity (O'Brien and Wehr, 2010; Wu et al., 2015; Köse et al., 2014). The higher background concentration of NH_4-N in the summer was most likely due to increased temperatures stimulating ammonification in water along with other co-occurring ecological processes (Racchetti et al., 2011). This could also be a result of impaired water quality from nitrogen enrichment from the surrounding urban watershed (Dodds and Smith, 2016). Visual observations indicated the presence of periphyton mats in the water column and on rip rap during the summer sampling period. Bacterially mediated reactions are sensitive to seasonal changes in stream water temperature, which affect the nitrate availability in the water column. The relationship between water temperature and nitrification within stream ecosystems is a well-researched topic. Studies show that temperature has a significant positive correlation with nitrate loss, meaning higher temperatures create more denitrification reactions within a stream system (Hill, 1988; Rasmussen et al., 2011). Processes involved in nitrate loss could be a consequence of a hydrologically disconnected floodplain, a lack of riparian area, or a result of the constant stormwater pipe discharge from upstream; however, Wolf et al. (2013) found that hydrological connectivity increases nitrate inputs. Additionally, excess TN and TP loads are common in developed areas, where stormwater runoff and soil-bound phosphorus are carried off impervious surfaces (Carpenter et al., 1998), potentially impacting nutrient cycling in stream systems.

The Redfield ratio is a well-established principle in ecology that describes the stoichiometric proportion of TN to TP in aquatic ecosystems, where a ratio of 16:1 is typically observed. This ratio is central to the principle of nutrient limitation, which states that the element in shortest supply relative to demand will limit biological productivity. However, the stoichiometric proportion of TN to TP in an urban stream can be highly variable due to increased nutrient inputs from anthropogenic sources, which can lead to an alteration of the typical Redfield ratio. A P-limited system is typical for surface waters. The RES and URE reaches exceeded the biological need for nitrogen for each sampled event (Table 5). This is common for urban streams because wastewater inputs and stormwater runoff have increased the downstream N exported in streams in recent years, which has influenced N demands (Reisinger et al., 2016). The lower ratio in the summer than winter in the RES and URE reaches could be due to the utilization of nutrients due to planktonic demand. Higher temperatures create a higher biological demand for nutrients. However, there is a higher concentration of TP in the water column, yet the TN:TP ratio shows that there is a demand for phosphorus. These unparalleled relationships could reflect the complex process during the growth and decay of algae when transitioning from summer to winter seasons.

4.4 Conductivity analysis and anomalies

The RES reach took the longest time to reach a plateau during early and late July (1 h 31 min and 1 h 30 min), most likely due to the combined effects of low discharge (Table 2), less rainfall (Table 3), stream restoration features such as riffles, pools, step pools, and higher temperatures in the summer month (Table 4). Conductivity is closely related to water temperature, and during the URE sampling period, March showed higher conductivity than other winter months. This is likely because the water temperature in March was seven degrees higher than in January and four degrees higher than in February (Table 5). The summer months showed a significant increase in mean conductivity (Figure 4), likely due to higher temperatures and lower discharge rates, which concentrated salts in the water column (Hayashi et al., 2012).

Diurnal fluctuation in conductivity has been documented in small or first-order streams in connection with enhanced photosynthetic processes, salt concentration via evaporation/evapotranspiration (evaporitic enrichment), or daily discharge variations from wastewater treatment plants during daylight hours (Hayashi et al., 2012; Calles, 1982). The elevated conductivity of the studied urban stream could be explained by these studies; however, the increase in conductivity over time in summer requires further examination, including evaluation of the influence of temperature on conductivity in urban environments. Additionally, the influence of groundwater discharge is believed to be greater during the dry season (summer) in perennial river systems (Le Maitre and Colvin, 2018). The RES and URE reaches experienced the lowest average precipitation in the summer (Table 3), making this period a dry season; however, the influence of groundwater on conductivity cannot be confirmed, only hypothesized.

5 Conclusion

The RES and URE urban streams retained SRP, although their streambed was predominantly small cobble and gravel. SRP retention likely occurred because Richland Creek is a P-limited system, reflected in the TN:TP ratio, with the slow-moving waters characteristic of a first-order watershed. In general, the summer results clearly indicated greater SRP and NH_4-N retention in the RES reach than in the URE reach, highlighting the effectiveness of restoration on nutrient retention. Significant retention of NO_3-N + NO_2-N was not observed in either the RES or the URE reaches, indicating that these sites may be a source of downstream NO_3-N + NO_2-N transport.

These whole-stream nutrient (e.g., SRP and NH_4-N) TE% experiments revealed that this stream traps more SRP and NH_4-N during the summer than winter. The RES reach was more effective than the URE reach when compared for nutrient TE%, suggesting that restoration has positive impacts on nutrient retention. The nutrient TE% in engineered pools at the RES reach indicated that SRP and NH_4-N uptake were more effective in the summer. Additionally, engineered pools in the RES reach were more efficient at assimilating NH_4-N than the other nutrients.

While the current study provides some guidance, initial data, and assessment of the method to quantify the effectiveness of stream restoration on nutrient dynamics, nutrient retention, whole-stream nutrient TE%, and nutrient TE% in engineered pools within streams, additional datasets in multiple restoration settings across different regions are required to make regulatory and policy recommendations. Additional research in multiple stream restoration settings [e.g., across different types and sizes] with analysis and interpretational data would be beneficial to creating such a regulatory practice. To assess the efficacy of stream restoration on nutrient dynamics, it is essential to account for the influence of periphyton, microbial activities, microhabitats, and microbiomes. The results from this study could be included when designing stream restoration features and nutrient reduction designs that could be linked to stream restoration credits. Therefore, conducting experiments during the summer, when these processes are most active, is likely to yield critical insights into their efficacy. When conducting nutrient injection studies to understand the efficacy of nutrient retention in stream restoration, it is important to ensure that the reach length adheres to the principles outlined in the Stream Solute Workshop (1990).

Data availability statement

The raw data supporting the conclusion of this article will be made available by the authors, without undue reservation. Should a manuscript be developed based on this data, the authors and the data collectors request appropriate acknowledgment and authorship.

Author contributions

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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The author(s) declare that no Generative AI was used in the creation of this manuscript.

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