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Regenerative stormwater conveyance (RSC) for reducing nutrients in urban stormwater runoff depends upon carbon quantity and quality

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Abstract

Regenerative stormwater conveyance (RSC), a relatively new stormwater management approach, is extensively implemented throughout the mid-Atlantic for nutrient control, but little is known of its pollutant reduction capabilities and controlling factors. This study examined effects of organic carbon (C) quantity and quality on stream water quality and nutrient retention at two RSCs near Annapolis, Maryland, USA by comparing longitudinal changes in water quality at paired restored and unrestored stream reaches, and conducting lab experiments simulating RSC processes. Results showed that RSCs consistently had lower dissolved oxygen saturation (DO%) and pH relative to nearby unrestored streams, probably due to release of labile dissolved organic carbon (DOC). At one RSC, with high nitrate (NO_3^-) inputs, retention of N (16–37%) and release of DOC (18–54%) were observed with the highest retention of N during summer, and the rates of N retention and DOC release were larger than that of the adjacent unrestored tributary (N: 5–8%, DOC: <18%). At another RSC site with lower NO_3^- concentrations, N retention and DOC release were not apparent. Mesocosm experiments showed that NO_3^- retention varies with organic C quantity and quality depending on incubating temperature; retention of total N did not increase with organic C due to release of other N species (e.g., organic N). Lab mesocosms showed an increase in the release of soluble reactive phosphorus (SRP) with increasing organic C quantity and quality. However, field measurements did not show any evidence of SRP release at RSCs. The changes in SRP concentrations in streams seemed to be a function of iron levels and leaf litter inputs, but control factors for SRP warrant further investigation. This study suggests that RSC as a restoration

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approach may be effective for reducing N depending upon C quantity and quality as well as water temperature and N levels.

Graphical Abstract

alt-text: Unlabelled Image

Keywords

Regenerative stormwater conveyance; Nitrogen; Phosphorus; Organic matter; Dissolved oxygen

1 Introduction

Excess nutrient (nitrogen and phosphorus) loading from agricultural and urban watersheds to surface waters is considered one of the main causes of eutrophication (Smith, 2003). Stream restoration, stormwater control measures (SCMs), and best management practices (BMPs) are increasingly implemented for nutrient retention and removal (Collins et al., 2010; Mayer et al., 2010; Newcomer Johnson et al., 2016). To improve performance of these practices, ecological and environmental engineers created novel designs such as regenerative stormwater conveyances (RSCs) to improve nutrient and sediment reduction efficiencies (Brown et al., 2010; Cizek et al., 2017). RSC systems are open-channel, sand seepage filtering systems that utilize a series of shallow pools, riffle weir grade controls, native vegetation, and a sand channel designed to reduce erosion, increase stormwater retention, capture sediments, and allow for nutrient transformation in the stream bed (Brown et al., 2010). They are commonly implemented in incised headwater channels or stormwater drainage ditches. RSCs add permeable sand to the stream bed along with a mix of organic matter as a microbial carbon source to increase rates of biogeochemical processes such as denitrification (Brown et al., 2010; Cizek et al., 2017). Despite the implementation of RSCs for over a decade, there is little information available concerning their nutrient and sediment reduction capabilities (Palmer et al., 2014; Williams et al., 2016). The type of sand and carbon added in constructing RSCs can affect performance; however, there is no standard

guide for selecting sand or organic matter types. Sands containing iron are often used because of cost and availability, but which may lead to unintended water quality conditions (Williams et al., 2016). Wood chips, cheap and abundant, are often chosen as the organic matter type to mix with the sands. Later, leaf litter, another organic carbon (C) source, may be captured and buried in step pools due to reduced flow velocity (Kochi et al., 2009). Relative to wood chips, leaf litter has a lower C to N (C: N) ratio and is therefore generally a higher quality substrate for the microbes which are responsible for nutrient biogeochemical cycles (Duan et al., 2014). Organic C in RSCs can lower surface or groundwater pH and dissolved oxygen (DO) (Keller et al., 2008; Duan et al., 2014), and thereby may affect nitrogen (N) removal via microbial denitrification (Thomas et al., 1994; Saleh-Lakha et al., 2009), a redox process that requires establishment of anoxic conditions and can be significantly influenced by carbon type and source (Newcomer et al., 2012; Duan et al., 2014). However, the low DO and reducing conditions may favor anaerobic bacterial reduction of Fe(III) or Mn(III) and release of iron (Fe^{2+}) or manganese (Mn^{2+}) in solution, which are sensitive to changes in reduction potential and pH (Williams et al., 2016). In general, P and some trace metals are sorbed onto surfaces of Fe/Mn (hydro)-oxides in suspended particles or bed sediments under oxidizing conditions, but can be desorbed under anoxic conditions when Fe/Mn (hydro)-oxides are reduced to soluble Fe^{2+} or Mn^{2+} (House and Denison, 2002). Meanwhile, P remobilization is also a function of pH (Jensen and Andersen, 1992), water temperature (Duan and Kaushal, 2013), and flow conditions (Duan et al., 2016). A recent study showed that large amounts of Fe^{2+} were released from many RSCs in Maryland and Washington, DC (Williams et al., 2016), which can cause a decrease in benthic substrate habitat for aquatic macroinvertebrates (Wellnitz et al., 1994), decreases in water exchange due to plugging of sediment pores (Ghiorse, 1986), and changes in the redox potential that cause release of toxic metals (Morin et al., 2003). Therefore, sources (i.e. quality) and optimal amounts (i.e. quantity) of organic C used in the sand seepage beds of RSCs must be identified and tested to determine whether there are unintended consequences or trade-offs in retention of N and P.

The objective of this study was to test whether RSCs can be an effective stormwater management or restoration approach for retaining both N and P, and to examine the relationship between nutrient retention and C quantity and quality. We hypothesized that the addition of excessive organic C as either wood chips or leaf litter in sand seepage beds may cause decreases in DO and pH at RSCs due to the release and oxidation of organic C. Low DO may result in more efficient removal of N but may, in turn, cause sediment remobilization of P that had been sorbed to Fe/Mn (hydro)-oxides. Furthermore, the balance of N removal and P release are also interactively affected by pH, water temperature, and their respective concentrations in stream water (Jensen and Andersen, 1992; Thomas et al., 1994; Saleh-Lakha et al., 2009; Wu et al., 2014), which may change with seasons. Therefore, we also hypothesized that P release and N retention can coincide under bacterially-mediated, iron-reducing conditions but that the rates of P release or N retention will vary by season. Understanding the influence of C quantity, quality and other controlling factors on nutrient retention or release in RSC systems is critical to improving the overall effectiveness of this novel technique for nutrient attenuation in stormwater management.

2 Material and methods

A combination of in-situ monitoring of RSCs water quality and lab mesocosm experiments simulating RSC processes were used to test our hypotheses. The field experiment consisted of paired tributary reaches, one treated with an RSC design and one unaltered which served as a control, at two stream locations. Streams were sampled across seasons to examine the effects of RSC construction on stream DO%, dissolved organic carbon (DOC), and nutrient levels. Downstream, longitudinal changes in water quality were examined seasonally to determine if the RSC systems were net sources or sinks of N, P, or organic C. Dissolved elemental iron (Fe) was analyzed as a possible control on P levels. Stream water was compared to groundwater to examine the role of groundwater as a source, sink, or zone of nutrient transformation. Because an oxygen deficit occurs during low-flow conditions when stream water becomes stagnant, our study focused on sampling during low-flow periods. To supplement field monitoring, mesocosm experiments representing various organic C sources and stream bed materials were constructed to test the influence of three sand types and the quantity of two organic C types (leaves and wood), on nutrient retention.

2.1 Study sites

We chose to study two streams with RSCs and associated control tributaries located in Anne Arundel County near Annapolis, Maryland, USA (Fig. 1), where multiple RSCs have been constructed and the RSC designs are generally similar. Our two RSC study sites, Central Sanitation and Crofton were constructed in a manner where one tributary of the stream was treated with the RSC approach and another tributary was left untreated and was examined as a control. The streams were located in the Coastal Plain physiographic province with similar urban and suburban land use. The soils near these streams vary somewhat but were similar because they have similar parent materials, having formed predominately in glauconitic marine deposits, other fluviomarine deposits, or recent alluvium (Soil Survey Staff, 2018), situated on geologic formations of acidic sands layered between confining layers of sandstone, silts, and clays (Cleaves et al., 1968).

The Central Sanitation RSC and control tributaries are among the headwaters of Severn River, which flows into Chesapeake Bay and is located in the Potomac Group geologic formation, composed of interbedded quartzose gravels, protoquartzitic to orthoquartzitic argillaceous sands, and white, dark gray and multicolored silts and clays (Cleaves et al., 1968). The Central Sanitation tributaries are located in an urban area with a high concentration of impervious surface and stormwater runoff from public buildings and parking lots as well medium-density residential areas. The Central Sanitation RSC, completed in 2012, was part of a larger effort with implementation of multiple SCMs to retrofit aging stormwater infrastructure (<http://ecosystemrestoration.com/central-sanitation/>, accessed on March 1, 2018). Iron flocculation widely occurred throughout the Central Sanitation RSC (personal observations). Most of stream section was free of tree canopy, and floating and emergent aquatic plants abundantly grew in the step pools of the RSC. According to Anne Arundel County (<http://dev.aacounty.org/departments/public-works/wprp/forms-and-publications/CentralSanitationSPSC.pdf>, accessed on March 1, 2018), retention rates of total N (TN), total P (TP), and total suspended solids at the Central

Sanitation RSC were approximately 40.0%, 59.9%, and 79.4%, respectively. The Central Sanitation reference tributary joins the restored stream downstream of the RSCs, and no iron flocculation was found in this stream. Central Sanitation RSC and control tributaries were approximately 670 m and 595 m in length, respectively.

The Crofton site is located on the Cretaceous Matawan geologic formation which consists of dark gray, micaceous, glauconitic, argillaceous, fine-grained sand and silt (Cleaves et al., 1968). Both tributaries are within the watershed of Little Patuxent River, a tributary of Patuxent River, which flows to Chesapeake Bay. The land use of the watersheds is high-density residential, with a large drainage area within the Crofton golf course. The RSC, completed in 2009, was developed to transform a degraded stream (e.g., with bank erosion and channel incision) to improve stability, water quality, and in-stream and riparian habitats (<http://www.biohabitats.com/projects/crofton-tributary-stream-restoration/>, accessed on March 1, 2018). The RSC design was similar to that of Central Sanitation, but the whole stream section was covered with tree canopy, and no aquatic plants were observed growing in the step pools. Unlike Central Sanitation, iron flocculation seldom occurred in the Crofton RSC (personal observation). The Crofton RSC and control tributaries were approximately 1070 m and 3140 m in length, respectively (Fig. 1).

2.2 Field monitoring on stream and ground water quality

Stream and groundwater of the four tributaries were monitored seasonally during base flow conditions (July 2016, October 2016, February 2017 and May 2017) to capture variability in water temperature, hydrology, and biogeochemistry. Base flows were operationally defined as the flow condition when precipitation does not occur for at least four days. We surveyed the whole section of the four tributaries on July 2016. A stream section without any visible water input (e.g., pipes) was selected for each tributary because additional water inputs may also affect the longitudinal pattern of water quality. Synoptic samples were collected longitudinally at 5 sites along the flow path of the selected sections to identify sources or sinks of nutrients and organic carbon (Fig. 1). Stream flow above and below the selected tributary sections was measured with a portable flow meter (Hach Flow FH950 Flow), and the difference was usually <5%. So, we assumed that the longitudinal variation in concentrations of nutrients or DOC indicated sources or sinks. If the streams are sources, concentrations of nutrients or trace elements would increase longitudinally, whereas, if streams are sinks, concentrations should decrease downstream.

Approximately 500-mL of grab samples of surface stream water were taken at five sites along each selected section of four tributaries for synoptic sampling, totaling 20 samples each season. Groundwater was sampled in order to examine if water quality of stream water is affected by groundwater inputs. The groundwater samples were taken in the streambed within the stream cross-section at the same locations where the surface water samples were taken. The difference in water quality between stream water and groundwater was used to indicate possible biogeochemical transformations, as well as possible direction of solute fluxes based on concentration gradient. Groundwater was collected using a push point sampler (<http://www.mheproducts.com/>, accessed on March 1, 2018), which allowed for rapid in situ pore water sampling. This tool consists of a 91 cm push point, 0.6 cm in

diameter made of stainless steel and a syringe assembly. The sampler was inserted into the substrate to a depth of ca. 30 cm to minimize streamflow and groundwater interactions so a true sample of groundwater was sampled. Groundwater samples were collected through a syringe to 250 mL acid pre-washed HDPE bottles, and the bottles were rinsed with groundwater three times prior to the collection of the final sample. The bottles were filled up without headspace and capped quickly after collection, in order to minimize exchange with atmospheric oxygen that may cause oxidation of Fe(II). The sampler and the syringe were rinsed after each use, first with stream water and then DI water. In order not to disturb bed sediment, samples were collected from downstream site to upstream site. Each sampling took about 30 min, and all samples were collected within one day.

At each site, water temperature ($^{\circ}\text{C}$), dissolved oxygen saturation (DO%), pH, and specific conductance (μS) of stream and ground water were measured in situ. Specific conductance (SC), which is usually higher in groundwater and wastewater, was used to indicate groundwater and/or wastewater inputs. Water temperature, pH, and SC were measured with a multi-probe meter (Oakton PCSTestr 35; Oakton Instruments, Vernon Hills, IL). Resolutions and accuracy of temperature, pH and SC were 0.1 degree ($\pm 0.5^{\circ}\text{C}$), 0.1(± 0.1), and 0.1 μS ($\pm 1\%$ full-scale), respectively. DO% was measured using a YSI 550A (YSI Inc., Yellow Springs, OH) with resolution and accuracy of 0.1 mg/L and ± 0.3 mg/L, or 0.1% and $\pm 2\%$ if reported as % air saturation.

In the field and during transportation to the lab, all samples of stream water and groundwater were preserved on ice. In the lab, stream and groundwater were filtered on the same date of sample collection with glass-fiber filters (pore size 0.7 μm). Filtered samples were kept at 4 $^{\circ}\text{C}$ for two weeks for fluorescence analysis, or acidified with concentrated nitric acid (BDH Aristar plus trace metal grade) to 1% HNO_3 for analysis of Fe. If the chemical analysis could not be finished within two weeks, samples were frozen until analysis. Although the samples were filtered quickly and no headspace was left in the storage bottles, additional Fe flocculate precipitated in a few groundwater samples after filtration. In such cases, additional metal grade nitric acid was added until all flocculate was dissolved. The acidified samples were kept at room temperature until analysis.

2.3 Mesocosm experiments with RSC construction materials

In order to test whether components (e.g., Fe and organic carbon) of RSC bed construction material affect nutrient transformation and retention, mesocosm experiments were conducted using different types of sands and organic materials (wood chips and leaves). RSC beds generally consist of sand and woodchip. Bed The sand component shall meet the AASHTO-M-6 or ASTM-C-33, 0.02 in. to 0.04 in. in size. For woodchips, use aged, shredded hardwood chips/mulch. The woodchips shall be added to the sand mix, approximately 20 % by volume, to increase the organic content and promote plant growth and sustainability. Therefore, we chose treatments of 20% and 5% wood chips to test the effects of high and low organic matter composition on N retention and P release. The wood chips of this study were collected from the banks of the Crofton RSC, the same types that were used to build this RSC. Although leaf litter is seldom used in RSC construction materials, we observed abundant leaf litter accumulating in many RSC pools, a phenomenon that can create hot

spots of denitrification (Groffman et al., 2005). Relative to wood chips, organic matter from leaves is more labile and causes faster release of DOC to stream water (Mafongoya et al., 1998). In this study, autumn-shed leaves were collected shortly after abscission in October 2016 from the floodplain of the Crofton RSC within 5 m of the wetted stream channel.

In addition to organic carbon, we choose three types of sands for stream bed substrates; Fe-containing sands (high Fe), carbonate sands (high Ca and Mg), and silicate sands (low Fe and low Ca/Mg). Elemental Fe was emphasized in this study because Fe can be reduced and thus mobilized under anoxic conditions, thereby affecting release of SRP and other trace elements that may be sorbed to Fe minerals. Release of Ca/Mg is possible if carbonate sand would be used, resulting in Ca/Mg release and a subsequent increase in pH and SC of stream water. Fe-containing sands were collected from a beach at Sandy Point State Park in Maryland, USA, and commercially available carbonate and silicate sands were obtained from Tri-State Stone and Building Supply Inc.

Each type of sand was mixed with leaf litter or wood chips (air-dried at room temperature) to make four organic/sand mixtures; 1) pure sands, 2) sands with 5% wood chips by volume, 3) sands with 20% wood chips, and 4) sands with 20% leaf litter. Thus, there were 16 combinations of organic matter and sand mixtures (e.g., Fe-containing sands only, Fe-containing sands+5% wood chips, Fe-containing sands+20% wood chips, Fe-containing sands+20% leaf litter, carbonate sands only, carbonate sands+5% wood chips, carbonate sands+20% wood chips, carbonate sands+20% leaf litter, silicate sands only, silicate sands +5% wood chips, silicate sands+20% wood chips, silicate sands+20% leaf litter) plus controls (water only, 5% wood chips, 20% wood chips, 20% leaf litter) each sampled in duplicate. The mixtures were put into PVC tubes (30 cm in depth) one side sealed with a rubber cap, and the PVC tubes were filled up with the mixtures to approximately one third the height of the tube. Stream water from Central Sanitation (collected one day before the incubations) was slowly added to tubes using a pipette. An aliquot (approximately 250 mL) of the stream water used for incubating the medium was saved (filtered and refrigerated) for water quality analysis. Mesocosm experiments were conducted in an incubator at two controlled temperatures (4 °C and 30 °C), representing winter and summer conditions. The experiments ran for 7 days in stagnant conditions (no shaking) to represent a period of base flow, because during base flows there was no or minor surface flow between the pools (yet the pools contain standing water). At the end of the incubations, the incubation water was measured for pH and DO%, and then filtered with GF/F filters for TDN, NO_3^- , SRP, and DOC analysis. The differences in TDN, NO_3^- , SRP, and DOC before and after incubations were used to estimate the rate of fluxes from RSC construction materials. All mesocosm experiments were conducted in duplicate.

2.4 Chemical analyses

SRP and NO_3^- were determined on a Lachat nutrient automated analyzer (a QuikChem 8500 Series 2 FIA System; Hach Company, Loveland, CO) according to the USEPA Standard Methods - Method 365.1 and Method 353.2 (O'Dell, 1993a; O'Dell, 1993b). DOC and TDN were analyzed on a Shimadzu total organic C analyzer with a total N module - TNM-1 (TOC-V CPH/CPN), by using high temperature catalytic oxidation and

chemiluminescence methods, respectively (Alvarez-Salgado and Miller, 1998). For all DOC and nutrient analyses, 3–5 injections were run for each sample to obtain a standard derivation 0.2. Dissolved Fe was analyzed on a Shimadzu Ion Coupled Plasma-Optical Emission Spectrometer (ICPE-9820) according to the US EPA Standard Method 200.7, Revision 4.4 (Martin et al., 1994). Because water samples were filtered, dissolved Fe measured here was presumably dominated by Fe (II) that was in dissolved form. For all above chemical analyses, a check standard and a blank were analyzed after every 15 samples.

DOC quality was characterized with fluorescence on a FluoroMax-4 spectrofluorometer (Horiba Jobin Yvon, Edison NJ, USA) (Cory et al., 2010). Water samples were scanned for fluorescence emissions (300–600 nm with a 2 nm increment) with an excitation range of 240–450 nm at 5 nm increments. Fluorescence excitation emission matrices (EEMs) were instrument corrected, blank subtracted, and normalized by the water Raman signal following the method of Cory et al. (2010). To characterize dissolved organic matter (DOM) composition and estimate its sources, fluorescence EEMs were analyzed to obtain the fluorescence index (FI, McKnight et al., 2001) and humification index (HIX, Zsolnay et al., 1999). FI, the ratio of the fluorescence intensity at 450 nm to that at 500 nm with excitation at 370, was used to indicate the relative abundance of terrigenous vs. microbial DOM (McKnight et al., 2001). Typical values for FI in microbially sourced DOM are ~1.9 while FI values of terrestrially sourced DOM are ~1.4. HIX was used to distinguish between the humic and autochthonous (microbially-derived) nature of DOM in the sample (Zsolnay et al., 1999; Ohno, 2002). HIX is the ratio of the integrated fluorescence intensity within the emission range of 435–480 nm divided by the integrated fluorescence intensity within the emission range 300–345 nm, at excitation 254 nm (Ohno, 2002). High HIX values (>10) correspond to strongly humified/aromatic DOM, principally of terrestrial origin, while low values (<4) indicate more autochthonous DOM. Water samples were allowed warm to room temperature before EEM scans. Humic acid standards from Suwannee River were analyzed to check the accuracy of the UV/Vis spectrometer. Quinine sulfate dissolved in 0.5 M H₂SO₄ was used to check the accuracy of the fluorimeter.

2.5 Data analysis and statistical analyses

Statistical analyses were performed using SPSS 10 (IBM Corporation, New York, NY, USA). Significant differences in water quality parameters (temperature, DO%, pH, and SC), and concentrations and fluxes of nutrients (TDN, NO₃⁻, and SRP) between RSCs and control unrestored stream sites were compared using a repeated measures Analysis of Variance (ANOVA) test with season as a factor. For lab mesocosm experimental incubations, significant differences in water quality parameters and nutrient concentrations were also tested among treatment combinations. Data were checked for normal distribution prior to ANOVA, and data were transformed or outliers were discarded if these data were not normally distributed. We used Pearson's correlation analysis to check for collinearity between SRP and Fe. We considered correlation coefficients significant when p-values were 0.05. If the relationships were not linear ($p > 0.05$), exponential, logarithmic or power non-linear regressions were generated. In field measurements, longitudinal changes in concentrations of nutrients and DOC within each RSC and control stream reach were

examined using linear regressions of these variables against distance. We identified significance at $p = 0.05$.

3 Results

3.1 Water quality and nutrient concentrations in RSCs and control tributaries

DO% in both RSCs was significantly lower than in the control, unrestored tributaries across seasons at Crofton ($20.8 \pm 4.9\%$ to $59.9 \pm 6.5\%$ vs. $40.8 \pm 4.9\%$ to $113.6 \pm 1.9\%$) and Central Sanitation ($37.5 \pm 2.5\%$ to $64.0 \pm 2.0\%$ vs. $88.8 \pm 1.7\%$ to $103.6 \pm 0.7\%$) (Fig. 2a-b; One-way ANOVA, $p < 0.05$). DO% at Crofton was highest in winter and lowest in fall, while relatively less seasonal variability was observed at Central Sanitation. At both sites, pH in RSCs was lower than in the control, unrestored tributaries across seasons (Fig. 2c-d; One-way ANOVA, $p < 0.05$). Unlike DO% and pH, water temperature, SC, SRP, and TDN were not consistently different between RSCs and control tributaries across seasons at either Crofton or Central Sanitation (One-way ANOVA, $p > 0.05$; Table 1). TDN concentrations at Crofton were lowest in fall and highest in summer, while TDN seasonal variability at Central Sanitation was relatively minor (Table 1). Concentrations of SRP were generally highest in fall or in spring (e.g., Crofton RSC; Table 1).

DOC concentrations were significantly higher at the RSC-treated tributary than the control tributary at Central Sanitation across seasons, but the opposite effect was observed at Crofton (Fig. 2e-f; One-way ANOVA, $p < 0.05$). Despite this difference, RSCs were consistently significantly higher in FI but lower in HIX than in the control unrestored tributary across seasons at both Crofton and Central Sanitation (Fig. 2g-j; One-way ANOVA, $p < 0.05$). Across sites, the highest FI and lowest HIX values occurred in fall 2016, and the values were close to that of fresh leaf litter collected during fall (Fig. 2g-j).

3.2 Longitudinal changes in concentrations of DOC, nitrogen, and SRP

DOC concentrations at the Central Sanitation RSC section (from the first to the last sample station) increased downstream by 18–54% across seasons, along with concurrent downstream decreases in TDN by 16–37% (Fig. 3a, c). The largest increase in DOC concentrations (by 54%) occurred in fall, while the largest decrease in TDN concentrations (by 37%) occurred in summer when temperatures were highest. Downstream increases in DOC and decreases in TDN also occurred at the Central Sanitation control tributary, but DOC increases ($< 18\%$) and TDN decreases (5–8%) were less relative to that of the RSC (Fig. 3b and d). The slopes of the TDN decreases and DOC increases (0.11 to 0.29 and 0.17 to 0.47) at the RSC section were significantly higher than that at the control tributary (0.05 to 0.06 and 0.06 to 0.11) (One-way ANOVA, $p < 0.05$). There were no clear longitudinal patterns (due to scatter in the data points) in DOC or TDN concentrations at Crofton RSC or control sites (data not shown).

Unlike DOC or TDN, no trends in SRP (scatter in data points) were observed at either RSCs or control tributaries at Central Sanitation (Fig. 3e-f) and Crofton sites (data not shown). Instead, SRP correlated positively with Fe in summer with the highest temperatures ($r^2 = 0.45$, $n = 20$, $p < 0.05$; Fig. 4a), and in spring when temperatures were second highest ($r^2 =$

0.30, $n = 20$, $p < 0.05$; Fig. 4d). In winter, with lower temperatures, SRP concentrations were lowest and SRP varied inversely to Fe ($r^2 = 0.63$, $n = 20$, $p < 0.05$; Fig. 4c). In fall, when SRP concentrations were highest, the relationship between SRP and Fe concentrations was scattered ($r^2 = 0.12$, $n = 20$, $p > 0.05$; Fig. 4b).

3.3 Stream water and groundwater comparisons

DO% in groundwater were consistently lower than in stream water at all RSC and control sites ($p < 0.05$; One-way ANOVA; Fig. 5a-d). Opposite to DO%, DOC concentrations in groundwater were generally higher than that in stream water ($p < 0.05$; One-way ANOVA); the exception was the Crofton control tributary where stream water DOC was the highest. Moreover, groundwater DOC of RSCs was significantly higher than that of control tributaries at both sites ($p < 0.05$; One-way ANOVA Fig. 5e-h).

At Crofton, groundwater NO_3^- was as low as stream water NO_3^- at the RSC and the control tributaries (Fig. 5i-j). Groundwater TDN, however, was always higher than stream water TDN ($p < 0.05$; One-way ANOVA; Fig. 5m-n), suggesting higher groundwater TDN in other forms (ammonium or dissolved organic nitrogen). At Central Sanitation RSC and control sites, groundwater NO_3^- was usually lower than stream water NO_3^- ; the same occurred for groundwater TDN ($p < 0.05$; One-way ANOVA) (Fig. 5k-l, o-p). However, concentrations of groundwater and stream water SRP were not consistently different among sites ($p > 0.05$, One-way ANOVA; Fig. 5g-t).

3.4 Lab mesocosm experiments

DOC fluxes from RSC construction material at 30 °C were generally positive (denoting release), and increased with increasing substrate organic C quantity (from 0 to 5% to 20% wood chips) and quality (from 20% wood chip to 20% leaf litter) (Fig. 6a). DOC release did not vary significantly across sand substrate types. DOC release at 30 °C was significantly higher than at 4 °C across organic substrates (One-way ANOVA, $p < 0.05$; Fig. 6b).

With an increase in DOC release across organic substrates, DO% in the leaching water generally decreased with increasing organic C quantity and quality, but the lowest DO% occurred in 20% wood chip (Fig. 6c). DO% values at 30 °C were significantly lower than that at 4 °C across organic substrates (One-way ANOVA, $p < 0.05$; Fig. 6d). Similarly, pH in the leaching water decreased with increasing organic C quantity and quality, and the pH values at 30 °C were significantly lower than at 4 °C (One-way ANOVA, $p < 0.05$; Fig. 6e-f).

Like DOC, SRP fluxes from RSC construction materials at 30 °C were positive (denoting release), and SRP release increased with increasing organic C quantity and quality. SRP fluxes were highest in Fe-containing sand mesocosms with 20% leaf litter added (Fig. 7a; one-way ANOVA, $p < 0.05$). SRP release from leaf litter at 30 °C was much higher than at 4 °C (One-way ANOVA, $p < 0.05$) (Fig. 7b).

At 30 °C, NO_3^- fluxes were positive (denoting release) in mesocosms without organic matter addition, but the values decreased sharply to negative values (denoting retention) when organic matter of either type or quantity was added (Fig. 7c). At 4 °C, the rate of

changing NO_3^- flux from positive to the negative across the organic matter gradient was slower. For example, NO_3^- fluxes were still positive when 5% or 20% wood chips were added but retention increased substantially in the 20% leaf treatment (Fig. 7d). The fluxes of TDN at 30 °C were negative (denoting retention) with 5% or 20% wood chips added (similar to NO_3^- fluxes), but the values were positive (denoting release) in incubations with 20% leaf litter (Fig. 7e). At 4 °C, TDN fluxes from construction materials were all positive (denoting release) (Fig. 7f).

4 Discussion

4.1 Release and source of DOC in RSCs

In constructing RSCs, wood chips are added to increase the organic content of sand seepage beds (Brown et al., 2010). Later on, leaf litter from the riparian zone naturally falls and accumulates in RSCs due to reduced flow rate (Kochi et al., 2009). From wood chips or leaf litter, DOC is leached to stream water and thereby affects nutrient transformations and water quality (Groffman et al., 2005). The released DOC is readily assimilated by heterotrophic bacteria (Meyer et al., 1987) that play a critical role in transforming N and P, and metals like Fe and Mn (Williams et al., 2016). In our mesocosm experiments, we observed increased DOC release with increasing organic carbon quantity (0 to 5% to 20% wood chip) and type (wood chips vs leaf litter) (Fig. 7a), demonstrating that organic matter quantity and quality are an important factor controlling for DOC release. In the field, we observed downstream increases in DOC concentration by 18%–54% at Central Sanitation RSC across seasons (Fig. 3a). Because there were only minor streamflow inputs (<5%) in this RSC stream section, these downstream increases can roughly be considered as DOC release from the RSC substrate. Meanwhile, changes in conductivity were usually <6% and the values could be positive or negative, once again suggesting groundwater input was minor. Interestingly, we also observed at the Central sanitation site that DOC increases downstream (denoting DOC release) in the control stream but the release was less (<18%) relative to that of the RSC (Fig. 3b), suggesting greater sources of labile organic matter in the RSC than in the control tributary, perhaps due to wood chips that were initially added and leaf litter that was accumulated later. Similar to this study, Williams et al. (2016) also observed that DOC increased substantially and steadily in a recently constructed RSC.

We found no downstream increases in DOC at either the Crofton RSC or control tributaries. Moreover, stream water DOC concentrations of the RSC were lower than that of the control tributary (Fig. 2e), which was unexpected. The higher stream water DOC at the reference tributary was probably due to upstream inputs rather than release from substrate, because DOC concentrations were lower in groundwater than in stream water at this site (Fig. 5f). Moreover, we found that groundwater DOC concentrations of both RSCs were consistently higher than that of groundwater in control tributaries (Fig. 5e-h), suggesting that organic matter added to the stream bed in RSCs contributes C to groundwater. Another possible reason is that RSC implementation often raises the water table by filling deeply incised channels during construction (Williams et al., 2016), thus likely retarding decomposition in the vadose zone that provides a larger source area for DOC to reference streams.

Our fluorescence data provided additional information on sources of DOC in RSCs. We found that FI was always highest in fall when leaf litter was more available across sites, and the FI values in fall were closest to that of fresh leaf litter (Fig. 2g-j). This seasonal variability of FI reflected the fact that leaf litter was fresh and more available in fall relative to other seasons (Duan et al., 2014). McKnight et al. (2001) used higher FI (~1.9) to indicate microbially sourced DOM and lower values (~1.4) for terrestrial DOM. This is consistent with our observation that FI values of wood chips were close to 1.4. In this study, we also found that fresh leaf litter was high in FI, and the higher FI values in stream water can be attributed to fresh leaf litter inputs. We cannot exclude the possibility that organic carbon inputs were from autochthonous sources because of their high FI values (>1.9), but evidence suggests that autochthonous inputs were likely minor. First, FI was not highest in summer when autochthonous production was highest. Secondly, FI values were not significantly higher in Central Sanitation RSC than Crofton RSC (Fig. 2g-h), where aquatic plants were far denser. By using FI as indicator of leaf litter input, we can interpret the difference in FI between RSCs and control tributaries. That is, FI values in RSCs were consistently higher and closer to the values of leaf litter than in the control tributaries (Fig. 2g-j), likely because RSCs are effective in retaining and burying leaf litter in step pools due to reduced flow velocity (Kochi et al., 2009). Thus, leaf litter can be an important source of DOC in the RSCs, especially in older projects like those of this study that were 5–7 years old and which may have had several seasons to accumulate leaf litter. We can draw similar conclusion from the patterns of HIX: 1) values were lowest and closest to that of fresh leaf litter in fall when fresh leaf litter was more available; 2) values at RSCs were closer to that of fresh leaf litter relative to control tributaries. However, HIX values of stream water were higher than that of both fresh leaf litter and wood chips, suggesting DOC inputs from other sources that were higher in HIX values (likely, soil pore water).

4.2 Decreases in DO and pH in RSCs

We observed that both DO% and pH in the RSCs were consistently lower than in control tributaries at both sites (Fig. 2a-d). Although pH values of RSCs were in normal ranges, DO concentrations were usually lower than 4 mg/L (except in winter) which may affect transformation and release of N, P, C and trace metals (e.g., Duan et al., 2014; Williams et al., 2016). The lower DO% in RSCs can be partially attributed to larger DO consumption due to decay of DOC from wood chips and leaf litter. This is supported by our mesocosm experiment where we observed that DOC release increased while DO decreased with increasing organic matter quantity and quality, or when incubation temperature increased from 4 °C to 30 °C (Fig. 6). The co-occurrence of DO% decreases with DOC release can likely be ascribed to biological oxygen demand during bacterial oxidation of the released DOC. Decreases in DO due to oxidation of DOC leached from leaf litter have been reported previously (Duan et al., 2014). Additional evidence for the effect of DOC on DO was from field DO% measurements, showing that the lowest DO% was in fall (Fig. 2a-b) when there was more labile organic matter available from fresh leaf litter. The co-occurrence of pH decreases and DOC releases during the mesocosm experiment can be attributed to the fact that organic acids (e.g., humic and fulvic acids) were leached from wood chip or leaf litter (Hartley and Buchan, 1979; Pocknee and Sumner, 1997). Low pH in streams, rivers, and lakes has been attributed to high levels of organic carbon (Ishikawa et al., 2006; Keller et al.,

2008). Therefore, excessive organic matter added or accumulated in RSC sand seepage beds (wood chips and leaf litters) may cause decreases in DO and pH of RSCs due to release and oxidation of DOC.

Our data also suggest that DOC was not the sole reason for the decreases in DO and pH in RSCs. For example, the Crofton RSC had lower DOC than the control tributary, but DO and pH were still higher than the control tributary (Fig. 2). Thus, other factors, such as reduced water-atmospheric exchange and groundwater seepage, may also play a role in DO and pH dynamics. In RSCs, stream flow is substantially reduced (Brown et al., 2010), as is the turbulence that causes stream water-atmosphere oxygen exchange. Meanwhile, RSCs usually cause an increase in groundwater levels (Williams et al., 2016) and stream water residence time (Brown et al., 2010). The increased groundwater levels and prolonged residence times may also cause lower DO in RSCs due to decreases in stream water-atmospheric exchange for less surface turbulence. Additionally, the effect of seepage of DO-depleted groundwater may influence stream water DO because DO% in groundwater was consistently lower than in stream water, and groundwater DO% of the RSCs was even lower relative to that of control tributaries (Fig. 5a-d).

4.3 Nitrogen retention in RSCs

Nitrogen retention through assimilation and/or denitrification is an objective of RSC construction and is the primary reason for adding organic matter to the streambed substrates. DOC released from organic matter provides an energy source for bacteria responsible for both NO_3^- immobilization and denitrification (Starr and Gillham, 1993; Newcomer et al., 2012). Meanwhile, mineralization of high amounts of DOC from organic matter supplementation can result in low DO concentrations under conditions of low re-aeration, thus providing an anaerobic environment for denitrification to occur (Bonin and Raymond, 1990; Korom, 1992). The effect of organic C quantity and quality on DO depletion and N transformation has been shown in prior studies (Newcomer et al., 2012; Duan et al., 2014). In the organic matter addition simulation, NO_3^- flux decreased along with DOC release and DO consumption when we increased organic matter quantity and quality (Fig. 7c-d), once again highlighting the effect of organic carbon and DO level. Moreover, we found NO_3^- was almost totally retained under low DO% (<40%) during the 7-day high temperature incubation (30 °C) even with 5% wood chips added, suggesting 5% wood chips is sufficient for NO_3^- retention at higher temperatures. However, NO_3^- retention for the 5% treatment was poorer than that for 20% treatment under 4 °C (Fig. 7d). We still cannot make any conclusion regarding which treatment is better, because there is a big gap between these two temperature and mesocosms with less temperature interval are needed. Furthermore, mesocosms differ from in situ flowing water conditions. Stagnant conditions of the mesocosms could create different redox conditions than if fresh stream water was cycling through more often. Thus, better design of further mesocosm experiments is needed in order to determine ideal ratio of wood chips to sands for RSC construction materials.

The larger retention of TDN (mainly NO_3^-) in the Central Sanitation RSC than in the control tributary was consistent with mesocosm experiments, showing that the addition of organics to RSCs may enhance N retention in streams. At this site, approximately 16–37%

of TDN was retained in the tributary treated with an RSC, while only 5–8% of TDN was retained in the control tributary with similar length. The higher N retention rates in the RSC were consistent with the higher DOC, lower DO concentrations, and larger downstream increases in DOC in the RSCs. Together, these observations suggest that denitrification may be occurring. We cannot exclude the possibility of autotrophic assimilation because of the high density of plant biomass (Nymphaeaceae family) in the RSC pools (personal observation). Additional evidence of the larger TDN retention was the larger TDN concentration gradients from stream water to groundwater in RSCs than in reference tributaries (Fig. 5k-i, o-p). These larger TDN concentration gradients suggest that more TDN was retained from groundwater as it passed through the bed sediment in RSC than in control reaches. The larger TDN retention in the RSC than control could also be a result of differences in water residence time, flow rate, area of water-sediment interface and uptakes of aquatic plants. In general, RSC increases water residence time for N retention due to creation of step pools. Stormwater flow rates decrease in RSC step pools owing to transfer of stormwater to shallow groundwater through infiltration (Brown et al., 2010). Because the RSC step pools usually have larger surface area of traverse cross-section profiles than the normal channel, RSC increases area of water-sediment interface that may affect N transformation. Finally, N uptake by dense aquatic plants in the RSC step pools also accounted for this difference in TDN between RSC and control, because no or much less aquatic plants were found in the control tributary.

Unlike Central Sanitation, we observed no apparent N retention in the Crofton RSC. There were no detectable downstream decreases along the stream. The lack of evidence for effective N retention within the RSC at Crofton was expected, however, because of much lower NO_3^- concentrations at this site than at Central Sanitation (Fig. 5). Prior studies (Trevors and Starodub, 1987; Arango et al., 2007) have shown that N retention via denitrification is a function of NO_3^- concentrations, and the effect of substrate organic C on N retention is apparent only when NO_3^- concentration is high, e.g., exceeding half-saturation constants. Here, NO_3^- concentration ($< 0.15\text{mg/L}$) at Crofton RSCs was an order of magnitude lower than that at Central Sanitation, so N retention at this site may be negligible and limited by NO_3^- . Duan et al. (2014) also reported lack of N retention at a NO_3^- -depleted stream despite high DOC in a Baltimore forested watershed. Additional evidence of minor retention of NO_3^- in the Crofton RSC was the small NO_3^- concentration gradients from stream water to groundwater in RSCs (Fig. 5i, k). These small gradients suggest that much less NO_3^- can diffuse from stream to groundwater in bed sediment for N transformation at this RSC. In contrast to NO_3^- , groundwater TDN concentrations at Crofton sites were higher than in stream water (Fig. 5m-n), suggesting there were diffusion gradients of TDN from groundwater to stream. So, bed sediment could be a source of TDN to stream water at this site. In our mesocosms, we also found a large fraction of TDN released from fresh leaf litter, along with a large amount of DOC (Fig. 7e). Thus, in tributaries of Crofton with low NO_3^- concentration but high groundwater TDN, RSCs may become a N source due to release of TDN from bed sediment.

4.4 Phosphorus transformations in RSCs

It is known that high DOC and anoxic conditions may cause release of SRP from stream substrates (Duan et al., 2014; Duan et al., 2016), because Fe/Mn hydroxides, onto which P can be sorbed, are unstable under anaerobic conditions (House and Denison, 2002). In lab mesocosm experiments, we observed release of SRP from substrates with organic matter addition, and the SRP release increased with increasing organic matter quantity and quality (Fig. 7a-b). Moreover, the SRP release generally corresponded to the lower DO levels (Fig. 6c-d), supporting our hypothesis that anoxic conditions in RSCs may cause SRP release from substrate (Duan et al., 2014). The higher values of SRP release from leaf litter versus wood (Fig. 7a-b) were consistent with previous studies showing that leaf litter is an important P source in stream ecosystems (Duan et al., 2014). Previous studies have also shown much higher P contents in leaf litter (0.17–0.30%; Kwabiah et al., 2001) than in wood chips (<0.05%; Romero et al., 2005; Zule and Dolenc, 2012). Moreover, we found that the highest value of SRP release occurred from Fe-containing sands (Fig. 7a), further supporting the idea that Fe/Mn oxides in the sands can facilitate release of SRP under anoxic conditions. We cannot exclude the effect of decreased pH on higher SRP release from leaf litter or Fe-containing sands, because more P was released under acidic conditions relative to neutral pH (Wu et al., 2014). In addition to the effect of sand type and organic matter, SRP release at 30 °C was significantly higher than that at 4 °C also agreeing with previous studies showing that temperature can stimulate SRP release from leaf litter or stream bed sediment (Duan et al., 2012; Duan and Kaushal, 2013; Duan et al., 2014). It is surprising though not to see much net SRP release in the very anoxic 20% woodchip treatment with Fe-containing sand (Fig. 7a-b), and the reasons warrant further examinations. Results from mesocosm experiments help to explain some of our field observations. For example, SRP concentrations in all RSCs and control tributaries were highest in fall (Table 1), consistent with the extremely high SRP release in leaf litter incubation. Leaf litter may play a more important role in shaping seasonal variability of stream SRP than sediment release that is dependent on water temperature.

Despite the evidence of SRP release in our mesocosm experiment, we did not observe consistent differences in stream water SRP between RSCs and control tributaries (Table 1). We also did not observe any consistent downstream increases in SRP concentrations (denoting release) at any of the sites (Fig. 3e-f). To our surprise, there was no consistent difference in SRP concentrations between stream water and groundwater (Fig. 5q-t), despite consistently lower DO values in groundwater (Fig. 5a-d). We speculate that release of SRP from RSC substrate might occur in situ, but the released SRP was probably sorbed rapidly onto particles (e.g., Fe/Mn hydroxides, organics, clays), onto precipitating Fe flocculate (Williams et al., 2016), or assimilated by bacteria or plants. For example, Duan et al. (2016) found that stormwater ponds in suburban/urban watersheds of Baltimore, Maryland, USA were significantly higher in particulate P (PP) and dissolved organic P (DOP) than the control stream sections, while the difference in SRP was minor. The high PP and DOP in the stormwater ponds were most likely derived from SRP that was released from the substrate, because there were no other sources of PP or DOP to the stormwater ponds. The second speculation was that reactive P in RSC substrates was mobilized and exported from the RSC systems with Fe/Mn hydroxides during storm events, because the RSCs are 5 to 7 years old.

Our preliminary data on the relationship between SRP and Fe provided limited information on P transformation in RSCs. We found that SRP was positively correlated with Fe in summer and spring (Fig. 4a-d) when water temperatures were higher (Table 1). This suggests that both SRP and Fe were mobilized under higher temperatures because P release is temperature dependent (Duan and Kaushal, 2013). In winter, when water temperatures were lower, SRP was the lowest and SRP decreased sharply with increasing Fe (Fig. 4), probably reflecting different controlling processes. In fall, when leaf litter input was more available, SRP concentrations were highest, and the relationship between Fe and SRP was probably affected by the leaf litter input. The role of Fe in SRP concentration across seasons warrants further investigation.

5 Conclusions

RSCs are generally effective but some aspects of RSCs can have negative water quality tradeoffs that engineers/designers should note and strive to minimize in their designs. RSC's may be effective at reducing NO_3^- but precise volumetric reductions are likely to be variable, as demonstrated by the substantial differences between our two field sites, depending on watershed characteristics and other limiting factors such as type and quantity of DOC and the initial ambient concentrations of NO_3^- which both can limit denitrification. We found no evidence for SRP release from RSCs in situ, but SRP release did occur in the mesocosm experiments owing to low DO% levels, especially in those with Fe-containing sands. Further study is needed to examine other forms of P, the role of Fe as a controlling factor for SRP levels in RSCs, and for P dynamics in newly-built RSCs. Managers may consider the trade-off in retention of N versus the release of P in RSCs. Fe-containing sands are not recommended for RSC construction.

Adding wood chips as an organic matter source for denitrifying microbes appears to be an effective choice in RSC construction. Much less DOC, P, and N was leached from wood chips than from fresh leaf litter. Wood chips were expected to provide a long-term, slowly degradable source of organic matter to RSCs. However, our fluorescence data showed a stronger signal of DOC inputs from leaf litter in RSCs, suggesting that the influence of wood chips may be reduced with time due to erosion by storm events. Replenishing this C source may be necessary as projects age or stream beds erode. Leaf litter from the riparian zone may contribute to the RSCs annually and become a relatively more constant organic matter source over time. Our mesocosm data suggest that leaves may have an important role in retaining N during colder temperatures and other studies (Groffman et al., 2005) show that leaf packs in urban streams can become hot spots of denitrification. Application of engineering methods such as RSCs to improve urban stream function will necessarily be constrained by surrounding land use and urban infrastructure which may not accommodate some channel designs. RSCs could be an important choice as a stormwater and/or nutrient BMP when trade-offs are considered.

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Highlights

- We examined effects of regenerative stormwater conveyances (RSCs) on water quality.
- RSCs reduced dissolved oxygen and pH in stream surface and ground water.
- RSCs retained nitrogen but retention varied with nitrogen concentration and seasons.
- Nitrogen retention varied with organic carbon quantity and quality and temperature.
- Release of soluble reactive phosphorus from RSCs warrants further investigation.

Fig. 1.

Two paired sampling sites (in red dots) with regenerative stormwater conveyance (RSC) and without RSC (Control) - Central Sanitation (panel B) and Crofton (panel C) in Anne Arundel county (panel A), Maryland. In each site, water samples were longitudinally collected at 5 sites along a stream section without any tributary input. Photos showed step pools in RSCs and normal channel in control tributaries. The yellow and red arrows in maps and photos show the directions of stream flow of RSCs and control tributaries, respectively. The numbers (1 through 4) are used to indicate sampling locations in both maps and photos. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Fig. 2. Comparison of mean dissolved oxygen (DO) concentration, pH, dissolved organic carbon (DOC) concentration, and DOC fluorescence index (FI) and humification index (HIX) between regenerative stormwater conveyance (RSC) and control at Crofton and Central Sanitation across seasons. Due to instrument failure, pH was not measured during fall 2016. The error bars represent for variability of the 5 selected sites (see Fig. 1).

Fig. 3.

Downstream changes in concentrations of dissolved organic carbon (DOC), total dissolved nitrogen (TDN), and soluble reactive phosphorus (SRP) at Central Sanitation sites with regenerative stormwater conveyance (RSC, left panels) and without (control, right panels). The trend lines without significant changes ($p > 0.05$) were in dash line. The 1 to 5 on the x axil are site number, and the arrow line indicates flow direction. For DOC and TDN, RSC sites showed larger slopes of downstream changes relative to control sites, suggesting larger rate of DOC release or TDN retention.

Fig. 4. Soluble reactive phosphorus (SRP) versus dissolved iron (Fe) in stream water for all sites across seasons. Equation of trend line or curve is provided for each season with R squared value and significance (p-value).

Fig. 5.

Comparison of concentration of dissolved oxygen (DO), dissolved organic carbon (DOC), nitrate (NO₃⁻), total dissolved nitrogen (TDN), and soluble reactive phosphorus (SRP) between stream water and groundwater across seasons. Groundwater TDN concentration was higher than stream water only at Crofton sites. The error bars represent for variability of the 5 selected sites (see Fig. 1).

Fig. 6.

Release of dissolved organic carbon (DOC) and changes in dissolved oxygen (DO) and pH in the end of 7-day mesocosm incubations of regenerative stormwater conveyance (RSC) construction materials with stream water. The left panels compare the incubations of 30 °C with different type rocks. The error bars represent for the range in duplicates. Right panels compare the incubations at 30 °C and 4 °C, and the error bars represent for the range of a combination of the 3 sand types. In left panels, * denotes significant difference than the other. In right panels, * denotes significant difference between 30 °C and 4 °C; “A” refers to significant difference between the left and the right with different organic treatments at incubating temperature 30 °C, while “B” indicates significant difference at incubating temperature 4 °C.

Fig. 7.

Changes in fluxes of soluble reactive phosphorus (SRP), nitrate (NO_3^-) and total dissolved nitrogen (TDN) at the end of 7-day incubation of regenerative stormwater conveyance (RSC) construction materials with stream water. The left panels compare the incubations of 30 °C with different type rocks. The error bars represent for the range in duplicates. Right panels compare incubations at 30 °C and 4 °C and the error bars represent for the range of a combination of the 3 sand types. In left panels, * denotes significant difference than the others. In right panels, * denotes significant difference between 30 °C and 4 °C; “A” refers to significant difference between the left and the right with different organic treatments at incubating temperature 30 °C, while “B” indicates significant difference at incubating temperature 4 °C.

Table 1

Comparison of mean/standard errors of water temperature, specific conductance (SC), and concentrations of total dissolved nitrogen (TDN) and soluble reactive phosphorus (SRP) between RSCs and control streams across seasons.

	Water temperature (°C)				SC ($\mu\text{S cm}^{-1}$)			
	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring
RSC-Crofton	23.9/0.1	6.3/0.5	10.3/0.4	21.2/0.4	199/7	253/6	247/4	220/2
Control-Crofton	24.3/0.1	4.8/0.2	12.5/0.4	21.9/0.2	224/7	288/11	245/3	201/2
RSC-C Sanitation	23.3/0.3	5.0/0.8	12.1/0.4	19.7/0.3	416/15	434/8	406/15	307/9
Control-C Sanitation	20.1/0.3	9.3/0.2	14.2/0.3	16.9/0.2	327/5	343/1	317/1	303/1
	TDN (mg/L)				SRP ($\mu\text{g L}^{-1}$)			
	Summer	Fall	Winter	Spring	Summer	Fall	Winter	Spring
RSC-Crofton	0.71/0.02	0.22/0.01	0.41/0.01	0.61/0.03	8.8/0.7	10.6/4.0	4.7/0.6	11.3/2.6
Control-Crofton	1.02/0.03	0.26/0.01	0.37/0.01	0.63/0.01	19.9/3.0	86.8/19.3	2.5/0.2	23.8/2.7
RSC-C Sanitation	4.24/0.33	3.79/0.13	3.81/0.25	2.61/0.14	3.2/0.9	9.3/4.7	0.7/0.2	1.7/0.2
Control-C Sanitation	3.23/0.04	3.15/0.04	3.53/0.04	2.96/0.04	3.6/0.6	4.7/0.6	4.4/0.7	2.7/0.3