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Stream Restoration Performance and Its Contribution to the Chesapeake Bay TMDL: Challenges Posed by Climate Change in Urban Areas

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Abstract In large part driven by total maximum daily load (TMDL) mandates, the restoration community in the Chesapeake Bay region has been implementing novel best management practices (BMPs) and stream restoration designs in urban areas, such as regenerative stream/stormwater conveyance (RSC) structures and stream-wetland complexes (SWCs). However, the nutrient and sediment reduction efficiencies of these novel designs are virtually unknown, and the possibility of increasing riverine flow in the Chesapeake Bay watershed associated with climate change this century necessitates an evaluation of their performance to develop and utilize those that optimize reductions in nutrient and sediment fluxes. We compare pre- and post-construction loads (total nitrogen (TN), total phosphorus (TP), and total suspended solids (TSS)) from RSCs (i.e., upland BMPs) and a SWC (i.e., stream restoration) constructed at the outflow of a highly developed watershed in the Coastal Plain physiographic province of Maryland. The largest of the two RSCs performed best in relation to expected nutrient and sediment reductions because of superior water retention capability. By the length of river reach restored, the SWC attained from 79 to 88% of its N reduction TMDL goal, but only 19 to 23 and 2.7 to 3.1% for TP and TSS, respectively; by watershed area, % attainments of TMDL goals were much lower. Results indicate that SWCs have the potential to curtail N loading from developed catchments, but additional water quality benefits may be limited. Climate change projections indicate that there will be an increased frequency of larger-volume storms that will result in an increase in stormflow runoff from urban areas, and increased pollutant loads will likely curtail potential gains made by efforts to achieve TMDL goals. Given the large-scale implementation of BMPs currently underway to accommodate the Chesapeake Bay TMDL, the restoration community needs to adopt a concerted strategy of building climate resilience into many types of urban BMPs to help attain and maintain loads at TMDL levels in anticipation of a progressively wetter climate throughout this century.

Keywords Best management practices (BMPs) · Climate change · Nutrients · Stream restoration · Total maximum daily load (TMDL)

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Introduction

In 2010, the EPA completed a total maximum daily load (TMDL) for Chesapeake Bay that identified total nitrogen (TN), total phosphorus (TP), and total suspended solids (TSS) load reductions needed to meet water quality standards (www.epa.gov/chesapeake-bay-tmdl). The TMDL included phase I watershed implementation plans (WIPs) developed by states contributing pollution runoff to Chesapeake Bay that outlined reductions expected of the wastewater, urban/stormwater, agriculture, and on-site sewage sectors. The states involved were subsequently charged with the development of phase II WIPs and the identification of local strategies to provide nutrient reductions. As part of the interim goal achievements, strategies responsible for an estimated 60% of the TMDL goals are to be implemented in 2017, and total implementation is expected by 2025.

In urban watersheds, stream restoration is being heavily relied upon to improve water quality and provide the necessary pollutant reductions to achieve TMDL goals. Whereas more traditional stream restoration designs focusing on physical modification and the stabilization of stream channels commonly result in limited ecological uplift and water quality improvements to streams (Selvakumar et al. 2010; Palmer et al. 2014), some studies indicate that designs that reconnect ground and surface water and provide labile organic carbon can reduce nitrogen (Newcomer-Johnson et al. 2016; Pennino et al. 2016). Consequently, novel stream restoration designs such as stream-wetland complexes (SWCs) are increasingly substituting traditional restoration designs in the Chesapeake region, especially in the Coastal Plain of Western Maryland (Filoso and Palmer 2011; Filoso et al. 2015). Although these techniques are specifically designed to moderate stormflow and improve water quality, there is a paucity of scientific information about their capacity to improve water quality and restore important biophysical processes that promote nutrient and sediment retention. Moreover, most studies evaluating best management practice (BMP) and stream restoration performance commonly focus solely on N reductions (Craig et al. 2008), even though sources of TP and TSS pollution from urban areas to Chesapeake Bay are proportionally similar to that of TN (i.e., 11, 19, and 25% of total TN, TP, and TSS inputs, respectively; Chesapeake Bay Program (CBP) phase 5.3.2 watershed model for 2012 assessment) and thus included in the Chesapeake Bay TMDL.

Effective stormwater management in urban watersheds of Chesapeake Bay has become an even more relevant issue given the potential for dramatic hydrological alterations with climate change (Pyke et al. 2008; Najjar et al. 2009, 2010; Williams et al. 2010; US EPA 2013; Williams 2014). For instance, regional increases in air temperature and the amount and intensity of precipitation that occurred in the last century (Groisman et al. 2004) are expected to continue in the NE USA throughout the twenty-first century (IPCC 2013; Karl et al. 2009; Melillo et al. 2014; Blunden and Arndt 2015; Liang et al. 2015). A variety of models can be used to elucidate complex interactions and feedbacks associated with future climate scenarios (Lempert et al. 2006; Sarewitz et al. 2000; Volkery and Ribeiro 2009) and develop strategies for building resilience to climate change (Sarewitz et al. 2000; Fischbach et al. 2015). Nevertheless, there is still high spatiotemporal uncertainty in multidecadal climate change forecasts (Raisanen 2007; Hawkins and Sutton 2009), particularly with regard to potential impacts on watershed hydrology (Sarewitz et al. 2000; Bouraoui et al. 2002; Caldwell et al. 2012). For example, while rainfall amount and intensity are expected to increase in the Chesapeake Bay watershed, downscaled global circulation models (GCMs) and watershed models (i.e., SWAT) indicate that higher temperatures and associated potential evapotranspiration (PET) will, depending on the model, substantially increase or decrease total annual runoff (Najjar et al. 2009, 2010; US EPA 2013). Hence, the capacity to inform water resource managers with accurate estimates of a runoff response to long-term climate change is still lacking and this limitation complicates the development of effective strategies (i.e., infrastructure) for managing climate risk (Cox and Stephenson 2007).

The high probability that some regions of the Chesapeake Bay watershed (e.g., Susquehanna River basin) will experience more precipitation and surface runoff with climate change this century (US EPA 2013; Melillo et al. 2014; Liang et al. 2015) will likely hamper efforts to achieve and maintain TMDL goals (Williams 2014). The likelihood of increased runoff warrants an evaluation of various BMP and stream restoration designs to develop and utilize those that optimize reductions in nutrient and sediment fluxes and to determine the extent to which these should be implemented beyond those planned to achieve TMDL goals in order to maintain pollutant inputs at TMDL levels.

Herein, we evaluate a 7-year time series of pre- and post-construction loads from a combination of headwater BMPs (regenerative stream/stormwater conveyances (RSCs)) and a stream restoration (SWC) to estimate load reductions from a highly urbanized catchment in the Coastal Plain of Western Maryland. The main objective was to quantify the contribution of load reductions from upland BMPs versus restoration of the mainstem stream channel. Prognostic GCM variables and precipitation trends were used with the CBP watershed and estuarine models to determine potential climate effects on streamwater runoff and estuarine water quality. Nutrient and sediment reduction efficiencies of BMPs in the CBP watershed model were increased to evaluate the level of BMP implementation that will be required to offset pollutant load increases expected from increased stormflow runoff in predominately urban areas due to climate change. We elucidate the relative effectiveness of upland BMPs and an elaborate lowland stream restoration at mitigating potentially larger flows and solute loads expected in a wetter climate and provide guidance to water resource managers concerning (1) the extent to which mitigation efforts will be needed to offset climate change impacts and (2) strategies for improved resiliency.

Study Sites

The Cypress Creek watershed is located in the Magothy River drainage basin on the western shore of Maryland (lat/long 39.076054°, −76.542692°; Fig. 1). This watershed has undergone various land use changes that ultimately resulted in it being listed as a high-priority restoration site in Anne Arundel County's (AA Co.) WIP. At least part of the watershed was cultivated in a corn-tobacco-corn/wheat rotation from 1660 to 1780 and subsequently wheat and corn through 1880 which coincides with the peak of farmland area (Schneider 1996). Although abandonment of farmland generally ensued thereafter resulting in a proportional increase in forest in AA Co., it is known that much of the study watershed was being used for agriculture in the early 1900s and a series of in-stream ponds and four cement weirs were constructed in the 1930s. An abrupt increase of urbanization in the 1960s increased sediment loadings to Cypress Creek, and the pond embankments breached in the1970s resulting in stream downcut through the sediment deposits. By the 1990s, Cypress Creek was deeply incised, with active head cuts and severe bank erosion. The upper intertidal zone of Cypress Creek has had to be dredged due to extensive bedload transport associated with this erosion and provided some impetus for its restoration.

The area of the Cypress Creek watershed monitored in this study is 143 ha, which is about 98% developed with 45% impervious cover (Williams and Filoso 2015). The SWC constructed in the lower mainstem to the estuarine intertidal zone is 3.6 ha and is a hybrid design that incorporates a large headwater wetland followed by a braided stream network and step-pool conveyance (Fig. 1). Small, lateral RSCs were incorporated into the SWC design. The RSCs are a type of step-pool conveyance system that include boulder weirs and stone cobble riffles that separate individual ponding basins, and a sand/wood chip matrix (usually <20% organic matter by volume) within the ponding basins to enhance stormwater percolation and biological activity (Brown et al. 2010; Williams et al. 2016). It appears that most of the organic matter (i.e., wood chips) in the RSCs of this study was applied superficially to steep embankments of the structures. Construction of the SWC resulted in a loss of forest habitat but a net gain of 1.3 and 1.8 ha of wetland and riparian forest, respectively. The subcatchment areas where the RSCs were implemented are 15 and 8 ha (located on Leelyn Dr. and Isaiah Dr.; henceforth, RSC1 and RSC2, respectively). The subbasins of the RSCs represent about 16% of the basin area monitored.

The percentages of major land use categories for the Cypress Creek catchment and other urban areas of Maryland in our analyses were determined using a combination of 2010 land use data (Mid-Atlantic Regional Earth Science Applications Center, MA-RESAC) and existing boundary delineations. Delineations for the Cypress Creek watershed were obtained from the MD Department of Planning (MDP) and Anne Arundel Co. Department of Public Works (AA Co. DPW). Delineations and land use for the smaller RSC subcatchments were derived from the conceptual design plans provided by the AA Co. DPW. Watershed boundaries for the Patuxent and other river segments of the Hydrologic

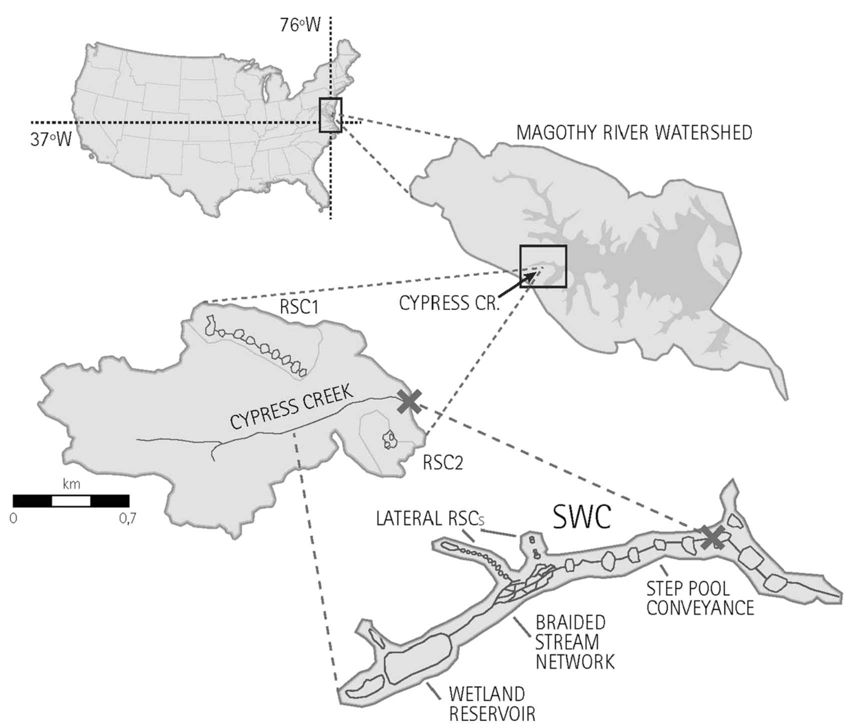


Fig. 1 The Magothy River watershed on the western shore of the Chesapeake Bay. Total area of Cypress Creek upstream of the main stem monitoring station is 143 ha. The drainage areas of *RSC1* and *RSC2* are 15 and 8 ha, respectively. The area of the SWC restoration is 3.6 ha. Monitoring stations were located at the outflows of each drainage area and that of the SWC (designated by *X*) was upstream of the step pool conveyance which extends to the intertidal interface

Simulation Program Fortran (HSPF) watershed model (phase 5.3.2) were obtained from the Chesapeake Community Modeling Program (ches.communitymodeling.org/models/CBPhase5/datalibrary/watershed-GIS-data.php).

Pre- and Post-Construction Periods in the Cypress Creek Watershed

Pre-construction sampling at RSC1 occurred from December 2010 to early July 2011. Sampling was curtailed during the construction period from July to August 2011 and resumed in the post-construction period. Pre-construction stormwater runoff at the outflow of RSC2 was collected from March 2011 through April 2012. The post-construction period began in July 2012. Pre-construction monitoring of the Cypress Creek mainstem (SWC) commenced in June 2008. Construction lasted from June 2012 through February 2013, at which time the post-construction phase began. Post-construction monitoring at all three sampling stations ended in October 2014.

Methods

Sampling

Water samples were collected with discharge measurements immediately downstream of each RSC. Water samples and instantaneous discharge data were collected from the outflow of the SWC during baseflow (monthly through October 2014) and stormflow (event-based) periods; the RSCs in this study did not have baseflow. Baseflow conditions were defined as periods of low flow when the effect of precipitation on stream flow was minimal (i.e., approximately 2 days after a storm event) and stream stage was relatively stable. Storm events were defined as a measurable precipitation event (>0.254 mm) with a ≥12-h antecedent dry period. Stormflow samples were collected over the rising and falling limbs of the stormflow hydrograph using automated samplers (Teledyne Isco 6712). Stormflow sampling was generally done twice per season to account for seasonal variation.

Water samples were collected in acid-washed high-density polyethylene (HDPE) bottles and filtered either immediately or within a few hours after collection. All samples were stored on ice after collection while being transported to the Chesapeake Biological Laboratory (CBL) where they were refrigerated at 4 °C. At the laboratory, samples not filtered in the field were filtered within 12 h using glass-fiber filters (nominal pore size of 0.7 μm) to separate dissolved from particulate constituents. Dissolved and particulate samples were stored in a freezer at CBL prior to analyses.

Field Measurements

A combination of tipping bucket (Hobo RG3-M) and bulk (Tenite) rain gauges was used at two sites to measure precipitation volume (August 2011 to December 2014). One site was located about 0.56 km from RSC1 and about 0.92 km from the SWC. These gauges were unobstructed by objects such as overhanging trees and power lines. The tipping bucket rain gauge was inspected and its logger downloaded every one to two months, whereas the plastic gauge was routinely monitored on a single event to biweekly basis. Air temperature was recorded continuously at 5-min intervals by the tipping bucket logger. The logger associated with the tipping bucket rain gauge was protected by a solar radiation shield (Onset Hobo RS1) to record accurate air temperatures. Stream temperatures were recorded continuously at all sites at 5-min intervals using Onset HOBO water level loggers (U20-001-04). These pressure transducers were also used to record continuous stage height at the outflows at each sampling station; separate loggers located inside the stream autosampler's fiberglass housing were used to record barometric pressure, which was subtracted from the stage data.

Instantaneous discharge was measured using the cross-sectional area method (Gordon et al. 2004). Discharge measurements were done immediately after water collection during the monthly baseflow sampling events and also during variable water stages for stormflow. Instantaneous discharge data for each site were used to create rating curves and convert continuous stage data into discharge (L s^{-1}).

Field measurements included dissolved oxygen (DO) concentrations (mg L^{-1}) and conductivity (μS cm^{-1}) from the SWC for part of the growing season in 2013 and 2014 (Hobo probes U26-001 and U24-001, respectively). Probes were installed downstream of the stone cobble riffle where the Isco strainer was located, from 10 to 20 cm below the water surface. Diel concentrations of DO were used to estimate stream metabolism (Roberts et al. 2007) and the influence of restoration on aquatic trophic status.

Lab Analyses and Data Processing

Water samples were commonly analyzed for nitrate, ammonium, total dissolved N, and particulate N (NO_3, NH_4, TDN, and PN, respectively); dissolved organic N (DON) is TDN minus the inorganic fractions, and TN is the sum of TDN and PN. Other constituents analyzed were phosphate and total dissolved phosphorus (PO_4 and TDP, respectively), TSS, chloride and sulfate (Cl and SO_4, respectively), and dissolved organic and particulate carbon (DOC and PC, respectively). Particulate phosphorus (PP) was analyzed on a subset of stormflow and baseflow samples to calculate the PP/PN ratio, which was determined to be a good indicator of relative concentrations of PP and PN in other streams of the area (unpublished data). Ratios were used to estimate PP for all samples having PN concentrations, and these estimates were used to calculate TP.

Methods and detection limits of those constituents analyzed at the Nutrient Analytical Services Laboratory (NASL) are available online (nasl.cbl.umces.edu). Other constituents were measured in a separate laboratory at the CBL, including NO_3, Cl, and SO_4 that were determined using a Dionex ion chromatograph (ICS-1000). Particulate N and C were measured with a PerkinElmer 2400 CHN elemental analyzer.

Volume-weighted means (VWMs) were used when representative discharge data were available for the majority of stormflow events sampled. Volume-weighted mean concentrations were calculated as follows: VWM = $(\Sigma C_i Q_i) / \Sigma Q_i$, where Ci is the observed solute concentration of instantaneous stream flow i, Qi is the discharge volume (L) estimated for the interval between sample collections, and the denominator is the Σ of discharge volume. Flow-weighted means (FWMs) were used for baseflow (SWC) and were calculated similar to VWM but with Qi representing instantaneous discharge (L s^{-1}) at the time the sample was collected. No samples collected during the construction periods at any site were used in our pre- and post-construction loading comparisons. Statistical differences with normally distributed data were determined using two-tailed t tests and, for all others, with the Mann-Whitney rank sum test. Significance ($p \leq 0.05$) and non-significance are designated with an asterisk and NS, respectively. All tests were done using SigmaPlot 12.

Load Reductions

Although arguable, stream restoration is considered by some in the restoration community as a BMP. However, this term implies that the best available technology and management strategy are incorporated into the stream restoration design to achieve the best outcome, which is not always the case. We argue that many urban stream restorations in the Chesapeake Bay watershed now vastly transform ecosystems in an attempt to enhance specific ecosystem functions (e.g., denitrification) and support desirable ecosystem services (Palmer et al. 2014). However, because of their novelty, our understanding of the extent to which various stream restoration designs and their position within a watershed can maximize pollutant reductions while simultaneously providing some degree of ecological uplift is still unclear. Because there are many types of stream restoration designs with unknown levels of effectiveness, to categorize all designs as BMPs is problematic. Thus, we refer herein to the SWC as a stream restoration technique rather than a BMP since we still know so little of its potential benefits as a management practice in an urban context.

Load reduction estimates for the BMPs and stream restoration in this study were done by standardizing the hydrology for the pre- and post-construction periods and then modifying the pre-construction hydrology with estimated runoff reductions measured from the RSCs and SWC in the post-construction environment (Supplementary Material). Minimum and maximum estimates of possible reductions in flow associated with retention and loss in the RSC and SWC were used to then calculate a range of potential nutrient and sediment reductions. Maximum reduction of flow volume in the post-construction environment was estimated as a combination of total water retention in the RSCs and loss due to enhanced PET from the constructed wetland area of the SWC (2 ha). Estimates of evaporation (NOAA 1982; Abtew and Melesse 2012) from the ponded water surfaces of the created wetland were used to estimate annual water loss from this area. An estimate for the minimum reduction of flow volume was calculated assuming that half the water retained in the RSCs subsequently contributed to baseflow and that the wetland area contributing to PET was 1.3 ha.

Modeling

The CBP's linked watershed and estuarine models were used to ascertain the impacts of changing climate for end-of-the-twenty-first century projections of pollutant runoff to Chesapeake Bay. In the first modeling exercise, the last three decades of observed precipitation data (hourly) in the CB watershed was obtained from NLDAS-II, a comprehensive high-resolution climate reanalysis of gauge-only data, and gridded data were spatially aggregated to the phase 5 CBP watershed model land segments. For each land segment, seasonal quantile thresholds of rainfall intensity were identified and, for consistency, quantiles were based on the average rainfall from 1991 to 2000. Precipitation intensity based on these quantiles was calculated for the period of 1980 to 2014, and seasonal regression slopes were used to compute mid-twenty-first century rainfall projections. The watershed model was run using (a) a rainfall projection to 2050 based on a 30-year trend analysis at seasonal and land-segment scales, (b) an ensemble of six GCMs that project increases in air temperature, and (c) a modified representation of transpiration due to changes in stomatal responses to higher ambient CO_2 concentrations (Keenan et al. 2013; Butcher et al. 2014).

Based on the results of the NLDAS-II analysis, the second set of modeling scenarios were generated using three GCMs. These models were selected because they project expected increases in flow in more urban (i.e., developed) areas where impervious surfaces have a positive influence on stormflow runoff. Prognostic variables derived from the ensemble of these three GCMs were used in the CBP watershed model (HSPF phase 5.3.2): BCCR-BCM2.0 (Bergen Climate Model, version 2), CSIRO-Mk3.0, and CCSM3 (Community Climate System Model, version 3.0). Projections were made to the period from 2087 to 2095 (A2 scenario, 9-year average of scenarios), and model output was compared to a base calibration scenario (average of 1991–1999) and WIP scenario (average of 1991–1999), the latter representing the TN, TP, and TSS reductions expected under the full implementation of the TMDL in 2025. Additionally, to evaluate the effectiveness of BMP implementation in urban areas, all nutrient and sediment efficiencies (i.e., the capacity of a BMP to reduce each constituent) in the urban BMPs used in the CBP watershed model were increased by 30%. This increase represents a hypothetical yet reasonable increase in both BMP efficiency and increased implementation of BMPs in urban areas expected this century. As with the NLDAS-II scenarios, a modified representation of transpiration (Keenan et al. 2013; Butcher et al. 2014) was used. Thus, modeled scenarios were generated with and without this transpiration factor to give a possible range in projected changes to nutrient and sediment loads with climate change by the end-of-the-twenty-first century.

Modeling climate change is inherently difficult and response variables will change depending on the type of model and parameterization used. For example, there are various PET formulations available that affect runoff projections (e.g., Hamon, Oudin, Penman-Monteith). The PET formulation of Hamon used in our modeling exercises is relatively high compared to other formulations and thus has a relatively large decreasing effect on runoff. By contrast, the effect of CO_2 concentrations on stomatal resistivity was relatively large in our projections, which has the opposite effect of increasing runoff. Thus, our end-of-the-twenty-first century projections were done both with and without the transpiration factor, and those with this factor were included in part to counteract potentially exaggerated reductions in runoff due to using Hamon PET.

Results

Representativeness of Storm Sampling

There were 195 events ≥5.1 mm (those typically resulting in stormflow) recorded with the tipping bucket and bulk rain gauges over the post-restoration period (March 2013 to December 2014). On-site precipitation records are only available from August 2011, and 2012 was a drought year (762, 1178, and 1386 mm from 2012 to 2014, respectively), so there were only two stormflow samples collected during the drought period. Although contingent on antecedent precipitation, storm sizes <5.1 mm generally did not result in measureable stormflow. Sampling effectiveness of stormflow events varied according to categories of different storm sizes. For example, storm sizes ranging from 5.1 to 10.2 mm represented 32% of the total number of storm events that occurred over the post-restoration period (Fig. 2). Of these storm events, 3% were sampled for stormflow and this category represented 1% of total stormflow runoff. By contrast, storm sizes >50.1 mm represented only 3% of the total number of storm events that occurred over the post-restoration period, with 70% (five of seven) sampled for stormflow (Fig. 2). This category represented 44% of the total stormflow runoff over the post-restoration period. Stormflow sampling effectiveness during the pre-restoration period of 2008 to 2011 was likely similar to that of the post-restoration period since the number of storms sampled and seasonal distributions were similar, and mid- to large-sized storm systems were typically targeted for sampling during both periods.

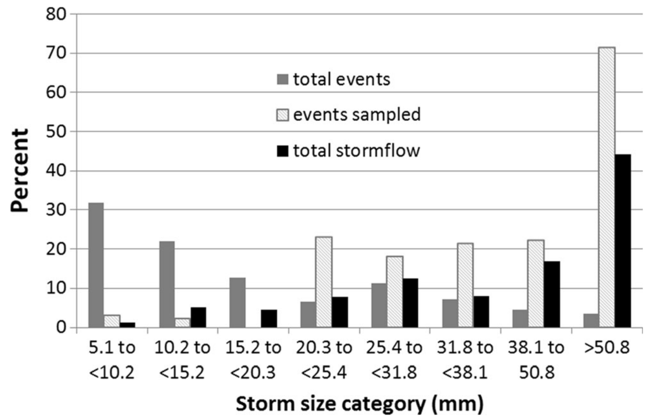


Fig. 2 Percentages of the sampling effectiveness at the SWC for different storm-size categories in the post-restoration period (March 2013 to December 2014) when we had an on-site tipping bucket rain collector installed. A total of 195 storm events >5.1 mm occurred over the post-restoration period, and these typically resulted in measurable stormflow runoff. Storm sizes >50.8 mm represented only 4% of the total number of storm events but accounted for over 44% of total stormflow runoff; 70% (five of seven) were sampled

Baseflow and Stormflow

A total of 7 and 11 stormflow samples were collected from RSC1 in the pre- and post-construction periods, respectively. A comparison of median TN, TDP, and TSS concentrations in the pre- and post-construction periods for RSC1 indicates that TN and TDP were commonly lower in the post-construction period, whereas TSS was higher (Fig. 3a–c). The VWM TN concentration of stormflow at RSC1 in the post-construction period was approximately half of that in the pre-construction period (Table 1). Ammonium, DON, and TDN concentrations were larger as a percentage of TN in post- compared to the pre-construction means. Post-construction VWM TDP concentrations were lower than those in the pre-construction period, with PO_4 accounting for a larger proportion of TDP in the post-construction period (Table 1)

By contrast, a total of 5 and 16 stormflow samples were collected from RSC2 in the pre- and post-construction periods, respectively. Median concentrations of TN and TSS at RSC2 were higher in the post- than in the pre-construction period (Fig. 3d–f), whereas TDP was slightly lower. Mean

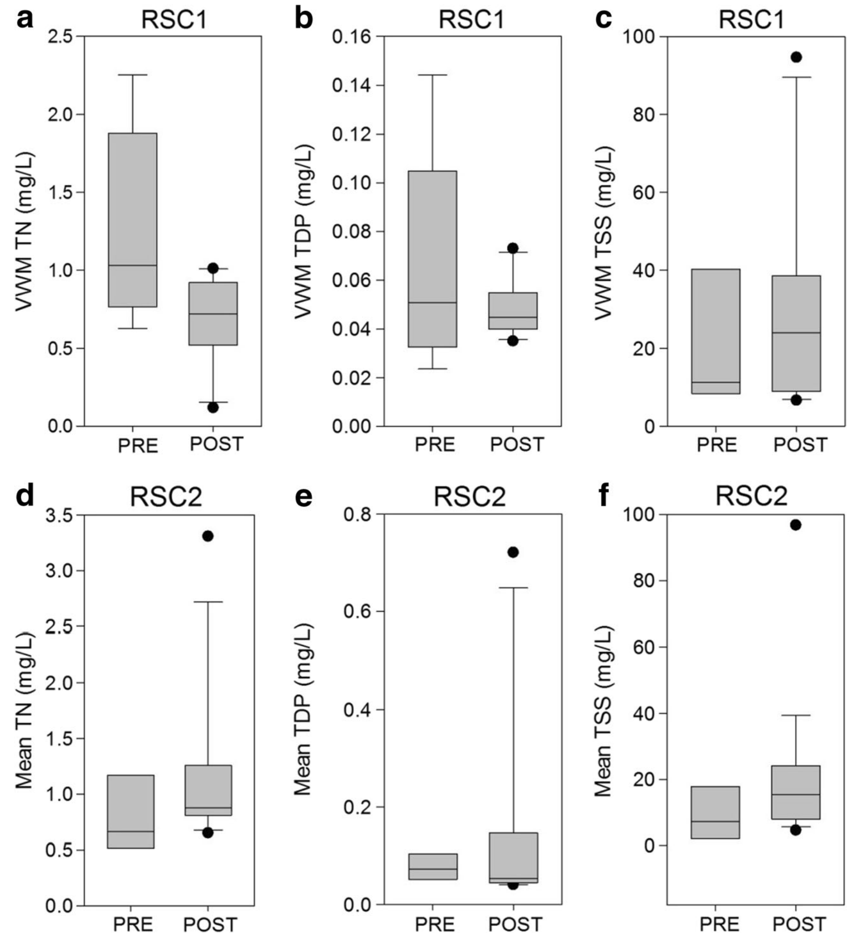


Fig. 3 **a**–**f** (*Left to right*, *top to bottom*) box and whisker plots of VWM TN, TDP, and TSS concentrations (mg L^{-1}) in the RSCs for all stormflow events sampled during pre- and post-construction periods. Box and whisker plots show the median, bounded by the 25th and 75th percentiles. *Bars* depict the minima and maxima, while *dots* depict the mild and extreme outliers associated with individual storm events

TN and TDN concentrations for the pre-construction period at RSC2 were 0.8 and 0.52 mg L^{-1}, respectively (Table 1). Total N was mostly composed of dissolved fractions while PN contributed 35 and 41% of the pre- and post-construction total, respectively. Mean concentrations of all the N fractions were higher in the post- than in the pre-construction period (Table 1), as were those of PO_4 and TDP. Total dissolved P exhibited a proportionally larger increase than PO_4. The mean TSS concentration was about two times higher in the post- compared to the pre-construction period (Table 1).

About 77% of the annual discharge at the SWC sampling station was composed of stormflow. A total of 19 and 24 stormflow events were sampled in the pre- and post-restoration periods of Cypress Creek, respectively. Median concentrations of TN and TSS in stormflow of the SWC were lower, whereas TDP was higher in the pre- compared to the post-restoration period (Fig. 4a–c). Median TSS concentrations were only slightly lower in the post-restoration period, and the higher median concentration of TDP was accompanied by higher variability. Dissolved N fractions constituted about 78% of TN in the pre-construction period, in contrast to about 52% in the post-construction period (Table 1). Volume-weighted mean TN concentrations in the post-construction period were lower than those of the pre-construction period, but the relative contributions of PN to TN were similar (47 and 43%, respectively). Although the proportional decrease in VWM NO_3 concentrations from the pre- to the post-construction periods was only slightly higher than for NH_4 (56 vs 50%, respectively), the decrease in NO_3 had the largest impact on the decrease in TN concentrations (Fig. 5a). TDP was higher in the post-construction period compared to the pre-construction period as a result of higher PO_4 concentrations (Table 1, Fig. 5b). Volume-weighted mean TSS concentrations decreased by 32% from 64 to 43 mg L^{-1} (Table 1).

A total of 35 and 30 baseflow samples were collected in the pre- and post-restoration periods of Cypress Creek, respectively. Samples affected by groundwater pumping in October and November 2010 and sewage overflows in May 2010 (pers. comm., Paul Dumar, Broadneck WRF, AA Co. Department of Public Works) were omitted. Median concentrations of TN in baseflow of the SWC during the post-construction period were lower than those in the pre-construction period, whereas those of TDP and TSS were higher (Fig. 4d–f). While TN concentrations in baseflow were similar in the pre- and post-construction periods, the form of N exported changed dramatically (Fig. 6a). Dissolved N fractions in baseflow of the SWC

Table 1 Mean stormflow (SF) and baseflow (BF) concentrations for all constituents

Metric	Flow	Cypress Creek–SWC			RSC1			RSC2		
		Pre	Post	% change	Pre	Post	% change	Pre	Post	% change
TN	BF	0.78	0.82**	*5*	–	–				
	SF	1.15	0.84*	*−27*	1.32	0.65**	*−51*	0.80	1.19**	*49*
PN	BF	0.06	0.30**	*400*						
	SF	0.54	0.36	*−33*	0.39	0.19**	*−51*	0.21	0.44**	*110*
TDN	BF	0.75	0.48	*−36*	–	–				
	SF	0.90	0.44*	*−51*	0.99	0.50	*−49*	0.52	0.71	*37*
NO_3	BF	0.41	0.06**	*−85*						
	SF	0.39	0.17*	*−56*	0.37	0.13**	*−65*	0.17	0.20	*18*
NH_4	BF	0.11	0.08	*−27*						
	SF	0.12	0.06	*−50*	0.03	0.02	*−33*	0.09	0.19*	*111*
DON	BF	0.22	0.34*	*55*						
	SF	0.20	0.22	*10*	0.57	0.34	*−40*	0.29	0.32	*10*
TP	BF	0.02	0.08**	*300*						
	SF	0.17	0.13	*−24*	0.18	0.11	*−39*	0.15	0.27*	*80*
PP	BF	0.02	0.07	*250*						
	SF	0.15	0.10	*−33*	0.11	0.06	*−45*	0.07	0.12	*71*
TDP	BF	0.002	0.011*	*450*						
	SF	0.022	0.029	*32*	0.07	0.05	*−29*	0.08	0.15	*88*
PO_4	BF	0.001	0.002*	*100*						
	SF	0.009	0.015	*67*	0.03	0.04	*33*	0.06	0.09	*50*
DOP	BF	0.002	0.010*	*400*						
	SF	0.013	0.014	*8*	0.04	0.01	*−75*	0.02	0.06	*200*
TSS	BF	4.9	14.6**	*198*	–	–				
	SF	63.6	43.1	*−32*	21.9	22.9	*5*	9.4	20.2**	*115*
Cl	BF	48.2	57.3**	*19*						
	SF	59.9	25.9	*−57*	23.0	3.1**	*−87*	22.5	24.2	*8*
SO_4	BF	8.3	7.2**	*−13*						
	SF	6.4	3.4	*−47*	2.5	1.4	*−44*	1.7	2.1	*24*
PC	BF	0.8	3.3**	*313*						
	SF	7.0	4.2	*−40*	5.6	1.6*	*−71*	1.9	3.4	*79*
DOC	BF	2.4	10.2**	*325*						
	SF	4.9	6.1	*24*	6.4	9.1	*42*	3.0	45.9**	*1430*

Stormflow concentrations are expressed as volume-weighted means (VWMs; i.e., Cypress Creek and post-construction RSC1), flow-weighted means (FWMs; pre-construction RSC1), or the mean of individual storms sampled (RSC2). Baseflow is expressed as FWMs. Values are in milligrams per liter. The change from the pre- to post-construction periods is given as a percent decrease (−) or increase (+) over the pre-construction concentration, and statistically significant differences are designated by asterisks (*$p \leq 0.05$; **$p \leq 0.01$)

constituted about 93% of TN in the pre-construction period, in contrast to 56% in the post-construction period (Table 1). During the pre-construction period, about half of baseflow N was in the form of NO_3, in contrast to the post-construction period when the dominant form changed to DON. Post-construction DON and PN concentrations increased substantially to about 41 and 37% of TN, respectively; DON showed a clear increase at the beginning followed by a decrease at the end of the growing season (Fig. 6a). By contrast, NH_4 concentrations decreased to 72% of its pre-construction values and NO_3 decreased from 0.41 to 0.06 mg L^{-1} (Table 1). Concentrations of TDP were higher in the post-construction period compared to the pre-construction period, with most of the increase a result of higher dissolved organic P (i.e., TDP minus PO_4) concentrations (Table 1). As with DON, DOP and TSS concentrations showed distinct seasonality (Fig. 6b, c). The concentration of TSS was higher in the post- compared to the pre-construction period by about 10 mg L^{-1}, whereas those of Cl and SO_4 were higher and lower in the post- compared to the pre-construction period, respectively (Table 1).

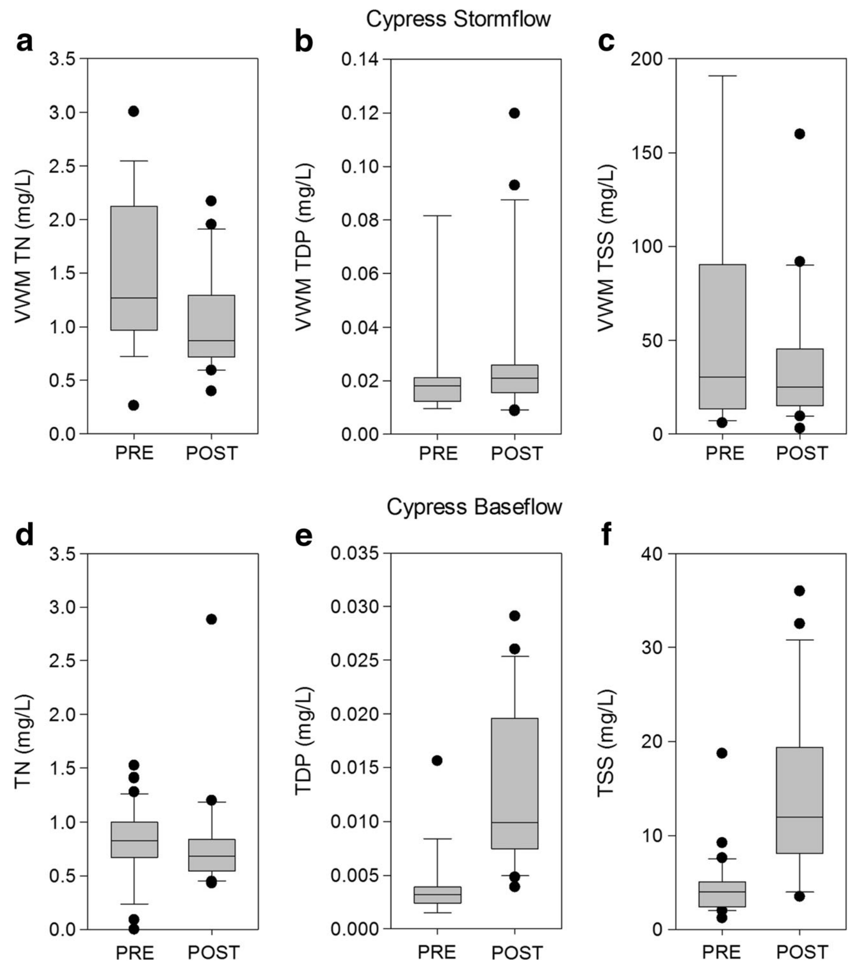


Fig. 4 **a**–**f** (*Left to right*, *top to bottom*) box and whisker plots of VWM TN, TDP, and TSS concentrations (mg L^{-1}) in stormflow and baseflow for all events sampled during the pre- and post-construction periods at Cypress Creek. Box and whisker plots show the median, bounded by the 25th and 75th percentiles. *Bars* depict the minima and maxima, while *dots* depict the mild and extreme outliers associated with individual storm events

Dissolved Oxygen and Stream Metabolism

Several Coastal Plain wetland streams sampled as part of another study were used to provide comparisons with Cypress Creek (143 ha, 98% urban land use, lat/long 39.075860°, −76.535976°). Gross primary production (GPP) was higher in the SWC compared to that of Dividing Creek (forested stream that served as a comparison site; 89 ha, 87% urban land use, lat/long 39.050569°, −76.515527°), Howard's Branch (SWC constructed in 2001; 96 ha, 49% urban land use, lat/long 39.021134°, −76.548014°), and Parker's Creek (natural wetland; 3237 ha, 16% urban land use, lat/long 38.536101°, −76.521857°; Fig. 7). In 2014, DO probes were installed at both sites to collect a longer time series of data throughout the summer and fall. Data expressed as daily averages show high variability of DO concentrations in the SWC (Fig. 8) compared to Dividing Creek that tended to stay above the 5 mg L^{-1} criteria threshold for freshwater systems. Dissolved oxygen concentrations in Cypress Creek exhibited occasional anoxia and hypoxia and sometimes decreased following storms presumably because the stormwater pulse flushed hypoxic bottom water out of the series of upstream step pools.

Stream Temperature

Stream temperature recorded simultaneously with stage using data loggers allowed us to make comparisons between pre- and post-restoration periods. Average streamwater temperatures were calculated using all available data for the SWC above 20 °C in each year. The comparison indicates that there was an abrupt increase in streamwater temperature, mostly in the summer, after the construction period ended in March 2013 (Fig. 9). This increase is in contrast to air temperatures (data not shown) that decreased by about 1.2 °C each year from the summer of 2012 (pre-construction) through 2014. Average streamwater temperatures (June to September 2008 to 2014) were higher (*) in the post-construction period by about 2 °C, and maximum temperatures in the pre- and post-construction periods averaged 29 and 33 °C, respectively, likely because of increased solar radiation and a lack of shading from a forest canopy in the post-construction period (Fig. 9).

Load Reductions

Decreases in pollutant loading expressed as area yields generally occurred for each constituent in both RSCs (Table 2). In

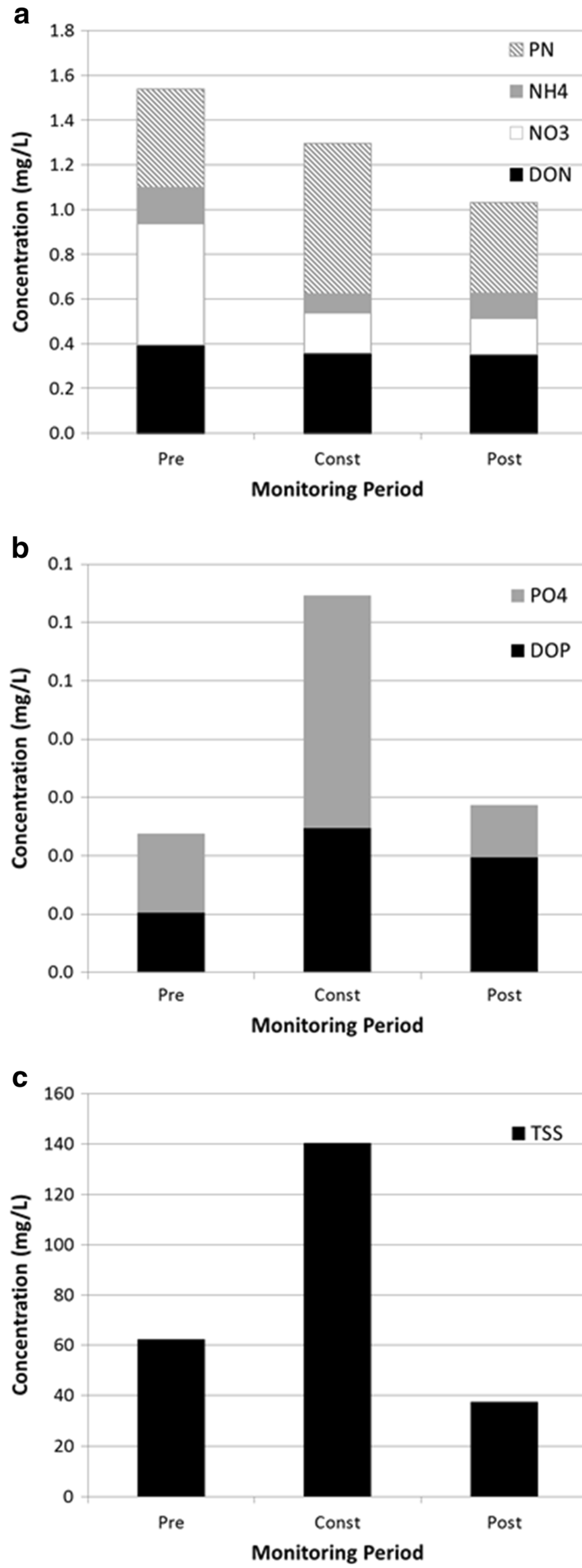


Fig. 5 **a**–**c** (*Top to bottom*) concentrations of N and P fractions and TSS ▶ in stormflow during the pre- and post-construction and construction periods at Cypress Creek

RSC2, by contrast, relatively small loading increases were observed for NH_4, PN, DOP, TSS, and DOC. Area yields in the SWC decreased with all constituents except for DON, TDP, PO_4, DOP, and DOC (Table 2).

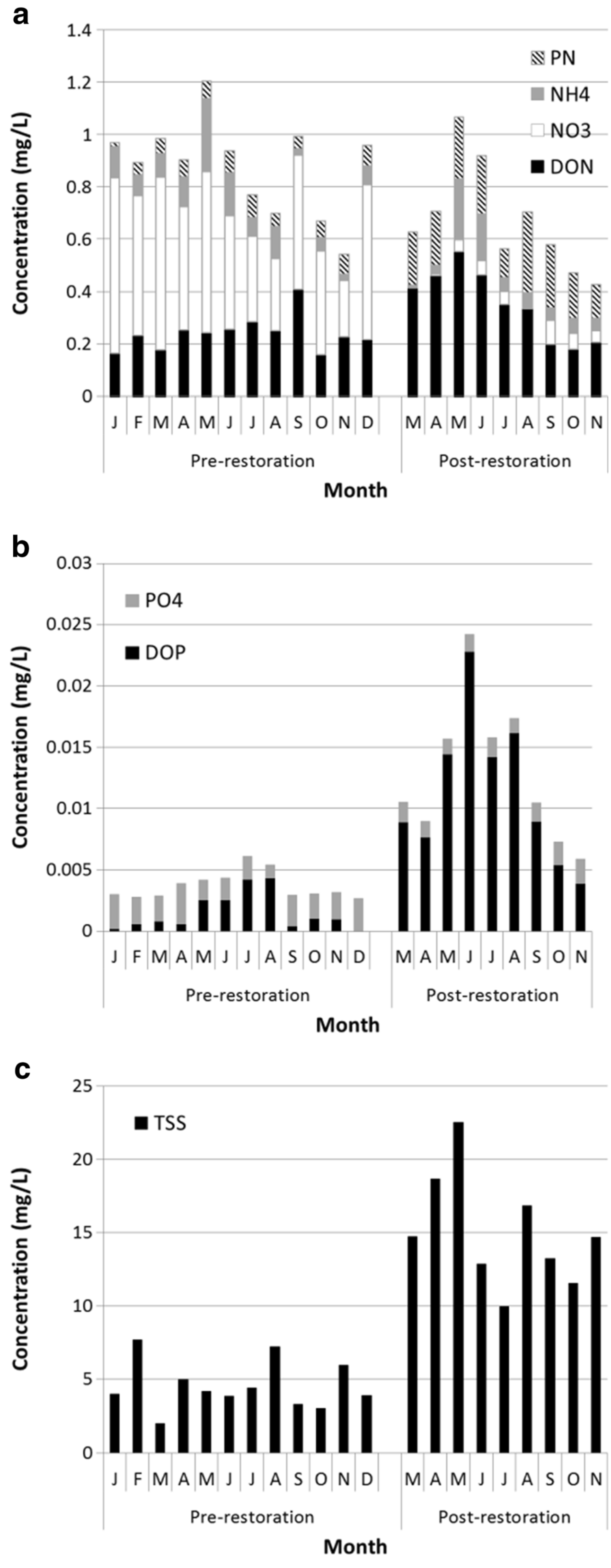
Modeling Pollutant Loads and Estuarine Response

Using NLDAS-II precipitation as forcing functions in the CBP watershed model produced spatially explicit forecasts of pollutant loads and runoff by mid-century. The analysis showed positive trends in the upper tenth percentile of the precipitation distribution across the Chesapeake Bay watershed, and these were statistically significant (Yactayo et al. 2015). The model projected increases of 5.5, 12.6, 22, and 66% for temperature, PET, stormflow, and sediment export, respectively, with a 12% increase in precipitation for MD (Fig. 1, Supplementary Material).

We used the average of the GCMs with and without the modified transpiration factor to represent the range of projected increases in runoff and nutrient and sediment loads (Table 3). Part of our scenario analysis included increasing the BMP effectiveness in a wetter climate by 30%, representing not only an increase in BMP efficiency but also overall implementation, to draw inferences about how such changes will mitigate expected increases in loads by the end-of-the-twenty-first century. To further elucidate the effects of changing climate in urban watersheds of MD, area yields of four basins (the largest of which was the entire Patuxent River watershed; Fig. 2b in Supplementary Material) with varying proportions of urban land cover (37 to 79%; Fig. 2a in Supplementary Material) were compared and these generally showed decreases from the base scenario to the WIP scenario, followed by an increase expected in a wetter climate. The average increase in runoff for the late-twenty-first-century scenarios with and without the effect of decreased transpiration associated with changes in stomatal resistance were from 4 to 7% and from 24 to 33%, respectively, with increasingly larger values for catchments with more urban area in both cases. When the nutrient and sediment efficiencies of all urban BMPs used by the CBP watershed model were increased by 30% (Table 4), decreases in the area yields of TN (i.e., the % return to TMDL values) ranged from 7 to 14% (lowest) and from 10 to 44% (highest). Reductions generally increased in proportion to the amount of developed land in each basin (Table 3, Fig. 3 in Supplementary Material)

Lastly, the effects of climate change on the Patuxent River estuary were evaluated. For example, a 14% increase in runoff resulted in a 180% increase in hypoxic volume (i.e., DO <2 mg L^{-1}) by the end-of-the-twenty-first century.

Fig. 6 **a**–**c** (*Top to bottom*) monthly average TN, TDP, and TSS ▶ concentrations in the baseflow of Cypress Creek during the pre- and post-construction periods. Total N is partitioned into DON, NO_3, NH_4, and PN, and TDP into DOP and PO_4. Pre-construction concentrations include samples collected since 2008. The construction period occurred from July 2012 to March 2013



Remediation through the implementation of urban BMPs only resulted in a 9% decrease in this hypoxic volume.

Discussion

Effects of Upland RSCs and Stream Restoration on the Flow and Water Quality of Cypress Creek

That we are aware, our study provides the first comparisons of pollutant reduction and performance capabilities of upland stormwater runoff BMPs (RSCs) versus a lowland stream restoration (SWC) in relation to TMDL goals for TN, TP, and TSS in the Chesapeake Bay region. Data were collected in the pre- and post-construction periods of Cypress Creek while a nearby forested catchment stream was used to provide some comparative data in the post-construction period (e.g., DO).

The novel RSCs implemented in the watershed retained and infiltrated considerably more runoff than the original structures, reducing the volume of stormwater conveyed by 63% (Table 2). The reduction in water volume resulted in load reductions for TN, TP, and TSS of 3, 0.94, and 33.6 kg ha^{-1} $year^{-1}$ (Table 2). The minor decrease in TP and increase in TSS in RSC2 were likely due to the erosion of loosely compacted soils and organic matter (i.e., wood chips and organic-rich topsoil) placed on steep hillslopes of the structure.

In the stream channel, the implementation of the SWC resulted in lower peak flows and stormwater runoff velocities, but a much smaller reduction in total runoff compared to the RSCs when normalized by drainage area (i.e., 331 vs 1439 m^3 ha^{-1} $year^{-1}$, respectively). Considering that the decrease in the total runoff observed in the stream channel was in part a result of improved stormwater retention in the RSCs, removing this storage from the stream channel decreases this value to 104 m^3 ha^{-1} $year^{-1}$. Water residence time in the stream channel also increased after the SWC implementation (i.e., >2 days after a storm event). However, the total runoff in the post-construction period was generally within 10% of that observed during storms of similar sizes and intensity in the pre-construction period (data not shown).

We attribute the relatively small change observed in total runoff in the SWC to the perennial nature of Cypress Creek. The SWC was designed to shunt initial stormflow runoff into the large wetland area at the upper reaches, where some is temporarily stored and slowly drains into the braided stream network (mid-reach) and ponds of the conveyance system (lower reach). Because the ponding basins of the lower

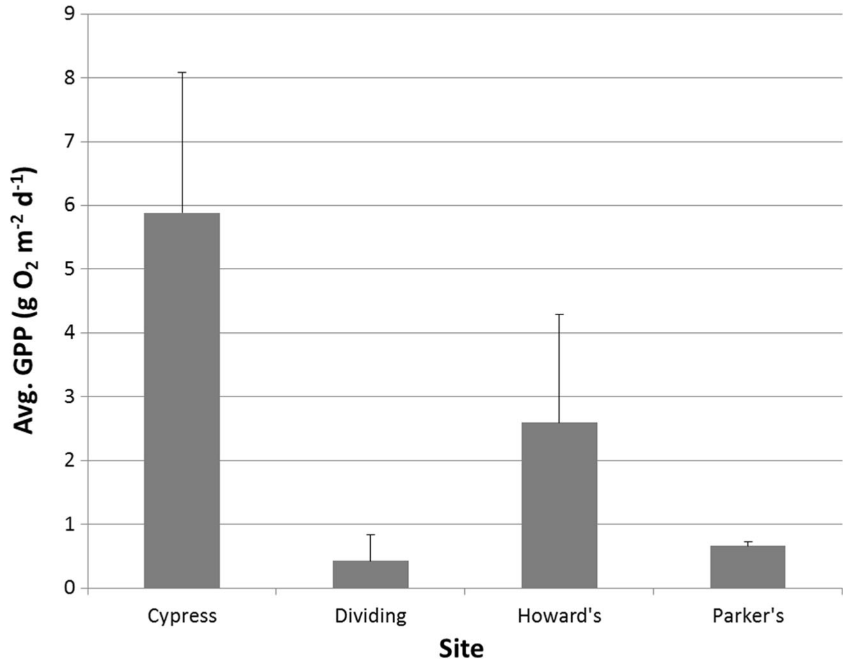


Fig. 7 Average gross primary productivity (*GPP*) with standard *error bars* for several Coastal Plain creeks in MD. Dividing Creek is a forested first-order creek, whereas Howard's Branch (Coastal Plain stream wetland complex constructed in 2001) and Parker's Creek (natural Coastal Plain stream wetland complex) are wetland systems. Dissolved oxygen and conductivity probes were deployed several times in 2013 (Cypress Creek: June 11–26, July 23–August 13, and September 11–25; Dividing Creek: May 5–16, August 13–September 4, and September 27–October 18) at least 1 day after storm activity

SWC are shallow (<1.5 m) and usually full from a combination of residual stormflow and baseflow, the system's capacity to infiltrate and retain additional stormflow is limited. Therefore, while the SWC increased baseflow runoff above that observed in the pre-construction period by about 6%, as well as water residence time during smaller storm events, its capacity to moderate flow and increase water residence time during large storm events appears to be relatively small. The increase in baseflow is likely a result of stormwater runoff retained in the upstream reservoir of the SWC and the RSCs that percolate stormwater runoff to the groundwater table which later emerges as bank seepage into the stream. One benefit of this increased stormflow infiltration is baseflow buffering capacity, as evidenced by the no-flow periods that were observed in the pre- but not the post-restoration period. Including the water loss attributed to PET in the created wetland areas (Supplementary Material), post-construction baseflow would be about 17% higher than in the pre-construction environment.

Large storms are responsible for exporting most of the annual loads of nutrients and sediments from urban streams in the region (Filoso et al. 2015). Therefore, if the capacity of restored streams to moderate stormflow during large storms is relatively small, their capacity to process nutrients and retain sediments are also likely compromised since such processes are a function of streamflow velocity and water residence time (Hall et al. 2009; Mulholland et al. 2009). Thus, because water storage capacity remains relatively constant (except for very

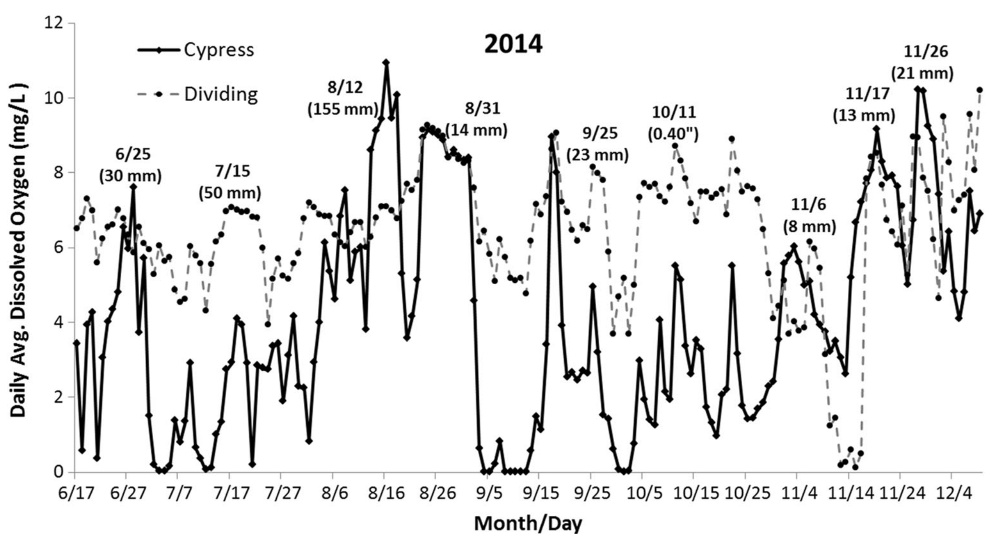


Fig. 8 Time series of daily average DO in Cypress Creek (SWC) and Dividing Creek (forested control) from June through December 2014. Storm event sizes (mm) are indicated for various dates

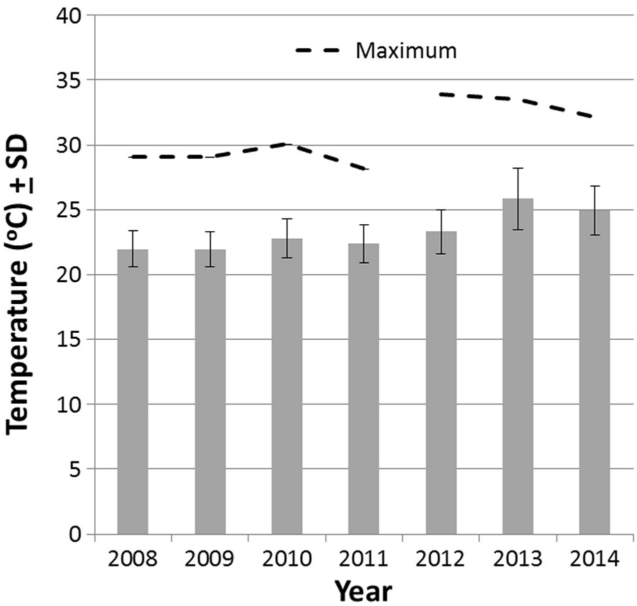


Fig. 9 Average streamwater temperatures for all records above 20 °C available from August to September (2008 to 2014) with corresponding *error bars*. Measurements were obtained from in-stream pressure transducers recording temperature and absolute pressure at 5-min intervals. Construction of the SWC was from June 2012 through March 2013

dry periods) in perennial systems such as the SWC of Cypress Creek, during larger storms, the system is simply conveying a combination of stormwater runoff and flushed water that had been stored temporarily in the system's reservoir and multiple ponding basins. Nevertheless, the runoff from smaller-sized storms can be stored from days to weeks in the system depending on antecedent rainfall, which can promote nutrient uptake and loss. Consequently, baseflow concentrations are the most informative in terms of how much the SWC potentially processes and transforms N and P and stores sediments in this lowland system.

Dilution effects on solute concentrations must also be evaluated. The upstream reservoir of the SWC undoubtedly has a large influence on the solute composition of baseflow and stormflow at our downstream monitoring station. The magnitude of this influence was estimated using the more conservative solutes of Cl and SO_4 to approximate a dilution factor. The dilution factor was estimated by first removing the data from December through March to reduce potential effects of road salts. This factor allowed us to evaluate which constituents were predominately influenced by dilution or other mechanisms (e.g., the uptake or loss of N). The ratio of FWM concentrations indicated that there was a three- to fourfold decrease

Table 2 Summary of runoff, catchment area, and area yields for all study sites

Site	Cypress Creek and SWC			RSC1			RSC2		
	Pre	Post	±	Pre	Post	±	Pre	Post	±
Runoff (m^3 $year^{-1}$)	968,537	921,066	−47,471	36,339	10,823	−25,516	15,356	8276	−7081
Area (ha)	143.3			15.0			7.9		
Area yields (kg ha^{-1} $year^{-1}$)									
TN	7.1 (7.3)	5.4	−1.8 (−2.0)	3.2	0.5	−2.7	1.5	1.3	−0.3
TDN	5.7 (5.9)	2.9	−2.8 (−3.0)	2.4	0.4	−2	1	0.7	−0.3
NO_3	2.6 (2.7)	0.9	−1.7 (−1.8)	0.9	0.1	−0.8	0.3	0.2	−0.1
NH_4	0.76 (0.79)	0.4	−0.35 (−0.38)	0.1	0	−0.1	0.2	0.2	0
DON	1.3 (1.4)	1.6	0.3 (0.2)	1.4	0.2	−1.1	0.6	0.3	−0.2
PN	2.9 (3.0)	2.2	−0.6 (−0.8)	0.9	0.1	−0.8	0.4	0.5	0.1
TP	0.91 (0.94)	0.76	−0.15 (−0.18)	1.03	0.23	−0.8	1.29	1.15	−0.14
PP	0.79 (0.82)	0.6	−0.2 (−0.22)	0.87	0.19	−0.68	1.09	0.95	−0.14
TDP	0.11 (0.12)	0.16	0.05 (0.04)	0.16	0.04	−0.12	0.2	0.2	−0.003
PO_4	0.047 (0.048)	0.074	0.028 (0.026)	0.07	0.03	−0.05	0.1	0.1	−0.01
DOP	0.069 (0.072)	0.084	0.015 (0.012)	0.09	0.01	−0.08	0.1	0.1	0.01
TSS	331.2 (343.1)	235	−96.2 (−108.1)	53.1	16.6	−36.5	18.2	21.1	2.9
Cl	378.2 (391.7)	213.2	−165 (−178.5)	55.7	2.2	−53.5	43.7	25.4	−18.4
SO_4	45.4 (47.0)	27.5	−17.8 (−19.5)	6.1	1	−5.1	3.3	2.2	−1
PC	37.1 (38.4)	25.6	−11.5 (−12.9)	13.5	1.2	−12.4	3.7	3.6	−0.1
DOC	28.4 (29.4)	45.2	16.7 (15.7)	15.6	6.6	−9	5.9	48.1	42.2

Changes between the pre- and post-construction periods for area yields indicate decreases as negative numbers and increases as positive numbers. Minimum and maximum reductions are presented with the latter in parentheses. Detailed methods for how each value was calculated are provided in the Supplementary Material section

Table 3 Area yields of runoff, nutrients, and sediment generated from the CBP watershed model for the following periods: base calibration (1991–1999), watershed implementation plan signifying the expected nutrient and sediment decreases with full implementation of the TMDL, A2 scenario (2087–2095), and A2 scenario with a 30% increase in urban BMP efficiency. End-of-the-twenty-first century scenarios were generated using prognostic variables of the CCSM3, CSIRO, and BCCR GCMs

Scenario	Base	WIP	Range			
			A2	A2 + 30% BMP	(2087–2095) minus (1991–1999) WIP	% reduction from 30% increase in BMPs
	Years					
	1991–1999	1991–1999	2087–2095	2087–2095		
Runoff (mm)						
Patuxent (2471 km^2)	357	362	375–447	375–447	14–85	0
Bowie (910 km^2)	364	370	392–476	392–476	21–106	0
Little Patuxent (102 km^2)	478	492	523–656	523–656	31–163	0
Baltimore County (168 km^2)	584	558	599–705	599–705	41–147	0
NO_3 (kg ha^{-1} year^{-1})						
Patuxent	3.8	2.3	2.7–3.1	2.6–3.1	0.4–0.9	6–12
Bowie	3.7	2.2	3.2–3.4	3.2–3.3	1.0–1.2	5–7
Little Patuxent	4.4	2.4	4.9–4.9	4.7–4.7	2.5–2.5	7–9
Baltimore County	5.3	2.7	4.1–6.1	3.9–5.8	1.4–3.4	10–15
TN (kg ha^{-1} year^{-1})						
Patuxent	7.3	5	4–7	4–7	0.5–2	9–18
Bowie	8.1	5.9	8–9	8–9	1–3	7–10
Little Patuxent	12.4	9.5	13–16	12–15	1–6	11–44
Baltimore County	14.3	7.5	11–14	10–13	2–6	14–26
TP (kg ha^{-1} year^{-1})						
Patuxent	0.55	0.45	0.6–0.7	0.6–0.7	0.16–0.25	8–11
Bowie	0.01	0.01	0.4–0.5	0.4–0.5	0.4–0.48	4–10
Little Patuxent	0.85	1.05	1.4–1.9	1.4–1.8	0.39–0.85	3–8
Baltimore County	1.02	0.48	0.6–1.2	0.6–1.1	0.17–0.7	14–18
TSS (kg ha^{-1} year^{-1})						
Patuxent	328	220	313–539	303–528	93–319	4–10
Bowie	409	327	573–921	558–899	246–594	4–6
Little Patuxent	1451	1323	2154–3239	2073–3110	831–1915	7–10
Baltimore County	1218	329	698–1569	576–1369	370–1241	16–33

The column "(2087–2095) minus (1991–1999) WIP" indicates the increase in each parameter expected with climate change relative to the WIP scenario, and the % reduction indicates how much a 30% increase in urban BMP efficiency (representing increased efficiency and distribution) will be able to curtail the expected increase associated with climate change. Basins represent the Patuxent River watershed, the Patuxent watershed to the UGSG gauging station at Bowie (head-of-tide), the Little Patuxent watershed, and Gwynn Falls in Baltimore County, respectively (Fig. 2 in Supplementary Material)

in Cl and SO_4 from the pre- to the post-construction periods. We surmise that this is predominately a dilution effect that occurs when more concentrated baseflow entering the reservoir from upstream is diluted by the less concentrated residual stormflow runoff that was retained in the reservoir during the most recent storm events (Table 1). For example, assuming that only half of the decrease observed in NO_3 from the pre- to the post-restoration period is a result of dilution, the combined increase of DON and PN exceeds the remainder, suggesting that there is little loss due to denitrification. The change from the pre- to the post-restoration period in the predominant form of N from NO_3 to DON and PN suggests that the restoration promoted the uptake of inorganic N by primary producers, thus limiting nitrification and subsequent denitrification. This effect is also evidenced by the strong seasonality of DON concentrations in baseflow (Fig. 6a). Alternatively, supposing that no dilution of upstream NO_3 from the reservoir occurred, then up to a third of the decrease in NO_3 from the pre- to the post-restoration period could be attributed to denitrification.

Similar to DON, seasonal variability is apparent in post-construction TDP and TSS concentrations (Fig. 6b, c) which is in part due to nutrient uptake by primary producers and the transformation of inorganic to organic forms. Moreover, hypoxic/anoxic conditions in the step pools during quiescent periods

Table 4 Urban BMPs and nutrient retention efficiencies used in the CBP watershed model. Note that while urban stream restorations are not on this list of BMPs, they are included in the watershed model. Both physiographic provinces have a reduction of 0.075 and 0.068 lb ft^{-1} for TN and TP, respectively, whereas TSS has a reduction of 15.13 lb ft^{-1} in the Coastal Plain and 44.88 lb ft^{-1} in the Piedmont. All BMP efficiencies were increased by 30% in the end-of-the-twenty-first century climate projections and were capped at 0.999 when the 30% increase exceeded 1.0

Urban BMPs in watershed model	BMP short name	Existing BMP efficiency			30% increase in BMP efficiency		
		TN	TP	TSS	TN	TP	TSS
Bioretention—C/D soils not underdrain	BioRetNoUDAB	0.8	0.85	0.9	0.999	0.999	0.999
Bioretention—A/B soils underdrain	BioRetUDAB	0.7	0.72	0.8	0.91	0.975	0.999
Bioretention—C/D soils underdrain	BioRetUDCD	0.25	0.45	0.55	0.325	0.585	0.715
Bioswale	BioSwale	0.7	0.75	0.8	0.91	0.975	0.999
Dry detention ponds and hydrodynamic structures	DryPonds	0.05	0.1	0.1	0.065	0.13	0.13
Dry extended detention ponds	ExtDryPonds	0.2	0.2	0.6	0.26	0.26	0.78
Urban filtering practices	Filter	0.4	0.6	0.8	0.52	0.78	0.999
Urban infiltration practices—no sand/vegetation, not underdrain	Infilration	0.8	0.85	0.95	0.999	0.999	0.999
Urban infiltration practices—with sand/vegetation, not underdrain	InfiltWithSV	0.85	0.85	0.95	0.999	0.999	0.999
Permeable pavement without sand/vegetation—A/B soils not underdrain	PermPavNoSVNoUDAB	0.75	0.8	0.85	0.975	0.999	0.999
Permeable pavement without sand/vegetation—A/B soils not underdrain	PermPavNoSVUDAB	0.45	0.5	0.7	0.585	0.65	0.91
Permeable pavement without sand/vegetation—C/D soils underdrain	PermPavNoSVUDCD	0.1	0.2	0.55	0.13	0.26	0.715
Permeable pavement with sand/vegetation—A/B soils not underdrain	PermPavSVNoUDAB	0.8	0.8	0.85	0.999	0.999	0.999
Permeable pavement with sand/vegetation—A/B soils underdrain	PermPavSVNoUDAB	0.5	0.5	0.7	0.65	0.65	0.91
Permeable pavement with sand/vegetation—C/D soils underdrain	PermPavSVUDCD	0.2	0.2	0.55	0.26	0.26	0.715
Retrofit stormwater management	RetroSWM	0.25	0.35	0.65	0.325	0.455	0.845
Stormwater management by eras 2002 to 2010 MD	SWMEra0210	0.3	0.4	0.8	0.39	0.52	0.999
Stormwater management by eras 1985 to 2002 MD	SWMEra8502	0.17	0.3	0.4	0.221	0.39	0.52
Vegetated open channels—A/B soils not underdrain	VegOpChanNoUDAB	0.45	0.45	0.7	0.585	0.585	0.91
Vegetated open channels—C/D soils not underdrain	VegOpChanNoUDCD	0.1	0.1	0.5	0.13	0.13	0.65
Wet ponds and wetlands	WetPondWetland	0.2	0.45	0.6	0.26	0.585	0.78

(see below) may cause PO_4 to desorb from sediments thereby increasing P availability to primary producers. Such a PO_4-desorption effect could partially explain why concentrations of TDP and TSS in baseflow of the SWC increased after construction.

The increase in particulates in baseflow (e.g., primary production in ponds; see below) and the modest decrease in stormflow from the pre- to the post-construction periods are in part responsible for the unexpectedly modest decrease in TSS concentrations resulting from the restoration. Although this could alternatively be due to an increase in PC in the post-construction environment, our data indicate that PC is only about 10% of TSS in both periods of the study. Otherwise, the modest decrease in TSS in stormflow is likely due to recurring disturbances in the headwater of the RSC1 catchment that contributed to higher TSS concentrations throughout the study, as well as highly erosive hillslopes of the RSC structure (i.e., soil stabilization fabric was not used). The disturbances in the RSC1 catchment were primarily from a parking lot that was used to stockpile dirt, rock, and sand used for construction that were constant sources of silt in stormflow runoff of RSC1 during the post-construction period.

Higher GPP in the SWC is also indicative of disturbed stream environments (Bunn et al. 1999) where higher rates of primary production are commonly observed in streams polluted with organic wastes and nutrients. Rates of GPP in the SWC were higher than those of the other sites used in our comparison (Fig. 7), likely a consequence of riparian forest removal and increased light availability and nutrients in the ponding basins. Increases in primary production can also produce hypoxia in quiescent waters as the organic matter is mineralized and biological oxygen demand (BOD) increases, as observed in SWC shortly after the deployment of DO probes following storm events (data not shown). Higher summer water temperatures (Fig. 9) are likely contributing to episodic hypoxia in the ponding basins, and structural modifications have affected metabolic parameters, as evidenced by higher GPP in Cypress Creek compared to Dividing Creek and re-engineered and natural stream wetland complexes (i.e., Howard's Branch and Parker's Creek, respectively) in the Coastal Plain. Higher streamwater temperatures, rates of primary production, and incidents of hypoxia are undesirable consequences of this type of stream restoration design but may decrease in severity as the system matures and the riparian forest provides more shading.

The Relative Performance of BMPs and Stream Restoration in Urban Coastal Plain Watersheds of Chesapeake Bay

Tributaries of the Coastal Plain physiographic province comprise a large portion of the drainage network in the state of Maryland and are particularly vulnerable to erosion due to the absence of structural control from bedrock combined with a surficial lithology dominated by highly erodible materials (Reger and Cleaves 2008). Moreover, lowland Coastal Plain channels are the last conduit for the transport of nutrient and sediment to the Chesapeake Bay and, consequently, as with Cypress Creek, have been particularly targeted for the implementation of novel stream restoration techniques over the last decade (Filoso et al. 2015).

Not surprisingly, there are high expectations with regard to the potential nutrient reduction capabilities of BMP implementation and stream restoration in the Chesapeake Bay watershed. We determined the nutrient and sediment reductions in relation to TMDL goals to see how well the implementation performance met expectations. The RSCs combined with the SWC resulted in quantifiable decreases in pollutant loads to the Cypress Creek mainstem and, ultimately, the Magothy River estuary, yet both fell short of expected goals. For example, RSC1 decreased TN, TP, and TSS by 90, 4, and 40% of the annual projected decreases, whereas RSC2 only decreased by 12, 1, and 2%, respectively. Poor performance on the part of TP and TSS was, as mentioned previously, in large part due to sediment and organic matter inputs from steep hillslopes of the RSC structures that were unstable throughout the post-construction period.

For stream restorations, real versus anticipated annual reductions (i.e., with completed TMDL implementation) of these pollutants can be determined by either watershed area or the length of river reach restored (AA Co. 2012). One important caveat to consider regarding the estimates for the SWC is that the reductions realized thus far are partially attributed to the RSCs rather than the SWC itself. Thus, a similar SWC implemented without the similarly effective headwater BMPs would show lower load reductions. For example, the total BMP stormflow load reductions for RSC1 and RSC2 are 43, 13, and 523 kg for TN, TP, and TSS, respectively, which are about 15, 11, and 4% of the total reductions estimated at the SWC, respectively. Converting the SWC estimates (not including BMP reductions) into pollutant reductions per length of reconfigured stream reach (length = 1.5 km), values range from 143 to 162 kg, 6 to 9 kg, and 8842 to 9979 kg TN, TP, and TSS km^{-1} year^{-1}, respectively. Compared to the reduction goals for severely degraded streams (A.A. Co. 2012), these numbers represent a maximum of about 88, 23, and 3% of the TMDL goals for TN, TP, and TSS, respectively. Determined by watershed area, % attainments of TMDL goals were much lower for each constituent. In terms of cost-effectiveness, RSC1 cost \$418,000 and reduced TN export by 41 kg year^{-1} (14 kg N/\$100,000). The SWC cost \$1.7 million and decreased TN export by 288 kg year^{-1} (17 kg/\$100,000).

As with any restoration project of this size, there are positive and negative outcomes and trade-offs that need to be evaluated in order to select and implement stream restoration designs that perform best on a larger scale. Positive outcomes of the Cypress Creek SWC are a decrease in TN export due to a combination of uptake and loss and an increase in wetland habitat. Negative outcomes include a decrease in forest habitat, an increase in streamwater temperature, an increase in PO_4 (likely from episodic hypoxia/anoxia), and much smaller reductions in TSS than expected. The latter is likely from a combination of residual disturbance effects of the construction, an increase in the incidence of iron (Fe) flocculate (Williams et al. 2016) and an increase in primary productivity as evidenced by higher DON and DOP concentrations in the post- compared to the pre-construction period and higher GPP compared to other constructed and reference wetland settings (Fig. 7). Despite its lackluster performance reducing TP and TSS, we emphasize that the sites monitored in this study may show improved retention of TP and TSS as they stabilize and mature (viz., with vegetation that stabilizes surface soils and reduces erosion), and the unit cost of pollutant reductions will quickly decrease considering the cumulative reductions that are bound to occur over the course of many years. Alternatively, the pools will likely fill with sediment and clog with fine particles over time, thus compromising TSS retention. Moreover, although the mineralization of decomposing vegetation (e.g., root wads) and organic matter added to the structures (e.g., surface layer of wood chips and topsoil on RSCs) that initially boosts DOC concentrations may be responsible for enhanced NO_3 losses via denitrification, it is likely that NO_3 losses will diminish in this system over several years as labile DOC decreases. Such nutrient dynamics still need to be properly characterized for these systems thus emphasizing the need for monitoring well beyond the first 2 years after restoration. Regardless, there is reason to be concerned about the lackluster performance of this SWC and others (Filoso et al. 2015), particularly in relation to whether these structures should be implemented as a means of attaining TMDL goals given their cost and limited contribution toward attaining these goals for some constituents.

Projecting Changes in Stormflow Runoff Due to Climate Change

While the upland RSCs helped reduce pollutant loads in the main channel of Cypress Creek, the limited restoration performance of the SWC implemented is a concern not only in terms of its contribution toward attaining TMDL goals but also in terms of what this portends with changes in runoff that may occur with climate change this century. Although a decrease in runoff is possible in a wetter climate due to warmer temperatures and higher rates of PET (particularly in more agricultural and forested catchments), this decrease will more than likely be restricted to baseflow, not stormflow, in many urban catchments. For example, the larger proportion of impervious surfaces in urban areas compared to other general land uses will likely increase stormflow runoff in these catchments. Thus, increased precipitation will exacerbate nutrient and sediment loads to and from urban streams, thereby contributing to the degradation of aquatic environments.

Many recognize that climate change will present challenges to the restoration and recovery of Chesapeake Bay. However, that we are aware, there is only one other study that attempts to address how much BMP implementation can be expected to mitigate the increase in pollutant loads associated with climate change, albeit projected to mid-century (Fischbach et al. 2015), that will likely curtail gains made toward achieving TMDL goals. The questions we asked were (1) how much will projected increases in surface runoff from urban areas increase pollutant loads and (2) how much additional BMP implementation will be needed to offset the increases above TMDL goals?

Scenarios for the mid-twenty-first century using the NLDAS-II forcing functions in the CBP watershed model give one example of the potential impacts of climate change. Projected increases in precipitation and stormflow runoff for the state of Maryland are 12 and 26%, respectively (Fig. 1 Supplementary Material). The increased runoff projected with this modeling exercise roughly agrees with the projected increase in precipitation by the GCMs for Maryland (16 and 15%, respectively), albeit total runoff is considerably lower in the GCM scenarios. Although the magnitude of the change in precipitation varies, results of this analysis generally corroborate with other recent studies using GCMs (Fischbach et al. 2015) and higher-resolution NARCCAP models (Kunkel et al. 2013) that project, on average, increases in precipitation intensity and annual precipitation in many areas of the Chesapeake Bay watershed. Thus, an increase in surface runoff can be expected to occur with a larger frequency of larger-sized storm events in more urban catchments that typically have higher runoff coefficients (i.e., the ratio of runoff to precipitation) than in less-developed watersheds. Scenarios from our GCM analysis evaluated for the different watersheds in MD support the idea that climate effects will be amplified in urban areas and exacerbated by urban sprawl that is projected to continue through this century in the Chesapeake Bay watershed (www.mwcog.org/uploads/committee-documents/aV5fVlxX20080430092814.pdf; accessed September 2016), because runoff increases in watersheds with proportionally more impervious surface (Table 3).

More importantly, our analysis indicates that the feasibility of implementing BMPs on a scale that will counter the increase in stormflow runoff with climate change will be daunting. For example, the ensemble of GCMs projects that 14 to 26, 14 to 18, and 16 to 33% of the gain in TN, TP, and TSS loads can be mitigated with a 30% increase in BMP efficiency in Baltimore County, a watershed that is about 80% urban (Table 3). The lower end of the range is representative of BMP effectiveness with higher runoff projected in scenarios that include the increase in stomatal resistance in response to higher CO_2 concentrations, which reduces transpiration, thereby increasing runoff. To put the magnitude of necessary mitigation into context, the decreases in TN loading from the suite of RSCs and SWC in our study would have to be implemented at two to three times its current scale to keep loads at TMDL goal levels, a prospect that is impractical given the highly urban configuration of the catchment.

It is not surprising that with increased precipitation resulting from climate change, much more BMP implementation will be needed to keep nutrient and sediment reductions potentially achieved with TMDL implementation at the levels anticipated. However, our analysis provides estimates of potential changes so that the water resource management community can proactively and more effectively mitigate climate impacts. The model scenarios evaluated herein generally agree that precipitation will increase by about 5 to 10% in many parts of the Chesapeake Bay watershed. And although previous climate change projections for this region have indicated that the runoff response could either positive or negative (Najjar et al. 2009), we argue that the response will generally be positive in urban areas due to extensive impervious surfaces. Thus, the net increase in runoff from larger rainfall events projected to occur with changing climate will generally exceed the net decrease in runoff from higher temperatures and resultant PET. Our estimates indicate that the increase in stormflow runoff from urban areas will range from 10 to 20%.

Conclusions and Recommendations

Our study indicates that efforts to achieve TMDL goals have resulted in large-scale stream restorations in sensitive mainstem stream reaches at the estuarine interface and provide some but not all of the expected benefits, particularly with regard to water quality. Although implemented prior to TMDL mandates were in effect, other studies evaluating N and sediment reduction capabilities of SWCs indicate that the performance of the restoration reach itself can be limited (Filoso and Palmer 2011; Filoso et al. 2015). Moreover, while the benefits of stream restoration designs such as SWCs are often touted, such as the creation of wetland habitat, potential disadvantages also need to be considered. For example, such restorations are sometimes built at the expense of existing forested habitat and riparian corridors that are potentially highly effective at reducing N via denitrification in riparian zones (Peterjohn and Correll 1984).

By contrast, this study and others (e.g., Williams et al. 2016) indicate that RSCs implemented in degraded headwater catchments hold great promise as a means by which nutrient and sediment loads from degraded streams and drainage ditches in urban catchments can be curtailed, particularly if more is done to limit residual disturbance effects that exacerbate sediment runoff. Accordingly, more stringent guidelines should be rigorously adopted so that construction is done in a manner that prevents excessive sediment runoff during the construction and post-construction periods. These guidelines should include the mandatory use of stabilization fabric (e.g., coir) on the hillslopes of BMP ponding basins, an erosion control technique that was not employed in the RSCs monitored in this study and which was directly responsible for excessive sediment and wood chip export throughout the construction and post-construction periods, thereby compromising performance. Based on these findings of this study, we also recommend that a *top-down* approach to restoration be adopted in order to achieve and maintain nutrient and sediments loads at the levels expected with full TMDL implementation. For example, headwater stream restorations should be given preference to stream restoration and BMPs planned for downstream reaches (i.e., upstream restorations should be done first) not only because of the documented effectiveness of headwater streams (Peterson et al. 2001; Mulholland et al. 2008) but also to ensure that the retention capacity and infiltration rates of downstream BMPs are not compromised by excessive siltation from upstream construction.

While stream restoration is an important part of the overall portfolio of nutrient and sediment reduction strategies (i.e., including many types of urban BMPs) that need to be implemented to achieve TMDL goals, we are in uncharted territory with regard to the implementation of very large and expensive stream restorations that have unknown benefits, particularly at the estuarine interface. By contrast, there is a growing base of literature that supports the contention that stream restoration in headwaters is very effective at reducing nutrient and sediment loads (Williams et al. 2016; Williams and Filoso 2016). Unfortunately, the advent of the Chesapeake Bay TMDL has pressured the regulatory community to extend permits for novel techniques of stream restoration implementation without having a solid understanding of their performance and sustainability. Until we understand more about the effectiveness of novel stream restoration techniques, the emphasis should be on implementing headwater stream restorations and urban BMPs to meet TMDL goals. We believe that efforts to achieve TMDL goals should not be undercut by implementing large-scale stream restorations such as SWCs until we have a better understanding of the real advantages and potential disadvantages (i.e., tradeoffs) to such structures. Thus, more research determining stream restoration and BMP effectiveness and sustainability is urgently needed.

Regarding climate change, projections indicate that a higher frequency of larger-sized storm events will result in a 10 to 20% increase in runoff from developed catchments in the Chesapeake Bay watershed this century. One likely possibility of how total runoff will increase is that higher air and water temperatures and rates of PET with climate change will decrease baseflow but produce a proportionally larger amount of stormflow runoff from urban catchments due to the increased frequency of larger-sized storms, resulting in a net increase in total runoff. Thus, an adaptive management strategy that evaluates unavoidable urban sprawl in tandem with increased runoff expected under wetter climatic conditions is necessary. At this time, the relative agreement of model projections indicate that a sound strategy should emphasize more extensive implementation and retrofitting of upland stormwater BMPs with larger storage capacities and infiltration rates (Pyke et al. 2011) to help mitigate the expected increase in surface runoff from urban areas. Increased infiltration will concomitantly increase groundwater storage that can provide a more constant source of baseflow runoff during drier periods likely to occur with increased air temperatures and PET.

It is clear that predicting the effects of climate change is difficult and complex. Thus, refined projections of potential changes in precipitation and stormwater runoff that will occur with climate change will be needed using a large ensemble of climate models to periodically re-evaluate the cost of infrastructure required to maintain pollutant loads at their TMDL levels (Fischbach et al. 2015). Proactive mitigation used to manage increasing pollutant fluxes from urban areas should be implemented, recognizing that the uncertainties of current watershed and climate models are relatively high. In particular, projected changes in the magnitude of pollutant loads will certainly change as models are refined (e.g., the subsequent version of the CBP watershed model, phase 6, will adjust TMDL goals, especially for TP). Although potential errors in current climate forecasts may be used by some as a justification for postponing investments in new or improved infrastructure, we argue that given the current consistency among climate projections indicating that there will be an increase in the frequency of larger-sized storms and total annual precipitation associated with climate change in many areas of the Chesapeake Bay watershed, expanding green infrastructure and enhancing the storage capacities and infiltration rates of upland BMPs henceforth implemented to accommodate a 10 to 20% increase in stormflow runoff from urban areas are warranted. This recommendation is particularly relevant, given the large-scale stream restoration and BMP implementation efforts now being funded in order to support the Chesapeake Bay TMDL. Such enhancements will not be prohibitively expensive and can serve the dual purpose of not only helping to achieve TMDL goals for Chesapeake Bay more expediently but also bolstering resilience to climate change effects in urban areas expected this century.

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