2019

Characterizing water quality and hydrologic properties of urban streams in central Virginia

Rikki Lucas
Virginia Commonwealth University

CHARACTERIZING WATER QUALITY AND HYDROLOGIC PROPERTIES OF URBAN STREAMS IN CENTRAL VIRGINIA

A thesis submitted in partial fulfillment of the requirements for the Degree of Master of Science in Biology at Virginia Commonwealth University

By
Rikki L. Lucas
BS in Environmental Science, University of Mary Washington, 2013

Advisor: Dr. Paul A. Bukaveckas
Professor, VCU Department of Biology and Center for Environmental Studies

Virginia Commonwealth University
Richmond, Virginia
January 2019

ACKNOWLEDGEMENTS

This research was supported in part through funding from the City of Richmond, Department of Public Utilities. Field studies were performed with the invaluable help and perseverance of Xin-Chen “Liu”, Elizabeth Keily, Spencer Tassone, and Edward Carneal. Thank you also for your limitless patience in the face of disaster—equipment failures, endless rain, lost supplies, disappearing streams, and everything in between. I thank William “Mac” Lee for his help and unflappable demeanor when handling all of my equipment woes. Thank you to Dr. Paul Bukaveckas for his patience and expertise throughout this project.

I would like to extend my overwhelming gratitude to Douglas Moyer and Jennifer Rapp of the U.S. Geological Survey: Virginia-West Virginia Water Resources office for their endless advice, guidance, training, and resources. To all my loved ones—your patience knows no bounds. Thank you for your endless support.

TABLE OF CONTENTS

ABSTRACT

CHARACTERIZING WATER QUALITY AND HYDROLOGIC PROPERTIES OF URBAN STREAMS IN CENTRAL VIRGINIA

By: Rikki L. Lucas, M.S.

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Biology at Virginia Commonwealth University.

Virginia Commonwealth University, 2019.

Advisor: Dr. Paul A. Bukaveckas, Professor, VCU Department of Biology and Center for Environmental Studies

The objective of this study was to characterize water quality and hydrologic properties of urban streams in the Richmond metropolitan area. Water quality data were analyzed for six urban sites and two non-urban sites. Geomorphological surveys and conservative tracer studies were performed at four urban sites and one non-urban site to describe intra- and inter- site variability in transient storage, channel geomorphology, and related hydrologic parameters. Urban sites showed elevated concentrations of nitrogen and more variable TSS concentrations relative to reference sites. Urban channels were deeply incised with unstable banks and low sinuosity. Little Westham Creek exhibited the greatest transient storage. This site was characterized by large, deep pools and therefore it is likely that transient storage was associated with surface water storage. Transient storage was low at all other sites, particularly for the study reach at Reedy Creek, which flowed through a concrete channel. Lowest transient storage was observed at this site in spring, though higher values were measured in summer corresponding to the presence of biofilms, A lower, more naturalized section of the concrete channel was found to have greater transient storage suggesting the possibility of passive restoration of concrete channels in urban environments. This study documents variability in the structure and function of urban streams. Restoration projects should work to improve impairments that are specific to each site at both the reach and watershed scale to maximize the efficacy of restoration.

INTRODUCTION

As urban centers grow, so does the effect of urbanization on the surrounding landscape, including freshwater ecosystems. Streams in urban catchments are particularly vulnerable to impacts associated with landcover change (Walsh et al. 2005). The physical, biological, and biogeochemical impairments that occur in urban streams are collectively referred to as the "urban stream syndrome" (Meyer at al. 2005, Walsh et al. 2005). Symptoms of the urban stream syndrome include altered baseflow and unstable hydrology, high levels of contaminants and nutrients, reduced nutrient retention (Mueller Price et al. 2015), channel incision, reduced channel complexity, and altered vegetative and benthic communities (Violin et al. 2011).

Impervious surfaces increase runoff during precipitation events, creating highly variable (i.e., "flashy") and occasionally high discharge conditions in urban streams (Walsh et al. 2005). This in turn erodes and simplifies stream channels, homogenizes stream bed sediments, and reduces flow variation within the channel (Sudduth et al. 2011). As a result, important hydraulic attributes, such as transit time and transient storage, are altered. Transit time, the time taken for a water molecule to pass through a length of channel (Soulsby et al. 2011), is tied to storage and exchange processes (e.g., presence of pools and riffles, debris dams, etc.). Transient storage is the temporary delay in transport within surface zones (such as side pools, eddies, debris dams, and dense vegetation stands) and the subsurface hyporheic zone (Runkel 1998). Accumulated organic matter and benthic algae can also act as storage zones (Jin et al. 2005). The rate of exchange of solutes between storage zones and the main channel determine the opportunities for biogeochemical processes such as denitrification (Kaushal et al. 2008), as well as a stream's ability to maintain and support ecosystem functions such as nutrient retention (Sudduth et al. 2011). These characteristics are mainly affected by the physical properties of the channel, such as stream discharge, water velocity (Jin et al.2005), morphology, sediment grain size and hydraulic conductivity, and channel heterogeneity, all of which are negatively influenced by urbanization (Ryan et al. 2010). Increasing discharge results in greater water velocity and diminished transient storage, thereby minimizing the influence of stream biogeochemical processes (Harvey and Wagner 2000, Jin et al. 2005). The degradation of hydrologic and hydraulic attributes of urban streams in turn degrades the geomorphology (Harman et al. 2012), often reducing the stream's ability to retain and remove nutrients and other contaminants (Mulholland et al. 1994) and the stream's ability to support diverse biotic communities (Walsh et al. 2005).

Urbanized areas are associated with greater runoff and diminished in-stream capacity for storage and retention. The combined effects increase the mass transport of nutrients, sediments, and contaminants to downstream waterways. Because of this, urban streams are often targeted for restoration. There is considerable interest in stream restoration as a means to mitigate nutrient transport, especially in the Chesapeake Bay region where stream restoration is included in state watershed implementation plans for achieving mandated nutrient load reductions (Kaushal et al. 2008). Typical stream restoration projects involve rebuilding the stream channel to mimic natural systems by installing meanders and obstructions to create diverse flow conditions and by lowering stream bank height to connect the stream to its floodplain (Bukaveckas 2007, O'Connor et al. 2010). Stream restoration activities that increase connectivity between surface water, groundwater, and floodplains are intended to improve ecosystem functions, such as nutrient storage and removal (Mueller Price et al. 2016) by increasing contact time with biogeochemically active areas (Craig et al. 2008). However, several studies have noted a deficit of evidence to support the efficacy of such projects (Bernhardt et al. 2005, Kaushal et al.

2008, O’Connor et al. 2010, Violin et al. 2011, and others). Further studies are needed to characterize hydrologic functioning in urban streams to better inform decision-making regarding their potential for successful restoration.

The objective of this study is to characterize the hydrologic properties of urban streams in the Richmond metropolitan area. An assessment of differences in physical structure and hydrologic functioning provides a basis for assessing impairment in urban streams, which may be useful for setting restoration targets. The specific goal was to assess variation in hydrologic functioning among urban streams, and over time, in response to changing discharge conditions. By performing multiple tracer injection experiments at each site, and at multiple sites, I was able to characterize intra- and inter- site variability in transient storage and related hydrologic parameters. In addition, data are presented to characterize water quality conditions in these streams to better understand the range of water quality conditions among urban streams.

METHODS

Study Sites

Seven urban streams were studied in the Richmond metropolitan area: Broad Rock Creek, Gillies Creek, Pocosham Creek, Rattlesnake Creek, Reedy Creek, Little Westham, and Upham Brook (Figure 1). Broad Rock, Gillies, Pocosham, Rattlesnake, Reedy, and Little Westham are tributaries of the James River, and Upham Brook is a tributary of the Chickahominy River. These are considered to be “urban streams” based on their location within a major metropolitan city and the significant human modification to their channels (Figure 2). Rattlesnake and Broad Rock have channelized study reaches, and the entirety of the study reach at Reedy Creek is lined with concrete. Little Westham and Upham Brook have impoundments located upstream of the study reach. Developed areas constitute more than 60% of land cover in the seven watersheds (Table 1).

Reference sites were selected based on the availability of prior water quality monitoring, their comparability to urban sites, the presence of less than 10% urban cover in the watershed, and channel morphology that is amenable to injection studies. Reference sites chosen for water quality comparisons were Kimages Creek (Charles City county) and Fine Creek (Powhatan county). The reference site for hydraulic evaluation (injection experiments) was Stagg Creek (Hanover county).

Water Quality

The 7 urban sites are monitored by Virginia Commonwealth University every two weeks for water quality, nutrients, *E. coli*, and discharge as part of a study conducted for the City of Richmond (Table 2). Little Westham was excluded from water quality analysis, due to the lack of long-term monitoring data. Water quality data are collected using a YSI model 6600 Water Quality sonde. Nitrogen, phosphorus, and chloride concentrations are analyzed using a Skalar San Plus System analyzer. *E. coli* analysis was performed via a membrane filtration method using modified mTEC agar, where the number of colonies is counted following a 24-hour incubation period (Environmental Protection Agency 2009). Monitoring at Kimages Creek is performed every two weeks as part of the long-term monitoring program at the VCU Rice Center (no discharge data available for this site). Monitoring at the Fine Creek reference site is performed once per month by Randolph Macon College using methods similar to those for the VCU sites.

Channel Morphology

Surveys were conducted at four urban sites—Broad Rock, Little Westham, Rattlesnake, and Reedy— to characterize channel morphology, grain size distribution, and habitat features. The upstream boundary was chosen by locating an area of constricted flow (for tracer injections) that maximized the downstream extent of the channel uninterrupted by tributary inputs. The longitudinal boundaries of habitat features (riffles, runs, pools) were identified and tabulated. Water surface elevation was surveyed using a survey level and rod at the midpoint of each habitat feature (Vermont Water Quality Division, 2009). Discharge was measured in conjunction with the channel surveys, and during solute injection experiments.

Using the feature boundaries marked during the longitudinal survey, cross-sectional profiles were measured according to U.S. Geological Survey methods (1998). These data were collected on the same day as the longitudinal surveys to minimize the effects of differing water surface elevation. At the midpoint of each channel feature, in-stream vegetation and habitat cover were noted. Bank height, angle, vegetative cover, and substrate were noted and given a score to characterize bank stability (bank stability index). Effective bankfull width and height, and height to the top of each bank (potential bankfull capacity) from the water surface were measured at each riffle to calculate a bank height ratio (Harman et al. 2012). Wetted width was recorded, and depth was measured at 25%, 50%, and 75% width distance across the stream (USGS 1998). The thalweg depth and location relative to the left bank were recorded. Where applicable, the maximum pool depth and its location relative to the left bank and the cross-section were measured. Pebble count surveys (Wolman 1954) were performed at each cross-section such that the total number of samples counted along the reach equaled 100. Canopy closure was measured with a spherical densiometer at each transect, and calculated as a percent for the entire reach. Channel complexity was determined by counting the number of transitions between different channel features (pool, riffle, run) for each reach, then normalized by converting the number of transitions to number of transitions per 100 m (Violin et al. 2011). Transit time, volume, and velocity for each feature was calculated based on cross-section measurements, and summarized by feature type (pool, riffle, run). Volume, surface area, and transit time were also derived for the entire reach. Gradient was calculated as the change in elevation from the top of the reach to the bottom divided by the reach length. To estimate channel incision, the bank height ratio (BHR) was calculated using cross-section data from riffles (Harman et al. 2012). The ratio is calculated as the depth from the top of the lowest bank to the thalweg divided by the depth from effective bankfull elevation to the thalweg. A stream is considered to be significantly incised above BHRs of 1.5. To characterize overall bank conditions, bank angle, height, dominant substrate, and vegetative cover were classified into value ranges and scored according to USGS (1998) protocol. These scores were then tallied out of a possible value of 22. Banks with scores of 4 to 7 are considered stable, scores of 8 to 10 are at risk, scores of 11 to 15 are unstable, and scores of 16 to 22 are very unstable.

Hydrology

Conservative tracer injections were performed at four urban sites (Broad Rock, Little Westham, Rattlesnake, and Reedy) and one reference site (Stagg) to determine characteristics of solute transport (Stream Solute Workshop 1990). A conservative tracer is a substance that does not react biotically or abiotically with the environment (Stream Solute Workshop 1990). Reach lengths varied among sites but ranged between 86 and 230 m. A concentrated solution (~125 g L^{-1}) of NaCl, a conservative tracer, was pumped into the stream at a discharge-dependent rate such that background specific conductance (SpC) was raised to at least 10 $\mu S\ cm^{-1}$ above background value. The upstream boundary was set so that the tracer was pumped into a constricted area of flow to encourage uniform distribution of the tracer vertically and laterally within the stream channel. The salt solution was injected over a 20 min period, and conductivity was measured at upstream and downstream locations to characterize the passage of the pulse. The pulse was measured every thirty seconds at a point below the injection where the water was sufficiently mixed (10-50 m downstream) and at the downstream end of the reach using Onset HOBO Fresh Water Conductivity Data Loggers. Data were recorded until all tracer had moved beyond the study reach, i.e., background SpC conditions returned. The rate of lateral inflow or outflow was calculated as the difference between upstream and downstream discharge divided by the reach length. Pre-injection background measurements of SpC were subtracted from readings during and after the injection to obtain the increase in SpC due solely to the injection (Mulholland et al. 1994). The background-corrected downstream conductivity values were used to define the breakthrough curve (BTC), which was further analyzed to parameterize hydrologic processes.

For the injection performed on 4/18/2018 at Reedy Creek, two BTCs were obtained: one spanning a 168 m length of the concrete channel (Figure 3a), and a second which extended a further 62 m downstream (i.e., 168 to 230 m; Figure 3b). The lower section of the concrete channel is more "naturalized" due to the accumulation of substrates (sediment and pebble) that have allowed plants to grow within the channel. There is also a large area of backwater at the transition from concrete to open channel (230 to 250 m). To determine whether the naturalized portion of the concrete channel differed from conditions upstream, separate BTCs were analyzed.

Data Analysis

Conservative solute analysis

A hydrologic model of one-dimensional transport with inflow and storage was used to determine the properties of solute transport during the injection (OTIS; Runkel 1998). The model accounts for five physical hydrologic processes: bulk transport (advection), dispersion (spreading out), lateral inflow, transient storage (temporary solute storage), and storage zone exchange (rate of exchange between storage zones and channel flow) (Stream Solute Workshop 1990). Transient storage processes are represented by the area of the storage zones and the storage zone exchange coefficient.

The OTIS model solves for the following coupled differential equations for the main channel (1) and storage zone (2):

1) $$\frac{\partial C}{\partial t} = \frac{Q \partial C}{A \partial x} + \frac{1 \partial}{A \partial x}\left(AD\frac{\partial C}{\partial x}\right) + \frac{q_{LIN}}{A}(C_L - C) + \alpha(C_S - C)$$

2) $$\frac{dC_s}{dt} = \alpha \frac{A}{A_S}(C - C_S)$$

where A is the main channel cross-sectional area (m^2), A_S is the storage zone cross-sectional area (m^2), C is the concentration of the solute in the main channel (mg L^{-1}), C_L is the concentration of the solute in late flow (mg L^{-1}), C_S is the concertation of the solute in the storage zone (mg L^{-1}), D is the dispersion coefficient (m^2 sec^{-1}), Q is the volumetric flow rate (m^3 sec^{-1}), q_{LIN} is the lateral flow rate (m^3 sec^{-1} m^{-1}), t is time (s), x is distance (m), and α is the storage zone exchange coefficient (s^{-1}). Discharge was assumed to be steady throughout the injection experiment. Using the measured discharge and the BTC, four parameters (A, A_S, D, and α) were iteratively adjusted to obtain a line of reasonable fit through observed data for each injection (Stream Solute Workshop 1990). These predicted values were then optimized using a modified version of OTIS, OTIS-P, that uses Nonlinear Least Squares algorithms to fit the predicted BTC to the observed data. OTIS-P was run at least three times to determine the optimal set of parameter values. Parameters were considered optimized if: 1) predicted values did not change between model runs, 2) if the coefficient of variation of the estimated parameter was less than fifty percent, and 3) the output of the model using the optimized parameters fit the observed values visually (used as a check for model error). The experimental Damköhler (*Dal*) number was used to assess the suitability of the reach length for analysis, and the reliability of the estimated transient storage parameters (Bukaveckas 2007, Rana et al. 2017):

$$Dal = \frac{\alpha\left(1 + \frac{A}{A_S}\right)L}{V}$$

where *L* (m) is the length of the study reach and *V* is the average stream velocity calculated from the BTC. *Dal* is considered acceptable if between 1.0 and 10 (Wagner and Harvey 1997). A comparison of *Dal* values among reaches will also allow determination of whether the reaches are appropriate for comparison (Gooseff et al. 2013).

Several parameters that describe hydrologic characteristics of a reach are calculable from the values A, A_S, D, and α. The ratio of storage zone area to channel area, A_S/A, normalizes the relative size or extent of storage zones in the reach for comparison among sites. Using the BTC, the median travel time (MTT) and average solute velocity were calculated. The fraction of median transit time due to transient storage standardized to a reach length of 200 m (F_{med}^{200}) was calculated to describe the relative influence of transient storage zones on flow (Runkel 2002).

Statistical analysis

Statistics were performed using R software version 1.1.453. *E. coli* data were log-transformed to normalize the distribution before analysis. Two-way analysis of variance (ANOVA) was performed using site, month, and their interaction terms to explain variation in water quality (nutrients, TSS, and *E. coli)*. One-way ANOVAs and t-tests were performed to detect statistically significant differences in hydraulic and geomorphologic characteristics among sites. Significance was considered when $p < 0.05$. For each stream, univariate linear regressions were used to test for relationships between A_S/A and F_{med}^{200} with discharge. Significance for these were considered when $p < 0.1$.

RESULTS

Water quality

Significant differences in water quality were observed among the streams included in this study. A two-way ANOVA showed that site, month, and their interaction term accounted for 72% of the variation in nitrate (Table 3). Median nitrate concentrations were higher among the 6 urban sites relative to the non-urban reference streams (Figure 4). Mean nitrate concentrations were greatest at Upham Brook (urban; mean = 1.30 ± 0.07 mg L^{-1}.) and lowest at Fine Creek (reference; mean = 0.138 ± 0.10 mg L^{-1}). Upham Brook also had the greatest average concentration of total nitrogen (1.75 ± 0.08 mg L^{-1}) and ammonia (0.156 ± 0.012 mg L^{-1}). Total nitrogen and nitrate concentrations were typically highest in May and June and lowest in August; ammonia concentrations were highest in January and lowest in October. TSS concentrations were generally similar across all sites (range of medians = 1.6 to 11.6 mg L^{-1}) with the exception of Pocosham Creek. This site was undergoing restoration during the period of sampling and exhibited a large increase in TSS from pre-construction (August to December 2017, mean = 7.2 ± 5.2 mg L^{-1}) to mid-construction (January to March 2018, mean = 122.1 ± 52.4 mg L^{-1}). Urban sites generally showed greater variability in TSS, compared to the reference sites and to each other, with site, month, and their interaction term accounting for 58% of the variation (Table 3).

TSS was generally greatest from February to May. TP and orthophosphate concentrations were similar across all streams, with the exception of Fine Creek, which showed significantly lower OP ($p < 0.0001$). *E. coli* concentrations varied by more than an order of magnitude at each site (range = 0 to 13,000 CFU per 100 mL) (Figure 5), with site and month accounting for 16% of the variation (Table 3).

Channel Morphology and Surface Water Storage

The surveyed urban streams differed in channel geomorphology and habitat complexity. Three of the sites exhibited generally similar characteristics, whereas Reedy Creek differed in a number of ways (Table 4). The stream bed at Reedy is predominantly exposed concrete, whereas the other sites had streambed materials comprised of sand (Rattlesnake) or a mix of sand and gravel (Little Westham and Broad Rock). Reedy Creek exhibited lower habitat complexity (number of habitat transitions = 3 vs. 8-10 at other sites) and less canopy cover (Reedy = 26%, other sites = 89 to 95%). Reedy Creek was also more incised, as indicated by a higher bank height ratio (4.8) relative to the other sites (range = 2.8 to 3.7), and had more stable banks (bank stability index =10.2 ± 0.1) relative to the other sites (range of reach average values = 11.7 to 11.8) (Figure 6).

The four sites differed in the quantity of surface water storage and the dominant water features (pools, riffles and runs). Little Westham contained the greatest volume (90.2 m^3) of surface water storage (other sites = 41.6 to 79.0 m^3; Table 5). Deep pools dominated at Little Westham, holding 88% of the total volume and accounting for 46% of the total surveyed reach length (Table 6, Figure 6). Other sites were dominated by riffles and runs. The dominant feature type at Broad Rock was runs, which carried 64% of the total volume and accounted for 61% of the total reach length. Reedy (concrete channel) was comprised solely of runs. Total water volume at Rattlesnake was more evenly distributed among its feature types (runs = 39%, riffles = 34%, and pools = 28%). Despite the greater volume of surface water storage, Little Westham exhibited a short water transit time (22 min) relative to the other reaches (41 to 84 mins) owing to its short reach length and higher discharge.

Hydrology

Tracer injections were conducted at five sites (three times per site) over a range of baseflow conditions (range = 0.011 to 0.076 m^3 sec^{-1}) during April to September 2018. Average discharge was lowest at Reedy (mean = 0.019 ± 0.018 m^3 sec^{-1}) and Rattlesnake (mean = 0.022 ± 0.004 m^3 sec^{-1}), and highest at Broad Rock (mean = 0.042 ± 0.018 m^3 sec^{-1}) and Stagg (0.046 ± 0.008 m^3 sec^{-1}). (Table 7). Modeled BTCs derived by OTIS-P resulted in a very good visual fit to the observed conductivity data (Figure 7). Hydraulic parameters were estimated within acceptable uncertainty limits (1 < *Dal* < 10, (Wagner and Harvey 1997) and coefficient of variation < 50%).

Three of five sites were generally similar in their storage-related parameters (Broad Rock, Rattlesnake, and Stagg), excluding Little Westham and Reedy (Table 7). Little Westham exhibited significantly greater transient storage zone size (A_S/A = 0.745 ± 0.080), transient storage zone exchange (α = 0.0676 ± 0.0201 min^{-1}), and fraction of median transit time due to storage standardized to 200 m (F_{med}^{200} = 38.8 ± 2.98%). Reedy had the lowest average A_S/A (0.114 ± 0.0330) and a low average F_{med}^{200} (6.66 ± 3.39%). The concrete channel at Reedy had little or no sediment and algae during the spring injections, which correspond to the lowest A_S/A for all injections (0.0706 on 4/18/2018 and 0.0938 on 5/14/2018). During the summer injection, thick algal mats and sediment accumulation was observed, with higher A_S/A (0.179). F_{med}^{200} also increased between spring (1.41 and 5.56%) and summer (13.01%) injections. Reedy and Little

Westham had the highest and lowest average solute velocity (0.279 ± 0.081 and 0.069 ±0.006 m sec^{-1}, respectively; other sites = 0.107 to 0.141 m sec^{-1}).

Discharge had variable effects on transient storage among sites (Figure 8). A_S/A increased with discharge at Rattlesnake and Stagg with only marginal statistical significance (gained storage area; $R^2 = 0.98$, p = 0.10 and 0.09, respectively), and F_{med}^{200} increased with discharge at Stagg ($R^2 = 0.99$, p = 0.07). There were no statistically significant relationships between storage parameters and discharge at Little Westham, Broad Rock, or Reedy. When all injections were combined, there were few significant relationships among calculated parameters. A_S/A at all sites ranged from 0.0706 to 0.847 (Table 7) and was not correlated to Q (Figure 9a). F_{med}^{200} ranged from 1.41 to 44.38%, and was also not correlated to Q (Figure 9b). There was a weak nonlinear negative relationship between solute velocity (range 0.0568 to 0.415 m sec^{-1}) and A_S/A ($R^2 = 0.36$, Figure 9c), indicating that as the size of the storage zone decreases, solute velocity increases.

At Reedy Creek, storage-related parameters in the lower (naturalized) section of the concrete channel were greater than in the upper channel (Table 8). MTT for the entire channel (1 to 230 m = 23.5 mins) was 2 ½ times greater than the upper section (1 to 168 m) alone. A_S/A was more than 2 ½ times greater (0.253) and α twice as great (0.0894 min^{-1}) in the lower section than in the upper (0.0706 and 0.0300 min^{-1}, respectively). F_{med}^{200} increased from 1.41% in the upper section to 16.97% in the lower section.

DISCUSSION

Data collected from urban streams in the Richmond area were generally consistent with expectations based on the urban stream syndrome. Urban sites showed elevated concentrations of nitrogen and more variable TSS concentrations relative to reference sites. Urban channels were deeply incised with unstable banks and low sinuosity. These results agree with those of other studies. Both Meyer et al. (2005) and Sudduth et al. (2011) found average nitrogen concentrations in urban streams to be three times greater than in non-urban streams. Violin et al. (2011) also found that urban streams had deeper channels with less habitat variability than forested counterparts. While the results of this study fit the paradigm of the urban stream syndrome, there was considerable variability among the individual sites. Urban sites varied in terms of habitat complexity, substrates, and hydrologic functioning.

Comparisons of hydrologic function among urban streams are challenging because few studies provide data for multiple sites, and often lack temporal replication to assess site-specific variability. Instead, most studies pool data across urban sites to make generalized comparisons to non-urban sites. Variation in hydrologic functioning among urban streams may arise in part due to differences in criteria used to designate urban streams. Streams with as little as 5 to 10 % impervious cover in their watersheds can exhibit signs of degradation (Shueler et al. 2009). Because of this, conclusions of how urban streams behave, especially in relation to non-urban streams, often conflict. Von Schiller et al. (2018) found that average A_S/A was greater in their urban sites than in non-urban sites, yet Meyer et al. (2005) found no difference in transient storage between urban and non-urban sites. Sudduth et al. (2011) also found no significant difference in storage among studied stream types despite noting differences in physical structure between stream types, yet Gücker et al. (2004) found significant differences in hydrology and storage based on differences in stream morphology. Though the urban stream syndrome is an accepted paradigm, data supporting this are somewhat equivocal, which may be attributed in part to the fact that few studies have measured variability in hydrologic functioning among urban streams.

In assessing the hydrologic functioning of streams, it is important to consider the various metrics in relation to the underlying mechanisms affecting solute transport. For example, high rates of solute transport can arise from a lack of transient storage features (low A_S/A) or from low rates of exchange between active and transient storage zones (α). F_{med}^{200} accounts for changes in velocity, A_S/A, and α, such that stream reaches in which storage greatly affects water transport will have high values of F_{med}^{200}, and streams in which storage has little effect on transport will have low F_{med}^{200} values (Runkel 2002). F_{med}^{200} values at sites in this study ranged between 1.41 (Reedy) and 44.38% (Little Westham), with no statistically significant difference in average F_{med}^{200} values between Broad Rock (mean = 7.79 ± 3.12%), Rattlesnake (mean = 5.67 ± 1.13%), Reedy (mean = 6.66 ± 3.39%) or non-urban Stagg (mean = 8.37 ± 3.21%). These F_{med}^{200} values indicate that transient storage only had a significant effect on overall hydraulic transport at Little Westham (mean = 38.84 ± 2.98%).

Storage values in this study were within the range seen in literature from comparable coastal areas. For forested coastal plain streams, Jin et al. (2005) reported A_S/A of 0.17 to 1.08, α of 0.006 to 0.112 min^{-1}, and F_{med}^{200} of 9.5 to 38.9%. Ryan et al. (2010) reported A_S/A between 0.131 and 1.89 and F_{med}^{200} between 1.2 and 24.9% in urbanized watersheds of 40 to 60% impervious cover. They reported very low α values at their sites (4.15×10^{-4} to 9.6×10^{-3} min^{-1}), which are much lower than in this study. Claessens et. al (2010) in a study of a stream network near the location of Ryan et al.'s study reported α between 0.048 and 0.149 min^{-1}. The values found in this study lie within the range between these two studies.

One novel aspect of this study is that it includes a stream flowing through a concrete channel. Though ubiquitous in urban environments, these have not received much attention in studies of hydrologic function. A concrete channel has no hyporheic zone and, in the absence of in-stream structure (woody debris, gravel bars) would be expected to have low transient storage. This expectation is consistent with results from spring injections, which indicate very low storage in comparison to other sites. In summer, the accumulation of sediment and growth of biofilms was associated with an increase in transient storage, as indicated by values of A_S/A, α and F_{med}^{200} comparable to those of other urban streams. The lack of shading over the channel of Reedy Creek may enhance algal growth on the concrete substrate. It is likely that high discharge events in the winter-spring scour the channel, re-setting transient storage to the lower values observed in the spring. This seasonal switch to subsurface storage is also indicated by the relationship between storage parameters over time. During spring months A_S/A increased nominally (0.07 to 0.09) between runs, yet was accompanied by a threefold increase in alpha (0.03 to 0.09 min^{-1}) and some biofilm on the channel bed. While the size of the storage area remained relatively the same, the rate of exchange increased and thus F_{med}^{200} increased modestly as well. The increase in α through time indicates a change in the physical process driving transient storage (Ensign et al. 2005) as the channel developed subsurface storage. Results from the 4/18/2018 injection also illustrate the potential for higher than expected transient storage in concrete channels that are allowed to naturalize. The presence of a backwater area where the concrete channel transitions to a natural channel favors sediment deposition. The presence of substrates within the concrete channel and associated vegetation increases the size of the storage area and changes the dominant location of storage from surface to “subsurface” (as indicated by the increase in α from upstream to downstream).

In many urban areas, stream restoration options may be limited by the presence of concrete channels, which were built to move water quickly and prevent localized flooding. These channels are often actively maintained by removing accumulated materials (e.g., sediments, woody debris). Active restoration to restore these channels (e.g., by removing the concrete) is expensive and intrusive. However, data from this study shows that partial restoration may be possible without concrete removal by allowing material to accumulate within the channel. This could be done by discontinuing active maintenance of these channels, and through installation of in-stream structures (e.g., large woody debris).

Little Westham exhibited much greater storage than all other sites. This is attributable to the presence of extensive pools, which accounted for 88% of the total volume and almost half of the length of the study reach. The reach is thus dominated by surface storage. Both A_S/A and α were high at this site, and were associated with a large F_{med}^{200} (mean = 38.8%). This indicates that storage processes were more important at Little Westham than at all other sites in this study. The high storage values at this site were comparable to those reported from other sites with similar physical structure. For example, Gücker et al. (2004) reported on a “swampy” stream type that contained extensive non-flowing dead zones. Ryan et al. (2010) reported high A_S/A in their reach with the highest number of deep pools.

Little Westham is scheduled for restoration. Although the site currently has a high level of hydrologic functioning (i.e., high transient storage), it is in the form of surface water storage. In streams, biogeochemical processes are typically associated with substrates, such as biofilm and the hyporheic zone (McMillan et al. 2014, Mendoza-Lera and Datry 2017). Therefore, a restoration that resulted in a shift of transient storage zones from water column to benthos could result in an improvement in biogeochemical functioning, even if hydrologic functioning declined.

Most prior studies that tested for relationships between hydrologic parameters and discharge relied on data pooled across sites. However, individual sites may not respond in the same way as rising water levels may overwhelm some storage zones while bringing others into play. Because of this, I looked at site-by-site relationships, despite having only 3 datapoints. Only Rattlesnake and Stagg showed statistically significant relationships between storage parameters and discharge. A_S/A and F_{med}^{200} had a positive relationship with discharge at Stagg, and A_S/A had a positive relationship with discharge at Rattlesnake. Though the relationship was not significant, high R^2 values indicate a strong negative correlation between A_S/A and discharge at Broad Rock and between F_{med}^{200} and discharge at Little Westham. The results of Mueller Price et al. (2015) showed that hydrologic characteristics vary according to the geomorphology of the sub-reach but were sensitive to high discharge events. Ryan et al (2010) showed that storage characteristics vary with baseflow conditions. They saw a five-fold decrease in average A_S/A from low to high baseflow, and a decrease in spatial variability during high baseflow. Similar to the results of this study, others have also found no relationship between discharge and A_S/A among urban sites (Hart et al. 1999, Jin et al. 2005). This lack of a relationship across sites, and even among sites with similar discharges, may be the result of differences in channel geomorphology (Hart et al. 1999).

Based on these results, if hydrologic function is driven by transient storage, then the goal of restoration would be to incrementally increase storage in urban streams, rather than attaining a fixed value represented by reference sites. Urban streams such as those in this study are deeply incised, such that storage is likely low at high flow. Ryan et al. (2010) suggested that transient storage is more important and more prevalent under low baseflow conditions than high. A meta-analysis by Harvey and Wagner (2000) concluded that a concurrent increase in the rate of exchange and the size of storage occurs accounts for a reduction in transient storage during high flows across several studies. If restoring hydrologic function is dependent on increasing storage via geomorphic-based restoration, projects could consider designs that improve storage at high flow conditions when storage is lowest. This may also work to maintain structural integrity post-construction. Violin et al. (2011) suggested that changes in flow, particularly storm events, are the major structuring force in deep, sandy, and simplified channels, such as Rattlesnake and Little Westham. Considering the vulnerability of these channels to high flow events, restoration projects could consider designing the channels accordingly to reduce the damage from channel-shaping flow events. It is also important to note that geomorphological degradation is not necessarily indicative of overall degradation. Duncan et al. (2011) were unable to correlate structural degradation to habitat degradation, as in-stream habitat is influenced by upstream and watershed characteristics that do not follow the same timescale as channel-forming processes. Similarly, in this study, a wide range of storage conditions were seen across sites that were contained within a wide array of morphological conditions. Though Little Westham is slated for restoration, the results from this study show high hydrologic function that is often used as a predictor of ecologic function. Also, the concrete channel at Reedy, which is expected to have low storage and thus overall poor ecological function, was shown to support biota and develop seasonal storage zones with the potential to perform the ecological services expected of natural streams. Physical or hydrologic impairments may not necessarily indicate overall ecologic function.

In this study of seven urban and three non-urban streams, significant differences in water quality, most notably in concentration and variability, support the model of the urban stream syndrome. Urban streams were also incised and had unstable banks. Urban streams presented a wide range of morphologic and hydrologic conditions. Climate (precipitation and temperature) (Hale et al.2016), regional differences in infrastructure and historical landcover (Parr et al. 2016), and both regional and local differences in geology (Walsh et al. 2005) influence how a stream will respond to urbanization. The natural channel design of most restoration projects aims to increase habitat complexity and reconnect streams to the floodplain by reshaping the channel. These designs should reduce disturbance from stormflow and promote organic matter retention (Sudduth et al. 2011). Because of the localized differences that impair the structure and function of the streams in this study, there may not be an “ideal” stream to mimic when designing projects in urban areas. Projects should work to improve impairments that are specific to each site at both the reach and watershed scale to maximize the efficacy of restoration.

TABLES

Table 1. Land cover summary of study sites' watersheds and studies performed at each site. Urban sites are characterized by developed cover greater than 60%. Non-urban sites were chosen such that developed cover comprised less than 10% of the watershed.

		Studies performed	Mean Discharge ($m^3 sec^{-1}$)	Total Area (ac)	Developed %	Forest %	Other %
Urban	Broad Rock Creek	WQ, Geo, Inj	0.092	1,740	88	10	2
	Gillies Creek	WQ	0.195	9,087	76	13	10
	Little Westham *	Geo, Inj	0.049	1,843	63	37	0
	Rattlesnake Creek	WQ, Geo, Inj	0.020	725	76	24	0
	Reedy Creek	WQ, Geo, Inj	0.107	2,420	84	12	3
	Pocosham Creek	WQ	0.092	3,444	74	24	2
	Upham Brook	WQ	0.248	9,711	88	10	2
Non-urban	Fine Creek**	WQ	0.461	15,040	6	69	23
	Kimages Creek	WQ	*n/a*	2,982	2	84	16
	Stagg Creek *	Inj	*n/a*	3,328	6	54	40

* calculated via USGS Stream Stats GIS application.

**monitoring performed by Randolph-Macon College (Ashland, VA).

Table 2. Urban study sites are monitored every other week for the listed water quality parameters. Discharge is not monitored at Kimages, and *E. coli* is monitored at urban sites only.

Parameter	Units	Parameter	Units
Temperature	°C	Total nitrogen	mg L^{-1}
pH	n/a	Total phosphorus	mg L^{-1}
Specific Conductance	µS cm^{-1}	Nitrate + nitrite	mg L^{-1}
Dissolved oxygen	% and mg L^{-1}	Ammonia	mg L^{-1}
Turbidity	NTU	Ortho-phosphate	mg L^{-1}
Total suspended	mg L^{-1}	Chloride	mg L^{-1}
Discharge	m^3 sec^{-1}	E. coli	CFU per 100 mL

Table 3. p-value results of two-way ANOVAs testing the effects of site, month, and the interaction of each variable on TSS, nutrient, and *E. coli* data.

	Site	Month	Site * Month	R^2
TSS	<0.001	<0.001	<0.001	0.58
TN	<0.001	<0.001	0.40	0.32
NO_x	<0.001	<0.001	0.014	0.72
NH_3	<0.001	0.015	0.70	0.19
TP	0.022	0.40	1.00	0.065
OP	<0.001	0.0098	0.040	0.36
EC	0.01	<0.001	1.00	0.16

Table 4. Morphological characteristics of 4 urban streams located in Richmond, VA.

	Broad Rock	**Little Westham**	**Rattlesnake**	**Reedy**
Surveyed reach length (m)	211.5	87.6	216.1	234.5
# habitat transitions, normalized to 100m	8.3	10.5	9.0	3.0
% canopy closure	95	89	95	26
Average bank height ratio	2.8 ± 0.1	3.1 ± 0.3	3.7 ± 0.4	4.8 ± 1.0
Median substrate size (mm)	20.5	4.0	0.1	concrete
Average bank stability index	11.8 ± 0.2	11.7 ± 0.3	11.7 ± 0.2	10.2 ± 0.1
Gradient	0.0087	0.0045	0.0041	0.0060
Sinuosity	1.04	1.02	1.05	1.07

Table 5. Hydrologic properties of 4 urban streams located in Richmond, VA.

	Broad Rock	**Little Westham**	**Rattlesnake**	**Reedy**
Discharge (m^3 sec^{-1})	0.0123	0.0136	0.0079	0.0155
Reach surface area (m^2)	898.2	384.0	719.7	1191.3
Reach volume (m^3)	79.0	90.2	41.6	49.7
Reach transit time (mins)	84.1	22.0	67.9	40.9
Median reach velocity (m sec^{-1})	0.04	0.06	0.06	0.07

Table 6. Hydrologic properties of habitat feature types in 4 urban streams.

	Habitat feature type	Number of individual features	Total volume of features (m^3)	Average volume (m^3)	Average transit time (mins)	Average velocity (m sec^{-1})
Broad Rock	Pool	2	12.2	6.1 ± 1.5	8.3 ± 2.0	0.02 ± 0.001
	Riffle	7	16.4	2.3 ± 0.7	3.2 ± 0.9	0.05 ± 0.004
	Run	8	50.3	6.3 ± 2.4	8.5 ± 3.2	0.04 ± 0.009
Little Westham	Pool	3	79.5	26.5 ± 13.5	32.5 ± 16.5	0.01 ± 0.004
	Riffle	3	1.8	0.6 ± 0.1	0.7± 0.2	0.1 ± 0.04
	Run	3	8.8	2.9 ± 0.5	3.6 ± 0.6	0.06 ± 0.02
Rattlesnake	Pool	3	11.6	3.9 ± 1.0	8.2 ± 2.05	0.02 ± 0.007
	Riffle	8	14	1.8 ± 0.4	3.7 ± 0.8	0.07 ± 0.007
	Run	8	15.9	2.0 ± 0.4	4.2 ± 0.9	0.05 ± 0.006
Reedy	Pool	0	*n/a*	*n/a*	*n/a*	*n/a*
	Riffle	0	*n/a*	*n/a*	*n/a*	*n/a*
	Run	7	49.7	4.5 ± 1.06	5.5 ± 1.3	0.1 ± 0.02

Table 7. Summary of hydrologic parameters for conservative tracer injections. Values are average of three injections ± SE, except Reedy (end of concrete).

	Total reach length (m)	Effective reach length (m)	Q (m^3 sec^{-1})	MTT (mins)	Solute velocity (m sec^{-1})	Dal	As/A	α (min^{-1})	Dispersion (m^2 sec^{-1})	F_{med}^{200} (%)
Broad Rock	210	196.7 ± 1.7	0.042 ± 0.018	37.00 ± 9.82	0.107 ± 0.025	3.730 ± 1.572	0.219 ± 0.030	0.0177 ± 0.0053	0.367 ± 0.152	7.79 ± 3.12
Little Westham	86	77.0 ± 1.5	0.034 ± 0.008	21.25 ± 2.02	0.069 ± 0.006	3.168 ± 1.172	0.745 ± 0.080	0.0676 ± 0.0201	0.156 ± 0.095	38.84 ± 2.98
Rattlesnake	210	192.0 ± 0.00	0.022 ± 0.004	26.92 ± 5.93	0.141 ± 0.026	2.711 ± 0.848	0.202 ± 0.044	0.0171 ± 0.0017	0.518 ± 0.196	5.67 ± 1.13
Reedy	168	156.7 ± 0.7	0.018 ± 0.007	12.42 ± 4.26	0.279 ± 0.081	7.391 ± 2.253	0.114 ± 0.033	0.0662 ± 0.0184	0.149 ± 0.030	6.66 ± 3.39
Reedy (end of concrete)	62	62	0.033	17.00	0.225	2.803	0.253	0.0894	0.0255	16.97
Stagg	205	184.3 ± 3.7	0.046 ± 0.008	28.50 ± 1.80	0.119 ± 0.007	5.023 ± 0.199	0.173 ± 0.044	0.0284 ± 0.00664	0.199 ± 0.018	8.37 ± 3.21

values are average of three injections ± standard error, except downstream Reedy

Table 8. Summary of April 2018 injection at Reedy Creek. Reach was extended and divided into two sub-reaches.

	Total reach length (m)	Effective reach length (m)	MTT (mins)	Solute velocity (m sec^{-1})	As/A	α (min^{-1})	Dispersion (m^2 sec^{-1})	F_{med}^{200} (%)
Reedy (1-168m)	168	158	6.75	0.415	0.0706	0.0300	0.180	1.41
Reedy (end of concrete, 168-230)	62	62	17.0	0.225	0.253	0.0894	0.0255	16.97
Reedy (1-230m)	230	220	23.5	0.163				

FIGURES

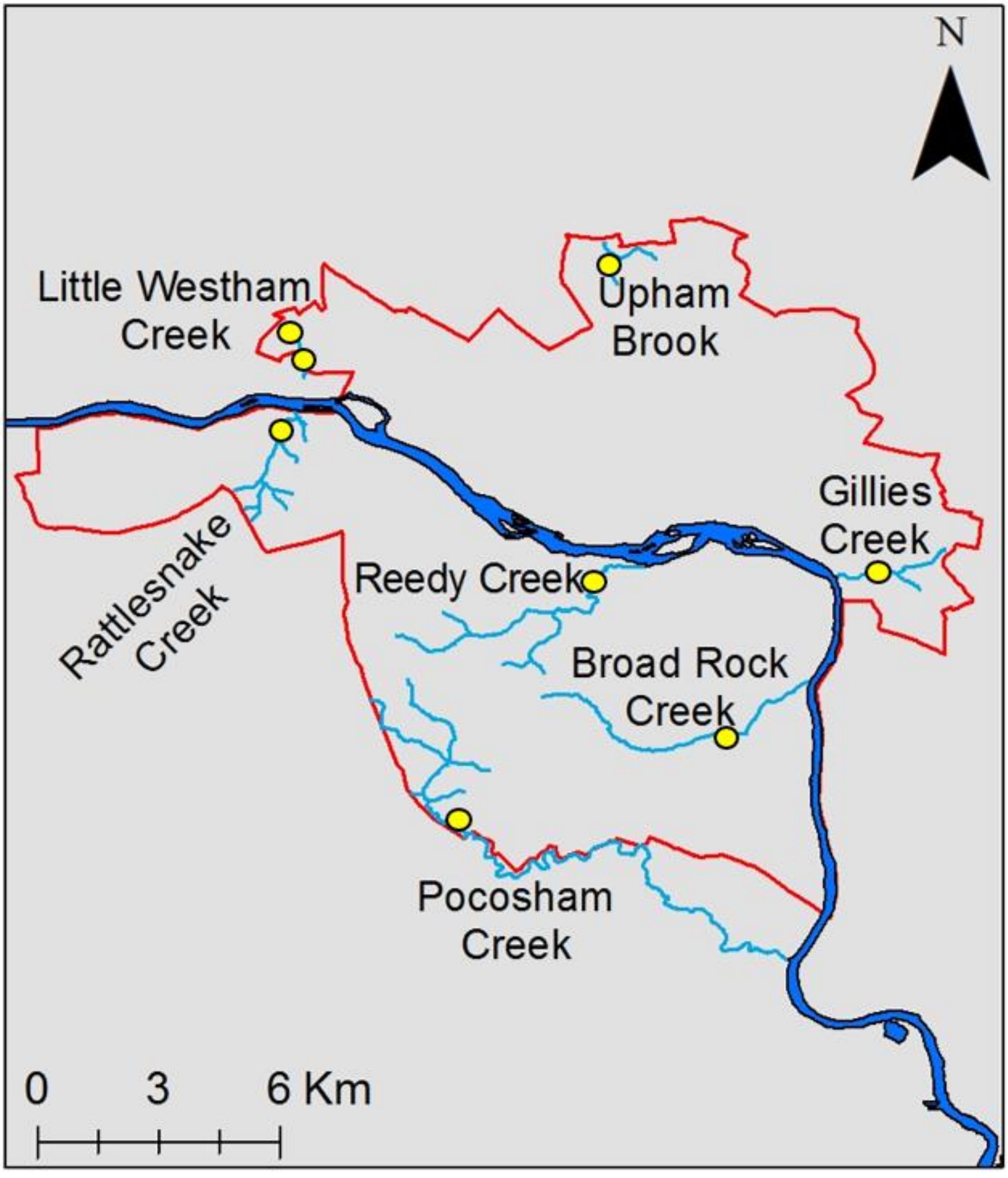

Figure 1. Locations of urban sampling sites (yellow circles) along their respective stream reaches. Non-urban sites Kimages, Fine, and Stagg Creek not shown.

Figure 2. Photos of non-urban (a) and urban (b, c, d, and e) reaches for tracer studies (sites a through e) and physical surveys (sites b through e only).

Figure 3. Upstream (a) and downstream (b) conditions in the concrete channel at Reedy. The upper 168 m for injections is characterized by a wide, shallow bed and open canopy that promotes algal growth in summer. 168 – 230 m conditions are more naturalized, due to the accumulation of substrates and plants.

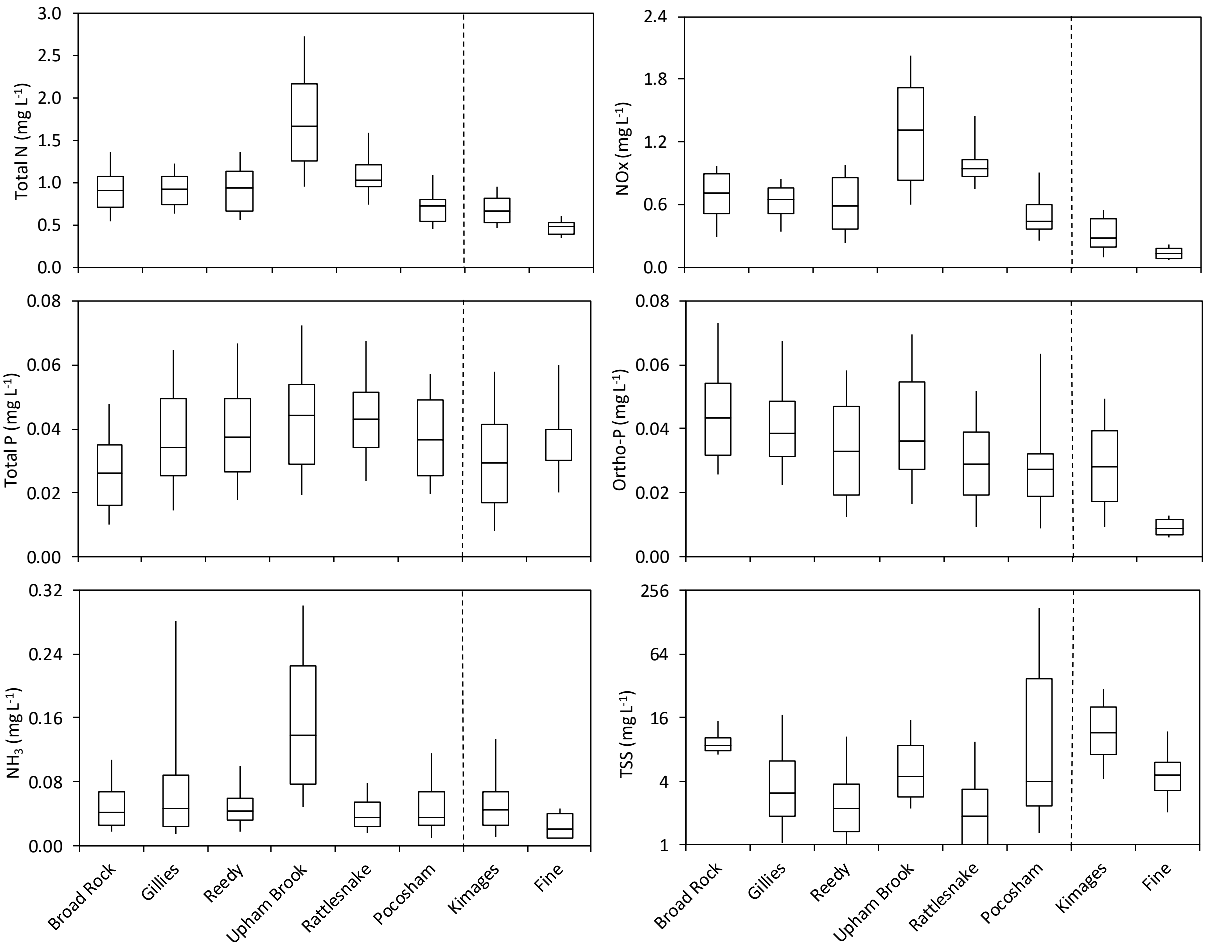

Figure 4. Variation in water quality among urban and non-urban streams in the Richmond metropolitan area. Error bars represent 10^{th} and 90^{th} percentiles. Kimages and Fine Creek provided as non-urban references. Sample sizes vary (Broad Rock, Gillies, Reedy, Upham Brook n = 64; Rattlesnake and Pocosham n = 25, Kimages n = 64, Fine n = 32).

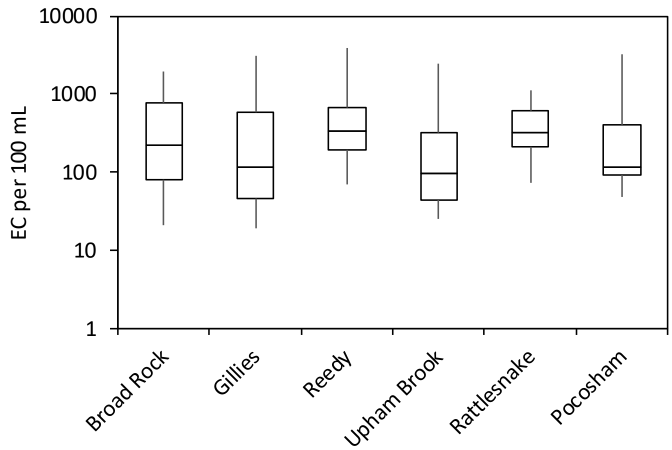

Figure 5. Box plots showing variability in E. coli among urban streams in the Richmond area. Sample sizes vary (Broad Rock, Gillies, Reedy, Upham Brook n = 63; Rattlesnake and Pocosham n = 25).

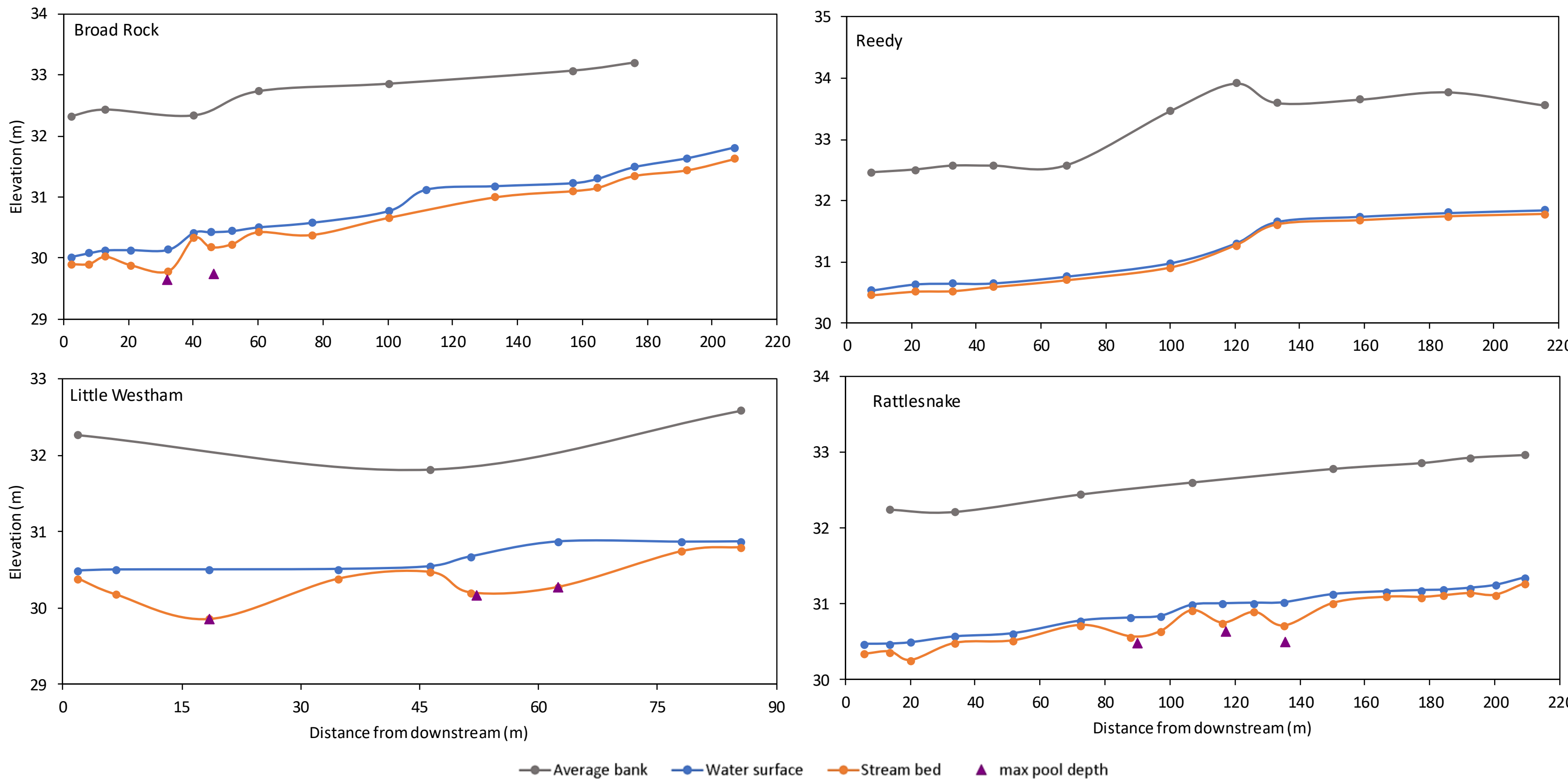

Figure 6. Longitudinal profiles of surveyed urban streams. Midpoint of features are marked with circles (●), and the deepest point of pools marked with triangles (Δ). Bank height ratios indicate deep incision at all sites. Pools are the dominant feature at Little Westham, containing the most volume and largest area. Reedy is the least diverse in terms of habitat features, comprised only of shallow, fast runs.

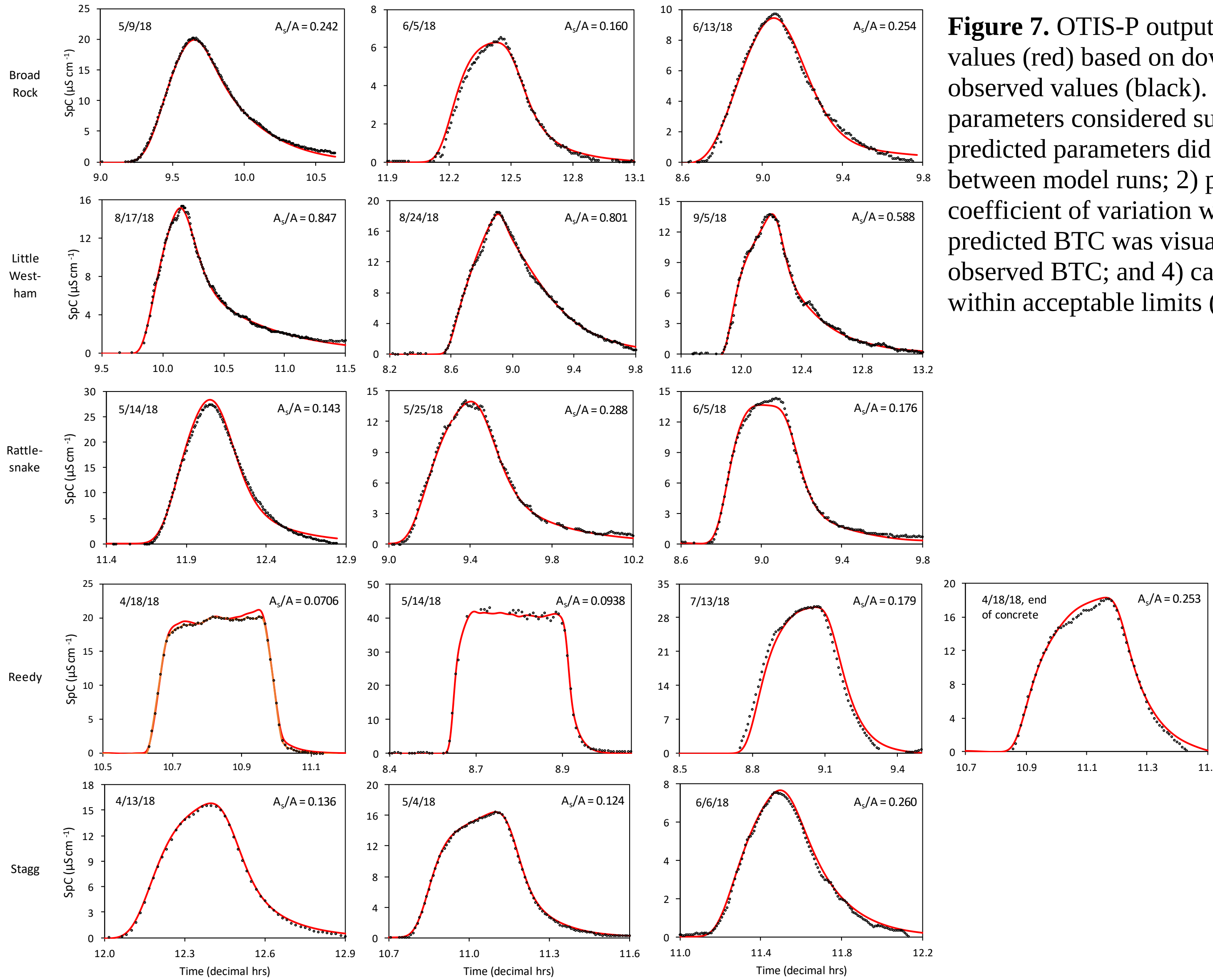

Figure 7. OTIS-P output of predicted values (red) based on downstream observed values (black). Model parameters considered successful if: 1) predicted parameters did not change between model runs; 2) parameter coefficient of variation was <50%; 3) predicted BTC was visually similar to observed BTC; and 4) calculated *Dal* was within acceptable limits (1 < *Dal* < 10).

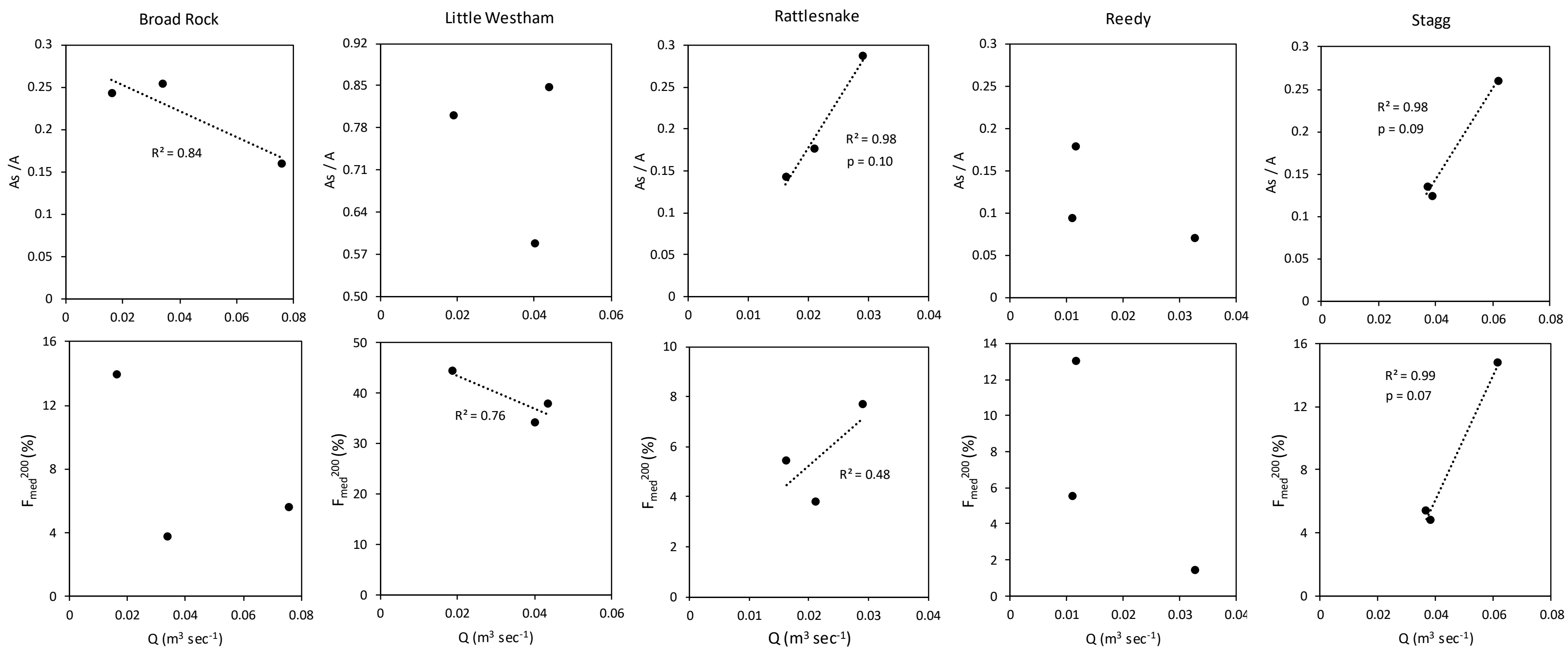

Figure 8. Relationships between select storage parameters (A_s/A and F_{med}^{200}) and discharge by site. Dotted line marks a moderate or strong relationship, and p values provided for statistically significant relationships ($p < 0.1$).

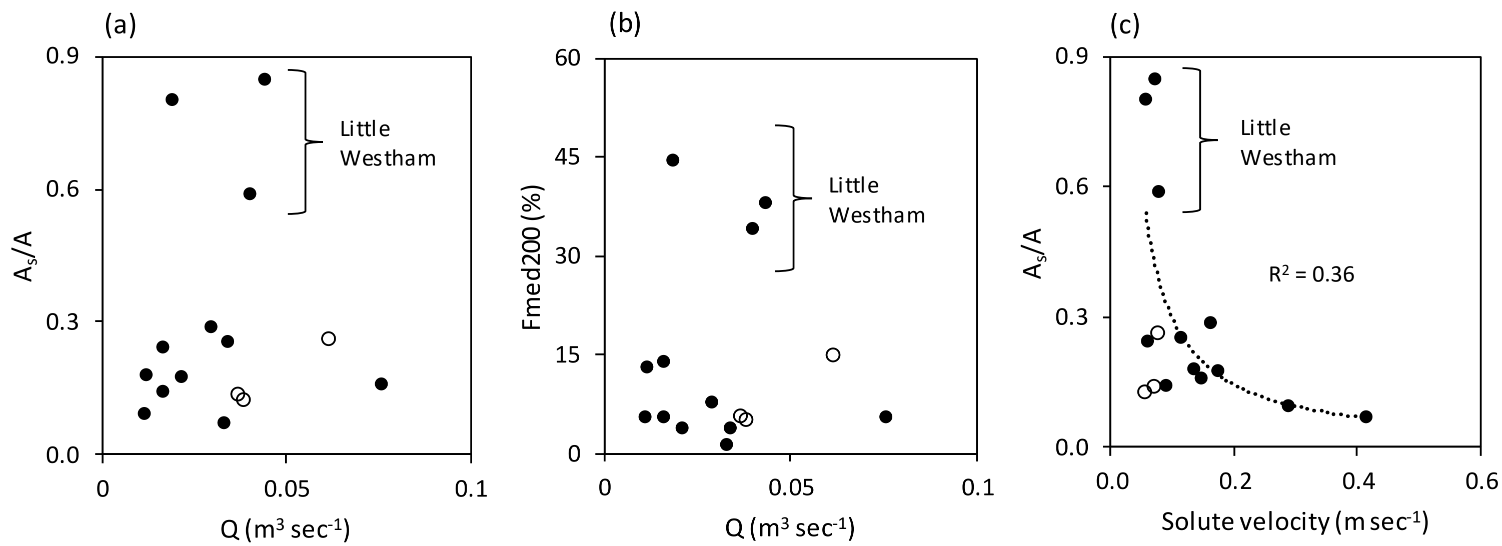

Figure 9. Relationships between a) discharge and A_s/A, b) discharge and F_{med}^{200}, and c) solute velocity and A_s/A among 4 urban streams located in the Richmond metro area. Hollow circles represent the non-urban site, Stagg Creek.

REFERENCES

Bernhardt, E. S., Palmer, M.A., Allan, J.D., Alexander, G., Barnas, K., Brooks, S., Carr, J., Clayton, S., Dahm, C., Follstad-Shah, J., Galat, D.,Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Katz, S., Kondolf, G.M., Lake, P.S., Lave, R., Meyer, J.L., Meyer, O'Donnell, T.K., Pagano, L., Powell, B., and Sudduth, E. (2005). Synthesizing U.S. river restoration efforts. *Science*, 308 (5722), 636–637.

Bukaveckas, P.A. (2007). Effects of channel restoration on water velocity, transient storage, and nutrient uptake in a channelized stream. *Environmental Science & Technology*, 41(5), 1570–1576.

Claessens, L., Tague, C.L., Groffman, P.M., and Melack, J.M. (2010). Longitudinal and seasonal variation of stream N uptake in an urbanizing watershed: effect of organic matter, stream size, transient storage and debris dams. *Biogeochemistry*, 98, 45-62.

Craig, L.S., Palmer, M.A., Richardson, D.C., Filoso, S., Bernhardt, E.S., Bledsoe, B.P., Doyle, M.W., Groffman, P.M., Hassett, B.A., Kaushal, S.S., Mayer, P.M., Smith, S.M., and Wilcock, P.R. (2008). Stream restoration strategies for reducing river nitrogen loads. *Frontiers in Ecology and the Environment*, 6(10), 529-538.

Duncan, W.W., Goodloe, R.B., Meyer, J.L., and Powell, E.S. (2011). Does channel incision affect in-stream habitat? Examining the effects of multiple geomorphic variables on fish habitat. *Restoration Ecology*, 19(1), 64-73.

Environmental Protection Agency (2009). Method 1603: *Escherichia coli* (*E. coli*) in water by membrane filtration using modified membrane-thermotolerant *Escherichia coli* agar (modified mTEC). Office of Water. Washington, D.C. *EPA-821-R-09-007*.

Gooseff, M.N., Briggs, M.A., Bencala, K.E., McGlynn, B.L., and Scott, D.T. (2013). Do transient storage parameters directly scale in longer, combined stream reaches? Reach length dependence of transient storage interpretations. *Journal of Hydrology*, 483, 16-25.

Gücker, B. and Boëchat, I.G. (2004). Stream morphology controls ammonium retention in tropical headwaters. *Ecology*, 85(10), 2818-2827.

Hale, R.L., Scoggins, M., Smucker, N.J., and Suchy, A. (2016). Effects of climate on the expression of the urban stream syndrome. *Freshwater Science*, 35(1), 421-428.

Harman, W., Starr, R., Carter, M., Tweedy, K., Clemmons, M., Suggs, K., and Miller, C. (2012). A Function-Based Framework for Stream Assessment and Restoration Projects. US Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Washington, D.C. *EPA 843-K-12-006.*

Hart, D.R., Mulholland, P.J., Marzolf, E.R., DeAngelis, D.L., and Hendricks, S.P. (1999). Relationships between hydraulic parameters in a small stream under varying flow and seasonal conditions. *Hydrological Processes*, 13, 1497-1510.

Harvey, J.W., Wagner, B.J. (2000). Quantifying hydrologic interactions between streams and their subsurface hyporheic zones. In *Streams and their Ground Waters.* Jones, Jeremey B and Mulholland, Patrick J. (eds.) Academic Press: New York, 3-44.

Jin, H.S. and Ward, G.M. (2005). Hydraulic characteristics of a small Coastal Plain stream of the southeastern United States: effects of hydrology and season. *Hydrological Processes*, 19(20), 4147–4160.

Kaushal, S.S., Groffman, P.M., Mayer, P.M., Striz, E.S., and Gold, A.J. (2008). Effects of stream restoration on denitrification in an urbanizing watershed. *Ecological Applications*, 18(3), 789–804.

McMillan, S.K., Tuttle, A.K., Jennings, G.D., and Gardner, A. (2014). Influence of restoration age and riparian vegetation on reach-scale nutrient retention in restored urban streams. *Journal of the American Water Resources Association*, 50(3), 626-638.

Mendoza-Lera, C., and Datry, T. (2017). Relating hydraulic conductivity and hyporheic zone biogeochemical processing to conserve and restore river ecosystem services. *Science of the Total Environment*, 579, 1815-1821.

Meyer, J.L., Paul, M.J. and Taulbee, W.K. (2005). Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society*, 24(3), 602-612.

Mueller Price, J.S, Bledsoe, B.P. and Baker, D.W. (2015). Influences of sudden changes in discharge and physical stream characteristics on transient storage and nitrate uptake in an urban stream. *Hydrological Processes*, 29(6), 1466–1479.

Mueller Price, J.S., Baker, D.W. and Bledsoe, B.P. (2016). Effects of Passive and Structural Stream Restoration Approaches on Transient Storage and Nitrate Uptake. *River Research and Applications*, 32, 1542–1554.

Mulholland, P.J., Steinman, A.D., Marzolf, E.R., Hart, D.R., and DeAngelis, D.L. (1994). Effect of periphyton on hydraulic characteristics and nutrient cycling in streams. *Oecologia*, 98(1), 40-47.

O'Connor, B.L., Hondzo, M. and Harvey, J.W. (2010). Predictive modeling of transient storage and nutrient uptake: implications for stream restoration. *Journal of Hydraulic Engineering*, 136(12), 1018–1032.

Parr, T.B., Smucker, N.J., Bentsen, C.N., and Neale, M.W. (2015). Potential Roles of past, present, and future urbanization characteristics in producing varied stream responses. *Freshwater Biology,* 35(1), 436-443.

Runkel, R. L. (1998). One-dimensional transport with inflow and storage (OTIS): a solute transport model for streams and rivers. *Water-Resources Investigations Report 98-4018.*

Runkel, R.L. (2002). A new metric for determining the importance of transient storage. *Journal of the North American Benthological Society*, 21(4), 529-543.

Ryan, R.J., Welty, C. and Larson, P.C. (2010). Variation in surface water–groundwater exchange with land use in an urban stream. *Journal of Hydrology*, 392(1-2), 1–11.

Schueler, T.R., Fraley-McNeal, L., and Cappiella, K. (2009). Is impervious cover still important? Review of recent research. *Journal of hydrologic engineering*, 4, 309-315.

Stream Solute Workshop (1990). Concepts and methods for assessing solute dynamics in stream ecosystems. *Journal of the North American Benthological Society*, 9(2), 95–119.

Soulsby, C., Piegat, K., Seibert, J., and Tetzlaff, D. (2011). Catchment-scale estimates of flow path partitioning and water storage based on transit time and runoff modelling. *Hydrological Processes*, 25(25), 3960–3976.

Sudduth, E.B., Hassett, B.A., Cada, P., and Bernhardt, E.S. (2011). Testing the field of dreams hypothesis: functional responses to urbanization and restoration in stream ecosystems. *Ecological Applications*, 21(6), 1972–1988.

U.S. Geological Survey (1998). Revised methods for characterizing stream habitat in the National Water-Quality Assessment Program. Raleigh, North Carolina, USA. *Water-Resources Investigations Report 98-4052*.

Vermont Water Quality Division (2009). Stream geomorphic assessment phase 3 handbook. Vermont Agency of Natural Resources, Department of Environmental Conservation.

Violin, C.R., Cada, P., Sudduth, E.B., Hassett, B.A., Penrose, D.L., and Bernhardt, E.S. (2011). Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. *Ecological Applications*, 21(6), 1932–1949.

Von Schiller, D., Martí, E., Riera, J.L., Ribot, M., Marks, J.C., and Sabater, F. (2008). Influence of land use on stream ecosystem function in a Mediterranean catchment. *Freshwater Biology*, 53, 2600-2612.

Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M., and Morgan II, R.P. (2005). The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society*, 24(3), 706-723.

Wolman, M.G. (1954). A method of sampling coarse river-bed material. Transactions of the American Geophysical Union, 35(6), 951-956.