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EVALUATING RESTORATION SUCCESS IN THE WATERSHED CONTEXT

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ABSTRACT

Urbanization has predictable effects on stream ecosystems that result in altered hydrology, increased nutrient and pollutant concentrations, decreased nutrient retention, and decreased biodiversity. Stream restoration seeks to re-establish the natural structure and function of stream ecosystems and practices that increase geomorphic complexity by adding features such as pools and riffles, which also increase transient storage. Transient storage refers to parts of the stream that slow down water movement or temporarily detain water and includes both the surface stream and the hyporheic (subsurface) sediments. Quantifying transient storage is important since it is an important determinant of temperature, nutrient retention, and biological diversity in stream ecosystems.

We investigated the effects of watershed land-use on the ecological benefits of stream restoration, by examining transient storage and the physical, chemical, and biological condition of restored and unrestored streams in forested and urbanizing watersheds. Given the importance of transient storage to overall ecosystem function, we ask the following question: *Does stream restoration in urbanizing and forested watersheds enhance transient storage and does this enhancement have a comparable impact on ecological parameters in both land uses?* We used a combination of conservative trace injections with modeling to quantify surface and hyporheic transient storage. We compared transient storage metrics with ecosystem metrics (macroinvertebrates, water quality, temperature) to investigate the relationship among restoration, transient storage, and ecosystem metrics.

Our results suggest that in low gradient, sand-bed alluvial streams, like those found throughout the North Carolina Piedmont, standard stream restoration practices are ineffective at creating hyporheic exchange, but can significantly increase in-channel transient storage in pools. Conversely, in steeper reaches with thin or absent alluvial cover, like those in the urban watersheds, stream restoration may create some hyporheic exchange where little or no hyporheic exchange existed prior to restoration. Among the field sites, hyporheic flux was driven by different factors, depending on the stream sediment size. Spatial variation in hyporheic flux in the forested unrestored stream was driven by variability in vertical head gradient whereas in the urban restored stream, which also has a smaller grain size compared to the forested unrestored stream, spatial variability in hyporheic flux had a direct relationship with variability in hydraulic conductivity.

We found weak relationships between hyporheic flux and ecosystem parameters. Temperature and benthic macroinvertebrates (abundance/diversity) were more strongly controlled by landscape characteristics such as canopy cover than transient storage. Overall nitrate was exported from all sites; however, export rates were lower in the restored compared to the unrestored streams. In contrast, total phosphorus was retained in these restored streams. Retention was not clearly related to hyporheic exchange since the site with the highest hyporheic flux was an exporter of both nutrients.

In this study restoration increased surface water storage which may have dissimilar effects on ecosystem health compared to increased hyporheic storage. If enhanced stream-groundwater connections are a desired restoration outcome, we will need to initiate new design and construction practices.

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1 INTRODUCTION

1.1 Effects of urbanization on stream ecosystems

Urbanization has predictable effects on stream ecosystems that have been well documented across multiple systems. Urban streams have flashier storm hydrographs, elevated nutrient concentrations, decreased rates of nutrient retention, decreased biological diversity, and higher concentrations of heavy metals and pesticides than their forested counterparts (Aber et al. 1989; Bernot and Dodds 2005; Walsh et al. 2005). These alterations have been summarized as the "urban stream syndrome" (sensu Walsh et al. 2005) and are the focus of many stream restoration programs.

Stream restoration seeks to re-establish the natural structure and function of stream ecosystems. In 2006, in North Carolina, restoration projects were completed at a cost of $427 million (Bernhardt et al. 2005) and annual expenditures are likely higher today. In urban ecosystems, the goal of restoration is often to stabilize grade control and decrease erosion. The techniques used to achieve these goals involve the placement of human-made structures such as j-hooks, v-shaped weirs, and log sills, and it is assumed that these structures also improve in-stream habitat and result in improved ecological integrity through increased biodiversity. Many researchers however, have challenged the value of stream restoration since several studies have indicated that current practices do not produce the desired biological improvements (Palmer et al. 2005; Palmer et al. 2010).

One of the challenges of understanding and improving urban streams is that while we have well documented the changes to streams from urbanization, researchers still do not have a clear understanding of the causative effects. Thus, any restoration will be hampered by not clearly understanding how restoration activities directly influence key stream ecosystem properties and processes.

1.2 Transient Storage

Transient storage refers to parts of the stream that slow down water movement or temporarily detain water (Figure 1). In the surface stream, transient storage occurs in pools and eddies, which can increase stream temperature and promote nutrient retention (Bukaveckas 2007). In the subsurface, transient storage is represented by the hyporheic zone, an area where stream water and groundwater mix (Triska et al. 1989; Edwards 1998).

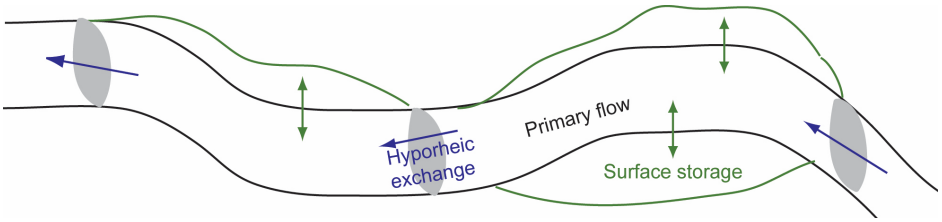

Figure 1: Conceptual diagram showing surface and subsurface locations of transient storage.

Stream restoration seeks to provide stream stability while re-establishing ecosystem services (Palmer et al. 2005) but its effect on transient storage and stream-groundwater interactions have been neglected (Bukaveckas 2007). Stream restoration may increase transient storage by

changing the stream's gradient and adding diverse geomorphic features such as pools and riffles. Such geomorphic changes may largely increase in-stream transient storage. Conversely, stream restoration may inhibit hyporheic exchange through sediment compaction during construction, reducing stream-groundwater interactions, and effectively removing the hyporheic zone from total transient storage.

Hyporheic flow is an important aspect of transient storage promoting the mixing of stream water with groundwater (Kalbus et al. 2007). This interaction promotes chemical reactions, provides a diverse habitat and regulates stream water temperature to stay constant all year around (White 1993; Boulton et al. 1998; Edwards 1998; Biksey and Gross 2001). Localized areas of upwelling and downwelling occur when streams do not have homogeneous bed slope due to their diverse geomorphology (Harvey et al. 1996). In natural streambeds, stream water recharges the subsurface where the bed slope decreases (e.g., flow from pool to riffle), and groundwater discharges to the stream at areas where an increase in streambed slope is seen such as flow from a riffle into a pool (White 1993). For example, this can be observed in patterns of hydraulic head and water temperature around a weir (Hester et al. 2009).

Interactions between stream and groundwater are driven by the vertical hydraulic gradient and the sediment's hydraulic conductivity, as expressed by Darcy's Law (Fetter 1999):

$$Q = -KA\left(\frac{dh}{dl}\right) \qquad (1)$$

where

Q = discharge (m^3)
K = hydraulic conductivity ($m\ s^{-1}$)
A = cross-sectional area (m^2)
dh = change in hydraulic head along the groundwater flowpath (m)
dl = length of the flow path (m).

Determining the magnitude and direction of groundwater gradients can be accomplished through measuring water levels in piezometers embedded in the streambed. The water level inside the piezometer is compared to the height of the streambed water, allowing the calculation of a vertical hydraulic gradient between the surface and subsurface water. Wells placed on the banks can help provide information on whether the stream is a gaining or losing stream, by comparing riparian ground water elevation to stream water elevation (Wroblicky et al. 1998; Baxter 2003; Wondzell 2006).

Hydraulic conductivity (K) is a measure of the ease of water movement through a porous media. It can be calculated by adding a known volume (slug) into piezometers using the Hvorslev slug test method (Fetter 1999). An advantage of the Hvorslev slug test is that it accounts for sediment anisotropic properties, which have been shown to produce Kh (horizontal hydraulic conductivity) that can be 10-100s of times larger than the Kv (vertical hydraulic conductivity) (Chen 2000). Another advantage of the Hvorslev test is that it is in situ.

Previously transient storage studies were described by mathematic explanations of the exchange between areas of water retention and the main channel (Bencala et al. 2011). Later, numerical

models were created to predict the retention behavior using conservative tracers. One of the most utilized models is the One dimensional Transport with Inflow and Storage model (OTIS). OTIS is used to estimate retention behavior by fitting Equations 2 and 3 to a tracer break through curve (Runkel and Broshears 1991). OTIS represents the stream as two compartments: the stream's main channel, described by advection and dispersion; and its storage zones which include hyporheic zones, pools and eddies.

$$\frac{\partial C}{\partial t} = -\frac{Q}{A} * \frac{\partial C}{\partial x} + \frac{1}{A} * \frac{\partial}{\partial x} * \left(AD * \frac{\partial C}{\partial x}\right) + \alpha(C_s - C) + q_l(C_L - C) \qquad (2)$$

$$\frac{dC_s}{dt} = \alpha * \frac{A}{A_s}(C - C_s) \qquad (3)$$

where

A - stream channel cross-sectional area (m^2)
A_S – storage zone cross-sectional area (m^2)
C – in stream solute concentration ($mass/m^3$)
C_L – solute concentration in lateral inflow ($mass/m^3$)
C_S – storage zone solute concentration ($mass/m^3$)
D – dispersion coefficient (m^2/s)
Q – volumetric flow rate (m^3/s)
t – time (seconds)
x – distance (m)
α – storage zone coefficient ($second^{-1}$)
q_l – lateral inflow rate (m^3/sm)

1.3 Research Goal and Objectives

The proposed research investigated the effects of watershed land-use on the ecological benefits of stream restoration, by examining transient storage and the physical, chemical, and biological condition of restored and unrestored streams in forested and urbanizing watersheds. We focused on transient storage because it can be an important determinant of stream temperature, nutrient retention, and biological diversity, and because restoration projects frequently impact surface and subsurface transient storage. Given the importance of transient storage to overall ecosystem function, we asked the following question: *Does stream restoration in urbanizing and forested watersheds enhance transient storage and does this enhancement have a similar impact on ecological parameters?* For this study, we selected nutrient retention and macroinvertebrate abundance/diversity as our ecological parameters.

The objectives of the project were to: 1) quantify and compare transient storage in restored and unrestored reaches of forested and urban streams; 2) quantify the physical, chemical and biological conditions of these streams and correlate these measures with degree of transient storage; and 3) evaluate the relative success of these restoration activities for urban and forested watersheds.

2 METHODS

2.1 Site Description

The study area lies within the Beaverdam Creek watershed (N 35°10′11″, W 80°59'16") in Mecklenburg County, North Carolina (Figure 2). The watershed has a total drainage area of 11.8 km^2. The study area has a humid subtropical climate, with 30-year normal monthly mean, minimum, and maximum temperatures of 16.3°C, 5.4°C, and 26.8°C, respectively, and a mean annual precipitation of 1104 mm. The Beaverdam Creek watershed drains into Lake Wylie, a water supply and hydropower reservoir on the Catawba River. Lake Wylie is classified as a eutrophic lake, due to excessive nutrient loading from urban and agricultural land uses in the surrounding watershed (Gagrani 2013). Watershed topography is characterized by gentle slopes in the upland areas, with narrow valleys and incised streams in mid-watershed positions, and broad floodplains near the watershed outlet. There is a mixed deciduous forest that has become established after a history of forest harvest and agricultural activities in the watershed (Eckardt, 2003). Bedrock in the watershed is metamorphosed quartz diorite and tonalite, gabbro, gabbronorite, and granodiorite (Goldsmith et al. 1988), and soils are predominantly sandy loam and fine sandy loam in texture (USDA NRCS 2010).

Four study reaches have been identified in subwatersheds of Beaverdam Creek, offering strong contrasts in restoration status and land use. Reaches BD1 (FOR) and BD4 (URB) are unrestored, while reaches BD2 (FOR-R) and BD3 (URB-R) underwent restoration in 2006. Restoration activities included channel re-patterning, installation of in-stream structures, and riparian zone planting with native species. The unrestored reaches reflect the cumulative effects of agricultural, forest harvest, and urban land use in their watersheds, and may be considered over-widened and incised. Forest cover in the watersheds of FOR and FOR-R is approximately 50%, with low density residential (<12% built upon area) covering 31-32% of the watersheds. More intensely developed areas are limited to the headwaters of the forested watersheds, and cover 17-21% of the area with the Charlotte-Douglas International Airport, I-485 and other roads, and commercial/industrial land cover. The URB reach has a similar land cover distribution to the FOR and FOR-R reaches, but the human land use is distributed more widely through the watershed. Downstream of the URB study reach, the human land use intensity increases sharply. The most intensely developed watershed is URB-R, with 36% forest cover, 56% high density residential, and 7.5% roads and institutions.

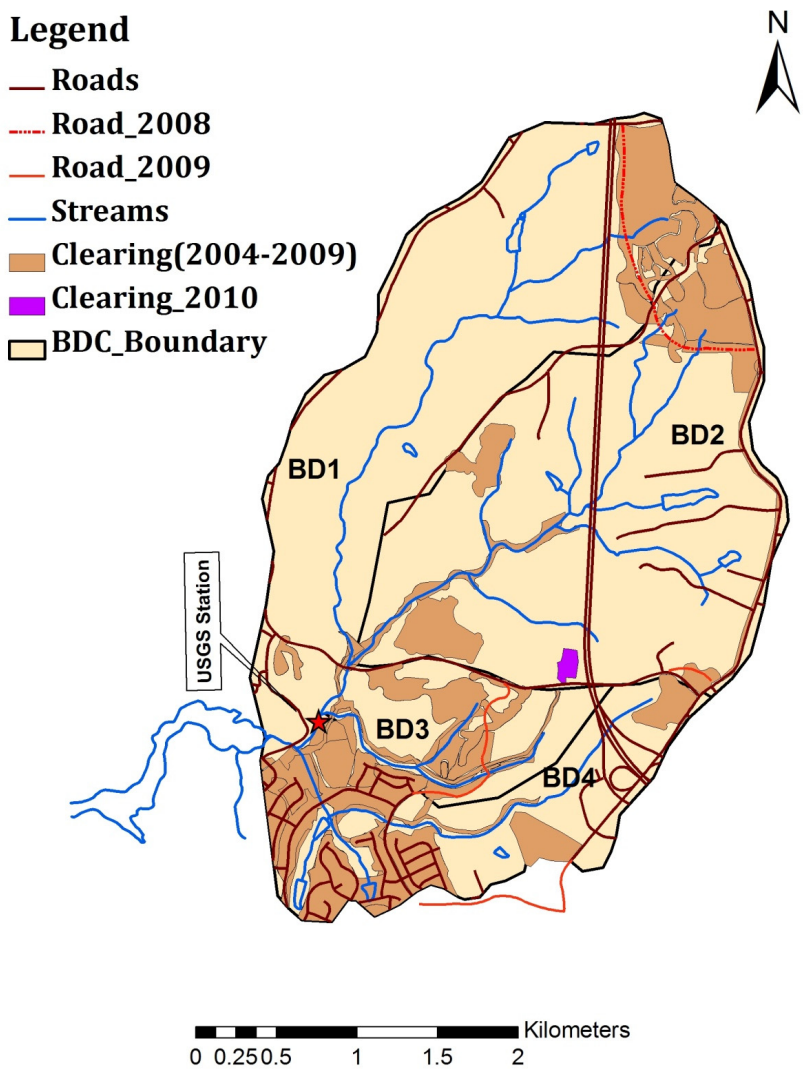

Figure 2: Location of field sites. FOR (BD1) and URB (BD4) are unrestored while FOR-R (BD2) and URB-R (BD3) are restored reaches. Figure reproduced from Gagrani 2013.

Each study reach is ≥ 100 m in length, representative of the overall stream conditions, and has no tributaries. Each reach has undergone past water quality research and has a stage gage at the downstream end of the stream (Gagrani 2013). The urban sites are predominantly bedrock with gravel and fine sediments, while the forested sites are predominantly sands and fine sediment in the stream. The forested watersheds are slightly larger than the urban watersheds, with FOR having a drainage area of 3.78 km^2 and FOR-R draining 4.74 km^2. URB-R has a drainage area of 0.93 km^2, while URB has a drainage area of 1.0 km^2. All streams are 1 m to 5 m wide. The urban reaches have a reserved riparian corridor acting as a buffer from local housing complexes.

2.2 Reach Characteristics

2.2.1 Canopy cover

A spherical crown densitometer was used to estimate canopy density in each study reach during summer 2012. Three representative transects throughout each reach were identified. At each of the transects, the densitometer was held in the middle of the stream at elbow height and far enough away from the user that her reflection is outside the grid but still seen in the mirrored sphere. The densitometer was then leveled, and the user closed her non-dominant eye. The number of dots where the sky was visible was counted, while dots where the upper canopy was visible were not counted. If there was understory interference, the measurement was moved upstream to a more suitable location. This measurement was completed while facing upstream, towards the left bank, towards the right bank and downstream at each transect. The average number of uncovered dots for each transect was calculated, multiplied by 1.036 and subtracted by 100 to get % overstory density. The four measurements taken at each transect were averaged together to find the canopy cover at that location. An average whole stream canopy density was then calculated by a weighted average of transect in relation to whole stream canopy cover.

2.2.2 Sediment Size

A sediment size analysis was completed for the top 10 cm of streambed sediment in each study reach (summer 2011), with the median diameter (D50) chosen to represent sediment size. Each reach had three representative transects spaced randomly throughout for sediment sampling. At each transect, nine samples were collected using a core that is driven 10 cm into the sediment. In the lab, each sample was thoroughly mixed, placed in a weighed metal pan, dried in an oven at 50° C until completely dry, and reweighed to get the sample's dry weight. The sample was then ashed in an oven at 500° C for 8 hrs to burn off the organics, then cooled and re-weighed. The difference between the sediment's dry weight and ashed weight was used to calculate the percentage of sediment that was made up of organic compounds. Next, the sample was placed though a separator until an unbiased sample of 100 g was obtained. The 100 g sample was placed into a sieve stack and shaken using a sieve shaker for >10 minutes. Sediment in each sieve was weighed, and a graph of the cumulative sediment size distribution was created (Table 1), allowing the calculation of the D50 of each sample.

2.2.3 Discharge

In a separate project, gauging stations were installed and maintained near the outlets of each sub-watershed. At each site, stage was recorded at 5 minute intervals using Odyssey water level loggers and converted into a discharge record using a stage-discharge rating curve. Discharge records for water year 2012 were obtained from Gagrani (2013). The gauging stations in FOR, FOR-R, and URB-R are immediately downstream of the study reaches. In URB, the gauging station is significantly downstream of the study reach, and has a drainage area of 192% relative to the study reach. Nonetheless, patterns of flow and no flow are observed to be similar between the study reach and the gauging station.

Table 1: Sieve ranges used to separate the composite sediment samples to quantify overall reach grain size.

Maximum	Minimum
> 8 mm	8 mm
8 mm	4 mm
4 mm	2 mm
2 mm	1 mm
1 mm	500 µm
500 µm	250 µm
250 µm	1.25 µm
1.25 µm	36 µm
36 µm	< 36 µm

2.2.4 Survey for longitudinal and cross-sectional profiles

Two sets of longitudinal and cross sectional profiles were collected during summer 2011 and recollected in summer 2012. Each field site was assigned an arbitrary benchmark near the downstream end of each reach on the stream bank. The bench marks were marked with a piece of rebar and their GPS locations were recorded.

In summer 2011, the survey was completed with an auto level, tripod, stadia rod and tape measures. One tape was placed in the thalweg with zero at the downstream end of the stream reach, and extended upstream for 100 meters. The tripod and auto level were set up at a location that would allow optimal view of the downstream of the reach and as far upstream as possible. The auto level was leveled, height of instrument found and a back sight to the benchmark measured. Cross sections were measured every five meters along the stream. At least ten points were measured at each cross-section, with more points in sections with complex topography. Each cross section included points at bank full, the thalweg, and the water edges. The edge of water measurements acts as a double check of measurements, as they should have the same exact elevation. However, the location of the water edge is specific to the date of measurement, as the wetted area changes quickly with discharge. When clear sight lines could not be extended any farther upstream, the tripod and auto level were relocated. Before the autolevel was moved, a fore sight was taken to an upstream turning point and marked with a flag. The tripod and autolevel were reset up as far upstream as possible, while still in view of the marked fore sight location. Once set up, a back sight was taken to the same turning point, and these measurements will be used to adjust the data collected after the turning point.

In summer 2012, the survey was completed using total stations, prism, rod, and tripods. The total station was set up similarly to the autolevel. Each time the total station was set up, the GPS location was collected; the direction was set internally north. Surveys with the total station did not have as many turning points as the autolevel, because the laser could see through thicker

brush than humans. Cross sections were collected every five meters, and notes were collected about bankfull, thalweg, current edge of water, and the top and bottom of the exposed pipe for every piezometer. This time the cross sections were in greater detail and additional points were taken between each cross section to characterize the pools in each reach. When the total station view became obstructed, a new location was selected and a turning point was established, as described above.

2.3 Transient Storage

2.3.1 Slug Tracer Injections

To compare seasonal changes in total storage, a sodium chloride (NaCl) slug injection was conducted each season at all sites. The goal of these injections was to create a breakthrough curve for each stream that was suitable to apply in OTIS-P to calculate total storage. For each injection a solution of NaCl and distilled water was mixed in the laboratory. Each injection consisted of a 20% salt solution that was placed on a mixer for multiple hours until the solution was dissolved. A 20% salt solution was chosen since it allows the NaCl to stay suspended in colder field conditions (Moore 2005). A reach was selected at each field site that allowed for adequate mixing in a short distance from the injection site and had no tributaries. Injection reach length varied depending on discharge where longer reaches were necessary to ensure mixing during low flow.

On the day of the injection, discharge and wetted area were collected at each site. An YSI datasonde model 6600vv/ 6000MS V2 or a Hydrolab datasonde 4a was placed at the downstream end of the study reach and recorded conductivity and temperature at 15 s intervals. A second datasonde was placed upstream of the injection site to record background data that was used to correct for natural changes in stream conductivity. A known volume of solution was then injected upstream the mixing reach over 30 seconds. The stream was left undisturbed until the conductivity returned to background levels.

2.3.2 Seasonal Piezometer Measurements

Hydraulic head differences between streambed and surface water were measured using piezometers systematically distributed throughout the reach in FOR, FOR-R, and URB-R sites. At the URB site, bedrock substrate made it unfeasible to install many piezometers, and likely prohibited significant ground water-surface water interactions.

Piezometers were constructed from ¾” inside diameter PVC pipe, with short piezometers cut to 91cm or 100 cm and longer piezometers cut to 150 cm. Each piezometer had a screen length of 3 cm from the bottom and was plugged with a washer and hex bolt to prevent sediment from entering the piezometer during installation. Short piezometers penetrated the streambed a minimum of 10 cm and a maximum of 35 cm at the time of installation. A smaller number of long piezometers were nested with short piezometers to examine the hydraulic head difference between streambed and surface water at different depths. These piezometers were driven into the streambed as far as possible, but not exceeding 125 cm.

Each piezometer identifies areas of upwelling or downwelling occurring in different areas of the stream. Upwelling was seen when the water inside the piezometer was higher than the surface water. Downwelling was seen when the stream water was higher than the groundwater level inside the piezometer. Hydraulic gradient was calculated by dividing the difference in the head

measurements by the depth of penetration to the top of the screened interval into the streambed. Vertical head gradients were measured for the fall, winter and spring seasons (2011-2012).

A falling head test was conducted in each piezometer to calculate hydraulic conductivity (Equation 4). The Hvorslev method calculates hydraulic conductivity using the radius of well casing, radius of well screen, length of well screen and the time it takes for the water level to fall to 37% of the initial change (Fetter 1994).

$$K = \frac{r^2 \ln\left(\frac{L}{R}\right)}{2\, LeTo} \quad (4)$$

where

K= hydraulic conductivity (cm s^{-1})
r = radius of well casing (cm)
R = radius of well screen (cm)
L_e = length of well screen (cm)
T_0 = time it takes for the water level to fall to 37% of the initial head difference (s)

The water flux (Darcy flux) was then calculated by multiplying hydraulic conductivity by the vertical head gradient (Equation 1).

2.3.3 Calculation of Transient Storage Metrics

OTIS-P quantifies total transient storage by using parameters estimated from a conservative tracer injection. From preliminary field trials we determined that constant rate injections were inappropriate for the small, low gradient streams used in this study due to the long time (often >8 hrs) to reach steady state conditions. Thus, we decided to use slug injections so that we could integrate a longer experimental reach compared to the constant rate method. OTIS-P improves upon the original OTIS model by incorporating an automated parameter estimation framework (Runkel and Broshears 1991). OTIS-P is a widely used model employed in streams ranging from coarse gravel step pools to sandy-silt, gentle gradient streambeds (Briggs et al. 2010; Bernot et al. 2006). OTIS-P allows inverse estimation of model transient storage parameters using the concentration data (electrical conductivity for this study) collected from the tracer injection . The model parameters include cross sectional area (A), storage zone area (A_s), exchange coefficient (α) and the stream dispersion coefficient (D). The storage exchange flux (q_s) is the product of As and α (Harvey et al. 1996).

SURFER, a krigging model, was utilized to create whole reach map of stream bed elevation, hydraulic conductivity, and flux. A survey was conducted to collect relative easting and northing of the base of the piezometers at FOR, FOR-R, and URB-R. Corresponding hydraulic conductivity and vertical head gradients were matched with the survey points and graphed.

Survey and field data were imported into SURFER's worksheets to produce grid files for hydraulic conductivity and vertical head gradient for each season. Kriging, a method for interpolating spatial data derived from regionalized variable theory (Oliver and Webster 1990), was applied to fill in the hull of the data. Values outside of the data points were not estimated to limit extraneous areas in calculations. Once the hydraulic conductivity grids and vertical head gradients were complete their grids were multiplied to create a flux map for each season. Point

values of hydraulic conductivity and vertical head gradient were also multiplied, and those point flux measurements were then kriged in SURFER. There was little difference in results between the two methods.

Once all the maps were completed, the measure tool was used to find the area of each contour interval that was downwelling. Downwelling areas were summed together to find the total area of downwelling for each specific discharge. Total downwelling flux was then calculated by finding the sum of the areas multiplied by their corresponding specific discharge class. The total hyporheic exchange flux was calculated by assuming that all downwelling was hyporheic flow since upwelling dominated the reach. Upwelling is comprised of hyporheic flow reentering the stream and groundwater gains.

Hyporheic exchange flux (q_{he}; $m^3\ s^{-1}\ m^{-1}$) was calculated by dividing the total hyporheic exchange flux (Q_{HE}) by the piezometer reach length (the length of stream where piezometers were installed; L_{pz}). Calculation of total hyporheic exchange flux (Q_{HE}) is described in Table 2. From these calculations the ratio of hyporheic exchange to storage exchange was found using the hyporheic exchange flux (q_{he}) and the storage exchange flux (q_s) from OTIS-P. Finally, the ratio of channel storage exchange to the total storage exchange (q_{cs}/q_s) was found by subtracting the hyporheic exchange flux (q_{he}) from the storage exchange flux (q_s) and dividing it by the storage exchange flux.

2.4 Benthic Macroinvertebrates

Benthic macroinvertebrates were collected using a modified version of the NC Division of Water Quality's standard qualitative sampling procedure (NCDENR 2006). Since our proposed sites have channel widths less than 4m, we used the abbreviated "Qual 4" method that includes collecting 4 composite samples: 1 kick sample, 1 bank sweep, one leaf pack sample, and visual collection from large rocks and logs. This procedure has been successfully used previously in Beaverdam Creek (Allan et al. 2013). Our modifications to this procedure involved using a dip net for both the kick sample and bank sweeps. Benthic macroinvertebrates were counted and identified to the family using a dissecting microscope. Relative abundance was calculated and organisms assigned a category of Rare (1-2 specimens), Common (3-9 specimens), or Abundant (>10 specimens). Additionally, we calculated EPT (Ephemeroptera + Plecoptera + Trichoptera) richness, EPT abundance, and total taxa richness (at the family level). Benthic macroinvertebrate metrics were compared across seasons and sites. Macroinvertebrate metrics were compared with flux parameters at each site.

Table 2: Summary of flux calculation parameters used in this study.

Parameter	Definition	Units
Total downwelling flux (S)	(K*dh/dl*A_b) Specific discharge (K*dh/dl) at piezometers is kriged in Surfer. The total area (A_b) within each specific discharge class is then determined and the total flux is calculated. These results are summed across all specific discharge classes that are downwelling.	m^3 s^{-1}
Total upwelling flux	Calculated as above, except only areas with upwelling fluxes are included.	m^3 s^{-1}
Total Hyporheic Exchange Flux (Q_{HE})	Total hyporheic exchange flux is assumed to be the smaller of downwelling or upwelling for each reach. In our case, downwelling is smaller. All downwelling is matched by upwelling in the reach, and "excess" upwelling represents groundwater gains. We have not quantified these gains for all streams and time steps, but examination of the flux maps shows the predominance of upwelling fluxes in the BD streams.	m^3 s^{-1}
Piezometer reach length (L_{pz})	Calculated based on the Euclidean distance between the upstream most and downstream most piezometers. Distance was measured along the thalweg of the stream channel.	m
Hyporheic exchange flux (q_{he})	Q_{HE}/L_{pz}	m^3 s^{-1} m^{-1}
Alpha (α)	storage zone exchange coefficient	s^{-1}
Area (A)	main channel cross-sectional area	m^2
Storage exchange flux (q_s)	A* α	m^3 s^{-1} m^{-1}
q_{he}/q_s	Ratio of hyporheic exchange to storage exchange	unitless
Channel storage exchange flux (q_{cs})	q_s-q_{he}	m^3 s^{-1} m^{-1}
q_{cs}/q_s	Ratio of channel storage exchange to total storage exchange	unitless

2.5 Water Quality and Temperature

Water quality was sampled at the downstream ends of each reach as part of the Long-Term Beaverdam Creek Watershed Study (Allan et al. 2013). Monthly baseflow samples were analyzed for nitrate and total phosphorous. Nitrate and total phosphorus were measured on a Dionex DX500 ion chromatograph.

Temperature probes were used to determine the differences in stream temperature along the length of the study reaches and the overall water temperature change across seasons. Hobo Tidbit temperature probes were placed in each stream reach in shallow pools that are deep enough to cover the temperature probe but shallow enough that the water appears to be well mixed (i.e., not stagnant). The Tidbits recorded data at five minute intervals, beginning in July 2011 and continuing through the 2012 water year. Some probe failures occurred resulting in data gaps, and when streams were dry, the probes recorded air temperature.

3 RESULTS

3.1 Reach Characteristics

3.1.1 Canopy Cover

Canopy cover was measured on 14 September 2012, prior to autumn leaf fall, and is considered representative of growing season stream shading. The unrestored streams had higher canopy cover than the restored streams (Table 3), with the URB-R having the lowest canopy cover. Many riparian trees were removed during restoration construction, a common practice that should be revised. This very limited dataset suggests that restoration can have a significant impact on riparian canopy, particularly where there is only a limited riparian buffer adjacent to the stream in a developed watershed.

There was some variability in canopy cover within each study reach, with FOR having consistently high canopy cover, FOR-R having lower canopy cover at the downstream end of the reach, and URB-R having the lowest recorded canopy cover (55%) at the upstream end of the study reach. At that site, restoration was accompanied by planting of conifers adjacent to the stream. These conifers, ~3-5 m tall, have done a very incomplete job of forming a canopy over and along the stream reach.

Table 3: Mean canopy cover for each study site. Data were collected 14 September, 2012.

	Canopy cover (%)
FOR	99
FOR-R	82
URB-R	50
URB	91

3.1.2 Streambed Sediment Size

Grain size was highest at the FOR site and decreased among FOR-R, URB-R and URB (Figures 3-6). At URB, the reach was incised down to bedrock where only small sediments trapped between bedrock outcrops could be sampled, and reflect the small grain size at this site. Grain size D50 was highest at FOR (1.5 mm) and decreased among FOR-R, URB-R and URB (0.75 mm, 0.54 mm , and 0.43 mm, respectively).

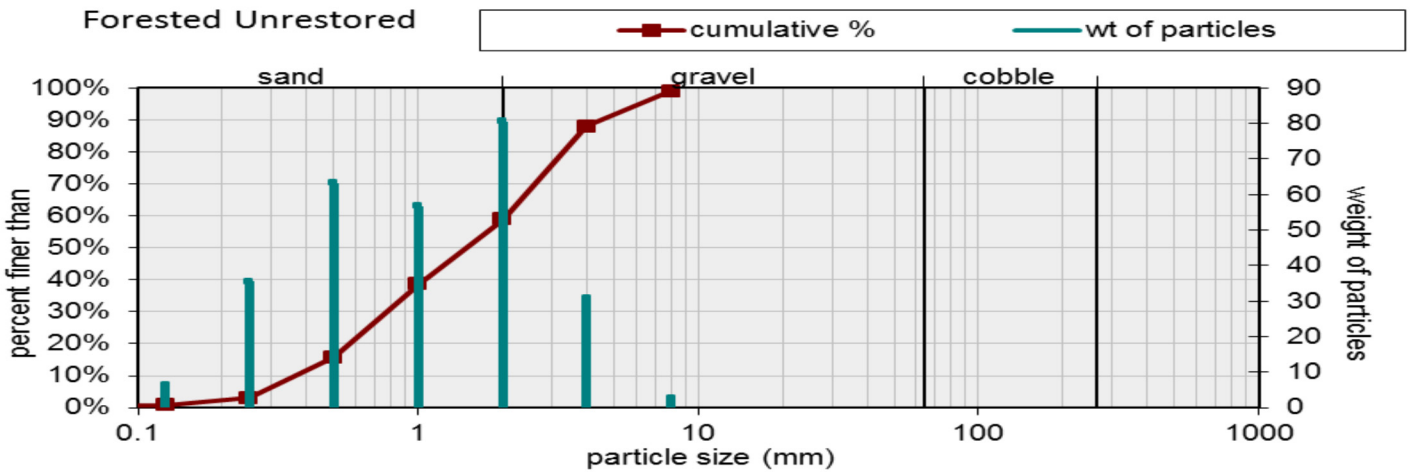

Figure 3: Composite grain size analysis for the top 10 cm in the forested unrestored (FOR) reach measured during fall 2011.

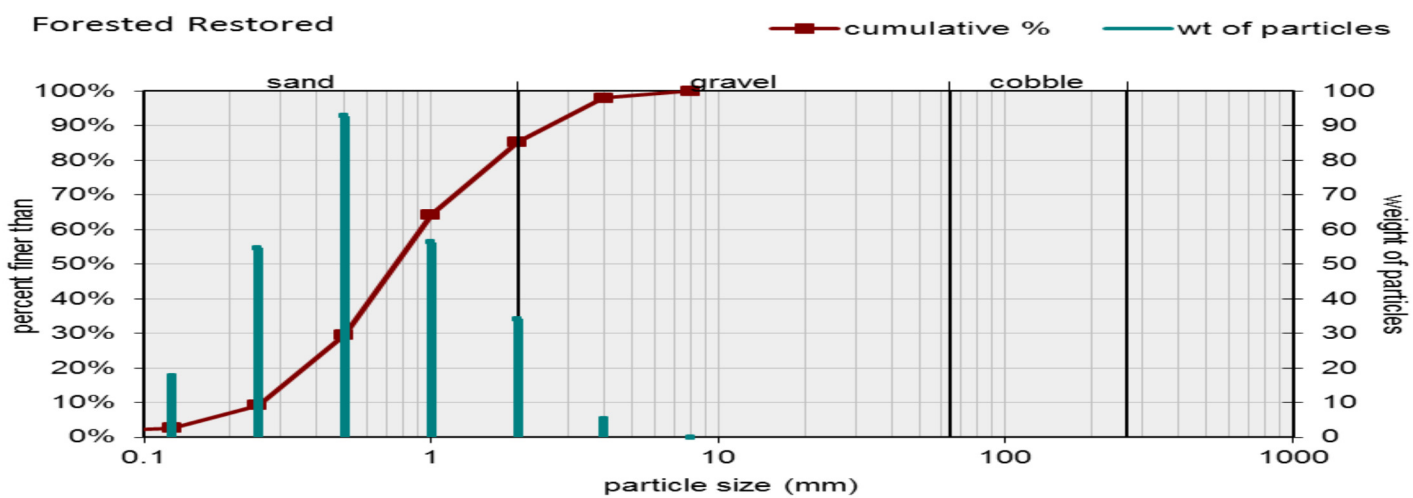

Figure 4: Composite grain size analysis for the top 10 cm in the forested restored (FOR-R) reach measured during fall 2011.

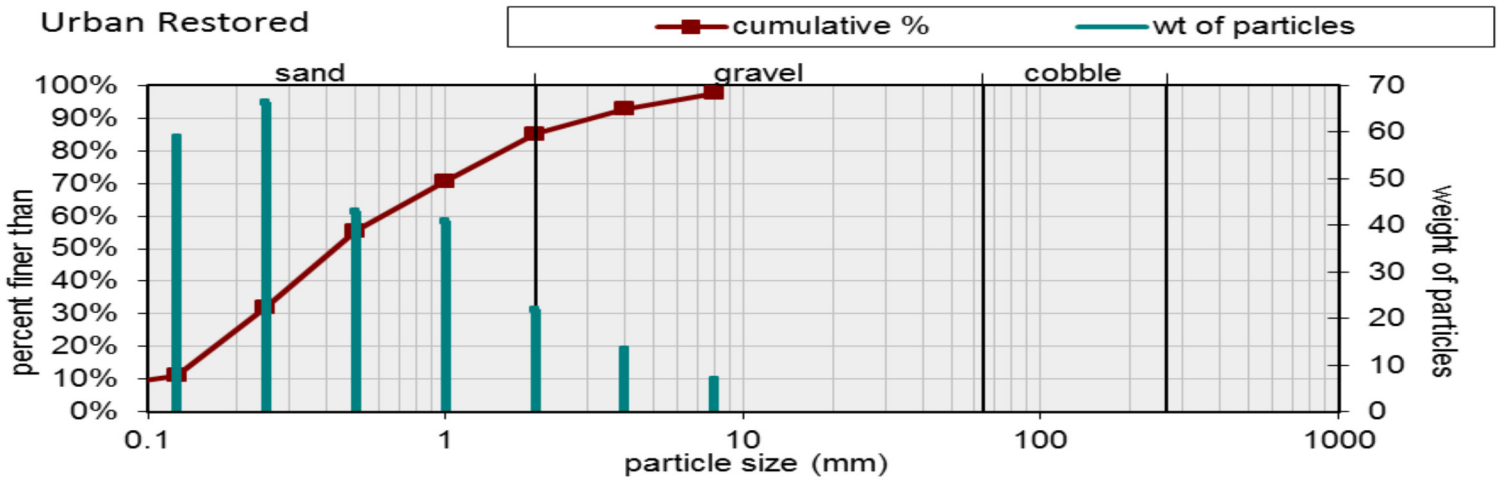

Figure 5: Composite grain size analysis for the top 10 cm in the urban restored (URB-R) reach measured during fall 2011.

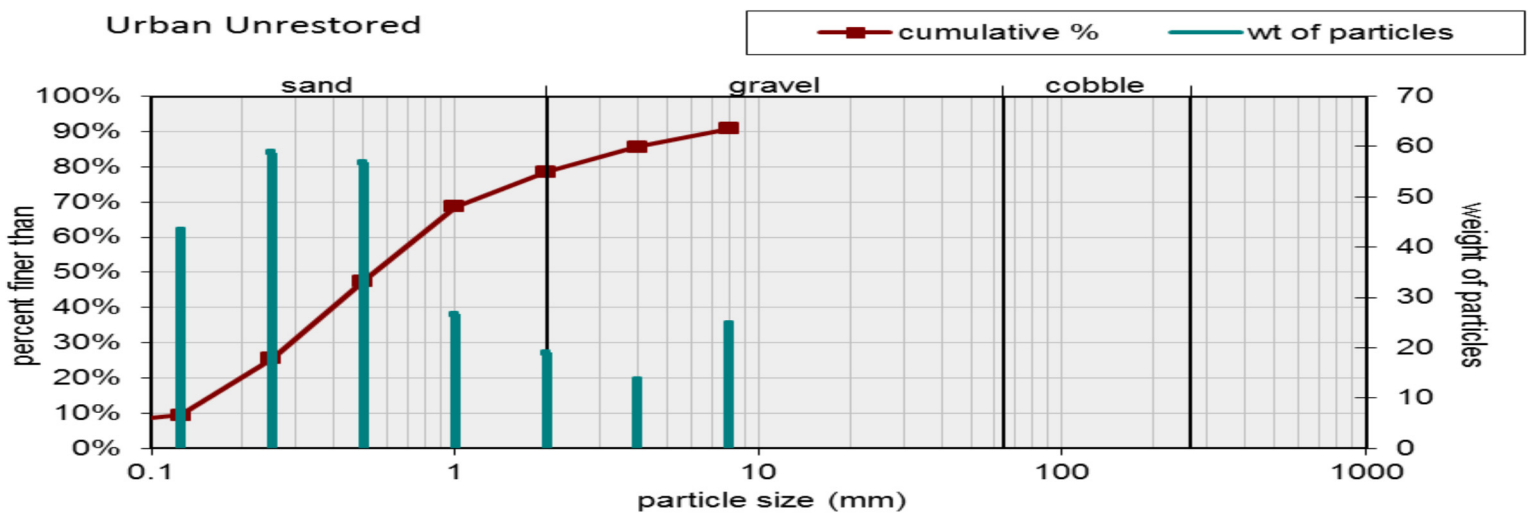

Figure 6: Composite grain size analysis for the top 10 cm in the urban unrestored (URB) reach measured during fall 2011.

3.1.3 Discharge

Discharge records were developed by Gagrani (pers. comm.) for stage recorders located near the outlet of each watershed (Table 4). During water year 2012 (October 1, 2011 to September 30, 2012), the highest flow event occurred on March 3, 2012 in FOR, FOR-R, and URB. In URB-R, the March 3rd event was slightly surpassed by a rain event on September 8, 2012. During peak flows, discharge was tens to hundreds of times greater than baseflow in the streams, as is typical for Piedmont streams.

Three of the streams had periods with no flow and dry stream beds, in water year 2012. In For and FOR -R, the dry conditions occurred between June and September, while in URB-R, dry periods were distributed throughout the year, with February and March having significant dry spells. In FOR and FOR -R, the dry periods were often continuous for days or weeks in a row, whereas in URB-R, the dry periods were much shorter and no 24-hour day was continuously dry. URB was perennial, never experiencing dry periods. This pattern has been observed in previous years and is attributed to leaky pipes, lawn irrigation, and slow releases from stormwater control measures in the URB watershed (Gagrani 2013). These anthropogenic effects may also contribute to reduced dryness at URB-R relative to FOR and FOR-R.

Table 4: Summary statistics for water year 2012 hydrographs in the study streams. Discharge (Q) is weighted by watershed size.

	FOR	**FOR-R**	**URB-R**	**URB****
Drainage area (km^2)	3.78	4.74	0.93	1.92
Mean Q ($L\ s^{-1}\ km^{-2}$)	3.2	3.2	1.7	7.3
Maximum Q ($L\ s^{-1}\ km^{-2}$)	225	468	366	224
Days in which the stream had no flow*	70	119	53	0

*Discharge recorded as 0 for any portion of the 24-hour period.
** There are no data for URB from 9/22/12 to 9/30/12.

Baseflow contributions to the stream hydrographs were determined via graphical hydrograph separations using the sliding interval of Pettyjohn and Henning (1979) and Sloto and Crouse (1996). Over the course of water year 2012, this hydrograph separation indicated that the four streams had variable proportions of baseflow, ranging from 31% in FOR-R to 73% in URB. In terms of absolute flows, URB-R had the lowest baseflow, averaging 0.7 L s^{-1} during water year 2012.

In the forested streams, summer months had the lowest proportion of their total hydrograph derived from baseflow, which likely reflects strong evapotranspirative demand by the forest vegetation during the growing season. In June-August, FOR had an average baseflow discharge of 1.3 L s^{-1}, while FOR-R had an average baseflow of 0.2 L s^{-1}. These extremely low flows, accompanied by 1/2 to 2/3 of days with no flow at all, made it extremely difficult to measure transient storage during summer months in the forested streams.

In URB-R, November and December had the lowest proportions of baseflow relative to the total hydrograph, while in URB-R the least baseflow dominated month was September. The shifts in

the seasonality of low baseflows may reflect changes in evapotranspirative water demand associated with urbanization and/or supplementation of baseflow via over-irrigation or leaky infrastructure. In June-August, URB-R averaged 1.0 L s^{-1} baseflow, while URB had 7.4 L s^{-1}.

3.1.4 Longitudinal and Cross-Sectional Profiles

From the surveyed longitudinal profiles, slopes of the low flow water surface, stream bed in the thalweg, and bankfull water surface were calculated for each ~5 m segment and for the overall reach. The coefficient of variability (standard deviation over average) of each slope was also calculated as a measure of the long profile variability (Table 5). The thalweg slope can locally reverse signs, reflecting unevenness in bed topography (e.g., between pools and riffles), while the bankfull and low flow water surfaces should have a consistent sign because water flows downhill (Figure 7). In each of the longitudinal profiles, some 5 m segments had upward sloping water surfaces, indicating measurement error during the survey. Therefore, we chose to calculate the coefficient of variability for the bankfull and low flow water surfaces only for those segments with downward slopes.

Table 5. Reach-averaged slope and coefficient of variability (CV) of bankfull water surface (BF WS), low flow water surface (LF WS), and thalweg, as surveyed in June 2011.

	FOR	FOR-R	URB-R	URB
Mean BF WS slope	8.00×10^{-4}	3.37×10^{-3}	1.85×10^{-2}	1.43×10^{-2}
Mean LF WS slope	2.46×10^{-3}	1.37×10^{-3}	2.06×10^{-2}	1.52×10^{-2}
Mean thalweg slope	1.99×10^{-3}	2.24×10^{-3}	2.49×10^{-2}	1.56×10^{-2}
BF WS CV	0.553	1.05	0.897	0.715
LF WS CV	0.896	1.07	1.12	0.717
Thalweg CV	14.6	8.93	2.44	2.43

Slopes at URB-R and URB were about 10 times steeper than at FOR and FOR-R. FOR and FOR-R had very similar reach-averaged slopes, while URB-R was slightly steeper than URB. In general, the reach-averaged thalweg and water surface slopes were similar, as is expected.

The two restored streams (FOR-R and URB-R) had somewhat greater water surface CV at both low flow and bankfull stages, suggesting that restoration is successful in increasing the heterogeneity of the water surface slope (Table 5). However, this water surface heterogeneity did not directly correspond to variability in the thalweg profiles. Forested, unrestored stream FOR had the most variable thalweg topography, while urban, unrestored URB had the least variable thalweg topography. The thalweg CV is similar between restored URB-R and unrestored URB, but the calculations in URB-R excluded three prominent rock structures >0.5 m high, built during restoration, so the stream bed topographic variability is actually higher than reported.

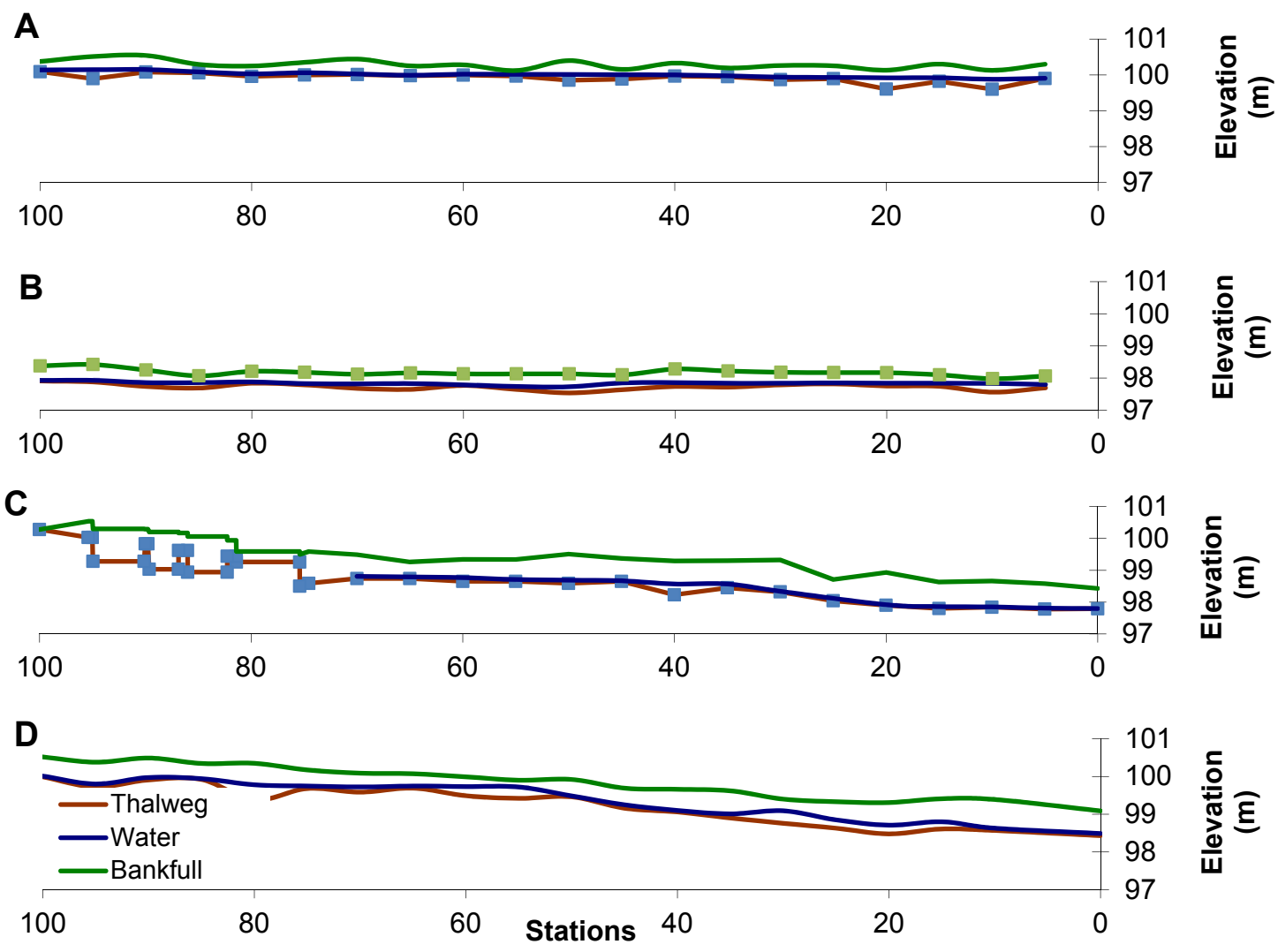

Figure 7: Summary of longitudinal profiles along the 100 m study reaches for (A) FOR, (B) FOR-R, (C) URB-R, and (D) URB in the Beaverdam Creek watershed. The zero station represents the most downstream site.

3.2 Transient Storage

3.2.1 Total Transient Storage (OTIS-P modeling)

Total stream storage parameters were found using OTIS-P for each stream in fall, winter and spring. Due to low or no flow during the summer period, we could not calculate transient storage during the summer season. The ratio of the cross-sectional area of the storage zone to the channel (As/A) highlights the different storage characteristics for each stream (Table 6). The forest restored site had higher As/A than the other three reaches. This could be a result of the stream having a low bed slope and discharge which combined with the abundant manmade structures creates large standing pools. The two unrestored sites (forested unrestored and urban unrestored) had similar normalized total storage (As/A), with the forest unrestored having slightly more storage than urban unrestored. At the forest unrestored, hyporheic flow was included in the total stream storage calculated by OTIS-P, while at urban unrestored, the stream is largely incised to bedrock, reducing the potential for hyporheic flow. While several large pools created significant in-channel storage, the urban restored had consistently lower As/A than the other three sites. In urban restored reach, the restoration strategy created large pools, which diverted flow under and around the structure during base flow conditions, under high flow conditions water would cascade down the pools and steps. These pools had their hyporheic exchange measured but the total storage not quantified as the retention time in the large pools was not measured because the long time required to label the pools with the salt injection were not feasible.

Table 6: Summary of OTIS-P parameters calculated during in this study. Parameters are described in Equations 2 and 3, where A = stream cross-sectional area; A_S = storage area; D = dispersion coefficient; α = storage zone coefficient; and As/A = storage zone coefficient corrected for cross-sectional area. Reach lengths varied depending on discharge and the necessary length needed to ensure maximum mixing.

	FOR			**FOR-R**			**URB-R**			**URB**		
	Fall	Winter	Spring	Fall	Winter	Spring	Fall	Winter	Spring	Fall	Winter	Spring
Discharge (m^3 s^{-1})	0.005	0.010	0.003	0.006	0.005	0.0045	0.002	0.002	0.002	0.022	0.090	0.010
A (m^2)	0.370	0.106	0.089	0.165	0.122	0.139	0.060	0.354	0.070	0.083	0.309	0.099
Reach length (m)	72	106	106	55	84	84	74	74	74	100	100	85
D (m^2s^{-1})	0.005	0.329	0.059	0.086	0.046	0.032	0.011	0.015	0.034	3.75	1.31	8.90
A_S (m^2)	0.389E-04	0.36	0.0198	1.94E-03	1.62 E-03	3.73E-03	0.057	0.029	0.015	1.36	0.036	0.025
α (s^{-1})	0.0001	0.0007	0.0004	0.095	0.002	0.0003	0.0062	0.0032	0.032	0.475	0.0004	0.082
A_S/A	16.45	3.396	0.222	0.012	0.013	0.027	0.950	0.082	0.214	16.386	0.117	0.253

3.2.2 Hydraulic Conductivity and Piezometric Hydraulic Head

Hydraulic conductivity ranges broadly within and between sites (Table 7). The unrestored forested site (FOR) had the most conductive sediments, even compared to its restored counterpart (FOR-R).

Vertical head gradient around structures were measured to identify areas of upwelling and downwelling within each stream (Figure 8). FOR had two debris dam structures, located 19 m and 33 m above the stage gage, which had downwelling above and upwelling below each structure. Upwelling hydraulic gradients in the FOR were smaller than in FOR-R, with maxima of 0.609 and 1.994 respectively. Conversely, the downwelling hydraulic gradients in the FOR were stronger than in FOR-R, with minima of -0.752 and -0.011 respectively (Table 8).

Table 7: Summary of hydraulic conductivity (cm s^{-1}) from the piezometers in each stream reach.

	FOR	**FOR-R**	**URB-R**
Total number of wells per site	37	46	18
Min	5.17E-06	1.03E-06	5.19E-06
Max	0.0292	0.0203	0.0129
Mean	0.0029	0.0007	0.0010
Median	7.88E-04	4.76E-05	6.82E-05
Variance	4.48E-05	8.98E-06	9.28E-06

FOR-R has mostly upwelling identified both upstream and downstream of structures. Upwelling above the structures, which ranged from 0.051 to 0.20, was smaller than the upwelling below the structures, which ranged from 0.3 to >0.51. The urban restored stream had areas of downwelling upstream of the top of the grade control structure and upstream of most steps within it. Upwelling was identified below the structure but not below each step, suggesting that water stayed in the subsurface until it reached the bottom of the structure.

Vertical head gradient was measured multiple times throughout the year to compare measurements seasonally. In all three reaches with piezometers, the largest mean and maximum values of upwelling were measured in the winter months (Table 8). The smallest upwelling was measured in fall for the FOR and URB-R while FOR-R stream had its smallest head gradient of upwelling in spring. The smallest downwelling was measured in winter for FOR but in spring for URB-R.

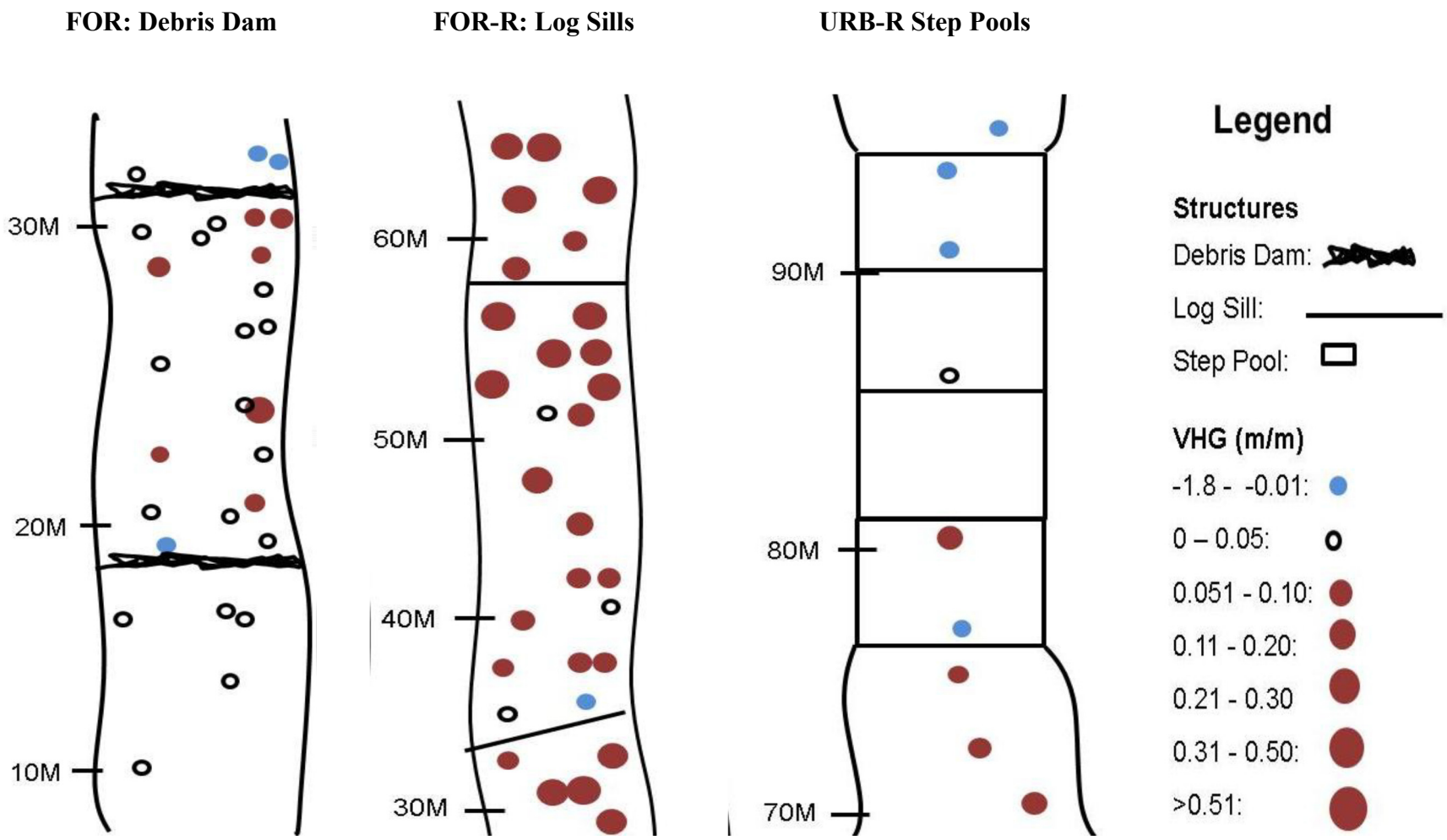

Figure 8: Vertical head gradients measured in Fall 2011 in the forested unrestored (FOR), forested restored (FOR-R), and urban restored (URB-R). Downwellings are indicated in blue and upwelling are in red. There are no data for the urban site (URB) due to the predominance of bedrock.

Table 8: Seasonal variability in hydraulic gradients at the forested unrestored (FOR), forested restored (FOR-R), and urban restored (URB-R) sites in Beaverdam Creek Watershed. There are no data for the urban site (URB) due to the predominance of bedrock.

	FOR			FOR-R			URB-R		
	Fall	Winter	Spring	Fall	Winter	Spring	Fall	Winter	Spring
Minimum	-0.725	-0.272	-0.592	-.0002	0.140	-0.011	-0.055	-1.783	-0.006
Maximum	0.007	0.599	0.610	0.00003	1.994	0.018	0.175	3.759	0.023
Mean	-0.031	0.063	0.053	0.000	0.951	0.002	0.015	0.213	0.003
Median	0.009	0.040	0.019	0.000	0.941	-0.0004	0.003	0.123	0.001

3.2.3 Transient Storage Metrics

The storage zone exchange coefficient (α) governs the rate in which water moves in and out of pools, eddies and hyporheic zones. FOR had the smallest storage zone exchange coefficient while URB had the highest storage zone exchange coefficient, despite the two streams having similar As/A values. The difference in exchange coefficients likely reflects the different mechanisms of storage in the two streams. As described above, URB had no hyporheic exchange because it sat directly on bedrock, while FOR had an alluvial streambed, with significant hyporheic exchange. When water enters the hyporheic zone it has to flow through sediment, which decreases velocity and potentially decreases the storage zone exchange coefficient. The two restored sites had a similar total storage zone exchange coefficients.

Combining the total storage area and exchange coefficient to calculate a total storage zone flux (Table 2; Table 9; q_s), we found that the bedrock URB site had the greatest storage flux and the FOR site had the lowest storage flux. The two restored streams had similar storage fluxes, and all results follow the same patterns as described for the exchange coefficient described above. We interpret these results to suggest that in-channel transient storage results in greater fluxes than hyporheic exchange and that stream restoration acts to produce similar transient storage dynamics even when different styles of structures are applied.

The hyporheic exchange flux (q_{he}) showed significant differences across sites and seasons (Table 9). Averaged across seasons, FOR-R had slightly greater hyporheic exchange flux than FOR and URB-R, but the hyporheic exchange flux showed strong seasonal variations in the restored reaches. In fall and spring, FOR had the greatest hyporheic exchange flux, but in winter FOR-R and URB-R were higher. Interestingly, there does not appear to be any correlation between discharge and hyporheic exchange flux in a given stream, so these seasonal variations require further investigation. No piezometers could be installed because of the extensive bedrock in the URB reach, so we assume that hyporheic exchange was negligible.

The ratio of the total storage exchange flux to hyporheic exchange flux (q_s/q_{he}) reflects the amount of a streams transient storage than can be attributed to the hyporheic zone. The hyporheic exchange flux can also be subtracted from the total storage exchange flux to calculate a channel storage exchange flux (q_{cs}). These measures integrate the various metrics described above to represent both the total amount of transient storage and the relative partitioning between in-channel and hyporheic transient storage.

Total storage flux and percent hyporheic flux varied among seasons for each of the streams. FOR and URB-R had the highest percent of hyporheic storage flux when the total storage flux was the smallest, and the smallest percent hyporheic storage flux in the seasons that had intermediate storage flux measurements. The FOR-R had the highest percentage of hyporheic flux relative to total storage flux in the winter, which was the intermediate total storage flux measurement and the smallest percent of hyporheic storage during the lowest total storage.

Table 9: Summary of exchange and storage parameters used in this study.

	FOR			FOR-R			URB-R			URB		
	Fall	Winter	Spring	Fall	Winter	Spring	Fall	Winter	Spring	Fall	Winter	Spring
Q_{HE} (m^3s^{-1})	1.84E-04	2.25E-04	6.74E-04	3.32E-08	1.49E-03	1.84E-05	6.89E-06	4.65E-04	1.33E-06			
Lpz (m)	78.54	78.54	78.54	82.39	82.39	82.39	99.53	99.53	99.53			
q_{he} ($m^3\ s^{-1}\ m^{-1}$)	2.34E-06	2.86E-06	8.58E-06	4.03E-10	1.81E-05	2.23E-07	6.92E-08	4.67E-06	1.38E-08			
q_s ($m^3\ s^{-1}\ m^{-1}$)	3.73E-05	7.22E-05	3.91E-05	1.56E-02	2.70E-04	3.94E-05	3.74E-04	1.09E-03	2.24E-03	3.93E-02	1.24E-04	8.04E-03
q_{he}/q_s	0.0626	0.0396	0.2195	2.58E-08	0.0668	0.0057	0.0002	0.0043	0.0000			
q_{cs} ($m^3\ s^{-1}\ m^{-1}$)	3.50E-05	6.94E-05	3.05E-05	1.56E-02	2.52E-04	3.92E-05	3.74E-04	1.09E-03	2.24E-03	3.93E-02	1.24E-04	8.04E-03
q_{cs}/q_s	0.9374	0.9604	0.7805	1.0000	0.9332	0.9943	0.9998	0.9957	1.0000	1.0000	1.0000	1.0000

3.3 Macroinvertebrates

Macroinvertebrates were present at all sites and dates and represented a range of diversity including both insect larvae and non-insect invertebrates (including microcrustaceans, crayfish, and oligochaetes). Overall, URB-R had the lowest numbers and richness of both total taxa and EPT taxa (Table 10). Seasonal differences were striking where the taxa and EPT abundance was highest during fall at the forested (FOR and FOR-R) sites and during winter at the urban (URB-R and URB) sites (Table 10). When comparing within a given landscape, both the FOR and FOR-R were similar for both total abundance and richness; however, FOR-R had slightly greater abundances of EPT taxa although similar richness. During fall, FOR-R had higher abundances of similar taxa and interestingly, included a high number of crustacean taxa such as *Holopedium* and *Daphnia*. Both urban sites had very low EPT richness; however, URB had high abundances of the few taxa that were present. When comparing the urban sites, the unrestored site (URB) had higher abundance and richness for both total taxa and the EPT taxa.

Table 10: Summary of taxa and EPT richness, and tax and EPT abundance from fall 2011 to spring 2012.

	FOR			**FOR-R**			**URB-R**			**URB**		
	F	W	Sp	F	W	Sp	F	W	Sp	F	W	Sp
Taxa richness	16	8	16	18	12	19	6	14	10	17	15	22
EPT richness	6	2	3	7	3	2	0	1	1	2	4	3
Taxa abundance	49	28	46	102	34	49	10	60	27	36	72	59
EPT abundance	19	11	5	38	12	11	0	1	3	2	24	14

Regardless of site and season, Dipteran larvae were the most common taxa (Figure 9). Whereas all EPT taxa were rare at URB-R, Trichopterans did occur at both the URB-R (spring) and URB sites (all seasons). In comparison, the EPT were more common at both forested sites and a seasonal separation was apparent with the Plecopterans dominating EPT taxa in the fall, Ephemeropterans during winter and Trichopterans during fall. Overall, taxa such as the snails/clams (*Bivalvia*) and worms (*Oligochaetes*) occurred at all sites but were most common in the urban sites (Figure 9) whether restored or unrestored.

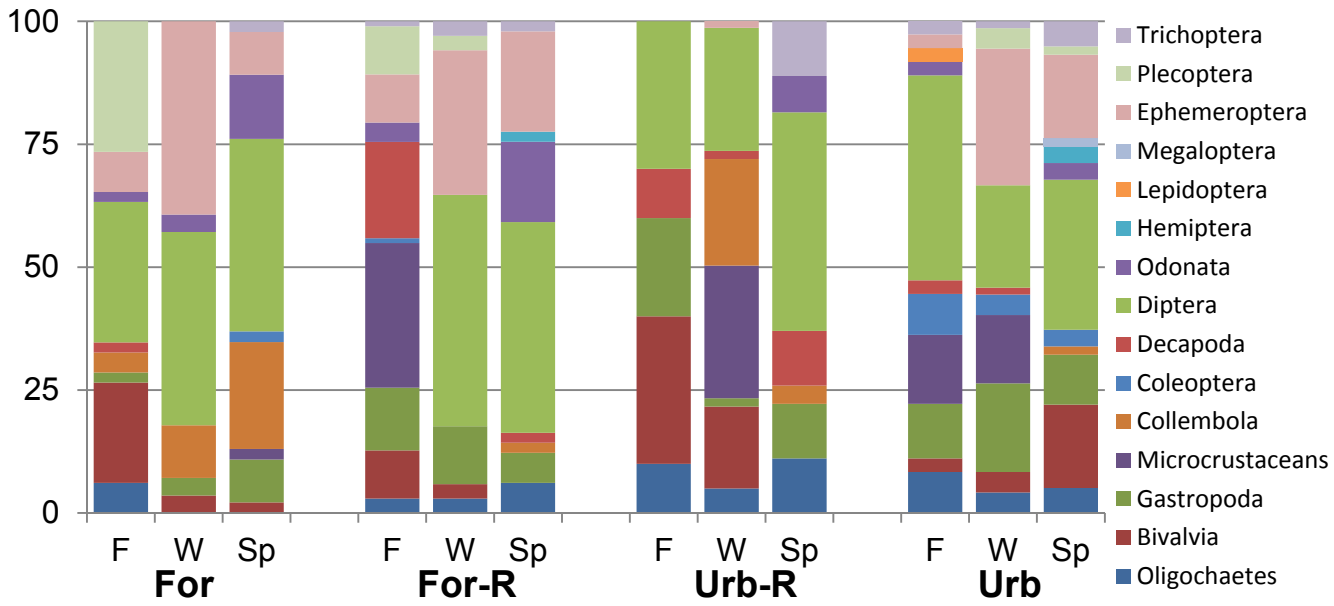

Figure 9: Seasonal changes in the percent composition of taxa from fall 2011 to spring 2012.

We can use changes in EPT abundance (Figure 10) and richness (Figure 11) across sites and dates to investigate the overall "health" of these sites since these taxa are considered to be sensitive to ecosystem alteration from urbanization. Overall, the EPT represent a greater proportion of the total invertebrate abundance during fall and winter at the forest sites compared to the urban sites. During winter at URB, EPT abundance was high and relatively similar to the forest sites and included the Plecoptera. While EPT did occur at the URB-R site, their contribution to overall abundance was always <10% and was zero during the fall sampling.

EPT richness as a function of the total community (Figure 11) was highest during fall in both FOR and FOR-R. During winter and spring, %EPT richness was relatively similar across all watersheds expect URB-R where it was low overall.

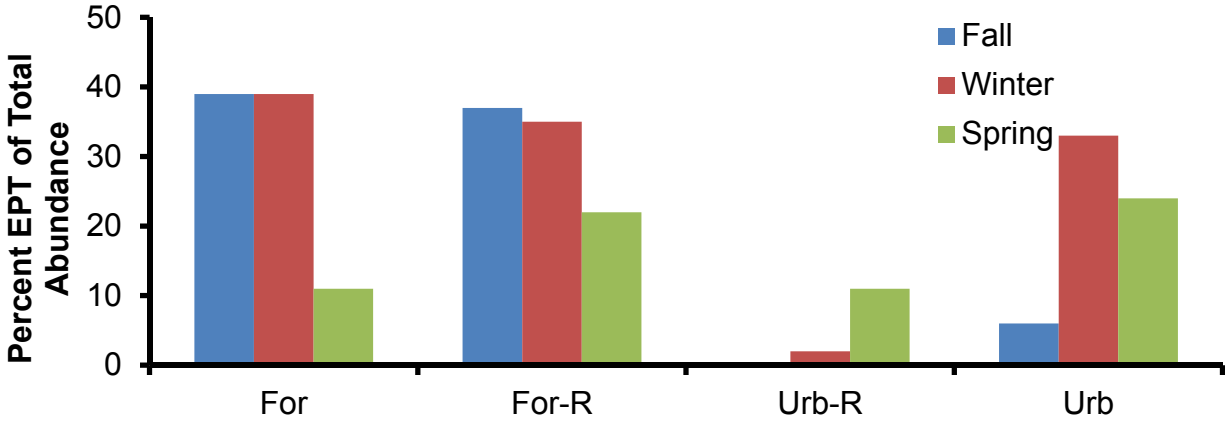

Figure 10: Changes in the percent abundance of EPT taxa relative to total abundance.

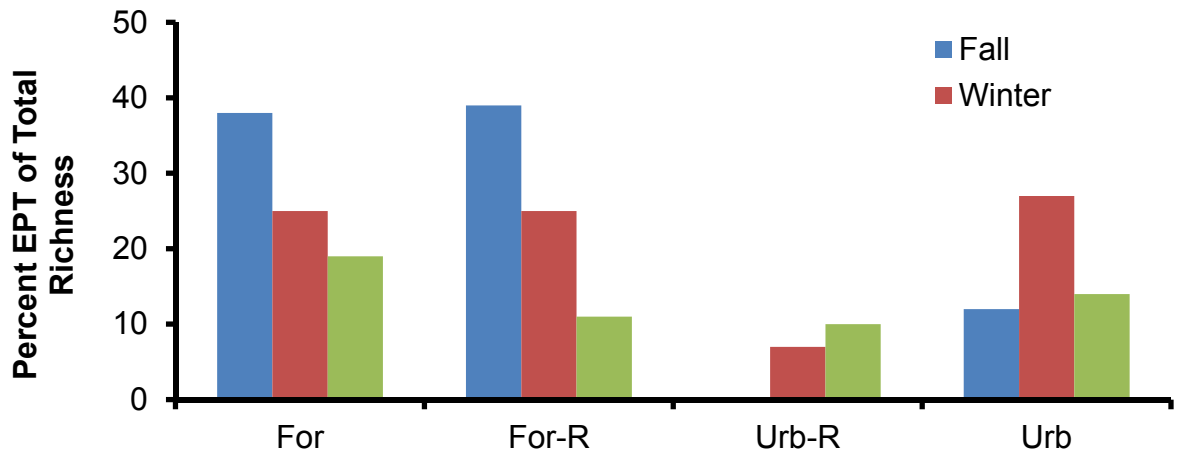

Figure 11: Changes in the percent richness of EPT taxa relative to total richness.

There was no significant relationship between the amount percent hyporheic exchange flux for either total/EPT abundance (Figure 12A) or richness (Figure 12B). While we predicted that sites with higher hyporheic exchange would have higher macroinvertebrate abundances and diversity this pattern was not supported in these sites.

A. B.

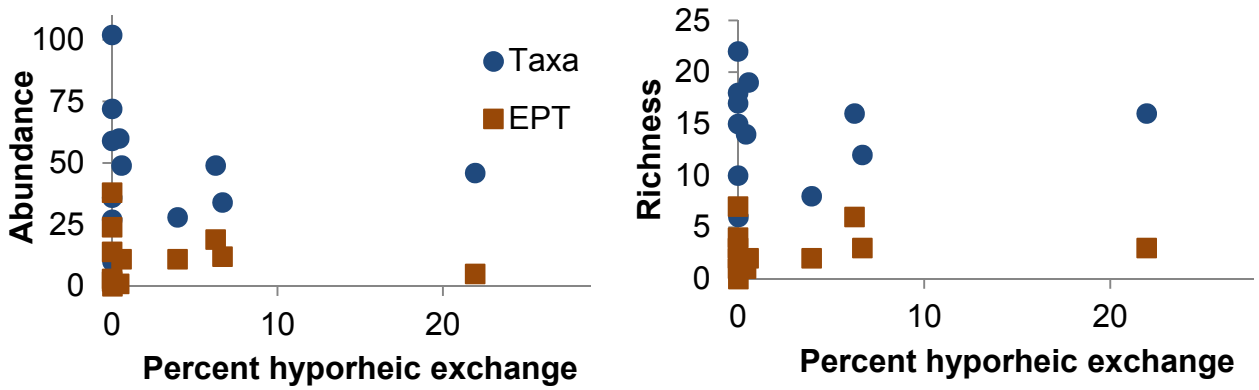

Figure 12: (A) Variability in total taxa/EPT abundance and (B) richness with change in percent hyporheic exchange across the 4 sample reaches (FOR, FOR-R, URB-R, URB).

At this reach scale, a more important variable in explaining macroinvertebrate patterns was changes in canopy cover (Figure 13 A, B). Overall, total taxa abundance (Figure 13A) and richness (Figure 13B) were higher in reaches with higher canopy cover (<80%). These reaches tended to be the unrestored reaches (FOR and URB) that had low (URB) to medium (FOR) percent hyporheic flux.

A. B.

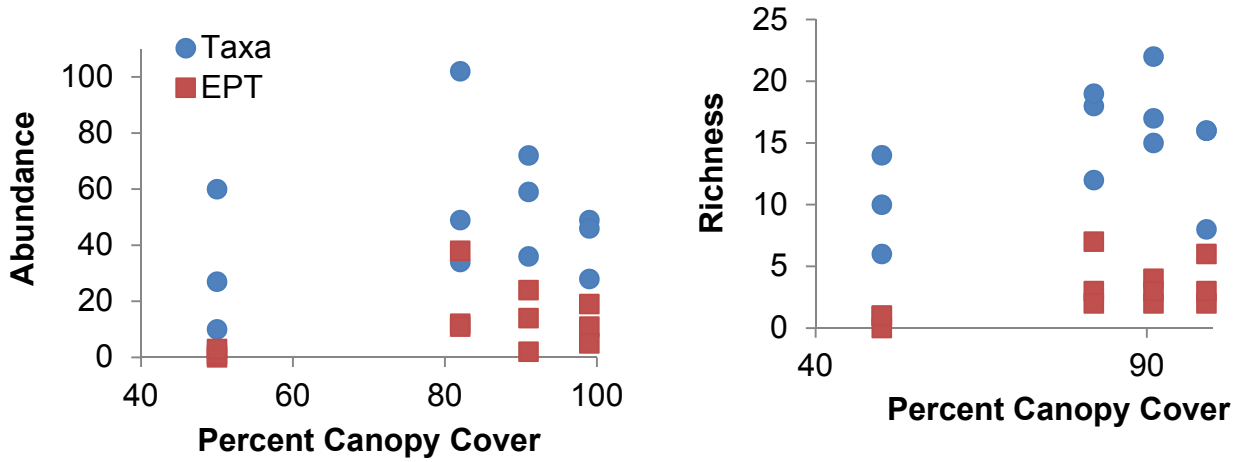

Figure 13: (A) Variability in total taxa and EPT abundance and (B) richness with changes in percent canopy cover across the 4 sample reaches (FOR, FOR-R, URB-R, URB).

3.4 Water quality and Temperature

3.4.1 Water quality

The long term changes in water quality from the Beaverdam Creek watershed can be found in greater detail in the recent research summaries (Allan et al. 2013; Gagrani 2013). We have focused our analysis on nitrate and total phosphorus since they were determined from the previous work to be the most ecological relevant to this study. For the 2011-21012 water year, nitrate yield (kg ha^{-1}) was 0.24, 0.09, 0.18, and 0.27 for the FOR, FOR-R, URB-R and URB watersheds. For that same time period, total phosphorus yield (kg ha^{-1}) was 0.23, 0.09, 0.34, and 0.49 respectively. Overall, total phosphorus yields over the long-term study were higher in the urban watersheds compared to the forested watershed with cumulative export 2.6-4.7 greater from the urban watersheds (Allan et al. 2013).

Within these watersheds reaches associated with the restoration (approximately equivalent in location to this study's FOR-R and URB-R sites) and smaller reaches in the unrestored watersheds (approximately equivalent in location to this study's FOR and URB sites) were monitored in greater detail to examine the effect of restoration on nutrient retention and/or export during baseflow and stormflow conditions. Data were collected during both the dormant (D) and growing (G) seasons. Greater details of these methods can be found in Allan et al. 2013 and Gagrani 2013. These data are summarized for nitrate (Figure 14) and total phosphorus (Figure 15) where positive numbers indicate net retention and negative number indicate net export. Since these data were collected in the previous water year (2010-2011) to this study, we have chosen to associate these nutrient patterns with our transient storage data instead of analyzing using statistical methods.

A. B.

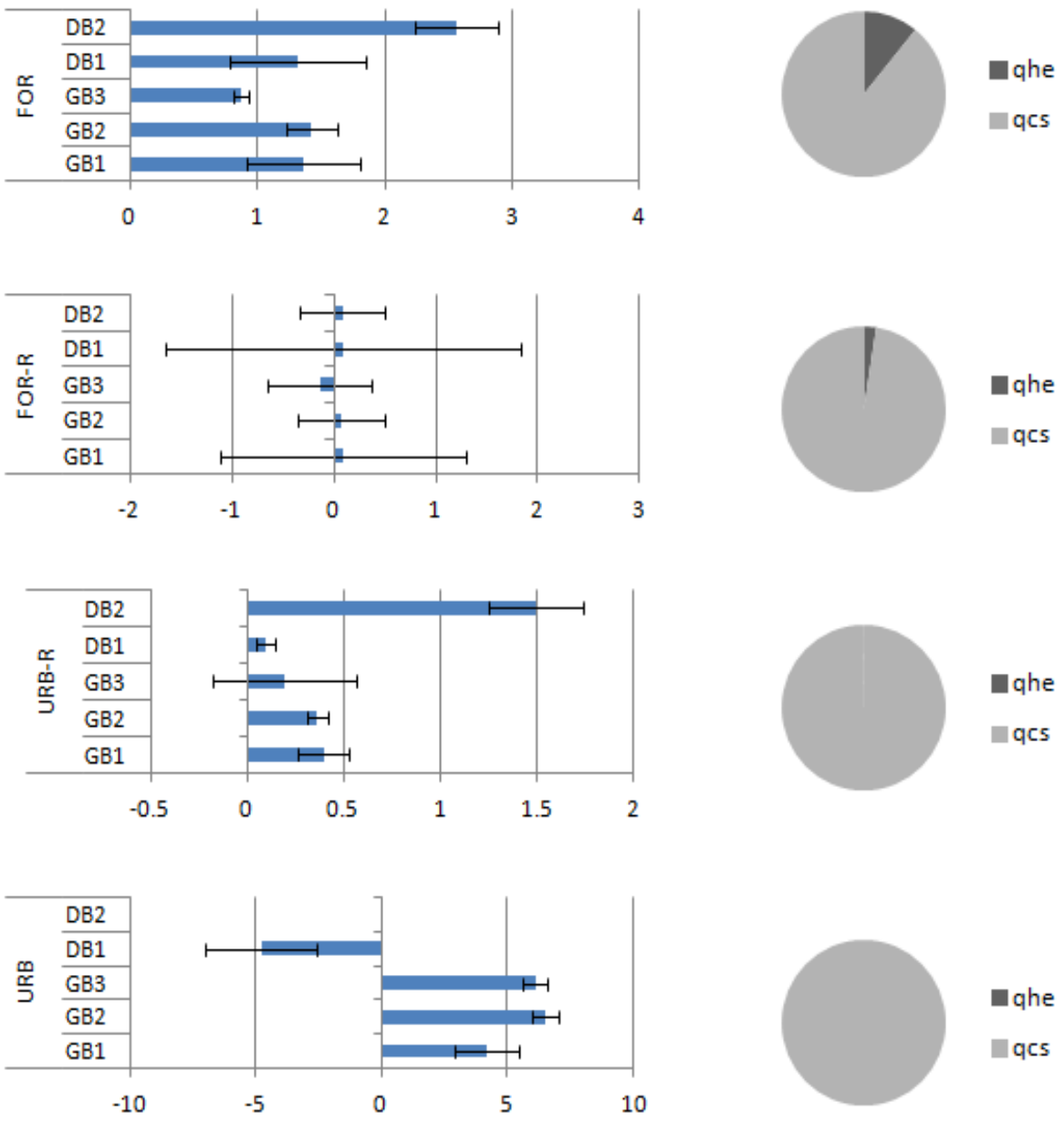

Figure 14: (A) Nitrate export (mg m^{-1} hr^{-1}) from subreaches similar in size and location to FOR, FOR-R, URB-R, and URB in the Beaverdam Creek Watershed. D = dormant season. G=growing season. Data were collected during the 2010-2011 water year (Allan et al. 2013; Gagrani 2013) where positive numbers indicate net export and negative numbers indicate net retention. (B) Percent of total transient storage as hyporheic exchange flux (q_{he}) and channel exchange flux (q_{cs}). Transient storage data were collected during the 2011-2012 water year during this study.

A. B.

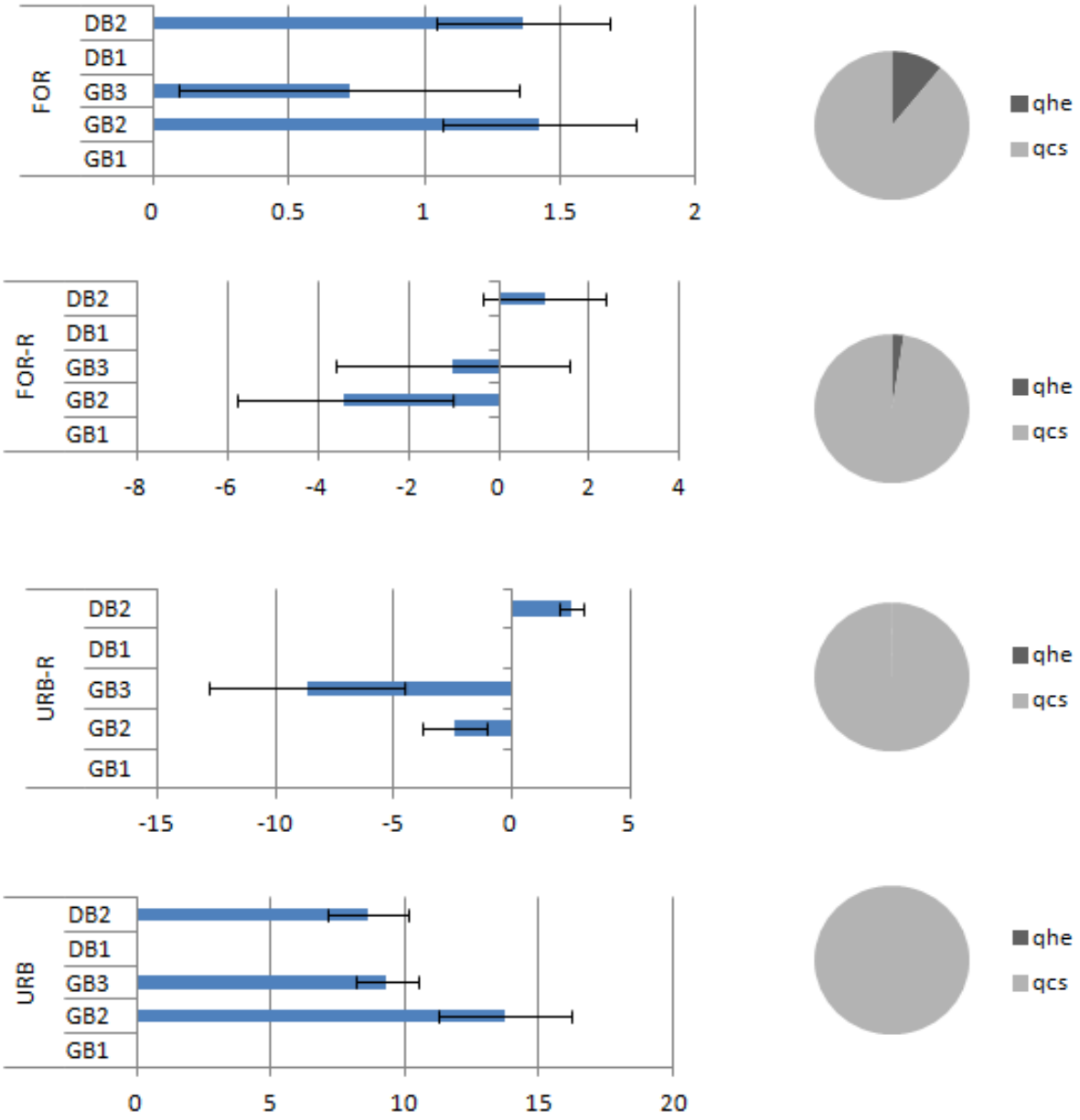

Figure 15: (A). Total phosphorus export (mg m^{-1} hr^{-1}) from subreaches similar in size and location to FOR, FOR-R, URB-R, and URB in the Beaverdam Creek Watershed. D = dormant season. G = growing season. Data were collected during the 2010-2011 water year (Allan et al. 2013; Gagrani 2013) where positive numbers indicate net export and negative numbers indicate net retention. (B). Percent of total transient storage as hyporheic exchange flux (q_{he}) and channel exchange flux (q_{cs}). Transient storage data were collected during the 2011-2012 water year during this study.

Overall, nitrate was exported from all sites regardless of season (dormant versus growing) with the highest rates in URB (Figure 14A). While this site had the greatest total transient storage, the storage was all in channel (Figure 14B). In contrast, URB-R had lower export rates compared to URB and higher percentage of hyporheic exchange flux. Interestingly, FOR-R had relatively low export and retention and was the site but had less hyporheic exchange flux than the FOR site.

Unlike nitrate, total phosphorus (Figure 15A) was exported from the unrestored sites (FOR and URB) and overall retained in the restored sites (FOR-R and URB-R). Similar to nitrate, the patterns of retention were not clearly related to either total transient storage or the percent hyporheic exchange flux (Figure 15B).

3.4.2 Temperature

Stream temperature is a function of the energy balance in the water column, and combines both atmospheric and hydrologic influences. Solar radiation is typically the dominant energy input to a stream, causing stream temperature to vary both spatially (as more radiation is received) and temporally (diurnal and seasonal cycles) (Poole and Berman 2001). Because of the close proximity of the four study sites, we assume that atmospheric variables that influence stream temperature are the same across sites. These include precipitation, air temperature, wind speed, solar angle, cloud cover, and relative humidity. Our study reaches were selected to exclude any tributaries, so we do not have to factor tributary flow and temperature into our understanding of stream temperature dynamics in our study. We did not explore the effects of topographic shading, but given the generally gentle topography of the Piedmont, we do not anticipate it to have a major effect at either the cross-site or within-stream scales.

Of the factors thought to influence stream temperature, our study design suggests vegetative shading, hyporheic exchange, in-channel transient storage, and groundwater inputs as the potential explanatory variables for temperature differences. Each of these factors could be influenced by stream restoration or watershed land use. Vegetative shading was quantified with densitometer measurements of canopy cover (see Section 3.1.1 for details), and higher amounts of shading typically lower maximum daily stream temperatures, but do not change mean or minimum daily stream temperatures (Johnson 2004). Hyporheic exchange has been shown to produce a range of possible stream temperature responses, which have been subdivided into "buffering" (reducing the diurnal or seasonal temperature range), "lagging" (increasing the wavelength of the diurnal or seasonal temperature cycle), and "cooling" (decreasing the mean temperature)(Arrigoni et al. 2008). In-channel transient storage could have the effect of increasing mean temperature, through additional time for radiative energy inputs, or buffering, by releasing warmed water from in-channel storage zones during cooler periods. However, we are unfamiliar with any previous study that has quantified the effects of in-channel storage on headwater stream temperatures. We assume that groundwater inputs have the same temperature across sites (~mean annual air temperature), but we recognize that the amount of groundwater upwelling within or just upstream of our study reaches may influence stream temperature dynamics by buffering diurnal or seasonal temperature signals.

In order to understand the effects of restoration and watershed land use, temperature probes were installed at the upstream and downstream ends of each reach. In FOR, FOR-R and URB (the URB-R probe was lost during the study), temperature probes were also installed in the middle of the reach. We focus on the daily time step (12:00 am to 11:55 pm) and define the following

parameters of interest: the maximum daily temperature (T_{max}), the mean daily temperature (T_{mean}), and the daily temperature range (T_{range}).

Temperature probes at the upstream end of each study reach were used to compare across study sites, because they offered the greatest number of common days in the record. Even though the comparison sites are at the upstream ends of the study reaches, watershed conditions and restoration conditions are similar throughout the 100 m the study reaches. There were 128 days in water year 2012 in which all four upstream probes recorded all measurements and all of the streams had water. Of these 128 days, 55 occurred in October-December 2011 (autumn), 61 occurred in March-May 2012 (spring), and 12 occurred in June-September 2012 (summer). In the summer, the number of dates with data across all sites was strongly affected by no flow periods, which were not always consistent between streams.

In the spring, the FOR-R upstream probe appeared consistently anomalously warm relative to the other streams, and comparison of the FOR-R upstream location to the mid-reach location indicates substantial apparent cooling within the study reach. Such a great amount of apparent cooling (2.6°C) over a short distance (45 m) suggests that there may have been problems with data quality at the upstream probe during this period (e.g., probe not fully within the water column at all times). However, even if the FOR-R mid-reach probe is used in replacement of the BD2 upstream probe for this period, FOR-R still had the greatest mean T_{mean}, T_{max}, and T_{range} of any of the streams during the spring (Tables 11, 12, 13).

During the spring and summer, the restored streams had T_{mean} and T_{max} that were warmer than the unrestored streams (Table 11, 12). Urban, restored URB-R had T_{mean} 1.1°C warmer than its urban, unrestored counterpart, while forested, restored BD2 had T_{mean} ≥2.0°C warmer than its forested, unrestored counterpart. During autumn, the T_{mean} and T_{max} were fairly homogeneous across all sites. These temperature differences likely result from differences in leafy canopy cover between the restored and unrestored sites. The restored sites had lower canopy cover than the unrestored sites (Table 3), and spring and summer are during the leaf-on period, while the autumn data were largely collected during leaf off conditions. Greater solar radiation reaches and heats the streams with lower canopy cover than the sites with higher canopy cover, leading to warmer stream temperatures. In the autumn, once the leaves have fallen, canopy cover makes little or no difference in the solar radiation received by the stream, and the cross-site temperature differences are minimal.

The seasonal mean T_{range} values do not show a consistent pattern across seasons or between streams. In the urban watersheds, T_{range} values were within 0.6°C of each other, each season, whereas the forested watersheds tended to have greater differences in their temperature ranges. In neither case, however, did the restored stream consistently exhibit a smaller or larger temperature range than the unrestored stream. The seasonally inconsistent cross-site patterns of T_{range} may in part be due to the influence of increased solar radiation on the restored streams, as described above. However, there was also significant variation in daily minimum temperatures (i.e., nighttime temperatures) across sites that cannot be explained by differences in canopy cover or residual heat energy from daylight hours. The factors controlling T_{range} in small headwater streams seem to require greater investigation. This finding echoes that of Johnson (2004).

Table 11: Seasonal mean T_{max} (°C) at the upstream end of each study reach. For FOR-R, the mid-reach location is shown (in parentheses) in spring to show the greater temperature variability along the reach than was found in the other sites.

	FOR	FOR-R	URB-R	URB
Autumn (n=55)	12.5	12.6	12.7	12.4
Spring (n=61)	18.4	23.6 (20.6)	19.3	18.5
Summer (n=12)	22.0	24.8	24.2	23.2

Table 12: Seasonal mean T_{mean} (°C) at the upstream end of each study reach. For FOR-R, the mid-reach location is shown (in parentheses) in spring to show the greater temperature variability along the reach than was found in the other sites.

	FOR	FOR-R	URB-R	URB
Autumn (n=55)	10.9	11.6	11.1	10.8
Spring (n=61)	16.8	21.5 (18.8)	18.3	17.2
Summer (n=12)	21.1	21.9	22.9	22.3

Table 13: Seasonal mean T_{range} (°C) at the upstream end of each study reach. For FOR-R, the mid-reach location is shown (in parentheses) in spring to show the greater temperature variability along the reach than was found in the other sites.

	FOR	FOR-R	URB-R	URB
Autumn (n=55)	3.3	1.9	2.9	3.2
Spring (n=61)	3.1	4.1 (3.6)	2.1	2.7
Summer (n=12)	1.7	4.7	2.3	1.7

4 DISCUSSION

4.1 Transient Storage

Our results suggest that in low gradient, sand-bed alluvial streams, like those found throughout the North Carolina Piedmont (Kolberg and Howard 1995), standard stream restoration practices may be ineffective at creating hyporheic exchange, but can significantly increase in-channel transient storage in pools. Conversely, in steeper reaches with thin or absent alluvial cover, like those in the urban watersheds, stream restoration may create some hyporheic exchange where little or no hyporheic exchange existed prior to restoration.

Differences in total storage between the restored and the unrestored sites were not seen. The hyporheic storage was found to be a small percentage of total transient storage for all four study reaches. These results disagree with previous studies by Bukaveckas (2007) and Gooseff et al. (2005). Bukaveckas (2007) found that transient storage in restored Indiana and Kentucky

agricultural streams had significantly higher normalized storage zone area than the channelized streams used as a comparison. However, that study used channelized streams with very few pools, while, in this project, the forested unrestored site has multiple debris dams creating pools and the urban unrestored site included large pools created by the bedrock form and boulders found in the stream reach. Gooseff et al. (2005) measured in-stream transient storage in bedrock and alluvial reaches of an Oregon mountain stream. In that study they found that the bedrock reach had less transient storage than the alluvial reach, but that the alluvial reach's storage behavior related to hyporheic flow while the bedrock reach's storage related to in-channel eddy exchange.

Among the three field sites, hyporheic flux seems to be driven by different factors, depending on the stream sediment size. Spatial variation in hyporheic flux in FOR were driven by variability in vertical head gradients due to structures, such as a down log or a debris dam where there is downwelling, while below a structure there is upwelling. When two structures were in close proximity to each other, downwelling is measured above both structures but is not as prevalent above the downstream structure, while upwelling is seen at both of these structures was seen in FOR-R at the debris dam between 18 and 33 m upstream from the stream gauge. In FOR-R, hyporheic flux appears to be limited by the small variability in vertical head gradients associated with the restoration structures and by the low hydraulic conductivity of the sediments. In the URB-R, which also has a smaller grain size compared to the FOR, spatial variability in hyporheic flux has a direct relationship with variability in hydraulic conductivity. This relationship appears to be stronger than that between head gradients and hyporheic flux. In the URB-R, downwelling only occurred upstream of a very large grade control structure, 75 m to 95 m above the stage gage, while other parts of the study reach only experienced upwelling. In summary, streams with fine or compacted sediments or bedrock appear to have hyporheic flux defined by variations in hydraulic conductivity, whereas streams with high hydraulic conductivity have their hyporheic flux driven by vertical head gradients found around in-stream structures.

Most measurements in this study indicated that hyporheic exchange flux was ≤6.25% of total storage flux. The methods used in this study are similar to Stofleth et al. (2008) who calculated hyporheic flow in small sand bed streams in Mississippi. They quantified that hyporheic exchange only accounts for a small percentage of the total storage (<5%) and did not increase with obstructions, but the obstructions did increase the total surface storage. Our study supports the finding that structures did not change the percent hyporheic storage and that hyporheic flow was a small portion of the total storage. Kasahara and Hill (2006) found that low channel gradient streams had hyporheic exchange flux as a percent of total transient storage of 0.005% to 0.08%, which is smaller than the hyporheic flow in streams in the Beaverdam Creek Watershed. This was related to the finer substrate that reduced the hydraulic conductivity and restricted hydraulic head gradients. Kasahara and Hill (2006) compared their study to a gravel bed mountain stream and found the alluvial, low gradient streams have smaller overall storage in the hyporheic zones. In comparison, using OTIS-P, Harvey et al (1996) determined that the hyporheic exchange was around 40% to 80% of the total storage in a mountain stream in Colorado.

We interpret our results, similar to other low gradient streams, to suggest that in-channel transient storage results in greater fluxes than hyporheic exchange and that stream restoration acts to produce similar transient storage dynamics even when different styles of structures are

applied. The differences seen between the FOR and FOR-R are most likely related to the difference in hydraulic conductivity restricting subsurface flow in the forested restored site.

4.2 Ecosystem Metrics and Transient Storage

Regardless of parameter (macroinvertebrates, nutrients, temperature) there was no significant relationship with any transient storage parameter (surface or subsurface). Overall, macroinvertebrate metrics and temperature were more closely controlled by variability in canopy cover among the study reaches. There was a potential relationship between restoration and total phosphorus retention; however, we were not able to correlate this pattern with hyporheic exchange flux.

Macroinvertebrate abundance and richness were comparable to other urban streams (Violin et al. 2011) in the Piedmont and similar to data collected by Allan et al. (2013) as part of a long-term water quality study in Beaverdam Creek. Differences in metrics between our results and Allan et al. (2013) are most likely due to the level of taxonomic description (we described to family not species) and the timing of sampling events. Overall, there was no relationship between the 4 macroinvertebrate metrics and transient storage – hyporheic or surface. While the percent EPT richness and abundance (of total richness and abundance) was higher with increased percent hyporheic flux (of total channel flux), sites with no hyporheic storage (URB) also had similar values. Macroinvertebrates respond to several parameters such as flow, habitat, and sediment transport; thus, it is difficult to determine the importance of groundwater-surface water interactions on the benthic community.

At the landscape level, the watersheds differed in percent canopy cover (see Section 3.1.1 for details) which may have an impact on macroinvertebrate response to restoration. During restoration, canopy cover is often removed and at our sites canopy cover was lowest at URB-R which also had the lowest EPT taxa richness and abundance. In forested systems, leaf litter inputs are important for structuring and maintaining macroinvertebrate communities (Wallace and Webster 1996; Wallace et al. 1997). When compared to percent canopy cover, there was a strong increase in the percent EPT (richness and abundance). Thus, URB-R, while having lower percent hyporheic flux compared to the forested sites, had higher hyporheic flux than the URB but lower macroinvertebrate diversity. We attribute this difference to the lower canopy cover in URB-R than URB. Thus, it is important to consider landscape attributes when determining the control on macroinvertebrate communities.

Long-term water quality dynamics in the Beaverdam Creek watershed have been detailed by Allan et al. (2013). Overall, increased material flux from the watersheds is related to the increased water yield from the conversion of forested to urban watersheds and fertilizer application in the URB-R and URB watersheds (Allan et al. 2013). Researchers have demonstrated that restoration increases (Buckaveckas 2007; Richardson et al. 2011; Gagrani 2013) or has little effect (Roberts et al. 2007; Sudduth et al. 2011) on in-stream nutrient retention. Using a mass-balance approach, Gagrani (2013) showed that restored reaches at Beaverdam Creek had increased nutrient retention compared to unrestored upstream reaches during summer. Unfortunately, we were unable to correlate these results to our transient storage measurements since low flow conditions during summer prevented us from measuring channel transient storage.

4.3 Relative success of restoration in the landscape context

In the forested alluvial stream reaches, restoration did not appear to significantly change the hyporheic exchange flux, but it did significantly increase total storage flux, so it increased surface transient storage substantially. The reason for lack of change in the hyporheic exchange fluxes is probably because natural drivers of hyporheic exchange (e.g., debris jams, bars) were removed but replaced by restoration structures that promoted the same amount of hyporheic exchange. However, despite producing little change in hyporheic exchange, restoration substantially modified the total transient storage flux by a significant increase in pools, eddies, and other in-channel transient storage zones. In the urban reaches, which had greater bedrock exposure in the channel, restoration significantly enhanced hyporheic exchange, even though it was still small relative to the forested, alluvial reaches Restoration in the urban watersheds did not enhance in-channel transient storage fluxes, because such fluxes were already large in the unrestored stream. While these findings could be interpreted in terms of land use, we suggest that the alluvial versus bedrock nature of the stream bed is of greater and more direct importance in explaining the effects of stream restoration on the amount and type of transient storage.

The flux calculations demonstrate that FOR, FOR-R and URB-R all have relatively similar hyporheic flux (~10^{-6} m^3 s^{-1} m^{-1}). The FOR-R has a slightly greater hyporheic flux compared to FOR, but some of that flux could be accounted for by the downwelling estimated by the kriging methods that overestimate the channel width (Osypian 2013). The FOR reach had the highest q_{he}/q_s, averaging ~ 11% (Table 9). FOR-R was found to have 2% average q_{he}/q_s, while URB-R had an average of 1.5%. The lower percent of hyporheic fluxes in the restored reaches could be the result of the increased pools created from structures placed in the stream during the restoration.

5 CONCLUSIONS AND RECOMMENDATIONS

Common goals of restoration are to create in-stream habitat, create a diverse channel form, increase the riparian zone and improving recreational purposes, the hyporheic zone is not taken into consideration. The hyporheic zone should be a consideration in restoration designs as it provides associated stream functions when it is present. Hyporheic zones provide habitat, promote nutrient cycling, buffer against pollutants, and moderate stream temperature. To enhance a stream's hyporheic zone, restoration techniques may include adding diverse morphologic features, large wood, coursing the sediment and replanting the riparian zone (Hester & Gooseff, 2010). While stream restoration does not restore the stream back to its original conditions, these practices have the ability to improve streams that have been degraded and increased hyporheic storage can add an additional dimension to these practices. Currently however, there are few studies that quantitatively compare hyporheic flux in restored and unrestored streams and the effect of that flux on ecological parameters.

In this study we asked if stream restoration in urbanizing and forested watersheds enhance transient storage and whether this enhancement has a similar impact on ecological parameters. We hypothesized that watershed land use (forested versus urban) affects the absolute and relative success of stream restoration. We predicted that the greatest impact of restoration would occur in urban streams (URB< URB-R) whereas comparing forested and urban restored streams would yield less improvement (FOR-R > URB-R) due to the mitigating impacts of the less disturbed

watershed. Our predictions were mostly upheld however, the role of hyporheic exchange flux was smaller than expected.

We found that URB-R had higher total transient storage flux, greater percentage of that flux as hyporheic exchange flux, higher macroinvertebrate abundance and diversity, lower nitrate export and higher total phosphorus retention compared to URB. Thus, restoration of urban streams does increase stream-groundwater interactions and potentially enhances ecological condition when compared to other unrestored urban streams. When FOR-R and URB-R sites were compared we found that FOR-R had higher total transient storage flux, greater percentage of that flux as hyporheic exchange flux, higher macroinvertebrate abundance and diversity, relatively neutral nitrate export and retention, and greater total phosphorus retention compared to URB-R. Thus, restoration of urban forested sites can potentially yield greater impacts on stream-grounwater interactions and ecological condition than restoration efforts in similar non-forested urban watersheds.

Overall, in this study, restoration did not increase hyporheic exchange flux in the restored watersheds. When averaged across seasons, restoration decreased the percent hyporheic exchange flux (of total transient storage flux) in the forested watersheds from 10% to 2% while there was a slight increase through restoration in the urban watersheds from 0% to 0.15%. We propose that these results are heavily influenced by both restoration design and construction practices. For example, restoration in URB-R was dominated by large grade control structures at the upstream end of the reach that contributed to in-channel storage but did not enhance hyporheic storage. In contrast, the FOR-R site had several locations of placed log sills that influenced movement of water into an out of the sediments. Regardless of these locations of potential head in FOR-R, the overall low hydraulic conductivity of the sediments resulted in slower connections between the surface and subsurface environments compared to the unrestored forested site (FOR).

A key question leading from this work however, is the role of varying types of transient storage (surface versus hyporheic) on ecosystem parameters such as nutrient retention where it is vital to understand the role of in-channel pools versus hyporheic flowpaths in the retention and transformation of nutrients. Furthermore, based on this study, we conclude that larger scale ecological parameters, such as macroinvertebrate communities and temperature, are being controlled by landscape processes such as canopy cover and not smaller stream-groundwater interactions.

Overall, we recommend that restoration design and construction be further improved to enhance hyporheic exchange fluxes in urban streams while maintaining a greater extent of the riparian zone during restoration to preserve in-stream biological communities.

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APPENDIX A: ABBREVIATIONS

A – channel cross-sectional area
As – storage zone area
α – storage zone exchange coefficient
D – stream dispersion coefficient
D50 – median grain size
dh – hydraulic head difference
dl – flowpath length
EPT - Ephemeroptera, Plecotpera, Trichoptera
FOR – forested unrestored site
FOR-R – forested restored site
K – hydraulic conductivity
L_{pz} – piezometer reach length
OTIS - One dimensional Transport with Inflow and Storage model
Q – discharge
q_{cs} – channel storage exchange flux
q_{cs}/q_s - ratio of channel storage exchange to total storage exchange
q_{he} – hyporheic exchange flux
Q_{HE} – total hyporheic exchange flux
q_{he}/q_s – ratio of hyporheic exchange flux to storage exchange flux
q_s – storage exchange flux
TN – total nitrogen
TP – total phosphorus
URB – urban unrestored site
URB-R – urban restored site

APPENDIX B: PUBLICATIONS AND PRESENTATIONS

Thesis

Osypian ML. 2013. Evaluating restoration effects on transient storage and hyporheic exchange in urban and forested streams. Master of Science in Civil and Environmental Program, UNC Charlotte, 66 pages.

Presentations

Jefferson AJ, Clinton SM, and Osypian ML. 2013. Transient storage versus hyporheic exchange in low gradient headwater streams. Annual Meeting of the American Geophysical Union. San Francisco, CA.

Osypian ML, Jefferson AJ, and Clinton SM. 2013. "Evaluating restoration effects on transient storage and hyporheic exchange in urban and forested streams." Annual meeting of the Society for Freshwater Science, Jacksonville, FL. (Special Session: Advances in groundwater and surface-groundwater interactions research)

Clinton SM, Osypian M, and Jefferson AJ. 2013. "Effects of urban stream restoration on transient storage and ecosystem function." Annual Meeting of the American Society of Limnolgy and Oceanography. New Orleans, LA.

Jefferson AJ, Clinton SM, Osypian ML, McMillan SK, and Tuttle A. 2013. "Evaluating the success of urban stream restoration in an ecosystem services and watershed context." Upper Midwest Stream Restoration Symposium (plenary/invited talk), LaCrosse, WI.

Jefferson AJ. 2013. "Evaluating the success of urban stream restoration in an ecosystem services and watershed context." Kent State University Department of Biological Sciences, Kent, OH.

Jefferson AJ. 2013. "Evaluating the success of urban stream restoration in an ecosystem services and watershed context." Denison University Department of Geosciences, Granville, OH.

Jefferson AJ. 2013. "Evaluating the success of urban stream restoration in an ecosystem services and watershed context." North Dakota State University Department of Geosciences, Fargo, ND.

Osypian ML, Jefferson AJ, and Clinton SM. 2012. "Evaluating restoration effects on transient storage and hyporheic exchange in urban and forested streams." Annual Meeting of the American Geological Society of America, Charlotte, NC.

Osypian ML, Jefferson AJ, and Clinton SM. 2012. "Evaluating restoration effects on transient storage and hyporheic exchange in urban and forested streams." Annual meeting of the Society for Freshwater Science, Louisville, KY.

Jefferson AJ. 2012. "Drainage network evolution driven by coupled changes in landscape properties and hydrologic response." The Johns Hopkins University Department of Geography and Environmental Engineering, Baltimore, MD.