

Changes in stream ecosystem structure as a
function of urbanization:
Potential recovery through stream restoration

By

Peter Cada

Date: _____

Approved:

Dr. Emily S. Bernhardt, Advisor

Masters project submitted in partial fulfillment of the
requirements for the Master of Environmental Management degree in
the Nicholas School of the Environment and Earth Sciences of
Duke University

2007

ABSTRACT

I documented reach scale changes in the physical structure of 12 stream channels in the summer months of 2006, comparing four small streams draining forested catchments with eight streams from developed watersheds of similar catchment size. Study sites in four of the urban streams are within recently implemented natural channel design restoration projects. To assess whether restoration projects increase stream habitat and flow heterogeneity and increase water exchange with floodplain and hyporheic sediments I compared reach-scale geomorphic (e.g. slope, cross section, degree of incision, variation in water depth) and hydrologic (e.g. transient storage volume (TS), surface-water groundwater exchange, fine scale variation in velocity) features of each stream. I used ArcGIS to compile watershed maps and to produce detailed maps of reach habitat for each stream, and the hydrologic model OTIS-P to estimate transient storage from field rhodamine releases. Minimally impacted reaches were found to have shallower average depths with a greater variation in depth than urban or restored stream reaches. Streams restored to provide habitat had the lowest flow habitat heterogeneity of the three stream classes. Channel incision was the only physical channel feature for which the urban restored streams were more similar to the forested streams than the urban degraded condition. Surprisingly, I was unable to detect significant differences in transient storage volume or hyporheic exchange between our three stream classes. My results suggest that restoration designs are placing inadequate attention on recreating the physical template seen in less degraded streams.

INTRODUCTION

The urbanization of a watershed degrades both form and function of stream ecosystems, often having immediate and rapid impacts that can be difficult to mitigate or correct (Booth and Jackson 1997). Watershed urbanization leads to dramatic changes in water and sediment fluxes into draining streams including flashier hydrographs, increased erosion, changes in sedimentation patterns (Niezgoda and Johnson 2005; Shields et al. 2003; Sudduth and Meyer 2006), and altered base flow levels (Booth and Jackson 1997; Meyer et al. 2005; Paul and Meyer 2001; Walsh et al. 2005b). Increased channel instability results, with more frequent and dramatic channel morphological changes such as bed scouring (May 1998), sedimentation (McMahon and Cuffney 2000), and channel instability (Shields et al. 2003). Urban streams tend to have elevated concentrations of nutrients and contaminants due to increased inputs from point and non-point sources, decreases in the efficiency of nutrient removal in riparian zones and the stream channel, and reduced base flow levels (Groffman and Crawford 2003; Meyer et al. 2005). This combination of high disturbance frequency, frequently scoured and homogenized bed materials and high nutrient loads leads to lower biotic richness (e.g. Sudduth and Meyer 2006). These impacts have been referred to collectively as the Urban Stream Syndrome (Paul and Meyer 2001; Walsh et al. 2005a), a term which recognizes the fact that all urban streams experience similar severe hydrologic, geomorphic and chemical alterations. While the mechanisms driving Urban Stream Syndrome are obviously complex and interactive, most impacts can be associated with a few major sources, with the most potent source being the delivery of urban storm water runoff via hydraulically efficient drainage systems (Walsh et al. 2005a). Sanitary sewer outflows, wastewater treatment plant effluents, and legacy pollutants can further degrade receiving streams (Walsh et al. 2005a).

Recognition of urban impacts on stream ecosystems has led to increased funding for projects attempting to mitigate urban impacts through active river restoration approaches (Bernhardt et al. 2005; Brown 2000; Carpenter 2003; Clarke 2003; Hassett et al. 2005; Kondolf 1996; Moerke and Lamberti 2004; Palmer et al. 2005; Walsh et al. 2005b). However, reversing harm caused by basin urbanization can be challenging because of the wide range of both local and watershed stressors in these environments and the severe constraints on restoration design options (Bernhardt 2007; Booth 2005). The morphologically based natural stream channel

design method, based on using a channel form to provide a stable stream channel, adequate storm water routing, and improved aquatic habitat, is one of the more common design techniques for restoring disturbed channels (Niezgoda and Johnson 2005; Rosgen 1994). Based on several assumptions (Niezgoda and Johnson 2005), these design methods focus on creating longitudinal slopes, sinuous meander characteristics, and cross-sectional profiles based on a chosen reference channel. This methodology has been criticized for giving inadequate consideration to underlying geomorphic processes, and consequently not recreating self-sustaining systems, but instead requiring continued investment of time and money (Clarke 2003). Clarke et al. observe that an ‘eco-hydromorphic’ approach, where spatial and temporal heterogeneity (fundamental characteristics of fluvial systems) are taken into account, can recreate a framework where sediment transport and nutrient dynamics can occur, allowing a more natural and dynamic restored ecosystem (“ecological success” sensu Palmer et al. 2005).

The relationships between fluvial system geomorphic and hydrologic processes and the ecosystem functions that many granting agencies hope to maximize via restoration (e.g. instream metabolism and nutrient uptake) remains a research frontier. It is generally agreed that streams with longer water residence time, higher hyporheic exchange, and high frequencies of in channel structural materials (large wood and boulders) will be more efficient at trapping organic materials (e.g. debris dams, side pools, hyporheic zone) and removing dissolved nutrients from the water column (Bilby and Likens 1980; Paul and Hall 2002; Wollheim et al. 2001). However, what is not known is the degree to which current restoration efforts can rehabilitate these ecosystem structural or functional properties. Most restoration efforts focus on affecting physical forms with the implicit or explicit goal of improving ecosystem function. Re-creating physical attributes of healthy stream ecosystems that ensure spatial and temporal heterogeneity are integral to develop and maintain species richness (Brooks 1991). Important geomorphological forms necessary to maintain specialized biota richness include riffles [e.g. small fish (Poff 1993) & benthic insects such as adult gyrid beetles (Poff 1993); mayfly nymphs, nematoceran larvae, beetle larvae (Palmer 1990)], rocky bank habitats [e.g. unionid mussels (Poff 1993)], hyporheic zones [e.g. rotifers, copepods, dipterans, nematodes (Palmer 1990)], debris dams [e.g. caddisfly, crane fly (Swank 1988)], and pools [e.g. young-of-year fish, neustonic insects, frog tadpoles, large fish (Poff 1993)].

The first objective of this study was to determine if urbanized streams (urban degraded - UD) have reduced habitat heterogeneity, flow (velocity and depth) heterogeneity, floodplain connectivity, riparian zone canopy coverage, and transient storage (Figure 1) relative to more natural (minimally impacted - MI) streams. The same data was collected and analyzed from stream reaches that have recently been restored in the Piedmont area (urban restored - UR). The objective of the second analysis was to determine if current restoration practices in the region have led to an increase of habitat and flow heterogeneity, and in-channel residence time relative to urban degraded conditions, thereby creating an ecosystem that has the potential to recover whole stream metabolism and ecosystem health. I expected to find the urban degraded streams to have the lowest levels of substrate and flow habitat heterogeneity, as well as the lowest transient storage volumes. The urban restored streams were expected to have higher levels of all metrics than the urban degraded streams, and we hypothesized that most metrics for these streams would be intermediate between the urban degraded and minimally impacted stream values.

METHODS

Site Selection

Sites were selected through consultation with the North Carolina Ecosystem Enhancement Program (EEP). EEP is a state organization with the mission to “restore, enhance, preserve, and protect the functions associated with wetlands, streams, and riparian areas,...”. With that in mind I asked their guidance in selecting their “best” restoration projects in the region. Urban degraded sites were selected because they either had similar watershed size and location, and/or they were slated for restoration in the near future. Sites were selected in this manner with the goal of determining the potential for restoration efforts to restore the physical structure of stream ecosystems, thus creating opportunities for ecosystem function recovery.

Site Descriptions

Our study included 12 sites located in the Raleigh-Durham-Chapel Hill area in the Piedmont of North Carolina. Of the 12 study sites four were small streams draining forested catchments (minimally impacted) with eight streams from developed watersheds. Study sites (urban restored) in four of the urban streams are within recently implemented natural channel

design restoration projects. Four blocks were created from the group of 12, each containing one urban degraded, urban restored, and minimally impacted stream of similar catchment sizes (Table 1, Figure 2). For each set of metrics, all streams within a block were sampled within one week with no intervening major storm events. In this way the blocking factor accounts for both differences in watershed size, and staged timing of field analyses. For each metric I performed an Analysis of Variance with stream type and block as factors to test for differences between the three stream types.

Hydrologic Tracer Studies

In May and June 2006 I performed short-term whole stream enrichment experiments using NaBr and rhodamine dye conservative tracer additions (Hall et al. 2002). We added rhodamine to reach a pre-determined goal concentration in each stream based on streamflow estimates calculated from the previous day. A solution of rhodamine and NaBr was pumped continuously into the stream with a Watson-Marlow fluid metering pump. We monitored rhodamine concentrations continuously with a YSI Optical Monitoring System (OMS) probe equipped with a rhodamine sensor. In each stream the sensor was installed prior to the release at a well mixed point in the channel ~45 minutes of travel time downstream of the addition site. Rhodamine concentrations were allowed to plateau (change less than 1 ug/L in a 10 minute period). At this point the pump was stopped and the probe continued to record until rhodamine concentrations returned to pre-release levels (or below detection)

Habitat Surveys

In July and August, 2006 the habitat of each reach was mapped (pool, riffle, run, debris dam, etc.) of all experimental reaches in the following weeks. This included observational decisions on longitudinal boundaries of differing habitats, a survey of five randomly chosen cross-sectional profiles, and a survey of the longitudinal slope. Additionally, the measurement of five velocity and depth values evenly spaced across the active channel, with a sixth measurement of depth and velocity in the thalweg, were done at 30 cross-section locations evenly spaced longitudinally, all within the same experimental reaches used in the conservative tracer release experiments.

Data Analyses

Habitat heterogeneity was determined by counting the number of transitions between different aquatic habitats (riffle, run, pool, and debris dam classifications) for each experimental reach. The transition counts were normalized for all reaches by converting the counts to number of transitions per a 100 meter reach length. The average number of transitions and standard error of the blocked average was determined for each blocked type (MI, UR, and UD). Velocity and depth point measurement averages for each reach were calculated and used to obtain an average for each reach type. The standard error of each blocked average was also calculated.

The ratio of active channel width to the active channel depth at the thalweg was determined from the field survey cross-section data for each experimental reach. Again, the averages and standard errors for the different blocks were calculated for comparison of the degree of incision between the types. Also, the maximum (smallest W:D ratio value for each stream) and minimum (largest W:D ratio value for each stream) incision value from the field survey data was calculated. Average percent of canopy coverage for each reach was determined from five values measured at the same longitudinal distances from the injection point as the five cross-sectional surveys. The average and standard errors for each blocked type's percent canopy coverage was calculated.

Hydrodynamic properties (e.g. transient storage) of each experimental reach were estimated using an one dimensional transport with inflow solute transport model known as OTIS-P (Runkel 1998). Similar to most applications of this type of model, uniform flow conditions were assumed for all experimental reaches. Therefore, tracer releases were not done after (or during) rain events until a stream's discharge had returned to stable levels. As described below, the storage zone area (A_s in m^2), cross-sectional area (A in m^2), dispersion coefficient (D in m^2/sec), storage zone exchange coefficient (α in sec^{-1}), and storage zone first-order decay coefficient (λ_s in sec^{-1}) were parameters modeled within OTIS-P for all 12 streams. Because rhodamine has the potential to adsorb to sediment surfaces (especially in the hyporheic zone) and photo-degrade (*pers. comm. Rob Runkel*), the mass of dye recovered was determined for each reach. This was done by dividing (the area under the measured dye concentration curve multiplied by the discharge at the injection site) by (the average rhodamine plateau concentration {only dilution corrected} multiplied by the discharge at the rhodamine probe site and multiplied by the duration of the injection). As below:

$$\begin{aligned} \text{Mass recovery} &= (\text{mass out}) / (\text{mass in}) \\ &= (\text{area under the measured curve.}) / (\text{dilution corrected plateau conc.} * \text{duration of injection}) \end{aligned}$$

Because the discharge values in the numerator and denominator are the same value they were left out of the equation altogether. The calculated mass correction factor was multiplied by the raw measured data to give a ‘mass corrected concentration data set’. Until the final estimates of D, A, A_s , and α were obtained this mass corrected data set was used as the measured concentration data input for OTIS-P iterations.

The first step in the modeling process is to use OTIS-P to iteratively model D (initially set to 0.05 for all streams) and A (initially set to the value obtained by dividing the estimated Q by the average velocity). Once the residual sum of squares (RSS) of the fit between modeled and field measured (mass corrected) rhodamine dye concentrations had stopped changing, the initial A_s value was entered as 20% of the final modeled A value from the first step. For all 12 streams, the initial α value was always set to 1.0×10^{-4} . Again OTIS-P was used to iteratively model all four parameter estimates until parameter and/or RSS convergence was achieved (except for the Northgate Park site – see results section). To account for any Rhodamine mass that may have been lost by sorption or photo-degradation the storage zone decay coefficient (λ_s) was estimated by fixing the OTIS-P estimates of D, A, A_s , and α while the model was iteratively run using the raw (not mass corrected) measured data set until the parameter and/or RSS convergence was attained. At this point all five of the parameters were set to be estimated (i.e. variable) and the model was run iteratively until parameter and/or RSS convergence was attained. From the OTIS-P model estimates the other commonly used transient storage metrics like the fraction of median travel time due to transient storage (F_{med}^{200}) (Runkel 2002) were calculated.

RESULTS

Minimally impacted (MI) streams had higher habitat diversity (measured as the # of longitudinal transitions between geomorphic habitats in a 100m reach) relative to the urban degraded (UD) and urban restored (UR) streams ($p < 0.002$) (Figure 3, Table 2). The average stream depth (calculated from ~50 point measurements per stream) was significantly lower for MI streams relative to UD and UR streams ($p < 0.00075$) (Figure 4). The average and the CV of the point measured velocity values showed no significant differences among the three stream types.

The average degree of incision (W:D ratio), the average maximum degree of incision (smallest W:D ratio), and percent longitudinal slope showed no statistically significant differences among any of the three blocks. However, the average percent canopy cover of the UR block was significantly lower (ANOVA, $p < 0.011$) than that of the MI or UD blocks (Figure 5). Finally, none of the storage zone metrics showed any significant differences between the stream types (Table 3).

As the percent impervious surfaces increase within 30 meters of the stream reaches there is an increasing trend of the maximum degree of incision (Figure 6). Additionally, the number of habitat transitions per 100 meter reach length appears to be a function of the percent impervious surface in the experimental site's watersheds (Figure 7).

DISCUSSION

The significant decrease in canopy coverage in the Urban Restored (UR) streams relative to the UD streams points to the need for prioritization of important riparian habitat attributes in both the design and the construction phase of stream restoration projects. As expected, the four Urban Degraded (UD) streams had significantly lower habitat heterogeneity relative to the Minimally Impacted (MI) streams. In contrast to our expectations the restored streams showed no significant increases in habitat diversity relative to the urban degraded stream conditions. As one of the main goals of stream restoration projects is the creation of habitat, it appears to be a major challenge to engineer a restored reach with significant heterogeneity of aquatic habitats. Flow habitat (depth only) heterogeneity was found to be less variable in restored reaches relative to urban degraded and minimally impacted reaches. In fact, restored reaches were deeper and had

less fine-scale variation in depth than their un-restored, urban counterparts. This fact points to the need for addressing habitat and fine scale flow-field heterogeneity during design and construction phases of stream restoration processes. Measurements from this study demonstrate that the restoration implementation can actually move a stream reach further away from reference conditions than the pre-project degraded state, by removing mature riparian trees and homogenizing the streambed .

Although channel incision estimates were not significantly different between stream types, it was encouraging to see that all of the restored projects had high w:d ratios indicating minimal channel incision. However, the large difference in canopy coverage found in the restored reaches with more variability in the coverage is a concern when taking the goal of ecosystem function recovery into consideration. Efforts to preserve existing trees and vegetation during the restoration process may accelerate the ecosystem recovery process.

Watershed land use characterization appears to be a better predictor of channel form as well as habitat diversity than stream type. These relationships to watershed land use point to the need for a holistic approach if stream restoration efforts are to be successful. In other words, watershed restoration is just as, if not more, important than only focusing on restoring stream ecosystem health through the natural stream channel design methods.

Given the dramatic differences in channel form and bed substrates across the urbanization gradient I was surprised to find that modeled transient storage estimates were not significantly different between stream types. I expected that the minimally impacted streams would have much higher transient storage relative to the UR and UD blocks, because I anticipated that the MI streams would have greater surface-subsurface exchange and a more extensive hyporheic zone than their counterparts in urban watersheds. It is unlikely that the urban streams have extensive hyporheic storage, thus I attribute my inability to detect differences in transient storage to a widely acknowledged shortcoming of the OTIS-P model (Harvey 2000; Harvey 1996). For example, Harvey et al. found that stream tracer approaches can fail to reliably characterize the hyporheic zone exchange rate at high base flows. And their modeling results were more sensitive to exchange with in-channel transient storage zones than with the hyporheic storage zone. Also, current tracer models like OTIS-P are unable to differentiate between storage zones in the channel (like slow flowing pools and backwaters) and storage within the streambed and riparian zone (hyporheic storage and groundwater exchange). I hypothesize that the location of transient

storage shifts from hyporheic and floodplain storage to in-channel storage in urban impacted watersheds. Quantitative GIS interpolation of the point measurements of depth and velocity are underway in order to adequately test this hypothesis. Importantly, if in-channel transient storage is the primary location of transient storage in these urban and restored reaches that leads to the conclusion that there is less opportunity for water-substrate interactions to occur and thus fewer opportunities for nutrient removal and uptake, as well as the creation and maintenance of critical habitat for benthic fauna.

Implications for Ecosystem Function

A primary motivation for conducting this study was to determine if urban stream restoration efforts in the state of NC are effective in their mission "to restore, enhance, preserve and protect the functions associated with wetlands, streams, and riparian areas..." (NC EEP mission statement). My analyses document that urbanization leads to deeper and slower flowing, low slope streambeds composed of homogeneous bed materials. Urban river restoration efforts appear to effectively reduce channel incision, but are ineffective at re-establishing heterogeneity of bed materials and flow habitats. In fact, for some metrics urban restored streams are even more homogenous than their degraded counterparts. While my assessment of these restoration efforts was principally focused on geomorphic forms and hydrologic connectivity, inferences can be drawn about the connections between my ecosystem form assessments and functional measures of these ecosystems. My observations mesh well with complementary work at these same sites, which has documented increased algal production in the urban streams, and significantly reduced aquatic macroinvertebrate diversity (E.B. Sudduth and C. Violin, unpublished data). By further homogenizing the streambed and reducing canopy cover during the implementation of restoration projects, we may actually be moving restored stream reaches off a direct recovery trajectory towards reference conditions.

Recommendations for the Practice

Current stream restoration efforts have focused on constructing stream channels that are fixed, well armored to stay fixed (albeit with natural materials), and rife with meander bends. These recent efforts have lacked the sophistication, and grounding in sound geomorphic and ecological science, to sufficiently address the complex task of restoring stream ecosystems

(Kondolf 2006). While acknowledging the constraints of most urban stream restoration projects, there are several ways in which practitioners can improve stream restoration design and construction.

The design, and more importantly, construction phase of a project can improve the recovery time of stream ecosystems by maintaining more of the pre-existing canopy coverage found in many urban stream systems. Although many of the trees in these riparian areas are non-native, and may have been an artifact of past land uses, they can be used to shade the stream while more shade tolerant, riparian species can grow to replace them.

Heterogeneity is an important component of the structure of natural streams that is lost through most land use intensification. Allowing the stream to function geomorphically as it once did can lead to restoration of ecosystem heterogeneity. To achieve and maintain ecosystem heterogeneity, restoration efforts must incorporate storm water management (not rely on the stream restoration itself) to restore the hydrograph back towards pre-development conditions. Also, even though it is a threat to the 'fixed' approach/philosophy of many restoration project designs, fine-scale flow fields (heterogeneity) are essential for geomorphic, hydrologic, and thus ecological (habitat) recovery. This can be done by including a larger diversity of substrate type and size in the restoration design, using the already prevalent root wads in the middle of the channel sticking straight up out of the bed (esp. at end of riffles/runs), and building a degree of 'failure' into these restored stream reaches. While channel failure may seem like project failure to many, it is a natural process seen in stream ecosystems of all regions throughout the world.

ACKNOWLEDGEMENTS

Special thanks to Dr. Emily Bernhardt for all of her input, insight, and assistance. And thanks to the 'Bernhardt Lab' at Duke University for the supplies, space, equipment, guidance, and opportunity to perform all of the exciting field experiments, data collection, and laboratory processing of field samples. Rob Runkel of the USGS was instrumental in the transient storage estimations using OTIS-P. For guidance and review of methods, thanks go to Dr. Martin Doyle of UNC Chapel Hill. My field and summer research work was funded by the North Carolina Ecosystem Enhancement Program (EEP). Special thanks to Mac Haupt at EEP. Other funding for this project, and additional and larger scale research studies involving this work was provided by NC Ecosystem Enhancement Program (NC EEP), NC WRRI, NSF Ecosystems, the John & Lucile Packard Foundation, and the EPA STAR Program. Field and lab work was done by Johanna Bernhardt, Peter Cada, Drew Goltermann, Brooke Hassett, Emily Bernhardt, Nikos Lessios, Jennifer Morse, Elizabeth Sudduth, and Christy Violin.

Table 1 – Site and watershed characteristics

Block	Status	Site Name	Reach Length (m)	Estimated Discharge (L/s)	Percent Dilution (%)	Watershed Size (km ²)	% Developed	% Impervious
1	minimally impacted	Stony Creek	100	0.66	27.3	6.9	24.4	3.4
	urban restored	Forest Hills	80	4.41	35.5	4.4	99.5	32.4
	urban degraded	Northgate Park	50	10.41	48.7	7.6	88.7	20.8
2	minimally impacted	Pots Branch	140	5.83	5.7	4.2	27.4	9.9
	urban restored	Abbott Creek	200	5.47	10.7	1.7	84.5	17.8
	urban degraded	Cemetary Creek	100	11.54	4.5	2.2	98	19.1
3	minimally impacted	Mud Creek Tributary	54	2.08	6.4	0.9	4.4	0.5
	urban restored	Rocky Branch	50	1.54	6.7	1.5	99.2	34.8
	urban degraded	Goose Creek	35	3.72	17.7	1.7	100	39.4
4	minimally impacted	Mud Creek Reach 4	102.5	11.58	12.2	4.1	58.6	9.5
	urban restored	Sandy Creek	60	12.00	5.3	6.7	76.9	16.8
	urban degraded	Mud Creek Reach 1	140	4.86	6.2	3.5	66.9	11

Table 2 – Average values used to analyze habitat and flow heterogeneity, floodplain connectivity, riparian zone canopy coverage, and hydrodynamics of minimally impacted, urban restored, and urban degraded stream type blocks. *p* - values are from single factor ANOVA's ('ns' = *p* > 0.10)

	Minimally Impacted	Urban Restored	Urban Degraded	<i>p</i>
number of habitat transitions per 100-meter reach length (#)	21.66	10.06	10.57	< 0.002
average depth from point measurements (m)	0.065	0.175	0.158	< 0.001
average %CV for depth point measurements (m)	109.3	73.7	83.0	< 0.076
average velocity from point measurements (m/s)	0.035	0.023	0.026	ns
average %CV for velocity point measurements (m)	209.2	139.4	237.0	ns
average degree of incision (W:D)	6.147	6.718	4.959	ns
average maximum degree of incision (smallest W:D)	4.743	5.023	4.403	ns
average longitudinal slope (%)	0.930	0.290	0.510	ns
average canopy coverage (%)	87.55	53.71	81.35	< 0.011
ratio of storage zone area to cross-sectional area (A_s/A)	0.595	0.601	0.558	ns
fraction of total reach volume occupied by the storage zone ($A_s/(A_s + A)$)	0.358	0.360	0.342	ns
median travel time due to transient storage (F_{med}^{200} , %)	0.350	0.353	0.335	ns

Table 3 – Example of outputs and metrics from OTIS-P (additional outputs/metrics in Appendix B)

Type	Stream	Block	As/A	As (As + A)	Damkohler Number (Dal)	Final Residual Sum of Squares	Ratio [Estimate/Std. Dev.]				
							D	(A)	(A _s)	(α)	(A _s)
MI	Mud 4	1	0.305	0.234	5.55	34.19	15.32	95.90	39.78	12.92	29.74
MI	Mud Trib	2	0.937	0.484	4.77	97.49	4.32	28.90	31.79	9.19	13.52
MI	Stony	3	0.714	0.416	147.68	95.47	17.85	135.50	134.50	32.88	65.97
MI	Potts	4	0.422	0.297	7.64	34.05	6.38	2.48	27.02	8.74	13.89
UR	Sandy	1	nodata	nodata	nodata	nodata	nodata	nodata	nodata	nodata	nodata
UR	Rocky	2	0.285	0.222	2.38	1281.01	4.47	26.71	9.69	2.61	0.58
UR	Forest Hills	3	0.894	0.472	5.26	3.46	8.50	28.53	18.81	7.83	19.30
UR	Abbot	4	0.451	0.311	19.01	5.94	2.67	36.77	17.86	8.21	17.08
UD	Mud 1	1	0.752	0.429	14.60	217.42	FIXED	76.61	67.88	18.46	34.13
UD	Goose	2	0.501	0.334	2.77	1311.23	6.14	50.96	8.57	6.19	0.06
UD	Northgate	3	0.201	0.168	2.90	25.77	NC	NC	NC	NC	NC
UD	Cemetery	4	0.780	0.438	6.86	6.23	1.44	22.02	19.79	6.51	10.47

MI – Minimally impacted stream; UR – Urban restored stream; UD – Urban degraded stream.

NC – No convergence for parameter estimation or residual sum of squares fit between modeled and measured dye concentrations.

LIST OF FIGURES

FIGURE 1. Expected transient storage estimation levels

FIGURE 2. Site and watershed locations

FIGURE 3. Habitat heterogeneity analysis

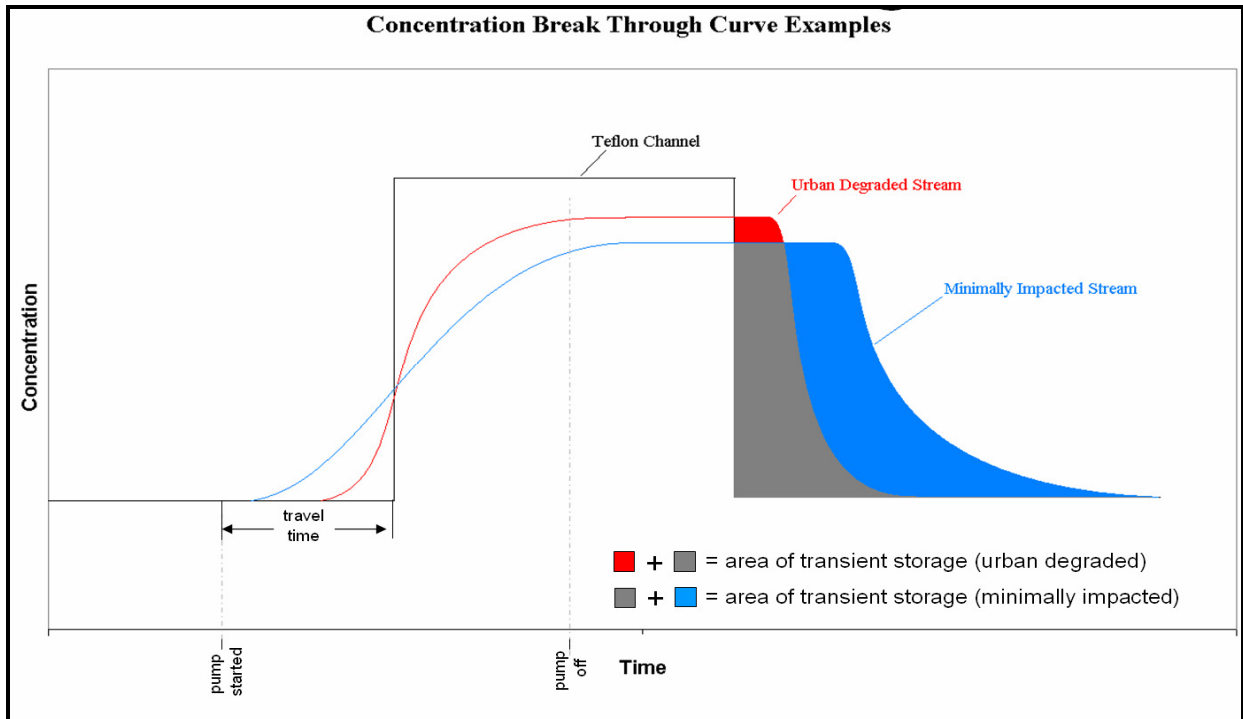
FIGURE 4. Average depths are from 186 point measurements. The coefficient of variation for each stream type group was used as a metric for depth heterogeneity.

FIGURE 5. Percent Canopy Coverage Analysis

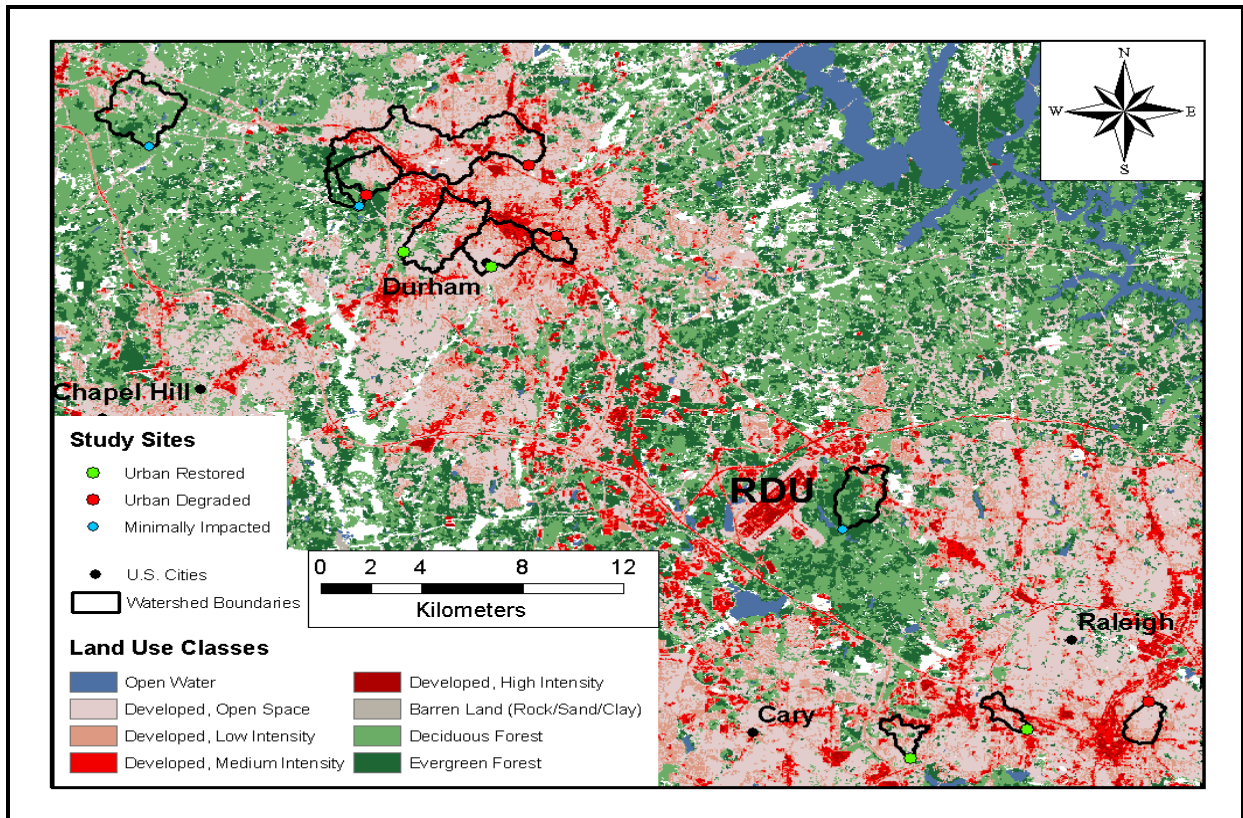
FIGURE 6. Riparian Buffer Imperviousness vs. Maximum Degree of Incision

FIGURE 7. Watershed Imperviousness vs. Habitat Transition Metric

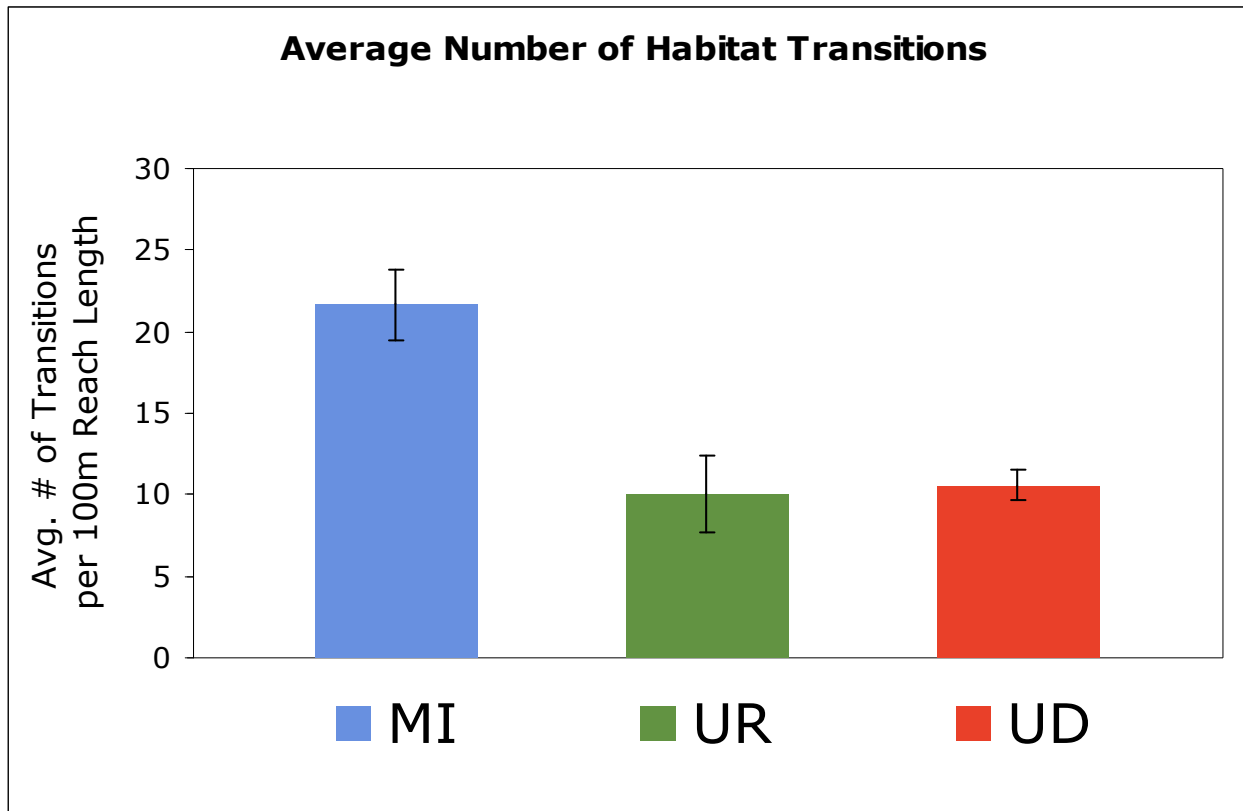
Figure_1



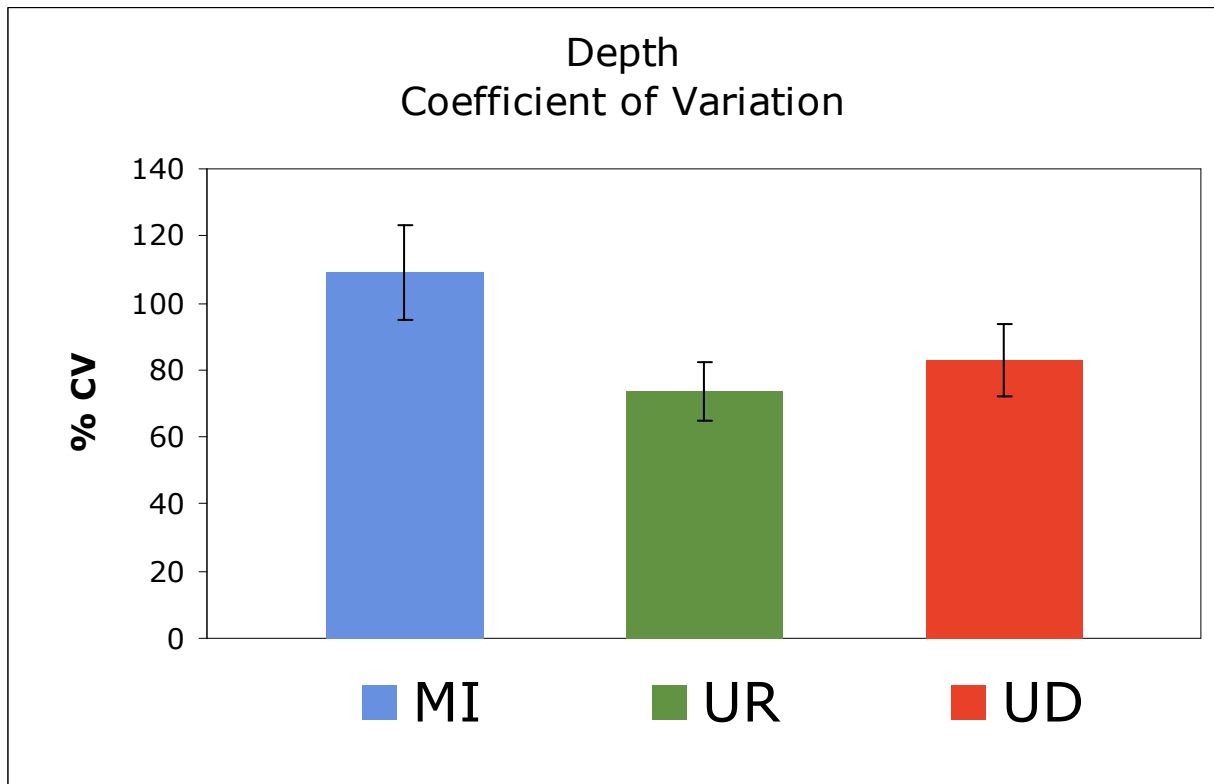
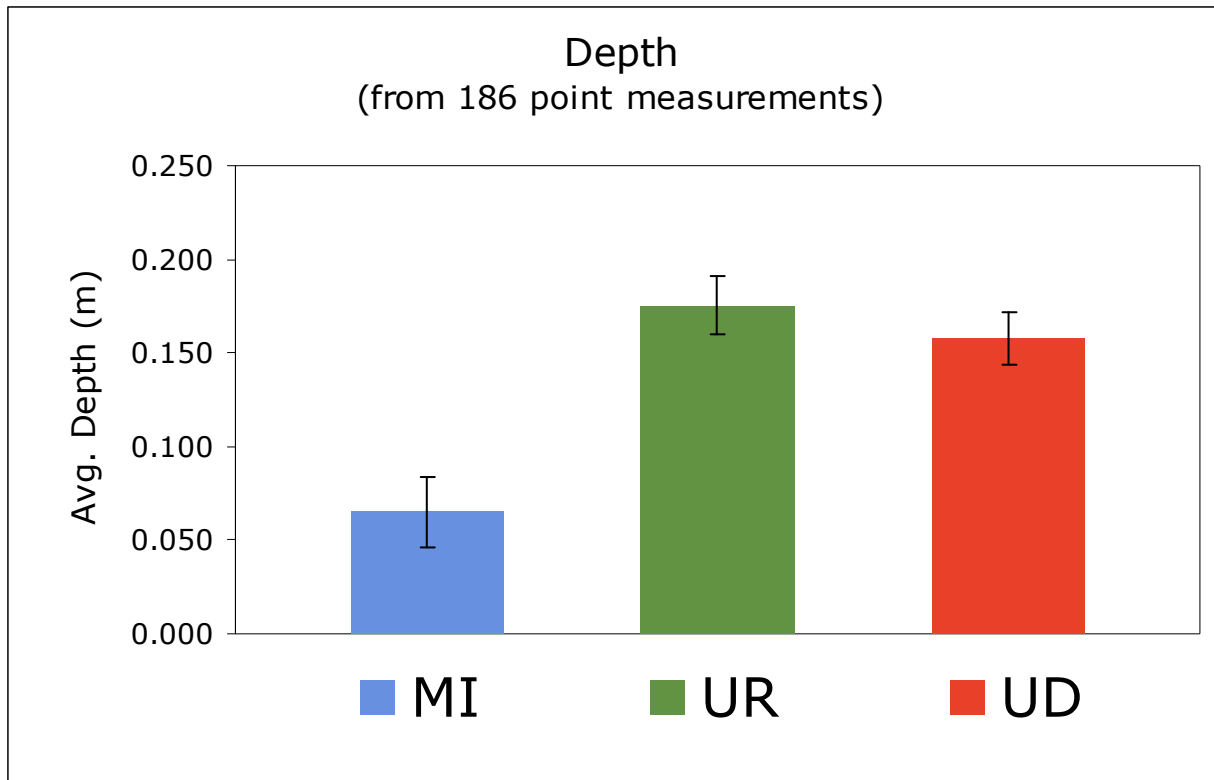
Figure_2



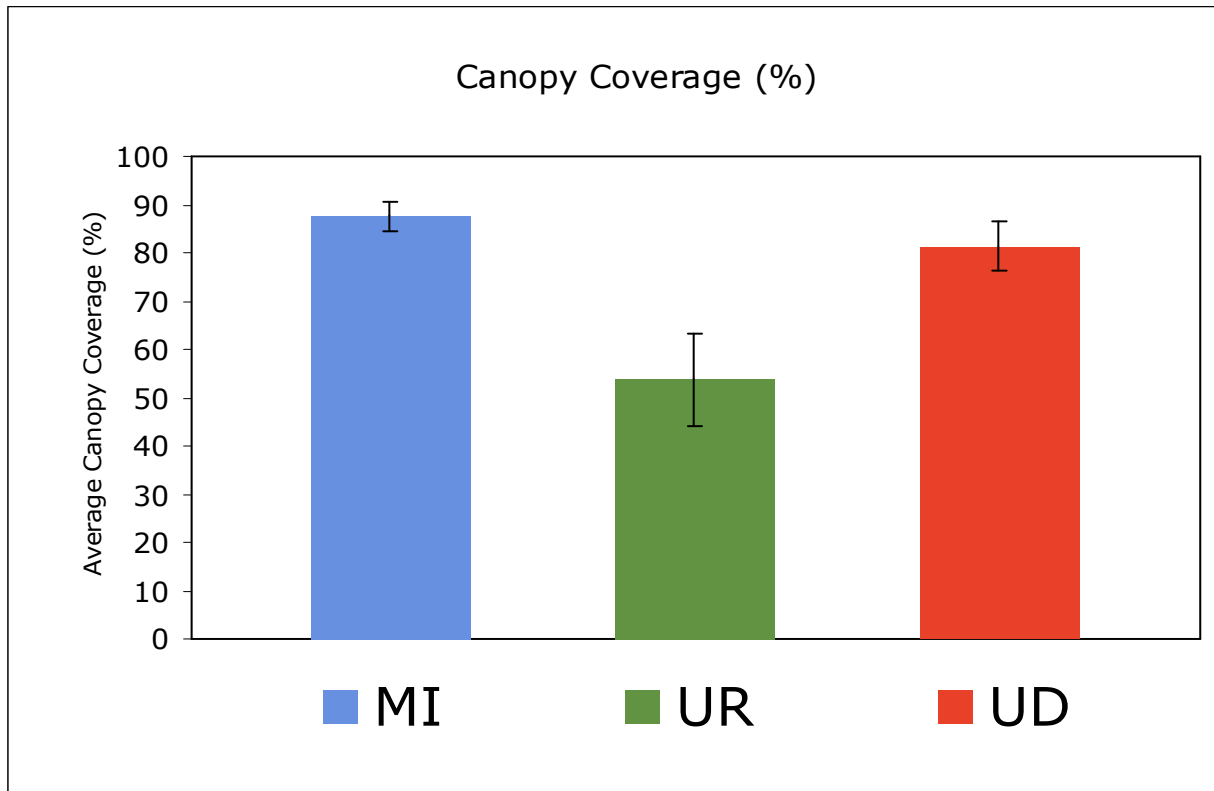
Figure_3



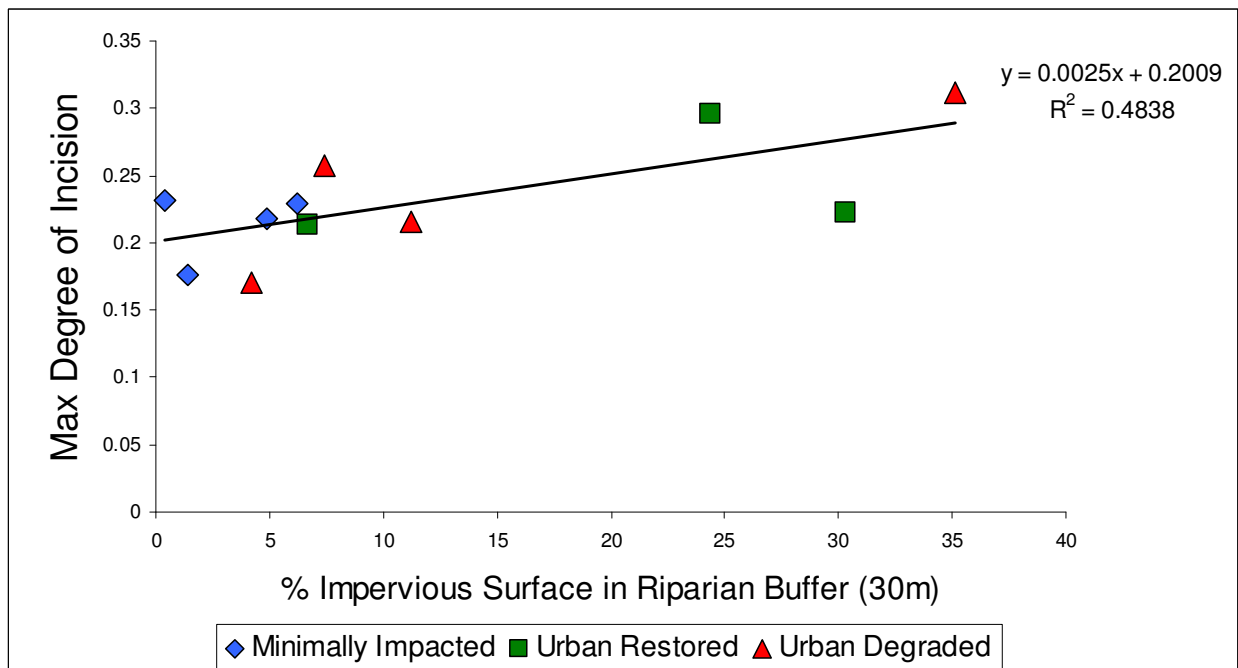
Figure_4



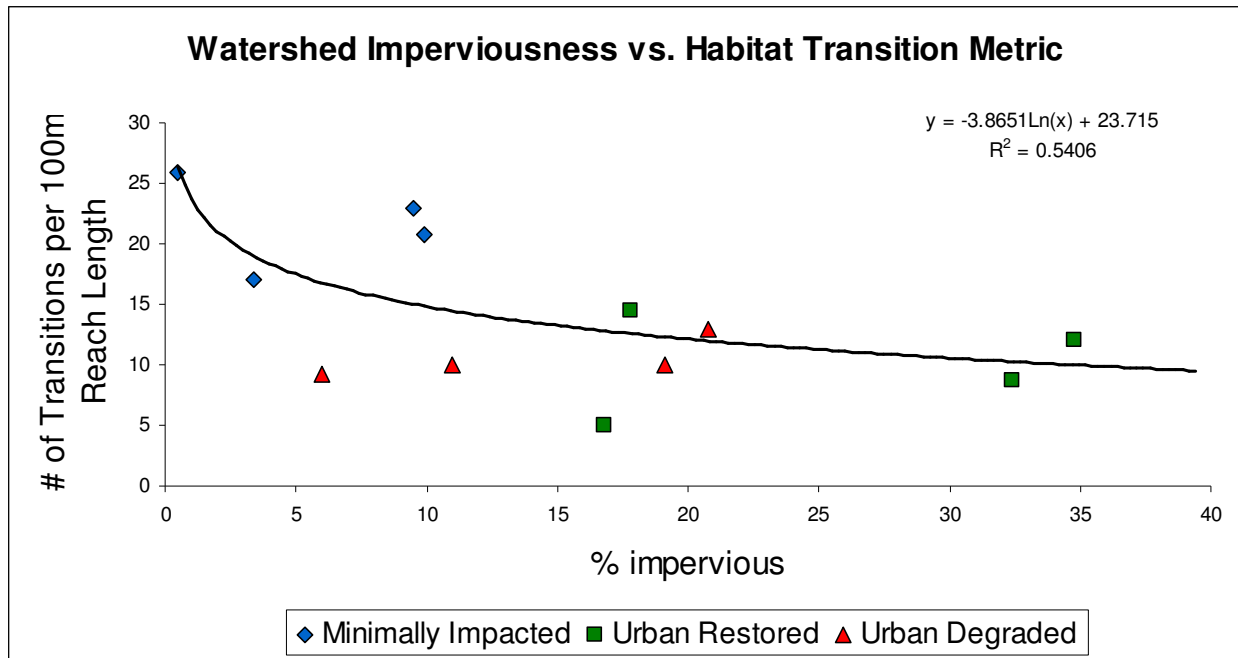
Figure_5



Figure_6



Figure_7



REFERENCES

- Bernhardt, E. S., and Margaret A. Palmer. 2007. Restoring streams in an urbanizing world. *Freshwater Biology* **52**: 738-751.
- Bernhardt, E. S. and others 2005. Ecology - Synthesizing US river restoration efforts. *Science* **308**: 636-637.
- Bilby, R. E., and G. E. Likens. 1980. IMPORTANCE OF ORGANIC DEBRIS DAMS IN THE STRUCTURE AND FUNCTION OF STREAM ECOSYSTEMS. *Ecology* **61**: 1107-1113.
- Booth, D. B. 2005. Challenges and prospects for restoring urban streams: a perspective from the Pacific Northwest of North America. *Journal of the North American Benthological Society* **24**: 724-737.
- Booth, D. B., and C. R. Jackson. 1997. Urbanization of aquatic systems: Degradation thresholds, stormwater detection, and the limits of mitigation. *Journal of the American Water Resources Association* **33**: 1077-1090.
- Brooks, S. S. A. A. J. B. 1991. Recolonization Dynamics of Benthic Macroinvertebrates after Artificial and Natural Disturbances in an Australian Stream. *Aust. J. Mar. Freshwater Res.* **42**: 295-308.
- Brown, K. B. 2000. Urban Stream Restoration Practices: An Initial Assessment. The Center for Watershed Protection, Ellicott City, MD.
- Carpenter, D. D., S.K. Sinha, K. Brennan., and L.O. Slate. Asce River Restoration Subcommittee on Urban Stream Restoration. . 2003. Urban Stream Restoration. *J Hydraulic Eng.* **July 2003**: 491-493.
- Clarke, S. J., L. Bruce-Burgess, and G. Wharton. 2003. Linking form and function: towards an eco-hydromorphic approach to sustainable river restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems* **13**: 439-450.
- Groffman, P. M., and M. K. Crawford. 2003. Denitrification potential in urban riparian zones. *Journal of Environmental Quality* **32**: 1144-1149.
- Hall, R. O., E. S. Bernhardt, and G. E. Likens. 2002. Relating nutrient uptake with transient storage in forested mountain streams. *Limnology and Oceanography* **47**: 255-265.
- Harvey, J. W., and B.J. Wagner. 2000. Quantifying hydrologic interaction between streams and their subsurface hyporheic zones, p. 3-44. *In* J. B. Jones, and Mulholland, P.J. [ed.], *Streams and Ground Waters*. Academic Press.
- Harvey, J. W., B.J. Wagner, and K.E. Bencala. 1996. Evaluating the reliability of the stream tracer approach to characterize stream-subsurface water exchange. *Water Res. Research Journal* **32**: 2441-2451.
- Hassett, B., M. Palmer, E. Bernhardt, S. Smith, J. Carr, and D. Hart. 2005. Restoring watersheds project by project: trends in Chesapeake Bay tributary restoration. *Frontiers in Ecology and the Environment* **3**: 259-267.
- Kondolf, G. M. 1996. A cross section of stream channel restoration. *Journal of Soil and Water Conservation* **51**: 119-125.
- . 2006. River restoration and meanders. *Ecology and Society* **11**.
- May, C. W. 1998. The cumulative effects of urbanization on small streams in the Puget Sound lowland ecoregion. *PSWQA Puget Sound Research '98, Conference Proceedings*, Seattle, WA.

- McMahon, G., and T. F. Cuffney. 2000. Quantifying urban intensity in drainage basins for assessing stream ecological conditions. *Journal of the American Water Resources Association* **36**: 1247-1261.
- Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society* **24**: 602-612.
- Moerke, A. H., and G. A. Lamberti. 2004. Restoring stream ecosystems: Lessons from a midwestern state. *Restoration Ecology* **12**: 327-334.
- Niezgoda, S. L., and P. A. Johnson. 2005. Improving the urban stream restoration effort: Identifying critical form and processes relationships. *Environmental Management* **35**: 579-592.
- Palmer, M. A. 1990. Temporal and spatial dynamics of meiofauna within the hyporheic zone of Goose Creek, Virginia. *J. N. Am. Benthol. Soc.* **9**: 17-25.
- Palmer, M. A. and others 2005. Standards for ecologically successful river restoration. *Journal of Applied Ecology* **42**: 208-217.
- Paul, M. J., and R. O. Hall. 2002. Particle transport and transient storage along a stream-size gradient in the Hubbard Brook Experimental Forest. *Journal of the North American Benthological Society* **21**: 195-205.
- Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. *Annual Review of Ecology and Systematics* **32**: 333-365.
- Poff, N. L., M.A. Palmer, P.L. Angermeier, R.L. Vadas, Jr., C.C. Hakenkamp, A. Bely, P. Arensburger, and A.P. Martin. 1993. Size structure of the metazoan community in a Piedmont stream. *Oecologia* **95**: 202-209.
- Rosgen, D. L. 1994. A CLASSIFICATION OF NATURAL RIVERS. *Catena* **22**: 169-199.
- Runkel, R. L. 1998. One dimensional transport with inflow and storage (OTIS): a solute transport model for streams and rivers. US Geological Survey Water-Resources Investigation Report 98-4018. USGS, Denver, Colorado. (Available from: <http://co.water.usgs.gov/otis>).
- . 2002. A new metric for determining the importance of transient storage. *J. N. Am. Benthol. Soc.* **21**: 529-543.
- Shields, F. D., R. R. Copeland, P. C. Klingeman, M. W. Doyle, and A. Simon. 2003. Design for stream restoration. *Journal of Hydraulic Engineering-Asce* **129**: 575-584.
- Sudduth, E. B., and J. L. Meyer. 2006. Effects of bioengineered streambank stabilization on bank habitat and macroinvertebrates in urban streams. *Environmental Management* **38**: 218-226.
- Swank, W. T. A. D. A. C., Jr. 1988. *Ecological Studies Vol. 66: Forest Hydrology and Ecology at Coweeta*. Chapter 19 – Aquatic Invertebrate Research – D.B. Wallace. reprint available at: <http://cwt33.ecology.uga.edu/publications/299.pdf>.: 257-268.
- Walsh, C. J., T. D. Fletcher, and A. R. Ladson. 2005a. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. *Journal of the North American Benthological Society* **24**: 690-705.
- Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan. 2005b. The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society* **24**: 706-723.
- Wollheim, W. M. and others 2001. Influence of stream size on ammonium and suspended particulate nitrogen processing. *Limnology and Oceanography* **46**: 1-13.

Appendix A – Additional outputs and metrics from OTIS-P modeling iterations

Type	Stream	Block	Rhodamine Mass Recovery	Damkohler Number (Dal)	Storage Zone Area (A _s)	Channel Cross- Sectional Area (A)	Storage Exchange Coefficient (α)	Transient Storage Decay Coefficient (λ _s)	As/A	$\frac{As}{(As + A)}$	Average Distance Traveled Before Entering Hyporheic (L _s) - meters	Percent of Reach Traveled Before Entering Storage Zone	Fraction of Median Travel Time Due to Transient Storage (%) (F _{med} - Runkel 2002)	"Standardized" Fraction of Median Travel Time Due to Transient Storage (%) (F _{med} ²⁰⁰ - Runkel 2002)
MI	Mud 4	1	0.8912	5.55	0.149	0.49	3.00E-04	1.20E-04	0.305	0.234	78.99	77%	16.99%	21.52%
MI	Mud Trib	2	0.9205	4.77	0.090	0.0955	9.30E-04	8.20E-05	0.937	0.484	23.42	43%	43.56%	48.37%
MI	Stony	3	0.8265	147.68	0.049	0.0688	5.90E-03	8.70E-05	0.714	0.416	1.63	2%	41.65%	41.65%
MI	Potts	4	0.9487	7.64	0.095	0.2247	4.20E-04	5.90E-05	0.422	0.297	61.74	44%	26.62%	28.53%
UR	Sandy	1	nodata	nodata	nodata	nodata	nodata	nodata	nodata	nodata	nodata	nodata	nodata	nodata
UR	Rocky	2	0.9464	2.38	0.042	0.1478	1.10E-04	1.30E-05	0.285	0.222	94.48	189%	9.12%	19.52%
UR	Forest Hills	3	0.7213	5.26	0.211	0.236	5.80E-04	3.50E-04	0.894	0.472	32.22	40%	43.26%	47.11%
UR	Abbot	4	0.8863	19.01	0.077	0.17	9.50E-04	9.26E-05	0.451	0.311	33.87	17%	30.98%	30.98%
UD	Mud 1	1	0.7292	14.60	0.109	0.145	1.50E-03	1.20E-04	0.752	0.429	22.34	16%	42.83%	42.91%
UD	Goose	2	0.9306	2.77	0.492	0.982	1.00E-04	9.10E-07	0.501	0.334	37.88	108%	20.13%	33.21%
UD	Northgate	3	0.6021	2.90	0.204	1.0132	1.00E-04	6.90E-07	0.201	0.168	102.75	206%	6.46%	14.37%
UD	Cemetery	4	0.9517	6.86	0.276	0.354	9.80E-04	3.90E-05	0.780	0.438	33.26	33%	41.64%	43.70%