ABSTRACT

CARTER, MELANIE DAWN. Stream Assessment and Constructed Stormwater Wetland Research in the North Creek Watershed. (Under the direction of Jean Spooner and Robert Evans.)

North Creek is a small urban stream that flows through Centennial Campus, NCSU, Raleigh, NC. The watershed is urban (40% impervious), with extensive campus development in progress. The channel is incised and unstable, and has experienced erosion, elevated sediment loads, and water quality degradation due to stormwater runoff effects. The purpose of this study was to evaluate stream conditions over two years in three reaches of North Creek. Streambank erosion rates ranged from 0.3 to 1.7 ft/yr, greater than predicted rates using a streambank erodibility model, (Rosgen 2001). BEHI assessments categorized erosion potential in High to Extreme categories for most of the stream length. The streambed experienced localized aggradation within upper reaches, but incised (1-3 ft) in the lower reach. Headcut migration was responsible for the incision, with eroded material deposited downstream. The majority of channel erosion resulted from two large storm events.

Substrate characteristics were measured over time using four methods: pebble counts, pavement/subpavement, particle embeddedness, and pool fine sediment intrusion. All methods, except the pavement/subpavement samples, provided evidence of fine sediment accumulation. TSS concentrations and turbidity were measured during most storm events using two methods. Suspended sediment yields were observed to be greater using a single-stage sampler (1320 lb/ac/yr) compared to an automated sampler (542 lb/ac/yr), due to differences in sampling along the storm hydrograph. Results from single-

stage samplers established along the stream were used to establish spatial trends. There was a significant increase in sediment parameters as discharge and channel length increased. A partial annual sediment budget was derived, with estimates of suspended sediment load (94 tons), streambank erosion load (100 tons), and streambed erosion load (-173 tons). Although overestimations were possible, the streambed acted as a net sediment output, compared to an input in this degraded stream budget. A preliminary water quality evaluation in North Creek suggested that pollutants, (sediment, nutrients, metals, and organic PAH compounds), were elevated during stormflows. Benthic macroinvertebrate sampling indicated that taxa richness and relative abundance were reduced, with chironomids comprising the majority of organisms. Biological habitat was deficient, demonstrated by detailed habitat maps of stream features, organic matter quantifications, and substrate parameters compared between North Creek and a reference reach, Avent Creek.

Due to stormwater runoff pollution within the North Creek watershed, the purpose of this study was also to design, construct, and evaluate two constructed stormwater wetlands (CSWs) on the North Creek floodplain. Wetland 1 (0.25 ac) and Wetland 2 (0.05 ac) captured runoff from two watersheds (6 ac) comprised of campus buildings and parking lots. Stormwater treatment of sediment and nutrients was evaluated during initial stabilization for three months post-construction. The study found suspended sediment generation in both CSWs during the first month, but concentrations and loads were reduced afterwards between the inlet and outlet. Net sediment load reduction occurred in Wetland 2 (-29%), compared to net sediment generation in Wetland 1 (+78%).

Differences were mostly attributed to greater total sediment loads entering Wetland 2 (3065 lb/ac) compared to Wetland 1 (400 lb/ac). Substrate measurements indicated that sediment was accumulating within the pools, and erosion was not evident in the remaining regions. For nutrients treatment, both CSWs reduced nutrient (TKN, NH_4, NO_3, and TP) concentrations and loads after two months. The net nutrient loads were reduced throughout the study in Wetland 1, while Wetland 2 generated NH_4 and TP. Polyacrylamide (PAM) was applied (15 lb/ac) to Wetland 1 but not Wetland 2 during post-construction hydromulching to assess PAM erosion control and nutrient retention performance. The effects of PAM could not be evaluated, however, due to greater sediment and nutrient inputs to Wetland 2, the variability in substrate between the CSWs, and the lack of correlation between sediment and phosphorus.

STREAM ASSESSMENT AND CONSTRUCTED STORMWATER WETLAND RESEARCH IN THE NORTH CREEK WATERSHED

by

Melanie Dawn Carter

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Approved by:

Co-chair of Advisory Committee	Co-chair of Advisory Committee
Advisory Committee Member	Advisory Committee Member
Advisory Committee Member	Minor Committee Member

DEDICATION

This doctoral degree and dissertation are dedicated to the two most influential people in my life, my mother and my son.

To my mother, I dedicate this dissertation in honor of all the encouragement and unconditional love that she gave to me throughout my life; the work ethic that she instilled in me from the beginning; and for the devotion she has always had for my success.

To my son Ely, I dedicate this dissertation to him for providing the most brilliant light on this path, shining during both the happiest and darkest moments.

BIOGRAPHY

Melanie Carter was born and raised in Lynchburg, Virginia. She received her B. S. in Biology, with a minor in Environmental Science from Virginia Polytechnical Institute and State University, Blacksburg, Virginia in December 1996. She received her M. S. in Aquatic Biology from the University of Alabama, Tuscaloosa, Alabama in December 1999. Deciding to continue her education in the field of environmental engineering, she began her doctoral studies at North Carolina State University, Raleigh, NC in May 2000.

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Water Quality also helped with data collection and biological identification, including Dave Wojnokski Dave Lenat, and Mark Vander Borgh. For the reference reach Avent Creek, site access was given by the landowner Mr. Wayne Senter, who was very supportive.

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A special thank you must be extended to my committee chairs, who believed in me, challenged me, and guided me through the research and design. I would like to thank Dr. Robert Evans, for agreeing to be my committee co-chair and committing to seeing me through the degree process from the beginning to the end. I would like to thank Dr. Jean Spooner, who throughout this whole journey is the person who deserves the most recognition for her time and effort. She served as my research advisor, my mentor, and she was always there for me with invaluable support along the long road to degree completion.

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LIST OF ABBREVIATIONS

°C	degrees Celsius
°F	degrees Farenheit
ac	acre
ac-ft	acre feet
bdl	below detectable limits
BEHI	bank erosion hazard index
BMP	best management practice
cfs	cubic feet per second
cm	centimeter(s)
COT	College of Textiles
CSW	constructed stormwater wetland
d	day
dia	diameter
EGRC	Engineering Graduate Research Center
EL	elevation above sea level
ft	foot or feet
g	gram
ha	hectares
hr	hour(s)
ht	height
in	inch(es)
kg	kilogram(s)
L	liter(s)
lb	pound(s)
m	meter(s)
mi	mile
mg	milligram(s)
mL	milliliter(s)
mm	millimeter(s)
mS	millisiemen(s)
NBS	near-bank shear stress
NCSU	North Carolina State University
RES II	Research II Building
s	second(s)
SOC	stormwater outfall channel
sq ft	square feet
sq mi	square miles
SQRT	square root
SWMP	stormwater management pond
W1	constructed stormwater wetland #1
W2	constructed stormwater wetland #2
yr	year

RESEARCH INTRODUCTION

North Creek is a small second order stream that flows through the center of Centennial Campus, NCSU to Lake Raleigh in southwest Raleigh, NC. The watershed (0.5 mi^2) is urbanized with 40% impervious surface , and development occurring on another 11% of the area during this research study. The stream and its tributaries have experienced many negative impacts from development and urban stormwater runoff. All of the tributaries have been buried, diverted through stormwater pipes, or converted to stormwater ditches. Increased sediment loads from campus development and urban hydrology have contributed to an imbalance in the water and sediment transport functions of the stream. North Creek is vertically incised and unstable, with excess channel erosion suspected of contributing additional sediment load downstream.

North Creek, as part of the backwaters of Lake Raleigh and Walnut Creek, is considered an impaired water body in North Carolina, under the Section 303(d) of the Clean Water Act, based on impaired biological integrity caused by urban runoff and storm sewer discharges. The purpose of this study was to evaluate stream impairment by measuring channel erosion, substrate characteristics, suspended sediment, water quality, and biological integrity. This stream assessment occurred over a two-year period, between August 2002 and August 2004, in order to evaluate changes in stream stability over time.

In response to the urban stormwater pollution in this watershed, two constructed stormwater wetlands were designed and constructed as best management practices (BMPs) on the floodplain of North Creek. The wetlands were evaluated for stormwater treatment, including sediment and nutrient reductions. The constructed stormwater wetlands

monitoring began after construction in August 2004 and continued through October 2004, in order to measure sediment and nutrient changes during initial stabilization. Polyacrylamide (PAM) was applied to the bare wetland substrate after construction as a secondary BMP, to provide erosion control during the study. The PAM application was evaluated for sediment and nutrient retention at the wetland sites.

CHAPTER 1: GENERAL CHARACTERISTICS OF THE NORTH CREEK WATERSHED

This chapter presents a general discussion of North Creek watershed and stream characteristics that affect the stream assessment and constructed stormwater wetland research. The purpose of collecting these data was to define the study site, and to use this information as background and reference in later chapters for data analysis and result interpretation.

Watershed and Stream Corridor Description

North Creek is a small second order stream that flows through the center of Centennial Campus, NCSU to Lake Raleigh in southwest Raleigh, NC. The total exposed stream length is about 2775 ft, with a watershed area estimated to be 0.5 mi^2 (Figure 1.1). The watershed is highly urbanized, with 40% covered by impervious surfaces of the developed campus and surrounding residential and commercial business (Table 1.1). Significant development of Centennial Campus began around 1990 and will continue in the next several decades. During the two-year study, many construction projects were underway, contributing another 37 acres (11%) of impervious surface to the watershed upon completion (Figure 1.1). The development projects on Centennial Campus and other portions of the watershed are listed in Table 1.2, with their associated footprint. Stormwater from these development projects drains to North Creek through various culverts and associated outfall channels. Along with watershed development, the stream is also influenced by City of Raleigh sewer lines that run parallel to the entire length of North Creek on the left bank, looking downstream. This sewer line is exposed in several locations due to excessive lateral streambank erosion.

North Creek is divided into four distinct reaches based on substantial in-stream structures. Reach 1 stretches from the Avent Ferry Road culvert to the Varsity Drive road culvert (Figure 1.2). The watershed above Avent Ferry Road is completely piped and covered by residential and commercial land use. North Creek is daylighted at the end of the Avent Ferry road culvert into a plunge pool. The streambanks within this reach are vertical and steep throughout most of the reach, with the exception of a section near the Varsity Drive culvert due to channel alterations during culvert installation. The riparian buffer along the left bank is essentially absent due to excessive erosion; site development of the York Properties along Avent Ferry Road; and the City of Raleigh sewer line easement. The riparian buffer on the right bank with mature hardwood trees is the widest along the stream corridor (average 175-ft width). The stream thalweg (420-ft length) has a low gradient and is relatively straight except for the significant meander bends near the beginning and end of the reach, suggesting past channelization. The reach ends at the large Varsity Drive bottomless culvert (160-ft length), with an artificial large boulder lining and concrete floodplain bench designed as a footpath for a future greenway.

Reach 2 stretches from the Varsity Drive road culvert to the Research Drive road culvert (Figure 1.3). The streambanks are moderately steep throughout the reach, except near the Varsity Drive culvert due to alterations during culvert installation and City of Raleigh sewer line maintenance. The riparian buffer on the left bank is composed of a thin strip of hardwoods (~10-ft width) endangered by streambank erosion and the greenway landscaped boundary installed mid-study. The riparian buffer on the right bank is larger (~80-ft width), also containing hardwoods. The stream thalweg (1070-ft length) has a low

gradient and is relatively straight in this reach with only two significant bends, both held in place by exposed saprolite layers. The reach ends at the large Research Drive road culvert (110-ft length), filled with about 2-4 ft of sediment along the main flow path.

Reach 3 stretches from the Research Drive culvert to Stormwater Management Pond #3 (SWMP #3) (Figure 1.4). The streambanks are shorter in this reach and heavily influenced by artificial stabilization performed near the beginning of campus development. The eroding streambanks show remnant large cobble, and a check dam was present near the end of the reach that was removed by flow during the study. Old plastic erosion control matting was also along the banks in the lower portion of the reach, seen currently tangled in a tree debris jam just before the pond. The riparian buffer along this reach is relatively wide on both sides (30 to 200 ft), with a mixture of hardwoods and open grassed floodplain near the end of the reach. The steam thalweg (670-ft length) is also relatively straight due to channelization, with one significant bend mid-reach held in place by the exposed saprolite in this region. In the past, the channel was heavily armored with large cobble and small boulders for most of the reach length in the past. At the beginning of the study, a steep headcut had migrated mid-reach with armoring along a relatively steep grade present upstream. At the headcut, the stream flowed into a deep (>5 ft) pool with fine sediment and gravel lining the remainder of the low gradient reach. The reach widens near the end at the entrance to the in-stream stormwater ponds, with significant backwater effects from these ponds.

The portion of North Creek considered Reach 4 (variable 700-ft length) extends from SWMP #3 to Lake Raleigh (Figure 1.1). SWMP #3 and #4 are contiguous and held

in place by a large concrete weir. North Creek flows over the weir and remains a single channel for only a short distance until backwater flooding from the lake induces braiding. These ponds were dredged prior to the study (2000), and have fluctuating boundaries due to sediment deposition. This reach was not considered for research purposes or stream assessment due to the combination of ponding and lake backwater effects.

During the study, North Creek was compared to a reference stream for several research topics, including water quality, biological integrity, and substrate characteristics. Avent Creek (Harnett County, NC) was used as the reference. This stream is a tributary to the Cape Fear River located in the same Northern Outer Piedmont (45f) Ecoregion as North Creek (USEPA Ecoregion Maps), classified for similar biological and geological, vegetation, and other abiotic features. The stream reach (~500-ft) was located in a forested watershed (249 ac) on private property owned by Mr. Wayne Senter, just southwest of Fuquay Varina, NC off of Hwy 42 (Figure 1.5).

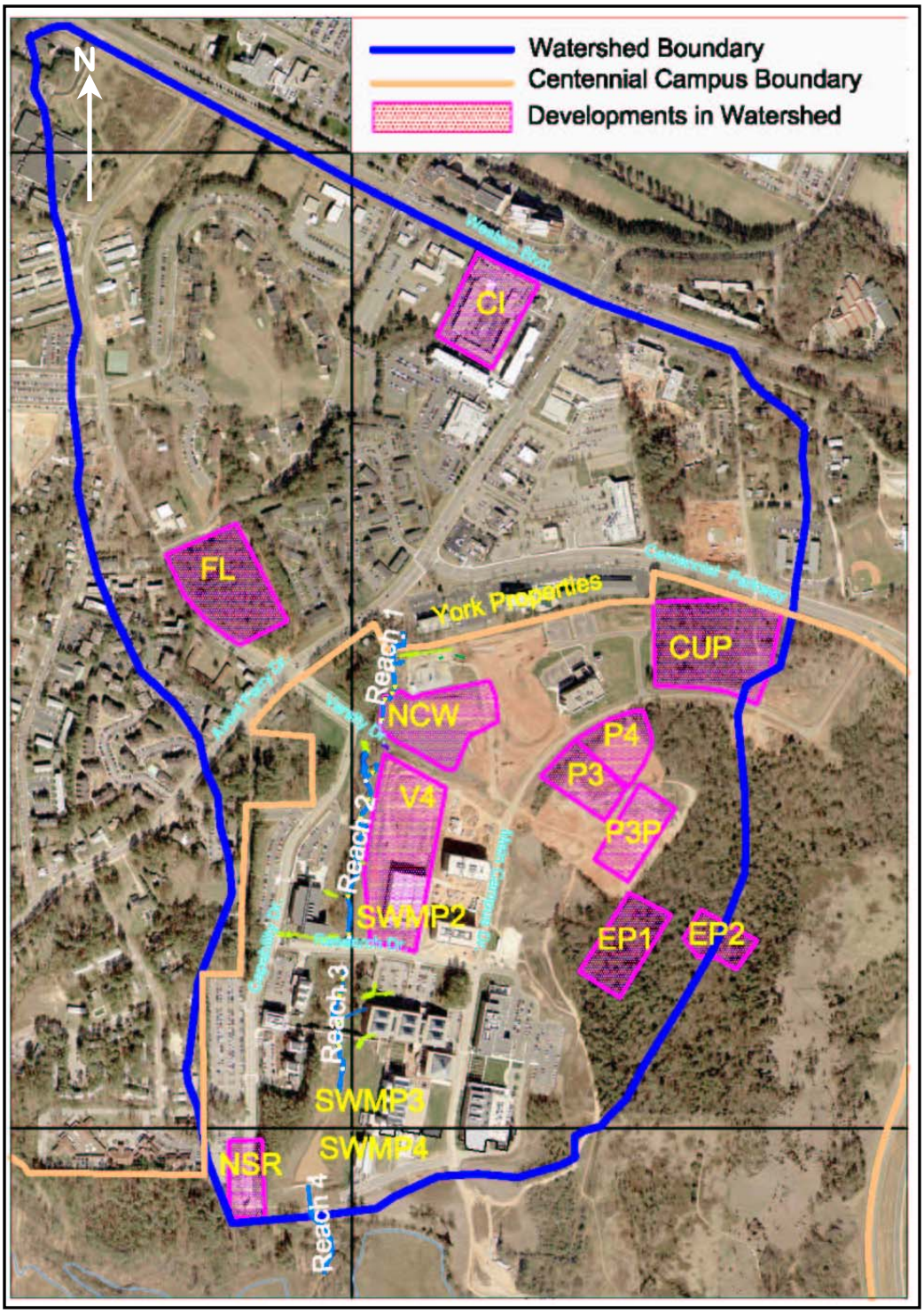

Figure 1.1: The North Creek Watershed (Reference: Aerial photo, Wake County, NC GIS Department; Digitized Layers, NCSU Facilities; Illustrated by Ian Jewell, NCSU Water Quality Group). Scale: 1 in = 775 ft.

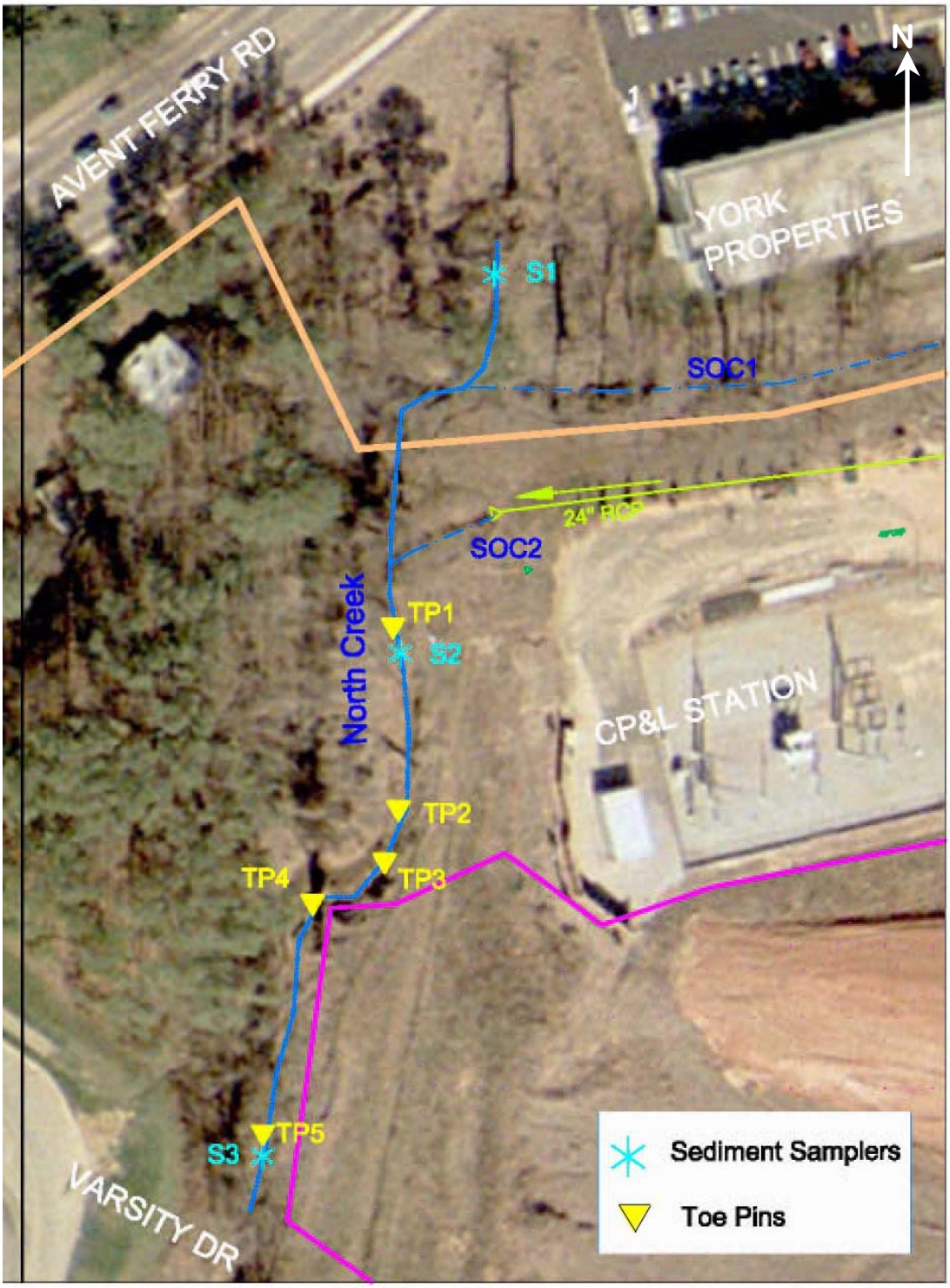

Figure 1.2: North Creek Reach 1 and Research Locations (Reference: Aerial Photo, Wake County, NC GIS Department; Digitized Layers, NCSU Facilities; Illustrated by Ian Jewell, NCSU Water Quality Group). Scale 1 in = 80 ft.

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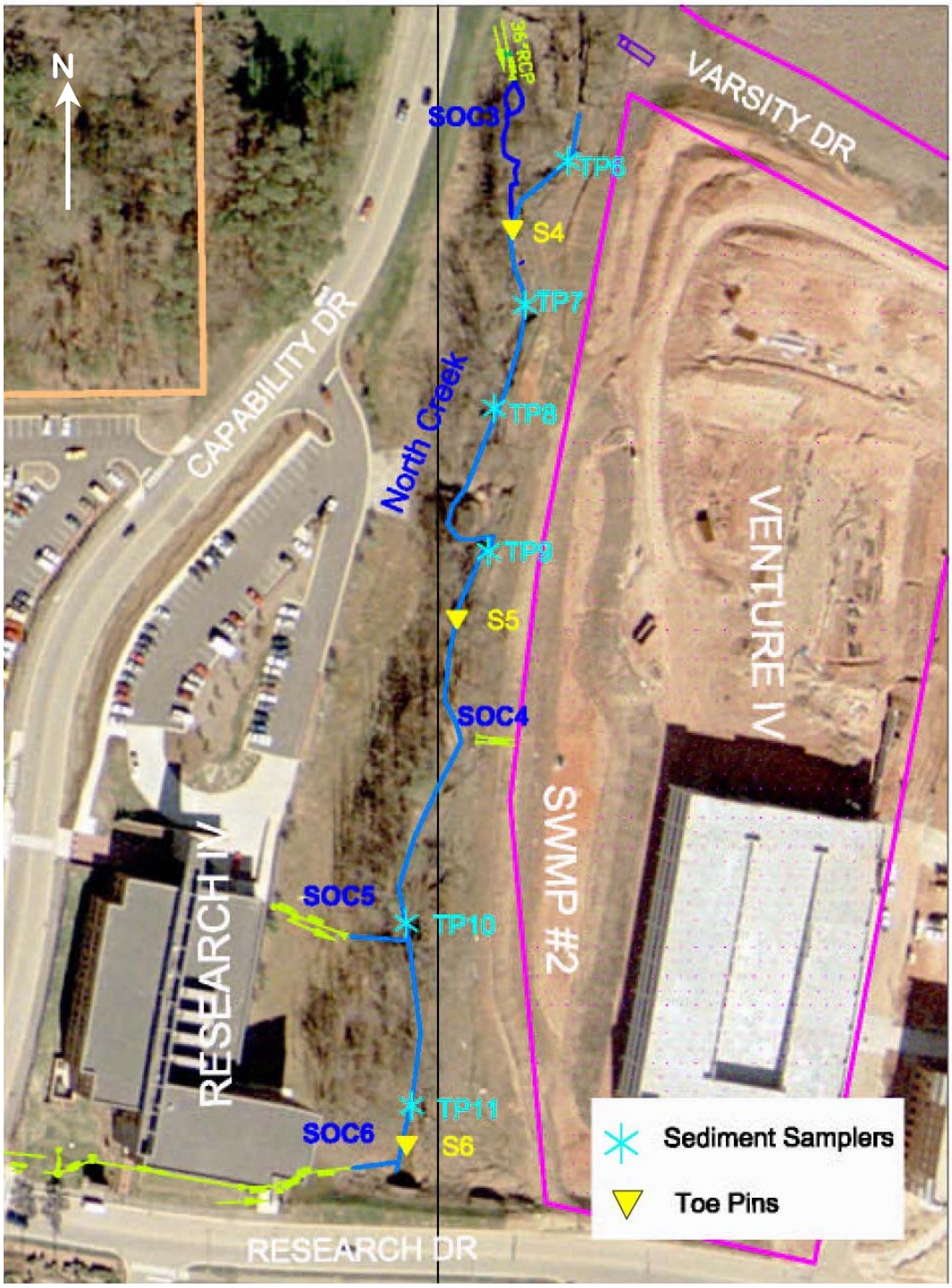

Figure 1.3: North Creek Reach 2 and Research Locations (Reference: Aerial Photo, Wake County, NC GIS Department; Digitized Layers, NCSU Facilities; Illustrated by Ian Jewell, NCSU Water Quality Group). Scale: 1 in = 125 ft.

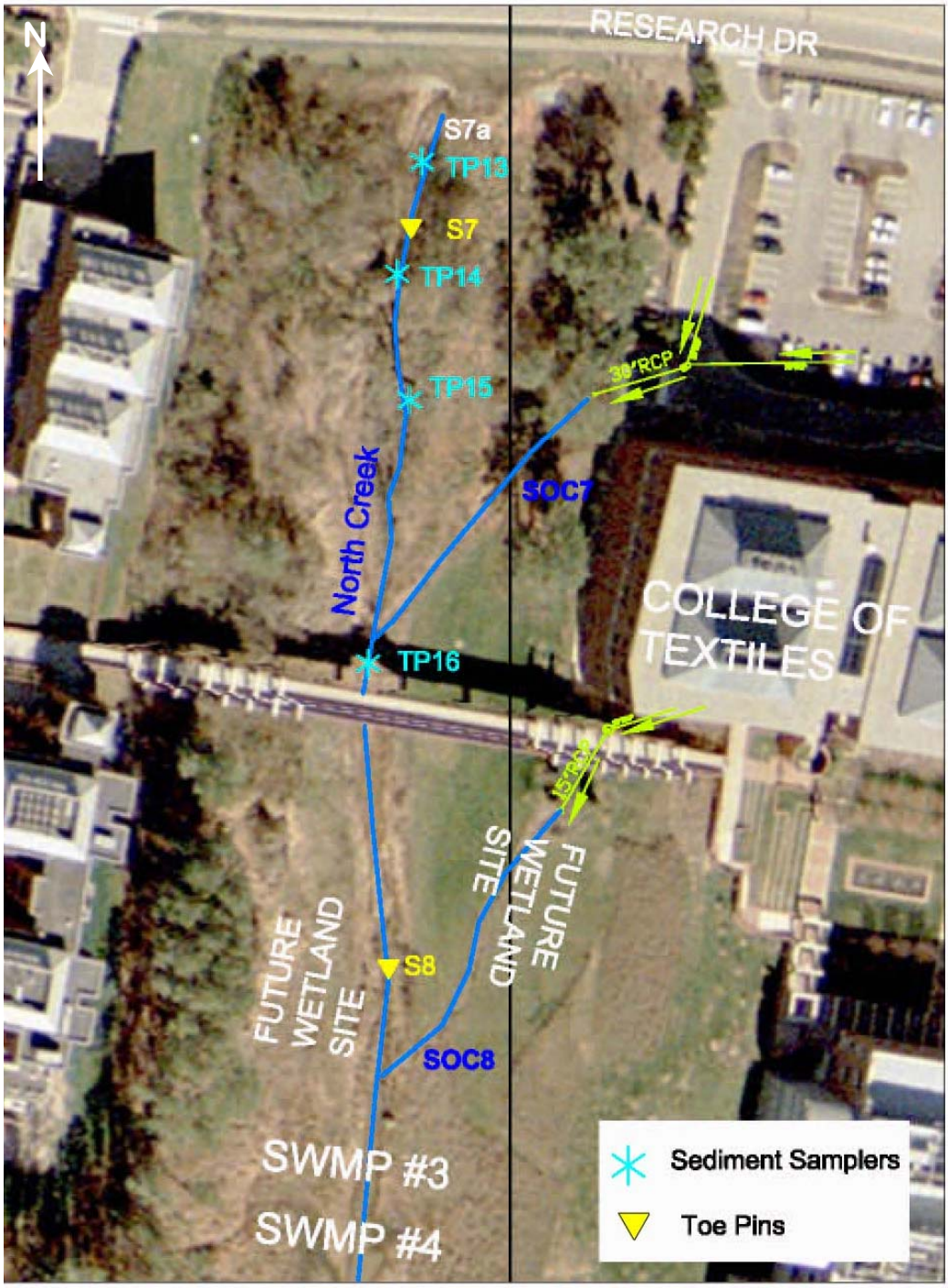

Figure 1.4: North Creek Reach 3 and Research Locations (Reference: Aerial Photo, Wake County, NC GIS Department; Digitized Layers, NCSU Facilities; Illustrated by Ian Jewell, NCSU Water Quality Group). Scale: 1 in = 90 ft.

Table 1.1: Land Uses in the North Creek Watershed.

Watershed Land Use	Area (Acres)	% Watershed Area
Impervious Surface (roads, parking lots, rooftops)	135	40
Lawn and Open Space	162	46
Wooded	50	14
Total	347	100

Table 1.2: North Creek Watershed Development Between August 2002 and August 2004 (Reference: Centennial Campus Partnership Office).

Development Project	Map Code	Start Date	End Date	Footprint (Acres)
The Campus Inn -- Residential Housing (Private)	CI	2003	2005	4.0
Flex Lab (NCSU)	FL	2002	2004	5.0
Centennial Campus Sites				
College of Engineering Phase 1	EP1	1/1999	4/2004	3.0
Stormwater Management Pond #2	SWMP2	1/2003	2004	1.0
Venture IV with Parking Deck	V4	2001	2004	5.0
Central Chiller/ Steam Plant Phase 1	CUP	2002	2004	6.5
North Carolina Wildlife Resources Building	NCW	2/2000	2005	4.0
College of Engineering Phase 2	EP2	2/2003	4/2004	1.5
North Shore Residential Phase 1	NSR	3/1999	3/2006	1.5
Partners III	P3	4/2002	4/2004	2.0
Partners III, COE 1&2 Parking Deck	P3P	4/2002	4/2004	2.0
Partners IV	P4	4/2003	3/2007	2.0

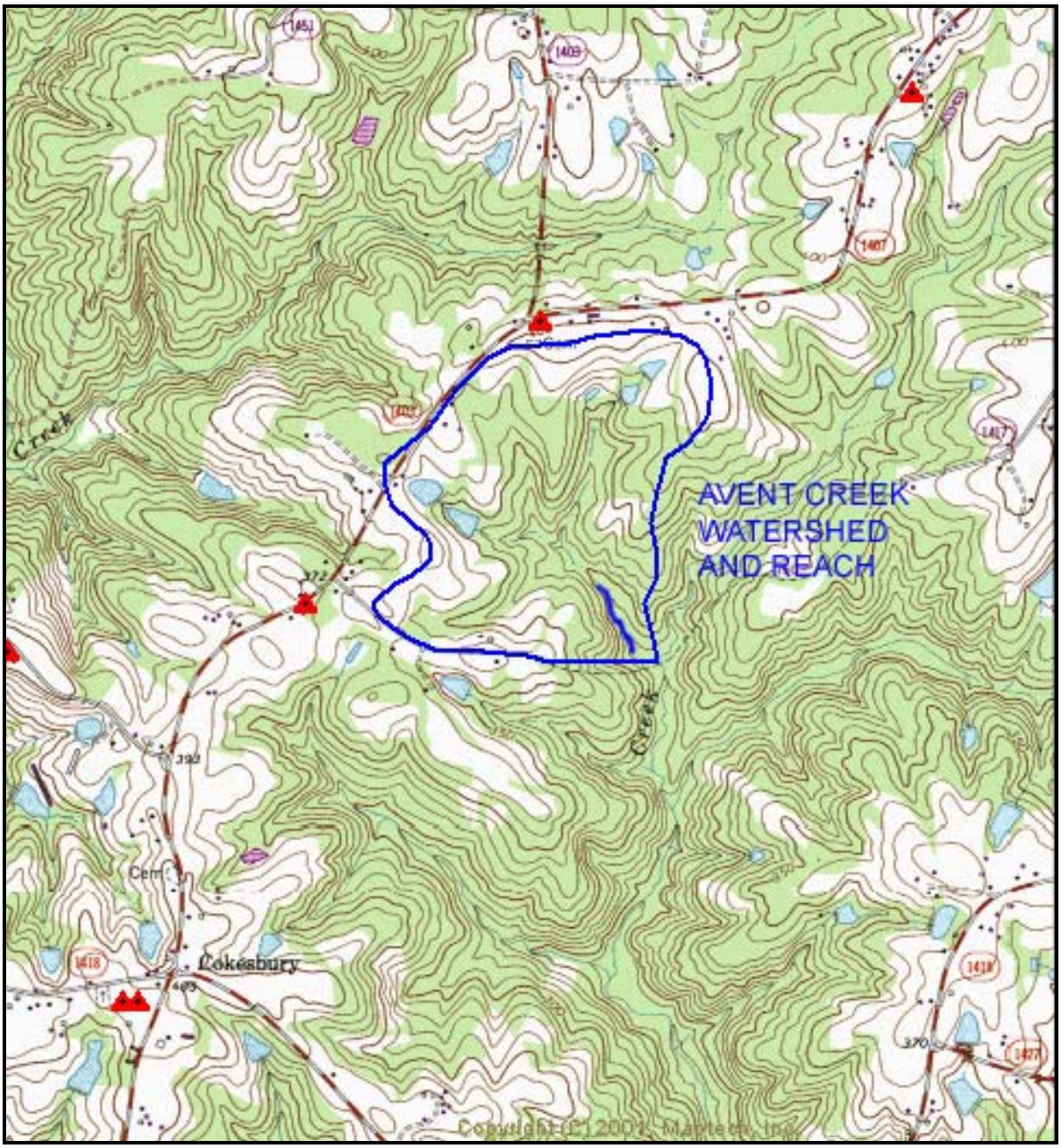

Figure 1.5: The Avent Creek Reach, Watershed, and Location Map. Approximate Scale: 1 in = 0.2 mi.

Watershed and Stream Corridor Soils

A soils map of the North Creek watershed is shown in Figure 1.6, with soil descriptions summarized from the Wake County, NC Soil Survey (SCS-USDA, 1970). Based on the soil survey, the predominant soil series in the watershed are Cecil, Worsham, Colfax, and Mantachie. Surface soil textures are either sandy loam or clay loam, with sandy loam most prevalent along drainageways, tributaries, and the main channel. Characteristics of each series are provided in Table 1.3.

Due to the high degree of channel incision and degradation in the upper reaches of North Creek, the streambed elevation is below the soil profile exposing saprolite layers. Saprolite is derived from the chemical weathering of bedrock, which can remove a large portion (60%) of the rock mass by solution without affecting volume. The amount of mass lost per volume of rock during saprolite formation is dependent on groundwater availability and its rate of movement (Horton and Zullo, 1991). The erosion potential of the saprolite layer exposed in North Creek could be dependent on the rate and degree of weathering, the shear stress of streamflows, and water chemistry of surface and groundwater. This saprolite layer typically has blocky and platy edges, and removed blocks easily crush to coarse sand.

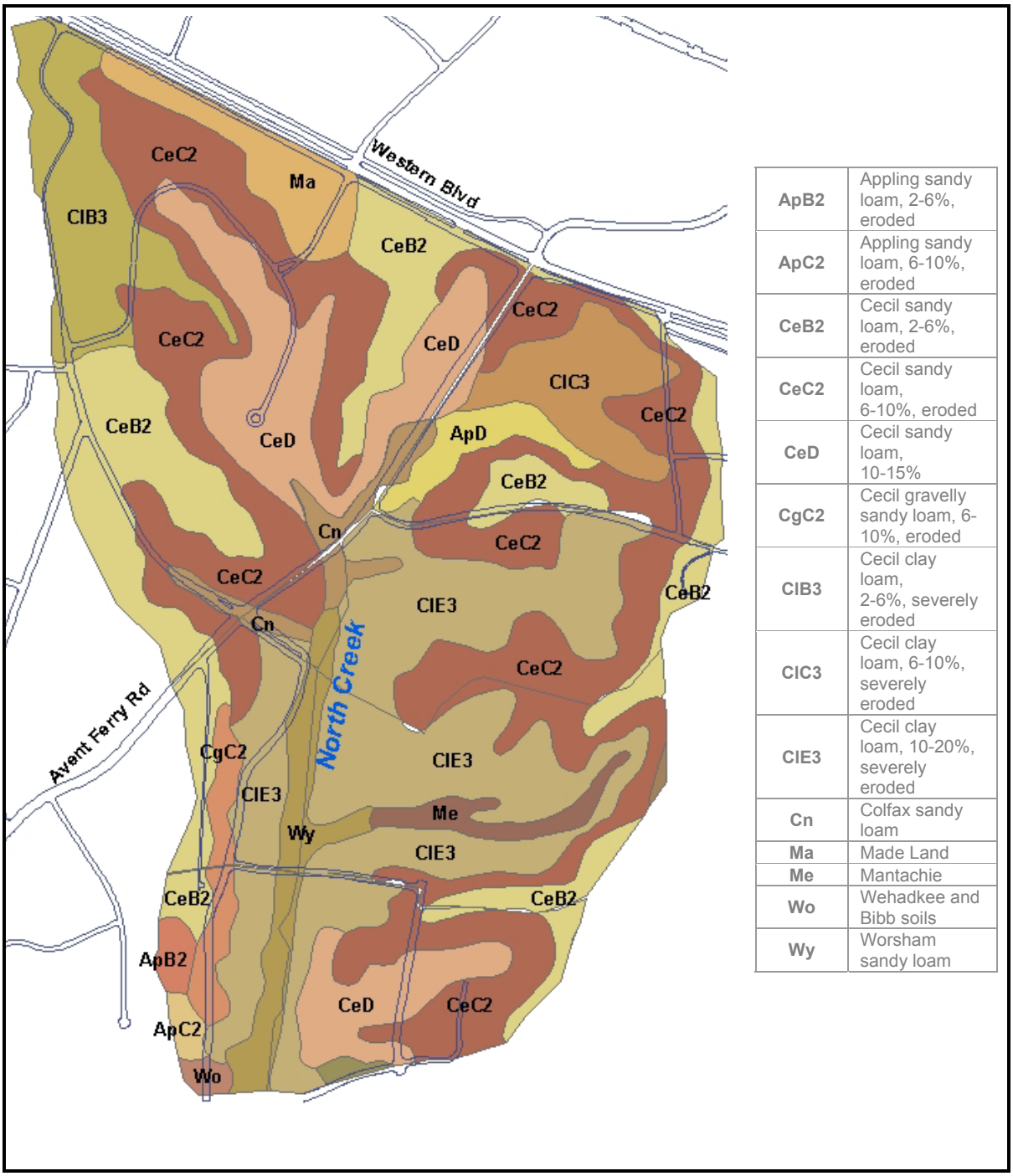

Figure 1.6: North Creek Watershed Soils (Data: Wake County, NC GIS Dept; Map provided by Kris Bass, NCSU Water Quality Group).

Table 1.3: Characteristics of the Most Common Soil Series Present in the North Creek Watershed (Wake County, NC Soil Survey 1970).

Soil Series	Classification	Infiltration/ Surface Runoff	Permeability	Drainage	Depth to Seasonal High Water Table (ft)	Depth to Bedrock (ft)
Worsham	Clayey, mixed, thermic Typic Ochraquults	Good/Medium to Slow	Moderately Slow	Poorly	Near Surface	5 - 15
Colfax	Fine loamy, mixed, mesic Aquic Fragiudults	Good/Medium to Slow	Moderately Slow	Somewhat Poorly	1.5	5 - 15
Mantachie	Fine-loamy, siliceous, acid, thermic Aeric Fluventic Haplaquepts	Good/Slow to Medium	Moderate to Moderately Rapid	Somewhat Poorly	2	5 - 15
Cecil sandy loam	Clayey, kaolinitic, thermic Typic Hapludult	Good to fair (eroded)/Medium to Rapid	Moderate	Well	Generally below solum	5 - >15
Cecil clay loam	Clayey, kaolinitic, thermic Typic Hapludult	Poor/Rapid to Very Rapid	Moderate	Well	Generally below solum	5 - >15

Watershed Hydrology

In small urban watersheds, the hydrology is predominantly controlled by precipitation and stormwater runoff and their associated effects on streamflow (Haan *et al.,* 1994). Due to the large amount of impervious surface in the North Creek watershed, the streamflow responds immediately and rapidly to storm events with steep peaks in the hydrographs. Baseflow is maintained by groundwater inputs, which can be reduced in urban watersheds due to limited infiltration (Simmons and Reynolds, 1982). Baseflow is relatively low in North Creek (about 0.3 cfs) compared to stormwater flows, with some portions in the lower reach dry during the summer months.

The hydrology of North Creek is heavily influenced by stormwater runoff discharged directly into North Creek or onto the floodplain (Figure 1.1). This stormwater is delivered to North Creek in stormwater conveyance systems that are difficult to map in urban settings. Installations and changes are commonly not documented adequately. In Reach 1, there are two significant stormwater outfall channels (SOCs) entering near the head of the reach (Figure 1.2). The first channel (SOC1) flows along the York Property boundary (500-ft length) and drains an unknown portion of Mission Valley and the York apartments in the upper watershed. This channel is steep and relatively wide in places due to excessive erosion. The second channel (SOC2) is a severely eroding draw that drains from a culvert about 50 ft from North Creek. The subwatershed is unknown in size due to lack of stormwater conveyance system data, but extends east to the Chiller/Steam Plant Site (CUP) built during the study.

In Reach 2, there are four significant stormwater outfall channels along the reach (Figure 1.3). The first channel SOC3 (70-ft length) enters on the right bank just past the Research Drive culvert. This channel formed due to stormwater erosion caused by flow from two culverts at the corner of Research Drive and Capability Drive, which drain Varsity Drive, portions of Avent Ferry Road and associated apartments, and the new Flex Lab construction (FL) to the west. Over the course of the study, this channel eroded into North Creek, taking a 25-ft section of streambank out and contributing numerous downed trees into the channel. The second stormwater outfall (SOC4) was installed during the study and discharges the outflow from SWMP #2 as it intercepts about 25-ac of watershed drainage on the eastern side. The culvert was installed within the left streambank with a large riprap apron spilling into the stream. The third and fourth stormwater outfall channels (SOC5 and SOC6) enter on the right streambank from culverts draining Research IV building and parking areas to the west. These are small channels (50-ft length), and relatively stable with dense riparian vegetation and low gradient.

In Reach 3, there are two significant stormwater outfall channels at the end of the reach (Figure 1.4). The first channel is SOC7 (210-ft length), flowing diagonally across the floodplain and entering the left streambank. The ephemeral tributary creates a braided stream pattern with mid-channel bar upon entrance. Stormwater enters the eroding channel from a culvert by the College of Textiles chiller facility that drains the above parking lot and portions of the building (< 5-ac area). The second channel downstream is SOC8 (160-ft length), which also drains stormwater from the COT building (1.6 ac area) and runs diagonally across the floodplain downstream. This channel was eroding as well as causing

streambank erosion at the entrance to North Creek until removal by stormwater wetland construction, described in later chapters of this dissertation.

Elements of watershed and stream hydrology measured in this study included precipitation and streamflow. Rainfall data were obtained for the entire study period from the NOAA National Weather Service Regional Forecast Office (Research III, Centennial Campus, Raleigh, NC). These data were collected every six hours using a tipping bucket rain gauge at the facility. Daily precipitation between August 2002 and December 2004 is shown in Appendix 1, Figures A1.1-A1.3, for the two years of the stream assessment research and for the three months during the constructed stormwater wetland research. Monthly rainfall totals were summarized (Table 1.4), along with the significant storm events that occurred during the study (Table 1.5). To help verify local campus data, hourly precipitation data were also obtained from the Lake Wheeler Road Field Laboratory (NC State Climate Office, NC CRONOS Database) and used for comparison.

Streamflow in North Creek was measured continuously starting in April 2003 just below the Research Drive culvert. An automated bubbler flow meter (Teledyne ISCO Model 4230) was installed in coordination with an automated water sampler (Teledyne ISCO Model 3700). The station was initially at S7 (Figure 1.4) until the end of July 2003 when a large storm and associated streambank erosion endangered its position. The station was moved closer to the culvert (S7a) after this. Streamflow measurements between April 2003 and August 2004 are provided in Appendix 1, Figure A1.4 and Figure A1.5, along with precipitation data for comparison. Time intervals where the streamflow data was missing are indicated by a star. During the largest rainfall events, the streamflows were

not recorded due to an intake tube washout caused by the high discharges. The largest recorded discharge is included in Table 1.5, associated with a rainfall event of 0.87 inches.

Prior to the installation of the automated flow meter, manual streamflow measurements were taken at both sampling stations using a hand-held flow meter (Swoffer Model 3000) during various size storms. The resulting stage-discharge curves and best-fit power equations were used to calibrate the automated flow meter (Appendix 1, Figure A1.6 and Figure A1.7). Stage heights for both flow measurement methods were obtained from established gage plates, which were also used to verify the accuracy of the automated stage measurements during sample collection.

Table 1.4: Monthly Rainfall Totals (Inches) for the North Creek Study Interval (Reference: NOAA, Raleigh Regional Office).

	Stream Research Intervals		Wetland Research Intervals	Normals
Month	**2002-2003**	**2003-2004**	**2004**	**30-yr Average**
August	5.37	8.57	9.97	4.02
September	1.59	3	5.41	3.19
October	8.33	3.92	2.14	2.86
November	3.74	1.62		2.98
December	5.67	3.38		3.24
January	2.31	1.64		3.48
February	4.89	3.53		3.69
March	5.29	2.52		3.77
April	4.77	2.38		2.59
May	4.33	2.66		3.92
June	6.52	3.43		3.68
July	7.38	4.27		4.01
Total	**60.19**	**40.92**	**17.52**	**41.43**

Table 1.5: Significant Rainfall Events (>3.0 in) and the Maximum Streamflow Recorded During the North Creek Study.

Storm Event Date	Rainfall (in)	Streamflow (Rainfall)
10/11/2002	4.23	> 148 cfs (0.87 in)
7/29/2003	3.56	
8/8/2003	3.61	

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CHAPTER 2: CHANNEL EROSION MEASUREMENT AND PREDICTION

INTRODUCTION

Channel Erosion

Channel erosion creates many significant problems in a watershed through the associated loss of surrounding property, lowering of the groundwater table, and increased hillslope instability. Channel erosion also leads to significant problems within the stream by contributing excess sediment derived from both the streambanks and streambed (Fischenich 1989; Lawler *et al*., 1997). Streams experiencing channel erosion lose valuable functions, including their ability to carry water and sediment effectively, and their ability to provide ecological habitat. The major cause of channel erosion in small urban watersheds is related to land use changes. Development increases impermeable surface area, reducing stormwater infiltration and storage. Stormwater runoff volumes and peak discharges within the stream are increased considerably. As a response to these flashy stormflows, stream channels will erode due to increased shear stress (Dunne and Leopold,1978; Leopold *et al*., 1992). Development also alters the sediment dynamics of a stream channel. During the initial stages of development, large sediment loads are contributed in stormwater runoff, derived from construction sites and unstable hillslopes (Wolman and Schick,1967; Goldman *et al*., 1986). Once development has been completed, the large volumes of stormwater runoff become devoid of sediment. This "hungry" water gains sediment from erosion of the streambanks and streambed. For small urban watersheds (< 10 sq mi), the change in channel dimensions can continue for long

periods of time, even after watershed development has been completed (Hammer, 1972; Whipple *et al*., 1981).

In addition to watershed changes, channel erosion can also be facilitated by in-stream modifications such as channelization to increase development area and to decrease localized flooding. In a meandering channel, stability is maintained by the balance of erosion on outer banks and deposition on the inner banks (Leopold *et al*., 1992). The morphological results of channelization include a shortened channel length and increased channel gradient. Streamflow velocities are increased, leading to channel instability and downstream erosion that will typically migrate upstream as a headcut into the altered reach.

The processes that lead to channel instability and erosion in response to a particular disturbance tend to occur in a predictable pattern over time. These dynamic processes performed by streams over time have been described as channel evolution, with several stages identified (Schumm *et al*., 1984; Simon, 1989). The undisturbed stable channel (Stage I) experiences a disturbance such as watershed change or channelization (Stage II). Channel degradation (Stage III) is initiated in order to adjust to the disturbance, which in urban watersheds is the increased discharge rates associated with stormwater runoff. During this stage, degradation is characterized by vertical incision, reduction in channel gradient, and an increase in streambank height. Streambank instability progresses as banks reach their critical heights and angles (Simon and Darby, 1999). Eventually mass wasting and slumping of bank material occurs due to the added weight, initiating the channel widening process (Stage IV). Channel degradation migrates upstream in the form of a

headcut, while aggradation occurs downstream as eroded material is deposited at the toe and transported downstream with the flow (Stage V). Since the channel downstream has been degraded with a reduction in channel slope, the flow velocity and, hence, the sediment transport capacity is reduced. The material aggraded from upstream is readily deposited in this downstream channel, but not in the same magnitude as it was removed by degradation (Simon, 1992). The final stage of channel adjustment is quasi-equilibrium (Stage VI), when the channel has widened and bank slopes have flattened to their fullest potential. Riparian vegetation is once again established on the banks and aggraded surfaces, and a new channel is established at a lower bed elevation. This new channel has a reduced gradient, allowing extended and elongated meanders to establish.

The ultimate outcomes of channel erosion and instability are changes in channel bed elevation, channel gradient, and channel size through aggradation or degradation. A disturbed channel generally has less resistance to shear stress, and will change its cross section over time in order to effectively carry the range of discharges and sediment loads delivered. Erosion is a mechanism employed by the channel to attain balance between the shear stress applied by stormflows and the resistance of the stream banks and channel bed. The culmination of channel evolution is typically an over-widened channel with a new smaller floodplain and channel established inside. The new channel and floodplain are sized appropriately for the bed elevation, and allow for adaptations to discharge and sediment supply from the watershed (Dunne and Leopold, 1978). This channel readjustment following disturbance and erosion usually requires a longer time period than the disturbance that initiated the process. During this recovery time, channel erosion and

adjustment have been shown to significantly increase sediment loads delivered downstream, particularly in urban watersheds (Wolman and Schick, 1967; Hammer, 1972; Trimble, 1997).

Streambed Erosion

In small urban channels, degradation of the channel bed is primarily a result of increased stormflows introduced into the channel with excess flow energy and increased shear stress. These stormflows have larger sediment transport capacities, with the ability to capture and move greater amounts and sizes of substrate particles in the overlying water (Simon and Darby, 1999). Channel incision typically occurs with an increase in stormflows, creating changes in the longitudinal profile that are dependent on the degree of disturbance (Schumm *et al*., 1984). During the degradation process, high sediment loads can be contributed downstream while a headcut typically migrates upstream.

In response to disturbance, several mechanisms resist and control vertical incision of the channel. Soils at the bed elevation play an important role in the rate of degradation, with cohesive silts and clays shown to be less resistant than sands that tend to encourage widening versus deepening (Schumm, 1977). Channels may also be underlain with bedrock, resistant saprolite, or have an artificial armoring layer that inhibits vertical erosion (Simon and Darby, 1999). Resistance to channel degradation can be affected by a change in the bed elevation downstream due to aggradation, impoundments, or an increase in stream sinuosity. If the bed elevation is raised downstream, the channel gradient can be reduced and limit bed erosion caused by streamflow shear stress. The channel bed will

never fully recover through aggradation, however, because degradation rates are always more rapid and more extensive (Simon, 1992).

Streambed degradation and changes in stream gradient over time can be measured using longitudinal profile surveys of the reach. Channel incision and localized aggradation can be readily identified through changes in bed elevation over stream length. The longitudinal profile is an effective tool to indicate how stable the stream is over time, and what stages in channel evolution currently exist. Profiles can also provide information on rates of headcut migration and incision, and may be used to estimate sediment yield from channel bed erosion.

Streambank Erosion

Streambank erosion is defined as the detachment, entrainment, and removal of streambank material, either as individual grains or as aggregates, by fluvial or subaerial processes (Lawler *et al*., 1997). The two main forces acting on streambanks that cause bank erosion and bank failures are hydraulic forces and geotechnical forces.

Hydraulic force is created by the stress of flowing water, and erosion occurs when this stress exceeds the critical sheer stress of the bank soil materials (Thorne, 1982; Simon, 1992). Hydraulic forces include velocity and velocity gradients, boundary shear stress, up and down-welling currents near the bank, and eddy circulations (Rosgen, 2001). Factors that influence hydraulic force include streamflow; the distribution of stream power and shear stress in the channel; and the intensity of turbulence (Keller *et al*., 1990). The erodibility of the streambank by these forces depends on resistance of the soil material to detachment and entrainment by the fluid shear applied. Erosion due to hydraulic forces

usually involves the removal of both individual particles and aggregates, with resistance determined by the physical and chemical bonding forces between soil particles (FISRWG, 1998). Evidence of these forces includes scour marks along the bank and excessive undercutting (USACE, 1981).

Geotechnical forces affect streambanks when the force of gravity exceeds the shear strength of the bank material. Bank erosion due to these forces usually occurs as bank collapse or mass failure of bank materials compared to entrainment of particles and aggregates by hydraulic forces (FISRWG, 1998). These forces are related to chemical weathering of soil, and to the amount of moisture within the banks (USACE, 1981; Thorne, 1982). Changes in subsurface moisture content occur during stormflow rise and recession, and during changes in the groundwater table elevation. Streambanks with higher moisture contents have been shown to experience an increase in erosion (Wolman, 1959; Hooke, 1979), and bank collapse can be significant after high flows due to saturation of the soils (Leopold *et al*., 1992). Saturation adds weight to the bank material, reduces cohesive forces between soil particles, and increases pore pressure within the bank as the water drains out of the profile (FISRWG, 1998). Evidence of geotechnical erosion can found by the presence of soil blocks derived from the upper bank profile deposited at the bank toe; seepage or piping of water from the bank face; or by significant areas of complete bank collapse (USACE, 1981).

There are many factors that can influence the effects of hydraulic and geotechnical forces on streambanks. The soil profile composition and degree of stratification are important bank characteristics that determine susceptibility to erosion. Channel geometry

is another important factor that influences the stability of banks, including the width to depth ratio, bank height, and bank angle (Keller *et al*., 1990). Incised streams with steep banks loose floodplain access during high flows, leading to bank erosion resulting from the altered distribution of shear stress and to very high velocity gradients (high boundary stress) near the streambanks (Rosgen, 1996). Another factor affecting bank erosion is the relative amount and composition of riparian and streambank vegetation. Vegetation can physically stabilize streambanks and uptake excess moisture to the extent of the root zone, but may contribute to excess loading stress if the streambank is unstable (FISRWG, 1998; Thorne and Osman, 1988). The vegetative portion of plants exposed to the streamflows has been shown to reduce erosion by decreasing the shear stress and localized velocity of high flows with an increase in friction (Kondolf and Curry, 1984; Shields *et al*., 1995; Harmel *et al*., 1999). Simon and Collison (2002) used a bank stability model and found that mechanical and hydrological effects of both riparian trees and grasses were important in determining streambank stability.

Measured streambank erosion rates recorded in the literature have been variable and dependent on the size of the stream. Most streambank erosion assessments found in the literature were in larger watersheds and rivers. Extreme streambank erosion rates have been observed in the Gila River, Arizona (50 m/yr), and the Toutle River, Washington (100 m/yr) (Simon, 1992). The erosion rates along North Creek are expected to be smaller than for a large river, but larger than a comparable watershed without urban development.

Streambank Erosion Prediction: BEHI Assessment and Streambank Erodibility Model

Quantification of actual bank erosion rates usually requires extensive field measurements over long time intervals. Current research has focused on deriving models to help predict streambank erosion. Although computer models for assessing bank erosion potential are becoming increasingly available, the most common model used in existing condition assessments is the bank erosion hazard index (BEHI) that provides direct, rapid assessment of streambanks along a particular reach (Rosgen, 1993a). BEHI is typically used in North Carolina for standard stream assessments to evaluate existing impaired conditions and to help better design stream restoration projects. The index was designed for widespread application, but may not be as applicable to modified streams with site-specific influences and to streams that experience less than or greater than bankfull stormflows (Rosgen, 1996).

The BEHI assessment allows relative estimates of bank erosion potential based on five variables that quantify bank characteristics and observations of soil profile composition (Rosgen, 1996). The first variable is the bank height to bankfull height ratio, which estimates vertical stability. Short banks are more stable compared to steep banks due to geotechnical forces such as gravity. The second variable is the ratio of rooting depth to bank height, and the third variable is the relative density of these roots estimated as a percentage of bank surface area. Streambanks with deeper rooting depths and higher root densities are more stable due to the anchoring ability of the roots. The fourth variable is an estimate of percentage surface area protected by vegetation and debris. When greater amounts of bank surface area are exposed to high flows, there is a greater potential for

bank material to be detached and entrained by the flow. The fifth parameter is bank angle, as measured from the top of bank. Larger bank angles are typically an indication of bank instability and an increase in geotechnical bank failure potential. After the five variables of the BEHI assessment are measured, a risk rating number between 1 and 10 is assigned to the variable based on ranges of measurements or percentages. The ratings are then tabulated and a category of the index is assigned for total numerical ranges. There are six categories ranging from very low to extreme potential for bank erosion. Adjustments to the numerical total are made for general bank soil composition and stratigraphy.

For a more inclusive streambank erodibility model, the BEHI assessment has been coupled with estimates of near-bank shear stress (NBS) and an associated index (Rosgen, 1996). The NBS index evaluates the erosion potential from hydraulic forces applying stress to the streambanks during stormflows. Estimations of near-bank shear stress should take into account the stage of flow, the channel slope, the channel width to depth ratio, and the relative location along the reach with attention to channel planform features. Near-bank shear stress will vary depending on the presence of riffle or pool, and large depositional features are particularly important because they create high velocity gradients and shear stress on the bank due to the increased flow depth and local slope (Rosgen, 1996).

Measurements of near-bank shear stress can be performed using three different methods (Rosgen, 1996). Velocity gradients can be estimated for the reach by determining vertical velocity profiles with corresponding velocity isovels and used to estimate shear stress (Leopold *et al*., 1992; Rosgen, 1996). A simpler method uses the ratio of near-bank

area to total area at the bankfull stage, in order to get a rough estimate of near bank shear stress. The third method is similar, but utilizes the ratio of near-bank shear stress to the mean shear stress applied across the entire bankfull channel. Average shear stress within streams can be calculated by: $\tau = \gamma\ d\ s$; where γ is the specific weight of water, d is the average depth of flow, and s is the channel slope (Ward and Trimble, 2004). Since the specific weight of water and channel slope are constant, the near-bank shear stress ratio can be reduced to the average depth of the near-bank third divided by the average depth of the bankfull cross-section. This ratio compares the effects of the streamflow closest to the streambank to the overall shear stress applied to the channel. For all three methods, only estimates are possible due to the variability within a stream reach. These estimates are generally accepted, however, due to the impractical nature of exact measurements for the entire stream reach. Similar to the BEHI portion of the model, quantities of near-bank shear stress are converted to a bank erosion risk rating system based on ranges of values (Rosgen, 1996).

The streambank erodibility model developed using the BEHI and NBS classifications was derived based on observed erosion rates during bankfull discharges in Midwest streams (Front Range, Colorado and Yellowstone National Park, Montana). For each region, an independent streambank erodibility relationship was determined (Rosgen, 1993b; Rosgen, 2001). The model allows erosion rate predictions in order to estimate the amount of sediment contributed from streambanks in a given reach. The potential lateral erosion rates predicted can be multiplied by bank height and channel length of similar condition to yield an annual sediment volume derived from stream (Rosgen, 2001).

This model has been tested in different streams and rivers with variable results. The Weminuche River, Colorado (Rosgen, 2001) and the Mitchell River in North Carolina (Harmon and Jessup, NCSU Water Quality Group, personal communication) were found to have similar erosion rates to those predicted by the Colorado streams relationship due to the similar meandering alluvial channels. The East Fork San Juan River, Colorado was found to have similar erosion rates to those predicted by the Yellowstone National Park stream relationship due to the similar braided channel pattern. Maggie Creek, Nevada (Rosgen, 1996) did not have similar measured erosion rates between 1992-93 to those predicted by the model relationships due to excess rainfall and snowmelt flows during the assessment period. A study conducted on the Illinois River, Oklahoma also measured higher bank erosion rates than those predicted by the model (Harmel *et al*., 1999). These results may have been due to the different stream type (stream classification, Rosgen, 1996), different soils comprising the river banks, and the larger than bankfull stormflows experienced during the study. Research conducted at NCSU by Patterson (2001), evaluated streambank erosion along cross-sections in seven stream locations in North Carolina. The average bank erosion rates estimated in that study were less than rates predicted by the model based on the Colorado data. Both Harmel *et al*., (1999) and Patterson (2001) found poor correlations between the BEHI classification and erosion rates measured based on the relationship between near-bank shear stress, BEHI, and corresponding erosion rates.

Channel Erosion Study

The purpose of this study was to measure channel erosion over a two-year period in a small urban stream undergoing intensive watershed development. The research included estimates of both streambed and streambank erosion along the three reaches of North Creek delineated by large in-stream road culverts. The purpose of this research was also to apply a common streambank erodibility model in North Creek, which uses the BEHI and NBS indices to predict annual streambank erosion rates. These predicted erosion rates derived from the model were compared to observed erosion rates. The objectives of this channel erosion study were:

1) To determine if the channel bed elevation measured by longitudinal profiles changed over time in the three North Creek reaches and to compare changes in the three reaches based on channel evolution concepts.

2) To measure streambank erosion using cross-sections and toe-pins along each North Creek reach, and to compare the annual erosion rates between reaches based on channel evolution concepts.

3) To classify the streambanks using the BEHI assessment tool and to evaluate changes in classification over time.

4) To compare streambank erosion rates predicted by the streambank erodibility model with streambank erosion rates measured in the North Creek reaches.

METHODS

Streambed Erosion

The longitudinal profiles of the North Creek reaches were surveyed in January 2003 and July 2004, with an additional survey of Reach 3 in November 2003. Reach 1 was surveyed between the first stormwater outfall channel (SOC1) along the NCSU property boundary and the entrance to the Varsity Drive culvert. Reach 2 was surveyed between the Varsity Drive culvert exit and the entrance to the Research Drive culvert. Reach 3 was surveyed between the Research Drive culvert exit and the pedestrian bridge (See Chapter 1, Figure 1.1). The longitudinal profile was composed of channel bed elevation measurements in the thalweg and water surface elevation measurements taken using a manual level (Topcon AT-G7 Auto Level) and survey rod. Survey points were collected at the head of riffle, head of pool, the maximum depth of the pools, and at other significant changes in bed elevation. The longitudinal station for each elevation measurement was recorded, using a surveyor tape stretched through the thalweg. Benchmarks along the reaches were also measured to help coordinate surveys over time. The surveys were input into Excel for graphical comparison and aligned using the benchmarks. Actual elevations of the benchmarks were derived from a digitized Centennial Campus map in AutoCAD Land Desktop3 (Autodesk, 2003), and applied to the longitudinal profiles.

Streambank Erosion

Eight permanent cross-sections were established in August 2002, and surveyed every three months for the first year, then every six months the second year. The sites were selected based on visual erosion potential and an effort was made to assess the variety

of bed features, including pool, riffle, and run in each reach. The surveys were performed using a manual level (Topcon AT-G7 Auto Level) and survey rod. Elevation measurements were recorded along a measuring tape stretched between two permanent pins placed beyond the tops of bank on the terrace. All significant features and changes in slope within the cross-section were measured.

Sixteen permanent toe-pins were established along North Creek at the base of individual streambanks. The toe-pins were made of 4-ft long aluminum pipe and hammered into the substrate as much as possible to a maximum depth of 2-ft. The toe-pin sites were initially selected to evaluate the effects of kudzu on streambank stability before and after herbicide application. The toe-pin sites were randomly placed, with no regard to erosion potential. Toe-pins in Reach 1 and Reach 2 were installed in December 2002, while toe-pins in Reach 3 were not installed until August 2003 after significant storms initiated large amount of erosion in this reach. Bank profile surveys were performed every six months by taking horizontal measurements between the bank face and a vertical survey rod placed along the pin. Horizontal measurements were collected about every half-foot from the lowest bottom elevation to the top of the bank.

For both cross-sections and toe-pin bank profiles, survey data were input into Excel to estimate the amount of bank material lost over the time interval between measurements, and to estimate changes to the channel cross-sectional area.

BEHI Assessment and Streambank Erodibility Model

Streambanks on both sides of the channel in each reach of North Creek were evaluated using the five BEHI assessment parameters in July 2002 and January 2003. The

parameters evaluated were: bankfull height to bank height ratio; rooting depth to bank height ratio; rooting density; percent surface protection by vegetation or debris; and bank angle (Rosgen 1996). Banks were measured for total height and bankfull height as measured from the thalweg. In most areas, bankfull heights could not be identified accurately due to extensive erosion. The Piedmont regional urban curve for bankfull mean depth was used to estimate the bankfull height for the corresponding drainage area when no reliable indicators were available (Doll *et al*., 2002). Rooting depth was measured using a measuring tape stretched from the top of bank down to the extent of root penetration. Rooting densities and percentages of surface protection were both estimated by subjective observations. Bank angles were estimated using the measuring tape and survey rod laid against the bank, with protractor measurement of angle.

Contiguous streambanks with similar characteristics were grouped into measured sections using the streambank erosion potential criteria. Within each designated section, the five parameters were evaluated and the bank soil profile was evaluated using the "feel" method (Brady, 1990) to determine texture in each horizon. Measured and estimated values for the five parameters were assigned a rating between 1 and 10. The ratings were tallied, and adjusted for dominant soil composition. The rating totals were then assigned BEHI categories ranging from Very Low to Extreme, based on established total value ranges (Rosgen, 1996). The streambank sections with assigned categories were input into AutoCAD Land Desktop3 (Autodesk, 2003), overlaying the digitized stream map provided by the professional channel survey in April 2003. Total stream length for each classification in each reach was determined for both assessments and compared over time.

The near-bank shear stress index (NBS) category was determined for each section of the stream classified by the BEHI assessment. Near-bank shear stress was calculated as the ratio of shear stress in the near-bank region to the shear stress in the bankfull channel. The equation for shear stress was $\tau=\gamma_w*d*s$; where γ is the specific weight of water (62.4 lb/ft^3), d is the mean depth in feet, and s is the slope of the water surface as a fraction (Leopold *et al*., 1992). For this application, the specific weight of water and the water surface slope were assumed to be equal across the channel width. The mean depth nearest the designated bank (1/3 bankfull width) was divided by the mean bankfull depth of the entire channel to estimate the near-bank shear stress ratio.

The average flow depths required for shear stress calculations were estimated by deriving cross-sections within each BEHI section. The cross-sections were extracted from the digitized stream map in AutoCAD and from the surveyed cross-sections used for bank erosion measurements. Cross-sections were drawn about every 25-ft along the stream, with at least 1, but no more than 3 cross-sections per each BEHI section. Some cross-sections were used for both banks if there was overlap between the BEHI sections on opposite sides of the stream. The cross-sections were then imported into Excel to determine average channel depths in each BEHI section. The first step to estimating depths was to estimate the bankfull cross-sectional area. The bankfull channel area was estimated using the hydraulic geometric relationships for urban streams in the North Carolina Piedmont (Doll *et al*., 2002). The drainage area (mi^2) for each BEHI section was estimated, and the bankfull cross-sectional area was then calculated using the equation $60.34(DA)^{0.65}$. Bankfull elevations were estimated using field indicators (Dani Wise-

Johnson, NCSU Water Quality Group, personal communication), but were difficult to identify accurately in most locations due to eroding streambanks. The elevations measured in the field (1.6-2.0 ft), were similar to elevations estimated using the urban regional curves.

Using the Excel spreadsheet, the bankfull cross-sectional area was fitted to the cross-sectional profile, and the mean bankfull depth was determined. The bankfull width for each cross-section was divided into thirds, and the mean depth from the outer third of the channel width near the bank of interest was estimated. The mean near-bank bankfull depth was divided by the mean bankfull depth of the entire cross-section, and an average near-bank shear stress ratio for all cross-sections within each BEHI section was determined.

An NBS category (Very Low to Extreme) was then assigned to each BEHI section using the determined value of the near-bank shear stress ratio and established value ranges for each category (Rosgen, 1996). The predicted erosion rates for each section were determined using published plotted relationships between BEHI and NBS (Rosgen, 2001), and compared to average erosion rates measured at the cross-section locations and the toe-pin locations during the study.

Statistical Analysis

Correlations, regression equations, and descriptive statistics were performed using JMP 5, a statistical package for data analysis (SAS Software, 2002).

RESULTS AND DISCUSSION

Streambed Erosion

The streambed was surveyed in January 2003 and July 2004 to develop longitudinal profiles in each reach of North Creek and to compare substrate elevations over time. The profiles were aligned using common benchmarks, and relative elevations measured in the field were corrected to actual elevations using the known benchmark elevation data. For each North Creek reach, factors thought to favor degradation during the course of this study included an increase in stormflow quantity and intensity coinciding with continued watershed development. The longitudinal profiles of the stream thalweg and water surface were expected to show this degradation through an increase in channel bed slope, through vertical incision following headcut migrations, or through significant localized scour within the profile.

Reach 1

The longitudinal profiles in Reach 1 were difficult to compare due to potential misalignments between the two surveys. The July 2004 survey was about 14-ft longer along the thalweg than the survey in January 2003, and significant pools and head of riffles did not align well between the two surveys (Figure 2.1). The misalignments could indicate errors in one of the two surveys, considering the relatively short channel length assessed. Changes in thalweg length and migrations in riffle and pool locations could be possible through addition of meander bends and a dynamically shifting substrate (Leopold *et al*. 1992), but these changes were assumed to require longer time periods than for this study.

Based on the survey data collected, there appeared to be considerable localized aggradation in both riffles and pools within this reach (Figure 2.1). Aggradation was visually observed during the study in many sections of the reach, but accumulations were not specifically measured using scour chains or substrate pins. The two cross-sections present within the profile (S2, S3) were near the beginning and end of the reach, and could only provide information on channel bed changes in those locations. Neither cross-section indicated aggradation or degradation of the bed over time, however (See Appendix 2, Figure A2.2 and Figure A2.3). The final bed elevation at the Varsity Drive culvert remained constant between the two surveys, even though localized bed elevations were fluctuating upstream. The overall channel bed slope was similar (0.0039-0.0042) in both survey from head of riffle near the stormwater outfall channel entrance to the culvert invert at Varsity Drive. The water surface elevation was consistently higher along the reach in the summer survey compared to the winter survey, while the slope was the same (0.0042).

Channel bed degradation in Reach 1 was not supported by the longitudinal profile data. Although the survey had potential errors, the data combined with observations during the study indicated that localized aggradation was probably occurring in both riffles and pools. The aggraded or shifting sediments deposited on the streambed were from two potential sources. One source was the relatively large suspended sediment load entering from the Avent Ferry culvert and the steeper stormwater outfall channels near the upstream end of the reach (See Chapter 4, Table 4.2). The fine sediment could have deposited upon entering Reach 1 since it had a comparably lower gradient that could have reduced the flow velocity (Leopold *et al*., 1992). Another source of sediment for aggradation was

eroded streambank material that slumped to the toe or were entrained during storm events and deposited during stormflow recession (Lawler *et al*, 1997). The cross-section and toe-pin surveys discussed in the streambank erosion section indicate that lateral streambank erosion was occurring within this reach (Figure 2.5-2.11).

The minimal channel bed degradation observed in Reach 1 could be due to several factors. The large instream bottomless culvert at Varsity Drive provided grade control for the channel bed slope in Reach 1. This culvert was heavily armored with small boulders and had a floodplain bench to dissipate stormflow energy. Headcut migrations from downstream would have had difficulty passing the culvert to access the upstream reach. Another factor was the existing degree of vertical incision that had occurred prior to the study, and the saprolite that was exposed on the channel bed in most portions of the reach. Saprolite is weathered bedrock, with usually lower erosion potential compared to fine sediment or gravel channel beds (Horton and Zullo, 1991).

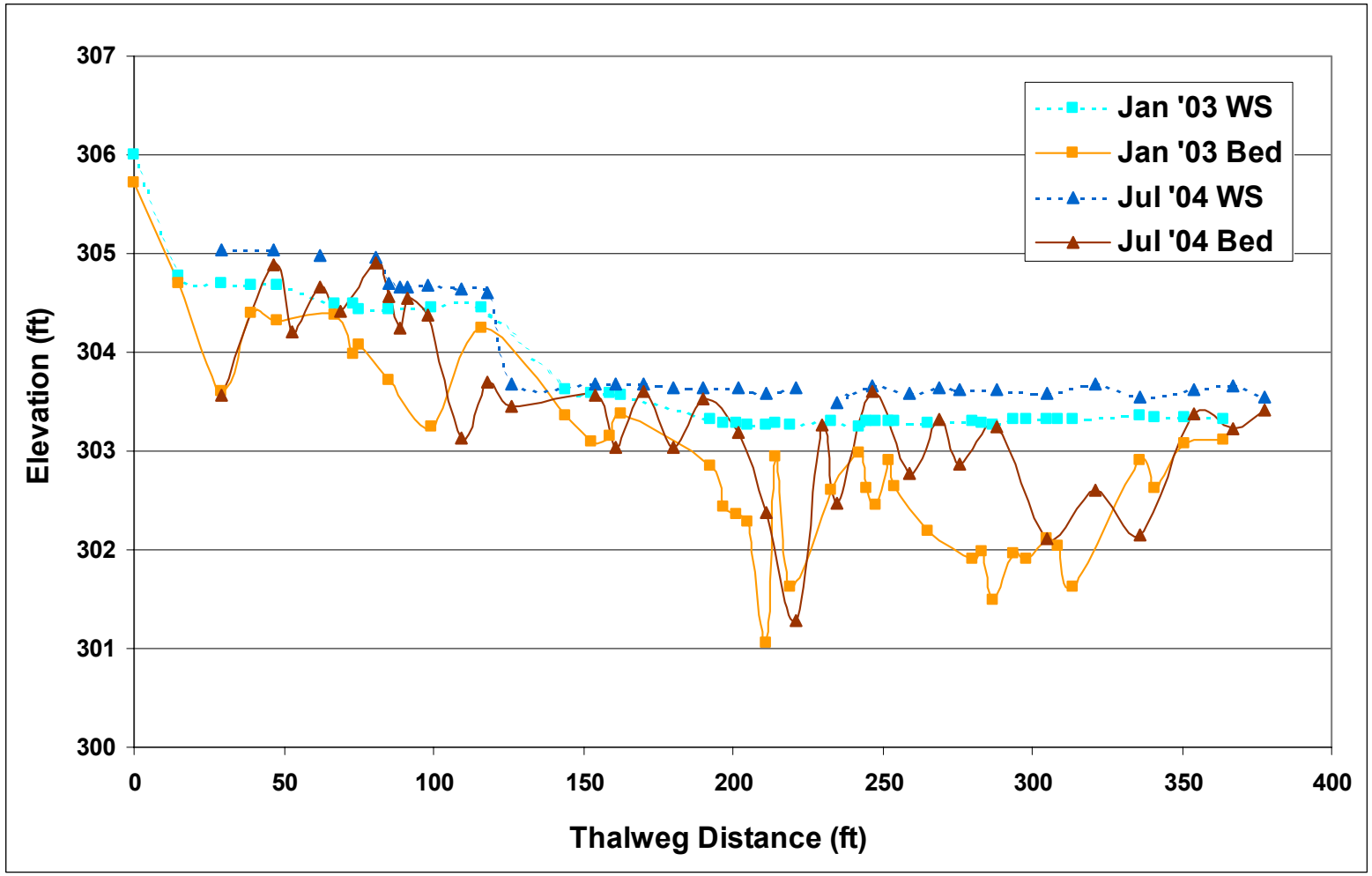

Figure 2.1: Longitudinal Profile Survey Comparison for Reach 1. Water Surface (WS) and Streambed (Bed) Elevations Shown.

Reach 2

The longitudinal profile survey in July 2004 was about 13-ft longer than the survey in January 2003 (Figure 2.2), but channel features did align well between the two surveys. The discrepancy in the profile length was considered minimal compared to the relatively long stretch of channel surveyed (about 1100 ft). The channel bed in the second survey was not consistently higher or lower compared to the first survey. The bed elevation was higher in most of the major pools during the second survey, but head of riffle elevations were not much different in the majority of the reach. Based on the two profiles surveyed over time, aggradation appeared to be occurring within the reach, mainly in the pools of the upper portion of the reach. There were three cross-sections located within the profile that had changes in the streambed elevation relatively consistent with the profiles at these locations. S4 (Station 695 ft) had bed scouring that was identified in both the cross-section surveys (Figure A2.4) and the profiles during comparable time periods. S5 (Station 1150 ft) and S6 (Station 1625) had relatively little change in bed elevation along the thalweg in both the cross-section surveys (Figure A2.5 and Figure A2.6) and the profile surveys.

Similar to Reach 1, the overall channel bed slope and water surface slope were relatively consistent over time. The channel bed slope was slightly higher in the January survey (0.0072) compared to the July survey (0.0067), but these changes were considered very small based on the low gradient existing within Reach 2. The slope of the water surface profiles were similar between surveys (0.0064-0.0066), even though the water surface elevation was higher in the summer survey compared to the winter survey.

The longitudinal profile results in Reach 2 were similar to Reach 1 and indicated localized aggradation within the reach, without evidence of channel bed degradation. The aggraded or shifting sediments deposited on the streambed were probably contributed from the same sources suspected in Reach 1, stormwater outfall channels and streambank erosion. The cross-section and toe-pin surveys in Reach 2 indicated lateral streambank erosion was occurring over time, as discussed in the following streambank erosion section (Figure 2.12-2.20). The lack of overall channel bed degradation that was observed within this reach was probably attributable to several factors, similar to those in Reach 1. The overall grade in this reach was most likely controlled by the large in-stream culvert under Research Drive. The presence of the culvert probably limited the migration of an observed headcut just downstream in Reach 3. Reach 2 also had a relatively high degree of vertical incision, particularly in the upper portions of the reach, which exposed saprolite along the channel bed. The saprolite could have limited scouring and streambed degradation in this reach, due to its resistant properties (Horton and Zullo, 1991).

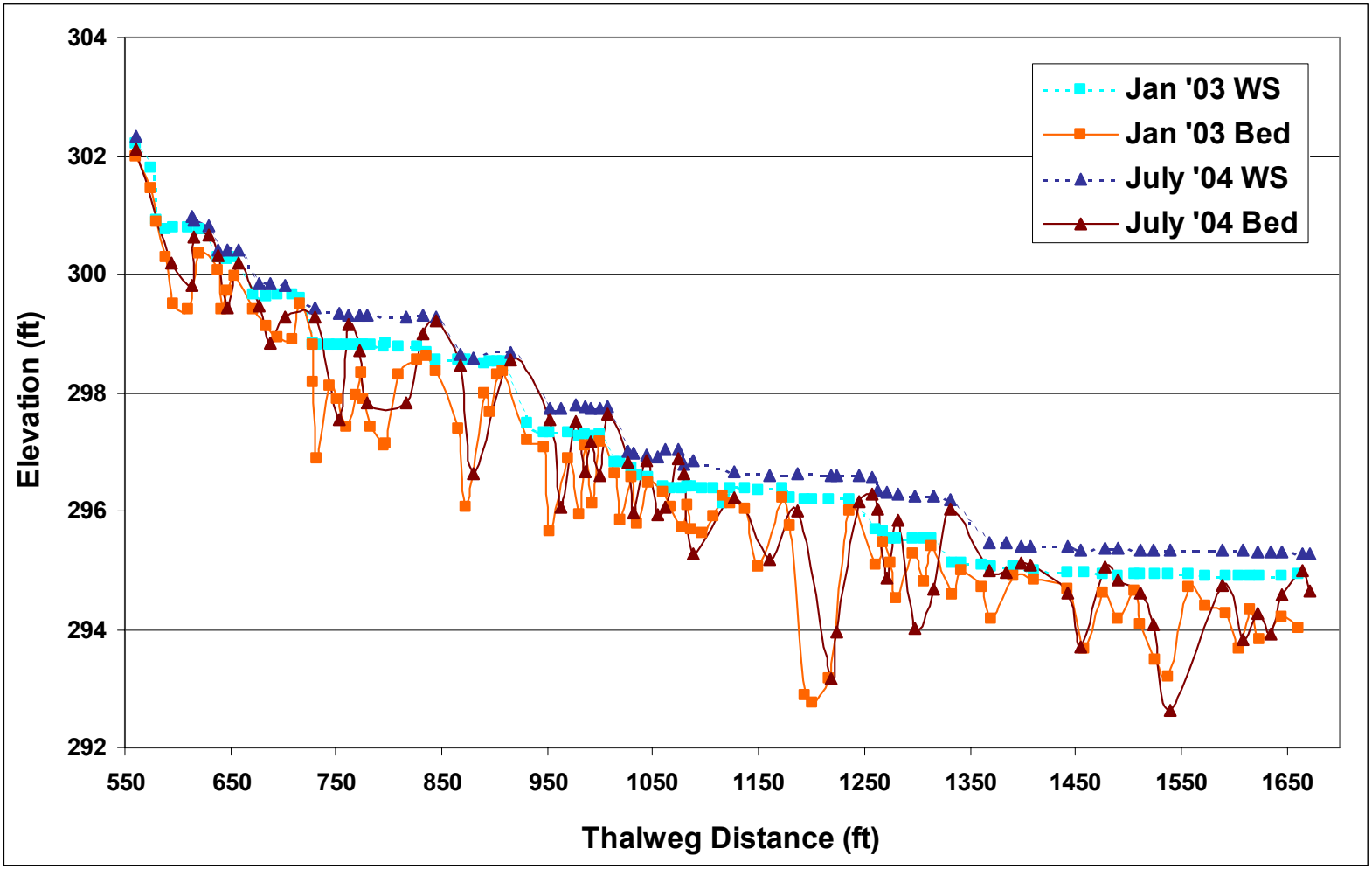

Figure 2.2: Longitudinal Profile Survey Comparison for Reach 2. Water Surface (WS) and Streambed (Bed) Elevations Shown.

Reach 3

The longitudinal profiles in Reach 3 only differed by 1.5-ft in length between surveys and the surveys were well aligned (Figure 2.3). The surveys indicated considerable channel bed adjustments occurring within this reach throughout the entire length. The upper artificially lined portion of this reach (Station 1785 to Station 2020) experienced vertical incision between 1 and 3 ft in depth, with scouring of a large pool (Station 1860) over time where the headcut migration stopped (Figure 2.4). This vertical incision was verified by the cross-section surveys at S7 (station 1880) during the same time period (Appendix 2, Figure A2.7). Based on survey data and observations, the majority of streambed degradation in this section occurred during two large storm events at the end of July and beginning of August 2003.

In the lower section of Reach 3 (Station 2020 to 2210), aggradation was considerable beginning with the large pool at the end of the upper lined reach, which was likely created by a significant headcut migrating upstream. This large pool aggraded about 3.5-ft between the January 2003 and November 2003 survey, with filling continuing over time to the July 2004 survey (Figure 2.4). Downstream of this pool, the channel bed seemed to fluctuate in the November 2003 survey, with net aggradation by July 2004. The change in channel bed slope over time reflected the shift in substrate elevation. Although there was no cross-section within the longitudinal profile of the Reach 3 lower section, S8 was located just downstream of the profile and showed considerable fluctuation in channel bed elevation over time due to localized aggradation.

The overall channel bed slope in the upper reach above the headcut was increased from 0.0250 during the first survey to 0.0282 during the last survey, while the slope in the lower reach was decreased from 0.0099 to 0.0067. These changes in slope reflect the headcut migration in the upper reach, and the aggradation in the lower reach. Comparing the water surface profiles, the elevation was lower in the upper half of the reach but higher in the lower half in the second survey. Changes in water surface elevation coincided with changes in bed elevation. The change in overall water surface slope over time was from 0.0221 to 0.0209.

Extensive streambed degradation was observed in the upper section of Reach 3. The large cobble lining in this section was apparently not adequate to protect the channel once vertical incision was initiated by the intense stormflows and progression of a migrating headcut at the end of the stone-lined section. The aggradation observed in the lower portion of Reach 3 was not expected, however, at the beginning of the study. Sediment was expected to be deposited in the Research Drive culvert, in the large pool at the beginning of the lower section, and finally in SWMP #3 and #4 downstream of the reach. The measured aggradation could have been caused by a combination of factors during the study. The most important factor was the sediment and substrate lost from erosion in the upper reach over a relatively short time interval. This large amount of material probably exceeded the suspended and bedload sediment transport capacity due to large particle size and quantity of material (Dunne and Leopold, 1978). Another factor encouraging deposition in this lower section was the lower channel slope (0.009 ft/ft)

compared to the upstream portion (0.025). The change in slope would have slowed velocity of stormflows entering the lower section, which could have increased deposition.

Another factor that could have contributed more sediment and aggradation in the lower section of Reach 3 was the backwater effect created by the SWMPs downstream. Backwater behind a channel obstruction (SWMP weir) will raise the elevation of the water and may increase turbulence during stormflows where backwater and progressive stormflows meet (Munson *et al*. 1998). The S8 cross-section had high lateral erosion rates and shifting channel sediment bars, providing evidence of this backwater effect. SWMP #3 was observed to be filling in with sediment, which may have extended the backwater effect further upstream. The upstream distance affected by the backwater was not determined in this study, however, but could have stretched upstream into the lower longitudinal profile surveyed in Reach 3.

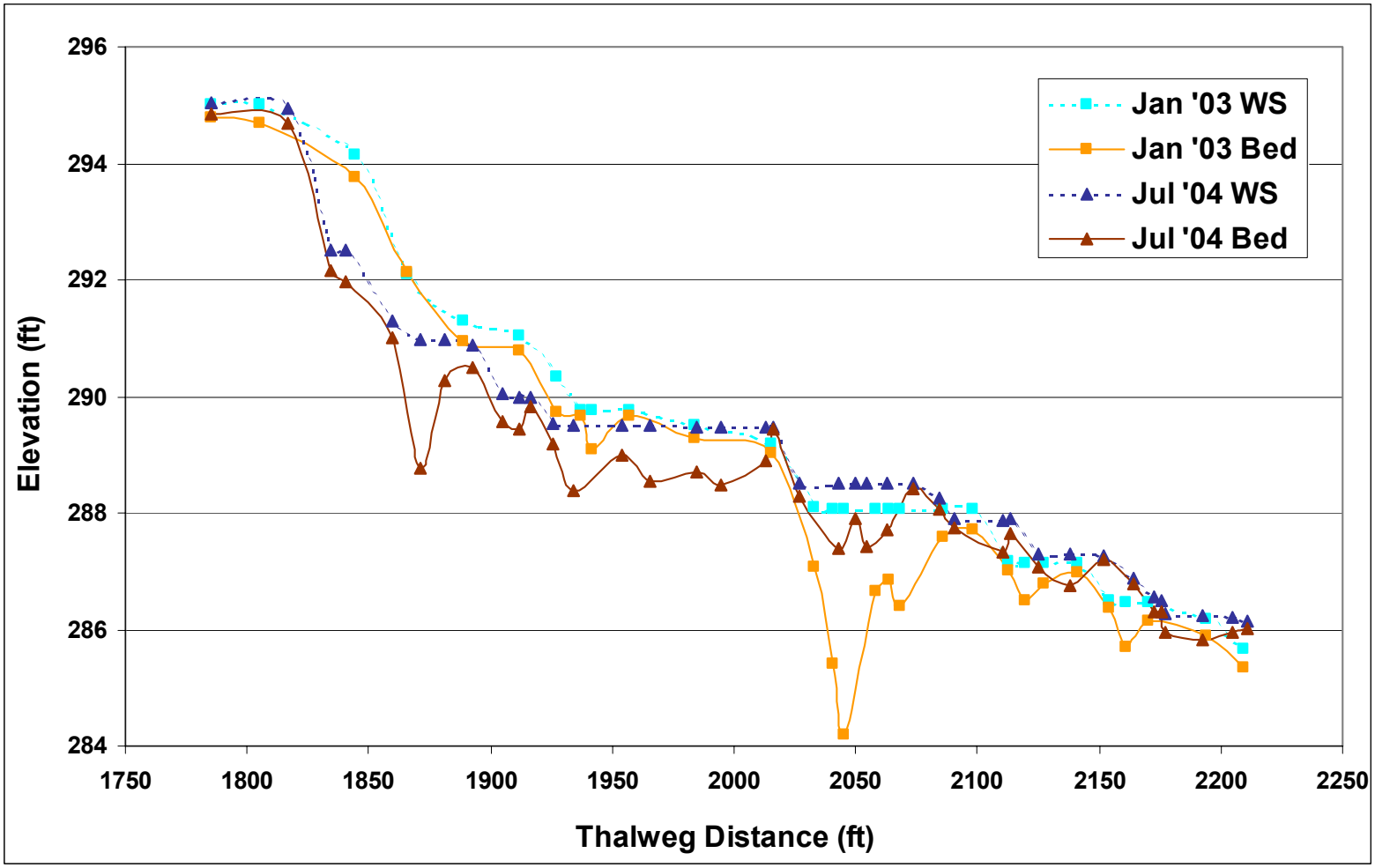

Figure 2.3: Longitudinal Profile Survey Comparison for Reach 3. Water Surface (WS) and Streambed (Bed) Elevations Shown.

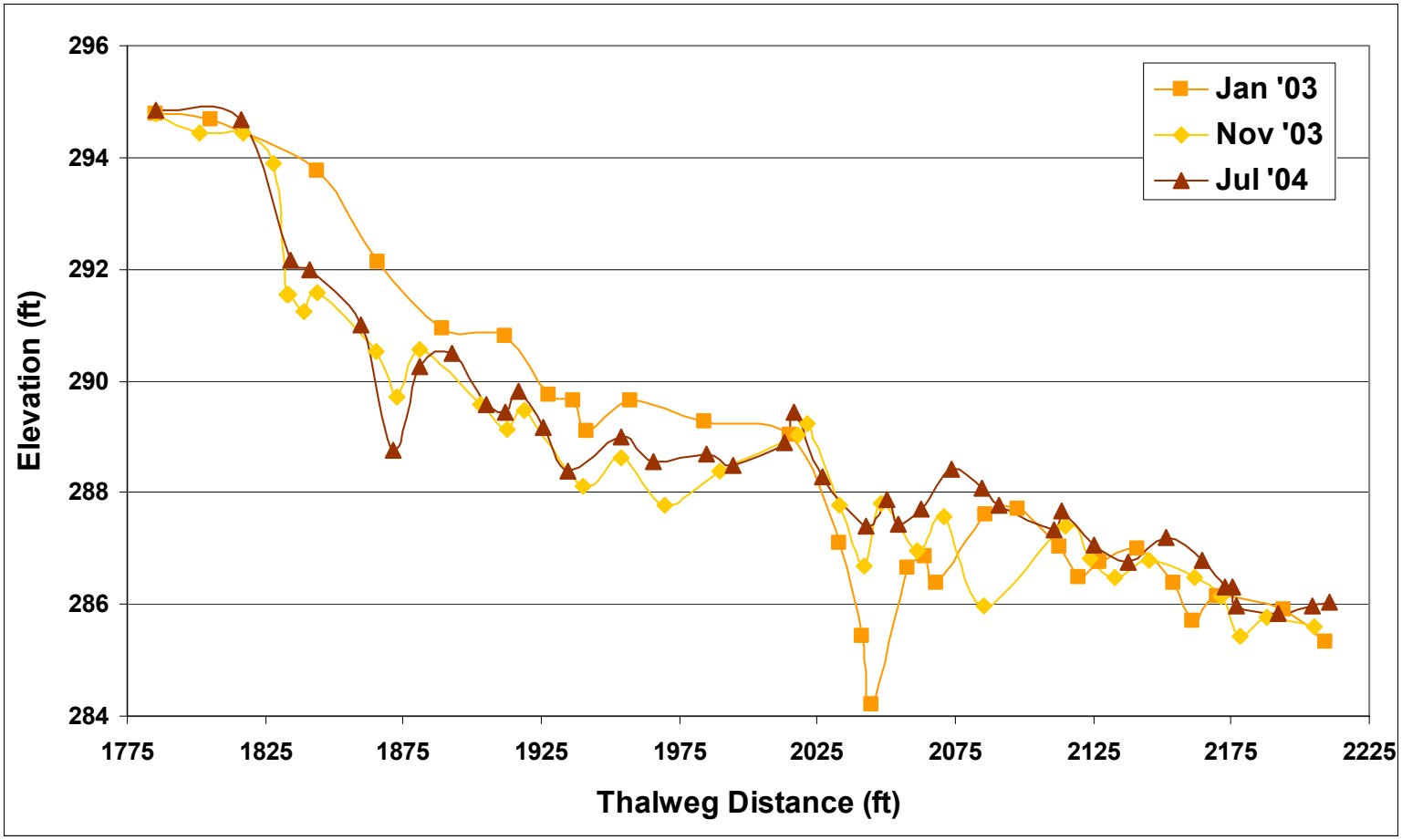

Figure 2.4: Longitudinal Profile Survey of the Channel Bed in Reach 3 Over Time.

Streambed Channel Evolution

Based on channel evolution concepts (Schumm *et al*., 1984; Simon, 1989), streambed degradation patterns were expected to be different between the three reaches in response to watershed development disturbance. Reach 1 was expected to have the smallest amount of degradation due to the existing degree of vertical incision, followed by Reach 2 with slightly less existing incision (Stage IV). Channel degradation was not observed in these upper reaches, however. Reach 3 was expected to have the greatest amount of streambed degradation due to the shorter bank heights and steeper slope (Stage III). Streambed erosion occurred but only in the upper portion of the reach. The lower portion experienced aggradation and widening, characteristic of a premature Stage V without the occurrence of Stage IV.

The typical streambed patterns of channel evolution were not observed in this study due to several site-specific stream characteristics. One reason for the deviation from the typical evolution pattern may have been the influence of watershed stormwater retention structures associated with the development sites upstream. Stormwater retention could have limited the effects of "disturbance" in this watershed, by mitigating the increase in stormwater runoff. The effects of the increased stormwater volume added over a longer time period may not be indicated in the streambed profile due to reduced shear stress, but may be indicated with the streambank profiles due to increased exposure to stormflows (Bledsoe, 2002). In the lower portion of Reach 3, the in-stream ponds were suspected of altering the evolution pattern through backwater effects, which could have caused lateral erosion before vertical incision occurred. The weir elevation could also have controlled

the grade of this lower Reach 3 section, similar to the effects of the large in-stream culverts in Reach 1 and Reach 2.

Streambank Erosion

Streambank erosion was measured in North Creek between August 2002 and August 2004 using cross-section and toe-pin surveys. Annual cross-section surveys are included in the text to summarize changes over time, while complete survey data are provided in Appendix 2, Figure A2.1-A2.8. The second "annual" survey was performed on 9/24/03, later than one year after the initial survey. Data collection at the beginning of August 2003 was attempted, but a large storm during the survey removed sampling equipment at the cross-sections that had to be replaced before the survey could be resumed. The lower cross-sections were surveyed during the first week of August 2003, and these data are provided in the Appendix figures. Cross-section areas between established top of bank elevations were determined from each survey for comparison over time. Toe-pin surveys are also included below, with the exception of two locations (TP 4 and TP 12) where the toe-pins were lost during storm events.

For each reach, average annual streambank erosion rates were determined using the survey data. Factors thought to most influence streambank erosion in each North Creek reach during the study included incised channel conditions upstream and the predicted increase in stormflow intensity and volume derived from continued development in the watershed.

Reach 1

Three cross-sections and four toe-pins were established in Reach 1 (See Chapter 1, Figure 1.2), where streambank heights were characteristically steep; between 6 and 14 ft on average. Cross-section 1 (S1) was located just after the Avent Ferry Rd. culvert near the end of a deep plunge pool. The left streambank was covered by large cobbles (12-in dia), and remained relatively stable over time (Figure 2.5). The right bank was bare of vegetation and nearly vertical; held in place by a fallen tree and tree roots from the top of bank. This bank eroded over time near the toe with the initiation of an undercut. Streambank changes were small relative to the large cross-section area, as indicated by little change in cross-sectional area (See Figure 2.27, presented at the end of the Streambank Erosion Section).

Cross-section 2 (S2) was located mid-reach just after SOC2 entered North Creek. The cross-section was located at a pool, just before a large lateral bar at the base of the right bank. The streambanks were near vertical and devoid of vegetation, except for weedy vines hanging from above. The left streambank was relatively stable over time, but the right streambank eroded along its entire vertical extent (Figure 2.6). During the first year an undercut formed near the toe, and the upper portion appeared to have slumped during the latter half of the second year (Appendix A2, Figure A2.2). The cross-section area fluctuated during the study, but indicated an overall trend of increasing area due to erosion (Figure 2.27). Some of the fluctuation in both S1 and S2 cross-section areas was due in part to the large size of the cross-sections and the difficulty associated with measuring steep vertical banks accurately.

Cross-section 3 (S3) was located just upstream of the Varsity Drive culvert, and had evidence of alteration during culvert installation with shortened streambanks (Figure 2.7). This cross-section was selected due to the construction at the top of the terrace during the study (North Carolina Wildlife Resources Headquarters), and sewer line excavation by the City of Raleigh in August 2002. Both streambanks had large stone (6-12 in dia) at the bases, similar to the large stone at the culvert entrance. Although the right streambank was relatively stable over time, the left streambank fluctuated between surveys. Variation along this bank resulted from sediment deposition from a small ephemeral channel that entered near the cross-section. The channel drained the construction site diversion ditch located upslope and deposited sediment at the entrance to North Creek. The cross-section area remained relatively constant during the first year, but decreased during the second year due to sediment deposition (Figure 2.27).

Toe-pins along Reach 1 were placed between S2 and S3 (See Figure 1.2), and results were variable between surveys. The right streambank measured by TP 1 (Figure 2.8) and the left streambank measured by TP 2 (Figure 2.9) had streambank material additions over time mid-profile without supporting aggraded material below, suggesting errors within the surveys. The upper profiles had evidence of erosion over time, but this could not be confirmed due to differences in measurement heights between surveys. The bank toes composed of saprolite were relatively stable, with some sediment aggradation indicated in TP 1 during the 12/03 survey. The most dramatic change in streambank profile within this reach was observed at TP 3 (Figure 2.10). Lateral erosion of the right bank occurred along the entire profile, with a loss of over 4 ft during the study. The right

streambank surveys associated with TP 5 showed variable results, similar to the first two toe-pins (Figure 2.11). An undercut developed after the first six months, but disappeared in the latter surveys. There may have been some erosion mid-profile**,** but variability in the surveys made the material loss uncertain. The survey of the upper profile portion was consistent over time, except for an expected survey error (6/03).

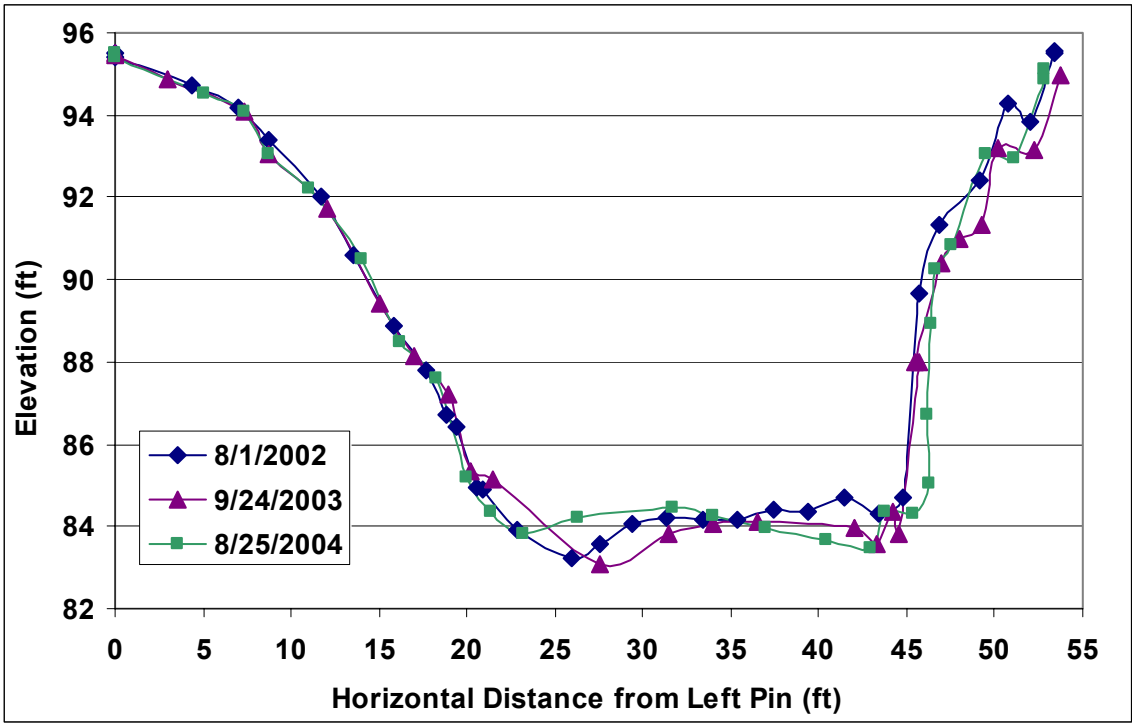

Figure 2.5: Annual Surveys for Cross-Section 1 (S1).

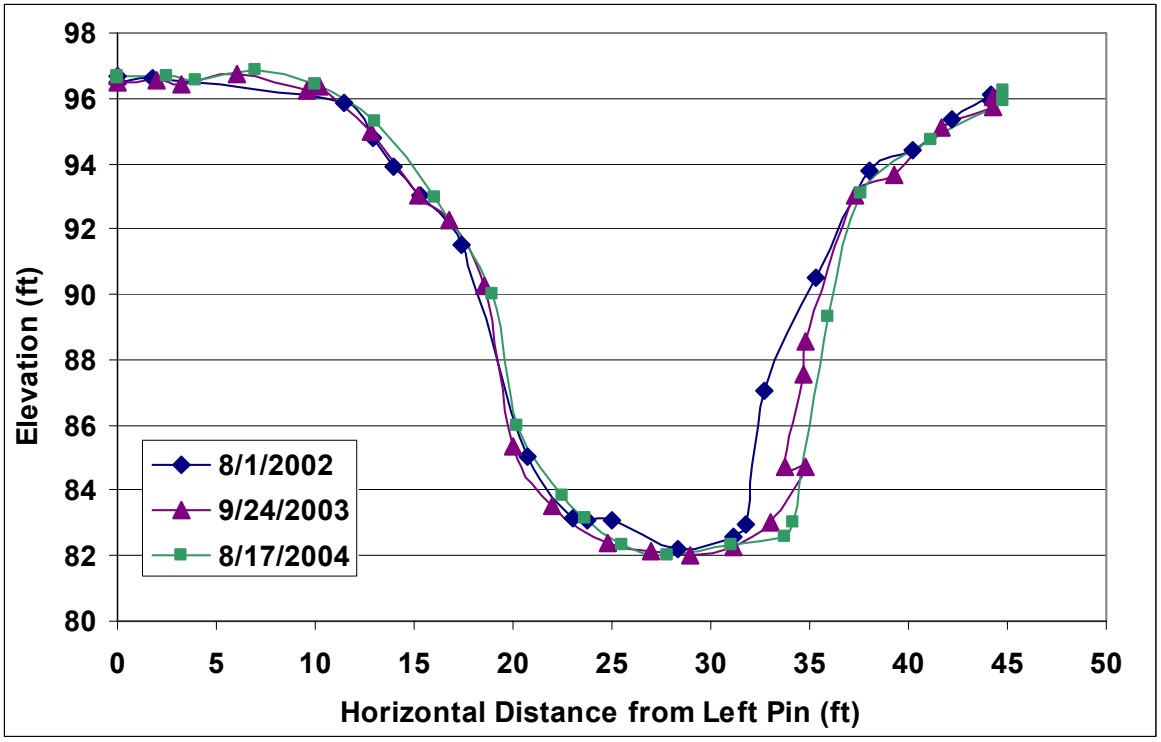

Figure 2.6: Annual Surveys for Cross-Section 2 (S2).

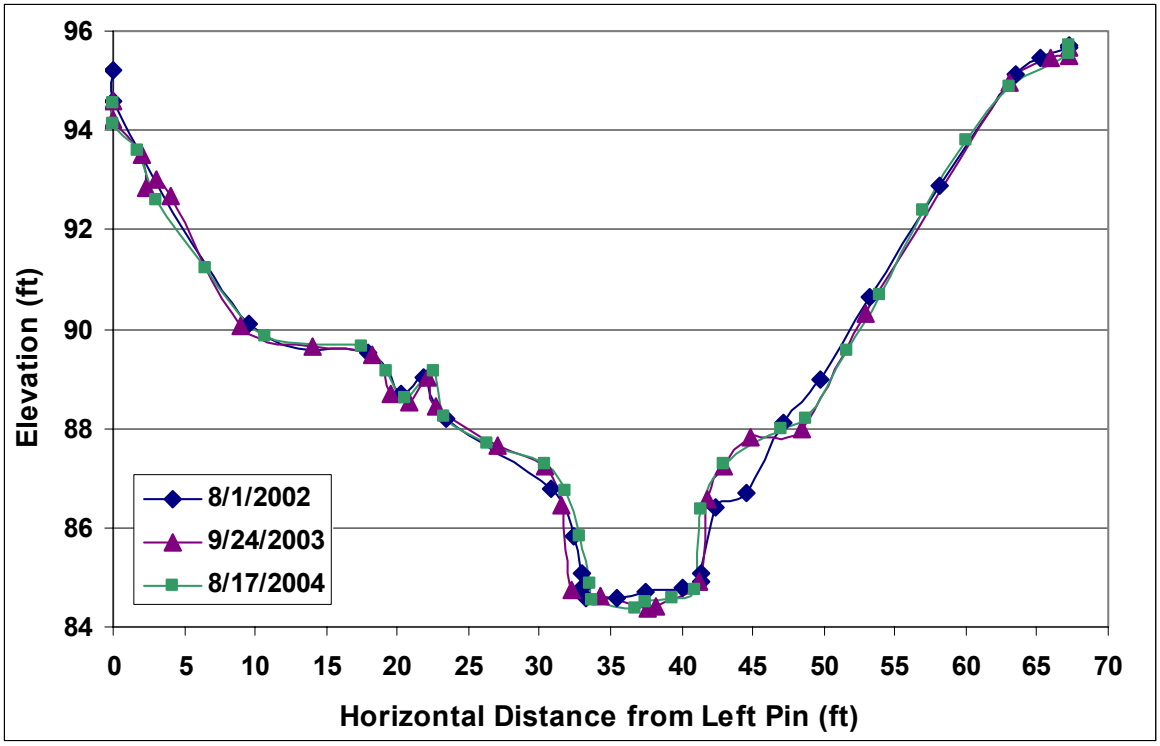

Figure 2.7: Annual Surveys for Cross-Section 3 (S3).

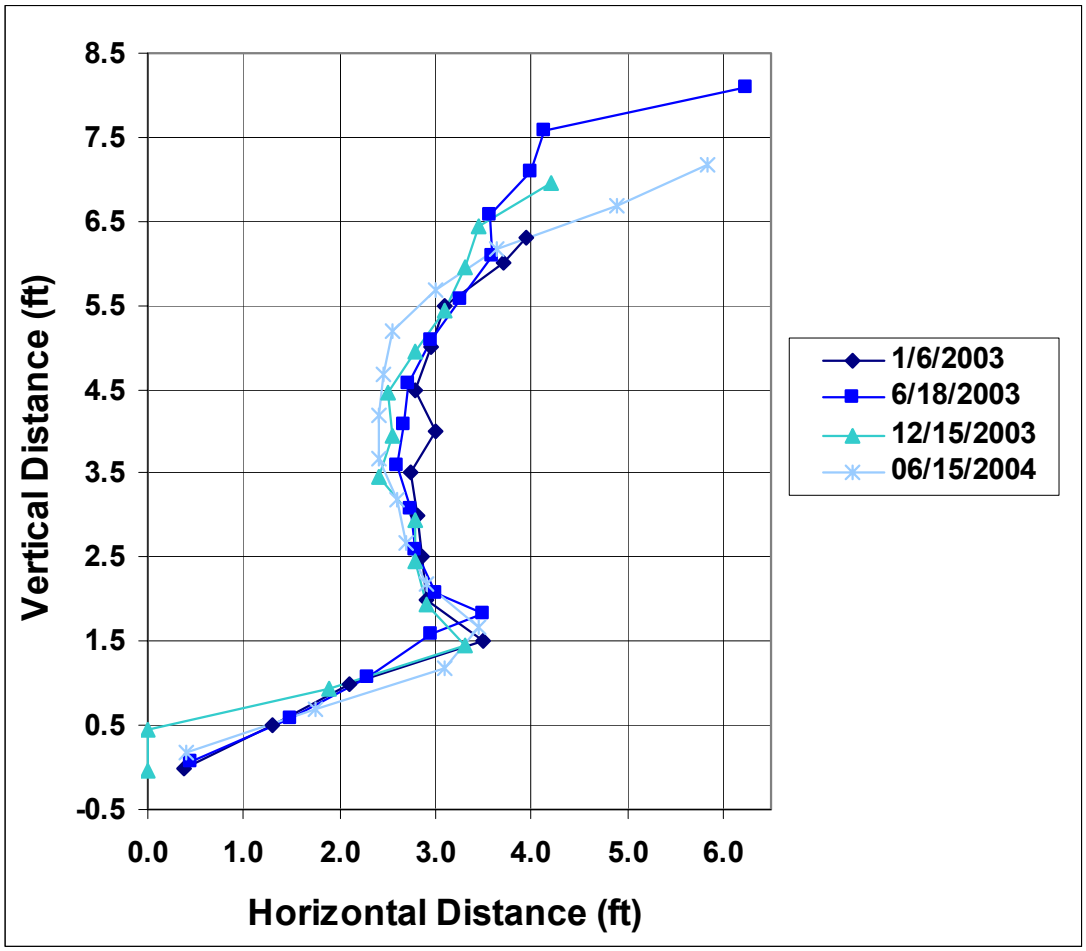

Figure 2.8: Right Streambank Profile at Toe Pin 1, in Reach 1.

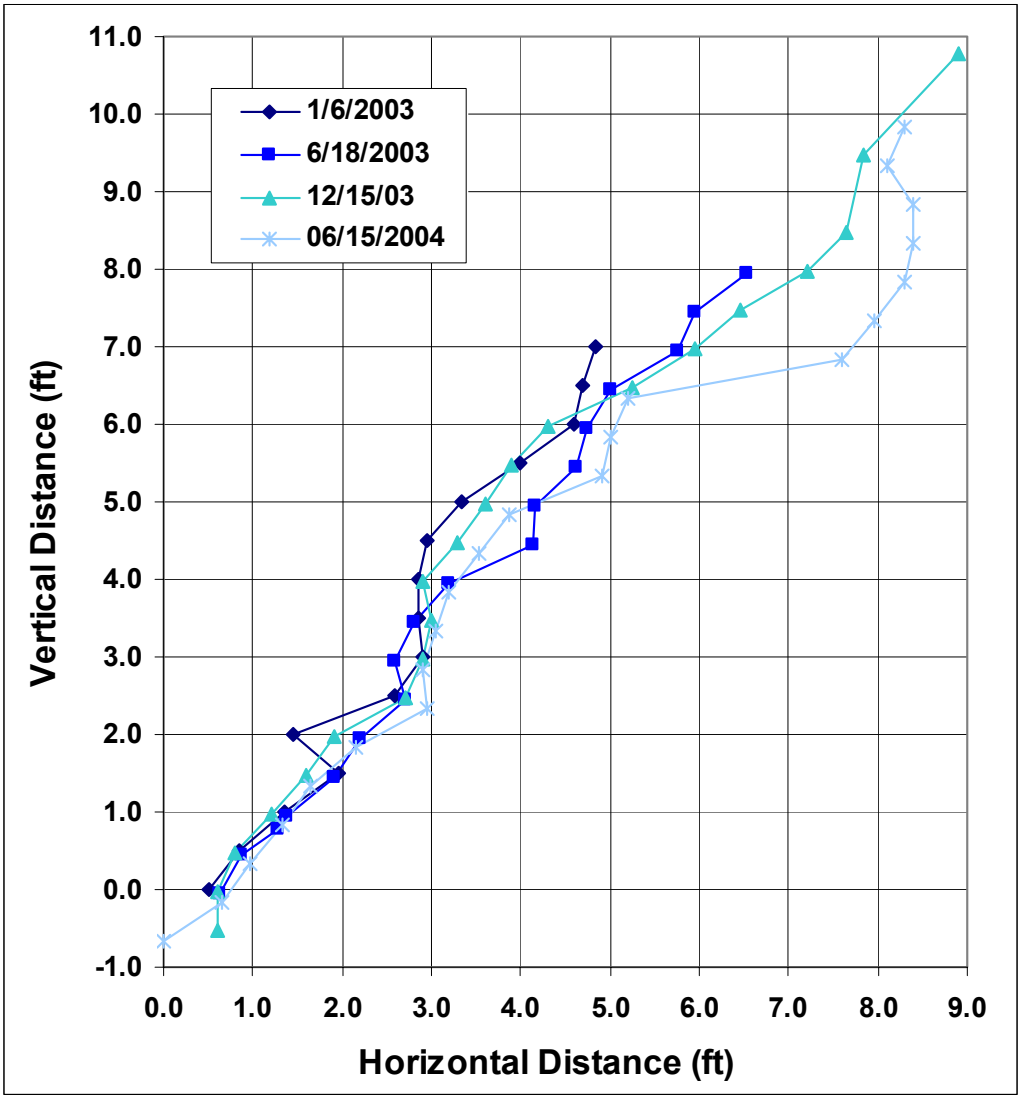

Figure 2.9: Left Streambank Profile at Toe Pin 2, in Reach 1.

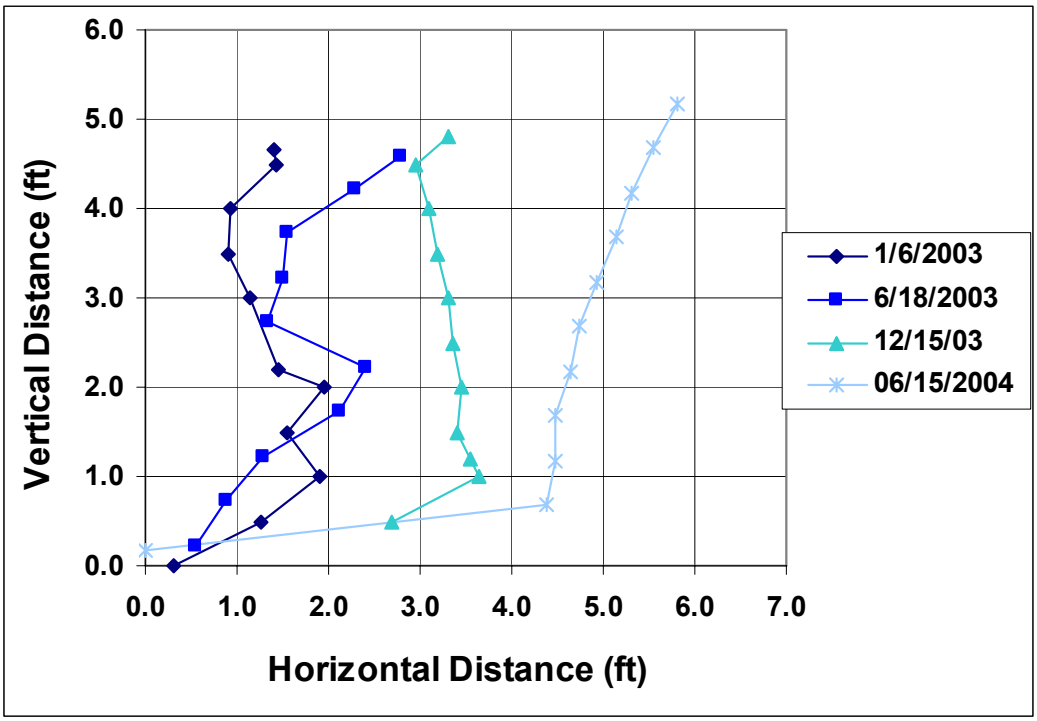

Figure 2.10: Right Streambank Profile at Toe Pin 3, in Reach 1.

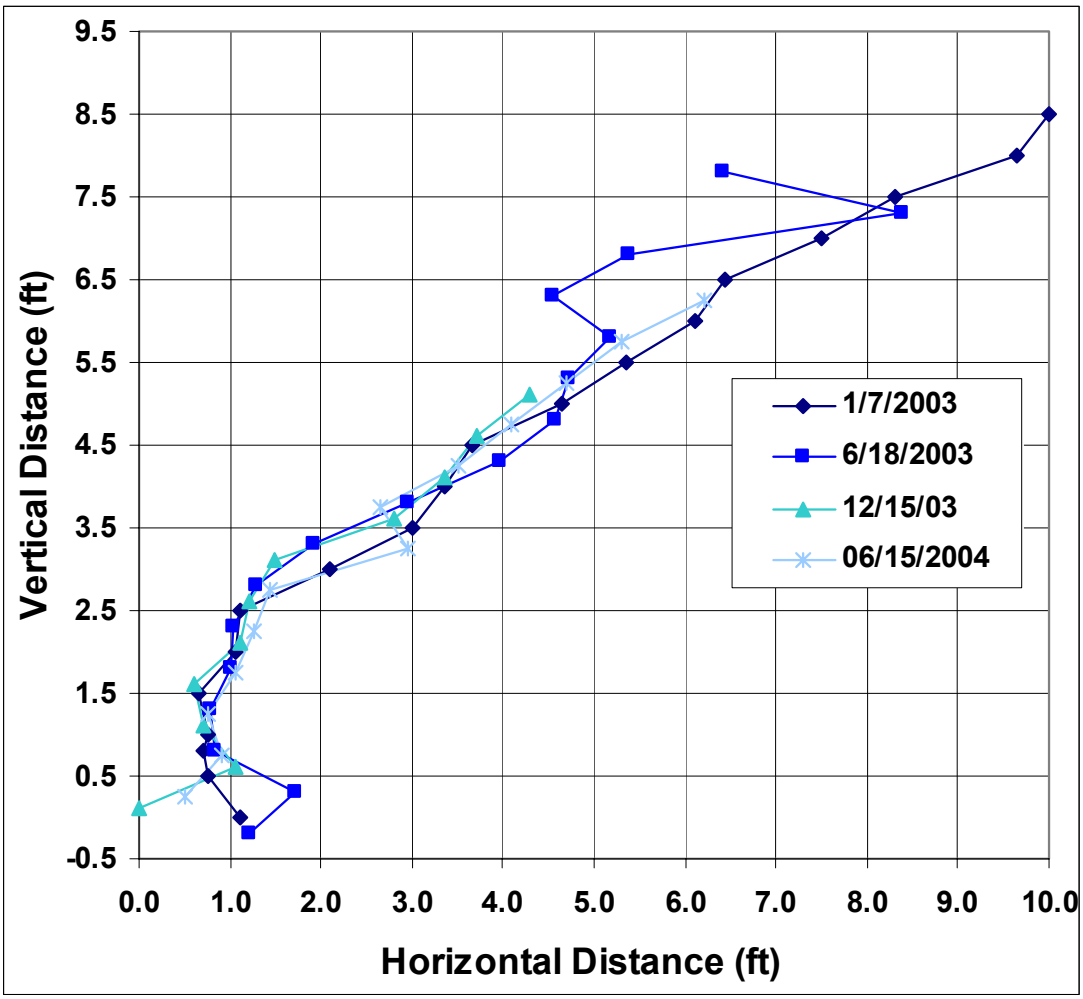

Figure 2.11: Right Streambank Profile at Toe Pin 5, in Reach 1.

Reach 2

Three cross-sections and six toe-pins were located in Reach 2 (See Chapter 1, Figure 1.3), which had moderately steep streambanks 3-8 ft in height. Cross-section 4 (S4) was located in the first meander bend downstream of the Varsity Drive culvert, with a pool undercutting the right streambank and a lateral bar along the left streambank. Thick tree roots stretched across the undercut region on the right bank (Figure 2.12), and made

quantification of its dimensions difficult. Surveys either omitted the undercut (5/03), or yielded variable estimates of its depth and dimension (Appendix A2, Figure A2.4), limiting the evaluation of erosion. The left bank lateral bar eroded over time, with a portion of the increase in cross-sectional area reflecting this erosion (Figure 2.27). Just upstream of this bend, a stormwater outfall channel entered North Creek. The channel progressively eroded during the study, and contributed many felled trees into the stream during the fall of 2003. Stormflows were diverted by the obstruction toward the left streambank, probably causing the erosion of the lateral bar. The channel bed was degraded on the left but sediment filled in the pool on the right, creating a more homogeneous bed profile in what was a typical meander bend cross-sectional pattern.

Cross-section 5 (S5) was located near the end of a long run in the middle of the reach. A large tree with exposed roots just downstream created and maintained a localized undercut area on the left bank with a pool. The right bank was flanked by saprolite in the lower two feet. Erosion occurred during the first year along the left bank and along the bed near the right bank, even with saprolite present. The cross-section remained relatively constant during the second year (Figure 2.13), and the cross-section area remained relatively stable over time, with an overall gain of only a few square feet (Figure 2.27).

Cross-section 6 (S6) was located just before the Research Drive culvert. The permanent left pin was lost just after installation during sewer line construction in August 2002. The left streambank was demolished during this time and rebuilt with a large stone (6 in dia) apron. The first survey for comparison was performed in September 2002, after sediment material filled the once large pool during bank replacement. On the left

streambank, a ditch formed just above the rock during the first year. The bank toe eroded over time for an increase in bank height (Figure 2.14). A mid-channel bar just in front of the right streambank was removed during the first year, and sediment filled the pool at the toe of the bank. The right bank maintained lateral stability over time, with little change in the top of bank location. The cross-sectional area of S6 fluctuated over time, presumably due to patterns of left bank degradation and aggradation at the right bank toe (Figure 2.27).

The toe-pin surveys were spread throughout the reach (See Figure 1.3), and were variable between surveys. The right streambank at TP 8 (Figure 2.17) and TP 11 (Figure 2.20), and the left streambank at TP10 (Figure 2.19), experienced notable erosion along most of the profile measurements. Lateral adjustments were most evident in the December 2003 surveys. The remainder of the toe-pins surveys were inconsistent, making them difficult to interpret. The right streambank associated with TP 6 appeared to have some erosion over time mid-profile, but the bank toe surveys were variable over time (Figure 2.15). Variation at the bank toe could have been a result of survey error or due to variability in aggradation between storms. Dense weedy vegetation was present mid-profile and could have affected scour and deposition patterns. The left streambank associated with TP 7 was not consistently surveyed during the study, but appeared relatively stable with portions of saprolite near the bottom (Figure 2.16). The TP 9 streambank profile was highly variable between surveys, that was suspected to be more a result of survey error than actual bank migration patterns (Figure 2.18).

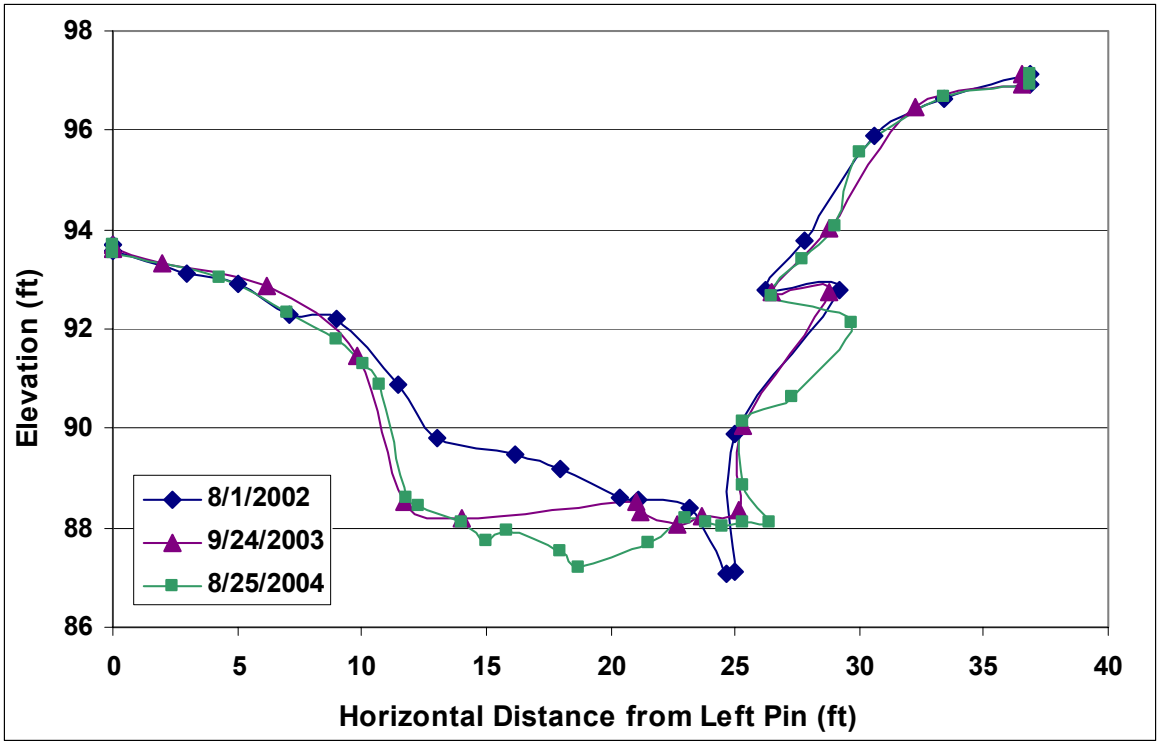

Figure 2.12: Annual Surveys for Cross-Section 4 (S4).

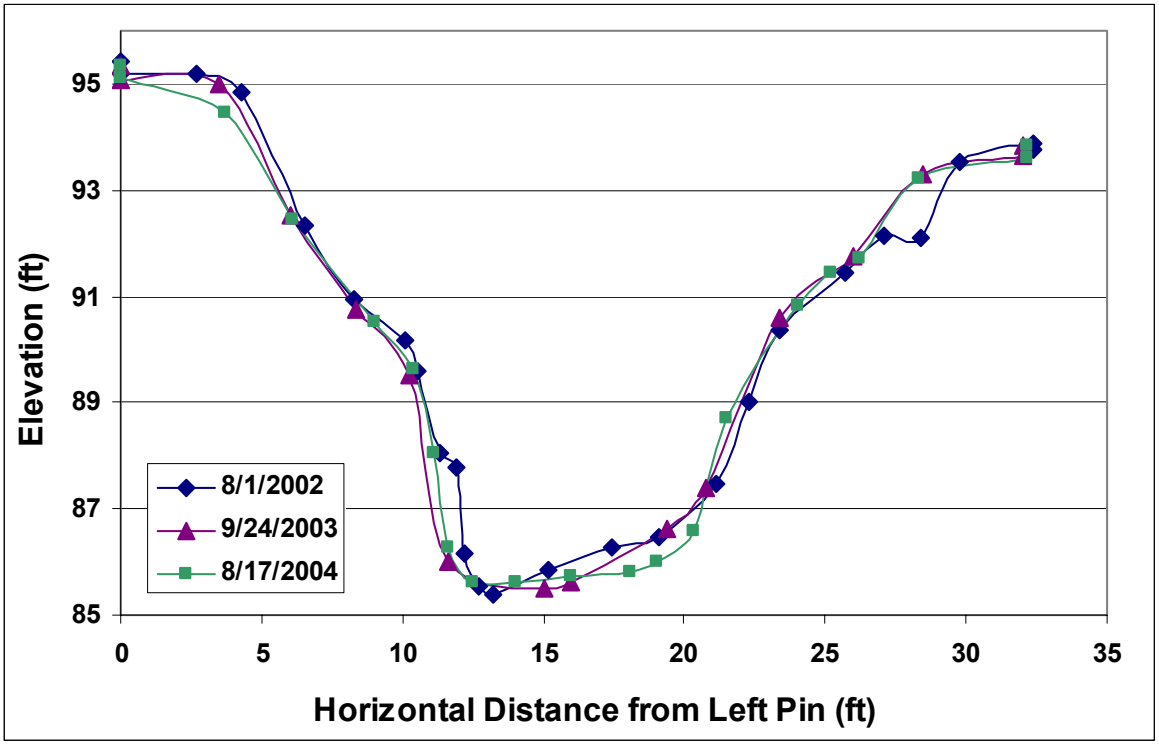

Figure 2.13: Annual Surveys for Cross-Section 5 (S5).

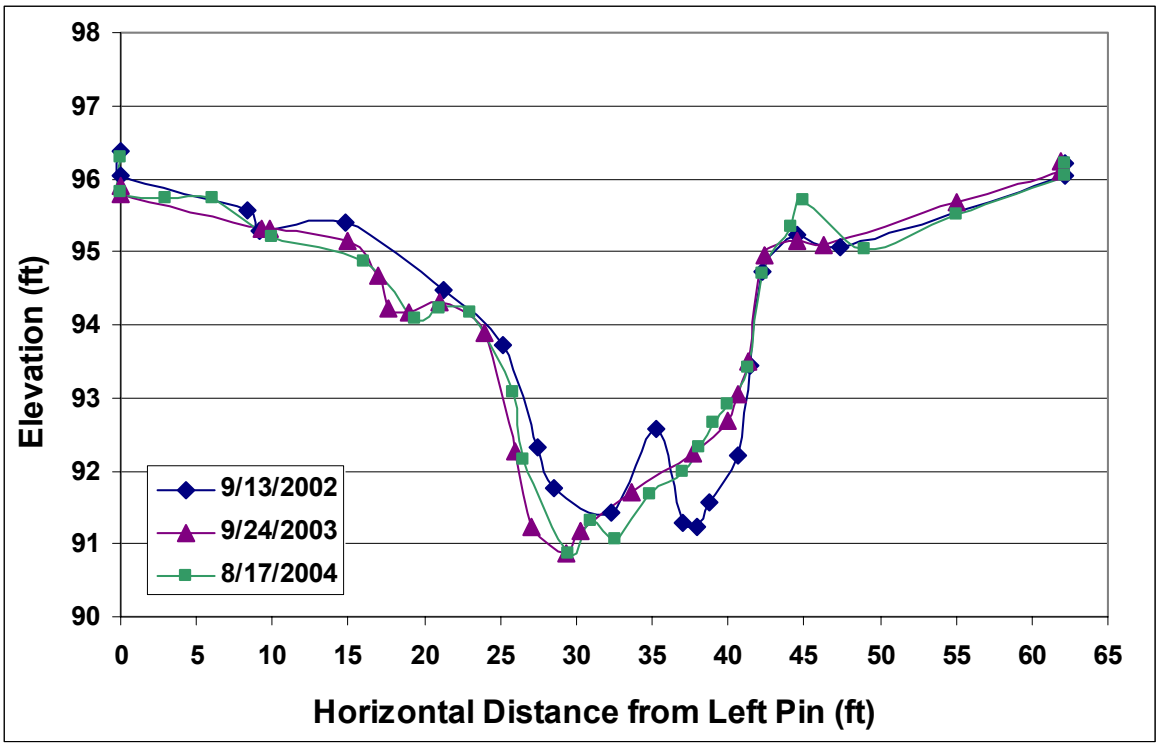

Figure 2.14: Annual Surveys for Cross-Section 6 (S6).

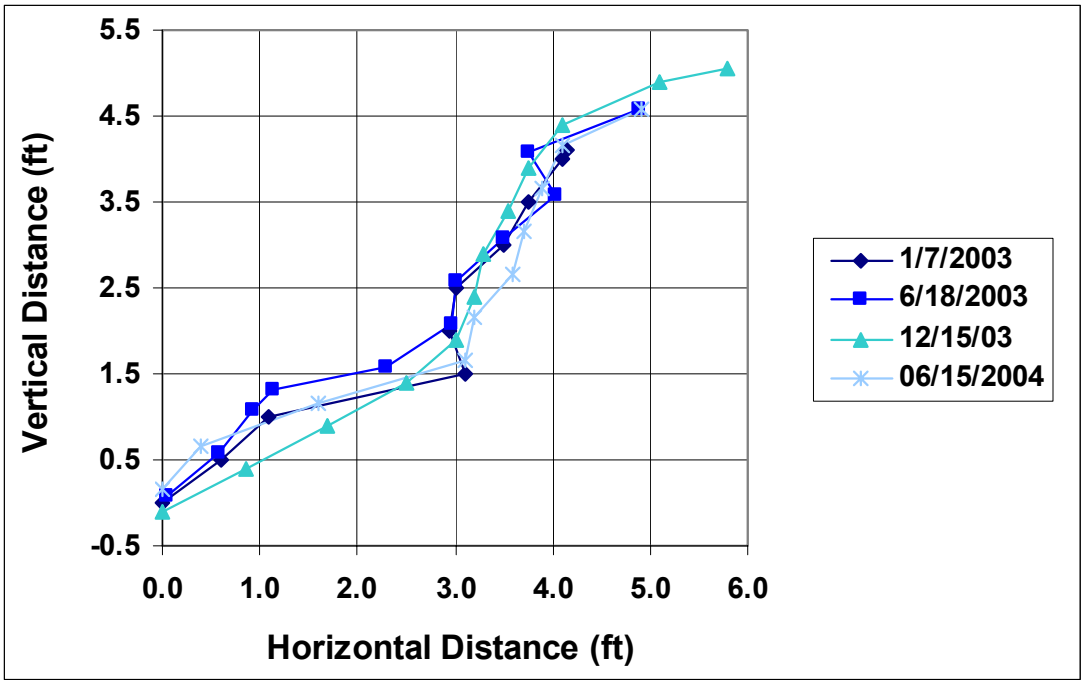

Figure 2.15: Right Streambank Profile at Toe Pin 6, in Reach 2.

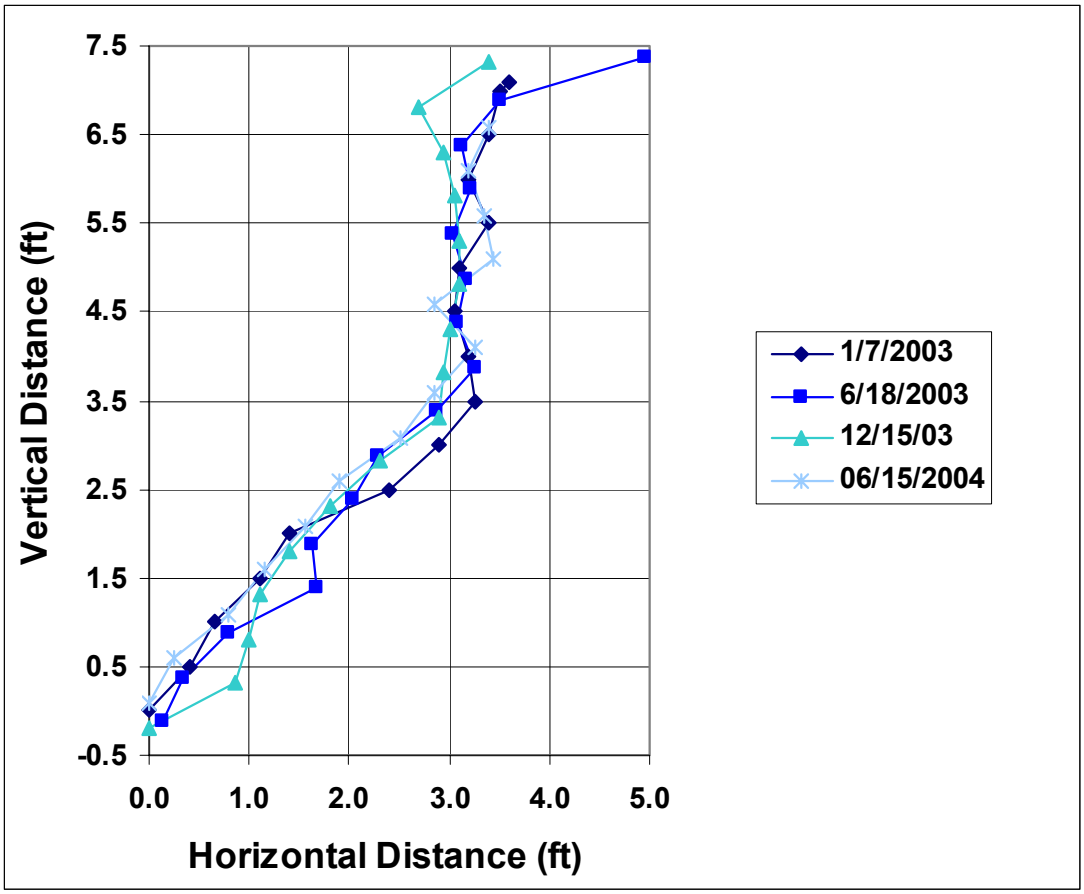

Figure 2.16: Left Streambank Profile at Toe Pin 7, in Reach 2.

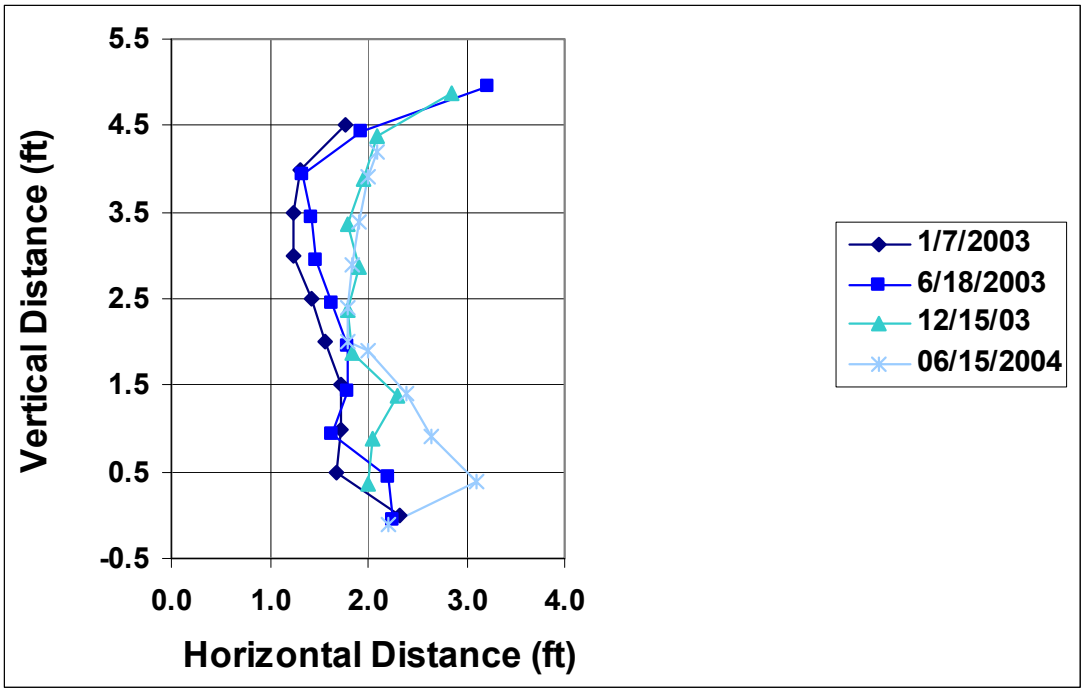

Figure 2.17: Left Streambank Profile at Toe Pin 8, in Reach 2.

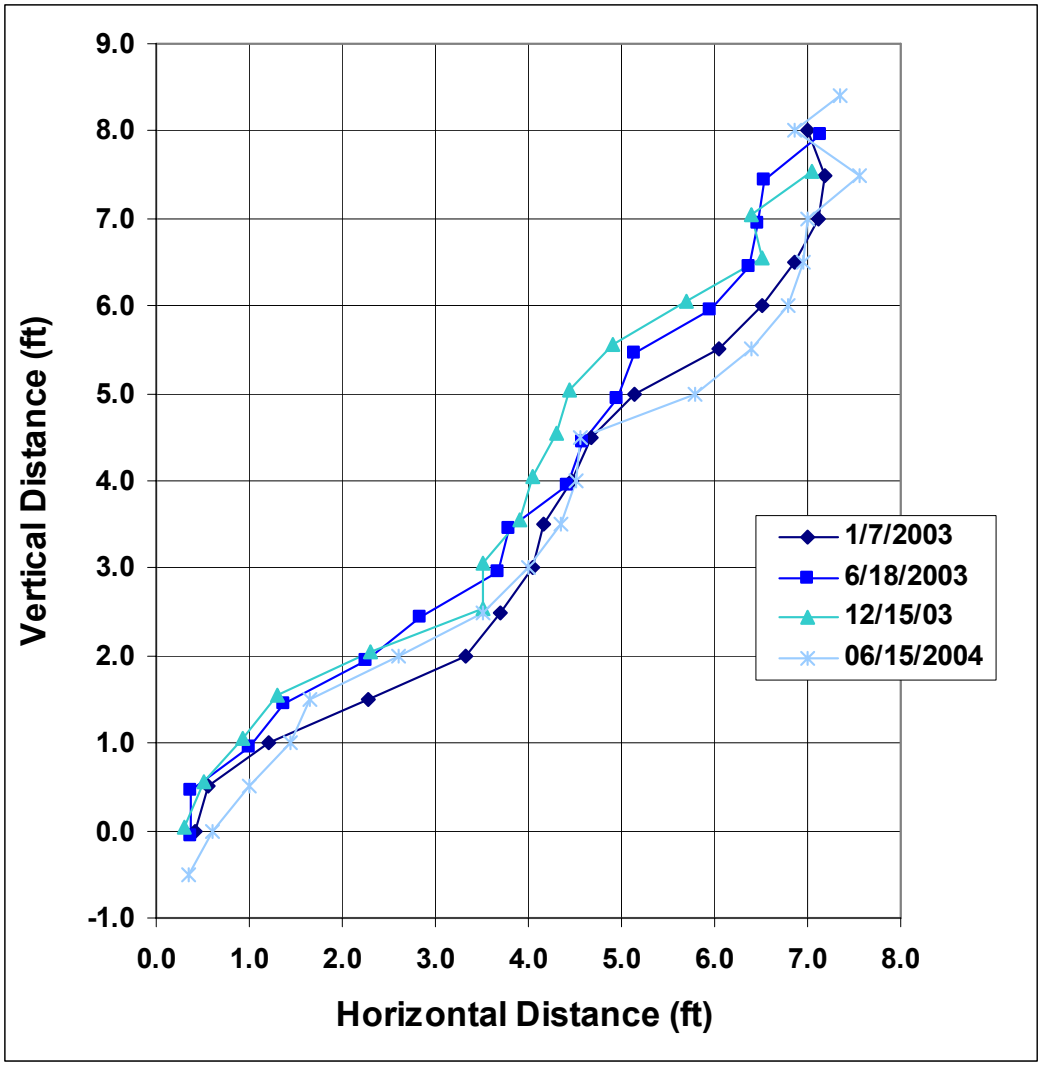

Figure 2.18: Left Streambank Profile at Toe Pin 9, in Reach 2.

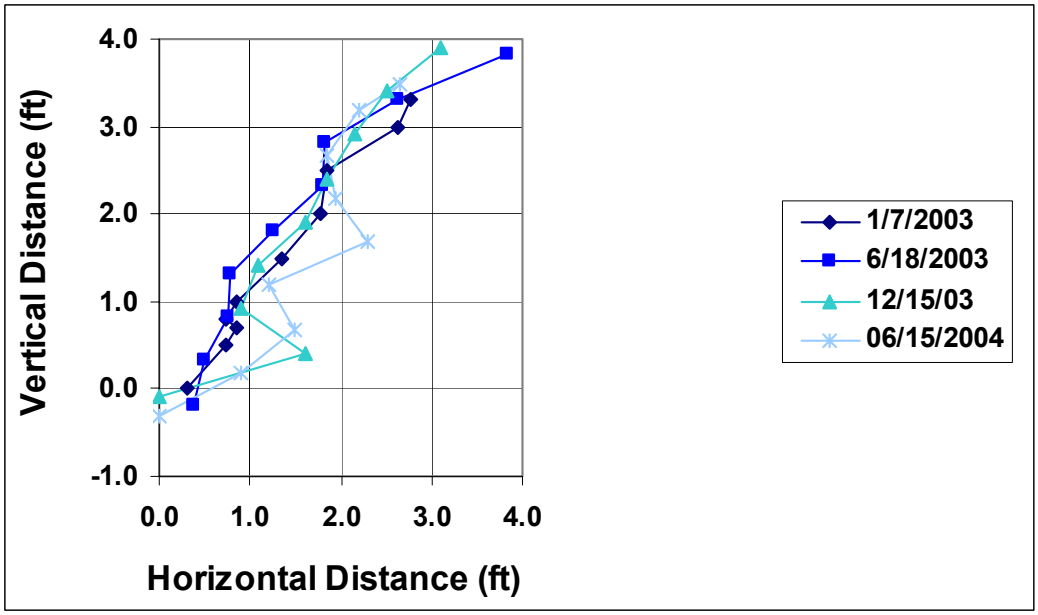

Figure 2.19: Left Streambank Profile at Toe Pin 10, in Reach 2.

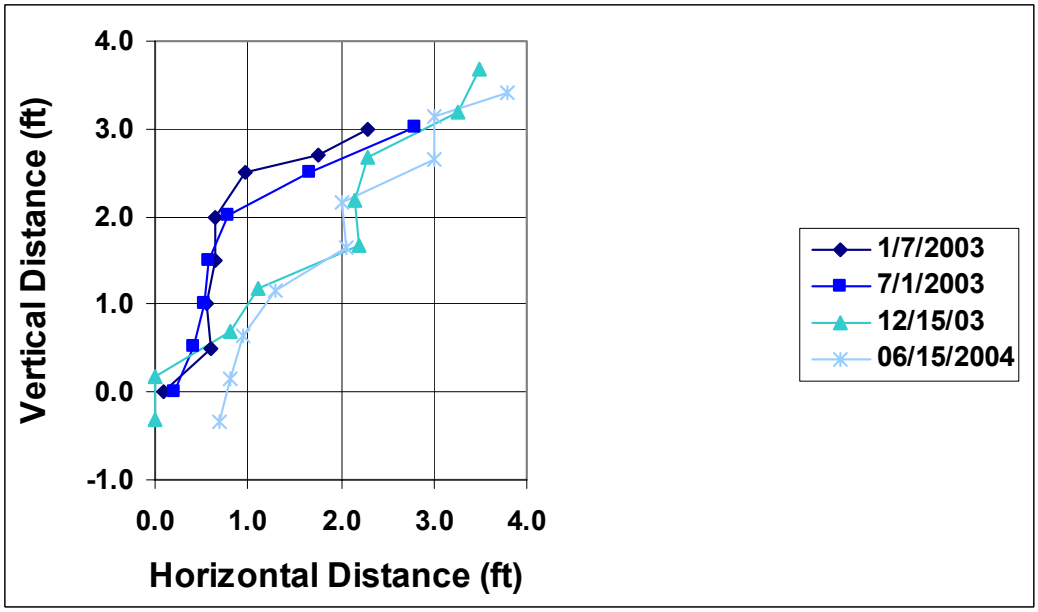

Figure 2.20: Right Streambank Profile at Toe Pin 11, in Reach 2.

Reach 3

There were two cross-sections and four toe-pins located within Reach 3 which streambanks were only moderately incised at the beginning of the study (Chapter 1, Figure 1.4). Cross-section 7 (S7) was located midway along the stone-lined portion of the reach. This cross-section eroded on the left bank associated with vertical incision during the first year, but remained stable over the next year (Figure 2.21). The survey performed in 8/03 verified that incision occurred before this sampling date (Appendix 2, Figure A2.7). The right bank also eroded laterally along the entire length, but only during the second year. The increasing cross-sectional area measurements over time indicated the considerable lateral streambank and vertical streambed erosion (Figure 2.27). Exposed cobbles within the streambanks were observed during the second year on both sides as bank material eroded, suggesting the streambanks were altered prior to the study with artificial stone additions.

Cross-section 8 (S8) was located just after the pedestrian bridge before SWMP #3. This cross-section experienced considerable lateral erosion on the left bank during the first year of the study, losing about 8 feet (Figure 2.22). The majority of lateral migration occurred during the first three months of the study (Appendix 2, Figure A2.8). The vertical height of the left bank remained relatively constant during migration, however, with some aggradation over time from slumped bank materials. The slumped bank materials were observed to have been deposited at the bank toe in blocks and slabs, but carried away during the next several storms. Construction debris was exposed in both the bank and shifting lateral bar, suggesting the floodplain soil profile had been altered prior to

the study. Compared to the left bank, the right streambank gained lateral footage with an associated shifting lateral bar that developed along the bank (Figure A2.8). The bar replaced a pool in front of the right bank during the first three months of the study. The cross-section area did not change much over time, due to the erosion on the left bank offset by the aggradation along the right bank (Figure 2.27).

The toe-pins in Reach 3 were installed later than in the upper two reaches, specifically to measure changes in channel morphology after the vertical incision event in August 2003. The toe-pin surveys in Reach 3 all showed erosion between August 2003 and December 2003, but results were variable during the next six months. TP 13 was located near the Research Drive culvert and experienced excessive lateral erosion of the right streambank due to a felled tree and collected debris at this location (Figure 2.23). Further lateral erosion did not occur between December 2003 and June 2004, as a result of flow deflection by the massive rootwad in front of the newly exposed bank face. The right streambank associated with TP 14 was located along an extensive run just before the end of the stone-lined section and experienced relatively uniform erosion along the profile during the first survey interval (Figure 2.24). TP 15 was located near the right streambank just after the large pool at the end of the stone-line channel. The bank profile eroded over time in the mid-profile region, with addition of material near the toe suspected to be the slumped material from above (Figure 2.25). TP 16 was located just before the pedestrian bridge on the left bank downstream to where SOC7 entered. The left streambank profile eroded the most at the bank toe, probably due to the SOC influence (Figure 2.26).

The differences between the December and June surveys appeared to be consistent between toe-pin locations, with the June 2004 surveys all slightly closer to the stream. This suggested that a survey error may have occurred, but it is not clear whether the December or June surveys were potentially affected. During the December survey of the last three toe-pins, small to large cobbles were noted within the streambank profiles of Reach 3, exposed by erosion. The cobble exposure suggested streambanks along the entire reach were amended at some point before the study.

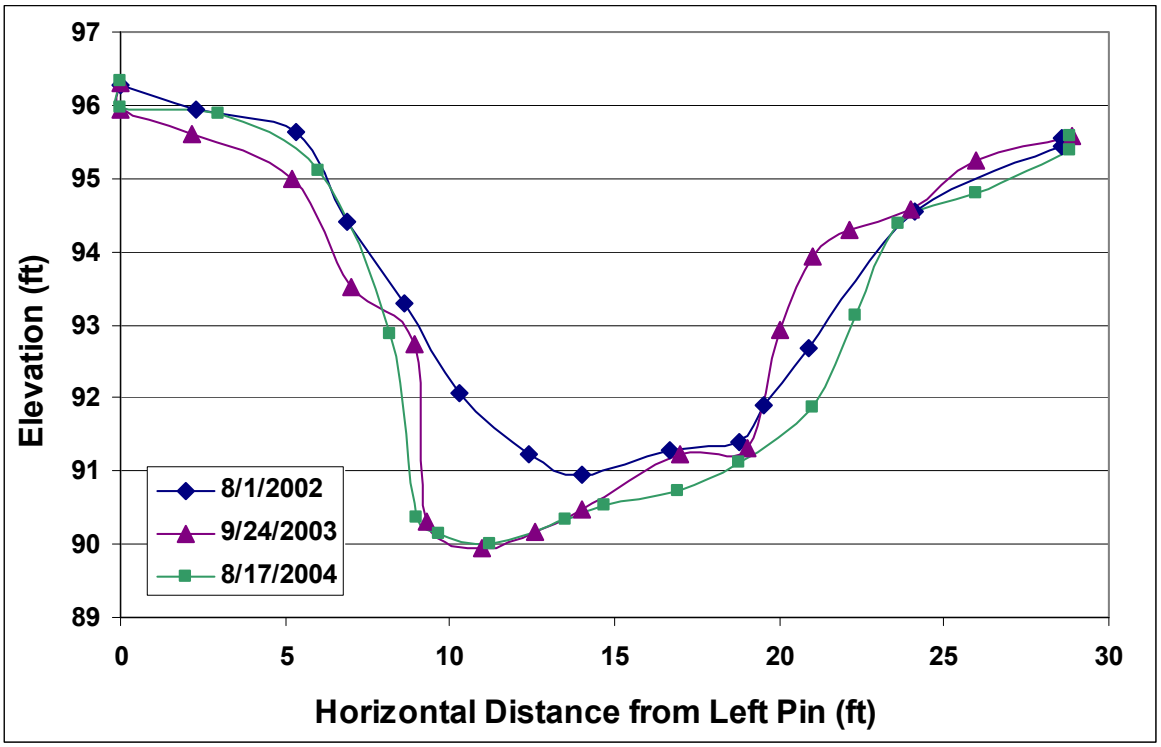

Figure 2.21: Annual Surveys for Cross-Section 7 (S7).

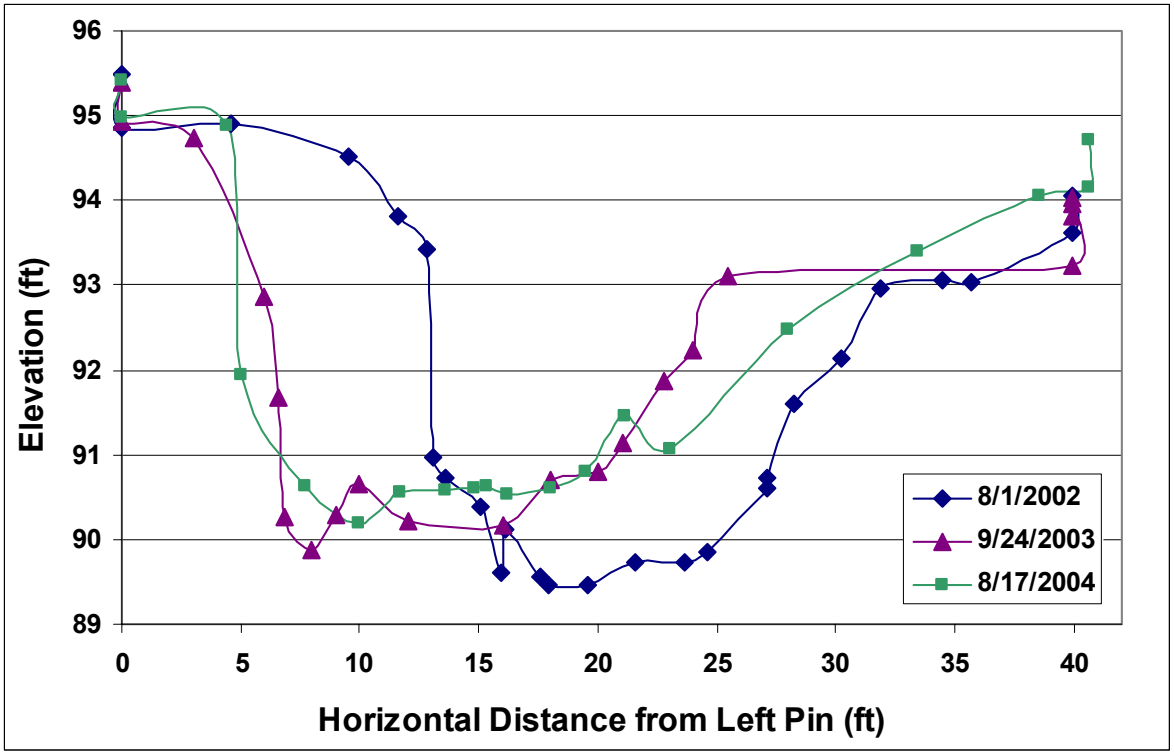

Figure 2.22: Annual Surveys for Cross-Section 8 (S8).

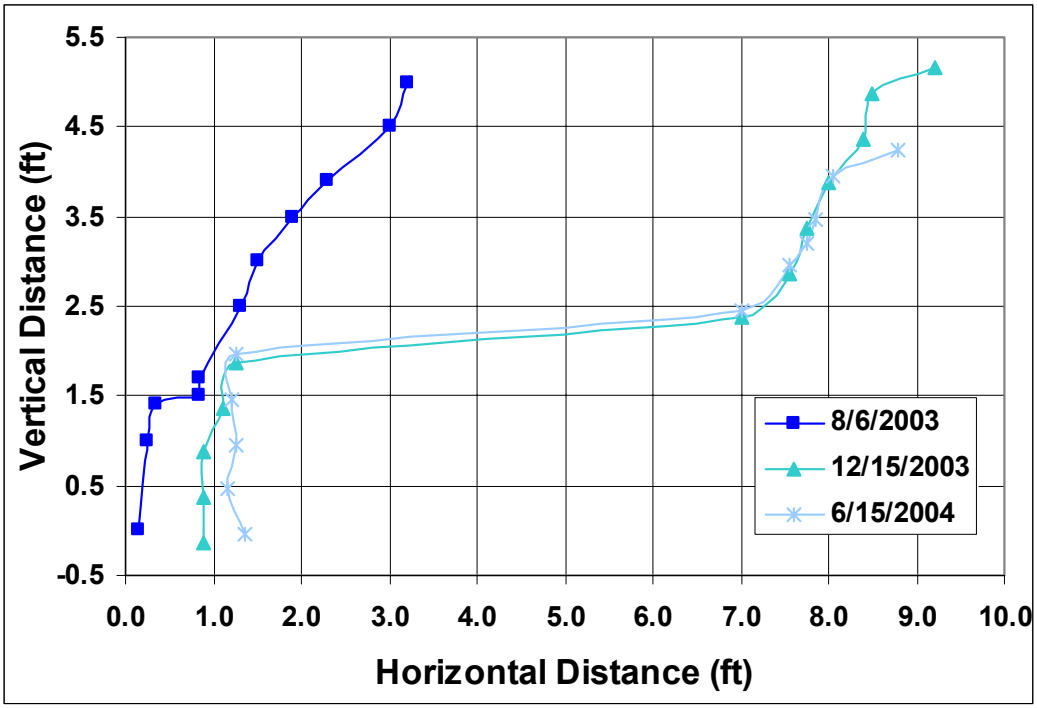

Figure 2.23: Right Streambank Profile at Toe Pin 13, in Reach 3.

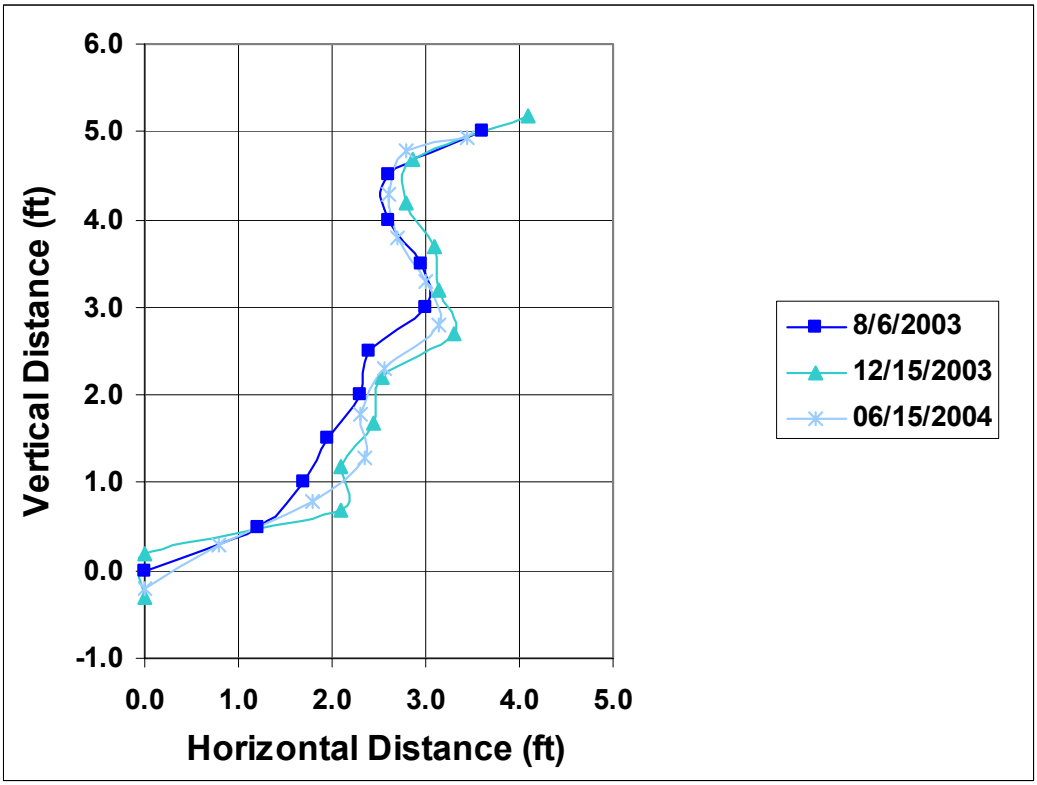

Figure 2.24: Right Streambank Profile of Toe Pin 14, in Reach 3.

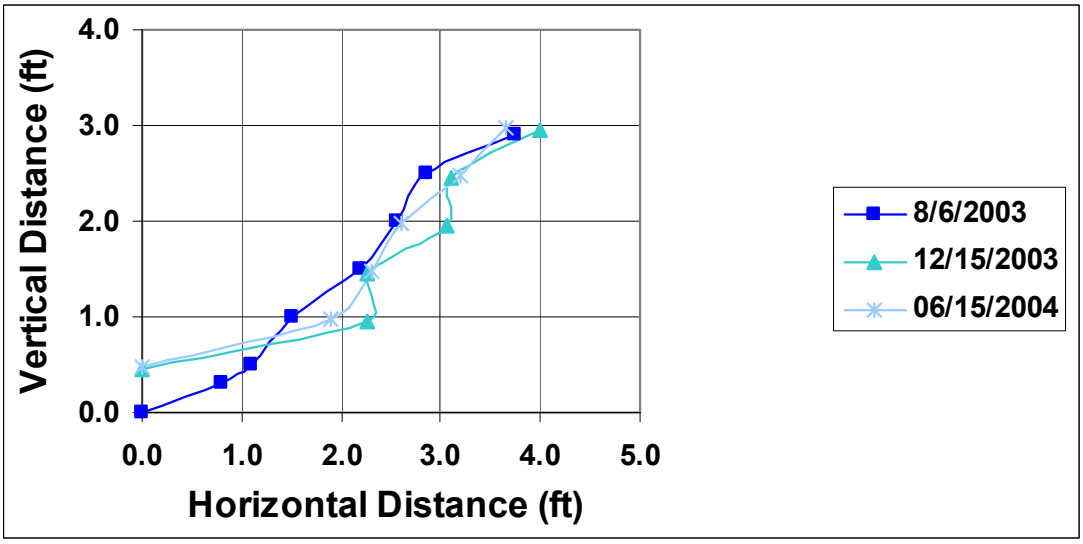

Figure 2.25: Right Streambank Profile at Toe Pin 15, in Reach 3.

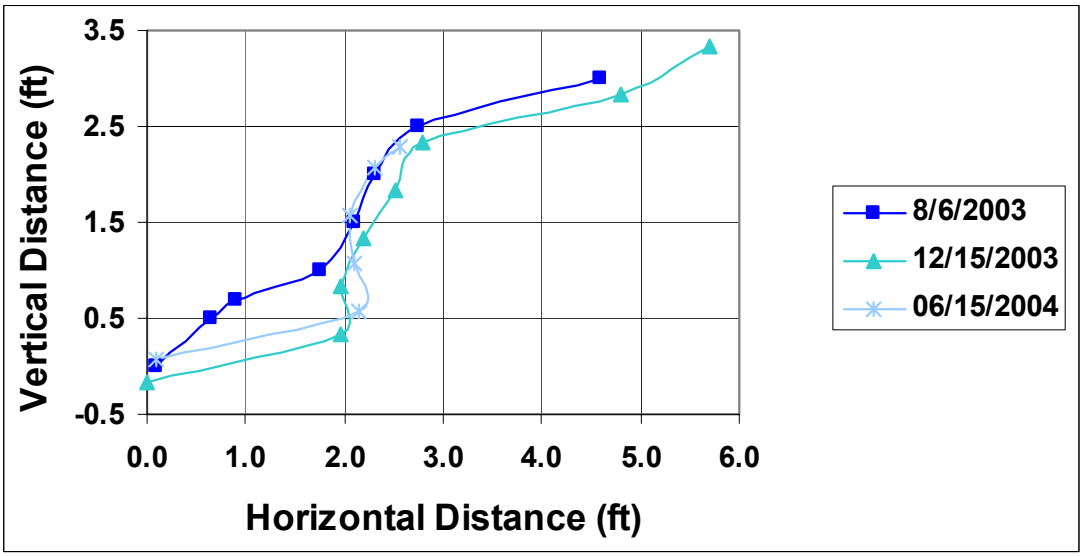

Figure 2.26: Left Streambank Profile at Toe Pin 16, in Reach 3.

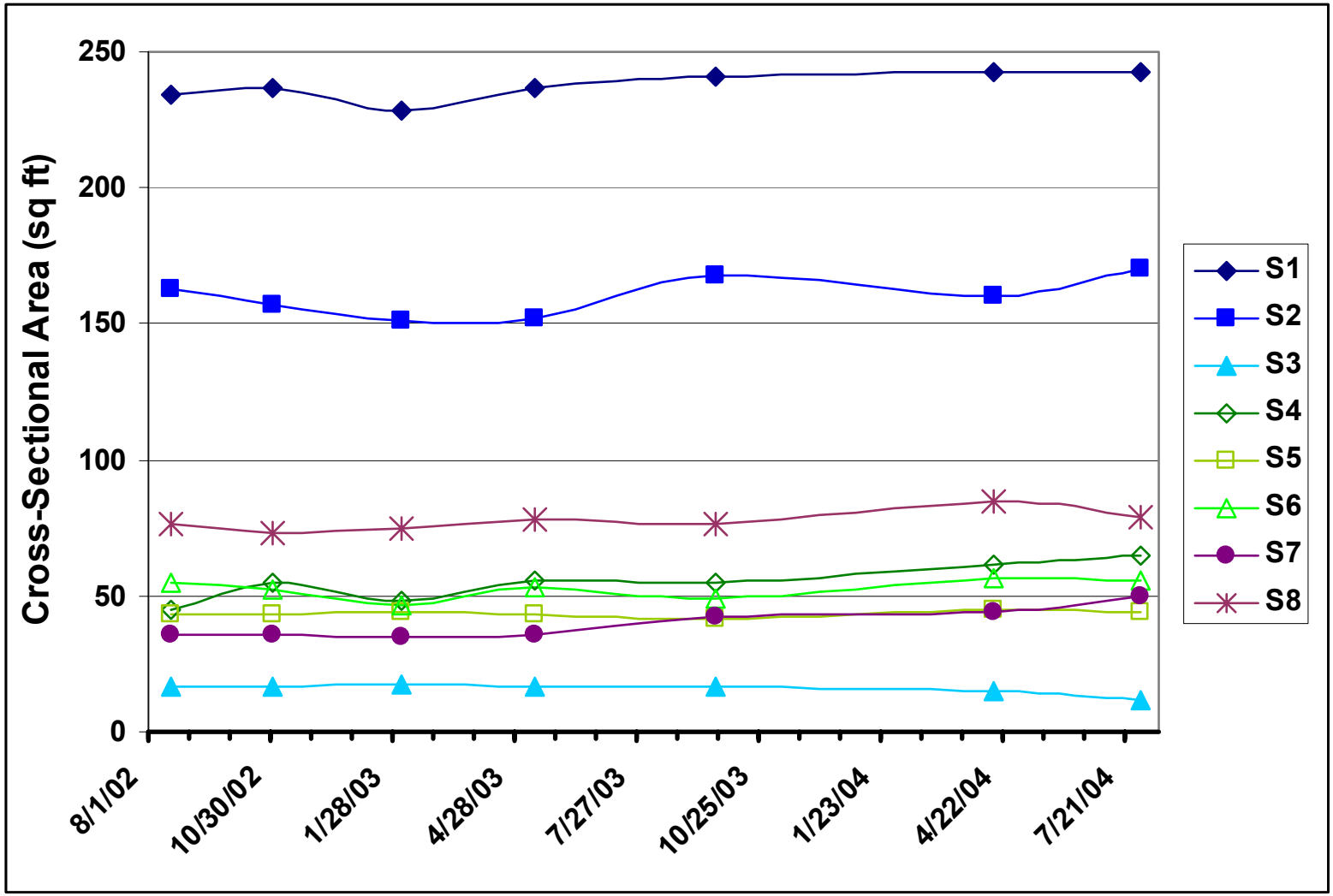

Figure 2.27: Cross-Sectional Areas Between Top of Bank Elevations Between August 2002 and August 2004.

Streambank Erosion Summary

A summary of the streambank erosion data measured in each of the North Creek reaches is provided in figures. The average cumulative lateral changes (lateral ft) in streambank profile for each cross-section and toe-pin were determined in order to compare average linear amounts (ft) lost or gained between surveys and between reaches. The figures (Figure 2.28-2.30) show that there was more erosion than aggradation in each reach, based on the number of points below zero and the number of negative slopes between points. Based on these figures, the extent of streambank erosion was greatest in Reach 3 (Figure 2.30); while similar between Reach 1 (Figure 2.28) and Reach 2 (Figure 2.29). These figures also highlight the effects of the two intense storms in late July and early August 2003. During this time, the lateral changes in most streambank profiles had a downward slope indicating erosion.

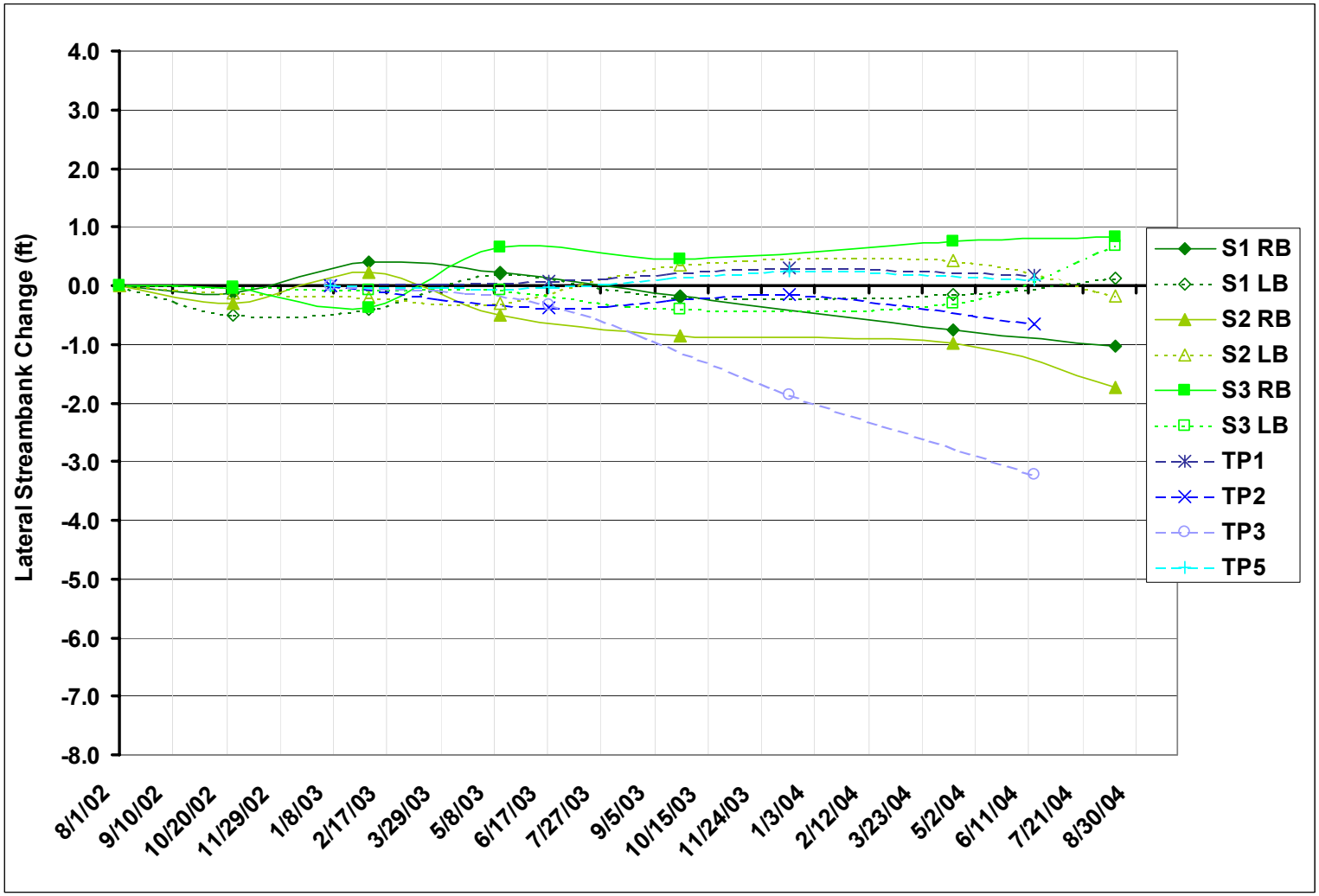

Figure 2.28: Average Cumulative Lateral Streambank Change Over Time in Reach 1.

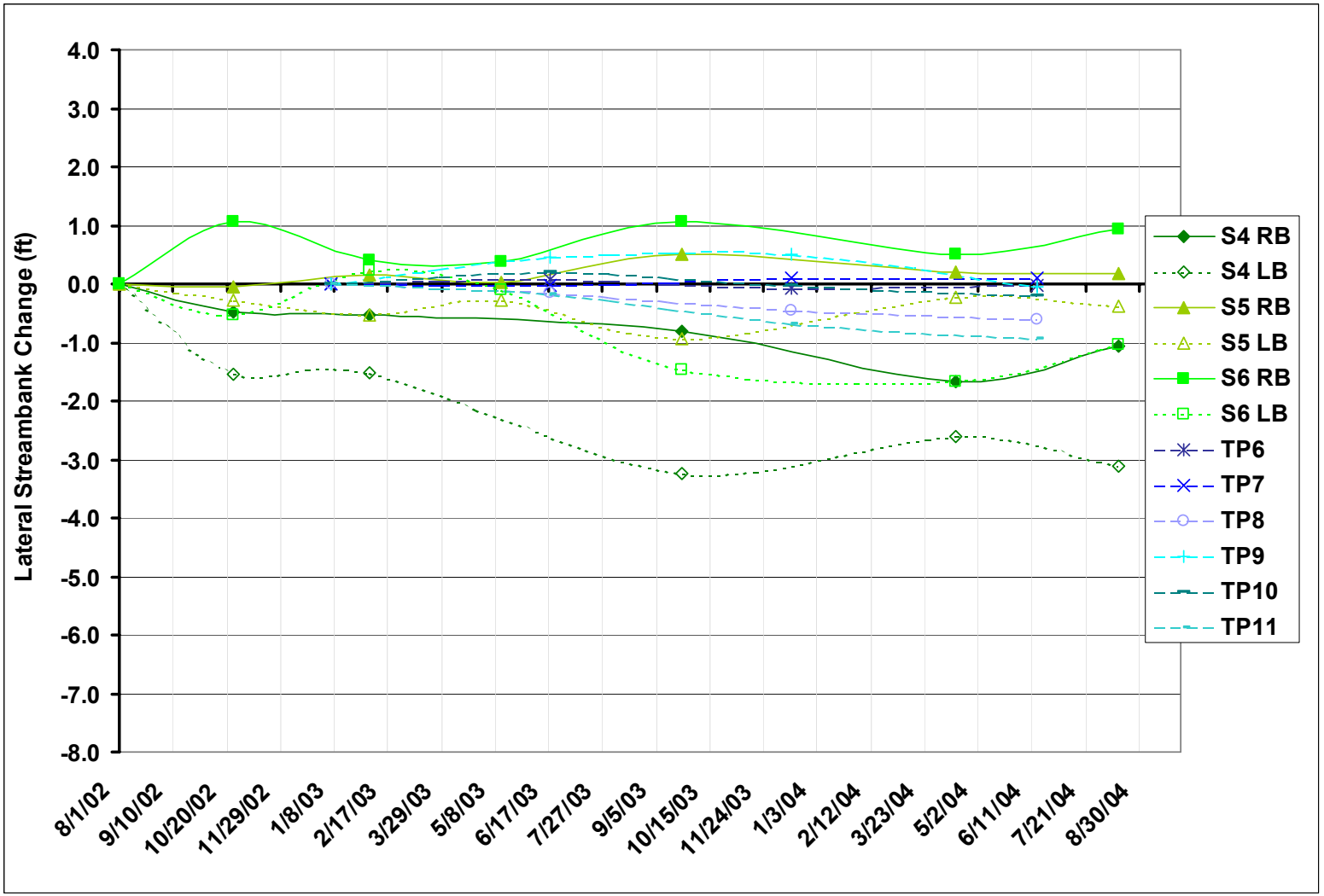

Figure 2.29: Average Cumulative Lateral Streambank Change Over Time in Reach 2.

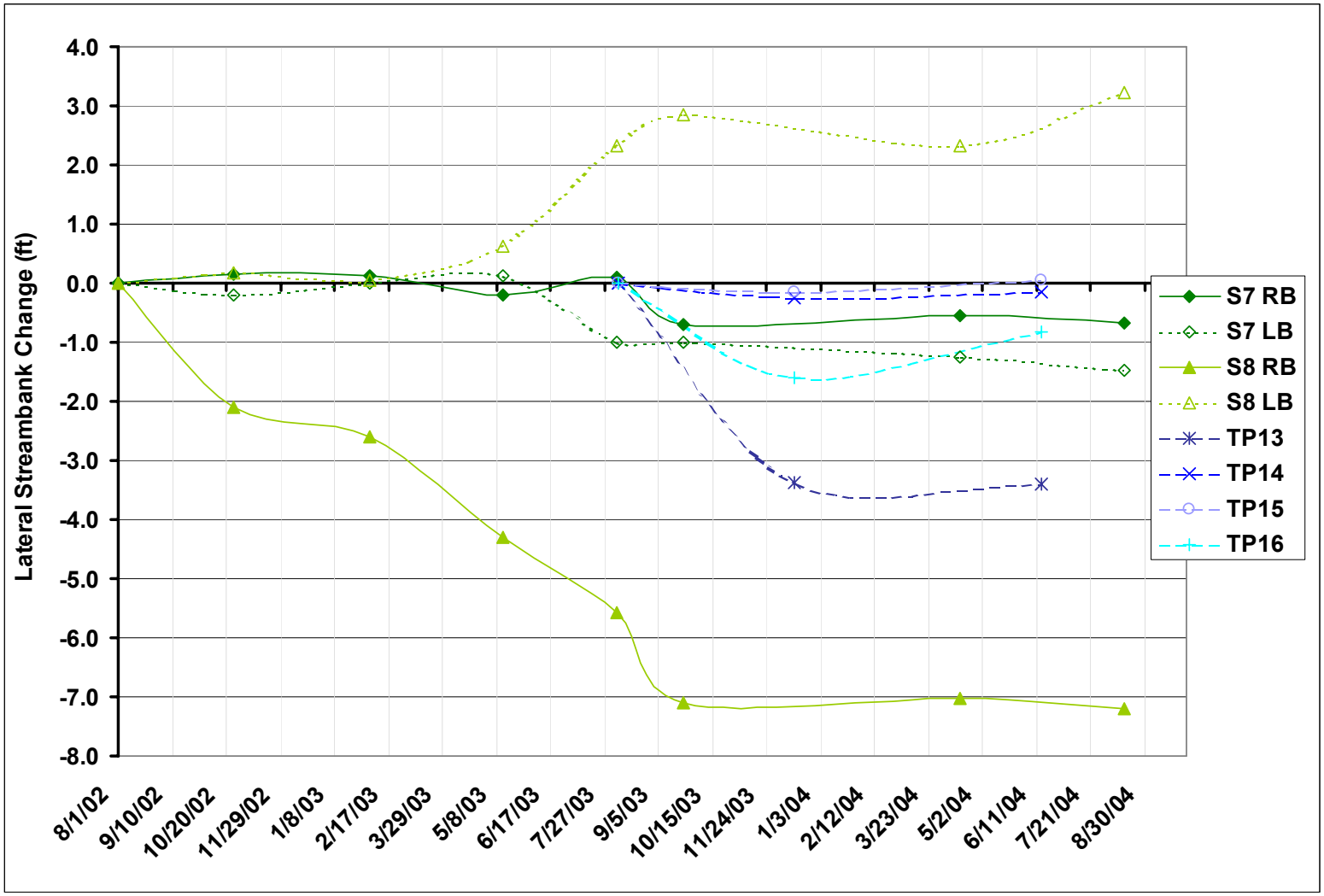

Figure 2.30: Average Cumulative Lateral Streambank Change Over Time in Reach 3.

Streambank Erosion Rates

The average lateral change in ft/yr for each streambank profile was calculated. These average erosion rates at each sampling location during the two-year study were used to determine average annual erosion rates (ft/yr) for each reach. Annual rates for individual years could not be calculated since the cross-sections and toe-pins were surveyed at different intervals and over different time periods. All three North Creek reaches had positive average erosion rates for the two-year study (Table 2.1). Reach 1 and Reach 2 had similar average erosion rates (0.3 ft/yr), while Reach 3 had the highest average erosion rate (1.5 ft/yr). These average rates were comparable to another small urban stream in the Raleigh area. Richland Creek (watershed 0.7 sq mi) experienced erosion rates of 0.42 – 1.09 ft/yr (Patterson, 2001). The North Creek erosion rates were also comparable to those found for two small Virginia streams in Rappahannock River basin, during watershed residential and commercial development (Bass and Strickler, 2000).

The average annual erosion rates at the different measurement locations in Reach 1 ranged from –0.48 to 2.19 ft/yr, and in Reach 2 ranged from –0.57 to 1.80 ft/yr. In both reaches, only half of the measurement locations experienced net erosion over time. Reach 3 had average erosion of the streambank profiles at almost all locations, ranging from 0.16 to 5.07 ft/yr lost. Only one location showed net negative erosion over time, with an annual average addition of about 1.68 feet to the profile. The installation of the toe-pins after channel instability was initiated by the vertical incision probably affected the average

erosion rate in this reach. Had these toe-pins been installed in 1/03, the average would probably have been reduced but not less than the upper reaches.

To compare the variability between streambank profile surveys in each reach, a variability chart was derived for all of the erosion rates measured during the study (Figure 2.31). Each point on the chart represents an average lateral change in streambank profile between two surveys for a particular sampling location. Positive values indicate erosion, while negative values indicate addition of materials to the streambank profile. All three reaches were highly variable, with both positive and negative erosion rates measured in each reach. The most variable reach was Reach 3, with variability decreasing moving upstream through the reaches.

The variability in measured erosion rates over time at each location could be attributed to several factors. Many of the streambank profiles fluctuated between positive and negative erosion rates during the study. This fluctuation could have been caused by bank material slumping or mass wasting during stormflow recessions, then collecting at the toe (USACE, 1981). Differential suspended sediment deposition caused by streamflow changes may also have introduced variability at the bank toe (Leopold *et al*., 1992). Survey errors contributed another source of variability within the surveys. The same channel points were not measured between cross-section surveys and undercut regions proved difficult to measure with vertical rods. The toe-pin surveys had potential vertical measurement errors due to changing substrate and debris damage to the toe-pins. Potential horizontal measurement errors could have occurred when aligning the tape exactly perpendicular to the streambank and at the same angle from the vertical rod.

Variability within reaches between sampling locations was also evident during this study. Streambanks exposed to higher hydraulic shear stress would have higher erosion rates. The shear stress applied by stormflows was probably different for all streambank profiles measured, depending on the thalweg location, channel width, and bed feature (Keller *et al*., 1990). Other channel features that affected localized stormflow velocities through flow diversion included debris jams (ex. S2, S4, S8 downstream) and SOCs entering near a measurement location (ex. S1, S4, TP 16). Another source of variability within reaches was attributed to vegetation. Streambanks associated with large trees were either stabilized by the roots (ex. S4 right bank, S5), or destabilized by tree fall (ex. TP 13), (Thorne and Osman, 1988). Most of the streambank faces were vegetated with the weedy vines (ex. kudzu) that either anchored small tendrils or grew down the bank from upslope. These vine masses could have decreased shear stress during summer, but their role in winter when the vegetative portion was dead was unknown.

The difference in average erosion rates between reaches during the study was attributable to differences in channel morphology, substrate type, and streambed slope. The relatively steep, vertical streambanks in Reach 1 and Reach 2 (Table 2.1) were expected to have the greatest erosion rates. Erosion was predicted to occur through the process of undercut formation from increased stormflow velocities, followed by slumping and mass wasting of bank materials above due to geotechnical forces (USACE, 1981). In Reach 1, there was evidence of undercut formation in the S1, S2, and TP 3 profiles over time, with slumping of bank material from above. In Reach 2, undercutting was observed

at S4, S5, TP 8 and TP 10 over time. The rate of this undercutting process was probably affected by the presence of exposed saprolite near the bank toes

In Reach 3, the average erosion rates were expected to be the smallest compared to upstream reaches due to moderate bank heights and armoring of the upper portion by large stone. The average streambank erosion rates were highest in Reach 3, however, due to mostly site-specific influences. The rapid vertical incision from headcut migration in the upper reach during July and August 2003 created streambank instability as the armor layer was rapidly removed. Bank heights and angles were increased, and channel widening occurred (ex. S7, TP 14). Saprolite was not exposed in this reach, and the cohesive bank materials were easily eroded after the artificial armor layer was breeched. The upper portion of Reach 3 also had a much higher slope compared to the rest of North Creek, which would have increased streamflow velocity and shear stress in this stream section.

In the lower portion of the reach past the significant headcut, the streambank profiles also experienced considerable erosion. TP 16 demonstrated undercutting from SOC7 stormwater entering upstream. The large amount of erosion at S8 (cumulative 8-ft lateral) was believed to be caused by hydraulic forces from SWMP backwater (Munson 1998). This cross-section created the most erosion rate variability for the reach, with large erosion rates on the left streambank offset by large aggradation rates on the right streambank. Over time a large debris jam of old erosion control matting and trees developed downstream of this cross-section, and also contributed to the differences in erosion between streambanks.

Table 2.1: Average Streambank Erosion Rates Measured along North Creek During the Two-Year Study, with Positive Values Indicating Erosion.

Reach 1	Bank Height (ft)	Average Erosion Rate (ft/yr)
S1 RB	6.0	0.32
S1 LB	10.0	-0.12
TP1	6.0	-0.14
S2 RB	9.5	0.89
S2 LB	8.0	0.21
TP2	7.0	0.48
TP3	4.5	2.19
TP5	5.0	-0.07
S3 RB	2.0	-0.48
S3 LB	2.5	-0.34
Average	***6.1***	***0.30***
Reach 2	**Bank Height (ft)**	**Average Erosion Rate (ft/yr)**
TP6	4.0	0.01
S4RB	8.0	0.46
S4LB	3.5	1.80
TP7	7.0	-0.06
TP8	4.5	0.42
TP9	7.5	-0.02
S5RB	4.0	-0.11
S5LB	5.0	0.30
TP10	3.0	0.11
TP11	3.0	0.64
S6RB	3.0	-0.57
S6LB	2.5	0.36
Average	***4.6***	***0.28***
Reach 3	**Bank Height (ft)**	**Average Erosion Rate (ft/yr)**
TP13	3.2	5.07
S7RB	2.5	0.59
S7LB	3.5	0.65
TP14	5	0.35
TP15	2.9	0.16
TP16	2.5	2.21
S8RB	3.5	4.48
S8LB	2.5	-1.68
Average	***3.2***	***1.48***

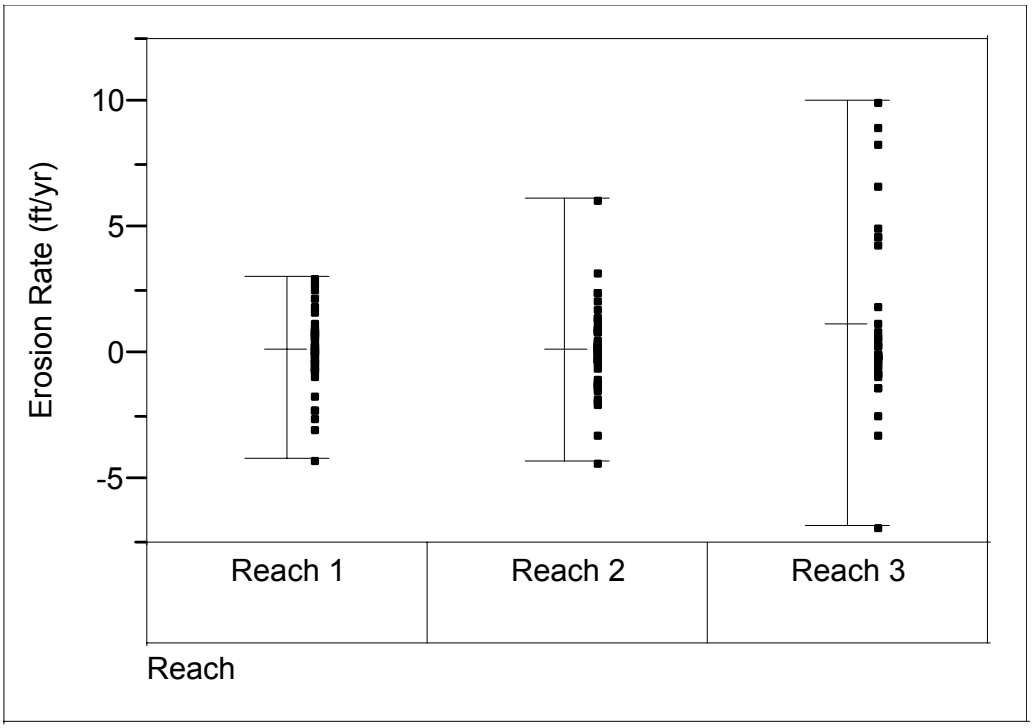

Figure 2.31: Variability Chart for Erosion Rates in the North Creek Reaches, with Cross Bars Showing the Mean Value and the Range of Values. Positive Values Suggest Erosion, while Negative Values Suggest Aggradation.

Streambank Channel Evolution

The streambank erosion results deviated from the expected patterns predicted by channel evolution concepts (Schumm *et al*., 1984; Simon, 1989). The upper reaches were thought to be in Stage IV of channel evolution, when vertical incision reaches a maximum and channel widening begins. Reach 3 was thought to be in Stage III, with vertical incision progressing as stormflows increased. The expected patterns were based on the assumption that watershed disturbance was continuing in the form of development.

The deviations may have been caused by the potential overestimation of watershed disturbance from development, as discussed in the streambed erosion section. Stormwater

retention associated with development sites (37 ac) and SWMP #2 may have reduced stormwater velocities and runoff volumes for less channel erosion. Studies have shown, however, that slow release of stormwater over time may not decrease streambank erosion due to the increased streambank exposure to low or moderate flows (Richardson *et al*., 1990; Bledsoe, 2002). These reduced flows could still entrain streambank materials and cause undercutting near the toe. Increased bank moisture is also known to cause geotechnical instability in streambanks (Thorne, 1982).

The deviations from the channel evolution pattern were also caused by the many unnatural conditions present within the reaches. The streambed and streambanks were all altered in the past by channelization, and Reach 3 appeared to be amended with artificial substrates along the entire reach at one time. These localized erosion control measures could have accelerated erosion downstream, or at the application site once instability was initiated (Dunne and Leopold, 1978). Entrainment of the large artificial stone in the upper reach of Reach 3 accelerated the evolution process in this reach, with changes progressing rapidly from Stage III to Stage IV during the study. Unnatural influences were also provided by the large in-stream structures along the stream, including the road culverts and the in-line SWMPs, that altered stormflow velocities through grade control and channel width. The SWMPs could have been highly influential on streambank erosion in the lower portion of Reach 3 due to backwater effects. This portion of the reach went from a Stage III to a Stage V, without the expected vertical incision of Stage IV.

Streambank Erosion Prediction

BEHI Assessments

The BEHI assessment performed along the stream length required several assumptions, held consistent between the two surveys. Kudzu, honeysuckle, and other vines were the predominant vegetation along steep banks. These vines were not used to define rooting depth but were considered in the surface protection estimation due to their presumed role in stormflow energy dissipation. The saprolite along the base of streambanks in Reach 1 and Reach 2 was considered as sandy soil material rather than bedrock for soil composition adjustments. Even though the material was thought to decrease streambank and streambed erosion, it was highly weathered in most areas and crumbled easily to touch when moist. In portions of the stream where top of bank elevation was not clearly defined, the first significant break in slope moving up the bank was chosen for bank height determination. This assumption fit well with the steep, near vertical streambanks along most of the stream length.

The total channel lengths classified in each BEHI category for each reach of North Creek are shown in Table 2.2. BEHI classification maps with distances along the thalweg of each reach are provided in Figure 2.32-2.34, with the 2002 assessment line closest to the stream on each side. In both BEHI assessments, no streambank was classified as Low or Very Low erosion potential. The streambank erosion potential determined by the BEHI assessment was expected to increase over time in each reach, with an increase in the stream length classified within the Very High and Extreme categories, due to increased streambank erosion over time.

Reach 1 had an increase in stream length in the Very High category to offset a decrease in the High category (Table 2.2), that predominantly occurred mid-reach and near the Varsity Drive culvert along both streambanks (Figure 2.32). Reach 2 demonstrated a shift toward Extreme and Very High erosion potential over time. In the July 2002 survey, the streambanks were all classified as either Moderate or High (Table 2.2). The shift was variable between streambanks and locations along the reach (Figure 2.33). The total stream length was shorter during the second survey due to the stormwater outfall channel near the Varsity Drive culvert (Station 830-ft) that removed about 35 linear feet of streambank through erosion and loss of trees at the top of bank. Reach 3 increased in erosion potential through addition of Extreme category sections of streambank. The stream length added in the Extreme category in Reach 3 was equal to the total length of stream classified as Extreme or Very High in Reach 2 (Table 2.2). Extreme classifications were added along both streambanks in the upper portion of the reach and on the left streambank near the end of the reach (Figure 2.34).

The comparison of shifting categories between the reaches showed that Reach 3 shifted the most, followed by Reach 2 and Reach 1, with all reaches increasing in bank erosion potential based on category lengths. The differential shift moving downstream was thought to be caused by the increase in streambank erosion moving downstream, which could have influenced BEHI variables associated with vegetative cover, bank heights, and bank angles. The differential shift between reaches could also be attributed to the existing bank erosion hazard associated with the steeper vertical streambanks in the upper reaches at the beginning of the study.

The proportions of each BEHI variable estimated during the assessments were compared to help determine which of these variables were most responsible for the shift in categories between surveys (Table 2.3). The greatest change over time in variable proportion for all reaches was the amount of surface protection. The average rating increased, which indicates less surface protection present on the streambanks and greater erosion potential. The total bank height to bankfull height ratio and the streambank angle were relatively constant, except in Reach 3 where an increase in bank height and angle was observed after the rapid vertical incision in August 2003. The rooting depth to bank height ratio variable was the only variable to decrease average ratings in each reach over time, which indicates that the rooting depth or the vertical extent of roots within the bank was greater and there was less erosion potential. In Reach 1, the greatest change was the decline in the rooting depth to bank height ratio variable, which affected the decline in the total average BEHI rating over time. In Reach 2 and Reach 3, the most change occurred with the surface protection variable increase, which affected the increase in total average BEHI rating in each reach over time. All total average ratings were in the High streambank erosion potential category.

The shifts in BEHI categories were due to both objective and subjective variable changes within the assessment. Objective measurements such as bank heights and angles were only influenced by changes in bank profile features, but the subjective measurements had a potential for bias. Bias was considered minimal due to the long time period between assessments and through the use of several surveyors allowing discussion and multiple perspectives of the subjective variables. In all three reaches, however, the rooting depth to

bank height depth increased over time even though root density and surface protection declined. A greater root depth may have been recorded in the 2004 winter survey due to more root exposure when vegetation was dead compared to the summer survey when rooting depth was obscured by vegetation.

The relationship between measured average annual erosion rates and the associated BEHI ratings determined during the 2004 survey is shown in Figure 2.35. Only positive erosion rates were included in this comparison. The erosion rates were not correlated with the BEHI rankings used to assign erosion potential categories ($r^2 = 0.01$). There were no BEHI rankings in the Very Low or Low categories (< 19.5), and only one BEHI ranking in the Moderate category (20.0-29.5); therefore, the comparison was not complete. The full set of categories could not be compared in North Creek due to the higher erosion potential of streambanks. These lower categories would be absent in most urban streams, and should be considered when applying the BEHI assessment. The lack of a relationship between measured erosion rates and BEHI ratings could also have been caused by site-specific factors present in North Creek that were not present in the streams used to derive the BEHI classification. These influences may have included the urban watershed, in-stream structures, and the assumptions made concerning BEHI variables.

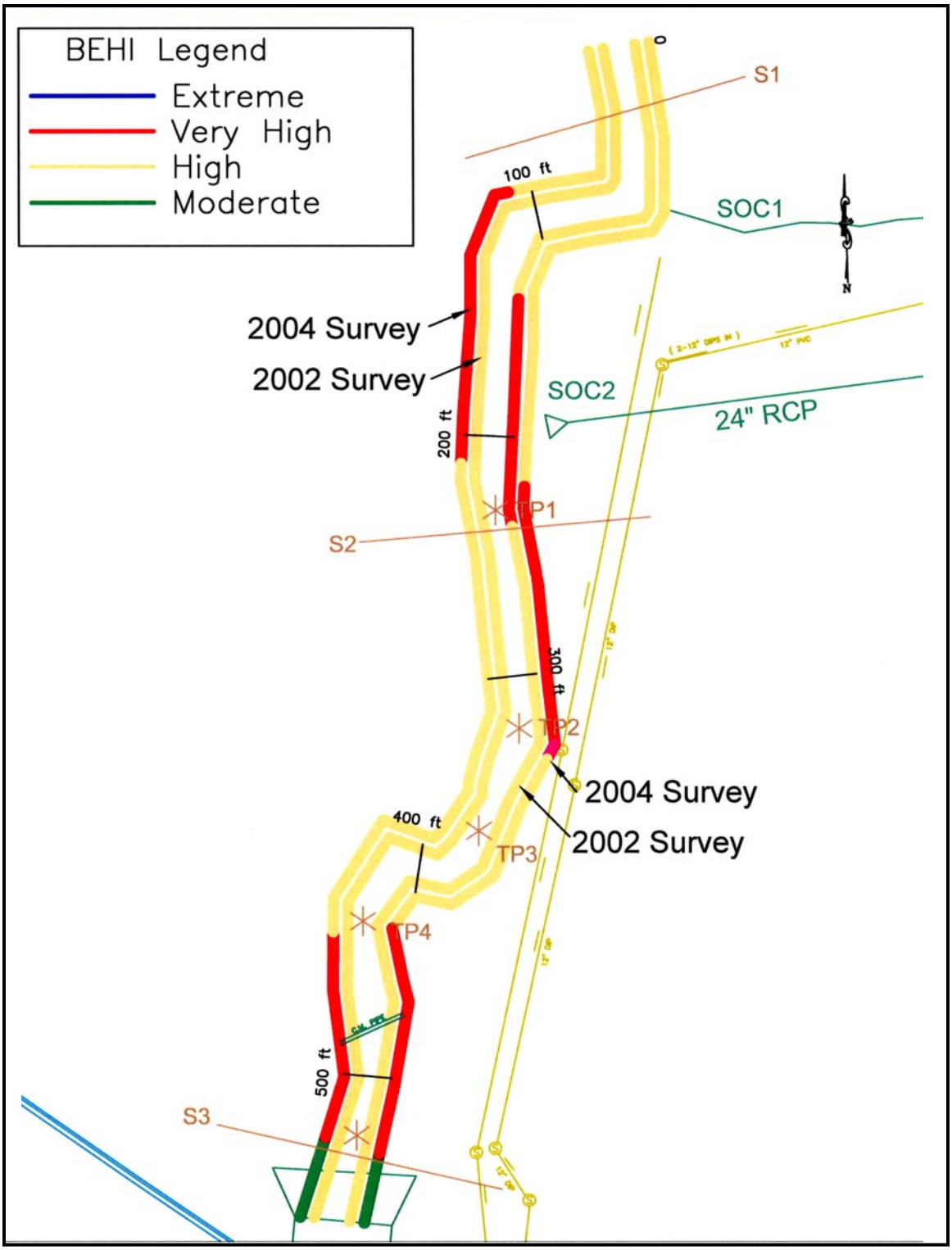

Figure 2.32: BEHI Categories Along North Creek Reach 1 Streambanks (Illustrated by Ian Jewell and Joseph Puckett, NCSU Water Quality Group). Scale 1 in = 60 ft.

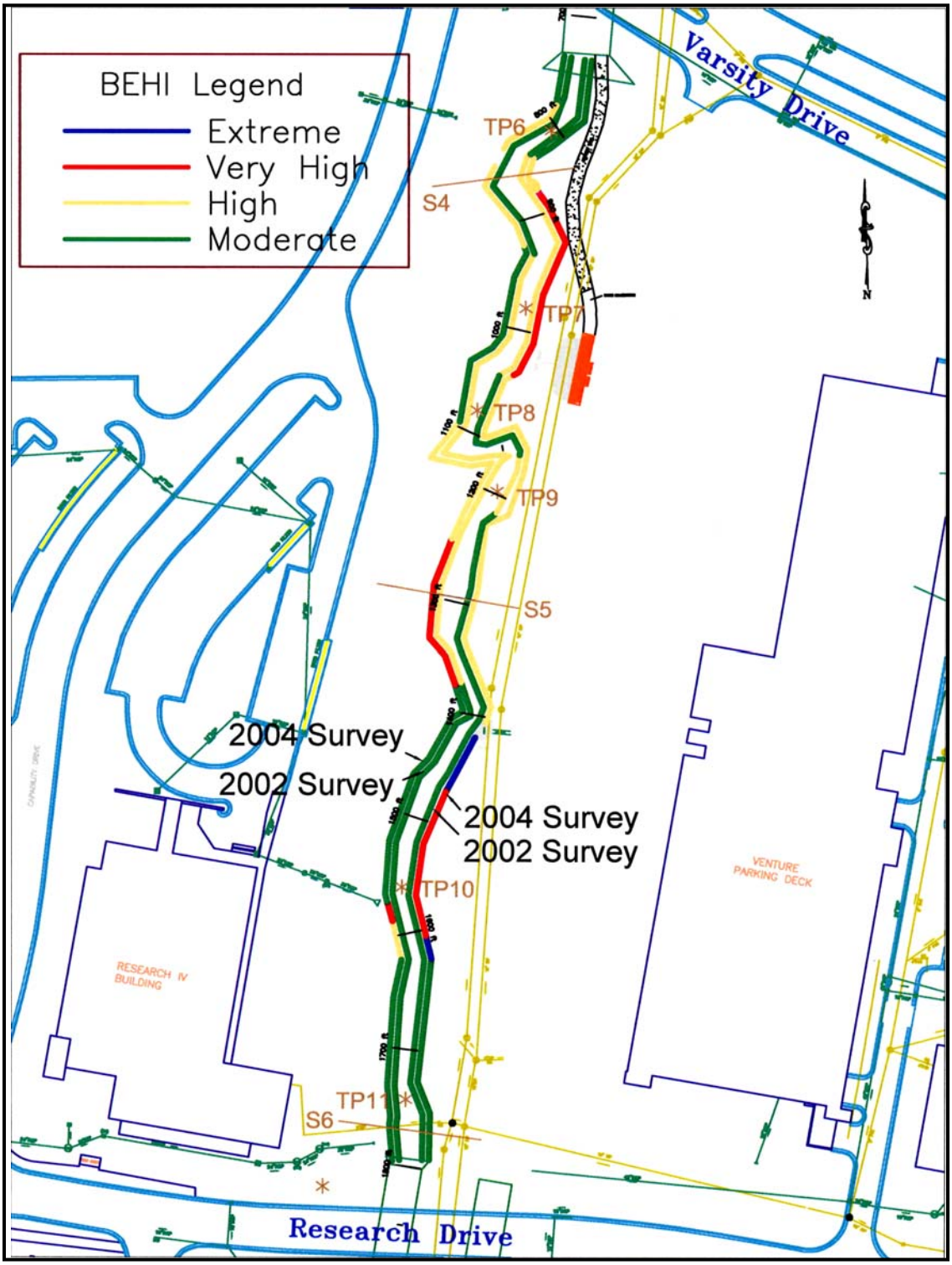

Figure 2.33: BEHI Categories Along North Creek Reach 2 Streambanks (Illustrated by Ian Jewell and Joseph Puckett, NCSU Water Quality Group). Scale 1 in = 125 ft.

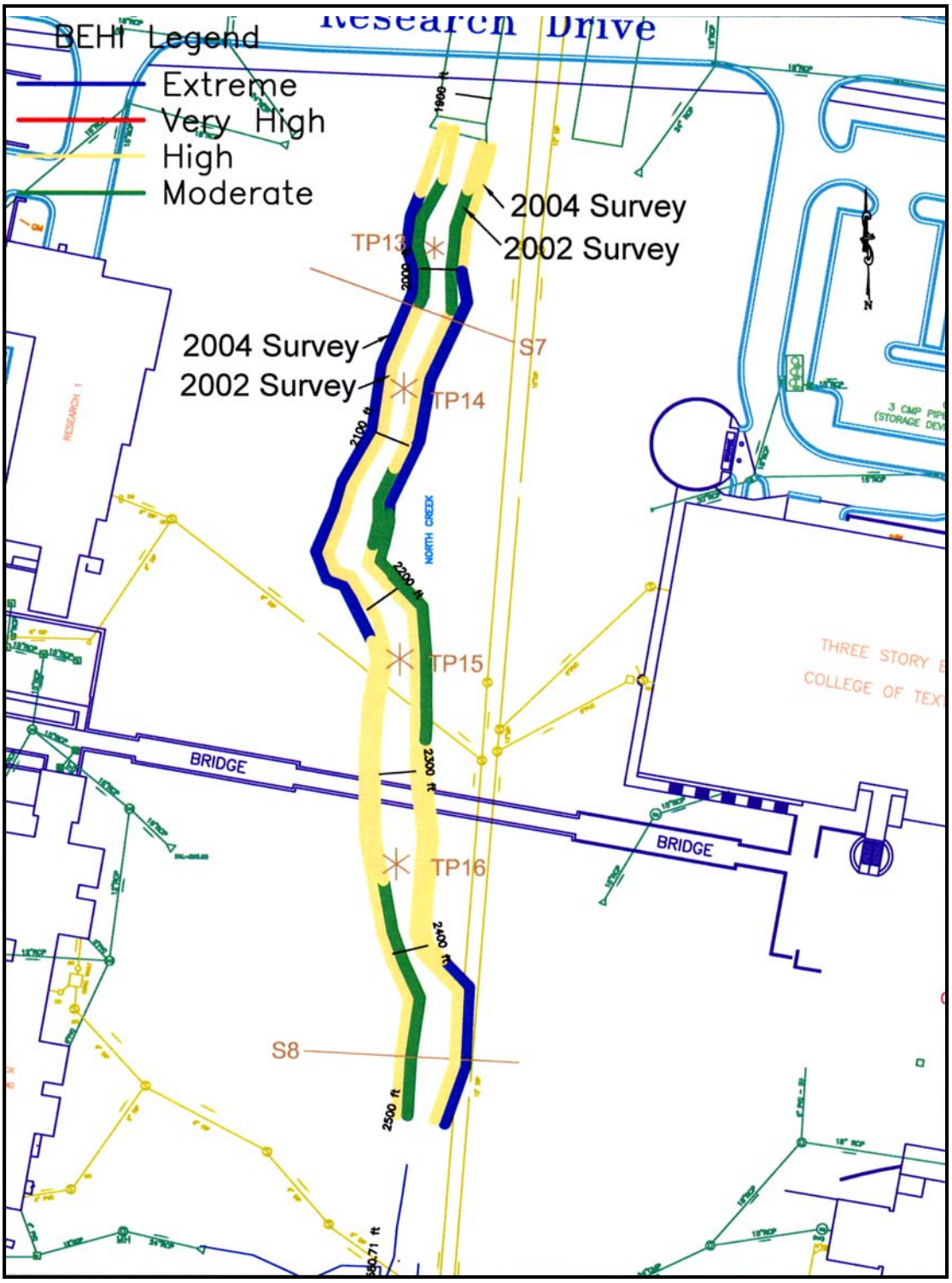

Figure 2.34: BEHI Categories Along North Creek Reach 3 Streambanks (Illustrated by Ian Jewell and Joseph Puckett, NCSU Water Quality Group). Scale 1 in = 60 ft.

Table 2.2: Total Stream Lengths Classified in Each BEHI Category per Reach in Both the 2002 and 2004 Assessments.

	Reach 1 Stream Length (ft)		Reach 2 Stream Length (ft)		Reach 3 Stream Length (ft)	
BEHI Category	**Jul 2002**	**Jan 2004**	**Jul 2002**	**Jan 2004**	**Jul 2002**	**Jan 2004**
Extreme	0	0	0	70	0	520
Very High	97	400	0	450	0	0
High	1023	660	711	720	853	550
Moderate	0	60	1409	845	357	140
Low	0	0	0	0	0	0
Very Low	0	0	0	0	0	0
Total	1120	1120	2120	2085	1210	1210

Table 2.3: The Proportion of Each BEHI Variable in the Total BEHI Ranking for Each North Creek Reach During the Two Surveys.

	Reach 1		Reach 2		Reach 3		Total	
BEHI Variable	**2002**	**2004**	**2002**	**2004**	**2002**	**2004**	**2002**	**2004**
Bank Height / Bankfull Height Ratio	10.00	9.12	9.59	8.72	5.86	7.79	8.67	8.72
Root Depth / Bank Height Ratio	8.34	5.07	5.85	4.40	6.51	5.28	6.73	4.78
Root Density	5.97	7.55	6.13	7.11	7.88	8.75	6.57	7.52
Bank Angle	4.61	4.26	3.87	4.48	3.99	6.36	4.11	4.70
Surfuce Protection	4.01	5.87	2.57	6.04	1.71	6.63	2.73	6.07
Soil Composition Adjustment	5.00	4.89	5.00	5.00	5.00	5.00	5.00	4.97
Total BEHI Rating	35.93	34.62	30.83	33.94	30.95	39.81	32.28	35.09

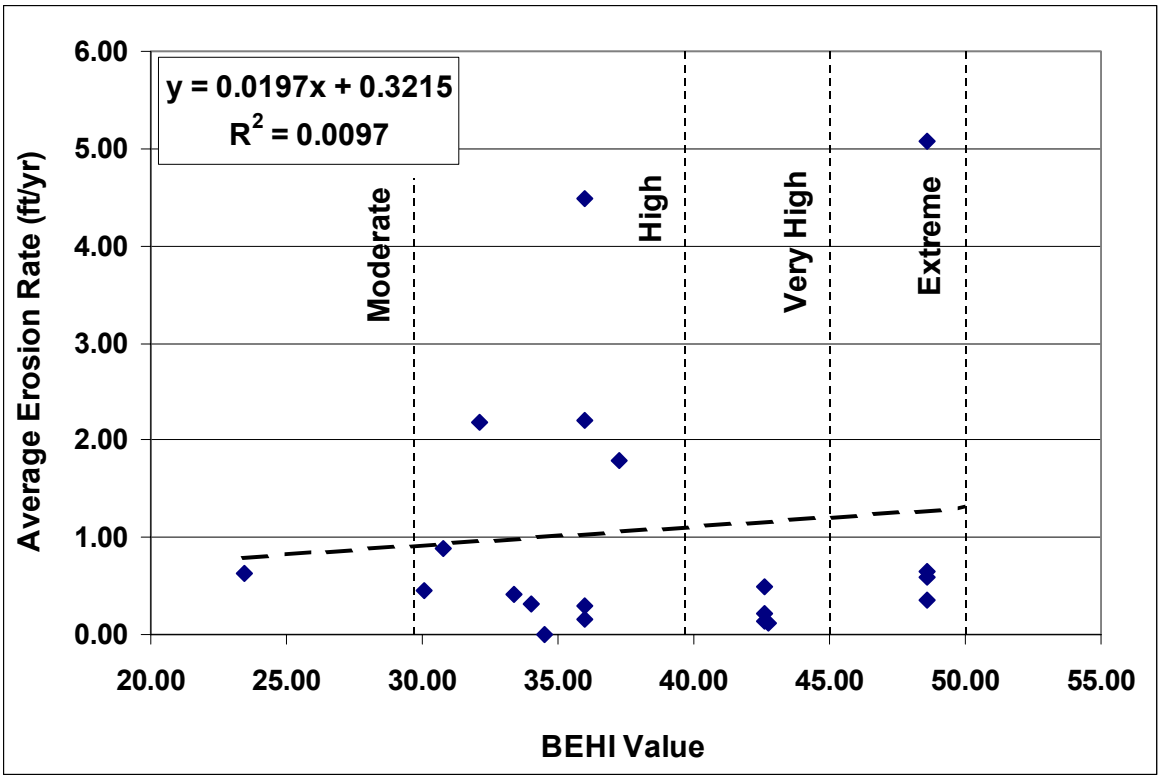

Figure 2.35: The Relationship Between Average Streambank Erosion Rates and BEHI Ratings For Cross-Section and Toe-Pin Locations along North Creek, with Regression Equation.

Streambank Erodibility Model

The use of the streambank erodibility model (Rosgen 2001) to predict bank erosion rates in North Creek was evaluated. The predicted erosion rates (ft/yr) for North Creek were based on BEHI and the NBS index values and a published erodibility chart (Colorado data from Rosgen 2001; Appendix 2, Figure A2.9). BEHI and NBS categories for each streambank profile are provided in Table 2.4, along with the predicted erosion rates.

Differences between measured average erosion rates (ft/yr) in each reach and predicted erosion rates by the model were thought to be influenced by the urban watershed

surrounding North Creek compared to the Colorado streams used to derive the model. The average erosion rate predicted was smaller than the observed rates in Reach 1 and Reach 3, but not in Reach 2 (Table 2.4). Ten of the thirty streambank profiles measured had negative average erosion rates, which affected the difference between the overall observed and predicted rates in each reach. There was no relationship between the predicted and measured erosion rates (Figure 2.36), even if negative erosion rates were not considered.

The differences between the average predicted and observed erosion rates in Reach 1 and Reach 2 could not be explained, considering these reaches were similar in streambank characteristics. The difference between the average predicted and measured rates in Reach 3 was most likely attributed to the site-specific influences, including tree fall at TP 13, the rapid vertical incision due to headcut migration, and the backwater effects of the downstream SWMPs. In both Reach 1 and Reach 3 where the average predicted rate was less than the measured rate, the NBS categories were all either Very Low to Low. These low NBS ratings reduced the predicted streambank erosion rates in the model, and were prevalent in these two reaches in straight sections (run features) with relatively homogeneous in cross-sections. Based on the inconsistent data within North Creek reaches, watershed development and the high proportion of impervious surfaces were not supported as causes of model discrepancy in North Creek, even though they may have been a factor.

Similar to the Rosgen (2001) model, an erodibility chart for the North Creek streambanks was derived based on the relationship between BEHI and NBS ratio, and average bank erosion rates (ft/yr) (Figure 2.37). The average erosion rates that were

negative or zero were not used to derive this relationship. The chart showed the scatter in the data across the higher BEHI categories. There was no correlation between the NBS ratio and the measured erosion rates for each BEHI category evaluated, (High, $r^2 = 0.17$; Very High, $r^2 = 0.24$; Extreme, $r^2 = 0.09$). Moderate category data were insufficient to analyze. Poor correlations were also observed in the Upper Illinois River, Oklahoma (Harmel *et al*., 1999), and in various North Carolina streams evaluated by Patterson (2001). Both of these studies evaluated streambanks in the higher categories, during periods of time with streamflows above bankfull flows.

North Creek streambank erosion rate data was combined with streambank erosion data from Patterson (2001), which evaluated seven different streams across the state. The resulting erodibility chart with data combined is shown in Figure 2.38. Both studies lacked sufficient data in the lower BEHI categories, and both studies presented considerable scatter in the data for the higher BEHI categories. Using the combined data set, correlation between the NBS ratio and the measured erosion rates for each BEHI category was assessed and no correlation was found for each BEHI category ($r^2 = 0.13$ or less). Moderate category data were again not analyzed due to limited data for North Creek.

The poor correlations could be a result of data analysis within the higher BEHI categories only. Streambanks in the lower categories should be added to the data set before relationships between the model variables can be established. The poor correlations may also have indicated that this streambank erodibility model was not the most appropriate for a small urban stream. The urban watershed conditions; differences in soils and vegetation; and differences in stream size may all have influenced the poor correlation

using the elements of the streambank erodibility model based on Colorado data. The model could be altered with further investigation of streambank erosion rates and variables affecting rates in small urban watersheds.

Table 2.4: Measured Erosion Rates Compared to Predicted Erosion Rates Using the Streambank Erodibility Model.

Reach 1	BEHI 2004 Category	NBS Category	Average Erosion Rate (ft/yr)	Predicted Erosion Rate (ft/yr)
S1 RB	High	Low	0.32	0.18
S1 LB	High	High	-0.12	0.49
TP1	Very High	Very Low	-0.14	0.10
S2 RB	High	Very Low	0.89	0.10
S2 LB	Very High	Very Low	0.21	0.10
TP2	Very High	Very Low	0.48	0.10
TP3	High	Very Low	2.19	0.10
TP5	Very High	Low	-0.07	0.18
S3 RB	Moderate	Very Low	-0.48	0.02
S3 LB	Moderate	Very Low	-0.34	0.02
Average			***0.30***	***0.14***
Reach 2	**BEHI 2004 Category**	**NBS Category**	**Average Erosion Rate (ft/yr)**	**Predicted Erosion Rate (ft/yr)**
TP6	High	Low	0.01	0.18
S4RB	High	Low	0.46	0.18
S4LB	High	Very Low	1.80	0.10
TP7	Very High	Low	-0.06	0.18
TP8	High	Extreme	0.42	1.45
TP9	High	Extreme	-0.02	1.45
S5RB	Very High	Very Low	-0.11	0.10
S5LB	High	Extreme	0.30	1.45
TP10	Very High	Low	0.11	0.18
TP11	Moderate	Very Low	0.64	0.02
S6RB	Moderate	Very Low	-0.57	0.02
S6LB	Moderate	Low	0.36	0.02
Average			***0.28***	***0.44***
Reach 3	**BEHI 2004 Category**	**NBS Category**	**Average Erosion Rate (ft/yr)**	**Predicted Erosion Rate (ft/yr)**
TP13	Extreme	Very Low	5.07	0.18
S7RB	Extreme	Very Low	0.59	0.18
S7LB	Extreme	Low	0.65	0.46
TP14	Extreme	Very Low	0.35	0.18
TP15	High	Very Low	0.16	0.1
TP16	High	Very High	2.21	0.8
S8RB	High	Very Low	4.48	0.1
S8LB	Extreme	Low	-1.68	0.46
Average			***1.48***	***0.31***

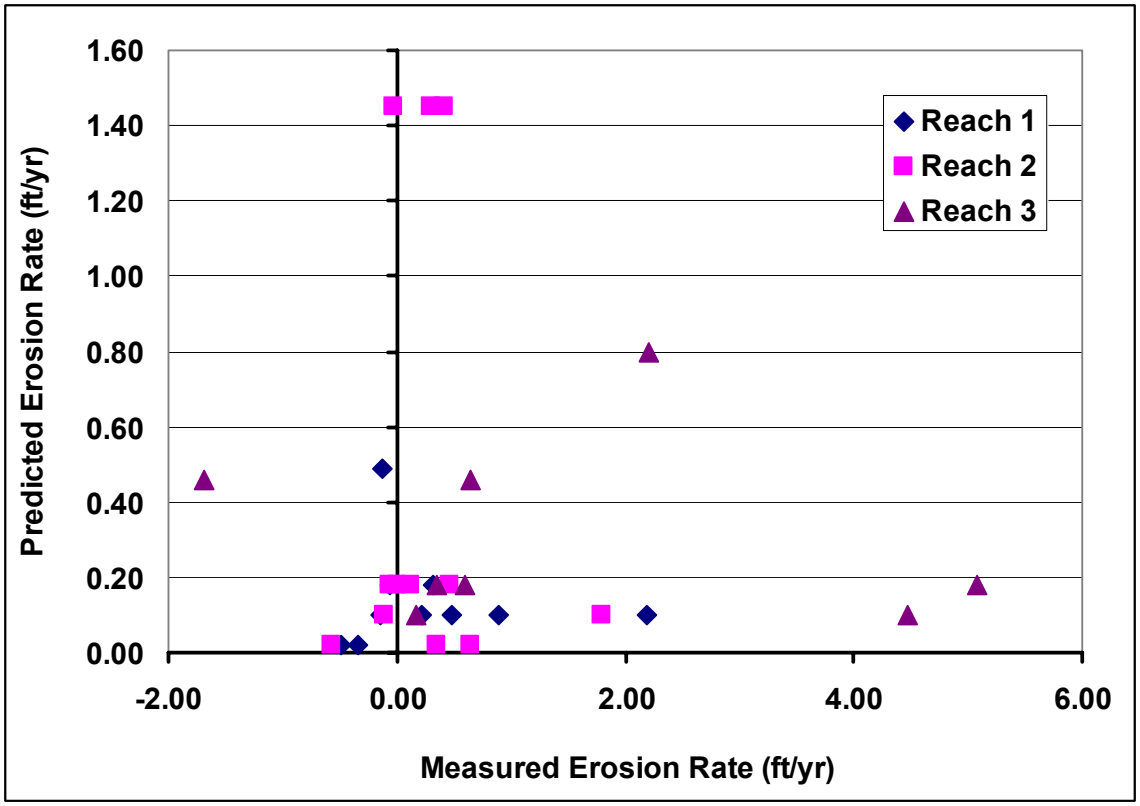

Figure 2.36: Predicted Erosion Rates Compared to Measured Erosion Rates in the North Creek Stream Reaches, with Positive Values Indicating Erosion.

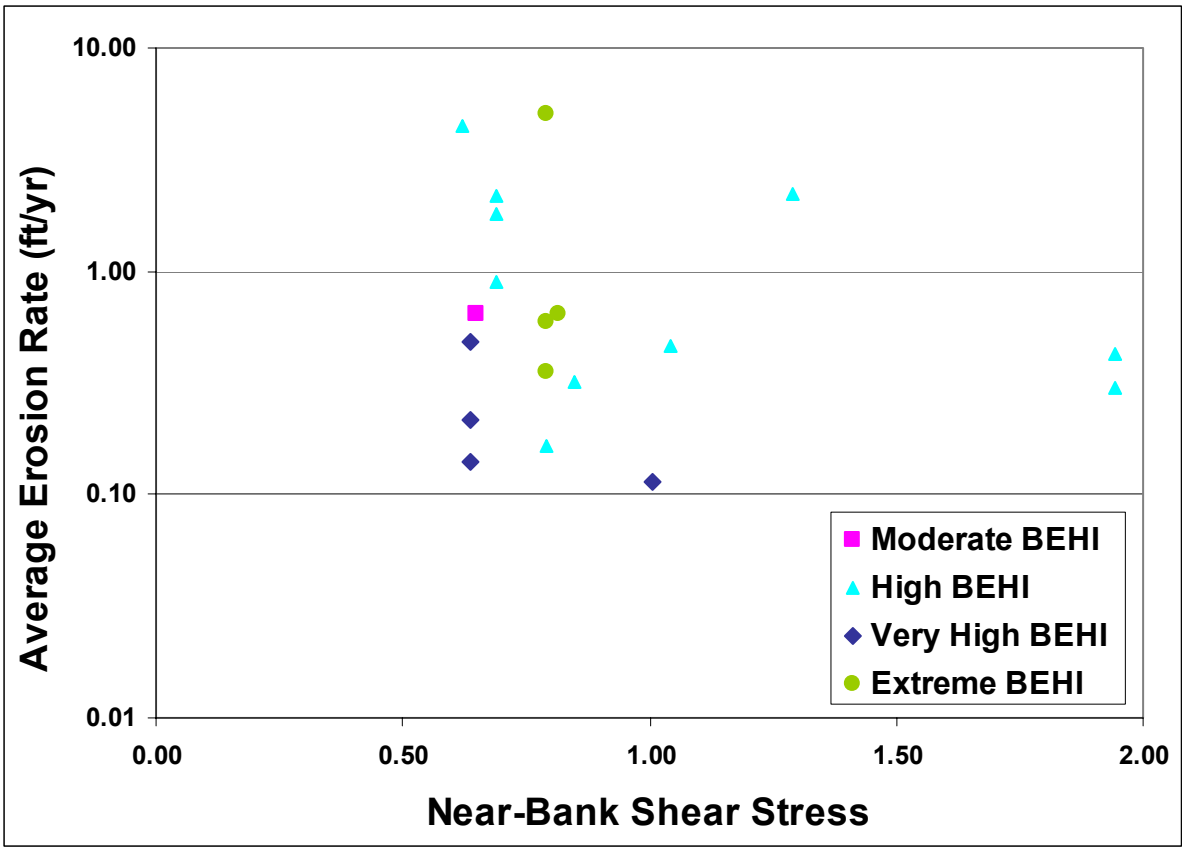

Figure 2.37: Streambank Erodibility Chart of North Creek Based on the Relationship Between BEHI Categories (Points) and NBS Values Corresponding with Erosion Rates.

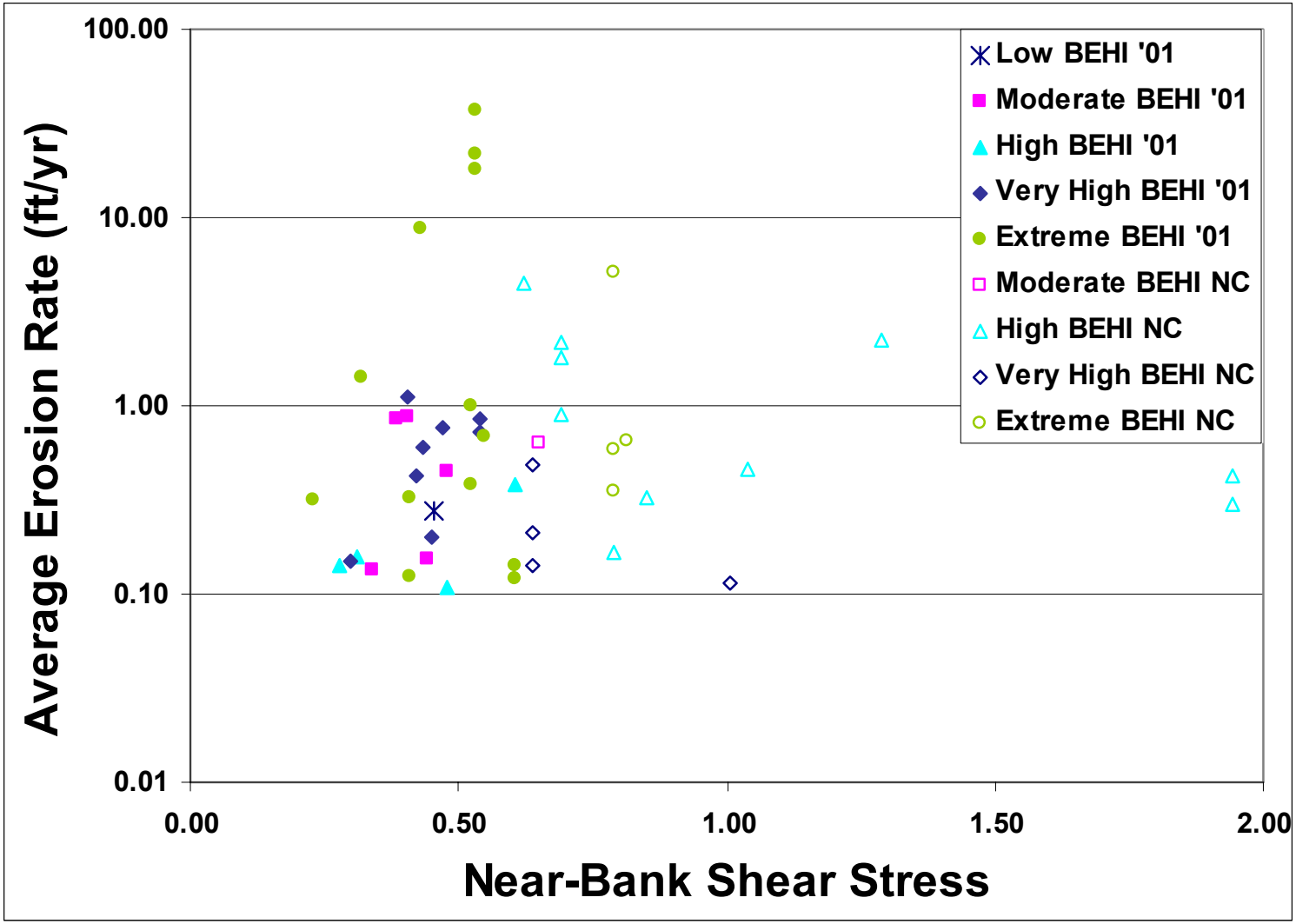

Figure 2.38: Comparison of Patterson (2001) Erodibility Chart Data and North Creek Erodibility Chart Data.

CONCLUSIONS

Channel erosion occurred in all three reaches of North Creek during the study period between August 2002 and August 2004. Reach 1 and Reach 2 experienced lateral streambank erosion at an average rate of 0.3 ft/yr, but there was not enough evidence to indicate streambed erosion. The overall bed slope was consistent, but the profile data suggested localized aggradation within both reaches. The aggradation was from sediment deposition probably derived from the combined effects of stormwater runoff sediment load and streambank erosion. Channel erosion in these upper reaches was thought to be limited by the presence of saprolite along the bed and lower banks, and by the grade control provided by the large in-stream culverts. In Reach 3, channel erosion was the greatest, with substantial vertical incision (1-3 ft) of the streambed followed by channel widening (average 1.7 ft/yr) in the upper portion after the two large storms in the summer of 2003. The change in this portion of North Creek was mostly caused by the steep slope in combination with a headcut migration that removed the artificial stone armor layer throughout. The lower portion of Reach 3 evolved rapidly through degradation and aggradation cycles, due to influences of unstable artificial substrates, channelization, and backwater from in-stream ponding. At the lower cross-section, the most dynamic channel migration was observed, with 8 ft lateral loss on the left bank and 6 ft lateral gain on the right bank, demonstrating the sediment transport imbalance in the stream.

The greatest amount of channel erosion occurred as a result of two consecutive storm events (July 29, 2003 and August 8, 2003), with intensities greater than 3 in/hr. Without these two storm events, the amount of streambed and streambank erosion would

have been reduced during the study. A single large event in October 2002 (intensity greater than 3.7 in/hr) had more minor erosive effects on the channel, except in the lower portions of Reach 3. During the study, large flashy flows had the greatest effect on erosion rates and channel stability, particularly when they occurred in sequence.

The channel erosion occurring in North Creek was expected to be a result of the urban watershed, and the continuing watershed development with the associated increase in stormwater runoff and high sediment loads. These factors probably had an effect on channel erosion rates, but to a lesser extent than the large storms. In this watershed, the effects of development on channel erosion may have been reduced due to stormwater retention facilities at each development site, or due to the short study period that would not have captured the effects of stormflow velocity and volume increases. Since streamflow could not be measured before and after development that was continuous during the study, there was no way to isolate what specific effects stormwater retention and stormwater alterations had on channel erosion. This study was also initiated following considerable historical erosion, which was not quantified.

The BEHI assessments demonstrated an increase in bank erosion potential over time in each of the reaches. The majority of streambanks were classified in the Very High or Extreme categories by 2004, which was believed to be a result of increased channel erosion and its effects on the BEHI variables. The streambank erodibility model (Rosgen, 2001) was applied to North Creek. The resulting predicted average streambank erosion rates (ft/yr) were less than measured rates in Reach 1 and Reach 3, but about the same as rates measured in Reach 2. The relationship between near-bank shear stress, BEHI, and

corresponding erosion rates (erodibility chart) was evaluated in North Creek. The NBS index values and erosion rates in each BEHI category were not correlated, but only the higher BEHI categories could be evaluated in North Creek. The relationships may have been better correlated if the full range of BEHI categories were present, similar to the model. When other North Carolina stream data (Patterson 2001) were combined with the North Creek values, the relationships were also not correlated for the higher categories present. The streambank erodibility model might not be suitable for small urban streams such as North Creek, with site-specific watershed and in-stream structure influences. The model may also not be suitable for streams with active streambank erosion in the higher BEHI categories. Additional streambank erosion data are needed in the lower erosion potential categories to test the model for urban streams. The problem is that urban channels rarely have streambanks with low erosion potential due to vertical incision and channel erosion caused by the stormwater runoff. It is suggested that for small urban streams, that predicted streambank erosion rates should be adjusted or that a more complex model should be employed to account for the urban site- specific influences.

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CHAPTER 3: SUBSTRATE ASSESSMENT AND BEDLOAD TRANSPORT

INTRODUCTION

Substrate Assessment

The substrate of a stream is an important property to define because it helps characterize the morphology of a steam, stream behavior, and habitat availability (Rosgen, 1996). The dimension, pattern, and longitudinal profile of a stream are all influenced by the substrate. In meandering channels with riffle/pool sequences, substrate in the streambed tends to be coarsest in riffles and the upstream end of bars due to zones of increased shear stress during high flows (Bunte and Abt, 2001a). The heterogeneity of bed material size is what creates the riffles and pools of the meandering channel (Leopold *et al*., 1992). The type of substrate present also affects the relative amount of vertical incision and corresponding channel erosion.

Stream behavior is controlled by the substrate, including factors such as channel resistance to hydraulic forces from stormflows, and channel sediment transport capabilities (Rosgen, 1996; Bundt and Abt, 2001a). Substrate material comprises the bedload portion of the total sediment load carried by the stream. (The other portions of the sediment load, including washload and suspended load, will be discussed in Chapter 4). Bedload is the sediment moving by saltation, rolling, or sliding in flow layers just a few particle diameters thick above the streambed (Simons and Senturk, 1992; Ward and Trimble, 2004). The substrate particles usually remain in contact with the bed due to their weight relative to the flow velocity (FISRWG, 1998). The bedload proportion is usually only about 5-25% of the total sediment load, but can be the most important portion due to its influence on the

shape stability and hydraulic character of the stream (Dunne and Leopold, 1978; Simons and Senturk, 1992). The distance traveled by the substrate as bedload is dependent on the sediment transport capacity of a given streamflow and the particle size (Haan *et al*., 1994), and usually comprises particle sizes greater than 1.0 mm (Dunne and Leopold, 1978).

The substrate composition and particle size distribution of a stream are dependent on several factors, including the soils and geology surrounding the channel; streamflows; channel gradient; and sediment supply from the watershed and from within the channel (Bundt and Abt, 2001a). Changes in substrate composition over time have been used to detect spatial and temporal trends resulting from watershed activities or disturbances. The substrate of a gravel and cobble-bed stream can be characterized using many different methods, depending on the particular information about the substrate that is desired.

The surface layer can be described using a modified pebble count technique (Wolman, 1954; Rosgen 1993) that measures random particle sizes in different stream features. The pebble count has been used in many water quality monitoring programs (ex. USEPA, MacDonald *et al*., 1991), to compare lotic environments before and after watershed impacts, and to compare one stream to another (Potyondy and Hardy, 1994). The method is currently part of the Rosgen Level II stream classification (Rosgen 1996), which has increased its utilization in stream assessments and restoration.

The surface layer in riffles can also be described using a particle embeddedness method, commonly known as cobble embeddedness (McDonald *et al*., 1991). The method quantitatively measures the extent to which large particles are buried by deposited fine sediments relative to the plane of the streambed (McDonald *et al*., 1991; Bundt and Abt,

2001a). This method is referred to as particle embeddedness in this study, since both gravel and cobble particle sizes are measured. This method has been a common parameter measured by aquatic biologists to assess fine sediment quantities within fish and macroinvertebrate riffle habitats (Luedtke and Brusven, 1976; Burns and Edwards, 1985).

The pavement and subpavement layers within the riffles can be described to determine if fine sediment is accumulating within and below the surface layer, and to determine sediment transport capacity of a stream (Andrews, 1983). In gravel and cobble streams, the pavement is typically coarser than the subpavement due to the selective scouring of fines from the surface during high flows (Bunte and Abt, 2001a). The relative degree of channel armoring by the pavement layer is a good indicator of substrate stability, and the relative sediment supply contributed from the watershed and channel (Bunte and Abt, 2001b; Dietrich *et al*., 1989). The range of particle sizes in the subpavement is usually an indication of what is mobilized during bankfull discharges (Rosgen, 1996).

The measurement of fine sediment intrusion in the pools is another substrate characterization technique that compares the volume of fine sediments within a pool to the total pool volume (Hilton and Lisle 1993). Fine sediments are deposited in pool features, backwater regions, and near obstructions, during low to moderate flows. The relative quantities of fine sediment within pools are affected by sediment supply characteristics and streamflows (Bundt and Abt, 2001a).

Sediment Transport (Bedload)

There are countless numbers of sediment transport equations and methods derived for streams (Haan *et al*., 1994). Graf (1971) describes three general equation types for

bedload transport: DuBoys-type that use shear stress relationships; Schoklitsch-type that use discharge relationships; and Einstein-type that use lift forces and statistical relationships. In this research, shear stress relationships during bankfull flow are utilized to define sediment transport capacity. The critical shear stress (τ_{ci}), a measure of the force required to move and transport a given particle size resting on the streambed (Andrews, 1983), can be determined using ratios of pavement and subpavement particles. This critical shear stress is dimensionless and has been shown to range between 0.02 and 0.28 in gravel and cobble-bed streams that are undisturbed (Andrews, 1983). The critical shear stress can also be used to estimate bankfull mean depths and bankfull water surface slopes required to entrain and transport a particular particle size during bankfull discharges (Rosgen, 1996). Stream competence is the size of the largest particle that a stream can move under a given set of hydraulic conditions, such as those that occur during bankfull flows (FISRWG, 1998). The competence of a stream can be verified using a Shields curve modified with additional data collected since the curve was established (Shields, 1936; Leopold *et al*., 1992; Rosgen, Colorado data). The shear stress that occurs at bankfull flow, a function of flow depth and water surface slope, can then be used to estimate the largest particle that will move (Doll *et al*., 2004).

Substrate Study

The purpose of this study was to describe the substrate in North Creek and Avent Creek using four substrate analysis methods. Pebble counts and pavement/subpavement samples were performed near the beginning and end of the study to evaluate substrate stability and changes in substrate sediment composition. Particle embeddedness and pool

intrusion of fine sediments were measured in each stream to compare the fine sediment accumulation in the riffle and pool features. The purpose of this study was also to calculate preliminary sediment transport relationships during bankfull flows in each stream using the pavement/subpavement data and established equations. The objectives of the North Creek substrate study were:

1. To describe the substrate in the North Creek reaches and in the reference reach, Avent Creek, using four sampling methods: pebble count, pavement/subpavement, particle embeddedness, and pool fine sediment intrusion.

2. To evaluate surface substrate stability and change in North Creek using the pebble count and pavement/subpavement methods.

3. To evaluate the sediment transport capacity of North Creek and Avent Creek using the pavement and subpavement data, and the critical shear stress calculations.

4. To determine if the largest particle size mobilized by the bankfull shear stress as predicted using a modified Shields curve was different than the largest particle size measured in the subpavement samples from North Creek and Avent Creek.

METHODS

Pebble Count

Three reaches of North Creek were visually assessed for relative proportion of channel features, including the amount of riffles, pools, and run areas before substrate analysis. The modified Wolman pebble count (Wolman, 1954; Rosgen, 1993) was performed in October 2002 and January 2004 along the stream reaches. Ten particles were sampled from ten transects in each reach (N=100) that were distributed across channel features in proportion to their presence within the reach. The particles were selected at random, by walking along the transect tape and selecting a particle at boot-tip using a vertical finger. The particles were selected between bankfull stages on each streambank, at estimated even intervals across transects to total ten particles. Particles greater than 2 mm were measured across their intermediate axis using a metric ruler, while particles smaller than 2 mm were measured by rubbing the sample between the fingers and estimating the dominant grain size. Each particle was classified into a size category of the standard Wentworth classification, and tallied for each stream feature (riffles, pools, and runs). A cumulative size distribution curve was derived for each channel feature and for the total pebble count in each reach, showing the percent finer for each size class. The curves were compared between reaches and over time to evaluate temporal and spatial changes in substrate. A pebble count was also performed in the Avent Creek reach in both riffles and pools in October 2002 for comparison to North Creek.

For this substrate analysis, particles less than 2.0 mm (clays, silts, and sands) were considered fine sediments that could intrude into the predominantly gravel-bed stream

bottom or deposit on top of the substrate (Chapman, 1988). For this substrate analysis, saprolite within the streambed and streambank was characterized based on the particle sizes found from crushed aggregates due to the degree of weathering of the material.

Pavement and Subpavement

Pavement and subpavement samples were collected from riffle substrates in October 2002 and January 2004 in North Creek using the collection method outlined in Rosgen (1993), for Rosgen Level II Stream Classification. One riffle was sampled in each reach during the first sampling, and two riffles during the second sampling. Two riffle samples were collected in Avent Creek during October 2002 for comparison.

The samples were collected using a bottomless bucket laid on the substrate within each riffle near the head of riffle where the coarsest portion was located, off center from the thalweg. The particles exposed on the surface were removed as the pavement sample and separated into a sample bag. The intermediate diameter of the largest particle collected in the pavement sample was measured. The subpavement below the surface layer was collected to a depth of twice this pavement particle diameter. The bucket was pushed and shimmied into the substrate until reaching this depth, taking care to remove all particles in the bucket area to the specified depth. The subpavement was collected into separate sample bags and the diameter of the largest particle was again measured within this sample to make sure it was smaller than the largest pavement particle. If the subpavement particles were larger than the largest pavement particle, the sample was discarded and another location was selected within the riffle.

After the particle samples were collected, they were brought back to the lab and emptied onto flat trays for air drying over several days. The pavement and subpavement samples were then sieved using standard metal mesh sieves (USA Standard Testing Sieves, ASTM E-11 Specifications) ranging from 16mm (2.5in) to 0.063mm (0.0025in) following the Wentworth standard size categories. Particles larger than 16 mm were measured along their intermediate axis using a metric ruler. Material in each particle size class was weighed to determine the cumulative particle size distribution curve for the pavement and the subpavement samples.

The average cumulative size distribution curves were compared between pavements and subpavements, between each North Creek reach over time, and between North Creek and Avent Creek. The degree of channeling armoring was estimated by dividing the median particle size of the pavement by the median particle size of the subpavement. The ratios typically range between 1 and 2, with ratios close to 1 indicative of a higher sediment supply to the stream and close to 2 indicative that the sediment transport capacity within the channel is greater than the sediment supply. Ratios less than 1 suggest that the surface is covered by the large sediment supply, and ratios greater than 2 suggest that the surface has had an introduction of large particles (e.g. artificial rip-rap) that are not mobilized effectively by larger flows (Bunte and Abt, 2001a).

Particle Embeddedness

The percent particle embeddedness by fine sediment in riffles was determined in North Creek and Avent Creek in March 2003 using a technique adapted from Skille and King (1989) for cobble embeddedness. The technique was referred to as particle

embeddedness rather than cobble embeddedness due to the gravel substrate in both streams. Eleven riffles were assessed in each stream and chosen randomly along the reaches. The riffles were subjectively designated as well developed, moderately developed, or poorly developed based on internal riffle features including crest differentiation and particle sorting.

Before sampling, the thalweg length of each riffle was measured. A hoola hoop (3.8 sq ft) was laid within the riffles, and 25-35 random particles greater than 6.3 mm were selected for measurement. The particles were grabbed at the plane of the bed between fingertips and removed. The vertical extent above the bed and the total particle height were measured based on the orientation of the particle as it was removed from the streambed. The percent embeddedness was then determined for each sample as the ratio of the embedded vertical height below the bed surface to the total vertical height of the particles within each hoop. The average embeddedness for both the North Creek and Avent Creek stream reaches were determined to compare the substrate between streams and to evaluate the habitat availability within the riffle features.

Pool Fine Sediment Intrusion

The percent volume of fine sediment intrusion into pools was determined in North Creek and Avent Creek in March 2003 using the method adapted from Hilton and Lisle (1993). Eight pools were selected at random in each stream and measured longitudinally from the crest at the end of the upstream riffle to the crest at the head of the next riffle downstream. In North Creek where riffles did not always follow pools, the crest of the next feature was used. The depth to the downstream crest was measured to correct water

surface depths influenced by current flow conditions. The downstream crest depth was assumed to be the residual pool depth, creating a flat surface across the pool for volume determinations.

A grid system was created across the pools using a series of measuring tapes. A measuring tape was stretched through the thalweg of the pool, and another horizontal measuring tape was stretched at a 90° angle across the pool width. Cross-sections along the length of the pools were surveyed at regular intervals, with the number of intervals chosen based on pool length. Cross-section points were also selected at regular intervals based on the length, with perimeter points collected. A steel rebar rod with tenths of feet increments marked was used to measure both depth to the fine sediment surface and depth to the pool bottom from the water surface. The rod was forced into the sediments until it reached the resistant armored layer of the pool bottom.

The survey points for each pool were put into an AutoCAD program to calculate the volumes of the total pool area and the volumes of the fine sediment within the pools. The ratio of fine sediment volume to total pool volume was determined and multiplied by 100 to determine the % fine sediment intrusion. The average percent fine sediment intrusion in the pools of North Creek and Avent Creek were determined for comparison, with variability within each stream evaluated.

Bedload Sediment Transport

The ratios of the median particle sizes from the pavement samples to the median particle sizes of the subpavement samples ($d_{50}/đ_{50}$) were determined in each reach of North Creek and in Avent Creek. The critical dimensionless shear stress (τ^*_{ci}) was determined

using Equation 1 for ratios between 3.0 and 7.0, and using Equation 2 for ratios outside of this range.

Equation 1: $\tau^*_{ci} = 0.0834\ (d_{50} / đ_{50})^{-0.872}$

Equation 2: $\tau^*_{ci} = 0.0384\ (d_i / d_{50})^{-0.887}$; where d_i is the largest particle diameter in the subpavement.

The mean bankfull depth (d_r) and bankfull water surface slope (s_r) required to entrain the largest particle in the subpavement in each reach were calculated using Equation 3 and Equation 4, with all diameters measured in feet:

Equation 3: $d_r = \{1.65(\tau^*_{ci})(d_i)\} / s_e$; where s_e is the existing estimated bankfull water surface slope.

Equation 4: $s_r = \{1.65(\tau^*_{ci})(d_i)\} / d_e$; where d_e is the existing average bankfull mean depth.

These calculated values were compared to existing condition estimated values of bankfull mean depth and water surface slope in order to show whether bankfull flows in North Creek were capable of moving the largest particle in the subpavement. The bankfull water surface slope was estimated as the baseflow water surface slope measured using the longitudinal profile evaluations in each reach of North Creek (see Figure 2.1-2.3) and in Avent Creek (Appendix A3, Figure A3.1). The bankfull mean depth in North Creek was

estimated using the urban regional bankfull hydraulic geometry relationship curves and associated equations for the Piedmont of North Carolina and the drainage area of each reach (Doll *et al*., 2002). The bankfull mean depths estimated by the regional curve were similar to measured depths using field indicators (1.6-2.0 ft) for the corresponding drainage area, but these field indicators were few and not very reliable due to the excessive erosion in North Creek (Dani Wise-Johnson, NCSU Water Quality Group, personal communication and unpublished data). The bankfull mean depth for Avent Creek was identified in the riffle cross-section surveyed (Appendix 3, Figure A3.2), and verified using the rural regional bankfull hydraulic geometry relationship curves and associated equations for North Carolina and the drainage area of the reach (Harman *et al*., 1999).

The stream competence was verified by calculating the bankfull shear stress (τ) using Equation 5 below, and determining the largest particle mobilized within the subpavement sample corresponding to this shear stress in a modified Shields curve (Appendix 3, Figure A3.3).

<u>Equation 5</u>: $\tau = \gamma R S$; where γ is specific weight of water (62.4 lbs/ft^3), R is hydraulic radius of the riffles (ft), and S is average water surface slope (ft/ft).

The average hydraulic radius of each reach was determined using riffle cross-sections derived in AutoCAD (AutoCAD Land Desktop3), from the professional stream survey completed of North Creek in April 2003. The bankfull areas (calculated from the regional curves mentioned above) were fitted into the cross-sections to determine the

wetted perimeter at bankfull depth. The bankfull area divided by the wetted perimeter yielded the hydraulic radius for each cross-section, and an average hydraulic radius was determined for each reach. In Avent Creek, a riffle cross-section was surveyed manually and similar procedures were performed to determine the hydraulic radius (Appendix 3, Figure A3.2). The predicted largest particle diameters were then compared to the measured largest particle diameters in each reach of North Creek and in Avent Creek to determine if bankfull flows were mobilizing these substrate particles.

Statistical Analysis

All statistical analysis was performed using JMP 5, a data analysis statistics package (SAS Institute, 2002). The particle embeddedness data and % pool intrusion data were compared between the two streams using box and whisker plots, and the t statistic for mean comparison, assuming unequal variance.

RESULTS AND DISCUSSION

Pebble Counts

The median particle sizes for the cumulative pebble counts and pebble counts performed in each feature of the North Creek reaches are shown in Table 3.1. The riffle d_{50} was a good indication of the stream substrate type, with the sizes in Reach 1 and Reach 2 indicating that North Creek is a fine to medium gravel-bed stream (4.0-16.0 mm). Reach 3 was not considered for overall substrate description because of the artificial large cobble and small boulder lining in the upper portion and the lack of pebble count data in the lower portion. Avent Creek was a fine gravel-bed stream (4.0-8.0 mm) based on the riffle pebble count.

The cumulative particle size distribution curves for the two complete pebble count surveys performed in each reach of North Creek are shown in the following figures. Reach 1 had a higher proportion of fine particle sizes (<2.0 mm) in the second survey, although the remaining larger particles were coarser over time (Figure 3.1). The proportion of fine sediments in Reach 1 was 62% in the first survey followed by 77% in the second survey. Reach 2 showed the same pattern as Reach 1, with 60% of the particle size classes considered fine sediment, followed by 66% in the second survey (Figure 3.2). Reach 3 had a much finer substrate in the second cumulative survey, with 15% of the particles less than 2.0 mm in the first survey followed by 42.5% in the second survey (Figure 3.3). The median particle size class for the cumulative pebble counts declined in each reach of North Creek over time (Table 3.1).

The cumulative pebble count in the reference reach, Avent Creek, indicated that 48% of the particle sizes were considered fine sediments (Figure 3.4), which is less than Reach 1 and Reach 2 but not Reach 3 of North Creek. The median particle size class was 2.0 mm, the boundary between sand (fine) and gravel (coarse) sediments (Table 3.1).

The pebble count results for the major stream features, including riffle, pool, and run transects, are plotted to further compare the reaches (Figure 3.5-3.7). There was a general pattern of declining particle size in substrate features based on the pebble counts in each reach over time. In Reach 1, the d_{50} of the riffles, pools, and runs all declined between surveys (Table 3.1). The cumulative particle size distribution curves for the features indicated that all features in Reach 1 had a smaller proportion of fine sediments (<2.0mm) in the first survey (riffle:20%, pool:70%, run 75%) compared to the second survey (riffle:53%, pool:90%, run:86%), with the greatest difference in the riffle fine particle compositions between surveys (Figure 3.5). The cumulative particle distribution curves for the features indicate that all features were finer during the first survey compared to the second survey for particles larger than fine gravel (Figure 3.6). All features in Reach 2 had a smaller proportion of fine sediments (<2.0mm) in the first survey (riffle:40%, pool:73%, run 65%) compared to the second survey (riffle:43%, pool:80%, run:80%), with the most difference in the run features of this reach. In Reach 3, the d_{50} of the riffles and pools decreased considerably between the first and second survey, with run features added during the second survey that were not present in the profile at the beginning of the study (Table 3.1). The riffles and pools during the first survey resembled more of a step/pool type sequence, artificially created without the typical steep streambed slope associated.

The cumulative particle distribution curves show that all features were generally finer during the second survey compared to the first survey (Figure 3.7). The pool features had a smaller proportion of fine sediments (<2.0mm), but the riffle features had an increase in fine sediments between the first survey (riffle:20%, pool:10%) and the second survey (riffle:0%, pool:90%).

Comparing stream features within the reaches, the pools tended to have the largest amounts of fine sediments followed by the runs then the riffles, as expected. Avent Creek also had a larger amount of fine sediments in the pools (52%) compared to the riffles (44%), with differences between the amounts of fine sediments in these two features smaller compared to the reaches of North Creek (Figure 3.4).

The differences in the pebble count surveys performed indicated that more fine sediment was present in the stream features during the second survey compared to the first survey. The results of the study could not evaluate whether these changes were caused by an increased sediment load from the watershed due to several factors associated with the sampling. There were only two pebble count sampling dates available for comparison in each reach, with additional surveys required to establish a trend. Survey locations and feature proportions were also different between sampling dates, because these surveys were performed by different surveyors independently. Another variable that could influence the comparison was the preceding storm events that could affect fine sediment distribution in stream features. There was comparatively little rainfall in North Creek in the weeks prior to sampling during each survey, but a large storm occurred in mid-October (4.2 in) between the North Creek and Avent Creek sampling dates that could have affected

the fine sediment distribution in Avent Creek. The median particle size comparisons between the two streams were inconclusive. These results may have been controlled by differences in sediment load, but differences in soils, geology, feature proportions and water chemistry could have also played a role (Horton and Zullo, 1991; Bundt and Abt, 2001a). Additional survey data were required in Avent Creek to show changes over time for comparison to North Creek, along with additional information on site variables that could have influenced the substrate particle size distributions.

In North Creek, the largest change in surface substrate occurred in Reach 3, but these changes appeared to be site-specific compared to the upper reaches. Erosion of the artificial stone lining from a rapid headcut migration in July and August 2003 exposed the remnant channel bed in several areas. Further surveys are needed in this portion of Reach 3 to measure changes in the newly exposed substrate.

Table 3.1: Comparison of Cumulative and Feature Pebble Counts using the Median Particle Size Class (mm).

	Reach 1		Reach 2		Reach 3		Reference
	10/02	*1/04*	*10/02*	*1/04*	*10/02*	*1/04*	*10/02*
Cumulative d_{50}	0.5	0.062	0.5	0.125	128	8.0	2.0
Riffle d_{50}	11.3	1.0	5.7	11.3	128	90	4.0
Pool d_{50}	0.125	0.062	0.25	0.125	90	0.125	0.125
Run d_{50}	0.5	0.062	0.5	<0.62	NA	5.7	NA

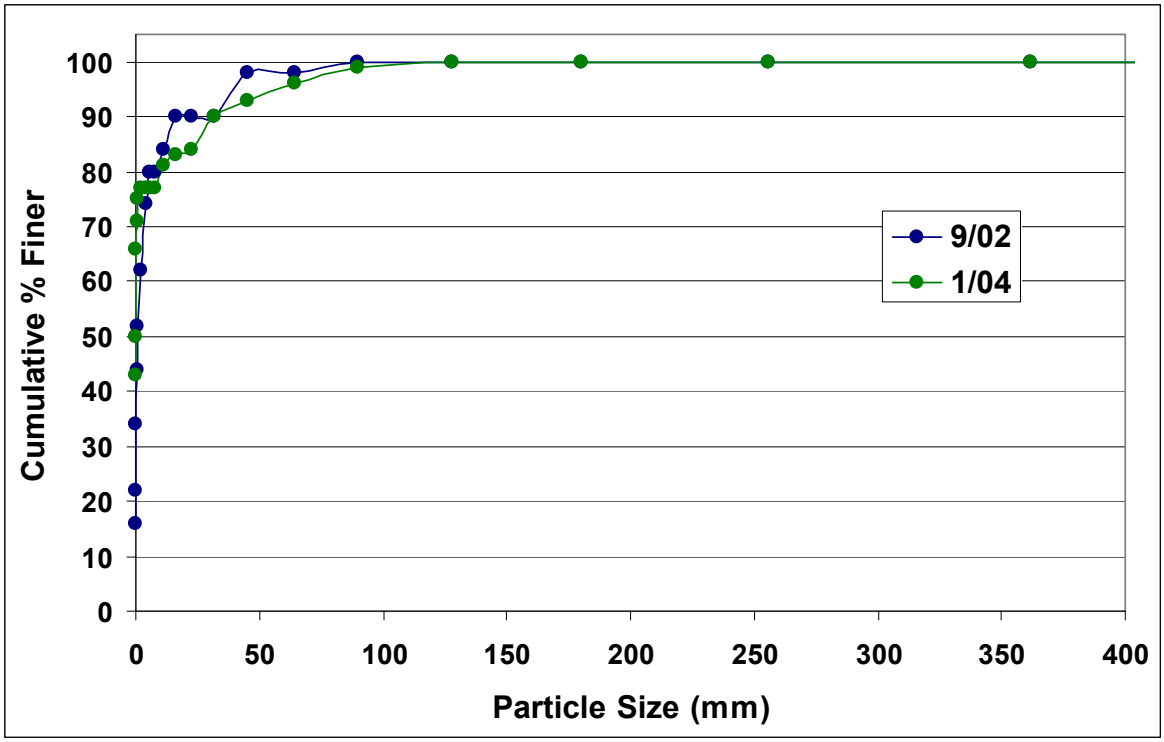

Figure 3.1: Particle Size Distributions for Cumulative Pebble Counts for Reach 1, North Creek.

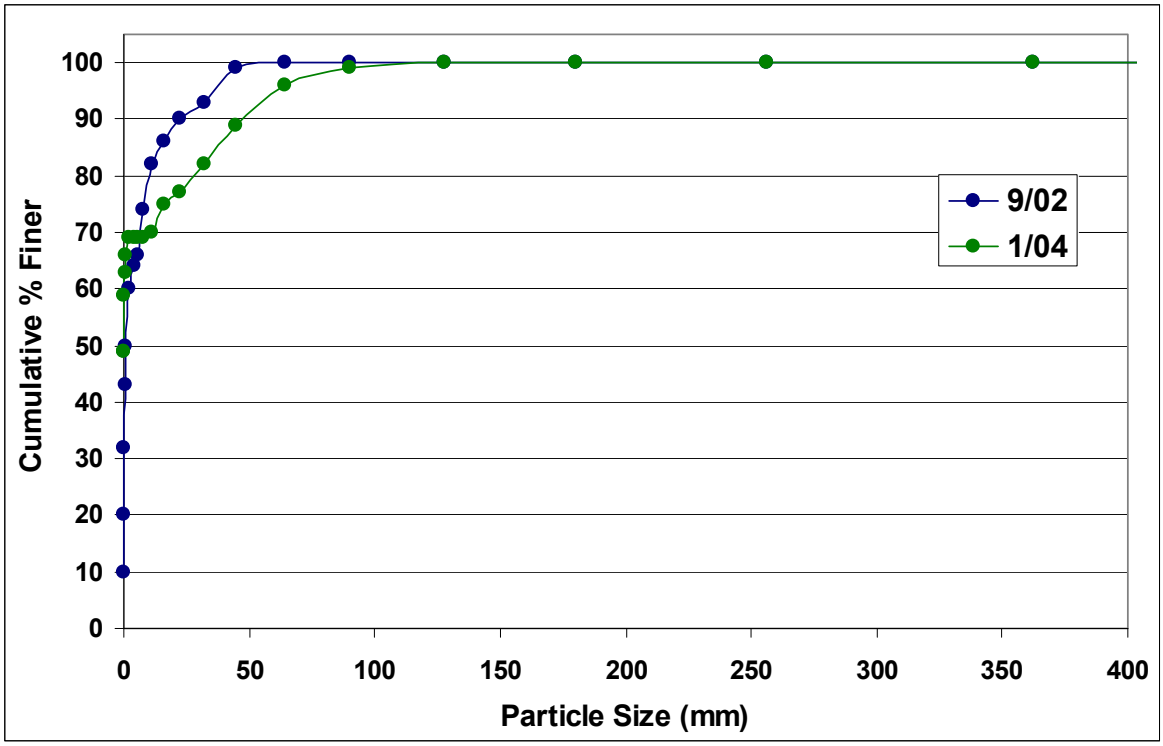

Figure 3.2: Particle Size Distributions for Cumulative Pebble Counts for Reach 2, North Creek.

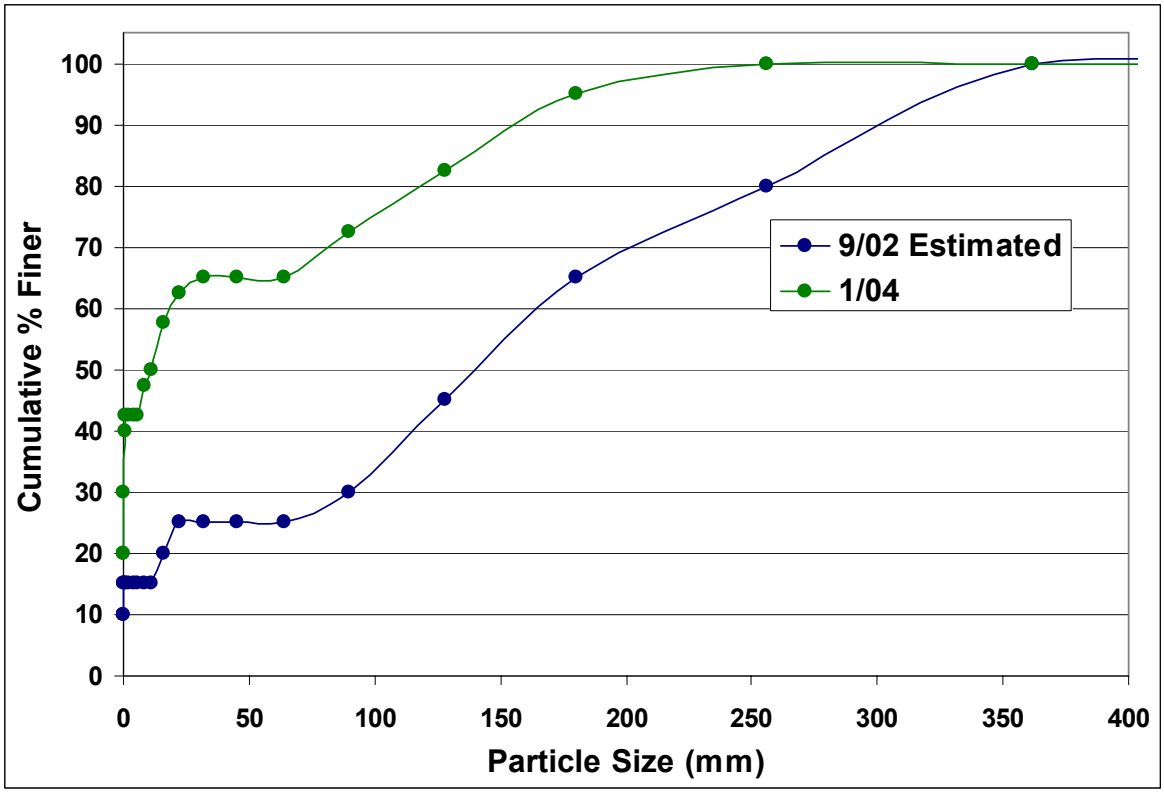

Figure 3.3: Particle Size Distributions for Cumulative Pebble Counts for Reach 3, North Creek.

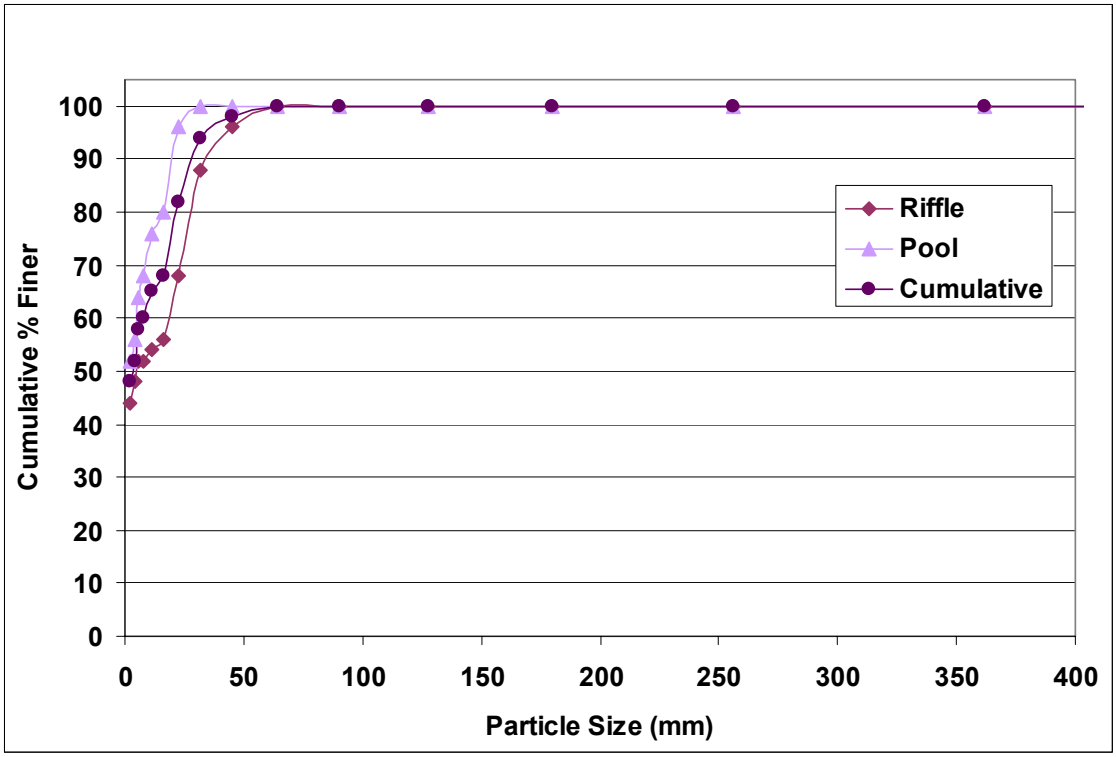

Figure 3.4: Cumulative and Feature Pebble Counts for the Reference Reach, Avent Creek.

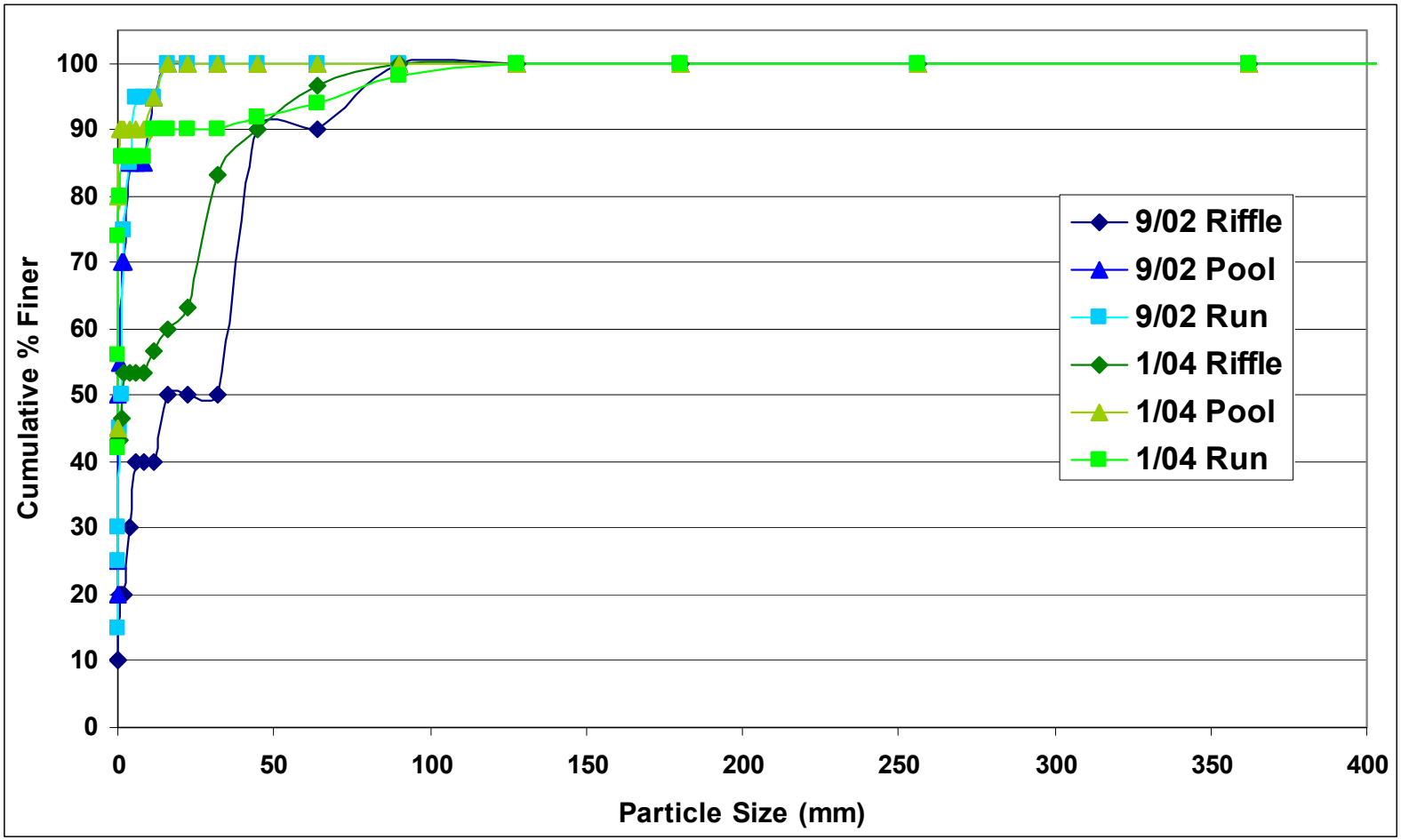

Figure 3.5: Particle Size Distributions for Feature Pebble Counts in Reach 1, North Creek.

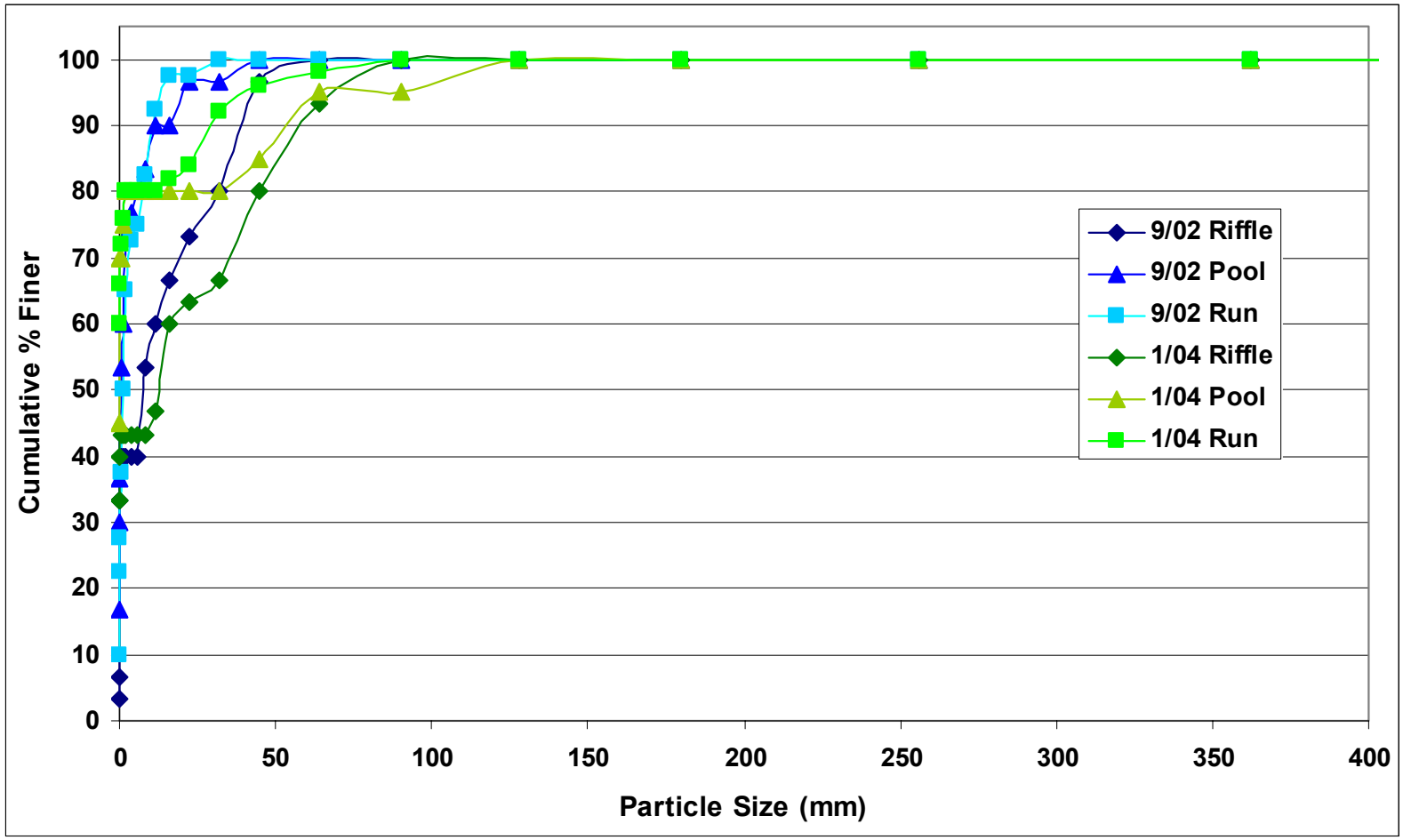

Figure 3.6: Particle Size Distributions for Feature Pebble Counts in Reach 2, North Creek.

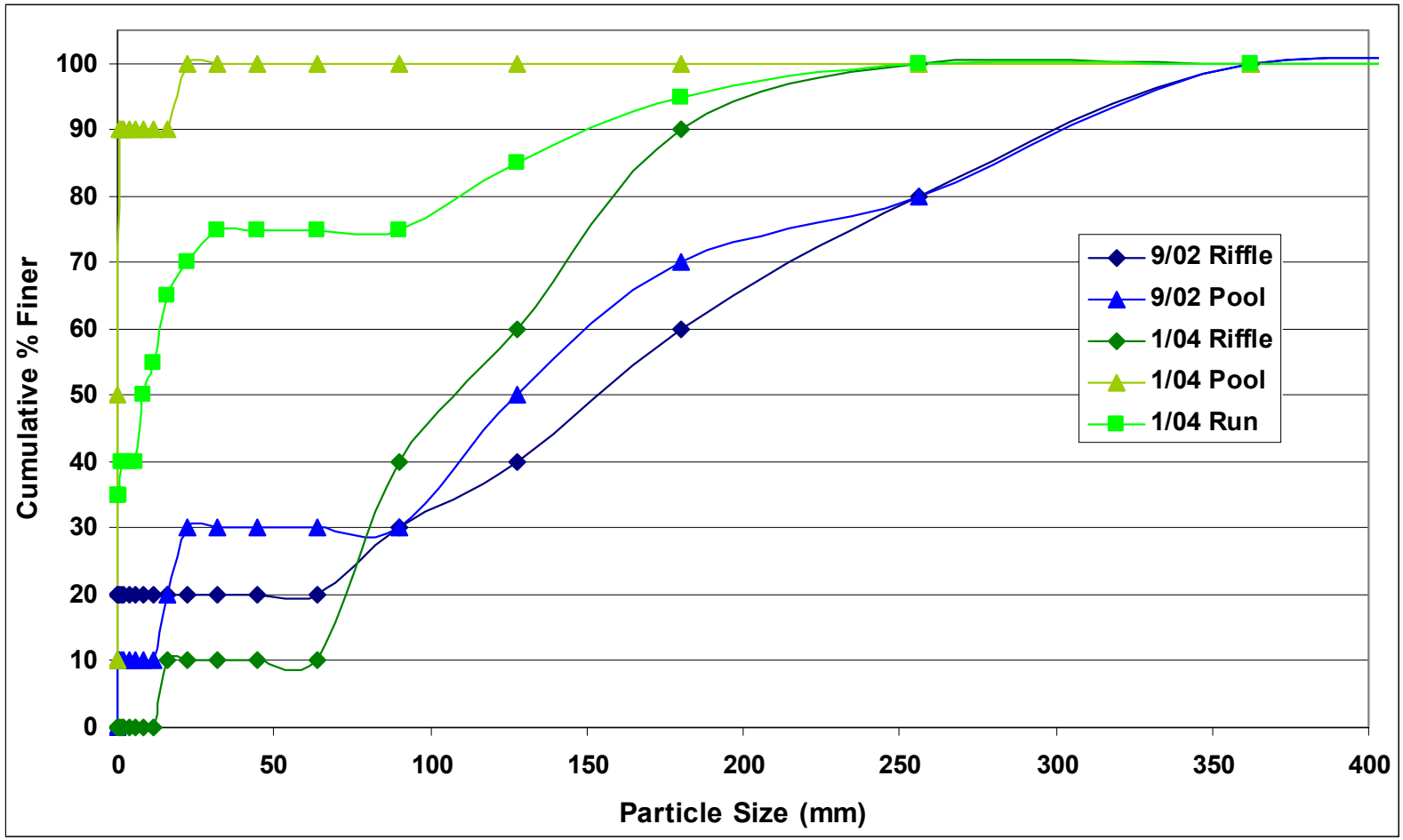

Figure 3.7: Particle Size Distributions for Feature Pebble Counts in the Upper Portion of Reach 3, North Creek.

Pavement/Subpavement

Pavement particle sizes distributions tend to be coarser than subpavement particle size distributions in gravel and cobble-bed streams due to the selective scouring of fines from the surface during high flows (Bundt and Abt, 2001a). In both North Creek (Figure 3.8-3.10) and Avent Creek (Figure 3.11), the pavement samples were consistently coarser than the subpavement samples in the cumulative particle size distributions. The median particle sizes were all larger compared to the subpavement samples, while the percent fine sediment was consistently lower in the pavement samples (Table 3.2). In Avent Creek, the d_{50} of both the pavement and subpavement were similar to or smaller than the d_{50} of the samples in North Creek (October sampling date). The percent fine sediment was smaller in the pavement samples compared to all North Creek reaches, and was larger in the subpavement samples except for in Reach 2 (Table 3.2).

In Reach 1, the pavement had a larger d_{50} during the first survey (Figure 3.8). The percent fine sediment within the Reach 1 pavement remained constantly low during both surveys (Table 3.2). The subpavement material was similar for particles smaller than coarse gravel, but appeared to have a larger proportion of coarser particles above this size in the first survey (Figure 3.8). The median particle size class remained constant, but the percent fine sediment increased (Table 3.2). In Reach 2, the pavement was coarser in the second survey for particle sizes smaller than small cobble, while the subpavement was coarser for all size classes (Figure 3.9). The d_{50} declined between surveys for the pavement but increased in the subpavement. The percent fine sediment declined between surveys for both sample layers (Table 3.2). In Reach 3, both the pavement and

subpavement samples were consistently finer (Figure 3.10), with a d_{50} smaller for both sample types. There were no fine sediments collected during both surveys in the pavement samples, while the absence of fine sediments in the subpavement was only increased slightly (Table 3.2).

The overall results of the pavement sampling did not indicate fine sediment accumulation. The median particle size of the pavement samples was reduced in each reach, but the percent fine sediment was either constant (Reach 1 and Reach 3) or declined (Reach 2). The subpavement results also indicated that fine sediment accumulation was not consistently occurring in the North Creek reaches. Fine sediment accumulation in the subpavement was only fully supported in Reach 3, where extensive vertical incision exposed portions of the subpavement that probably became the pavement in the following samples. The lack of fine sediment accumulation in the North Creek subpavement could be due to the scouring effects of the larger stormflows extending into this layer.

The October sampling date comparison between pavement/subpavement samples in North Creek (Reach 1 and Reach 2) and Avent Creek was expected to show a smaller median particle size and larger percent fine sediment within the cumulative particle size distributions in the North Creek riffles. The particle sizes in both the pavement and subpavement samples were generally smaller in Avent Creek compared to North Creek. Differences could have been due to the elevated stormflows occurring over time in North Creek that might have selectively scoured the smaller particles from the pavement compared to Avent Creek. The artificial stone influence in North Creek may also have affected the comparison. As discussed in the pebble count section, the Avent Creek data

could not be used to determine fine sediment effects in North Creek without additional samples over time and more information of the different site variables between the two streams.

Similar to the pebble count surveys, the pavement and subpavement samples were few and not collected from the same riffles between sampling dates. Different locations were utilized to avoid any disturbance from the previous sample collection influencing the next. The riffles were relatively small and not well defined in North Creek, so the potential for substrate disturbance with sampling was increased. Pavement and subpavement changes should be further evaluated with a larger number of samples in each reach and in the same locations to order to determine if fine sediment accumulation is not occurring. Similar amounts of precipitation occurred in the previous weeks before each sampling date. Sampling should be conducted after different precipitation patterns in order to assess variable stormflow influences on the pavement and subpavement.

Table 3.2: The Median Particle Size Class (mm), The Largest Particle Size Diameter (mm), and the Percent of Fine Sediments (< 2.0mm) in the Pavement and Subpavement Samples Over Time.

	Reach 1		Reach 2		Reach 3		Reference
	10/02	*1/04*	*10/02*	*1/04*	*10/02*	*1/04*	*10/02*
Pavement d_{50}	64	45	90	64	256	128	32
Subpavement d_{50}	16	16	6.3	16	180	64	6.3
Pavement d_{100}	85	110	110	110	256	225	68
Subpavement d_{100}	80	55	82	92	180	105	54
Pavement % Fine Sediment	0.8	0.8	3	0.2	0	0	0.3
Subpavement % Fine Sediment	19	23	26	8	0	3	22

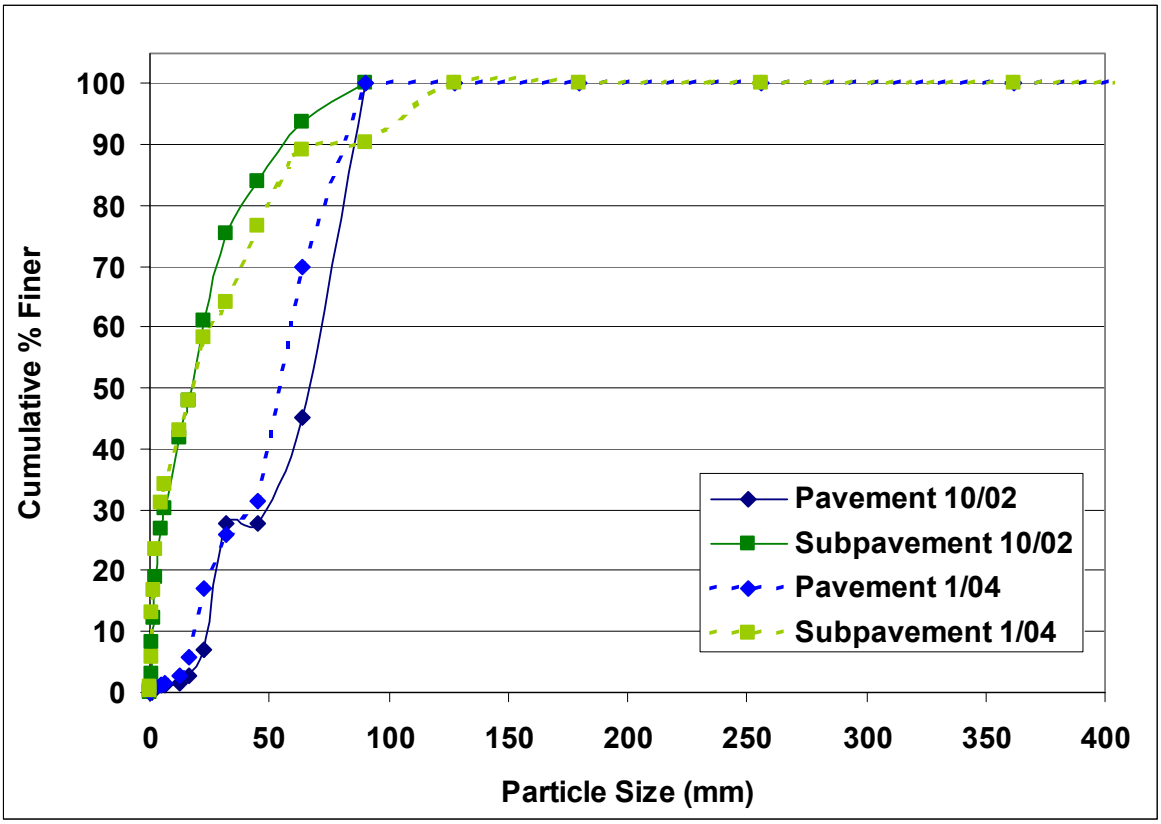

Figure 3.8: Pavement and Subpavement Particle Size Distributions for Reach 1, North Creek.

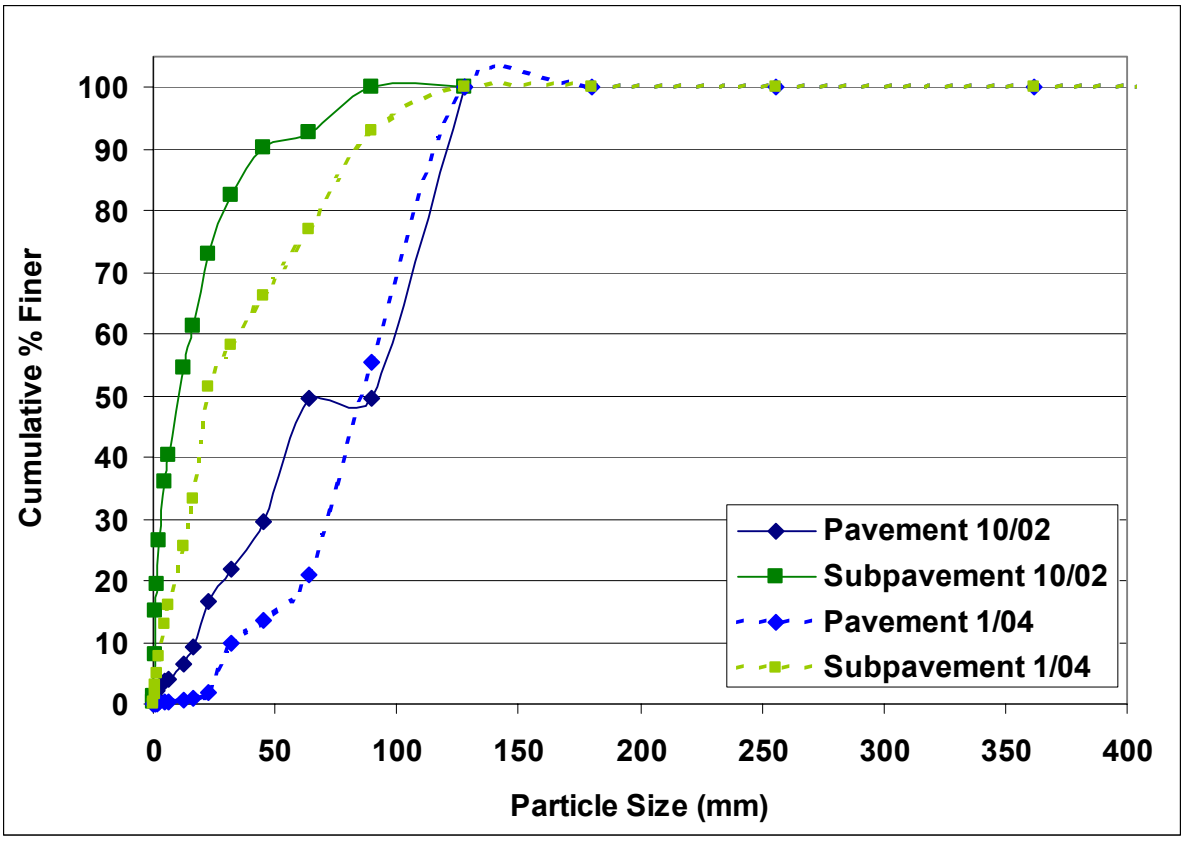

Figure 3.9: Pavement and Subpavement Particle Size Distributions for Reach 2, North Creek.

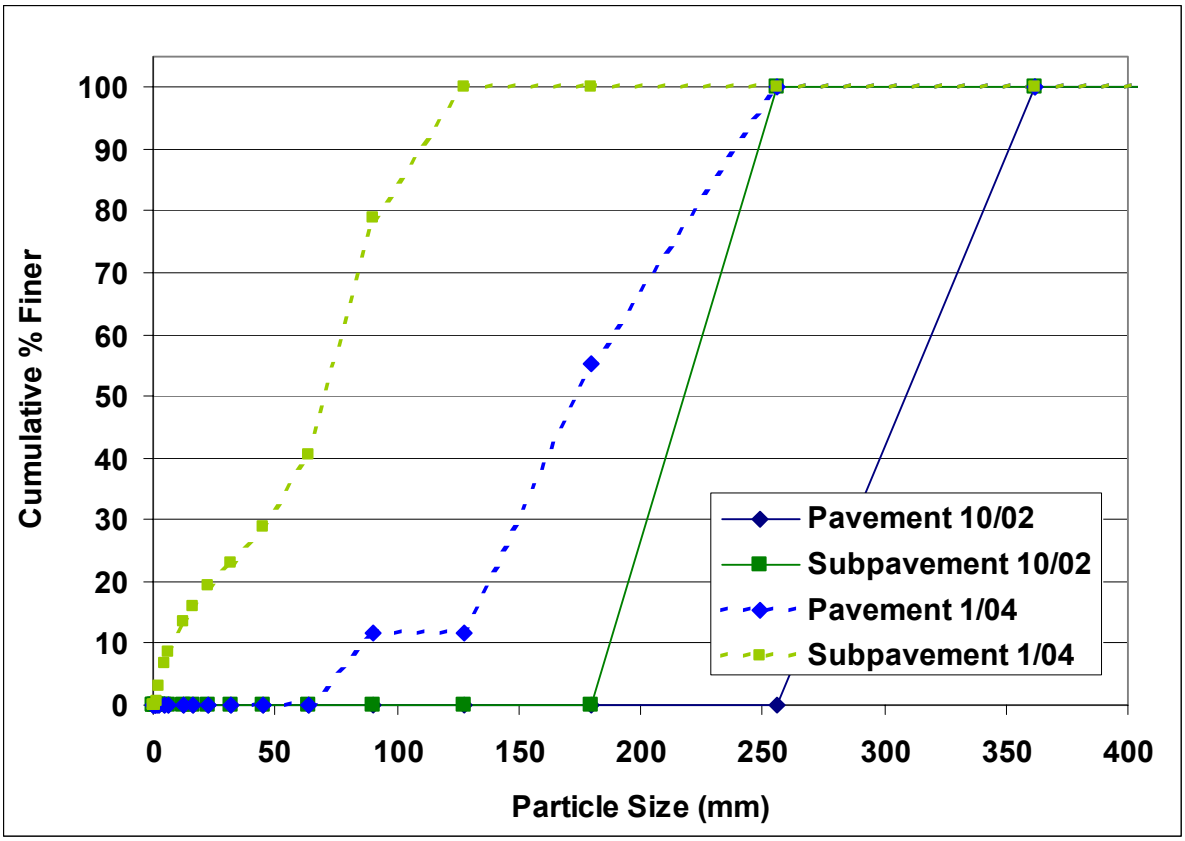

Figure 3.10: Pavement and Subpavement Particle Size Distributions for Reach 3, North Creek.

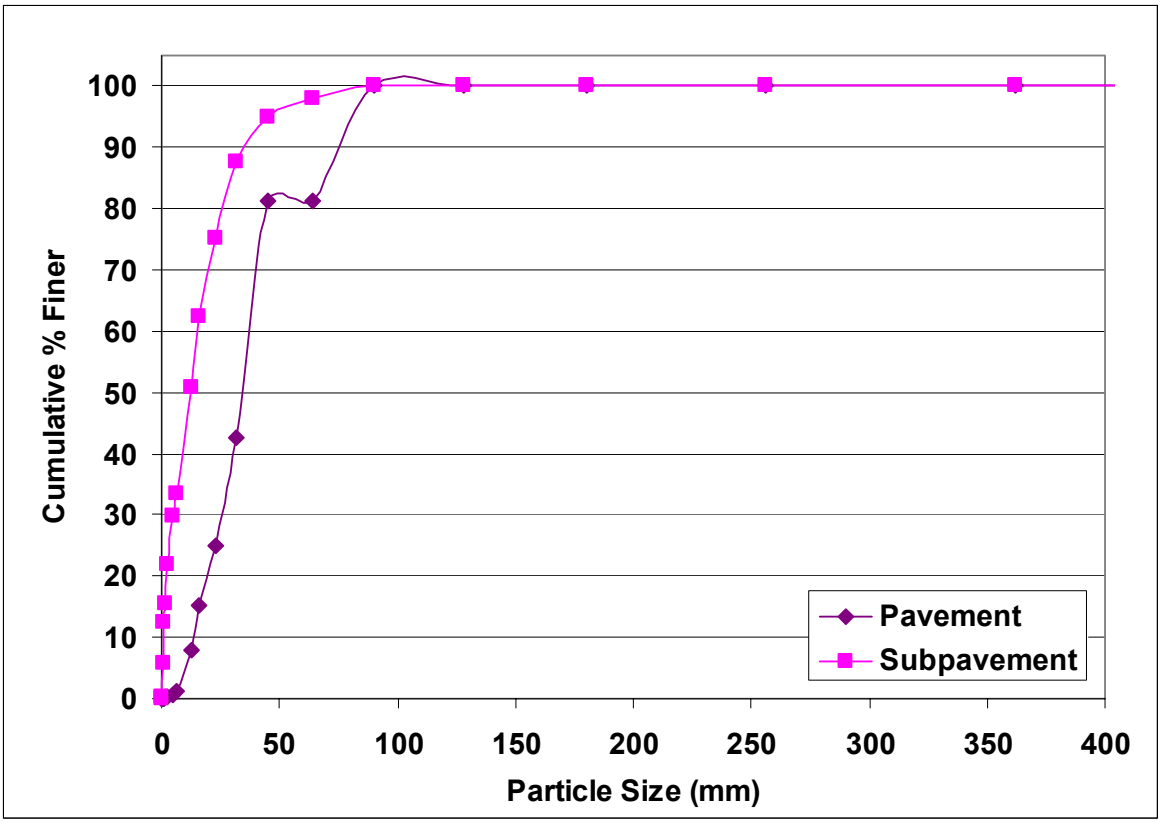

Figure 3.11: Pavement and Subpavement Particle Size Distributions for the Reference Reach, Avent Creek.

Particle Embeddedness

Particle embeddedness is a method used to compare riffle substrate habitat between streams, and can indicate fine sediment load differences between streams (McDonald *et al*., 1991). Eleven riffles with various channel lengths were measured in both North Creek and Avent Creek in the same week of March 2003, the week following a large storm event (1.5 in). The particles measured in both streams were greater than 6.3 mm in diameter, with a relatively even distribution between medium gravel and cobble sizes. The average percent of fine sediment embeddedness of cobble and gravel particles was significantly

greater in the North Creek riffles (24.8%) compared to the Avent Creek riffles (8.9%), based on the t-test statistic ($p \leq 0.05$). North Creek had greater variability in % particle embeddedness, with median values (29.5%, North Creek; 7.5%, Avent Creek) also indicating more embeddedness in North Creek (Figure 3.12).

The greater percent embeddedness in North Creek indicated that there was more fine sediment present between the large particles in the North Creek riffles compared to the Avent Creek riffles. This fine sediment was probably settled during stormflow recession in both streams. The results suggested that there was a greater fine sediment load during stormflows in North Creek compared to Avent Creek, which was expected to be derived from watershed activities and channel erosion. The average particle embeddedness in North Creek was only about 25%, which is lower than the embeddedness values reported in the literature that affected fish and macroinvertebrates (Luedtke and Brusven, 1976; Chapman and McLeod, 1987). The effects of particle embeddedness on aquatic organisms have been shown to be variable, however, depending on the species present (McDonald *et al*., 1991). Higher particle embeddedness proportions could have been measured in North Creek without the influence of increased stormflow velocities derived from the urban watershed.

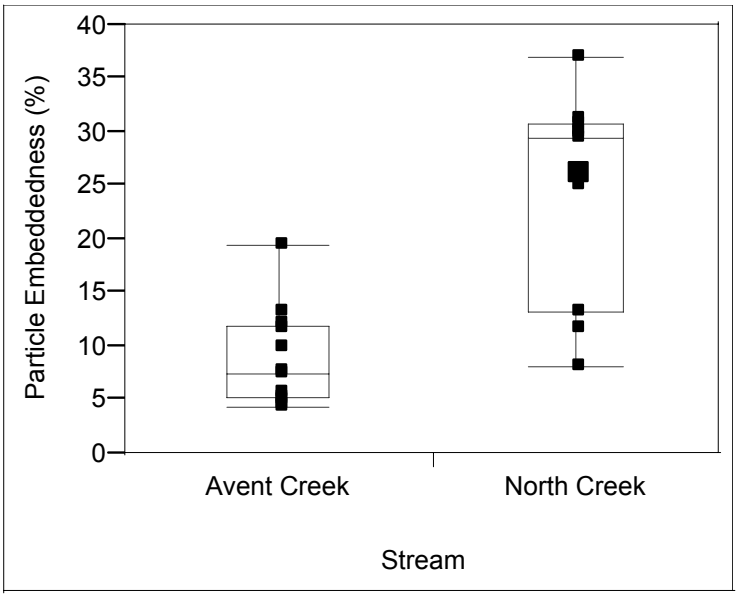

Figure 3.12: Box and Whisker Plot of % Particle Embeddedness in the Riffles of North Creek and Avent Creek, Showing Minimum and Maximum Values, with Bars Indicating 10% and 90% Quantiles, and Box Delineating Median Value, 25% and 75% Quantiles.

Pool Fine Sediment Intrusion

The percent fine sediment intrusion into pools can be used to evaluate patterns of fine sediment deposition and to compare the fine sediment load between streams (Lisle and Hilton, 1999). The finest suspended materials, including silts and clays, tend to settle out first within the backwater areas and pools during flow recession and during moderate to low flows (Bunte and Abt, 2001a). Eight pools were evaluated in both North Creek and Avent Creek in March 2003. The average pool size in North Creek was about 2.7 ft^2, while in Avent Creek the average size was about 2.2 ft^2. The average percent of fine sediment intrusion was 37.2% into the North Creek pools and 25.5% into the Avent Creek pools based on pool and sediment volumes. The samples in each stream were not

significantly different, however (t-test statistic, $p \geq 0.05$), with median values similar between the two streams (24.3%, North Creek; 22.3%, Avent Creek). The variability in North Creek was much higher than in Avent Creek (Figure 3.13).

The percent fine intrusion into North Creek pools was expected to be greater than the fine sediment in Avent Creek pools due to a greater fine sediment load in North Creek. The mean % pool intrusion was not significantly greater in North Creek, which might suggest that the fine sediment load was not higher in this stream. Observations and sediment measurements during stormflows indicated, however, that the fine sediment load was increased (See Chapter 4). The reason for lower % fine sediment intrusion than expected along with the greater variability in North Creek was probably the flashy stormflows and the unstable stream conditions. The elevated stormflow velocities derived from the urban watershed could have repeatedly scoured the pools, with disproportionate deposition as the unstable channel bed was shifted during the larger flow events.

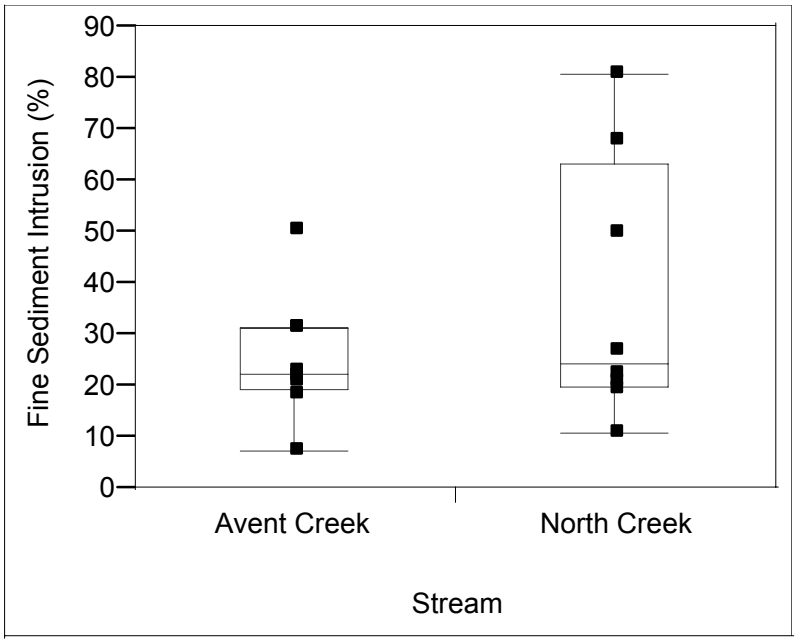

Figure 3.13: Box and Whisker Plot of % Fine Sediment Intrusion in the Pools of North Creek and Avent Creek, Showing Minimum and Maximum Values, Bars Indicating 10% and 90% Quantiles, and Box Delineating Median Value, 25% and 75% Quantiles.

Substrate Methods Comparison

Although substrate analysis was difficult to interpret due to sample number and location variances, the use of four different substrate analysis methods during this study provided more information about the substrate character than each method alone. The surface substrate was compared using all of the methods, and the data collectively supported increased sediment deposition. There was only one method used to evaluate the subpavement; so results could not be verified in the study.

Pavement samples were collected towards the middle of the riffle, and probably biased towards the larger (some artificial) stones in the riffles compared to the bankfull

cross-section evaluated by the pebble count. The fine sediments surrounding the lower regions of these particles (as indicated by particle embeddedness) would probably be left behind as subpavement when the larger surface particles were removed. Fine sediments were difficult to collect with a large particle surface layer. For urban streams such as North Creek with heterogeneous riffle particle sizes influenced by artificial large substrates and fine sediment deposition, surface layer substrate analysis methods need to account for the large size distribution. Particle embeddedness and pebble counts that evaluate a larger riffle area and allow better sampling of the finer sediments could be more effective.

The differences between the pool pebble counts and the % pool intrusion were probably related to sample depth. The pebble count evaluated the increase in fine sediment on the surface, most likely deposited during flow recession from the previous storm. The % fine intrusion evaluated the volume of sediment within the pools, which was affected by the elevated stormflows in this urban stream. The sediments within the pools could have been increasing in fine particle sizes due to the fine sediment load carried by the stormflows, but not increasing in fine sediment volume due to scour by the stormflows. In an urban stream, substrate assessments should account for the increased stormflow velocities as well as the increased sediment loads.

Bedload Sediment Transport

The variables used in the sediment transport equations and the results from these evaluations are provided in Table 3.3. The ratio of the median particle size of the pavement to the median particle size of the subpavement was used to determine the degree of channel armoring (Bunte and Abt, 2001a). The ratios of the median particle size in the

pavement to the subpavement were highest during the first Reach 2 sampling and relatively low for Reach 3 during both samplings. In natural streams, the ratios typically range between 1 and 2 with values close to 1 indicative of a high sediment supply and values close to 2 indicative of a sediment transport capacity greater than the sediment supply. For higher ratios greater than 2, the surface has most likely had an introduction of large particles (ex. artificial rip-rap) that are not mobilized effectively by larger flows. For both North Creek and Avent Creek, the ratio was greater than 2 except for the earlier sample in Reach 3 (1.4) where the ratio indicated a relatively even balance between sediment supply and sediment transport capacity. The results support the influence of artificial substrate added in Reach 1 and Reach 2 during large stormflows from amended streambanks and stormwater outfalls. The results in the upper portion of Reach 3 did not indicate channel armoring in this reach, even though the reach was artificially lined. The reason was probably due to the depth of the artificial substrate, which extended below the subpavement depth and caused the ratio to be deceptively low. The elevated ratio in Avent Creek could not be explained, however, since this stream had no evidence of artificial substrate influence.

Bedload sediment transport was estimated for bankfull flow since this discharge is typically thought to be the channel forming discharge where the largest substrate particles are initially entrained for movement (Rosgen, 1996). Critical shear stress at bankfull was calculated using the ratio of pavement to subpavement median particle sizes (ratio between 3 and 7), or using the ratio of the largest subpavement particle to the median pavement particle (ratio<3, ratio>7). The critical shear stress was relatively low except for during the

first sampling of Reach 2 and during both samplings of Reach 3, due to the elevated ratio between the largest subpavement particle and the median pavement particle required in the equation. The critical shear stress remained relatively constant over time in North Creek, except for the elevated value in Reach 2 during the first sampling.

The bankfull critical shear stress estimates were used to determine the required bankfull water surface slope and bankfull mean depth in the riffles needed to move the largest particle of the subpavement (Andrews 1983; Doll *et al*., 2004). Water surface slope measured using the longitudinal profile in each reach was comparably low in Reach 1, Reach 2, and in Avent Creek, but greater in Reach 3 due to the greater channel bed slope in the upper portion. The required slope for bedload transport at bankfull flow was lower for all reaches except Reach 3 during the first sampling when the artificial stone layer was still intact. The mean bankfull depth required for bedload transport was lower than the estimated bankfull mean depth in each reach, except during the first sampling in Reach 2 and Reach 3.

The required values were expected to be greater than the estimated values in North Creek due to the influence of artificial substrate added along the channel. The artificial substrate was expected to increase the largest particle size of the surface and subsurface layers compared to the natural channel bed material typically entrained during bankfull flows. These results were observed in Reach 2 and Reach 3 using the first substrate sampling data. The remainder of the data showed, however, that the required bankfull water surface slope and bankfull mean depth calculated were less than estimated values. The variability in the results suggested that the artificial substrate could have been

influential in some reaches, but not throughout the entire stream. In portions of the stream where particle ratios were smaller, the results suggested that the substrate is mobilized more frequently in this urban channel than bankfull recurrence intervals would predict.

In Avent Creek, the differences between predicted bankfull water surfaces slopes and bankfull mean depths and the measured values in Avent Creek were expected to be relatively small due to the natural substrate and the fact that the bankfull elevation was near the top of streambank. The results showed, however, that the predicted bankfull water surface slope and bankfull mean depth were less than the measured values. These results could not be explained using the substrate data collected. For both North Creek and Avent Creek, the limited number of pavement and subpavement samples could have affected the results due to high variability in substrate, particularly in North Creek. The substrate sampling should capture the variability in substrate particle size, especially when using these variables for design purposes.

The stream competence in both North Creek and Avent Creek was verified using a modified Shields curve to predict the largest particle of the subpavement mobilized during bankfull flow. This curve utilizes bankfull shear stress estimates to predict the largest subpavement particle size mobilized by the bankfull discharge (Shields, 1936; Doll *et al*., 2004). The bankfull shear stress calculated for each reach in North Creek and in Avent Creek was comparable, except for in Reach 3 where the slope and hydraulic radius were both larger. The bankfull shear stress was also relatively constant over time in North Creek due to the small changes in slope between sampling dates, and due to the use of average hydraulic radius values for lack of detailed riffle surveys on sampling dates. The

predicted largest particle of the subpavement mobilized by the bankfull shear stress was smaller than the observed largest particle size averaged between samples in all reaches except Reach 3. The bankfull shear stress available in this reach was capable of moving particles 3-4 times the measured sizes in the subpavement.

The largest subpavement particle size predicted was expected to be smaller than the measured largest particle size in North Creek due to the artificial substrate influence. The results were as expected in Reach 3 with the artificial substrate influences and steep slope, but opposite in Reach 1 and Reach 2. The bankfull shear stress was capable of moving the largest subpavement particles in these two reaches, suggesting that the artificial substrate added was not influential on the bedload sediment transport capacity, and that the substrate was mobilized by stormflows at higher frequencies than the bankfull events.

In Avent Creek, the predicted and measured largest particle sizes were expected to be similar due to bank height to bankfull height ratio near 1. The predicted particle size was less than the observed subpavement largest particle size, which could not be easily explained considering the stream characteristics. For all sediment transport evaluations, the largest subpavement particle sizes collected in Avent Creek caused the deviations from expected bankfull sediment transport performance. Increasing the number of samples might affect the sediment transport estimates in this stream. Another problem with using these sediment transport equations in Avent Creek may be the substrate type. The median particle size was a fine gravel (2.0 mm), which is the borderline between a gravel and fine sediment bed channel. The equations are derived for natural cobble and gravel-bed streams, and may not have been suitable in Avent Creek. All other substrate and channel

dimension parameters assessed indicated that this reference stream was stable, even though the sediment transport evaluation in this study did not fully support this.

Table 3.3: Sediment Transport Variables and Calculation Results for North Creek and Avent Creek.

	Reach 1		Reach 2		Reach 3		Avent Creek
	10/02	1/04	10/02	1/04	10/02	1/04	10/02
Ratio d50/đ50	3.19	3.89	8.49	3.90	1.42	2.57	2.83
τ^*_{ci}	0.014	0.012	0.065	0.012	0.062	0.062	0.025
Channel Slope (ft/ft)	0.0041	0.0042	0.0066	0.0064	0.0223	0.0209	0.007
Mean Bankfull Depth (ft)	1.64	1.64	1.80	1.80	1.97	1.97	1.62
Required Slope (ft/ft)	0.0037	0.0019	0.0100	0.0028	0.0307	0.0179	0.0046
Required Bankfull Depth (ft)	1.48	0.76	2.74	0.80	2.71	1.68	1.06
Average Hydraulic Radius (ft)	1.8		1.8		2.2		1.2
Bankfull Shear Stress (lb/ft^2)	0.46	0.47	0.74	0.72	3.06	2.87	0.52
Measured Average Largest Particle Diameter Mobilized (mm)	80	50	66	81	180	105	54
Predicted Largest Particle Diameter Mobilized (mm)	28	30	50	48	475	450	32

CONCLUSIONS

The substrate analysis in North Creek provided information about the morphology of the channel, the habitat condition, and the hydraulic and sediment transport characteristics. Evaluating the substrate during this study provided information on the potential effects of development in the watershed, the effects of artificial substrate added to the streambanks and streambed, and the effects of channel erosion observed within the stream.

The overall results of the four substrate evaluation methods collectively provided evidence of elevated fine sediment loads in North Creek and their effects on the substrate. Pebble count surveys and particle embeddedness measurements showed that fine sediment accumulated within the riffles. Pebble count surveys showed that fine sediment had accumulated within the pools. Pavement and subpavement samples and % pool fine sediment intrusion did not substantiate an increased fine sediment load in North Creek, however. The comparisons of substrate samples were difficult to compare in this study based on the limited number of samples collected on each sampling date and over the study period. The short-term precipitation events prior to the sampling dates appeared to be similar, but storm event variability could have been a long-term factor and should be considered when further evaluating substrate characteristics and changes in this stream. Comparisons between North Creek and Avent Creek were also inconclusive due to variable results, potentially different site characteristics affecting substrate sizes, and the lack of measurements in Avent Creek over time.

Although effective results were not generated by the substrate analysis in the two streams, the utilization of the four different methods provided valuable insight into which methods were more suitable for site characteristics. The variable results indicated that more than one method should be used in a stream substrate assessment, and that the most appropriate methods should be considered based on stream characteristics. In this study, comparisons were made between surface substrate evaluation methods, but not for subsurface methods since there was only one method (subpavement). The results indicated that substrate should be evaluated across the entire cross-section, at more locations and on more sampling dates along the reach to account for the high variability present in urban channels. The methods chosen must account for both increased sediment load and increased stormflow velocities in these stream types.

Bedload sediment transport evaluations in this study were performed using bankfull shear stress calculations and pavement and subpavement particle ratios in both North Creek and Avent Creek. The ratios indicated that the artificial stone added along the channel for stabilization were introduced into the riffles and probably influential in both the substrate and bedload sediment transport evaluations. Estimation of bedload sediment transport capacity in North Creek during bankfull flow using pavement and subpavement particle sizes also indicated variability in the substrate particle sizes. In samples where the particle size ratios were not in the 3-7 range, the measured water surface slopes and mean bankfull depths were greater than the predicted values for mobilizing the bed materials. This occurred in Reach 2 and Reach 3 during the first sampling date, where larger particle sizes in the samples required more shear stress to initiate mobilization. The other portions

of the stream with less influence from artificial substrates at the substrate sampling locations (Reach 1 and the second samples in Reach 2) had measured slopes and mean depths at bankfull less than predicted. This suggested that the substrate was mobilized more frequently at flows less than bankfull. Comparisons between measured largest subpavement particle sizes and predicted sizes indicated similar potential influences of the larger artificial substrates in portions of North Creek. The overall bedload sediment transport results in North Creek showed the effects of variable substrates, and indicated that more substrate sampling of the pavement and subpavement may be required before utilizing this information in stream designs.

The bedload sediment transport analysis in Avent Creek suggested that the stream was not stable, even though the channel morphology and substrate characteristics provided evidence of stability. The sediment transport imbalance in this stream was thought to be a result of either limited variability in the subpavement samples used for analysis or due to the application of sediment transport equations not suitable for the fine to very fine gravel substrate. Similar to North Creek, the pavement and subpavement samples indicated variability that should be verified with more than two samples per reach.

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CHAPTER 4: SUSPENDED SEDIMENT, SEDIMENT YIELD, AND SEDIMENT BUDGET

INTRODUCTION

Suspended Sediment

Suspended sediment is the sediment that remains suspended in the water column for extended periods of time due to turbulence provided by streamflow (Simons and Senturk, 1992). Sediment suspended is usually less than 0.5 mm in diameter, including clay, silt, and small sand particle size classes (Dunne and Leopold, 1978; Beschta, 1996). Sediment suspended in the water column is a function of particle size, the amount of particles available, and the amount of streamflow turbulence (Hynes, 1960). Sediments with smaller particle sizes tend to remain suspended longer than larger particles, according to Stokes law (Marshall *et al*., 1996). Once sediment is suspended, it maintains its position in the water column by turbulence (Dunne and Leopold, 1978).

Suspended sediment concentrations in streamflow can vary significantly over space and time. Vertical and horizontal gradients develop along the stream cross-section, with concentrations usually highest near the streambed compared to the surface, and higher concentrations near the center of the channel compared to near the streambanks (Graf 1971; Haan *et al*., 1994). These gradients are caused by velocity gradients and the turbulence of streamflow (Graf, 1971). Suspended sediment concentrations also vary temporally along the hydrograph. Maximum concentrations usually occur at or near the peak discharge, then decrease rapidly during the recession limb (Beschta, 1987). In small urban streams with intense stormflows, the suspended sediment spatial gradients may not be as steep due to the turbulence of the streamflow and the smaller channel dimension.

Variability in suspended sediment concentrations between the rising limb and falling limb of the hydrograph may also not be as great in small urban streams. The increase during the rising limb would be contributed from the "first flush", when stormwater runoff from the watershed carries the available loose sediment from land surfaces to the stream (Beschta, 1987). Suspended sediment could continue to be elevated during the falling limb, however, due to channel erosion contributions. During stormflow recession, the channel continues to be exposed to shear stresses, and the streambanks experience additional geotechnical stresses from bank moisture (USACE, 1981; Whipple *et al*., 1981). In urban streams with significant streambank erosion, elevated suspended sediment concentrations could be maintained throughout the entire hydrograph.

Sources and Effects in Urban Streams

Excess sediment contributed to streams and rivers is a leading cause of stream degradation in the United States (USEPA, 1998). Suspended sediment can be contributed to streams from both the surrounding watershed and from within the channel itself. In urban and developing watersheds, suspended sediment is derived from soil erosion upslope. Erosion occurs through rainsplash on bare surfaces and through sheetwash across bare and impervious surfaces (Dietrich *et al*., 1982). Sediment derived from the watershed can be deposited as colluvium on the hillslopes, deposited as alluvium on the floodplain, or delivered directly to the stream in stormwater runoff (Dunne and Leopold, 1978; Dietrich *et al*., 1982). In urban watersheds, most of the suspended sediment is probably delivered directly to the stream channel by stormwater runoff pipes and ditches, with little opportunity for deposition in the landscape.

The average total suspended sediment (TSS) concentration in urban stormwater runoff has been estimated to be about 100 mg/L (USEPA, 1983). During watershed construction activities, however, TSS concentrations can be magnified significantly to greater than 1000 mg/L (Wolman and Schick, 1967; Goldman *et al.,* 1986). Watershed development has been shown to substantially affect the stream sediment yield, through both construction and addition of impervious surfaces (Dunne and Leopold, 1978).

Sediment derived from within the stream occurs through the processes of channel erosion and the shifting of channel bars. In urban streams, stormflows tend to have larger volumes and peak discharges along with elevated sediment loads (Dunne and Leopold, 1978). Suspended sediment concentrations during stormflows may continue to be elevated for long time periods after development because of channel instability, which is caused by altered stormflows and an imbalance in the sediment carrying capacity. After development, the predominant sediment source in urban watersheds becomes channel erosion (Whipple *et al.*, 1981). Rocky Branch on the NCSU campus was found to have sediment concentrations of 700-1000 mg/L during storm events, with little construction occurring in the watershed at the time (Duda *et al.*, 1979). Lenat and Crawford (1994) found average TSS concentrations of 440 mg/L in Marsh Creek, an urban stream north of Raleigh with a fully developed watershed.

The increased suspended sediment concentrations in streams have many negative effects on channel stability, substrate characteristics, sediment transport capacity, and habitat availability for aquatic organisms. The stream cross-section adjusts its shape to convey the increased stormflow volumes and velocities in urban watersheds. The stream

cross-section responds by vertical incision, then channel widening as erosion progresses (Schumm *et al*., 1984). The stream cross-section must also adjust to an increased suspended sediment load that deposits during flow recession. The sediment transport capacity of the stream is exceeded, and aggradation of fine sediments at the base of streambanks and along the channel bed results. The channel responds by widening, as the cross-section must adjust to convey the same stormflows delivered (Simon, 1989).

Detrimental effects of large suspended sediment loads on aquatic organisms have been well documented, including studies with fish (Karr, 1981) and macroinvertebrates (Reed, 1977; Lenat and Crawford, 1994). High suspended sediment concentrations lead to elevated turbidity of the stream water, blocking light for photosynthesis and for feeding and reproduction (Hynes, 1960; Hynes, 1970). Although there are no suspended sediment concentration standards for North Carolina, the water quality standard for turbidity in non-trout waters is less than 50 NTUs (NCDENR-DWQ, 2004). Suspended sediments also cause habitat problems when they settle to the substrate during stormflow recession. The substrate becomes blanketed with the fine particles and/or embedded as the interstitial spaces between larger particles are filled. The habitat potential of the substrate decreases for benthic macroinvertebrates and periphyton, along with negative impacts within the hyporheos zone just below the substrate (Richards and Bacon, 1994). Large suspended sediment concentrations are also a concern due to the potential associated pollutant load (Hynes, 1960).

Suspended Sediment Measurements

Both suspended sediment and turbidity of streamflow can be measured from water samples collected during storm events. One sampler type is the single-stage suspended sediment sampler, or "bottles-on-a-stick". These independent samplers are designed to collect stream samples automatically during the rising stage of storm flows (IACWR, 1961). The samplers are composed of a supporting post with sample bottles attached at various stage heights. The sample bottles are fitted with a stopper and two copper tubes, one for water intake and one for air exhaust as the sample is collected (Appendix 4, Figure A4.1). Another method to collect water samples for suspended sediment and turbidity measurements is to use an automated pumping sampler that can collect stream samples at specified time or flow volume intervals (Beschta, 1996). These samplers have an intake that is anchored to the streambed or floating free at a particular location and height. Water samples are collected using suction from a pump.

Both methods have associated advantages and disadvantages for sediment and water sampling. Single-stage sediment samplers obtain depth-integrated samples during stormflows, while automated samplers are anchored at a fixed vertical height. Single-stage samplers collect suspended sediment data near the surface, while automated samplers collect data usually near the bed if anchored. Suspended sediment concentrations tend to be higher near the bed compared to the surface (Beschta 1996); therefore, single stage samplers may underestimate sediment parameters. Sample collection may require more time and effort with single stage samplers, since samples must be collected before the next event. Automated samplers can collect samples from multiple events, reducing sample

collection time. The single-stage samplers are less expensive in equipment costs, and they require no power source to maintain.

For both sampling methods, intakes are subject to blockage by debris or large sediment that can prevent sample collection or affect the sample concentration. The velocity of the water entering the intake must also be the same as the velocity of the streamflow. If the velocity entering the intake nozzle is slower than the stream flow, then sediment will settle for an enriched sample. If the velocity is faster, such as during surges or rapid rises in flow, the sediment will tend to wash out of the nozzle and the sample will be diluted (Edwards and Glysson, 1999). To ensure that the velocities are the same, the right sampler and intake diameter should be chosen for the stream measured, based on the stream velocities, the expected particle sizes, and the potential water-surface surge (IACWR, 1961).

Stream Sediment Yield and Budget

The total sediment yield (volume/time) from a watershed is the rate at which sediments from both watershed and channel sources pass a particular point in the drainage system (Dunne and Leopold, 1978). Sediment yield is composed of both suspended sediment and bedload sediment moving along the channel bed. Sediment yields can increase with greater rainfall and greater stormwater runoff, but the relationship may not be as straightforward in all watersheds (Foster *et al.*, 1990). Some watersheds may have a significant amount of sediment storage, as a portion of the sediment is redeposited downslope or downstream from the original sediment source. Sediment derived from both watershed and channel sources in urban watersheds can be deposited in channel bars and

along the toe of streambanks; deposited on the floodplain; or carried out of the drainage basin as sediment yield (Trimble, 1983).

To help define sediment yield in a watershed, a sediment budget can be derived to describe the entire sediment system. A complete sediment budget is a quantitative statement of sediment production from the watershed and channel; sediment transport and storage; and sediment discharge rates or yields from the basin (Dietrich *et al*., 1982; Trimble, 1983). In order to derive a sediment budget, transport processes and storage elements must be quantified, and the linkages between the two components must be identified (Dietrich *et al.,* 1982). The typical method of constructing sediment budgets is to estimate sediment yield, hillslope erosion, and channel erosion, with the assumption that a steady state relationship exists between what is transported out of the basin by the stream and what is derived from erosion (Peart and Walling, 1988). Sediment yield is commonly estimated by integrating streamflow hydrographs and sediment rating curves, which relate sediment concentrations as a function of discharge (Dunne and Leopold, 1978; Prestegaard, 1988). Variability in sediment concentrations associated with discharges can be due to location within the hydrograph, variation in sediment sources, different rainfall intensities, and other factors associated with sampling (Beschta, 1987). In small urban watersheds, hillslope erosion may be negligible since a significant proportion of the watershed is covered by impervious surface and most of the stormwater runoff is collected in stormwater conveyance structures. A measure of sediment suspended in stormwater runoff could be an adequate estimate of sediment yield from the watershed. Channel

erosion in small urban watersheds can be extensive (See Chapter 2), and sediment loads derived from this source should be measured in the sediment budget.

The typical sediment budget described above has been shown to be incomplete in some watersheds because it neglects sediment storage (Trimble, 1977; Trimble 1983). The most common place for sediment storage in watersheds is on the floodplain (Leopold *et al.*, 1992). In urban streams, floodplain storage may not be significant or long-term for several reasons. These streams are typically incised with limited stormflow floodplain access for sediment deposition. Stormwater runoff is usually delivered directly to the channel without exposure to upslope areas and floodplains. When stormwater runoff is allowed floodplain access, the larger discharges may cause floodplain erosion and discourage sediment deposition and storage.

Sediment yields and sediment budgets have been shown to be influenced by watershed size. Yields tend to be higher per area for smaller watersheds, because less sediment gets trapped in the watershed floodplains, upslope areas, and in the channel sediment bars compared to larger watersheds (Dunne and Leopold, 1978). Brune (1948) found that sediment yield per area tends to increase with a decrease in drainage area, an increase in stormwater runoff, and an increase in landscape area disturbed. Based on this concept, a sediment budget for a small urban stream may be adequate to include only streambank erosion, streambed erosion, and sediment yield as suspended sediment load and bedload.

Suspended Sediment and Sediment Yield Study

The purpose of this study was to estimate total suspended sediment (TSS) concentrations and turbidity levels in North Creek between August 2002 and August 2004. Samples were collected during baseflow, and during most storm events using two methods, single-stage sediment samplers and an automated sampler. The results from these two methods were evaluated and compared. For each storm event, the suspended sediment load was estimated and compared between the methods. Annual suspended sediment yields were also determined for the North Creek watershed above Research Drive.

The purpose of this study was also to provide information about portions of a sediment budget for North Creek. A complete annual sediment budget would include:

$$\text{Sediment Yield}_{\text{(suspended + bedload)}} + \text{Sediment Storage}_{\text{(upslope + floodplain + stream bars)}} = \text{Sediment Load}_{\text{(streambank)}} + \text{Sediment Load}_{\text{(streambed)}} + \text{Sediment Load}_{\text{(watershed runoff)}}$$

For this study, only certain elements of the budget were determined during the study period; therefore, a complete budget was not possible with the data collected. The suspended sediment yield was determined, along with the sediment load attributed to streambank and streambed erosion. Even though not all elements were assessed, these three components were compared to provide some insight into the sediment system of the watershed. The suspended sediment yield and sediment budget study objectives were:

1) To compare the average total suspended sediment (TSS) concentrations and average turbidity measurements during stormflows and baseflows.

2) To evaluate the trends in stormflow average TSS concentrations and average turbidity moving downstream and with stage elevation during a two-year period.

3) To compare average TSS concentrations and average turbidity values in samples collected using two methods: a single-stage sediment sampler and an automated sampler.

4) To estimate the annual suspended sediment yield using data from the two sampling methods over a two-year period.

5) To estimate the annual sediment loads from streambank erosion and streambed erosion over a two-year period.

6) To derive a preliminary sediment budget for North Creek, using the suspended sediment yield and channel sediment load estimates.

METHODS

Suspended Sediment and Turbidity

Baseflow grab samples were collected at four locations (S1, S3, S6, and S8) along North Creek for total suspended sediment (TSS) concentration and turbidity measurement (See Chapter 1, Figure 1.1). The samples were collected five times during the two-year study.

Stormflow samples were collected for TSS and turbidity analysis using two methods along North Creek. Single-stage sediment samplers were placed at the eight permanent cross-section locations established along North Creek in three reaches (See Chapter1, Figure 1.2-1.4). Samples were collected during storm events between August 2002 and August 2004 at all eight locations (S1-S8) during the first year, and at four locations (S1, S3, S6, and S8) during the second year. The support posts for the samplers were constructed of industrial size U-channel, or 2x4 wood planks nailed to protruding tree roots. The posts were driven into the ground to a depth of about 2-4 feet below the substrate. Sample bottles were 500-ml plastic Nalgene bottles (32mm opening), attached to the posts using cable ties. Intake and exhaust tubes were made of ¼-in diameter copper tubing inserted into rubber stoppers, with height and curvature dimensions of the US U-59B Model (IACWR, 1961; Figure 40), for small watershed size and heterogeneous sediment. The sample bottles were placed facing flow at 0.5-ft intervals along the posts to a maximum height of 3 feet above baseflow water surface. A gage plate was located near each sampler to direct bottle placement, and for recording actual stage above substrate.

An automated sampler (Teledyne ISCO Model 3700) was installed just downstream of the Research Drive culvert. An automated bubbler flow meter (Teledyne ISCO Model 4230) was attached to the sampler, signaling sample collection when the water level was above a specified height, on a flow-weighted basis (75,000-100,000 gallons). The flow meter was calibrated using stage-discharge curves derived for the cross-section (See Appendix 1, Figure A1.6 and A1.7). For each storm between April 2003 and August 2004, the samples from the rising limb and the falling limb were composited, depending on the size of the storm, and the time intervals for composites were recorded.

For comparison of the two methods, the automated sampler and the S7 single-stage sediment sampler were located in the same section of stream. The automated sampler was located at the same station as the S7 sampler between April 2003 and late July 2003. Due to two large erosive storms at the end of July and beginning of August, the automated sampler had to be moved about 100-ft upstream to S7a (See Figure 1.4) for additional intake protection and substrate stability.

Grab samples were collected at the stormwater outfall channels (SOCs) along North Creek during two storm events (8/12/2004 and 8/14/2004) near the end of the study. These samples were collected at the culverts draining into the SOCs (See Chapter 1, Figure 1.2-1.4 for locations) for a general idea of TSS concentrations and turbidity measurements derived from the watershed. These measurements were not used for sediment load calculations or watershed runoff sediment estimates.

All water samples were brought back to the lab and analyzed for total suspended sediment (TSS) concentration and turbidity measurement. For TSS, the samples were

vigorously shaken and 25-ml to 100-ml aliquots were filtered using pre-weighed glass microfiber filters (Whatman 934AH, 1.5µm pore size, 47 mm). The filters were dried at 230°F (110°C) for 48 hrs (Cole Palmer Model 05015-54 Laboratory Oven), and then re-weighed to the nearest hundreth gram (Mettler Toledo AB54). TSS concentration (mg/L) was determined by subtracting the oven-dried filter weight from the filter tare weight, then dividing the resulting sediment weight by the sample volume filtered.

Turbidity (NTUs) of each sample was measured using a turbidity meter (Hach 2100P Turbidimeter). Aliquots were taken after vigorously shaking the sample and placed in appropriate size sample vials for readings. Fingerprints and excess water were wiped from the vial before reading, and samples were measured twice for an average turbidity determination.

The average TSS concentrations and turbidity values were compared between baseflow and stormflows to determine the effects of stormwater runoff on suspended sediment quantities. The comparisons were made at the baseflow sampling locations (S1, S3, S6, and S8), and average stormflow values were obtained from two storm events (rainfall >0.30-in) occurring near the baseflow sample date in order to make effective temporal and spatial comparisons.

The average TSS concentration determined from samples collected during each storm at each single-stage sediment sampler location were compared to determine trends with increasing stage and downstream distance along North Creek. The relationship between the automated sampler average TSS concentrations and average turbidity values for the entire storm and the single-stage sediment sampler (S7) for the rising limb of the

hydrograph was determined. Sediment rating curves were generated for each sampling method at this stream location for storms with measured data. The average TSS concentrations were plotted versus peak discharge estimates for each storm event. Peak discharge was determined using peak stage measurements and the stage-discharge relationships at S7 and S7a (see Chapter 1, Appendix 1).

Suspended Sediment Load

The suspended sediment load was determined for each storm event during the two-year study at the automated sampler and S7 single-stage sampler locations. The average TSS concentrations for each storm sampled using both methods were multiplied by the total streamflow volume estimated from the flow meter. The loads were reported in tons of sediment.

Sediment Budget

In this study, suspended sediment yield, streambank erosion sediment load, and streambed erosion sediment load were determined as a portion of the annual sediment budget.

Suspended Sediment Yield

The annual sediment yields between August 2002 and August 2004 were determined by summing the suspended sediment loads during each storm. The storm events not measured were estimated from hydrologic and TSS relationships. For flow volume estimates when the flow meter was not operating, the relationship between precipitation and streamflow volume was used. In the North Creek watershed, this relationship was well correlated ($r^2 = 0.86$, $p<0.05$) even with differences in intensity due

to the high concentration of impervious surfaces and the small watershed size (Appendix 4, Figure A4.2). For average TSS concentration estimates during storms not measured, the sediment rating curves were used to estimate average TSS concentrations at both locations. The peak discharge was determined from maximum stage estimates at the S7 single-stage sampler, or from the maximum stage estimates at the automated sampler, or from the relationship between precipitation and peak discharge. This relationship between precipitation and peak discharge estimated using the flow meter at the automated sampler location was also well correlated ($r^2 = 0.56$), due to watershed characteristics (Appendix 4, Figure A4.3).

Streambank Erosion Sediment Load

The streambank erosion sediment load was estimated using the streambank erosion data from cross-sections and toe-pins (See Chapter 2), and bulk density measurements at each streambank measurement location. Bulk density of each significant soil texture horizon within the streambank profile was determined at each cross-section and toe-pin location. The profiles were classified for general soil texture using the "feel" method (Brady 1990). The soil composition for cross-sections and toe-pin profiles are provided in Appendix 4, Table A4.1 and A4.3 respectively.

Core soil samples were collected from each layer in the profile of the cross-section streambanks. Four core samplings were performed using two different cores with fixed volumes. The first two sets of bank samples were collected using a standard-sized metal cylindrical soil core with a 3-in diameter and 3-in height. At some of the locations, dry streambank soil samples were difficult to collect with this core due to effort required to

insert the core into the bank profile. The second two sets of bank samples were collected using a soil auger with a 2.75-in barrel diameter and 5.5-in barrel height as the coring device. The end barrel was isolated using a thick cardboard insert to establish a fixed volume above the curved end pieces. Some error was expected with both methods due to the disturbance of the soil during core insertion. After collection, the soil samples were brought back to the lab and allowed to air dry for several days. Each sample was then oven dried 230 °F (110°C) for 48 hours, and weighed after cooling to the nearest hundredth of a gram. The bulk density of each sample was determined (g/in^3) by dividing the total oven-dry weight by the volume of the sample core, and converted to lb/ft^3 for load calculations.

Soil samples with suspected errors due to a partial core sample obtained were not used to calculate the average bulk density of the profile. The average bulk densities for each typical soil texture were determined along North Creek (Appendix 4, Table A4.2). These average values were used to estimate bulk densities for the horizons classified at each toe-pin location.

The sediment load at each streambank profile location was calculated by first multiplying the lateral change (ft) at each half-foot vertical interval of the streambank by the bulk density (lb/ft^3) determined for that elevation within the profile. The average lateral amount (lb/ft^2) of soil material lost or gained within the streambank profile between surveys was determined. This average lateral loss was multiplied by the streambank height of each profile to yield an estimate of sediment lost or gained per streambank area (lb/ft) between surveys. The sediment lost between surveys was totaled for each streambank

profile over time. The average amount of sediment lost at each streambank measurement location within a reach was determined, and multiplied by the stream reach length to yield pounds of sediment lost over the time period. Annual rates were determined by adjusting the pounds lost over the entire study for an annual time period. For cross-sections, annual sediment loads were calculated using data from both years, and for each year between August 2002 and September 2003, and between September 2003 and August 2004 based on survey intervals. For toe-pins, annual sediment loads could only be determined using data from both years due to survey intervals. All sediment loads were converted to tons of sediment per year for comparison to average TSS yields.

Streambed Erosion Sediment Load

The sediment load contributed from streambed erosion in each reach was determined using the longitudinal profile surveys, stream habitat assessment maps, and substrate characterization data. The longitudinal profile survey data from January 2003 and July 2004 (See Chapter 2, Figure 2.1-2.3) were input into AutoCAD (AutoCAD Land Desktop3) and stream feature lengths were overlain to divide the profiles into feature sections. The stream features (riffle, run, and pool) and their locations along each reach were determined using stream habitat assessment maps (See Chapter 5, Figure 5.4-5.6). An estimate of the vertical change in the streambed between longitudinal profiles was determined (ft^2) for each feature section. This areal estimate was multiplied by the average stream bottom width (ft) in each section determined using an extensive channel survey performed in April 2003 (Al Prince, Professional Surveyor). The volume of streambed change (ft^3) in each feature was then totaled for each reach.

Bulk density measurements were determined for the three stream features in each reach. Core samples were collected from the substrate of two runs and two pools in each reach using a cylindrical metal standard core (3-in dia, 3-in ht). The core samples were oven-dried to determine weight per core volume (g/in^3) and converted to lb/ft^3, as described above for the streambank bulk density samples. The average run and pool bulk densities were determined as the average of the two samples collected in each reach. The substrate bulk density within the riffles was determined using the subpavement particle size analysis, as an indication of particle sizes mobilized from the riffles (See Chapter 3, Figure 3.8-3.10). The combined weight of particles in each subpavement sample collected in January 2004 was divided by the volume of the sample. Sample volume was determined by the area of the bottomless bucket sampler times the depth of sample collection, which was two times the largest pavement particle diameter. The average bulk density (g/in^3) was estimated for the riffles in each reach, and converted to lb/ft^3 for calculations. The average bulk density measurements in each feature for each reach are provided in Appendix 4, Table A4.4.

To estimate the streambed sediment load, the total volume of streambed change (ft^3) in each feature was multiplied by the feature bulk density (lb/ft^3). The total streambed sediment load was determined for each reach of North Creek by summing the sediment loads from each feature within the reach. The sediment load was converted from pounds to tons for comparison with streambank sediment load and the suspended sediment yield.

Statistical Analysis

All statistical analyses were performed using JMP 5 (SAS Institute, 2002). Correlations were performed using linear regression analysis. Box and Whisker plots were used to compare TSS concentrations and turbidity measurements between sampling locations. Analysis of Covariance models were used to determine the effects of sampling station and stage height on TSS concentrations and turbidity measurements. The least squares means (LS Means) were determined to compare the mean TSS and turbidity measurements at the average stage measurement for all stations.

RESULTS AND DISCUSSION

Total Suspended Sediment and Turbidity

Stormflow Estimates

Baseflow and stormflow sediment concentrations and turbidity were compared to determine the effects of stormwater runoff on these sediment parameters in North Creek. Average TSS concentrations and turbidity measurements in both baseflow and stormflow samples were compared at the four sampling locations where baseflow grab samples were collected (Table 4.1). Average stormflow values were determined by taking the average of two storm events with greater than 0.30-in of rainfall near the baseflow sampling date.

The average TSS concentrations and turbidity values were greater during stormflows compared to baseflow measurements at all four sampling locations during the study. The only exception occurred at S1 during the last sampling date, where average TSS and turbidity were greater during baseflow. Observations made on the baseflow sampling date indicated that turbid discharge was entering at the Avent Ferry culvert upstream from S1, not associated with a storm event. The discharge at the upstream culvert was diluted moving downstream, with a general decrease in average TSS and turbidity between locations. Based on the results of the comparison, stormwater runoff and stormflows were indicated as a source of the suspended sediment load in North Creek.

To get a general idea of the stormwater runoff sediment contributions along North Creek, TSS concentrations and turbidity values were determined from samples collected at significant stormwater culverts draining to North Creek. The samples were only collected during two storms near the end of the study (See Chapter 1, Figure 1.2-1.4), but could be

used to indicate what portion of the watershed was potentially contributing high suspended sediment loads. The culverts and their associated TSS and turbidity values are shown in Table 4.2. The average TSS and turbidity measurements were not generally consistent between storms, which occurred close together in time. The average values were much greater during the first storm in the upper watershed, but results were highly variable moving downstream. The trend seemed to be a greater suspended sediment and turbidity contribution from the upper watershed, down to the SWMP #2 (SOC4), where most watershed development had been occurring. The elevated TSS and turbidity measurements at the SOC8 were due to a newly constructed stormwater wetland, discussed in Chapter 6. The results of the culvert sampling demonstrated that stormwater runoff was a contributing factor to the suspended sediment load for North Creek. The watershed sediment load entering North Creek from the culverts could not be estimated, however, due to the limited number of samples and the variability measured between storms.

Table 4.1: Average Total Suspended Sediment (TSS) Concentrations and Turbidity Measurements During Baseflows and Stormflows.

		Baseflow		Stormflow	
Baseflow Sampling Date	Location	TSS (mg/L)	Turbidity (NTUs)	TSS (mg/L)	Turbidity (NTUs)
7/24/2002	S1	No Data	18	228	84
	S3	9	18	466	131
	S6	11	39	No Data	No Data
	S8	13	36	1453	540
10/18/2002	S1	No Data	29	142	91
	S3	11	25	262	144
	S6	7	29	724	218
	S8	10	21	593	232
1/24/2003	S1	35	7	73	70
	S3	17	9	340	162
	S6	55	24	360	253
	S8	10	10	411	157
3/13/2003	S1	4	15	230	66
	S3	33	56	243	129
	S6	19	40	414	139
	S8	19	35	372	164
6/28/2003	S1	186	148	146	78
	S3	100	111	306	149
	S6	96	99	379	146
	S8	76	122	646	247

Table 4.2: Average Total Suspended Sediment (TSS) Concentrations and Turbidity Measurements at Culverts Discharging to Stormwater Outfall Channels Along North Creek. SOC8 was the Outlet to a Constructed Stormwater Wetland in August 2004.

		8/12/2004		8/14/2004	
Reach	Stormwater Outfall Culverts	TSS (mg/L)	Turbidity (NTU)	TSS (mg/L)	Turbidity (NTU)
1	Avent Ferry Culvert	4028	875	79	47
	SOC1	524	326	25	20
	SOC2	110	284	346	281
2	SOC3A	221	123	64	65
	SOC3B	16	96	19	5
	SOC4	43	56	464	446
	SOC5	34	42	12	10
	SOC6	29	30	4	8
3	SOC7	39	15	9	9
	SOC8	122	118	19	36

Spatial Trends

The eight single-stage sediment samplers established along the stream reach were used to provide information on the spatial variability of TSS concentrations and turbidity measurements in North Creek moving downstream and with an increase in discharge. The single stage sediment data were analyzed to establish spatial trends using the log values of both TSS concentrations and turbidity values, since the log (base 10) values were normally distributed. The box and whisker quantile plots of log TSS concentrations (Figure 4.1) and log turbidity measurements (Figure 4.2) suggested an increase in sediment parameters progressing downstream, with large variability at each station.

To test the effects of both stage height above water surface and station on suspended sediment parameters, an Analysis of Covariance model was performed for log TSS concentrations and log turbidity measurements (JMP 5, SAS Institute, Inc.). For TSS

concentrations, the full model interaction term (stage * station) was found to not be significant ($p < 0.2025$), so it was dropped and the reduced model was analyzed. The slopes of the regression lines were similar for each station demonstrating the response of TSS concentrations to an increase in stage height. The regression plot for log TSS concentrations is shown in Figure 4.3, derived from the reduced model. The slopes of the regression lines at each station are positive, indicating an increase in TSS concentration with an increase in stage height. For turbidity measurements, the full model interaction term (stage * station) was significant ($p < 0.0378$) indicating that at least one of the stations responded differently to an increase in stage. The regression plot for log turbidity is shown in Figure 4.4, showing that not all of the regression lines were parallel. The slopes of the regression lines at each station are positive, indicating an increase in turbidity values with an increase in stage height moving downstream. The analysis of covariance analysis confirmed that there was a significant difference between TSS concentrations and turbidity measurements after adjustments for stage height.

The predicted log TSS concentration and turbidity values were compared across stations in a least squares means (LS Means) plot for log TSS concentrations and log turbidity values (Figure 4.5). The LS Means represented the mean log values of TSS and the mean log values of turbidity at each station, evaluated at the average value of all stage measurements. The least squares means contrast analysis indicated that the LS Means for both TSS concentrations and turbidity had a significant increasing linear relationship moving downstream ($p < 0.05$).

Based on the statistical analysis, the suspended sediment concentrations and turbidity of the stream water significantly increased moving downstream with an increase in stage height during storm events. An increase in stage height indicates an increase in discharge moving downstream. As more stormwater runoff was contributed to the channel, more suspended sediment was also contributed to the stream water. The results of the analysis of covariance alone did not indicate which sediment source, stormwater runoff or channel erosion, delivered the majority of sediment. Further information from the streambank and streambed erosion sediment loads were needed to determine the relative sediment contributions in the stream.

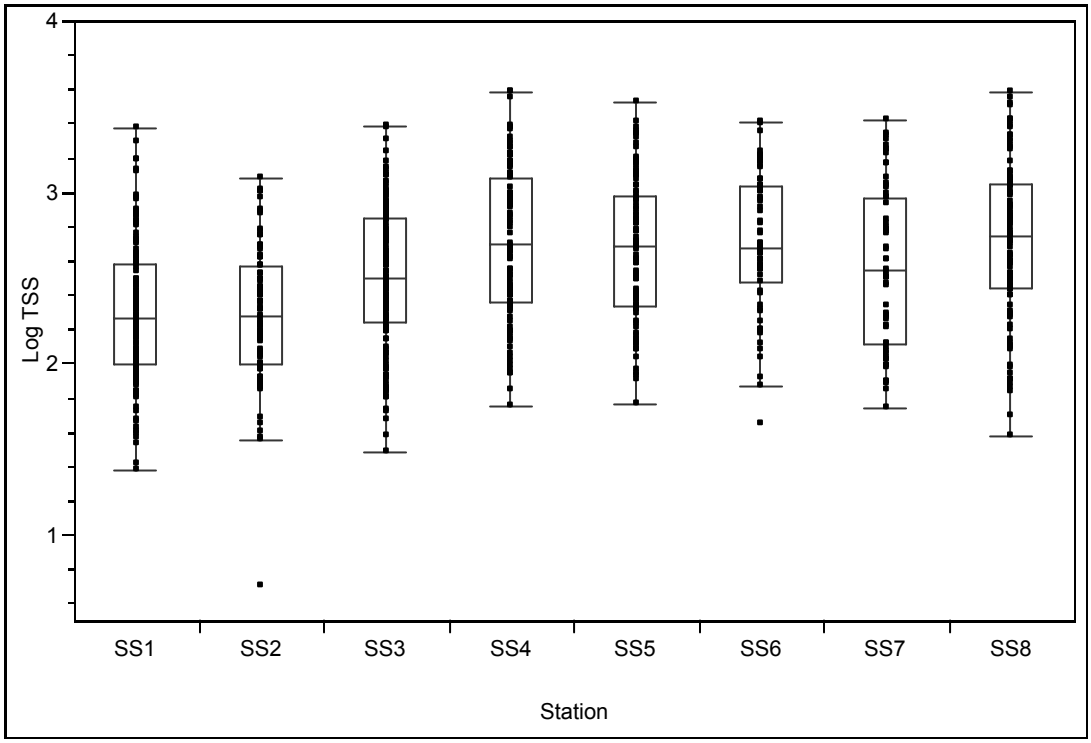


Figure 4.1: Box and Whisker Plots of Total Suspended Sediment (TSS) Concentrations at Each Sampling Location. The Plot Depicts the Minimum and Maximum Values, Bars Showing the 10% and 90% Quantiles, and Box Delineating the Median Value, 25% and 75% Quantiles.

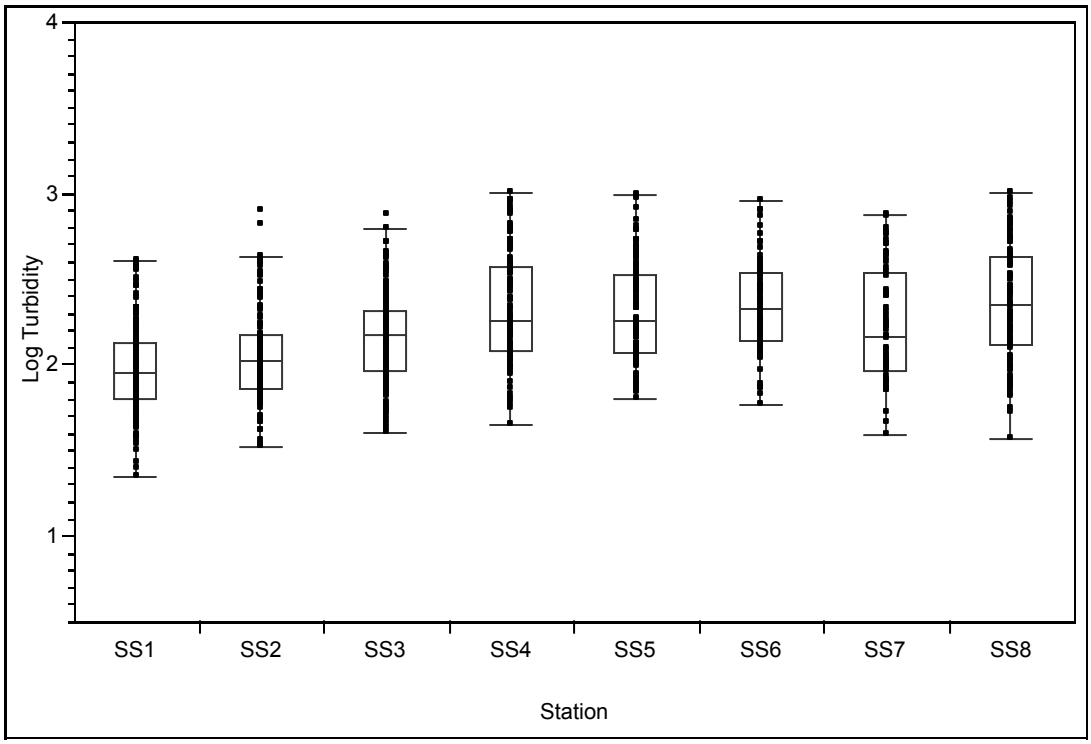


Figure 4.2: Box and Whisker Plots and Mean Diamonds of Turbidity Measurements at Each Sampling Location. The Plot Depicts the Minimum and Maximum Values, Bars Showing the 10% and 90% Quantiles, and Box Delineating the Median Value, 25% and 75% Quantiles.

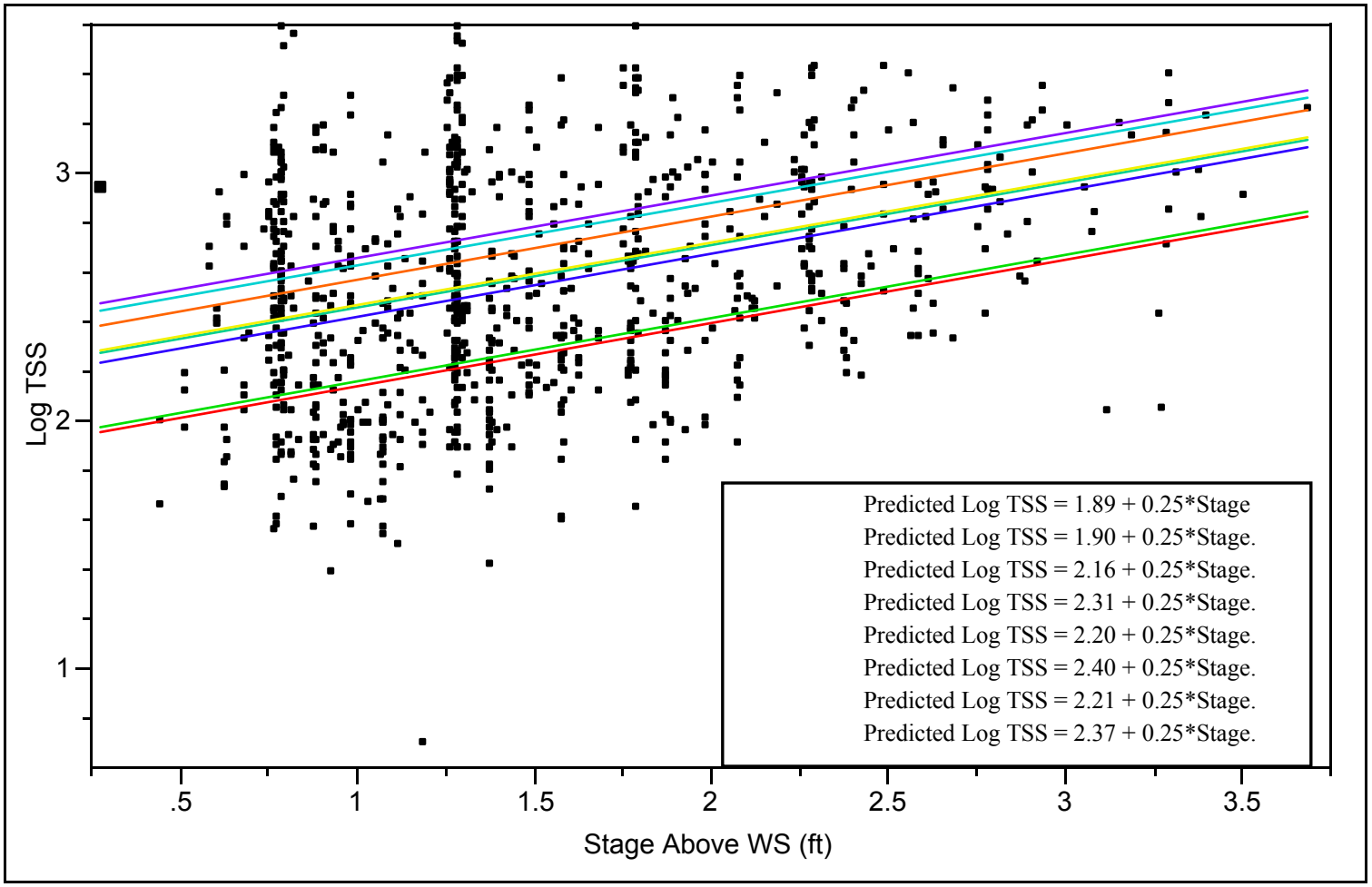


Figure 4.3: Regression Plot for the Log TSS Concentrations and Stage Height Reduced Analysis of Covariance Model, with Regression Equations.

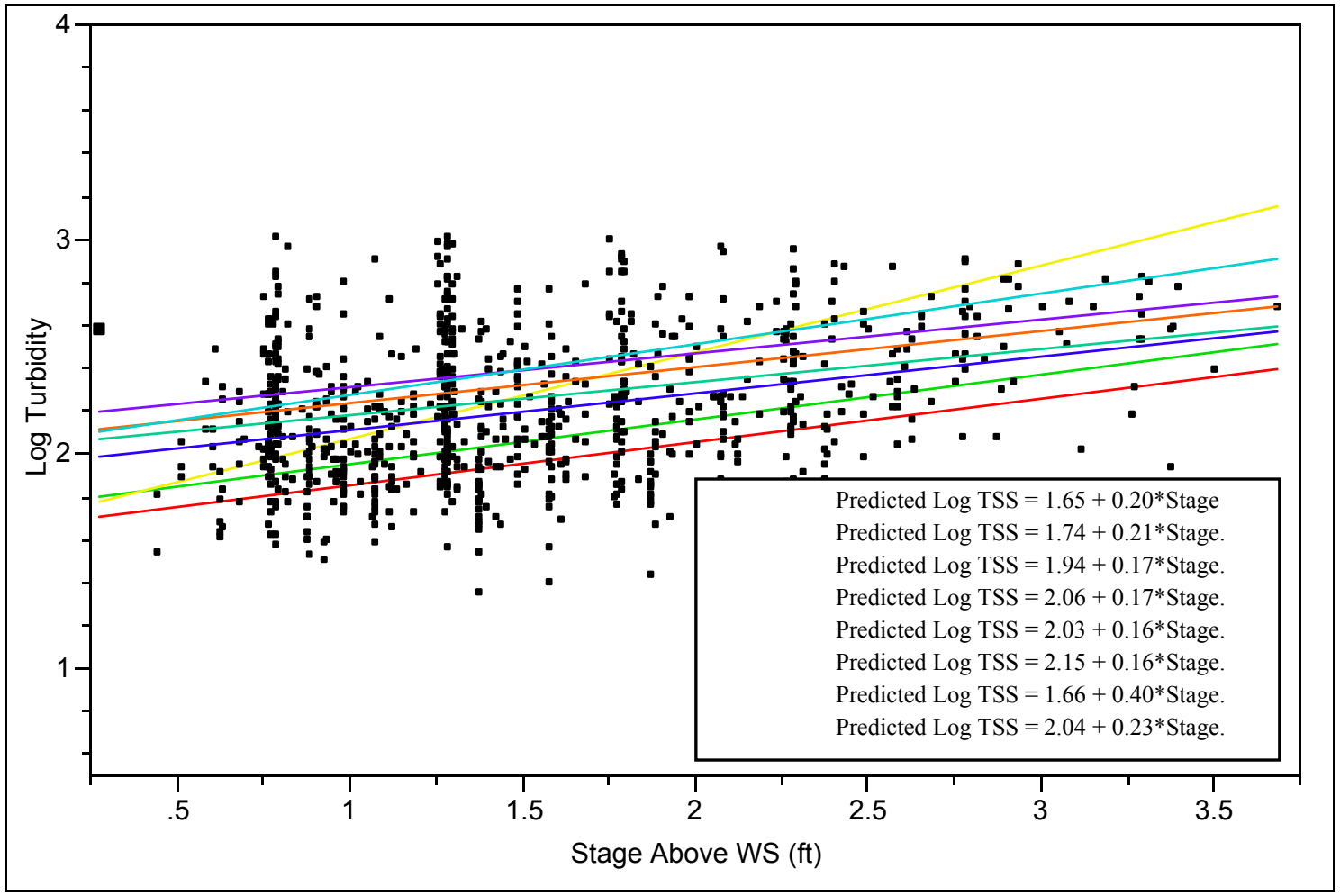


Figure 4.4: Regression Plot for the Log Turbidity Values and Stage Height Analysis of Covariance Full Interaction Model at Each Station, with Regression Equations.

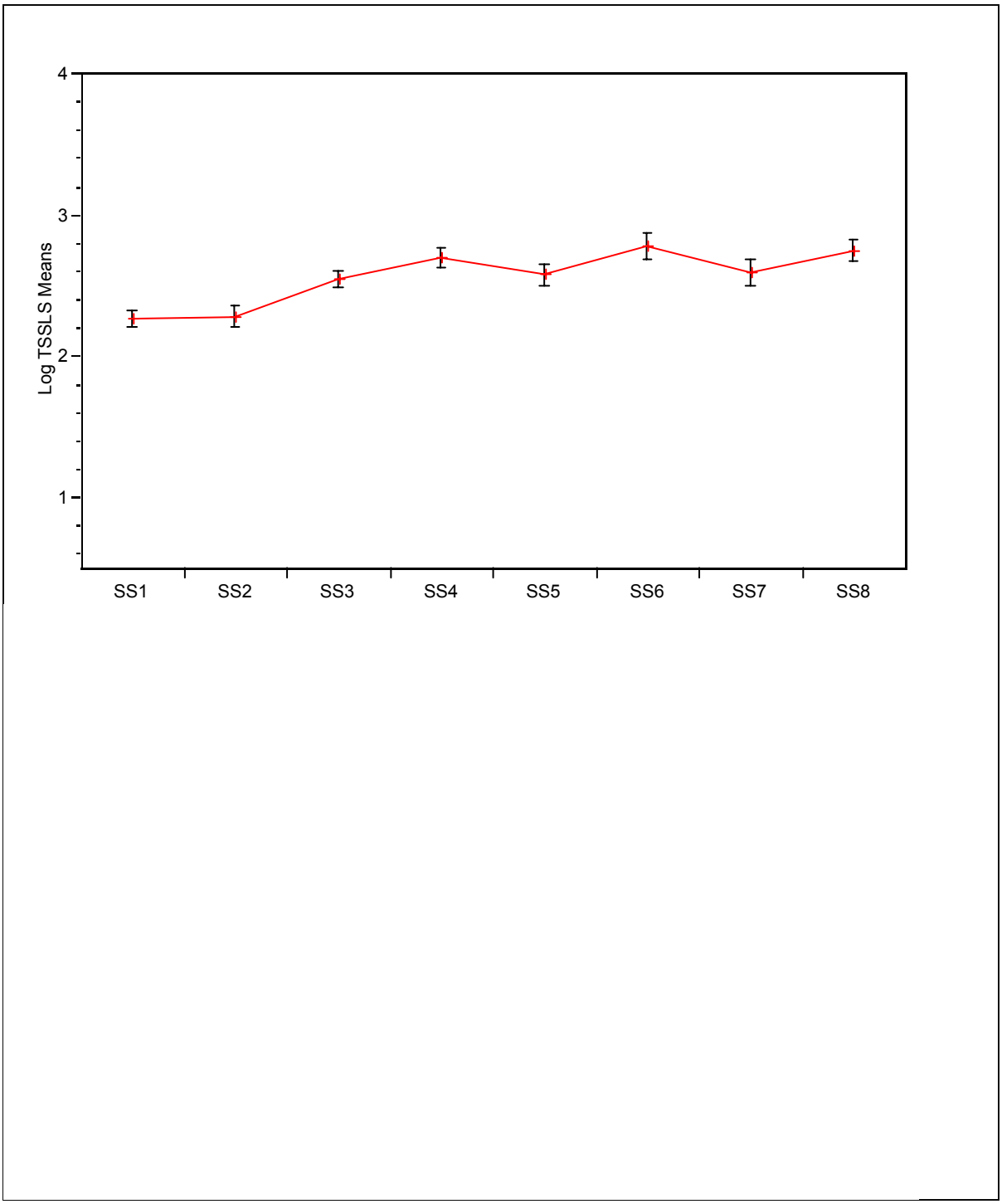


Figure 4.5: Least Squares Means Plots of Predicted TSS Concentrations and Turbidity Measurements at the North Creek Sampling Locations.

Sampling Methods Comparison

The relationship between sediment parameters measured using a single-stage sampler and an automated sampler at the same location (near S7) was determined in order to compare the two methods. The single-stage samplers collected stormflows at different vertical heights as the stage rose, but only during the rising limb of the hydrograph. The automated sampler collected stormwater at one fixed height near the substrate, but over the entire hydrograph. The average TSS concentrations for both methods were estimated for each storm measured by taking the average of all concentrations measured from samples within a storm. The concentrations were not adjusted for spatial variability along the cross-section, since intakes were located off-center near the same streambank, and because the sediment was assumed well mixed based on the relationship between channel size and streamflow velocities.

A sediment rating curve was generated for both the single-stage sediment sampler (S7) and the automated sampler (ISCO) using available discharge data. The average TSS concentration during each measured storm event was plotted versus the peak discharge estimated using the top bottle height for the single-stage sampler and the flow meter data for the automated sampler. Average TSS concentrations for both methods were well correlated with peak discharge measurements based on the regression equation, with r^2 = 0.75 for S7 (Figure 4.6), and r^2 = 0.57 for ISCO (Figure 4.7). TSS concentrations were much higher for the single-stage sampler compared to the automated sampler. When the average TSS concentrations of both methods were compared between the two methods for the same storm events, there was a significant relationship between the two methods

(Figure 4.8). The concentrations were again well correlated ($r^2 = 0.76$), suggesting there might be a consistent factor showing the difference between the measured concentrations using each method.

The factor causing the difference between the two methods was probably the suspended sediment concentration of the falling limb. The majority of average TSS concentrations estimated using the automated sampler were obtained by compositing the rising limb samples and the falling limb samples, then taking the average. The single-stage sampler only measured TSS concentrations during the rising limb, which appeared to be greater than the TSS concentrations during the falling limb based on method comparisons. Further data analysis is required to confirmed differences in the rising and falling limb of samples collected using the automated sampler. There were no studies found in the literature that compared these two methods for obtaining suspended sediment concentrations; therefore, the relationships found in this study could only be assumed from the data measured in North Creek.

After evaluating the relationship between the two sampling methods, the TSS concentrations estimated by the automated sampler were considered more appropriate in this study for sediment load determinations. The associated problems with each method also influenced the higher confidence in the automated sampler. During several large storms, the automated sampler intake became buried and collected enriched samples. This enrichment was not as frequent as the suspected enrichment or flushing that was suspected to have occurred with the single-stage samples. These sample bottles were frequently uncorked or surrounded by debris at stage elevations greater than a foot above water

surface. Many of the storms also had multiple peak discharges, especially during winter, spring, and remnant tropical storms, that could have affected single-stage samples once they were initially collected (Edwards and Glysson, 1999). For an improved comparison between the two methods, a manual sampler should be used to obtain width and depth integrated samples along the cross-section and during different storm events for comparison.

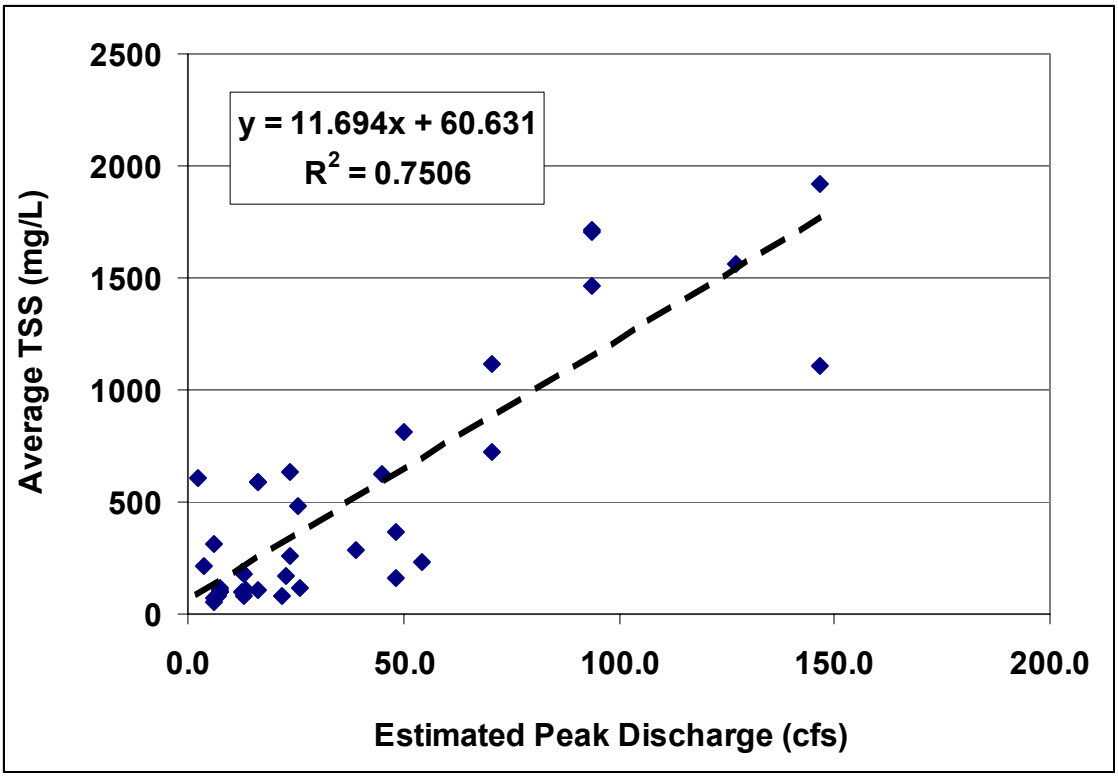


Figure 4.6: Sediment Rating Curve and Best-Fit Regression Equation for Average Total Suspended Sediment (TSS) at the S7 Single-Stage Sampler Location.

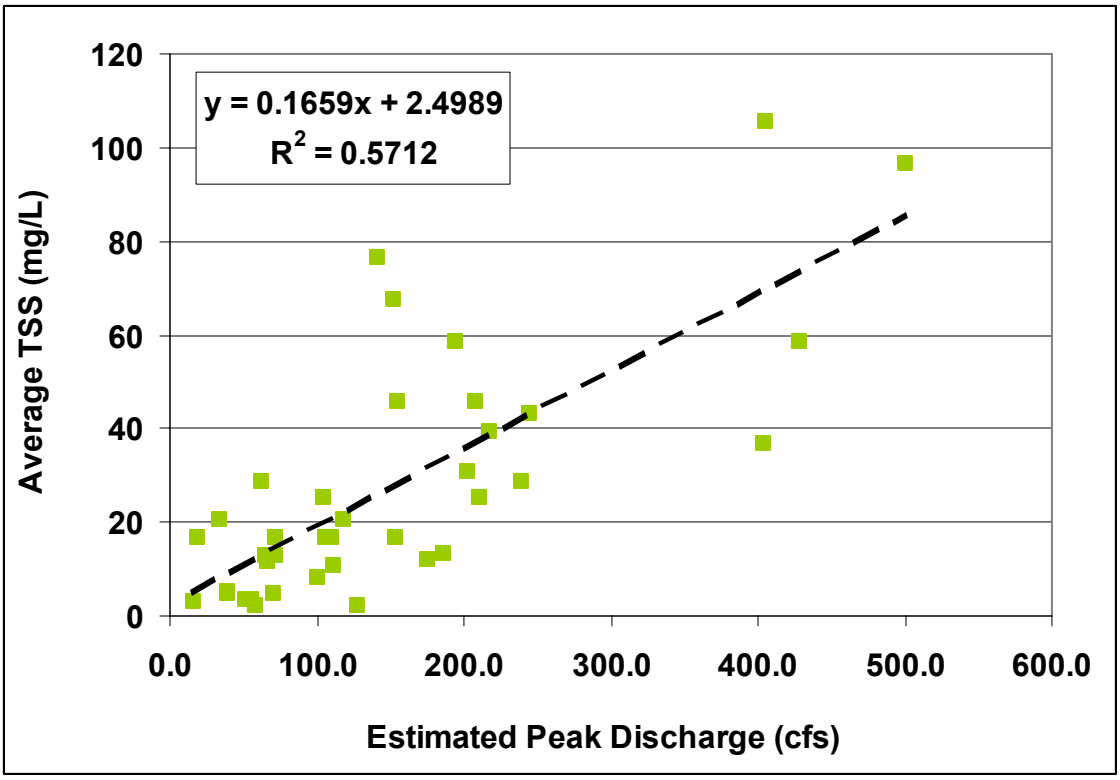


Figure 4.7: Sediment Rating Curve and Best-fit Regression Equation for Total Suspended Sediment (TSS) at the Automated Sampler Location.

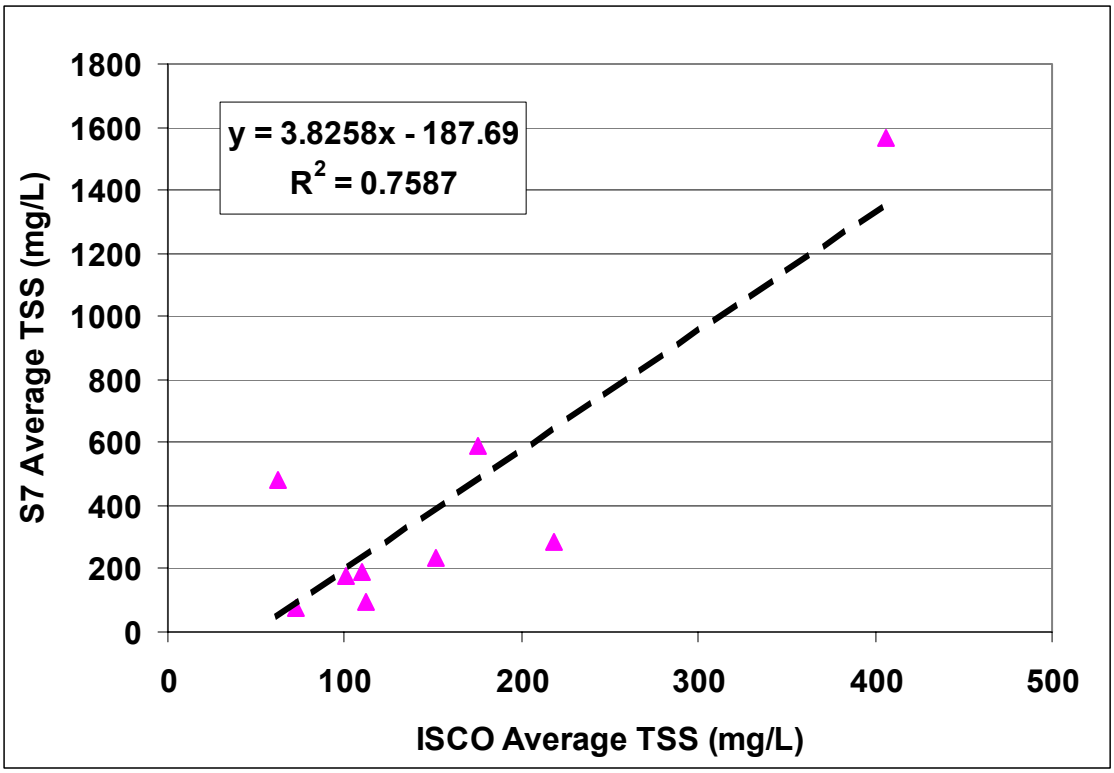


Figure 4.8: The Relationship Between the Average Total Suspended Sediment (TSS) Concentrations at the Single-Stage Sampler (S7) and the Automated Sampler (ISCO), with Best-Fit Regression Equation.

Suspended Sediment Yield and Sediment Budget

The complete sediment budget for North Creek would include the following elements:

Sediment Yield $_{(\text{suspended + bedload})}$ + Sediment Storage $_{(\text{upslope + floodplain + stream bars})}$ = Sediment Load $_{(\text{streambank})}$ + Sediment Load $_{(\text{streambed})}$ + Sediment Load $_{(\text{watershed runoff})}$.

During this study, the three components of this sediment budget measured were suspended sediment yield on the left side of the equation (outputs), and the sediment load attributed to streambank erosion and streambed erosion on the right side of the equation (inputs).

Suspended Sediment Loads

The estimated suspended sediment yields (tons/yr) at the TSS collection locations were determined by totaling the average TSS loads that were measured or estimated during each rainfall event. Annual sediment yields were estimated for each year during the study (August 15th to August 15th), as well as for the entire study period. The annual sediment yields estimated using measurements from the single-stage sampler were about 2.4 times the yields estimated using the automated sampler during both years of the study (Table 4.3). The average annual sediment yields based on watershed size were 1320 lb/ac/yr using the single-stage samplers and 542 lb/ac/yr using the automated sampler. These yields are similar to another small urban stream study of Marsh Creek located north of Raleigh, estimated as having about 1175 lb/ac/yr (Lenat and Crawford, 1994).

Estimated sediment yields during the first year were over three times higher than those estimated during the second year for both sampling methods. The temporal variability was most likely attributed to the amount and intensity of rainfall during the first year compared to the second. The total rainfall for the first year (68 in) was much greater than the 30-yr normal average of 41.4 in (NC SCO, Raleigh, NC), compared to the second year (37 in). The first year also had three large, intense storms in October 2002 (>3.66 in/hr), in July 2003 (>3.56 in), and August 2003 (>2.89 in), based on hourly rainfall data and observations near the site (NC SCO, Lake Wheeler Field Lab Station, Raleigh, NC). The second year lacked intense storms, with most all storms less than 1 in/hr. High intensity storms could have lead to greater sediment loads caused by rainsplash and sheetwash of sediment in the watershed, and due to the larger stormwater volumes and

velocities experienced by the receiving stream (Schwab *et al.*, 1993). These larger storms are also not well retained by watershed sediment basins, designed to hold much smaller storm events. Another reason for the inter-year variability could have been differences in watershed construction activities.

Table 4.3: Estimated Annual Suspended Sediment Yields at the Single-Stage Sediment (S7) and the Automated Sampler (ISCO) Locations.

Single-Stage Sampler Estimates	Sediment Yield (tons)
August 2002-2003	354
August 2003-2004	105
Average Annual Sediment Yield	229
Automated Sampler Estimates	**Sediment Yield (tons)**
August 2002-2003	141
August 2003-2004	46
Average Annual Sediment Yield	94

Streambank and Streambed Sediment Loads

The sediment loads from streambank erosion were estimated using the cross-section and toe-pin streambank profile data. Annual average erosion rates were determined for each survey, and the amount of sediment lost or gained was estimated using the bulk density estimates and the total channel length in each reach. For cross-sections, annual sediment loads were estimated between August 15, 2002 and September 24, 2003, then through August 15, 2004 for the second year (Table 4.4). The estimated sediment loads followed a different pattern in Reach 1 compared to the lower reaches. Reach 1

cross-sections had similar amounts of degradation during both years, with an average sediment loss of 35 tons/yr. Reach 2 and Reach 3 had greater sediment loads during the first year compared to the second year, with about twelve times more sediment lost during the first year. The inter-year variability in these latter two reaches was probably attributed to the greater rainfall amounts during the first year, and the two intense storms in July and August 2003. Inter-year variability could also have been caused by variable aggradation patterns along the streambank toe, and variable influence of geotechnical forces within the streambanks that could have caused erosion.

The annual sediment load derived from streambank erosion was also estimated for the entire two-year study period (Table 4.4). The average annual sediment loads for cross-section surveys increased moving downstream from Reach 1 to Reach 3. The estimated annual sediment load from the toe-pin profiles could only be estimated for the entire study due to different sampling dates. These average sediment loads were similar to the cross-section average annual loads over the two-year study between reaches. When the toe-pins and cross-sections were considered collectively in each reach, the average annual sediment load increased moving downstream from Reach 1 (45 tons) to Reach 2 (55 tons) to Reach 3 (101 tons). Reach 3 had the greatest sediment load contribution, as well as the greatest sediment load per reach stream length (0.14 tons/ft). Reach 1 lost an estimated 0.12 tons/ft while Reach 2 only lost an estimated 0.05 tons/ft due to its much longer stream thalweg length. These sediment load loss rates were surprisingly similar between reaches, differing less than one tenth a foot, which suggests that similar hydraulic and geotechnical forces were contributed to the streambanks along the entire stream length. The total estimated

sediment load from streambank erosion in North Creek was 200 tons/yr, using the average values for the two years of data.

The sediment loads from streambed erosion during the two-year study were estimated using the longitudinal profiles taken in January 2003 and July 2004, and bulk density measurements of the three different profile features. The sediment loads are provided below in Table 4.5, with negative numbers indicating estimated net aggradation. For Reach 1 and Reach 2, sediment appears to have been added through aggradation to the stream thalweg. The accumulation of sediment occurred in all three features evaluated along the longitudinal profile. Only the pools in Reach 1 showed a net loss of two tons during the study. In Reach 3, sediment was eroded from the riffles and runs in the upper portions of the reach, probably during the drastic vertical incision in August 2003. The pools increased in sediment volume, mainly due to the sediment accumulation in the large pool at the end of the upper reach.

For all three reaches, an estimated 173 tons of sediment was estimated to be added to the streambed using the longitudinal profile comparisons. These upper reach aggradation results were not expected in this urban stream with erosive stormflows, and sediment load from the streambed was considered as an input variable to the sediment system. Based on the data, the streambed actually served as a net output, probably storing sediment derived from the watershed and streambanks. The largest aggradation was estimated in Reach 1, which could be an overestimation due to several factors. The longitudinal profile surveys used to estimate sediment volume change did not align well between the major riffles and pools even when using comparable benchmarks, which

suggests there might have been errors in one of the surveys. The sediment was also added to the riffles and runs and not the pools, where suspended sediment tends to deposit first. Even though the Reach 1 results could be overestimated, aggradation probably did occur in the upper reaches due to the elevated sediment load in stormwater runoff contributed from the upper portions of the watershed, and due to the streambank erosion measured in these reaches.

Table 4.4: Estimated Streambank Erosion Sediment Loads for Cross-Section and Toe-Pin Locations Over Time Periods During the Study. Stream Reach Lengths are Included for Comparison.

	Time Interval	Reach 1	Reach 2	Reach 3
Cross-section Average Bank Material Lost (tons/yr)	**August 2002-2003**	30	114	158
Cross-section Average Bank Material Lost (tons/yr)	**August 2003-2004**	42	9	14
Cross-section Average Bank Material Lost (tons/yr)	**August 2002-2004**	35	70	98
Toe Pin Average Bank Material Lost (tons/yr)	**August 2002-2004**	54	40	104
Total Average Annual Streambank Sediment Load (tons/yr)	**August 2002-2004**	45	55	101
Stream Thalweg Length (ft)		*370*	*1100*	*715*

Table 4.5: Estimated Streambed Erosion Sediment Loads for North Creek, with Negative Numbers Indicating Aggradation.

Sediment Loads (tons)	Reach 1	Reach 2	Reach 3
Riffle	-77	-8	4
Run	-66	-9	32
Pool	2	-16	-26
Total	-140	-33	10

Sediment Budget Summary

A summary of the North Creek sediment budget components is provided in a diagram with flow paths of sediment between potential inputs and outputs (Figure 4.9). The suspended sediment yield is given for both sampling methods at the comparable station below Research Drive. The sediment loads for the streambanks and streambed are given as a total for Reach 1 and Reach 2, based on the location of the suspended sediment yield data.

The suspended sediment yield was the only output parameter within the budget measured during this study. Bedload sediment yield that traveled through the same cross-section of stream with the suspended load was not measured. Both sediment sampler intakes were elevated above the substrate and not large enough for expected substrate particle sizes. The bedload yield has been estimated to account for 5-25% of the total sediment load within a stream (Simons and Senturk, 1992). This portion of the sediment budget might have been important in North Creek, and should have been evaluated had the goal of a sediment budget been established at the beginning of the study.

Another important output not evaluated in this sediment budget was the amount of sediment storage in the watershed and within the stream channel. Upslope storage within the watershed was considered negligible for the majority of the watershed area. The stormwater conveyance system collected stormwater from impervious surfaces for delivery directly to North Creek before much opportunity for deposition occurred. The exception to this assumption was the storage within sediment basins in the watershed. These structures served about 37 acres of the watershed being developed, and another 25 acres of the watershed through diversion into SWMP #2. The amount of sediment and stormwater retained by these structures was not determined, so their effects are not known within the North Creek sediment budget. Floodplain sediment storage was another portion of the budget not evaluated. This portion was considered negligible for the watershed due to incised conditions along most of the stream length. Only stormflows during the three largest storms in October 2002, July 2003, and August 2003 were observed to reach the floodplain in the lower portions of Reach 2 and along Reach 3. Some of the stormwater outfall channels located on the floodplains were actually observed to contribute sediment from gully erosion during the study, indicating that the floodplain may serve more as a sediment input versus output where erosion is occurring. In-stream sediment storage in lateral and mid-channel bars was another portion of the sediment budget not evaluated for most of the channel. Some of the smaller bars were measured in the cross-sectional surveys, and shown to be dynamic in degradation and aggradation over time. The larger bars were not measured, however, and could have served as a significant depositional area for sediment during streamflow recession of larger storms.

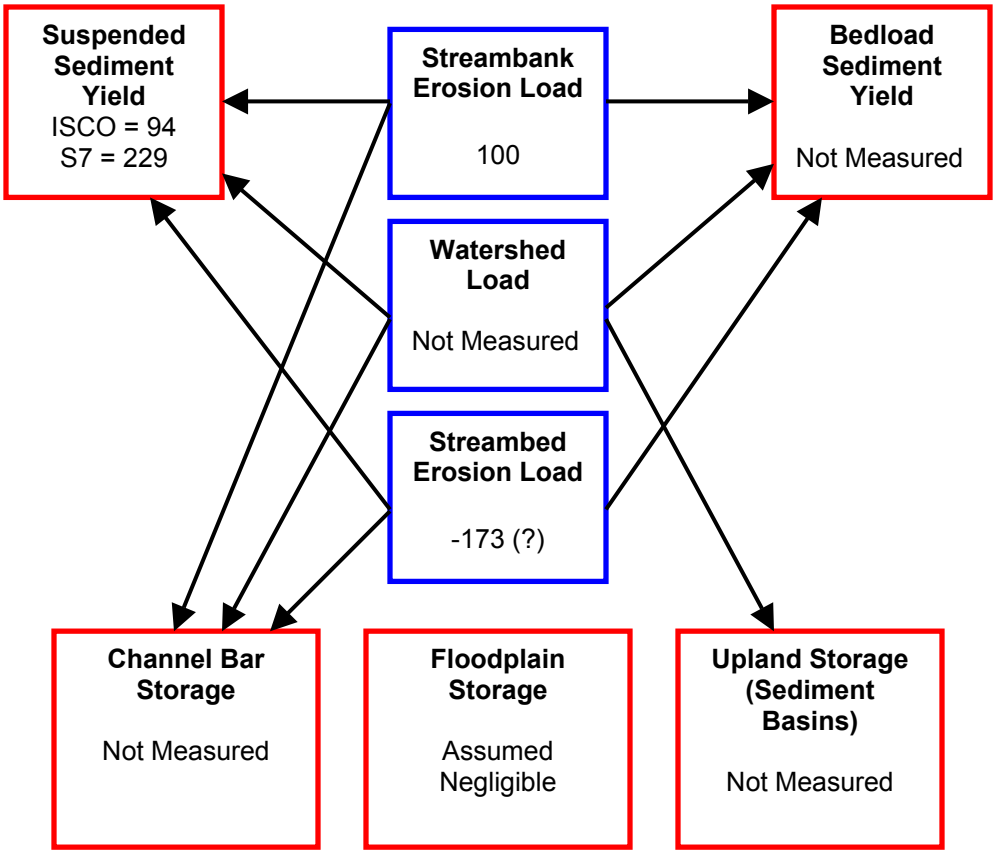
The inputs in the North Creek sediment budget were the sediment loads contributed from streambanks, the streambed, and the watershed runoff. Streambank sediment loads were large, compared to the overall suspended sediment yield. The proportion of streambank materials contributed to the suspended sediment yield was unknown, however. The streambank load could have contributed to both the suspended and bedload sediment yields, as well as contributed to in-stream storage in the channel bars. The streambed sediment load acted as an output during this study, and not an input as indicated in the sediment budget equation. Sediment derived from both the watershed and streambanks could have been stored along the streambed, reducing the annual sediment yield due to slower release of stored sediment over time.

The remaining sediment load input in the North Creek sediment budget was watershed load contributed from stormwater runoff. This portion of the budget was not measured, although grab samples from riparian culverts near the end of the study suggested that this load could be substantial, particularly in the upper portions of the watershed where most of the development was occurring. The watershed load input into the sediment system was expected to either leave as suspended sediment or bedload yield, or be deposited within the stream channel in sediment bars. The majority of the sediment composing the watershed runoff load was expected to be in the smaller particle size classes from upslope soil erosion. Upslope soils were mostly Cecil sandy loam or Cecil clay loams with mainly clay subsurface textures (See Chapter 1, Figure 1.5). These finer particles derived from soil erosion, particularly on disturbed development sites, were

expected to remain suspended in stormflows and be captured during suspended sediment sampling and compose a substantial proportion of the suspended sediment yield.

The sediment budget results during this study would have been different if estimates were made at the end of the stream, including Reach 3. The streambanks and streambed were very dynamic in this reach, with a net sediment load loss of 110 tons/yr. Although the streambed indicated aggradation in the lower reach, the headcut migration in the upper reach caused net degradation. These two loads would probably have composed a larger portion of the suspended sediment yield, and altered the budget numbers even if they didn't alter the budget concepts revealed in the first two reaches.

Figure 4.9: The North Creek Sediment Budget with Sediment Flow Paths Indicated. Estimated Annual Suspended Sediment Yields (tons) and Total Annual Sediment Loads (tons) from Streambank and Streambed Erosion are Given for North Creek Reach 1 and Reach 2. Potential Inputs of Sediment are in Blue and Potential Outputs of Sediment are in Red.



CONCLUSIONS

The TSS concentrations and turbidity values along North Creek were greater during stormflows compared to baseflows, and were found to be elevated at the stormwater culverts contributing runoff to the stream, particularly in the upper portion of the watershed. The results suggested that a portion of the sediment load in North Creek was derived from the watershed in addition to channel erosion. TSS concentrations and turbidity measurements were found to increase significantly moving downstream and with an increase in stage height, as an indication of increasing discharge. As more stormwater runoff was delivered to the stream, the suspended sediment quantities increased. The source of the sediment could have been watershed runoff or channel erosion. The suspended sediment yields were high for a small urban watershed, and comparable to other urban stream studies that documented urban stream suspended sediment.

During this study, suspended sediment and turbidity were measured using two different sampling methods. The single-stage sampler estimates using rising limb data were consistently greater than the automated sampler estimates that accounted for the entire hydrograph. The relationship between average TSS concentrations estimated by the two methods suggested that there could be a consistent factor causing the difference between them, but further data analysis is required for method comparison. A manual sampler should be used, however, to verify results of the two methods by obtaining both width and depth-integrated samples.

Several components of the sediment budget were measured for North Creek during this research. Even though the complete budget was not assessed, the components

measured provided valuable information about the sediment system within the watershed in the upper two reaches. The measured components revealed that the suspended sediment yield was substantially elevated in this stream, and that streambanks were a major sediment contributor within the system. It was estimated that bedload accounted for about 25% or less of total sediment yield. Even if all of the bedload were composed of streambank sediment, the proportion of suspended sediment load derived from streambanks would still have been very large compared to other sources.

The aggradation estimated using the streambed load over time revealed that portions of the streambank materials were most likely deposited near their source. Even though the aggradation volume was thought to be overestimated due to survey inconsistencies in Reach 1, there was still net aggradation along the streambed in North Creek during the study. In-stream sediment bar storage was suspected to also be elevated considering the changes in the thalweg of the streambed profiles. For future research on small urban stream budgets, an estimate of the bedload sediment yield and in-stream sediment storage is recommended to provide adequate information on the sediment dynamics of the system. It is also suggested that the longitudinal profile not only assess the thalweg as the deepest point along the cross-section, but also the highest point along the cross-section for better substrate volume estimates and for information on in-stream sediment bars.

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CHAPTER 5: WATER QUALITY AND BIOLOGICAL ASSESSMENT

INTRODUCTION

Urban Stream Water Quality

The major cause of water quality degradation in urban streams is non-point source pollution, contributed by stormwater runoff (USEPA, 1992). The increased impervious surface cover and reduced infiltration associated with development delivers larger stormwater runoff volumes and peak discharges, along with larger pollutant loads to the receiving streams. Common pollutants in stormwater runoff include excess sediment, excess nutrients, metals, toxic organic compounds, and harmful microorganisms associated with human and animal waste (Schueler, 2000). These pollutants can be added in large pulses during storm events, and can be persistent in the stream system through association with the sediments, organic materials, and stream organisms. Pollutants may have direct biological effects, as well as indirect effects through influence on general water quality parameters such as pH, dissolved oxygen, temperature, and turbidity. Beginning in the early part of the 20th century, the major pollutants delivered to urban streams were recognized and their effects on the biological components of streams were being studied (Hynes, 1960). Besides detriment to aquatic life, contamination of receiving waters can also pose health hazards to humans through drinking water supplies, during recreational uses, and from food consumption.

To address local water quality issues associated with pollutants, North Carolina has established water quality standards for all freshwaters through legislation (NCDENR-DWQ, 2004; 15A NCAC 02B.0211). The standards are based on the classification of the

water body, and were derived to help protect the desired uses. North Creek is designated as a Class C stream, affording it protection for secondary recreation; fishing; aquatic life including propagation and survival; and wildlife habitat. Sources of pollution that preclude any of these uses on either a short-term or long-term basis are considered a violation of the North Carolina water quality standard.

Sediment

Excess sediment contributed to streams and rivers is a leading cause of stream degradation in the United States (USEPA, 1998). Sources of sediment from urban watersheds include soil erosion from bare, unstable landscapes and particle collection from impervious surfaces. Average total suspended sediment (TSS) concentrations in urban stormwater runoff have been estimated at around 100 mg/L (USEPA, 1983). During construction activities in the watershed, however, TSS concentration and sediment load can be magnified significantly (Wolman and Schick, 1967; Goldman *et al*., 1986). After development, TSS concentrations may continue to be elevated due to channel instability caused by large stormflows and the imbalanced sediment carrying capacity. The predominant sediment source in urban watersheds eventually becomes streambed and streambank erosion (Whipple *et al*., 1981). Rocky Branch on the urban NCSU campus and Beaverdam Creek in Raleigh, NC, were found to have average sediment concentrations of 700-1000 mg/L during storm events, with little construction occurring in the watersheds at the time (Duda *et al*., 1979).

Increased TSS concentrations in streams due to both watershed activities and channel erosion have many negative effects on aquatic organisms, including fish (Karr,

1981) and macroinvertebrates (Reed, 1977; Lenat and Crawford, 1994). The amount of sediment suspended in the water column is a function of particle size, the amount of particles suspended, and the amount of streamflow turbulence (Hynes, 1960). When high concentrations of fine sediments are suspended, particularly during stormflows, this leads to elevated turbidity of the stream water. Turbidity is the relative cloudiness of water and can negatively impact light penetration. Aquatic algae and vegetation require adequate light for photosynthesis, while macroinvertebrates and fish require visibility for feeding and reproduction (Hynes, 1960; Hynes, 1970). Although there is no TSS standard for North Carolina, the water quality standard for turbidity in non-trout waters sets a limit of 50 NTUs (NCDENR-DWQ, 2004).

Suspended sediments also cause problems as the stormflows recede, and the sediment settles onto the substrate. The substrate becomes embedded as the interstitial spaces between larger particles are filled. The habitat potential of the substrate decreases for benthic macroinvertebrates and periphyton, along with negative impacts within the hyporheos zone just below the substrate. This zone is an important refuge during extremely high and low flows, and can be negatively affected by fine sediment intrusion and a reduction in dissolved oxygen (Richards and Bacon, 1994). Large suspended sediment concentrations are also a concern due to the potential pollutant load associated with sediment. Pollutants that commonly associate with soil particles include metals, toxic organic compounds, and nutrients such as phosphorus. These pollutants can be stored within the sediments of the streambed, and slowly released over time for long-term effects on stream biota (Hynes, 1960). The contaminated sediment can be resuspended during

large stormflows or release the pollutants during changes in water chemistry (Weiss *et al*., 1981; Wetzel, 1983; Sharma *et al*., 1994).

Nutrients

In healthy streams, nutrient concentrations are generally low due to uptake by plants and organisms, and lower watershed inputs. Sources of excess nutrients, particularly nitrogen and phosphorus, in the watershed include fertilizers, pesticides, organic matter, artificial organic compounds, and waste products. These nutrients can be added to urban streams through both stormwater runoff and rainfall. Ammonium (NH_4) and nitrate (NO_3) are inorganic forms of nitrogen that are relatively non-persistent and non-cumulative in stream systems (Hynes, 1960). In healthy streams, the ammonium concentration is usually below 0.1 mg/l, and the nitrate concentration is generally below 0.5 mg/l. Organic nitrogen compounds found in streams are diverse in nature, associated with biomass or artificial sources. These compounds may be susceptible to microbial decomposition or become persistent in the water column and sediments. Higher concentrations of both inorganic and organic nitrogen may be indicative of sewage or industrial contamination (Hynes 1960). Nitrification, which is the conversion of ammonia to nitrate, occurs in aerobic conditions provided by adequate streamflow. Denitrification, which is the conversion of nitrate to nitrogen gas, requires anaerobic conditions that can occur within the bottom sediments and with stagnant water (Wetzel, 1983).

Phosphorus concentrations in healthy streams are generally below 0.1 mg/l, with higher concentrations indicative of eutrophication (Hart, 1974). Phosphorus is present in stream water as inorganic phosphate (PO_4) or as organic phosphorus compounds.

Inorganic phosphate is readily absorbed by the biotic components, and associated with bottom sediments. Phosphorus adsorbs readily to suspended clay particles and can form complexes, chelates, and insoluble salts with suspended metals present in the water column. The bound phosphorus maintains its relationship due to aerobic conditions prevalent in streams that prevent the release into the water column (Arthington *et al*., 1982). Organic phosphorus, like nitrogen, is also derived from biomass or artificial sources. Some of the compounds are broken down or assimilated by the biota, and some become persistent in the water column and sediments.

Excess nutrients are a problem in urban waters due to the increase in productivity and eventual eutrophication of downstream waters (Hynes, 1970). Microbial communities, including algae and bacteria, become enhanced with excess nutrients. These communities can have negative effects on benthic macroinvertebrates and fish by blanketing substrates and blocking light penetration. As the colonies mature and decay, decomposition of detritus associated with increased productivity depletes oxygen concentrations in the water column and has further detrimental effects on aquatic biota.

Metals

Elevated metal concentrations in urban streams can pose a serious health threat to both aquatic life and humans. High concentrations of metals are typically found in urban runoff, from watershed sources that include industrial effluents, vehicle products and emissions, sewage effluents, and construction activities (Arthington *et al*., 1982). Metals can be dissolved in the water column or associated with suspended and deposited sediments. The North Carolina water quality standards for metals vary, and are specific to

each metal found in the aquatic environment (NCDENR-DWQ, 2004). Harmful effects of elevated metal concentrations toward aquatic organisms in receiving waters are well documented, including fish, benthic macroinvertebrates, and plants (Bryan, 1976; Weiss *et al*., 1981; Jones and Holmes, 1985). Metals can be directly absorbed from the water column by diffusion across organism membranes. Aquatic macroinvertebrates can also be exposed to metals by ingestion of food particles and sediment (Weiss *et al*., 1981). Organisms most affected by high metal concentrations appear to be those associated with the sediment (Smock, 1979). High concentrations of metals within sediments have been shown to reduce benthic macroinvertebrate density and alter community structure (Grumiaux *et al*., 1998; Richardson and Kiffney, 2000). Metals also have the potential for bioaccumulation within trophic levels, to perpetuate negative effects moving up the food chain (Clements, 1991). The toxicity of metals and their rate of absorption by aquatic biota is determined by the chemical form and concentration of the particular metal; the presence of other metals or complexing agents; water temperature; pH; and the relative size of the organism (Bryan, 1976; Weiss e*t al*., 1981).

Toxic Organic Compounds:PAHs

Organic compounds represent the most diverse group of water quality pollutants available in the environment; therefore, it is nearly impossible and financially prohibitive to measure the concentration of all compounds. Polycyclic aromatic hydrocarbons (PAHs) are a common group of organic compounds measured due to their carcinogenic effects and to their prevalence in the environment (Bjőrseth and Ramdahl, 1985). These compounds are comprised of two or more fused benzene rings (McIntosh *et al*., 2004). PAHs enter

receiving waters through stormwater runoff, atmospheric deposition, and sewage effluent (Van Metre *et al*., 2000). They are typically hydrophobic, and readily bind to sediments for slow release over time into the water column (Hoffman *et al*., 1984). The primary anthropogenic sources of PAHs include products from the incomplete combustion of fossil fuels, wood, and coal; automobile emissions; power stations; and other industrial activities (McIntosh *et al*., 2000). There have been over 120 PAHs identified in urban pollution, and the US Environmental Protection Agency has selected sixteen PAHs as priority pollutants with the highest carcinogenic effects (USEPA, 1984). PAHs that have larger numbers of benzene rings usually have the greatest carcinogenic effects (example: 5-ring benzo[a]pyrene) and pose the largest threat to human health (Liu *et al*., 2002; McIntosh *et al*., 2004). A study conducted by the USGS National Water Quality Assessment (NAWQA) Reconstructed Trends Program evaluated sediment cores from ten lakes and reservoirs in urban areas for PAH concentrations. The study indicated that although PAH loads from industrial and coal emissions have decreased due to regulations and improved efficiency, the PAH concentrations have increased over the last two decades (Van Metre *et al*., 2000). Currently, the primary source of PAHs in urban stormwater and receiving water sediments is motor vehicle emissions, which have increased dramatically in urban and suburban areas (Van Metre *et al*., 2000; ATSDR, 1990). Vehicular sources of PAHs include vehicle exhaust, tire wear, asphalt and associated coatings, and various lubricants (Sharma *et al*., 1994).

In North Carolina, there is no specific water quality standard for PAH concentrations in Class C waters. The standard for bodies of water utilized as water supply

sources is 2.8 μg/l (NCDENR-DWQ, 2004), and can be used as a comparative standard concentration. Harmful effects of elevated PAH concentrations have been shown for humans (USEPA, 1984), fish (Herbes, 1977) and benthic macroinvertebrates (ex. USEPA, 1993a and USEPA, 1993b). The effects of PAHs have been shown to be synergistic, with negative effects increasing when benthic macroinvertebrates were exposed to PAH combinations (Verrheist *et al*., 2001). PAHs have also been shown to have photo-induced toxicity effects on benthic macroinvertebrates when exposed to UV light at lower concentrations (Hatch and Burton, 1999).

General Water Quality Parameters

Pollutants in stormwater runoff combined with altered stream hydrology can have negative effects on the general water quality parameters of streams. Aquatic biota require certain parameter conditions for optimal health and reproduction. One important parameter is adequate dissolved oxygen (DO) concentrations for respiration. DO concentrations are dependent on factors such as streamflow, water temperature, and biological metabolism. Oxygen is dissolved in the water column from the atmosphere by the turbulence of flowing water. For small streams, DO is usually at or near saturation during turbulent conditions (USEPA, 2001; Hynes, 1970), and typical concentrations are around 10 mg/l (Hart, 1974). Streams with adequate flow also provide a continuous source of dissolved oxygen for stream biota as the water aids in gas exchange in the vicinity of the organism during metabolism (Ruttner, 1926). Temperature affects DO concentrations based on the inverse relationship between water temperatures and the solubility of gases (USEPA, 2001), with colder water temperatures usually increasing the potential amount of

oxygen dissolved in the water column. Biological metabolism affects the amount of oxygen based on relative rates of photosynthesis and respiration. Algae and plants produce oxygen in photosynthesis during the day, and consume oxygen at night. Heterotrophic organisms remove oxygen from the water column during respiration, when organic matter is decomposed (Hynes, 1960; USEPA, 2001).

The North Carolina water quality standard for non-trout waters is 5 mg/L dissolved oxygen as the average concentration (NCDENR-DWQ, 2004). Concentrations at this level can be found in streams with stagnant pools caused by impoundments or loss of meanders, and in streams with excess microbial productivity and decomposition from nutrient enrichment. Lower DO levels can cause oxygen deficiency in fish and macroinvertebrates, and indirectly affect organism health through the influence on other pollutants (Hynes, 1960). Reduced oxygen concentrations near the substrate may lead to the release of pollutants bound to sediment, such as phosphorus, metals, and toxic organic compounds.

The pH of water indicates the relative acidity or alkalinity, using a logarithmic scale between 1 and 14 (USEPA, 2001). Low pH values indicate acidic conditions, while high pH values indicate alkaline conditions. The pH of the stream water can be influenced by geology, soils in the watershed, and pollutants contributed through stormwater runoff. The pH of stream water affects many chemical and biological processes, including pollutant availability in the water column. Aquatic biota are most diverse between pH values of 6.5 and 8.0 (USEPA, 2001), and North Carolina water quality standards specify an acceptable range between 6 and 9, depending on normal values observed in nearby

waters (NCDENR-DWQ, 2004). Extreme pH values can cause disassociation of sediment and pollutants, and can be intolerable to aquatic organisms (Wetzel, 1983; Hynes, 1960)

Conductivity is the measure of water's ability to pass an electric current and is dependent on the total concentration of dissolved elctrolytes. These substances include anions such as chloride, nitrate, sulfate, phosphate; and cations, such as sodium, magnesium, calcium, iron, and aluminum. The geology of the channel is the primary natural factor controlling conductivity, with flow through sediments derived from granite bedrock usually having lower conductivity than flow through channel substrates dominated by clay minerals. The most suitable conductivity measurement for fish and benthic macroinvertebrates is 0.15-0.50 (mS/cm). Although conductivity itself doesn't negatively affect organisms, changes in conductivity over time can indicate increase in pollutant loading. Sewage inputs tend to increase conductivity measurements, while oils, grease, and organic compounds tend to decrease measurements from ambient levels.

The rates of most biological and chemical processes are influenced by water temperature. Higher temperatures accelerate metabolic processes in microorganisms, leading to increased oxygen consumption (Hynes, 1960). Aquatic organisms usually require a specific temperature range for optimum health, with temperature affecting the susceptibility of the organism to toxic compounds and disease. Temperatures in stream water can change due to weather; removal of the vegetative canopy; the presence of stagnant water; urban stormwater runoff; and groundwater inflows (USEPA, 2001). The North Carolina water quality standard states that the temperature may not exceed 2.8 °C

above the natural water temperature, with a maximum of 32 °C in the Piedmont and Coastal Plain regions (NCDENR-DWQ, 2004).

Benthic Macroinvertebrates as Bioindicators

Although water quality guidelines are in place for many water quality parameters and pollutants found in urban streams, quantifying and monitoring urban stream conditions are difficult. Aquatic systems are inherently variable in nature, with fluctuations in discharge and pollutant storage. The pollutant concentrations within the water column and associated with the sediments are inconsistent, depending on factors such as streamflow, the relative state in which the pollutant exists, and general water quality parameters. The amount and rate of pollutant loading from sources outside the aquatic system is also highly variable, depending on the frequency and duration of storm events, the timing and release rate from the source, and changes occurring to the landscape. Attempts to summarize accurately the large temporal and spatial variability through water quality monitoring can become expensive and time consuming, especially in urban streams with a diverse array of pollutants. Due to these variability issues, the use of biological indicators for water quality monitoring has developed.

Biological communities integrate the effects of multiple environmental stresses and express the cumulative impacts through community changes (Plafkin *et al*., 1989). Benthic macroinvertebrates have been used extensively as biological indicators of overall stream health because they usually occupy a fixed habitat, have relatively longer life cycles than microbes, and occur in all aquatic habitats (Hynes, 1960). The taxonomy and ecology of these organisms is also relatively well documented, with literature available on species and

responses to specific pollutants. Due to heterogeneous communities with diverse trophic levels and microhabitats, there is a high likelihood that some portion of the community will be affected by a change in water quality. Benthic macroinvertebrates are also good indicators because they reflect both short-term and a long-term stream conditions (Reynoldson, 1984). These organisms can be affected by both large pulses of pollutants and by smaller, more continuous pollutant loads (Duda *et al*., 1979). Several studies have shown that benthic macroinvertebrates can be more reliable in urban settings than chemical or physical parameters for stream water quality assessment (Hynes, 1960; Resh and Unzicker, 1975; Benke *et al*., 1981; Jones and Clark, 1987; Lenat and Crawford, 1994). While chemical and physical parameters are limited to the moment in time and the place where they are measured, biological parameters can integrate conditions over space and time with less collection effort (Hynes, 1960; Metcalfe-Smith, 1992).

However, there are disadvantages to using only biological indicators for water quality assessment. It is not possible to gain information on all pollutant types and quantities using biological parameters (Hynes, 1960); therefore, some chemical and physical parameters should be incorporated in monitoring. Another disadvantage is that biological parameters can respond to other factors not associated with water quality degradation (Metcalfe-Smith, 1992). As biological indicators, benthic macroinvertebrates are also influenced by availability of habitat, availability of food resources, and natural variations in hydrology (Voshell, 2002). These factors should be assessed when sampling benthic macroinvertebrates and evaluating the communities present.

Benthic Macroinvertebrate Habitat

Benthic macroinvertebrates can occupy a diversity of habitats in aquatic systems. In stream reaches, habitats can include riffles, pools, undercut regions, subsurface sediments, and various organic materials. Substrate size and type have a significant influence on macroinvertebrate distribution and abundance. Many of these organisms prefer larger substrates due to the ease of attachment, ability to move in the interstitial pore spaces between particles and due to the protection provided from predators and stormflows (Minshall, 1984; Voshell, 2002).

In addition to habitat requirements, benthic macroinvertebrates also require organic resources for food acquisition, including decomposing leaf materials, microbial films on hanging roots in undercuts and other solid substrates, and decomposing woody debris (Voshell, 2002). One way to assess the health of benthic macroinvertebrate communities is to evaluate their functional feeding group (FFG) composition. A FFG is a group of organisms with similar behavioral and morphological adaptations for obtaining food resources. Changes to the FFG assemblage or the absence of a FFG in the stream environment may indicate water quality or habitat problems operating through the food chain (Merritt and Cummins, 1996), and may help evaluate the ability of a stream system to retain organic matter and sediment (Cummins, 1988). The most common FFGs are scrapers that remove periphyton or algae from hard surfaces; predators that eat living tissue; collector-gatherers that feed on organic matter in sediment or detritus; collector-filterers that filter particulate organic matter from the water column; and shredders that break-up coarse organic matter such as woody debris and leaf particles (Cummins, 1974).

In small forested streams, the FFG assemblage is generally dominated by shredders and collectors due to readily available allochthonous materials contributed from the canopy. Scrapers increase with stream size as a result of increased substrate size, light penetration, and resulting growth of benthic algae. With an increase in fine organic particles and flows, such as occur in larger rivers, the assemblage may be dominated by collector-gatherers (Cummins, 1988). During environmental stress, shifts in FFG composition can occur with a general decline in the proportion of shredders, scrapers, and predators that may require more complex habitats and sources of organic matter compared to FFGs such as small collectors that can utilize fine sediment habitats and microbial resources.

Changes to habitat over time in urban streams have been shown to negatively impact benthic macroinvertebrates. Channel erosion and deposition of fine sediment remove available substrate within the channel and decrease habitat diversity. Important habitats are eliminated when headwaters and small streams are graded or confined in stormwater pipes. Natural vegetated habitats from which these tributaries play an important role in attenuating peak discharges in watersheds, while providing sediment and floodwater storage (Resh and Goldhaus, 1983). In contrast, organic resources may be limited in urban channels due to removal of riparian vegetation by development and elevated peak discharges, and due to the flashy stormflows carrying coarse materials downstream (Gore, 1994). Streamflow is an important factor controlling the benthic macroinvertebrate community because it provides food and oxygen, and it controls the organisms' ability to maintain positions within habitats (Hynes, 1970; Voshell, 2002). During watershed development, the hydrology of the stream is usually altered. Larger

stormflows with higher peak discharges are contributed from impervious surfaces, while smaller baseflows occur between storm events due to channel incision and reduced infiltration (Simmons and Reynolds, 1982). During frequent flashy events, the organisms may not withstand the velocity and drift downstream. During reduced baseflow, the habitat in the stream is reduced as the wetted perimeter declines and the macroinvertebrates may move out of the substrate and drift to find better food and space (Gore, 1994). As a result of channel morphology and hydrology changes, the larger species with longer life cycles tend to be replaced by smaller, opportunistic species with shorter life cycles (Menzel *et al*., 1984; Shields *et al*., 1994; Shields *et al*., 1995). The reduced diversity and abundance in these communities make them ecologically unstable, and potentially more susceptible to catastrophic storm events and pollutant exposures.

The Effects of Urbanization on Benthic Macroinvertebrates

There have been many studies concerning the effects of urbanization on benthic macroinvertebrates. Watershed development has been shown to negatively effect natural, diverse populations of unharmful aquatic insects, while creating new habitats for less desirable species such as mosquitoes (Resh and Goldhaus, 1983). Macroinvertebrate communities have been impacted in urban watersheds even when there were no suspected sewer leaks or other point source discharges, indicating that stormwater runoff was a significant factor (Lenat *et al*., 1979; Jones and Clark, 1987; Stepenuck *et al*., 2002). In North Carolina, Duda *et al*. (1979, 1982) found that taxa richness was significantly decreased in streams flowing through urban areas. The number of more sensitive EPT taxa (members of the insect orders Ephemeroptera, Plecoptera, and Trichoptera) declined, while

more tolerant species of chironomids (order Diptera) and oligochaetes increased. Chironomids and oligochaetes have been the most common benthic macroinvertebrates found downstream of contaminated effluents (Hynes, 1960). Lenat and Crawford (1994) compared water quality parameters and benthic macroinvertebrate communities between different land uses. The benthic macroinvertebrate community showed indications of stress in the urban stream through reduced taxa richness; fewer unique species; higher biotic indices, and lower abundance values.

The relative amount of watershed development has been found to affect community composition, taxa diversity, abundance, and the overall biological health of benthic systems. Benke *et al*. (1981) evaluated the impacts of urbanization in 21 watersheds of the Atlanta area and found a negative relationship between degree of urbanization and taxa richness. Taxa richness seemed to be positively associated with the amount of vegetation left within the watershed, and the age of the development. Jones and Clark (1987) assessed the impacts of urbanization as determined by watershed human population density in 22 Northern Virginia streams. They found that population density had little effect on total abundance of benthic invertebrates, but significant shifts in the species composition and a decline in genus diversity. As watersheds increased in population densities, there was an increase in the abundance of chironomids and tolerant caddisflies, while the numbers of Ephemeroptera, Coleoptera, Megaloptera, and Plecoptera declined. Stephenuck *et al*. (2002) assessed benthic macroinvertebrate communities in 43 Wisconsin streams with varying degrees of imperviousness. The effects were evaluated using the Hilsenshoff biotic index, a diversity index, and functional feeding groups (Cummins,

1974). The amount of imperviousness was negatively correlated with the metrics evaluated, and degradation of biological communities was observed at relatively small amounts of impervious area. Good stream quality was found below 8% imperviousness with a threshold occurring between 8 and 12% imperviousness. Even small increases in percent impervious surface were found to be associated with large degradations in benthic macroinvertebrate communities.

Water Quality and Benthic Macroinvertebrate Study

Walnut Creek and the backwaters of Lake Raleigh are designated as impaired waters in North Carolina under the Section 303(d) of the Clean Water Act, based on impaired biological integrity caused by urban runoff and storm sewers. Since North Creek is a tributary to Lake Raleigh and has multiple stormwater runoff inputs along its reaches, it was suspected of being impaired and to be contributing to the problems in the Walnut Creek watershed. The North Creek watershed has about 40% impervious cover, and is currently experiencing further development on Centennial Campus, NCSU.

The purpose of this study was to determine the general water quality condition of North Creek, and to perform a biological assessment using benthic macroinvertebrates for biological integrity evaluation. The water quality of the stream and the amount of available habitat were evaluated to help interpret the biological data collected. All of these data will be used as a background existing condition for assessment of stream restoration and stormwater retrofit designs in an upcoming improvement project for the North Creek watershed. For the water quality and benthic macroinvertebrate study, the objectives were:

1) To evaluate the state of water quality during stormflows in North Creek using pollutant concentrations and established water quality standards.

2) To determine the differences between pollutant concentrations measured during stormflows and baseflow.

3) To evaluate general water quality parameters measured during baseflow and compare values to established water quality standards.

4) To measure and compare benthic macroinvertebrate taxa richness, relative abundance, and functional feeding group assemblages in North Creek and Avent Creek.

5) To measure and compare habitat availability in North Creek and Avent Creek, using habitat surveys, organic material sampling, and substrate evaluations in riffles and pools.

6) To introduce benthic macroinvertebrates into North Creek from Avent Creek and to evaluate their survival and persistence in order to further define biological integrity and the effects of water quality and habitat availability in North Creek.

METHODS

Stream Water Quality

Stream samples and measurements for water quality analysis were collected at four locations along North Creek (See Chapter 1, Figure 1.1). The first location (S1) was just after the Avent Ferry culvert where the stream is initially day-lighted. The second location (S3) was at the end of Reach 1 just before the Varsity Drive culvert. The third location (S6) was at the end of Reach 2 just before the Research Drive culvert. The fourth location (S8) was just before Stormwater Management Pond #3 at the end of Reach 3. The sampling plan was based on available funding; therefore, samples were not collected on a regular basis following an experimental plan. Samples were collected whenever money or laboratory time was available, particularly for metals and PAHs.

Stream samples and measurements were also collected in a forested reference reach, Avent Creek (Harnett Co., 0.4 sq mi drainage area). The samples were collected in a riffle and a pool along a 500-ft reach with comparable drainage area (See Figure 1.5).

Total Suspended Sediment and Turbidity

Total suspended sediment (TSS) concentrations and turbidity levels were measured in baseflow conditions using grab samples collected at the four locations along North Creek. Baseflow samples were collected five times between 2002 and 2003, at each location. In Avent Creek, TSS concentrations and turbidity levels were measured during baseflow at the two locations, on two sampling dates between 2002 and 2003. Stormwater samples were collected and measured for TSS concentrations and turbidity levels in North Creek during the majority of storm events between August 2002 and August 2004 (See

Chapter 4). Monthly average TSS concentrations and average turbidity values were determined for storm events that occurred during the months when baseflow samples were collected. The samples were collected using single-stage sediment samplers at various stage heights during the rising limb of the hydrograph. The specific methods for TSS concentration and turbidity measurements are presented in Chapter 4, along with a detailed assessment of these two parameters during storm events.

Nutrients

Baseflow and stormwater samples were evaluated for nitrate, ammonium, and orthophosphate concentrations (mg/L) by the NCSU-BAE Environmental Analysis Laboratory. Baseflow nutrient concentrations were measured in North Creek and Avent Creek by collecting grab samples. Samples were collected during October 2002 and March 2003 at the described locations. In North Creek, stormflow nutrient concentrations were determined from stormwater samples collected using single-stage sediment samplers at the four described locations. Samples were collected during two storm events (February 2003 and June 2003) with greater than 1-in of rainfall. The stormwater samples were taken from the rising limb of the hydrograph and were a composite of two different water levels above the substrate.

Metals

Baseflow and stormwater samples were evaluated for metal concentrations in North Creek by the NCSU Soil Analysis Laboratory. Metal concentrations were measured as mg/kg sediment suspended within the samples, and not as a concentration within the water column. Baseflow samples were collected in April 2003 as grab samples from the four

described locations. Stormflow samples were collected using single-stage sediment samplers during two storm events (January and February 2003) with greater than 1-in of rainfall. During the first storm, the individual location samples were composited in error, and only average values for the entire stream reach were determined. The second stormwater samples were taken from the rising limb of the hydrograph and were a composite of two different water levels above the substrate.

Organic PAHs

Baseflow and stormwater samples were evaluated for PAH concentrations in North Creek by the NCSU Department of Molecular and Environmental Toxicology. Fifty-three different PAH concentrations (ηg/L) were measured, including the 16 USEPA Priority Pollutant PAHs. Baseflow samples were collected March 2003 as grab samples from the four described locations. Stormflow samples were collected using single-stage sediment samplers during two storm events (February 2003 and June 2003) with greater than 1-in of rainfall. The stormwater samples were taken from the rising limb of the hydrograph and were a composite of two different water levels above the substrate.

General Water Quality Parameters

Dissolved oxygen, specific conductivity, pH, and temperature were measured at the four locations in North Creek and at the two locations in Avent Creek. Parameters were measured using a multiple probe water quality meter (YSI MPS 556) six times between 2003 and 2004 in North Creek. These parameters were measured in Avent Creek once during 2003. The pH probe was calibrated using pH 4.0 and pH 7.0 buffers. The

conductivity probe was calibrated using a standard solution (1000 μS/cm). The dissolved oxygen probe was calibrated to 100% saturation using a barometer.

Benthic Macroinvertebrate Sampling

Benthic macroinvertebrates were collected seasonally in both North Creek and Avent Creek between July 2002 and May 2004. Avent Creek was established as a reference reach, as part of the same Northern Outer Piedmont Ecoregion (classified by NCDENR) and was bioclassified as excellent by the North Carolina Division of Water Quality in the Cape Fear Basinwide Plan (NCDENR-DWQ, 1999). In North Creek, macroinvertebrates were collected in Reach 1 and Reach 2, with the addition of Reach 3 during the summer 2004 sampling. No samples were taken during the summer of 2003 due to large portions of the stream being dry during a drought. In Avent Creek, macroinvertebrates were collected from the 500-ft section of stream with comparable drainage area.

Macroinvertebrates were collected using the North Carolina Standard Qualitative Method. Two kick-net samples were taken in riffles; three dip-net sweeps in undercuts; two leaf packs were elutriated and partitioned; pool sediments were sampled using a fine mesh sampler; and visual removal of animals from cobbles, algal mats, and other coarse particulates was done in each reach. The standard method was used for all samplings except the May 2004 collection that used the Qual 4 method in North Creek (NCDENR-DWQ, 2003). This method is an abbreviated form of the standard method for use in small streams, and was deemed to be more appropriate after initial observations of the community in North Creek. The Qual 4 method includes one kick net, one sweep net, one

leaf pack, and visual collections of macroinvertebrates. For both methods, organisms were sampled in approximate proportion to their abundance, with no more than ten individuals of recognizable taxa collected.

During sampling, the benthic macroinvertebrates were preserved in 95% ethanol and brought back to the lab for identification to the lowest taxonomic level possible. The abundance of each taxon was recorded, using the following categories for number of individuals: Rare (<3); Common (3-9); and Abundant (>9). For most sampling dates, the habitats in which each organism was found were noted. The functional feeding group of each genus was also designated as scraper, collector-filterer, collector-gatherer, predator, or shredder and compared between streams and sample dates.

Benthic Macroinvertebrate Introduction

In April 2003, benthic macroinvertebrates collected in Avent Creek were introduced into Reach 2 of North Creek. Macroinvertebrates were sampled in Avent Creek using the Standard Qualitative Method, modified by sampling all organisms within an allotted fifteen-minute time interval for each collection technique. Pre-sampling of the Avent Creek benthic macroinvertebrate community was performed in the lower half of the 500-ft reach, two days prior to the introduction. Macroinvertebrate sampling of the introduced organisms was then performed in the upper half of the 500-ft reach on the day of the introduction (April 16, 2003) using the same sampling method. Benthic macroinvertebrates collected for introduction were transferred into a large, flat plastic container (1.0-ft W x 2.5-ft L x 0.5-ft H), with stream water and fine organic materials. Large predators were separated into a smaller container to avoid loss of prey. The

transport container was then aerated using a hand pump for the 45-minute trip to North Creek. The macroinvertebrates were transferred into North Creek immediately by pouring a portion of the container sample into each riffle along Reach 2.

Exploratory sampling was performed two weeks and one month after introduction to search for introduced species. All available habitats, including riffles, undercut areas, organic materials, and pools, were sampled in Reach 2. Any organisms found in North Creek that could not be identified in the field to genus level (e.g. Chironomids) were brought back to the lab for further identification.

Biological Habitat Evaluation

Available biological habitat was mapped in each reach of North Creek and Avent Creek using design software (Autodesk, 2003) and digitized layers that showed stream contours, top of banks, and culverts along the reaches. Riffle, pool, run, and undercut lengths were measured in the field and drawn on the map, along with mid-channel and lateral bars, large woody debris, and rip-rapped sections. Comparisons were made between channel feature lengths in each stream.

Available organic substrates were quantified from each reach in North Creek and Avent Creek just after litterfall and during the following spring. Samples were collected within the same week, with no rainfall in the preceding days. Leaf and wood materials were collected or measured along randomly chosen transects within each reach. There were 2 transects in Avent Creek, and ten transects in North Creek for the fall sampling. Due to uneven selection of stream features during the first sampling, the number of transects was increased to five in Avent Creek during the spring sampling and an effort

was made to sample the nearest pre-designated pool or riffle to the stream station selected in both streams. Leaf material was collected from a bottomless 5-gallon bucket placed on the streambed. Samples were collected at ¼, ½, and ¾ the distance along each transect within the wetted perimeter, and placed in labeled paper bags. The leaf material was oven-dried 110 °C for 48hrs, and then weighed to estimate the dry weight of leaf material per area. Stem diameters and lengths of submerged woody debris were measured within one foot above and one foot below the transect line. The total exposed surface area and volume of woody substrate per stream area was then estimated, assuming the materials were cylindrical in shape. Comparisons of leaf material mass and woody debris surface area and volume per stream area were made between North Creek and Avent Creek.

Substrate characterization and analysis was also performed in North Creek and Avent Creek during the study. Pebble counts were collected within cross-sections using the Wolman Method (Wolman, 1954), with collection techniques modified to account for feature proportion (riffle, pool, and run). Pebble count surveys (100 count) were performed in October 2002 in the three reaches of North Creek and in Avent Creek. The median particle size class was determined using the Wentworth particle size classification for the cumulative pebble count survey and for the pebble counts performed within the riffles only. Cobble embeddedness (%) was determined in a variety of riffle plots as an indication of fine sediment accumulation within the riffle substrate. Pool intrusion of fine sediment (%) was also quantified in a variety of pools in each stream. Detailed methods for techniques are described in Chapter 3, with average results reported within this chapter for stream comparisons and macroinvertebrate habitat interpretation.

RESULTS AND DISCUSSION

Stream Water Quality

For all pollutants and general water quality parameters, the number of samples evaluated during baseflows and stormflows was small, and statistical analysis was not performed. The purpose of the sampling was to get a general idea of what variables might be affecting the water quality in North Creek and to provide information for biological assessment interpretations. More water quality sampling will be required to verify trends and to establish pollutant reduction goals for North Creek.

Total Suspended Sediment and Turbidity

Average TSS concentrations and turbidity measurements during baseflow are shown in Table 5.1 for North Creek and Avent Creek. In North Creek, average values for both parameters were relatively low until June 2003. Although the range of values indicated that high levels were present throughout North Creek, the values decreased moving downstream from the Avent Ferry Road culvert. Observations made during the June 2003 sampling indicated that turbid discharge was entering from the Avent Ferry Road culvert. The source of this discharge was not specifically identified, but further investigation in the following days led to the conclusion that the discharge was derived from an upstream development site during release of stormwater retained within a sediment detention basin. Although there is no water quality standard for TSS concentrations, turbidity measurements in the June 2003 samples exceeded the North Carolina water quality standard of 50 NTU (NCDENR-DWQ, 2004).

Baseflow samples collected in Avent Creek had lower average TSS concentrations and turbidity measurements compared to North Creek on comparable sampling dates. The number of samples collected in Avent Creek was less than the number collected in North Creek; therefore, a consistent trend could not be established between the two streams.

The average monthly TSS concentrations and average monthly turbidity measurements were determined using storm events sampled during the same months as baseflow samples were collected. There were 2-5 storm events within each month evaluated. The average monthly TSS concentrations (Figure 5.1), and average monthly turbidity measurements (Figure 5.2), were consistently higher during stormflows compared to baseflows in North Creek. The sampler at S6 was not installed during August 2002 due to sewer line construction by the City of Raleigh, so no data were available at this location. Average turbidity values were greater than the 50 NTU standard during all storm events evaluated (NCDENR-DWQ, 2004). The elevated stormflow sediment values were expected due to observations and grab sample data from stormwater outfall channels (SOCs) entering along North Creek (See Chapter 4, Figure 4.2). Average TSS concentrations and average turbidity values were elevated in the stormwater runoff, particularly in the upper half of the watershed where most of the development was occurring. These development sites (37 ac) are shown in Figure 1.1 (See Chapter 1), and their relative footprints are given in Table 1.2. Many other studies have also shown that development within the watershed can elevate TSS concentrations in the receiving streams (Reed, 1980; Wolman and Schick, 1967; Goldman *et al*., 1986). The elevated stormflow sediment values were also expected due to the streambank erosion measured during this

study (See Chapter 2). Stormflow TSS and turbidity measurements continued to be elevated moving downstream even when the SOC suspended sediment values declined, providing further evidence of streambank erosion influence on suspended sediment quantities during stormflows. Other studies in urban settings found similar results, even after watershed development had ceased (Whipple *et al*., 1981; Lenat and Crawford 1994)

Table 5.1: Average Total Suspended Sediment (TSS) Concentrations and Turbidity Measurements for North Creek (N = 20), and Avent Creek (N = 4) During Baseflow, with Range of Values in Parentheses.

North Creek		
Date	**TSS (mg/L)**	**Turbidity (NTU)**
7/24/2002	11 (9 – 13)	31 (18 – 39)
10/18/2002	9 (7 – 11)	25 (21 – 29)
1/24/2003	29 (10 – 55)	13 (7 – 24)
3/13/2003	19 (4 – 33)	37 (15 – 56)
6/28/2003	115 (76 – 186)	120 (99 – 148)
Avent Creek		
Date	**TSS (mg/L)**	**Turbidity (NTU)**
8/7/2002	1	6
3/26/2003	6	7

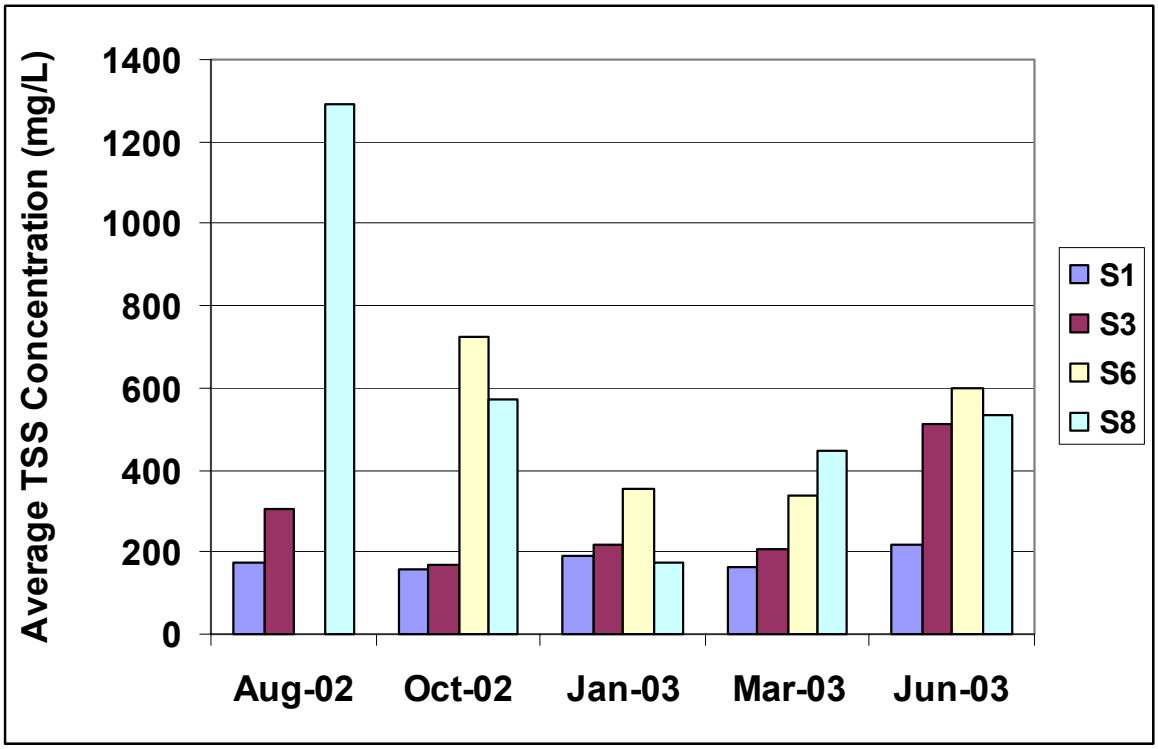

Figure 5.1: Average Monthly Total Suspended Sediment (TSS) Concentrations in North Creek During Storm Events. Value Missing for S6, August 2002

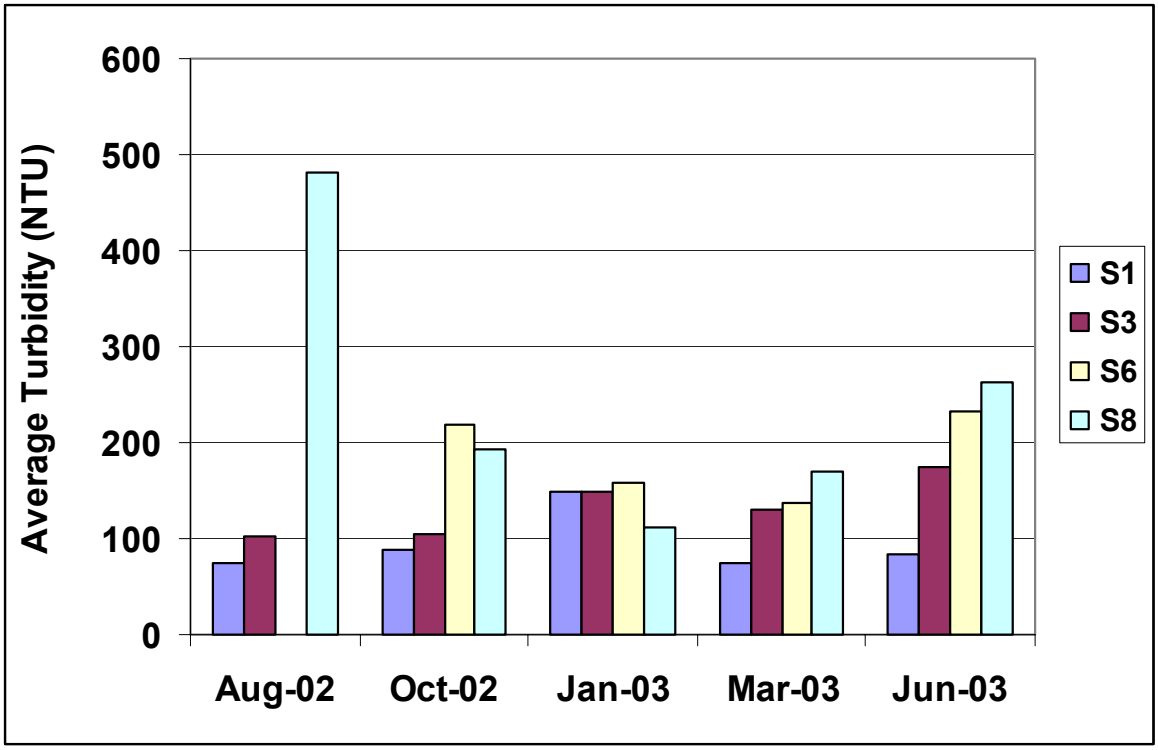

Figure 5.2: Average Monthly Turbidity Measurements in North Creek During Storm Events. Value Missing for S6, August 2002.

Nutrients

Average nutrient concentrations measured in North Creek and Avent Creek are presented in Table 5.2, based on samples collected during baseflow on five dates, and samples collected during two storm events. Concentrations were variable for all three nutrients, with no trend observed between sampling locations or between sampling dates for both stormflow and baseflow measurements. There was little difference between stormflow and baseflow average nutrient concentrations in North Creek, based on the range of measured values. North Carolina does not have a water quality standard for nutrients, as given by a required stormflow concentration in Class C waters. The average

nutrient concentrations measured were typical of stormwater values observed in other studies (USEPA, 1983, Line *et al*., 1997), except for average nitrate-nitrogen concentrations that were considerably lower than the average values measured in urban stormwater. A comparison of North Creek values with reference stream values during baseflow showed that there was also little difference between the two streams for the samples collected.

The nutrient concentrations measured during storm events in North Creek were expected to be greater than nutrient concentrations measured during baseflow due to the large amount of landscaped area on campus and on-going watershed development. Any excess fertilizer, along with suspended sediment, can be collected by stormwater runoff and delivered to the stream for elevated nutrient concentrations during stormflows (Schueler, 2000). There was indirect biological evidence during the study of increased nutrient amounts in North Creek during baseflow, including high concentrations of "sewage fungus" and algae. Sewage fungus is a collection of microorganisms common in streams with high nutrient and organic loadings (Hynes, 1960). Sewage fungus and bacterial presence (orange and metallic sheens) were also noted during baseflow near exposed sewer line portions in Reach 1 SOC1 and in the lower portion of Reach 3 near the S8 cross-section. The biological presence suggested that nutrients and reduced iron were available from either groundwater or sewer line leakage, which was not confirmed during this study. The results of the baseline nutrient sampling were inconclusive, due to the high variability and limited samples.

Table 5.2: Average Nutrient Concentrations (mg/L) in North Creek and Avent Creek, with Range of Values in Parentheses. For Two Stormflow Events, N = 8; For Two Baseflow Events, N = 8 for North Creek and N = 4 for Avent Creek.

	Ammonium-N	Nitrate-N	Orthophosphate
North Creek (Stormflow)	0.2 (<0.02 - 0.7)	0.4 (0.1 - 0.9)	0.2 (0.1 - 0.5)
North Creek (Baseflow)	0.1 (<0.02 - 0.4)	0.4 (<0.02 - 0.8)	0.1 (<0.02 - 0.4)
Avent Creek (Baseflow)	<0.02(<0.02)	0.4 (0.3 - 0.4)	0.2 (<0.02 - 0.4)

Metals

Average metal concentrations (mg/kg soil) measured in North Creek during two storm events and once during baseflow are shown in Table 5.3. There were many inconsistencies in the metal sampling; therefore, the results could only yield information about the metals potentially present and elevated within the system. Four metals were only measured during the second storm event sampled, so their concentrations only reflect results from one storm event. Comparisons between stormflow and baseflow concentrations were not feasible for seven of the twelve metals, due to the difference in thresholds for minimum concentrations or due to lack of measurement. Average metal concentrations during storm events were also not comparable between sampling locations, due to a compositing error during analysis for the first storm when samples were combined from all four locations. The second storm was the only event where sample data were evaluated at each location separately.

The results of the samples collected indicated that at least three of the metals (Cu, Pb, and Ti) were elevated compared to baseflow (Table 5.3). Zinc (Zn) was also elevated during both baseflow and stormflows. Of these four metals, copper, lead, and zinc have been shown to negatively affect aquatic organisms at the concentrations observed within the sediment (Grumiaux *et al*., 1998; Richardson and Kiffney, 2000). There were no studies found that evaluated titanium, but this is a relatively new metal of concern for research, associated with vehicular products (Dr. Wayne Robarge, NCSU Soil Science Department, personal communication).

Since metal concentrations were measured per mass of suspended sediment, the North Carolina water quality standards could not be used to evaluate metal concentrations because these standards were based on concentrations in the water column. Table 5.3 gives average probable effects levels (PELs) for certain metals evaluated by the USEPA and NOAA for sediment quality criteria (Ingersoll *et al*., 2000; Long and Morgan, 1991). The PEL standards were determined from laboratory tests on certain macroinvertebrates, and indicate relative sediment effect concentrations (SECs) for the metals evaluated. Most all of the metals measured in North Creek either approached or surpassed the PELs. Metals such as titanium and vanadium did not have standards established at this time. The levels measured for most of the metals assessed in North Creek were comparable to other urban stream sediments evaluated in the United States, and potentially indicate contamination (Rice, 1999).

Table 5.3: Average Metal Concentrations in North Creek and Water Quality Standards for Metals, with Range of Values in Parentheses. For Storm Events, N = 5, and for Baseflow Events, N = 4.

Metals	Average Stormflow Concentrations (mg/kg)	Average Baseflow Concentrations (mg/kg)	Standard Concentrations (mg/kg)
Arsenic (As)	<3*	<30 (<17 - <50)	33[1]
Antimony (Sb)	<4*	<30 (<17 - <50)	25[2]
Beryllium (Be)	2 (2 - 3)	<30 (<17 - <50)	
Cadmium (Cd)	1* (0.5 - 2)	<30 (<17 - <50)	5[1]
Copper (Cu)	177 (169 - 183)	109 (63 – 194)	150[1]
Lead (Pb)	68 (43 - 127)	41 (28 - 52)	130[1]
Nickel (Ni)	19 (15 - 27)	103 (90 – 117)	49[1]
Zinc (Zn)	314 (236 - 459)	623 (490 - 800)	460[1]
Selenium (Se)	<8*	<30 (<17 - <50)	
Titanium (Ti)	975 (872 - 1141)	578 (70 - 982)	
Vanadium (V)	97 (81 - 110)	Not Measured	
Chromium (Cr)	38 (30 - 57)	Not Measured	110[1]

* Metal only measured during the second storm event
[1] Probable Effect Concentrations, Ingersoll *et al*., 2000 (USEPA).
[2] Probable Effect Concentrations, Long and Morgan, 1991 (NOAA).

Organic PAHs

Polycyclic aromatic hydrocarbons (PAHs) are organic compounds that can be derived from a variety of watershed activities, including fuel combustion, industrial emissions, and vehicular emissions (McIntosh *et al*., 2000). Even though regulations have improved industrial emissions, PAHs in urban sediments have remained elevated due to the increase in vehicle traffic (Van Metre *et al*., 2000). The average PAH concentrations measured from composite samples during two storm events and one baseflow sampling in North Creek are presented in Table 5.4. Average stormflow concentrations were higher than average baseflow concentrations for all PAHs measured, except for those that had concentrations below detectable limits in both sample types. In North Carolina, there is no standard for PAHs in Class C waters. The average concentrations measured were elevated above detectable limits for all of the 53 PAH compounds measured. For baseflow samples, concentrations were all relatively low (below detectable limits), with only naphthalene and C1-naphthalenes surpassing the standard.

Due to the importance of the USEPA 16 priority pollutant PAHs and their potential carcinogenic effects (USEPA, 1984), these concentrations are commonly summed for the best indication of toxicity potential. In North Creek, the total concentration of these organic compounds was very high (Table 5.5), particularly near the Avent Ferry culvert (S1). Comparing concentrations seasonally, summer total values were found to be typically greater than winter total values. Whatever the causes and extent of PAH variability, the sampling showed that most of the PAH concentrations measured were high, especially considering the toxic effects these compounds could have together and during

exposure to sunlight (Hatch and Burton, 1999; Verrheist *et al*., 2001), and their affinity for sediment storage (ATSDR, 1990). Further sampling should be performed to measure these toxic organic compounds, based on the preliminary findings.

Table 5.4: Average Polycyclic Aromatic Hydrocarbon (PAH) Concentrations in North Creek, with Range of Values in Parentheses. Bold PAHs Indicate USEPA 16 Priority Pollutant PAHs. The "bdl" Designation Indicates Below Detectable Limits. For Stormflow Samples, N = 7; For Baseflow Samples, N = 4.

Polycyclic Aromatic Hydrocarbon (PAH)	Average Stormflow (ηg/L)	Average Baseflow (ηg/L)
1-methylfluorene	4 (bdl - 13)	bdl
1-methylnaphthalene	8 (1 - 28)	1
1-methylphenanthrene	50 (bdl - 56)	bdl
2,3,5-trimethylnaphthalene	bdl	bdl
2,6-dimethylnaphthalene	23 (4 - 65)	bdl
2-methylnaphthalene	11 (bdl - 40)	2
acenaphthene*	24 (bdl – 139)	bdl
acenaphthylene*	2 (bdl - 14)	bdl
anthracene*	10 (bdl – 53)	bdl
benz[a]anthracene*	92 (bdl - 623)	bdl
benzo[a]pyrene*	129 (bdl – 875)	bdl
benzo[b]fluoranthene*	188 (bdl - 1297)	bdl
benzo[e]pyrene	136 (0.1 – 930)	bdl
benzo[g,h,i]perylene*	124 (bdl - 828)	bdl
benzo[k]fluoranthene*	162 (bdl – 1122)	bdl
biphenyl*	1 (bdl - 4)	bdl
C1-chrysenes*	40 (bdl - 280)	bdl
C1-dibenzothiophenes	36 (15 - 65)	bdl
C1-fluoranthenes/pyrenes	89 (bdl - 588)	bdl
C1-fluorenes	7 (bdl - 22)	bdl
C1-naphthalenes	24 (2 - 72)	4
C1-phenanthrenes/anthracenes	132 (bdl - 543)	bdl
C2-chrysenes	24 (bdl - 166)	bdl
C2-dibenzothiophenes	59 (15 - 116)	bdl
C2-fluorenes	bdl	bdl
C2-naphthalenes	54 (bdl - 219)	bdl
C2-phenanthrenes/anthracenes	66 (bdl - 341)	bdl

Table 5.4 (Continued)

C3-chrysenes	14 (bdl - 97)	bdl
C3-dibenzothiophenes	67 (bdl - 130)	bdl
C3-fluorenes	bdl	bdl
C3-naphthalenes	17 (bdl – 69)	bdl
C3-phenanthrenes/anthracenes	22 (bdl - 142)	bdl
C4-chrysenes	bdl	bdl
C4-naphthalenes	6 (bdl - 44)	bdl
C4-phenanthrenes/anthracenes	bdl	bdl
chrysene*	49 (3 – 278)	bdl
coronene	26 (bdl - 182)	bdl
dibenz[a,h]anthracene*	24 (bdl – 165)	bdl
dibenzofuran	15 (bdl - 72)	bdl
dibenzothiophene	25 (5 – 70)	bdl
fluoranthene*	432 (4 - 2785)	bdl
fluorene*	4 (bdl - 21)	bdl
indeno[1,2,3-c,d]pyrene*	132 (bdl - 918)	bdl
naphthalene*	18 (5 – 41)	bd1
perylene	33 (bdl - 189)	bdl
phenanthrene*	141 (bdl – 824)	bdl
pyrene*	273 (bdl - 1809)	bdl
retene	1 (bdl – 5)	bdl

Table 5.5: The Total Average Concentrations (ηg/L) for the 16 EPA Priority Pollutant PAHs Added Together at the Four Sampling Locations in North Creek.

Sample Location	Stormflow Summer 2003	Stormflow Winter 2003	Baseflow
S1	11649	375	4
S3	46	274	3
S6	39	79	3
S8	177	N/A	2

General Water Quality Parameters

General water quality parameters measured in North Creek and Avent Creek are presented in Table 5.6 for DO, conductivity, pH, and temperature. These variables were expected to show impaired water quality, similar to the pollutant concentrations based on urban watershed activities. All parameters measured were within the acceptable ranges, however, for the North Carolina water quality standards (NCDENR-DWQ, 2004) and other established values within the literature. There were no site-specific trends observed in North Creek between sampling locations, and the range of values measured on each sampling date shows that there is variability along the reach. Ranges were omitted in Avent Creek where no variability was observed.

Dissolved oxygen concentrations showed a negative relationship with temperature seasonally, with the lowest values recorded during the highest temperatures in the summer of 2003. Average measurements taken after January 2003 were all below the 10-12 mg/L range, which is the expected concentration in streams with adequate flow and saturated conditions (Hart, 1974). Although dissolved oxygen concentrations were above the

minimum requirement (4 mg/L), measurements approached the average 5 mg/L standard in the summer and fall (NCDENR-DWQ, 2004). These low values were probably due to stagnant water in the pools measured, elevated temperatures, and the presence of high sewage fungus concentrations (with their high respiration rates) during these times (Hynes, 1970). These low concentrations suggest some biological stress could be present within North Creek for fish and benthic macroinvertebrate respiration. DO levels in Avent Creek were near saturation, as expected for a stable stream in an undisturbed watershed.

Average conductivity measurements were greater in North Creek compared to Avent Creek, but all values were in the acceptable range (0.15-0.50 mS/cm) for biological health. There was no distinct change over time in conductivity during baseflow to indicate pollutants, but conductivity was expected to be higher due to the clay mineralogy of the channel (USEPA, 2001). Measurements during storm events might have provided more information about the effects of pollutants on conductivity.

The average pH values measured were between 6.8 and 7.7, well within the acceptable range. Errors occurred with the pH probe in January 2003 and March 2003, and the data were deemed not reliable. The average pH values were slightly higher in North Creek compared to Avent Creek, but differences were not certain due to different sampling dates. The pH values were expected to deviate from neutral more than the observed values due to pollutant influence (USEPA, 2001), but there was not enough evidence with the data collected during baseflow. Soils with relatively high clay contents can contribute relatively acidic pH values to water flowing through them (Brady, 1990). The average pH values measured in Avent Creek were slightly acidic, presumably due to clay-dominated

soils, and relatively comparable to North Creek. Samples collected during stormflows might have provided more information on watershed stormwater runoff effects.

Temperature measurements were all within the expected range for the season and comparable in both streams on the shared sampling time period. Temperatures were not impaired during the study even during stagnant water conditions, probably due to the riparian buffer strip left along most of the North Creek reach (Hynes, 1960).

Table 5.6: Average General Water Quality Parameters in North Creek and Avent Creek, with Ranges in Parentheses. For North Creek, N = 12 on Each Sampling Date; For Avent Creek, N = 6.

North Creek				
Date	**Dissolved Oxygen (mg/L)**	**Conductivity (mS/cm)**	**pH**	**Temperature (ºC)**
1/22/2003	10.6 (9.7 - 11.2)	0.2 (0.1 - 0.2)	7.1 (6.5 - 7.0)	5.7 (4.4 - 8.1)
1/31/2003	10.7 (9.7 - 11.9)	0.2 (0.2 - 0.3)	N/A	6.4 (5.5 - 7.8)
3/13/2003	8.3 (6.8 - 10.8)	0.2 (0.2)	N/A	14.8 (12.4 - 17.4)
6/14/2003	6.4 (5.1 - 7.6)	0.1 (0.1 - 0.2)	6.7 (6.7 - 6.8)	25.0 (23.8 - 26.1)
9/17/2003	8.5 (5.8 - 11.6)	0.1 (0.1 - 0.2)	7.3 (6.5 - 8.4)	22.1 (19.6 - 23.6)
2/25/2004	7.7 (7.4 – 8.0)	0.2 (0.2 - 0.3)	7.7 (7.5 - 7.9)	7.1 (6.6 - 7.9)
Avent Creek				
Date	**Dissolved Oxygen (mg/L)**	**Conductivity (mS/cm)**	**pH**	**Temperature (ºC)**
3/22/2003	10.7	0.1	6.6	14.5

Benthic Macroinvertebrate Sampling

The genera found in the streams during each season sampled are provided in data tables in Appendix 5, with North Creek data shown in Table A5.1-Table A5.5 and Avent Creek data shown in Table A5.6. All organisms were insects except for a few genera that were common listed at the bottom of the tables. The functional feeding groups are listed for each genus, as well as the relative abundance (Rare, Common, or Abundant) and habitats where the organisms were collected (Riffle, Undercut, and Leaf Packs). Pool habitat was not noted due to the small number of individuals collected in this habitat over the two-year study in both streams. Benthic macroinvertebrates collected in pools were accounted for, however, in the relative abundance values.

The benthic macroinvertebrate genera data for each sampling event show that the community in North Creek was dominated by chironomids during each sampling event. Chironomids (Diptera) were greatest in proportion of genera (27-75%) for all sampling dates, and had the greatest relative abundance on all sampling dates, except the summer of 2002. The dominant taxa in Avent Creek were different depending on season, and taxa were spread across the insect orders for large diversity as evidenced by the following low proportions for dominant taxa. In summer, beetles were dominant (Coleoptera 27%); in fall and winter, dipterans were dominant (Diptera 28% and 23% respectively); and in spring, mayflies were dominant (Ephemeroptera 23%). Although the numbers of dipteran genera were greatest during the fall and winter, they were not considerable in abundance compared to the other taxa.

The predominant habitat type utilized in North Creek was variable between sampling dates and reaches. In fall the majority of benthic macroinvertebrates were found in the undercut regions (75%). In the spring of 2003, the majority were found in the riffles (83%), but there was no consistent habitat preferred during the spring of 2004. In winter, the predominant habitat was undercut (86%) in Reach 1 and leaf packs (60%) in Reach 2. The undercut habitat appears to be the most common resource, with the majority of organisms in at least one reach found here during all four seasons evaluated. In Avent Creek, the predominant habitat was riffles with 68-75% of the genera found here throughout the study.

A summary of benthic macroinvertebrate diversity and relative abundance is provided in Table 5.7. Taxa richness was greater in Avent Creek compared to North Creek for all sampling dates. There were 2-10 times more genera found in Avent Creek, except during the fall. The proportion of EPT genera to total number of genera was also considerably higher in Avent Creek (28-55%) compared to North Creek (0-20%). The relative abundance of individuals was estimated using the numbers assigned to each abundance category (Rare = 3, Common = 6, and Abundant = 10). The relative abundance was much higher in Avent Creek, with 3-10 times more individuals collected in Avent Creek except during the fall. The relatively low number of individuals and genera collected during the fall of 2002 in Avent Creek was unusual compared to other seasons, and could not be explained. Average number of genera for the entire North Creek ranged from about 9-15, with a small decline observed over time between 2002 and 2004. Average relative abundance ranged from 19-91 individuals, with the addition of Reach 3 during the spring of 2004 contributing the highest

number. There was no observed trend in average relative abundance over time for these sampling dates.

The benthic macroinvertebrate community characteristics in North Creek were similar to other urban stream studies (Duda *et al*., 1979, Benke *et al*., 1981, Jones and Clark, 1987). Chironomids were dominant due to their known tolerance to pollutants and their ability to effectively adapt to limited resources and reduced habitat diversity (Epler, 2001, Merritt and Cummins, 1996). The benthic macroinvertebrate community in Avent Creek was shown to be diverse and organisms abundant, comparable with the NCDWQ biological assessments made in 1999 (Cape Fear River Basinwide Plan; NCDENR-DWQ, 1999).

Benthic macroinvertebrate functional feeding group (FFG) assemblages were compared in both streams during the seasons for all reaches combined (Figure 5.3). Avent Creek was expected to have a greater diversity in FFGs due to the diversity in taxa present, while North Creek was expected to have limited FFG distribution (Merritt and Cummins, 1996). The results of this study indicated, however, that there was no distinct difference in functional feeding group assemblage between the two streams. For both streams, no more than 40% of the genera were in one FFG or another, and the majority of genera were collector-gatherers or predators. In North Creek, collector-gatherers were dominant in the winter and spring, while predators were dominant in summer and fall. In Avent Creek, collector-gatherers were greatest for all seasons except summer where proportions of collector-gatherers and predators were equal. There was also no trend in FFG with smaller proportions of genera. These FFGs, including scrapers, shredders, and collector-filterers were all relatively low in North Creek. In Avent Creek, the collector-filterers were the

smallest proportion for all seasons except during the fall when shredders were lowest just before litterfall. In Avent Creek, the proportion of shredders increased in winter after litterfall, but decreased in North Creek.

The reasons for the lack of differences in FFG assemblages between the two streams are unclear. Explanations could include variables such as size, substrate type, the nature of chironomid tolerance in North Creek. Stream size and substrate are known to affect FFG assemblage (Cummins, 1974). Both streams had comparably small wetted perimeters during baseflow, and relatively small substrate in the riffles (gravel particle sizes) compared to cobble-bed streams. Greater differences in FFGs would probably be observed if size and substrate were more variable. Scrapers were limited in both streams as a result of periphyton communities, but for potentially different reasons (Cummins, 1988). In Avent Creek, the extent of periphyton was limited due to the closed dense canopy over the small stream. In North Creek, the canopy was open in portions due to riparian disturbances but the particular periphyton communities that are usually present in polluted streams (increase in filamentous algae) are not as suitable food resources for benthic macroinvertebrates (Cummins, 1992). Due to the presence of a thick canopy in Avent Creek, the number of shredders was also expected to be greater than observed. The lack of shredders is difficult to explain, since organic matter was observed to be readily available.

The FFG assemblage in North Creek was expected to be dominated by collector-gatherers and collector-filterers, due to their predominance in urban streams with greater fine particulate food resources (Cummins, 1992). Chironomids have been shown to be a significant component of these two FFGs in urban streams (Jones and Clark,1987; Lenat and

Crawford, 1994). The relatively large amount of chironomids in North Creek may also have affected FFG composition, but in a different manner than expected. Although chironomids were expected to be predominantly collector-gatherers, these organisms are also known for their capacity to out-compete other taxa due to their versatility in food resource utilization (Pinder, 1986; Merritt and Cummins, 1996). The chironomid genera present in North Creek were very tolerant, highly versatile, and present in a variety of FFGs, which probably influenced the FFG assemblage diversity (Appendix 5, Table A5.1-Table A5.5).

Table 5.7: Taxa and Relative Abundance of Benthic Macroinvertebrates in North Creek and Avent Creek Between 2002-2004. Abundance Estimated by Summing Rare (3), Common (6), and Abundant (10) Categories.

		Sample Year	Number of Total Genera	Number of EPT Genera	Individual Abundance (Estimated)
Summer (Jul-Aug)	North Creek, Composite Reach	2002	15	3	19
	Avent Creek	2002	30	10	127
Early Fall (Oct)	North Creek, R2 only	2002	12	2	88
	Avent Creek	2002	18	6	96
Spring (Apr-May)	North Creek, R1	2003	6	0	38
		2004	4	0	72
	North Creek, R2	2003	12	0	78
		2004	10	0	79
	North Creek, R3	2004	12	2	121
	Avent Creek	2003	40	21	221
Winter (Dec)	North Creek, R1	2003	7	0	37
	North Creek, R2	2003	10	1	40
	Avent Creek	2002	40	18	330

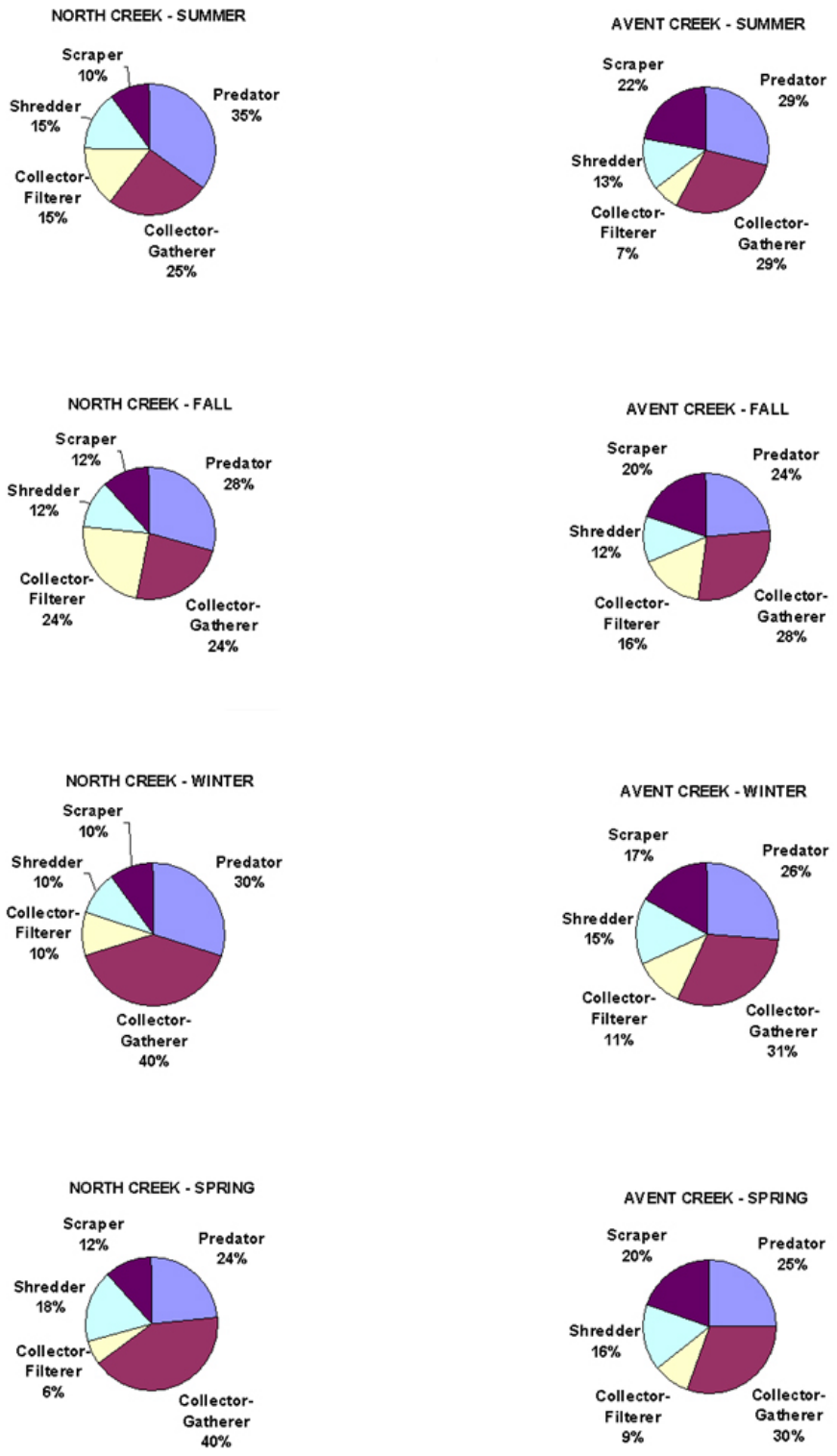

Figure 5.3: Comparison of Benthic Macroinvertebrate Functional Feeding Groups in North Creek and Avent Creek Between 2002-2004.

Benthic Macroinvertebrate Introduction

The benthic macroinvertebrates collected during the April 2003 sampling in Avent Creek were presumed to be the organisms transferred into Reach 2, North Creek. The sampling time per each collection technique was kept constant between the pre-introduction sampling and the collection of introduced organisms. Based on the pre-sampling, it is estimated that about 220 benthic macroinvertebrates were collected and transferred to North Creek.

The results of the study indicated that the benthic macroinvertebrate introduction into North Creek from Avent Creek was not detectable, based on the absence of signature individuals from North Creek following introduction. Extensive searches carried out two weeks and one month after introduction in all habitats of Reach 2 did not reveal any characteristic individuals present in Avent Creek but not present in North Creek. Organisms found in North Creek were restricted to the dipterans, particularly *Tipula* and chironomids. Chironomids were collected alive and brought back to the lab for identification. *Lopescladius* and *Paratendipes* found in Avent Creek were not present in the North Creek collections. *Polypedilum* was collected from North Creek, but was characteristic of both streams and not conclusive.

The number of introduced individuals was expected to decline over time in North Creek, but the complete disappearance after only a few weeks was not expected. There are several potential reasons why the introduction was not successful, and several factors that may have caused the rapid loss of the introduced taxa. Although the post-introduction sampling was extensive, organisms might still have been missed during collection. The

elevated pollutants and streamflows during storm events may have also caused the introduced macroinvertebrates to drift (Merritt and Cummins, 1996). During the two weeks after release, there were three significant rainfall events (0.15, 0.62, and 0.34 in). Turbid stormwater was also released into Reach 2 during this time period from SWMP #2 located mid-reach during maintenance and stormflows (See Chapter 1, Figure 1.1). Although turbidity measurements were not measured during the two weeks, values from this pond have ranged between 56 and 446 NTU at other sampling times. Organisms could have drifted passed the Research Drive culvert, which was not sampled. Another factor affecting benthic macroinvertebrate survival could have been habitat availability. Limited riffle and undercut habitats (discussed in the following section) would have affected the indicator organisms. The EPT taxa typically require clean riffles with little sediment accumulation for effective respiration and food acquisition (Merritt and Cummins, 1996). Other taxa require well-defined undercut habitat for predation (Odonates), and many organisms use the undercut regions for protection during large flows.

Benthic Macroinvertebrate Habitat

The habitat composition in North Creek and Avent Creek was evaluated using habitat feature maps, organic matter sampling, and substrate assessments. Habitat features were mapped for all reaches in North Creek (Figure 5.4-5.7) and in Avent Creek (Figure 5.8). The most noticeable difference between the two streams is the large amount of stream length classified as relatively homogeneous runs in North Creek compared to the heterogeneous riffle/pool sequences in Avent Creek. Avent Creek has more meandering pattern and considerably more undercut and large woody debris habitat per length

compared to North Creek. Another observation from the habitat assessment is the lack of extensive lateral bar formation in North Creek, particularly opposite the pool areas. North Creek shows its urban influence with the large amount of stream length affected by artificial stone additions (termed "rip-rap"). These artificial stones were observed within the riffles, particularly in the lower portion of Reach 3 where the eroded stone-lined channel upstream resulted in deposition of the large stones downstream as the grade was reduced.

The total stream length of each bed feature, including riffle, pool, run, and undercut, was determined in each stream using the thalweg length. Table 5.8 shows the proportion of each feature compared to the total stream length in the study reaches of North Creek and Avent Creek. The proportions indicate that the planform of North Creek is dominated by stretches of runs, while Avent Creek is uniformly divided between riffles, pools, and runs for enhanced habitat diversity. The undercut habitat proportion is over five times greater in Avent Creek compared to North Creek. The enhanced riffle/pool sequences, undercut regions, and large woody debris in Avent Creek were most likely attributable to an undisturbed watershed and a larger, more densely vegetated riparian buffer.

Average organic material quantities per stream area were estimated in both streams during a fall and spring of the study (Figure 5.9). Just after litterfall, the average mass of leaf materials was much larger in North Creek compared to Avent Creek. These results were contradictory to observations of leaf litter quantities in the two streams, and were thought to indicate problems with the sampling method and transect selection. The cross-

sections were in riffles, pools, and runs in North Creek; but were in riffles only in Avent Creek. Organic matter tends to collect disproportionately in pools and along snags in a stream (Allan, 1995); therefore relative sampling of these features could affect results. For the spring sampling, the sampling method was improved by spreading the transects across comparable features in both streams. The transect stations were still selected at random, but sampling of the nearest riffle or pool to the transect was designated before arriving at the streams. The average leaf mass per stream area was about five times in Avent Creek compared to North Creek in the spring.

The results of the woody debris sampling were the same for both seasons and transect selection methods. The estimated average surface area and volume of woody debris was much greater in Avent Creek compared to North Creek during both the fall and spring sampling. Woody debris decreased in North Creek, but increased in Avent Creek between sampling dates.

The amount of organic material present in North Creek was less than the amount in Avent Creek. The limited quantity of coarse organic matter in North Creek was probably due to flashy stormflows combined with a relatively homogeneous channel bed (increased run length) that could reduce retention. Avent Creek retained organic matter better and for a longer period of time after litterfall, due to heterogeneous patterns and large woody debris present to snag coarse materials (Allan, 1995). The riparian buffer was also more extensive in Avent Creek, providing more coarse organic matter input to this stream.

The substrate assessment results for the two streams are summarized in this habitat evaluation, with detailed results of substrate characterization provided in Chapter 3. For

all three methods utilized, the number of samples was limited and changes in hydrology could have affected the results. The median particle size class (d_{50}) was determined for the cumulative pebble counts and for the pebble counts performed within riffles in each reach of North Creek and in Avent Creek (Table 5.10). The cumulative and riffle median particle size class were larger in Reach 3 of North Creek due to artificial lining with stone (12 in dia) in the section surveyed. This section of the reach was dry during baseflow, and not used to compare the two streams.

Both North Creek and Avent Creek were considered gravel bed streams based on the median particle size in the riffles. The riffle median particle size classes were larger in North Creek compared to Avent Creek, while the cumulative median particle size classes are smaller in North Creek compared to Avent Creek. The pebble count results suggested that the substrate in these two streams was comparable for benthic macroinvertebrate habitat evaluations.

The average percent of large particle embeddedness by fines in the riffles was 24.8% in North Creek and 8.9% in Avent Creek (see Chapter 3, Figure 3.12). The particle embeddedness was greater in North Creek, and may be indicative of a larger fine sediment load from development in this watershed (See Chapter 4) and increased streambank erosion (See Chapter 2). The increased particle embeddedness showed that fine sediment was filling the interstitial spaces between particles in North Creek riffles more than in Avent Creek. Increased particle embeddedness has been shown to limit the habitat potential of cobble and gravel substrates (Luedtke and Brusven, 1976) as well as the limit habitat access to the underlying hyporheos zone (Richards and Bacon, 1994).

The average percent of pools filled with fine sediments was 37.2% in North Creek and 25.5% in Avent Creek (see Figure 3.13). The overall difference between the two streams was not as large as expected, however, even though more fine sediment was observed and measured in the water column of North Creek compared to Avent Creek. The lack of extensive fine sediment intrusion in the North Creek pools indicated that the pools had adequate depth for fish habitat (Lisle and Hilton 1999). Fine sediment depths were most likely controlled by the intense stormflows from the developed watershed.

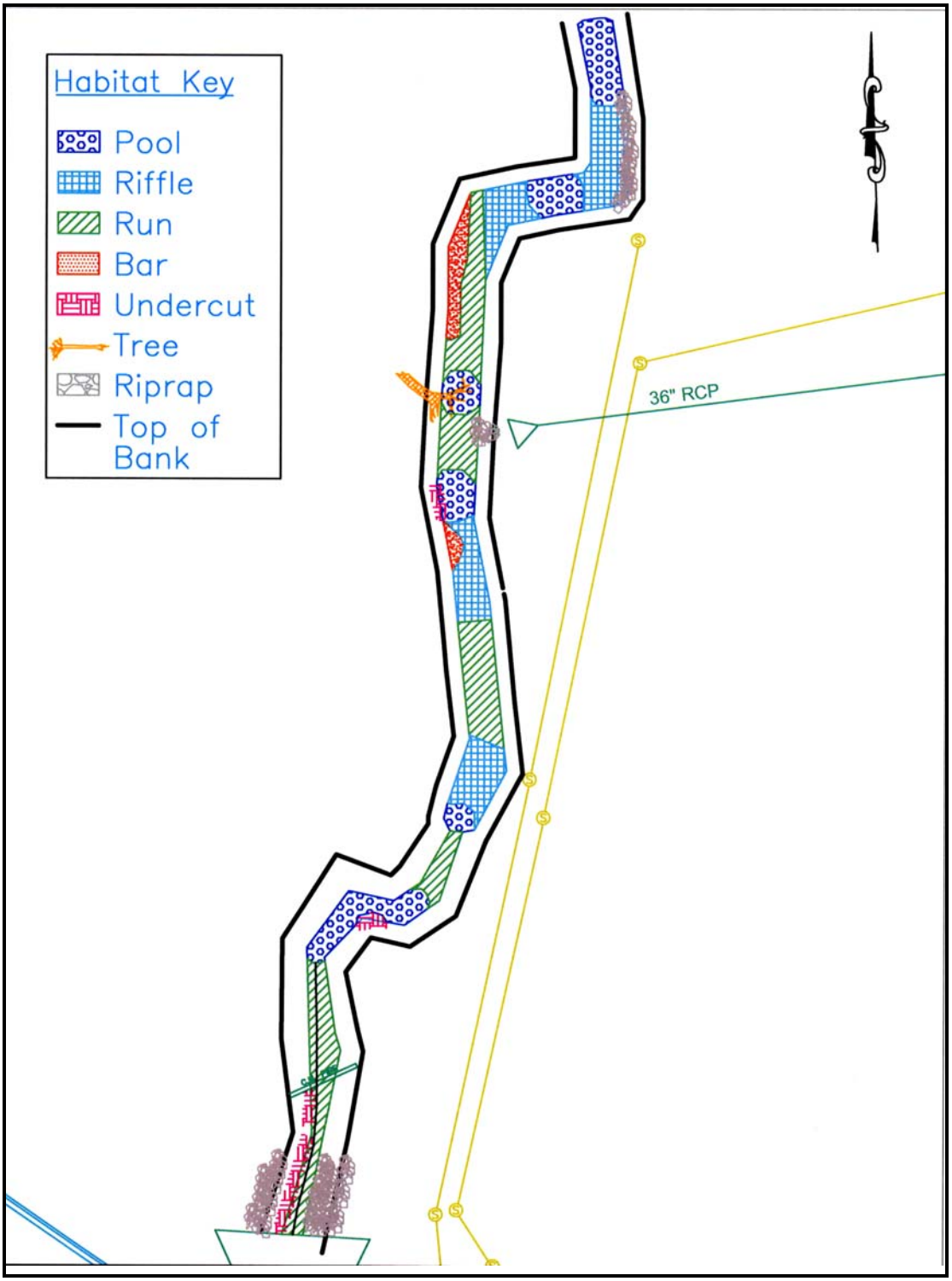

Figure 5.4: Reach 1, North Creek Habitat Assessment Map (Illustrated by Ian Jewell, NCSU Water Quality Group). Scale: 1 in = 60 ft.

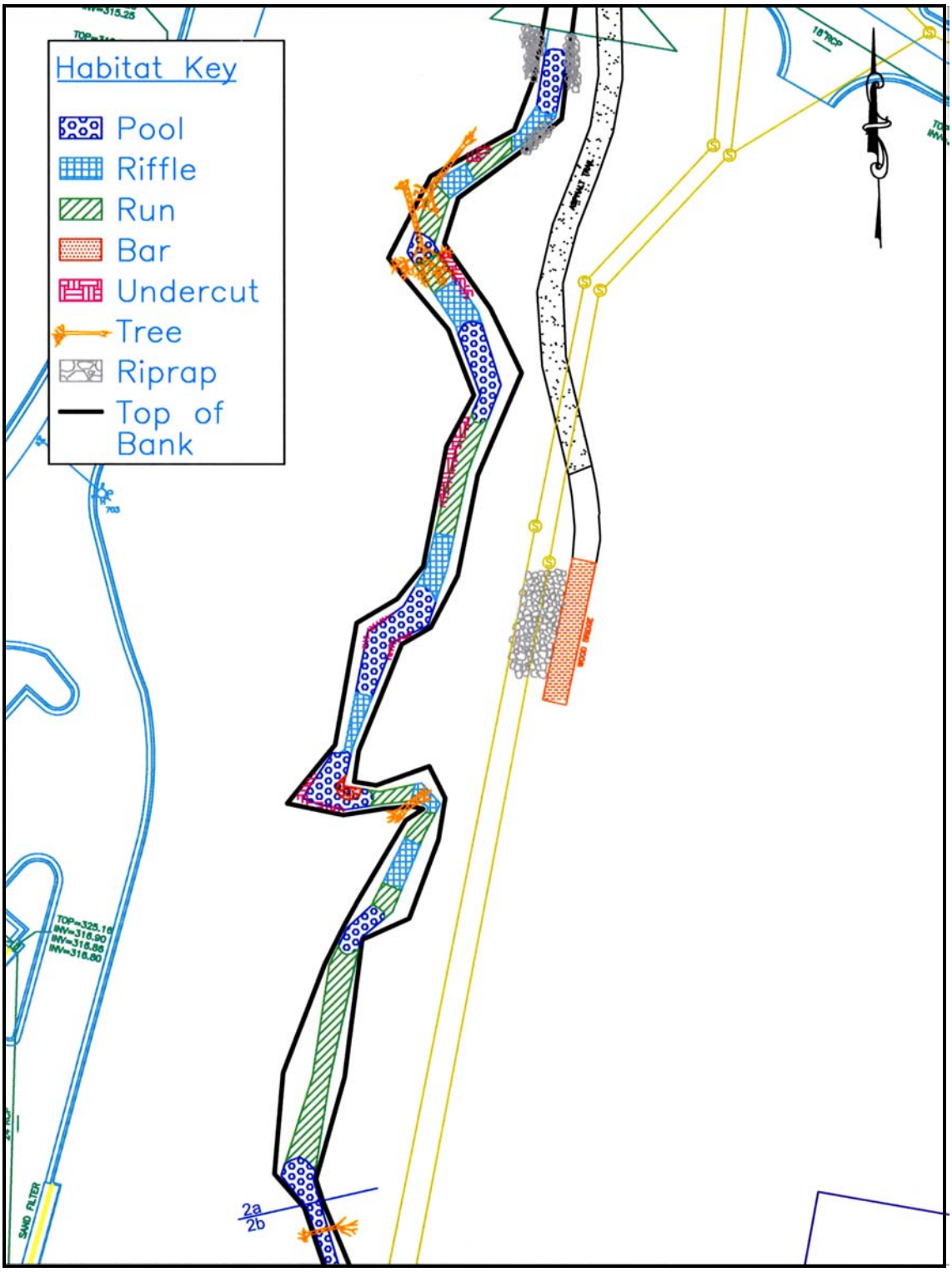

Figure 5.5: Upstream Portion of Reach 2, North Creek Habitat Assessment Map (Illustrated by Ian Jewell, NCSU Water Quality Group). Scale: 1 in = 60 ft.

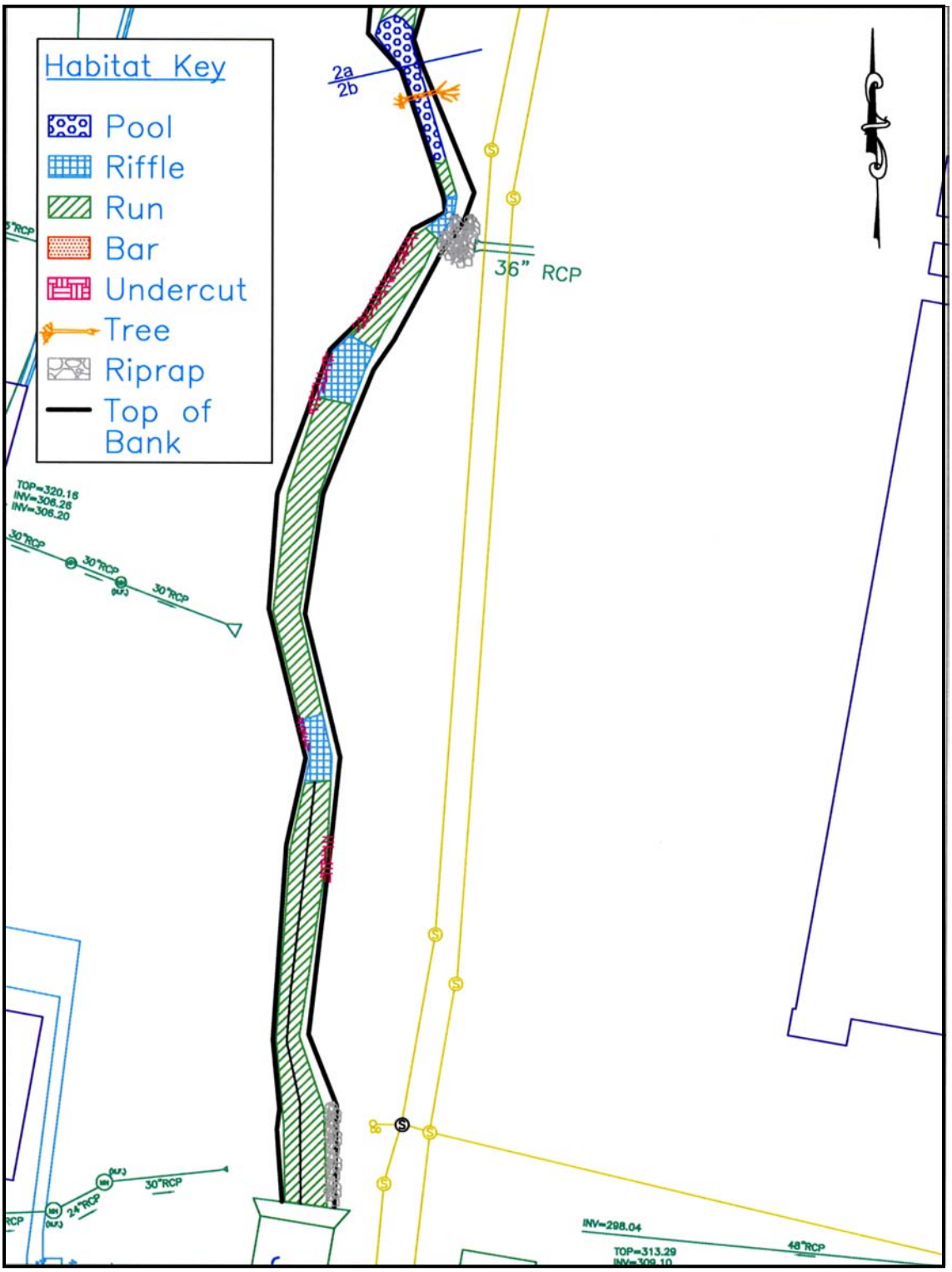

Figure 5.6: Downstream Portion of Reach 2, North Creek, Habitat Assessment Map (Illustrated by Ian Jewell, NCSU Water Quality Group). Scale: 1 in = 60 ft.

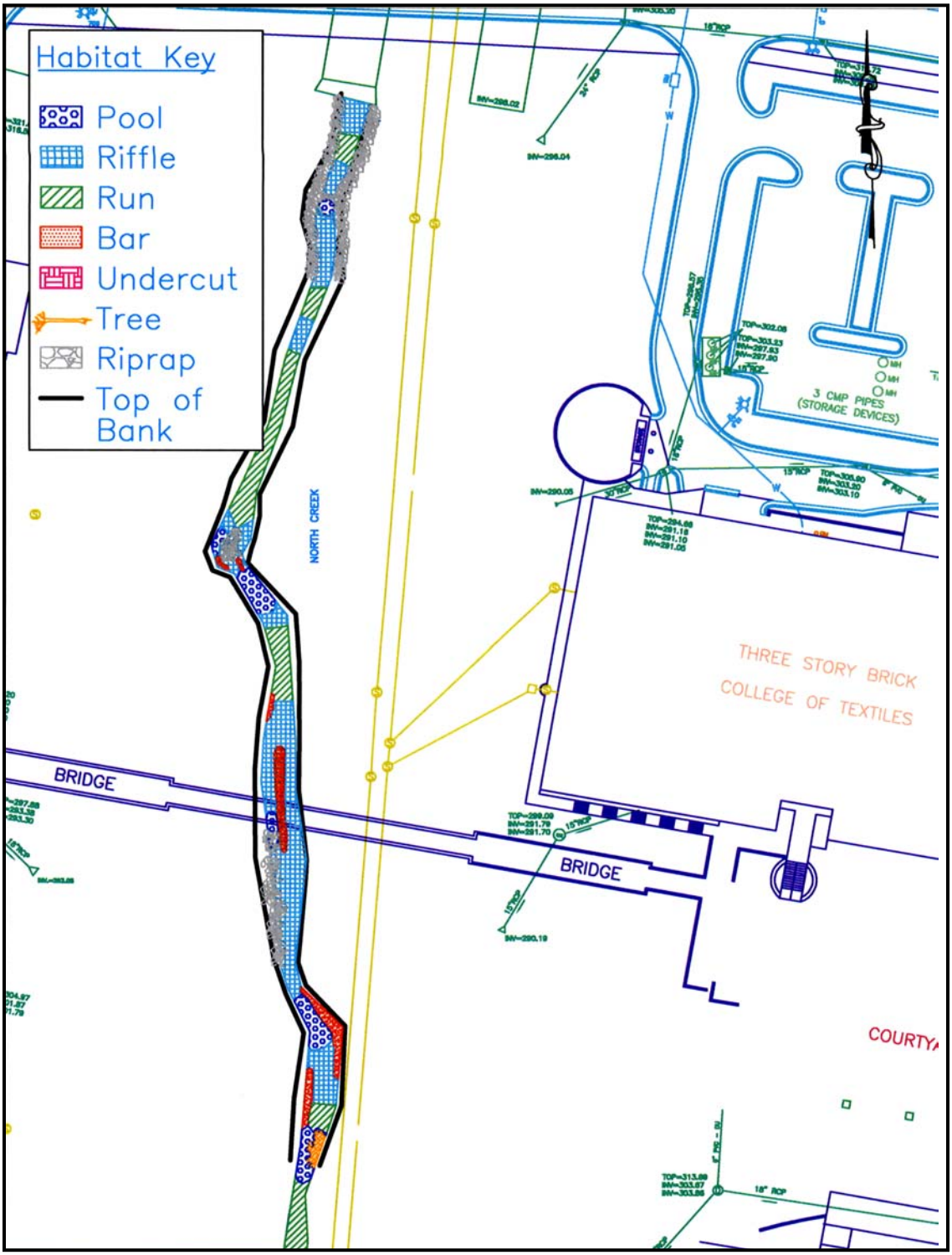

Figure 5.7: Reach 3, North Creek Habitat Assessment Map (Illustrated by Ian Jewell, NCSU Water Quality Group). Scale: 1 in = 80 ft.

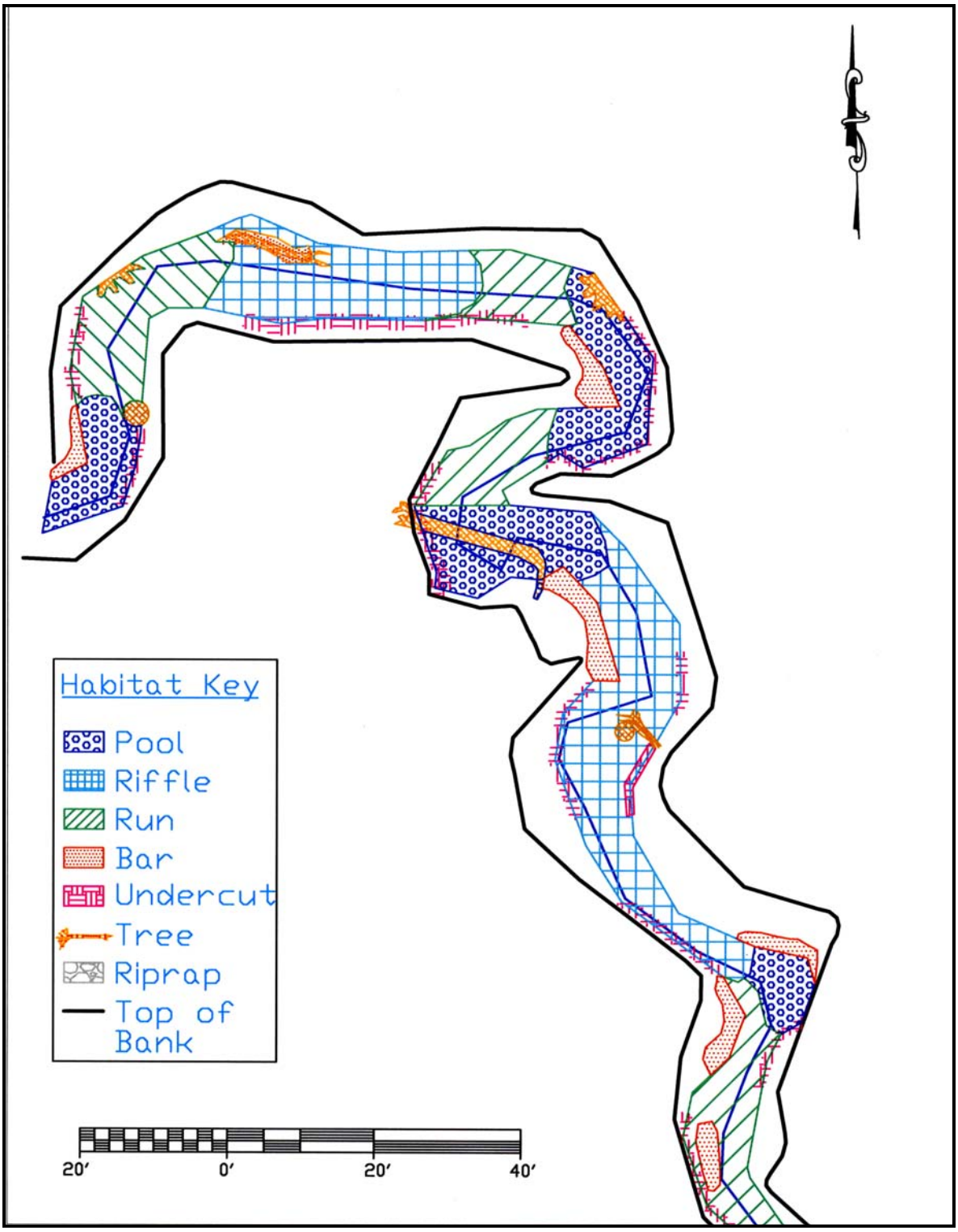

Figure 5.8: Avent Creek Habitat Assessment Map (Illustrated by Ian Jewell, NCSU Water Quality Group). Scale 1 in = 20 ft.

Table 5.8: The Proportion of Channel Feature Habitat Length in North Creek and Avent Creek.

Habitat Feature	North Creek (% Thalweg Length)	Avent Creek (% Thalweg Length)
Riffle	24	34
Pool	23	32
Run	51	34
Undercut	13	71

Table 5.9: Average Organic Matter Quantities per Stream Area (ft^2) in North Creek and Avent Creek during Fall (12/02) and Spring (3/03).

	North Creek	Avent Creek
Material Sampled	*After Litterfall*	
Leaf Material (g/ft^2)	8.99	0.99
Wood Volume (ft^3/ft^2)	0.001	0.02
Wood Surface Area (ft^2/ft^2)	0.02	0.30
	Spring	
Leaf Material (g/ft^2)	0.53	2.63
Wood Volume (ft^3/ft^2)	0.0001	0.04
Wood Surface Area (ft^2/ft^2)	0.01	0.68

Table 5.10: Pebble Count Median Particle Size Classes (d_{50}) for Cumulative Surveys and Riffle Surveys.

Stream	Cumulative (mm)	Riffle (mm)
North Creek, Reach 1	0.5 (Coarse Sand)	11.3 (Medium Gravel)
North Creek, Reach 2	0.062 (Very Fine Sand)	5.7 (Fine Gravel)
North Creek, Reach 3	128 (Large Cobble)	128 (Large Cobble)
Avent Creek	2.0 (Very Fine Gravel)	4.0 (Fine Gravel)

CONCLUSIONS

The preliminary water quality sampling of North Creek demonstrated that the water quality in North Creek was potentially impaired during stormflows based on pollutant concentrations (sediment, metals, and PAHs) measured during this study. High pollutant concentrations were consistent with stormwater runoff concentrations found in other urban stream watersheds, and generally did not meet the water quality requirements established by the North Carolina standards for Class C freshwaters (NCDENR-DWQ, 2004). The results indicated that further sampling would be necessary for all pollutants evaluated, since all were elevated during the study. The general water quality parameters were all within an acceptable range for biological health and for the North Carolina water quality standards. Some dissolved oxygen concentrations were found to be approaching minimum concentration requirements for biota, however, and results for most of the general parameters should be confirmed with stormflow sampling.

The benthic macroinvertebrate habitat assessment in North Creek also showed extensive impairment of this stream. Habitat degradation included lack of heterogeneous riffle/pool sequence and meander pattern due to channelization and intense stormflows; limited undercut regions and organic substrates due to poor riparian vegetation presence and function; and altered substrate resulting from fine sediment loads and artificial substrate additions. The comparison to Avent Creek further demonstrated the loss of habitat function most likely resulting from influences of the urban watershed and poor riparian buffer surrounding North Creek.

Based on water quality and habitat assessments, the diminished diversity and abundance within the benthic macroinvertebrate community was expected. The biological integrity of North Creek was impaired, with data supporting the federal 303d listing as part of the Clean Water Act. Community results of this study were similar to other urban stream studies and the comparison to Avent Creek further supported the main cause of impairment as development and urban stormwater runoff from the watershed. The benthic macroinvertebrate introduction effort suggested that the current stream conditions were unsuitable for biological health. The results of this study indicated that both water quality degradation and habitat availability were potentially responsible for loss of ecological function. The results did not, however, indicate which factor was more influential on the benthic macroinvertebrate community.

One of the goals of this research was to provide preliminary data for planned watershed enhancement projects that will include stormwater treatment, stormwater retention, and stream restoration. Further evaluation will be performed after watershed and stream modifications to determine what factors are most influential for maintaining biological integrity in urban watersheds, and how to best improve the quality and quantity of stormwater runoff and biological habitats in urban systems. Benthic macroinvertebrate introductions should also be attempted after the project to investigate further the use of this technique to enhance the biological community of impaired systems.

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CHAPTER 6: CONSTRUCTED STORMWATER WETLAND DESIGN, CONSTRUCTION, AND ESTABLISHMENT

INTRODUCTION

Constructed Stormwater Wetland Functions

Wetlands are defined by the US Army Corps of Engineers as those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions (Environmental Laboratory, 1987). For a constructed wetland to function like a natural wetland, it must be designed to support wetland hydrology, hydrophytic vegetation, and hydric soils. The environmental and ecological functions and services desired for a constructed stormwater wetland (CSW) includes stormwater treatment, stormwater retention, and the establishment of aquatic and terrestrial habitat.

Stormwater Treatment

Constructed stormwater wetlands have been increasingly utilized in urban and developed watersheds as best management practices (BMPs) for improving the water quality of stormwater (DeBusk and DeBusk, 2001). Common pollutants associated with stormwater derived from developed watersheds include excess nutrients, excess sediments, heavy metals, toxic organic compounds, and harmful microorganisms associated with human and animal waste (Bingham, 1994). The ability of CSWs to reduce these pollutants in stormwater has been well documented in the literature. Contaminant removal is achieved through a variety of physical, biological, and chemical processes that occur within the wetlands. Wetlands offer a unique environment, combining attributes of

microtopography, fluctuating water levels, and a biological community of microorganisms and plants, that can carry out these processes effectively and in an efficient manner. Pollutant reduction and removal mechanisms have been described in several published references (Schueler, 1992; Bingham, 1994; Debusk and Debusk, 2001) and will be summarized in this review.

The physical mechanisms occurring within CSWs for stormwater treatment are associated with the variable topographic features and the plant community. Sedimentation is one physical process, when suspended sediment settles out of the water column due to gravity and the reduction in flow velocity. Particles suspended in the water column, include clays, silts, and organic matter. These particles increase turbidity and carry a net negative charge that promotes association with metals, nutrients, and toxic organic compounds with opposite charges. Most sedimentation occurs in the deeper pool regions that act as small settling basins and these pools are commonly placed at the inlets and outlets for maximum sediment removal and ease in long-term maintenance. The suspended sediment removed from the water column can be stored on the wetland bottom for long periods of time and permanently removed through pool maintenance. Another physical mechanism for pollutant removal is filtration by the wetland vegetation. Dense plant communities within the shallow water and shallow land regions obstruct the flow path and create localized reductions in flow velocity. Suspended particulate contaminants can be removed by settling, while floating organic compounds such as oils and grease can be skimmed from the surface by plant surfaces (Ferlow, 1993).

Chemical mechanisms occurring in CSWs can remove both particulate and dissolved contaminants from stormwater. Particulates can adsorb to above-ground plant tissues and the associated microbial biofilms that utilize the vegetation for substrate. Aquatic plants (DeBusk and DeBusk, 2001) and biofilms (Atlas and Bartha, 1993) have charged surfaces that can attract charged particles such as clays and metals. These bound pollutants can eventually settle to the bottom sediments upon plant senescence and microbial decay. This mechanism is enhanced in CSWs due to increased stormwater residence time and to the relatively large amount of shallow water area in contact with vegetation. Dissolved pollutants such as metals can be removed through the formation of precipitates that can flocculate from the water column for sediment storage. Removal of dissolved pollutants can also occur in the opposite direction, through volatilization (e.g. Ammonia), or through aerosol formation when the wind blows across the water surface. Another chemical process is photochemical degradation of complex organic contaminants by sunlight. This mechanism is promoted by shallow water depths and by the increased stormwater residence time. Once this degradation process is initiated, the potential for pollutant removal by microbial decomposition is enhanced.

Biological mechanisms that treat stormwater in CSWs are performed by the microorganism and plant communities. Wetlands are well known for their enhanced biological productivity compared to other aquatic environments (Wetzel, 1983) and provide excellent habitats for a diverse group of microorganisms including bacteria, algae, and fungi. The success of the microbial community is due to an abundant carbon supply from detritus and plentiful substrate surface area. At the soil-water interface, microbial

decomposition of organic matter removes oxygen to create an anaerobic layer within the sediments. A productive aerobic/anaerobic zone is created for interactive microbial metabolism. The aerobic layer above the anaerobic layer functions to keep phosphorus and metals bound to the sediments that have settled. Nitrification in the aerobic zone and denitrification in the anaerobic zone are coupled to convert soluble nitrogen to nitrogen gas (Richardson and Vepraskas, 2000). Microorganisms can also degrade complex organic compounds during decomposition, but these processes are variable with time depending on the pollutant. Biological components also have the ability to directly absorb dissolved pollutants during metabolism. Vegetative uptake occurs mainly through the roots within the sediments, but microorganisms colonizing the above ground portions of the plant can mineralize and absorb dissolved contaminants directly from the water column. Although the storage of these pollutants within living tissue is considered short-term due to leaching during senescence and decay, recycling can be extensive within the complex biological communities and lengthen the storage time within the overall system (Wetzel, 1983).

Stormwater Retention

Watershed urbanization increases the proportion of impervious surfaces, with stormwater removal from these surfaces designed to be as rapid as possible to reduce on-site flooding. As a result, stormwater is delivered to receiving waters with limited infiltration, increased runoff volume, and with a larger peak discharge. This urban runoff causes channel instability in receiving streams, with an increased sediment load contributed during channel erosion (Dunne and Leopold, 1978). Best management practices such as CSWs can reduce stormwater impacts by providing stormwater retention.

Stormwater can be stored within the wetlands using berms and adjustable outlet structures for slow release to the receiving waters.

The topography of CSWs allows for effective storage of stormwater during both small and large storm events. As stormwater enters CSWs, it flows through the deep pools and shallow water channels that meander through the wetland for a decline in flow velocity and dissipation of flow energy. These features are designed for maximum pollutant removal during the initial "first flush" of stormwater from the watershed. As the stormwater volume increases, shallow land and upland features become inundated and can provide significant storage during larger storm events. When the wetland is near its storage capacity it behaves like a detention pond, with the amount of storage determined by the combined berm heights or overall depth of the wetland below the ground surface (NCDENR, 1995; Hunt and Doll, 2000).

Aquatic and Terrestrial Habitat

The combined dense vegetation, shallow water, and deep pool features of CSWs provide habitat for both aquatic and terrestrial organisms. This habitat diversity can produce a more balanced trophic system for control of undesirable species such as fecal organisms (Bingham, 1994), or mosquitoes (Tennessen, 1993). Permanent water in the deep pools and shallow water regions can support fish and macroinvertebrates even during drought. Although CSWs may not be as diverse or as productive as natural wetlands, they can provide natural settings within urban landscapes to help maintain corridors between the natural ecosystems disturbed by development (Schueler, 1992).

Constructed Stormwater Wetland Design

The design and establishment of a CSW to treat and manage stormwater in an urban watershed presents many challenges. Stormwater can contribute very dynamic hydraulic and pollutant loads to the wetland (Bingham, 1994). These structures must be crafted for adequate retention of both small and large storm events, with variable intensities and durations. They must also be designed for the most effective treatment of stormwater with highly variable and unpredictable pollutant loads. For maximum long-term benefit in urban watersheds, a CSW must be designed with minimal need for repair and maintenance. Wetland functions evolve over time, and disturbances through maintenance may disrupt this development (Ferlow, 1993). CSW design must also address the time required for wetland functional development, which can take as long as 1-5 yrs to approach the levels of a natural wetland system (Ferlow, 1993).

North Creek CSW Design

The purpose of this chapter is to describe the design process used for two constructed stormwater wetlands on the North Creek floodplain and to evaluate the designs as implemented. Evaluation was based on how well the design elements were performing and based on the three goals of stormwater treatment, stormwater retention, and habitat enhancement. These wetlands were constructed using USDA grant funds and NCSU stormwater improvement funds. The following elements for design and installation were addressed:

1. Location Selection
2. Size Determination

3. Internal Wetland Features
4. Wetland Depth and Lining
5. Inlet and Outlet Structures
6. Storm Hydrographs and Storm Modeling
7. Sediment and Erosion Control
8. Wetland Vegetation

The methods presented below are a summary of the official design plan, with detailed design plan sheets and a construction project manual submitted to the NCSU Construction Management Department. These documents are available in the NCSU Facilities Plan Room. Design took place between October 2003 and March 2004, with construction between July 6, 2004 and August 2, 2004. The CSWs were designed with guidance from Dr. William F. Hunt, PE, from the Biological and Agricultural Engineering Department, with design drawings contributed by Joseph Puckett, Ian Jewell, and Todd Flippen, and James Blackwell from the NCSU Water Quality Group. Wetland vegetation was planned by Karen Hall from the NCSU Water Quality Group. Recommendations and information concerning the substrate and liners was provided by Mark Rice from the Biological and Agricultural Engineering Department. The wetlands were constructed by North State Environmental, Inc.

METHODS

Location Selection

Throughout the North Creek watershed, stormwater is released from culverts into detention ponds, onto the floodplains near the stream channel, and directly into the stream. A location was sought to optimize stormwater treatment with respect to available surface area and funding. Along the North Creek corridor, a relatively large expanse of floodplain area on each side of the stream was left undeveloped and designated by the university as natural area. These areas are located near the instream stormwater management ponds just before North Creek enters Lake Raleigh (Figure 6.1). These floodplain areas were ideal for constructed stormwater wetlands for several reasons. The soils were composed of construction fill material of low quality, evidenced by little vegetation colonization in the five years left undeveloped. The topography was relatively flat on both floodplains, reducing grading cost. Both areas also had easily divertable untreated stormwater released in close proximity for diversion. Another reason for choosing the floodplains was the extensive lateral streambank erosion occurring in North Creek within this reach. Increasing stormwater retention on the floodplains could help alleviate stress on the streambanks.

The available surface area and stormwater near each floodplain was evaluated for CSW installation. On the eastern floodplain (Figure 6.2), the estimated surface area available for wetland land use was about 11,500 sq ft, determined by several constraints. The northern extent was limited by a pedestrian bridge and its support columns, while the southern extent was limited by an existing stormwater pipe and sewer line. A proposed

greenway path, as part of the NCSU Master Plan, controlled the eastern boundary. The western boundary was drawn by an established City of Raleigh sewer line that required a 15-ft easement on each side. There were two stormwater sources near the eastern floodplain identified for treatment feasibility. A 15-in culvert released stormwater near the northern boundary into an eroding ditch that stretched diagonally across the floodplain to the stream. This stormwater source could be directly intercepted by a wetland. The culvert (COT inlet) drained stormwater from the College of Textiles roof and lawn areas. The other stormwater source was a 24-in culvert that released runoff into stormwater management pond #3 (SWMP #3). This stormwater source could be intercepted by upslope pipe redirection. The created culvert (EGRC inlet), would drain portions of the Engineering Graduate Research Center building and brickyard.

On the western floodplain of North Creek (Figure 6.3), the wetland surface area available was estimated to be about 6000 sq ft. The northern extent was again limited by the pedestrian bridge, while the southern boundary was limited by another City of Raleigh sewer line. The Neuse River riparian buffer requirement along North Creek limited the extent of the eastern boundary, and the western boundary was limited by a wooded region with a relatively steep slope. There were also two stormwater sources near the western floodplain identified for treatment feasibility. A 24-in culvert released stormwater upslope and south of the proposed wetland area and into a stone lined channel that stretched to SWMP #3. This stormwater source could be indirectly intercepted by a wetland using a grassed waterway, due to the steep slope between the culvert and floodplain. This culvert (RES II inlet) drained stormwater from portions of the Research II building and front

parking lot. An 18-in culvert released stormwater upslope and north of the proposed wetland area within the forested area. This culvert drained the Research I building, parking lot, and lawn areas. Upon investigation, there was no discreet pathway evident for stormwater flow between this culvert and the floodplain. The stormwater pipe was determined to be broken above the outlet, and most of the stormwater infiltrated before arriving at the outlet. This stormwater source was not purposely intercepted due to the unpredictable nature of its subsurface flow, but additional storage area was provided within the wetland for any stormwater that may have resurfaced downslope.

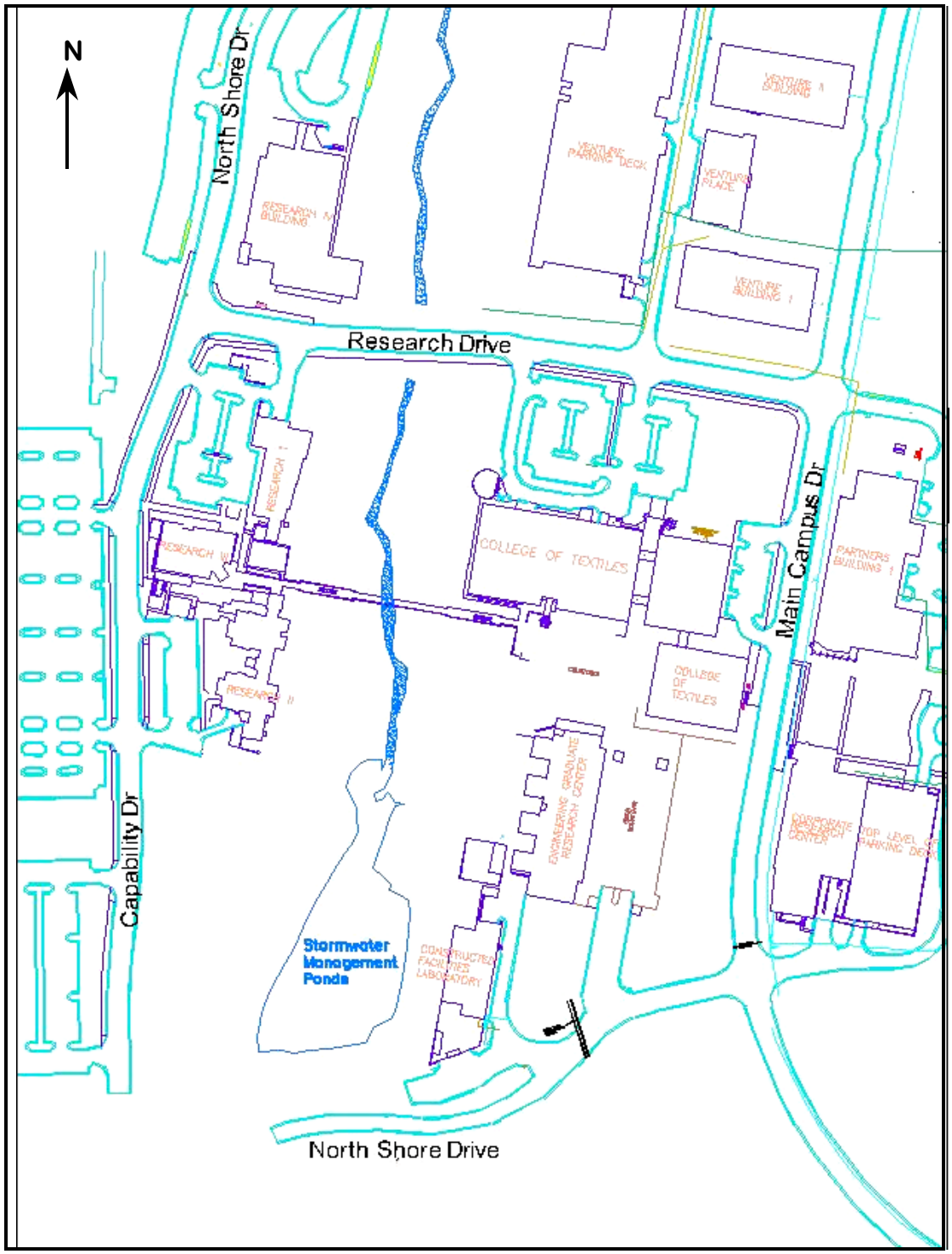


Figure 6.1: North Creek Floodplains for Constructed Stormwater Wetland Sites (Digitized Map by NCSU Facilities; Figure by Ian Jewell, NCSU Water Quality Group). Scale: 1 in = 250 ft.

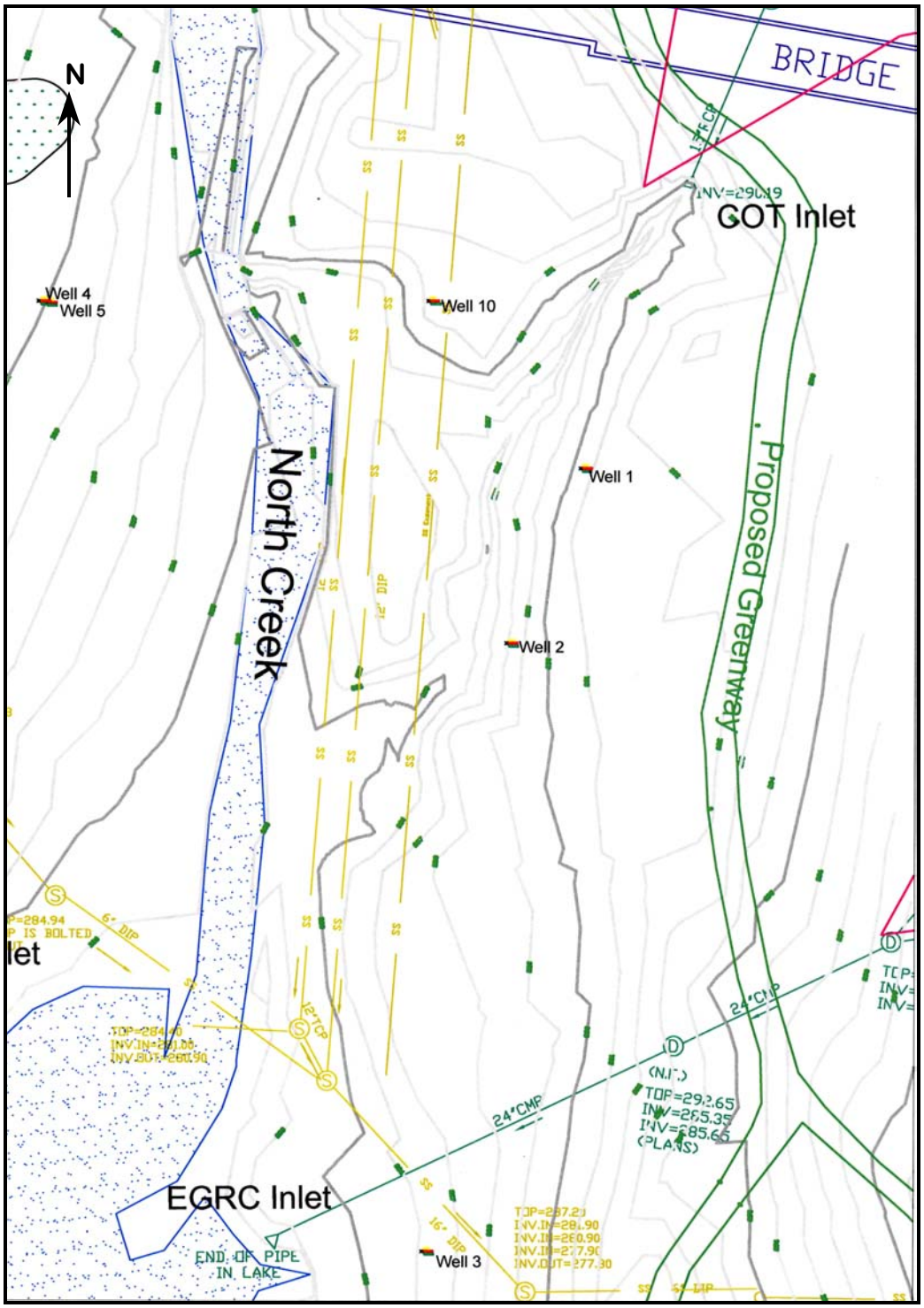


Figure 6.2: Constructed Stormwater Wetland 1 Existing Site Conditions with Groundwater Monitoring Well Locations (Digitized Map by NCSU Facilities; Figure by Ian Jewell and Joseph Puckett, NCSU Water Quality Group). Scale 1 in = 30 ft.

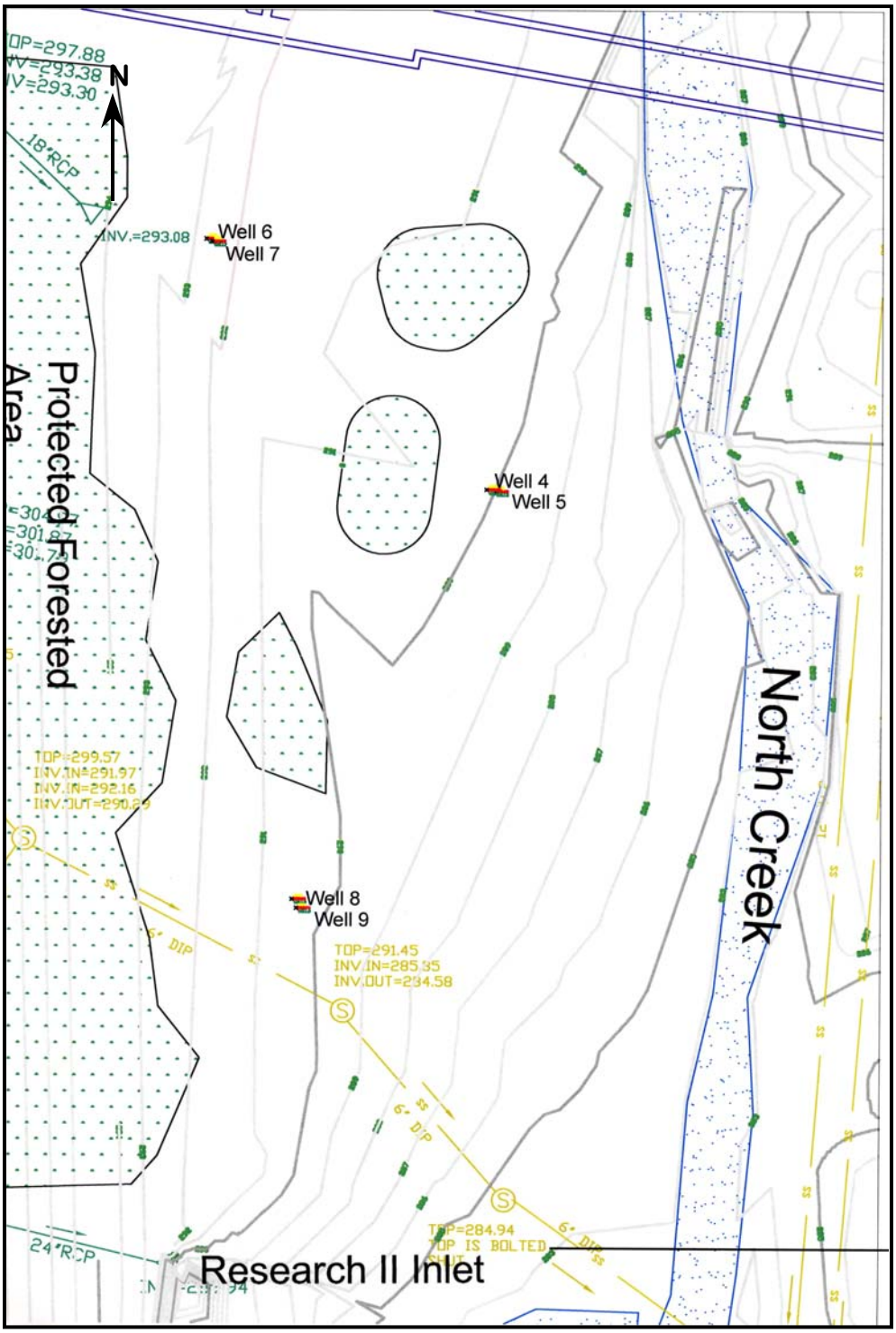


Figure 6.3: Constructed Stormwater Wetland 2 Existing Site Conditions with Groundwater Monitoring Well Locations (Digitized Map by NCSU Facilities; Figure by Ian Jewell and Joseph Puckett, NCSU Water Quality Group). Scale 1 in = 30 ft.

Size Determination

One of the goals of CSW design was to treat the "first flush" of stormwater runoff, which has been shown to capture the largest proportion of pollutants derived from the watershed. The first flush is commonly considered runoff from the first 1-in of precipitation falling on the watershed and can be used to size CSWs for treatment (NCDENR, 1995). To determine this runoff volume, the watershed areas were first delineated using both topography and stormwater routing information. The runoff volume was then calculated based on land uses within the watersheds.

The watersheds for the two North Creek CSWs were delineated using AutoCAD Land Desktop3 (Autodesk, 2003) with digital Centennial Campus maps that included current topography, structures, paved surfaces, and utilities (Provided by NCSU Facilities). The subwatersheds at each inlet culvert were determined, then field verified with ground-truthing during rainfall events (Figure 6.1).

After the watershed areas were estimated, the runoff volume calculated using the SCS Curve Number Method (SCS, 1972). Curve numbers were assigned to the different land uses within the watershed, with designated land use areas and associated curve numbers shown in Appendix 6, Table A6.1 and Table A6.2. The runoff depth from each land use area was then calculated using the equation $R = (P - 0.2S)^2 / (P + 0.8S)$; where R is runoff depth (in), P is precipitation from the first flush (in), and S is the ultimate soil storage (in) determined by dividing 1000 by the curve number and subtracting 10. Once the first flush runoff depth from each land use was calculated, the runoff volume was

determined by multiplying depth times land use area. The total runoff volume from the stormwater sources was then estimated by adding runoff volumes from each land use.

Using the stormwater volume information, the required CSW sizes were determined and evaluated for floodplain location feasibility. Wetland surface areas needed to hold the first flush stormwater were estimated by dividing the runoff volume by the average depth of a constructed stormwater wetland (0.75-ft). For the CSW on the eastern floodplain (W1), the surface area required was about 11,100 sq ft, while the surface area required for the CSW on the western floodplain (W2) was about 2,230 sq ft. All calculated values for size determinations are provided in Appendix 6, Table A6.1 and Table A6.2. The results of the calculations showed that both CSWs could be placed on the floodplains within the constraints presented.

Internal Wetland Features

The internal wetland features desired in the CSWs were those that mimicked the topography found in natural wetlands, while effectively treating and storing stormwater without erosion (Schueler, 1992). The four general components of CSWs are Deep Pools, Shallow Water, Shallow Land, and Upland regions (Hunt and Doll, 2000). Their placement within the two North Creek CSWs is shown in Figure 6.4 and 6.5.

Deep pools are generally placed at stormwater inlets (forebays) and just before outlets. They function to dissipate flow energies, promote sedimentation, provide storage during periods of drought, and provide mosquito predator habitat. They usually comprise about 10% of the surface area, and range from 2-3 ft in depth below the permanent pool

elevation. In both wetlands, forebays were installed at each concentrated stormwater inlet point and just before the outlet structures.

Shallow Water regions of CSWs are usually about 0.5-1.0 ft deep below the permanent pool elevation, and comprise about 40-50% of the total surface area. These areas are important for wetland function, providing shallow water and a channel to direct and extend the flow path of stormwater. In this region, obligate wetland vegetation thrives and aerobic/anaerobic interface zones along the substrate flourish with microbial decomposition. In the North Creek CSWs, the Shallow Water channels were given as much meander and surface area as possible for the space provided.

Shallow Land areas of CSWs start at the permanent pool elevation and extend about one foot above. These regions are inundated during stormflows and comprise about 30-40% of the total surface area. Diverse, facultative vegetation grows and heterogeneous topography develops, providing substrate for stormwater treatment during high flows. For both North Creek CSWs, Shallow Land areas extended from the Shallow Water channel to the base of the berms.

Upland regions are the fourth component of CSWs and comprise about 10% of the surface area. These regions provide added vegetation diversity, a buffer between the wetlands and surrounding landscape, and can promote storage within the wetland boundary. In the North Creek CSWs, Upland areas were the surrounding berm slopes (3:1) that were utilized for retaining stormwater within the wetlands and for protection of the wetland structure from erosive streamflows accessing the floodplain during large events.

During North Creek CSW construction, the internal wetland features were created from existing soil material stockpiled from initial excavation. Topsoil collected from a nearby development site was added to the wetland bottom (3-in) to reach the design elevation for each feature and to provide a better growth medium for wetland vegetation.

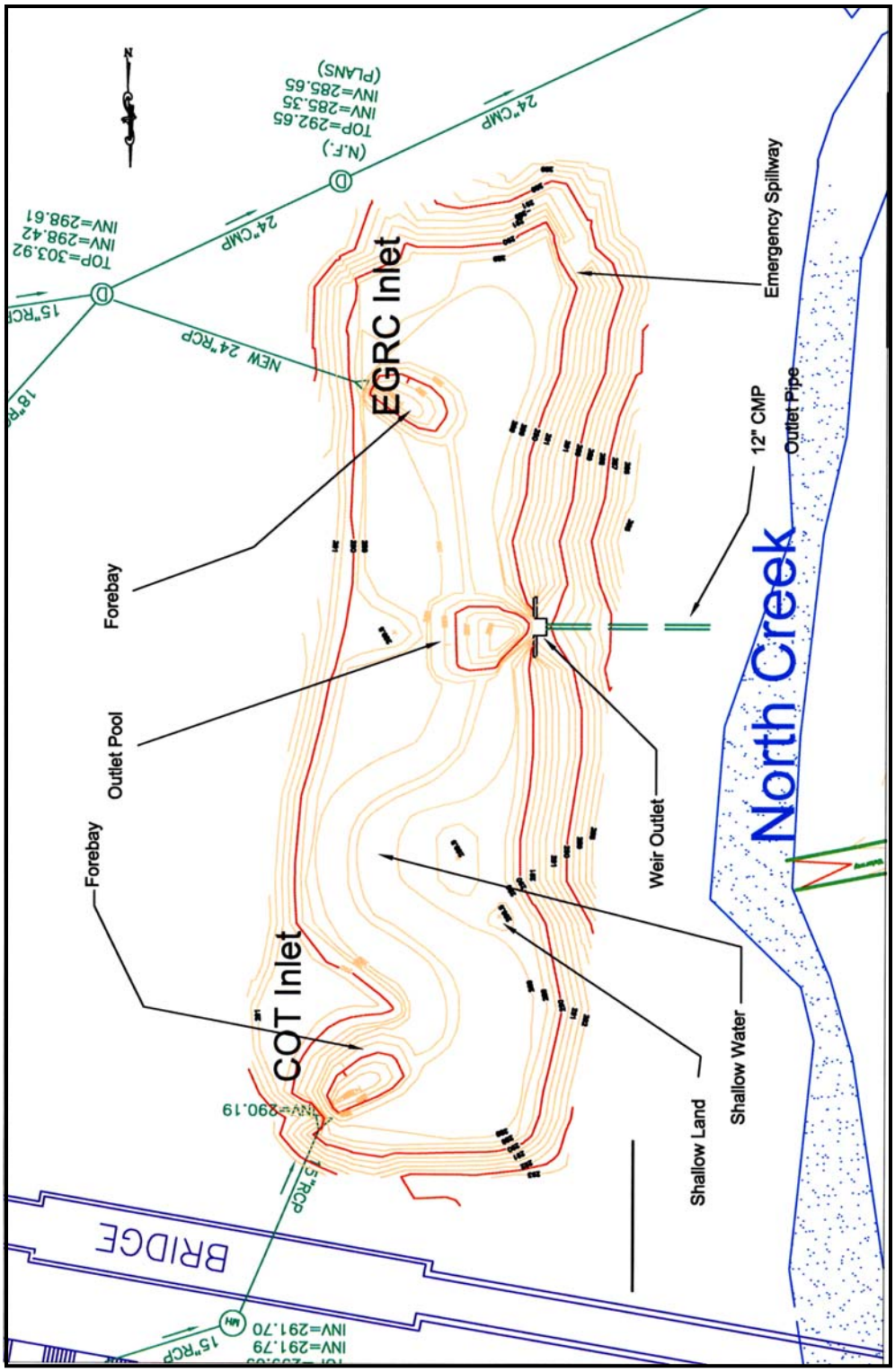


Figure 6.4: Constructed Stormwater Wetland 1 Design (Figure by Joseph Puckett and Ian Jewell, NCSU Water Quality Group). Scale 1 in = 35 ft.

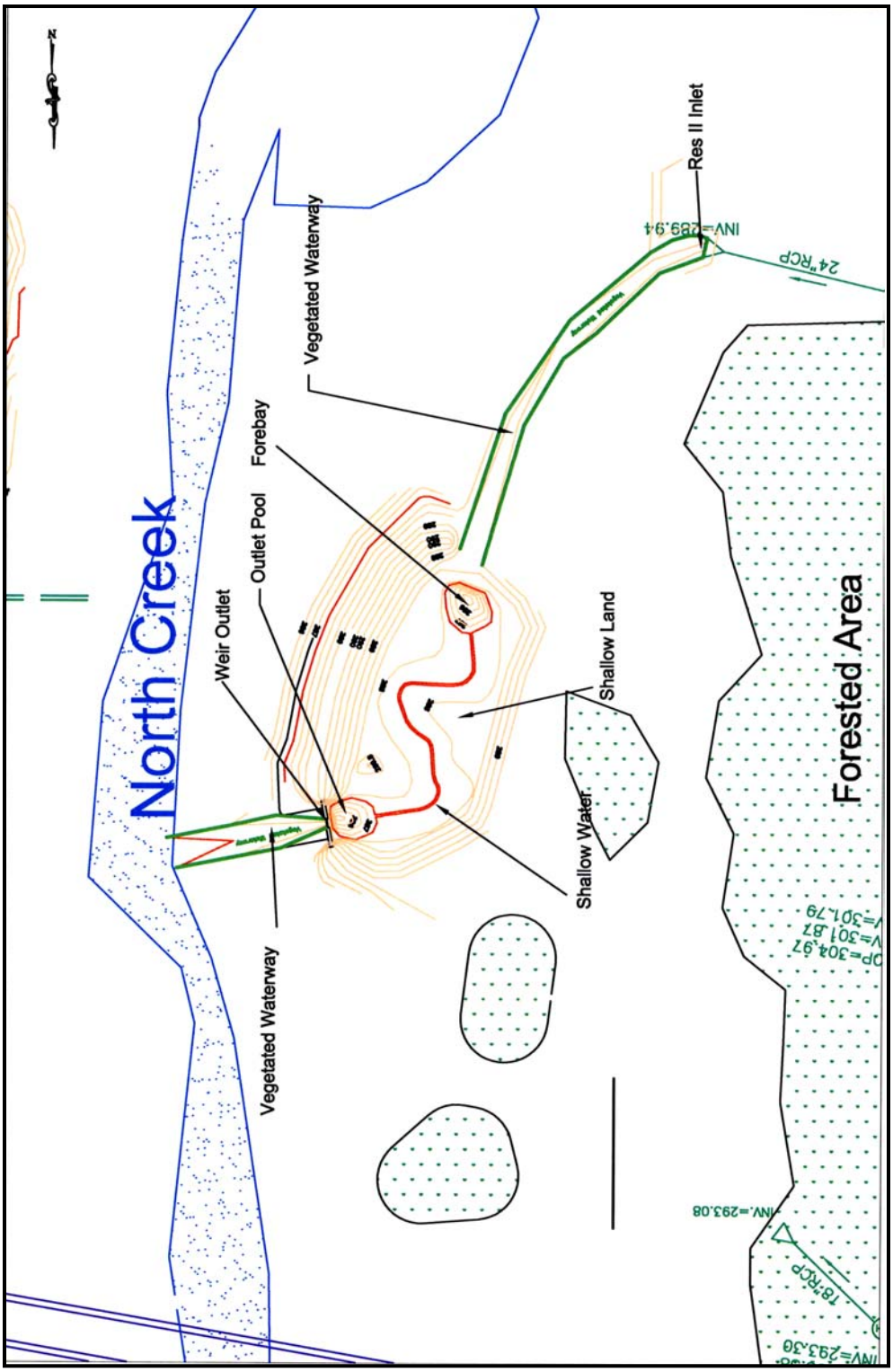


Figure 6.5: Constructed Stormwater Wetland 2 Design (Figure by Joseph Puckett and Ian Jewell, NCSU Water Quality Group). Scale 1 in = 35 ft.

Wetland Depth and Lining

Saturated conditions can be maintained within CSWs using stormwater and groundwater inputs. The depth below the landscape is generally determined based on elevation of these water sources, along with factors such as topography.

Groundwater table elevations below the North Creek CSW sites were estimated to determine the feasibility of interception. Groundwater monitoring wells were installed in September 2003 and monitored periodically through March 2004 before and after storm events. Four wells with depths ranging between 2.6 ft and 4.8 ft below ground surface were installed on the W1 floodplain (Figure 6.2), with two wells at shallow depths (Well 2 and 3) and two wells at deeper depths (Well 1 and 10). Six wells ranging between 2.3 ft and 5.8 ft below ground surface were installed on the W2 floodplain (Figure 6.3), with three wells at shallow depths (Well 5, 7, and 9) and three wells at deeper depths (Well 4, 6, 8). Each well was comprised of 2.0-in diameter PVC pipe, with $1/16^{th}$ in diameter holes drilled along the portion below ground surface. The wells were encased by sand, sealed at the surface using bentonite clay, and capped to prevent rainfall intrusion into the pipe. Water level measurements were taken manually, using a metal measuring tape and baby powder to identify groundwater depth below the surface.

The groundwater table elevations measured within 24 hrs after rainfall events are shown in Appendix 6, Table A6.3 and Table A6.4. The shallow wells indicated that there was a perched water table near the surface, but the deeper wells indicated that groundwater table elevations were generally between 3 and 6 ft below the ground surface on each floodplain. Due to evidence of a relatively deep seasonal low water table, it was decided

that the CSWs would be perched above groundwater and receive inputs only from surface waters. Perching the wetland reduced the depth required for excavation and helped set the permanent pool elevations at EL 289.0 ft in W1, and EL 288.0 ft in W2 based on the existing lowest topography.

In order to maintain saturated conditions using surface waters, the material beneath the wetlands had to provide a relatively impermeable barrier to reduce seepage loss. The North Creek CSWs were required to hold water for at least 40 days with no rainfall to ensure plant survival during drought periods. To meet these conditions, the permeability would need to be less than 9.9×10^{-3} in/hr. The soils present on site at the expected depth of wetland excavation were collected and evaluated for their suitability by a soil geotechnical consultant (GeoTechnologies, Inc., Raleigh, NC). The soils on the W1 site were sandy silty clays with a permeability of 1.4×10^{-6} in/hr, and the soils on the W2 site were silty fine to medium sands with a permeability of 3.1×10^{-3} in/hr. Soils with sandy textures are generally not suitable to hold water compared to clay soils, as indicated by the measured permeabilities for the W2 site. Although soils on the W1 site were adequately impermeable, construction debris and evidence of heterogeneous fill material was found while the samples were collected. Based on these observations and the soil test results, it was decided that a liner was needed on both CSW sites.

Liners can be constructed using synthetic materials, or by augmenting the existing soil. Synthetic liners include plastic or geomembrane liners made of polyurethane, polyethylene, PVC, or other materials; concrete liners; and geosynthetic clays (NRCS, 1997). With all synthetic liners, the largest disadvantages are the required expertise to

install them and the potential errors associated with installation. Synthetic materials can be more expensive due to the materials themselves and due to the specialized contractors required for proper installation. Plastic or geomembrane liners can puncture or tear as a result of equipment or imperfections in the substrate below, and seam alignment can be cumbersome. Concrete liners and geosynthetic clays may also crack or sink due to properties of the underlying soil (NRCS, 1997). The synthetic liners were determined to be too expensive and more prone to significant leakage if errors occurred.

Augmentation of existing soils can be accomplished by adding clays to the wetland bottom and sides followed by compaction to form a nearly impermeable layer. A typical clay used for ponds and lagoons is bentonite, which has the ability to swell approximately 15 times its original volume when exposed to water (NRCS, 1997). There are many advantages to augmenting the bottom substrate of constructed wetlands using bentonite. The grain size of the bentonite can be selected to best suit the in-situ soils, and grains can be blended within the soil compared to a single vulnerable boundary on top of the soil. Changes to the substrate soil such as settling or decomposition will not alter the ability of the bentonite to swell. Another advantage is that the impermeability of the soil would be enhanced even if there were errors associated with installation or inadequate amounts of bentonite added.

A bentonite clay liner was selected for installation under each North Creek CSW to maintain saturated conditions. The relative grain size of bentonite was selected for the particular soil types present. On the W1 site, Wyoming sodium bentonite clay with a 30-50 mesh size (CETCO C/S Granular) was applied for sealing. On the W2 site, Wyoming

sodium bentonite clay with an 8-20 mesh size (CETCO Volclay Crumbles) was applied for sealing. The bentonite clay was applied uniformly at a thickness of 1/3-in, using a drop spreader. It was applied to dry substrate along the wetland bottom and extended to half the berm elevation. The liner was created at about 1.5 ft below the final design elevation by tilling the clay into the in-situ soil to a depth of 4 in or greater. According to NRCS Practice Standard 521C for non-waste water containment, the minimum depth of the bentonite layer should be 4 in for overlying water depths less than 8 ft. After mixing, the bentonite-amended layer was compacted to 95% Maximum Standard Proctor dry density using tracks on heavy equipment in order to establish the liner. Stockpiled soil and topsoil was then added above the liner to contour internal wetland feature contouring.

Inlet and Outlet Structures

An inlet culvert (EGRC inlet) was added to W1 on the southern end to divert stormwater that formerly flowed into SWMP #3 (Figure 6.4). A new 24-in concrete pipe was added at the manhole, while the former pipe was closed off to redirect water. The pipe was extended downslope to the permanent pool elevation at a 10% slope and fitted with a flared end unit to help dissipate energy at the end of the pipe. Due to the potential erosive energy of stormwater exiting both culverts into W1, a stone-lined apron was added at the end of each culvert (COT inlet and EGRC inlet) extending into the forebays. Both the forebays and stone-lined aprons were sized to dissipate the maximum discharge expected from each culvert (NCDNRCD, 1988; ARC, 2001).

In W2, the stormwater from the RES II inlet culvert was diverted into the CSW using a vegetated waterway (Appendix 7, Figure A7.2). A forebay was added to dissipate

energy from this culvert before entry into the vegetated channel. The channel was a triangular vegetated swale, lined with erosion control matting. The dimensions were 1-ft deep by 8-ft wide, as determined using the peak discharge for the 10-yr, 24-hr design storm and Manning's equation with additional freeboard added and verification of the permissible velocity (NCDNRCD, 1988; Malcom, 1995). The vegetated waterway was brought down to the permanent pool elevation at the inlet forebay of W2, with a berm placed on the eastern side to prevent short-circuiting.

The primary outlet structures for both wetlands were concrete weirs with adjustable weir slats and an orifice at the permanent pool elevation. The outlet designs for both CSWs are provided in Appendix 7, Figures A7.3 – A7.5. The 9-in wide concrete weirs extended to the top of berm elevations, EL 291.5 ft in W1 and EL 290.0 ft. In W1 an additional concrete splash box was added to the back of the weir for further energy dissipation and to improve the aesthetics of the tall weir in the landscape. In W1 the weir slats were made of 2x6 planks of TREX plastic-wood composite board (TREX Company, Inc.). In W2 the weir slats were made of untreated 2x6 wood planks coated with Trojan Beta-Z wood sealer (Envirosafe Manufacturing Corp.), a non-toxic treatment. The different materials were used for evaluation of more environmentally-sound alternatives to chemically pre-treated wood. The weir slats were interlocking, and placed in an aluminum track for relative ease of adjustment. The weir slats extended to 1-ft below the top of the concrete weir in both wetlands. An orifice was placed within the weir slat at the permanent pool elevation, EL 289.0 ft in W1 and EL 288.0 ft in W2. The orifice was composed of a PVC pipe with a down-turned end and protective wire-mesh box to block

debris. The orifice pipes were sized to slowly release stormwater volume between the top of the weir slats and the permanent pool elevation over a 3-d period. The orifice equation was used to predict orifice discharge (Malcom, 1995): $Q = C_d A(2gh)^{(1/2)}$; where Q is the orifice discharge (cfs), A is the area of the orifice pipe (sq ft), C_d is the coefficient of discharge equal to 0.6 for typical pipes, g is the acceleration due to gravity (ft/s^2), and h is the height of water over the orifice elevation (ft). The orifice size was determined based on the calculated time-step drawdown using trial and error design orifice diameters and orifice discharge. The orifice diameters were 1.75 in for W1 and 0.75 in for W2.

Stormwater was carried from the base of the W1 splash box to North Creek using a 12-in corrugated metal pipe with stone protection along the streambank. A vegetated waterway was constructed to carry stormwater from the weir of W2 to North Creek with stone protection along the streambank. As a secondary outlet, an emergency spillway protected by stone and permanent erosion control matting was added to W1 on the southern end of the wetland closest to the stormwater management ponds. The spillway was 0.5 ft below the top of berm and 8 ft wide. W2 did not have a separate emergency spillway due to its relatively small size and excess storage in the forested area on the western side.

Storm Hydrographs and Storm Modeling

The HEC-HMS model was utilized to predict peak discharge and total flow from the watersheds where stormwater was derived (USACE, 2003). Predictions were made before and after wetland installation to estimate the effects of the CSWs on the inflow and outflow hydrographs. For this model the watershed origins were placed at existing and

proposed outlets to North Creek. CSWs were accounted within the model by replacing existing conveyance channels with reservoirs.

For both wetlands, the peak flow and total flow were estimated using a basin model and a meteorological model appropriate for each design storm (1-yr, 2-yr, and 10-yr 24-hr storms). Peak flow was estimated in the model using the SCS Curve Number method, with the lag time estimated to be 0.6 * (time of concentration). The watershed time of concentration was estimated using the Kirpich equation (Kirpich, 1940). Baseflow was considered to be zero since the source was stormwater. In the basin model, the watersheds were designated as sub-basins; the ditches, pipes, riprap channels, and vegetated waterways were designated as stream reaches with different Manning roughness coefficients; and the wetlands were designated as reservoirs with area measurements for each elevation up to the berms. Data for the meteorological model were derived from precipitation measured at RDU Airport, Raleigh, NC (NOAA National Weather Service, Raleigh, NC).

The results of the HEC-HMS model predictions are shown in Appendix 6, Table A6.5. The times of concentration increased with the addition of the CSWs, while the peak discharge and total flow decreased for all three design storms. The increase in the time of concentration and attenuation of peak discharges are due to the extended flow path and storage provided by the CSWs. The total flow reduction was not as large compared to the peak discharge reductions, suggesting that the stormwater retention provided by the CSWs is not significant enough to yield a marked decrease in outflow volume. Due to the perched conditions and limited infiltration, this result was anticipated for these wetlands.

The model predicts that the CSWs will retain stormwater and reduce total flow when compared to the pre-existing conditions.

Design stormflows were evaluated for both CSWs, using design storms and design outlet features to determine the maximum elevations reached during each storm. The inflow hydrographs for the 1-yr, 10-yr, and 25-yr 24-hr design storms were routed through each CSW using the outlet structures. The orifice equation was used to calculate orifice discharge (provided above). The basic weir equation was used for the weir and emergency spillway in W1 to calculate weir discharge (Malcom, 1995): $Q = C_w L H^{(3/2)}$; where Q is the discharge (cfs), C_w is the weir coefficient equal to 3.0 for free overfall, L is the length of the weir slats (ft), and H is the height of the water over the weir slats (ft). A time-step calculation series was then performed for each design storm in each wetland using Excel spreadsheets. Based on the stormflow retention predictions, all three design storms would pass over the weir structures, and both wetlands would be able to hold the 25-yr, 24-hr design storm without utilizing the emergency spillway (W1) or overtopping the weir concrete (W2).

Inflow and outflow hydrographs in each wetland were evaluated after construction to help determine if the HEC-HMS model predictions and the storm routing predictions were valid. Differences between the inflow and outflow volumes during storm events were also evaluated to define and interpret reductions in total flow, as predicted by the HEC-HMS model.

Sediment and Erosion Control

Several measures were implemented to control erosion during and after construction. Temporary construction entrances were installed or enhanced at the beginning of construction access paths. A temporary stream crossing was added to North Creek to protect streambanks and substrate during equipment and material transfer from one site to the other. Silt fencing was installed between the CSW sites along the stream and surrounding the stockpiled soil. Construction phasing was also implemented, to avoid excess erosion associated with excavation and soil movement on-site and to minimize the length of time for bare soil exposure.

After construction was complete, an erosion control system was applied by hydromulching all exposed surfaces. This system was designed by U.S. Environmental Protection Services (USEPS, Greensboro, NC). The system was composed of the trademark products H_2OLD (Greensboro Agricultural Products, Greensboro, NC, patent pending) and Biosol (Rocky Mountain Bio Products, Denver, Colorado, patent pending), PAM (APS 700 Series Silt Stop, APS Polymers, Inc., Woodstock, GA), standard wood chip mulch, lime, water, and seed. The general contents of the trademark products were obtained from US EPS, and are shown in Appendix 7, Table A7.1 and Table A7.2. The goal of the erosion control system was to enhance soil properties, establish vegetation rapidly, and slowly release nutrients for plant consumption. H_2OLD is a product developed to increase the water holding capacity and nutrient retention of soil. It contains micronutrients and minerals, including Phillipsite, a natural zeolite that improves the cation exchange capacity and buffer the pH of the soil. The chelating properties and negative

charges associated with the minerals help sequester added nutrients for slow release. Biosol is a natural organic fertilizer, using a penicillin fermentation by-product as the key ingredient. The wood mulch was added for organic matter, nutrients, and substrate; and the seed was incorporated for vegetation establishment.

Wetland Vegetation

Permanent wetland and upland vegetation seed mixes were added to both CSW sites through hydromulching (Appendix 7, Table A7.3). The "wet" species seed mix was applied to the interior of the wetlands, while "dry" species seeds were applied to the berms and surrounding areas disturbed by construction. All permanent seed was applied at the recommended rate per acre for each species. Temporary annual seed, Brown top millet, was also included in the hydromulching seed mix for surface protection at a rate of 30 lb/ac. Additional hydromulching occurred about a month and a half after construction (September 17, 2004) to revegetate bare spots where seed had washed away. The seed mixtures were the same as above, with the addition of rye grass seed (winter annual) composing about half of the temporary seed application.

Permanent wetland vegetation, including potted and plug-form shrubs, herbs, sedges, and grasses (Appendix 7, Table A7.3; NCDENR-DWQ, 1997), were planted within each CSW two months after construction (October 1, 2004). These plants were added in the fall to avoid drought, and to allow root establishment over winter. Vegetation was planted at approximately 18-in on center, within the shallow water and shallow land and along the periphery of the pools.

RESULTS AND DISCUSSION

Location Selection and Size Determination

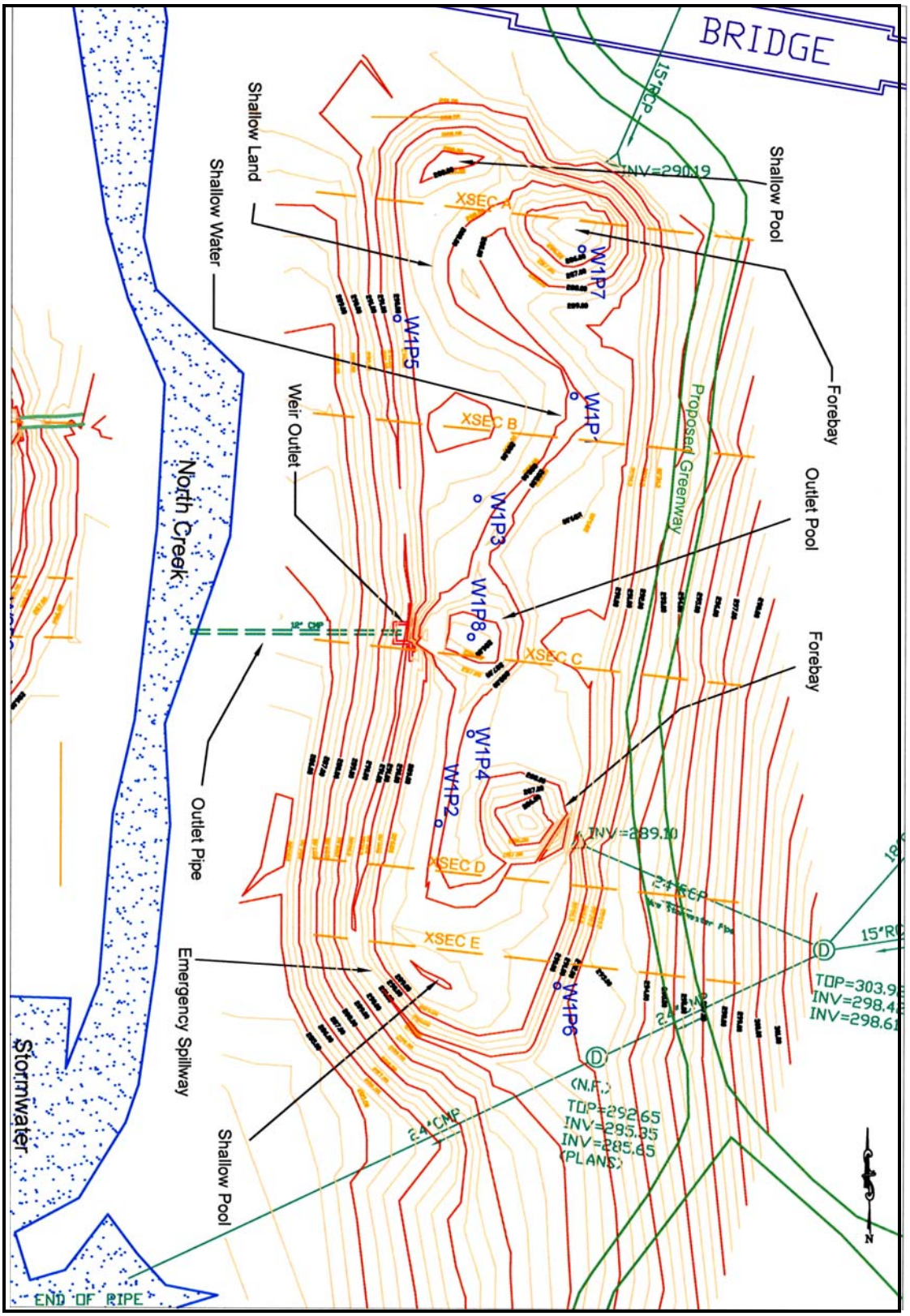
The locations selected for CSW installation were appropriate for stormwater acquisition. Although within the Neuse River riparian buffer (50-ft), placements of the CSWs posed the minimum effects on the buffer as possible. The CSW sizes were adequate to hold the stormwater delivered during the three month period, without filling the wetland volume to capacity.

Internal Wetland Features

As-built surveys were performed after construction to evaluate the finished CSWs based on the designs (Figure 6.6 and Figure 6.7). The pools in W2 were deeper than designed, but this had no detrimental effects on the total storage of the wetland. Actual permanent pool elevations in both wetlands were as designed, with minimal adjustment required of the orifice elevation in both wetlands. At permanent pool the water elevation fills the pools and Shallow Water regions in both wetlands as designed.

The proportion of each wetland feature in the CSWs for the design and the as-built survey were similar. In Wetland 1 the Shallow Water areas were 45% of the surface area compared to the designed 35% of the area due to added vernal pools in the northern and southern corners. These shallow pools were added because there was a shortage of appropriate soil for contouring the Shallow Land region.

Figure 6.6: Constructed Stormwater Wetland 1 As-Built Survey with Research Cross-Sections and Field Plots (Figure by Ian Jewell, NCSU Water Quality Group). Scale 1 in = 35 ft.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Forebay
Outlet Pool
Forebay
Proposed Greenway
XSEC A
XSEC B
XSEC C
XSEC D
XSEC E
W1P7
W1P5
W1P
W1P3
W1P8
W1P4
W1P2
W1P6
Shallow Land
Shallow Water
Weir Outlet
Outlet Pipe
Emergency Spillway
Shallow Pool
North Creek
Stormwater
INV=289.10
2.4"RCP
15"RC
TOP=303.9
INV=298.4
INV=298.61
(N.F.)
TOP=292.65
INV=285.35
INV=285.65
(PLANS)
2.4"CMP
END OF PIPE

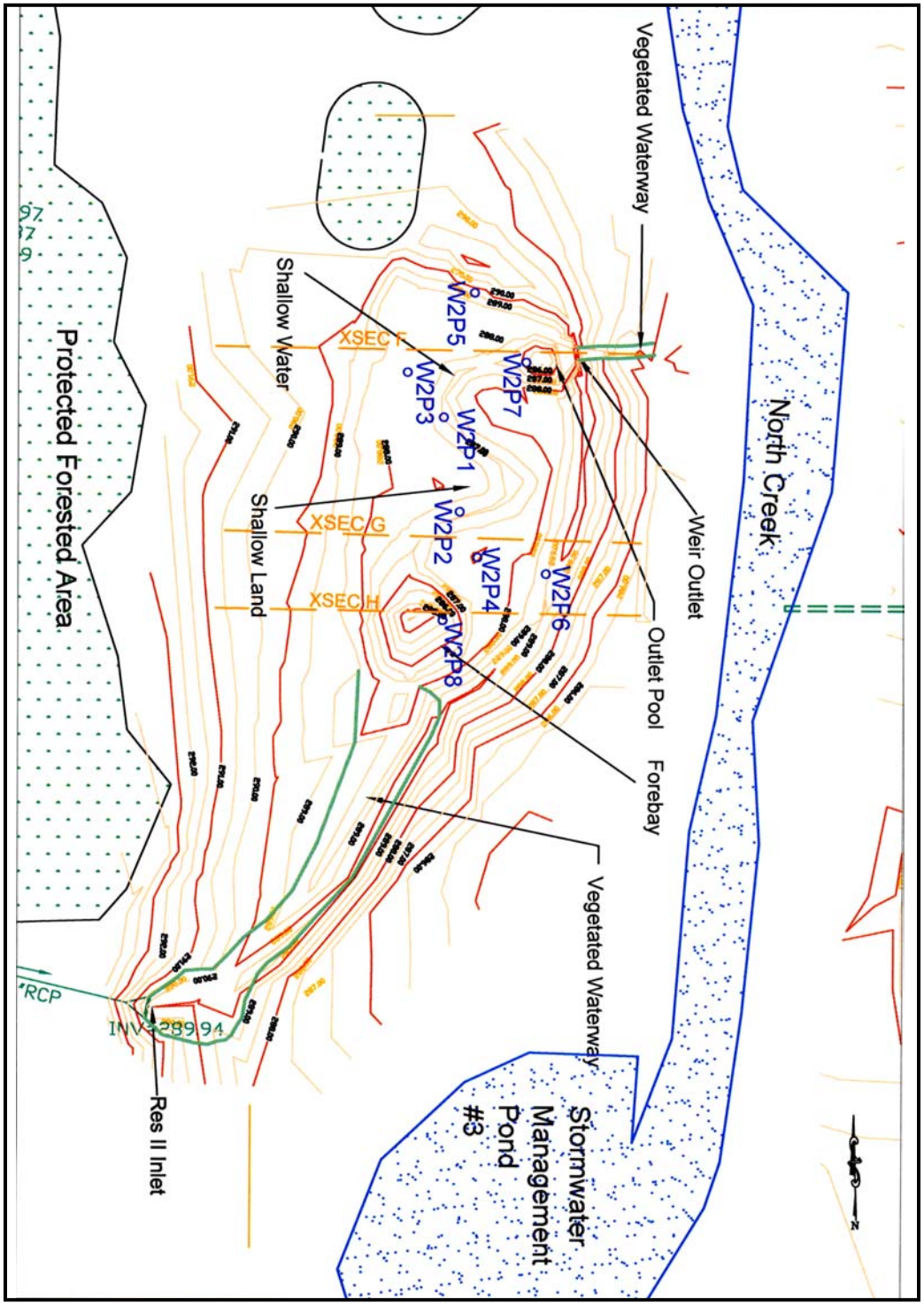

Figure 6.7: Constructed Stormwater Wetland 2 As-Built Survey with Research Cross-Sections and Field Plots (Figure by Ian Jewell, NCSU Water Quality Group). Scale 1 in = 35 ft.

Wetland Depth and Lining

Both wetlands held stormwater throughout the three-month period between storm events. The bentonite liner appeared to be functioning, although there were no long periods without rainfall to fully evaluate the performance.

The use of bentonite as a liner posed a potential problem within the North Creek CSWs. The liner was not installed at the design depth (3.0-3.5 ft below design substrate elevation) specified in the plans. It was instead installed only 0.5-1.5 ft below the surface, with the shallowest installation depths within the pools. This shallow liner affected the pH of the stormwater retained. When the substrate was disturbed for planting or data collection, bentonite was observed suspended in the water column. The pH of the permanent pool water was elevated after disturbance (8.72-11.85), but returned to neutral when the water was sampled a few weeks later (See Chapter 8, Table 8.1). The bentonite from the liner installation was thought to have produced the elevated pH values, based on the pH values of other elements added to the substrate, based on the soils and based on the pH of the stormwater entering the wetland (Appendix 6, Table A6.6).

The problem associated with the bentonite can be addressed by installing the liner uniformly below all features, with a soil buffer between the bottom substrate and liner of at least 1.5-ft. With proper installation depths and minimal substrate disturbance during research, bentonite lining is highly recommended due to low cost and performance.

Inlet and Outlet Structures

The inlet structures in each wetland were evaluated by observation for flow conveyance and energy dissipation. In W1 stormwater was released through each culvert

without erosion to the surrounding berm and forebay areas. In W2 stormwater released from the RES II culvert caused some erosion to the low berm of the plunge pool, but the elevation of the berm in this area was restored through repairs on September 17, 2004. Along the vegetated waterway, a low spot along the berm (288.5) caused short-circuiting of stormwater during large storm events. During the repairs, the berm elevation along the vegetated waterway and the eastern side of the CSW was raised to EL 290.0 in all locations initially below this elevation. The premature loss of stormwater was eliminated after the repair. At the W2 forebay, there was no evidence of erosion from stormwater entering the CSW.

The outlet structures in each wetland were also evaluated by observations and through the flow balance equation (See Chapter 7). There was no erosion in the surrounding areas around the concrete weirs. The weir slats in both wetlands leaked throughout the study. This leakage was evident from observations, as well as from the flow balance equation when the change in storage was much higher than the outflow volume estimated using orifice and weir discharge (See Chapter 7 for a detailed discussion of flow balance). Saw dust was poured just before the weir slats in the pools to help seal the slats, and wood wedges were added along the top slats to tighten the fit in the slat channel. In W1, the TREX boards bowed and were not able to withstand the pressure of the stormwater, even after bolting an aluminum channel (¼-in thickness) in the middle for support. These boards were replaced by untreated plywood (8/26/04) without the environmental seal for future comparison with the W2 weir slats. Leakage was still evident but diminished from this point. In W2 the weir slats experienced leakage but were

adequate to support the stormwater and the leakage rate diminished. Table 6.1 shows the estimated proportion of the total outlet volume during composite storm events that was released through the weir slats as leakage (See Chapter 7 for Methods). The decline in W1 leakage after August 22 was due to weir board replacement (Trex replaced by untreated wood) and aluminum channel reinforcement in the center. The percent leakage decreased over time in both CSWs and is expected to diminish as the wood is sealed with silts and biological materials.

The outlet pipe and vegetated waterway conveying stormwater to North Creek were both stable throughout the study, with no evidence of erosion along the streambank attributed to these structures. There were no storms to evaluate the performance of the W1 emergency spillway, but it is assumed that this structure's stability will increase with time as mature vegetation develops on the surrounding berm and within the erosion control matting.

Table 6.1: Estimated Amount of Leakage at the Outflow (% Total Outflow Volume) During Each Composite Storm Event.

Storm Date	W1 % Leakage	W2 % Leakage
Aug 12-16	91	80
Aug 21-22	92	84
Aug 29-31	72	67
Sept 6-8	56	64
Sept 14-18	44	71
Sept 27-Oct 3	42	55
Oct 13-Oct 15	41	22

Storm Hydrographs and Storm Modeling

During the study period, there were about 30 individual storm events ranging from 0.02 to 2.11 in (Figure 6.8). The total precipitation during the study was 17.5 in, which is much higher than the 30-yr normal precipitation of 10.0 in estimated for the three months (NC State Climate Office, Raleigh, NC). Many of the storm events were associated with five remnant tropical storms moving through the area, with smaller intensities compared to thunderstorms that usually occur in late summer and early fall.

Inflow and outflow hydrographs for both CSWs were determined using the flow volume data over 15-minute intervals, estimated from the stage-storage relationship and the flow balance equation (See Chapter 7). The inflow and outflow hydrographs for each wetland indicated a reduction in peak discharge between the inlets and outlets. Example inflow and outflow hydrographs are shown for a relatively large storm (Aug 21-22, 1.83 in, 0.4 in/hr) in W1 (Figure 6.9) and W2 (Figure 6.10). For this storm, the peak discharge in W1 was reduced from 5.2 cfs to 0.34 cfs (93%), and the peak discharge in W2 was reduced from 1.1 cfs to 0.4 cfs (64%). Stormwater was retained efficiently within the CSWs over time, even with significant leakage observed at the weir structures. The measured peak discharge reductions are similar to the predictions made using the HEC-HMS model for three design storms (Appendix 6, Table A6.5). W1 had predicted reductions in peak discharge for a 1-yr, 24-hr design storm of 83% between the inlet culvert and outlet, while W2 had predicted peak reductions of 68% .

The ability of the CSWs to reduce the total flow was evaluated by comparing the inflow and outflow volumes for composite storm events during the study. In W1 the total

flow was reduced between the inlet and the outlet by an average of 35%, while the total flow in W2 was reduced by an average of 39%. The measured average reduction in total flow was greater than the range predicted using the HEC-HMS model for design storms in W1 (8-18%) (Appendix 6, Table A6.5). For W2, the measured average total flow was within the range of total flow predicted (14-45%). Differences in the model predictions and the measured total flow reductions can be explained by site-specific factors not accounted for in the model. Stormwater volume lost to high evaporation rates or to seepage through the berms above the liner height was not accounted for in the model. The leakage through the weir slats was also not accounted for using the model, and there were uncertainties when estimating inflow and outflow volume due to this high leakage rate.

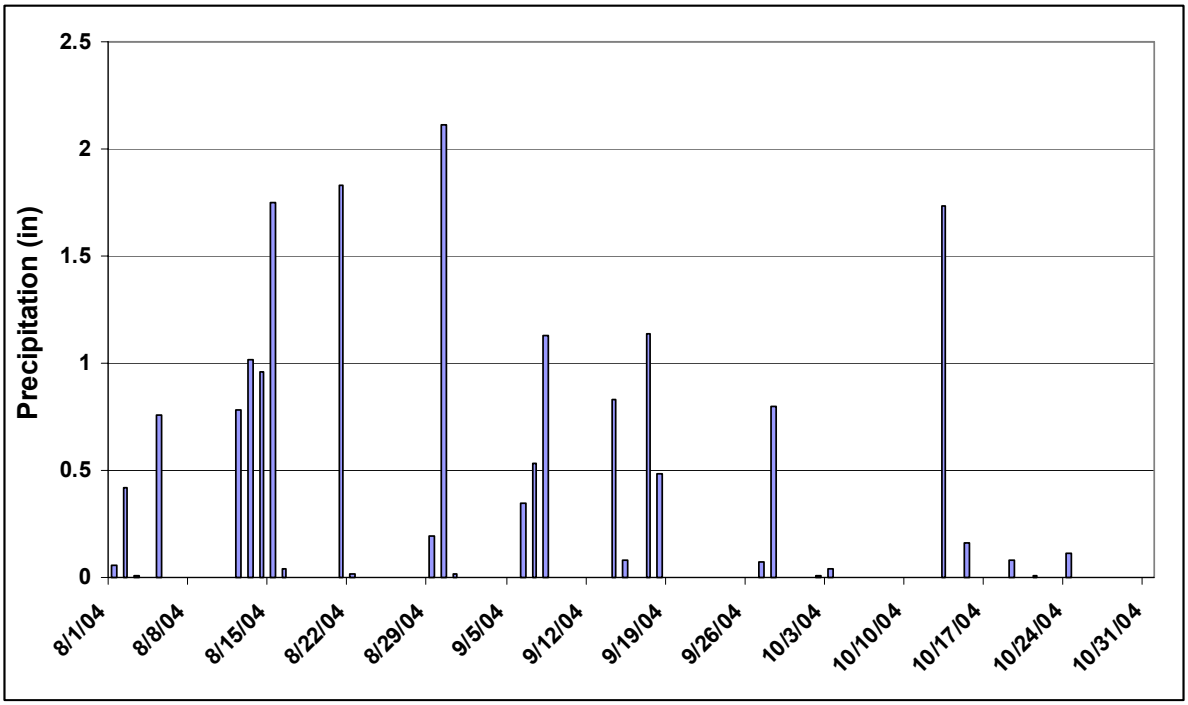

Figure 6.8: Precipitation Events During the Three Month Study (Reference: NOAA Raleigh Regional Office).

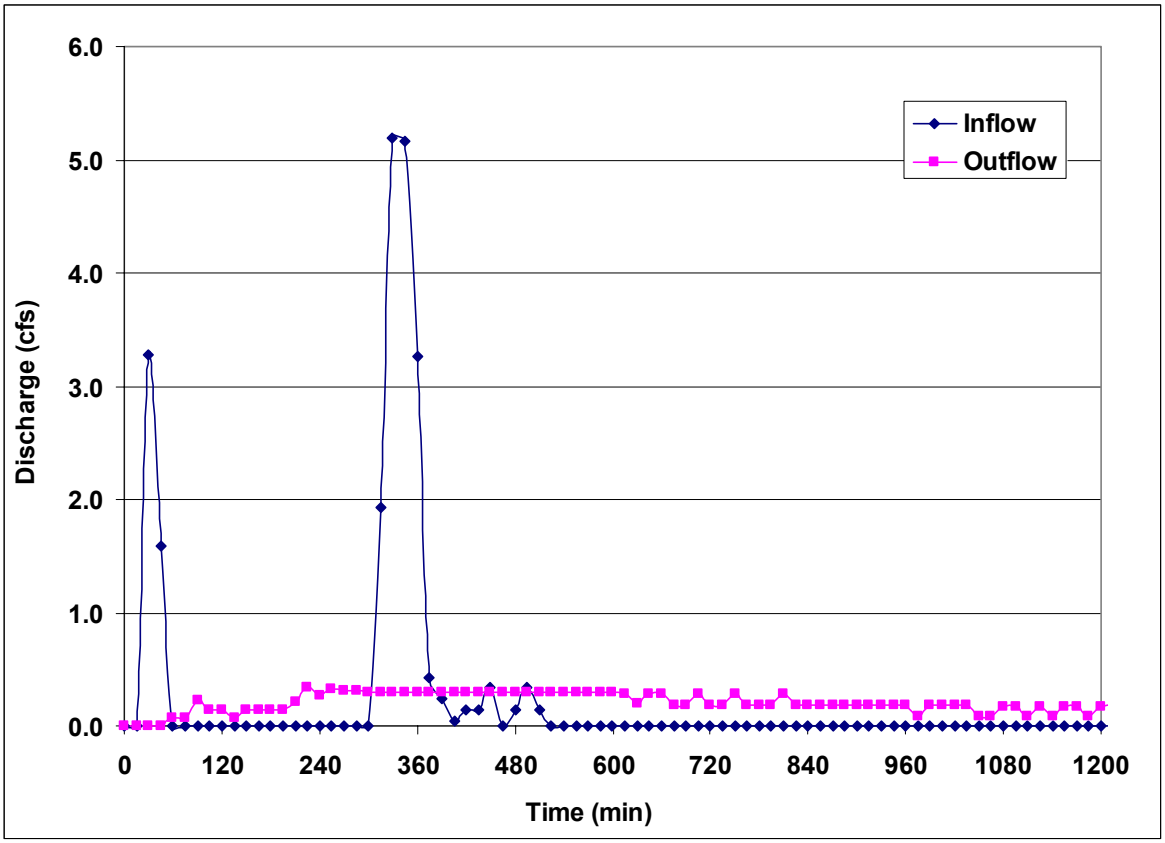

Figure 6.9: Inflow and Outflow Hydrograph for a Large Storm (8/21/04) in Wetland 1.

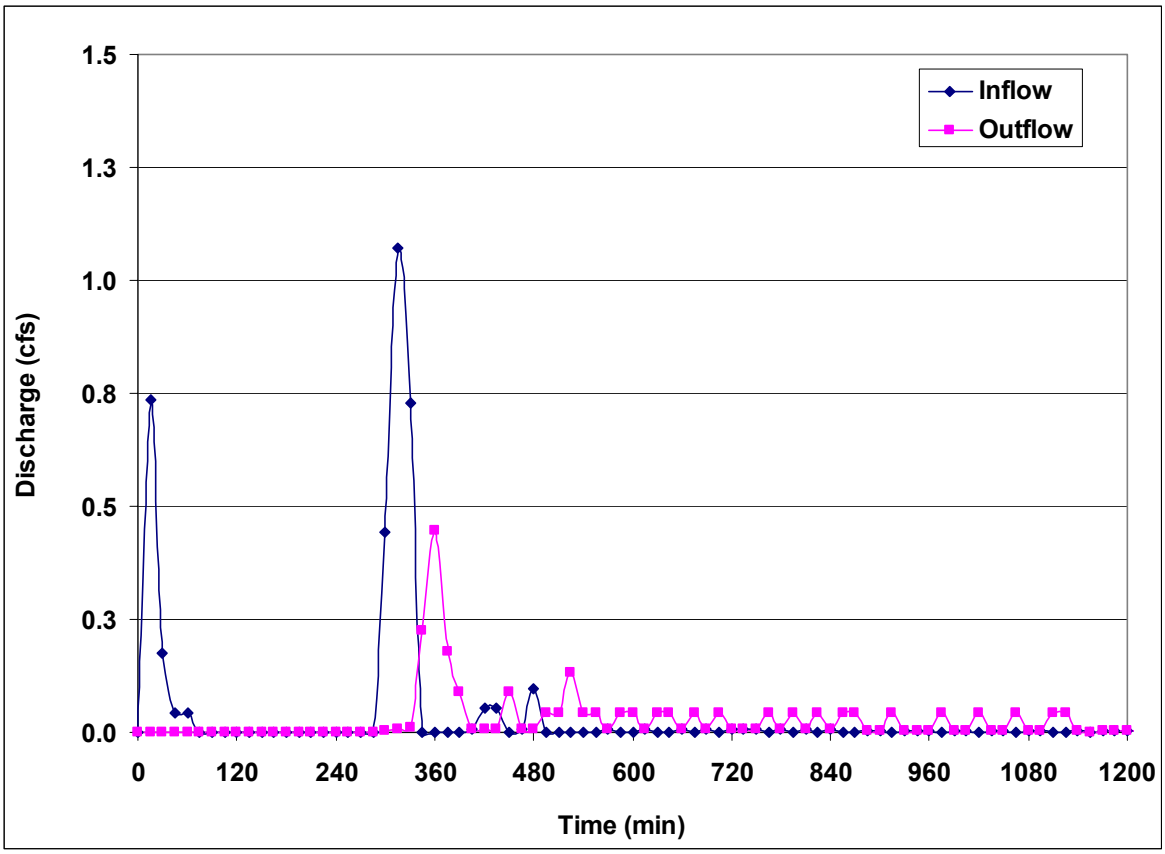

Figure 6.10: Inflow and Outflow Hydrographs for a Large Storm (Aug 21-22) in Wetland 2.

Sediment and Erosion Control

The evaluations of the CSWs for water quality improvements, including sediment and nutrient retention during the initial stabilization period are detailed in Chapter 7 and Chapter 8. This discussion focuses on erosion control observations and vegetation establishment during this study.

The hydromulch erosion control system applied initially was successful at establishing dense temporary vegetation on the berms and into the shallow land areas of both wetlands, and on the upslope areas of W1 disturbed by construction. Photologs are provided in Appendix 7 for W1 (Figure A7.6) and W2 (Figure A7.7) documenting the state of vegetative growth during the study. The Brown top millet emerged within three days, and provided cover during vegetative growth or as fallow through the end of the study. The establishment of temporary vegetation was sparse, however, along the corridors between both wetlands and North Creek, along the construction access path to W1, and in areas surrounding W2. These areas were the last to be treated (August 2, 2004), and a moderate precipitation event followed that evening. The hydromulching was successful on areas treated first (July 29, 2004), with no precipitation for two days. Poor vegetation establishment may also have occurred along these areas due to repeated exposure to equipment and to the high level of compaction along the access paths to both wetlands. A second hydromulch application was performed on all areas where vegetation was not established (Sept 17, 2004). The density of temporary vegetation improved but was not comparable to the berms and W1 upslope areas. The density of temporary vegetation

within the wetland interior was also sparse, but this was expected due to the inundated conditions that were not suitable for these annual species.

Erosion of sediment along the disturbed streambanks was evident during the initial few storms due to poor temporary vegetation establishment and disturbance from the temporary stream crossing. Additional erosion control matting and seed was applied to these areas, with observed reduction in rill formation and sediment entering North Creek during storm events.

Wetland Vegetation

Temporary seeding results are discussed above through evaluation of the hydromulch erosion control system. Sparse emergence of the permanent seeds was observed, but results are not expected until next spring due to the nature of the species selected. The wetland vegetation planting could also not be evaluated during the study period based on the nature of the species.

CONCLUSIONS

CSWs are designed for three main functional goals: stormwater treatment, stormwater retention, and habitat enhancement. In this chapter, the North Creek CSW designs were described and evaluated. Observations and measurements of eight design elements were collected over a three month period after construction. Evaluation of the first stormwater treatment goal will be detailed later in Chapter 7 and Chapter 8 through study of sediment and nutrients. The second goal of stormwater retention was evaluated in this chapter. Both CSWs were successful at retaining stormwater from a variety of storms for slow release over time. The CSWs were able to decrease the peak discharge entering North Creek from the stormwater sources. The third goal of habitat enhancement was not evaluated during the study period and will require more long-term observations and measurements due to plant community dynamics.

In summary, the North Creek CSWs were successfully implemented on the floodplains, and stormwater from the surrounding Centennial Campus (6.1 ac) was effectively captured and retained by these structures.

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CHAPTER 7: EVALUATION OF SEDIMENT AND SUBSTRATE IN CONSTRUCTED STORMWATER WETLANDS DURING INITIAL STABILIZATION

INTRODUCTION

Sediment in Constructed Stormwater Wetlands

Excess sediment contributed to streams and rivers is a major water quality pollutant in the United States (USEPA, 1998). Common sediment sources in urban watersheds are soil erosion, where stormwater entrains sediment from bare, unstable soils associated with construction or gully formation, and particles collected from impervious surfaces. In urban watersheds, stormwater runoff volume and peak discharge are increased, and large pollutant loads including sediment are delivered to the receiving waters (USEPA, 1983). Excess sediment increases turbidity, leads to channel instability and an imbalance in sediment transport capacity, and can contribute significant amounts of other pollutants such as nutrients, metals, and toxic organic compounds that are associated with soil particles (FISRWG, 1998).

Over the past few decades, best management practices (BMPs) have been utilized to reduce the amount of sediment in stormwater. Detention ponds and settling basins are used in areas with large sediment loads like construction sites. These structures are composed of an inlet, a deeper pool region to slow velocity and promote sediment deposition, and an outlet to deliver stormwater at a suitable elevation for the receiving waters (Schwab *et al*., 1993). These structures may be limited in their ability to remove finer particles that remain suspended throughout the retention period and other pollutants that do not settle (Schueler, 2000). For urban watersheds with existing watershed

development and moderate sediment loads, a more suitable BMP may be a constructed stormwater wetland (CSW). CSWs can be enhanced for stormwater retention and sedimentation during larger storm events, similar to a detention pond or settling basin. In addition to sedimentation, these structures have the benefits of variable topography, microhabitats, and dense vegetation that are more effective at stormwater treatment (DeBusk and DeBusk, 2001). CSWs can also reduce the fine suspended particles with associated pollutants that are difficult to settle in pond environments (Campbell and Ogden, 1999). Sediment removal rates observed in CSWs range from 63-94% (Schueler, 1992; Brown and Schueler, 1997; Strecker *et al*., 1992), indicating that these structures are effective at suspended sediment treatment.

For suspended sediment removal, the dominant mechanism in CSWs is sedimentation in the deep pools. Sediment is settled out of the water column by gravity when flow velocity is reduced and deposited onto the wetland substrate for long-term storage (Bingham, 1994). Another sediment removal mechanism occurs in the Shallow Water regions, where dense vegetation physically obstructs the flow path. Suspended sediment and other particulates are filtered out of the stormwater through friction and localized reductions in flow velocity (Bingham, 1994). Plants and their associated microbial biofilms can also adsorb suspended sediment from the water column due to the charged surfaces held by both biological surfaces and clays (Atlas and Bartha, 1993; DeBusk and DeBusk, 2001). The adsorbed sediment can flocculate to the bottom, or fall during plant senescence and microbial sloughing.

Sediment associated with CSWs can be derived from several sources. One source is sediment carried by stormwater runoff entering directly from the surrounding watershed through pipes, ditches, or overland flow. For developed watersheds, these sediment loads are expected to be moderate from paved surfaces, but can be substantial during certain watershed activities. Sediment may be derived from localized construction, from erosion of landscaped areas or gullies, swales, and ditches that route stormwater, or from the breakdown of impermeable surfaces such as pavement and roofs. Average total suspended sediment (TSS) concentrations found for urban stormwater runoff typically range between 80-224 mg/L (USEPA, 1983; MDE, 2000)

Another potential source of sediment associated with CSWs may be generated from within the wetland boundary. The sediment collected within the inlet forebay can be resuspended during stormflows. Forebay regions should be maintained at the frequency dictated by the expected sediment load contributed from the watershed to prevent resuspension. Large stormflows that exceed the retention capacity of the CSW may also create a source of sediment by eroding internal topography and berms. CSWs can be designed to pass these large stormflows through protected spillways, or to route them around CSWs for erosion prevention.

The amount of sediment generated within CSWs may depend on the relative stability of the wetland. During the initial construction and establishment period, the bottom substrate can be unstable before mature vegetation establishes. Without anchoring by roots and vegetative surface protection, this substrate material is vulnerable during stormwater additions and can suspend into the water column. The loss of wetland

substrate is a problem for several reasons. The sediment load generated at the outlet during this initial stabilization period could be significantly greater than the inlet sediment load derived from developed watersheds. Depending on the amount of sediment eroded from within the wetland and the amount of time over which this erosion occurs, long-term stormwater treatment rates may be affected. Loss of wetland substrate can also reduce wetland vegetation colonization. This substrate is usually composed of topsoil or amended with nutrients to promote the rapid establishment of vegetation. When the substrate is removed, nutrient loads at the outlet could increase while vegetation establishment is reduced to perpetuate the erosion potential. Another consequence of substrate loss is the resulting loss of microtopography and internal features that provide the essential treatment mechanisms for CSWs and vegetative diversity.

Based on concerns for substrate erosion and sediment generation within CSWs during the initial stabilization period after construction, a secondary BMP utilized for erosion control can be employed. In this study, the secondary BMP chosen was an application of PAM on the wetland substrates after construction and before inundation.

Polyacrylamide (PAM) as a Secondary BMP

Polyacrylamide (PAM) is synthetic, water-soluble polymer used for decades as a soil conditioner and flocculent in agriculture (Seybold, 1994; Sojka and Lentz, 1996; Lentz and Sojka, 1997). The utilization of PAM has been shown to enhance soil aggregation and infiltration, to flocculate suspended particles, and to reduce suspended sediment in runoff water (Sojka and Lentz, 1996; Roa-Espinosa, 1999). PAM has been shown to be a practical erosion control method based on the low application rates required for

effectiveness (Sojka and Lentz, 1996), the lower cost compared to other erosion control practices (Peterson *et al*., 2003), its non-toxic effects in the environment (Barvenik, 1994), and its relative ease of application over large areas for a variety of site conditions (Lentz and Sojka, 1997).

PAM polymers are composed of long chains of repeating subunits that can bind to soil particles. PAM can enhance soil cohesion and form stable aggregates by creating effective bridges between soil particles (Laird, 1997). When PAM is suspended in the water column, it can act as a settling agent for fine particles that are suspended by binding these particles together. The flocculation of suspended fines not only removes them from the water column but also reduces their ability to intrude into soil pores and reduce infiltration (Sojka and Lentz, 1996).

There are several mechanisms that facilitate the binding of PAM molecules and soil particles. Cation bridging (example: PAM-Ca-Clay bond) occurs between negatively charged PAM molecules and clay particles (Theng, 1982; Laird, 1997). Divalent cations associated with water help bind the PAM and soil through Van der Waals forces (Seybold, 1994; Sojka and Lentz, 1996). This bridging interaction is dependent on the availability of exchangeable cations, the amount of clay, the pH, and the size of the PAM polymer (Theng, 1982; Lu *et al*., 2002). PAM can also bind directly to negative clay particles that have broken edges with protonated nonbridging aluminol groups (Laird, 1997). The long chain length of the PAM polymer also facilitates the binding mechanism by providing many sites for soil particle associations along a single large molecule (Laird, 1997). The effectiveness of PAM depends on the amount of binding sites available along the molecule

and the relative amount of fine particles composing the soil (Malik and Letey, 1991; Seybold, 1994; Lu *et al*., 2002).

PAM properties can be selected for the most effective performance in erosion control based on soil type (Nadler *et al*., 1996). PAM should have a high molecular weight (12-15 million grams per mole) for increased chain length (Lentz and Sojka, 1996). The anionic form of PAM should be utilized with less than 0.05% acrylamide monomer due to toxic effects found for the cationic form (Goodrich *et al*., 1991; NRCS, 2001) and due to the enhanced ability of the anionic form to uncoil in solution (Laird, 1997). Recommended charge densities for PAM are between typically 8-35% by weight and are dependent on the type of soil at the application site (Sojka and Lentz, 1996; NRCS, 2001).

The application of PAM in the environment can be accomplished by spreading the polymer when dry across the soil surface or by dissolving it in water for spraying. NRCS (2001) recommends an application rate of less than 200 lb/ac per yr to be applied at a concentration of 10 ppm for agricultural purposes. Sojka and Lentz (1996) found application rates of 1 lb/ac were effective in furrow irrigation water for each application. For erosion control applications on bare surfaces, a rate of 20 lb/ac was recommended (Dr. Rich McLaughlin, NCSU, personal communication). The use of application rates larger than those recommended by NRCS (2001) have been shown to reduce the effectiveness of PAM and increase erosion (Sojka and Lentz, 1996, Roa-Espinosa, 1999).

The interaction of PAM and the environment has been well studied. The PAM polymer is relatively resistant to microbial decomposition, until broken down by a combination of biological, photochemical, and physical degradation. The acrylamide

portion is readily biodegradable once exposed and can be utilized as a nitrogen source by soil microorganisms (Kay-Shoemake and Watwood, 1996). Studies on the interaction of PAM and plants found no adverse effects on seedling emergence or plant growth, with essentially no absorption by plants (Wallace and Wallace, 1986). Plant seedling emergence, growth, and yield were all improved when PAM was applied with a nutrient solution (Wallace, 1987). The release of PAM into receiving waters has also been shown to have no adverse affects on aquatic life (Barvenik, 1994). Although the cationic form of PAM can be toxic to fish (Goodrich *et al*., 1991), the anionic form is safe for fish at high concentrations (McCollister *et al*., 1965). Krautter *et al*. (1986) showed no adverse effects on daphnids, midges, and three types of fish with levels below 100 mg/L in intermittent flow. PAM has also been shown non-toxic to humans and is considered an EPA Acceptable Drinking Water Additive as long as it complies with the legal limit of less than 0.05% acrylamide monomer (Sojka and Lentz, 1996).

The mobility of PAM after application is of concern when considering off-site effects as well as the length of time PAM is effective where applied. The adsorption of PAM to soil appears to be irreversible once the soil is dry (Laird, 1997). Mobility of PAM within the soil has been shown to be limited, with minimal leaching over time (Nadler *et al*., 1994). Furrow irrigation studies have shown that average PAM concentrations leaving agricultural fields were below detectable limits in the receiving waters when applied according to NRCS recommended standards (Lentz *et al*., 1998). The persistence of PAM within the soil is variable and dependent on soil and site conditions as well as frequency and concentration of the application. Nadler *et al*. (1994) found PAM in the

upper 25 cm of the sandy loam and clay loam soils after 10 months, but Fox and Bryan (1992) found the effects of PAM only lasted for 6 weeks in semi-arid regions. Both Hayes (2003), and Soupir *et al*. (2004) observed diminishing results during their short-term construction site studies, although no measurements of residence time were made. The limited persistence of PAM may be a disadvantage when utilizing it in erosion control applications, and reapplication may be required.

History of PAM Research

Studies concerning the use of PAM began over 50 years ago in agricultural practices (Seybold, 1994), and have developed over time in number and applications. PAM was first used in the western United States for soil conditioning, and then added to furrow irrigation water where it successfully reduced sediment yields (over 90% in most studies), total phosphorus loss, and pesticide concentrations in the runoff water (Sojka and Lentz, 1996). Other applications over the past few decades include sprinkler irrigation (Bjorneberg *et al*., 2000), and many industrial applications, including wastewater treatment and paper and paperboard production (Seybold, 1994). During the last ten years, research on the utilization of PAM for erosion control during development has increased. PAM has been applied to steep slopes and bare surfaces to reduce sediment loss and runoff volume, while facilitating vegetation establishment. NRCS recommends the use of PAM for erosion control on sites where adequate vegetation cannot be established in a timely manner (NRCS, 2001).

The results from the research evaluating PAM for erosion control are variable, with various application rates and application methods employed. Roa-Espinosa (1999)

evaluated the effects of PAM on steep slopes at a construction site with rainfall simulation. PAM was applied dry and dissolved in solution, on both wet and dry soils, with and without the addition of mulch and seed. Sediment yield was reduced in runoff water for all PAM applications, with the largest reductions (average 93%) when PAM was applied in solution (123.5 lb/ac), to dry soils, and in combination with mulch and seed. Similar simulated experiments were conducted at the University of Georgia on steep slope construction plots (Glazer, 2001). There was no significant difference in sediment loss between plots receiving a hydroseed mixture with PAM (40 lb/ac application rate), and plots receiving the hydroseed mixture alone.

Another study by Soupir *et al*. (2004) at Virginia Tech evaluated the loss of sediment on construction field plots under simulated rainfall. PAM was applied both dry and dissolved, with different liquid application rates (half recommended, recommended 12-37 lb/ac, double recommended). Comparisons were made between hydroseeding with and without PAM, and between straw mulch applications. The overall results of the study found that straw mulch alone was most effective for reducing total suspended sediment (TSS) in runoff water. TSS concentrations were reduced by 90% with the straw mulch alone, followed by the hydroseeding without PAM (50%). Both liquid and dry PAM applications were comparatively effective at sediment reduction, but doubling the recommended application rate actually increased runoff and sediment loss. All treatment methods showed diminishing results over time, and none of the treatments were able to reduce total runoff volumes.

Research on construction sites subjected to natural rainfall conditions during the wet season was conducted by the Washington Department of Transportation (Tobiason *et al*., 2001). PAM was applied alone, with hydromulch, and with straw applications. For all treatments, PAM significantly reduced turbidity in runoff water from disturbed sites for six weeks or more. The PAM and hydromulch combination performed the best (94-99% reduction), followed by the use of PAM alone (88-90% reduction). Due to differences between application doses, it was recommended that application be tested for the optimal application rate.

Research on construction sites in North Carolina subject to natural rainfall was also conducted at NCSU (Hayes, 2003). PAM was applied alone and in seed/mulch combinations at the recommended rates for the PAM product (7.9-10 lb/ac), and half the recommended rates. The results showed that the mixture of PAM, seed, and mulch decreased erosion rates significantly, but not as much as seed and mulch applied alone in comparison to bare soil. Seed and mulch alone were most effective at reducing runoff, turbidity, and sediment loss from the sites. The effects of PAM were more significant on steeper slopes, with increased effectiveness at the higher rates and diminishing effectiveness of PAM observed over time.

Besides construction site applications, PAM has been applied for erosion control on unstable substrates subjected to concentrated flows. Peterson *et al*. (2003) looked at the effects of PAM applied in an experimental earthen waterway on erosion and headcut migration at the USDA-ARS Hydraulic Engineering Research Unit in Stillwater OK. The results showed sediment yield reductions (93-98%), and reductions in advancement rates

of headcuts formed by knickpoints (reduced from 17.8 m/hr in untreated to a maximum of 0.6 m/hr in PAM treated channel). The PAM application rate was high, at 439 lb/ac and was applied to the channel soils before each exposure to flow.

The Application of PAM in CSWs

There have been no studies published to date on the use of PAM in association with CSWs. The expected effects of PAM must be derived from agriculture and erosion control research, as well as account for the heterogeneous nature of CSW features and their variable exposure to stormwater.

The application of PAM on the berms and surrounding upslope surfaces of CSWs is comparable to its use for erosion control on construction sites with bare soils and steep slopes. These regions are subjected to the erosivity by rainsplash and during sheetflow runoff across the surface. The results of PAM applications on these types of sites were variable for sediment reduction in the runoff water and dependent on the application method and rate.

The application of PAM on the Shallow Land and at the base of the Upland berms of a CSW may be comparable to its use in furrow irrigation and experimental waterways, based on intermittent but frequent inundation. For both methods, however, the PAM application was performed during each inundation. The experimental waterway research also used PAM application rates much higher than recommended rates for erosion control. The reduction in sediment loss for both of these PAM applications was significant, and results from furrow irrigation studies demonstrated the added benefit of increased infiltration through soil conditioning facilitated by the PAM. Similar reductions in

sediment and improvements in infiltration are possible for these internal wetland features. The effectiveness of PAM in the CSW application may be dependent on the frequency of inundation, and persistence of PAM associated with the soil in these features.

The application of PAM in the Shallow Water and Deep Pool regions of CSWs was not comparable to other erosion control studies due to inundated conditions. During the initial inundation after PAM application, PAM not associated with sediment could be dissolved in the water column and aid in the flocculation of suspended sediment. It is uncertain, however, if this flocculation will occur and if flocculation will persist after the initial inundation.

CSW Sediment and PAM Study

The purpose of this study was to evaluate the substrate stability and sediment removal capacity of CSWs during the initial stabilization period after construction. The performances of two CSWs (W1 and W2) on the floodplain of North Creek were assessed based on their ability to remove and retain sediment from stormwater, and based on their ability to maintain substrate stability. The purpose of this study was also to evaluate the use of PAM as a secondary BMP for erosion control and wetland substrate stabilization. The wetland sites were treated using hydromulch with (W1) and without (W2) PAM after construction for comparison.

The following objectives were derived for the study:

1. To determine if total suspended sediment (TSS) concentrations, turbidity, and TSS loads were reduced between the inlet and outlet of each wetland.

2. To determine if there was a difference between the TSS concentrations, turbidity, and TSS loads released at the outlets of W1 and W2.

3. To determine what substrate changes were occurring over time in W1 and W2, including elevation, bulk density, and particle size composition.

4. To determine if the PAM application within and surrounding W1 provided erosion control and substrate stability.

METHODS

Polyacrylamide (PAM) Application

After internal wetland contouring was completed and topsoil added (July 29, 2004), hydromulch with permanent and temporary seed was applied to all surfaces within the wetlands and on the surrounding berms and disturbed areas using the U. S. Environmental Protection Services erosion control system (see Chapter 6 and Appendix 7 for details). PAM (APS 700 Series Silt Stop, APS Polymers, Inc., Woodstock, GA) was added to the mixture and applied on only the W1 site at a rate of 15 lb/ac, which was greater than the manufacturer's recommended application rate of 10 lb/ac. Hydromulching occurred on July 29, 2004 for the W1 site and August 2, 2004 for the W2 site, with significant rainfall events occurring on July 31, 2004 (1.07 in) and on August 2, 2004 (0.42 in) after PAM application. An additional hydromulch application was performed on September 17, 2004 to surrounding patchy areas of both wetlands where temporary vegetation had not established. This hydromulch mixture contained PAM (15 lb/ac), but was not applied to the W2 interior.

Weir slats were installed one week after initial hydromulching to allow temporary vegetation establishment within the wetlands. Two storm events occurred during this time period allowing stormwater to fill the Deep Pools and Shallow Water regions.

Total Suspended Sediment and Turbidity

Before wetland construction, grab samples were collected during a large storm (0.75 in) to get an idea of average TSS concentrations and turbidity levels at the existing culvert inlets and outlets to North Creek. The EGRC inlet culvert for W1 was not installed

until wetland construction, so no preliminary data were available. This culvert was manually sampled on 8/2/04, 8/12/04, and 8/15/04 to compare inlet concentrations with the established COT culvert.

After wetland construction was completed, suspended sediment samples were collected using automated samplers (Teledyne ISCO Model 3700). Samples were collected between August 2, 2004 and October 31, 2004 from the inlet and outlet of each wetland. In W1 the inlet intake tube was placed within the College of Textiles inlet culvert near the edge, and the outlet intake tube was placed on a rod immediately in front of the outlet weir about mid-height between the orifice and base of the weir slats. The EGRC culvert was not sampled and assumed to have similar sediment inputs as the COT culvert due to similar watersheds. In W2 the inlet intake tube was placed within the Research II culvert near the edge, and the outlet intake tube was placed just in front of the outlet weir at a comparable elevation to W1. The samplers were activated and deactivated using flow actuators placed near the intake tubes. Samples (500 ml) were collected every 15 minutes after activation and composited every four hours or less during the rise in wetland stage, and every 8 hours or less during the 2-3d falling stage to determine the best average TSS and turbidity measurements for each storm. To verify the accuracy of the automated samplers, manual grab samples were collected during the first few storms at all inlets and outlets for comparison.

After sample collection, the water samples were processed in the NCSU Water Quality Group Laboratory for total suspended sediment (TSS) concentration and turbidity. For TSS determination, 100 ml aliquots were extracted from each sample after vigorous

shaking and filtered using glass microfiber filters (Whatman 934-AH, 47mm). The filters were dried at 110°C for 48 hrs, and weighed to determine the sample concentration (mg/L). Turbidity measurements (NTU) were performed on aliquots from each sample after vigorous shaking using a turbidity meter (Hach 2100P Turbidimeter).

Sediment loads were calculated at the inlets and outlets of each wetland for each storm event using TSS concentration data and stage data. Stage heights in each wetland were recorded every 15 minutes within the outlet pools using a water level logger (Global Logger Model WL15, ±0.2% accuracy). The logger was anchored at the bottom of the pool within 1½ in PVC pipe with small holes along the pressure transducer, attached to a cinder block for stability. The storage volume at each 0.25-ft contour interval was estimated using the as-built survey and AutoCAD Land Desktop3 (Autodesk, 2003). Using the stage and volume data, a stage-storage function was then derived for each wetland in order to determine the storage volume at corresponding elevations over time (Appendix 6, Figure A6.1 and A6.2). The stage-storage function, $S = K_s * Z^b$, was determined from the log-log plot of cumulative storage vs. stage; where S is the storage in cu ft, Z is the stage in ft, K_s is the antilog of the intercept and b is the x-variable coefficient for a fitted regression line of the plot (Malcom, 1995). Once this equation was established for each wetland, the stormwater volume stored at each stage recorded over time was calculated.

To calculate sediment load, the volumes of stormwater entering and leaving the CSWs were calculated. The inflow and outflow volumes for each wetland over time were estimated using the basic flow balance equation, dS/dt = Inflow – Outflow; where dS/dt is

the change in storage volume over the change in time (Schwab *et al*., 1993). The change in storage volume was estimated for each 15-minute interval of stage height data using the stage-storage function.

The outflow volume from each wetland was determined as the sum of stormwater released by the orifice, by the weir, through evaporation, and from estimated leakage through the weir slats during each time interval. Infiltration was assumed negligible in this study due to the liner. The standard orifice equation (Malcom, 1995) was used to calculate orifice discharge, $Q = A*C_d*SQRT(2*g*h)$; where Q is the orifice discharge (cfs), A is the area of the orifice pipe (sq-ft), C_d is the coefficient of discharge equal to 0.6 for typical pipes, g is the acceleration due to gravity (ft/s^2), and h is the height of water over the orifice elevation (ft). The basic weir equation (Malcom, 1995) was used to calculate weir discharge, $Q = C_w * L * H^{3/2}$; where Q is the discharge (cfs), C_w is the weir coefficient equal to 3.3, L is the length of the weir slats (ft), and H is the driving head of water over the weir slats (ft). The volume for each fifteen-minute time interval released by the orifice and weir was determined by multiplying the discharge by time. The amount of outflow volume attributed to evaporation was estimated using average pan evaporation data (NC State Climate Office, Raleigh, NC), determined from a 50-yr data record at a station in Chapel Hill, NC. The average evaporation for August (0.3 in/d), September (0.2 in/d), and October (0.2 in/d) was multiplied times the CSW surface area, and converted to cubic feet per each 15-minute time interval.

The volume of leakage through the weir was estimated using change in storage over the 15-minute increments and assumptions. Leakage was only determined if the

change in storage was negative (loss of volume). Leakage could not be estimated when the volume was increasing; therefore, leakage volume per time interval was assumed zero during inflows. When the CSWs were losing volume, leakage was estimated as the change in storage during each 15-minute interval (positive value) minus volumes lost to orifice flow, weir flow, and evaporation. This method effectively accounted for leakage during weir replacement and maintenance, and between storms as the water level continued to drop much faster than evaporation could be occurring.

Once the total outflow volume was estimated using the decline in storage, the inflow volume was estimated by adding the outflow volume to the change in storage as described by the water above balance equation. Inflow volumes were only determined when the change in storage per 15-minute time interval was positive. The inflow intervals were checked using rain gage data (NOAA Raleigh Regional Office, Centennial Campus, NCSU) and recorded observations from a location in the watershed.

The flow balance equation estimations of inflow and outflow volumes described above had a high level of uncertainty due to lack of flow data at the inlet and outlet, and due to the large amount of outflow volume observed to be lost through leakage. The inflow and outflow estimates were considered adequate for the CSWs and load estimations, due to watershed and stormwater runoff characteristics. Inflows were only determined when there was an increase in storage during each time interval. This estimate neglects the time periods when inflow volume is being contributed but not in greater amounts than what is leaving as outflow. This error occurred usually during the initial portions and end portions of storm events when there was a small amount of inflow. The

error was minimal for most storm events, due to the rapid delivery of stormwater runoff from the watershed. Stage increased rapidly after the onset of precipitation. Another source of uncertainty was the estimation of leakage only when the stage was falling. Leakage was still occurring during inflow periods, but could not be estimated when the change in volume was increasing during storm events. The error was again considered minimal due to the rapid runoff response to precipitation, and the short duration of most inflow events.

Once the inflow and outflow volumes for each 15-minute time interval were determined, the total suspended sediment load at each sampling location was then estimated by multiplying the measured TSS concentrations at the inlet and outlet by the inflow and outflow volumes, respectively. For load calculations during each 15-minute time interval, the TSS concentrations were converted from mg/L to lb/cu ft. The total sediment load was then determined for each storm event at the inlet and the outlets of each wetland by summing the loads during the rising and falling stages above the permanent pool. Storm events had to be composited when the permanent pool elevation was not reached before the next storm event occurred.

Topsoil Surveys and Wetland Plots

Five cross-sections were established in W1 and three cross-sections were established in W2 (See Chapter 6, Figure 6.6 and 6.7). The cross-sections were surveyed initially before stormwater was added, then two months and three months post-construction. Surveys were performed using an electrical total station (Topcon GTS-211D) and survey rod. Survey data were input into AutoCAD Land Desktop3 (Autodesk,

2003) to estimate changes in wetland bed elevation over time. Cross-sectional areas between top of berm elevations were determined using interval-area summations in Microsoft Excel.

Field plots were established in both wetlands just after construction to measure substrate changes in wetland features over time (See Chapter 6, Figure 6.6 and 6.7). Each plot was composed of two rebar stakes (2-ft length, with at least 1 foot below the surface). The length of stake exposed was measured immediately after construction and again a month afterwards. Two plots were established per wetland feature in each wetland (Deep Pool, Shallow Water, Shallow Land, and Upland/Berm) and averages were determined for each feature in each wetland during the sampling events.

Bulk Density and Particle Size Analysis

Soil samples were taken in each plot to determine bulk density and particle size composition using a standard cylindrical metal standard core (3-in dia, 3-in h). For inundated plots, the samples were collected by easing the core into the substrate until it was flush with the substrate, then capping the bottom with the palm of the hand while lifting the sample out of the substrate. The core sample was quickly emptied into a plastic bag, with as minimal water loss as possible. Substrate samples were taken just after construction, and at one month and three months post-construction.

The soil samples were taken to the NCSU Water Quality Group Laboratory and air dried for three days. The samples were then oven dried 230 °F (110 ºC) for 48 hrs. Sample weights were recorded before and after oven drying, with sample bags rinsed clean to avoid any sample loss. Bulk density was determined by dividing the oven-dry weight

(g) by the volume of the soil core (g/in^3), and converted to pounds per cubic foot for comparison.

The particle size composition of each sample was determined using 100g of the original oven-dried soil sample. Each sample was ground using a mortar and pestle to separate individual particles as much as possible. Samples were then sieved using standard metal mesh sieves (U.S.A. Standard Testing Sieves, ASTM E-11 Specifications) ranging from 16 mm to 0.063 mm following the Wentworth size categories. Since the samples were relatively small, the sieves were manually shaken for at least a minute per sieve before the soil material in each sieve was collected. Soil from each sieve size class was weighed, and compared to the total weight of the sample before sieving to ensure minimal loss of soil material during processing.

Fine particles passing through the last sieve that were < 0.063 mm, including fine sand, silt, and clay, were collected separately for particle size analysis. Fine particles were mixed to a final concentration of 1% by volume or less, in a solution of distilled water and sodium hexametaphosphate (5% solution, 50 g/L) as the dispersant. The suspension was a 4:1 mixture of distilled water to dispersant.

Particle size analysis was performed using a settling method (Marshall *et al*. 1996). Settling times for sand, silt, and clay were determined using Stokes equation in metric units: $t = (18\eta h)/(\rho - \rho_0)gd^2$); where t (s) is time for a particle to fall h distance (cm), ρ is particle density (g cm^{-3}), ρ_0 is density of water, g is acceleration due to gravity (cm s^{-2}), d is the average equivalent sphere diameter (cm), and η is viscosity of water (Pa s)

(Loveland and Whalley, 1991). For these samples, sand settled out in 40s, silt in 8.5 hr, and the remainder to the sample was assumed to contain the clay fraction.

To separate particle sizes, the samples were shaken vigorously and poured into the bottom-withdrawal burette with the settling time beginning immediately. Each settled particle class was separated by opening the burette valve and collecting the soil into a beaker. The contents of the beaker were poured into pre-weighed aluminum tins and the volume of liquid recorded to determine the added weight of the dispersant for each sample. The samples were then oven-dried at 110 °C for 48 hours and weighed. The weight of each particle size fraction was determined by subtracting the tin weight and the estimated dispersant weight from the total. The percentage sand, silt, and clay in each sample was determined on a weight basis using the particle size analysis data and the sieve data for larger sand sized particles. Particle sizes larger than sand were disregarded in the percentage determination.

Statistical Analysis

All descriptive statistics and correlations were performed using JMP 5, a statistical analysis package (SAS Institute, 2003).

RESULTS AND DISCUSSION

Total Suspended Sediment Concentrations and Turbidity

TSS and turbidity samples were collected starting the last day after construction. Due to sampler malfunctions, there were several missed samples at various locations during the initial weeks, but an effort was made to sample each storm at both the inlets and outlets. Missing bars in the figures represent missing data, not measured zero values. The samples were composited over 4-hr intervals at the inlets and composited over 4-8 hr intervals at the outlets over the 1-3 d retention time. The average TSS concentration and turbidity measurement during each storm sampled at the inlets and outlets was determined.

The TSS concentrations entering W1 were generally less than those entering W2 during the three months (Figure 7.1). Inlet average TSS concentrations in W1 remained relatively constant below 25 mg/L for all storms, except for the last storm where sediment additions were much higher (227 mg/L). Inlet average TSS concentrations in W2 remained relatively constant and generally below 50 mg/L for all storms until after 9/17/04, when the average values increased (162-207 mg/L) for the remainder of the study (Figure 7.1). Average turbidity measurements at the inlets followed similar patterns as the average TSS concentrations (Figure 7.2). At the W1 inlet, average turbidity was below the 50 NTU standard for generally all storms except the last event. At the W2 inlet, average turbidity was also below the standard until the several storms after 9/17/04 where turbidity ranged between 81 and 212 NTUs.

The increased TSS concentrations and turbidity measurements observed in W1 during the last storm may have been caused by sediment collecting in the culvert around

the intake line. There was no observed change in the watershed to explain this increase. The increased TSS concentrations and turbidity measurements observed after 9/17/04 at the W2 inlet were probably caused by site disturbances upslope near the stormwater grates by Research II. Heavy equipment entered the site near this grate for wetland repairs on 9/21/04, and utility work occurred at this time upslope of the stormwater intakes in association with the North Shore residential development.

Average TSS concentrations at the outlets were much higher than inlet measurements during the first few weeks after construction in both wetlands (Figure 7.1). The greatest average concentration was measured during the first storm, followed by a decline in each successive storm. Average outlet concentrations were more than three times greater in W2 compared to W1 during the first three storms. Throughout the entire study, average outlet concentrations were greater in W2 compared to W1. After the first month of storm events, the average outlet concentrations were below 50 mg/L even with the large increase in average inlet concentrations measured in the last several storm events. Average turbidity measurements again followed similar patterns as average TSS concentrations (Figure 7.2). Outlet average turbidity measurements were generally below the 50 NTU standard by the end of the first month in both wetlands.

Elevated TSS concentrations and turbidity measurements at the outlets were expected, especially during the first few weeks, due to substrate instability. Stormwater was diverted into both wetlands with bare soils exposed. The loose soil particles at the surface of the topsoil layer were probably easily suspended and delivered to the outlet. Over time, the substrate stability increased through the establishment of temporary

vegetation on the Upland berms and Shallow Land regions and through the loss of all loose particles near the surface. Inundated portions, including the Shallow Water and Deep Pool regions, also could have stabilized over time with reductions in suspended sediment contributions. The existing ponded water could have dissipated the entering stormwater energy to reduce erosion and encourage deposition rather than resuspension of topsoil.

Throughout the study, the average TSS concentrations and turbidity values measured at the outlet of W2 were greater than the average measurements at the outlet of W1. For storm events during the last month of the study, both CSWs were reducing the average TSS concentrations and average turbidity values between the inlet and outlet. Even though W2 had greater average TSS concentrations at the outlet compared to W1, the percent reductions were generally much higher in W2 (87%-89%) compared to W1 (18-36%). The exception occurred during the last storm measured (10/24/04), when average inlet concentrations were reduced more at the outlet of W1 (96%) compared to W2 (72%) (Figure 7.1). The average inlet TSS concentration in W1 during this storm was not consistent with all other storms, however. The same trend was observed for the average turbidity values during the last four storm events measured (Figure 7.2). Differences between the performance of the two CSWs may be attributed to the amount of sediment entering at the inlet to each wetland. TSS loads per wetland acre were calculated to provide more information about the differences between the two CSWs, with results and discussion to follow.

A few preliminary stormwater samples were collected before wetland construction for a general idea of differences with addition of the CSWs. These initial samples were

relatively low in average TSS concentrations (21-33 mg/L) and in average turbidity measurements (13-37 NTU) at both the inlets and the outlets to North Creek. During initial stabilization, the TSS concentrations and turbidity measurements were greater than the preliminary samples at the outlets. By the end of the study, the outlet sediment amounts were similar to the preliminary samples. Due to the limited number of samples prior to construction, the effects of the CSWs on sediment reduction from the stormwater sources on the floodplain could not be adequately assessed. The results suggest, however, that the CSWs stabilized by the end of the study.

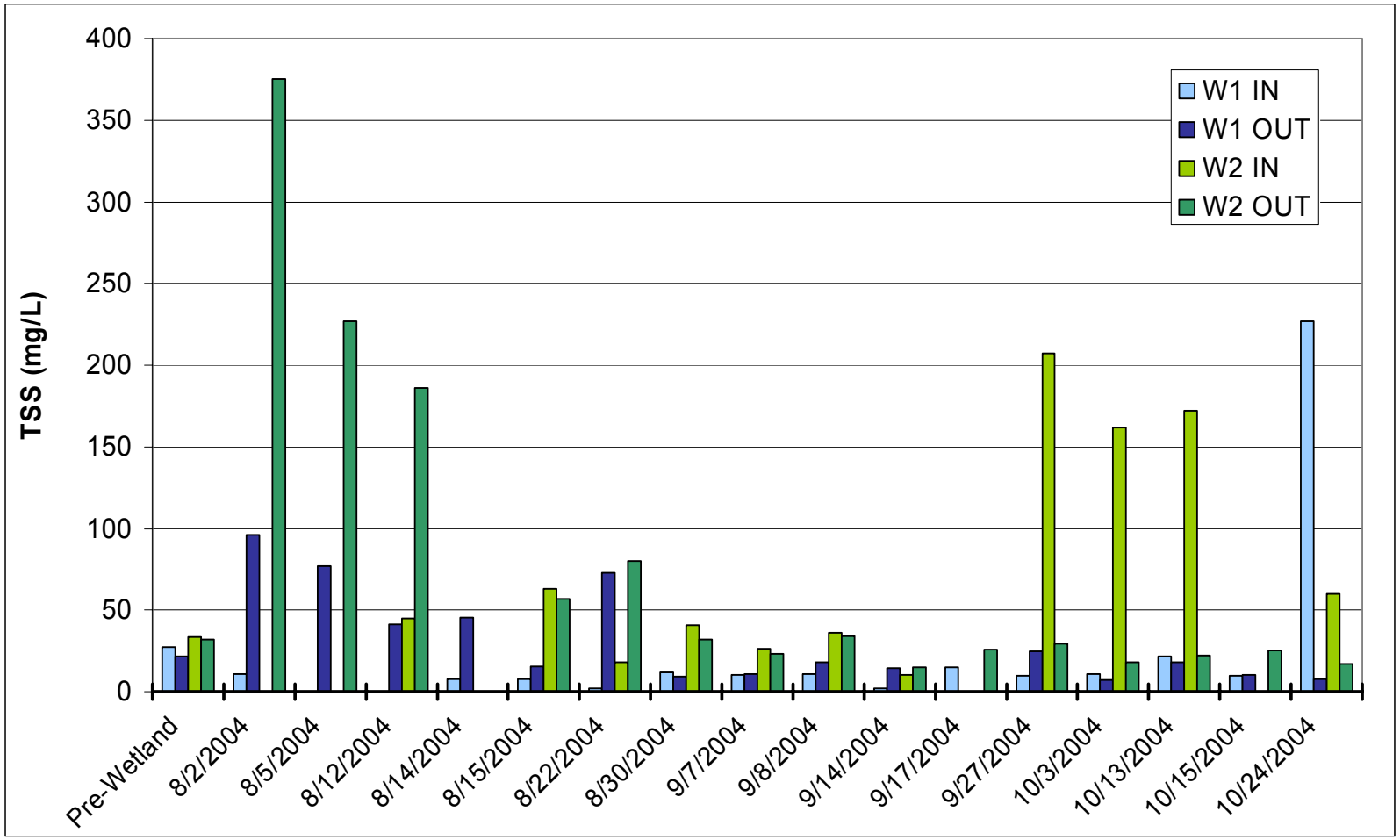

Figure 7.1: Total Suspended Sediment (TSS) Concentrations Measured at the Inlet and Outlet of Wetland 1 (W1) and Wetland 2 (W2) Between August and November 2004.

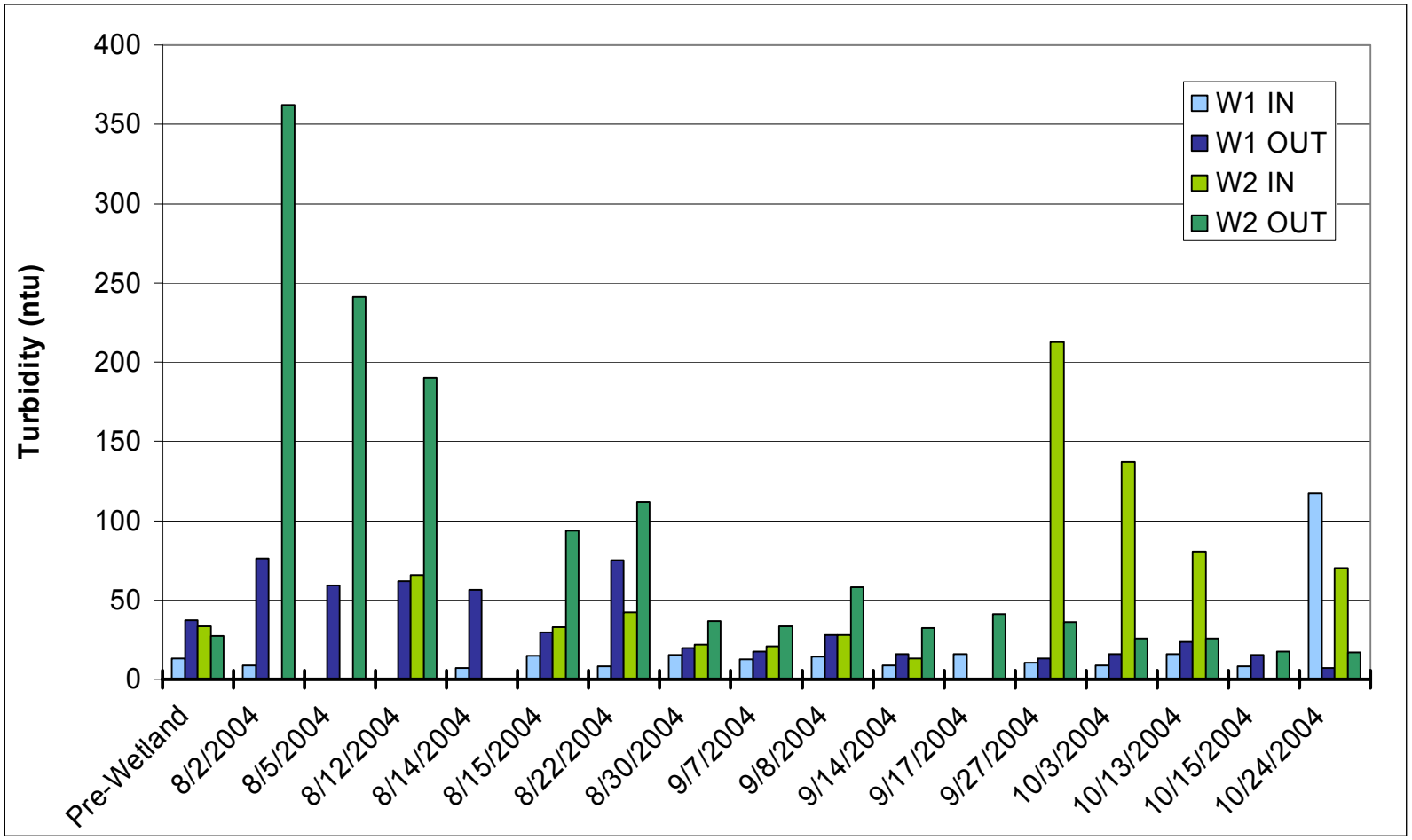

Figure 7.2: Turbidity Measurements at the Inlet and Outlet of Wetland 1 (W1) and Wetland 2 (W2) Between August and November 2004.

Total Suspended Sediment Loads

TSS loads at the inlets and outlets of the CSWs were determined by multiplying the average TSS concentrations by the estimated inflow and outflow amounts during composited storm events. Individual storms had to be composited when a storm event occurred before the CSWs fully drained from the preceding storm event. In this case outflow volumes could not be differentiated between individual storms. The individual and composite storm events used to estimate TSS loads are listed in Table 7.1, with total precipitation amounts and estimated largest intensities during the time period. Due to equipment errors, average TSS concentration data were missing for individual storm events at some of the sampling locations. The missing values had to be estimated using average concentrations measured during individual storm events within the composited group of storms. Stage measurements were not available for the storm events on 8/2/04 and 8/5/04 to compute TSS loads; therefore total TSS loads could not account for these two initial storms.

The TSS loads determined for each wetland during composite storms were divided by the CSW surface area to help eliminate the differences based on wetland size (Figure 7.3). The last two storm events during the study (10/19 and 10/24) did not produce enough stormwater runoff to generate outflow, but were included in the figure to indicate the low inlet TSS loads measured for both wetlands. The outlet TSS loads are divided into two quantities: 1) the estimated amount lost through the orifice and over the weir and 2) the estimated amount lost through leakage from the weir slats.

Inlet TSS loads per acre were all relatively low in W1, with the highest load per acre (101 lb/ac) contributed during the first measured storm (Figure 7.3). Inlet TSS loads per acre in W2 were elevated during the first storm and during the last two composite storms that generated outflow. The highest load (1458 lb/ac) was observed during the Oct 13-15th composite storm (Figure 7.3). For all composite storms, the inlet TSS loads per acre were greater in W2 compared to W1, similar to the results of the TSS concentration data.

Outlet loads were highest from W1 during the first two composite storms and declined for the remainder of the study (Figure 7.3). Outlet loads from W2 were greatest during the first two storms and during the last significant composite storms measured (10/13/04 – 10/15/04). For most of the composite storms measured, W2 outlet loads were greater than W1 outlet loads, similar to the results of the TSS concentration data.

During the initial two composite storms, the inlet TSS loads were less than the outlet TSS loads in both wetlands (Figure 7.3). The CSWs were generating sediment internally from the substrate. By the end of the study, the inlet TSS loads were greater than the outlet TSS loads, for a net reduction in sediment within both wetlands. The CSWs appear to be performing some of the sediment removal function of wetlands, most likely through sedimentation in the Deep Pools and Shallow Water regions. Table 7.2 summarizes the differences between inlet and outlet TSS loads for each CSW, with negative values indicating reductions in TSS loads at the outlets. The last two storm events during the study were not compared due to the lack of outflow. The reductions in

TSS load per wetland acre during the last composite storm events were considerably greater in W2 compared to W1.

The cumulative TSS loads over the three-month study are shown in Table 7.3. In W1, the cumulative TSS load at the outlet was about 1.8 times greater than the cumulative load at the inlet. Sediment appeared to be generated in this CSW between the inlet and the outlet. Based on the cumulative TSS load over time (Figure 7.4), the elevated TSS loads at the outlet compared to the inlet were consistent after the first composite storm. The cumulative TSS load increased at a constant rate for both the W1 inlet and outlet. In W2, the cumulative TSS load was reduced by 29% between the inlet and the outlet (Table 7.3). The TSS load accumulated over time is shown in Figure 7.4. The reduction in outlet load compared to inlet load occurred during composite storm 7 (10/13 – 10/15). Before this storm, the cumulative W2 inlet TSS load was less than the cumulative outlet TSS load, similar to W1.

The TSS load estimations provided additional information in describing the differences in sediment between the two CSWs. TSS loads per wetland acre were elevated at the CSW outlets compared to the inlets during the first portion of the study, similar to TSS concentrations. Both CSWs were generating sediment during these two composite storm events. Over time, both CSWs showed reductions in the amount of sediment leaving at the outlets. Although the outlet TSS loads were greater for W2, the inlet TSS loads were also much greater. W2 had a larger percent reduction in TSS load during the last month compared to W1 (Figure 7.3), similar to the results of the TSS concentrations. Due to the

large differences between inlet TSS loads in the two CSWs, it was not possible to assess which CSW was performing better at stormwater TSS load reduction.

In addition to variable stormwater suspended sediment characteristics, there were several other differences between the two CSWs that may have caused differences in sediment generation and reduction during the study. Topsoil conditions were observed to be different in the two CSWs. W1 had a more compact substrate due to heavy equipment exposure near the end of construction. The equipment crossed W1 after topsoil application, in order to reach W2 during its topsoil application.

The application of hydromulch and PAM in W1 and hydromulch alone in W2 may also have caused internal differences. The presence of PAM on the steep Upland berms (3:1 slope) and Shallow Land regions may have decreased erosion from rainsplash and sheet flow. Research conducted on steep bare slopes at development sites using hydromulching and PAM at recommended application rates has shown that this combination is more effective at reducing turbidity than hydromulching alone (Tobiason *et al*., 2001). Another study with mulch and PAM combinations showed reductions in TSS concentrations and loads in runoff water (Roa-Espinosa, 1999), but at much higher application rates. Other studies have shown that the reductions with PAM were not greater than hydroseeding alone (Glazer, 2001, Soupir *et al*., 2004), application of straw mulch alone (Soupir *et al*., 2004), or with mulch and seed alone (Hayes, 2003). These studies had similar application rates compared to the W1 site, based on recommended rates associated with the PAM products. All studies demonstrated, however, that reductions in sediment in the runoff water were greater with the addition of PAM compared to bare slopes.

Another internal factor causing differences between the wetlands may have been the establishment of temporary vegetation. The internal berms and Shallow Land regions of W1 had much denser vegetation established compared to W2. This vegetation could also have reduced erosion in these areas and enhanced sedimentation as stormwater inundated the areas. Better vegetation establishment in W1 may have been a result of differences in topsoil condition and nutrient content. This was not supported by observations made during planting that indicated the topsoil on the Upland berms and Shallow Land were compact and very dry between storms in W1 compared to W2. The nutrients applied during hydromulching were also the same between CSWs. Vegetation establishment may also have been enhanced by the presence of PAM (Wallace, 1987; Wallace and Wallace, 1986), through soil stabilization and increased infiltration during the initial weeks after construction.

Another difference between the two CSWs was the presence of the vegetated waterway, conveying stormwater from the inlet culvert to the wetland forebay. The amount of suspended sediment deposition occurring in this structure was not evaluated, and may have removed a considerable amount of suspended sediment from the stormwater.

The effects of the PAM application in this study could not be assessed due to the differences in inlet sediment amounts and due to the numerous site differences between the wetlands. The application of PAM at higher than the recommended rate could have reduced erosion within W1. There was not enough evidence with the TSS and turbidity data collected, however, to support or refute the ability of PAM to enhance erosion control in this environment.

An important loss of stormwater from the wetlands during this study was through leakage from the weir slats. The proportions of the outlet TSS loads that were estimated to have been released through leakage during the composite storm events were considerable in both CSWs (Figure 7.3). The total outlet load attributed to leakage was about 81% in W1 and 69% in W2 (Table 7.3). This large amount of leakage was observed during storm events, and was greater in W1 due to the two weir slat changes during the study. The new weir slats were not able to seal as rapidly as the weir slats in W2 that were present during the entire three months.

If leakage affected either outlet volume or outlet sediment loads, then all of the sediment load results at the outlet would have been overestimated during the study. The effects of leakage were not expected to be a significant factor in the accuracy of sediment load estimations, however. The outlet volume estimated using the flow balance equation would be the same regardless of the manner in which the stormwater was released; therefore, the amount of leakage was assumed to have no effect on the total outlet volume estimated during storm events. Leakage through the weir slats was also assumed not to affect the outlet sediment concentrations, based on the absence of sediment accumulation in front of the weir. The TSS concentrations were assumed the same in stormwater leaving through the orifice and over the weir as leaving through cracks within the weir slats. Comparisons were made between outlet loads attributed to design structures and outlet loads attributed to leakage for later comparison of the CSWs when significant leakage was eliminated.

Table 7.1: Composite Storm Events Used to Estimate Inlet and Outlet Loads for Both Wetlands.

Composite Storm Number	Storm Dates	Precipitation (in)	Largest Estimated Intensity for Individual Storm (in/hr)
1	8/12 - 8/16	4.55	0.25
2	8/21 - 8/22	1.85	0.45
3	8/29 - 8/31	2.32	0.29
4	9/6 - 9/8	2.01	0.25
5	9/14 - 9/18	2.53	0.27
6	9/27 - 10/3	0.92	0.23
7	10/13 - 10/15	1.89	0.38
8	10/19	0.08	0.05
9	10/24	0.11	0.01

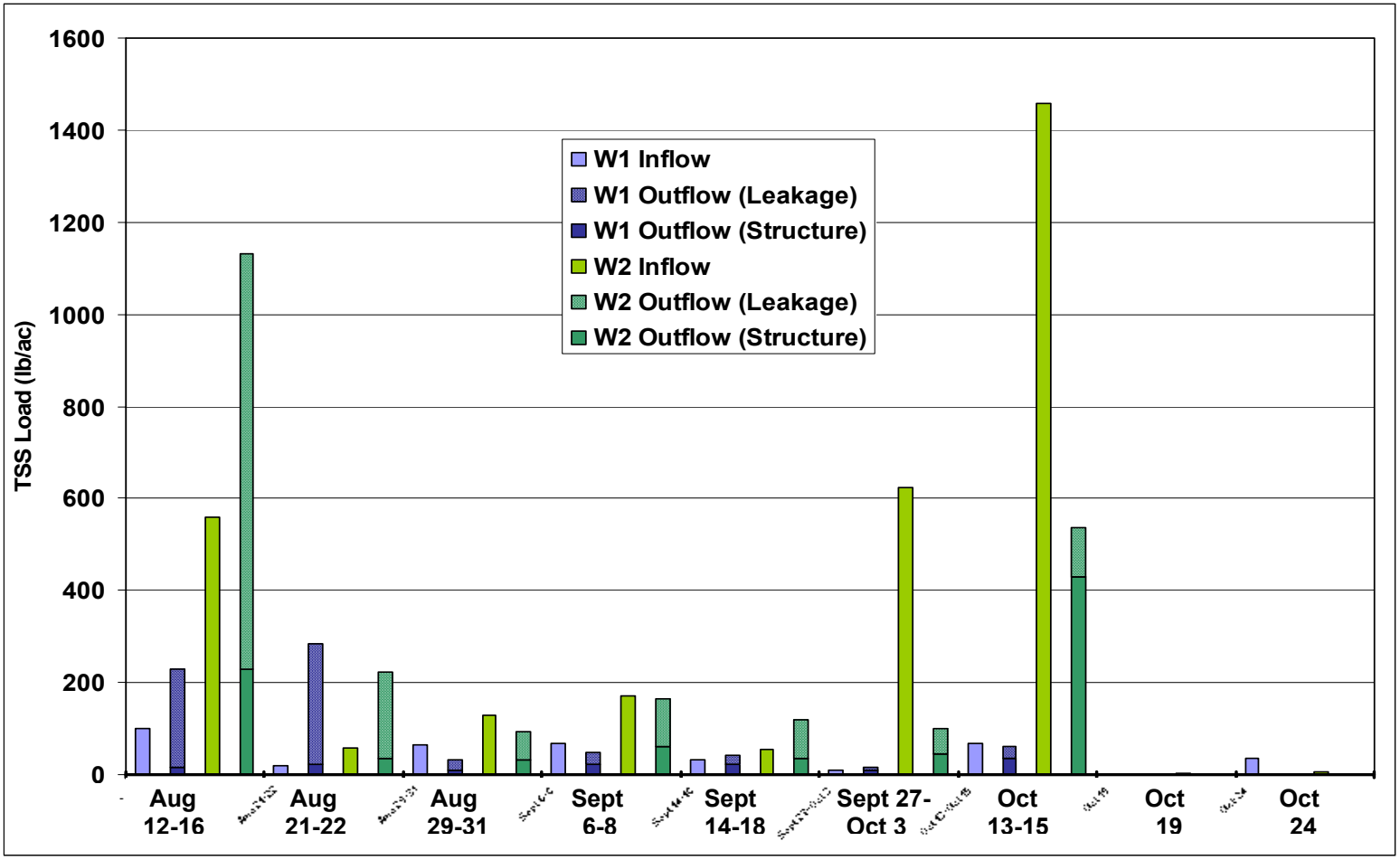

Figure 7.3: Total Suspended Sediment (TSS) Loads per Acre of Wetland Measured in Wetland 1 (W1) and Wetland 2 (W2) for Composite Storms Between August and November 2004.

Table 7.2: The Difference Between Measured Inlet and Outlet TSS Loads per Wetland Acre After Wetland Construction, with Negative Loads Indicating Reduction.

Storm Date	W1 TSS Load Difference (lb/ac)	W2 TSS Load Difference (lb/ac)
8/12 – 8/15	128	573
8/21 – 8/22	265	165
8/29 - 8/31	-32	-34
9/6 - 9/8	-18	-6
9/14 - 9/18	8	63
9/27 - 10/3	6	-523
10/13 - 10/15	-8	-1128
10/19	No Outflow	
10/24	No Outflow	

Table 7.3: Estimated Total Loads per Wetland Acre for Total Suspended Sediment (TSS) in Wetland 1 and Wetland 2 for the Three Month Study.

	Wetland 1 (lb/ac)	Wetland 2 (lb/ac)
Inlet	401	3065
Total Outlet	714	2165
Outlet - Structures	136	619
Outlet - Leakage	578	1503

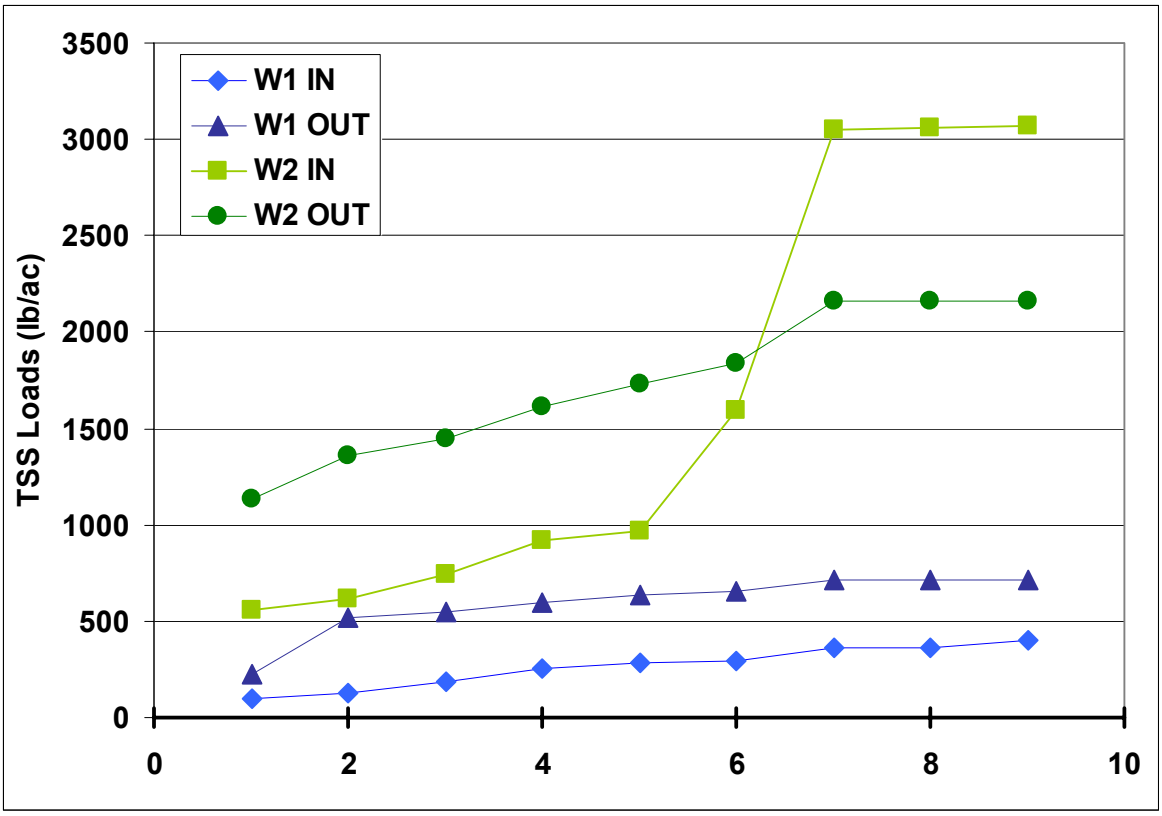

Figure 7.4: Cumulative Suspended Sediment (TSS) Loads per Acre of Wetland Estimated for All Composite Storms (1-9) in Wetland 1 (W1) and Wetland 2 (W2).

Wetland Topsoil Surveys

Cross-sections were surveyed in both wetlands before stormwater was added, after two months, and after three months (See Figure 6.6 and Figure 6.7). These surveys were performed to indicate if erosion was occurring during initial stabilization, and to indicate areas where sediment accumulation was occurring. The surveys are shown in Figure 7.5-7.12, and berm-to berm areas are presented in Table 7.4. The cross-sectional surveys that stretched across outlet weirs (X-S C and X-S F) were amended to compute the most accurate berm-to-berm areas. The elevation of the weir was extended vertically up to the

berm elevation for these surveys over time. In Wetland 2, loose material was added to the top of the eastern berm (X-S H) during mid-September repairs. The berm elevation was raised to the new height for the first survey, following the contour of the other two surveys in order to gain the most accurate cross-sectional area comparisons. An error occurred during the second survey of X-S A (W1) when a wrong permanent pin was used at one endpoint; therefore, this survey was not included.

The cross-sectional surveys and changes in cross-sectional areas did not indicate any considerable erosion or accumulation over time in the two CSWs. Minor changes in elevation did occur, however, and may indicate trends in substrate changes over time. An increase in cross-sectional area may indicate erosion of substrate material, while a decrease may indicate deposition of substrate material.

In W1, all of the cross-sections decreased in berm-to-berm area except X-S B. X-S C and X-S E had the largest reductions in area (8.9% and 10.5%, respectively). X-S C crosses the outlet Deep Pool region and shows an increase in elevation along the slope to the weir over time (Figure 7.7). X-S E showed the largest reduction, but there was no one specific feature that indicated the change (Figure 7.9). There was not much change in the X-S A, B, and D cross-section profiles (Figure 7.5, 7.6, 7.8), even though X-S A crosses the inlet forebay in W1.

In W2, two cross-sections showed an increase in berm-to-berm area (X-S F and X-S G), while the third cross-section (X-S H) showed a decline (Table 7.4). X-S F crossed the outlet Deep Pool region and had the largest reduction in berm-to-berm area (7.3%). This reduction was evident in the cross-section profile as an increase in elevation within

the pool bottom and within a depression at the base of the berm (Figure 7.10). The cross-sectional profile for X-S G showed fluctuation over time, with a decline in elevation in the Shallow Water area during the second month, but elevation rise in this region by the third month (Figure 7.11). The decrease in X-S H berm-to-berm area over time is shown in the cross-sectional profile as erosion at the base of the berm (Figure 7.12).

Wetland plots were placed in each wetland feature to determine patterns of substrate change in specific areas of the wetlands (See Figure 6.6 and Figure 6.7). The plot stakes were surveyed initially, and each month afterwards for the 3-month study. For each of the four major wetland features, a total of four stakes were measured in each wetland. The average elevation change was determined for each wetland feature and plotted over time in W1 (Figure 7.13) and in W2 (Figure 7.14). All plot stakes were assumed undisturbed during the study, except for W2P5 on the W2 interior berm where loose material was added during repairs in mid-September. This plot was not included in the average berm elevation changes for W2.

For both wetlands, the substrate elevations surrounding the stakes did not change very much, with a maximum deviation of only 1.5 inches in the Deep Pools. In W1, the average substrate elevation increased in the Deep Pools and decreased in the Shallow Water after the first month, but then remained constant for the remainder of the study (Figure 7.13). Elevations remained relatively constant for the shallow land and berm plots. In W2, the average substrate elevation increased the most in the Deep Pool regions by the second month and then declined. The Shallow Water also showed an increase in the

average substrate elevation during the first month and then declined afterwards. The Shallow Land and Upland berm plots remained relatively constant throughout the study.

The cross-sections and field plots indicated that substrate elevations were primarily increasing in the Deep Pool regions, where sedimentation was expected to cause the largest change in substrate elevation. A similar magnitude increase was observed in both CSWs (average 1.5-in), with cross-sections showing more net accumulation within the outlet Deep Pool regions. Inlet forebays did not show the same accumulation, probably as a result of scouring as the stormwater entered. The largest differences between the two CSWs were observed during initial surveys in the Shallow Water regions. These regions were expected to increase in elevation due to sedimentation. W2 had an initial increase, but W1 had initial erosion of these regions. This erosion in W1 was observed during the first few storms that flowed through the W1 channels without weir slats, with coarse substrate exposure and rills forming near the outlet Deep Pool.

The Shallow Land and Upland berm features were expected to show the effects of PAM application in W1. These features were expected to remain stable in elevation in W1, but show erosion in W2 where PAM was not applied. In both CSWs, the Shallow Land and Upland berm features were relatively stable over time. There was no evidence that PAM affected the stability of these areas in W1. Other factors that may have influenced stability in these regions include compaction during construction and the establishment of dense temporary vegetation over time. These factors may have reduced the erosion from rainsplash and sheetwash of stormwater on the exposed surfaces. In CSW

environments, the shear stress of the stormwater as it filled the wetland may also not be adequate to entrain topsoil particles in these regions.

The overall results of the surveys and plots suggest that the majority of the substrate is increasing in elevation. The rise in substrate levels may indicate that sedimentation is occurring, as suspended sediment is being deposited on the wetland bottom in both CSWs. The surveys and plot measurements had some error associated during data collection. Cross-section points were not surveyed at the same intervals and disturbance of the substrate during both the surveys and the plot measurements may have affected the elevation measurements. Other errors associated with the inundated plot measurements could have occurred due to the limited visibility of the stakes and difficulty associated with measuring depths accurately under water.

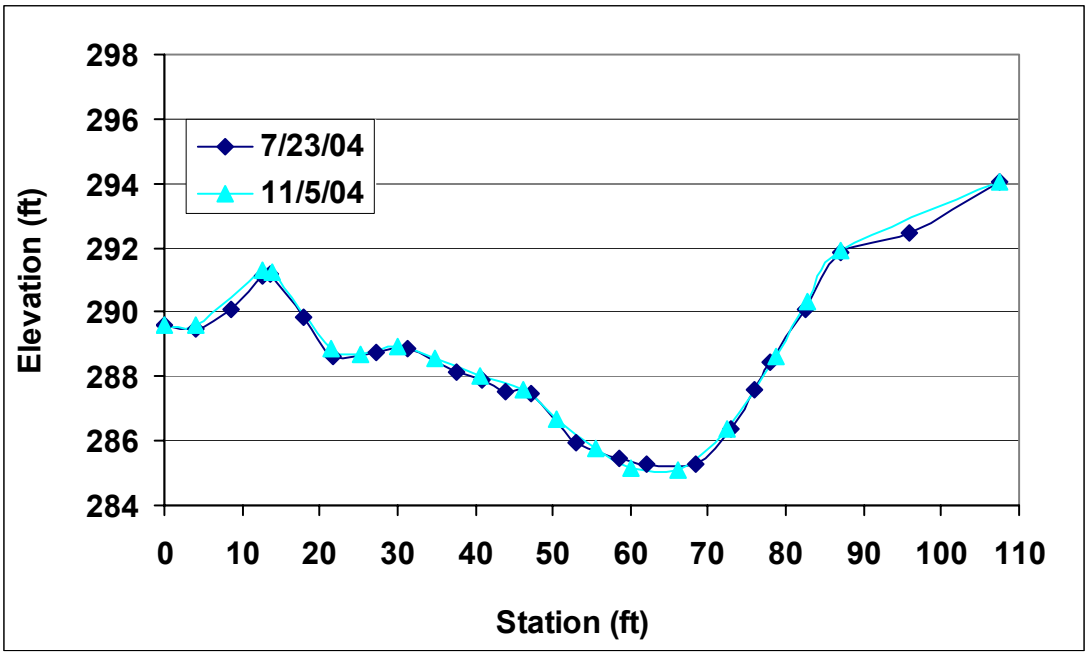

Figure 7.5: Cross-Sectional Survey for X-S A in Wetland 1.

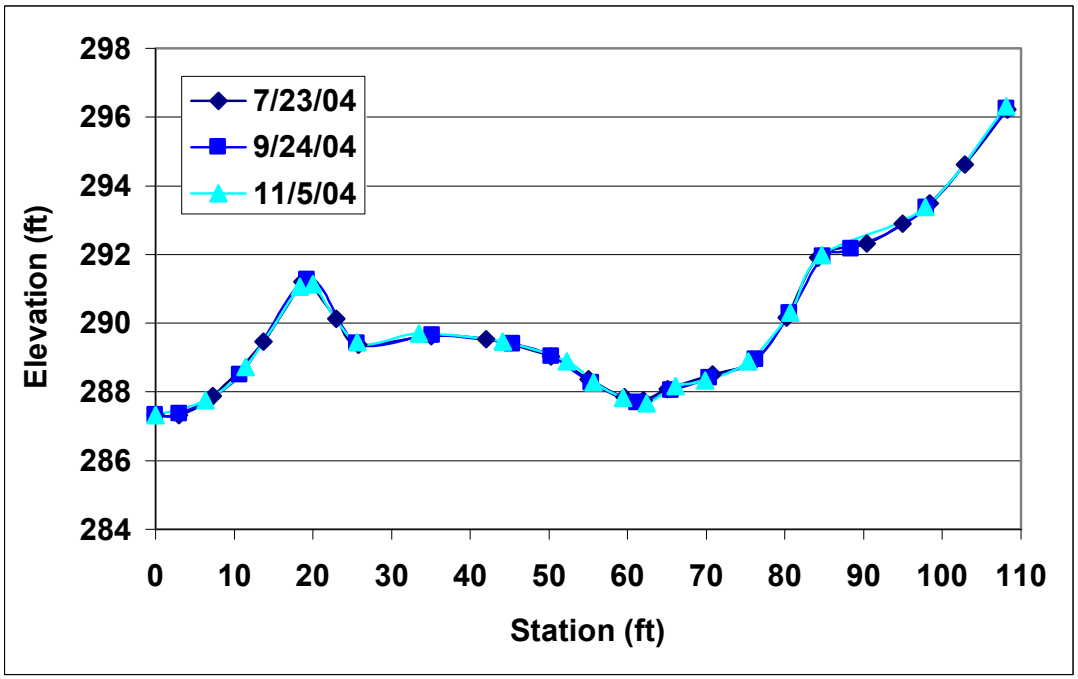

Figure 7.6: Cross-Sectional Survey for X-S B in Wetland 1.

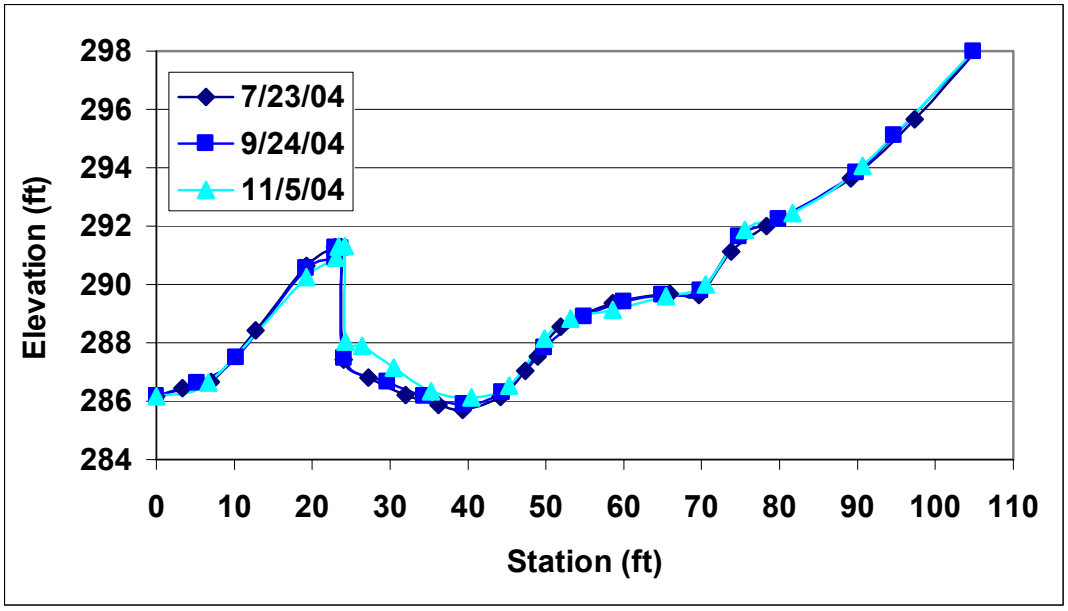

Figure 7.7: Cross-Sectional Survey for X-S C in Wetland 1.

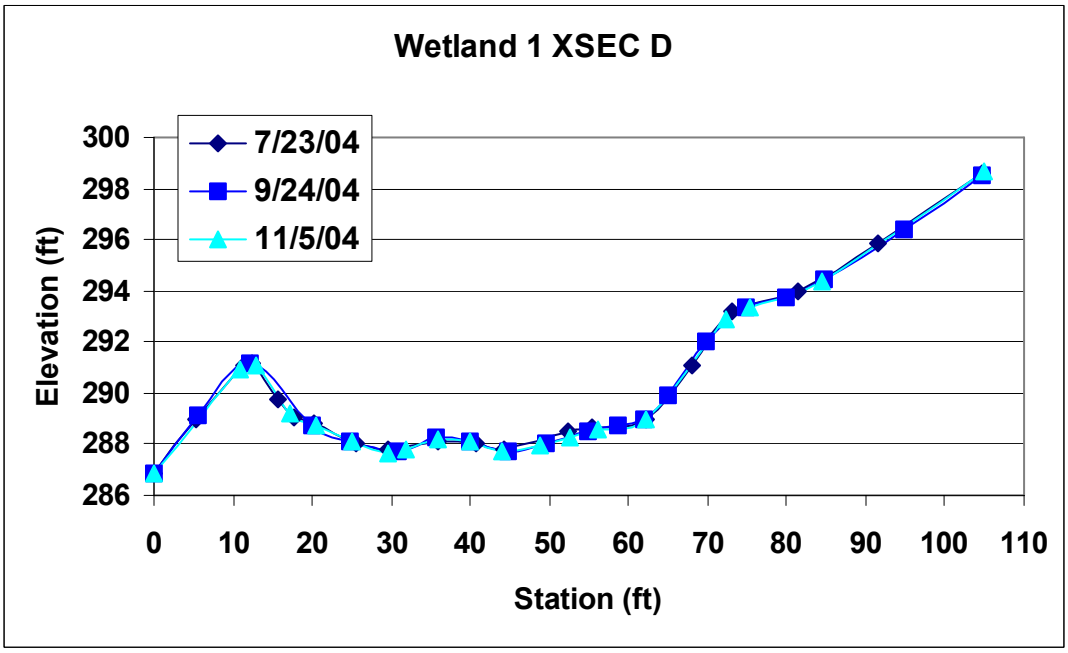

Figure 7.8: Cross-Sectional Survey for X-S D in Wetland 1.

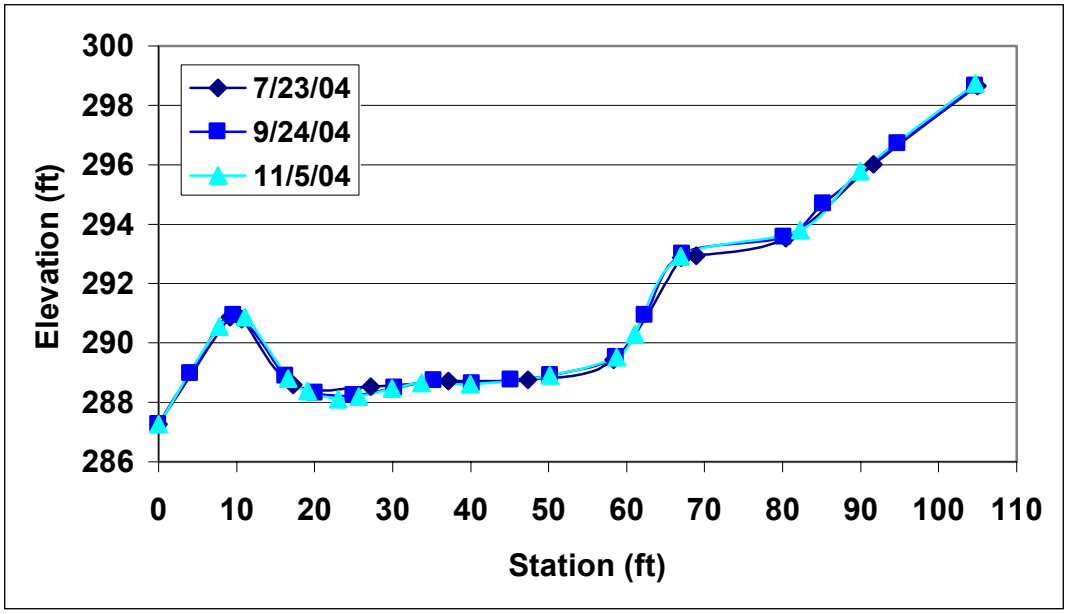

Figure 7.9: Cross-Sectional Survey for X-S E in Wetland 1.

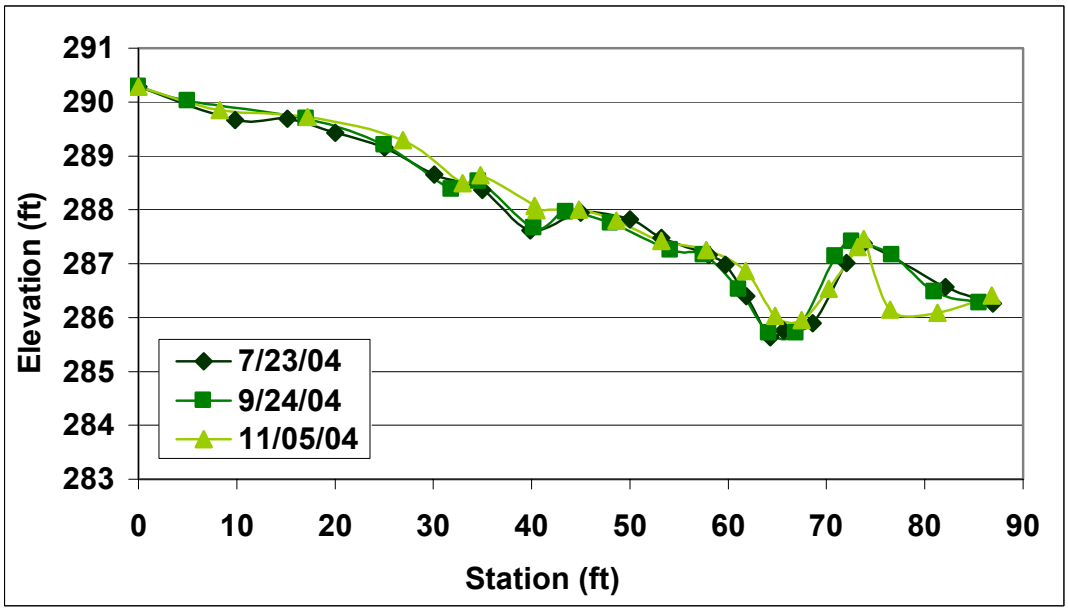

Figure 7.10: Cross-Sectional Survey for X-S F in Wetland 2.

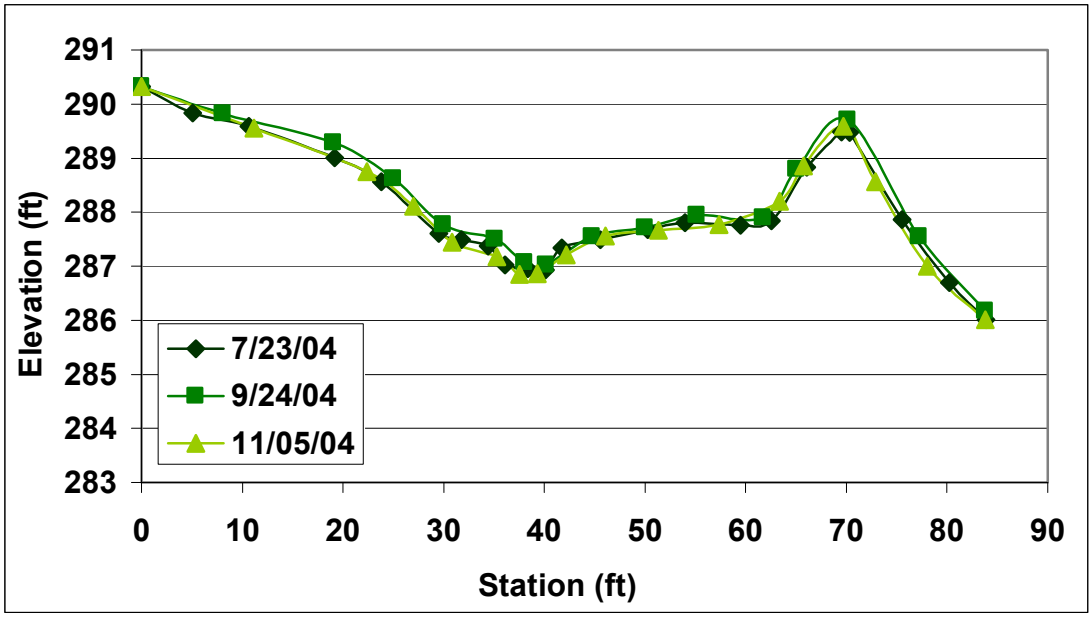

Figure 7.11: Cross-Sectional Survey for X-S G in Wetland 2.

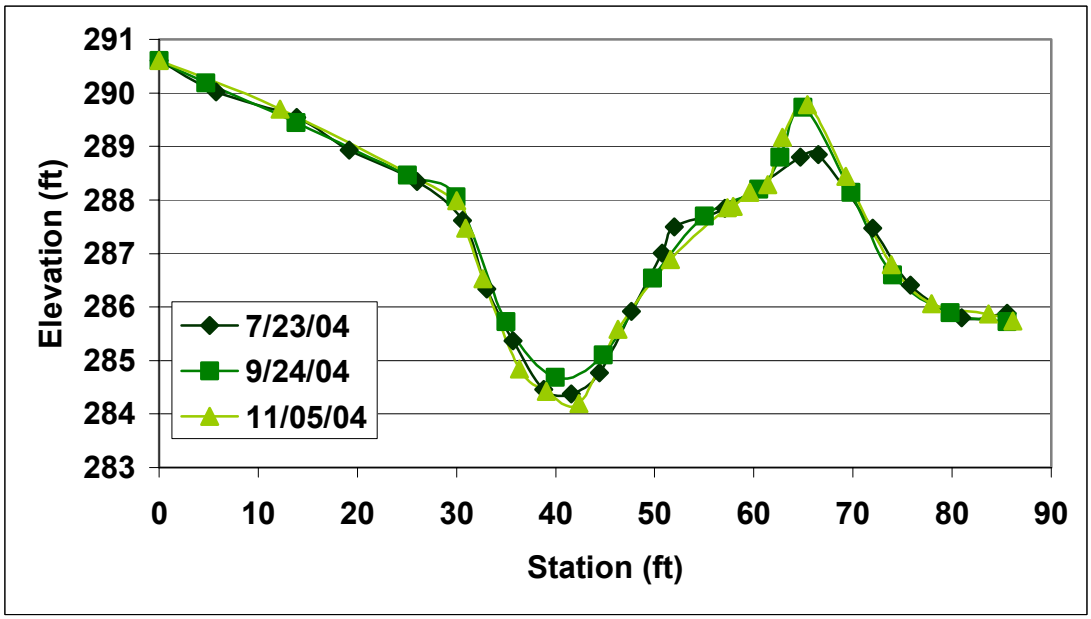

Figure 7.12: Cross-Sectional Survey for X-S H in Wetland 2.

Table 7.4: Cross-Sectional Areas (sq ft) for Wetland 1 and Wetland 2 over the Study Period with Total Percent Reductions.

Cross-Section	Initial (7/23/04)	Month 2 (9/24/04)	Month 3 (11/05/04)	Total % Reduction
Wetland 1				
X-S A	240.2	ERROR	238.2	0.8
X-S B	121.7	128.9	124.5	-2.4
X-S C	156.9	153.5	143.0	8.9
X-S D	137.1	137.9	133.9	2.3
X-S E	109.0	101.4	97.5	10.5
Wetland 2				
X-S F	128.1	127.8	118.8	7.3
X-S G	88.7	81.9	85.4	3.6
X-S H	124.1	125.2	129.5	-4.4

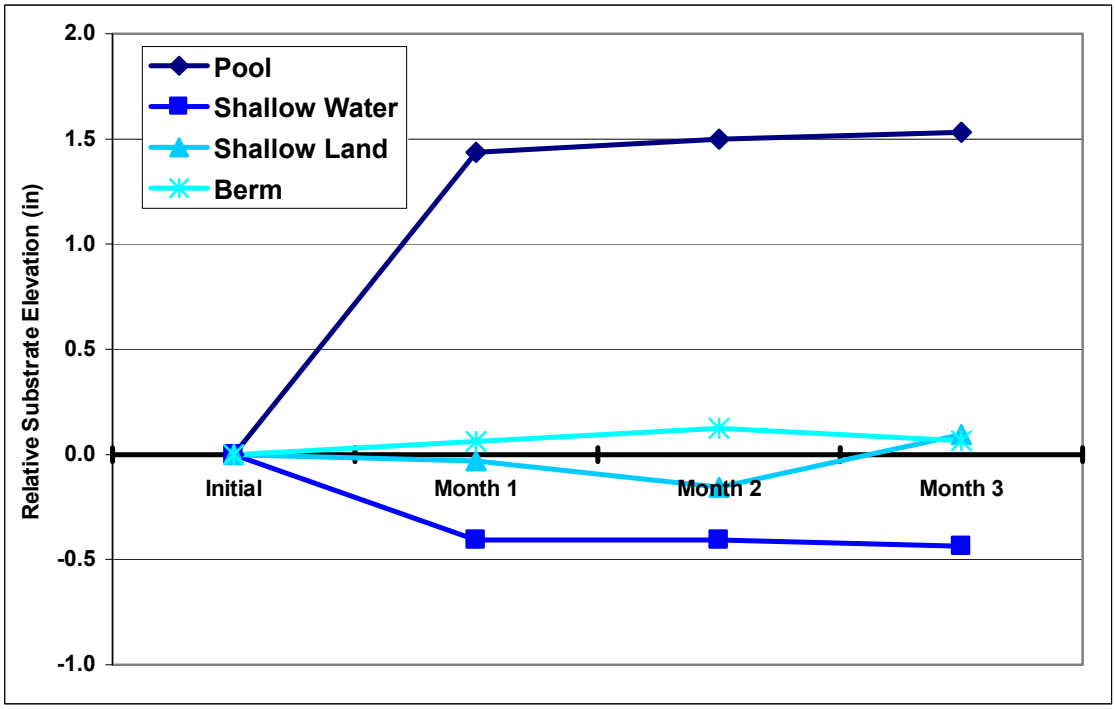

Figure 7.13: Average Substrate Elevation Changes Measured with Plot Stakes in Internal Features of Wetland 1.

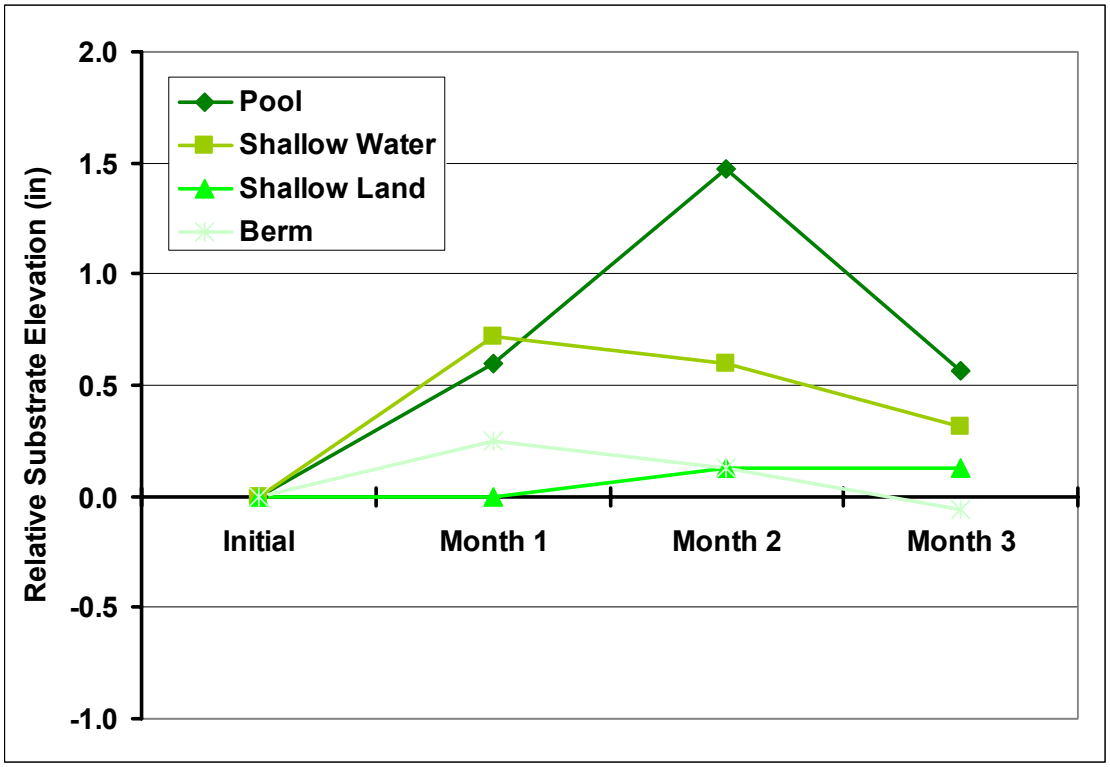

Figure 7.14: Average Substrate Elevation Changes Measured with Plot Stakes in Internal Features of Wetland 2.

Bulk Density and Particle Size Analysis

Bulk density samples were collected within the feature plots over time to provide additional information on substrate changes as sediment eroded or accumulated in the CSWs. The samples were collected using a cylindrical soil core initially after construction (8/6/04), after one month and after three months. Within the inundated plots, the samples were difficult to collect due to the large amount of water within the core. Efforts were made to collect the samples without significant loss of water, but the first Deep Pool samples were discarded. These samples had extensive sample collection errors, as the water was allowed to drain before placing the sample in the collection bag. The berm data

for W2P5 was also omitted due to the additions of topsoil during the mid-September repairs, even though the overall results were the same when this plot was considered. Average bulk density measurements were determined for each feature in the two CSWs by averaging the results of the substrate samples collected in each plot present in that feature.

In W1, the average bulk density increased over time in the Shallow Land and Upland berm regions, but remained relatively constant in the Shallow Water and Deep Pools (Figure 7.15). The average bulk density increased the most in the Shallow Land plots, particularly between the first and third month. In W2, the average bulk density decreased within the Deep Pool, Shallow Land, and Upland berm plots (Figure 7.16). In the Shallow Water regions, the average bulk density initially decreased along with the other features, but then increased by the end of the study for little net change.

The Deep Pools and Shallow Water regions were expected to decrease in bulk density due to loose sediment material deposited during sedimentation. Only the Deep Pools in W2 showed this trend during the study. During sample collection, accumulation of loose material was observed by the change in substrate consistency under foot. Addition of loose material may not have been adequate, however, to alter the bulk density of the substrate. Due to missing initial sample data in the Deep Pool regions, changes in bulk density could not adequately evaluated for this feature.

The results of the bulk density measurements provided some information about substrate changes in the features of the CSWs. The data collected did not, however, provide information on the effects of PAM application in W1. The ability of PAM to reduce erosion was not supported or refuted by the bulk density measurements, due to the

other possible variables present. Overall, the bulk density results were not consistent particularly during initial assessments. Samples collected immediately after construction should have had similar bulk densities due to the same topsoil applied to all surfaces. Differences suggest that the topsoil was variable in composition, or the method of application was different between features and between wetlands (i.e. more compaction). There were also many errors associated with sample collection methods. Bulk density measurements in Deep Pools were difficult due to the amount of water within the sample, and missing data affected interpretation. The substrate samples were also collected to a depth of 3 in, which may have been too deep to measure changes occurring at the surface over a short time period. The soil core may also have collected in-situ soil beneath the topsoil layer, due to variability in topsoil application and disturbance during sample collection. In-situ soils were heterogeneous and much different in soil texture compared to the topsoil. To improve the bulk density assessments in the CSWs, more refinement of topsoil application methods and more refinement of techniques used for sample collection are needed. A longer time period of assessment may also be warranted to show significant changes in bulk density.

Particle size analysis was performed using the settling method for all bulk density samples collected to show what changes occurred due to addition or removal of substrate material in the CSWs. Average percent sand, silt, and clay were determined from the samples in each feature for both CSWs. Particles larger than sand were removed before settling analysis.

In all of the samples analyzed, sand was the dominant particle size class with greater than 70% of the total weight. The clay content was generally found to be less than 5% by weight for all samples analyzed. For both CSWs, the change in particle content was small with differences in composition less than 10% during the study. The changes were also inconsistent over time in many of the features, with fluctuation in particle size classes between sampling dates.

In W1, the average percent sand fluctuated between sampling dates in the Deep Pool, Shallow Land, and Upland berm plots (Figure 7.17). These fluctuations in sand content by weight were 7% or less. The Shallow Water plots had a consistent decrease in average percent sand throughout the study, with an average change of about 9%. For all of the W1 samples, the changes in silt were greater than the changes in the clay content. In W2, the average percent sand increased in the Deep Pool and Shallow Water regions, but decreased in the Upland berm samples by 10% (Figure 7.18). The Shallow Land content fluctuated with a decrease in average percent sand after a month, but no net change by the end of the study. When the sand content was decreased, there was not a consistent pattern of replacement by silt or clay content between samples.

Based on the results of this study, changes in particle size content were small and there was considerable fluctuation in many of the regions. The particle size analysis did not provide adequate data during the time period to evaluate substrate changes. The role of PAM in erosion control could not be evaluated with the data collected. There were several factors that could have contributed to the inconsistency found in the particle size analysis. The topsoil added to the wetland surface was variable in both bulk density and particle size

composition, based on comparisons of initial samples immediately after construction. The suspended sediment entering the CSWs was also variable. Portions were derived from impervious surfaces in both watersheds, and from localized construction activity upslope from W2. These sources probably contributed different particle sizes, with coarser particles expected from the impervious surfaces but finer particles expected from upslope disturbances. Sedimentation was probably occurring in both wetlands, but the sediment deposited was variable in particle size class. Resuspension and redistribution of sediment in the CSWs would also have altered the particle size composition between sampling dates and could not be evaluated with the sampling method employed. As with the bulk density determinations, the sample depth (3 in) may have been too deep for assessing changes at the surface. To improve particle size analyses for CSW comparisons, more refinement of uniform topsoil applications and more refinement of technique for substrate surface evaluations are needed. A longer time period of assessment may also improve results and demonstrate whether the variability was due to substrate changes or due to variability between samples.

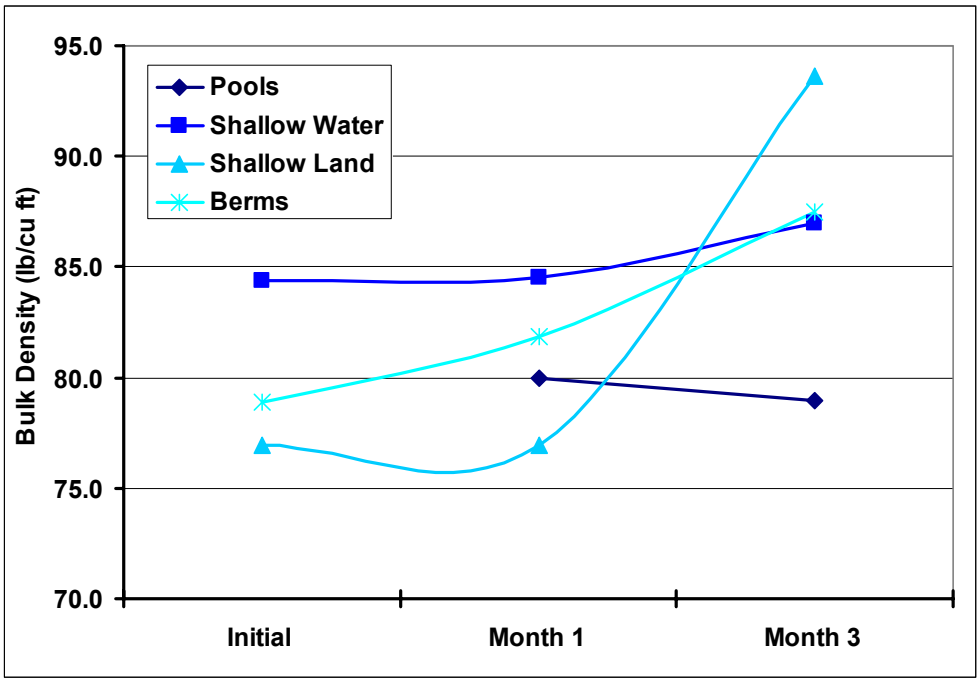

Figure 7.15: Changes in Substrate Average Bulk Density within Internal Feature Plots of Wetland 1.

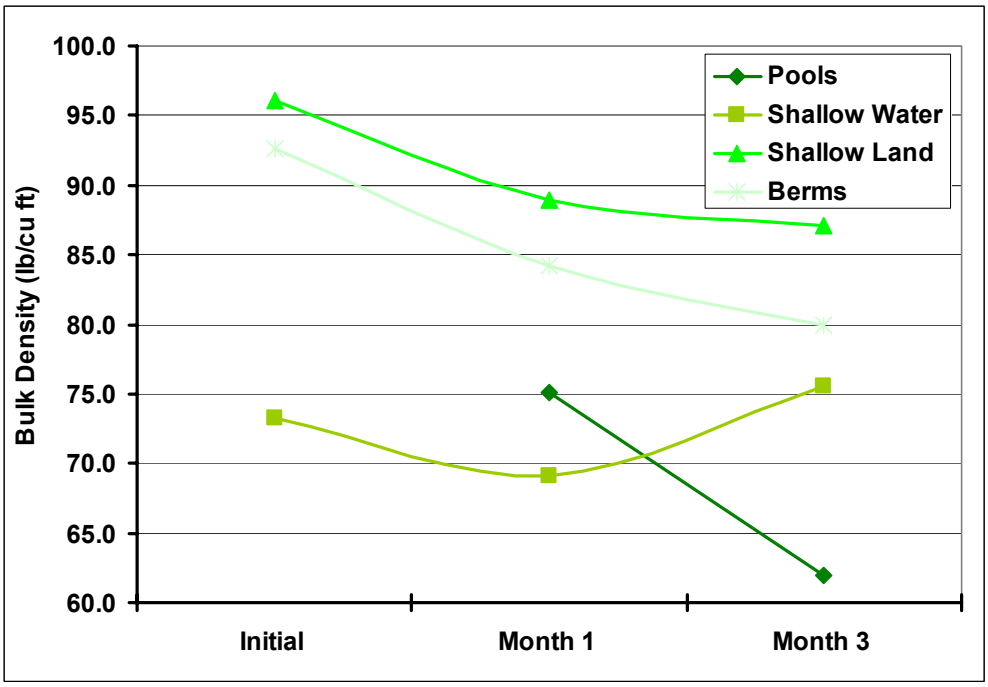

Figure 7.16: Changes in Substrate Average Bulk Density for Internal Feature Plots within W2.

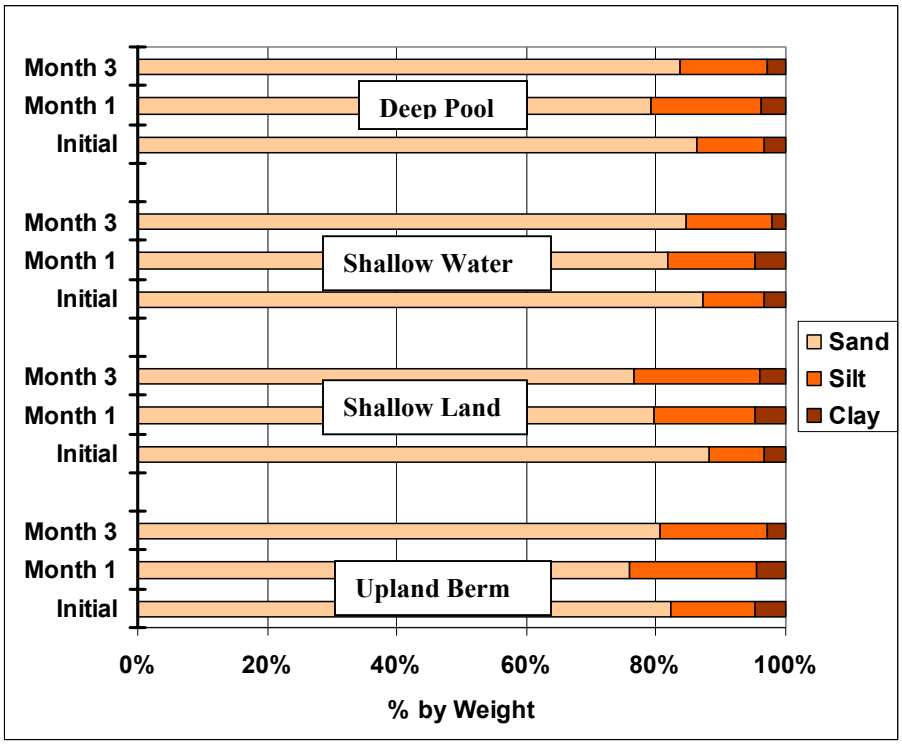

Figure 7.17: Particle Size Analysis of Plot Samples in Internal Features of Wetland 1.

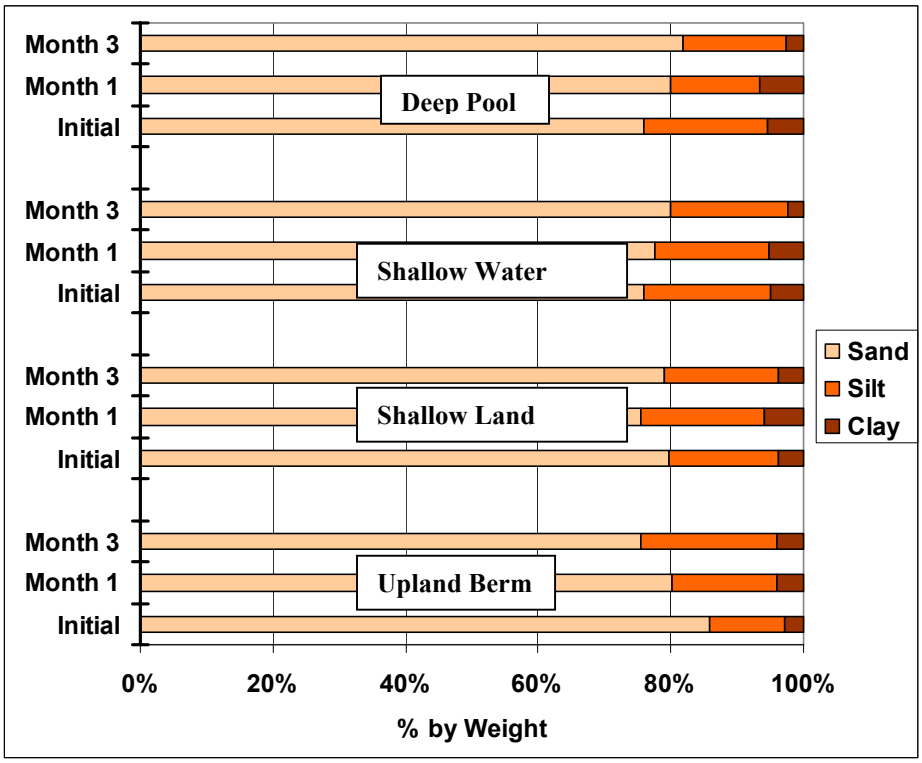

Figure 7.18: Particle Size Analysis of Plot Samples in Internal Features of Wetland 2.

CONCLUSIONS

The purpose of this study was to evaluate CSW performance during the initial stabilization period after construction. The time period required for CSWs to develop their stormwater treatment ability is important due to the ultimate goal of sediment reduction in receiving waters using BMPs. The results of this study demonstrated that CSWs are capable of generating more sediment than they remove from stormwater during the first several weeks after construction. By the end of the study, however, both CSWs were capable of reducing suspended sediment and turbidity from stormwater. The total sediment load reductions observed by the end of three months indicated that W2 had achieved a net positive stormwater treatment effect for sediment removal but W1 had not. The percent reductions in TSS concentration, turbidity measurements, and TSS load were higher for W2 during the last month. The data did not indicate that W2 was performing better than W1, however, due to the higher sediment amounts in the stormwater entering W2, and due to site differences between the two CSWs.

The substrate measurements collected through topsoil surveys and field plots during this study provided evidence that sedimentation was occurring inside the wetlands, particularly in the Deep Pool regions. The internal features of the CSWs were relatively stable during the study, with little erosion indicated by the substrate elevation measurements. Other substrate evaluations performed in this research, including bulk density measurements and particle size measurements, did not provide adequate evidence to indicate whether erosion or deposition was occurring in the internal features. The

results were variable and inconsistent, possibly due to problems associated with sampling methods or due to the limited time period of the study.

Given a longer time period of evaluation, both CSWs are expected to achieve stormwater treatment for suspended sediment based on the observed sedimentation and based on the observed reductions in TSS in the last month of the study. Stormwater treatment is also expected to be enhanced with the establishment of mature permanent vegetation, beyond the levels observed during this study. Long-term average sediment reductions (TSS) are expected to exceed 65% based on results of other studies (Schueler, 1992; Strecker, 1992; Brown and Schueler, 1997). A longer time period is needed, however, to assess the benefits of these BMPs for sediment removal.

The purpose of this study was also to evaluate the use of PAM as a secondary BMP to prevent erosion and reduce sediment runoff during the initial stabilization period of CSWs. The effects of PAM on wetland substrate stability could not be adequately evaluated in this study, due to the number of variables that were different between the CSWs. The stormwater sediment load entering W2 was much greater than the load entering W1, which limited comparisons. Differences in the particle size composition of the suspended sediment in the stormwater entering the CSWs could also have affected sedimentation rates, with larger particles settling more than finer particles. W2 conveyed stormwater through a vegetated waterway before reaching the forebay. The amount of suspended sediment deposited in this waterway was unknown, and may have influenced the higher sediment reductions observed. The bulk density and particle size analysis also indicated that there might be differences in the topsoil application within the two CSWs.

Compaction of the substrate was expected to be greater in W1 because of heavy equipment exposure after topsoil addition, as observed during sample collection in the Shallow Land and Upland berm regions.

To further investigate the performance of CSWs during initial stabilization, and the use of PAM in CSW applications, it is recommended that the CSWs being compared have more of the variables that were potentially causing differences in this study controlled. Using CSWs with less site variability, or conducting the research within cells of the same wetland would produce more comparable results. Substrate analysis techniques should be improved, for concentrated measurements of the surface only. All measurements should be conducted over a longer time period to indicate if the variability observed is attributed to sediment fluctuations or to the limited number of samples. More extensive preliminary sampling would also help better evaluate the improvements in stormwater sediment treatment at the particular installation site.

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CHAPTER 8: EVALUATION OF NUTRIENTS IN CONSTRUCTED STORMWATER WETLANDS DURING INITIAL STABILIZATION

INTRODUCTION

Nutrients in Constructed Stormwater Wetlands

One purpose of a CSW is to treat stormwater for excess nutrients contributed from the watershed. Pollutant loads, including nutrients, tend to be higher in urban stormwater runoff due to larger runoff volumes and higher concentrations contributed from development and landscaped settings (Bingham, 1994; Schueler, 2000). Nitrogen and phosphorus are the predominant nutrients of concern in developed watersheds. Excess nitrogen and phosphorus carried in stormwater cause eutrophication of receiving waters. Algal growth becomes stimulated, leading to an increase in turbidity and to the depletion of dissolved oxygen in the water column during microbial decomposition of the excess biomass (Wetzel, 1983; Bingham, 1994).

Nitrogen can be found in aquatic environments in inorganic forms, including nitrogen gas (N_2), nitrate (NO_3^-), and ammonia or ammonium (NH_3 , NH_4^+), and in organic forms associated with living and detrital biomass and synthetic compounds (Wetzel, 1983). In urban watersheds, sources of nitrogen in stormwater runoff include inorganic and organic nitrogen associated with fertilizers, pesticides, herbicides, industrial wastes, sediment, and human or animal wastes (Debusk and Debusk, 2001; Richardson and Vepraskas, 2001). Nitrogen is contributed to CSWs predominantly through stormwater runoff, in dissolved or particulate forms and can be associated with sediment. Other sources include nitrogen gas from the atmosphere, which can be added to the water column through nitrogen fixation by bacteria and algae. This process forms ammonia that is

rapidly incorporated into organic biomass and can occur in both aerobic and anaerobic conditions in shallow waters (Wetzel, 1983). Nitrogen can also be added from the atmosphere through both wet and dry deposition, in both dissolved and particulate forms. Within wetlands, nitrogen sources include plants and aquatic organisms during metabolic functions and during leaching upon senescence or death. Typical total nitrogen concentrations in natural wetlands are around 1 mg/L, due to the decomposition of excess organic matter (Wetzel, 1983).

Phosphorus can be found in aquatic environments in both an inorganic and organic form. The predominant inorganic form of phosphorus, orthophosphate, can be directly utilized by aquatic plants and organisms, or adsorbed to various cations, sediment, or organic compounds. The organic forms of phosphorus are associated with plant and animal biomass and various organic chemical compounds (Wetzel, 1983). Phosphorus is mainly contributed to CSWs through stormwater runoff, derived from similar watershed sources as nitrogen. Phosphorus has a high affinity for sediment, particularly fine sediments such as clays, and concentrations have been found to be related to the amount of suspended sediment in runoff water (Carter *et al*., 1974). There is no significant gaseous form of phosphorus; therefore, the atmosphere is not a significant source. Within wetlands, phosphorus can be cycled by release from plants and microorganisms, and by release from bottom sediments or cations in anaerobic conditions (Richardson and Vepraskas, 2001). Phosphorus associated with suspended sediment can dissolve in the water column, becoming readily available for uptake (Carter *et al*., 1974). Typical phosphorus concentrations in natural wetlands are around 0.01-0.05 mg/L, with the large

majority of phosphorus found in organic form associated with biota or in the fixed mineral form associated with the bottom sediments. The inorganic dissolved form, orthophosphate is rapidly adsorbed by aquatic biota, and is found in very low concentrations (Wetzel, 1983).

Nutrient removal from stormwater contributed to CSWs can occur through several physical, biological and chemical processes. The most important mechanism for nitrogen removal happens at the soil-water interface where both aerobic and anaerobic zones coexist (Debusk and Debusk, 2001). Nitrification by select bacteria converts ammonia to nitrate in the oxygenated region just above the sediments. Denitrification in the underlying anaerobic zone converts the nitrate to nitrogen gas, as microorganisms associated with the diverse sediment microtopography utilize the nitrate during decomposition of organic matter (Richardson and Vepraskas, 2001). Most of the nitrogen gas produced by denitrification is released to the atmosphere, with nitrogen fixation usually limited, resulting in permanent removal of nitrogen. Denitrification has been shown to be the most significant removal mechanism in wetlands for nitrogen (Johnston, 1993; Debusk and Debusk, 2001).

Although phosphorus cannot be permanently removed in a gaseous form, such as the denitrification process for nitrogen, the most significant mechanism for long-term phosphorus storage is through the associations with bottom sediments. Phosphorus bound to particulates, including sediment, metals, and organic matter, can be removed from the water column during sedimentation in deep pools where the flow velocity decreases or in shallow areas where plants physically obstruct the flow path (Johnston, 1993; Bingham,

1994). Although anaerobic conditions in wetland soils can cause the release of phosphorus from sediment and associated cations, the aerobic zone above the soil-water interface reverses the process and leads to reassociation, creating a balance in phosphorus storage (Wetzel, 1983). Phosphorus is bound to soils over a wide range of pH values. Acidic conditions promote the adsorption of orthophosphate with iron and aluminum in soils, while basic pH values allow phosphorus to precipitate as calcium or magnesium phosphates (Richardson and Vepraskas, 2001). Relatively large concentrations of phosphorus can be stored in the sediments over a long time period, with opportunity for microbial and plant root uptake.

For both nitrogen and phosphorus, another important removal mechanism is storage in plant and animal biomass (Debusk and Debusk, 2001). Organic nitrogen and phosphorus compounds can be mineralized to ammonia and orthophosphate, respectively, and readily assimilated by microorganisms and plants in the water column and sediments (Debusk and Debusk, 2001; Richardson and Vepraskas, 2001). Although the storage of both nutrients in microorganisms is usually considered a short-term process due to their short life cycles, efficient recycling of nutrients within mature colonies can effectively remove the nutrients from the water column over a long-term period (Wetzel, 1983). Plant uptake through roots and storage can also be regarded as both a short-term and long-term process (Johnston, 1993). Plant senescence during fall and winter, and plant decomposition can return nutrients to the water column, but the nutrients can again be recycled with no net effect on either nitrogen or phosphorus concentrations (Debusk and Debusk, 2001). Plants can offer a permanent removal of nutrients from wetlands if they

are harvested, but they must be replaced to continue their nutrient attenuation role. Complex organic forms of nitrogen and phosphorus derived from synthetic compounds are common pollutants in urban stormwater runoff. These compounds can be removed from the water column by sedimentation, and may be decomposed to varying degrees by microbial metabolism when associated with the bottom sediments.

Nutrient removal rates have been evaluated in many CSW studies over both long and short time periods. Schueler (1992) reviews long-term pollutant removal rates for stormwater wetlands in the Mid-Atlantic Region, and estimates the average removal rate for total phosphorus (TP) is 45%, and for total nitrogen (TN) is 25%. Other estimates from natural and constructed wetlands suggest the range is 40-60% for total phosphorus, and 21-76% for total nitrogen (Bingham, 1994; Brown and Schueler, 1997; DeBusk and DeBusk, 2001). These rates are long-term, however, and may not apply to CSWs during the initial establishment period where all removal mechanisms have not developed.

The amount of stormwater nutrient treatment and the amount nutrient generation during the initial period after construction of the wetlands is of concern and has not been sufficiently evaluated. Just after construction, some of the most important removal mechanisms for both nitrogen and phosphorus are not yet available including mature vegetation. The nutrient removal efficiency of the CSW may be diminished during this time, delivering untreated stormwater at the outlet. Just after construction, topsoil and fertilizers are usually added to the bare substrate in order to establish vegetation. Without the protection of mature vegetation, nutrients associated with these additions are vulnerable to the erosive forces of the stormwater. In this situation, the CSW may be delivering more

nutrients at the outlet compared to the incoming stormwater. The concern is that nutrient generation during the initial stabilization period may be significant, and overshadow the nutrient attenuation benefits over the life span of the wetland.

Based on these concerns for substrate erosion and nutrient generation within CSWs just after construction, a secondary BMP utilized for erosion control can be used for erosion control. In this research, the secondary BMP chosen was an application of PAM.

Polyacrylamide (PAM) as a Secondary BMP

Polyacrylamide (PAM) characteristics and associated research are detailed in Chapter 7. Most research has focused on the ability of PAM to reduce sediment loss from agricultural fields and construction sites. Sediment reduction can also indicate nutrient reduction in runoff waters due to the close association between phosphorus and some forms of organic nitrogen with suspended sediments. Several studies have measured nutrient reductions coupled with PAM applications, with most attention given to phosphorus due to its affinity for suspended sediment.

PAM applications during agricultural irrigation have been shown to reduce nutrients in runoff, through reduction of soil erosion and through the retention of applied materials such as fertilizers and pesticides (Bahr and Steiber, 1996; Sojka and Lentz, 1996). Lentz *et al*. (1998) found reductions in TP, PO_4, and NO_3 in furrow irrigation runoff from bean fields treated with two different PAM treatment methods. Total phosphorus mass was reduced by an average of 86% and 91%, and PO_4 mass was reduced by an average of 77% and 86%. There was no significant difference in nitrate-nitrogen losses between PAM treatments and irrigation without PAM, and this was assumed due to

the reduced runoff volume from the PAM-treated plots. The reduction in sediment concentrations was reflected by reductions in phosphorus, but these two pollutants were only moderately positively correlated. The ability of PAM to decrease nutrient losses seemed to be related to its capacity to decrease sediment concentrations in runoff, and to reduce runoff by improving infiltration rates. For sprinkler irrigation applications, PAM has also been successful at nutrient reductions. Bjorneberg *et al*. (2000) found that PAM-treated irrigation water and straw applied to agriculture fields both controlled sediment, phosphorus, and runoff volumes significantly. Reduction results were improved when both were applied together.

Soupir *et al*. (2004) evaluated the loss of both nutrients and sediment from construction field plots subjected to simulated rainfall. PAM was spread on the plots dry and dissolved in water at different concentrations. Comparisons were made between hydroseeding with and without PAM and straw mulch applications without PAM. The overall results of the study found that both PAM applications and straw mulch applications reduced nutrients and sediment in runoff water. Straw mulch alone, however, was the most effective for reducing nutrients, including nitrate, TKN, ammonium, total phosphorus, and orthophosphate, as well as sediment in runoff water. Liquid and dry PAM applications were comparatively effective at nutrient reductions, but liquid applications at half the recommended application rate reduced nitrogen associated with TSS, and total nitrogen more than other PAM methods. All treatment methods showed diminishing results over time, and none of the treatments were able to significantly reduce runoff volume.

The Application of PAM in CSWs

There have been no studies published to date on the use of PAM in association with CSWs, and as indicated above, there has been limited research on the reduction of nutrients when PAM was used for erosion control. The predicted effects of PAM on nutrient dynamics within CSWs are largely based on the predicted effects of PAM on sediment reduction (See Chapter 7) and the relationship between sediment and nutrients in CSWs. For nutrients that don't readily associate with sediment (NO_3 and NH_4), PAM may contribute to their retention by improving infiltration on Shallow Land and Upland berm regions, or have no effect on their quantities.

The application of PAM to the substrate may reduce the loss of nutrients derived from the topsoil and the fertilizers applied during hydromulching for vegetation establishment. These nutrients include both organic and inorganic forms of nitrogen and phosphorus. The capacity to reduce nutrient losses may be related to the ability of PAM to bind soil and other charged nutrient particles, forming aggregates. These aggregates could withstand the sheer stress of rainsplash and sheetflow better, and loss to erosion could be reduced. PAM has also been shown to enhance vegetation establishment in agriculture (Wallace and Wallace, 1986; Wallace, 1987; Lentz and Sojka, 1996), which may influence nutrient retention in CSWs upon establishment of vegetation.

The effects of PAM application in inundated regions including Deep Pool and Shallow Water regions are not known. During initial inundation, some of the PAM could be dissolved in the water column and available for flocculation of fine sediments and their associated nutrients, but the effects have not been studied in inundated conditions.

The ability of PAM to reduce nutrients in CSWs may diminish over time without reapplication. The amount of time for PAM effectiveness is not clear, but a decline in effects over time has been noted in erosion control applications (Soupir *et al*., 2004). Agriculture applications through irrigation typically reapply PAM during each treatment (Lentz and Sojka, 1996). The persistence of PAM has been shown to be dependent on soil type and suitability of PAM for the soil type and application rates (Sojka and Lentz, 1996). The amount of exchangeable cations associated with the soil and water may also affect the performance of PAM molecules (Seybold, 1994). Once the PAM molecules have become saturated with bound sediment particles, they may no longer be effective for erosion control and soil aggregation without reapplication. In CSWs, the persistence may also be dependent on the frequency of inundation and the erosive force applied by the stormwater.

CSW Nutrients and PAM Study

The purpose of this study was to evaluate nutrient retention and generation within CSWs during the initial stabilization period after construction. The performances of two CSWs (W1 and W2) on the floodplain of North Creek were evaluated based on their ability to remove and retain nutrients without the majority of stormwater treatment mechanisms established. Details of the CSWs are provided in Chapter 6. Nutrients evaluated in this study include total Kjeldahl nitrogen (TKN) as an indication of organic nitrogen, nitrate, ammonium, and total phosphorus. As part of this evaluation, a secondary BMP was utilized in the form of a PAM application (15 lb/ac) for erosion control. The purpose of this study was also to evaluate the effectiveness of PAM for nutrient reduction during initial stabilization of CSW environments.

The following study objectives were derived for nutrient assessments in the CSWs:

1) To determine if there is a difference between the general water quality parameters measured within W1 and W2.

2) To determine if there is a reduction in nutrient concentrations and nutrient loads between the inlet and the outlet of each CSW.

3) To determine if there is a difference between the nutrient concentrations and nutrient loads at the outlet of W1 and the outlet of W2.

4) To determine if the PAM application in W1 affected nutrient retention.

METHODS

General Water Quality Parameters

Temperature, specific conductivity, dissolved oxygen concentrations and pH measurements were collected using a multiple probe meter (YSI 556 Multiprobe System). The meter was calibrated using standard methods for all relevant variables. General water quality measurements were collected during the afternoon on two sampling dates (9/21/04, and 10/26/04). Data were collected at the substrate elevation and near the surface to determine average values within two pools and two shallow water areas per wetland. Due to higher than expected pH values measured on 10/26/04, additional pH measurements were taken on 11/17/04 using a Styrofoam float to suspend the meter out into the body of the wetland without disturbance of the substrate. For better pH data interpretation, the pH values of various materials added to the wetland substrate were collected from the manufacturers (Appendix 6, Table A6.6). Soil samples were collected from the topsoil before application, and after three months of wetland residence within the pools and shallow water areas. These samples were analyzed by the North Carolina Department of Agriculture (NCDA Agronomic Division, Raleigh, NC). Stormwater pH measurements were also taken from samples collected during storm events near the end of the study to help interpret the results.

Hydromulching and PAM Application

Following construction, an erosion control system composed of fertilizers, organic substrates, and permanent and temporary seed was applied to the wetlands and surrounding disturbed areas by hydromulching. Details of this hydromulch mixture are provided in Chapter 6, and nutrient contents are provided in Appendix 7, Table A7.1 and A7.2). PAM (APS 700 Series Silt Stop, Applied Polymer Systems, Inc.) was added to the hydromulch mixture (15 lb/ac), for application within W1 and the surrounding disturbed areas, but was not added to the mixture for application on the W2 site (Details provided in Chapter 7).

Nutrient Concentrations

Stormwater samples collected by an automated sampler before and after construction for TSS concentration and turbidity measurement were used to determine nutrient concentrations at the inlet and outlet of both wetlands (See Chapter 7). Nutrients were measured in stormwater samples from individual storm events on 8/13/04, 9/8/04, 9/28/04, 10/3/04, and 10/13/04 with precipitation greater than 0.8 in for all events except for the 10/3/04 storm (0.04 in). Nutrients were only measured for a storm if all automated samplers functioned properly during sample collection. Bottles in the automated samplers were washed initially and mid-study using LiquiNox detergent to avoid contamination. Portions of the stormwater samples were collected into containers washed with LiquiNox detergent during the first two storms, then collected in acid-washed containers provided by the analysis lab for the remainder of the storms due to contamination concerns.

Nutrient concentrations in stormwater samples were determined by the NCSU BAE Environmental Analysis Laboratory. All samples were collected within 24 hours of the

storm event and taken to the lab for analysis. Two composite storm samples at each sampling location were created with two replicates, and analyzed for total Kjeldahl nitrogen (TKN), ammonium (NH_4-N), nitrate (NO_3-N), and total phosphorus (TP). Blank samples were submitted in replicate for three of the storm events. Blanks contained either distilled water or deionized water, and were placed in containers that were acid washed by the lab.

Pre-construction samples were collected during one storm event from the existing stormwater culvert inlets and outlets to North Creek on the floodplains. These grab samples were collected in Liquinox washed containers, and analyzed by the NCSU BAE Environmental Analysis Laboratory for the same four nutrients as stormwater samples.

Nutrient Loads

Nutrient loads per wetland acre were determined for each of the four nutrients measured during the storms events sampled, following the methods used for TSS in Chapter 7. Nutrient loads were determined at the inlets and outlets of each wetland for each storm event sampled, starting from the rise of the wetland stage above permanent pool and ending with the return to permanent pool elevation. Storm events were composited if the permanent pool elevation was not restored before the next storm event occurred. Nutrient concentrations for storm events in the composite with no data collected were estimated using concentrations measured during another storm event within the composite. Since only a small number of storms were measured, nutrient loads were only determined for composite storm events with nutrient concentration data available. Total

nutrient loads for the study period were not determined because of limited data and the high variability of the nutrient concentrations.

Statistical Analysis

All descriptive statistics and correlations were performed using JMP 5, a statistical analysis package (SAS Institute, 2002).

RESULTS AND DISCUSSION

General Water Quality Parameters

The results of the general water quality parameters measured in the Deep Pool and Shallow Water regions between storm events are shown in Table 8.1. The parameters were measured near the surface and along the substrate at two locations in each feature to determine an overall average. Temperature measurements were higher during the second sampling due to higher air temperatures. Conductivity measurements were consistent in both wetlands between sampling events and were in the expected range for surface waters.

Dissolved oxygen concentrations increased over time in both features of the CSWs. By the second sampling date, all measurements indicated saturation (≥ 10 mg/L) except the lower average dissolved oxygen concentrations in the Deep Pool regions of W2. Dissolved oxygen concentrations were measured to indicate if anaerobic conditions were developing near the lower substrate due to inundation and microbial decomposition. Based on the results, anaerobic conditions have not yet been established, unless below the substrate surface.

The pH was relatively acidic during the initial sampling, but increased significantly during the second sampling for both features in the CSWs. Due to the exceptionally basic values, a third sampling event was performed. These measurements were taken using a Styrofoam floating device, to avoid substrate suspension. These pH measurements were higher than the first sampling, but only slightly basic compared to the second event values. The pH values associated with the substrate, materials added during construction, and the stormwater entering and leaving the wetlands are provided in Appendix 6, Figure A6.6.

The pH values of the soils were all acidic, even after three months residence time in the wetland. Stormwater entering and leaving the wetland was also acidic. The bentonite used to seal the wetland had the highest pH range, with the hydromulching products, PAM and H_2OLD, being less basic. The elevated pH values were most likely due to the bentonite entering the water column during substrate disturbance.

The results of the pH measurements did not indicate influences on nutrients directly, but high pH values measured during substrate disturbance may affect the wetland vegetation. A decline in wetland vegetation establishment within the wetland can affect nutrient concentrations by limiting plant uptake in the future (DeBusk and DeBusk, 2001). Sustained high pH values near the substrate may lead to a temporary release of nutrients such as phosphorus bound to the sediment (Richardson and Vepraskas, 2001).

Table 8.1: Average CSW General Water Quality Parameters Measured in the Pool and Shallow Water Regions Between Storms at Two Water Elevations.

	Wetland 1			Wetland 2		
	Deep Pool			Deep Pool		
	9/21/04	10/26/04	11/17/04	9/21/04	10/26/04	11/17/04
Temperature (deg C)	19.5	19.4		17.3	19.1	
Conductivity (mS/cm)	0.1	0.1		0.1	0.1	
Dissolved Oxygen (mg/L)	8.3	10.7		4.5	7.2	
pH	7.4	9.0	7.6	6.2	8.7	7.2
	Shallow Water			Shallow Water		
	9/21/04	10/26/04	11/17/04	9/21/04	10/26/04	11/17/04
Temperature (deg C)	19.2	20.9		17.2	20.6	
Conductivity (mS/cm)	0.1	0.1		0.1	0.1	
Dissolved Oxygen (mg/L)	7.6	10.5		4.8	11.9	
pH	7.3	9.1	7.7	6.6	11.8	7.5

Nutrient Concentrations and Loads

Nutrient concentrations were determined from two composite samples per storm event at each sampling location. Average nutrient concentrations per storm event were determined by averaging the replicated composite samples. Nutrient loads were estimated for each nutrient measured, during the composite storm events when nutrient samples were collected. Composite storms were a collection of storm events occurring in close time proximity, when the outflow from one storm could not be distinguished from the outflow of the next storm. The outlet nutrient loads were divided into estimated amounts lost through the designed outlet structures (orifice and over top the weir) and amounts lost to leakage from the weir slats. Limited pre-construction sampling was performed at the existing inlets and outlets on the floodplains. These samples were used for general

comparison only, with the understanding that nutrient concentrations were probably much more variable in the pre-construction stormwater conveyance channels.

Total Kjeldahl Nitrogen

Average TKN concentrations were similar between the inlets and outlets of both CSWs for all storms measured, except the first storm after construction and the last storm (Figure 8.1). During the first storm (8/12/04), W1 inlet average concentrations were more than twice the outlet average concentrations. In W2 the outlet concentration was three times higher than the inlet concentration. During the last storm (10/13/04), the maximum average concentration (6.0 mg/L) was measured at the W2 inlet. This average concentration was reduced by about 45% between the inlet and outlet of W2. The average concentration at the W1 inlet was also elevated, with a 24% reduction at the outlet. After the first storm, both inlet and outlet concentrations measured in W2 were all higher than those measured in W1. The percent reductions in average TKN concentration were also higher in W2 compared to W1.

The pre-construction storm sample at both inlets and outlets to the stream had lower average concentrations compared to post-construction storm samples. Blanks submitted for quality control with storm samples were all relatively low compared to the average concentrations measured during storms.

The TKN loads per wetland acre determined for composite storm events are shown in Figure 8.2. During the first storm sampled after construction, W1 had a larger amount of TKN entering than leaving, while W2 exhibited the opposite trend. All loads per wetland area were relatively low compared to the last storm, when a large load was

contributed at the W2 inlet. This load was reduced at the W2 outlet by about 28%, compared to a 12% reduction in W1. The total TKN loads per wetland acre estimated for all composite storm events sampled are shown in Table 8.2. The total TKN load at the W2 inlet was about 2.5 times the load estimated at the W1 inlet, and outlet differences were even greater. The estimated TKN load reduction was greater for W1 (40%) compared to W2 (2%).

The differences between average TKN concentrations and estimated TKN loads were mainly due to the large average concentrations measured during the last storm (10/13/04). In this case, estimating concentrations for the other storm (10/15/04) in the composite using the concentrations from the previous storm measured may not have produced an accurate TKN load for W2. The average TKN concentration measured at the inlet during the last storm was believed to be appropriate, however, based on the elevated average TKN concentration at the outlet. If this concentration was due to an isolated event within the watershed, then the percent reduction in total estimated TKN load would have been similar to reductions in W1. The cause of the elevated TKN concentrations in W2 during the last storm was unknown.

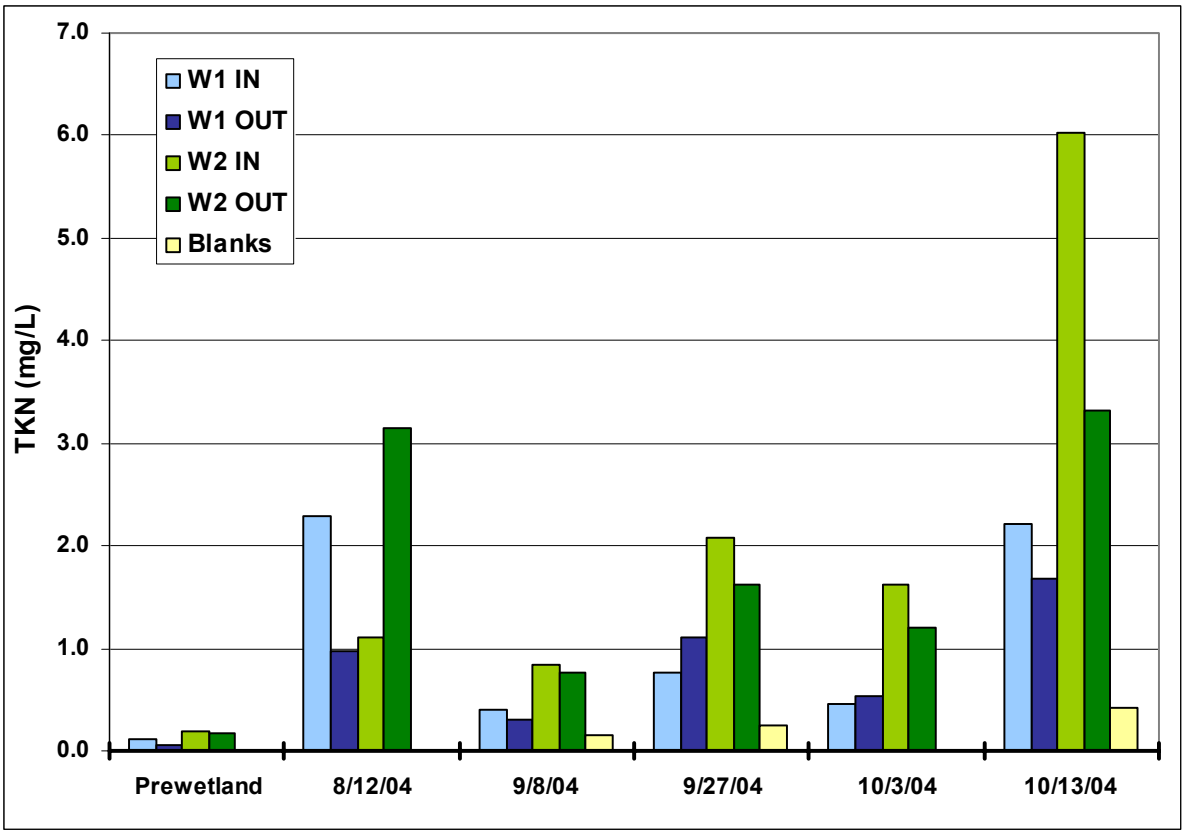

Figure 8.1: Average Total Kjeldahl Nitrogen (TKN) Concentrations Measured During Storms in Wetland 1 (W1) and Wetland 2 (W2).

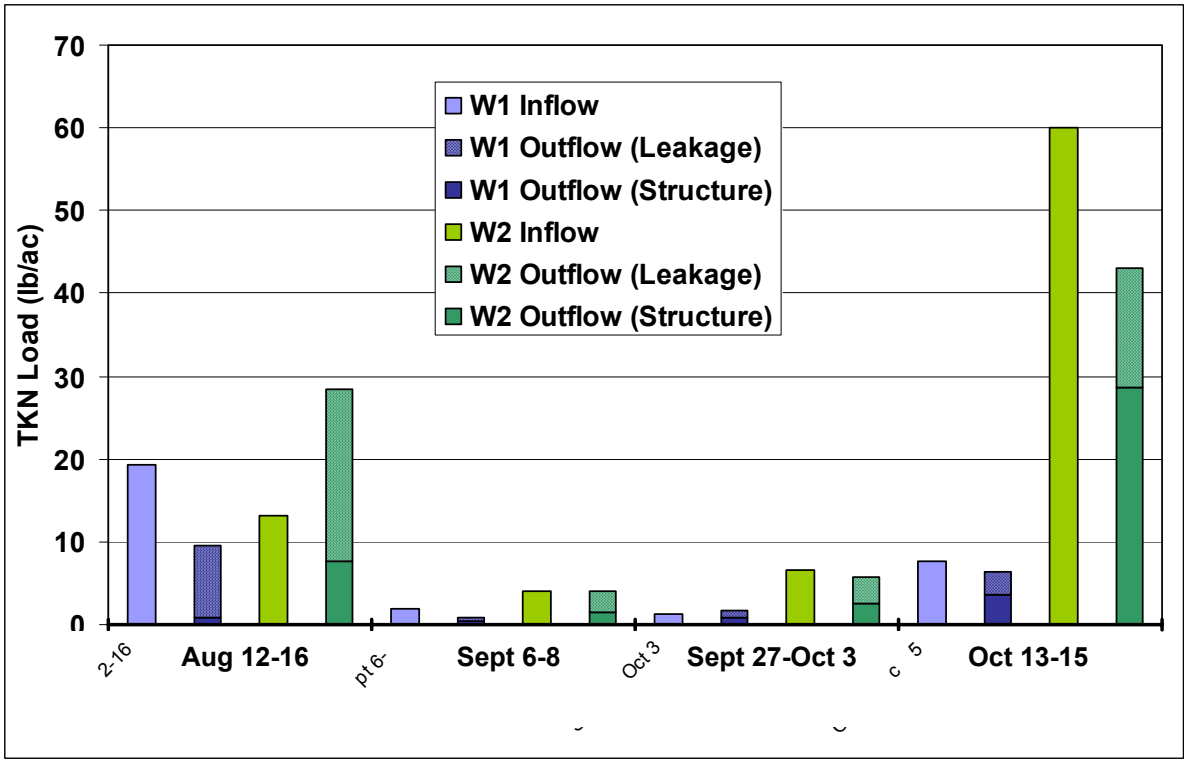

Figure 8.2: Total Kjeldahl Nitrogen (TKN) Loads per Acre of Wetland Measured in Wetland 1 (W1) and Wetland 2 (W2) for Composite Storms.

Ammonium-Nitrogen

Average NH_4-N concentrations were below 0.8 mg/L in both wetlands before and after construction (Figure 8.3). During storm events, the greatest concentrations were generated at the W2 outlet during the first and last storm measured. All inlet samples were generally below 0.1 mg/L in both CSWs, except in the first storm at the W1 inlet. W1 also had all outlet average concentration below 0.1 mg/L. The blank samples had little ammonium-nitrogen, helping to verify the storm sample concentrations.

Ammonium-nitrogen loads per wetland acre showed similar trends to loads measured during storm events (Figure 8.4). The largest load per acre was observed at the

outlet of W2 during the first storm measured, about 7 lb/ac. The highest load per acre for W1 was observed at the inlet during the first storm (about 0.2 mg/L), but relatively little was contributed or released in this wetland during the remainder of the storms.

The total NH_4-N load per wetland acre estimated for all storms was very low for W1 inlet and outlet, and at the W2 inlet (Table 8.2). The NH_4-N estimated load per acre released at the W2 outlet was over 10 times more than the estimated inlet amount. This elevated outlet load in W2 was mostly attributed to the elevated NH_4-N average concentrations during the first and last storm. Due to small average concentrations measured for NH_4-N, any increase would be considered large based on percent changes. Similar to the results of the TKN estimated loads, NH_4-N load estimations for the composite storms may have been overestimated at the W2 outlet by applying these average concentrations to all storms in the composite. If the elevated average concentrations were isolated events, then the difference between the W2 estimated outlet load and estimated inlet load may not be as great as calculated. Even with the potential errors associated with the composite storm estimations, the elevated concentrations at the W2 outlet observed during the study would still probably have shown an increase in NH_4-N load per acre between the inlet and the outlet.

The NH_4-N average concentrations (< 0.8 mg/L), and loads (< 8.0 lb/ac) were relatively small during this study, as expected. Based on these results, the TKN average concentrations and loads estimated during this study can be used as an indication of organic nitrogen amounts.

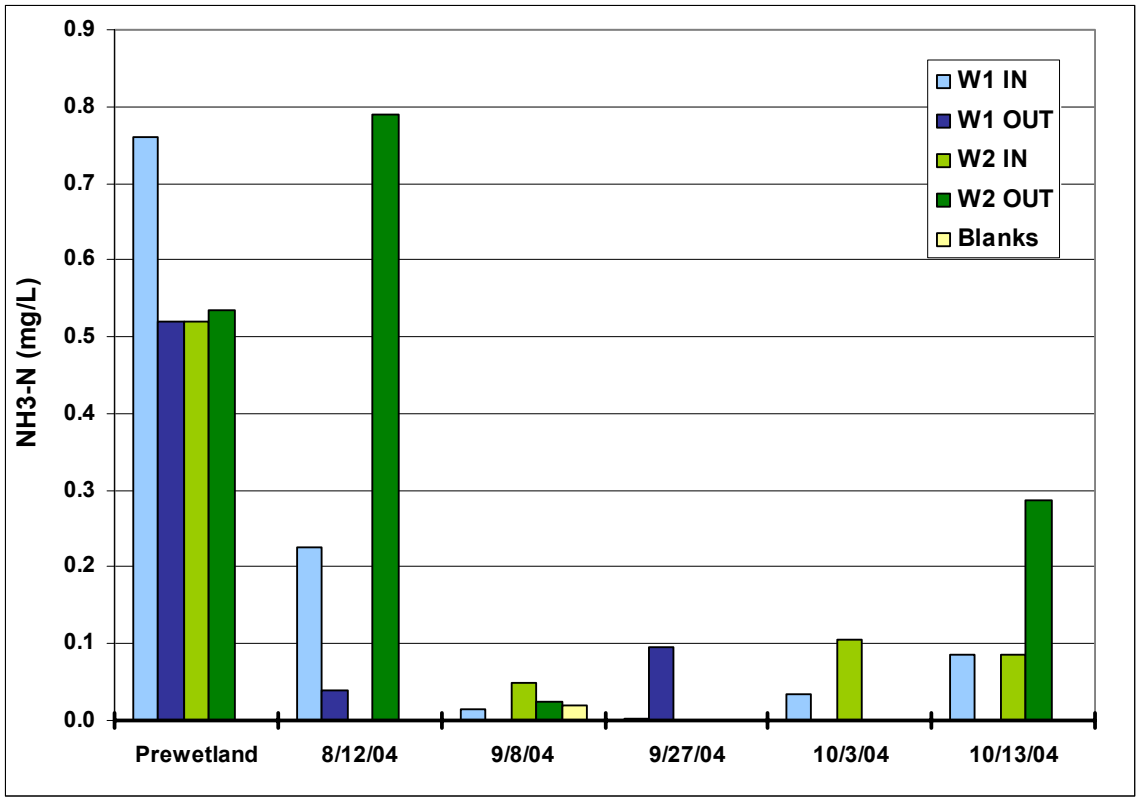

Figure 8.3: Average Ammonium-Nitrogen (NH_4-N) Concentrations Measured During Storms in Wetland 1 (W1) and Wetland 2 (W2).

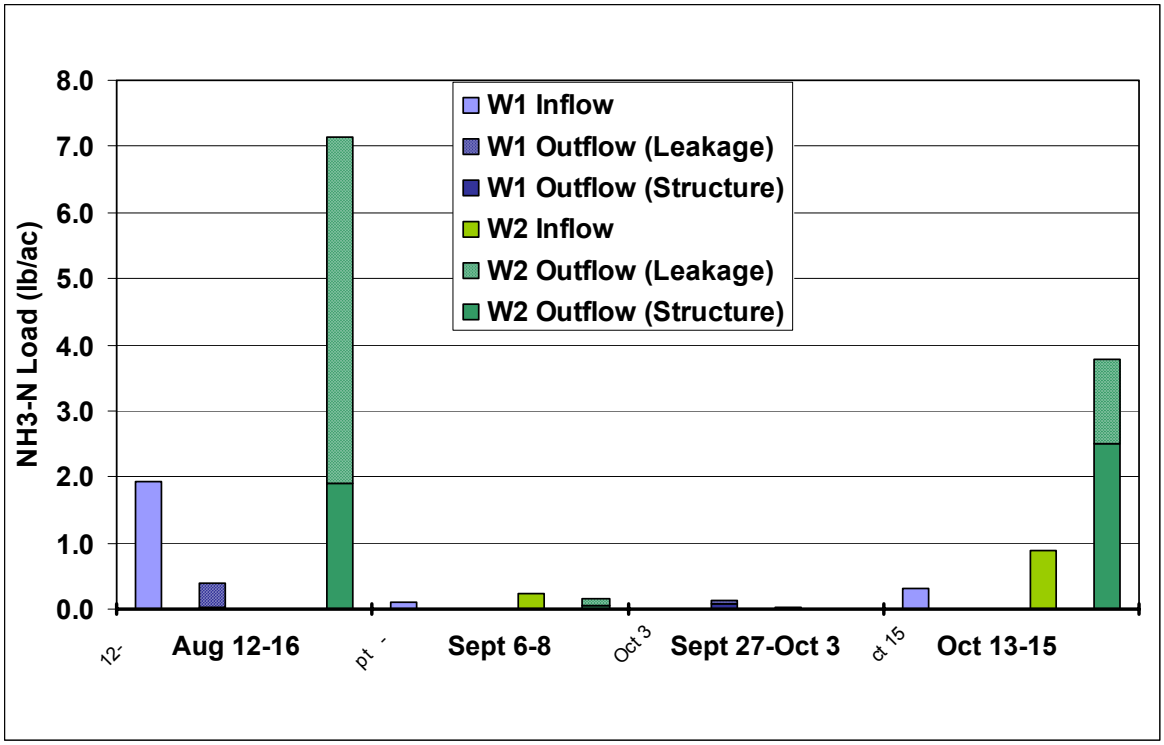

Figure 8.4: Ammonium-Nitrogen (NH_4-N) Loads per Acre of Wetland Measured in Wetland 1 (W1) and Wetland 2 (W2) for Composite Storms.

Nitrate-Nitrogen

Average NO_3-N concentrations measured were all less than 2.5 mg/L before and after construction. The largest average concentration was measured at the W1 inlet (2.4 mg/L) during the first storm (Figure 8.5). Average NO_3-N concentrations were generally greater at the inlets compared to the outlets for both wetlands during the study. A higher average concentration was typically contributed at the W1 inlet compared to the W2 inlet. Pre-construction average concentrations were greater than those measured after construction only at the outlets. Blank samples had relatively little NO_3-N measured and were used as quality control standards for storm samples.

Nitrate-nitrogen loads per wetland acre were all below 10 lb/ac during each storm, except for the relatively large load per acre measured at the W1 inlet during the first composite storm (Figure 8.6). The cause of this elevated inlet NO_3-N load in W1 is unknown, with no observed changes in watershed activity before these composite storms. Nitrate-nitrogen loads per acre were reduced between the inlet and outlet for all storms in both wetlands. The greatest reduction (85%) occurred in W1 during the first storm. The total NO_3-N estimated load per acre reductions were greater in W1 (86%) compared to W2 (56%), but inlet loads and outlet loads were similar in the two wetlands (Table 8.2).

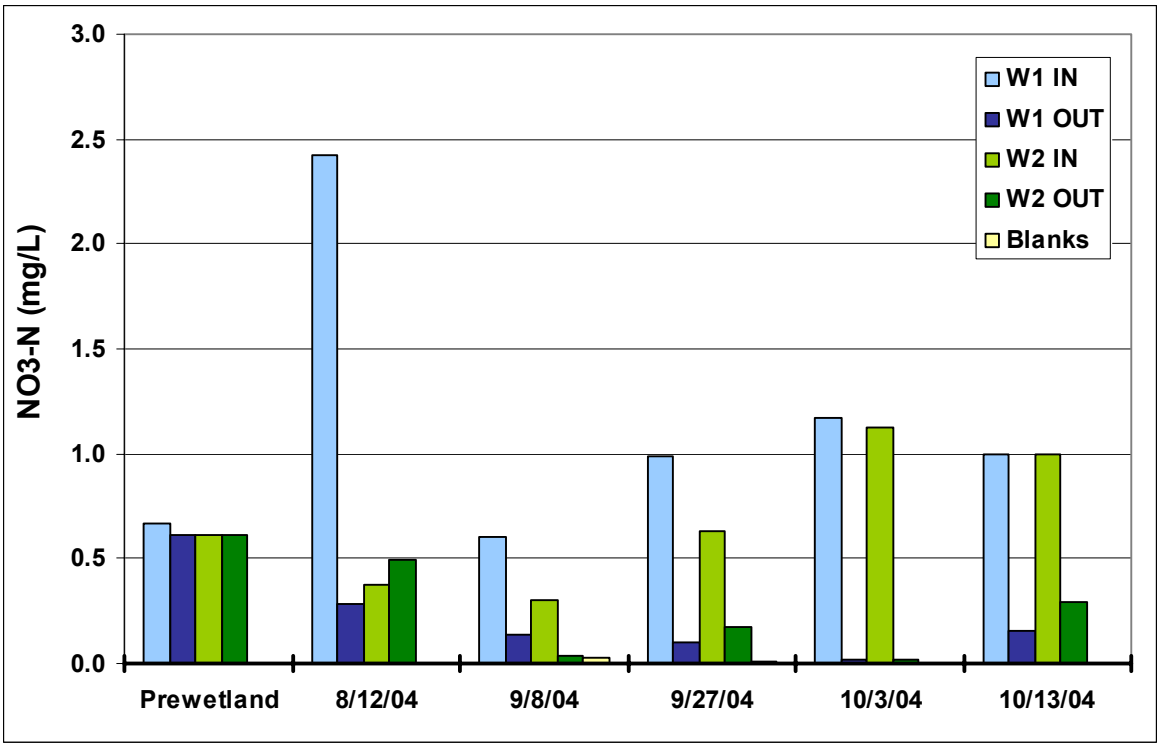

Figure 8.5: Average Nitrate-Nitrogen (NO_3-N) Concentrations Measured During Storms in Wetland 1 (W1) and Wetland 2 (W2).

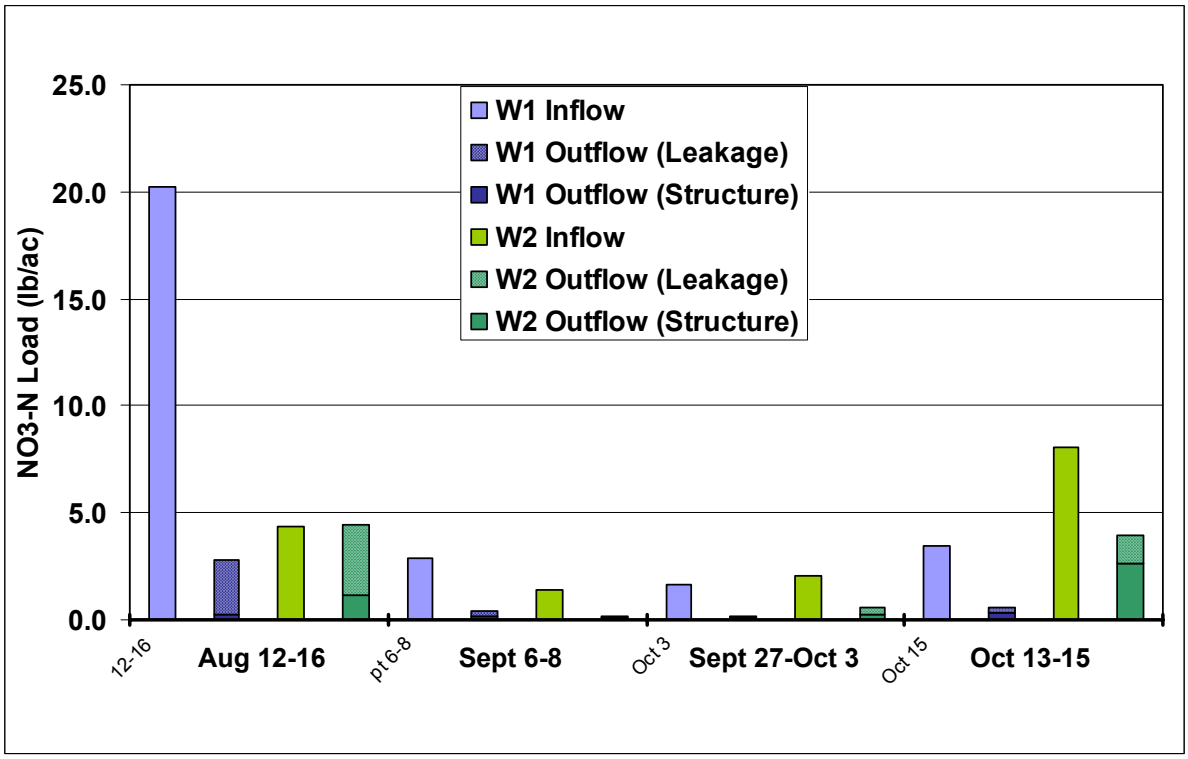

Figure 8.6: Nitrate-Nitrogen (NO_3-N) Loads per Acre of Wetland Measured in Wetland 1 (W1) and Wetland 2 (W2) for Composite Storms.

Total Phosphorus

Average TP concentrations were below 0.6 mg/L for all storms measured (Figure 8.7). Similar average concentrations were contributed at the inlets compared to release at the outlets for both wetlands except for the first storm. During this storm, W1 had a much larger average TP concentration at the inlet compared to the outlet, while W2 demonstrated the opposite pattern. There was no consistent pattern of average concentration reduction between storms. Elevated TP concentrations were also not observed during the last storm as with other nutrients. Pre-construction average TP concentrations were generally less

than post-construction measurements at both the inlet and outlet. The blank samples were relatively high in TP concentration for two of the storms measured, which must be considered when analyzing these results. Due to small average concentrations observed for both CSWs, any change in concentration would seem large when considering percent reduction as with the ammonium-nitrogen data.

Total phosphorus loads per acre of wetland were high during the first storm for both wetlands at the inlets and the outlets (Figure 8.8). W1 reduced the TP load by 30%, while W2 increased the TP load by about 35%. A large TP load contribution was made at the W2 inlet during the last storm, with nearly double the load per acre at the outlet compared to the inlet. For all storm events measured, the loads per acre were greater at the outlet of W2 compared to W1, and W2 appeared to generate TP between the inlet and the outlet during the study. The estimated total loads per wetland acre for all composite storms evaluated were similar or equal between the inlets and outlets of both CSWs (Table 8.2). The total TP loads in W2 were not much higher than the total TP loads in W1 due to the relatively low average TP concentrations measured during individual storm events.

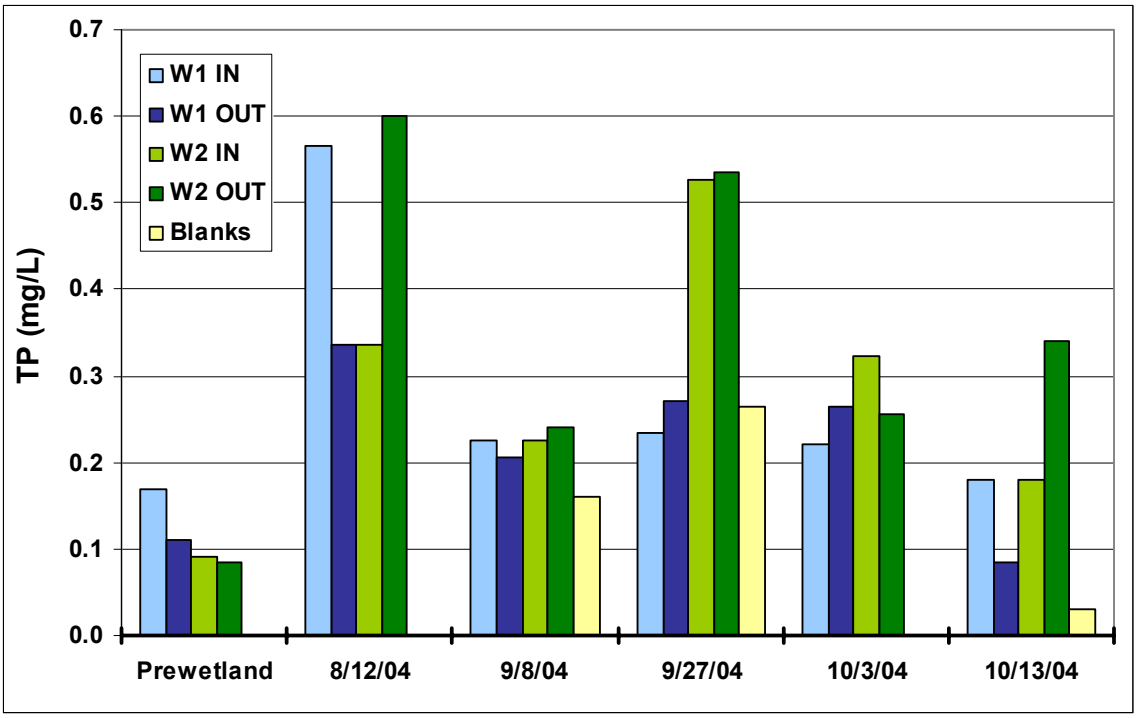

Figure 8.7: Average Total Phosphorus (TP) Concentrations Measured During Storms in Wetland 1 (W1) and Wetland 2 (W2).

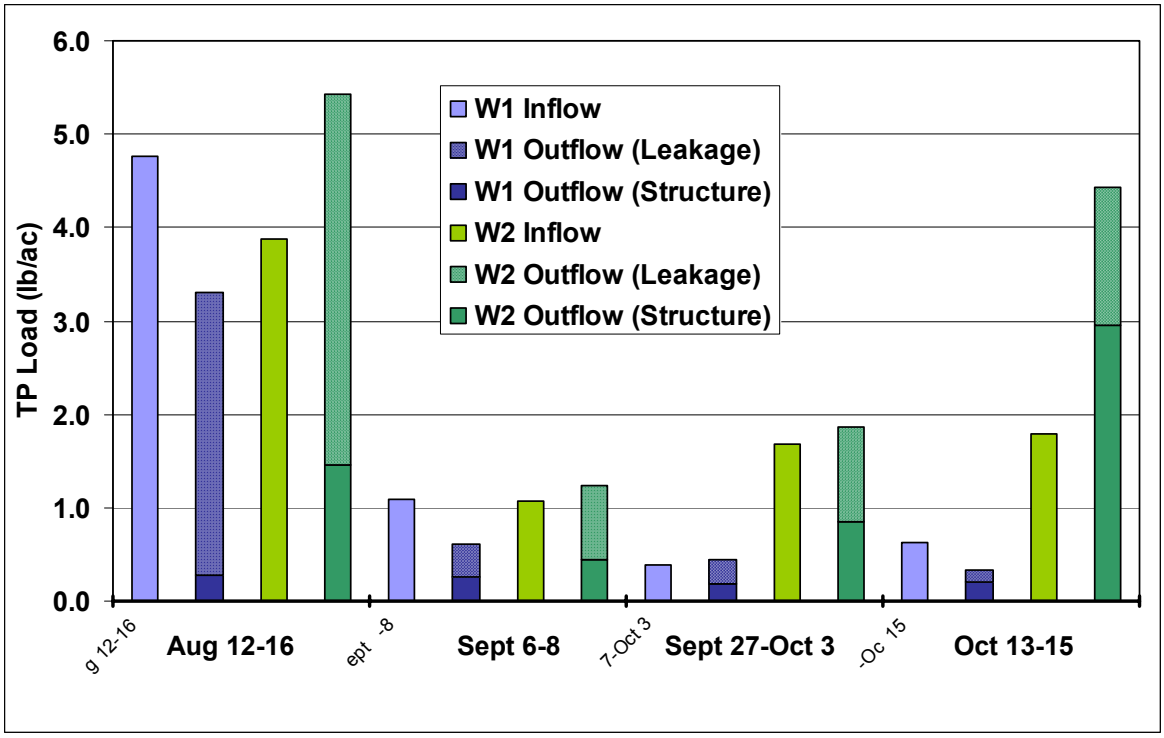

Figure 8.8: Total Phosphorus (TP) Loads per Acre of Wetland Measured in Wetland 1 (W1) and Wetland 2 (W2) for Composite Storms.

Table 8.2: Estimated Total Nutrient and Sediment Loads per Acre of Wetland for Composite Storms Evaluated During the Study Period.

	Wetland 1 (lb/ac)	Wetland 2 (lb/ac)
Inlet TKN	30	83
Outlet TKN	18	81
Difference	- 40%	- 2%
Inlet NH_4-N	2	1
Outlet NH_4-N	1	11
Difference	- 50%	+ 1000%
Inlet NO_4-N	28	16
Outlet NO_4-N	4	9
Difference	- 86%	- 56%
Inlet TP	7	8
Outlet TP	5	13
Difference	- 29%	+63%
Inlet TSS	247	2812
Outlet TSS	354	1727
Difference	+ 43%	- 61%

Nutrient Sources and Nutrient Retention

The nutrient reduction efficiencies of these two CSWs were comparable to published data, even though these literature studies were based on long-term averages (Schueler, 1992; Strecker *et al*., 1992; Brown and Schueler, 1997; DeBusk and DeBusk, 2001). Both wetlands performed similar to established CSWs for TKN reduction (15-48%) and for nitrate reduction (15-75%). W1 performed better than expected in ammonium-nitrogen reduction (literature average 10-43%), while W2 contributed ammonium to the stormwater leaving. The ammonium-nitrogen concentrations were so

low, however, that even a minute increase would seem considerable when evaluating proportions. Total phosphorus reductions in W1 were also similar to the published data (37-90%), but W2 did not appear to be successful at TP retention. These initial nutrient reduction rates are a positive indication of wetland function in general, and are expected to improve as mature vegetation establishes.

For all of the nutrients measured during this study, the average concentrations in the stormwater entering the CSWs were generally within the expected range for urban runoff (USEPA, 1983). The average inlet concentrations for all nutrients in both CSWs were highly variable during the study, however. During the first storm measured (8/12/04), nutrient average concentrations were elevated at the W1 inlet compared to other concentrations measured in following storm events. These concentrations may have been due to fertilizer application within the watershed, particularly the COT lawn area upslope, but no specific activity was observed during this time. The last storm event measured (10/13/04) showed elevated TKN average concentrations in W2 that were not observed in previous storm events. The source of this organic nitrogen was unknown, with no specific activity observed before this storm. CSW repairs and hydromulching were performed on 9/21/04, near the upslope stormwater intake, and may have had a delayed affect as a result of the larger storm on 10/13/04. W2 had elevated TP concentrations during a middle storm (9/27/04), but elevated concentrations measured in the blanks submitted with these samples may suggest the TP average concentration was overestimated.

The nutrient average outlet concentrations and loads per wetland acre were expected to decline over the course of the study, as the CSWs stabilized. During the initial

storm event, nutrients were expected to be generated within both CSWs due to the unstable substrate, lack of mature vegetation, and the addition of nutrients during hydromulching activities. By the last storm measured, nutrients were expected to be reduced at the outlets as a result of substrate stabilization, addition of vegetation, and depletion of the fertilizers applied during hydromulching.

The outlet average nutrient concentrations in W1 were similar throughout the study, with only TP showing a notable decline over time. W1 had reductions in all nutrient concentrations and loads during the study, with relatively low nutrient concentrations and loads at the outlet. W1 appeared to retain nutrients throughout the study, with little to no generation of nutrients from within the CSW. The total estimated nutrient loads for all composite storms indicated that W1 was retaining nutrients, with both lower inlet and outlet total loads compared to W2.

The outlet average concentrations and loads in W2 were greater than the inlet amounts for all nutrients during the first storm, indicating that nutrients were probably being generated within this CSW. By the end of the study, NH_4-N and TP average concentrations and loads were still elevated at the W2 outlet compared to the inlet, suggesting continued generation of these two nutrients from within the CSW. The total estimated NH_4-N loads and TP loads indicated that these nutrients were not reduced between the inlet and the outlet in W2. Organic nitrogen as a portion of TKN and NO_3-N total loads were reduced and appeared to be retained by the CSW, but the percent reductions were lower compared to those observed in W1.

Differences in nutrient retention between the two wetlands could be attributable to several factors. Inlet nutrient concentrations and loads for all nutrients were greater in W2 compared to W1. Nutrients were readily available in both CSWs from hydromulching materials and topsoil; therefore, additional large amounts of nutrients might not have been utilized by the temporary vegetation and microorganisms within W2. For both CSWs, the hydromulch components were assumed to be the same in nutrient content and availability between the two sites. Differences in topsoil nutrient availability may have occurred due to differences in topsoil condition. The topsoil was observed to be more compacted in W1 compared to W2, due to additional heavy equipment exposure near the end of construction. Compaction could have decreased sediment suspension with associated nutrients, particularly phosphorus. The topsoil was also acquired from a long-term tall piling from a nearby construction site, and may have had differential nutrient contents stratified throughout the pile.

Another factor that could have contributed to differences between the two CSWs was the second hydromulching (9/21/04) on upslope and surrounding areas of W2. This re-application was performed due to sparse temporary vegetation establishment at this site. Stormwater runoff into the CSW from these areas may have carried additional nutrients from hydromulching products that were not accounted for in the inlet culvert measurements. The initial hydromulch application was more successful in W1 based on the denser temporary vegetation establishment on the berms and upslope areas surrounding the CSWs. This vegetation may also have increased nutrient retention through plant uptake and enhanced deposition of suspended sediment in stormwater runoff.

Another observed difference between these two CSWs was the presence of thick algal mats in W2 throughout the study. Algae were present in the water column and along the substrate where the permanent pool covered the bottom starting in August. Although algal colonies increased in W1 by the end of the study, the surface area covered was always considerably less than in W2. The algal presence between storm events demonstrates that there were more nutrients available within W2 compared to W1. Prolific algal colonies can assimilate large amounts of both nitrogen and phosphorus through absorption and diffusion and store these nutrients in organic form as biomass (Vymazal, 1995). Algae along with bacteria can also fix nitrogen from the atmosphere to increase total nitrogen in the wetland system (Wetzel, 1983). During senescence and decay, algae settle to the bottom substrates where decomposition releases the nitrogen and phosphorus back into the water column for recycling and perpetual growth of the microbial communities (Wetzel, 1983, Richardson and Vepraskas, 2001). This substrate level decomposition was supported by the lower dissolved oxygen concentrations in the Deep Pool regions of W2 (Vymazal, 1995). In the CSW environment, this nutrient cycling probably continued until the next storm event washed the suspended algae and associated nutrients through the outlet.

All of these factors listed above could be responsible for higher TP amounts at the outlet of W2. The source of elevated NH_4-N at the outlet of W2 was unexpected and difficult to explain. During senescence, algae can release NH_4-N into the water column, but this ammonium is usually rapidly sequestered by the community (Vymazal, 1995). Elevated concentrations during the last storm could still be contributed from the algal

community, if large enough amounts were released into the water column. The biological factor could not explain the first storm results, however, due to the absence of significant algal mats at this time. Errors in nutrient analysis and nutrients introduced during sample collection might explain the unusual elevation in NH_4-N during the first storm.

Nutrient and Sediment Relationships

Sediment results for both CSWs are detailed in Chapter 7. Total suspended sediment (TSS) loads per acre estimated for the composite storms evaluated for nutrients were increased between the inlet and outlet of W1, while reduced between the inlet and outlet of W2 (Table 8.2). These trends were different for the nutrients measured during the study, with W1 reducing nutrient loads and W2 generating phosphorus and ammonium. Since suspended sediment was known to associate readily with phosphorus, the relationships between average TSS concentrations and average TP concentrations measured at the inlets and outlets of both CSWs during the same storm events were evaluated. The average TSS concentrations and average TP concentrations were poorly correlated at all four sampling locations (Figure 8.9). There appears to be other factors besides sediment influencing the relative amounts of phosphorus in the CSWs. Organic nitrogen as a portion of TKN also has the ability to associate with suspended sediment, depending on the type of compound present. The relationship between average TKN concentrations and average TSS concentrations during the measured storm events was also poorly correlated (Figure 8.10).

The amount of nutrients estimated to have been released at the outlets through leakage was relatively high during the study, similar to the sediment study. As discussed

in the previous chapter, the leakage rates were not expected to have affected the amount of nutrients leaving the CSWs. The nutrient loads at the outlet would have been the same even if leakage were not a factor. The leakage amounts were included in this chapter for future comparisons over time when leakage is expected to cease.

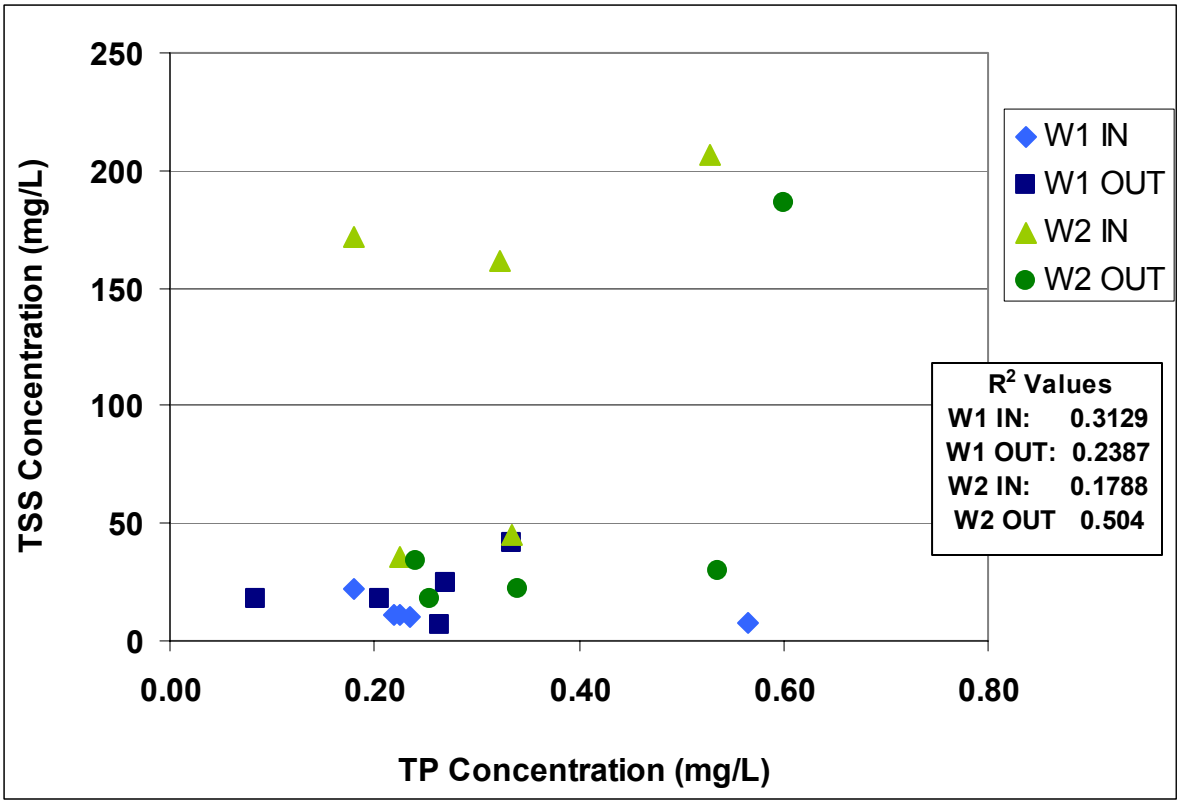

Figure 8.9: Correlations Between Total Suspended Sediment (TSS) Concentrations and Total Phosphorus (TP) Concentrations at the Inlet and Outlet of Wetland 1 (W1) and Wetland 2 (W2).

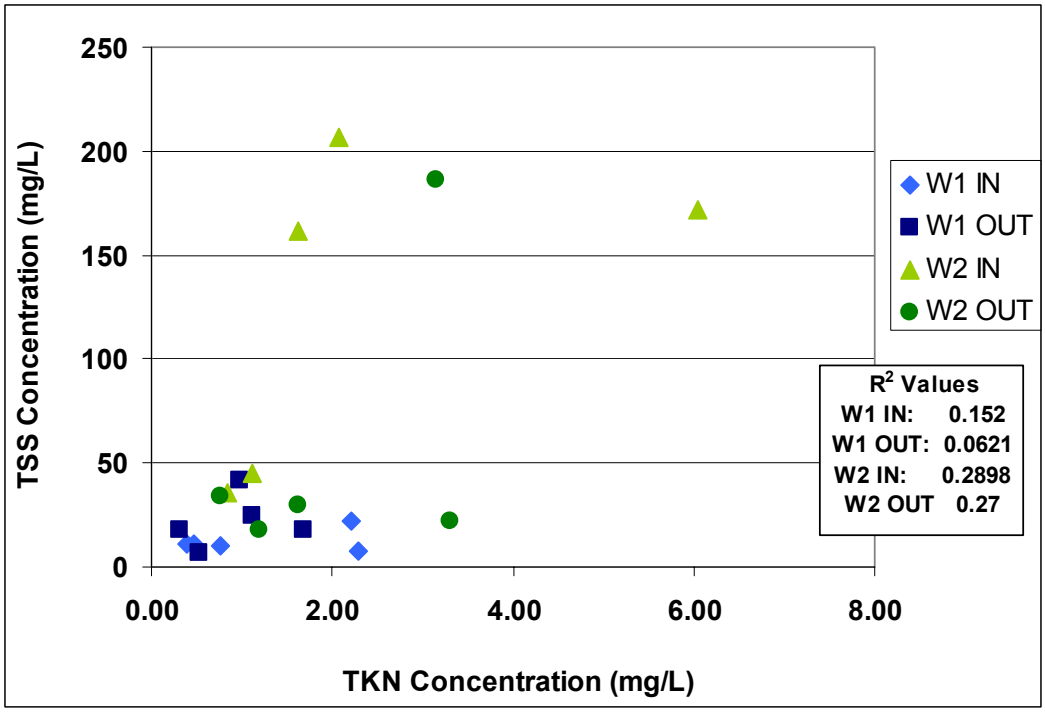

Figure 8.10: Correlations Between Total Suspended Sediment (TSS) Concentrations and Total Kjeldahl Nitrogen (TKN) Concentrations at the Inlet and Outlet of Wetland 1 (W1) and Wetland 2 (W2).

PAM and Nutrient Retention

The application of PAM in W1 was expected to aid nutrient retention, particularly phosphorus, due to its abilities in erosion control and suspended sediment reductions in runoff water. Other studies have shown variable results with PAM applications and nutrient reductions. In furrow irrigation applications with frequent PAM dosing, the total phosphorus was found to be reduced 86-92%, (Lentz *et al*., 1998; Lentz *et al*., 2002) with sediment reduction and phosphorus reduction found to only be moderately correlated. Soupir *et al*. (2004) evaluated the loss of nutrients from construction field plots and found reductions in total phosphorus loads were an average of 13-33%, and reductions in total

nitrogen loads (including sediment bound nitrogen) were an average of 18-59% with PAM applications. The nutrient reductions in this study for W1 are comparable to this study, but the amount of variables that were different between the two CSWs limits establishment of a relationship between the PAM application and the nutrient reductions observed.

The nutrient data collected during this study could not support or refute the influence of PAM in nutrient reductions observed in W1. The PAM may have helped reduce nutrient release from the hydromulch fertilizers and topsoil in W1, compared to W2 where no PAM was applied with these additives. The increased nutrient concentrations and loads at the outlet of W2 along with the enhanced algal presence in W2 indicate that nutrients, particularly total phosphorus, were more available in the water column of this CSW. The PAM influence may have been short-lived, only during the initial weeks of the study. The poor relationship between phosphorus and sediment concentrations and the opposite trends in TP and TSS reductions between the inlet and outlet of W2 suggest that PAM was not an important factor in nutrient reduction during the entire study. More research is needed to determine the effects of PAM on the hydromulch components and topsoil. Further study should be conducted in CSWs that have less variable differences, or in cells of the same CSW with similar stormwater nutrient inputs.

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CONCLUSIONS

The purpose of this study was to measure nutrient concentration and nutrient loads in the North Creek CSWs during the initial stabilization period. The study found that both CSWs were capable of reducing some nutrients during stormwater treatment between the inlet and outlet. W1 reduced all four nutrients evaluated, while W2 did not reduce NH_4-N and TP. There were several factors related to incoming stormwater nutrient content and site differences that could have caused the differential retention of nutrients. Based on the presence of enhanced algal communities in W2 in combination with elevated nutrient amounts at the outlet, there was evidence that this CSW had higher nutrient concentrations in the water column. Due to the relatively low concentrations of NH_4-N and TP measured throughout the study (<0.8 mg/L), sample analysis errors or sample collection errors could have affected the results with only small changes to the concentrations. A longer time period of sampling and more storm events measured could provide additional information about these two nutrients in both CSWs.

The purpose of this study was also to evaluate the effects of PAM application during hydromulching on nutrient retention. The PAM was applied in W1 at a rate slightly higher than recommended (15 lb/ac). Based on the nutrient data, W1 did not appear to generate nutrients between the inlet and outlet during the study, and W1 had higher reduction rates compared to W2 during the majority of storm events evaluated. The nutrient reduction in W1 could not be solely attributed to the PAM application, however, due to the numerous variables that were different between the two CSWs, including higher inlet nutrient concentrations in W2, application of additional nutrients upslope of W2, and

the relative compaction of the topsoil in W2 compared to W1. The effects of PAM could also not be determined during this study based on the poor correlation between the nutrients (TP and TKN) and sediment amounts. The ability of PAM to reduce nutrient loss is associated with its ability to reduce sediment loss. During this study, TSS concentration and load results showed opposite trends compared to the nutrient concentrations and load results.

The study indicates that PAM application may be beneficial during the initial stabilization period of CSWs, but further research is needed to determine the specific contribution PAM applications make to nutrient reductions.

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