ABSTRACT

GREGG, SIMON BENJAMIN. Mountains to the Piedmont: Urban Thermal Pollution Mitigation and Regenerative Stormwater Conveyance Design Review and Optimization. (Under the direction of Dr. William F. Hunt, III).

Land use changes and urbanization negatively impact aquatic environments. Urbanized watersheds discharge higher runoff volumes and larger peak flows, while exporting pollutants, including sediment, oils and grease, heavy metals, nutrients, and pathogens. Thermal enrichment is a lesser-studied stormwater pollutant that has substantial impacts on cold-water streams. A study in Hendersonville, North Carolina, was conducted to assess the performance of a rainwater harvesting (RWH) system, a level spreader vegetative filter strip (LSVFS), and a bioretention cell (BRC) for temperature and thermal load reduction. All systems received thermally-enriched runoff from roof top and paved areas. BRC buffered maximum and, to a lesser extent, median event temperatures, while reducing flow volumes and thermal loads. BRC treatment reduced thermal stress on trout populations, resulting in a 27% chance that maximum outlet temperatures would exceed 21^{o}C, the avoidance threshold for brook trout (*Salvelinus fontinalis*). The RWH appeared to thermally stratify, which resulted in significantly hotter water temperatures exiting the storage tank overflow. Typical RWH design, with the outlet determining maximum storage capacity, increased the likelihood of effluent temperature exceeding the 21^{o}C trout threshold. LSVFS displayed minimal ability to buffer influent temperatures, resulting in a 75% chance of maximum effluent temperatures exceeding 21^{o}C.

Regenerative Stormwater Conveyance (RSC) is a new Stormwater Control Measure (SCM) which integrates urban stormwater management with stream restoration techniques to simultaneously convey stormwater and treat pollutants. RSCs are most often used to stabilize eroded streams, gullies, and stormwater outfalls by filling the channel bed with sand and

woodchip media. An undersized RSC was constructed with an exfiltration trench downstream of a dry pond in the North Carolina Piedmont to stabilize an advancing headcut and provide pollutant treatment. Two inlets and one outlet were monitored for flow and select pollutants. True Event Mean Concentrations (EMCs) were unable to be collected due to flow-measurement challenges; instead Representative Concentrations (RCs) were collected. Influent calibration allowed analysis of 15 hydrologic events revealing no volume reduction via RSC treatment due to poorly drained underlying soils, limited surface ponding, constant media saturation, and a large loading rate. Water quality treatment, however, was modest for most analytes. Storm flow Removal Efficiencies (REs) for TN, TP, and TSS were 3%, 26%, and 69%, respectively; TSS RC reduction was significant. Base flow REs for TN, TP, and TSS were 22%, 3%, and 11%, respectively; TN and TSS RC reductions were significant. RSC temperature mitigation was similar to that of BRC. While this RSC likely eliminated headcut migration and channel erosion, other benefits such as hydrologic mitigation and pollutant removal were muted, likely by its extremely small size relative to its drainage area.

A review and synthesis of RSC literature was conducted. Depending on site constraints (i.e., available disturbance, right-of-way, economics, etc.) loading ratios should be adjusted to accommodate the design storm; however, effective treatment of typical NC water quality volumes is possible when loaded at 80-120:1. Conversion of surface flows to sub-surface seepage is an important factor RSC employs for treatment. A new metric is proposed, the Storage Ratio (ratio of surface storage to media storage volumes), to compare designs and provide insight into the partitioning of storage volumes; storage ratios should range from 0.5-1.5:1. Media saturation, whether purposefully-designed, from base flow, or due to groundwater intrusion, inhibits surface-to-seepage flow conversion; saturated media volumes should not be included in total storage volumes. Media composition can vary depending on usage in well-drained or saturated conditions; but should generally follow specifications for BRCs. Type, size, and number of in-stream structures will be site dependent and based on pool to pool spacing and vertical step limitation guidelines. Vegetation inclusion appears to be a benefit.

Mountains to the Piedmont: Urban Thermal Pollution Mitigation and Regenerative Stormwater Conveyance Design Review and Optimization

by
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DEDICATION

This thesis is dedicated to my family, our environment, and the journey to a positive impact on the Earth.

BIOGRAPHY

Simon Gregg was born August 20, 1990 in Huntsville, Alabama to Pam and Danny Gregg. The youngest of three siblings, Micah, Kata, and Simon are each approximately two-years apart and remain close friends. From an early age, Simon and Micah raced motocross; these experiences gave Simon an opportunity to work with his hands, gain experience with mechanical systems, and learn the responsibilities of equipment maintenance. Through this early passion, he also learned to channel his focus in order to overcome obstacles and achieve a desired goal; this is a trait he carries with him today.

Simon's uncle farms in Minnesota and he has spent time helping plant and harvest; giving him a valuable perspective and a connection to hardworking farmers in America and our food systems. Growing up in Alabama (the most ecologically diverse state east of the Mississippi River) and spending copious amounts of time in the Sipsey Wilderness (the first Wilderness area east of the Mississippi River), it's no wonder Simon's experiences shaped him to be a Biological and Agricultural Engineer.

Simon studied Biosystems Engineering at Auburn University and was fortunate to meet Adeline Lerner, his best friend and partner in all things. Simon and Adeline grew their friendship during travel to New Zealand and a month long stay in the South Pacific Fijian Island of VoroVoro with the Mali tribe. After graduation, Simon met Dr. Hunt and found support from the Stormwater Engineering Group. Adeline and Simon moved the North Carolina, and both joined graduate programs at NC State University in Landscape Architecture and Biological and Agriculture Engineering, respectively, in 2016. Their tenure has been filled with growth, cross-pollination, collaboration, challenges, and opportunities. They look forward to a bright future together.

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I want to thank the BAE graduate student family past and present; you all make the difficult times achievable. To my parents and family, I love you all. Thank you for your love and support! Finally, Adeline, I would likely be someone different had we not met at Auburn. Thank you for your enduring love and support! I am so grateful to have you as my partner in all things, and I am so excited to enter a new phase of life with you!

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List of Acronyms

ADP	Antecedent Dry Period
AEL	Annual Export Load
ALR	Annual Load Reduction
ARI	Annual Recurrence Interval
AVM	Area Velocity Meter
BLR	Baseflow Load Reduction
BRC	Bioretention Cell
BUA	Built Upon Area
C	Carbon
CBC	Cavalry Baptist Church
CEC	Cation Exchange Capacity
CI	Confidence Interval
CISR	Constructed In-Stream Riffle
CN	Curve Number
Co	County
Cu	Copper
CWA	Clean Water Act
DCIA	Directly Connective Impervious Area
DNRA	Dissimilatory Nitrate Reduction to Ammonia
DO	Dissolved Oxygen
EL	Event Load
ELR	Event Load Reduction

EMC	Event Mean Concentration
EPA	Environmental Protection Agency
ET	Evapotranspiration
EVAP	Evaporation
HSG	Hydrologic Soil Group
HUC	Hydrologic Unit Code
IWS	Internal Water Storage
JLR	Jordan Lake Rules
LID	Low Impact Development
LOI	Loss on Ignition
LS	Legacy Sediments
LSVFS	Level Spreader-Vegetative Filter Strip
MS4s	Municipal Separated Storm Sewer System
N	Nitrogen
NC	North Carolina
NCCA	National Coastal Condition Assessment
NCDEQ	North Carolina Department of Environmental Quality
NCDOT	North Carolina Department of Transportation
NH_3	Ammonia
NH_4^+	Ammonium
NOAA	National Oceanic and Atmospheric Administration
NO_2^-	Nitrite
$NO_{2,3}$-N	Nitrite-Nitrate

NO3-	Nitrate
NPDES	National Pollutant Discharge Elimination System
NRCS	Natural Resource Conservation Service
NSW	Nutrient Sensitive Water
OM	Organic Matter
ON	Organic Nitrogen
Ortho-P	Orthophosphate
P1	Priority 1
P4	Priority 4
Pb	Lead
PBP	Particle Bound Phosphorous
P-Index	Phosphorous-Index
PM	Particulate Matter
PN	Particulate Nitrogen
PVC	Polyvinyl Chloride
RC	Representative Concentration
RCP	Reinforce Concrete Pipe
RDU	Raleigh-Durham
RE	Removal Efficiency
RSC	Regenerative Stormwater Conveyance
RSE	Residual Standard Error
RWH	Rainwater Harvester
SCM	Stormwater Control Measure

SHWT	Seasonable High Water Table
SLR	Simple Linear Regression
SPSC	Step Pool Storm Conveyances
SWC	Seepage Wetland Complex
TAN	Total Ammoniacal Nitrogen
TFC	The Free Clinics
TI	Total Imperviousness
TKN	Total Kjeldahl Nitrogen
TLR	Total Load Reduction
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Sediment
TVA	Tennessee Valley Authority
UCAT	Uniform Continuous Above Threshold
U.S.	United States
USACE	U.S. Army Corps of Engineers
WQ	Water Quality
WQV	Water Quality Volume
WVDEP	West Virginia Department of Environmental Protection
Zn	Zinc

CHAPTER 1 . LITERATURE REVIEW: STORMWATER POLLUTANTS, MITIGATION STRATEGIES, AND TREATMENT PERFORMANCE

1.1. Abstract

Urbanization decreases infiltration, shallow interflow, and evapotranspiration which increases runoff volumes and velocities. Altered runoff regimes degrade stream ecosystems while exporting sediments, hydrocarbons, nutrients, pathogens, heavy metals, and thermal pollution. Hydrologic modification reduces stream base flows and alters stream morphology, stability, and ecosystem function. Stormwater Control Measures (SCMs) are documented to mitigate the hydrologic and water quality effects of land use change with varying degrees of success. Stream restoration is a tool to reduce sediment export, improve aquatic habitat, and increase nutrient sequestration in-stream. A review of the hydrologic and water quality treatment processes and the benefits provided by select SCMs' and stream restoration practices are presented in this chapter.

1.2. Introduction

In urban areas, vegetated landscapes are replaced by impervious hardscapes such as buildings, sidewalks, or roads. Imperviousness increases runoff volume while exporting sediments, hydrocarbons, nutrients, pathogens, heavy metals, and thermal pollution to receiving waters (Davis et al., 2003; Davis et al., 2006; Hunt et al., 2012; Jones et al., 2012; Kim et al., 2003). Increased frequency of high flow runoff events damage stream ecosystems and contribute to channel erosion and instability (Leopold, 1994; Meyer et al., 2005; Walsh et al., 2005). Water is required to support life; additionally, industry processes such as power generation, food processing, and manufacturing rely on surface and/or aquifer water.

As of 2014, 54% of the world's 7.2 billion inhabitants lived in urban areas with projections to add another 2.5 billion more people to urban populations by 2050 (United Nations, 2014). North Carolina is among the top 5 U.S. states in net migration population growth with 81,000 individuals moving to the state between 2015 to 2016. Total population growth during the same period was 112,000, a growth rate of 1.1%, surpassing the national average of 0.7% (Tippett, 2017). Population growth and migration to urban centers increases demand for land-use change, development, and resource consumption. Industry, agriculture, and drinking water systems compete for water resources and in some water scarce areas the water rights have become highly contested (Chattahoochee Riverkeeper, 2015; Hood & Sommer, 2016). In 2011, 45% of the U.S.'s gross domestic product was generated in coastal communities along oceans and Great Lakes (U.S. EPA, 2015). In the North Carolina Mountains, trout fishing had a total economic impact of $383.3 million and supported approximately 3,600 jobs in 2014 (Responsive Managment & Southwick Associates, 2015).

Nutrients and metals are of particular concern because of their persistence and bioavailability (Herngren et al., 2005; Ladson et al., 2006). Novotny (2003) reported that urban runoff from highway and urban diffuse sources were responsible for 30% of impaired stream reaches and 45% of impaired lakes. Trout require cold-water habitats-streams with mean weekly temperatures less than 22^{o}C (Armour, 1991). Cold-water habitats are threatened by land use change, riparian buffer loss, and urbanization due to increased thermal loads (Beschta et al., 1987; Rossi & Hari, 2007; Jones et al., 2012). Urban runoff contains many pollutants that can degrade water quality including: nutrients, suspended sediments, heavy metals, pathogens, and thermal loads (Davis et al., 2006; Jones et al., 2012; Leopold, 1994; Meyer et al., 2005; Walsh et al., 2005).

1.3. Urbanization and Hydrologic Impacts

The effects of urbanization on watershed hydrology are well documented; increased runoff volumes and peak flows result when infiltration and evapotranspiration are reduced or eliminated as vegetation is removed and soil is compacted and covered with impervious surfaces (Figure 1-1) (Davis et al., 2003; Leopold, 1994; Novotny, 2003; Paul & Meyer, 2001; Pitt et al., 2009). Conventional stormwater management focuses on efficient drainage design; conveyance systems using ditches and culverts deliver runoff directly to receiving waters with little or no attenuation or treatment (Burns et al., 2012). The resulting Directly Connective Impervious Areas (DCIAs) rapidly convey flows directly to receiving waters decreasing time to peak discharge, increasing flow velocities, and increasing peak discharge (Burns et al., 2012; Davis et al., 2006; Ladson et al., 2006; Leopold, 1994; Walsh et al., 2005). High velocity urban runoff increases stream scour and erosion of streambed and banks (Burns et al., 2012; Novotny, 2003; Ladson et al., 2006; Leopold, 1994; Walsh et al., 2005).

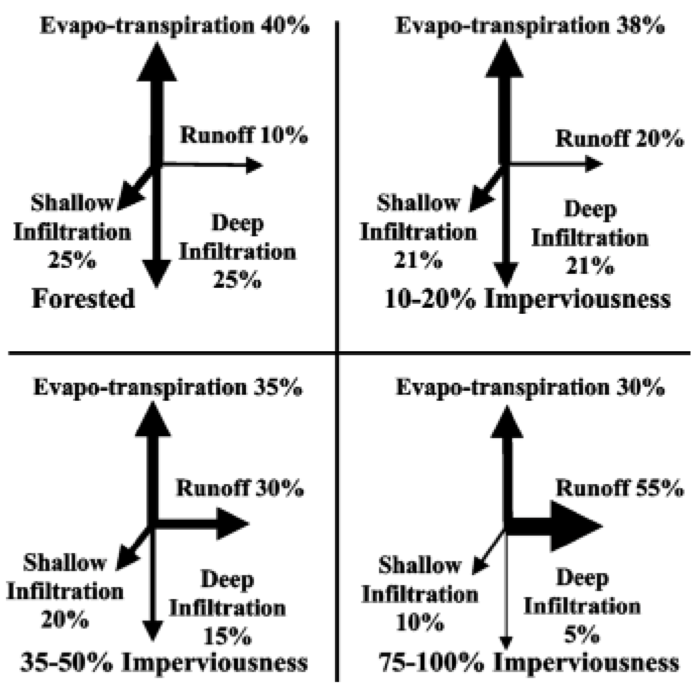


Figure 1-1. Hydrologic Impacts of Urbanization (Paul & Meyer, 2001)

1.4. Urbanization and Water Quality Impacts

1.4.1. The First Flush & Water Quality Volume

Metcalf and Eddy (1916) documented higher pollutant mass loading, mainly organic waste and horse manure, early in runoff events. This is the result of pollutant build-up on impervious surfaces during dry periods before subsequent wash-off during runoff events, referred to as the first flush. Pollution has changed since the early 1900s; we have moved away from horse drawn carriages to automobiles, busses, and trains, all of which contribute pollution to the urban environment. The first flush, however, is still experienced; although, some studies note its inconsistent effect in combined sanitary and storm sewer discharges (E.P.A., 1983; Ellis, 1986; Sansalone & Cristina, 2004). Despite this, the first flush is used to improve SCM design. Kayhanian et al. (2012) defines the first flush as 80% of pollutant mass delivery in the first 30% of runoff. While, Novotny (2003) provides the rule of thumb of 60% of pollution load carried in the first 40% of runoff.

Understanding the first flush has improved stormwater treatment with the development of the water quality volume (WQV), which represents the required capture volume needed to treat most stormwater pollutants. SCMs designed to capture 19- to 38-mm of rainfall will treat the majority of pollutants while limiting bypass (N.C. DEQ, 2017; Sansalone & Cristina, 2004). The North Carolina Department of Environmental Quality (NCDEQ) Stormwater Design Manual defines the WQV as the first 38-mm of rainfall in Coastal Counties and 25-mm of rainfall elsewhere in the state (N.C. DEQ, 2017). Design and treatment for the WQV results in the capture and treatment of 75% of annual storms plus the heaviest pollutant loads from storms exceeding the WQV; annual runoff treatment is approximately 85% to 90% (Sansalone & Cristina, 2004).

1.4.2. Pollutants

Varying gradations of Particulate Matter (PM) size are seen in urban runoff from large coarse PM (>75 μm) to fine PM (<75 μm); PM less than 45 μm is considered dissolved (Kim & Sansalone, 2008; Vaze & Chiew, 2004). PM in urban runoff is the result of upland erosion, atmospheric deposition, and land use activities (Freer-Smith et al., 1997; Lu et al., 2008; Novotny, 2003). Davis et al. (2001) found vehicle brake wear and tire abrasion to be a major source of copper and zinc, respectively, in urban roadway-runoff. Herngren et al., (2005) studied urban runoff along the Gold Coast region in Queensland, Australia, finding up to 90% of sediments in road side deposits were less than 75 μm in size; the largest portion was between 0.45- and 75-μm. Sansalone et al. (1998) collected lateral surface flow runoff from a highly-trafficked highway in Cincinnati, Ohio, where solids ranged from 1- to 10,000-μm; particle sizes ranging from the minimum detection limit, 2-μm, to 8-μm were most rapidly washed from the pavement (Sansalone et al., 1998).

Vaze & Chiew (2004) found less than 15% of Total Nitrogen (TN) and Total Phosphorous (TP) associated with particles larger than 300 μm. Vaze & Chiew (2004) also determined substantial variability in dissolved fractions of TN and TP ranging from 20-50% and 20-30%, respectively; most particulate bound nitrogen and phosphorous was attached to PM between 11-150 μm (Vaze & Chiew, 2004). Nitrogen and phosphorus are important pollutants because of their bioavailability and potential to cause eutrophication in receiving waters (Novotny, 2003).

Metals in stormwater have similar particulate affinities as nitrogen and phosphorus. Researchers have found Total Suspended Sediment (TSS) strongly correlated with metal concentrations (Herngren et al., 2005; Maniquiz-Redillas & Kim, 2016; Novotny, 2003).

Hallberg et al. (2007), determined TSS was a good indicator of metal concentrations in winter runoff, while in summer the correlation is strongest for aluminum, copper, iron, magnesium, nickel, zinc, and possibly lead. Maniquiz-Redillas & Kim (2016) found that for TSS concentrations less than 50-mg/L metals were predominantly particulate bound, noting an increase in total metals as the concentration of TSS increased.

Novotny (2003) identified a one meter strip along roadside medians and gutters as the location that accumulates more than 90% of roadway total solids. Herngren et al., (2005) studied mechanically-cleaned streets finding that only particles greater than 250 μm were effectively removed. The inability to mechanically remove PM less than 250 μm from roadsides highlights the need for treatment systems to manage these pollutants in runoff. Sansalone (1999) noted that treatment system design depends on the particle size gradation. If pollutants are mainly dissolved, treatments with sorption and precipitation abilities are needed, while filtration and sedimentation will be better for particulate bound constituents (Sansalone, 1999).

1.4.3. Thermal Load

The thermal regime of streams and rivers are influenced by many factors including: atmospheric conditions, topography, stream discharge, and streambed characteristics (Caissie, 2006). Low-order perennial headwater streams often experience little diurnal or temporal temperature change due to large groundwater inputs and riparian shading (Vannote et al., 1980); ephemeral streambeds lack streamflow and groundwater inputs at times and, thus, experience greater diurnal temperature variations (Vannote et al., 1980; Constantz et al., 2001). During runoff events, ephemeral stream beds cool substantially (Constantz et al., 2001). Alternatively, large rivers have little riparian shading and are less influenced by groundwater, therefore, they experience the maximum variation in daily temperatures (Caissie, 2006). Stream temperatures vary widely in space and time (Caissie, 2006). Ecosystem stability is directly linked to the stability of the physical system; low-diversity headwater biotic assemblages maintain stability because diel and temporal variations are low and few species can function in the narrow temperature range with limited nutritional base (Vannote et al., 1980).

Natural heat exchange processes take place at the air-water interface and the streambed-water interface when changes in stream flow are negligible (Caissie, 2006). Heat flux at the air-water interface occurs via solar radiation (net short-wave radiation) and net long-wave radiation, evaporative heat flux, and convective heat transfer, other minimal components include precipitation and friction (Caissie, 2006). Net radiation is often the most dominant heating component and highlights the importance of riparian vegetation in minimizing solar gain (Caissie, 2006). Stream bed interactions are mainly a function of geothermal loading and are likely more important in headwater streams due to the greater dominance of groundwater (Caissie, 2006).

Impervious surfaces and some SCMs are sources of thermal pollution (Jones et al., 2012; Jones & Hunt, 2010; Woodward, 2011). Jones et al. (2012) explored the effect of parking lot composition and layout on stormwater runoff temperature (Figure 1-2). Parking lot runoff experienced seasonal fluctuations with thermally enriched runoff generated during peak summer months, June-August, in afternoon storm events (Jones et al., 2012). The presence of tree canopy and shading was determined to reduce median and maximum runoff temperatures by 0.3^{o}C and 3.2^{o}C, respectively (Jones et al., 2012).

Figure 1-2. Hot-Mix with Shading (Left); Chip-Seal (Right) (Jones et al., 2012)

1.5. Urbanization and Streams

Urbanization replaces vegetated landscapes with impervious hardscapes such as buildings, sidewalks, and roads. Urban areas discharge higher runoff volumes and larger peak flows (Davis et al., 2003; Davis et al., 2006; Hunt et al., 2012; Jones et al., 2012; Ladson et al., 2006). Conventional stormwater management and efficient drainage alters infiltration, evapotranspiration (ET), and surface and sub-surfaces flow regimes (Figure 1-1); the lack of watershed storage and connectivity to streams increasing total runoff volume while reducing the time to peak discharge (Burns et al., 2012; Fletcher et al., 2014; Meyer et al., 2005). This reduces stream base flow and alters stream morphology, stability, and ecosystem function (Burns et al., 2012; Fletcher et al., 2014; Meyer et al., 2005). Collectively, the aforementioned characteristics are referred to as the urban stream syndrome (Walsh et al., 2005). Research to mitigate the impacts of urbanization often focuses on hydrologic matching to a pre-urban or pre-development hydrologic condition by reducing DCIA. Runoff retention and infiltration practices reduce the frequency of runoff events and provide treatment of a design volume (Burns et al., 2012; Davis, 2008; Fletcher et al., 2014; Ladson et al., 2006; Walsh et al., 2009).

Land use changes, whether for agriculture or urbanization, decrease deep and shallow infiltration, evapotranspiration, and increase surface runoff and soil erosion resulting in sediment aggregation in streams and on floodplains (James, 2013; Novotny, 2003). Legacy Sediments (LS) are sediments that were deposited in valley bottoms and along stream channels following human disturbances such as deforestation, agriculture, or mining (James, 2013). The accumulation of post-settlement sediments on floodplains may be thick, covering pre-settlement soils with a young stratified soil profile (James, 2013). The effect of settlement and land use change will vary with climatic region, in arid climates installation of lawns, parks, and vegetation can reduce runoff and decreased sediment delivery to streams (Novotny, 2003). Large accumulations of sediments may progress through facies changes from rills and gullies, to cobble and gravel bed streams in steep valleys, to floodplains and channel fill along large rivers, to fine-grained deposits in slack-water environments; incision and abandonment of legacy sediments on floodplains and in streams continue to be reworked and delivered downstream (James, 2013). Erosion and deposition processes do not occur uniformly across a drainage basin; headwaters and trunk channels may experience different sequences of erosion and fill at different rates (Leopold, 1994).

<u>1.5.1. Headcut Formation</u>

Flashy urban runoff hydrographs produce high velocities that exceed critical shear stress of stream beds and banks, initially degrading streams and then aggrading downstream (Walsh et al., 2005; Leopold, 1994). Steep changes of bed slope in headwaters or confined within the channel banks are referred to as headcuts and knick points, respectively (Bennett et al., 2000). Headcut formation and migration in vegetated spillways can be understood in 3 phases (Temple & Hanson, 1994):

1. loss of vegetation and flow concentration,
2. exceedance of critical shear stress, producing downward and downstream erosion, forming a headcut vertical face, and
3. upstream migration of headcut.

In urban drainage systems, discharge of concentrated flows at high velocities to ditches and earthen channels produce conditions analogous to phases 1 and 2, beginning headcut formation. Headcut retreat will advance upstream at a constant rate in homogenous soils (Flores-Cervantes et al., 2006). Migration in stratified soil is characterized by episodic failure and mass wasting associated with undercutting of a plunge pool of an upper, more cohesive soil layer. The rate of migration depends on a soil's resistance to cantilever mass failure and scour hole geometry (Stein & LaTray, 2002).

1.5.2. Morphology

Stream and river cross-sectional characteristics are a function of flow, the quantity and character of sediments, and the composition of materials that make up the channel bed and banks, including vegetation (Leopold, 1994). Stream channels work toward dynamic equilibrium migrating laterally as the flow exerts shear stress (and erosion) on the outside of meanders, while simultaneously depositing sediments on the inside of meanders building a point bar (Leopold, 1994). Stream channels are maintained by a characteristic discharge called the bankfull or effective discharge. The bankfull discharge is empirically determined to reoccur approximately every 1.5-years for natural streams (Leopold, 1994; Rosgen, 1994). Stream flow is comprised of groundwater and surface water; therefore, the bankfull discharge is influenced by changes in the runoff regime of a drainage basin. The runoff regime is influenced by many factors including: climate and land-use. After land-use change or urbanization, watershed peak discharge may exceed bankfull discharge more frequently; the new flow regime will cause the stream to work toward new channel characteristics that maximize energy dissipation and equalize with the new flow capacity (Leopold, 1994; Novotny, 2003; Rosgen, 1994).

1.5.3. Ecology

Vannote et al. (1980) translated the energy equilibrium theory used by geomorphologist to a biological analog, The River Continuum Concept; it is used for understanding ecologic structure and function in streams. The concept states that geomorphic processes regulate macroinvertebrate functional feeding groups via energy inputs, and organic matter transport, storage, and use. Stream systems represent a continuum of variables moving from headwaters to river mouths, requiring biological communities to adapt to energy dissipation patterns and characteristics of the physical system (Vannote et al., 1980). Headwater ephemeral and perennial streams are the most connected to the landscape, representing a maximum interface with the terrestrial environment (Vannote et al., 1980); riparian shading and detritus input reduce autotrophic production resulting in a lower gross primary production to community respiration (P/R) ratio. Moving downstream, shading decreases and autotrophic production becomes more important leading to an increase in the P/R ratio (Vannote et al., 1980). Headwater streams function predominantly to accumulate, process, and transport terrestrial material downstream. These rich systems boast the highest diversity of soluble organic matter and heterogeneous assemblages of labile and refractory dissolved components that represent a significant source of energy, locally and downstream (Vannote et al., 1980).

In medium to large rivers primary production becomes depth and turbidity limited and river morphology forces benthic macroinvertebrate communities to adopt new locomotion and feeding mechanisms. A shift in the dominance of shredders to collectors depicts the reduced importance of riparian inputs and a greater reliance on the downstream flux of energy and materials (Vannote et al., 1980). Biological systems are organized and reorganized to take advantage of energy leaks from upstream systems. New functional groups take advantage of leakage while trending toward an equilibrium between most efficient use of energy inputs and uniform rate of energy processing throughout the year. Energy gains and losses are typically due to low probability cataclysmic events and slow channel forming process (Vannote et al., 1980). However, any perturbations in the hydrologic system (e.g., land use, climate, stream characteristics) will shift the river continuum upstream or downstream, eliminating some species or allow others to enter the system depending on the location, magnitude, and type of disturbance (Poff et al., 1997; Vannote et al., 1980).

River ecosystem stability refers to a river's ability to reduce energy fluctuations during environmental variations while structure and function of the ecological community are maintained; therefore, community stability is tied to the instability of the physical system (Vannote et al., 1980). Southwood (1977) during his presidential address to the British Ecological Society discussed the implications of habitat stability on species persistence. He expressed the idea that predictable habitats are relatively permanent and exhibit relatively low favorability variance, while unpredictable habitats have greater favorability variance, and thus, longer periods of unfavorable conditions. He continued saying that a species will only persist if favorable conditions for breeding are longer than the species' generation time (Southwood, 1977). This reiterates the sensitivity of aquatic biota and their communities to terrestrial impacts and the hydrologic system, expressed by Vannote et al. (1980) and reinforces the need for mitigation of human induced change to the hydrologic cycle.

1.6. Existing Regulations

In 1972, the Clean Water Act (CWA) was established to provide a framework for protecting U.S. water resources (EPA, 2016). Under the National Pollutant Discharge Elimination System (NPDES), it is the local municipalities' responsibility to implement a plan for monitoring and permitting of point and non-point source discharges of effluent or runoff by municipalities, industrial facilities, or other entities to waters of the state (EPA, 2016; NCDENR, 2007).

NPDES Phase 1 Municipal Separated Storm Sewer System (MS4s) permitting began in 1990 and required municipalities of 100,000 or greater to develop and implement stormwater management programs to increase public knowledge of stormwater issues, detect and eliminate illicit discharges, perform land use and sewer system mapping, and analytically monitor system dynamics (EPA, 2016). Session Law 2004-163 required NPDES Phase 2 MS4s permitting in 1995; Phase 2 extended NPDES coverage to municipalities of less than 100,000. Requirements include: minimums in public education, outreach on stormwater-related issues, public involvement and engagement, detection and elimination of illicit discharges, construction site stormwater runoff controls, post-construction stormwater management in new and redevelopments, and pollution prevention (GANC, 2003).

NPDES requires adoption of minimum standards in post-construction management of stormwater for new development and redevelopment. Rules apply to new and redevelopment project areas that exceed 0.4-ha (1-ac); projects are categorized as low or high density depending on a 24% Built Upon Area (BUA) threshold. Low and high-density developments both require the use of (1) vegetated conveyances to the maximum extent possible, (2) minimum 9.1-meter (30-foot) buffers between BUA and perennial and intermittent streams, and (3) deed restrictions and protective covenants to ensure communities maintain municipally-approved plans for stormwater management. In addition to low density development requirements, high density development requires the uses of structural SCMs for treatment of the water quality event, 38 mm in coastal counties and 25 mm elsewhere in North Carolina. Crediting of SCMs is focused on the reduction of nutrients and TSS (NCDEQ, 2017).

The goal of the CWA is to protect and restore the chemical, physical, and biological integrity of the nation's waters by requiring states to monitor, analyze, and develop water quality standards for their waters (EPA, 2016). Water quality standards are comprised of a designated use, the criteria or numeric pollutant threshold, and anti-degradation policy (U. S. EPA, 2009).

Numerous agencies monitor and report the health of U.S. water resources; the Environmental Protection Agency's (EPA) *National Coastal Condition Assessment* (NCCA) is a series of probability survey reports documenting the condition of nearshore coastal, estuarine, and Great Lake waters of the United States for four indices: biological quality, water quality, sediment quality, and ecological fish tissue quality. These reports are important for understanding current conditions, trends, and allocating resources to protect the integrity and productivity of U.S. water resources.

1.7. Stormwater Mitigation

1.7.1. Hydrology

Conventional stormwater management focuses on efficient drainage design and conveyance of impervious runoff directly to receiving waters with little or no attenuation or treatment (Burns et al., 2012). The negative impacts of efficient drainage and urbanization can be mitigated by interception, detention, and infiltration of runoff volumes (Brown & Hunt, 2011; Davis, 2008; Sanders, 1986). Infiltrating SCMs, (e.g. bioretention cells, pervious pavement, suspended pavement) generally provide volume and peak flow reductions as runoff is detained in a ponding zone or moves through a media zone (Table 1-1). Rainwater Harvesters (RWHs) provide detention or retention and water for a designated used.

Table 1-1. SCM Hydrologic Treatment

Stormwater Control Measure	*Volume Reduction (%)*	*Peak Flow Reduction (%)*	*Source*
Bioretention Cell	<1-96	1-80	Li et al. (2009) Hatt et al. (2009)
Pervious Pavement	<1-100	30-92	Drake et al. (2013) Wardynski et al. (2013)
Suspended Pavement	NA	62	Page et al. (2015)
Regenerative Stormwater Conveyance	(10)-100	21-96	Koryto et al. (2017, 2018) Cizek et al. (2017, 2016)
Rainwater Harvester	0-88	0-93	Gee & Hunt (2016) Jones & Hunt (2010)

( ) Values in parenthesis represent an increase in volume

These SCMs may utilize pretreatment or a forebay to dissipate influent energy and remove coarse sediments, while surface, or temporary ponding, allows capture of a design volume (Herngren et al., 2005; Sansalone & Cristina, 2004; Tuccillo, 2006). In North Carolina, coastal county SCMs are designed to capture a 38-mm water quality event, while a 25-mm events are captured elsewhere in the state (NC DEQ, 2017). Unlined media based SCMs include two volume reduction mechanisms: (1) ET and (2) exfiltration to subsoils (Table 1-2).

Table 1-2. Observed Hydrologic Treatment Mechanisms for Bioretention Cells

Treatment Mechanisms	*Contribution to Volume Reduction (%)*	*Influenced By*	*Source*
Evapotranspiration	4-78%	Storm depth, duration, and intensity; vegetation type and planting density; and IWS	Brown & Hunt (2011) Li et al. (2009) Hatt et al. (2009) Wadzuk et al. (2015)
Exfiltration to Subsoils	0-96%	Storm depth, duration, and intensity; type of subsoils; and IWS	Archer et al. (2002) Brown & Hunt (2011) Li et al. (2009)

SCM media generally consist of sand, soil fines, and organic matter (OM); media may be homogenous, well-mixed, or stratified with the least permeable layer controlling the infiltration rate (Hsieh & Davis, 2005a, 2005b). Rapid infiltration is important to limit bypass; however, poor nutrient treatment is associated with high (> 25 cm/hr) saturated hydraulic conductivity (k_{sat}) values (Davis et al., 2010; Hunt et al., 2012; Kim et al., 2003). Media infiltration and saturated hydraulic conductivity rates can be adjusted with an increase or decrease in the fraction of fines (silt and clay) (Hsieh & Davis, 2005a). An Internal Water Storage (IWS) zone, created using an elevated outlet, can improve volume and peak flow reductions by increasing temporary storage in the media profile (Brown et al., 2011). Overflow (i.e., bypass) in Bioretention Cells (BRCs) is linked to the hydraulic conductivity of underlying soil, depth to IWS, and the surface infiltration rate (Brown & Hunt, 2011; Hunt et al., 2006). Surface mulching with a hardwood mulch can prevent clogging and preserve surface infiltration rates (Hsieh & Davis, 2005b, 2005a). Wardynski & Hunt (2012) found 53% of BRCs were moderately to severely undersized; under-sizing will reduce hydrologic and water quality performance by increasing the frequency of overflow (Brown & Hunt, 2011).

Loading ratios (watershed area : SCM area) for BRCs typically range between 14:1 – 20:1 but can be larger (e.g. 45:1) (Davis, 2008; Hunt et al., 2012); lower loading rates can yield volume reductions greater than 50% depending on underlying soils (Davis, 2008; Dietz & Clausen, 2005; Hunt et al., 2006). BRC is the least expensive SCM for small watersheds (< 1.6 ha or 4 ac) in residential and commercial areas in the NC piedmont and coastal plains (Wossink & Hunt, 2003).

Preservation or restoration of watershed vegetation and urban forest canopy will increase canopy cover, reduce runoff via increased initial abstraction, and provide rainfall storage within tree canopies delaying time to peak discharge (Deak et al., 2014; Inkiläinen, et al., 2013; Xiao & McPherson, 2011); however, compacted urban soils limit trees' ability to grow, access water, and thrive and often lead to premature death (Smiley et al., 2006).Vegetation and plantings in SCMs can improve infiltration rates by creating macropores around plant roots and improve volume reductions via ET (Li et al., 2009; Wardynski & Hunt, 2012).

Wet and dry detention basins and stormwater wetlands provide hydrologic mitigation by capturing and storing runoff in large surface depressions and releasing effluent over a longer period (Herb et al., 2009; Jones & Hunt, 2010; Lieb & Carline, 2000). RWHs have been studied for their water quality and hydrology benefits (Doan & Davis, 2017; Gee & Hunt, 2016; Jones & Hunt, 2010; Pelak & Porporato, 2016). Studies of RWHs have shown their potential to substantially reduce runoff volumes and peak flows while simultaneously reducing potable water demands by greater than 80% in the southeast region (Steffen et al., 2013); however, lack of water usage (underutilization) leads to diminished performance and excessive overflow when storage capacity is not available for runoff volumes (Jones & Hunt, 2010).

1.7.2. Nitrogen

Particulate nitrogen in the form of Total Kjeldahl Nitrogen (TKN), Ammonia (NH_3), and Organic Nitrogen (ON) are readily adsorbed to soil particles and OM making them available for inter-event biological treatment or removed via settling and surface filtration (Davis et al., 2010); however, the majority of nitrogen (50-70%) is dissolved and moves easily through the soil profile (Novotny, 2003; Vaze & Chiew, 2004). Soluble nitrogen, Nitrite (NO_2^-) and Ammonium (NH_4^+), must undergo oxidation and nitrification to form Nitrate ($NO3^-$) before anaerobic reduction and denitrification are possible. In aerobic environments, these molecules oxidize readily. Total Ammoniacal Nitrogen (TAN) and Nitrite-Nitrate ($NO_{2,3}$-N) are plant available nutrients readily available to autotrophic organisms and contribute to algal blooms in rivers, lakes, and streams.

Nitrogen removal is accomplished to varying degrees via all 5 SCM nutrient treatment mechanisms (Table 1-3, Table 1-4). IWS zones create saturated conditions within the soil profile using an elevated outlet (Brown et al., 2011). Inundation of the soil profile depletes dissolve oxygen and allows denitrification to occur; NO_3^- reduction produces dinitrogen and nitrous oxide gas (Birgand et al., 2007; Novotny, 2003). Localized areas of anoxic conditions and denitrification are thought to persist even in well-drained SCMs (Hunt et al., 2006). Denitrification and Dissimilatory Nitrate Reduction to Ammonia (DNRA) are competing processes under anoxic conditions depending on available organic carbon and nitrate concentration; denitrification is favored when organic carbon is limited while DNRA is favored when nitrate is limited (Davis et al., 2010; Tiedje et al., 1982). Microbial mineralization of organic nitrogen is dependent on the Carbon:Nitrogen (C:N) ratio of growth substrate and the microbes need for nitrogen; at C:N ratios less than 25:1 decomposers typically export TAN (Hodge et al., 2000). Denitrification results in nitrogen loss as dinitrogen and nitrous oxide gases (Tiedje et al., 1983); DNRA conserves nitrogen with the production of TAN (Tiedje et al., 1983).

Table 1-3. SCM Nutrient Treatment Mechanisms

Treatment Mechanisms	*Influenced By*	*Source*
(1) Sedimentation	Total volume and geometry of surface ponding or pool and particle size	Davis et al. (2010) Miguntanna et al. (2013) Sansalone & Cristina (2004) Tuccillo (2006)
(2) Filtration	Hydraulic conductivity of, and depth to, least permeable soil and the surface infiltration rate	Dietz & Clausen (2005) Brown et al. (2011)
(3) Adsorption	Fraction of media fines (silt and clays); Cation Exchange Capacity; Media P-Index	Hunt et al. (2006) Kim et al. (2003) Hsieh & Davis (2005a) Hardy et al. (2009)
(4) Denitrification	Infiltration rate, saturated hydraulic conductivity, and residence time; nitrate concentration; media OM content and C:N ratio; and oxygen concentration	Birgand et al. (2007) Novotny (2003) Dietz & Clausen (2005) Kim et al. (2003) Tiedje et al. (1983)
(5) Assimilation by Vegetation	Plant species and planting density	Bratieres et al. (2007) Read et al. (2008) Turk et al. (2017)

Table 1-4. SCM TN Treatment

Stormwater Control Measure	*Total Nitrogen Reduction* (%)*	*Source*
Bioretention Cell	(79)-80	Davis et al. (2006) Hunt et al. (2006) Dietz & Clausen (2005) Brown & Hunt (2011) Hatt et al. (2009)
Pervious Pavement	73	Smolek & Hunt (2015)
Suspended Pavement	35-42	Page et al. (2015)
Regenerative Stormwater Conveyance	0-76	Cizek et al. (2016) Koryto et al. (2018, 2017) Palmer et al. (2014) Browning (2008)
Level Spreader-Vegetative Filter Strip	(17)-32	Winston et al. (2011)

* Reporting varies; Removal Efficiencies or Event Load Reductions
( ) Values in parenthesis represent a pollutant increase

1.7.3. Phosphorus

Phosphorous treatment relies on treatment mechanisms 1-3 and 5 (Table 1-3). The P-Index is a reliable predictor of medias' ability to bind or leach phosphorus (Hunt et al., 2006). The P-Index is a measure of the amount, or saturation, of phosphorus in the soil (Hardy et al., 2009). The P-index of a soil is rated high, medium, or low based on its saturation level of phosphorus. Low P-index soils are desirable so they have more binding sites available to adsorb phosphorus (Hunt et al., 2006). The dissolved fraction of Total Phosphorus (TP), Orthophosphate (Ortho-P), is removed via adsorption processes. Ortho-P adsorption to soil and OM particles is controlled by amorphous iron and aluminum oxides; increasing the fraction of media fines can increase Ortho-P binding sites (Davis et al., 2010; Hsieh & Davis, 2005a). Increased media flow will increase sorption of Ortho-P provided sufficient binding sites exist; sorbed pollutants are available for subsequent inter-event biological treatment or assimilation (Davis et al., 2010). SCMs using media with a medium or high P-index are likely to leach phosphorus (Hunt et al., 2006; Passeport et al., 2009). Vaze & Chiew (2004) reported approximately two-thirds of TP is particle bound phosphorous (PBP) in urban runoff; the remaining one-third (dissolved phosphorus) can be sorbed to media iron and aluminum oxides and OM. SCMs reliably remove (> 75%) TP provided effective sedimentation for PBP and low P-indices for adsorption are in place (Table 1-5).

Table 1-5. SCM TP Treatment

Stormwater Control Measure	*Total Phosphorous Reduction* (%)*	*Source*
Bioretention Cell	(957)-83	Davis et al. (2006) Hunt et al. (2006) Dietz & Clausen (2005) Brown & Hunt (2011) Hatt et al. (2009)
Pervious Pavement	95	Smolek & Hunt (2015)
Suspended Pavement	72	Page et al. (2015)
Regenerative Stormwater Conveyance	7-77	Cizek et al. (2016) Koryto et al. (2018, 2017)
Level Spreader-Vegetative Filter Strip	(27)-40	Winston et al. (2011)

* Reporting varies; Removal Efficiencies or Event Load Reductions
( ) Values in parenthesis represent a pollutant increase

1.7.4. Total Suspended Solids

TSS removal is accomplished via treatment mechanisms 1 and 2 (Table 1-3). Effective sedimentation requires reduction of flow velocities sufficient that particles settle via gravity (Stokes' Law); large, dense particles are readily settled compared to smaller, low density particles (Davis et al., 2010). Sufficient storage is necessary to achieve conditions suitable for sedimentation and to prevent pollutant resuspension (Davis et al., 2010; Miguntanna et al., 2013). Pretreatment or a forebay is used to capture large particles that are readily settled from stormwater flows (Herngren et al., 2005; Sansalone & Cristina, 2004; Tuccillo, 2006). Surface filtration within soil media removes fine particles from urban runoff (Davis, 2007; Hsieh & Davis, 2005; Sansalone & Buchberger, 1995).

Media-based SCMs are effective (> 85%) at removal of TSS (Table 1-6); however, media clogging and wash-out can inhibit TSS removal. Hsieh & Davis (2005a, 2005b) found that stratified media resulted in the removal of different pollutants, with the least permeable layer controlling the infiltration rate; TSS was captured by the mulch layer and oils, grease, and lead were retained in the soil media (Hsieh & Davis, 2005a, 2005b). Media layers separated by filter fabric are at a high risk for clogging (Siriwardene et al., 2007). Siriwardene et al. (2007) reported clogging occurred more rapidly when there were fluctuations in water levels; PM less than 6 μm was the main cause of clogging. Often hardwood mulch is used to prevent clogging of the surface layer and add sorption sites (Hsieh & Davis, 2005b, 2005a; Maniquiz-Redillas & Kim, 2016). However, mulch with a high d60/d10 ratio (>15.4) can disperse and clog media reducing infiltration rates (Hsieh & Davis, 2005a). Maniquiz-Redillas & Kim (2016) found sediments mostly attached to wood chips versus sand, gravel, or geotextiles. Maniquiz-Redillas & Kim

(2016) determined the removal of heavy metals was influenced greatly by the concentration of TSS and the rate of flow.

Table 1-6. SCM TSS Treatment

Stormwater Control Measure	*Total Suspended Solids Reduction* (%)*	*Source*
Bioretention Cell	(170)-97	Hunt et al. (2006) Brown & Hunt (2011) Hatt et al. (2009) Hsieh & Davis (2005b)
Pervious Pavement	98	Smolek & Hunt (2015)
Suspended Pavement	> 86	Page et al. (2015)
Regenerative Stormwater Conveyance	10-92	Cizek et al. (2016) Koryto et al. (2018, 2017) Palmer et al. (2014) Browning (2008)
Level Spreader-Vegetative Filter Strip	51-72	Winston et al. (2011)

* Reporting varies; Removal Efficiencies or Event Load Reductions
( ) Values in parenthesis represent a pollutant increase

1.7.5. Thermal Load

The effects of urbanization on water quality are well documented; however, less research documents the impact of stormwater runoff on the thermal regime of streams (Table 1-7). Wardynski et al. (2014) completed an assessment of 10 temperature-based water quality standards to evaluate metrics for evaluating the thermal performance of SCMs with the goal to determine the best metric or combination of metrics to evaluate SCM temperature mitigation for the protection of cold-water habitats. Performance metrics included:

1. event mean temperature reduction within an SCM,
2. instantaneous maximum inlet and outlet SCM temperature,
3. biological temperature thresholds as a method of SCM evaluation for both short and long-term exposure to thermally enriched waters,
4. duration exceeding local cold-water reference stream temperatures (temperature-duration plots),

5. Uniform Continuous Above Threshold (UCAT) analysis,
6. probabilistic determination of threshold temperatures from local monitoring of cold-water streams,
7. groundwater temperature as an SCM effluent temperature metric,
8. thermal load reduction within an SCM,
9. application of Low Impact Development (LID) principles as surrogate for temperature-based metrics, and
10. direct monitoring of thermal impacts of urban development on in-stream temperature.

Table 1-7. SCM Thermal Mitigation

Stormwater Control Measure	*Effect on Runoff Temperatures*	*Source*
Bioretention Cell	Some ability to buffer maximum and, to a lesser extent, median runoff temperatures	Jones & Hunt (2009)
*Pervious Pavement**	77-100	Wardynski et al. (2013)
Wet Pond	Increased temperature variance compared to reference reach; source of thermal load; thermally stratified; 1.2ºC increase in effluent temperature; all temperatures higher than 21ºC	Lieb & Carline (2000) Jones & Hunt (2010) Herb et al. (2009)
Wetland	Source of thermal load; wetland vegetation benefited cooler bottom temperatures; bottom temperatures only, lower than 21ºC	Jones & Hunt (2010)
Regenerative Stormwater Conveyance	Minimal buffering of runoff temperatures	Koryto et al. (2017)
Level Spreader-Vegetative Filter Strip	Can buffer maximum runoff temperatures; function of VFS length and shading	Winston et al. (2011)
Under-Ground Detention	Median 1.6ºC temperature reduction in summer months; did not meet MD standards	Natarajan & Davis (2010)

* Thermal Load Reduction (%)

The UCAT method evaluated by Wardynski et al. (2014) is an adaption of other methods for identification of habitat events via magnitude, frequency, and duration from time-series data (Capra et al., 1995; Castelli et al., 2012; Parasiewicz, 2008). Castelli et al., (2012) applied UCAT analysis to 33-years of thermal time-series data to identification the frequency of rare habitat thresholds. SCM evaluation methods vary in data required and ease of analysis, some require long-term species supporting reference reach data while others only use SCM effluent temperatures.

Jones & Hunt (2009) found bioretention systems can reduce the initial spike of heated runoff but media temperatures were unable to adapt to cooler runoff later in the event (Table 1-7); thermal equilibrium between runoff and media was reached within 60 cm of the media surface limiting the soils ability to cool runoff after the initial thermal spike and pavements had cooled (Jones & Hunt, 2009). Furthermore, BRC soils were not cooled by runoff later during storm events (Jones & Hunt, 2009). Some BRCs were unable to reduce influent and effluent temperatures (Jones & Hunt, 2009); all runoff temperatures were significantly higher than the 21^{o}C trout threshold (Jones & Hunt, 2009).

Jones et al. (2012) began initial exploration of the effect of parking lot composition and layout on runoff temperature. Runoff temperatures experience seasonal fluctuation with the hottest runoff events happening during peak summer months, June-August, during afternoon storms (Jones et al., 2012). Runoff temperatures were significantly warmer than the 21^{o}C threshold for North Carolina brook trout (Jones et al., 2012). Tree canopy shading reduced parking lot runoff temperatures, although not significantly; the effect of asphalt composition (e.g., chip-seal, a lighter colored asphalt, and regular hot-mix) did not have a significant effect on runoff temperatures (Jones et al., 2012). Median runoff temperatures from the chip-seal and hot-mix asphalts were 20.6^{o}C and 22.0^{o}C, respectively (Jones et al., 2012). Jones et al. (2012) described the hot-mix catchment as “aged and light-in-color with partial shading” possibly resulting in reduced runoff temperatures. Jones et al. (2012) also found a significant reduction in maximum runoff temperature when surface collection was compared to sub-surface conveyance temperatures (Jones et al., 2012).

Lieb & Carline (2000) conduced a 42-day bioassay to understand if metal concentrations from detention pond effluent were causing benthic macroinvertebrate impairment observed on Thompson Run, Pennsylvania. The bioassay included exposure of caged *Gammarus minus* to in-stream detention pond effluent. Data were collected on stream temperatures and metal concentrations in a reference reach, at the pond outlet, and upstream (Lieb & Carline, 2000). Pond effluent exhibited a higher temperature variation (9.8^{o}C—10.8^{o}C) than the reference reach (10.1^{o}C—10.4^{o}C) (Table 1-7)(Lieb & Carline, 2000). Hourly temperature change during stormflow at the reference reach did not exceed 0.2^{o}C while pond effluent hourly temperature change reached 6.6^{o}C; alternatively, reference reach base flow hourly temperature change never exceeded 1^{o}C (Lieb & Carline, 2000).

Walsh et al. (2005) discussed the urban stream syndrome and highlights reduced base flow and increased in-stream diel temperature variation as an effect of urbanization. Morphologic changes such as channel widening from more frequent high flow events can contribute to the loss of riparian forest increasing direct solar radiation and stream heating (Caissie, 2006; Vannote et al., 1980; Walsh et al., 2005).

1.8. Stream Restoration

Often restoration efforts are undertaken in watersheds with highly altered hydrologic cycles resulting from impoundments and land use changes (Poff et al., 1997). The natural flow regime (i.e., quantity, timing, and variability of flows) regulates ecological processes via (1) magnitude, (2) frequency, (3) duration, (4) timing, and (5) rate of change of hydrologic conditions (Poff et al., 1997). Full restoration of natural flow is often impossible, but recognition of the variability of stream flow, and incorporation of the five components of natural flow into management, is an advancement over conventional management (Poff et al., 1997). Specific restoration goals should be developed within the context of existing conditions and watershed disturbances, and likely include establishment of self-sustaining, rather than, pristine conditions (FISRWG, 1998).

Self-sustaining conditions (i.e., dynamic equilibrium) are developed through an understanding of watershed characteristics and the hydrologic and hydraulic processes therein, and the geomorphic processes that occur along the stream corridor (FISRWG, 1998). The conceptual, dominant (or channel-forming) discharge, is responsible for maintenance and development of the stable geomorphic channel; dominant discharge may be used to design channel geometry (e.g., width, depth, shape) (FISRWG, 1998; Leopold, 1994). Using long-term water and sediment data, the dominate discharge can be estimated; the calculated measure is called effective discharge (FISRWG, 1998). A field estimation of effective and dominate discharge, the bankfull discharge, is the discharge above which water leaves the channel and enters the floodplain (FISRWG, 1998; Leopold, 1994).

Channel geometry and discharge will influence stream competence, tractive or shear stress, and stream power (FISRWG, 1998). Excess or limited shear will produce channel degradation or aggradation, over time, respectively. A stable channel will transport its sediment load while maintaining bed and banks (FISRWG, 1998). Valley slope, stream planform (i.e., channel length, sinuosity, meander wavelength, belt width, radius of curvature), bed features (i.e., riffles, pools, steps), bed substrate, and channel geometry are all parameters used to restore a stable stream reach (Doll et al., 2003; FISRWG, 1998; Leopold, 1994).

The most common stream restoration method uses Rosgen's (1994) classification of streams and rivers to provide consistent and reproducible descriptions of the most probable channel morphology (Doll et al., 2003). Rosgen (1994) includes four levels of classification: Geomorphic Characterization, Morphological Description, Stream Conditions, and Validation to move from broad qualitative analysis to detailed quantitative assessment of a stream reach (Figure 1-3).

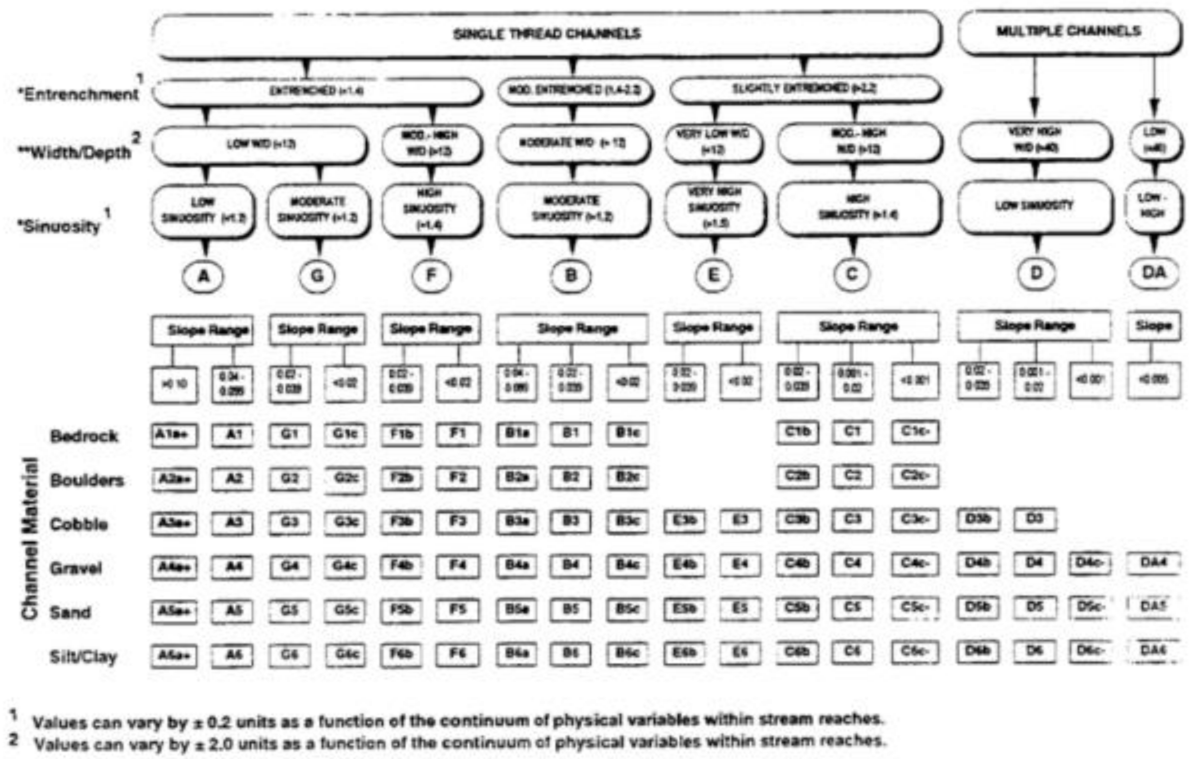


Figure 1-3. Natural River Classification Key (Rosgen, 1994)

Understanding the probable natural condition and the current conditions provides insight to develop a plan for stabilization, restoration, and long-term stability of a stream reach (Doll et al., 2003; Leopold, 1994; Rosgen, 1994). Rosgen (1994) methods are adapted for urban areas. Boulder-weirs, step-pool sequences, large woody debris additions, hydrologic reconnection of the floodplain, increased sinuosity, and live staking or armoring of unstable banks are common urban restoration practices (Booth, 2005; Craig et al., 2008; Kaushal et al., 2008; Schueler & Brown, 2004).

In-stream erosion can account for large portions of sediment yield (approximately 67% at San Diego Creek, California) (Trimble, 1997). Small streams, less than 3rd order, where low and moderate flows export large TN loads, have the greatest potential for TN reductions (Craig et al., 2008). Stream restoration can increase denitrification rates, improve stream habitat, and reduce sediment export (Table 1-8) (Booth, 2005; Craig et al., 2008; Filoso & Palmer, 2011; Kaushal et al., 2008; Trimble, 1997).

Table 1-8. Stream Restoration Results

Method	*Effect of Restoration*	*Estimated Removal Rates*	*Source*
Increase In-stream Carbon, Contact Between Water and Benthos, and Hydrologic to Floodplains	Increased hydraulic and hydrologic residence time, and denitrification potential	2.2-78 mg-N/m^2-d	Craig et al. (2008)
Geomorphic Restoration	Increased hydraulic residence and hydrologic connectivity	(5.1) – 5.9 g-N/ m^2-yr ~0 – >200 μg-N/kg*-yr	Filoso & Palmer (2011) Kaushal et al. (2008)
Short- & Long-Term Enhancements for Urban Streams	Improvement of habitat elements vs. restoration of habitat forming processes	NA	Booth (2005)

* kg sediment

1.9. Regenerative Stormwater Conveyance

Regenerative Stormwater Conveyance (RSC) is a new, innovative, approach to managing stormwater and protecting streams. Patented in 2009 as a system and method, subsequent design recommendations were made by Anne Arundel County, Maryland, and followed by the West Virginia Department of Environmental Protection (WVDEP) (Underwood, 2002; Flores et al., 2012; WVDEP, 2012). RSCs integrate urban stormwater management and stream restoration engineering techniques to convey and treat non-point source pollutants (Flores et al., 2012) (Figure 1-4). A main objective of RSC installation is to stabilize eroded gullies and headcuts by filling gullies and reducing local bed slope (Bennett et al., 2000; Flores et al., 2012). A combination of Priority 1 (P1) and Priority 4 (P4) stream restoration techniques (Figure 1-5) are used to fill and stabilize headcuts with RSC systems (Doll et al., 2003). The streambed is elevated, similar to P1 restorations, by filling the existing headcut with coarse sand blended with OM; channel stabilization is accomplished with boulder-weir and riffle-cascade sequences, similar to P4 restorations, to provide grade control and support scour pools for energy dissipation during high flow events (Doll et al., 2003; Flores et al., 2012).

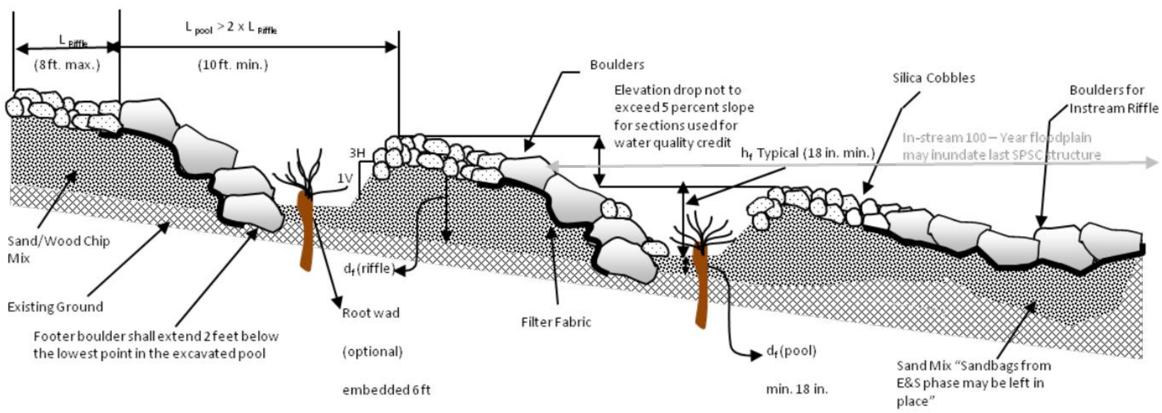


Figure 1-4. Typical RSC Cross-Section (Flores et al., 2012)

Pollutant treatment mechanisms include (1) settling in surface storage (ponded water), (2) filtration and (3) sorption processes in the media layer, (4) assimilation by vegetation, and (5) microbiological treatment; volume reduction mechanisms include (1) ET and (2) infiltration to subsoils (Cizek et al., 2016; Cizek et al., 2017; Davis et al., 2010; Flores et al., 2012; Koryto et al., 2018; Koryto et al., 2017).

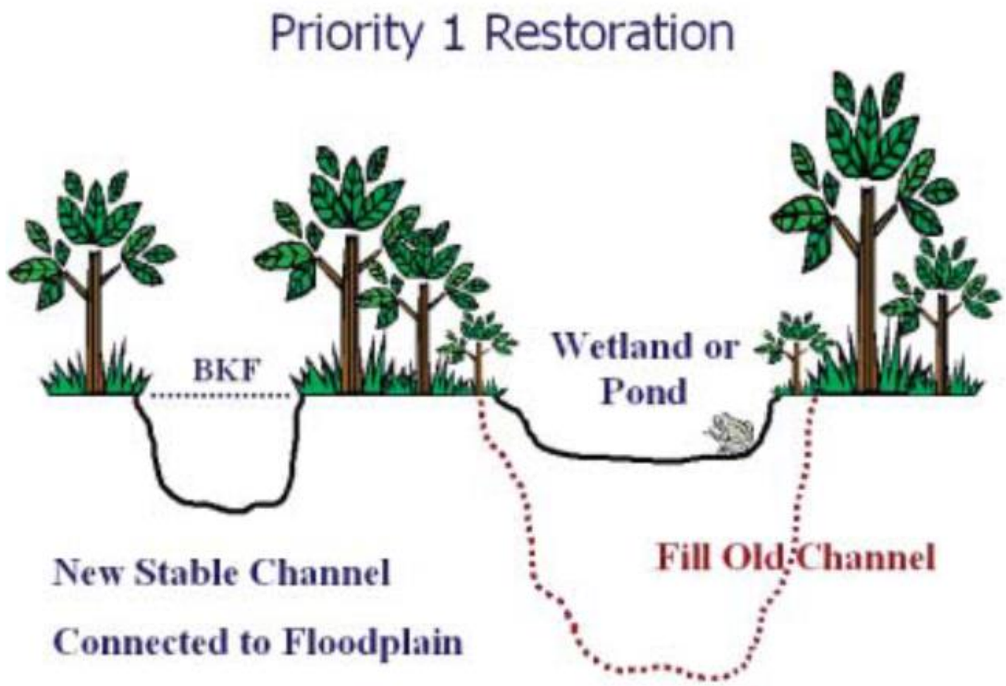


Figure 1-5. P1 Stream Restoration Cross-Section; (Doll et al., 2003)

Sand filtration and media treatment of stormwater runoff are widely adopted practices; design standards and performance expectations exist to support the permitting and regulatory process (Barrett, 2003; Davis et al., 2003; Davis, 2008; Gallo, Moore, & Wywrot, 2012; Hunt et al., 2012; Wardynski & Hunt, 2012). Design recommendations for RSCs include safe conveyance of 100-year, 24-hour storm flows via floodplain connection and energy dissipation by in-stream geomorphic structures (Flores et al., 2012; WVDEP, 2012). Adoption of this practice continues despite the lack of scientific consensus around design characteristics or expected treatment results.

1.10. Conclusions

Land use change and urbanization result in substantial negative impacts to downstream waters, ecology, and habitats (Leopold, 1994; Novotny, 2003; Poff et al., 1997; Rosgen, 1994; Vannote et al., 1980). Numerous pollutants and processes contribute to downstream degradation; however, these phenomena are the focus of much research and effort to develop effective mitigation strategies (Davis et al., 2003; Davis et al., 2006; Hunt et al., 2012; Jones et al., 2012; Ladson et al., 2006). This thesis will (1) analyze three SCMs (RWH, BRC, and LSVFS) for their ability to mitigate thermally-enriched runoff and provide recommendations for the protection of cold-water streams in the North Carolina Mountains (Chapter 2); (2) analyze field monitoring data from an undersized RSC constructed in-series (downstream of) a dry pond in Morrisville, North Carolina (Chapter 3); and (3) conduct an extensive review and analysis of RSC research, from which, design guidance and recommendations will be synthesized (Chapter 4).

1.11. References

Archer, N. A. L., Quinton, J. N., & Hess, T. M. (2002). Below-ground relationships of soil texture, roots and hydraulic conductivity in two-phase mosaic vegetation in South-east Spain. *Journal of Arid Environments*, *52*(4), 535–553. https://doi.org/10.1006/jare.2002.1011

Armour, C. L. (1991). Guidance for evaluating and recommending temperature regimes to protect fish instream flow. *U.S. Fish and Wildlife Service, Biological Report*, *90*(22), 1–13.

Barrett, M. E. (2003). Performance, Cost, and Maintenance Requirements of Austin Sand Filters. *Journal of Water Resources Planning and Management*, *129*(3), 234–242. https://doi.org/10.1061/(ASCE)0733-9496(2003)129:3(234)

Bennett, S. J., Alonso, C. V, Prasad, S. N., & Romkens, M. J. (2000). Experiments on headcut growth in concentrated flows typical of upland areas. *Water Resources Research*, *36*(7), 1911–1922.

Beschta, R., Bilby, R., Brown, G., Holtby, L., & Hofstra, T. (1987). Stream temperature and aquatic habitat: fisheries and forestry interactions. *Streamside Management: Forestry and Fishery Interactions*, 191–232.

Birgand, F., Skaggs, R. W., Chescheir, G. M., & Gilliam, J. W. (2007). Nitrogen Removal in Streams of Agricultural Catchments—A Literature Review. *Critical Reviews in Environmental Science and Technology*, *37*(5), 381–487. https://doi.org/10.1080/10643380600966426

Booth, D. B. (2005). Challenges and prospects for restoring urban streams: a perspective from the Pacific Northwest of North America. *Journal of the North American Benthological Society*, *24*(3), 724. https://doi.org/10.1899/0887-

3593(2005)024\[0724:CAPFRU\]2.0.CO;2

Bratieres, K., Fletcher, T. D., Deletic, A., & Zinger, Y. (2008). Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study. *Water Research*, *42*(14), 3930–3940. https://doi.org/10.1016/j.watres.2008.06.009

Brown, R. A., & Hunt, W. F. (2011). Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads. *Journal of Environmental Engineering*, *137*(11), 1082–1091. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000437

Browning, M. O. (2008). The Efficacy of Urban Stream Restorations to Improve Water Quality Across a Spectrum of Design Approaches. *Thesis Submitted to George Mason University*, 118.

Burns, M. J., Fletcher, T. D., Walsh, C. J., Ladson, A. R., & Hatt, B. E. (2012). Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform. *Landscape and Urban Planning*, *105*(3), 230–240. https://doi.org/10.1016/j.landurbplan.2011.12.012

Caissie, D. (2006). The thermal regime of rivers: A review. *Freshwater Biology*, *51*(8), 1389–1406. https://doi.org/10.1111/j.1365-2427.2006.01597.x

Capra, H., Breil, P., & Souchon, Y. (1995). A new tool to interpret magnitude and duration of fish habitat variations. *Regulated Rivers: Research & Management*, *10*(2–4), 281–289. https://doi.org/10.1002/rrr.3450100221

Castelli, E., Parasiewicz, P., & Rogers, J. N. (2012). Use of Frequency and Duration Analysis for the Determination of Thermal Habitat Thresholds: Application for the Conservation of Alasmidonta heterodon in the Delaware River. *Journal of Environmental Engineering*, *138*(August), 886–892. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000520.

Chattahoochee Riverkeeper. (2015). Tri-State Water Conflict. Retrieved from https://chattahoochee.org/our-work/tri-state-water-conflict/

Cizek, A. R., Hunt, W. F., Winston, R. J., & Lauffer, M. S. (2017). Hydrologic Performance of Regenerative Stormwater Conveyance in the North Carolina Coastal Plain. *Journal of Environmental Engineering*, *143*(9), 05017003. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001198

Cizek, A. R., Hunt, W. F., Winston, R. J., Waickowski, S., Narayanaswamy, K., & Lauffer, M. S. (2016). Water Quality and Hydrologic Performance of a Regenerative Stormwater Conveyance (RSC) in the Piedmont of North Carolina. *Journal of Environmental Engineering*, *Submitted*(8), 1–11. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001344

Constantz, J., Stonestrom, D., Stewart, A. E., Niswonger, R., & Smith, T. R. (2001). Analysis of streambed temperatures in ephemeral channels to determine streamflow frequency and duration. *Water Resources Research*, *37*(2), 317–328. https://doi.org/10.1029/2000WR900271

Craig, L. S., Palmer, M. A., Richardson, D. C., Filoso, S., Bernhardt, E. S., Bledsoe, B. P., … Wilcock, P. R. (2008). Stream restoration strategies for reducing river nitrogen loads. *Frontiers in Ecology and the Environment*, *6*(10), 529–538. https://doi.org/10.1890/070080

Davis, & , Robert G. Traver, Hurt, W. F. (2010). Improving Urban Stormwater Quality: Applying Fundamental Principles. *Journal of Contemporary Water Research & …*, *146*(1), 2–10. https://doi.org/10.1111/j.1936-704X.2010.00387.x

Davis, A. P. (2007). Field Performance of Bioretention: Water Quality. *Environmental Engineering Science*, *24*(8), 1048–1064. https://doi.org/10.1089/ees.2006.0190

Davis, A. P. (2008). Field Performance of Bioretention: Hydrology Impacts. *Journal of*

Hydrologic Engineering, *13*(2), 90–95. https://doi.org/10.1061/(ASCE)1084-0699(2008)13:2(90)

Davis, A. P., Shokouhian, M., & Ni, S. (2001). Loading estimates of lead, copper, cadmium, and zinc in urban runoff from specific sources. *Chemosphere*, *44*(5), 997–1009. https://doi.org/10.1016/S0045-6535(00)00561-0

Davis, A. P., Shokouhian, M., Sharma, H., & Minami, C. (2006). Water Quality Improvement through Bioretention Media: Nitrogen and Phosphorus Removal. *Water Environment Research*, *78*(3), 284–293. https://doi.org/10.2175/106143005X94376

Davis, A. P., Shokouhian, M., Sharma, H., Minami, C., & Winogradoff, D. (2003). Water quality improvement through bioretention: lead, copper, and zinc removal. *Water Environment Research : A Research Publication of the Water Environment Federation*, *75*(1), 73–82. https://doi.org/10.2175/106143003X140854

Deak Sjöman, J., & Gill, S. E. (2014). Residential runoff – The role of spatial density and surface cover, with a case study in the Höjeå river catchment, southern Sweden. *Urban Forestry & Urban Greening*, *13*, 304–314. https://doi.org/10.1016/j.ufug.2013.10.007

Dietz, M. E., & Clausen, J. C. (2005). A field evaluation of rain garden flow and pollutant treatment. *Water, Air, and Soil Pollution*, *167*(1–4), 123–138. https://doi.org/10.1007/s11270-005-8266-8

Doan, L. N., & Davis, A. P. (2017). Bioretention–Cistern–Irrigation Treatment Train to Minimize Stormwater Runoff. *J. Sustainable Water Built Environ.*, *3*(2), 1–10. https://doi.org/10.1061/

Doll, B. a., Grabow, G. L., Hall, K. R., Halley, J., Harman, W. A., Jennings, G. D., & Dani E. Wise. (2003). Stream Restoration: A Natural Channel Design Handbook, 128 pp.

https://doi.org/(11/19/2014)

Drake, J. A. P., Bradford, A., & Marsalek, J. (2013). Review of environmental performance of permeable pavement systems: State of the knowledge. *Water Quality Research Journal of Canada*, *48*(3), 203–222. https://doi.org/10.2166/wqrjc.2013.055

E.P.A., U. S. (1983). *Results of the Nationwide Urban Runoff Program, Vol. 1, Final Repot*. Washington, DC.

Ellis, B. J. (1986). Pollutional aspects of urban runoff, in Urban Runoff Pollution (pp. 1–38). New York: Springer-Verlag.

EPA. (2016). Summary of the Clean Water Act. Retrieved October 14, 2016, from https://www.epa.gov/laws-regulations/summary-clean-water-act

Filoso, S., & Palmer, M. A. (2011). Assessing stream restoration effectiveness at reducing nitrogen export to downstream waters. *Ecological Applications*, *21*(6), 1989–2006. https://doi.org/10.1890/10-0854.1

FISRWG. (1998). *Stream Corridor Restoration: Principles, Processes, and Practices*. https://doi.org/-0-934213-59-3

Fletcher, T. D., Vietz, G., & Walsh, C. J. (2014). Protection of stream ecosystems from urban stormwater runoff: The multiple benefits of an ecohydrological approach. *Progress in Physical Geography*, *38*(5), 543–555. https://doi.org/10.1177/0309133314537671

Flores-Cervantes, J. H., Istanbulluoglu, E., & Bras, R. L. (2006). Development of gullies on the landscape: A model of headcut retreat resulting from plunge pool erosion. *Journal of Geophysical Research: Earth Surface*, *111*(1), 1–14. https://doi.org/10.1029/2004JF000226

Flores, H., Mcmonigle, D., & Underwood, K. (2012). *Design Guidelines for Regenerative Step Pool Storm Conveyance*.

Freer-Smith, P. H., Holloway, S., & Goodman, A. (1997). The uptake of particulates by an urban woodland: Site description and particulate composition. *Environmental Pollution*, *95*(1), 27–35. https://doi.org/10.1016/S0269-7491(96)00119-4

Gallo, C., Moore, A., & Wywrot, J. (2012). Comparing the adaptability of infiltration based BMPs to various U.S. regions. *Landscape and Urban Planning*, *106*(4), 326–335. https://doi.org/10.1016/j.landurbplan.2012.04.004

Gee, K. D., & Hunt, W. F. (2016). Enhancing Stormwater Management Benefits of Rainwater Harvesting via Innovative Technologies. *Journal of Environmental Engineering*, *142*(8), 04016039. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001108

General Assembly of North Carolina. (2003). NC Session Law 2004-163 Senate Bill 1210. Retrieved from https://www.ncleg.net/EnactedLegislation/SessionLaws/HTML/2003-2004/SL2004-163.html

Hallberg, M., Renman, G., & Lundbom, T. (2007). Seasonal variations of ten metals in highway runoff and their partition between dissolved and particulate matter. *Water, Air, and Soil Pollution*, *181*(1–4), 183–191. https://doi.org/10.1007/s11270-006-9289-5

Hardy, D. H., Tucker, M. R., & Stokes, C. E. (2009). Crop Fertilization Based on North Carolina prepared by, (1).

Hatt, B. E., Fletcher, T. D., & Deletic, A. (2009). Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale. *Journal of Hydrology*, *365*, 310–321. https://doi.org/10.1016/j.jhydrol.2008.12.001

Henderson, C., Greenway, M., & Phillips, I. (2007). Removal of dissolved nitrogen, phosphorous and carbon from stormwater by biofiltration mesocosms. *Water Science and Technology*, *55*(4), 183–191. https://doi.org/10.2166/wst.2007.108

Herb, W. R., Mohseni, O., & Stefan, H. G. (2009). Simulation of temperature mitigation by a stormwater detention pond. *Journal of the American Water Resources Association*, *45*(5), 1164–1178. https://doi.org/10.1111/j.1752-1688.2009.00354.x

Herngren, L., Goonetilleke, A., & Ayoko, G. A. (2005). Understanding heavy metal and suspended solids relationships in urban stormwater using simulated rainfall. *Journal of Environmental Management*, *76*(2), 149–158. https://doi.org/10.1016/j.jenvman.2005.01.013

Hodge, A., Robinson, D., & Fitter, A. (2000). Are microorganisms more effective than plants at competing for nitrogen? *Trends in Plant Science*, *5*(7), 304–308. https://doi.org/10.1016/S1360-1385(00)01656-3

Hood, G., & Sommer, L. (2016). High Demand, Low Supply: Colorado River Water Crisis hits Across the West. *National Public Radio*. Retrieved from http://www.npr.org/2016/12/30/507569514/high-demand-low-supply-colorado-river-water-crisis-hits-across-the-west

Hsieh, C., & Davis, A. P. (2005a). Evaluation and Optimization of Bioretention Media for Treatment of Urban Storm Water Runoff. *Journal of Environmental Engineering*, *131*, 1521–1531. https://doi.org/10.1061/(ASCE)0733-9372(2005)131:11(1521)

Hsieh, C., & Davis, A. P. (2005b). Multiple event study of bioretention for treatment of urban storm water runoff. *Water Science and Technology*, *51*, 177–181. Retrieved from http://link.aip.org/link/?ASC/118/281/1%5Cnhttp://dx.doi.org/10.1061/40685(2003)281

Hunt, W. F., Davis, A. P., & Traver, R. G. (2012). Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design. *Journal of Environmental Engineering*, *138*(6), 698–707. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000504

Hunt, W. F., Jarrett, A. R. R., Smith, J. T., & Sharkey, L. J. (2006). Evaluating Bioretention Hydrology and Nutrient Removal at Three Field Sites in North Carolina. *Journal of Irrigation and Drainage Engineering*, *132*(6), 600–608. https://doi.org/10.1061/(ASCE)0733-9437(2006)132:6(600)

Inkiläinen, E. N. M., Mchale, M. R., Blank, G. B., James, A. L., & Nikinmaa, E. (2013). The role of the residential urban forest in regulating throughfall: A case study in Raleigh, North Carolina, USA. *Landscape and Urban Planning*, *119*, 91–103. https://doi.org/10.1016/j.landurbplan.2013.07.002

James, L. A. (2013). Legacy sediment: Definitions and processes of episodically produced anthropogenic sediment. *Anthropocene*, *2*, 16–26. https://doi.org/10.1016/j.ancene.2013.04.001

Jones, M., & Hunt, W. (2010). Effect of Storm-Water Wetlands and Wet Ponds on Runoff Temperature in Trout Sensitive Waters. *Journal of Irrigation and Drainage Engineering*, *136*(9), 656–661. https://doi.org/doi:10.1061/(ASCE)IR.1943-4774.0000227

Jones, M. P., & Hunt, W. F. (2009). Bioretention Impact on Runoff Temperature in Trout Sensitive Waters. *Journal of Environmental Engineering*, *135*(8), 577–585. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000022

Jones, M. P., & Hunt, W. F. (2010). Performance of rainwater harvesting systems in the southeastern United States. *Resources, Conservation and Recycling*, *54*(10), 623–629. https://doi.org/10.1016/j.resconrec.2009.11.002

Jones, M. P., Hunt, W. F., & Winston, R. J. (2012). Effect of Urban Catchment Composition on Runoff Temperature. *Journal of Environmental Engineering*, *138*(12), 1231–1236. https://doi.org/doi:10.1061/(ASCE)EE.1943-7870.0000577

Kaushal, S. S., Groffman, P. M., Mayer, P. M., & Gold, A. J. (2008). Effects of stream restoration on denitrfication in an urbanizing watershed. *Ecological Applications*, *18*(3), 789–804. https://doi.org/10.1890/07-1159.1

Kayhanian, M., Fruchtman, B. D., Gulliver, J. S., Montanaro, C., Ranieri, E., & Wuertz, S. (2012). Review of highway runoff characteristics: Comparative analysis and universal implications. *Water Research*, *46*(20), 6609–6624. https://doi.org/10.1016/j.watres.2012.07.026

Keith R. Underwood. (2002). United States Patent Application Publication - Regnerative Stormwater Conveyance System and Method, *1*(60). Retrieved from https://www.google.com/patents/US20090290936

Kim, H., Seagren, E. A., & Davis, A. P. (2003). Engineered Bioretention for Removal of Nitrate from Stormwater Runoff. *Water Environment Research*, *75*(4), 355–367.

Kim, J. Y., & Sansalone, J. J. (2008). Event-based size distributions of particulate matter transported during urban rainfall-runoff events. *Water Research*, *42*(10–11), 2756–2768. https://doi.org/10.1016/j.watres.2008.02.005

Koryto, K. M., Asce, M., Hunt, W. F., Arellano, C., & Page, J. L. (2018). Performance of Regenerative Stormwater Conveyance on the Removal of Dissolved Pollutants: Field Scale Simulation Study, *144*(6), 1–13. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001374

Koryto, K. M., Hunt, W. F., & Page, J. L. (2017). Hydrologic and water quality performance of regenerative stormwater conveyance installed to stabilize an eroded outfall. *Ecological Engineering*, *108*, 263–276. https://doi.org/10.1016/j.ecoleng.2017.04.041

Ladson, A. R., Walsh, C. J., & Fletcher, T. D. (2006). Improving Stream Health in Urban Areas by Reducing Runoff Frequency from Impervious Surfaces. *Australian Journal of Water*

Resources, *10*(1), 23–33.

Leopold, L. B. (1994). *A View of the River*. Cambridge, Massachusetts: Harvard University Press.

Li, H., Sharkey, L. J., Hunt, W. F., & Davis, A. P. (2009). Mitigation of Impervious Surface Hydrology Using Bioretention in North Carolina and Maryland. *Journal of Hydrologic Engineering*, *14*(4), 407–415. https://doi.org/10.1061/ASCE1084-0699200914:4407

Lieb, D. A., & Carline, R. F. (2000). Effects of urban runoff from a detention pond on water quality, temperature and caged gammarus minus (say) (amphipoda) in a headwater stream. *Hydrobiologia*, *441*, 107–116. https://doi.org/10.1023/A:1017550321076

Lu, S. G., Zheng, Y. W., & Bai, S. Q. (2008). A HRTEM/EDX approach to identification of the source of dust particles on urban tree leaves. *Atmospheric Environment*, *42*(26), 6431–6441. https://doi.org/10.1016/j.atmosenv.2008.04.039

Maniquiz-Redillas, M. C., & Kim, L.-H. L.-H. (2016). Understanding the factors influencing the removal of heavy metals in urban stormwater runoff. *Water Science and Technology*, *73*(12Maniquiz-Redillas, Marla C., and Lee–Hyung L.–H. Kim. “Understanding the Factors Influencing the Removal of Heavy Metals in Urban Stormwater Runoff.” Water Science and Technology 73.12 (2016): 2921–2928. Web. 16 Sept. 2016.), 2921–2928. https://doi.org/10.2166/wst.2016.153

Metcalf, L., and Eddy, H. (1916). *American sewerage practice, disposal of sewage, Vol. 3*. New York: McGraw-Hill.

Meyer, J. L., Paul, M. J., & Taulbee, W. K. (2005). Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society*, *24*(3), 602–612. https://doi.org/10.1899/04-021.1

Miguntanna, N. P., Liu, A., Egodawatta, P., & Goonetilleke, A. (2013). Characterising nutrients wash-off for effective urban stormwater treatment design. *Journal of Environmental Management*, *120*, 61–67. https://doi.org/10.1016/j.jenvman.2013.02.027

N.C. DEQ. (2017). North Carolina Stormwater Control Measure Credit Document. *North Carolina Department of Environmental Quality*. Retrieved from https://files.nc.gov/ncdeq/Energy Mineral and Land Resources/Stormwater/BMP Manual/SSW-SCM-Credit-Doc-20170807.pdf

Natarajan, P., & Davis, A. P. (2010). Thermal Reduction by an Underground Storm-Water Detention System. *Journal of Environmental Engineering*, *136*(5), 520–526. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000172

NC DEQ. (2017). Stormwater Design Manual.

NCDENR. (2007). Stormwater Best Management Practices Manual. North Carolina Division of Water Quality.

Novotny, V. (2003). *Water Quality: Diffuse Pollution and Watershed Management* (2nd ed.). Hoboken, New Jersey: John Wiley & Sons, Inc.

Page, J. L., Winston, R. J., & Hunt, W. F. (2015). Soils beneath suspended pavements: An opportunity for stormwater control and treatment. *Ecological Engineering*, *82*, 40–48. https://doi.org/10.1016/j.ecoleng.2015.04.060

Palmer, M. A., Filoso, S., & Fanelli, R. M. (2014). From ecosystems to ecosystem services: Stream restoration as ecological engineering. *Ecological Engineering*, *65*, 62–70. https://doi.org/10.1016/j.ecoleng.2013.07.059

Parasiewicz, P. (2008). HABITAT TIME SERIES ANALYSIS TO DEFINE FLOWAUGMENTATION STRATEGY FOR THE QUINEBAUG RIVER, CONNECTICUT AND MASSACHUSETTS, USA. *River Research and Applications*, *24*, 439–452. https://doi.org/10.1002/rra

Passeport, E., Hunt, W. F., Line, D. E., Smith, R. A., & Brown, R. A. (2009). Field Study of the Ability of Two Grassed Bioretention Cells to Reduce Storm-Water Runoff Pollution. *Journal of Irrigation and Drainage Engineering*, *135*(4), 505–510. https://doi.org/10.1061/ASCÊIR.1943-4774.0000006

Paul, M. J., & Meyer, J. L. (2001). Treams in the Urban Landscape. *Annual Review of Ecology and Systematics*, *32*(1), 333–365. https://doi.org/doi:10.1146/annurev.ecolsys.32.081501.114040

Pelak, N., & Porporato, A. (2016). Sizing a rainwater harvesting cistern by minimizing costs. *Journal of Hydrology*, *541*, 1340–1347. https://doi.org/10.1016/j.jhydrol.2016.08.036

Pitt, R., Asce, M., Chen, S., Asce, A. M., Clark, S. E., Asce, M., … Asce, A. M. (2009). Compaction ' s Impacts on Urban Storm-Water Infiltration, *134*(5), 652–658.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., … Stromberg, J. C. (1997). The Natural Flow Regime: A paradigm for river conservation and restoration. *BioScience*, *47*(11), 769–784. https://doi.org/10.2307/1313099

Read, J., Wevill, T., Fletcher, T., & Deletic, A. (2008). Variation among plant species in pollutant removal from stormwater in biofiltration systems. *Water Research*, *42*(4–5), 893–902. https://doi.org/10.1016/j.watres.2007.08.036

Responsive Managment, & Southwick Associates. (2015). *MOUNTAIN TROUT FISHING : ECONOMIC IMPACTS ON AND CONTRIBUTIONS TO NORTH CAROLINA ' S ECONOMY.*

Rosgen, D. L. (1994). A classification of natural rivers. *Catena*, *22*(3), 169–199.

https://doi.org/10.1016/0341-8162(94)90001-9

Rossi, L., & Hari, R. E. (2007). Screening procedure to assess the impact of urban stormwater temperature to populations of brown trout in receiving water. *Integrated Environmental Assessment and Management*, *3*(3), 383–392. https://doi.org/10.1897/1551-3793(2007)3[383:SPTATI]2.0.CO;2

Sanders, R. A. (1986). Urban vegetation impacts on the hydrology of Dayton, Ohio. *Urban Ecology*. https://doi.org/10.1016/0304-4009(86)90009-4

Sansalone, J. J. (1999). Adsorptive infiltration of metals in urban drainage - Media characteristics. *Science of the Total Environment*, *235*(1–3), 179–188. https://doi.org/10.1016/S0048-9697(99)00211-9

Sansalone, J. J., & Buchberger, S. G. (1995). An infiltration device as a best management practice for immobilizing heavy metals in urban highway runoff. *Water Science and Technology*. https://doi.org/10.1016/0273-1223(95)00546-Y

Sansalone, J. J., & Cristina, C. M. (2004). First Flush Concepts for Suspended and Dissolved Solids in Small Impervious Watersheds. *Journal of Environmental Engineering*, *130*(11), 1301–1314. https://doi.org/10.1061/(ASCE)0733-9372(2004)130:11(1301)

Sansalone, J. J., Koran, J. M., Smithson, J. A., & Buchberger, S. G. (1998). Physical Characteristics of Urban Roadway Solids Transported during Rain Events. *Journal of Environmental Engineering*, *124*(5), 427–440. https://doi.org/10.1061/(ASCE)0733-9372(1998)124:5(427)

Schueler, T., & Brown, K. (2004). Urban Stream Repair Practices. *Urban Subwatershed Restoration Manual Series: 4*, (November), 240 pgs.

Siriwardene, N. R., Deletic, A., & Fletcher, T. D. (2007). Clogging of stormwater gravel infiltration systems and filters: Insights from a laboratory study. *Water Research*, *41*(7), 1433–1440. https://doi.org/10.1016/j.watres.2006.12.040

Smiley, E. T. J. T. J., Calfee, L., Fraedrich, B. R., & Smiley, E. T. J. T. J. (2006). Comparison of structural and noncompacted soils for trees surrounded by pavement. *Arboriculture and Urban Forestry*, *32*(4), 164–169.

Smolek, A., & Hunt, W. (2015). *Performance of Permeable Pavement over a Tight, Clay Soil in Durham, North Carolina: Hydrology, Water Quality and the Calibration and Validation of DRAINMOD*. North Carolina State University.

Southwood, T. R. E. (1977). Habitat, the Templet for Ecological Strategies? *The Journal of Animal Ecology*, *46*(2), 336. https://doi.org/10.2307/3817

Steffen, J., Jensen, M., Pomeroy, C. A., & Burian, S. J. (2013). Water supply and stormwater management benefits of residential rainwater harvesting in U.S. cities. *Journal of the American Water Resources Association, 49*(4), 810–824. https://doi.org/10.1111/jawr.12038

Stein, O. R., & LaTray, D. A. (2002). Experiments and modeling of head cut migration in stratified soils. *Water Resources Research*, *38*(12), 20-1-20–12. https://doi.org/10.1029/2001WR001166

Temple, D. M., & Hanson, G. J. (1994). Headcut development in vegetated earth spillways. *Applied Engineering in Agriculture*, *10*(5), 677–682.

Tiedje, J. M., Sexstone, A. J., Myrold, D. D., & Robinson, J. A. (1982). Denitrification: ecological niches, competition and survival. *Antonie van Leeuwenhoek*, *48*(6), 569–583. https://doi.org/10.1007/BF00399542

Tippett, R. (2017). North Carolina population growth at highest levels since 2010. Retrieved from http://demography.cpc.unc.edu/2017/02/10/north-carolina-population-growth-at-highest-levels-since-2010/

Trimble, S. W. (1997). Contribution of stream channel erosion to sediment yield from an urbanizing watershed. *Science*, *278*(21 Nov.), 1442–1444.

Tuccillo, M. E. (2006). Size fractionation of metals in runoff from residential and highway storm sewers. *Science of the Total Environment*, *355*(1–3), 288–300. https://doi.org/10.1016/j.scitotenv.2005.03.003

Turk, R. P., Kraus, H. T., Hunt, W. F., Carmen, N. B., & Bilderback, T. E. (2017). Nutrient Sequestration by Vegetation in Bioretention Cells Receiving High Nutrient Loads. *Journal of Environmental Engineering*, *143*(2), 06016009. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001158

U.S. EPA. (2015). *National Coastal Condition Assessment 2010* (Vol. 1). Washington, DC: Office of Water and Office of Research and Development. Retrieved from http://www.epa.gov/national-aquatic-resource-surveys/ncca Cover

United Nations. (2014). World's population increasingly urban with more than half living in urban areas. Retrieved October 12, 2016, from http://www.un.org/en/development/desa/news/population/world-urbanization-prospects-2014.html

United States Environmental Protection Agency. (2009). National Water Quality Inventory: Report to Congress. *Water*, (January), 43. https://doi.org/http://www.epa.gov/owow/305b/2004report/

Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The River Continuum Concept.

Vaze, J., & Chiew, F. H. S. (2004). Nutrient loads associated with different sediment sizes in urban stormwater and surface pollutants. *Journal of Environmental Engineering*, *130*(4), 391–396. https://doi.org/10.1061/(ASCE)0733-9372(2004)130:4(391)

Wadzuk, B. M., Hickman, J. M., & Traver, R. G. (2015). Understanding the Role of Evapotranspiration in Bioretention: Mesocosm Study. *Journal of Sustainable Water in the Built Environment*, *1*(2), 04014002. https://doi.org/10.1061/JSWBAY.0000794

Walsh, C. J., Fletcher, T. D., & Ladson, A. R. (2009). Retention Capacity: A Metric to Link Stream Ecology and Storm-Water Management. *Journal of Hydrologic Engineering*, *14*(4), 399–406. https://doi.org/10.1061/ASCE1084-0699200914:4399

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan II, R. P. (2005). The urban stream syndrome : current knowledge and the search for a cure The urban stream syndrome : current knowledge and. *Journal of the North American Benthological Society*, *24*(3), 706–723. https://doi.org/10.1899/04-028.1

Wardynski, B. J., Asce, A. M., Winston, R. J., Hunt, W. F., Wre, D., & Asce, M. (2013). Internal Water Storage Enhances Exfiltration and Thermal Load Reduction from Permeable Pavement in the North Carolina Mountains. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000626

Wardynski, B. J., & Hunt, W. F. (2012). Are Bioretention Cells Being Installed Per Design Standards in North Carolina? A Field Study. *Journal of Environmental Engineering*, *138*(12), 1210–1217. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000575

Wardynski, B. J., Winston, R. J., Line, D. E., & Hunt, W. F. (2014). Metrics for assessing thermal performance of stormwater control measures. *Ecological Engineering*, *71*, 551–562. https://doi.org/10.1016/j.ecoleng.2014.07.068

West Virginia Department of Environmental Protection. (2012). West Virginia Stormwater

Management & Design Guidance Manual, 1–22.

Winston, R. J., Hunt, W. F., Osmond, D. L., Lord, W. G., & Woodward, M. D. (2011). Field Evaluation of Four Level Spreader – Vegetative Filter Strips to Improve Urban Storm-Water Quality. *Journal of Irrigation and Drainage Engineering*, *137*(March), 170–182. https://doi.org/10.1061/(ASCE)IR.1943-4774.0000173.

Wossink, A., & Hunt, B. (2003). The economics of structural stormwater BMPs in North Carolina. *Water Resources Research Institute of the University of North Carolina.*, (May).

Xiao, Q., & McPherson, G. G. (2011). Rainfall interception of three trees in Oakland, California. *Urban Ecosystems*. https://doi.org/10.1007/s11252-011-0192-5

CHAPTER 2 . THERMAL MITIGATION BY SELECT STORMWATER CONTROL MEASURES IN TROUT SENSITIVE WATERS OF WESTERN NORTH CAROLINA

2.1. Abstract

Land use changes and urbanization negatively impact aquatic environments. Urbanized watersheds produce downstream flooding, while exporting pollutants, including sediment, oils and grease, heavy metals, nutrients, and pathogens. Thermal enrichment is a lesser-studied stormwater pollutant that has substantial impacts on cold-water streams. Runoff heated by impervious surfaces produce in-stream thermal spikes exceeding the 21^{o}C avoidance threshold for brook trout (*Salvelinus fontinalis)*. Stormwater Control Measures (SCMs) mitigate many of the hydrologic and water quality impacts of increased imperviousness; however, their performance at thermal mitigation is less understood. A study in Hendersonville, North Carolina, was conducted to assess the performance of a rainwater harvesting (RWH) system, a level spreader vegetative filter strip (LSVFS), and a bioretention cell (BRC) for temperature and thermal load reduction. All systems received thermally-enriched runoff from roof top and paved areas. BRC buffered maximum and, to a lesser extent, median event temperatures, while reducing flow volumes and thermal loads. BRC treatment reduced thermal stress on trout populations, resulting in a 27% chance that maximum outlet temperatures would exceed 21^{o}C. The RWH studied herein appeared to thermally stratify which resulted in significantly hotter temperatures near the storage tank overflow. Typical RWH design, with outlet determining maximum storage capacity, increased the likelihood of effluent temperature exceeding the 21^{o}C trout threshold. LSVFS displayed minimal ability to buffer influent temperatures, resulting in a 75% chance of maximum effluent temperatures exceeding 21^{o}C.

2.2. Introduction

2.2.1. Importance of Thermal Regime

Stream networks evolve from headwaters to river mouth in which biological communities conform both structurally and functionally to energy dissipation patterns of the physical system (Vannote et al., 1980). Headwater streams represent the maximum interface with the terrestrial environment, created by the influence of riparian vegetation for shading and detrital inputs. Downstream reach assemblages are organized to take advantage of material leakage from upstream reaches via increasingly efficient processing of the smallest organic material; while predatory invertebrate groups are mainly constant as stream order increase, fish species diversity increases as streams progress from cold-water to warm water (Vannote et al., 1980).

Stream temperatures regulate aquatic life (Beschta et al., 1987). Headwater streams are mainly groundwater-dominated and experience little temperature change; the resultant biological assemblages are low in diversity because few species can function in the low temperature fluctuations and restricted nutritional base (Vannote et al., 1980). The thermal regime of a stream is influenced by many factors: atmospheric conditions, topography, land use, stream discharge, and streambed substrate (Caissie, 2006). Stream temperatures vary widely in space and time; mean daily temperature increases as stream order increases (Caissie, 2006). Spatial variability in confluences, seepage, and microhabitats occur widely while seasonal variability closely mimics air temperatures (Caissie, 2006). Cold-water ecological stability is maintained by low diel and annual temperature regimes; perturbations in the thermal regime shift the continuum upstream or downstream depending on location, magnitude, and type of disturbance, and species will only persist in an area as long as the habitat remains favorable for breeding longer than its gestation time (Vannote et al., 1980; Southwood, 1977). Elevated stream temperatures lead to higher stream metabolism and depletion of available nutrients; elevated temperatures can also alter the species composition along a reach and result in early insect larvae emergence (Beschta et al., 1987).

Thermal inputs from industry and thermonuclear reactors, and the artificial reduction of stream temperatures below large dams, affect fish migration timing (Beschta et al., 1987). Natural heat exchange processes take place at the air-water and water-streambed interface when tributaries, thermal effluent, and water extractions are negligible (Caissie, 2006). Heat flux at the air-water interface occurs via solar radiation (net short-wave radiation) and net long-wave radiation, evaporative heat flux, and convective heat transfer; other minimal components include precipitation and friction (Caissie, 2006). Solar radiation is often the most dominating heating component and highlights the importance of riparian vegetation in minimizing solar gain (Caissie, 2006). Stream bed interactions are mainly a function of geothermal loading and are likely more important in headwater streams due to the greater dominance of groundwater (Caissie, 2006).

2.2.2. Trout Waters

In 2014, North Carolina trout fishing had a total economic impact of $383.3 million and supported approximately 3,600 jobs (Responsive Managment & Southwick Associates, 2015). Western North Carolina is home to three trout species, the brown (*Salmo trutta*), brook (*Salvelinus fontinalis*), and rainbow (*Oncorhynchus mykiss*) trout (NC Wildlife Resource Commission, 2018). Trout require cold-water habitats, streams with mean weekly temperatures less than 22°C (Armour, 1991). Cold-water habitats are threatened by land use change, riparian buffer loss, and urbanization (Beschta et al., 1987; Rossi & Hari, 2007; Jones et al., 2012).

Fish are cold-blooded, so their environment directly impacts physiology and activity; sensitivity to external temperatures is dependent on species and life stage (Beschta et al., 1987). Coutant (1977) synthesized temperature preference data from field and laboratory studies into three endpoints for developing waterbody standards. Coutant's (1977) endpoints include: (1) final preferendum, (2) upper avoidance, and (3) lower avoidance tolerances (Armour, 1991; Coutant, 1977). Upper avoidance tolerances for brown, brook, and rainbow trout range from 20-22ºC; final preferendum range from 12-22ºC, depending on life stage (Coutant, 1977). Most temperature sensitive species native to North Carolina are acclimated to 15ºC (Ferguson, 1958) and have upper tolerable and incipient lethal limits.

2.2.3. Impacts of Stormwater Runoff

The effects of urbanization on water quality are well documented, increased runoff volumes and peak flows, decreased time to peak discharge, increased suspended sediment and nutrients, channel incision, scour, and streambed and bank erosion (Pitt et al., 2009; Burns et al., 2012; Davis et al., 2003; Davis et al., 2006; Novotny, 2003; Ladson et al., 2006; Leopold, 1994; Walsh et al., 2005); however, less research documents the impacts of stormwater runoff on the thermal regime of streams.

Jones et al. (2012) began initial exploration of the effect of parking lot composition and layout on runoff temperature. The results of this study showed that runoff temperatures experience seasonal fluctuation with the hottest runoff events happening during peak summer months, June-August. Runoff also exhibited diel effects with the warmest events during afternoon storms. Runoff temperatures were significantly warmer than the 21ºC threshold for brook trout (Jones et al., 2012). Tree canopy, shading parking lots, reduced median and maximum runoff temperatures by 0.3ºC and 3.2ºC, respectively. The effect of asphalt composition on runoff temperature was compared by capturing runoff from chip-seal, a lighter colored asphalt, and regular hot-mix. No significant difference between runoff temperatures was determined; median runoff temps from the chip-seal and hot-mix asphalts were 20.6^{o}C and 22.0^{o}C, respectively. The authors described the hot-mix catchment as aged and light-in-color with partial shading possibly resulting in reduced runoff temperatures. Jones et al. (2012) also found a significant reduction in maximum runoff temperature when surface collection was compared to in-pipe conveyance temperatures.

Lieb & Carline (2000) conducted a 42-day bioassay to understand if metal concentrations from detention pond effluent were causing benthic macroinvertebrate impairment observed on Thompson Run, Pennsylvania. The bioassay included exposure of caged *Gammarus minus* to in-stream detention pond effluent. Data were collected on stream temperatures and metal concentrations in a reference reach, in-stream (at the pond outlet), and upstream of the pond (Lieb & Carline, 2000). Temperature variation at the pond outlet exhibited a higher range (9.8^{o}C—10.8^{o}C) compared to variation at the reference reach (10.1^{o}C—10.4^{o}C). Hourly temperature change during stormflow at the reference reach did not exceed 0.2^{o}C while pond effluent hourly temperature change reached 6.6^{o}C. Hourly base flow temperature change at the reference reach never exceeded 1^{o}C (Lieb & Carline, 2000).

Walsh et al. (2005) discussed the urban stream syndrome and the effect of catchment urbanization on base flow reduction and increased in-stream diel temperature variation. Walsh et al. (2005) also discussed the loss of riparian forest and buffers as a source of increased direct solar radiation and stream heating; channel widening caused by frequent high flow events increases stream area for thermal gain.

2.2.4. Stormwater Regulations

In 1972, the Clean Water Act (CWA) was established to provide a framework for protecting U.S. water resources by preserving and restoring the chemical, physical, and biological integrity of the nation's waters by requiring states to monitor, analyze, and develop water quality standards (EPA, 2016). Under the National Pollutant Discharge Elimination System (NPDES), it is the local municipalities' responsibility to implement a plan for monitoring and permitting of point and non-point source discharges of effluent or runoff leaving municipalities, industrial facilities, or other entities to waters of the state (EPA, 2016; NCDENR, 2007).

NPDES requires adoption of minimum standards in post-construction management of stormwater for new development and redevelopment. Rules apply to new and redevelopment project areas that exceed 0.4-ha (1-ac); projects are categorized as low or high density depending on a 24% Built Upon Area (BUA) threshold. Low- and high-density developments both require the use of (1) vegetated conveyances to the maximum extent possible, (2) minimum 9.1-meter (30-foot) buffers between BUA and perennial and intermittent streams, and (3) deed restrictions and protective covenants to ensure developers pass on municipally-approved plans for stormwater management. In addition to low-density development requirements, high-density development must employ structural SCMs to treat the water quality event (38 mm in coastal counties and 25 mm elsewhere in North Carolina). NCDEQ crediting of SCMs focuses on the reduction of nutrients and Total Suspended Sediment (TSS) (NCDEQ, 2017); temperature mitigation is designated as an "other benefit" (NCDEQ, 2017). In the SCM Credit Document bioretention, rainwater harvesting, and level spreader-vegetative filter strip are assigned good, excellent, and poor designations for in-stream temperature protection, respectively (N.C. DEQ, 2017).

2.2.5. Thermal Mitigation by Stormwater Control Measures

Jones & Hunt (2009) evaluated four bioretention cells (BRCs) in western North Carolina to understand their effect on runoff temperatures and identify design improvements to better mitigate thermal pollution. This study found runoff temperatures during May through October in 2006 and 2007 were significantly higher than the 21°C trout avoidance threshold. BRCs studied were able to reduce the initial thermal spike of runoff; however, they were unable to acclimate to cooler runoff later in a storm event. One bioretention system showed an inability to consistently reduce runoff temperatures with no significant difference between median influent and effluent temperatures (Jones & Hunt, 2009). Thermal equilibrium between infiltrated runoff and soil media was reached within 60-cm, limiting the media's ability to reduce runoff temperatures after the initial thermal spike and pavements had cooled.

Jones & Hunt (2009) found that oversizing bioretention systems provided minimal additional thermal reduction benefits for runoff that left via the underdrain; however, larger bioretention systems discharge less outflow by converting more runoff to infiltration, effectively eliminating the thermal impact (Jones & Hunt, 2009). Hunt et al. (2012) in review of bioretention design, highlight Internal Water Storage (IWS) and longer retention times as methods to improve (1) subsoil heat transfer and (2) infiltration. Bioretention vegetation shades the soil surface reducing thermal input on upper soil layers and deeper soils are less influenced by ambient air temperatures and solar radiation (Hunt et al., 2012).

Winston et al. (2011) studied two Level Spreader-Vegetative Filter Strips' (LSVFSs) ability to reduce runoff temperatures. Vegetative filter strips were composed of volunteer vegetation, mainly warm weather grass and/or a wooded buffer and differed in length (7.6-m and 15.2-m). Median and maximum temperatures were reduced across both the short and long

LSVFSs. Effluent temperature variance was less than inlet variance; however, effluent variance was higher than what was observed for BRC (Winston et al., 2011). Median and maximum effluent temperatures from the short LSVFS were significantly greater than the 21°C trout threshold, while only the maximum effluent temperatures from the long LSVFS were significantly greater than the 21°C threshold. Winston et al. (2011) determined that influent temperatures were dependent on ambient air temperatures; they found LSVFS can reduce influent temperatures, with the greatest reductions in runoff temperature occurring at the beginning of storm events when the influent and soil temperature gradient is the highest (Winston et al., 2011).

Jones & Hunt (2010) evaluated the effect of stormwater wetlands and wet ponds in western North Carolina on runoff temperatures; SCM effluent temperatures were significantly higher than the 21°C trout threshold from June through September. Influent temperatures exhibited a peak early in the event and cooled as runoff continued. This study concluded wetland and wet ponds were sources of thermal pollution (Jones & Hunt, 2010). Lieb & Carline (2000) documented increased temperature variation and increased median in-stream temperatures downstream of a detention pond.

Jones & Hunt (2010) observed thermal stratification of a wet pond, resulting in significantly different temperatures at all study depths (0-, 40-, 80-, 120-cm). Wetland temperatures benefited from additional vegetation shading and increased evaporation resulting in temperatures deeper than 90-cm being significantly lower than the trout threshold. Jones & Hunt (2010) proposed modified outlets for both systems to discharge cooler water from deeper depths.

Herb et al. (2009) created a 1-D, unsteady model to simulate dry pond hydraulics and heat transfer properties to illustrate the effects of land use and zoning in watersheds with cold-water streams. Simulation results showed dry ponds were not effective at thermal pollution mitigation with outflow temperatures on average 1.2^{o}C higher than runoff temperatures. Long retention times allow accumulation of solar energy which is released in outflows (Herb et al., 2009). This study determined dry ponds were able to reduce the rate of heat export by an order of magnitude compared to inflow rates and peak in-stream temperature spikes of unmitigated runoff; however, the prolonged outflow increased the duration of temperature impacts (Herb et al., 2009).

Rainwater Harvesters (RWHs) are studied for their water quality and hydrology benefits (Doan & Davis, 2017; Gee & Hunt, 2016; Jones & Hunt, 2010; Pelak & Porporato, 2016). Studies of RWHs show their potential to substantially reduce runoff volumes and peak flows while simultaneously reducing potable water demands by greater than 80% in the U.S. Southeast (Steffen et al., 2013); however, lack of water usage (underutilization) diminished performance and produced excessive overflow when storage capacity was not available to capture runoff volumes (Jones & Hunt, 2010). Overflow occurred during 62-94% of rainfall events greater than 10 mm (Jones & Hunt, 2010).

Passive and active release systems allow drainage of the RWH storage tank ensuring capacity for runoff capture. Gee & Hunt (2016) determined passive release systems allowed 77% runoff capture (23% overflow); active release increased runoff capture to 88% (12% overflow) (Gee & Hunt, 2016). Kalz et al., (2010) studied the use of a rainwater cistern as a heat source and sink to support heating and cooling systems for a passive energy house in Germany. Minimal research has been conducted on the thermal mitigation potential of RWHs vis-à-vis stormwater runoff. Large runoff volume reductions reduce the volume of thermally-enriched runoff.

2.2.6. Performance Metrics for Thermal Mitigation by Stormwater Control Measures

Wardynski et al. (2014) evaluated ten temperature-based water quality metrics for SCMs (Table 2-1). Their goal was to determine the best metric or combination of metrics to evaluate SCM temperature mitigation. SCM evaluation methods vary in data required and ease of analysis.

Table 2-1. SCM Temperature Evaluation Metrics Adapted from Wardynski et al. (2014)

Method	*Description*	*Data Required*	*Pros*	*Cons*
Event Mean Temperature Reduction	$EMT = \frac{\int_0^{D_i} TQ\,dt}{\int_0^{D_i} Q\ dt}$	Instantaneous temperature & hydrology	More representative than event median temperatures	Only useful with ET & infiltration
Instantaneous Maxima	Event based inlet & outlet temperature maxima	Instantaneous temperatures	Minimal data collection; Temperature buffering capacity	Limited temporal or load use; Need downstream standards for comparison
Biological Temperature Threshold	Evaluation of short & long-term exposure to temperatures. Weight over duration	Instantaneous temperature, hydrology, & optimum & lethal temperatures	Considers stream ecology; Important for extended dewatering SCMs	Requires EMT calculation
Temperature-Duration Plot	Plot of temperatures exceeding biological maxima	Instantaneous temperature, flow duration, biological threshold & reference reach data	SCM can be designed to infrequently exceed threshold	Need species supporting reference reach; Does not consider continuous exposure duration
Uniform Continuous Above Threshold	Considers continuous duration above temperature thresholds	Long-term reference reach data & biological threshold	SCM can be designed to infrequently exceed threshold	Need species supporting reference reach; More conservative approach
Probabilistic Threshold Determination	Uses reference reach temperature regime to determined habitable range	Reference reach data & biological threshold	SCM effluent goals based on stream ecology	Need species supporting reference reach
Groundwater Temperature	Uses groundwater temperature as an effluent temperature metric	SCM effluent temperatures & groundwater temperatures	Minimal data collection	Produces conservative SCM targets
Thermal Load Reduction & In-Stream Impacts	Determines energy input to stream system	Hydrology, effluent temperature, & duration	Considered temperature & volume reduction within SCM	Does not consider in-stream mixing; Often not tied to biological thresholds
Low Impact Development as a Surrogate	Assumes treatment of WQ event is sufficient thermal mitigation	SCM hydrology & temperature data	Ease of application	Needs to be validated
Direct Monitoring of Thermal Impacts	Monitoring downstream of development	In-stream temperatures	Direct visualization of stream impacts	Needs biological reference conditions; "after-the-fact"

2.2.7. Mud Creek

Mud Creek (HUC# 040302) is 292.7-km^2 watershed within the French Broad River Basin. The Mud Creek watershed is comprised of forest, residential, agricultural, and industrial land uses. Major sources of stream impairment identified by the Mud Creek Watershed Council with data from NCDEQ and the Tennessee Valley Authority (TVA) are (1) volume, velocity, and quality of post-construction runoff from existing and new development; (2) pesticides, nutrients, sediment, bacteria, and other non-point source pollutants; (3) habitat degradation due to sedimentation, bank erosion, channelization, lack of riparian vegetation, loss of riffles or pools, loss of woody habitat and streambed scour; and (4) sediment from construction activities, unpaved road/driveways, forestry, mining and development (NCDENR, 2003). Mud Creek and some of its tributaries historically harbored trout populations; some streams remain trout waters (NCDENR, 2003). Trout stream classification is intended to protect high water quality standards for dissolved oxygen, temperature, and turbidity regardless of trout presences. The objectives of this chapter are to (1) monitor and analyze the performance of a RWH, a LSVFS, and a BRC for mitigation of thermally-enriched stormwater runoff and (2) to determine which practice best protects North Carolina trout waters.

2.3. Materials and Methods

2.3.1. Field Sites

Three field sites located in the North Carolina mountains were chosen for installation of SCMs with the goal to evaluate thermal pollution mitigation. All sites drain to tributaries of Mud Creek (HUC# 040302) in Hendersonville, North Carolina (Figure 2-1). A RWH was installed at The Free Clinics (TFC) and a BRC and a LSVFS were installed at the Calvary Baptist Church (CBC).

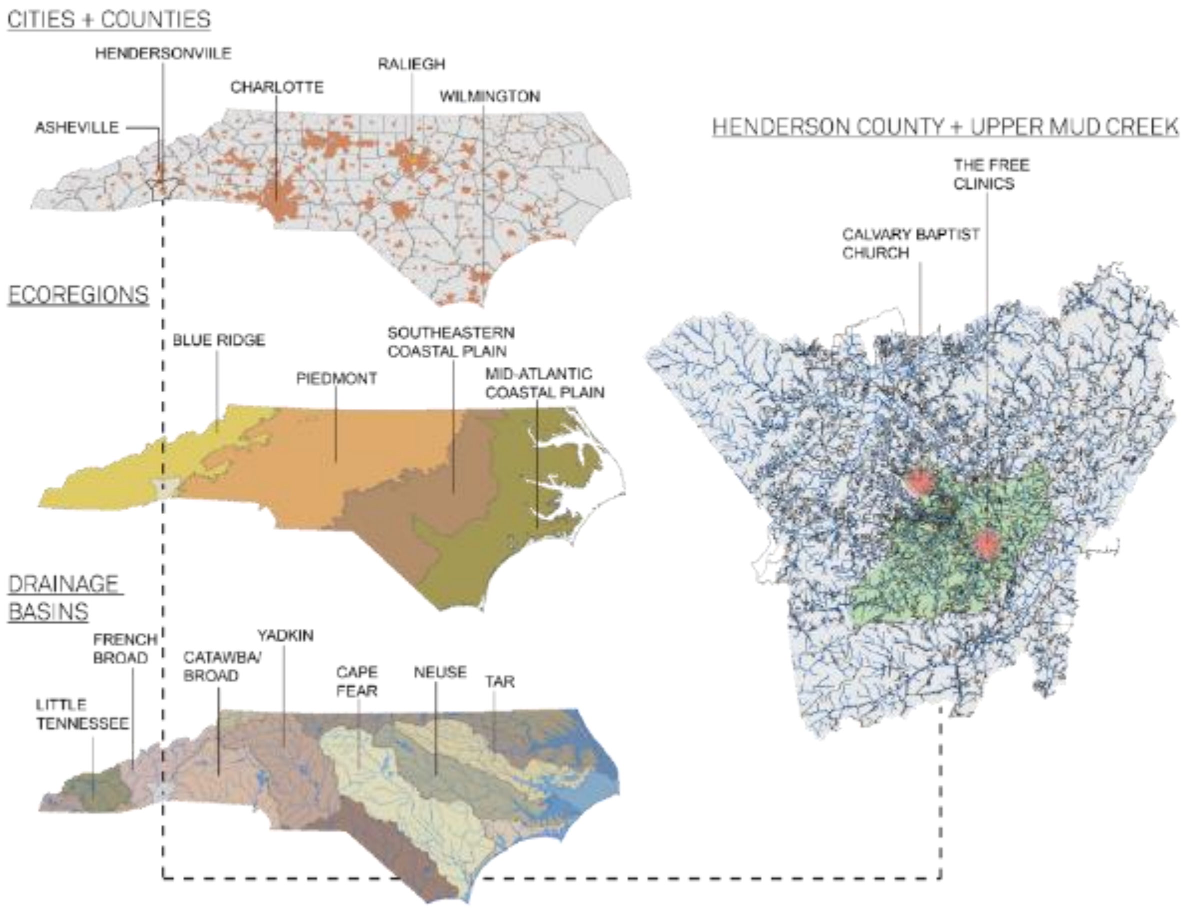


Figure 2-1. Regional Context Map

Located at 841 Case Street (35.292985, -82.428077) (Figure 2-2) along an unnamed tributary of King Creek, TFC operates a series of wooded walking trails and a teaching garden used to educate clients about the role of exercise and diet in disease prevention and health improvement. RWH installation at the TFC provides non-potable water for irrigation at the teaching garden.

Figure 2-2. TFC Site Context Map

The Calvary Baptist Church is located at 2501 Haywood Road (35.337229, -82.489899) along Brittain Creek (Figure 2-3). A BRC and a LSVFS were both installed to treat roof and parking lot runoff from the eastern and western halves of the property, respectively.

Figure 2-3. CBC Site Map

2.3.2. Sites Description

2.3.2.1. Rainwater Harvester

The RWH designed and installed at TFC (Table 2-2) uses a wet conveyance system, 10.2-cm PVC piping to deliver runoff to RWH inlet (Appendix A). A passive release (Gee & Hunt, 2016) was incorporated in the wet conveyance to prevent water ponding and freezing damage (Figure 2-4).

Table 2-2. RWH Design Elements

Parameter	*Value*
Drainage Area (m^2)	241.5
Land Use	Asphalt shingling
Imperviousness (%)	100
Curve Number	98
*WQ Event (m^3)**	5.8
Downspouts	4
Tank Capacity (m^3)	9.4

*Estimated runoff from a 25-mm event

Figure 2-4. Rainwater Harvester Layout

2.2.3.2. Level Spreader-Vegetative Filter Strip

The LSVFS was designed to treat runoff from the western portion of the CBC property (Table 2-3; Figure 2-5). Runoff is captured in a drop-inlet and conveyed via a 20-cm diameter, 5.2-m long PVC pipe to the LSVFS blind swale (Winston et al., 2011); the drop inlet contains a weir and bypass outlet for larger storm events. The LSVFS included a rock-lined blind swale with underdrain, concrete level spreader, gravel verge, vegetative filter strip, and an impermeable-lined monitoring trough (Figure 2-6).

Table 2-3. LSVFS Design Elements

Parameter	*Value*
Drainage Area (ha)	0.28
Land Use	Asphalt parking, shingling, & lawn
Imperviousness (%)	85
Curve Number	95
*WQ Event (m^3)**	37.5
LSVFS Avg. Drainage Area Slope (%)	5.6
LS Length (m)	9.1
VFS Length (m)	15.2
Surface Area (m^2)	138.7
LSVFS Longitudinal Slope (%)	5.0%
Vegetation	Fescue Sod

*Estimated runoff from 25-mm event

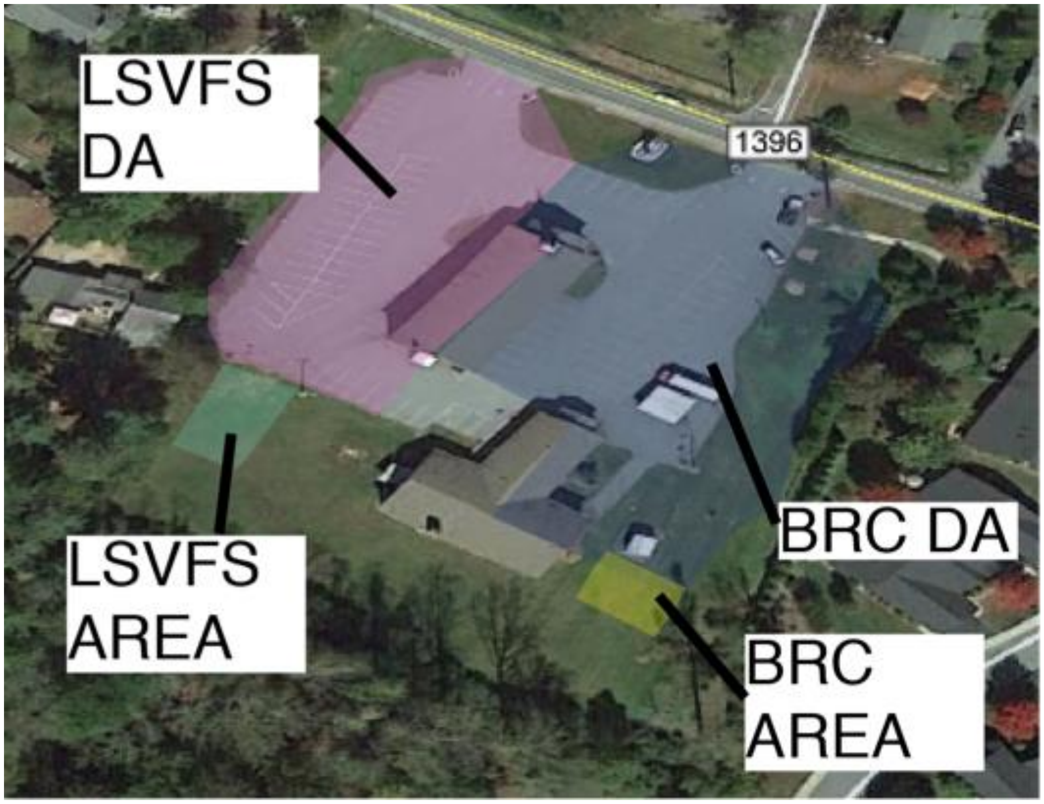


Figure 2-5. CBC Layout

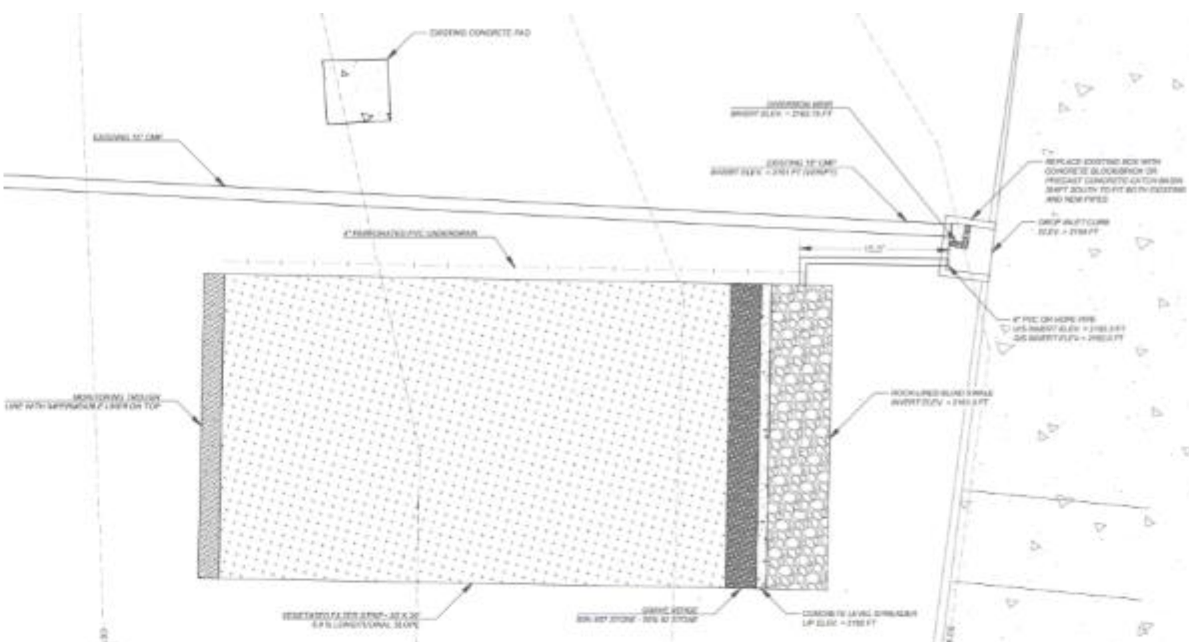


Figure 2-6. LSFVS Construction Drawing (J. Johnson personal communication, 2018)

2.2.3.3. Bioretention Cell

The BRC was designed to treat runoff from eastern portions of the CBC property (Table 2-4, Figure 2-5). Runoff flows from some roofing and parking areas on to lawn and a swale for conveyance to the BRC (Figure 2-5). Other portions of roof area drained directly to the BRC via downspouts and buried corrugated plastic pipe. The BRC includes 61-cm of media depth with fescue sod and 45-cm of internal water storage depth (Figure 2-7). The BRC surface area was 3.5% of the contributing watershed or a loading ratio of 29:1 (Table 2-4).

Table 2-4. BRC Design Elements

Parameter	*Value*
Drainage Area (ha)	0.42
Land Use	Asphalt parking, shingling, & lawn
Imperviousness (%)	57
Curve Number	88
*WQ Event (m^3)**	27.3
LSVFS Avg. Drainage Area Slope (%)	5.7
Width (m)	18.6
Length (m)	7.9
Bowl Volume (m^3)	44.9
Loading Ratio	29:1
Vegetation	Fescue sod

*Estimated runoff from 25-mm event

1) $Loading\ Ratio = \frac{Watershed\ Area}{SCM\ Area}$

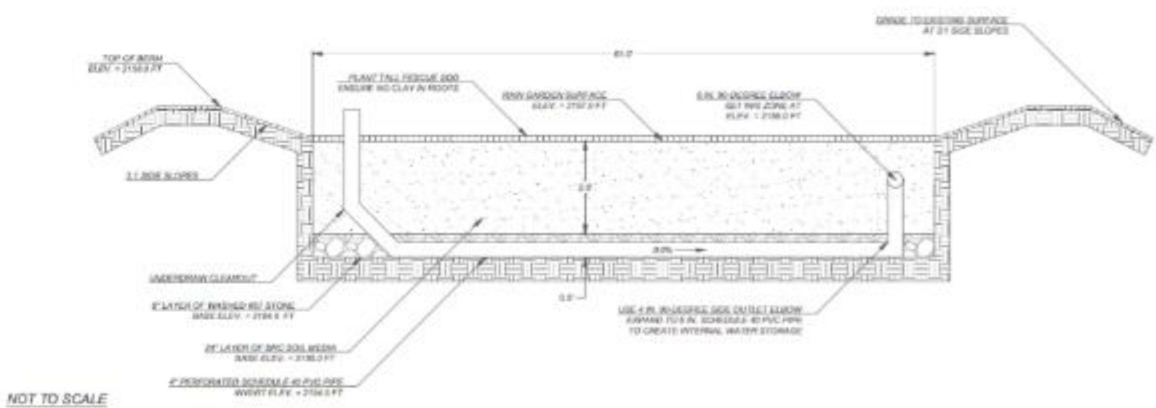


Figure 2-7. BRC Construction Drawing (J. Johnson personal communication, 2018)

2.3.3. Monitoring Design & Hydrologic Data

2.3.3.1. Rainwater Harvester

Hydrologic monitoring began August 28, 2017, and concluded November 12, 2017. At TFC, HOBO U20™ water level and temperature loggers were deployed at varying depths (1.0-,1.5-, and 2.0-m) within the storage tank; data were collected at 2-min intervals. A HOBO tipping bucket (0.025-mm/tip) and manual rain gauge were installed on September 12, 2017, and used to collect rainfall accumulation data; manual rain gauge accumulation depths were used to normalize tipping bucket measurements. HOBO U20™ sensors were attached to a PVC pipe inside the cistern (Figure 2-4). The atmospheric logger recorded ambient weather data and was used for atmospheric data correction; it was left outside the storage tank.

2.3.3.2. Level Spreader-Vegetative Filter Strip

HOBO U20™ water level loggers recorded inflow and outflow water depths and temperatures at 2-min intervals within the inlet and outlet blind swales beginning on August 28, 2017; atmospheric conditions were also recorded. Sensors were mounted within a PVC pipe and allowed to sit just above the soil surface; sensor depths were used to confirm the presence of flow (Figure 2-8). At CBC, a HOBO tipping bucket (0.025-mm/tip) and manual rain gauge was installed on September 12, 2017, and used to collect rainfall data; manual rain gauge accumulation depths were used to normalize tipping bucket measurements. Rain gauges at TFC and CBC were installed 0.9-m above the ground in areas free of overhanging branches or other obstructions.

Figure 2-8. LSVFS & Monitoring

2.3.3.3. Bioretention Cell

At the BRC, two wells were installed on August 28, 2017, within the BRC media using 3-cm diameter PVC pipe with 1-cm perforations and covered with a well sock; wells were installed to depths of 31-cm and 53-cm below the soil surface. On September 12, 2017, a weir box with stilling baffle and 60^{o} V-notch was installed on the underdrain of the BRC outlet (Figure 2-9). HOBO U20^{TM} water level and temperature loggers were installed in the well casings and the weir box; loggers recorded data at 2-min intervals. Inlet flow was measured by mounting a HOBO U20^{TM} logger within a 3-cm diameter PVC pipe and allowing it to sit just above the soil surface; sensor depths were used to confirm the presence of flow.

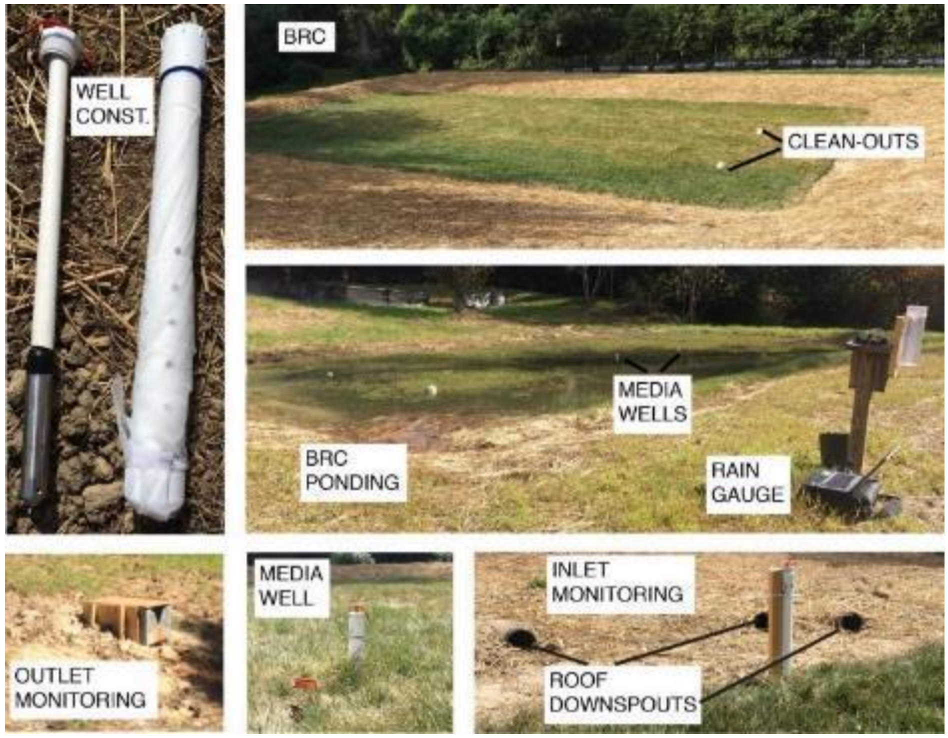


Figure 2-9. BRC & Monitoring

Due to an equipment malfunction, the tipping bucket rain gauge at TFC did not record from October 25, 2017, through November 22, 2017. Missing rainfall data were replaced with hourly rainfall data for the Bat Fork Creek weather station (BFCN7) (35.3118, -82.4352) in the NC DCP network located adjacent to the Hendersonville Airport-0A7 (6.6-km from CBC and 1.3-km from TFC).

2.3.4. Hydrology

Run-off volumes for discrete events were estimated using the NRCS Curve Number Method (Eq. 2.1 – 2.1.2) (NCDEQ, 2017). Composite Curve Number (CN) were used in areas of multiple land uses (Eq. 2.1.2).

$$Q = \frac{(P-0.2S)^2}{(P-0.2S)+S} * A * C \qquad \text{(Eq. 2.1)}$$

$$S = 25.4 * (\frac{1000}{CN} - 100) \qquad \text{(Eq. 2.1.1)}$$

$$CN = \frac{CN_i * A_i}{A_T} \qquad \text{(Eq. 2.1.2)}$$

Where,

Q = Runoff Volume (m^3)

P = Precipitation depth (mm)

A_i = Watershed area i (m^2)

A_T = Total watershed area (m^2)

S = Potential maximum retention (mm)

CN = Composite Curve Number

C = Conversion factor: $\frac{1\,m}{1000\,mm}$

i = area 1, 2, 3, …, n

<u>2.3.5. Thermal Load</u>

Thermal load estimations were made using Eq. 2.2 (Wardynski et al., 2014; Winston et al., 2011).

$$W = Q\rho TC \qquad \text{(Eq. 2.2)}$$

Where,

W = thermal load

Q = flow volume (m^3)

ρ = density of water at standard conditions (kg/m^3)

T = temperature (K)

C = specific heat of water 4.18-KJ/(kg-K)

<u>2.3.6. Statistical Analysis</u>

Statistical analysis was performed using R Software Version 3.3.3 (R Core Team, 2017); dedicated repositories document data and processing methods for each SCM and Brittain Creek (RWH (Gregg, 2018d), LSVFS (Gregg, 2018c), BRC (Gregg, 2018b), and Brittain Creek

(Gregg, 2018a)). Precipitation and portions of temperature data were determined to be non-normal via the Shapiro-Wilks Normality test; therefore, non-parametric methods were used throughout the analysis to maintain uniformity. Event medians and maximum were used during tests of significance. The Wilcoxon Signed-Rank Sum and Wilcoxon Rank Sum Test were used for determining statistical significance using a Type 1 error (α) of 0.05.

2.4. Results & Discussion

2.4.1. Weather Summary

SCM rainfall event discretization typically occurs when inter-event dry periods exceed 6-hours (Koryto, 2016; Sharkey, 2006); however, in this study rainfall event delineation was SCM dependent due to the effect of direct solar radiation and heating on monitoring locations. Discrete rainfall events for the RWH, the LSVFS, and the BRC were delineated after the last rainfall measurement that precede a dry period of 2-, 1-, and 6-hours, respectively. The RWH storage tank was black and unshaded (Figure 2-4); the LSVFS monitoring trough was lined with a black impermeable liner (Figure 2-8). It was evident that both the RWH and the LSVFS were substantially influenced by direct solar radiation which contributed to temperature increases. The BRC was less influenced by solar radiation.

Table 2-5 compares 30-year weather normals for Hendersonville with observations from both sites during the monitoring period. Observed average and maximum air temperatures during September and October were hotter than the long-term weather normal. November was hotter, and August was mixed; however, data collection did not capture the entire months of August and November. Sixty-four percent less precipitation fell in September than normal at TFC, while CBC was close to normal. Fifty-three percent and 163% more precipitation fell at TFC and CBC than October normals, respectively. Cumulatively over the monitoring period, TFC had 28% less precipitation than the same period in a normal year, while CBC had 79% more precipitation than a normal year.

Table 2-5. Observed Site and Hendersonville Weather Normals 1981-2010 (NOAA, n.d.)

Parameter	*August*	*September*	*October*	*November*
Normal Avg. Air Temperature (°C)	23.2	19.4	13.8	8.8
Observed Avg. Air Temperature (°C) (TFC/CBC)	19.3/19.2*	19.0/19.0	14.8/14.9	10.4/10.1*
Normal Maximum Air Temperature (°C)	28.2	24.8	19.9	15.1
Observed Maximum Air Temperature (°C) (TFC/CBC)	26.1/32.0*	38.8/38.8	36.2/36.2	31.2/32.1*
Normal Avg. Precipitation (mm)	129.0	119.4	88.1	116.3
Observed Rainfall Accumulation (mm) (TFC/CBC)	19.8/18.0*	42.2/114.3	134.9/232.2	27.7/31.8*

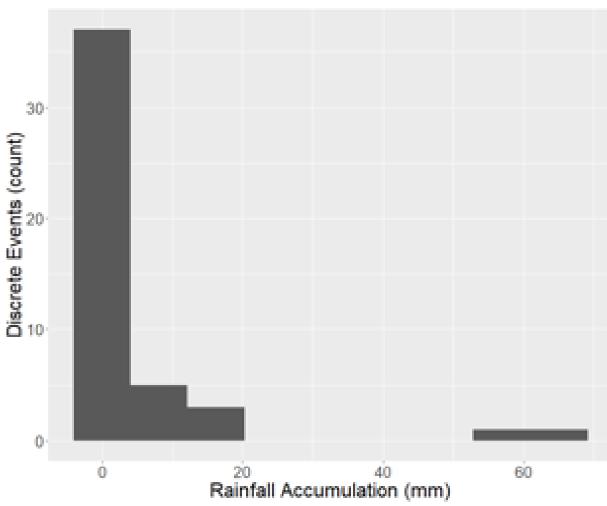
*Incomplete dataset; missing 08/01/2017-08/27/2017 and 11/23/2017-11/30/2017

During the approximately 3-month monitoring period between August 28, 2017, and November 22, 2017, the RWH, LSVFS, and BRC had 30, 54, and 29 discrete rainfall events, respectively (Table 2-6). CBC observed 171-mm more precipitation than TFC; 7.4-km separate CBC and TFC. The largest event at CBC (96.5 mm in 15.9 hr) occurred on October 23, 2017, and generated the largest 5-minute peak intensity of 74.5-mm/hr. That precipitation depth had a 2- to 5-year Annual Recurrence Interval (ARI); while that intensity was a 25- to 50-year ARI (NOAA National Weather Service, n.d.). At TFC, two discrete events produced 61.0- and 62.5-mm on October 8 and October 28, 2017, respectively; however, the maximum peak 5-minute intensity, 30.5-mm/hr, was generated by a 4.6-mm event on September 12, 2017. TFC had two discrete events with rainfall accumulations greater than or equal to 25.4-mm, while CBC had four. Event discretization for rainfall accumulation histograms was completed for the RWH (2-

hours, left) and the LSVFS (1-hour, right) (Figure 2-10). The lower threshold for discretization inflates the occurrence of smaller events; however, CBC observed more large events than TFC.

Table 2-6. TFC and CBC Rainfall Summary

Site	*Metric*	*Depth (mm)*	*Duration (hrs)*	*5-minute Peak Intensity (mm/hr)*	*Antecedent Dry Period (days)*
TFC	*Range*	0.25-62.5	1.2-20.0	3.1-30.5	0.04-16.4
	Median	1.27	4.3	7.6	0.79
	Sum	225	-	-	-
CBC	*Range*	0.25-96.5	1.0-22.8	3.1-74.5	0.02-6.82
	Median	0.25	3.7	3.3	0.61
	Sum	396	-	-	-



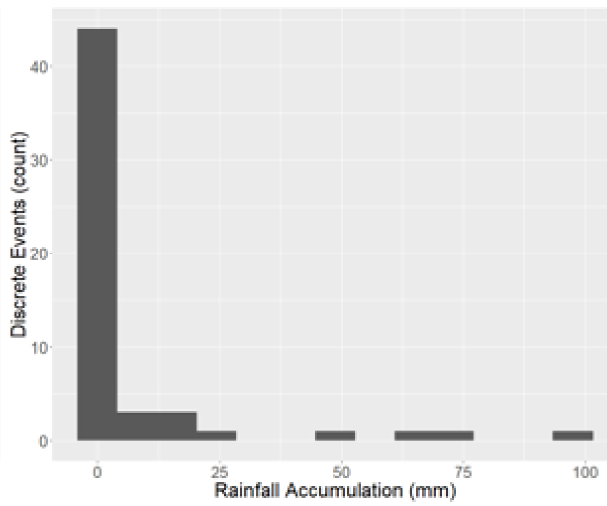


Figure 2-10. Discrete Event Rainfall Accumulation TFC (Left) CBC (Right)

2.4.2. Thermal Mitigation provided by SCM

2.4.2.1. Rainwater Harvester

The warmest temperatures consistently occurred at the top of the RWH (Table 2-7; Figure 2-11), while the coolest temperatures occurred at the bottom. Figure 2-11 illustrates the maximum temperatures observed at the middle (1.5-m) and top (1.0-m) of the tank. A 2-day lag between peak air temperature and that of water in the tank is apparent. This is attributed to direct solar radiation on the black tank surface and conduction to the water inside the tank likely exceeded convection to the atmosphere.

Table 2-7. Rainwater Harvester Event Temperatures at Each Measurement Depth

Month	*Median Temp (Depth 1.0-m) (°C)*	*Maximum Temp (Depth 1.0-m) (°C)*	*Median Temp (Depth 1.5-m) (°C)*	*Maximum Temp (Depth 1.5-m) (°C)*	*Median Temp (Depth 2.0-m) (°C)*	*Maximum Temp (Depth 2.0-m) (°C)*
August	23.7	24.8	23.3	24.3	22.2	23.0
September	21.6	24.6	21.5	23.5	21.0	21.5
October	21.3	22.6	21.1	22.2	20.6	21.3
November	14.7	17.1	14.4	16.5	12.8	16.1

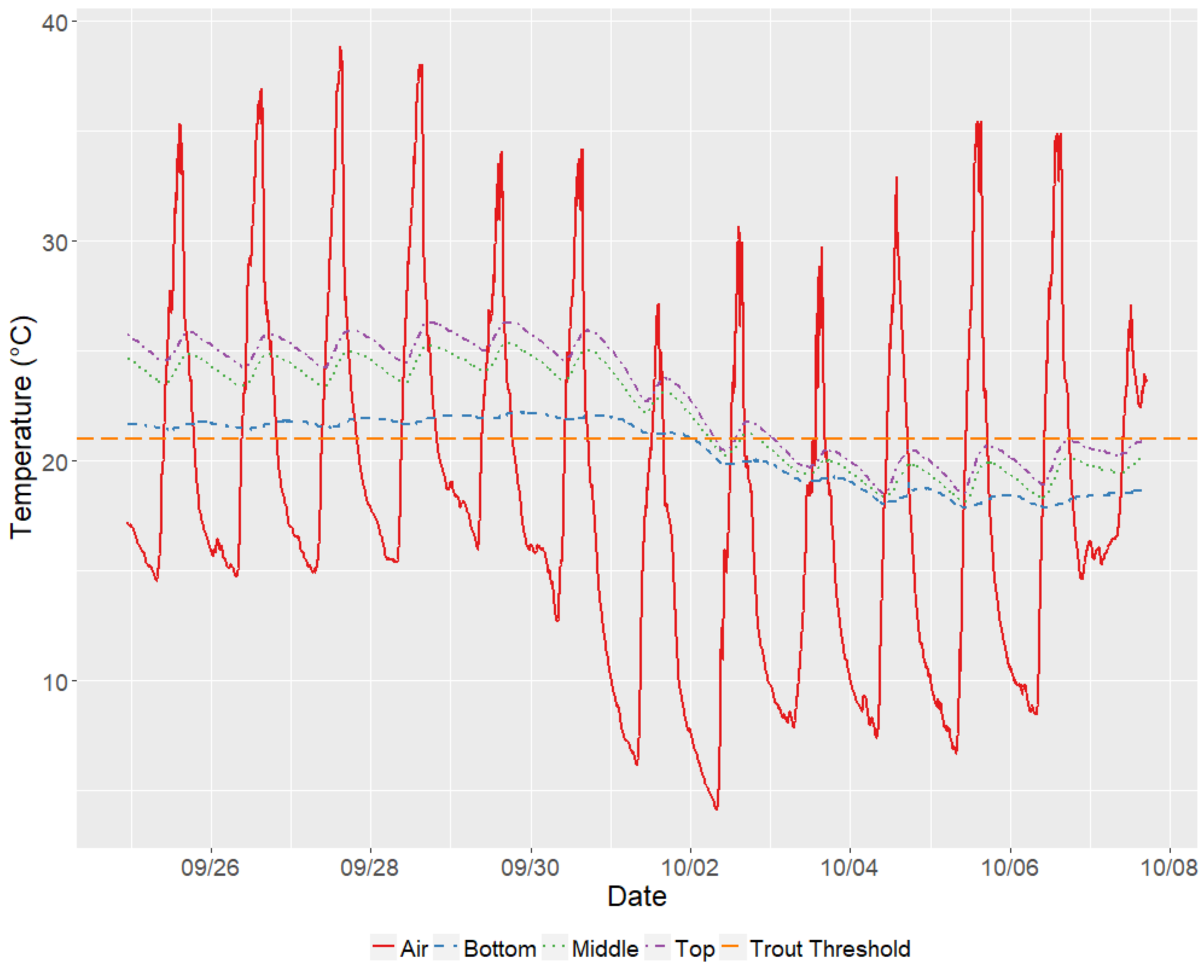


Figure 2-11. RWH Maximum Temperature Event

Figure 2-11 also highlights the stability of temperature and diurnal variation between depths within the tank from September 25—September 30, 2017. A decrease in thermal stratification occurs at the onset of October as air temperatures decrease and water temperatures within the RWH equalize. The RWH pump was installed on October 4, 2017, and could have contributed to tank mixing and temperature equalization across tank depths. Thermal stratification of a wet pond was observed by Jones & Hunt (2010). In stratified waterbodies where fish habitation is valued, aeration or similar disturbances are employed to break stratifications providing more habitable conditions for fish species (Novotny, 2003).

Figure 2-12 plots temperatures and cumulative rainfall over the monitored period. The degree of thermal stratification varied throughout the monitoring period; tank water temperature equalization occurred when cooler temperatures followed hotter periods. Tank volume reduction, either passive drawdown or designated use, should perturb tank volume and temperature profile preventing stratification persistence; this was displayed on October 4, 2017, (Appendix B) when a pump was installed and operated.

Figure 2-12. RWH Temperature & Rainfall Plot

Event median and maximum temperatures at the tank bottom were significantly cooler than the 21ºC trout avoidance threshold with 95% Confidence Intervals (CI) of 16.0 – 20.3ºC and 16.1 – 20.9ºC, respectively. Whereas, at the middle and top depths, temperatures were less frequently below the 21ºC trout threshold. The NRCS Curve Number Method was used to estimate that a 2.38-mm event was needed to fill the wet conveyance volume (Appendix A; Figure 2-4) and begin entering the tank, provided a sufficiently long Antecedent Dry Period (ADP). Several small (<2.38-mm) events that occurred after November 5, 2017, produced inflow to the tank and overflow due to short (< 4-days) ADP (Appendix B).

Analysis was conducted for events occurring before October 12, 2017, and events that occurred after (Figure 2-13); this division in the data was apparent due to the lack of water temperatures in exceedance of the trout threshold and represented the end of thermal-enrichment concerns (Figure 2-13). Considering only events (n=7) that (1) were greater than or equal 2.38-mm and (2) occurred before October 12, 2017, median and maximum temperatures from all 3 monitoring points were not significantly different from 21°C (Figure 2-14).

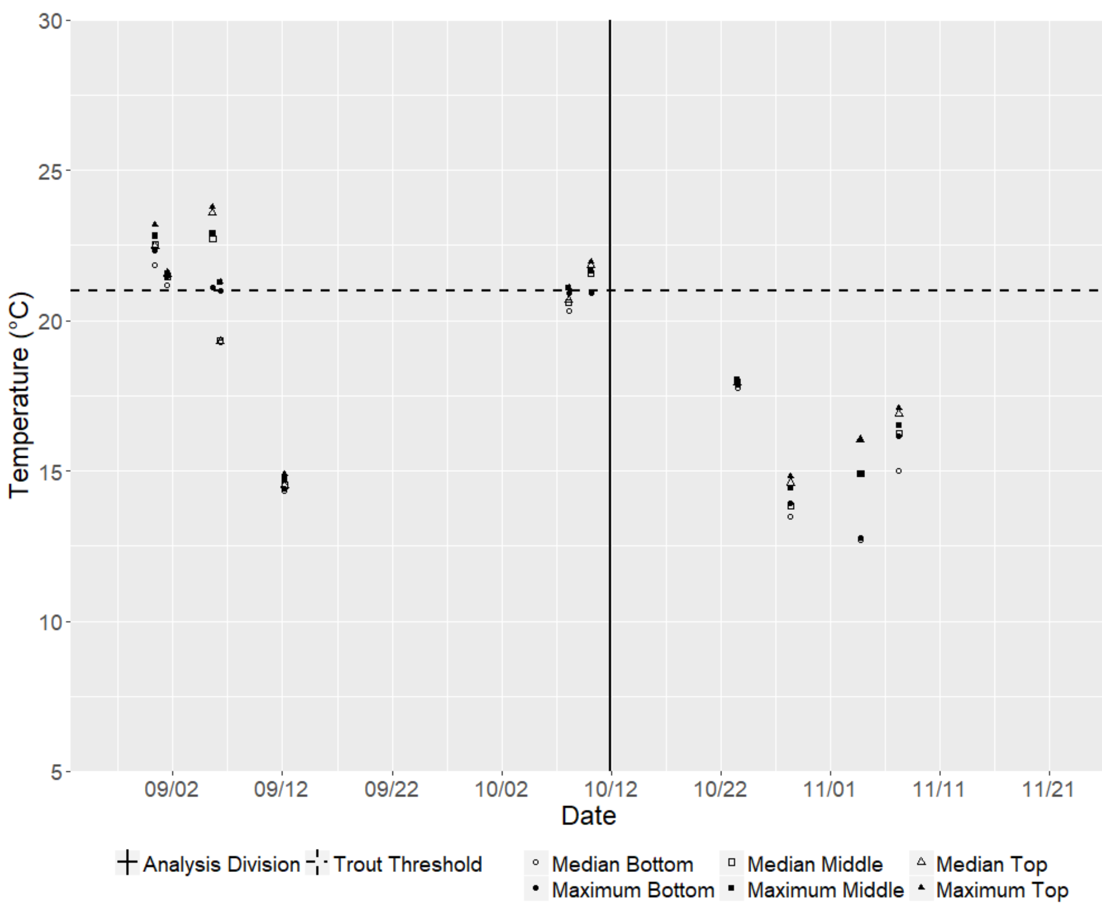


Figure 2-13. RWH Temperatures; Events ≥ 2.38 mm, August 28, 2017—September 22, 2017

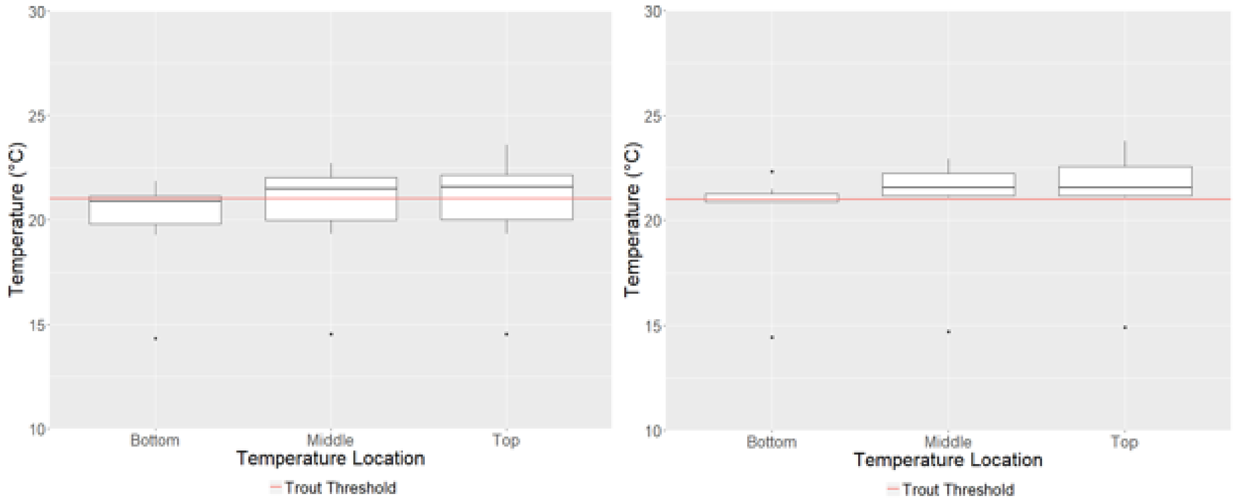


Figure 2-14. Median (Left) & Maximum (Right) RWH Temperatures; Events ≥ 2.38 mm Before October 12, 2017

Median temperatures at the bottom measurement point were significantly lower than those of the top and middle measurement locations; the middle and top depths were not significantly different. This is the result of thermal stratification within the RWH. The 95% CI of the median difference of bottom and top temperatures ranged from 0.2 – 1.5°C. Maximum event temperatures also differed significantly across the same measurement depths and exhibited similar temperature differences. Temperature variance (within event) is significantly different between middle and top monitoring locations. No such differences were observed (statistically) in other comparisons of variance. This likely results from mixing of influent nearest the inlet.

After October 12, 2017, events greater than or equal 2.38-mm (n=4) had median and maximum event temperatures that were significantly colder than 21°C (Figure 2-15); location of measurement was not significantly different for either median or maximum temperatures. Sensor depth for the monitoring period generally remained at depths associated with maximum tank capacity (set by the overflow invert), aside from short deviations above this elevation when overflow occurred (Figure 2-4; Appendix B). RWH storage tank passive release (Gee & Hunt, 2016) installation was not completed during the monitoring period; however, on October 4, 2017, a pump was installed to deliver water to a nearby garden. Tank depth decreased as the pump was tested; on October 8, 2017, a 182.9-mm rainfall event refilled the tank to maximum capacity and produced overflow (Appendix B).

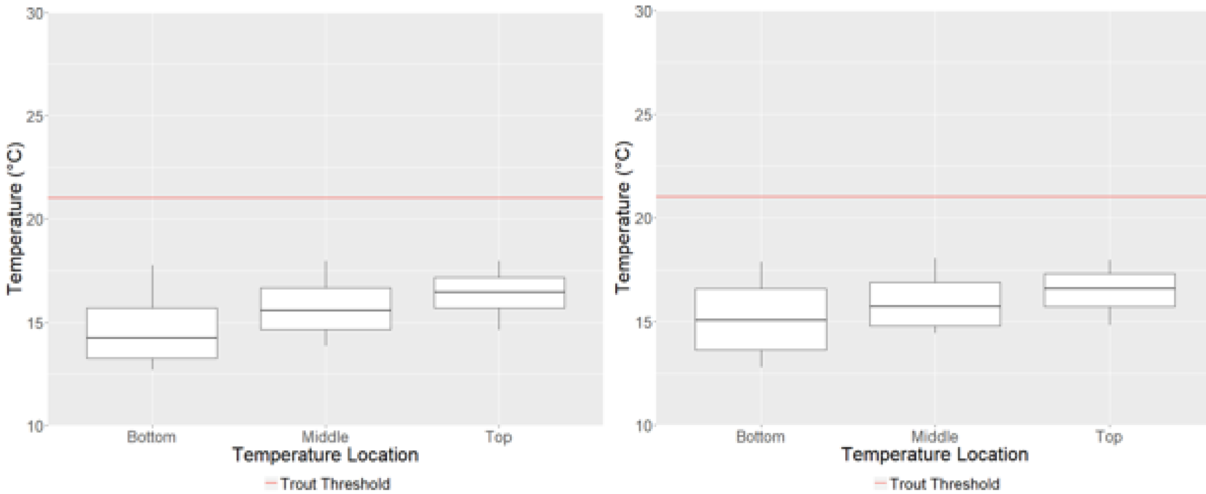


Figure 2-15. Median (Left) & Maximum (Right) RWH Temperatures; Events ≥ 2.38 mm After October 12, 2017

To quantify volume reduction via slow emptying of the wet conveyance, storage tank outflow was modeled as orifice discharge to the atmosphere; this is likely an underestimation, per system design (Appendix A). Orifice discharge was calculated using Eq. 2.3 & Eq. 2.3.1 (NCEES, 2013).

$$Q = C_d A \sqrt{2gh} \qquad \text{(Eq. 2.3)}$$

Where,

Q = orifice discharge (m^3/s)

C_d = coefficient of discharge (unitless), 0.51

A = office cross-sectional area (m^2)

g = gravity acceleration constant, 9.807-m/s^2

h = head above orifice center (m)

If the stage above the orifice was below the orifice center, discharge was calculated as follows:

$$Q = 4.464 C_d D z^{1.5} \quad \text{(Eq. 2.3.1)}$$

Where,

Q = orifice discharge (cfs)

C_d = coefficient of discharge (unitless), 0.51

D = office diameter (ft)

z = head above orifice invert (ft)

The median volume reduction before and after October 12, 2017, was 97% (Table 2-8); cumulative volume reductions was 96%. Substantial reductions in runoff volume were produced by (1) an initially empty tank, (2) one major drawdown during the pump installation, and (3) wet conveyance passive dewatering. The large wet conveyance volume (6.18-m^3) increased total storage capacity by 66% (Table 2-2). The estimated wet conveyance volume can capture runoff from a 2.38-mm rainfall event, 11% of the water quality volume (25.4-mm); however, the wet conveyance represents 40% of total storage capacity that will dewater inter-event becoming available to capture and store subsequent rainfall events. A precise estimation of conveyance drawdown rate is not possible due to unknown elevations; an as-built survey was not conducted.

Table 2-8. RWH Hydrologic Treatment; Events ≥ 2.38 mm Before and After October 12, 2017

Parameter	*Before*	*After*
Cumulative Runoff Volume (m^3)	2,649.4	2,166.0
Cumulative Outflow Volume (m^3)	98.2	106.0
Median Runoff Volume Reduction (%)	97	97
Cumulative Runoff Volume Reduction (%)	96	95

Davis (2008) compared the Rational Method runoff coefficient, C, for imperviousness (C=0.9) and undeveloped land (C=0.3) to justify a Low Impact Development (LID) target metric of 33% inflow volume reduction. By this metric, the studied RWH betters the LID target; Wardynski et al. (2014) emphasizes that meeting LID hydrologic goals is a surrogate for mitigating thermal loads. Achieving LID hydrology sufficiently supports salmonids.

Median rainfall event accumulation for the monitoring period was 1.27-mm with a median 0.79-day ADP (Table 2-6), small median rainfall accumulations likely allowed sufficient time for the wet conveyance to completely dewater, becoming available for the next rainfall event; maybe somewhat drier conditions (Table 2-5) improved RWH treatment.

Thermal load reductions are used to document temperature mitigation provided by SCMs, accounting for both reductions in temperature and volumes (Jones et al., 2012; Wardynski et al., 2013, 2014; Winston et al., 2011). Thermal load estimations (Eq. 2.2) for events before and after October 12, 2017, were calculated using the median event temperature at the top measurement depth estimated influent volumes, and calculated outflow volumes (Wardynski et al., 2014; Winston et al., 2011) (Table 2-9).

Table 2-9. RWH Thermal Load; Events ≥ 2.38 mm Before and After October 12, 2017

Parameter	*Before*	*After*
Cumulative Inflow Thermal Load (MJ)	3,258,586.5	2,622,664.0
Cumulative Outflow Thermal Load (MJ)	120,787.8	128,378.8
Median Thermal Load Reduction (%)	97	97
Cumulative Thermal Load Reduction (%)	96	95

Applying median event temperatures (1.0-m below the outlet invert) may be an underestimation of effluent thermal loads if temperature stratifications maintained higher than median temperatures near the outlet invert. Due to substantial runoff volume reductions, thermal load reductions were likewise substantial (97%) (Table 2-9); cumulative thermal load reduction was 95%. The impact of effluent thermal loads on salmonids will be dependent on in-stream flows and mixing (Wardynski et al., 2014).

2.4.2.1.1. Summary & Discussion

Rainwater harvesting is a SCM proven to reduce nutrient export and runoff volumes (Doan & Davis, 2017; Gee & Hunt, 2016; Jones & Hunt, 2010; Pelak & Porporato, 2016). RWHs can also reduce potable water consumption reducing the demand for centralized treatment and distribution systems (Gee & Hunt, 2016; Jones & Hunt, 2010; Steffen et al., 2013). In this study, a wet conveyance RWH displayed thermal stratification with significantly different temperatures at all measured depths, likely from underutilization; previous studies have cited over estimations of water demands, logistical challenges, operational changes, and public perception as causes for underutilization so that RWH storage tanks remain near maximum capacity (Jones & Hunt, 2010). The designated use for harvested water at TFC is watering a teaching garden; however, the monitoring period, August 28, 2017—September 22, 2017, took place near the end of the growing season and water demands were minimal. Delivery pump installation was not completed until one-half of the monitoring period had passed. Stormwater hydrologic mitigation depends on RWH storage tank capacity available for runoff capture during rainfall events. Passive and active release systems are shown to improve the hydrologic performance of RWH systems by creating separate detention and retention volumes; retention volumes are available for designated use while detention volumes are slowly released over several days and provide capacity to capture runoff during the next rainfall event (Gee & Hunt, 2016; Jones & Hunt, 2010; Steffen et al., 2013).

The RWH herein employed a wet conveyance system to deliver rooftop runoff to a RWH storage tank. The wet conveyance likely reduced temperatures reaching the RWH; previous research has documented significant temperature reductions in subsurface conveyance networks, up to 7^{o}C across a 168-m metal corrugated pipe culvert (Jones & Hunt, 2010). The wet conveyance was equipped with a passive drawdown which created a detention volume equal to 40% of total system capacity. Hydrologic mitigation of events less than 2.38 mm was achieved for low median rainfall event size (1.27-mm) and a sufficiently long median ADP (0.79-days) which retrieved storage capacity for subsequent rainfall. A moderately drier rainfall record than normal likely contributed to substantial (97%) runoff reduction. The RWH system bested the LID target (33%) for inflow volume reduction and by this metric is assumed successful at thermal mitigation (Davis, 2008; Wardynski et al., 2014).

Event median and maximum temperatures were compared to 21^{o}C (Coutant, 1977; Ferguson, 1958; Jones & Hunt, 2010), the avoidance threshold for brook trout, to analyze the RWH's thermal mitigation. Analysis was conducted for data collected before and after October 12, 2017; a division within the data representing the end of the warmer summer period. Event median and maximum temperatures at all measurement depths (1.0-,1.5-, and 2.0-m) were not significantly different from the 21^{o}C threshold; however, the passive release produced substantial thermal load reductions of 97% for events both before and after October 12, 2017. Regular usage of harvested water or passive release on the storage tank would increase runoff capture and reduce outflow and thermal loads further. The black RWH storage tank exhibited diel temperature variation likely for direct solar heating. Designers, concerned with thermal impacts, could improve thermal performance by siting the RWH tank in cool shaded areas or provide cover or shade structure to prevent solar gain, or use non-black materials.

2.4.2.2. Level Spreader-Vegetative Filter Strip

The warmest median temperatures occurred in August while the warmest maximum temperatures occurred in September (Table 2-10). The thermal dynamics of this LSVFS are illustrated when examining an event that on September 5, 2017 (Figure 2-16). Air, inlet, and outlet temperatures cooled as rainfall began, followed by a temperature spike of approximately 1^{o}C at the inlet and 3^{o}C at the outlet; similar thermal spikes at the onset of runoff were observed by Jones & Hunt (2010). The inlet blind swale is "armored" with rip-rap, capable of storing large amounts of thermal energy (Figure 2-8) (Akbari & Konopacki, 2005), which confounds the true temperature of inflow. Alternatively, the outlet monitoring trough had minimal thermal mass, and thus, exhibited a larger thermal spike. Throughout the morning of September 6, 2017, the outlet monitoring trough maintained a depth of approximately 11-cm; ponded water was likely warmed from direct solar radiation heating the black, impermeable trough liner (Figure 2-8). By midday, additional rainfall and runoff flushed the trough, cooling the discharge. Pulses of runoff entered the LSVFS throughout the morning of September 6, 2017, which minimally changed temperatures at the inlet sensor. Runoff around 10AM on September 6, 2017, overtopped the level spreader and produced a modest 1^{o}C rise in inlet temperature while cooling the outlet monitoring trough (Figure 2-16).

Table 2-10. LSVFS Median and Maximum Influent and Effluent Temperatures

Month	*Median Inlet Temperature (^{o}C)*	*Maximum Inlet Temperature (^{o}C)*	*Median Outlet Temperature (^{o}C)*	*Maximum Outlet Temperature (^{o}C)*
August	22.2	25.5	22.6	24.7
September	19.7	28.0	17.9	29.3
October	19.9	23.5	19.5	22.8
November	13.5	20.2	15.5	19.6

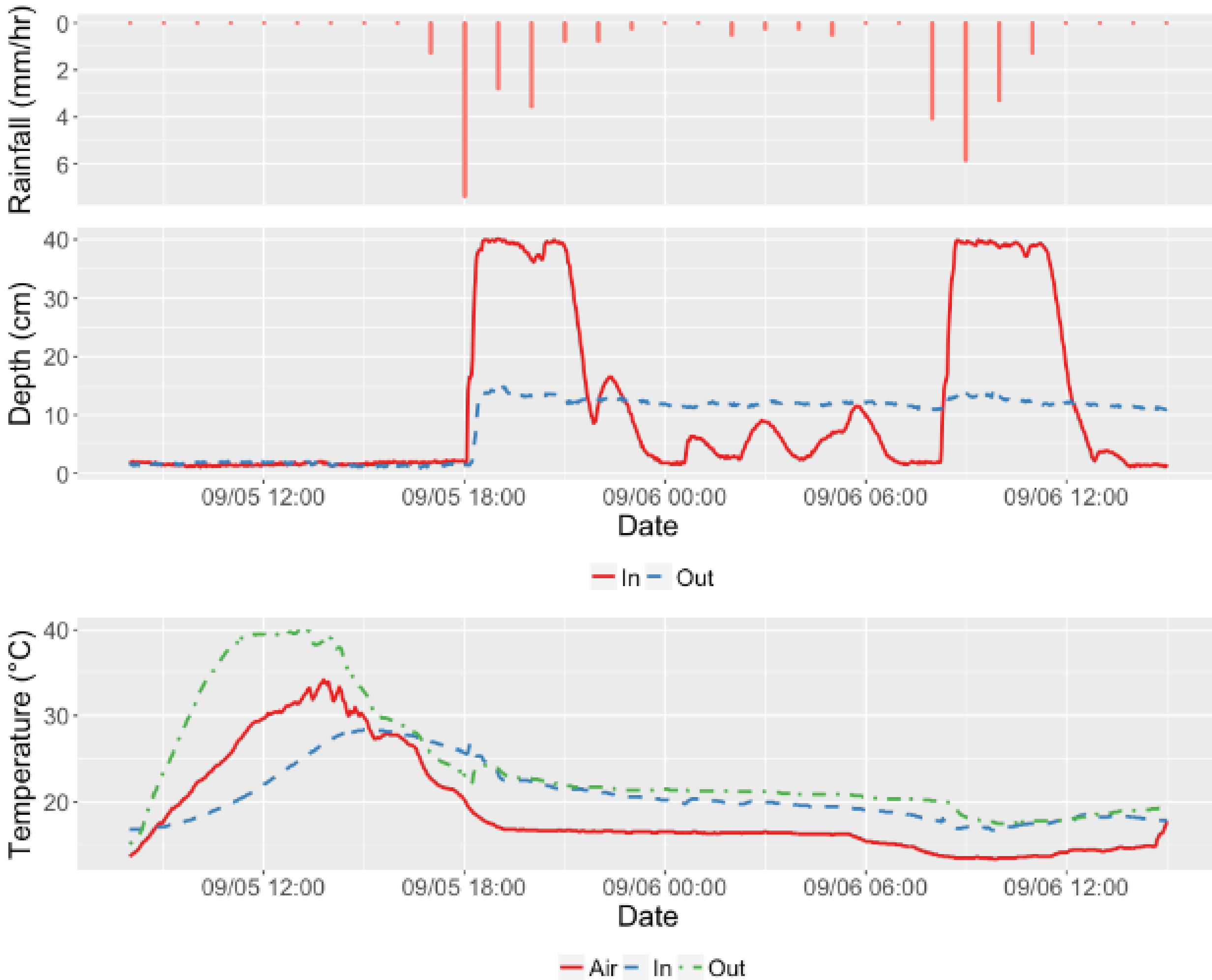


Figure 2-16. September 5, 2017, Thermograph; Rainfall (Top), Depth (Middle) and Temperature (Bottom) at Monitoring Locations

On August 31, 2017, a 16.0-mm event was followed by a 0.25-mm event; these overtopped the level spreader (Figure 2-17). A strong thermal flush was not observed; inlet temperatures remain within approximately 1°C of outlet temperatures throughout the event. This may be due to under-drainage of the blind swale limiting overtopping of the level spreader.

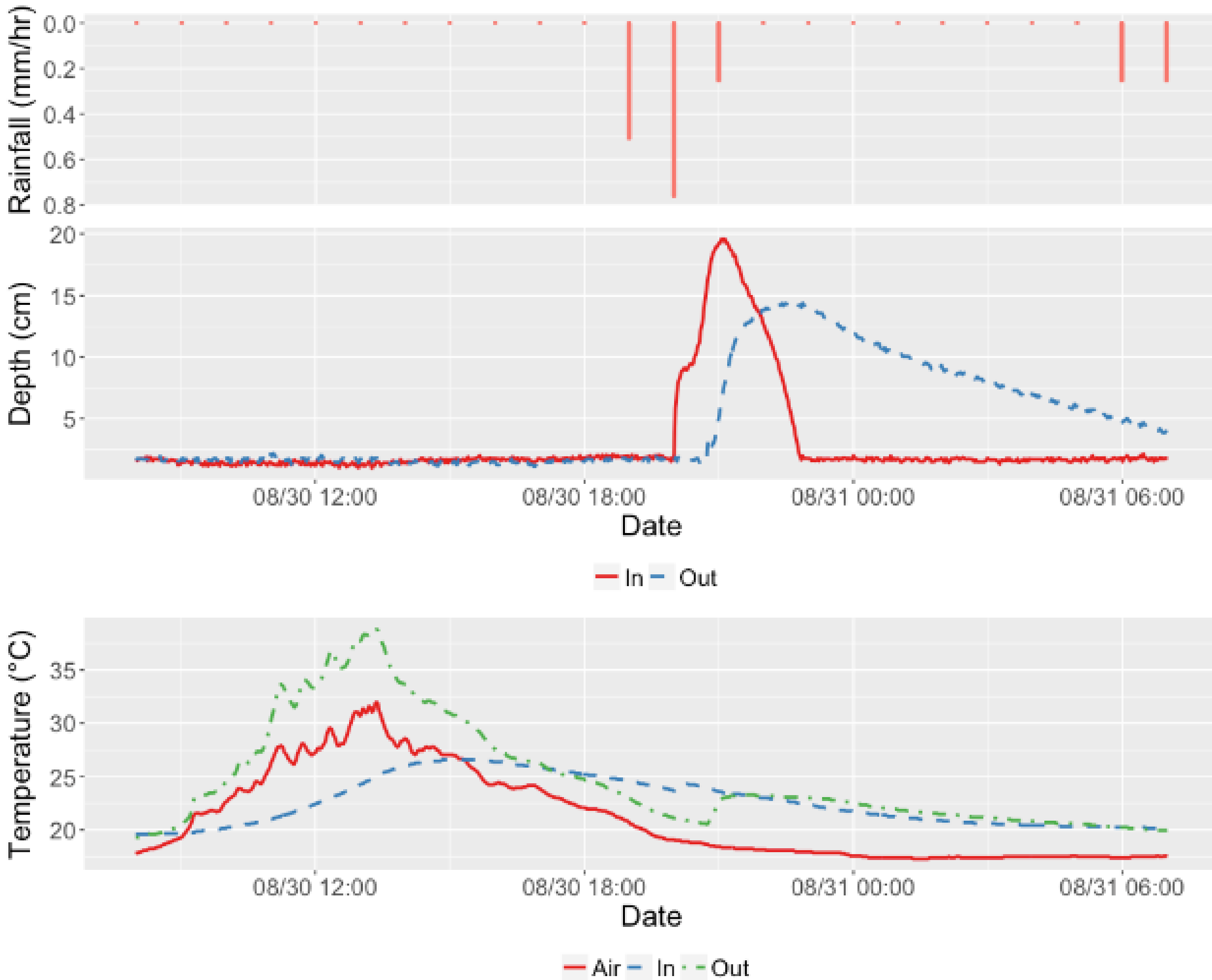


Figure 2-17. August 31, 2017, Thermograph; Rainfall (Top), Depth (Middle) and Temperature (Bottom)

The NRCS Curve Number Method (Eq. 2.1 – 2.1.1) was used to estimate the minimum event required to produce runoff and overtop the level spreader, 1.89 mm, for the contributing area (CN of 95)(NCDEQ, 2017). This estimation includes 0.67 mm of initial abstraction and 1.22 mm to fill the inlet blind swale and is likely an underestimation. Runoff volumes that do not exceed the inlet swale's storage capacity (3.39-m^3) are dewatered via underdrainage and discharged to the outlet monitoring trough (Figure 2-8). Events that were dewatered via under-drainage are likely thermally mitigated from contact with subsoil and subsurface conveyance; however, underdrainage temperatures were not collected.

Only comparing maximum and median event temperatures (as done herein), provides limited insight to thermal load reduction of this practice (Wardynski et al., 2014); however, the lack of reliable hydrologic data limits usable metrics. Twenty-three discrete events produced precipitation exceeding 0.67-mm, the estimated threshold to produce runoff, while only 15 events generated precipitation greater than or equal to 1.89-mm, the estimated depth required to overtop the level spreader. Events were divided into those occurring before October 12, 2017, and events that occurred after; this division represents the end of peak thermal-enrichment concerns due to the lack of runoff temperatures exceeding the trout threshold (Figure 2-18). Ten events produced greater than or equal to 1.89-mm prior to October 12, 2017. A scatter plot of inlet and outlet event median and maximum temperatures (Figure 2-18) illustrates the seasonal decrease in runoff temperatures.

For events greater than or equal to 1.89-mm before October 12, 2017, median and maximum inlet temperatures were not significantly different from 21^{o}C with a 95% CI of 19.2—22.4^{o}C and 20.4—25.0^{o}C, respectively. Similarly, median and maximum outlet temperatures were not significantly different from 21^{o}C with a 95% CI of 19.6—22.8^{o}C and 21.0—25.3^{o}C, respectively (Figure 2-19).

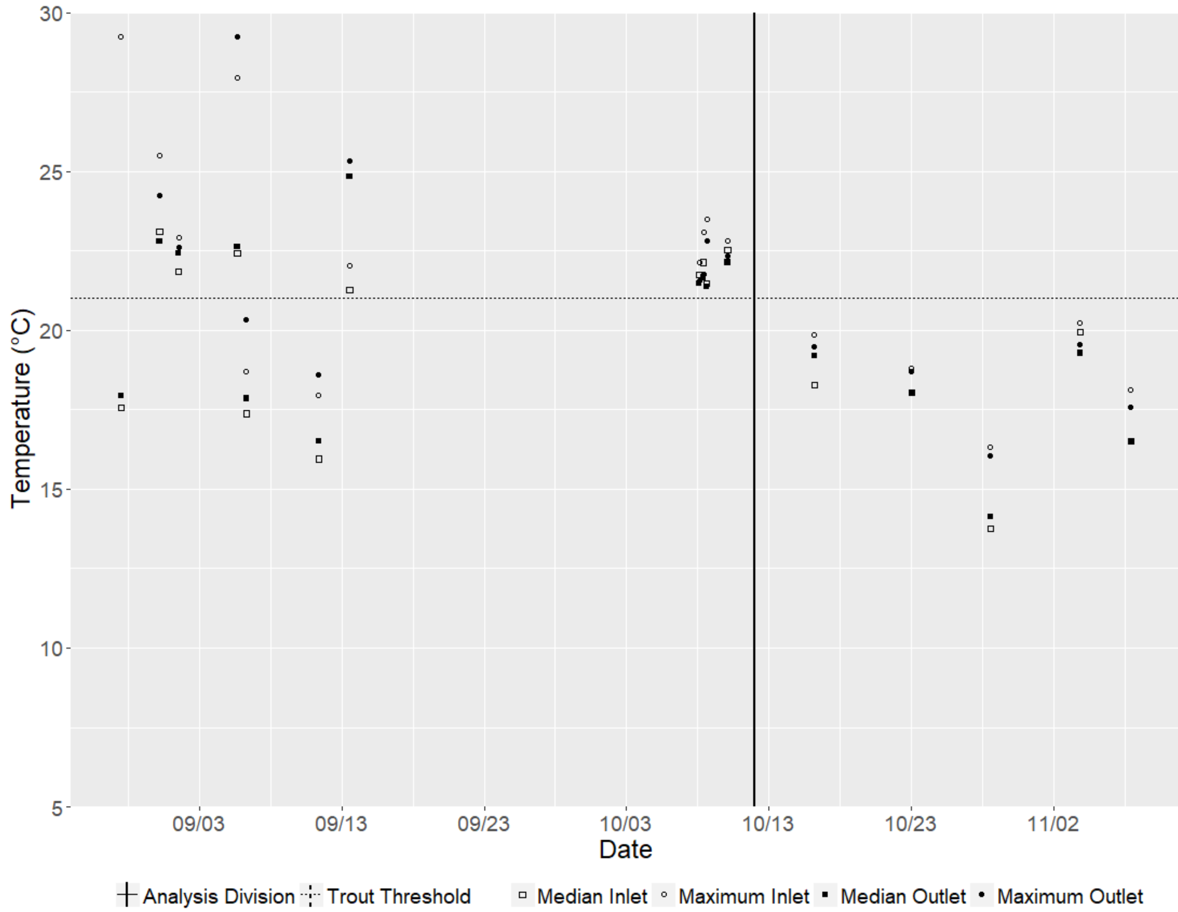


Figure 2-18. LSVFS Influent and Effluent Temperatures; Events ≥ 1.89 mm, August 28, 2017—September 22, 2017

Boxplots (Figure 2-19) highlight the distribution of median and maximum event temperatures in relation to the trout threshold of 21ºC (Coutant, 1977; Ferguson, 1958). Inlet temperatures (maximum and median) were 0.1ºC lower than those of the outlet; the difference was not significant.

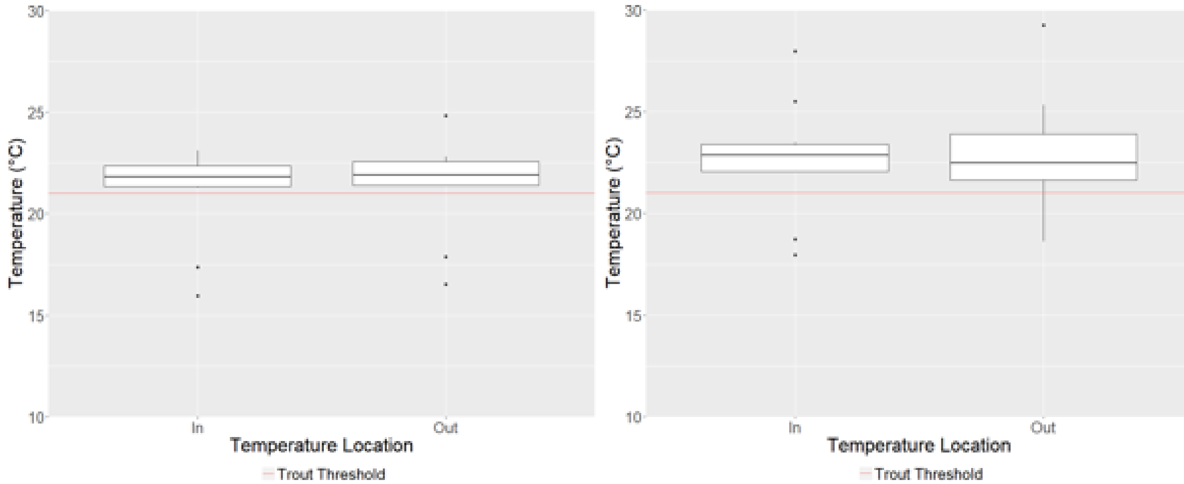


Figure 2-19. Median (Left) and Maximum (Right) Temperatures; Events ≥ 1.89 mm Before October 12, 2017

Median and maximum event temperatures were minimally impacted by LSVFS treatment; there was insufficient evidence to suggest effluent temperatures were more consistent as a result of LSVFS treatment, results seen in BRC monitoring studies (Jones & Hunt, 2009). Significant maximum temperature reductions were observed for both LSVFSs (7.6-m and 15.2-m) studied by Winston et al. (2011). The 15.2-m LSVFS studied by Winston et al. (2011) included 7.6-m of wooded buffer which showed improved median event temperature reductions over the shorter LSVFS; possibly displaying the benefit of additional length or shading (Winston et al., 2011). The LSVFS studied herein had 1.6 times the surface area (138.7-m^2)(Table 2-3) compared to the 15.2-m (86.5-m^2) LSVFS studied by Winston et al. (2011). The LSFVS studied herein is was in full sun (Figure 2-3). Winston et al. (2011) monitoring studied collected approximately 11-months of data during thermal stress periods between May and September in 2008 and 2009; more robust data was likely more effective at detecting changes in temperatures.

Events greater than or equal to 1.89-mm after October 12, 2017, were significantly (p-value = 0.0315) cooler than 21^{o}C for median and maximum inlet and outlet temperatures (Figure 2-20). Temperature variance was not significantly different after LSVFS treatment for events after this date.

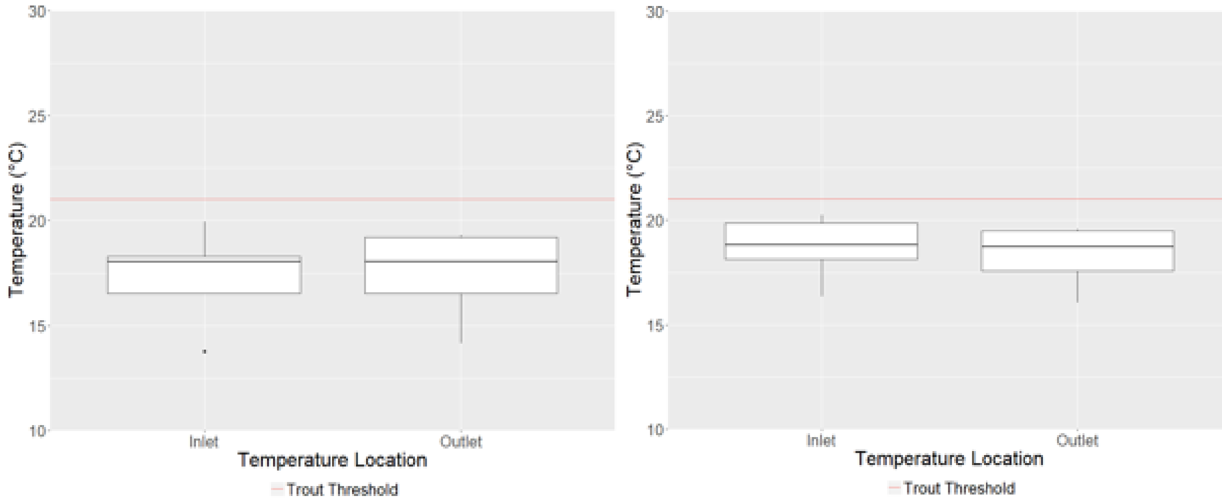


Figure 2-20. Median (Left) and Maximum (Right) Temperatures; Events ≥ 1.89 mm After October 12, 2017

A final illustration of how this LSVFS did not mitigate thermal pollution is found in Figure 2-21. Eighty percent of events greater than or equal to 1.89-mm before October 12, 2017, exceeded 21^{o}C at the inlet and outlet for both event median and instantaneous maximum temperatures.

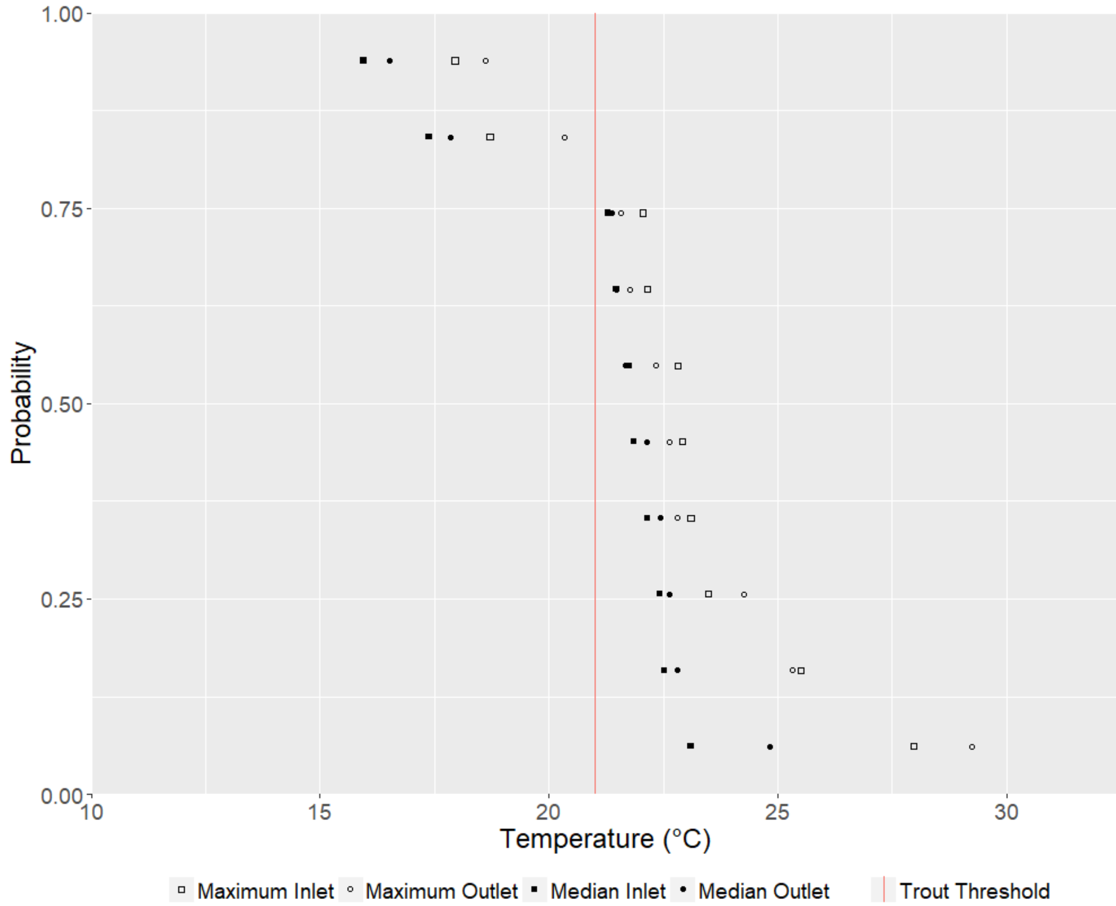


Figure 2-21. Inlet and Outlet Temperature Probability Exceedance; Events ≥ 1.89 mm Before October 12, 2017

2.4.2.2.1. Summary & Discussion

Monitoring of a 17.4-m LSVFS was conducted from August 28, 2017—November 22, 2017. Instantaneous influent and effluent temperatures were collected within the inlet blind swale and outlet monitoring trough; accurate hydrology data were not collected. Temperatures were compared to 21^{o}C (Coutant, 1977; Ferguson, 1958), the upper avoidance threshold for brook trout. Analysis of instantaneous maximum event and median event temperatures is most useful in conjunction with downstream temperature standards (Wardynski et al., 2014); however, in the absence of more robust data, this analysis provides useful insight to the performance of LSVFS as a thermal mitigation strategy.

Analysis was conducted for events greater than or equal to 1.89-mm collected before and after October 12, 2017; a division within the data representing the end of thermal impacts due to the lack of runoff temperatures exceeding the trout threshold. The LSVFS had minimal effect on runoff temperatures. The ability to discern LSVFS performance may have been diminished by monitoring location; a smaller LSVFS (86.5 m^2) studied by Winston et al. (2011) vis-à-vis the LSVFS herein (138.7 m^2) significantly reduced influent temperatures when a portion of the LSVFS was wooded and shaded. The system herein was fully exposed to the sun. The inlet blind swale was "armored" with rip-rap which showed some contribution of thermal energy to influent. Shading is thus apparently important to limit thermal export. Similarly, Jones et al. (2012) found tree canopy shading parking lot areas cooled median and maximum runoff temperatures by 0.3^{o}C and 3.2^{o}C, respectively. Shading of the filter strip and inlet blind swale will reduce soil and material temperatures, increasing the overall temperature reduction potential. Alternatively, inlet protections other than large amounts of rip-rap may improve thermal reductions. If thermal-enrichment in cold-water streams is a concern, results from this LSVFS do not support its selection as a thermal reduction SCM relative to other practices.

2.4.2.3. Bioretention Cell

Table 2-11. Bioretention Monthly Event Median and Maximum

Month	*Median Temp Inlet (°C)*	*Maximum Temp Inlet (°C)*	*Median Temp (31-cm Depth) (°C)*	*Maximum Temp (31-cm Depth) (°C)*	*Median Temp (53-cm Depth) (°C)*	*Maximum Temp (53-cm Depth) (°C)*	*Median Temp Outlet (°C)*	*Maximum Temp Outlet (°C)*
August[*]	21.2	30.2	23.7	24.5	24.1	24.4	-	-
September	20.3	34.9	20.6	24.5	20.8	22.5	18.4	25.1
October	17.0	33.5	18.8	21.6	19.7	21.9	17.3	25.8
November[*]	14.2	22.0	14.0	15.3	13.9	15.5	13.0	19.0

[*]Incomplete dataset; missing 08/01/2017-08/27/2017 and 11/23/2017-11/30/2017

Typical BRC thermal "behavior" is illustrated by the event on September 5, 2017. The maximum inlet temperature (34.9°C; Table 2-11) occurred at the onset of this 32.8-mm rainfall at 4:00PM (Figure 2-22). Air and inlet temperatures cooled rapidly until inlet temperatures were approximately 5°C warmer than air temperatures. Shallow (31-cm) and deep (53-cm) well temperatures (located in the center of the BRC between the inlet and outlet, Figure 2-9) varied diurnally with the shallow well exhibiting slightly more temperature variance. During this rain event, well temperatures displayed minor temperature fluctuations; the thermal load of influent was effectively "captured" by the BRC, demonstrating excellent treatment. Warmer surface soils have less ability to buffer temperatures than deeper subsoils (Jones & Hunt, 2009). Surface-ponded runoff infiltrated into the media within 24 hours of runoff inception, while the shallow and deep wells needed an additional 24 hours to dewater. Increased residence in the IWS zone will improve seepage, volume reductions, and thermal exchange with subsoils (Brown & Hunt, 2011; Jones & Hunt, 2009; Wardynski et al., 2013).

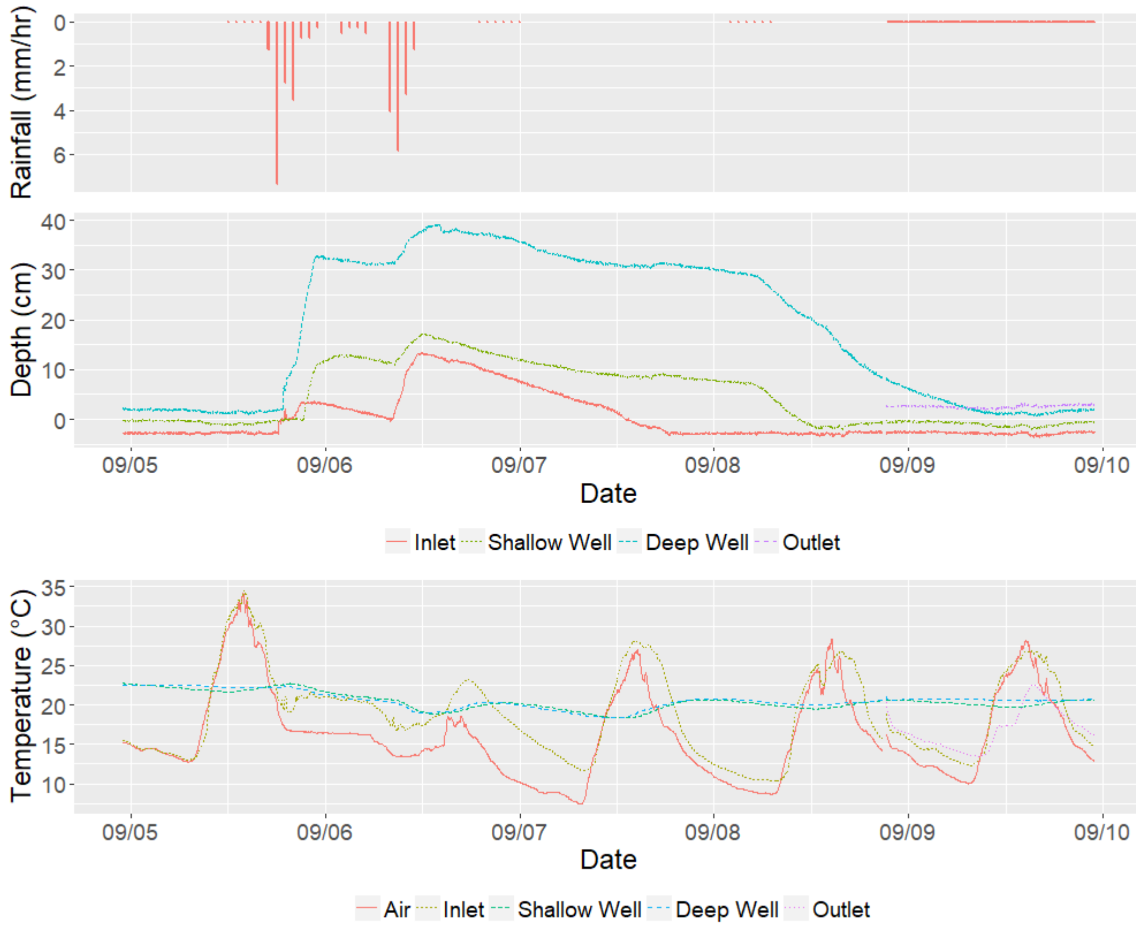


Figure 2-22. September 5, 2017, Thermograph; Rainfall (Top), Sensor Depth (Middle) and Temperature (Bottom)

On September 11, 2017, a 66.3-mm rainfall event occurred followed by a 4.0-mm event two days later. Daytime air temperatures preceding September 11, 2017, had fallen below those of the shallow and deep well by approximately 7ºC, highlighting the thermal lag between air temperatures and soil temperatures (Figure 2-23). Once ponding began, well temperatures decreased towards those of the air; outlet temperatures increased to those of the well, pre-event. This illustrated export of thermal energy stored in the BRC media. The wells began to exhibit diurnal trends after surface ponding and soil saturation occurred due to these back-to-back events. Once ponding ceased, wells and the outlet temperatures returned to minimal diel variability. Deep well (53-cm) temperatures exhibited the least impact from infiltrated runoff and had the quickest return to antecedent temperature; this phenomena was observed in previous field studies on BRC (Jones & Hunt, 2009).

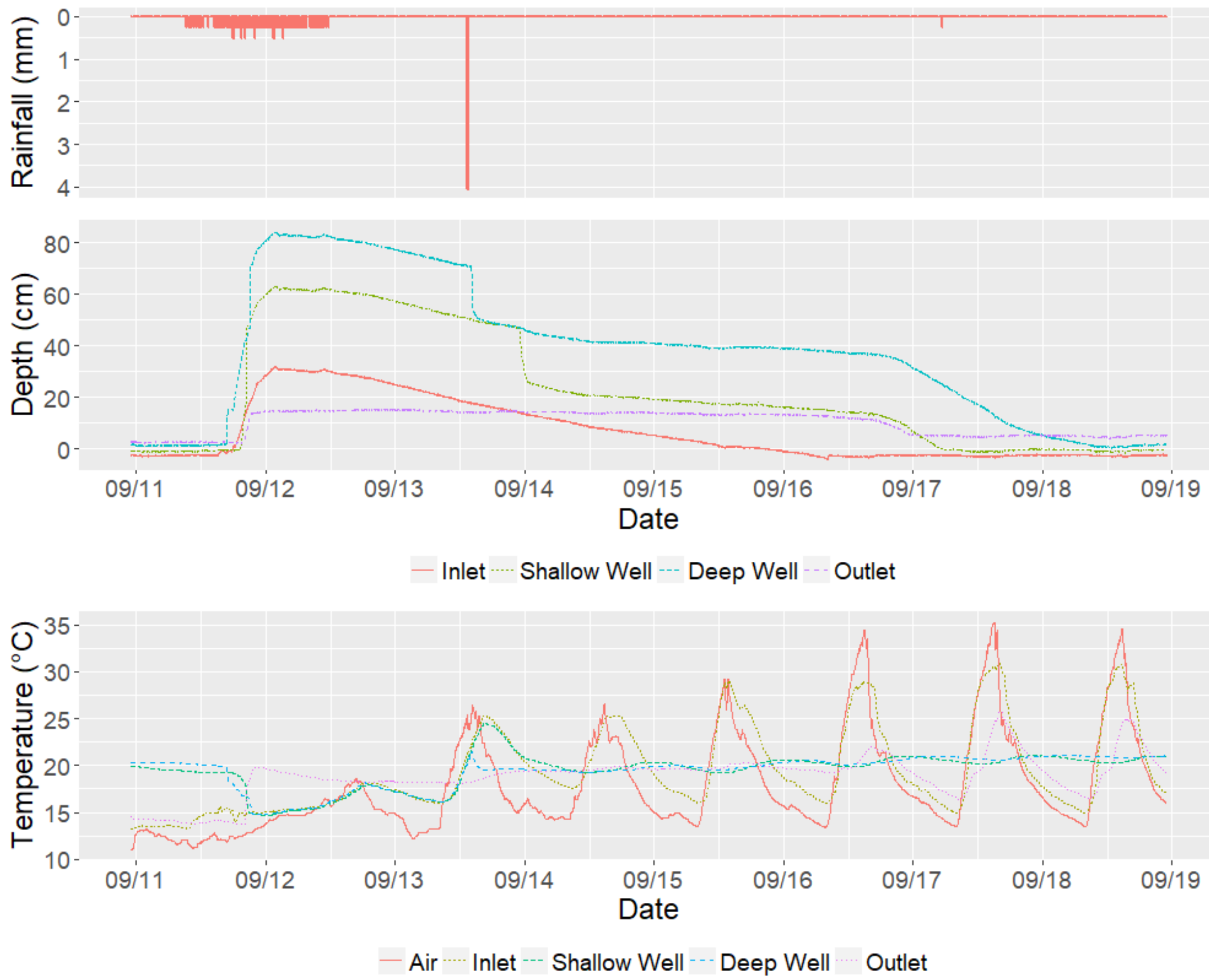


Figure 2-23. September 11, 2017, Thermograph; Rainfall (Top), Sensor Depth (Middle) and Temperature (Bottom)

The September 11, 2017, event is further presented with a temperature-duration plot of instantaneous inlet and outlet temperatures (Figure 2-23); the cummulative duration and the temperature are plotted on the abscissa and ordinate axies, respectively. The smoothed line plot was generated using the *loess* function in the *stats* package for local polynomial regression fitting (R Core Team, 2017) to help vizualize the scatter plot patterns. Temperature-duration plots illustrate the duration of temperatures exceeding a reference value (Wardynski et al., 2014). If long-term reference reach data are available for a salmonid-supporting stream reach, SCM effluent temperature-duration plots can be overlain with those of the reference reach for comparison. Influent temperatures exceeded 21°C for approximately 1 hour while effluent temperatures exceed 21°C for approximately 30 mins (Figure 2-24).

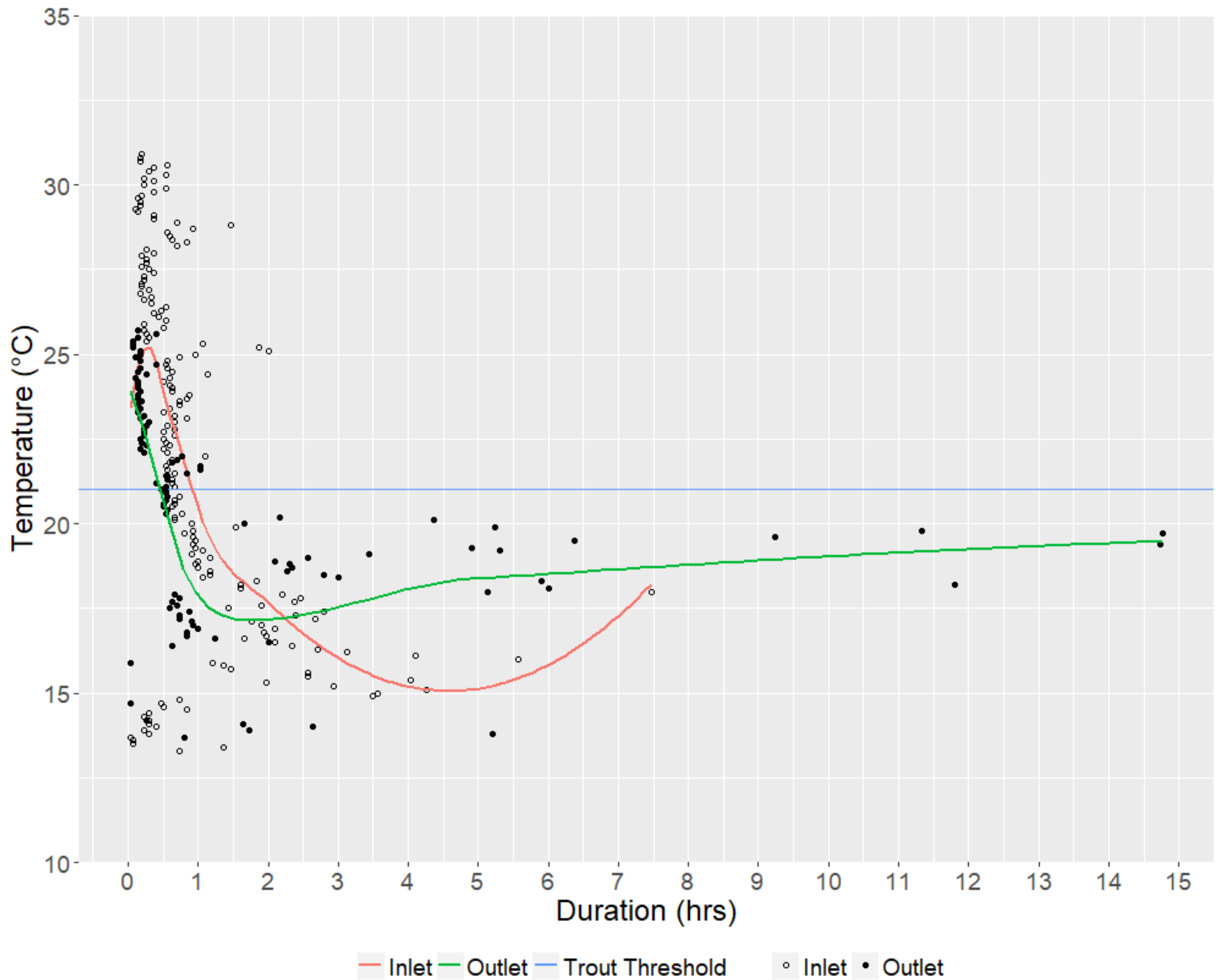


Figure 2-24. September 11, 2017, Temperature-Duration Plot

Instantaneous maximum and median event temperatures were used to determine temperature mitigation provided by this BRC as suggested by Wardynski et al. (2014); temperatures were compared to 21°C (Coutant, 1977; Ferguson, 1958). The NRCS Curve Number method (Eq. 2.1 – 2.1.1) was used to calculate precipitation depth required to generate runoff (CN of 88, Table 2-4)(NCDEQ, 2017); rainfall events less than or equal to 1.70-mm, the calculated threshold to generate runoff, were removed from the dataset. No runoff-generating events occurred between September 13, and October 5, 2017. Events were divided into two groups; those occurring before October 12, 2017, and those after (Figure 2-25). The selection of this date was previously justified. No influent or effluent temperatures exceeded 21^{o}C after October 12, 2017.

Six discrete events produced rainfall amounts exceeding 1.70-mm prior to October 12, 2017 (Figure 2-25). Median and maximum inlet temperatures were not significantly different from 21^{o}C. Additionally, median and maximum outlet temperatures were not significantly different from 21^{o}C with an 88% CI of 18.4—21.4^{o}C and 19.1—21.5^{o}C, respectively (Figure 2-26). The lack of significance is attributed to a small dataset (6 events). Moreover, only effluent 4 events were captured. The differences between inflow and outflow temperatures (both median and maximum) were not significant.

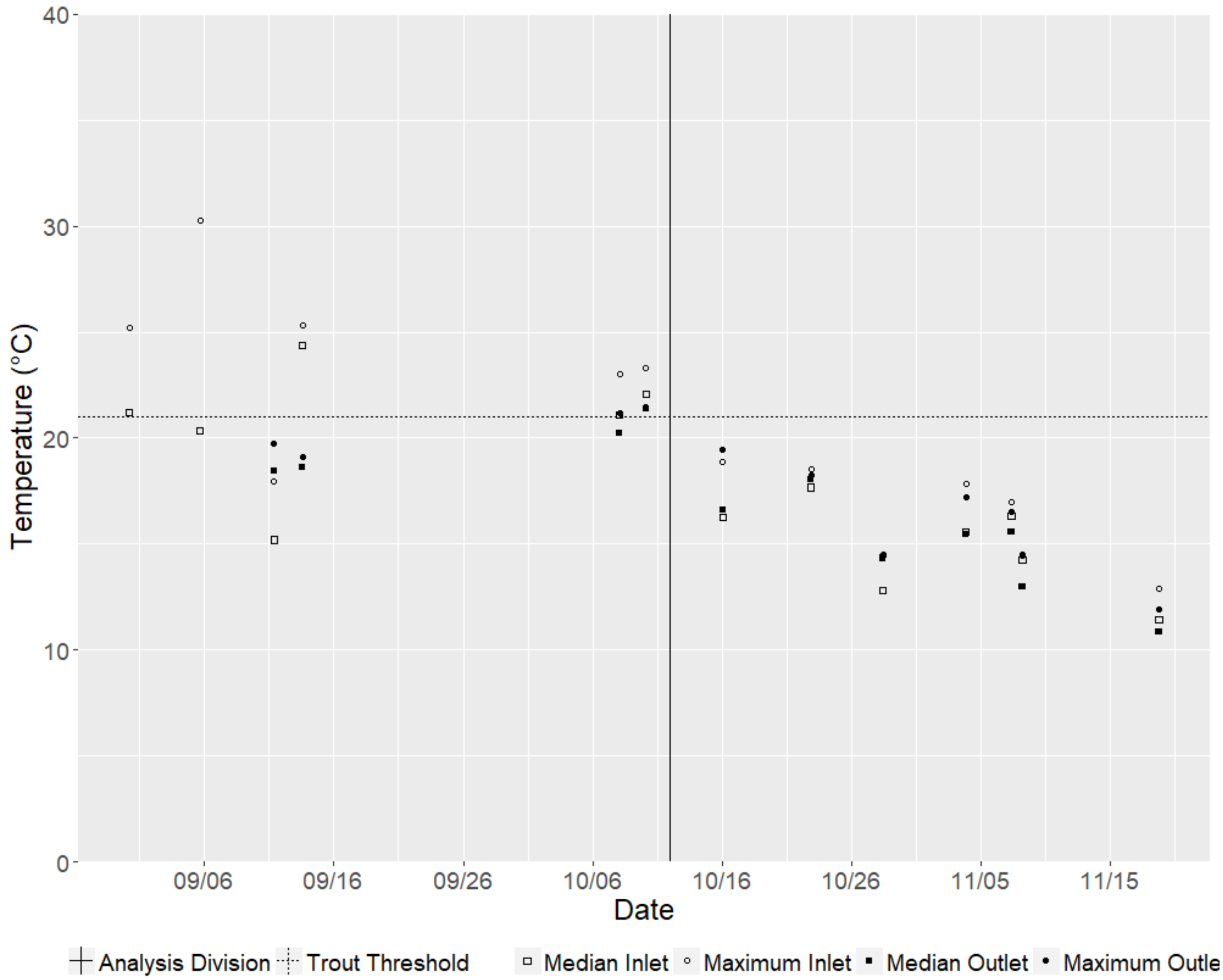


Figure 2-25. Median and Maximum Temperatures; Events ≥ 1.70 mm, August 28, 2017—September 22, 2017

Previous research monitoring BRCs in the NC mountains found SCMs with loading rates ranging from 6 – 25:1 significantly reduced maximum influent temperatures when influent was significantly warmer than 21°C (Jones & Hunt, 2009); median influent temperature reductions were inconsistent (Jones & Hunt, 2009). During their two summers of data collection, Jones & Hunt (2009) found median effluent temperatures were not significantly different from 21°C, while maximum effluent temperatures were significantly warmer than 21°C. Similar results were found by Long & Dymond (2014) studying BRC in Blacksburg, Virginia, where maximum and average influent temperatures were reduced significantly; however, the studied BRC did not consistently reduced effluent temperatures below the trout threshold. Long & Dymond (2014) specified volume reductions as an important thermal load reduction mechanism when cold-water habitat protection is a primary objective. Jones & Hunt (2009) identified minimal additional temperature buffering benefits of moderately oversize BRCs; however, Jones & Hunt (2009) did note that larger BRCs' have the ability to infiltrate greater portions of runoff improving thermal load reductions.

Median and maximum well temperatures were not significantly different than 21°C. There were no significant differences between any two monitoring locations for either temperature metric (median or maximum event temperatures) (Figure 2-26). The large watershed loading rate (29:1, Table 2-4) seems to have elevated media temperatures at all monitored locations. The BRC herein appears undersized and is thus unable to adequately buffer influent temperatures. BRC effluent maximum and median temperatures have a 27% and 15% chance, respectively, of exceeding 21°C for events greater than 1.70-mm before October 12, 2017 (Figure 2-25).

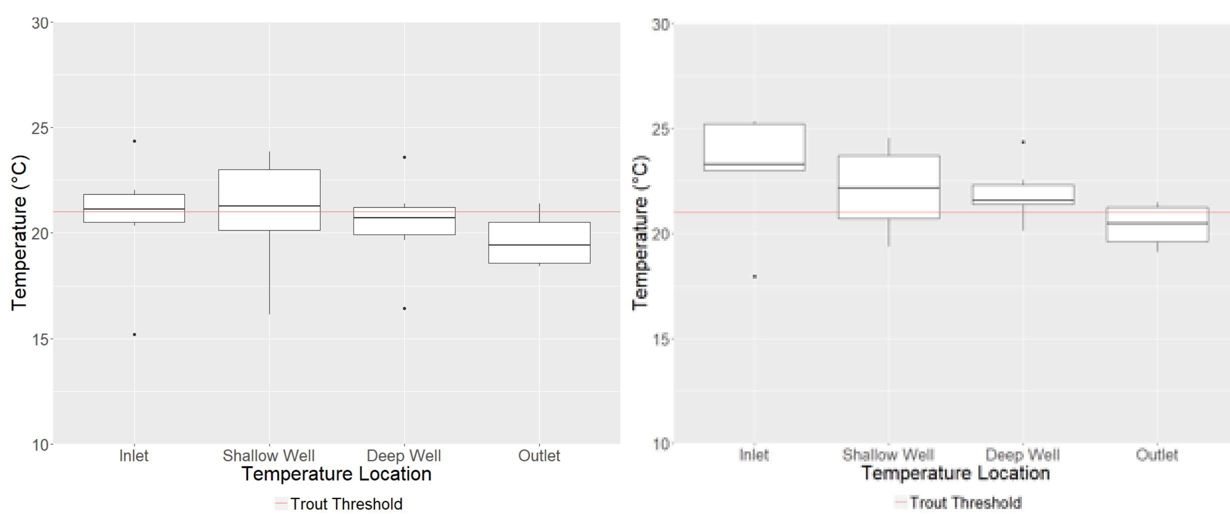


Figure 2-26. BRC Median (Left) and Maximum (Right) Temperatures; Events ≥ 1.70 mm Before August 28, 2017

Median and maximum event temperatures after October 12, 2017, were significantly cooler than 21ºC (Figure 2-27). Inlet and outlet median and maximum temperatures were not significantly different for runoff events after October 12, 2017.

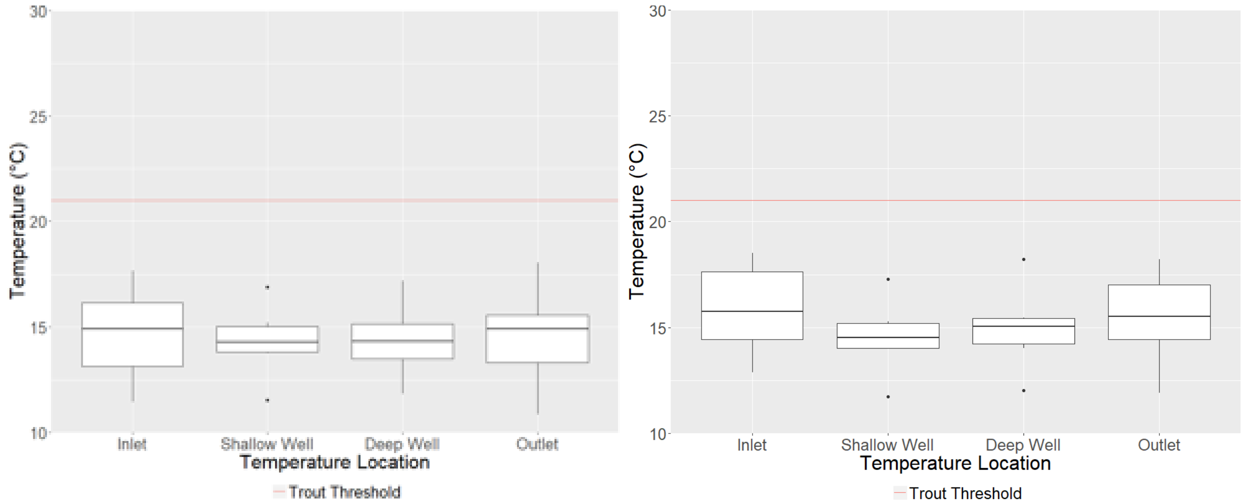


Figure 2-27. BRC Median (Left) and Maximum (Right) Temperatures; Event ≥ 1.70-mm After October 12, 2017

Using the NRCS Curve Number method (CN of 88) (NRCS, 1986) runoff volumes were calculated from discrete rainfall events. Outflow volumes were calculated using instantaneous depth measurements taken within the outlet weir box (Figure 2-9) using Eq. 2.4 & Eq. 2.4.1 (NCEES, 2013).

$$Q = 796.7h^{2.5} \qquad h \leq 0.114 \text{ m} \qquad \text{(Eq. 2.4)}$$

$$Q = 796.7h^{2.5} + CLh^{3/2} \qquad h > 0.114 \text{ m} \qquad \text{(Eq. 2.4.1)}$$

Where,

Q = orifice discharge (L/s)

H = stage above weir crest (m)

C = 1.84, coefficient of discharge (unitless)

L = length of rectangular weir crest (m)

Effluent volumes are likely an underestimation; leaks and scour around the weir box were observed during site visits. Additionally, at least one event before October 12, 2017, produced enough inflow to overwhelm the BRC bowl volume eroding the downslope embankment. System bypass, or overflow, was undocumented. Good to excellent hydrologic mitigation was observed for events greater than or equal to 1.70-mm before and after October 12, 2017 (Table 2-12); cumulative runoff reduction for the study period was 94%. Large rainfall events (2 of 4 events were greater than 55.0-mm) dampened hydrologic mitigation before October 12, 2017. Influent volume reductions greater than 50% are common for field monitored BRCs (Davis, 2008; Dietz & Clausen, 2005; Hunt et al., 2006; Li et al., 2009), however, volume reductions exceeding 90% are excellent. Davis (2008) compared the Rational Method runoff coefficient, C, for imperviousness (C=0.9) and undeveloped land (C=0.3) to justify a LID target metric of 33% inflow volume reduction. By this metric, the studied BRC betters the LID target; Wardynski et al. (2014) highlights LID performance (volume reduction) as a surrogate for thermal performance. Achieving LID hydrology also achieves thermal mitigation sufficient to support salmonids.

Table 2-12. BRC Hydrology Performance; Events ≥ 1.70-mm Before and After October 12, 2017

Parameter	*Before*	*After*
Cumulative Runoff Volume (m^3)	53,423	84,087
Cumulative Outflow Volume (m^3)	4,808*	3,336*
Median Runoff Volume Reduction (%)	79*	98*
Cumulative Runoff Volume Reduction (%)	91*	96*

* Does not account for bypass and leaks; overestimates volume reduction

Thermal load mitigation marries both changes in temperature and volumes (Jones et al., 2012; Wardynski et al., 2013, 2014). Thermal load estimations for events prior to and following October 12, 2017, were calculated using the median inlet and outlet event temperatures, estimated runoff volumes, and calculated outflow volumes using Eq. 2.2 (Wardynski et al., 2014).

Good to excellent hydrologic treatment yielded substantial thermal load reductions (Table 2-13). The impact of effluent thermal loads on salmonids is dependent on discharge mixing within the stream (Wardynski et al., 2014). The BRC herein mitigated thermally-enriched runoff. These results are congruent with other studies examining BRC thermal mitigation (Jones & Hunt, 2009; Long & Dymond, 2014).

Table 2-13. RWH Thermal Load Performance; Events ≥ 1.70 mm Before and After October 12, 2017

Parameter	*Before*	*After*
Cumulative Inflow Thermal Load (MJ)	9.5×10^7	10.1×10^7
Cumulative Outflow Thermal Load (MJ)	5.9×10^7	4.0×10^7
Median Thermal Load Reduction (%)	79	99
Cumulative Thermal Load Reduction (%)	94	96

2.4.2.3.1. Summary & Discussion

BRC monitoring was conducted from August 28, 2017—November 22, 2017. Instantaneous temperatures and depth measurements were collected at the inlet, shallow well (30-cm), deep well (53-cm), and outlet. Event median and maximum temperatures were analyzed to determine the temperature buffering capabilities of this BRC following methods proposed by Wardynski et al. (2014); temperatures were compared to 21°C (Coutant, 1977; Ferguson, 1958), an accepted threshold for trout. Analysis of maximum event and median event temperatures are most useful in conjunction with downstream temperature standards (Wardynski et al., 2014); however, in the absence of more robust data, this analysis provides useful insight to the performance of BRC as a thermal mitigation strategy. Analyses were conducted for events greater than or equal to 1.70-mm, the threshold to generate runoff, collected before and after October 12, 2017 (a date selected when runoff temperatures no longer exceeded that of the trout threshold).

Influent and effluent median and maximum temperatures were not significantly different from the 21^{o}C trout threshold. A similar study observed good buffering of maximum runoff temperatures and an inability to adapt to cooler runoff temperatures (Jones & Hunt, 2009). Jones & Hunt (2009) observed equilibrium between temperature of the media and runoff occurred at a depth of 60 cm. Once deeper than 60-cm, media was unlikely to reduce runoff temperatures; deeper media has a diminishing thermal reduction benefit although it likely enables more infiltration (Jones & Hunt, 2009). Similarly oversized BRCs will improve thermal load reduction by proportional increases in volume reduction (Brown & Hunt, 2011; Jones & Hunt, 2009).

The BRC herein was smaller than others studied (loading ratio 29:1) (Table 2-4) (Davis, 2008; Hunt et al., 2006; Jones & Hunt, 2009); despite being undersized, substantial volume reductions were achieved throughout the monitoring period (91% and 96% before and after October 12, 2017, respectively)(Table 2-12). The BRC bested the LID target (33%) for inflow volume reduction and by this metric was judged successful at thermal mitigation (Davis, 2008; Wardynski et al., 2014). Large volume reductions corresponded to proportionally large thermal load reductions (Table 2-13). The impact of effluent thermal loads on salmonids will be dependent on the on in-stream flows and mixing (Wardynski et al., 2014).

The simultaneous reduction of temperatures and volumes tended to reduce the duration of elevated temperatures (Wardynski et al., 2014). For the 70.3-mm rainfall event (Figure 2-23) beginning on September 11, 2017, BRC treatment reduced the duration of outflow exceeding the 21^{o}C trout threshold by 50% (from 1 hour to 30 minutes) (Figure 2-24).

The BRC demonstrated excellent treatment of influent thermal load observed by the minimal fluctuations of well temperatures during inflow. Deep well (53-cm) temperatures exhibited the least impact from infiltrated runoff and had the quickest return to antecedent temperature; this phenomena was observed in a previous field study on BRC (Jones & Hunt, 2009). These results are consistent with other studies of BRC performance for thermal reduction (Jones & Hunt, 2009; Long & Dymond, 2014). BRC is a versatile SCM with proven water quality treatment (Dietz & Clausen, 2005; Hunt et al., Eubanks, 2008; Jones & Hunt, 2009; Li et al., 2009; Long & Dymond, 2014).

2.4.2.4. Brittain Creek

In-stream temperartures and stage measurements were collected in Brittain Creek, adjacent to CBC, upstream of LSVFS and BRC discharge. Data were collected to ascertain if stream temperatures exhibited thermal enrichement from runoff events. The watershed (147.6 ha) was comprised of residential (average 0.2-ha lot size) and forested land uses. Watershed Total Imperviousness (TI) was estimated to be 25% using NRCS CN Method land use cover descriptions (NCDEQ, 2017). The upstream network included 3 wetponds or in-stream impoundments; wet ponds are known to conribute thermal loads and increased temperature variation (Jones & Hunt, 2010; Lieb & Carline, 2000).

Peaks of in-stream depth correspond to precipitation events; there is minimal evidence of in-stream temperature fluctuations corresponding to peaks or spikes in stream depth (Figure

2-28). The largest event at CBC (96.5 mm in 15.9 hr) occurred on October 23, 2017, and generated the largest 5-minute peak intensity (74.5 mm/hr). That precipitation depth had a 2- to 5-year ARI; while that intensity was a 25- to 50-year ARI (NOAA National Weather Service, n.d.). During this event a large tree fell on the stream bank and altered the HOBO monitoring position exposing it to air temperatures when stream stage fell. Data that recorded air temperatures were removed; temperatures recorded during data collection (when the probe was exposed to air) have also been removed from the plot.

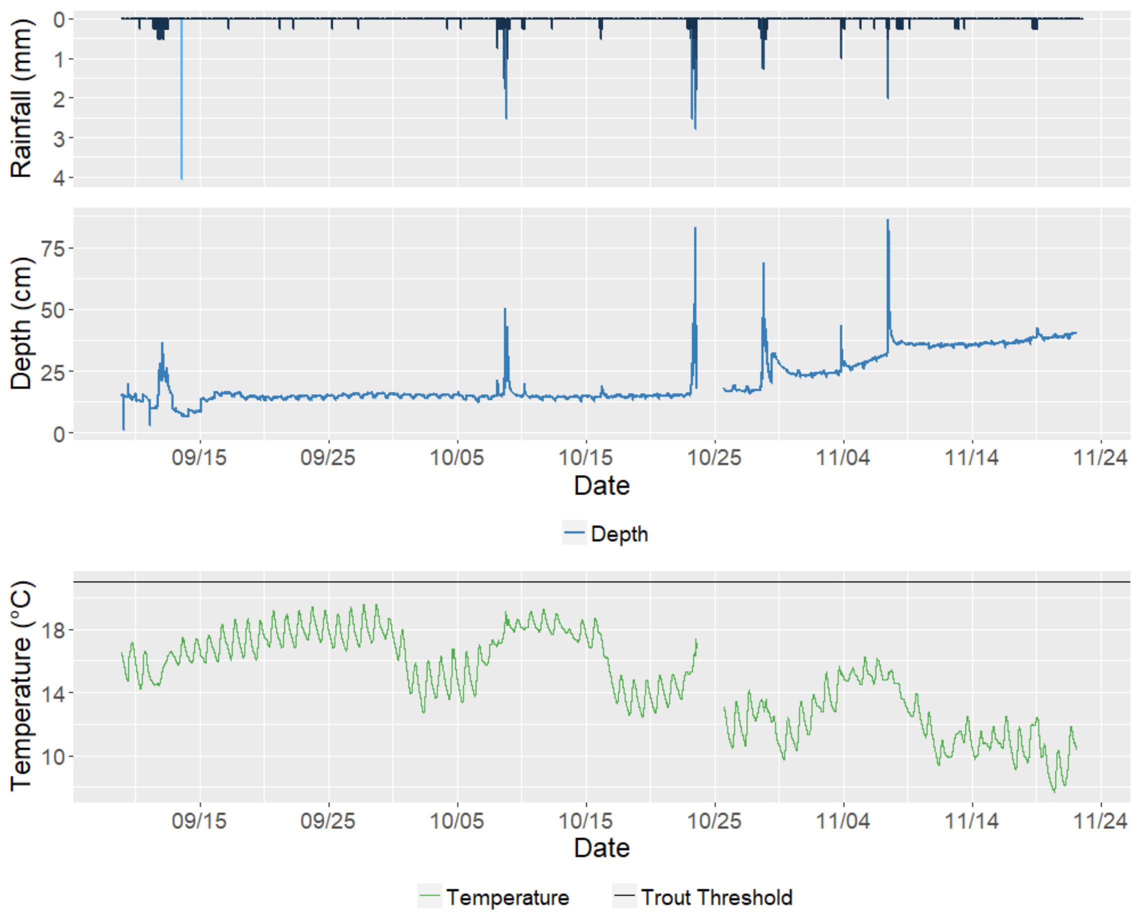


Figure 2-28. Rainfall (Top), Brittain Creek Depth (Middle) and Temperature (Bottom); August 28, 2017—September 22, 2017

In-stream temperatures never exceeded the 21ºC trout threshold during the monitoring period. A temperature-duration plot was created using in-stream temperatures and cumulative durations (Figure 2-29). Comparisons of SCM temperature-duration plots to the temperature-duration plot created for Brittain Creek would not be appropriate because Brittain Creek does not currently support brook trout; however, the comparison illustrates how temperature-duration plots can be used for comparisons. BRC effluent temperatures exceeded 21ºC for approximately 250 hours (Figure 2-29). The Brittain Creek temperature-duration plot could be compared to that of another trout-supporting streams to determine if Brittain Creek's thermal regime can support trout or other cold-water biota of interest.

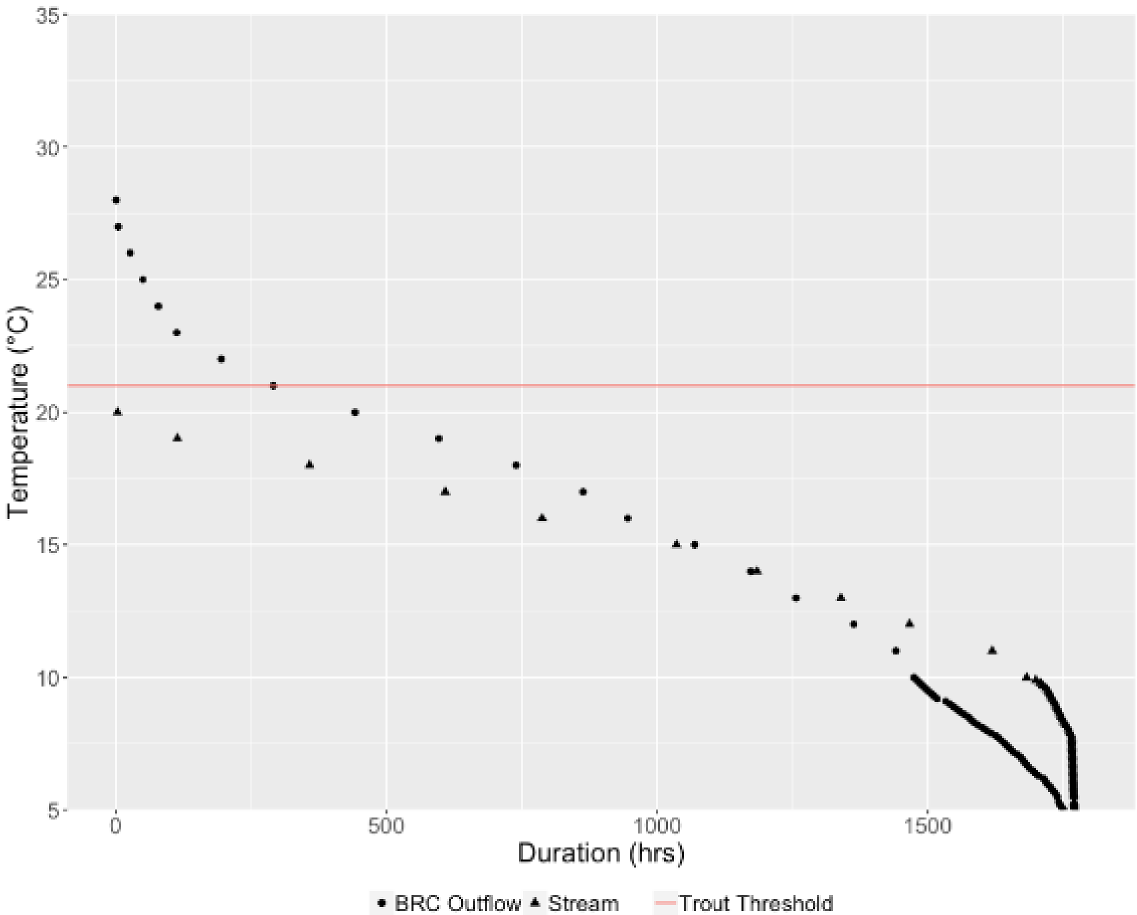


Figure 2-29. Brittain Creek Temperature-Duration Plot; August 28, 2017—September 22, 2017

Thermal impacts are likely not the sole cause of trout population recession; urbanization alters ecosystem function (Meyer et al., 2005). Urban streams experience flashy hydrographs, elevated pollutant concentrations, reduced baseflows, and increased sediment loads, in aggregate referred to as the urban stream syndrome (Pitt et al., 2009; Burns et al., 2012; Davis et al., 2003;

Davis et al., 2006; Novotny, 2003; Ladson et al., 2006; Leopold, 1994; Walsh et al., 2005). Strategic SCM implementation to restore catchment hydrology and reduce pollutant (including thermal) loads in conjunction with stream geomorphologic and habitat restoration will likely produce the best species conservation or reintroduction outcomes (Poff et al., 1997).

2.5. Recommendations & Conclusions

1. BRCs can buffer maximum and, to a lesser extent, median event temperatures, while reducing runoff volumes and, thus, thermal loads. Reductions in effluent temperature and volume reduce the cumulative-duration at elevated temperatures, minimizing thermal stress on trout populations. Of the 3 SCMs tested, BRCs are recommended for thermal mitigation.
2. With consistent usage, RWH has the potential to substantially reduce runoff volumes; alternatively, passive release systems allow the creation of detention (slow release) and retention (designated use) volumes which ensure available capacity for runoff capture and mitigation. The RWH herein displayed some evidence of thermal stratification which resulted in significantly hotter temperatures near the tank overflow; harvested rainwater usage will likely eliminate thermal stratifications, because mixing should occur as tank volume is used.
3. The RWHs wet conveyance was vital to its performance. The wet conveyance increased total storage capacity by 60%, provided detention and slow release of events up to 11% of the required design storm (25-mm), and 97% mitigation of cumulative runoff volumes and thermal loads over the monitored period. The wet conveyance can improve RWH system aesthetics and performance by “hiding” conveyance components below ground and producing additional runoff detention capacity.

4. The open grass LSVFS herein did not buffer influent temperatures and is not recommended for purposes of thermal mitigation. Perhaps when shaded or wooded, LSVFS performance will improve; however, based upon these results coupled with literature, it is doubtful that LSVFS thermal mitigation will surpass that of BRCs, or RWHs, that are designed and operated properly.
5. In areas of thermal sensitivity, material selection and shading can assist in mitigation of thermally-enriched runoff. Shaded SCMs will likely perform better than similar unshaded SCMs. A smaller LSVFS with some shading (Winston et al., 2011) buffered temperatures better than the system herein. This RWH storage tank was in full sun and exhibited large diel temperature variation. The BRC was grassed and exposed to full sun allowing the surface soil to be heated; shading surface soils will reduce solar heating and improve temperature buffer capabilities (Hunt et al., 2012; Jones & Hunt, 2009; Winston et al., 2011). Shaded parking areas have resulted in significantly lower runoff temperatures vis-à-vis unshaded parking areas (Jones et al., 2012). By maintaining stream buffers and riparian forest, and shading impervious areas and SCMs, the thermal regime of cold-water streams can be protected or restored (Caissie, 2006; Hunt et al., 2012; Jones & Hunt, 2009; Jones et al., 2012; Vannote et al., 1980; Winston et al., 2011).
6. Trout-supporting reference reach stream temperature data are needed to support the development of SCM effluent design standards. Additionally, these data could assist decision makers in diagnosis of (temperature or other water quality) trout population decline or absence.
7. Monitoring studies that capture a complete view of a watershed's energy balance are needed to better understand the effects of imperviousness on a stream's thermal regime.

Researchers should compare various watersheds of similar size at different degrees of development. Data should be collected throughout a watershed's hydrologic cycle to understand how ET, surface runoff, shallow interflow, and deep infiltration effect the energy balance of the receiving stream. Monitoring of SCMs should focus on these elements of the hydrologic cycle and seek to understand if SCMs can effectively mimic the natural processes that produce cold water habitats, what effect they have on a streams thermal regime, how does stormwater infiltration effect shallow interflow and groundwater delivery to streams, and what SCM design parameters influence its ability to protect a streams thermal regime.

8. Proposed and existing development can minimize thermal export by maintaining riparian buffers. Buffers provide stream shading and reduce direct solar radiation, minimizing diel temperature variation. Throughout a watershed, vegetation should be used to shade imperviousness and similarly reduce thermal gains. Shading impervious surfaces will minimize export of thermal energy during runoff events. Dispersed capture, treatment, and infiltration of runoff will reduce thermal export to receiving waters and minimize the impact of increased imperviousness. DCIAs should be targeted throughout a developed watershed for SCM retrofits due to its ability to rapidly deliver thermally-enriched runoff to receiving waters.

2.6. References

Akbari, H., & Konopacki, S. (2005). Calculating energy-saving potentials of heat-island reduction strategies. *Energy Policy*, *33*(6), 721–756. https://doi.org/10.1016/j.enpol.2003.10.001

Armour, C. L. (1991). Guidance for evaluating and recommending temperature regimes to protect fish instream flow. *U.S. Fish and Wildlife Service, Biological Report*, *90*(22), 1–13.

Beschta, R., Bilby, R., Brown, G., Holtby, L., & Hofstra, T. (1987). Stream temperature and aquatic habitat: fisheries and forestry interactions. *Streamside Management: Forestry and Fishery Interactions*, 191–232.

Brown, R. A., & Hunt, W. F. (2011). Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads. *Journal of Environmental Engineering*, *137*(11), 1082–1091. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000437

Burns, M. J., Fletcher, T. D., Walsh, C. J., Ladson, A. R., & Hatt, B. E. (2012). Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform. *Landscape and Urban Planning*, *105*(3), 230–240. https://doi.org/10.1016/j.landurbplan.2011.12.012

Caissie, D. (2006). The thermal regime of rivers: A review. *Freshwater Biology*, *51*(8), 1389–1406. https://doi.org/10.1111/j.1365-2427.2006.01597.x

Coutant, C. C. (1977). Compilation of Temperature Preference Data. *Journal of the Fisheries Research Board of Canada*, *34*(5), 739–745. https://doi.org/10.1139/f77-115

Davis, A. P. (2008). Field Performance of Bioretention: Hydrology Impacts. *Journal of Hydrologic Engineering*, *13*(2), 90–95. https://doi.org/10.1061/(ASCE)1084-0699(2008)13:2(90)

Davis, A. P., Shokouhian, M., Sharma, H., & Minami, C. (2006). Water Quality Improvement through Bioretention Media: Nitrogen and Phosphorus Removal. *Water Environment Research*, *78*(3), 284–293. https://doi.org/10.2175/106143005X94376

Davis, A. P., Shokouhian, M., Sharma, H., Minami, C., & Winogradoff, D. (2003). Water quality improvement through bioretention: lead, copper, and zinc removal. *Water Environment Research : A Research Publication of the Water Environment Federation*, *75*(1), 73–82. https://doi.org/10.2175/106143003X140854

Dietz, M. E., & Clausen, J. C. (2005). A field evaluation of rain garden flow and pollutant treatment. *Water, Air, and Soil Pollution*, *167*(1–4), 123–138. https://doi.org/10.1007/s11270-005-8266-8

Doan, L. N., & Davis, A. P. (2017). Bioretention–Cistern–Irrigation Treatment Train to Minimize Stormwater Runoff. *J. Sustainable Water Built Environ.*, *3*(2), 1–10. https://doi.org/10.1061/

EPA. (2016). Summary of the Clean Water Act. Retrieved October 14, 2016, from https://www.epa.gov/laws-regulations/summary-clean-water-act

Ferguson, R. G. (1958). The Preferred Temperature of Fish and their Midsummer Distribution in Temperate Lakes and Streams. *Journal of Fish Research*, *15*(4), 607–624. Retrieved from http://www.nrcresearchpress.com/doi/pdf/10.1139/f58-032

Gee, K. D., & Hunt, W. F. (2016). Enhancing Stormwater Management Benefits of Rainwater Harvesting via Innovative Technologies. *Journal of Environmental Engineering*, *142*(8), 04016039. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001108

Gregg, S. (2018a). Brittain-Creek. Retrieved from https://github.com/simonsaysgregg/Brittain-Creek

Gregg, S. (2018b). CBC-BRC. Retrieved from https://github.com/simonsaysgregg/CBC-BRC

Gregg, S. (2018c). CBC-LSVFS. Retrieved from https://github.com/simonsaysgregg/CBC-LSVFS

Gregg, S. (2018d). TFC-RWH. Retrieved from https://github.com/simonsaysgregg/TFC-RWH

Herb, W. R., Mohseni, O., & Stefan, H. G. (2009). Simulation of temperature mitigation by a stormwater detention pond. *Journal of the American Water Resources Association*, *45*(5), 1164–1178. https://doi.org/10.1111/j.1752-1688.2009.00354.x

Hunt, W. F., Davis, A. P., & Traver, R. G. (2012). Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design. *Journal of Environmental Engineering*, *138*(6), 698–707. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000504

Hunt, W. F., Jarrett, A. R. R., Smith, J. T., & Sharkey, L. J. (2006). Evaluating Bioretention Hydrology and Nutrient Removal at Three Field Sites in North Carolina. *Journal of Irrigation and Drainage Engineering*, *132*(6), 600–608. https://doi.org/10.1061/(ASCE)0733-9437(2006)132:6(600)

Hunt, W. F., Smith, J. T., Jadlocki, S. J., Hathaway, J. M., & Eubanks, P. R. (2008). Pollutant Removal and Peak Flow Mitigation by a Bioretention Cell in Urban Charlotte, N.C. *Journal of Environmental Engineering*, *134*(5), 403–408. https://doi.org/10.1061/(ASCE)0733-9372(2008)134:5(403)

Jones, M., & Hunt, W. (2010). Effect of Storm-Water Wetlands and Wet Ponds on Runoff Temperature in Trout Sensitive Waters. *Journal of Irrigation and Drainage Engineering*, *136*(9), 656–661. https://doi.org/doi:10.1061/(ASCE)IR.1943-4774.0000227

Jones, M. P., & Hunt, W. F. (2009). Bioretention Impact on Runoff Temperature in Trout Sensitive Waters. *Journal of Environmental Engineering*, *135*(8), 577–585.

https://doi.org/10.1061/(ASCE)EE.1943-7870.0000022

Jones, M. P., & Hunt, W. F. (2010). Performance of rainwater harvesting systems in the southeastern United States. *Resources, Conservation and Recycling*, *54*(10), 623–629. https://doi.org/10.1016/j.resconrec.2009.11.002

Jones, M. P., Hunt, W. F., & Winston, R. J. (2012). Effect of Urban Catchment Composition on Runoff Temperature. *Journal of Environmental Engineering*, *138*(12), 1231–1236. https://doi.org/doi:10.1061/(ASCE)EE.1943-7870.0000577

Kalz, D. E., Wienold, J., Fischer, M., & Cali, D. (2010). Novel heating and cooling concept employing rainwater cisterns and thermo-active building systems for a residential building. *Applied Energy*, *87*(2), 650–660. https://doi.org/10.1016/j.apenergy.2009.06.002

Koryto, K. M. (2016). *Evaluation of the hydrologic and water quality benefits of regenerative stormwater conveyance*. North Carolina State University.

Ladson, A. R., Walsh, C. J., & Fletcher, T. D. (2006). Improving Stream Health in Urban Areas by Reducing Runoff Frequency from Impervious Surfaces. *Australian Journal of Water Resources*, *10*(1), 23–33.

Leopold, L. B. (1994). *A View of the River*. Cambridge, Massachusetts: Harvard University Press.

Li, H., Sharkey, L. J., Hunt, W. F., & Davis, A. P. (2009). Mitigation of Impervious Surface Hydrology Using Bioretention in North Carolina and Maryland. *Journal of Hydrologic Engineering*, *14*(4), 407–415. https://doi.org/10.1061/ASCE1084-0699200914:4407

Lieb, D. A., & Carline, R. F. (2000). Effects of urban runoff from a detention pond on water quality, temperature and caged gammarus minus (say) (amphipoda) in a headwater stream. *Hydrobiologia*, *441*, 107–116. https://doi.org/10.1023/A:1017550321076

Long, D. L., & Dymond, R. L. (2014). Thermal Pollution Mitigation in Cold Water Stream Watersheds Using Bioretention. *Journal of the American Water Resources Association*, *50*(4), 977–987. https://doi.org/10.1111/jawr.12152

Meyer, J. L., Paul, M. J., & Taulbee, W. K. (2005). Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society*, *24*(3), 602–612. https://doi.org/10.1899/04-021.1

N.C. DEQ. (2017). North Carolina Stormwater Control Measure Credit Document. *North Carolina Department of Environmental Quality*. Retrieved from https://files.nc.gov/ncdeq/Energy Mineral and Land Resources/Stormwater/BMP Manual/SSW-SCM-Credit-Doc-20170807.pdf

National Council of Examiners for Engineering and Surveying. (2013). Fundementals of Engineering Suplimental-Reference Handbook. Clemson, SC.

NC Wildlife Resource Commission. (2018). What to Catch: Common Sport Fish Species. Retrieved from http://www.ncwildlife.org/Fishing/What-to-Catch

NCDENR. (2003). *Biological Impairment in the Mud Creek Watershed*.

NCDENR. (2007). Stormwater Best Management Practices Manual. North Carolina Division of Water Quality.

NCEQ. (n.d.). MUD CREEK LOCAL WATERSHED PLAN.

NOAA. (n.d.). Data Tools: 1981-2010 Normals.

NOAA National Weather Service. (n.d.). NOAA Atlas 14: Precipitation Frequency Data Server. Retrieved March 12, 2018, from https://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_cont.html

North Carolina Department of Environmental Quality. (2017). Stormwater Design Manual. Retrieved April 27, 2018, from https://deq.nc.gov/sw-bmp-manual

Novotny, V. (2003). *Water Quality: Diffuse Pollution and Watershed Management* (2nd ed.). Hoboken, New Jersey: John Wiley & Sons, Inc.

NRCS. (1986). Urban Hydrology for Small Watersheds TR-55. *USDA Natural Resource Conservation Service Conservation Engeneering Division Technical Release 55*, 164. https://doi.org/Technical Release 55

Pelak, N., & Porporato, A. (2016). Sizing a rainwater harvesting cistern by minimizing costs. *Journal of Hydrology*, *541*, 1340–1347. https://doi.org/10.1016/j.jhydrol.2016.08.036

Pitt, R., Asce, M., Chen, S., Asce, A. M., Clark, S. E., Asce, M., … Asce, A. M. (2009). Compaction ' s Impacts on Urban Storm-Water Infiltration, *134*(5), 652–658.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., … Stromberg, J. C. (1997). The Natural Flow Regime: A paradigm for river conservation and restoration. *BioScience*, *47*(11), 769–784. https://doi.org/10.2307/1313099

R Core Team. (2017). R: A Language and Environment for Statistical Computing. Vienna, Austria. Retrieved from https://www.r-project.org/

Responsive Managment, & Southwick Associates. (2015). *MOUNTAIN TROUT FISHING : ECONOMIC IMPACTS ON AND CONTRIBUTIONS TO NORTH CAROLINA ' S ECONOMY*.

Rossi, L., & Hari, R. E. (2007). Screening procedure to assess the impact of urban stormwater temperature to populations of brown trout in receiving water. *Integrated Environmental Assessment and Management*, *3*(3), 383–392. https://doi.org/10.1897/1551-3793(2007)3[383:SPTATI]2.0.CO;2

Sharkey, L. J. (2006). *The performance of bioretention areas in NC*. North Carolina State University.

Southwood, T. R. E. (1977). Habitat, the Templet for Ecological Strategies? *The Journal of Animal Ecology*, *46*(2), 336. https://doi.org/10.2307/3817

Steffen, J., Jensen, M., Pomeroy, C. A., & Burian, S. J. (2013). Water supply and stormwater management benefits of residential rainwater harvesting in U.S. cities. *Journal of the American Water Resources Association*, *49*(4), 810–824. https://doi.org/10.1111/jawr.12038

Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The River Continuum Concept.

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan II, R. P. (2005). The urban stream syndrome : current knowledge and the search for a cure The urban stream syndrome : current knowledge and. *Journal of the North American Benthological Society*, *24*(3), 706–723. https://doi.org/10.1899/04-028.1

Wardynski, B. J., Asce, A. M., Winston, R. J., Hunt, W. F., Wre, D., & Asce, M. (2013). Internal Water Storage Enhances Exfiltration and Thermal Load Reduction from Permeable Pavement in the North Carolina Mountains. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000626

Wardynski, B. J., Winston, R. J., Line, D. E., & Hunt, W. F. (2014). Metrics for assessing thermal performance of stormwater control measures. *Ecological Engineering*, *71*, 551–562. https://doi.org/10.1016/j.ecoleng.2014.07.068

Winston, R. J., Hunt, W. F., & Lord, W. G. (2011). Thermal Mitigation of Urban Storm Water by Level Spreader – Vegetative Filter Strips. *Thermal Mitigation of Urban Storm Water by Level Spreader-Vegetative Filter Strip*, *137*(8), 707–716. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000367.

CHAPTER 3 . PERFORMANCE OF REGENERATIVE STORMWATER CONVEYANCE IN THE NORTH CAROLINA PIEDMONT

3.1. Abstract

Regenerative Stormwater Conveyance (RSC) is a new Stormwater Control Measure (SCM) which integrates urban stormwater management with stream restoration techniques to simultaneously convey stormwater and treat pollutants. RSCs are most often used to stabilize eroded streams, gullies, and stormwater outfalls by filling the channel bed with sand and woodchip media. An undersized RSC was constructed with an exfiltration trench downstream of a dry pond in the North Carolina Piedmont to stabilize an advancing headcut and provide pollutant treatment. Two inlets and one outlet were monitored for flow and select pollutants. True Event Mean Concentrations (EMCs) were unable to be collected due to flow-measurement challenges, instead Representative Concentrations (RCs) were collected. Influent calibration allowed analysis of 15 hydrologic events; poorly drained underlying soils, limited surface ponding, constant media saturation, and large loading rates resulted in no volume reduction via RSC treatment. Water quality treatment was modest for most analytes. Storm flow Removal Efficiencies (REs) for TN, TP, and TSS were 3%, 26%, and 69%, respectively; TSS RC reduction was significant. Base flow Res for TN, TP, and TSS were 22%, 3%, and 11%, respectively; TN and TSS RC reductions were significant. While this RSC likely eliminated headcut migration and channel erosion, other benefits such as hydrologic mitigation and pollutant removal were muted, likely by its extremely (proportionally) small size.

3.2. Introduction

3.2.1. Land Use Change & Downstream Impacts

Landscapes and stream networks evolve from headwaters to river mouth which biological communities conform, structurally and functionally, to energy dissipation patterns of the physical system (Vannote et al., 1980). Headwaters, zero and first order streams, represent the maximum interface of water with the terrestrial environment in part due to riparian vegetation shading and the associated detrital inputs (Vannote et al., 1980). Urbanization replaces vegetated surfaces with impervious hardscapes such as buildings, sidewalks, and roads. Urban areas export sediment, hydrocarbons, nutrients, pathogens, heavy metals, and thermal pollution with higher runoff volumes and larger peak flows (Davis et al., 2003; Davis et al., 2006; Hunt et al., 2012; Jones et al., 2012; Ladson et al., 2006). Population growth and urban migration increases land-use change, development, and resource consumption. Industry, agriculture, and drinking water systems compete for water resources; in some water-scarce areas water rights have become highly contested (Chattahoochee Riverkeeper, 2015; Hood & Sommer, 2016).

Conventional stormwater management focuses on efficient drainage design where conveyance systems are designed to deliver impervious runoff directly to receiving waters with little or no attenuation or treatment (Burns et al., 2012). Stream and river cross-sectional characteristics are a function of flow, the quantity and character of sediments, and the composition of materials that make up the channel bed and banks, including vegetation (Leopold, 1994). Stream channels work toward dynamic equilibrium migrating laterally as the flow exerts shear stress on the outside of meanders simultaneously depositing sediments on the inside of meanders building point bars (Leopold, 1994). Stream channels are maintained by a characteristic discharge called the bankfull or effective discharge. The bankfull discharge occurs (on average) approximately every 1.5 years for natural streams (Leopold, 1994; Rosgen, 1994). Stream flow is comprised of groundwater and surface waters; therefore, the bankfull discharge is influenced by changes in a basin's runoff regime. Efficient drainage alters infiltration, evapotranspiration (ET), surface and sub-surface flow regimes, increasing total runoff volume while reducing the time to peak discharge; it reduces stream base flows, elevates in-stream nutrient and contaminant concentrations, and alters stream morphology, stability, and ecosystem function (Burns et al., 2012; Fletcher et al., 2014; Meyer et al., 2005). Collectively, the aforementioned characteristics are referred to as the urban stream syndrome (Walsh et al., 2005). Hydrologic modification of watersheds caused by increasing imperviousness shifts streams' flow regime producing morphological changes in streams' cross-sectional area, pattern, and bed features (Leopold, 1994). Research to mitigate the impacts of urbanization often focuses on hydrologic matching to a pre-urban or pre-development hydrologic condition by reducing Directly Connected Impervious Area (DCIA). Runoff retention and infiltration practices reduce the frequency and magnitude of runoff events and provide water quality (WQ) treatment of a design volume (Burns et al., 2012; Davis, 2008; Fletcher et al., 2014; Ladson et al., 2006; Walsh et al., 2009).

Flashy urban runoff hydrographs produce high velocities that can exceed critical shear stress of both stream bed and banks, producing stream degradation locally and aggradation downstream (Walsh et al., 2005; Leopold, 1994). Steep changes of bed slope in headwaters or confined within the channel banks are referred to as headcuts and knick points, respectively (Bennett et al., 2000). Headcut formation and migration in vegetated spillways can be understood in three phases (Temple & Hanson, 1994):

1. loss of vegetation and flow concentration,

2. exceedance of critical shear stress, producing downward and downstream erosion, forming a headcut vertical face, and
3. upstream migration of headcut.

In urban drainage systems, the discharge of concentrated flows at high velocities to ditches and earthen channels produce conditions analogous to phases 1 and 2, beginning headcut formation. Headcuts will advance upstream at a constant rate in homogenous soils (Flores-Cervantes et al., 2006). Migration in stratified soil is characterized by episodic failure and mass wasting after plunge pool undercutting of an upper, more cohesive soil layer; rate of migration is dependent on surface soil strength to resist cantilever mass failure and scour hole geometry (Stein & LaTray, 2002).

3.2.2. Stormwater Controls

In 1972, the Clean Water Act (CWA) was established to provide a framework for protecting U.S. water resources by preserving and restoring the chemical, physical, and biological integrity of the nation's waters by requiring states to monitor, analyze, and develop water quality standards for their waters (EPA, 2016). Under the National Pollutant Discharge Elimination System (NPDES), it is the local municipalities' responsibility to implement a plan for monitoring and permitting of point and non-point source discharges of effluent or runoff leaving municipalities, industrial facilities, or other entities to waters of the state (EPA, 2016; NCDENR, 2007).

NPDES requires adoption of minimum standards in post-construction management of stormwater for new development and redevelopment. Rules apply to project areas that exceed 0.4-ha (1-ac); projects are categorized as low or high density depending on a 24% Built Upon Area (BUA) threshold. Low and high-density developments both require the use of (1) vegetated conveyances to the maximum extent possible, (2) 30-foot buffers between BUA and perennial and intermittent streams, and (3) deed restrictions and protective covenants to ensure long-term municipally-approved plans for stormwater management. In addition to low density development requirements, high density development requires the uses of structural SCMs to treat the water quality event, 38 mm in coastal counties and 25 mm elsewhere in North Carolina. NCDEQ (2017) has metrics in place to calculate SCM treatment of nutrients and Total Suspended Sediment (TSS).

3.2.3. Regenerative Stormwater Conveyance

Regenerative Stormwater Conveyance (RSC) is a new, innovative, approach to managing stormwater and protecting streams. Patented in 2009 as a system and method, subsequent design recommendations were made by Anne Arundel County, MD, and followed by the West Virginia Department of Environmental Protection (WVDEP) (Underwood, 2002; Flores et al., 2012; WVDEP, 2012). RSC installation is growing across the Piedmont and Coastal Plains of the Mid-Atlantic physiographic regions under a variety of names including: Regenerative Step Pool Stormwater Conveyance, Bioinfiltration Stormwater Conveyance, and Coastal Plains Outfall (Cizek, 2014; Flores et al., 2012; WVDEP, 2012). RSCs integrate urban stormwater management and stream restoration engineering techniques to convey and treat non-point source pollutants (Flores et al., 2012) (Figure 3-1). A main objective of RSC installation is to stabilize eroded gullies and headcuts by filling gullies and reducing local bed slope (Bennett et al., 2000; Flores et al., 2012). A combination of Priority 1 (P1) (Figure 3-2) and Priority 4 (P4) (Doll et al., 2003) stream restoration techniques are used to fill or construct a new stream channel and stabilize headcuts with boulders, coarse aggregates, or in some RSC applications, pre-cast concrete structures, respectively (Cizek et al., 2017; Doll et al., 2003). The streambed is elevated, similar to P1 restorations, by filling the existing headcut with coarse sand blended with organic matter;

channel stabilization is accomplished with boulder-weir and riffle-pool sequences, similar to P4 restorations, to provide grade control, support scour pools for energy dissipation, and prevent degradation during high flow events (Doll et al., 2003; Flores et al., 2012).

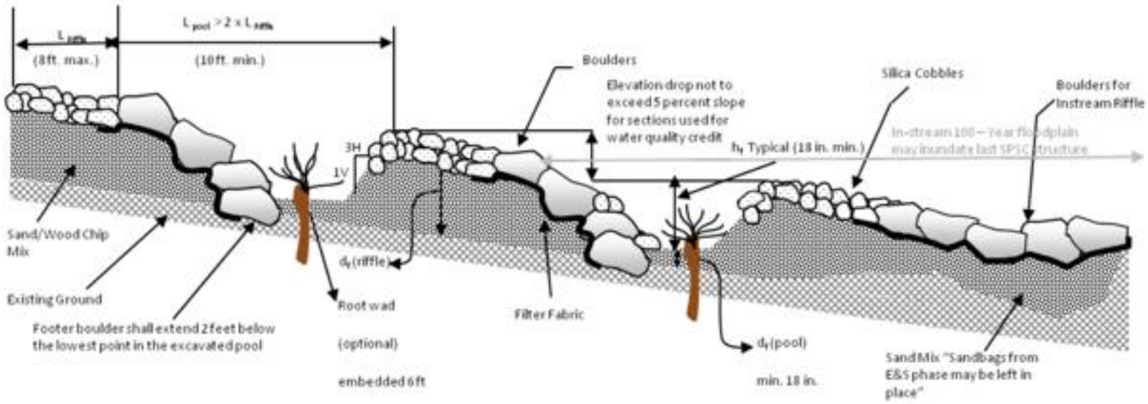


Figure 3-1. Typical RSC Profile (Flores et al., 2012)

Recommendations by Flores et al. (2012) specify RSCs be designed with available storage capacity in surface pool and media pore space to capture a design storm of 25 mm. Pollutant treatment mechanisms include: (1) settling in surface storage (ponded water), (2) filtration, (3) sorption in the media layer, (4) assimilation by vegetation, and (5) microbiological treatment; volume reduction mechanisms include: (1) ET and (2) exfiltration to subsoils (Cizek et al., 2016; Cizek et al., 2017; Davis et al., 2010; Flores et al., 2012; Koryto et al., 2018; Koryto et al., 2017).

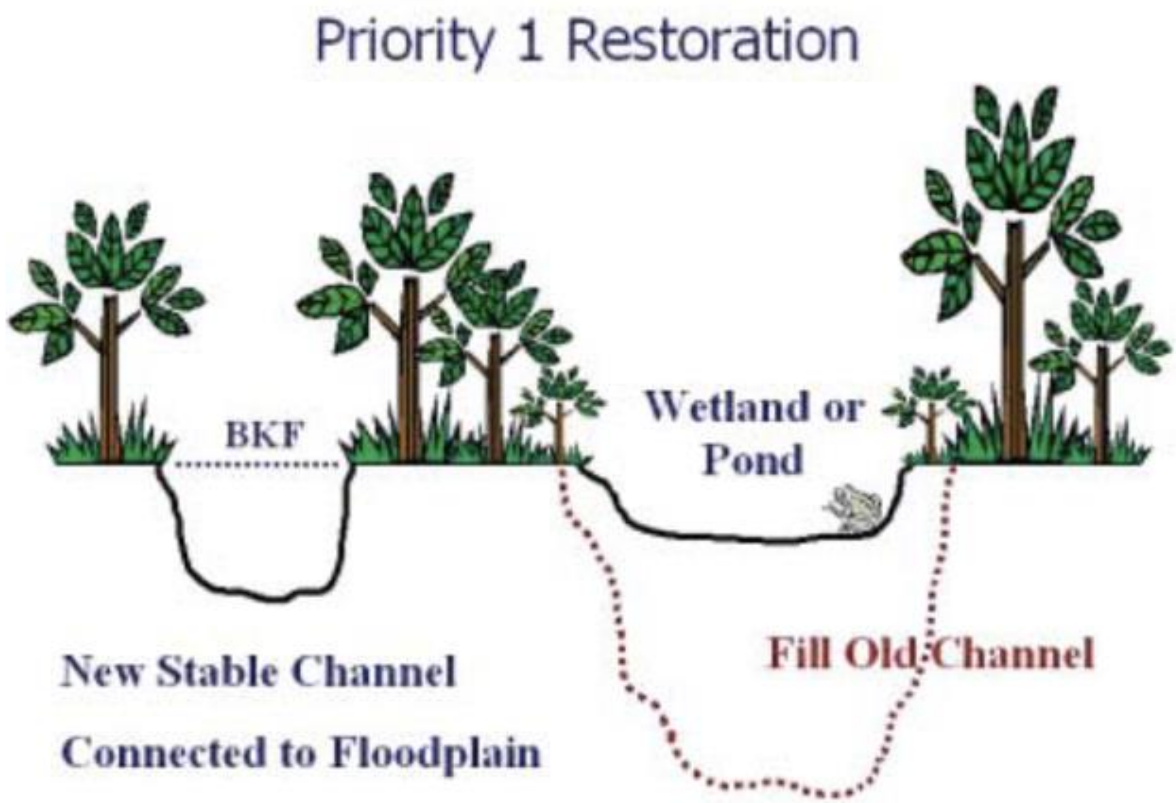


Figure 3-2. P1 Stream Restoration Cross-Section; (Doll et al., 2003)

Sand filtration and media treatment of stormwater runoff is a widely adopted practice with existing permit and regulatory processes, design standards, and performance expectations (Barrett, 2003; Davis et al., 2003; Davis, 2008; Gallo et al., 2012; Hunt et al., 2012; Wardynski & Hunt, 2012). Design recommendations for RSC include safe conveyance of 100-year, 24-hour storm flows via floodplain connection and energy dissipation by in-stream geomorphic structures (Flores et al., 2012; WVDEP, 2012). This sizeable conveyance design storm appears to protect against severe degradation associated with very large and intense events. Adoption of this practice continues despite the lack of scientific consensus around design characteristics or expected treatment performance. The objective of this chapter is to quantify (1) the hydrologic and (2) water quality impacts of an undersized RSC constructed in-series (downstream) of a dry pond in the North Carolina Piedmont.

3.2.4. Jordan Lake

Jordan Lake was created by damming the Haw and Deep rivers near their confluence; since its impounded in 1983, and designation as a Nutrient Sensitive Water (NSW), the lake has been rated eutrophic or hyper-eutrophic (NCDEQ, 2018a). The Jordan Lake Rules (JLR) went into effect on August 11, 2009, and were intended to protect and improve the water quality of the lake by addressing pollution from wastewater discharges, urban stormwater runoff, and agriculture and fertilizer applications (NCDEQ, 2018). Under the JLR, nutrient reduction goals vary depending on the sub watersheds (Figure 3-3, Table 3-1).

Table 3-1. Jordan Lake Rules Nutrient and TSS Reductions from 2001 Baseline and Target Nutrient Loads (GANC, 2009)

Sub-Watershed	*Total Nitrogen (%)*	*Total Phosphorous (%)*	*Total Suspended Solids (%)*	*Target TN Loads (kg/ha/yr)*	*Target TP Loads (kg/ha/yr)*
Haw	8	5	85	3.39	1.28
Upper New Hope	35	5	85	1.96	0.73
Lower New Hope	0	0	85	3.93	0.70

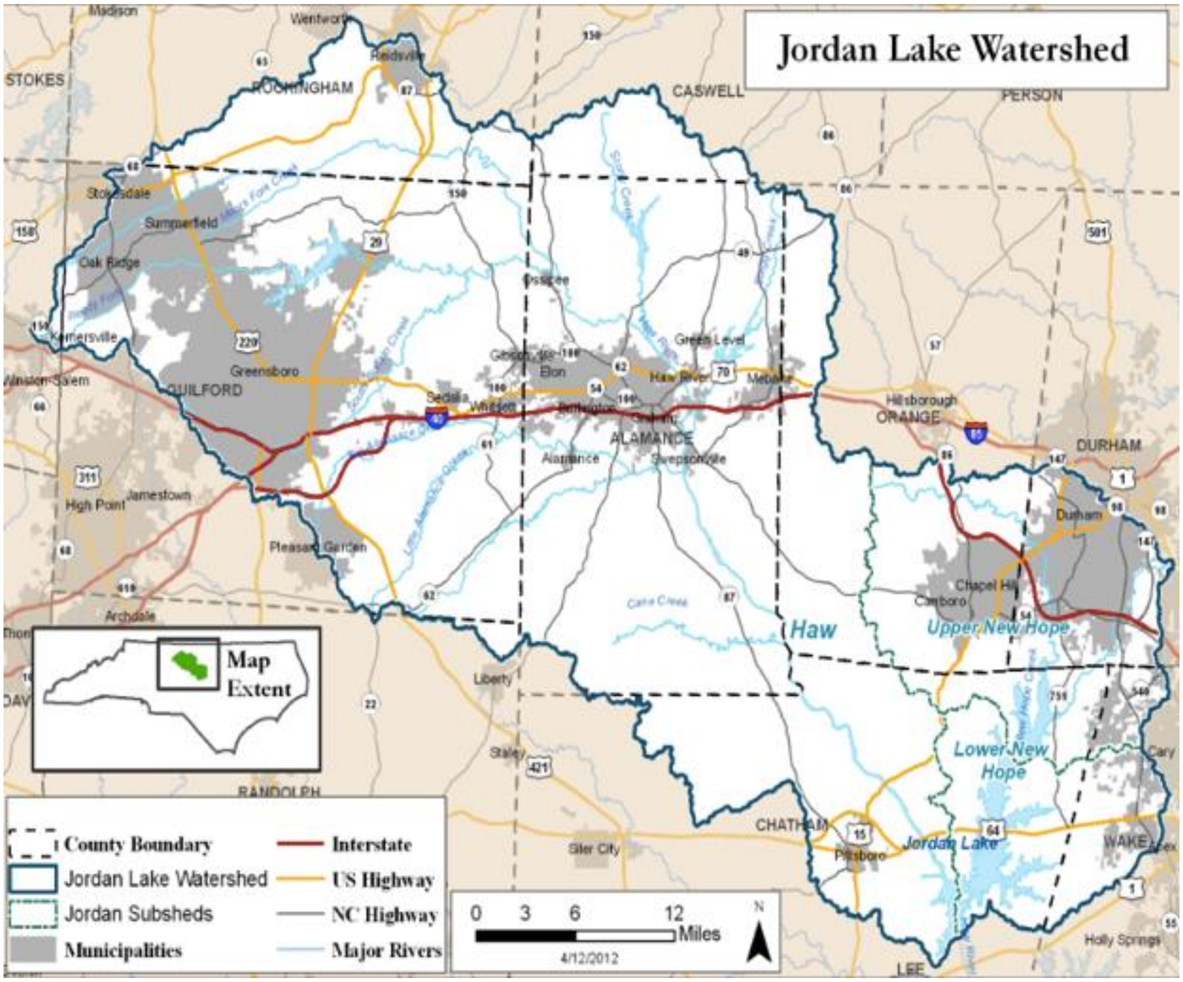


Figure 3-3. Jordan Lake Watershed (NCDEQ, 2018)

Target loading rates for new development under the JLRs for TN and TP are 1.96- and 0.73-kg/ha/yr, respectively (NCDEQ, 2018b). Under Session Law 2009-0484 Senate Bill 838, new developments unable to achieve the target loadings for TN and TP may offset the balance of TN and TP load reductions by off-site management; however, on-site TN loading may not exceed 5.35- and 8.92-kg/ha/yr for single family or duplex and multi-family or commercial development, respectively (GANC, 2009). The objectives of this chapter are (1) to monitor and analyze a RSC and (2) determine if an under-sized RSC in-series with a dry pond can reduce stormwater pollutants.

3.3. Materials and Methods

3.3.1. Field Site

An RSC in Morrisville, NC, was chosen for monitoring and treatment evaluation. The RSC was installed in a medium density residential neighborhood at the confluence of a dry pond outfall and a zero-order stream; the site forms an unnamed tributary to Kit Creek, Upper New Hope Creek, and Jordan Lake in Wake County (Figure 3-4).

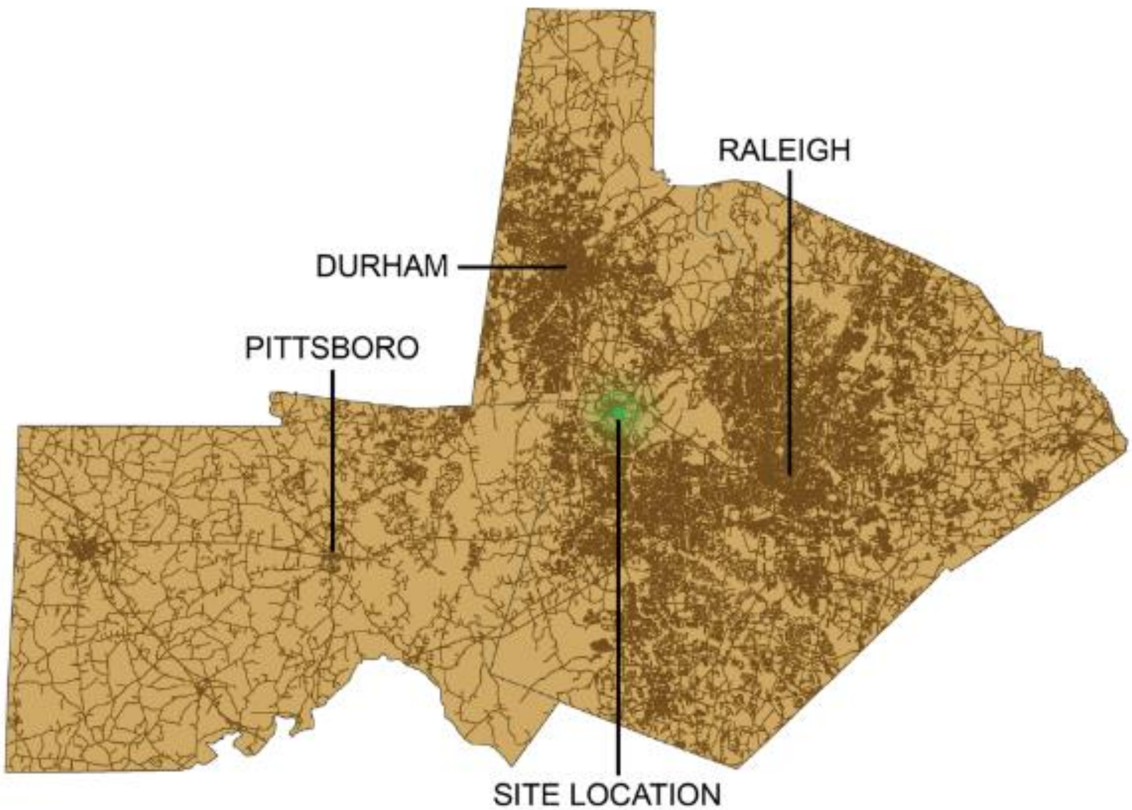


Figure 3-4. Site Context

The field site was located downstream of a dry pond outfall within a stream buffer in the Providence Place neighborhood, north of Mason Farm Rd (35.856720, -78.845823) (Figure 3-5). RSC installation stabilized a retreating headcut of the eroding tributary (Figure 3-6 B).

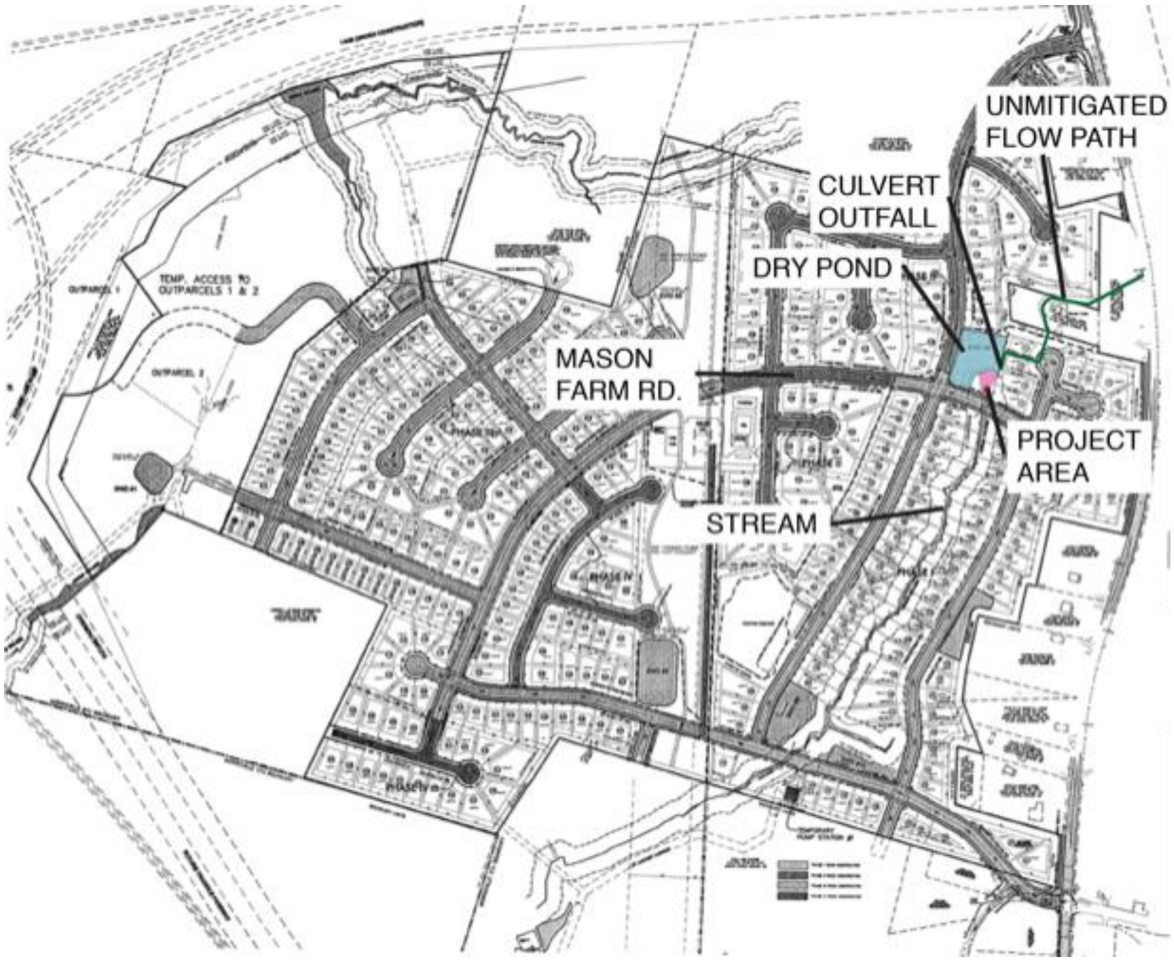


Figure 3-5. Site Map (Providence Place Construction Plans, Rice & Associates, 2005)

Figure 3-6. RSC Pre-Construction (A-B); Post-Construction (C-D)

3.3.2. Site Description

The drainage areas consisted of (1) medium density residential development which first passed through a dry pond (IN1) and (2) unmitigated urban and right-of-way drainage that was conveyed through reinforce concrete pipe (RCP) culvert and discharged near the dry detention basin outfall (IN2); some development drains directly into the project boundaries (Run-On) (Table 3-2; Figure 3-7). The unmitigated flow (IN2) disperses through a small wooded area before merging with flow from the detention basin (IN1) (Figure 3-8). Downstream of the IN1 and IN2 confluence the stream is conveyed under Mason Farm Rd via 1.2-m RCP, the stream forms an F or G type (Rosgen, 1994) channel draining portions of Providence Place before joining Kit Creek (Figure 3-5).

Table 3-2. Drainage Summary

Contributing Area	*Value (ha)*	*Curve Number*
IN1	7.2	86
IN2	12.4	84
Run-On	0.3	87

*CNs were calculated using aerial images and Eqs. 3.4-3.4.2

Figure 3-7. RSC Contributing Areas

Figure 3-8. Site Layout

3.3.3. RSC Construction

RSC construction took place Summer 2016, with two main objectives (1) stabilize an advancing headcut and (2) provide additional nutrient reduction in the Jordan Lake watershed. Headcut stabilization was accomplished using a series of boulder-cascades and Class B riprap to dissipate erosive flow velocities (Figure 3-6 D). RSC design features were used to convey dry pond effluent downslope; a channel was excavated and backfilled with sand and woodchip media (Figure 3-10 A), the exact composition of which is unknown (Figure 3-10 B). Boulder-weirs and riffle-pool sequences were constructed to form channel bed features, provide grade control, energy dissipation, and improve bed habitat (Figure 3-9; Figure 3-11).

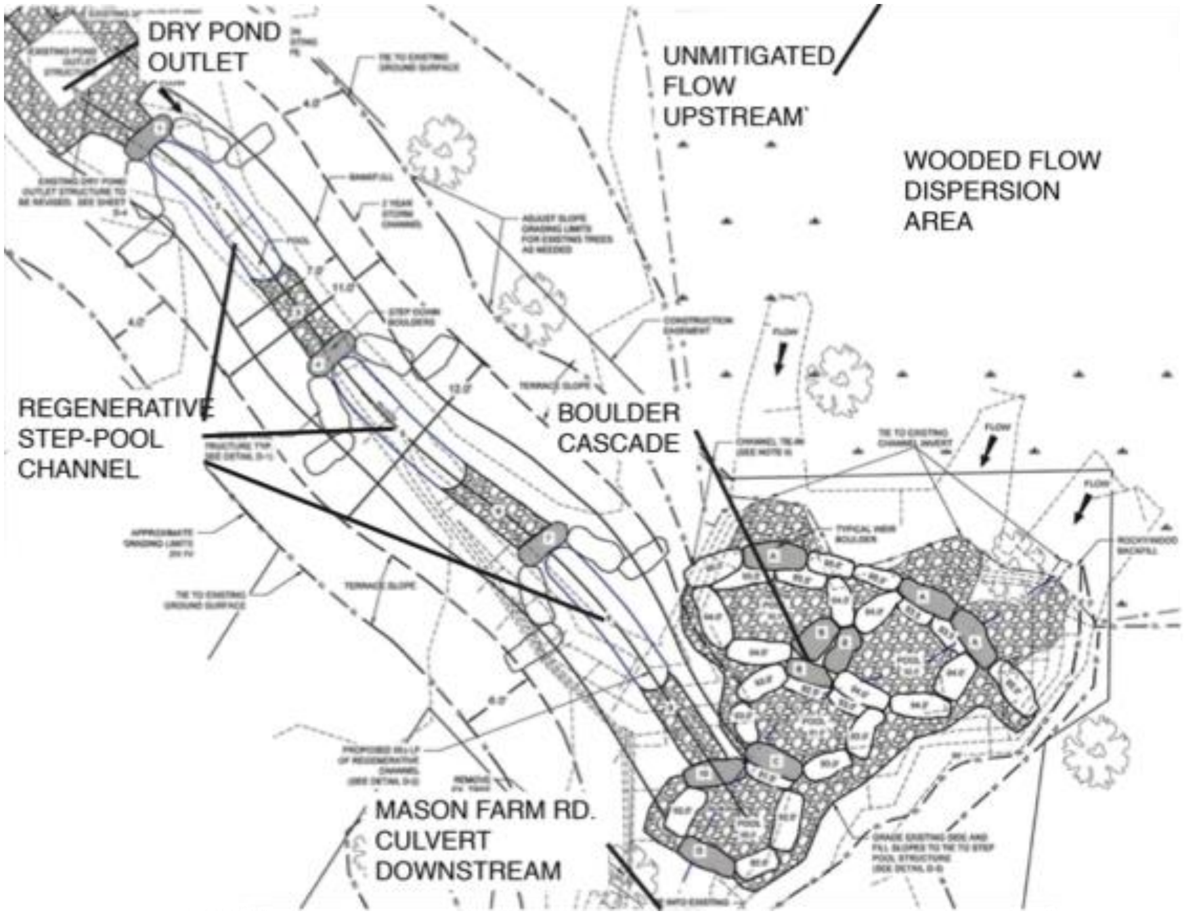


Figure 3-9. RSC Plan View (Stantec Construction Documents, 2016)

Figure 3-10. Exfiltration Trench Construction (A); Sand and Woodchip Media (B) (J. Baird, personal communication, 2017)


Figure 3-11. RSC 3 years post-construction

The RSC was constructed generally within the footprint of the existing gulley formed by the dry pond discharge (Figure 3-6 C); the small SCM area resulted in a design storm depth of 11.7-mm or 3.2% of the required capture volume in the North Carolina Piedmont (Table 3-3). A portion of flow entering this RSC initially flowed through a dry pond. Under-sizing in RSCs is common due to confined corridors or rights-of-way, construction within the existing gulley, and protection of riparian vegetation (Cizek, 2014; Cizek et al., 2016, 2017; Koryto et al., 2017). The RSC exfiltration trench was dug and filled with layers of washed #57 filter stone and sand and woodchip media; Class B stone and boulder-weir structures created bed forms (Figure 3-9; Figure 3-12). Shallow pool depths limited surface ponding and produced a low storage ratio, much lower than other RSCs studied (Brown et al., 2010; Cizek, 2014; Cizek et al., 2016, 2017, Koryto et al., 2018, 2017). An estimated 1.98-m^3 of surface ponding exists within the wooded wetland area where flow disperses downstream of IN2 (Figure 3-9).

Table 3-3. RSC Characteristics

Parameter	*System Values*
Design Depth (mm)	11.7
Loading Ratio[1]	25,000:1
Length (m)	16.8
Slope (m/m)	0.04
Surface Area (m^2)	4.04
Pool Depth (m)	0.10-0.16
Media depth (m)	0.61-1.52
Pond Storage (m^3)	0.3
Media Storage (m^3)	27.6
Storage Ratio[2]	0.01:1
Media Composition	Unknown

1) $Loading\ Ratio = \frac{Watershed\ Area}{SCM\ Area}$

2) $Storage\ Ratio = \frac{Surface\ Storage\ Volume}{Media\ Storage\ Volume}$

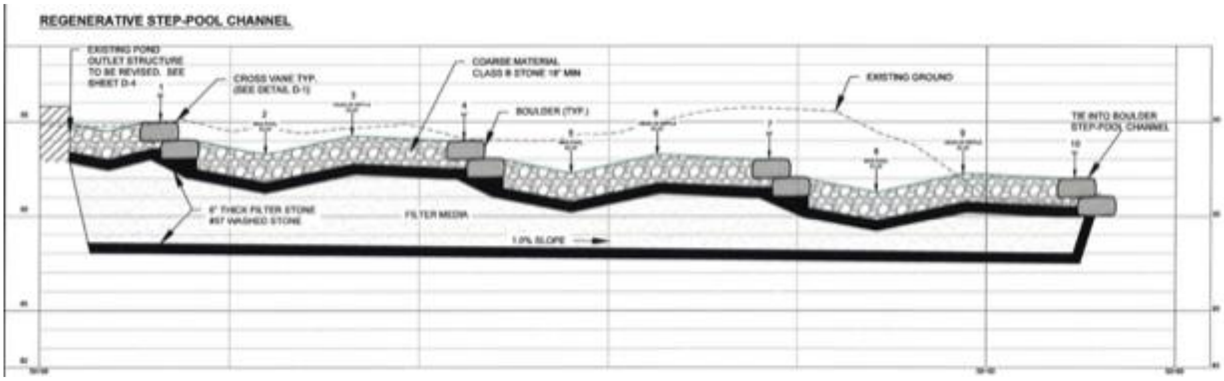


Figure 3-12. RSC Cross-Section (Stantec Construction Documents, 2016)

<u>3.3.4. Monitoring Design</u>

Hydrologic monitoring began July 12, 2017, and concluded July 9, 2018 (Table 3-4). On September 14, 2017, a concrete splash pad was installed around the dry pond outlet structure, upstream of the IN1 weir (Table 3-4 A); this improved the ability to measure low flows.

Table 3-4. Monitoring Design

Location	*Monitoring Period*	*Equipment Used*	*Parameters Analyzed*
IN1	July 12, 2017 – July 9, 2018	ISCO 6712 Automated Sampler with ISCO 730 Bubbler Flow Module, Manual + Tipping Bucket Rain Gauge, Base Flow Grab samples, & Cross-Channel Weir	TSS, NO2,3-N, TKN, TAN, TP, Ortho-P, Temperature, Water Level
DRYOUT (Pond)	May 25, 2018 – July 9, 2018	HOBO U20 Water Level Logger	Water Level
IN2	February 9, 2018 – July 9, 2018	ISCO 6712 Automated Sampler with ISCO 730 Bubbler Flow Module & Base Flow Grab samples, HOBO U20 Water Level Logger, &Cross-Channel Weir	TSS, NO2,3-N, TKN, TAN, TP, Ortho-P, Temperature, Water Level
OUT	July 12, 2017 – July 9, 2018	ISCO 6712 Automated Sampler with ISCO 750 Area Velocity Meter & Base Flow Grab samples	TSS, NO2,3-N, TKN, TAN, TP, Ortho-P, Temperature, Water Level
WELL	October 18, 2017 – July 9, 2018	HOBO U20 Water Level Logger	Water Level

On October 18, 2017, a groundwater well was installed into the media layer in pool 3 to monitor media saturation and water depth (Figure 3-14 B). The well casing was constructed of 3-cm diameter PVC pipe with 10-mm perforations wrapped with a well sock (Figure 3-14 A); the

HOBO logger was attached to a 1-cm PVC pipe and inserted into the casing to improve continuity of measurement depth over time. A manual and HOBO tipping bucket rain gauge (0.25-mm/tip) was installed 0.9-m above the ground, in an area free of overhanging branches or other obstructions, to measured rainfall depth (Table 3-4; Figure 3-8; Figure 3-13 D).

Figure 3-13. Monitoring Locations & Equipment: IN1 (A); OUT (B); Auto-Sampler (C); Rain Gauge (D); IN2 (E)

Figure 3-14. Media Well Casing & Well Sock (A); Well Installed (B)

On February 9, 2018, water quality sampling equipment was installed at the second, untreated inlet (henceforth referred to as IN2) (Figure 3-8;

Figure 3-13 E). Base flow sampling was conducted between April 5, 2018, and July 5, 2018, to characterize inter-event pollutant load. ISCO 6712 automated samplers with ISCO 730 Bubbler flow modules were used at the inlets (

Figure 3-13 C). An ISCO 6712 automated sampler with an ISCO 750 Area Velocity Meter (AVM) was used at the outlet (

Figure 3-13). Installation of a HOBO water level logger within the dry pond outlet structure (henceforth referred to as DRYOUT) was required for calibration of IN1 flow rates (

Figure 3-13 A), as knowing the depth of water in the pond enabled calculation of the flow through outlet orifices.

Flow-proportional composite sampling was attempted; however, hydrologic monitoring errors prevented accurate sample collection. Instead, composite samples were collected throughout runoff events producing a Representative Concentration (RC) for the sampled event; RCs are not necessarily Event Mean Concentrations (EMCs) because of the flow measurement inaccuracies. Compound contracted weirs were employed at IN1 and IN2 for flow calculation using stage-discharge relationships (Table 3-5; Eq. 3.1 – 3.2.1).

Table 3-5. Compound Weir Geometries

Location	*V-Notch weir*	*Contracted Weir*
IN1	60º height = 0.15-m	length = 0.61-m height = 0.46-m
IN2	60º height = 0.9-m	length = 1.07-m height = 0.31-m

$$Q_{IN1_{stage<0.15m}} = (1344 * IN1_{stage}^{2.5}) \quad \text{(Eq. 3.1)}$$

$$Q_{IN1_{stage>0.15m}} = (1344 * 0.15^{2.5} + 3970.8 * IN1_{stage}^{1.5}) - (1323.6 * IN1_{stage}^{1.5}) \quad \text{(Eq. 3.1.1)}$$

$$Q_{IN2_{stage<0.09m}} = (2868 * IN2_{stage}^{2.5}) \quad \text{(Eq. 3.2)}$$

$$Q_{IN1_{stage>0.09m}} = (2868 * 0.09^{2.5} + 7279.8 * IN1_{stage}^{1.5}) - (2647.2 * IN1_{stage}^{1.5}) \quad \text{(Eq. 3.2.1)}$$

3.3.5. Hydrologic Analysis

3.3.5.1. Water Balance

RSC water balance was conducted across the system as follows (Eq. 3.3):

$$Surface_{out} = Surface_{in} + RO + P - ET - Exfiltration \quad \text{(Eq. 3.3)}$$

Where,

$Surface_{out}$ = Includes surface outflow, change in subsurface, and surface (m^3)

$Surface_{in}$ = Surface inflow, directly measured and calculated sum of IN1 and IN2 (m^3)

RO = Run-on enters RSC as overland flow; estimated by SCS CN method (m^3)

P = Precipitation, estimated by direct rainfall on cell surface area (m^3)

ET = Evapotranspiration was estimated using daily Open Water Penman-Monteith reference values (NC Climate Office, 2018) (m^3)

$Exfiltration$ = estimated infiltration to surrounding subsoils (m^3)

3.3.5.2. Precipitation

Manual rain gauge measurements were used to scale tipping bucket rain gauge measurements over the same period because tipping bucket rain gauges notoriously under-predict rainfall during high intensity rainfall. Seasonal analysis was conducted based on astronomical seasons (e.g., Spring 03/01-05/31, Summer 06/01-08/31, Autumn 09/01-11/31, Winter 12/01-02/28).

3.3.5.3. Run-On

A run-on area of 0.7-ha (CN=87, Table 3-2) flows overland to the RSC without upstream measurement; run-on volumes for discrete events were estimated using the NRCS Curve Number Method (Eq. 3.4 – 3.4.1) (NCDEQ, 2017). Composite CN were used in areas of multiple land uses (Eq. 3.4.2).

$$Q = \frac{(P-0.2S)^2}{(P-0.2S)+S} * A * C \qquad \text{(Eq. 3.4)}$$

$$S = 25.4 * (\frac{1000}{CN} - 100) \qquad \text{(Eq. 3.4.1)}$$

$$CN = \frac{CN_i * A_i}{A_T} \qquad \text{(Eq. 3.4.2)}$$

Where,

Q = Runoff Volume (m^3)

P = Precipitation depth (mm)

A = Watershed area (m^2)

S = Potential maximum retention (mm)

CN = Composite Curve Number

C = Conversion factor: $\frac{1\,m}{1000\,mm}$

i = area 1, 2, 3, …, n

3.3.5.4. Evapotranspiration

RSC surface vegetation was not established during the monitoring period; however, IN2 flow disperses through a wooded area with some understory wetland vegetation of Common Rush (*Juncus effusus*). Vegetated cover and swamps were documented to have mixed effects on total open water evaporation (EVAP) (Idso, 1981; Munro, 1979). A crop coefficient (K_c) (Allen et al., 1998) was applied to daily open water Penman-Monteith EVAP depths, retrieved from the North Carolina State Climate Office for Raleigh-Durham (RDU) International Airport (site ID: KRDU), 5.5-km from the project site (NC Climate Office, 2018) to calculate ET over discrete event durations when rainfall was not occuring (Eq. 3.5) (Huffman et al., 2013).

$$ET_i = \sum_{i=1}^{n} K_c * EVAP \qquad \text{(Eq. 3.5)}$$

Where,

ET_i= Evapotranspiration for storm i (mm)

K_c= Wetlands Crop Coefficient, 1.2 (unitless)

$EVAP$ = Penman-Monteith Evaporation for Reference Site (mm)

i = storm event 1, 2, 3, …, n

3.3.5.5. Volume Metrics

For discrete rainfall events, changes in surface flow volumes were compared cumulatively from inlets to outlet; inflow, direct precipitation, and run-on runoff were summed to determine system inflow (Eq. 3.6).

$$V_{in} = V_{IN1} + V_{IN2} + Q + \frac{P}{1000} * A \qquad \text{(Eq. 3.6)}$$

Where,

V_{in} = Inflow volume (m^3)

V_{IN1} = IN1 observed flow volume (m^3)

V_{IN2} = IN2 observed flow volume (m^3)

Q = Run-on volume (m^3)

P = Direct Precipitation (mm)

A = RSC surface area (m^2)

Pre-event stage returned within 12-hrs of a storm's completion; therefore, volume reduction metrics were assessed based on influent and effluent flows during both the event duration and the following 12-hr drawdown period (Eq. 3.6.1).

$$VR = \frac{\sum_{i=1}^{n} V_{in-12,i} - \sum_{i=1}^{n} V_{out-12,i}}{\sum_{i=1}^{n} V_{in-12,i}} * 100 \qquad \text{(Eq. 3.6.1)}$$

Where,

V_{in-12} = Inflow volume (includes storm duration and following 12-hrs) (m^3)

V_{out-12} = outflow volume (includes storm duration and following 12-hrs) (m^3)

i = storm event 1, 2, 3, …, n

VR = Percent storm flow volume reduction (%)

3.3.6. Water Quality Analysis

3.3.6.1. Concentrations

Paired inlet samples were combined according to their fraction of contribution of total inflow (Eq. 3.7):

$$RC_i = fr_1 * RC_{1_i} + fr_2 * RC_{2_i} \qquad \text{(Eq. 3.7)}$$

Where,

RC_i = Representative Concentration (mg/L)

fr_1 = Median IN1 fraction of total flow, 0.32 (unitless)

RC_{1_i} = Representative sample concentration IN1 for storm i (mg/L)

fr_2 = Median IN2 fraction of total flow, 0.28 (unitless)

RC_{2_i} = Representative sample concentration IN2 for storm i (mg/L)

i = sampled event 1, 2, 3, …, n

Discrete event RC inflow and outflow samples were tested for significance and Removal Efficiency (RE) were determined (Eq. 3.7.1) for each sampled event. Cumulative probability plots were used to compare inlet and outlet RCs to ambient water quality targets (McNett et al., 2010).

$$RE_i = \frac{RC_{in_i} - RC_{out_i}}{RC_{in_i}} * 100 \qquad \text{(Eq. 3.7.1)}$$

Where,

RE_i = Removal Efficiency for storm i (%)

i = sampled event 1, 2, 3, …, n

RC_{in} = inflow representative concentration (mg/L)

RC_{out} = outflow representative concentration (mg/L)

RC ratios were used to determine whether inlet concentrations had been reduced at the point of discharge (Eq. 3.7.2):

$$RC\ Ratio = \frac{RC_{in}}{RC_{out}} \qquad \text{(Eq. 3.7.2)}$$

Where,

RC_{in} = Inflow representative concentration (mg/L)

RC_{out} = Outflow representative concentration (mg/L)

3.3.7. Statistical Analysis

Statistical analysis was performed using R Software Version 3.3.3 (R Core Team, 2017); a repository for data and documentation of processing methods has been established (Gregg, 2018). Hydrologic and water quality data were tested for normality and log-normality by visual inspection of histograms, quantile-quantile plots and Shapiro-Wilk normality test. Hydrologic measurements and temperature were determined to be non-normal. Base flow pollutants TKN, TAN, TP, and OP, and storm flow $NO_{2,3}$-N were determined to be normal; all other pollutants were non-normal. Visual inspection with histograms and normal quantile-quantile plots and the Shapiro-Wilks Normality test were used in normality determination; non-parametric methods were used for non-normal data. For pollutants with normal distribution, the Paired-Sample t Test were used (Helsel & Hirsch, 2002). Simple Linear Regression (SLR) was employed to study the relationships between various predictor and explanatory variables and improve the accuracy of hydrologic storm flow data (Helsel & Hirsch, 2002). Autocorrelation was tested with the Durbin-Watson test and Run's test (Helsel & Hirsch, 2002); correction was conducted by averaging data over larger time intervals. Test of significance were conducted using the Wilcoxon Signed-Rank Sum and Wilcoxon Rank sum Tests with a Type 1 error (α) of 0.05 (Wilcoxon, 1945).

3.4. Results & Discussion

3.4.1. Weather Summary

Discrete rainfall events occurred when dry periods of at least 12 hours followed rainfall cessation. During the approximately 12-month monitoring period between July 12, 2017, and July 9, 2018, 93 discrete rainfall events totaled 972.6-mm, 83% of average annual precipitation at RDU International Airport, 5.5-km away (Table 3-6) (U.S. Climate Data, 2018).

Table 3-6. Morrisville RSC Rainfall Summary

Event	*Parameter*	*Depth (mm)*	*Duration (hr)*	*5-min Max. Int. (mm/hr)*	*ADP (days)*
Rainfall (n=93)	*Range*	0.25-83.1	12.0-52.0	3.0-115.8	0.01-14.0
	Median	4.19	16.0	7.62	2.26
	Sum	972.6	-	-	-
Hydrology (n=15)	*Range*	6.4-38.6	1.0-20.4	13.7-109.7	0.04-4.9
	Median	21.3	14.1	27.4	1.9
	Sum	287.8	-	-	-
Water Quality (n=10)	*Range*	12.7-44.2	13.5-51.4	3.0-109.7	0.04-4.9
	Median	22.5	35.7	11.4	1.9
	Sum	253.5	-	-	-

The portion of Summer 2017, and Autumn-Winter 2017-2018, were dry with 71% and 56% average annual precipitation, respectively; while the portion of Summer 2018 was wet with 124% average annual precipitation (U.S. Climate Data, 2018). The largest event, 83.1 mm, occurred on April 15, 2018, and generated a 5-min maximum intensity of 76.2-mm/hr; this event accumulation exceeded a 1-year, 24-hour design storm (NOAA National Weather Service, n.d.). The largest 5-min maximum intensity of 109.7-mm/hrs occurred on July 5, 2018; this 5-min maximum intensity did not exceed a 1-year, 5-min intensity (NOAA National Weather Service, n.d.). Fifteen and 10 runoff events were analyzed for hydrology and water quality, respectively. The depths of these events represented 30% and 26% of total observed precipitation, respectively (Table 3-6). Compared to total observed rainfall, hydrologic and water quality events have smaller ranges and higher medians, except for Antecedent Dry Period (ADP).

3.4.2. Hydrology

3.4.2.1. Outlet Flow Measurement

Outflow velocity had a high degree of variability (Figure 3-15). This is representative of the conditions in which measurements were recorded. The outlet AVM was installed in an area of highly turbulent flow, approximately 1.8-m downstream of the culvert invert and approximately 1.8-m upstream of an in-culvert gravel bar (

Figure 3-13 B). Eddies and turbulence in the vicinity of the doppler ultrasonic sensor were recorded as rapid changes in velocity, both positive and negative (Figure 3-15)(Teledyne ISCO, 2013); negative velocities are representative of flow moving away from (upstream) the doppler sensor and are typical if eddies form. The 750 AVM module has reported velocity accuracy for a range of flow rates (Table 3-7; highlighted in Figure 3-15).

Table 3-7. 750 AVM Velocity Accuracy (Teledyne ISCO, 2013)

Accuracy (m/s)	*Velocity Range (m/s)*
±*0.03*	-1.5-1.5
±*2%*	1.5-6.1

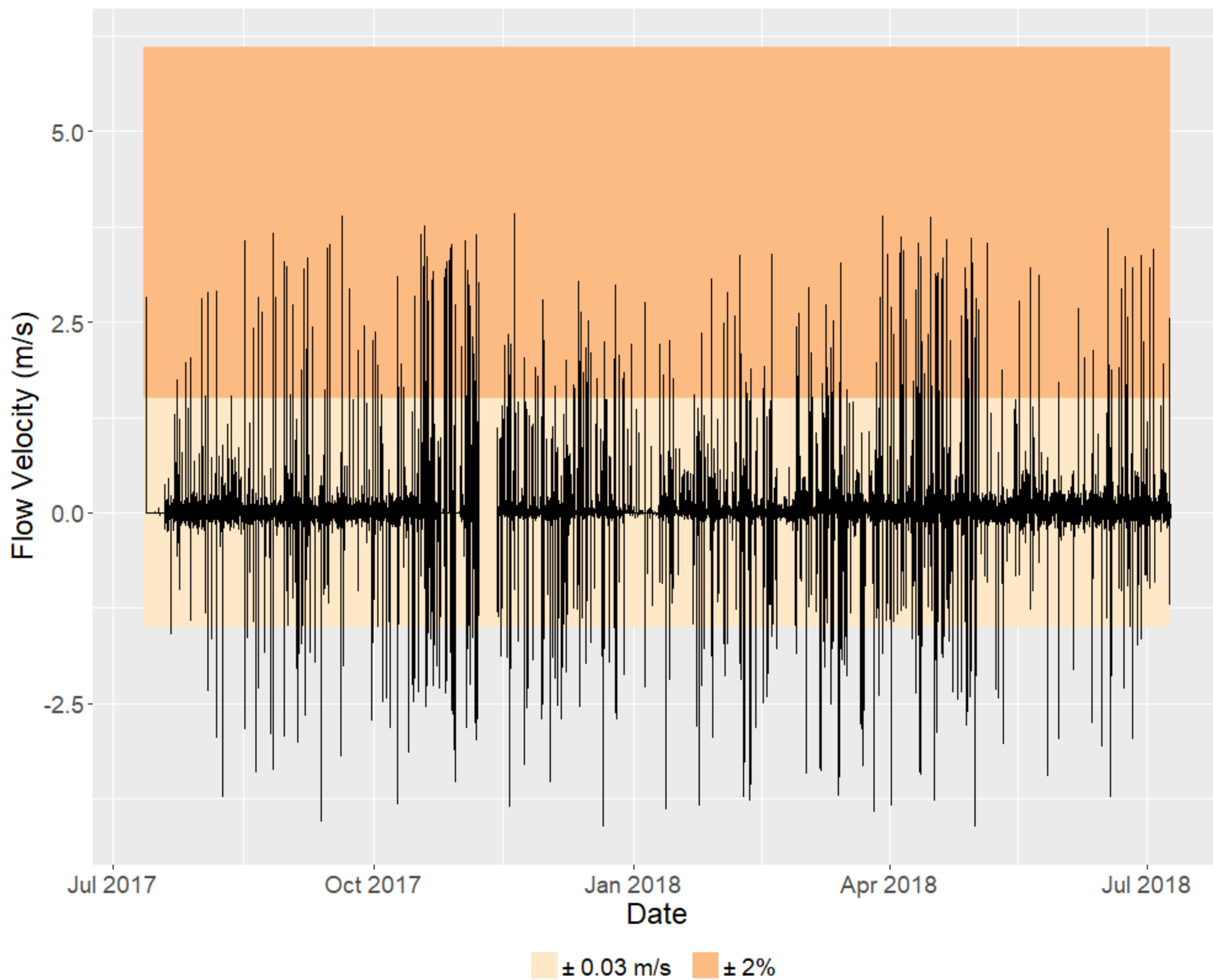


Figure 3-15. Outflow Velocity Profile

Positive velocity measurements are within the range for reported error; however, negative velocities exceed the range for reported error (Figure 3-15). Negative velocities outside the reported accuracy range likely introduce substantial error. Inflation of negative velocities over a year-long flow period will reduce cumulative flow volume. No additional data were recorded for purposes of AVM calibration or correction; therefore, no AVM calibration or corrections were possible.

3.4.2.2. Inlet Flow Measurement

The concrete splash pad surrounding the dry pond outlet structure improved the ability to measure low flows; however, at high flows (i.e., during storm events) water overtopped the splash pad (Figure 3-16 B) and flowed underneath the splash pad and weir, bypassed measurement at the IN1 weir, and later reemerged downstream (Figure 3-16 C).

Figure 3-16. DRYOUT Hydraulic Structures (A); Overtopping Slash Pad (B); Upwelling Downstream of Weir (C)

3.4.2.2.1. Storm Flow Correction

A HOBO pressure transducer was installed in the dry pond outlet structure to allow influent flow calculation based on the outlet hydraulic control structure. The dry pond outlet contained two—10.2-cm orifices and four—contracted broad-crested weirs with a total length of 4.21 m (Figure 3-16 A); under normal conditions the orifices were submerged. Influent calculations were made using (Eq. 3.9 & 3.9.1)(NCEES, 2013):

$$Q = 2\,C_d A\sqrt{2g(h_1 - h_2)} \qquad 5.1 \geq h_1(\text{cm}) \geq 19.2 \qquad \text{(Eq. 3.9)}$$

$$Q = \left(2\,C_d A\sqrt{2g(h_1 - h_2)}\right) + \left(C(L - 0.2h_1)\left({h_1}^{1.5}\right)\right) \qquad 19.2 \geq h_1(\text{cm}) \qquad \text{(Eq. 3.9.1)}$$

Where,

Q = orifice discharge (m^3/s)

C_d = orifice coefficient of discharge (0.41) (unitless)

C = broad crested weir coefficient of discharge (1.84) (unitless)

A = office cross-sectional area (m^2)

g = gravity acceleration constant, 9.807-m/s^2

h_1 = head above orifice center inside outlet structure (m)

h_2 = head above orifice center outside outlet structure (m)

L = broad crested weir length (m)

The submerged orifice flow calculation (Eq. 3.9) required upstream and downstream head measurements; geometric relationships between known inverts and field measurement locations were utilized to calculate flow. IN1 weir stage represented downstream head for orifice calculations.

Monitoring within the dry pond outlet structure was conducted from May 25, 2018, to July 9, 2018 (Table 3-4). Six events ranging in accumulation (6.35—54.1-mm) and 5-min maximum rainfall intensity (13.71—109.73-mm/hr) occurred during that time (Figure 3-17). On July 7, 2018, a 54.1-mm event with a 5-min maximum rainfall intensity of 83.8-mm/hr damaged the cross-channel weir and required its removal from the correction dataset; this event also prompted the end of data collection. Hydrologic data for the remaining 5 events were used to create a flow correction model for IN1 weir measurements (Figure 3-18).

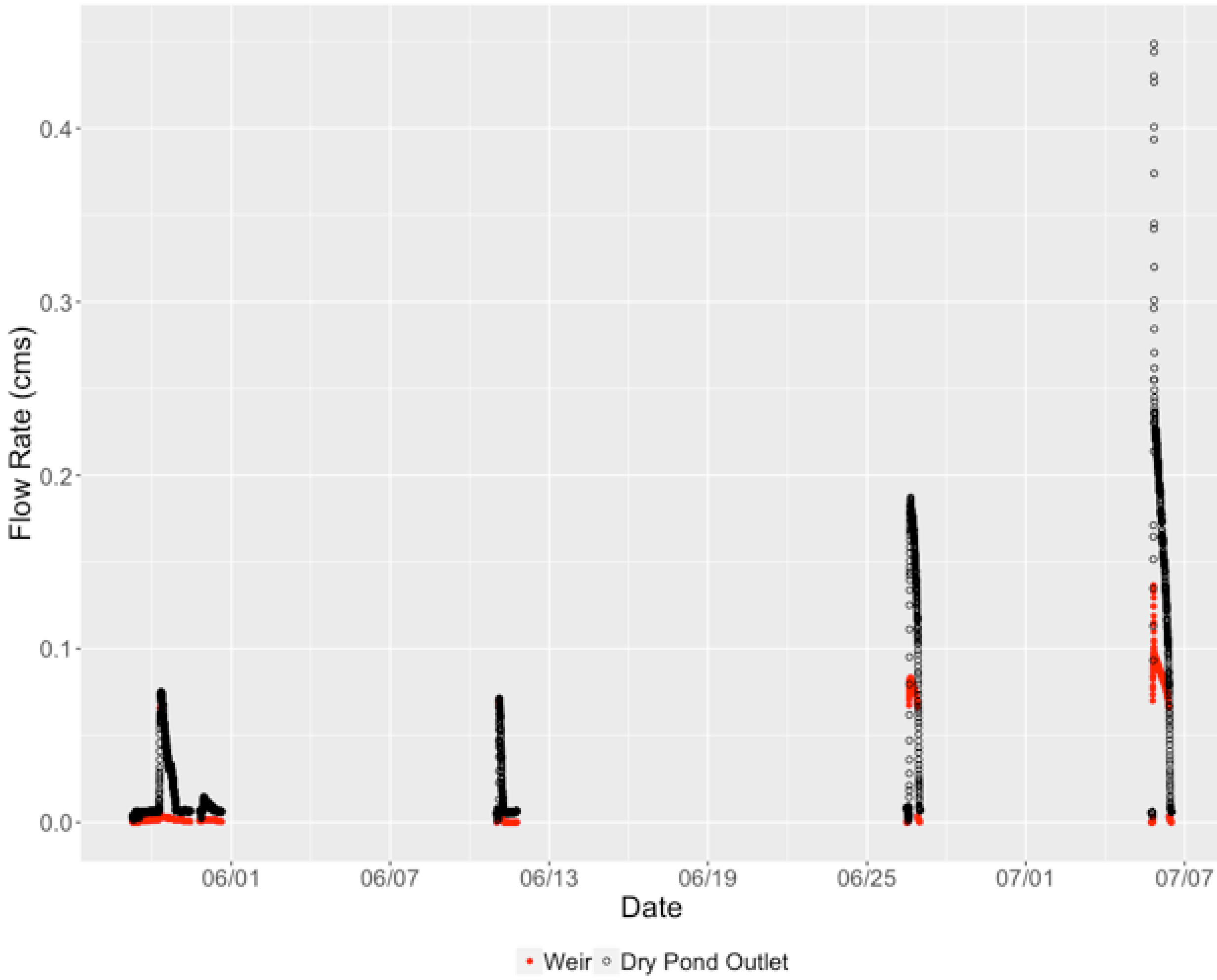


Figure 3-17. IN1 Calibration Events

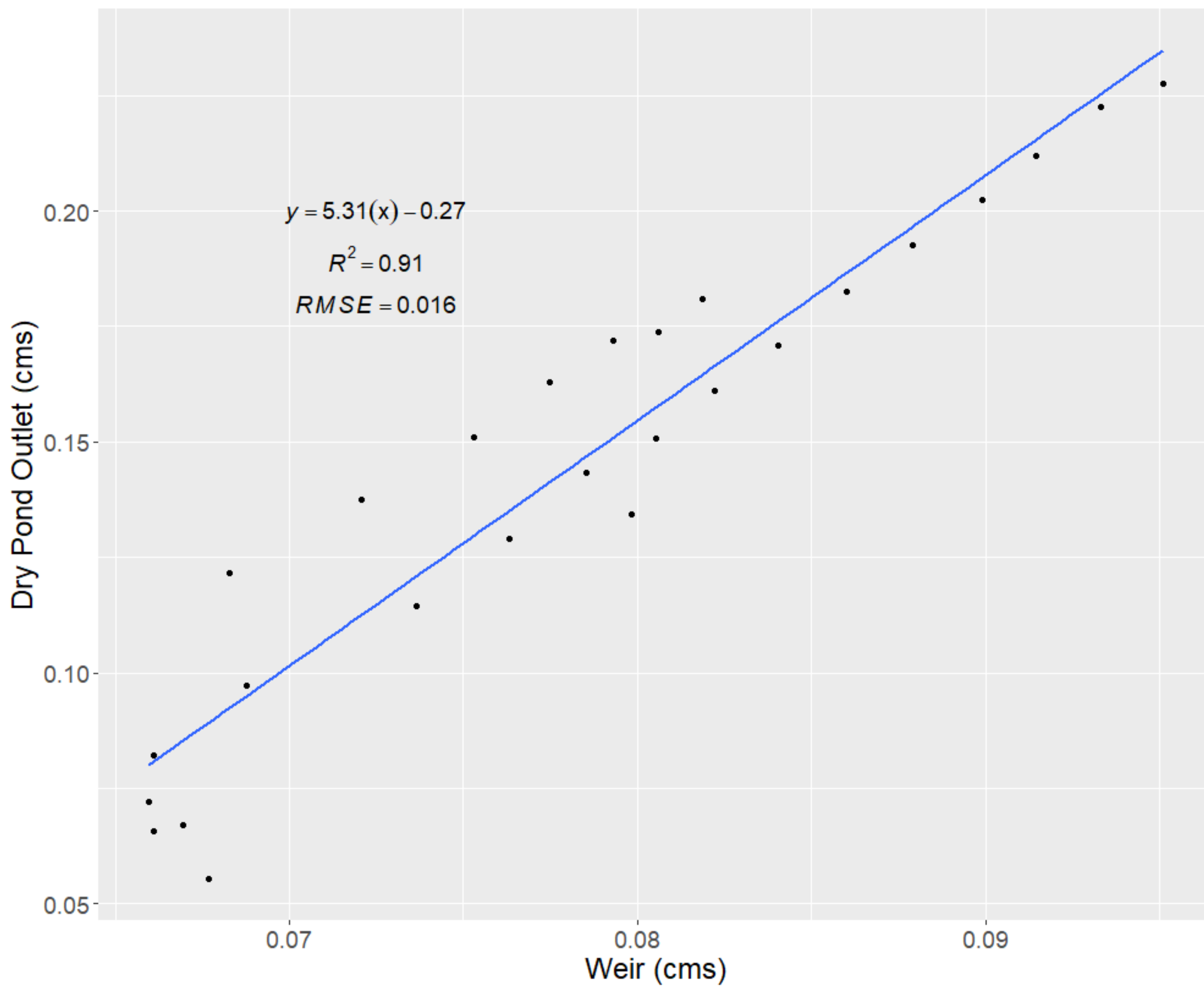


Figure 3-18. IN1 Event Flow Linear Correction Model

IN1 measurements were used as predictors, and DRYOUT measurements were used as the response (Figure 3-18). Predictors were determined to be normal when weir flow measurements less than or equal to 0.066 cms were excluded (Appendix D). This correction model is applicable to flow rates greater than or equal to 0.066 cms when discrete event rainfall accumulations and max 5-min intensities are within the range of 6.35—38.61-mm and 13.71—109.73-mm/hr, respectively. Twenty events met these criteria; however, the first five occurred before splash pad installation and were thus removed from analysis. The remaining fifteen events comprised the data for hydrologic analysis. IN1 flow linear correction induced some over prediction error during lower accumulation events (Figure 3-19).

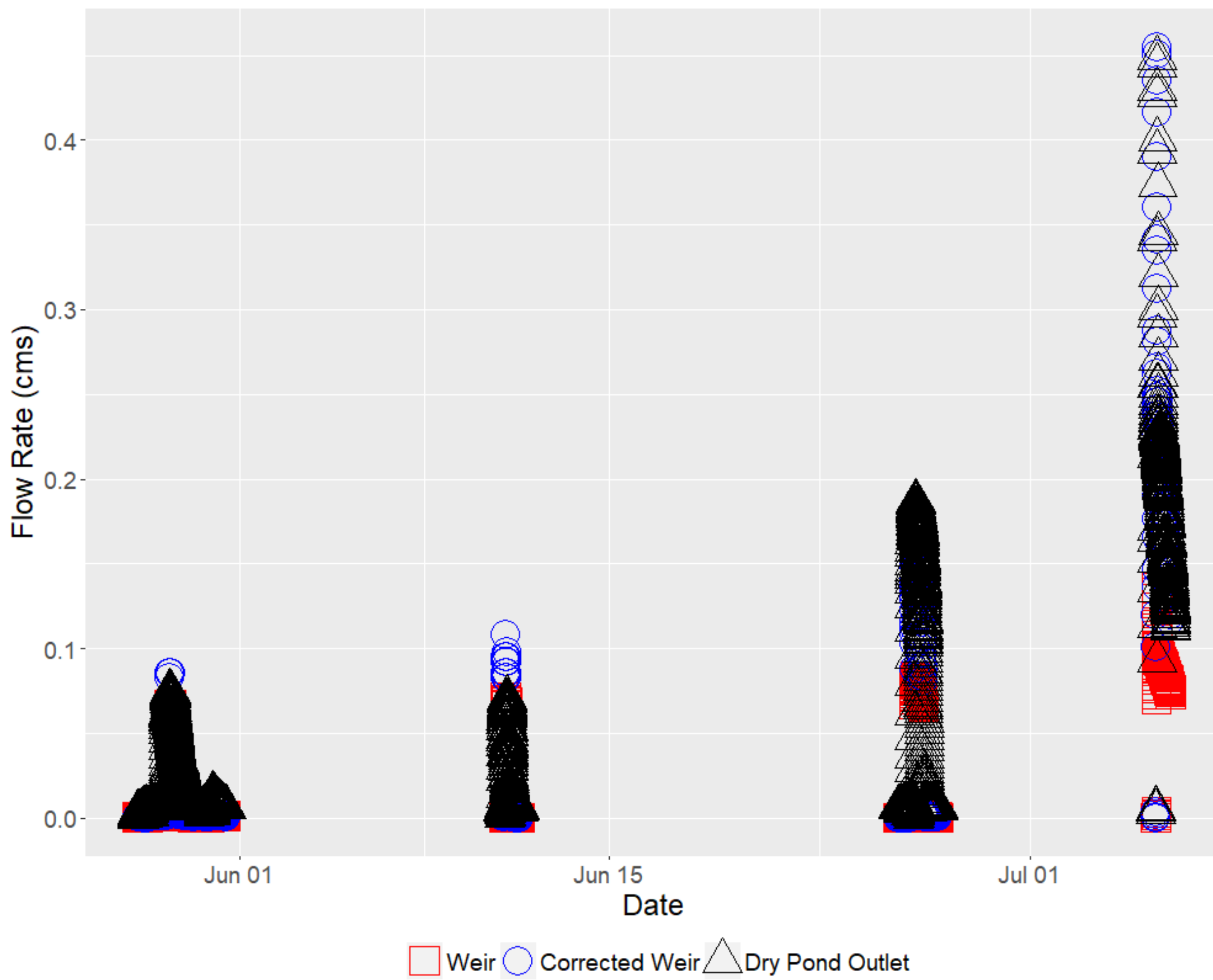


Figure 3-19. IN1 Corrected Flows

3.4.2.2.2. Base Flow Correction

Similar to event flows, base flow corrections were necessary to correct for seepage that was observed to occur between the splashpad and outlet structure/weir (Figure 3-16 B, Figure 3-20). DRYOUT (response variable) was compared to IN1 (predictor variable) during inter-event, low flow, periods.

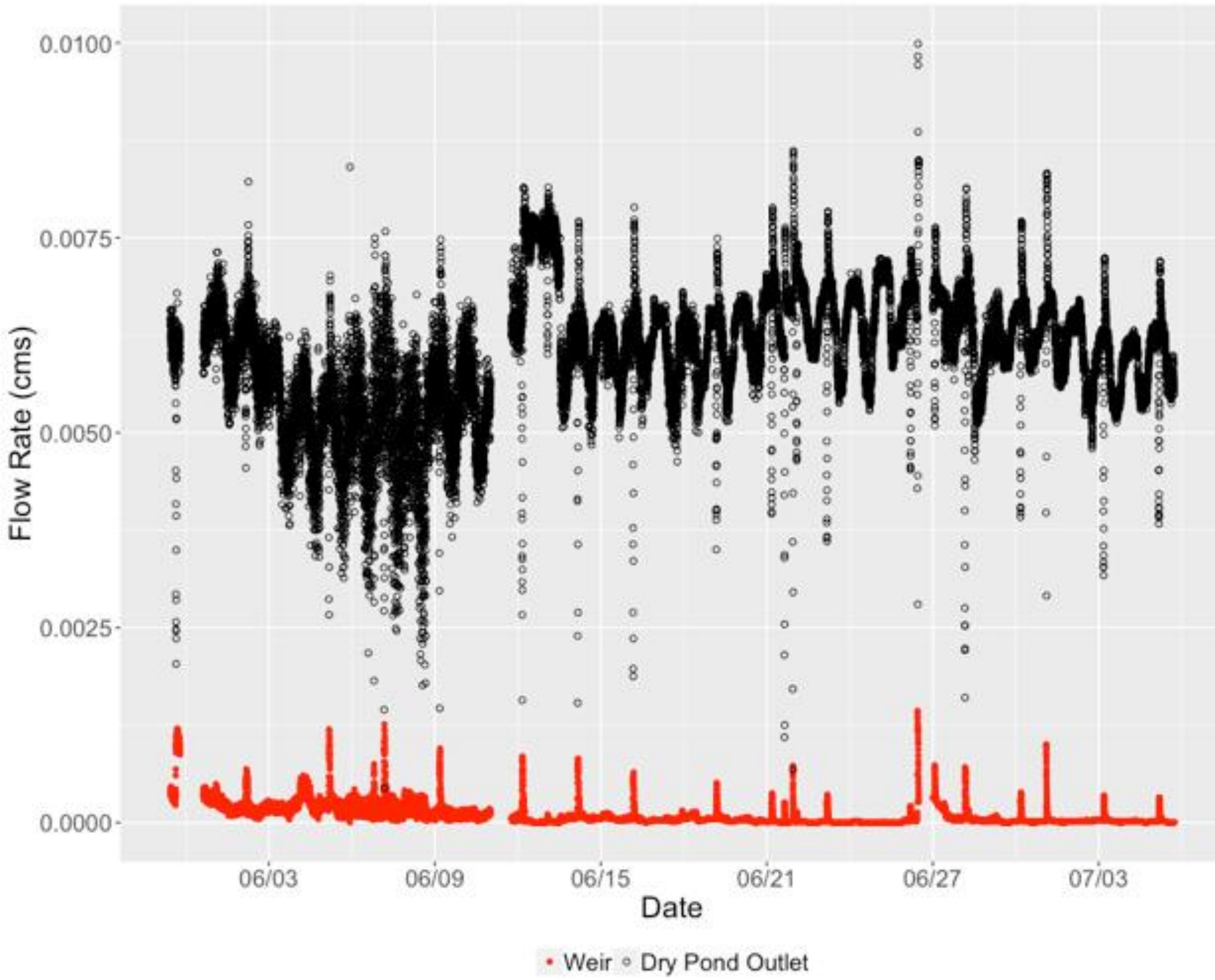


Figure 3-20. Baseflow Calibration Events

Predictor normality was assessed visually and determined to be acceptable (Appendix E). Model autocorrelation was corrected by averaging 2-min flow data over 24-hour periods. One outlier was removed substantially improving model fit (Figure 3-21). While this model is less accurate than that described in Figure 3-18 it was applied to weir base flow rates up to 0.0012 cms. Driving head for submerged orifice flow (Eq. 3.9) leaving the dry pond outlet structure had an inverse relationship with driving head for the IN1 cross-channel weir; the slope (-4.1) of this relationship is significantly different from zero and much larger than 1; this suggested that (1) the validity of the assumption that some water was unaccounted for as it flowed off the splash pad during low flows and (2) it should be corrected for a more accurate analysis.

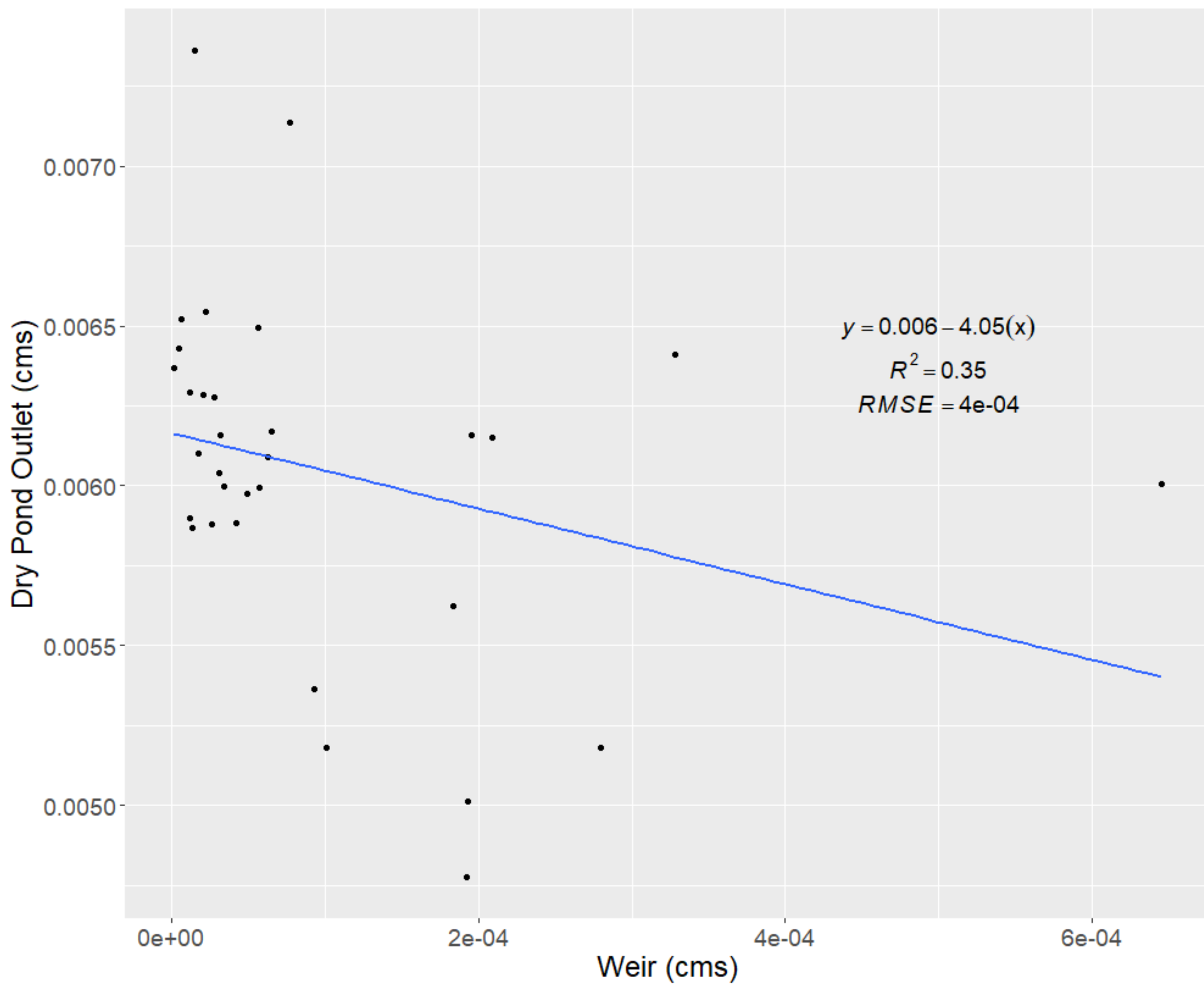


Figure 3-21. Baseflow Linear Correction Model

IN1 model predictions were applied to baseflows less than or equal to 0.0012 cms; higher flows are not corrected. Model application produced negative flow rates (Figure 3-22). During baseflow periods, all flow left the dry pond as submerged orifice flow; when stage increases at the cross-channel weir tailwater conditions occur in the outlet structure, restricting orifice flow. The correction model was unable to characterize the relationship between the dry pond outlet flows and those of the cross-channel weir under these conditions. Baseflow corrections were not applied; baseflow analysis was thus conducted using uncorrected IN1 cross-channel weir measurements, which probably means some values are under-predicted.

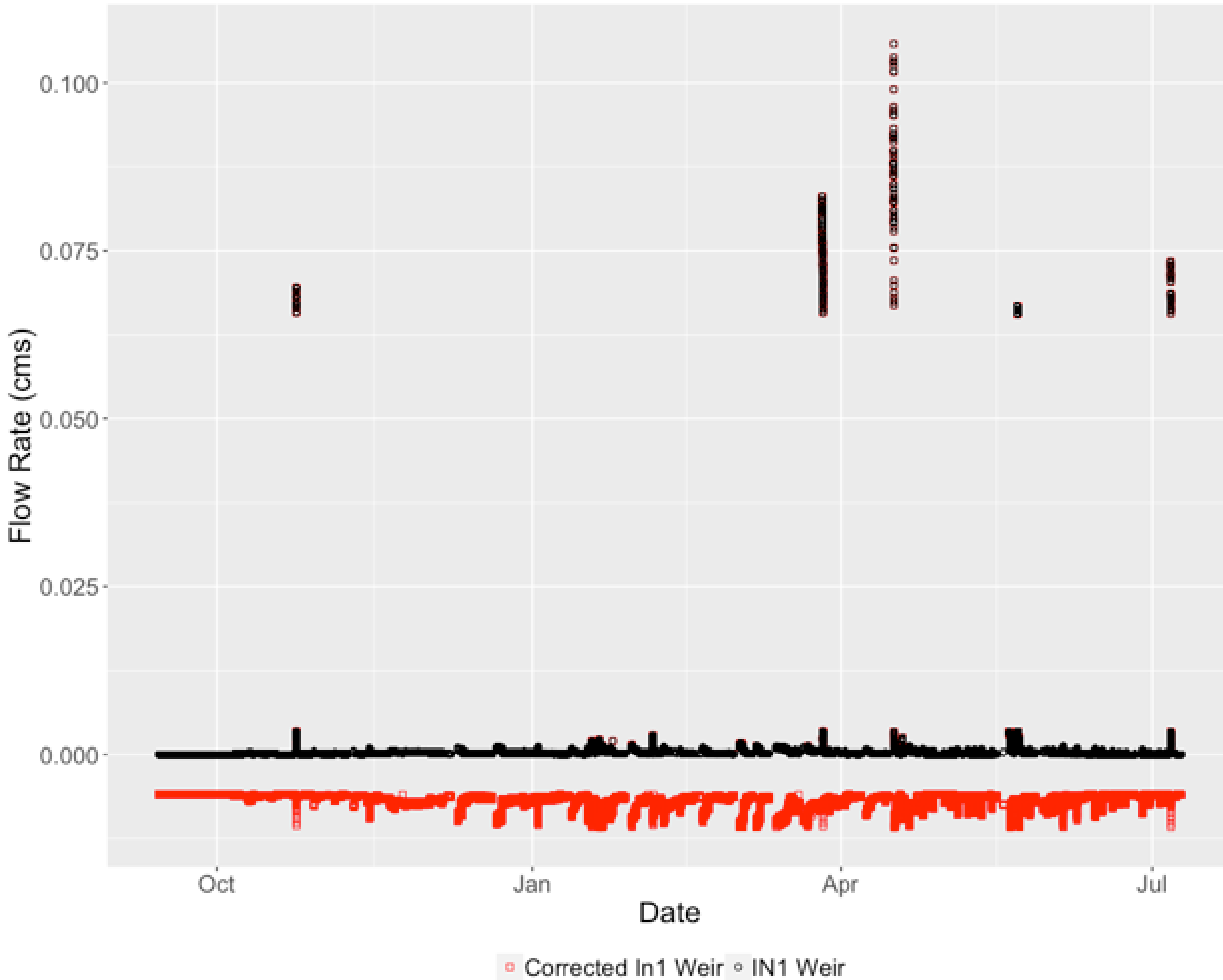


Figure 3-22. IN1 Baseflow Model Corrections

3.4.2.3. Exfiltration Estimation

Well stage recorded during inter-event periods was used to estimate exfiltration rate (Figure 3-23). Sixty-eight discrete ADPs ranging from 0.1—14-days occurred after well installation. Three data periods were removed from the dataset due to maintenance issues. Additionally, ADPs less than 2 days were discarded so that successive event flows did not complicate the calculation. Well stage varied seasonally with higher average stage during Autumn-Winter and lower average stage during Spring-Summer periods (Figure 3-23).

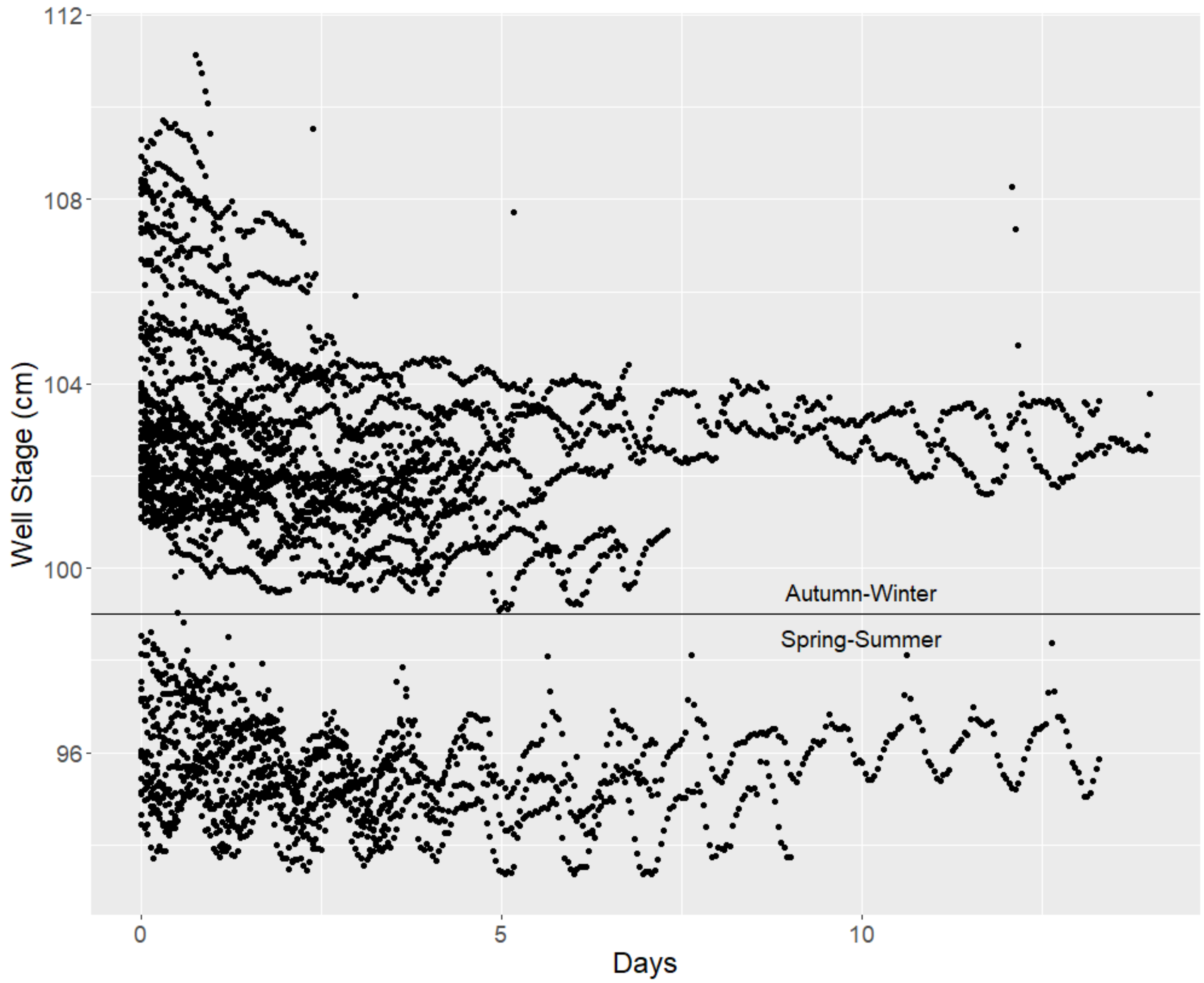


Figure 3-23. Well Stage Recession

A regression analysis using daily Spring-Summer well stage change (response variables) and well stage (predictor variables) was conducted (Figure 3-24); a quadratic expression provided the best model fit, although, model fit is poor ($R^2 = 0.35$). Regression of Autumn-Winter daily well stage change (response variables) and well stage (predictor variables) failed to achieve a better model fit (Appendix F). The Spring-Summer model provides an estimation of exfiltration to the surrounding soils that was used in further hydrology analysis. The global minimum exhibited in the model is possibly caused by slight groundwater intrusion or diurnal fluctuations in the water table seen in Figure 3-23.

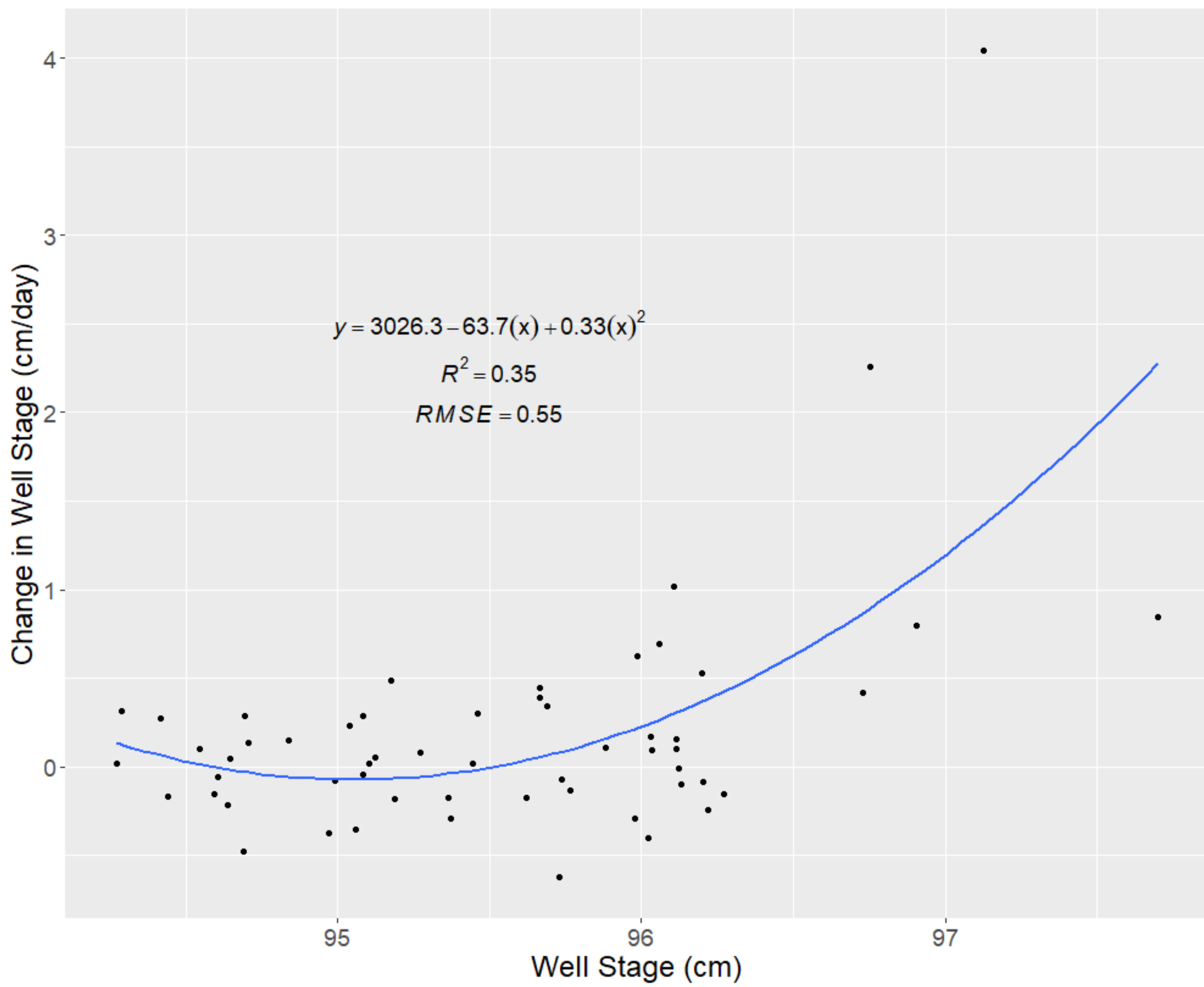


Figure 3-24. Spring-Summer Well Recession Rate

3.4.2.4. Hydrologic Treatment

Analyzed events (n=15) occurred between October 2017 and July 2018. Hydrologic event summary statistics (Table 3-6) used for analysis share a similar range as those of all monitored hydrologic events and corresponded to summary statistics used for inflow model correction; however, median analyzed hydrologic events are larger than that observed for all hydrologic events (Table 3-6).

Components of influent and effluent are represented as positive or negative fractions of total inflow volume (Figure 3-25). AVM uncertainty prevented AVM data use in hydrologic analysis. Water balance analysis was conducted assuming that all volume loss occurred as ET or exfiltration both of which were calculated for this site; the remaining inflow volume was discharge as outflow. Median contributions of total inflow from the IN1 and IN2 monitoring stations and run-on were 32%, 28%, and 40%, respectively; direct precipitation accounted for less than 0.01% of total inflow. Similarly, ET and exfiltration comprised less than 0.01%. ET and exfiltration were negligible; ET and exfiltration accounted for only 0.05% of total influent. The resulting outflow volume equals that of inflow.

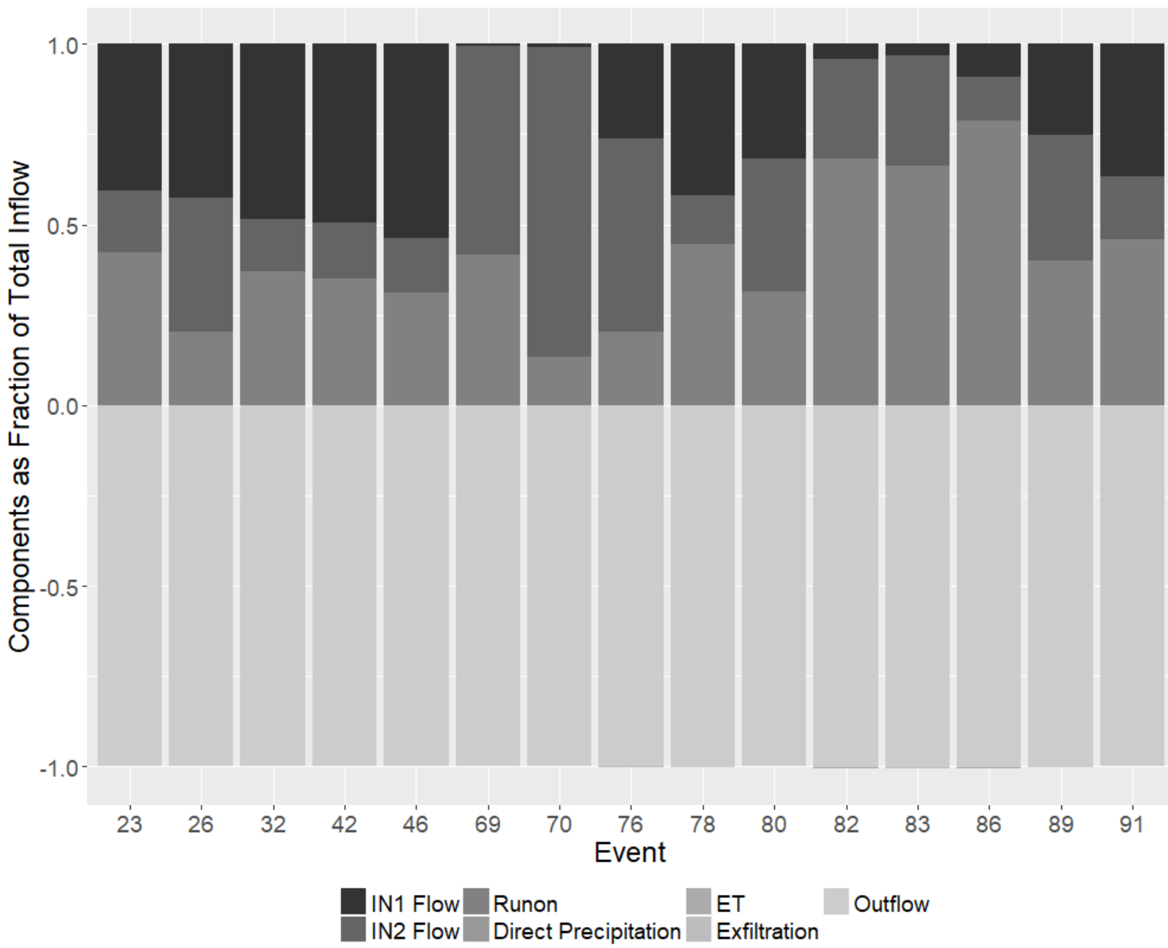


Figure 3-25. Event Water Balance

Peak flow analysis was not conducted due to measurement error and uncertainty in outflow velocity measurements, but would “at best” be quite minimal. This RSC is in-series (downstream) with a dry pond which provides some mitigation of peak inflow rates; however, under-sizing (3.2% require water quality volume), low storage ratio (0.01:1), large loading ratio (25,000:1), and HSG D underlying soils limited the ability of the RSC to mitigate flow (Table 3-3). Under-sizing is not uncommon in RSCs (Cizek, 2014; Cizek et al., 2016, 2017; Koryto et al., 2017) and poor hydrologic performance has been documented as a result (Koryto et al., 2017). Under-sizing limits storage capacity. Moreover, groundwater or baseflow induced saturation, as documented herein, also diminished total storage capacity and hydrologic treatment; without (or with minimal) surface and media storage, flow will continue downslope, bypassing treatment. Overflow or bypass in BRCs has been linked to the hydraulic conductivity of underlying soil, depth to Internal Water Storage (IWS), and the surface infiltration rate (Brown & Hunt, 2011; Hunt et al., 2006); these factors contribute similarly to RSC surface discharge. Effective (>70%) RSC surface-to-seepage conversion is capable when loaded at 80-120:1 provide no additional flow controls are used (Cizek et al., 2016; Koryto et al., 2018, 2017). When constructed downstream of hydrologic controls, RSCs could potentially provide treatment at higher loading rates. However, this RSC proved to be exceptionally small.

3.4.3. Water Quality

Inlet storm flow pollutant concentrations (n=10) showed considerable variation between monitoring locations (Table 3-8); IN1 produced significantly higher median pollutant concentrations for all analytes except $NO_{2,3}$-N. The main treatment mechanism employed by dry ponds is sedimentation; therefore, pollutant treatment provided by dry ponds are generally poor, however, some studies have observed above average dissolved pollutant treatment (Hathaway & Hunt, 2007; Stanley, 1996).

Table 3-8. RSC Storm Flow Inlet Summary

Pollutant	*Median (IN1/IN2) (mg/L)*	*Range (IN1/IN2) (mg/L)*	*Median Difference (% (mg/L))*	*Significance (p-value)*
TKN	4.2/1.1	1.7/0.9-21.7/2.2	74.4 (3.1)	**0.01***
TAN	0.3/0.2	0.1/0.06-0.7/0.3	52.4 (0.2)	**0.04***
$NO_{2,3}$-N	0.6/0.4	0.01/0.2-0.9/0.7	34.1 (0.2)	0.2
ON	3.7/1.0	1.3/0.7-21.0/1.9	74.4 (2.8)	**0.003***
TN	4.9/1.6	2.3/1.3-21.7/2.5	67.8 (3.4)	**0.002***
OP	0.08/0.07	0.07/0.05-0.1/0.1	30.5 (0.02)	**0.02***
PBP	0.7/0.1	0.2/0.09-5.4/0.2	79.2 (0.6)	**0.001***
TP	0.8/0.2	0.3/0.2-5.5/0.3	73.4 (0.6)	**0.002***
TSS	329.0/50.4	56.9/28.0-4153.3/125.7	85.5 (303.1)	**0.003***

Bold* Indicates significant difference between inlet pollutant concentrations

Biological treatment and assimilation by plants within the dry pond could lower $NO_{2,3}$-N concentrations at the IN1 station (Davis et al., 2010). The dry pond maintained base flow and saturated conditions promoting wetland vegetation, Common Rush (*Juncus effusus*) and Pickerelweed (*Pontederia cordata*), which likely produced some denitrification and nutrient reductions (Figure 3-26) (NCDEQ, 2017).

Figure 3-26. Dry Pond Wetland Vegetation

Hydrologic monitoring errors prevented accurate composite storm flow sampling; however, sampling did occur throughout storm durations. Therefore, reported sample concentrations are considered representative storm flow pollutant concentrations (RCs). Baseflow sampling consisted of grab sampling at IN1 and OUT monitoring stations; IN2 only received flow during storm events.

Ten base flow and 10 storm flow samples were collected (Table 3-9); 5 inlet and 7 outlet base flow TSS concentrations were below the Practical Quantitation Limit (PQL) of 2.5-mg. When TSS concentrations were below 2.5 mg an estimation was made using (Eq. 3.10).

$$TSS_{est} = \left(\frac{1000ml}{Volume\ Filtered}\right) * 2.5mg/L \qquad \text{(Eq. 3.10)}$$

Table 3-9. Water Quality Summary

Event	*Base Flow*		*Storm Flow*	
Pollutant	*Median (IN/OUT) (mg/L)*	*Range (IN/OUT) (mg/L)*	*Median (IN/OUT) (mg/L)*	*Range (IN/OUT) (mg/L)*
TKN	0.7/0.5	0.5/0.4-0.9/0.8	1.6/1.4	0.8/1.1-7.2/2.4
TAN	0.09/0.06	0.07/0.03-0.1/0.2	0.15/0.1	0.08/0.06-0.3/0.4
$NO_{2,3}$-N	0.1/0.1	0.06/0.04-1.4/0.8	0.3/0.6	0.2/0.0-0.4/0.7
ON	0.6/0.5	0.4/0.3-0.8/0.7	1.5/1.2	0.7/0.9-6.9/2.0
TN	0.9/0.7	0.5/0.5-2.3/1.5	2.0/2.0	1.1/1.2-7.4/3.0
OP	0.09/0.1	0.05/0.03-0.1/0.2	0.05/0.07	0.04/0.04-0.08/0.2
PBP	0.04/0.04	0.02/0.02-0.07/0.05	0.3/0.1	0.09/0.08-1.8/0.5
TP	0.1/0.1	0.07/0.05-0.2/0.2	0.3/0.2	0.1/0.1-1.8/0.6
TSS	3.3/2.8	2.9/2.8-9.8/4.9	116.3/37.9	34.1/18.0-1341.8/120.7

Base flow was more effectively treated than stormflow. All nitrogen species (except for TAN) were significantly reduced with REs around 15% or greater (Table 3-10); TSS influent concentrations were not substantially reduced (RE = 10.5%) but the change was significant. Phosphorous was not treated; no reductions were significant.

Table 3-10. RSC Pollutant Treatment

Event	*Base Flow*		*Storm Flow*	
Pollutant	*Median RE (%)*	*Significance (p-value)*	*Median RE (%)*	*Significance (p-value)*
TKN	16.5	**0.005***	4.0	0.4
TAN	13.8	0.6	10.9	0.7
$NO_{2,3}$-N	44.0	**0.03***	(77.0)	0.3
ON	21.4	**0.005***	7.4	0.2
TN	22.3	**0.01***	3.4	0.8
OP	(4.1)	0.4	(52.5)	**0.002***
PBP	23.0	0.2	37.0	**0.007***
TP	2.6	0.8	25.6	0.1
TSS	10.5	**0.02***	69.2	**0.002***

Bold* Indicates significant reduction in pollutant concentration
( ) ER values in parenthesis represent an increase in pollutant concentration

Storm flow pollutant treatment was poor; only PBP and TSS concentrations experienced significant reductions, REs equal to 37% and 69%, respectively. Approximately 67% of TP is PBP (Novotny, 2003; Vaze & Chiew, 2004); therefore, effective TSS treatment should correspond with the removal of large fractions of TP. Some TSS influent concentrations were extremely high (1,341 mg/L, Table 3-9), this contributed to an inflated removal efficiency. OP was significantly leached (53% concentration increase). Hydrologic monitoring errors and the timing of water quality sampling prevented corresponding hydrologic and water quality events. Therefore, estimations of Baseflow Load Reductions (BLR), Event Load Reductions (ELR), and Annual Load Reductions would contain substantial error and are considered inaccurate.

3.4.3.1. Media Analysis

Media OM provides sorption sites for hydrophobic organic compounds, heavy metals, OP, and ammonium (NH_4^+) (Davis et al., 2010; Hsieh & Davis, 2005a) and organic carbon for denitrification reactions (Davis et al., 2010; Tiedje et al., 1982). Media without incorporated carbon may have limited denitrification potential (Davis et al., 2010; Tiedje et al., 1982). Media core sample collection was attempted on July 2, 2018. The depth to media, persistent saturation, and coarse substrate inhibited core sample or clean media collection. Instead trowel samples were collected from a depth of greater than 0.5 m below the RSC bed surface in pool 3 (Figure 3-27 A); samples consisted of a heterogeneous blend of #57 stone, fine gravel, sand, and OM (Figure 3-27 B).

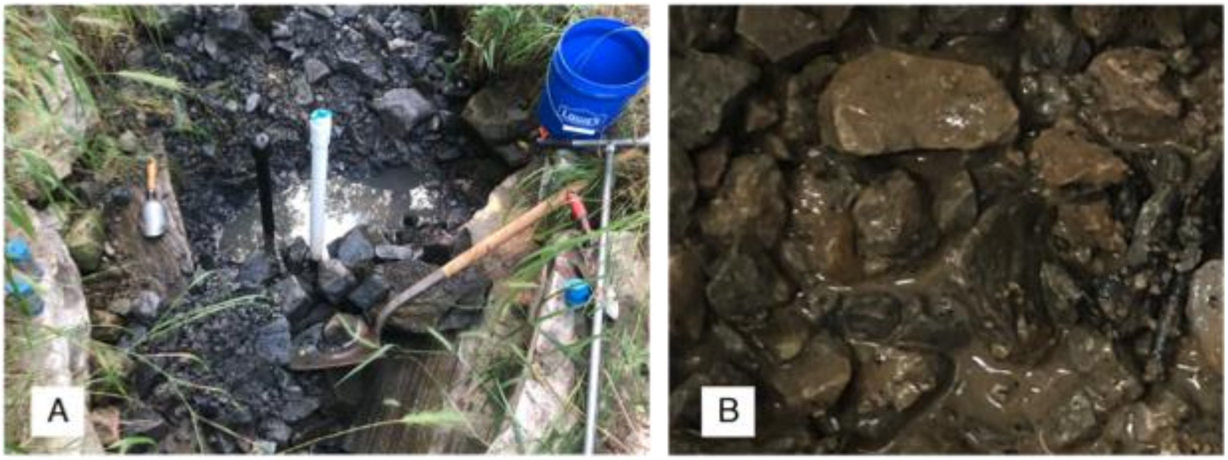


Figure 3-27. Sampling Attempt (A); Media Sample (B)

Loss on Ignition (LOI) analysis, in which samples are heated allowing volatile compounds to escape, was conducted on media samples. An average 4.16-mg/kg was lost, less than 0.01% of total mass (Table 3-11). Due to difficulty digging down to the sand-woodchip media layer, media samples are likely biased toward less OM; however, samples represent the total fill within the RSC exfiltration trench. Approximately 0.5 m of course Class B stone on the surface of the RSC bed and a total 0.3-m of #57 stone above and below the media layer (Figure 3-12) are likely subjected to anoxic conditions affected by prolonged saturation. While now ASTM C33 (Flores et al., 2012) or NCDOT S2 coarse sand mixes are recommended, a sand analysis conducted in Morrisville in 2013 concluded that neither standard was met herein.

Table 3-11. Loss on Ignition Media Analysis

Sample ID	*LOI (mg/Kg)*	*% Loss*
20180702-A	3.84	3.84EE^{-6}
20180702-B	4.14	4.14EE^{-6}
20180702-C	4.51	4.51EE^{-6}
Average	4.16	4.16EE^{-6}

<u>*3.4.3.2. TSS*</u>

All base flow inlet and outlet TSS concentrations were less than target TSS concentrations of 25-mg/L proposed by Barrett et al. (2004); however, 100% of storm flow inlet and 60% of outlet TSS concentrations exceeded target TSS concentrations (Figure 3-28). RSC treatment produced significant reductions in storm and base flow TSS concentration. Storm flow RE was 69% (Table 3-10). Most previous RSC research observed substantial TSS removal in the first pool, suggesting it acts as a defacto forebay (Cizek et al., 2016; Filoso, 2012; Koryto et al., 2017). As stated previously, this RSC was proportionally very small relative to its watershed, thus (1) it's possible that the entire SCM acted as a forebay and (2) any effects of a first cell were not captured.

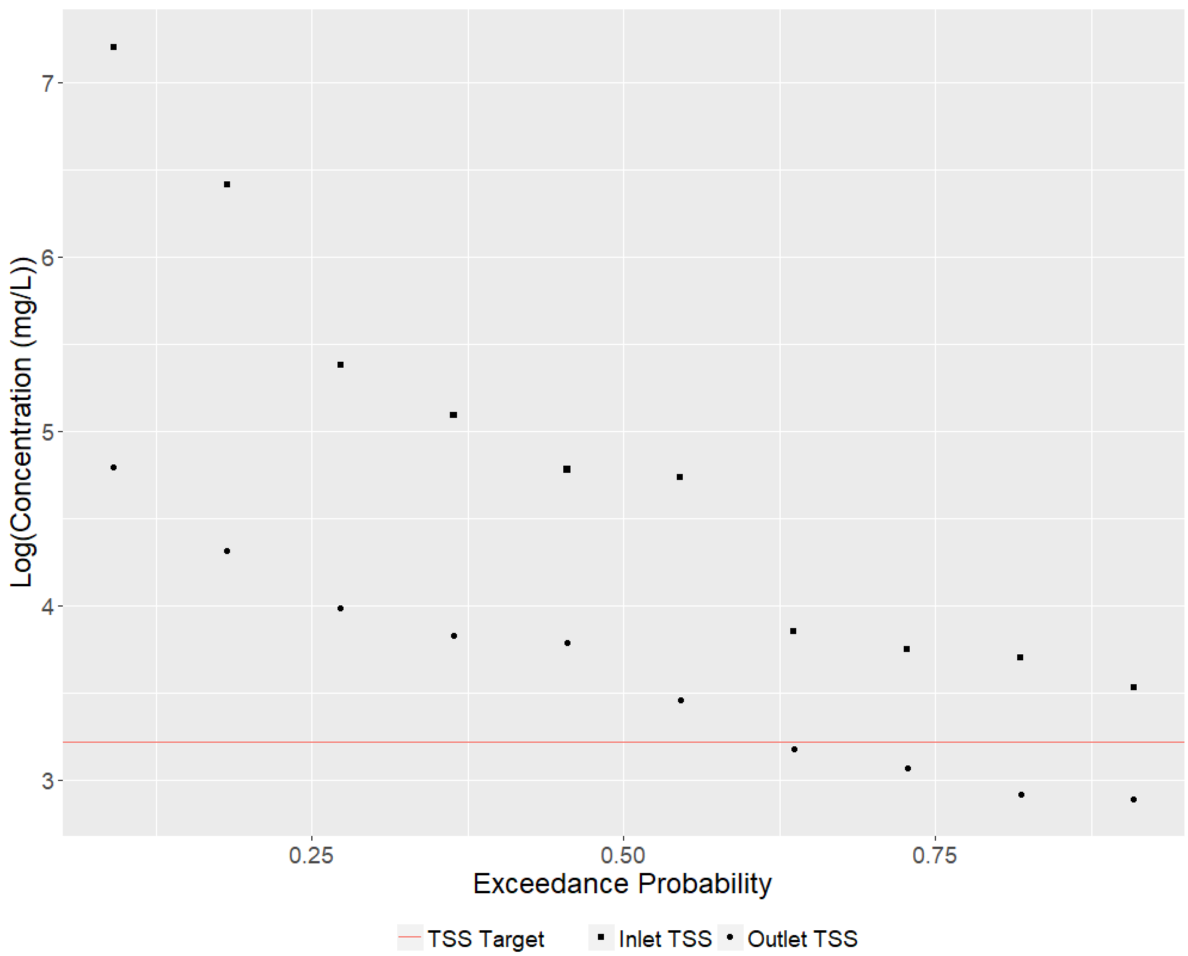


Figure 3-28. Storm Flow TSS Exceedance Probability

Effective sedimentation requires reduction of flow velocities sufficient that particles settle via gravity (Stokes' Law) to the pool bottom; large, dense particles are readily settled compared to smaller, low density particles (Davis et al., 2010). Sufficient pool storage is necessary to achieve conditions suitable for sedimentation and to prevent pollutant resuspension (Davis et al., 2010; Miguntanna et al., 2013). Minimal surface storage volumes were not conducive to TSS settling; however, a very low storage ratio (0.01:1) (Table 3-3) likely produced conditions supportive of surface-to-seepage conversion and surface filtration of TSS and particulate-bound pollutants.

Media saturation will inhibit filtration of influent and will diminish treatment as storm size increases and media saturation occurs in well-drained systems (Cizek et al., 2016; Koryto et al., 2017); however, in this study large in-stream geomorphic structures and up-stream flow control likely created substantial driving head and also reduced inflow rates. Both factors improved delivery of surface flows to the media zone. Log TSS RC ratio (Eq. 3.10) (response variables) versus Log-accumulation (predictor variables) failed to conclude that the proportion of influent TSS treated decreased as storm size increased (due to insignificant slope) (Appendix G) contrary to what was previously documented (Koryto et al., 2017); this could be the result of up-stream flow mitigation or small sample size (n=10).

3.4.3.3. Phosphorous

Approximately two-thirds of TP is PBP, therefore, effective removal of TSS will remove large fractions of TP (Novotny, 2003; Vaze & Chiew, 2004). Base flow TP effluent concentrations exceeded the “Good” ambient water quality standard of 0.11-mg/L (McNett et al., 2010) in 70% of grab samples (Figure 3-29) while storm flow TP concentrations exceeded the “Good” ambient water quality standard in 100% of samples. RSC treatment did not significantly reduce phosphorous analyte concentrations for base flow (Table 3-10). Storm flow RE for PBP was significant at 37% while OP concentrations *increased* significantly with RE of -53% (Table 3-10). Reductions of TP herein are attributed the capture of PBP.

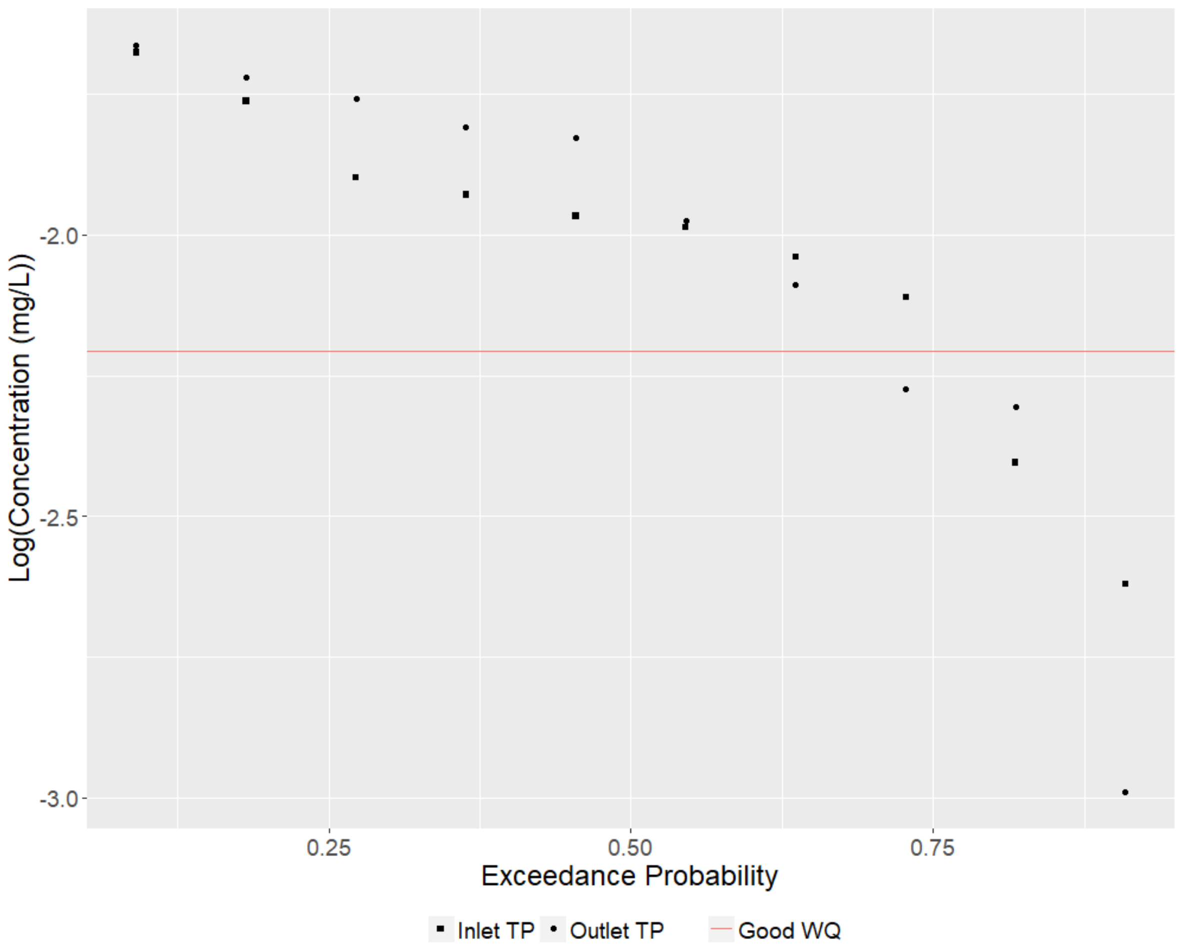


Figure 3-29. Base Flow TP Exceedance Probability

Phosphorous removal is dependent on effective treatment of PBP and soluble OP; treatment mechanisms include (1) settling, (2) surface filtration, (3) adsorption to soil and OM, and (4) plant assimilation (Cizek et al., 2016, 2017; Davis et al., 2010; Flores et al., 2012; Koryto et al., 2018, 2017). Reductions of TP concentrations ranging from 70-90% are common in field monitoring of media-based SCMs (Davis et al., 2006; Hunt et al., 2006; Page et al., 2015); the P-Index is a reliable predictor of a medias' ability to bind or leach phosphorus (Hardy et al., 2009; Hunt et al., 2006). Settling is likely not effective due to the low surface storage volume; however, a combination of the extremely low storage ratio (0.01:1) (Table 3-3), large in-stream structures (0.3-m step) (Figure 3-12), and upstream flow control likely improved surface-to-seepage conversion, and thus, surface filtration and adsorption to soil and OM. All of which produced good reductions of PBP. OP leaching in media based SCMs with high P-Indices are common (Davis et al., 2006; Hunt et al., 2006; Page et al., 2015); anaerobic conditions will affect redox conditions and potentially lead to OP release and leaching (Clark & Pitt, 2012). High media organic matter and redox conditions were not accurately assessed in this study; however, these factors may have contributed the OP leaching observed (Table 3-10). Log PBP RC ratio (Eq. 3.10) (response variables) versus Log accumulation (predictor variables) failed to conclude that the proportion of influent PBP treated decreased as storm size increased (due to insignificant slope) (Appendix H) contrary to what was previously documented (Koryto et al., 2017); this could be the result of up-stream flow mitigation or small sample size (n=10).

3.4.3.4. Nitrogen

Base flow TN effluent concentrations exceeded the "Good" ambient water quality standard of 0.99-mg/L (McNett et al., 2010) in 20% of grab samples (Figure 3-30); storm flow TN concentrations exceeded "Good" ambient water quality standard in 100% of samples. RSC treatment significantly reduced TN concentrations in base flow with a RE of 22%; however, storm flow treatment was non-existent (Table 3-10). The RSC studied herein is easily (1) the most undersized, (2) the only in-series with another SCM, and (3) the only designed with boulder-weir in-stream structures. The majority (50-70%) of nitrogen is dissolved (Novotny, 2003; Vaze & Chiew, 2004) and effective treatment occurs in the media layer. (1) Under-sizing (3.2% water quality volume) and overloading (25,000:1) will limit total storage capacity and minimize hydraulic retention time, diminishing TN treatment. (2) Upstream control has the potential to mitigate influent rates, acting as an equalizing basin, extending storm flow treatment over a longer duration, benefiting capture volume and loading rate. (3) Large boulder-weirs (0.3-

m step) likely improved downwelling rates (Hester & Doyle, 2008), and thus, pollutant delivery to treatment sites (Lawrence et al., 2013). During storm flow, under-sizing and large boulder structures (1 and 3) had the greatest influence on treatment; they likely produced short residence times and strong downwelling of oxygen rich water which inhibited denitrification. Large boulder structures benefited base flow NO3- treatment.

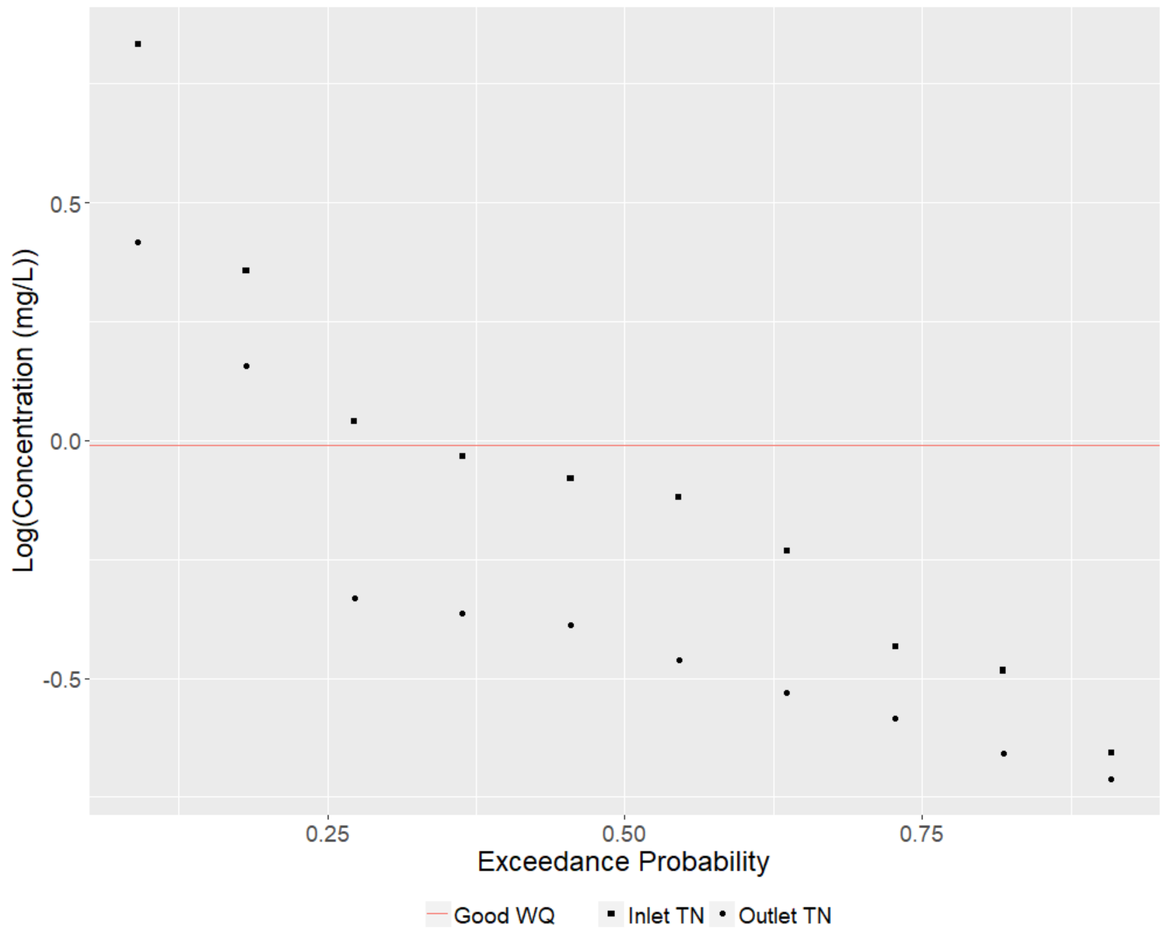


Figure 3-30. Base Flow TN Exceedance Probability Plot

TKN, which is the sum of TAN and ON, comprised the majority of TN effluent; TAN consists of NH_3 and an ionized form, NH_4^+. ON and NH_3 are readily adsorbed to soil fines and OM, making them available for inter-event biological treatment (Davis et al., 2010) or removed via settling and surface filtration. $NO_{2,3}$-N and NH_4^+ are soluble; nitrite (NO_2^-) and NH_4^+ must undergo oxidation and nitrification to form NO3-, respectively, before anaerobic reduction and denitrification are possible. In aerobic environments, these molecules oxidize readily. Koryto et al. (2017) observed aerobic conditions in media Dissolved Oxygen (DO) measurements during storm flows which persisted for 48 to 72 hours after storm flow; denitrification is, therefore, likely inhibited by downwelling of aerated influent. Hydraulic retention of NO3- is required for denitrification to occur.

Base flow maintained constant saturation of the media; prolonged media saturation in RSCs has contributed to inter-event leaching of TKN, TAN, and TN (Koryto et al., 2017). Engineered saturation zones or IWS zones used in BRC improve NO3- treatment by producing anoxic conditions (Brown et al., 2011; Davis, 2008; Kim et al., 2003). An engineered zone of saturation was used to promote wetland-like conditions which improved TKN, NO3-, and TAN treatment (Cizek et al., 2016). The RSC herein did not maintain any aerobic media; this may have diminished NO_2^- and NH_4^+ oxidation during storm flows.

Low media C:N ratios lead to the production of TKN and TAN in saturated media zones (Hodge et al., 2000; Kim et al., 2003; Tiedje et al., 1982). When OM decomposition occurs dissolved organic carbon, and turbidity have been exported (Kim et al., 2003). Denitrification and Dissimilatory Nitrate Reduction to Ammonia (DNRA) are competing processes under anoxic conditions that depend on available organic carbon and nitrate concentration; denitrification is favored when organic carbon is limited while DNRA is favored when nitrate is limited (Davis et al., 2010; Tiedje et al., 1982). Heterogenic microbial mineralization of organic nitrogen is dependent on the C:N ratio in the growth substrate and the microbes' need for nitrogen; at C:N ratios less than 25:1 TAN is typically exported (Hodge et al., 2000).

Denitrification causes nitrogen to leave the water and soil column as dinitrogen and nitrous oxide gases (Tiedje et al., 1983); DNRA conserves nitrogen with the production of TAN

(Tiedje et al., 1983). Koryto et al. (2017) observed a TAN increase of 390% during inter-event base flow, while $NO_{2,3}$-N concentrations were significantly lower during inter-event flow; these results somewhat align with those herein. $NO_{2,3}$-N was the only analyte which was not significantly reduced during event flow. This could be explained by highly reduced conditions during base flow and near-complete reduction of available nitrate coupled with the production of TAN via DNRA and OM decomposition. During storm flows, extreme aeration inhibits microbial reduction of NO3-. $NO_{2,3}$-N entering from IN2 does not receive full media treatment before reaching the OUT monitoring station; furthermore, $NO_{2,3}$-N was the only pollutant which was not significantly different between IN1 and IN2 (Table 3-8). $NO_{2,3}$-N-leaching in storm flows is possibly the result of OM decomposing during inter-event periods producing TKN and TAN, that then underwent oxidation as it is flushed from the media zone during higher storm flows. TAN and $NO_{2,3}$-N are plant available nutrients; however, vegetation did not persist in the RSC, so assimilation by plant material is not a major treatment mechanism.

Poor storm flow TN removal was caused by under-sizing (3.2% water quality volume) and overloading (25,000:1). Large hydraulic gradients created by the boulder-weirs (0.3-m steps) contributed to strong downwelling flux and hyporheic exchange; however, continuous saturation and limited total storage produced excessive bypass and limited hydraulic retention times. In short, there was not much opportunity during event flows for nitrogen transformation/removal. The extremely low storage ratio 0.01:1 (Table 3-3) provided adequate media storage and treatment for most base flow. Conversion of surface flows to subsurface seepage is essential for providing TN treatment when surface storage volumes are minimal. Surface filtration and adsorption of pollutants likely provided the most treatment benefits.

3.4.3.5. Temperature

Table 3-12. Monthly Event Temperature Summary

Year	*Month*	*Median IN1 Temp (°C)*	*Max IN1 Temp (°C)*	*Median IN2 Temp (°C)*	*Max IN2 Temp (°C)*	*Median WELL Temp (°C)*	*Max WELL Temp (°C)*	*Median OUT Temp (°C)*	*Max OUT Temp (°C)*
2017	*Oct.*	17.5	18.4	17.3	18.1	17.1	17.4	17.4	18.1
	Nov.	13.1	16.7	12.8	14.5	14.8	17.0	12.9	14.6
2018	*Feb.*	13.8	17.0	12.8	14.6	13.8	14.3	13.6	15.9
	Mar.	8.9	15.7	8.4	13.6	11.3	12.0	8.6	15.0
	April	17.2	23.7	15.7	20.6	14.8	18.5	16.4	20.6
	May	23.6	29.3	22.2	31.1	20.4	21.3	23.3	28.3
	July	25.5	31.0	24.5	31.6	23.6	23.8	25.5	27.7

Twenty-six events between October 2017 and July 2018 were analyzed for thermal mitigation; due to equipment malfunction, no events for December 2017, and January and June 2018, were analyzed (Table 3-12). The hottest month for median and maximum event temperatures was July 2018; the coolest month for median and maximum event temperatures was March 2018.

A representative summer event is illustrated in Figure 3-31. Starting at 2PM on July 6 and continuing into the afternoon of July 7, 2018, a 54.1-mm event produced the hottest median and maximum event temperatures observed during the study. Temperatures at the IN1, IN2, and OUT monitoring stations all had a small (< 2^{o}C) spike in temperature before a prolonged decrease in temperature. WELL temperatures showed little change, and OUT temperatures cooled down to those of the WELL. Daytime heating on the afternoon of July 7 increased IN1 and IN2 temperatures, but due to the small magnitude of additional rainfall the impact on outlet temperatures were minimal.

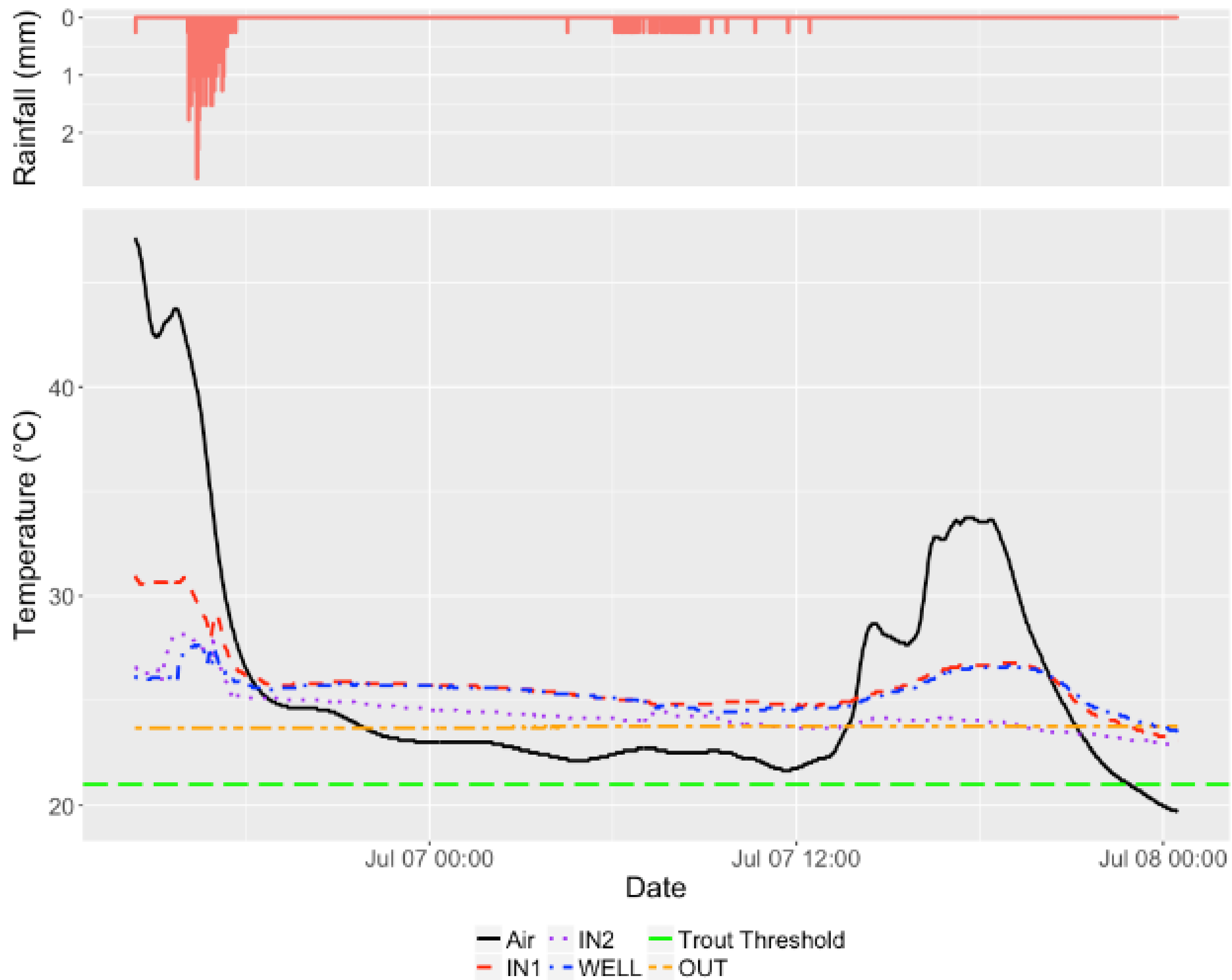


Figure 3-31. Hottest Median Temperature Event

An analysis of event median and instantaneous maximum temperatures (Wardynski et al., 2014) helped determine the temperature buffering capabilities of this RSC. In the absence of more robust data, this analysis provides useful insight to the performance of RSC as a thermal mitigation tool. There are no relevant temperature standards in the North Carolina Piedmont (because no trout waters exist). Researchers compare stormwater runoff temperatures to the threshold for brook trout (*Salvelinus fontinalis*) (21°C) (Coutant, 1977; Ferguson, 1958; Jones & Hunt, 2010, 2009; Jones et al., 2012; Lieb & Carline, 2000; Wardynski et al., 2014; Winston et al., 2011); trout are not present in Morrisville, but comparison will be made for reference (Figure 3-31). When considering events between May 1 and October 31 (the period of peak thermal stress for salmonids) (Wardynski et al., 2014), IN1 and IN2 median event temperatures (n=9) were significantly different; wet ponds are a known source of thermal pollution (occurs upstream of IN1), while sub-surface conveyance is observed to cool runoff temperatures (occurs upstream of IN2) (Jones & Hunt, 2010; Jones et al., 2012; Lieb & Carline, 2000). Based on a 95% confidence interval for the median difference, IN1 was 1.1ºC hotter than IN2. IN1 and IN2 maximum event temperatures were not significantly different.

IN1 and IN2 flows converge downstream of the WELL monitoring location; WELL measured the temperature of saturated media and not that of surface flows. Median and maximum IN1 event temperatures were significantly warmer (2.5ºC and 6.8ºC warmer, respectively) than that of the WELL. This highlights the RSCs ability to buffer median and maximum runoff temperatures; previously research by Koryto et al. (2017) documented buffering of maximum but not median runoff temperatures. BRC studies, likewise, have observed good buffering of maximum runoff temperatures, with an inability to adapt to cooler runoff temperatures in later stages of an event (Jones & Hunt, 2009). Equilibrium between BRC media temperatures and runoff temperatures occurred at depths of 60 cm, minimizing deeper medias' ability to reduce runoff temperatures; therefore, thermal reduction benefits diminished with deeper media (Jones & Hunt, 2009).

This analysis was confounded by the confluence of IN1 and IN2 flows; IN1 median temperatures are significantly greater than IN2, while maximum temperatures are not significantly different. IN1 median influent temperatures are buffered by the media beneath the step-pool channel (noted by significantly cooler median and maximum WELL temperatures). Thermally-enriched influent from IN2 receives minimal treatment and results in maximum OUT temperatures that are not significantly different than maximum IN2 temperatures. WELL temperatures (median and maximum) are significantly cooler than all other monitoring locations, highlighting the temperature buffering abilities of media filtration and seepage. RSC performs similar to BRC in its ability to buffer maximum event temperatures (Jones & Hunt, 2009; Koryto et al., 2017); although, the practice herein showed some evidence of median event temperature buffering. Analysis of thermal loads would allow more accurate accounting of thermal mitigation; however, without accurate effluent velocity data this analysis is not possible (Wardynski et al., 2014). If substantial reductions in flow volume had been achieved, proportional reductions in thermal load would have been observed.

3.5. Recommendations & Conclusions

1. Despite upstream flow control, adequate storage volume in the RSC surface and media zones is required to effectively treat pollutants (Cizek et al., 2016; Davis et al., 2010; Koryto et al., 2017). Saturated media volumes should not be included in storage calculations if a seasonally high water table or base flow is present, as both lead to media saturation (Koryto et al., 2017).
2. Pool storage to support effective sedimentation and surface-to-seepage flow conversion is important for pollutant treatment. High storage ratios should aid sedimentation but will likely limit media filtration, while low storage ratios more likely convert surface flows to seepage. However, if total storage is undersized or media saturation occurs the ratio does not matter; excessive bypass will occur (Cizek et al., 2016; Koryto et al., 2017).
3. Media OM and P-index are important to nutrient treatment. Excessive OM can lead to nitrogen leaching under prolonged saturation and nitrate-limited conditions (Koryto et al., 2017). High P-index media is likely to leach phosphorous (Hunt et al., 2006). Stormwater

nutrient treatment will likely be maximized when media OM is 10-15% and the P-index is less than 30.

4. RSCs are capable of buffering maximum event temperatures and, to a lesser extent, median event temperatures. Like BRC, RSCs buffer runoff temperatures as influent water exchanges heat energy with the sub-surface media; however, once effluent reaches temperature equilibrium with media, thermal mitigation stops. This occurs relatively quickly in grossly undersized RSCs.
5. Retrofit RSCs used in series with hydrologic control devices might be able to provide additional pollutant reduction benefits when employed as a stabilization technique or constructed with an exfiltration trench; however, additional research to understand appropriate usage of RSCs, design optimization under varying constraints (i.e. riparian vegetation protection, right-of-way access, and economics), long-term maintenance protocols, and expected lifespan of RSCs is needed.

3.6. References

Allen, R. G., Pereira, L. S., Raes, D., & Smith, M. (1998). *Crop Evapotranspiration - Guidelines for Computing Crop Water Requirements - FAO Irrigation and Drainage Paper 56*.

Barrett, M. E. (2003). Performance, Cost, and Maintenance Requirements of Austin Sand Filters. *Journal of Water Resources Planning and Management*, *129*(3), 234–242. https://doi.org/10.1061/(ASCE)0733-9496(2003)129:3(234)

Barrett, M., Lantin, A., & Austrheim-Smith, S. (2004). Storm Water Pollutant Removal in Roadside Vegetated Buffer Strips. *Transportation Research Record*, *1890*(1), 129–140. https://doi.org/10.3141/1890-16

Bennett, S. J., Alonso, C. V, Prasad, S. N., & Romkens, M. J. (2000). Experiments on headcut growth in concentrated flows typical of upland areas. *Water Resources Research*, *36*(7), 1911–1922.

Brown, R. A., & Hunt, W. F. (2011). Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads. *Journal of Environmental Engineering*, *137*(11), 1082–1091. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000437

Brown, T., Berg, J., & Underwood, K. (2010). Replacing Incised Headwater Channels and Failing Stormwater Infrastructure with Regenerative Stormwater Conveyance. In *Low Impact Development 2010: Redefining Water in the City*. https://doi.org/10.1061/41099(367)4

Browning, M. O. (2008). The Efficacy of Urban Stream Restorations to Improve Water Quality Across a Spectrum of Design Approaches. *Thesis Submitted to George Mason University*, 118.

Burns, M. J., Fletcher, T. D., Walsh, C. J., Ladson, A. R., & Hatt, B. E. (2012). Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform. *Landscape and Urban Planning*, *105*(3), 230–240. https://doi.org/10.1016/j.landurbplan.2011.12.012

Chattahoochee Riverkeeper. (2015). Tri-State Water Conflict. Retrieved from https://chattahoochee.org/our-work/tri-state-water-conflict/

Cizek, A. R. (2014). *Quantifying the Stormwater Mitigation Performance and Ecosystem Service Provision in Regenerative Stormwater Conveyance (RSC)*. North Carolina State Univeristy.

Cizek, A. R., Hunt, W. F., Winston, R. J., & Lauffer, M. S. (2017). Hydrologic Performance of Regenerative Stormwater Conveyance in the North Carolina Coastal Plain. *Journal of Environmental Engineering*, *143*(9), 05017003. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001198

Cizek, A. R., Hunt, W. F., Winston, R. J., Waickowski, S., Narayanaswamy, K., & Lauffer, M. S. (2016). Water Quality and Hydrologic Performance of a Regenerative Stormwater Conveyance (RSC) in the Piedmont of North Carolina. *Journal of Environmental Engineering*, *Submitted*(8), 1–11. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001344

Clark, S. E., & Pitt, R. (2012). Targeting treatment technologies to address specific stormwater pollutants and numeric discharge limits. *Water Research*, *46*(20), 6715–6730. https://doi.org/10.1016/j.watres.2012.07.009

Coutant, C. C. (1977). Compilation of Temperature Preference Data. *Journal of the Fisheries Research Board of Canada*, *34*(5), 739–745. https://doi.org/10.1139/f77-115

Davis, & , Robert G. Traver, Hurt, W. F. (2010). Improving Urban Stormwater Quality: Applying Fundamental Principles. *Journal of Contemporary Water Research & …*, *146*(1), 2–10. https://doi.org/10.1111/j.1936-704X.2010.00387.x

Davis, A. P. (2008). Field Performance of Bioretention: Hydrology Impacts. *Journal of Hydrologic Engineering*, *13*(2), 90–95. https://doi.org/10.1061/(ASCE)1084-0699(2008)13:2(90)

Davis, A. P., Shokouhian, M., Sharma, H., & Minami, C. (2006). Water Quality Improvement through Bioretention Media: Nitrogen and Phosphorus Removal. *Water Environment Research*, *78*(3), 284–293. https://doi.org/10.2175/106143005X94376

Davis, A. P., Shokouhian, M., Sharma, H., Minami, C., & Winogradoff, D. (2003). Water quality improvement through bioretention: lead, copper, and zinc removal. *Water Environment Research : A Research Publication of the Water Environment Federation*, *75*(1), 73–82. https://doi.org/10.2175/106143003X140854

Doll, B. a., Grabow, G. L., Hall, K. R., Halley, J., Harman, W. A., Jennings, G. D., & Dani E. Wise. (2003). Stream Restoration: A Natural Channel Design Handbook, 128 pp. https://doi.org/(11/19/2014)

EPA. (2016). Summary of the Clean Water Act. Retrieved October 14, 2016, from https://www.epa.gov/laws-regulations/summary-clean-water-act

Ferguson, R. G. (1958). The Preferred Temperature of Fish and their Midsummer Distribution in Temperate Lakes and Streams. *Journal of Fish Research*, *15*(4), 607–624. Retrieved from http://www.nrcresearchpress.com/doi/pdf/10.1139/f58-032

Filoso, S. (2012). Assessing the Effectiveness of Regenerative Stormwater Conveyance in reducing Sediment and Nutrient Loads to Three Mid-River Creeks in the Severn River watershed, (410), 1–22.

Fletcher, T. D., Vietz, G., & Walsh, C. J. (2014). Protection of stream ecosystems from urban stormwater runoff: The multiple benefits of an ecohydrological approach. *Progress in*

Physical Geography, *38*(5), 543–555. https://doi.org/10.1177/0309133314537671

Flores-Cervantes, J. H., Istanbulluoglu, E., & Bras, R. L. (2006). Development of gullies on the landscape: A model of headcut retreat resulting from plunge pool erosion. *Journal of Geophysical Research: Earth Surface*, *111*(1), 1–14. https://doi.org/10.1029/2004JF000226

Flores, H., Mcmonigle, D., & Underwood, K. (2012). *Design Guidelines for Regenerative Step Pool Storm Conveyance*.

Gallo, C., Moore, A., & Wywrot, J. (2012). Comparing the adaptability of infiltration based BMPs to various U.S. regions. *Landscape and Urban Planning*, *106*(4), 326–335. https://doi.org/10.1016/j.landurbplan.2012.04.004

General Assembly of North Carolina. (2009). NC Session Law 2009-484 Senate Bill 838.

Gregg, S. (2018). M-RSC. Retrieved from https://github.com/simonsaysgregg/M-RSC

Hardy, D. H., Tucker, M. R., & Stokes, C. E. (2009). Crop Fertilization Based on North Carolina prepared by, (1).

Hathaway, J., & Hunt, W. F. (2007). City of Charlotte Pilot BMP Monitoring Program - Little Sugar Creek - Westfield Level Spreader, (January), 22. Retrieved from http://charmeck.org/stormwater/StormWaterAgencies/Pilot SCMs/Wet Ponds/Shade Valley Pond - Final Report.pdf#search=shade valley pond

Helsel, D. R., & Hirsch, R. M. (2002). Statistical Methods in Water Resources (Techniques of Water-Resources Investigations of the United States Geological Survey, Book 4, Hydrologic Analysis and Interpretation, Chapter A3). *US Geological Survey*, 1–524. https://doi.org/10.2307/1269385

Hester, E. T., & Doyle, M. W. (2008). In-stream geomorphic structures as drivers of hyporheic exchange. *Water Resources Research*, *44*(3). https://doi.org/10.1029/2006WR005810

Hodge, A., Robinson, D., & Fitter, A. (2000). Are microorganisms more effective than plants at competing for nitrogen? *Trends in Plant Science*, *5*(7), 304–308. https://doi.org/10.1016/S1360-1385(00)01656-3

Hood, G., & Sommer, L. (2016). High Demand, Low Supply: Colorado River Water Crisis hits Across the West. *National Public Radio*. Retrieved from http://www.npr.org/2016/12/30/507569514/high-demand-low-supply-colorado-river-water-crisis-hits-across-the-west

Hsieh, C., & Davis, A. P. (2005). Evaluation and Optimization of Bioretention Media for Treatment of Urban Storm Water Runoff. *Journal of Environmental Engineering*, *131*, 1521–1531. https://doi.org/10.1061/(ASCE)0733-9372(2005)131:11(1521)

Huffman, R. L., Fangmeier, D. D., Elliot, W. J., & Workman, S. R. (2013). *Soil and Water Conservation Engineering*. St. Joseph, MI.

Hunt, W. F., Davis, A. P., & Traver, R. G. (2012). Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design. *Journal of Environmental Engineering*, *138*(6), 698–707. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000504

Hunt, W. F., Jarrett, A. R. R., Smith, J. T., & Sharkey, L. J. (2006). Evaluating Bioretention Hydrology and Nutrient Removal at Three Field Sites in North Carolina. *Journal of Irrigation and Drainage Engineering*, *132*(6), 600–608. https://doi.org/10.1061/(ASCE)0733-9437(2006)132:6(600)

Idso, S. B. (1981). Relative Rates of Evaporative Water Losses From Open and Vegetation Covered Water Bodies. *JAWRA Journal of the American Water Resources Association*, *17*(1), 46–48. https://doi.org/10.1111/j.1752-1688.1981.tb02587.x

Jones, M., & Hunt, W. (2010). Effect of Storm-Water Wetlands and Wet Ponds on Runoff

Temperature in Trout Sensitive Waters. *Journal of Irrigation and Drainage Engineering*, *136*(9), 656–661. https://doi.org/doi:10.1061/(ASCE)IR.1943-4774.0000227

Jones, M. P., & Hunt, W. F. (2009). Bioretention Impact on Runoff Temperature in Trout Sensitive Waters. *Journal of Environmental Engineering*, *135*(8), 577–585. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000022

Jones, M. P., Hunt, W. F., & Winston, R. J. (2012). Effect of Urban Catchment Composition on Runoff Temperature. *Journal of Environmental Engineering*, *138*(12), 1231–1236. https://doi.org/doi:10.1061/(ASCE)EE.1943-7870.0000577

Keith R. Underwood. (2002). United States Patent Application Publication - Regnerative Stormwater Conveyance System and Method, *1*(60). Retrieved from https://www.google.com/patents/US20090290936

Kim, H., Seagren, E. A., & Davis, A. P. (2003). Engineered Bioretention for Removal of Nitrate from Stormwater Runoff. *Water Environment Research*, *75*(4), 355–367.

Koryto, K. M., Asce, M., Hunt, W. F., Arellano, C., & Page, J. L. (2018). Performance of Regenerative Stormwater Conveyance on the Removal of Dissolved Pollutants: Field Scale Simulation Study, *144*(6), 1–13. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001374

Koryto, K. M., Hunt, W. F., & Page, J. L. (2017). Hydrologic and water quality performance of regenerative stormwater conveyance installed to stabilize an eroded outfall. *Ecological Engineering*, *108*, 263–276. https://doi.org/10.1016/j.ecoleng.2017.04.041

Ladson, A. R., Walsh, C. J., & Fletcher, T. D. (2006). Improving Stream Health in Urban Areas by Reducing Runoff Frequency from Impervious Surfaces. *Australian Journal of Water Resources*, *10*(1), 23–33.

Leopold, L. B. (1994). *A View of the River*. Cambridge, Massachusetts: Harvard University

Press.

Lieb, D. A., & Carline, R. F. (2000). Effects of urban runoff from a detention pond on water quality, temperature and caged gammarus minus (say) (amphipoda) in a headwater stream. *Hydrobiologia*, *441*, 107–116. https://doi.org/10.1023/A:1017550321076

McNett, J. K., Hunt, W. F., & Osborne, J. A. (2010). Establishing Storm-Water BMP Evaluation Metrics Based upon Ambient Water Quality Associated with Benthic Macroinvertebrate Populations. *Journal of Environmental Engineering*, *136*(5), 535–541. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000185

Meyer, J. L., Paul, M. J., & Taulbee, W. K. (2005). Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society*, *24*(3), 602–612. https://doi.org/10.1899/04-021.1

Miguntanna, N. P., Liu, A., Egodawatta, P., & Goonetilleke, A. (2013). Characterising nutrients wash-off for effective urban stormwater treatment design. *Journal of Environmental Management*, *120*, 61–67. https://doi.org/10.1016/j.jenvman.2013.02.027

Munro, D. S. (1979). Daytime energy exchange and evaporation from a wooded swamp. *Water Resources Research*, *15*(5), 1259–1265. https://doi.org/10.1029/WR015i005p01259

N.C. DEQ. (2017). North Carolina Stormwater Control Measure Credit Document. *North Carolina Department of Environmental Quality*. Retrieved from https://files.nc.gov/ncdeq/Energy Mineral and Land Resources/Stormwater/BMP Manual/SSW-SCM-Credit-Doc-20170807.pdf

National Council of Examiners for Engineering and Surveying. (2013). Fundementals of Engineering Suplimental-Reference Handbook. Clemson, SC.

NC Climate Office. (2018). Weather and Climate Database.

NC DEQ. (2017). Stormwater Design Manual.

NCDENR. (2007). Stormwater Best Management Practices Manual. North Carolina Division of Water Quality.

NCDEQ. (2018a). Jordan Lake Nutrient Strategy. Retrieved September 24, 2018, from https://deq.nc.gov/about/divisions/water-resources/water-planning/nonpoint-source-planning/jordan-lake-nutrient#Rules

NCDEQ. (2018b). Understanding the Rules. Retrieved August 20, 2018, from https://deq.nc.gov/about/divisions/water-resources/water-planning/nonpoint-source-planning/jordan-lake-nutrient-1

NOAA National Weather Service. (n.d.). NOAA Atlas 14: Precipitation Frequency Data Server. Retrieved March 12, 2018, from https://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_cont.html

North Carolina Department of Environmental Quality. (2017). Stormwater Design Manual. Retrieved April 27, 2018, from https://deq.nc.gov/sw-bmp-manual

Novotny, V. (2003). *Water Quality: Diffuse Pollution and Watershed Management* (2nd ed.). Hoboken, New Jersey: John Wiley & Sons, Inc.

Page, J. L., Winston, R. J., & Hunt, W. F. (2015). Soils beneath suspended pavements: An opportunity for stormwater control and treatment. *Ecological Engineering*, *82*, 40–48. https://doi.org/10.1016/j.ecoleng.2015.04.060

R Core Team. (2017). R: A Language and Environment for Statistical Computing. Vienna, Austria. Retrieved from https://www.r-project.org/

Rosgen, D. L. (1994). A classification of natural rivers. *Catena*, *22*(3), 169–199. https://doi.org/10.1016/0341-8162(94)90001-9

Stanley, D. W. (1996). Pollutant Removal by a Stormwater Dry Detention Pond. *Source: Water*

Environment Research, *68*(6), 1076–1083. https://doi.org/10.2307/25044810

Stein, O. R., & LaTray, D. A. (2002). Experiments and modeling of head cut migration in stratified soils. *Water Resources Research*, *38*(12), 20-1-20–12. https://doi.org/10.1029/2001WR001166

Teledyne ISCO. (2013). 750 Area Velocity Module.

Temple, D. M., & Hanson, G. J. (1994). Headcut development in vegetated earth spillways. *Applied Engineering in Agriculture*, *10*(5), 677–682.

Tiedje, J. M., Sexstone, A. J., Myrold, D. D., & Robinson, J. A. (1982). Denitrification: ecological niches, competition and survival. *Antonie van Leeuwenhoek*, *48*(6), 569–583. https://doi.org/10.1007/BF00399542

U.S. Climate Data. (2018). Raleigh-Durham International Airport, North Carolina. Retrieved from https://www.usclimatedata.com/climate/raleigh-durham-int.-arpt/north-carolina/united-states/usnc0559

Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The River Continuum Concept.

Vaze, J., & Chiew, F. H. S. (2004). Nutrient loads associated with different sediment sizes in urban stormwater and surface pollutants. *Journal of Environmental Engineering*, *130*(4), 391–396. https://doi.org/10.1061/(ASCE)0733-9372(2004)130:4(391)

Walsh, C. J., Fletcher, T. D., & Ladson, A. R. (2009). Retention Capacity: A Metric to Link Stream Ecology and Storm-Water Management. *Journal of Hydrologic Engineering*, *14*(4), 399–406. https://doi.org/10.1061/ASCE1084-0699200914:4399

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan II, R. P. (2005). The urban stream syndrome : current knowledge and the search for a cure The urban stream syndrome : current knowledge and. *Journal of the North American Benthological Society*, *24*(3), 706–723. https://doi.org/10.1899/04-028.1

Wardynski, B. J., & Hunt, W. F. (2012). Are Bioretention Cells Being Installed Per Design Standards in North Carolina? A Field Study. *Journal of Environmental Engineering*, *138*(12), 1210–1217. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000575

Wardynski, B. J., Winston, R. J., Line, D. E., & Hunt, W. F. (2014). Metrics for assessing thermal performance of stormwater control measures. *Ecological Engineering*, *71*, 551–562. https://doi.org/10.1016/j.ecoleng.2014.07.068

West Virginia Department of Environmental Protection. (2012). West Virginia Stormwater Management & Design Guidance Manual, 1–22.

Wilcoxon, F. (1945). Individual Comparisons by Ranking Methods. *International Biometrics Society*, *1*(6), 80–83. https://doi.org/10.2307/3001946

Winston, R. J., Hunt, W. F., & Lord, W. G. (2011). Thermal Mitigation of Urban Storm Water by Level Spreader – Vegetative Filter Strips. *Thermal Mitigation of Urban Storm Water by Level Spreader-Vegetative Filter Strip*, *137*(8), 707–716. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000367.

CHAPTER 4 . REGENERATIVE STORMWATER CONVEYANCE AVAILABLE RESEARCH AND SYSTEM PERFORMANCE

4.1. Abstract

Regenerative Stormwater Conveyance (RSC) is a new Stormwater Control Measure (SCM) which integrates urban stormwater management with stream restoration techniques to simultaneously convey stormwater and treat pollutants. RSCs are most often used to stabilize eroded streams, gullies, and stormwater outfalls by filling the channel bed with sand and woodchip media; however, applications which excavate an exfiltration trench to provide media volume have been documented. Conversion of surface flows to sub-surface seepage is an important factor RSC employs for treatment; therefore, a new metric is proposed, the Storage Ratio, to compare designs and provide insight into the partitioning of storage volumes. Media saturation, whether purposefully-designed, from base flow, or due to groundwater intrusion, inhibits surface-to-seepage flow conversion; saturated media volumes should not be included in total storage volumes. This chapter reviews and synthesizes available RSC research conducted in Maryland and North Carolina to provide guidance for regulatory and design purposes.

4.2. Introduction

4.2.1. Land Use Change & Downstream Impacts

Urbanization replaces vegetated landscapes with impervious hardscapes such as buildings, sidewalks, and roads. Urban areas export sediment, hydrocarbons, nutrients, pathogens, heavy metals, and thermal pollution with higher runoff volumes and larger peak flows (Davis et al., 2003; Davis et al., 2006; Hunt et al., 2012; Jones et al., 2012; Ladson et al., 2006). Population growth and migration to urban centers increases demand for land use change, development, and resource consumption. Industry, agriculture, and drinking water systems compete for water resources and in some water-scarce areas water rights have become highly contested (Chattahoochee Riverkeeper, 2015; Hood & Sommer, 2016). As of 2014, 54% of the world's 7.2 billion inhabitants lived in urban areas with projections to add another 2.5 billion more people to urban populations by 2050 (United Nations, 2014). North Carolina is among the top five U.S. states in net migration population growth with 81,000 individuals moving to the state between 2015 to 2016. Total population growth during the same period was 112,000, a growth rate of 1.1%, surpassing the national average of 0.7% (Tippett, 2017).

Conventional stormwater management focuses on efficient drainage design; conveyance systems are designed to deliver impervious runoff directly to receiving waters with little or no attenuation or treatment (Burns et al., 2012). Efficient drainage alters infiltration, evapotranspiration (ET), surface and sub-surfaces flow regimes, increasing total runoff volume while reducing the time to peak discharge; it reduces stream base flows, elevates in-stream nutrient and contaminant concentrations, and alters stream morphology, stability, and ecosystem function (Burns et al., 2012; Fletcher et al., 2014; Meyer et al., 2005). Collectively, the aforementioned characteristics are referred to as the urban stream syndrome (Walsh et al., 2005). Research to mitigate the impacts of urbanization often focuses on hydrologic matching to a pre-urban or pre-development hydrologic condition by reducing Directly Connected Impervious Area (DCIA). Runoff retention and infiltration practices reduce the frequency of runoff events and provide treatment of a design volume (Burns et al., 2012; Davis, 2008; Fletcher et al., 2014; Ladson et al., 2006; Walsh et al., 2009).

Flashy urban runoff hydrographs produce high velocities and exceed critical shear stresses of stream beds and banks, initially degrading streams and then aggrading downstream (Walsh et al., 2005; Leopold, 1994). Steep changes of bed slope in headwaters or confined within the channel banks are referred to as headcuts and knick points, respectively (Bennett et al., 2000). Headcut formation and migration in vegetated spillways can be understood in 3 phases (Temple & Hanson, 1994):

1. loss of vegetation and flow concentration,
2. exceedance of critical shear stress, producing downward and downstream erosion, forming a headcut vertical face, and
3. upstream migration of headcut.

In urban drainage systems, discharge of concentrated flows at high velocities to ditches and earthen channels produce conditions analogous to phases 1 and 2, beginning headcut formation. Headcut retreat will advance upstream at a constant rate in homogenous (Flores-Cervantes et al., 2006). Migration in stratified soil is characterized by episodic failure and mass wasting after plunge pool undercutting of an upper, more cohesive soil layer; rate of migration is dependent on surface soil strength to resist cantilever mass failure and scour hole geometry (Stein & LaTray, 2002).

4.2.2. Stormwater Controls

In 1972, the Clean Water Act (CWA) was established to provide a framework for protecting U.S. water resources by preserving and restoring the chemical, physical, and biological integrity of the nation’s waters by requiring states to monitor, analyze, and develop water quality standards for their waters (EPA, 2016). Under the National Pollutant Discharge Elimination System (NPDES), it is the local municipalities’ responsibility to implement a plan for monitoring and permitting of point and non-point source discharges of effluent or runoff leaving municipalities, industrial facilities, or other entities to waters of the state (EPA, 2016; NCDENR, 2007).

NPDES requires adoption of minimum standards in post-construction management of stormwater for new development and redevelopment. Rules apply to new and redevelopment project areas that exceed 0.4-ha (1-ac); projects are categorized as low or high density depending on a 24% Built Upon Area (BUA) threshold. Low and high-density developments both require the use of (1) vegetated conveyances to the maximum extent possible, (2) 30-foot buffers between BUA and perennial and intermittent streams, and (3) deed restrictions and protective covenants to ensure developers pass on and communities maintain municipally-approved plans for stormwater management. In addition to low density development requirements, high density development requires the uses of structural SCMs for treatment of the water quality event, 38 mm in coastal counties and 25 mm elsewhere in North Carolina. Crediting of SCMs is focused on the reduction of nutrients and Total Suspended Sediment (TSS) (NCDEQ, 2017).

4.2.3. Regenerative Stormwater Conveyance

Regenerative Stormwater Conveyance (RSC) is a new, innovative, approach to managing stormwater and protecting streams. Patented in 2009 as a system and method, subsequent design recommendations were made by Anne Arundel County, MD, and followed by the West Virginia Department of Environmental Protection (WVDEP) (Underwood, 2002; Flores et al., 2012; WVDEP, 2012). RSC installation is growing across the Piedmont and Coastal Plains of the Mid-Atlantic physiographic regions under a variety of names including: Regenerative Step Pool Stormwater Conveyance, Bioinfiltration Stormwater Conveyance, and Coastal Plains Outfall (Cizek, 2014; Flores et al., 2012; WVDEP, 2012). RSCs integrate urban stormwater management and stream restoration engineering techniques to convey and treat non-point source pollutants (Flores et al., 2012) (Figure 4-1). A main objective of RSC installation is to stabilize eroded gullies and headcuts by filling gullies and reducing local bed slope (Bennett et al., 2000; Flores et al., 2012). A combination of Priority 1 (P1) and Priority 4 (P4) stream restoration techniques (Figure 4-2) are used to fill and stabilize headcuts with RSC systems (Doll et al., 2003). The streambed is elevated, similar to P1 restorations, by filling the existing headcut with coarse sand blended with organic matter; channel stabilization is accomplished with boulder-weir and riffle-cascade sequences, similar to P4 restorations, to provide grade control and support scour pools for energy dissipation during high flow events (Doll et al., 2003; Flores et al., 2012).

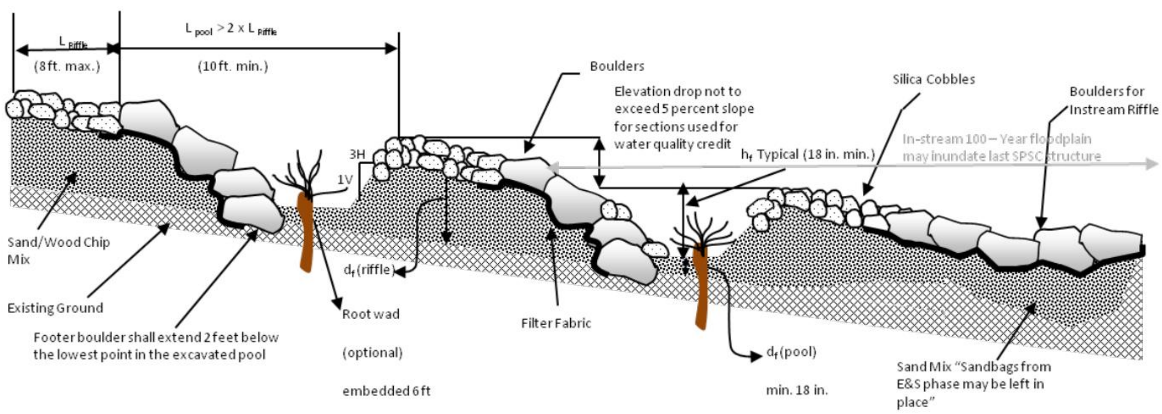


Figure 4-1. Typical RSC Cross-Section (Flores et al., 2012)

Recommendations by Flores et al. (2012) specify RSCs be designed with available storage capacity in surface pool and media pore space to capture a design storm of 25 mm. Pollutant treatment mechanisms include (1) settling in surface storage (ponded water), (2) filtration and (3) sorption processes in the media layer, (4) assimilation by vegetation, and (5) microbiological treatment; volume reduction mechanisms include (1) ET and (2) infiltration to subsoils (Cizek et al., 2016; Cizek et al., 2017; Davis et al., 2010; Flores et al., 2012; Koryto et al., 2018; Koryto et al., 2017).

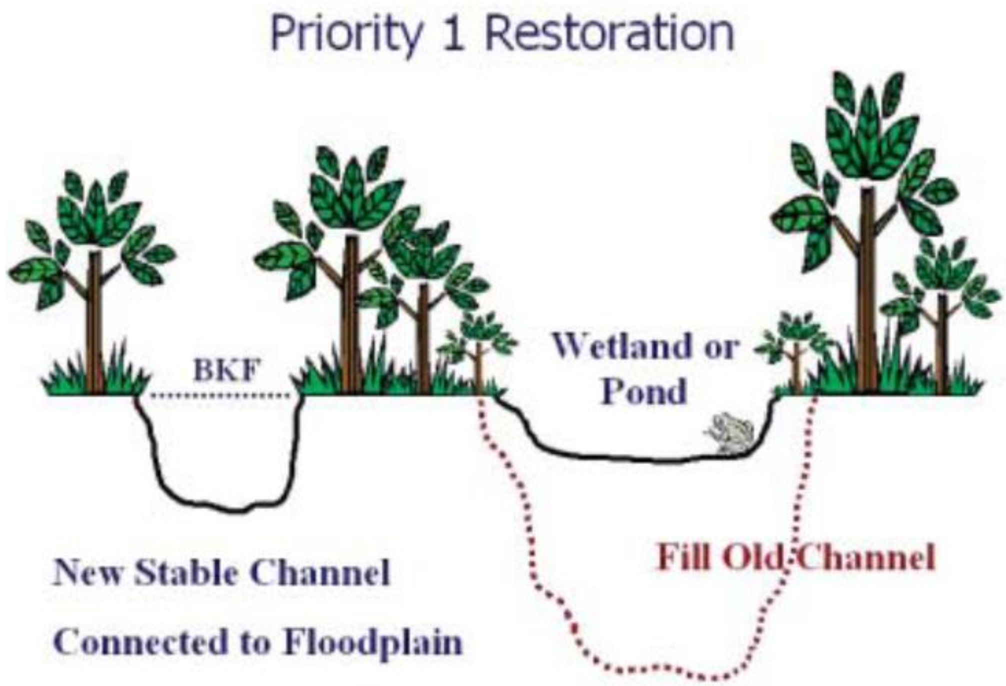


Figure 4-2. P1 Stream Restoration Cross-Section; (Doll et al., 2003)

Sand filtration and media treatment of stormwater runoff is a widely adopted practice; design standards and performance expectations exist to support the permitting and regulatory process (Barrett, 2003; Davis et al., 2003; Davis, 2008; Gallo, Moore, & Wywrot, 2012; Hunt et al., 2012; Wardynski & Hunt, 2012). Design recommendations for RSC include safe conveyance of 100-year, 24-hour storm flows via floodplain connection and energy dissipation by in-stream geomorphic structures (Flores et al., 2012; WVDEP, 2012). Adoption of this practice continues despite the lack of scientific consensus around design characteristics or expected treatment performance. The objective of this chapter is to synthesize available RSC research with the purpose to 1) provide North Carolina-centric design recommendations and 2) establish RSC treatment expectations under typical design configurations.

4.3. Previous RSC Research

4.3.1. Introduction

Several publications of field monitoring, simulated plot-scale events, and hydrologic modeling of RSCs focus on RSC hydrologic and water quality treatment (Table 4-1); published studies have been conducted in North Carolina and Maryland.

Table 4-1. RSC Report Summaries

Location & Source	*Publication Type*	*Method*	*Data*
Durham Co., NC (Koryto et al., 2017)	Peer-Review Article	Field Monitoring	Hydrology & Water Quality
Wake Co.-A, NC (Koryto et al., 2018)	Peer-Review Article	Plot-Scale Simulations	Hydrology & Water Quality
Brunswick Co., NC (Cizek et al., 2017)	Peer-Review Article	Field Monitoring	Hydrology
Alamance Co., NC (Cizek et al., 2016)	Peer-Review Article	Field Monitoring	Hydrology & Water Quality
Wake Co.-B, NC (Cizek, 2014)	Doctoral Dissertation	Plot-Scale Simulations	Hydrology & Water Quality
Wake Co.-C, NC (Chapter 3)	Master's Thesis	Field Monitoring	Hydrology & Water Quality
Anne Arundel Co.-A, MD (Brown et al., 2010)	Conference Proceeding	Hydrologic Modeling	Peak Flow Reductions
Anne Arundel Co.-B, MD (Palmer et al., 2014)	Monitoring Report	Field Monitoring	Hydrology & Water Quality
Anne Arundel Co.-C, MD (Palmer et al., 2014)	Monitoring Report	Field Monitoring	Hydrology & Water Quality
Anna Arundel Co.-D, MD (Browning, 2008)	Master's Thesis	Field Monitoring	Hydrology & Water Quality

Contributing drainage areas vary from 0.1 ha (plot-scale simulations) to 94.0 ha (field monitoring) with imperviousness ranging from 13-63% (Table 4-2). RSCs constructed in the Piedmont are frequently located in Hydrologic Soil Groups (HSGs) that are poorly drained (HSG D) while Coastal Plains RSCs are underlain by better drained soils (HSG A-B).

Table 4-2. RSC Site Conditions

Location & Source	*Ecological Region*	*Underlying Soils*	*Drainage Area (ha)*	*Proportion of Impervious Cover*
Durham Co., NC (Koryto et al., 2017)	Piedmont	HSG D	3.06	38%
Wake Co.-A, NC (Koryto et al., 2018)	Piedmont	HSG D	0.3	Unknown; Simulation
Brunswick Co., NC (Cizek et al., 2017)	Coastal Plains	HSG A	5.2	12%
Alamance Co., NC (Cizek et al., 2016)	Piedmont	HSG D	1.6	63%
Wake Co.-B, NC (Cizek, 2014)	Piedmont	HSG D	0.1	Unknown; Simulation
Wake Co.-C, NC (Chapter 3)	Piedmont	HSG D	19.9	24%
Anne Arundel Co.-A, MD (Brown et al., 2010)	Coastal Plains	HSG B	7.2	Unknown
Anne Arundel Co.-B, MD (Palmer et al., 2014)	Coastal Plains	HSG B	94.0	Unknown
Anne Arundel Co.-C, MD (Palmer et al., 2014)	Coastal Plains	HSG B	5.7	Unknown
Anna Arundel Co.-D, MD (Browning, 2008)	Coastal Plains	Unknown	86.6	37%

4.3.2. Design Elements

Design characteristics vary among RSCs researched (Table 4-3); some studies report design storm depths while others were calculated from total storage volumes available in surface ponding and media zones provided in the reference. Most of the Maryland studies do not report RSC system characteristics; this limits the ability to compare systems.

Table 4-3. Literature RSC Characteristics

Location & Source	*Design Storm Depth (mm)*	*Loading Ratio*[1]	*Storage Ratio*[2]	*Media Composition*	*Slope (%)*
Durham Co., NC (Koryto et al., 2017)	6.0	160:1	3.34:1	80% sand, 20% shredded mulch	10.0
Wake Co.-A, NC (Koryto et al., 2018)	25.0	72:1	0.87:1	ASTM C33 Sand	1.2
Brunswick Co., NC (Cizek et al., 2017)	31.6	310:1	1.47:1	ASTM C33 Sand	4.0
Alamance Co., NC (Cizek et al., 2016)	18.0	100:1	0.42:1	Sand	3.9
Wake Co.-B, NC (Cizek, 2014)	81.0	24:1	0.87:1	ASTM C33 Sand	1.2
Wake Co.-C, NC (Chapter 3)	11.7	25,000:1	0.01:1	Unknown	4.0
Anne Arundel Co.-A, MD (Brown et al., 2010)	25.0	NA	1.26:1	NA	Unknown
Anne Arundel Co.-B, MD (Palmer et al., 2014)	Unknown	114:1	Unknown	Unknown	Unknown
Anne Arundel Co.-C, MD (Palmer et al., 2014)	Unknown	Unknown	Unknown	Unknown	12.5
Anna Arundel Co.-D, MD (Browning, 2008)	Unknown	Unknown	Unknown	Unknown	0.8

1) $Loading\ Ratio = \frac{Watershed\ Area}{SCM\ Area}$
2) $Storage\ Ratio = \frac{Surface\ Storage\ Volume}{Media\ Storage\ Volume}$

4.3.2.1. Design Storm

Design guidance from Anne Arundel Co., Maryland, Department of Public Works (Flores et al., 2012) specifies capture of 25-mm water quality events in surface ponding and media storage; this Water Quality Volume (WQV) aligns with recommendations for other SCMs in the North Carolina (NC) Piedmont; 38-mm events are required to be captured and treated in NC coastal counties (NC DEQ, 2017). NC Bioretention Cell (BRC) design guidance does not allow inclusion of media pore space in available storage volume (NC DEQ, 2017). Wardynski & Hunt (2012) found 53% of NC BRCs were moderately to severely undersized; under-sizing will reduce hydrologic and water quality treatment by increasing the frequency of overflow (Brown & Hunt, 2011). Capture of the WQV wholly as surface storage provides a factor-of-safety in

BRC design (Brown & Hunt, 2011). Eliminating the inclusion of media pore space in design calculations of RSCs is not practical. RSCs are typically constructed in eroded gullies and confined riparian corridors where minimizing disturbance is often a project objective (Cizek et al., 2017; Filoso, 2012; Koryto et al., 2017). The Durham Co. RSC was designed to minimize impact on the surrounding riparian forest (Koryto et al., 2017); both the Durham and Alamance Co. RSCs were under-sized, 25% and 72%, respectively, compared to the required 25-mm event (Table 4-3) (Cizek et al., 2017; Koryto et al., 2017). The Wake Co.-C RSC was constructed to stabilize an advancing headcut in-series with (downstream of) a dry pond; the small SCM area resulted in a design storm depth of 11.7-mm or 3.2% of the required capture volume (Chapter 3). The Brunswick Co. RSC, located in the Coastal Plains, has a calculated design storm of 31.6-mm, 83% of the required 38-mm for SCMs in NC coastal counties (Cizek et al., 2017; NC DEQ, 2017). Design recommendations for RSCs include safe conveyance of 100-year, 24-hour storm flows via floodplain connection and energy dissipation by in-stream geomorphic structures (Flores et al., 2012; WVDEP, 2012). This sizeable conveyance design storm appears to protect against severe degradation associated with very large and intense events.

Cizek (2014) simulated 2-, 10-, and 50-year, 24-hour storm events in a plot-scale RSC; the simulated watershed was specified as a 0.1-ha urban development in Raleigh, NC. Storage volumes determined a design storm of 81.0-mm, more than 3 times greater than design requirements, approximately equal to the 2-year, 24-hour storm (Cizek, 2014). Simulated events by Koryto et al. (2018) were predicated on a 100%-sized RSC able to capture a 25-mm event for the hypothetical 0.3-ha watershed. Brown et al. (2010) reported model results for RSC design scenarios using 25-mm design storms.

4.3.2.2. Loading Ratio

Loading ratios relate the size of an SCM to that of its contributing area (Davis, 2008; Hunt et al., 2012). Under-sizing of the Durham Co. RSC resulted in a loading ratio of 160:1 (Koryto et al., 2017). The Wake Co.-C RSC has the highest loading ratio of 25,000:1 (Table 4-3); however, it received dry pond-mitigated flow. The Alamance Co. RSC was loaded at 100:1 (Cizek et al., 2017). Sufficient information was provided to estimate the Anne Arundel Co.-B loading ratio, 114:1; although, this was not reported (Palmer et al., 2014).

The relatively large SCM size for the simulated watershed in Cizek's (2014) plot-scale experiments yielded the smallest loading ratio, 24:1. Koryto et al.'s (2018) simulations on a 100% sized RSC with a hypothetical 0.3-ha watershed had a loading ratio of 72:1. Loading ratios for BRCs typically range between 14:1 – 20:1 but can be larger (e.g. 45:1) (Davis, 2008; Hunt et al., 2012).

4.3.2.3. Storage Ratio

An important presumption governing RSC treatment is the conversion of surface flows to subsurface seepage, which provides filtration, sorption, and biological treatment in the media zone; therefore, the ratio of surface-to-media storage (Eq. 4.1) may provide insight into RSC treatment potential. The storage ratio can be illustrated by two BRCs both treating a 464.5-m^2 watershed with SCM areas of 23.3-m^2 (20:1 loading ratio) (Figure 4-3). This metric is not reported for BRCs or RSCs but can be useful in comparing design scenarios.

$$Storage\ Ratio = \frac{Surface\ Storage\ Volume}{Media\ Storage\ Volume} \quad \text{(Eq. 4.1)}$$

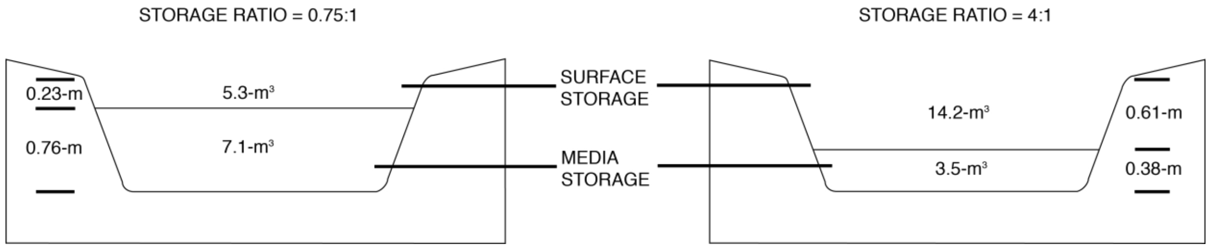


Figure 4-3. Storage Ratio Diagram

The Durham Co. RSC was designed within an existing gulley to minimally impact surrounding riparian forest; this resulted in the largest storage ratio, 3.34:1 (Table 4-3). The Alamance Co. RSC storage ratio was 0.42:1. The Wake Co.-C RSC has the smallest storage ratio, 0.01:1.

The storage ratio is independent of loading ratios and design storm sizing, as illustrated by Koryto et al. (2018) and Cizek (2014) using the same plot-scale RSC with a storage ratio of 0.87:1 despite differing simulated watershed areas and design storms. This metric provides designers a tool to understand the treatment implications of design options.

4.3.2.4. Media & Mulch

RSCs' ability to safely convey large flows while providing media treatment motivates designers to consider this practice; however, as with early BRC studies, proper media selection and design configuration is critical to achieving the desired system performance (Brown & Hunt, 2011; Davis et al., 2003; Kim et al., 2003). Contrary to BRC media specifications (N.C. DEQ, 2017), existing RSC media recommendations do not include specifications for fractions of fines or sorption of phosphorous (such as the P-Index) while recommending more than twice the percentage of organic matter (OM) (Table 4-4) (Flores et al., 2012). The P-Index, a measure of phosphorus bound to soil particles, has been identified as an important predictor of media's ability to bind or leach phosphorus (Hardy et al., 2009; Hunt et al., 2006). Media infiltration and saturated hydraulic conductivity rates are affected by the fraction of fines (Hsieh & Davis, 2005a); increasing the fraction of media fines usually improves Total Phosphorus (TP) removal by increasing the amorphous iron and aluminum oxides binding sites (Davis et al., 2010; Hsieh & Davis, 2005a). RSCs in Brunswick and Alamance counties and Wake Co.-A and -B did not contain a carbon source mixed in-media; instead, they contained a sand bed with 5-10-cm of triple shredded hardwood or a composted hardwood mulch layer on top of the RSCs (Cizek et al., 2016, 2017; Koryto et al., 2018). These systems were also hydro-seeded with North Carolina Department of Transportation (NCDOT) seed stabilization mix (Cizek et al., 2016, 2017; Koryto et al., 2018). Both (1) mulching and hydro-seeding and (2) compost-amending disturbed surfaces promote vigorous vegetation establishment and is recommended for temporary soil stabilization (Flores et al., 2012). During high flow events, if sufficient pool storage is not provided or steep slopes exist, mulch can be mobilized and washed from the RSC (Davis et al., 2010; Miguntanna et al., 2013).

Table 4-4. Media Comparison

Media Component	*BRC Specification*[1]	*RSC Specification*[2]
Sand	75-85%	80%
Fines (silt & Clay)	8-15%	NA
Organic Matter	5-10% (Pine Bark)	20% (Woodchips)
P-Index	≤ 30 in NSW* ≤ 50 elsewhere	NA

*Nutrient Sensitive Waters
1) N. C. DEQ (2017)
2) Flores et al. (2012)

The Wake Co.-C RSC included an in-media OM mixed with sand; however, exact OM type and media composition are unknown (Chapter 3). Media OM is an important component that provides sorption sites for hydrophobic organic compounds, heavy metals, orthophosphate (Ortho-P), and ammonium (NH_4^+) (Davis et al., 2010; Hsieh & Davis, 2005a) and organic carbon for denitrification reactions (Davis et al., 2010; Tiedje et al., 1982). Media without incorporated carbon may have limited denitrification potential (Davis et al., 2010; Tiedje et al., 1982).

4.3.2.5. In-Stream Structures

RSCs integrate urban stormwater management and stream restoration engineering techniques to convey and treat non-point source pollution (Flores et al., 2012) (Figure 4-1). Often an objective of RSC installation is to stabilize eroded gullies and headcuts by filling gullies and reducing local bed slope (Bennett et al., 2000; Flores et al., 2012). The streambed is elevated by filling the existing headcut with media; channel stabilization is accomplished with boulder-weir or riffle-pool sequences which provide grade control and support scour pools for energy dissipation during high flow events (Flores et al., 2012).

Flores et al. (2012) recommends 3 main design types for constructed Step Pool Storm Conveyances (SPSC) depending of valley slope: RSC, Seepage Wetland Complex (SWC), Constructed In-Stream Riffle (CISR) (Table 4-5). Design types attempt to create, or form over time, channel structures that mimic Rosgen A, B, or DA stream classifications (Figure 4-4) (Rosgen, 1994). Depending on valley slope and design constraints (e.g., available construction materials, project area, economics, etc.) riffle-pool or cascade sequences are used to convey stormflows downslope (Figure 4-5).

Table 4-5. Step Pool Storm Conveyance Design Configurations Flores et al. (2012)

Design Type	*Characteristics*	*In-Stream Structures*	*Bed Slope*
Regenerative Stormwater Conveyance	Rosgen A or B class stream	Shallow pools with riffle grade control, native vegetation, underlying media	> 2%
Seepage Wetland Complex	Rosgen DA	Riffle grade control, lateral shallow pools to promote highly connected floodplain	< 2%
Constructed In-Stream Riffle	Forms Rosgen DA Over Time	Rock riffles to encourage upstream sedimentation and floodplain connection; no media	Not Specified

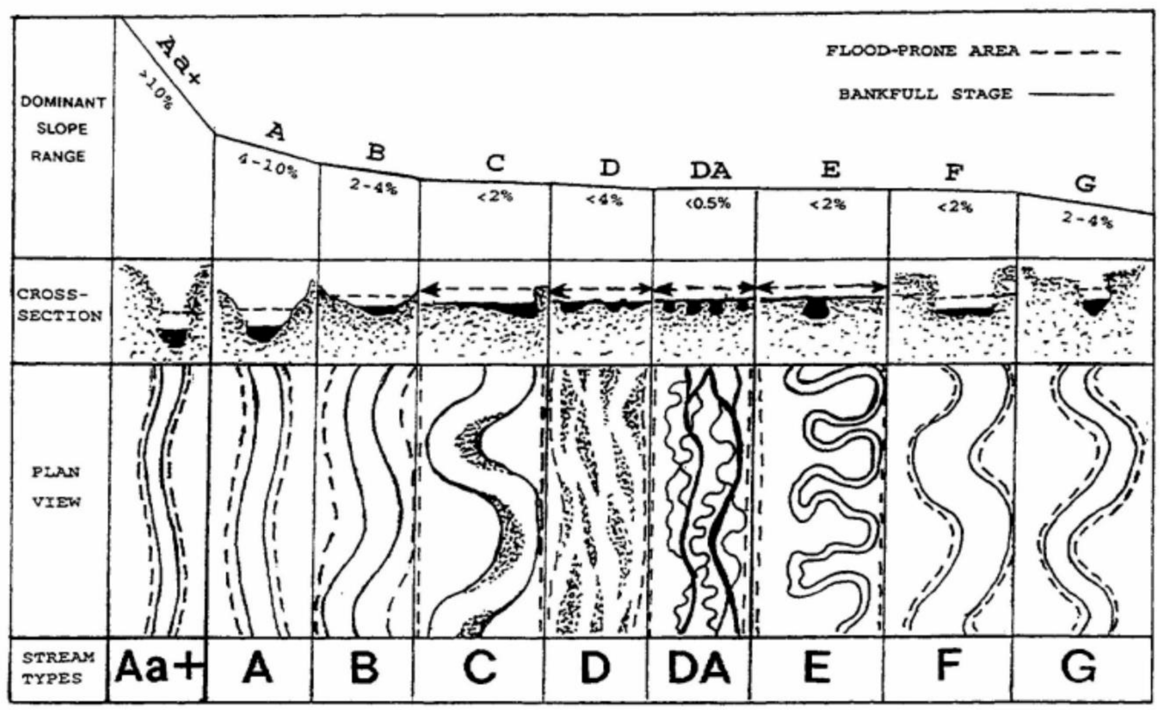


Figure 4-4. Major Stream Classes Rosgen (1994)

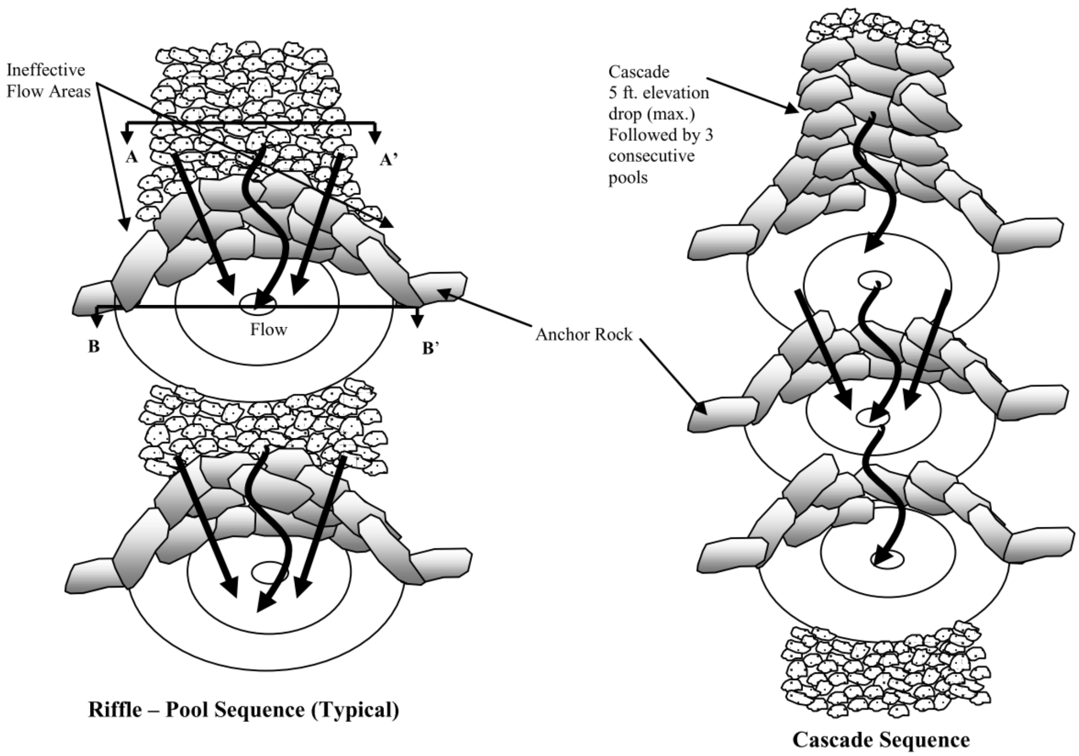


Figure 4-5. Typical Grade Control Structures Flores et al. (2012)

All design types are specified for stabilization of entrenched streams and gullies (Flores et al., 2012); however, only RSCs and SWC explicitly reference use of media to promote filtration of storm flows (Flores et al., 2012). Furthermore, design recommendations for SWCs only specify use of media to create berms and lateral pools adjacent to the channel where storm flows may pond and filter back into the channel (Figure 4-6) (Browning, 2008; Flores et al., 2012); use of in-channel media for SWC is unclear. CISRs promote sedimentation with the goal of filling gullies over time and reconnecting channels to adjacent floodplains (Flores et al., 2012). CISRs and SWC are sometimes employed together to divert flows over-bank into media berms and lateral pools (Palmer et al., 2014).

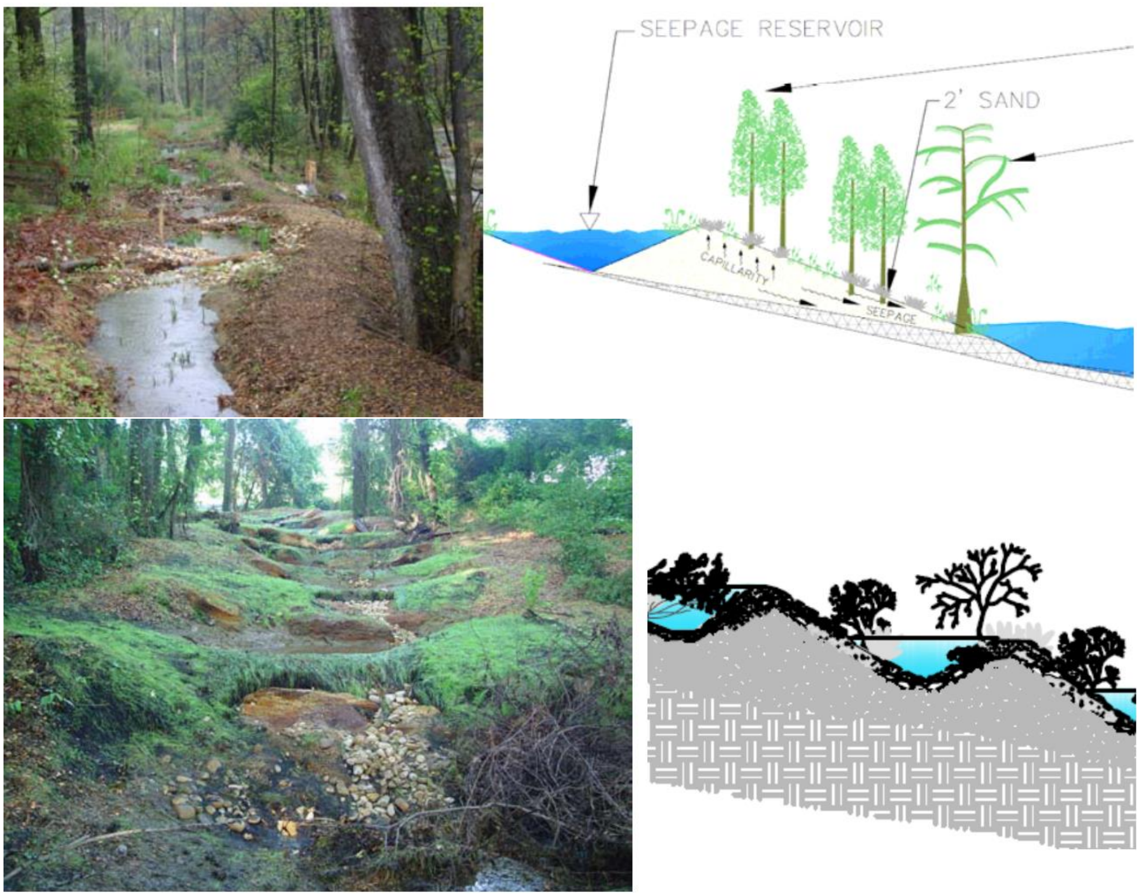


Figure 4-6. Seepage Wetland Complex (Top) v. RSC (Bottom) Comparison Browning (2008)

Treatment mechanisms for most media-based SCMs include: (1) settling in surface storage areas, (2) filtration, (3) sorption to media components, (4) assimilation by vegetation, and (5) microbiological treatment; volume reduction mechanisms include: (1) ET and (2) exfiltration to subsoils (Cizek et al., 2016; Cizek et al., 2017; Davis et al., 2010; Flores et al., 2012; Koryto et al., 2018; Koryto et al., 2017). Because both SWCs and RSCs are media-based SCMs, researchers have made treatment comparisons between their performance (Cizek et al., 2016, 2017; Filoso, 2012; Koryto et al., 2018, 2017). Anne Arundel Co.-B and -D were designed as SWCs (Browning, 2008; Filoso, 2012).

Figure 4-5 illustrates typical in-stream geomorphic structures; however, the Brunswick Co. RSC employed pre-cast concrete weirs (Cizek et al., 2017). Hester & Doyle (2008) studied weirs (i.e., impermeable channel-spanning obstructions, perpendicular to flow), steps (i.e., vertical drops in channel bed), and lateral structures (i.e., like weirs but greater height and do not completely span channel width) (Figure 4-7). In-stream geomorphic structure size, type, and slope have significant effects on hyporheic exchange metrics, down-welling rate, residence time, and down-welling depth (Hester & Doyle, 2008).

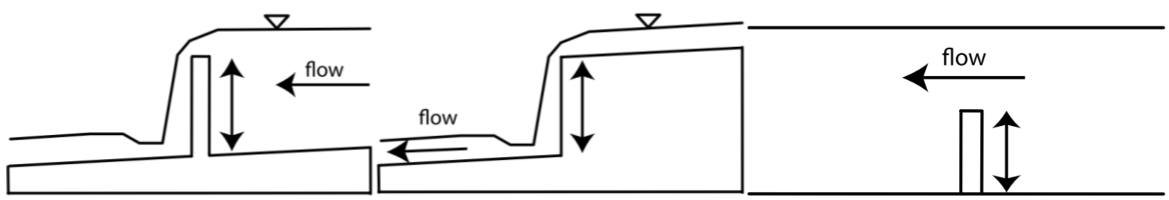


Figure 4-7. Simplified Illustrations of In-Stream Structures; Weir (Left), Step (Middle), Lateral (Right) Hester & Doyle (2008)

Down-welling flux rates (i.e., amount and depth reached by surface flows into stream bed) for weirs were twice as large as steps (Hester & Doyle, 2008). HEC-RAS and MODFLOW results highlight hyporheic exchange across channel distance (Figure 4-8)(Hester & Doyle, 2008).

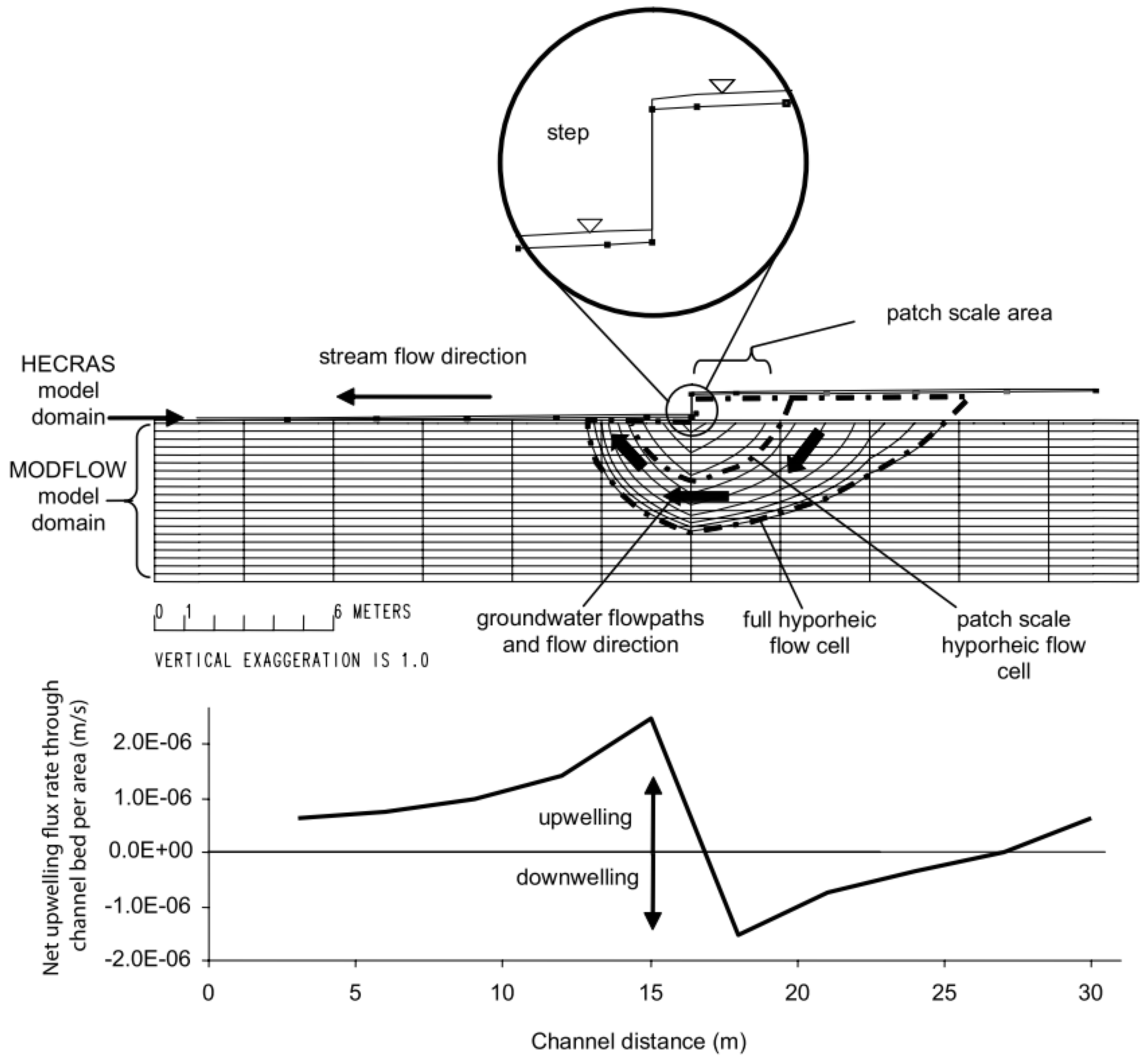


Figure 4-8. Longitudinal Profile with Hyporheic Exchange Flow Paths Hester & Doyle (2008)

Hyporheic exchange in natural streams is likely analogous to media filtration and seepage in RSCs when shallow groundwater is present or saturated conditions are prevalent. Hyporheic flow enhances advection of dissolved organic compounds and nutrients to microbial habitats, and thus, influences transformation rates (Lawrence et al., 2013). A better understanding of the interactions between various RSC design parameters (e.g., bed slope, bed substrate, media composition, etc.) and in-stream geomorphic structures will likely improve designers' ability to have RSCs infiltrate surface flows during saturated conditions.

4.3.2.6. Slope

RSC slope is generally dictated by existing site grade, and field studies represent this variability with slopes ranging from 0.8% to 10% (Table 4-3) (Cizek et al., 2016, 2017; Flores et al., 2012; Koryto et al., 2018); however, Anne Arundel Co. specifies that no water quality credit is available for RSCs with slopes greater than 5% (Flores et al., 2012). Steep slopes contribute to higher hydraulic gradients which force surface flow into subsurface pore space (Brown et al., 2010); however, once media pore space becomes saturated, surface and subsurface flow exchange likely behaves more similar to hyporheic exchange (Hester & Doyle, 2008). Hester & Doyle (2008) conducted multidimensional, steady-state, modeling of a small gaining stream's hyporheic response to in-stream geomorphic structures; this study determined that hyporheic exchange metrics, down-welling rate, residence time, and down-welling depth significantly decrease exponentially as channel slope increased.

4.3.2.7. Exfiltration Trench

RSC application has expanded beyond entrenched channels and gullies because of their ability to simultaneously convey large storms and provide pollutant treatment. RSCs are now constructed where no or minimal gullying existed prior to installation (Chapter 3; Cizek et al., 2016, 2017). These systems employ an exfiltration trench where media is placed prior to bed form and grade control installation. The exfiltration trench provides capacity for subsurface seepage and associated pollutant treatment (Figure 4-9) (Chapter 3; Cizek et al., 2016, 2017). In excavation of SCMs, the infiltration capacity of underlying soils should be preserved; excavation with the "rake" method, versus "smear" or "scoop," has been recommended for the final 0.3-m of excavation depth to preserve soil infiltration rates (Brown & Hunt, 2009; NC DEQ, 2017). This would presumably enable the RSCs' exfiltration trench to better exfiltrate.

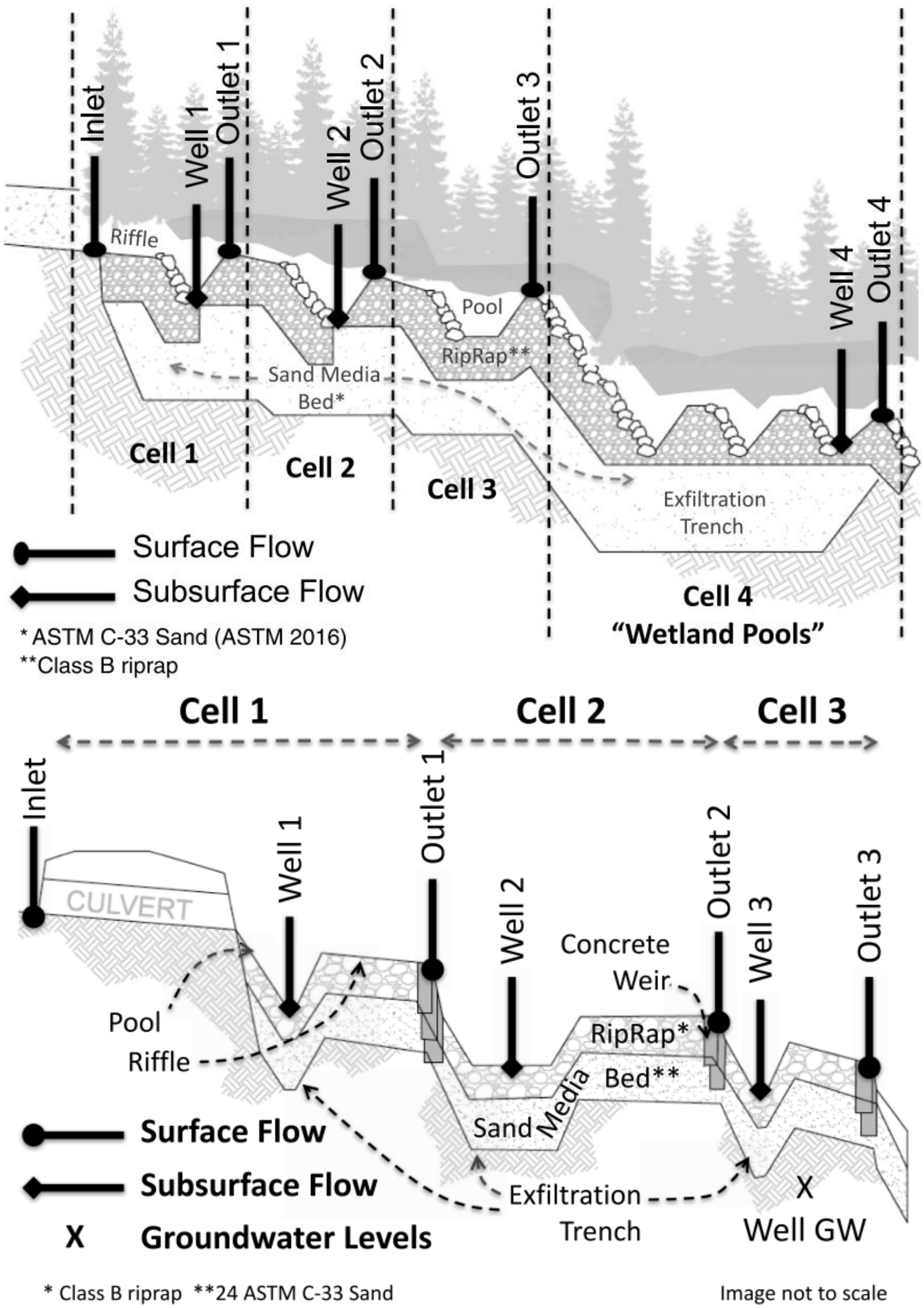


Figure 4-9. Exfiltration Trench ; Alamance Co. (Top) Brunswick Co. (Bottom) Cizek et al. (2016, 2017)

4.3.2.8. Forebay

Forebay pretreatment is required for some SCMs in NC; forebays promote influent stilling and energy dissipation allowing large particles and gross solids to settle (NC DEQ, 2017). Effective sedimentation requires reduction of flow velocities and creation of laminar flow so particles settle via gravity (Stokes' Law) to the pool bottom; large, dense particles are readily settled compared to smaller, low density particles (Davis et al., 2010). Forebays provide a location where a majority of influent sediment can be captured for ease of maintenance and removal (NC DEQ, 2017). Forebays are typically 0.5-0.8-m in depth, providing a deeper area, or plunge pool, near flow entry and decreasing in depth toward flow exit (Figure 4-10) (NC DEQ, 2017). Sufficient pool storage is necessary to create conditions suitable for sedimentation and to prevent pollutant resuspension (Davis et al., 2010; Miguntanna et al., 2013). Most RSC studies observe substantial sedimentation occurring in the first pool, suggesting it is a defacto forebay (Cizek et al., 2016; Filoso, 2012; Koryto et al., 2017). Forebay characteristics are not atypical of RSC pools (Figure 4-10, Figure 4-1); therefore, additional design and maintenance consideration is merited for the first RSC pool to ensure long-term treatment.

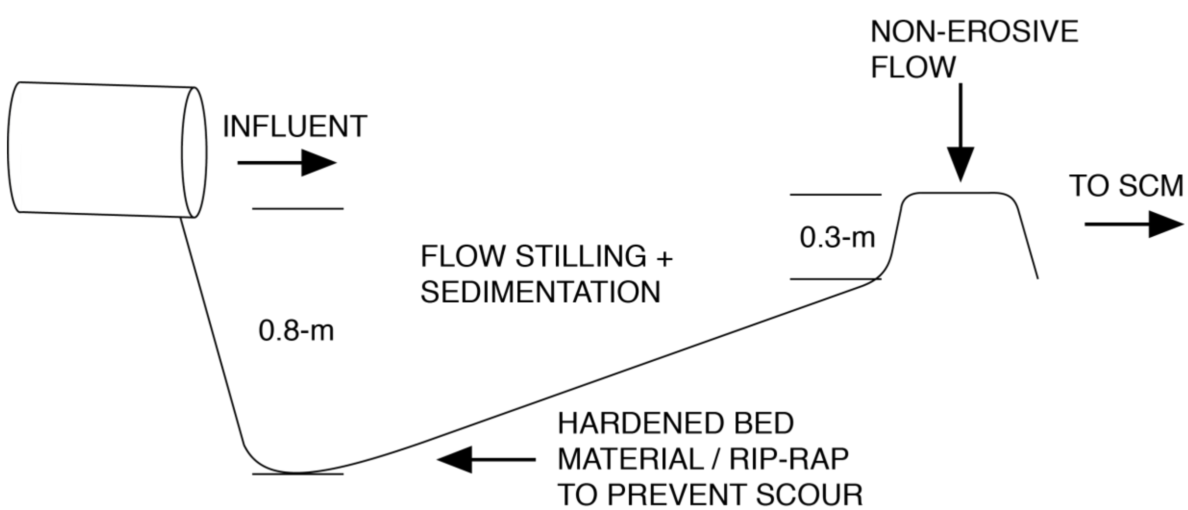


Figure 4-10. Typical Forebay Design Diagram

Forebays improve long-term performance of SCMs. In fact, SCMs with forebays are expected to increase sediment removal by a practice by up to 20 years over cognate SCMs without forebays (NC DEQ, 2017). Forebay clean-out procedures should be considered in design plans to allow non-erosive access and equipment staging (NC DEQ, 2017). Forebay sizing recommends approximately 20% of the design storm volume be held in the forebay (NC DEQ, 2017), but sizing of RSC forebays are not likely capable of satisfying 20% capture volume due to limited area in corridors and rights-of-way (Cizek et al., 2017; Filoso, 2012; Koryto et al., 2017).

4.3.2.9. Vegetation

Flores et al. (2012) specify diverse and dense native vegetation for use in appropriate wetness zones of SPSCs. Vegetation provides flow stilling, enhanced sediment removal and pollutant assimilation (Flores et al., 2012). Vegetation can improve infiltration rates in BRC by creating macro pores around plant roots and improve volume reductions via ET (Li et al., 2009; Wardynski & Hunt, 2012); however, few RSC studies have documented established vegetation (e.g., Browning, 2008; Cizek et al., 2016, 2017; Filoso, 2012; Koryto et al., 2018, 2017), possibly due to early successional monitoring. The Wake Co.-A and -B RSC was established with warm season grasses; however, it only received flow during simulation events. The Brunswick Co. RSC was armored with Class A rip-rap and did report the establishment of wetland vegetation in an area of prolonged saturation (Cizek et al., 2017). Portions of the Durham, Alamance, Brunswick Co. RSCs and the entire Wake Co.-C RSC experienced saturated conditions, similar to stormwater wetlands (Chapter 3; Cizek et al., 2016, 2017; Koryto et al., 2017).

Reserachers comparing plant species and cultivars determined plant selection will impact nutrient concentration and load reductions (Bratieres et al., 2007; Read et al., 2008; Turk et al., 2017). In high nutrient load bioretention systems, nitrogen and phosphorus sequestration in root and shoot growth can range from 0-11% (Turk et al., 2017). Planting in bioretention systems can increase ET rates and improve hydrologic treatment (Brown et al., 2013; Wadzuk et al., 2015). Vegetation in constructed wastewater treatment and urban stormwater wetlands assimilate nutrients and polluants reducing pollutant concentrations and loads (Kadlec, 1999; Rogers et al., 1991; Song et al., 2001). Hatt et al. (2009) vigorous vegetation growth significantly increased infiltrtion capacity; indicating vegetation root zone helped maintain media porosity (Hatt et al., 2009). Therefore, vegetation with comparatively larger root growth may perform better at maintaining RSC infiltration capacity and media porosity. Wetland planting recommendations for constructed stormwater wetlands apply a zonal planting method based on ponding depths (NC DEQ, 2017). The unique and variable conditions created in RSC applications (e.g., high flow velocities, prolonged media saturation, well-drained, temporary inundation) suggest more study is needed for proper plant selection guidance.

4.3.3. Study Methods

4.3.3.1. Hydrology

Table 4-6. Hydrologic Monitoring Comparisons

Location & Source	*Monitoring Period*	*Monitoring Design*	*Equipment Used*
Durham Co., NC (Koryto et al., 2017)	Feb. 1, 2015 – Mar. 31, 2016	Upstream/Downstream + 2 mid-reach stations at riffle heads	Compound Cross-Channel Weirs + HOBO U20 Water Level Logger, ISCO 750 Area Velocity Meter, & Manual + Tipping Bucket Rain Gauge
Wake Co.-A, NC (Koryto et al., 2018)	Oct. 26, 2015 – Nov. 21, 2015	Upstream/Downstream + Underdrain (seepage)	ISCO 6712 + ISCO 730 Bubbler Modules, Weir boxes, Simulated Hydrograph
Brunswick Co., NC (Cizek et al., 2017)	Oct. 2012 – Mar. 2014	Inflow + Outflow of all 3 pools	Compound Cross-Channel Weirs + HOBO U20 Water Level Logger, & Manual + Tipping Bucket Rain Gauge
Alamance Co., NC (Cizek et al., 2016)	July 10, 2013 – June 10, 2014	Inflow + Outlet of each pool, 1-3, & system outflow	Inlet ISCO 750 Area Velocity Meter, Compound Cross-Channel Weirs + HOBO U20 Water Level Logger, & Manual + Tipping Bucket Rain Gauge
Wake Co.-B, NC (Cizek, 2014)	Jan. 15, 2014 – Feb. 7, 2014	Inflow/Outflow + Underdrain (seepage) & Wells in pools 2 & 3	ISCO 4230 Bubblers + Weirs & Simulated Hydrographs
Wake Co.-C, NC (Chapter 3)	July 12, 2017 – July 9, 2018	Upstream/Downstream + Media Well Stage	ISCO 6712 + ISCO 730 Bubbler Modules, Cross-Channel Weirs + HOBO U20 Water Level Logger, ISCO 750 Area Velocity Meter, & Manual + Tipping Bucket Rain Gauge
Anne Arundel Co.-B, MD (Palmer et al., 2014)	2011 – 2012	Upstream/Downstream, Instantaneous Velocity measurement + Stage-Discharge Relationship	ISCO 6712, HOBO Water Level Logger, Onset HOBO RG3-M Tipping Bucket Rain Gauge, Velocity Measurements
Anne Arundel Co.-C, MD (Palmer et al., 2014)	Jan. 2011 – Sep. 2012	Downstream + Paired Watershed, Instantaneous Velocity measurement + Stage-Discharge Relationship	ISCO 6712, HOBO Water Level Logger, Onset HOBO RG3-M Tipping Bucket Rain Gauge, Parshall Flume & Velocity Measurements
Anna Arundel Co.-D, MD (Browning, 2008)	Oct. 2007 – Apr. 2008	Upstream/Downstream	Bimonthly measurement of cross-sectional area and velocity, Portable flow meter, & stream gauge

Hydrologic field studies of RSCs were conducted with similar methodologies (Table 4-6); generally, upstream/downstream monitoring designs are employed (Figure 4-11). Additional monitoring at some or all riffle heads allowed assessment of pool-by-pool flow mitigation. Weirs in conjunction with pressure transducer water level loggers are frequently used in field monitoring of SCMs (Figure 4-12) (Davis, 2008; Hunt et al., 2008; Page et al., 2015).

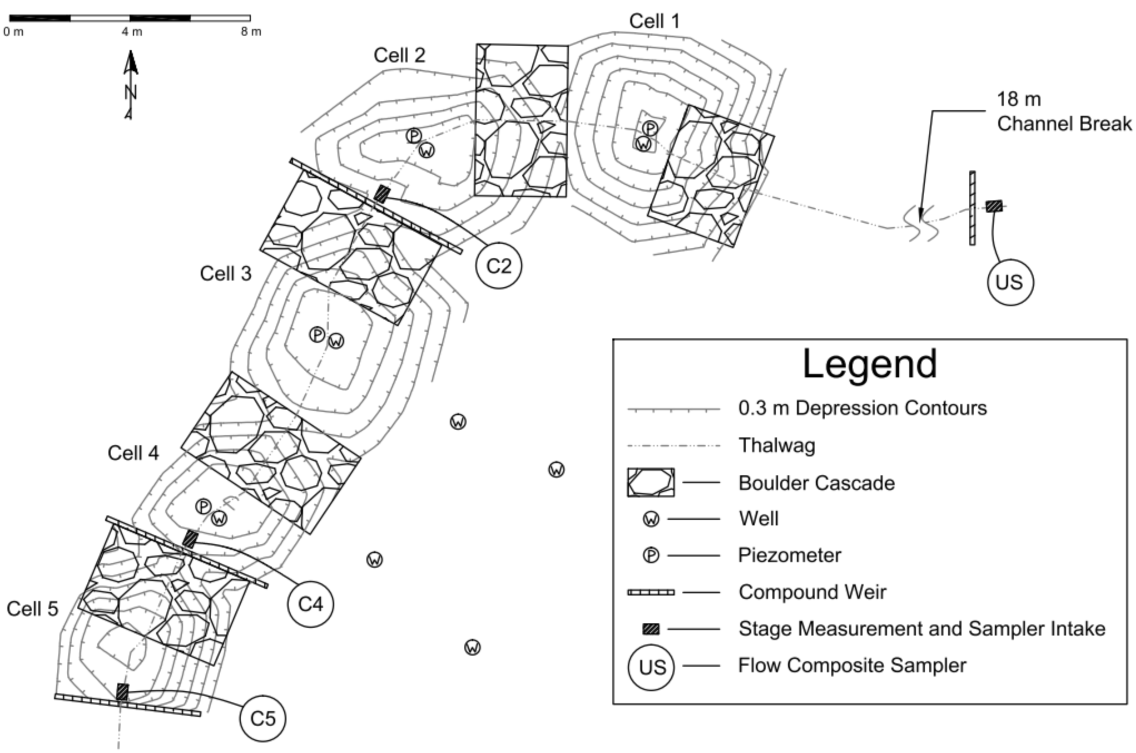


Figure 4-11. Typical Monitoring Design Koryto et al. (2017)

Figure 4-12. Typical Weir & Pressure Transducer in PVC Well (Koryto, 2016)

Monitoring of stream restoration often encompasses larger drainage areas and experiences higher flows, rendering in-stream hydraulic control structures infeasible. Palmer et al. (2014) used instantaneous velocities collected with a hand-held probe, the cross-sectional area method, and pressure transducers for stage measurement to determine stage-discharge relationships of monitoring locations; a Parshall flume was installed at one site to fix cross-sectional geometry. Stage-discharge relationships were used to determine stream discharge when stage was recorded (Palmer et al., 2014). Browning (2008) also employed stage, instantaneous velocity, and cross-sectional area measurements to construct a stage-discharge relationship across a range of flows.

Accurate hydrologic monitoring of RSCs is often more difficult than that for other SCMs, because of large flow rate variability. ISCO 6712 automated samples in combination with ISCO 730 bubbler modules and HOBO U20 water level loggers are commonly added to the monitoring design to allow stage measurement for stage-discharge relationships and flow rate determination. Many of these monitoring efforts experienced hydrologic monitoring errors from high influent velocities, measurement site aggradation, ponding and back water effects, autosampler malfunctions, and difficulties preventing preferential flow underneath cross-channel weirs (Chapter 3; Cizek, 2014; Cizek et al., 2016; Filoso, 2012; Koryto et al., 2017).

4.3.3.2. Water Quality

Table 4-7. Water Quality Monitoring Comparisons

Location & Source	*Monitoring Period*	*# of Samples*	*Equipment Used*	*Parameters Analyzed*
Durham Co., NC (Koryto et al., 2017)	Feb. 1, 2015 – Mar. 31, 2016	22	ISCO 6712 Automated Sampler w/ ISCO 730 Bubbler Flow Module & Monthly Base Flow Grab samples & HOBO U26 DO Data Logger	TSS, NO2,3-N, TKN, TAN, TP, Ortho-P, Cu, Pb, Zn, Temperature, and Dissolved Oxygen (DO)
Wake Co.-A, NC (Koryto et al., 2018)	Oct. 26, 2015 – Nov. 21, 2015	12	ISCO 6712 + ISCO 730 Bubbler Modules, Weir boxes, Simulated Hydrograph, & Nutrient Spikes	NO2,3-N, TKN, TAN, TN, Ortho-P, & TP
Alamance Co., NC (Cizek et al., 2016)	July 10, 2013 – June 10, 2014	20	ISCO 6712 Automated Sampler w/ ISCO 730 Bubbler Flow Module	TSS, TP, Ortho-P, TN, NO2,3-N, TKN, TAN, pH, Temperature
Wake Co.-B, NC (Cizek, 2014)	Jan. 15, 2014 – Feb. 7, 2014	9	ISCO 4230 Bubblers + Weirs, UV-Visual Spectrometer, Inlet & Outlet Grab Samples, Simulated Hydrographs & Nutrient Spikes	NO2,3-N, TKN, TAN
Wake Co.-C, NC (Chapter 3)	Feb. 9, 2017 – July 7, 2018	20	ISCO 6712 Automated Sampler w/ ISCO 730 Bubbler Flow Module & Base Flow Grab Samples	TSS, TP, Ortho-P, TN, NO2,3-N, TKN, TAN, Temperature
Anne Arundel Co.-B, MD (Palmer et al., 2014)	2011 – 2012	7	ISCO 6712, HOBO Water Level Logger, Onset HOBO RG3-M Tipping Bucket Rain Gauge, Velocity Measurements, Monthly Grab Samples	TSS, PN, Ortho-P, TDP, TDN, DON NO3--N, NH4+
Anne Arundel Co.-C, MD (Palmer et al., 2014)	Jan. 2011 – Sep. 2012	10	ISCO 6712, HOBO Water Level Logger, Onset HOBO RG3-M Tipping Bucket Rain Gauge, Parshall Flume & Velocity Measurements, Monthly Grab Samples	TSS, PN, Ortho-P, TDP, TDN, DON NO3--N, NH4+
Anna Arundel Co.-D, MD (Browning, 2008)	Oct. 2007 – Apr. 2008	16-30	Bimonthly samples & Hydrolab Water Quality Probe w/ Scout 2 data measurements	TSS, NO2,3-N, TAN, pH, Temperature, DO, Specific Conductivity

Monitoring parameters and sampling methods are similar among field monitored RSCs (Table 4-7); automated samplers were typically used during field monitoring. Grab sampling is less common in storm flow analysis because of its inability to characterize a complete event; however, inter-event baseflow sampling is common (Browning, 2008; Filoso, 2012; Koryto, 2016). Monitoring logistics, equipment malfunctions, and negligible outflow limited sample collection for some studies (Browning, 2008; Filoso, 2012). Plot studies have exclusively used time-incremental grab samples to quantify pollutant reduction throughout simulated storm events (Cizek, 2014). All studies examined nutrient and TSS capture (Table 4-7); Koryto et al. (2018) also assessed Copper (Cu), Lead (Pb), Zinc (Zn), and Dissolved Oxygen (DO) within the media zone. For storm simulations on a plot-scale RSC, dissolved nutrient species were created using spiked inflow (Cizek, 2014; Koryto et al., 2018). Cizek (2014) additionally employed a multiplexing ultraviolet visual spectrometer to record high frequency nitrogen species concentrations via light absorbance at 15-min intervals.

4.4. Study Results & Impact of Design Elements

4.4.1. Hydrologic Treatment

Hydrologic reporting is generally inconsistent among RSC studies (Table 4-8). Browning (2008) reported sampling on 30 bimonthly occasions; 4 were during rain or snow melt events, the remaining were during base flow (Table 4-7). The Durham Co. RSC had 81 discrete rainfall events producing 114% of normal annual precipitation; several events were discarded for snow melt and exceeding limits of calibration, resulting in 75 events analyzed for hydrology (Koryto et al., 2017) (Table 4-8). The Anne Arundel Co.-C RSC required 16.3 mm of precipitation to produce surface outflow, while the unrestored control reach produced runoff after 7.9 mm; 47 hydrologic events were observed during the monitoring period (Palmer et al., 2014). Ninety-three hydrology events were observed at the Wake Co.-C RSC with total accumulation of 972 mm; however, measurement error and necessary flow calibrations reduced the number of events analyzed to 15 (Table 4-8). The Alamance Co. RSC had 43 inflow producing events with the maximum 81-mm event producing a maximum 5-min peak intensity of 74-mm/hr (Cizek et al., 2016). At the Durham Co. RSC, the largest 24-hr accumulation, 93.7-mm, produced a maximum peak 5-min intensity of 138 mm/hr; both 24-hr accumulation and peak intensity were 2-year Average Recurrence Interval (ARI) events (Koryto et al., 2017).

Table 4-8. Hydrologic Event Comparison

Location & Source	*Observed Rainfall (mm)*	*Analyzed Events*	*Event Range (mm)*	*Median Event (mm)*
Durham Co., NC (Koryto et al., 2017)	1463	75	2.5-93.7	11.8
Wake Co.-A, NC (Koryto et al., 2018)	378*	12*	25-38*	31.5*
Brunswick Co., NC (Cizek et al., 2017)	665	27**	5.8-74.3**	23**
Alamance Co., NC (Cizek et al., 2016)	978	43**	NA-81**	NA
Wake Co.-B, NC (Cizek, 2014)	1161*	9*	87-171*	129*
Wake Co.-C, NC (Chapter 3)	287	15	6.4-38.6	21.3
Anne Arundel Co.-A, MD (Brown et al., 2010)	NA	3***	127-218****	NA
Anne Arundel Co.-C, MD (Palmer et al., 2014)	NA	47	7.9-172**	NA
Anna Arundel Co.-D, MD (Browning, 2008)	NA	30	NA	NA

* Simulated Events
** Inflow producing events
*** reported TR-20 models
**** (NOAA National Weather Service, n.d.) Anne Arundel County

Plot-scale RSC storm simulations, Wake Co-A and -B, were studied under controlled hydrologic conditions, aside from direct precipitation reported by Koryto et al. (2018), analyzed RSC events were simulated inflow hydrographs (Cizek, 2014; Koryto et al., 2018). Koryto et al. (2018) simulated 25- and 38-mm events while varying the Antecedent Dry Period (ADP) in a 2x2 factorial cross with 3 replicates per scenario resulting in 12 simulations (Table 4-8). These simulations represent 80th and 90th percentile events; it was postulated that hydrologic performance would be greater for events smaller than those simulated (Koryto et al., 2018). Cizek (2014) simulated 2-, 10-, and 50-year, 24-hour ARI storms from Raleigh, NC, corresponding to storm depths of 87-, 129-, and 171-mm, respectively (NOAA National Weather Service, n.d.). Nine event simulations were completed, 3 replicates for each design storm; equipment malfunctions resulted in a loss of some hydrologic data. Brown et al. (2010) reported model results for 3 events, 10-, 25-, and 100-year, 24-hour for Anne Arundel Co., MD storms corresponding to 127-, 158-, and 218-mm, respectively (NOAA National Weather Service, n.d.). Hydrologic treatment reporting varied with some studies reporting median treatment, results for outflow producing events, and ranges of simulations or modeled events (Table 4-9).

Table 4-9. Reported Hydrologic Treatment

Location & Source	*Volume Change (%)*	*Peak Flow Reduction (%)*
Durham Co., NC (Koryto et al., 2017)	+10	21
Wake Co.-A, NC (Koryto et al., 2018)	37-87*	40-93*
Brunswick Co., NC (Cizek et al., 2017)	94-100**	90-96**
Alamance Co., NC (Cizek et al., 2016)	78***	76***
Wake Co.-C, NC (Chapter 3)	0	NA
Anne Arundel Co.-A, MD (Brown et al., 2010)	NA	75****

+ Indicates volume increase
* Range of event simulations
** Outflow producing events; 74-665 mm
*** Median event results, surface flow
**** 25-yr, 24-hr model results

4.4.1.1. Surface-to-Seepage Flow Conversion

The conversion of surface flow to subsurface seepage appears to be an important function governing an RSC’s ability to treat pollutants. The Durham Co. RSC converted 50% of influent to seepage in the upper 2 well-drained pools (Table 4-10); a steep slope and a high storage ratio (Table 4-3) likely improved seepage conversion by creating high hydraulic gradients and deep (approximately 1-m) ponding depths; however, groundwater intrusion and hillslope dewatering (observed with transect wells) reduced media capacity by 40% causing seepage to reemerge as surface flow before the subsequent, C4, monitoring station (Koryto et al., 2017). Koryto et al.

(2017) reported a 10% increase in surface outflow over the monitoring period (Table 4-9), due to run-on (4.5%) and groundwater intrusion (5.5%). Twenty-four-hour outflow volumes and media drawdown rates varied significantly across seasons due to groundwater impacts or clogging over time (Koryto et al., 2017). The Alamance Co. RSC's pools 1-3 converted 71% of influent to seepage (Table 4-10); however, some seepage was reported to reemerge as surface flow downslope (Cizek et al., 2016). The Alamance Co. RSC exfiltration trench retained water in the media and promoted wetland conditions, this diminished seepage capacity (Cizek et al., 2016). The Brunswick Co. RSC had 27 inflow events but only 2 had surface outflow, Tropical Storm Andrea and a cloud burst with short (< 24-hour) ADP (Cizek et al., 2017); for outflow-producing events surface-to-seepage conversion was 94-100% (Table 4-9)(Cizek et al., 2017). The Wake Co.-C RSC was constructed with an exfiltration trench; baseflow maintained media saturation and surface ponding, which eliminated potential to mitigate hydrologic flows (Chapter 3).

Table 4-10. Well-Drained Surface-to-Seepage Conversion Rates

Location & Source	*Surface-to-Seepage Conversion (%)*
Durham Co., NC (Koryto et al., 2017)	50
Alamance Co., NC (Cizek et al., 2016)	71*
Brunswick Co., NC (Cizek et al., 2017)	94-100**

* Pools 1-3

** Outflow producing events; 74-665 mm

Table 4-11. RSC Design for Surface-to-Seepage Flow Conversion

Design Parameter	*Range*	*Guidance*	*Supporting Studies*
Loading Ratio (unitless)	80-120:1	RSC can provide effective treatment at 80-120:1; exact loading rate will depend on contributing, SCM areas and design volume.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Storage Ratio (unitless)	0.5-1.5:1	Lower storage ratios are more effective at surface-to-seepage conversion; however, surface storage is required to support sedimentation and prevent particle resuspension.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018) Cizek et al. (2017)
Media	Sand: >80% Mulch: $^{d60}/_{d10} < 15$	Higher fraction of sand will improve infiltration rates; homogeneous in-media mulch will prevent dispersion and clogging. Surface mulch may preserve infiltration rate; media storage volume base on media at field capacity	Hunt et al. (2006) Hsieh & Davis (2005a) Koryto et al. (2017) Cizek et al. (2016)
In-Stream Structures	Site Dependent	Larger in-stream geomorphic structures, riffles, steps, cascades, may improve hyporheic exchange	Hester & Doyle (2008)
Slope (%)	1-10	Dependent on site conditions; steeper slopes will require larger grade control structures and produce stronger hydraulic gradients.	Flores et al. (2012) Koryto et al. (2017) Cizek et al. (2016)
Exfiltration Trench	In Well-Drained Conditions	Can provide additional media volume to improve seepage capacity. Rake final 0.3-m of exfiltration to preserve subsoil infiltration	Cizek et al. (2016) Koryto et al. (2017) Brown & Hunt (2009) Chapter 3
Seasonal High-Water Table	≥ 0.6 m Above SHWT	Media saturation will render additional media volume ineffective; do not include saturated media in storage volume calculations.	Koryto et al. (2017) Cizek et al. (2016) Chapter 3
Forebay	Yes	Dedicate pool 1 to "Forebay/Pretreatment;" follow existing forebay design guidance. Benefits seepage conversion by preserving subsequent pools for seepage.	Cizek et al. (2016) Koryto et al. (2017) (NC DEQ, 2017)

4.4.1.1.1. Design Storm, Loading & Storage Ratios

The Durham Co. RSC was designed with 25% of the required capture volume and constrained to the existing gully, producing a large storage ratio, 3.34:1; the Alamance Co. RSC was designed with 72% the required capture volume and minimal site constraints, yielding a storage ratio of 0.42:1 (Table 4-3). The Durham Co. RSC's ability to convert surface flow to seepage was limited by severe under-sizing and a relatively high storage ratio; flow conversion was only evident during smaller events (< 12.7 mm) (Koryto et al., 2017). Larger events overwhelmed seepage capacity, producing overflow and surface discharge. The Alamance Co.

RSC was slightly under-sized vis-à-vis the design storm but did have a favorable storage ratio which effectively converted surface flow to seepage (Table 4-10 and Table 4-11). The Brunswick Co. RSC experienced surface clogging but converted surface runoff to seepage flow reliably (Cizek et al., 2017). Surface outflow only occurred during 2 of 27 inflow events (Cizek et al., 2017). The Brunswick Co. RSC had a large loading rate, 310:1 (Table 4-3), and was designed to capture 83% of the required volume (Cizek et al., 2017). The Wake Co.-C RSC was designed with only 3.2% the required capture, an extremely large loading rate (25,000:1), and upstream flow control; however, continuous base flow maintained media saturation and surface flow; potential storage was always "full."

Koryto et al. (2018) determined a significant location-by-ADP interaction representing the impact of soil moisture on surface outflow volumes; 3-day ADP retained an estimated 4.7 m^3 more volume than a 1-day ADP. Once soil storage capacity was exceeded, nearly all inflow remained on the surface (Koryto et al., 2018). Overflow or bypass in BRCs is linked to the hydraulic conductivity of underlying soil, depth to Internal Water Storage (IWS), and the surface infiltration rate (Brown & Hunt, 2011; Hunt et al., 2006); these factors also impact RSC surface discharge. The plot-scale Wake Co.-A RSC had relatively low loading and storage ratios, 72:1 and 0.87:1, respectively (Table 4-3) (Koryto et al., 2018). The Koryto et al. (2018)-simulated watershed was predicated on a 100%-sized RSC able to completely capture the 25-mm design storm; however, once inflow volumes exceeded media at field capacity storage (i.e., 36% available storage), subsequent inflow stayed on the surface (Koryto et al., 2018).

Effective (> 70%) RSC surface-to-seepage conversion is evident when loaded at 80-120:1 without additional upstream flow controls (Table 4-11) (Cizek et al., 2016; Koryto et al., 2018, 2017); when constructed downstream of hydrologic controls that mitigate flow rates, effective treatment may be capable at much higher loading rates (Chapter 3). Loading rates should be adjusted to accommodate the appropriate design storm. Lower storage ratios provide larger media volumes to support surface-to-seepage conversion; when storage ratios and design storms are too small, pool energy dissipation will be ineffective and RSC scour and degradation may occur (Chapter 3). Storage ratios of 0.5-1.5:1 provide good flow conversion while also maintaining enough surface storage to support sedimentation and prevent bed scour (Table 4-11) (Chapter 3; Cizek et al., 2016, 2017, Koryto et al., 2018, 2017).

4.4.1.1.2. Media & Mulch

Media composition and antecedent moisture conditions will impact an RSCs' ability to convert surface flows to seepage. The Alamance Co. RSC utilized a sand bed with surface mulching (Table 4-3); whereas, the Durham Co. RSC utilized the Flores et al. (2012)-recommended RSC media (Table 4-3, Table 4-4). Durham Co. RSC media K_{sat} values ranged from 1-55 cm/hr with a median 18 cm/hr. However, in-media mulch may have contributed to clogging; mulch with a high d60/d10 ratio (> 15.4) can disperse and clog media pore space (Hsieh & Davis, 2005a). Surface mulching and sand media at the Alamance Co. RSC may have prevented surface clogging and maintained higher K_{sat} values compared to the Durham Co. RSC, improving surface-to-seepage flow conversion (Hsieh & Davis, 2005a). The Brunswick Co. RSC was constructed within an existing gulley and filled with sand media (Table 4-3). The media bed was separated from Class A rip-rap with a geotextile fabric; rip-rap was covered with 50 mm of composted hardwood mulch and seeded with warm season grasses (Cizek et al., 2017). Higher fractions of sand (> 80%) provided rapid infiltration while homogeneous in-media mulch (d60/d10 ratio < 15) likely prevented dispersion and clogging (Table 4-11) (Cizek et al., 2016; Hsieh & Davis, 2005a; Hunt et al., 2006; Koryto et al., 2017). Media storage volumes should be calculated based on media at approximately field capacity (e.g., 36% available storage as in Koryto et al., 2018); if media saturation is expected, then media is not a part of design storage (Cizek et al., 2016; Koryto et al., 2018).

4.4.1.1.3. In-Stream Structures

Hyporheic exchange rates are influenced by the size and type of in-stream structures (Hester & Doyle, 2008); improved downwelling rates will increase surface-to-seepage conversion. Hyporheic exchange requires hydrologic neutral or gaining conditions within a stream reach (Hester & Doyle, 2008); otherwise, the reach loses surface flow to seepage and exfiltration (Cizek et al., 2016; Koryto et al., 2017). At the Alamance, Durham, and Brunswick Co. RSCs, pools not influenced by groundwater converted substantial portions of surface flow to media seepage (Cizek et al., 2016, 2017; Koryto et al., 2017). Hester & Doyle (2008) reported an exponential decrease in hyporheic exchange rates for a given weir height as slope increased. In-stream structure size, background groundwater discharge rate, hydraulic conductivity, and depth to bedrock were determined to be important factors when considering hyporheic exchange metrics (Hester & Doyle, 2008). The influence of hydraulic head and groundwater discharge, or media pore space availability, will contribute to surface—seepage flow exchange; depth to less permeable subsoils underlying RSCs is likely analogous to effects as bedrock depth. The pre-cast concrete weirs at the Brunswick Co. RSC, with footers in the media layer, may have produced deeper downwelling depths, versus typical riffle-weir grade controls. To counter the saturated conditions of the Durham Co. RSC, larger grade control structures, boulder-weirs or cascades, might have improved downwelling depths and residence times; the Alamance Co. RSC's cascade into the wetland pool likely improved surface-to-seepage conversion (Table 4-10). The Wake Co.-C RSC employed relatively large boulder-weir grade control structures, which may have improved hyporheic exchange under saturated media conditions. Type, size, and number of in-stream structures will be site dependent on engineering judgment (Table 4-11); designers must balance available material, slope, area, energy dissipation, storage creation, presence of saturation, etc., when selecting in-stream structures.

4.4.1.1.4. Slope

Any slope within the examined range (1.2% to 10%) can effectively convert surface flow to seepage, but how much conversion is dependent upon other factors. The Wake Co.-A RSC was constructed at a 1.2% slope, and contained 4 pools; surface outflow did not occur until the media capacity was exceeded (Koryto et al., 2018). The Alamance Co. RSC was constructed at a 3.9% slope with modest ponding depths (approximately 0.5-m) in pools 1-3 and had excellent at surface-to-seepage conversion (71%) (Table 4-3, Table 4-10) (Cizek et al., 2016). The Durham Co. RSC had the steepest slope, 10%, and converted substantial amounts of surface flow to seepage when saturated conditions were not present (Table 4-3, Table 4-10). Steep slopes contribute to higher hydraulic gradients which force surface flow into subsurface pore space (Brown et al., 2010). RSC slopes are dependent on site conditions; however, the size, type, and number of structures/ cells are part of engineering design. Steep slopes will require larger structures and produce higher hydraulic gradients, improving surface-to-seepage conversion (Table 4-11) (Cizek et al., 2016; Flores et al., 2012; Koryto et al., 2017).

4.4.1.1.5. Exfiltration Trench

Three of the tested RSCs included exfiltration trenches, with mixed results. The success of an exfiltration appears entirely dependent on one factor: can it remain unsaturated inter-event? The Alamance and Brunswick Co. RSCs converted surface flow-to-seepage flow in areas of well-drained conditions (Cizek et al., 2016, 2017); at the Alamance Co RSC, pool 4 promoted wetland conditions due to prolonged media saturation in the exfiltration trench (Cizek et al., 2016). Media saturation resulted in a 36% increase in surface flow from pool 4 (Cizek et al., 2016). At the Wake Co.-C RSC base flow maintained continuous saturation, eliminating media storage capacity, despite the presence of an infiltration trench. Media saturation from base flow or groundwater renders exfiltration trenches ineffective. Therefore, when designing for hydrologic mitigation, well-drained conditions are necessary. The Seasonable High Water Table (SHWT) should be avoided when installing infiltration trenches (Table 4-11) (Chapter 3; Cizek et al., 2016; Koryto et al., 2018). Moreover, when excavating during RSC construction, the infiltration capacity of underlying soils should be preserved, especially those underneath exfiltration trenches. It is recommended to follow methods described by Brown & Hunt (2009).

4.4.1.1.6. Forebay

Researchers have observed that the RSC's first pool acts as a defacto forebay, capturing a substantial portion of sediment (Cizek et al., 2016; Filoso, 2012; Koryto et al., 2017). Reduced infiltration rates at the Durham Co. RSC were likely caused by sedimentation in pool 1 (Koryto et al., 2017). Palmer et al. (2014) was forced to move inflow monitoring locations due to in-channel aggradation. Designers should consider dedicating the first pool of an RSC to be a forebay, to optimize sedimentation and system maintenance (Table 4-11)(NC DEQ, 2017).

4.4.1.2. RSC Hydrologic Design Optimization

Table 4-12. RSC Design for Mitigation of Peak Flow and Discharge Volumes

Design Parameter	*Range*	*Guidance*	*Supporting Studies*
Design Storm (mm)	25-38	RSC storage volume is the sum of surface and media storage; media storage based on media at field capacity.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Loading Ratio (unitless)	80-120:1	RSC can provide effective treatment at 80-120:1; exact loading rate will depend on contributing watershed, SCM areas and design volume.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Storage Ratio (unitless)	0.5-1.5:1	Effective detention of surface flows in-media will improve peak flow reductions. Water in the media zone is nearest to subsoils and exfiltration losses.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018) Cizek et al. (2017)
In-Stream Structures	Site Dependent	Structures that improve surface-to-seepage conversion will improve peak flow reductions.	Hester & Doyle (2008) Koryto et al. (2017) Cizek et al. (2016) Cizek et al. (2017)
Exfiltration Trench	When Applicable	Can provide additional media volume to support appropriate storage ratio. Rake final 0.3-m of exfiltration to preserve subsoil infiltration.	Cizek et al. (2016) Koryto et al. (2017) Brown & Hunt (2009) Chapter 3
Seasonal High-Water Table	≥ 0.6 m Above SHWT	Media saturation will render exfiltration trench ineffective; do not include saturated media in storage volume calculations.	Koryto et al. (2017) Cizek et al. (2016) Chapter 3
Vegetation	Yes, zonal planting areas	Vegetation may improve volume reductions by creating macro pores and increasing ET.	Li et al., (2009) Wardynski & Hunt, (2012) Hatt et al. (2009)

4.4.1.2.1. Design Storm & Loading Ratio

The loading ratio reflects the ability of the RSC to capture the design storm; if space is available, RSC designers can capture most, if not all, of the design storm (Koryto et al., 2018). Koryto et al.'s (2018) Wake Co.-A RSC was designed to capture a 25-mm water quality storm from a simulated watershed loaded at 72:1. Simulations determined both volume and peak flow reductions were dependent on storm-ADP scenario; 1-day ADP and a 38-mm event had volume and peak flow reductions of 37% and 40%, respectively, while a 25-mm event with a 3-day ADP had its volume and peak flow reduced by 87% and 93%, respectively (Table 4-9)(Koryto et al., 2018).

The Brunswick Co. RSC had a high loading ratio, 310:1, yet was sized to capture 83% of the WQV. Peak flow reductions ranged from 90%-96% (Cizek et al., 2017). HSG A underlying soils improved exfiltration rates allowing 94-100% of runoff volumes to be eliminated from surface flow (Table 4-9) (Cizek et al., 2017).

The Alamance Co. RSC yielded median influent volume and peak flow reductions of 78% and 76%, respectively (Table 4-9) (Cizek et al., 2016). Pools 1-3 of the Alamance Co. RSC were well-drained; cumulative volume and peak flow reductions increased until the cascade following pool 3 where saturated media in the exfiltration trench limited additional storage. The Alamance Co. RSC was sized to capture 72% of the required volume with a loading ratio of 100:1, well-drained upper pools and a favorable storage ratio contributed to good surface-to-seepage conversion but HSG D underlying soils limited exfiltration potential.

Hydrologic mitigation at the Durham Co. RSC, 8% and 49% for volume and peak flow reduction, respectively, were achieved for small events (<12.7-mm) (Koryto et al., 2017); larger events overwhelmed media storage capacity, yielding large overflow rates and volumes. Substantial under-sizing, 25% the required design storm, HSG D underlying soils (Table 4-2), and groundwater intrusion diminished the Durham Co. RSC's ability to mitigate larger storms.

Effluent measurement error at the Wake Co.-C RSC prevented peak flow reduction analysis; however, estimations of exfiltration and ET determined 0% volume reduction (Chapter 3). The Wake Co.-C RSC had HSG D underlying soils, only 3.2% the required capture volume, a 25,000:1 loading ratio, and constant media saturation. While the Wake Co.-C RSC was constructed in-series with a dry pond these design characteristics, inhibited volume reductions (Chapter 3). In short, the Wake Co.-C RSC was too small.

In previous studies on BRCs, overflow and bypass were linked to the hydraulic conductivity of underlying soil, depth to IWS, and surface infiltration rate (Brown & Hunt, 2011; Hunt et al., 2006). These factors contribute similarly to RSC outflow production, once media storage and surface ponding are exceeded, flow will bypass media treatment. Thus, adjusting the loading ratio to reflect the design storm to a sufficient size to produce surface and media volumes is an important design element. Koryto et al.'s (2018) study determined precisely this, significant main effects for location, storm depth, ADP, and location-by-ADP interaction illustrated the impact of soil moisture on surface outflow volumes and peak flow reductions; the longer 3-day ADP allowed the RSC to retained an estimated 4.7-m^3 more volume than the 1-day ADP (Koryto et al., 2018). High influent volumes resulted in saturation of subsurface pathways; once inflow volume exceeded media storage at approximately field capacity (i.e., 36% available storage), additional inflow volume yielded equivalent outflow volume (Koryto et al., 2018). Therefore, Koryto et al. (2018), suggest a more conservative media storage calculation based on field capacity, accounting of media pore water, versus total estimated porosity.

Runoff capture is necessary for volume reduction and peak flow mitigation to occur; RSCs are capable of effective volume and peak flow reductions when loading ratios are 80-120:1, provided it is not used in conjunction with other SCMs (Table 4-12) (Cizek et al., 2016; Koryto et al., 2018, 2017). When constructed downstream of other hydrologic controls or in parts of the coastal plains (where underlying soils are often sandy) effective treatment may be possible at much higher loading rates (Chapter 3). Lower storage ratios provide larger media volumes to support surface-to-seepage conversion; when storage ratios (and design storms) are too small, pool energy dissipation will be overwhelmed and thus ineffective. RSC scour and degradation may occur (Chapter 3). Storage ratios of 0.5-1.5:1 provide good flow conversion while also maintaining enough surface storage to support sedimentation and prevent bed scour (Table 4-12) (Cizek et al., 2016, 2017, Koryto et al., 2018, 2017). As stated previously, base flow or groundwater intrusion will cause media saturation, limit storage volume, and diminish hydrologic treatment; saturated media should not be included in storage calculations (Table 4-12) (Cizek et al., 2016; Chapter 3; Koryto et al., 2017).

4.4.1.2.2. In-Stream Structures

Delivery of surface flows to the media zone improves peak flow mitigation and increases the likelihood for exfiltration losses; in-stream structures that improve surface-to-seepage conversion will improve hydrologic treatment. The Brunswick Co. RSC produced 94-100% conversion of surface flow to subsurface seepage for events which yielded surface discharge (Table 4-9)(Cizek et al., 2017); surface outflow occurred during 2 of 27 inflow events (Cizek et al., 2017). Modest slopes (4%), deep ponding depths (approximately 1-m), and possibly pre-cast concrete weirs at the Brunswick Co. RSC, with footers in the media layer, may have contributed to deeper downwelling depths. A modeled predevelopment hydrograph suggests that no outflows (of 27 events) would have occurred during the monitoring period, the Brunswick Co. RSC achieved pre-development hydrology in 94% of events monitored (Cizek et al., 2017). Pools converted 84% of surface flow to seepage and 75% to exfiltration on an event basis; 6% of total runoff was converted to exfiltration while some seepage was reported to reemerge as surface flow downslope (Cizek et al., 2017). When surface outflow occurred, event basis volume reductions of 90-96% were observed (Cizek et al., 2017). Groundwater caused surface ponding and high-water levels in adjacent groundwater wells at the Brunswick Co. RSC (Cizek et al., 2017); geotextile clogging was also thought to contribute to extended ponding (Cizek et al., 2017).

The Wake Co.-A RSC was constructed with riffle-pool features at 1.2% slope; hydrologic simulation results for volume and peak flow reductions were good, 37-87% and 40-93%, respectively, depending on storm-ADP event simulation. At low slopes, riffle-pool channels are effective at delivering influent to the media zone.

The Durham Co. RSC was constructed on steep grade (10%) with typical boulder-cascade grade control structures; steep grade required large boulder-cascades which created deep ponding depths (approximately 1-m), large surface storage volume, and large storage ratio, 3.34:1. Upper pools in the Durham Co. RSC were well-drained producing significant volume and peak flow reductions of 50% and 40%, respectively; however, lower pools intersected the “Point of Intermittency” stream designation per U.S. Army Corps of Engineers (USACE) and NCDEQ (Koryto et al., 2017). The Durham Co. RSC experienced a 10% increase in surface flow (Table 4-9) due to run-on (4.5%) and groundwater intrusion (5.5%). Hillslope dewatering was observed (using transect wells) to reduced system storage capacity by 40% (Koryto et al., 2017). The impact of groundwater varied significantly across seasons determined by 24-hour outflow volume. Surface-to-seepage conversion and volume reduction was good in well-drained areas with large boulder-cascades; however, in saturated zones steps will likely convert surface flows to seepage more effectively (Hester & Doyle, 2008). Any benefits provided by in-stream structures can be undermined by groundwater intersection.

The Wake Co.-C RSC was constructed with relatively large boulder-weir structures creating an approximately 0.3-m step-pool channel. Under well-drained conditions effective surface-to-seepage conversion and modest volume reduction would be expected, provided a sufficient a design storm was used; however, under-sizing (3.2% of the WQV) and media saturation eliminated available storage (Chapter 3). Volume reduction did not occur, and measurement errors prevented peak flow analysis (Chapter 3).

Type, size, and number of in-stream structures will be site dependent and based on engineering judgment (Table 4-12); design decisions must balance available material, slope, area, energy dissipation, storage creation, etc. Designers should consider where saturation may occur and design to optimize surface-to-seepage conversion; subsurface seepage and the in-stream structure's ability to produce it appear to be an important factor governing RSC hydrologic treatment.

4.4.1.2.3. Exfiltration Trench

IWS in BRC can improve exfiltration rates and peak flow reductions (Brown & Hunt, 2011; Li et al., 2009); IWS's cognate in RSCs, exfiltration trenches, have the potential to provide the same benefits if designed and implemented properly. However, in the 2 RSCs with exfiltration trenches, the benefits of exfiltration trenches were muted.

Including an exfiltration trench at the Alamance Co. RSC yielded a 45% *increase* in surface outflow from pool 3 to the outlet of pool 4; however, the exfiltration trench did promote infiltration of 280-m^3 to the subsoils (Cizek et al., 2016). The Wake Co.-C RSC was constructed with an exfiltration trench to create a media bed (Chapter 3); volume reductions were negligible. Exfiltration trenches allow application of RSC methodology to areas where no or minimal gullying existed prior to installation (Chapter 3; Cizek et al., 2016, 2017). During SCM construction, the infiltration capacity of underlying soils should be preserved with methods recommended by Brown & Hunt (2009). If an exfiltration trench is to be included in an RSC for hydrologic mitigation, the SHWT must not intersect the exfiltration trench (Table 4-12) (Cizek et al., 2016; Koryto et al., 2018; Chapter 3). When considering the use of an exfiltration trench, the impact of media saturation (from base flow or groundwater intrusion) must be a factor.

4.4.1.2.4. Vegetation

Vegetation has been shown improve infiltration rates in BRC by creating macro pores around plant roots and improve volume reductions via ET (Brown et al., 2013; Li et al., 2009; Wadzuk et al., 2015; Wardynski & Hunt, 2012); however, few studies have documented established vegetation in RSCs (e.g., Browning, 2008; Cizek et al., 2016, 2017; Filoso, 2012; Koryto et al., 2018, 2017), possibly due to early successional monitoring. Heavy canopy cover, steep side slopes, and prolonged saturation in the reviewed RSCs diminished vegetation establishment and eliminated ET as an important hydrologic treatment mechanism (Chapter 3; Cizek et al., 2016, 2017; Koryto et al., 2017). Flores et al. (2012) specifies dense and diverse native vegetation for use in appropriate wetness zones of SPSCs. For example, wetland planting recommendations for constructed stormwater wetlands apply a zonal planting method based on ponding depths inwhich different species survive (NC DEQ, 2017). Hatt et al. (2009) vigorous vegetation growth significantly increased infiltration capacity; indicating vegetation root zone helped maintain media porosity (Hatt et al., 2009). Therefore, vegetation with comparatively larger root growth may perform better at maintaining RSC infiltration capacity and media porosity. RSC designers should adapt existing zonal planting (i.e., shallow water, temperay inundation, and deep water zones) recommendations for BRCs and stormwater wetlands to RSCs (Table 4-12)(NC DEQ, 2017).

4.4.2. Water Quality Treatment

RSC water quality treatment is highly variable (Table 4-13); similar to hydrologic treatment, water quality treatment is highly dependent on the conversion of surface flows to subsurface treatment. Pollutant treatment mechanisms include (1) settling in surface storage (ponded water), (2) filtration and (3) sorption processes in the media layer, (4) assimilation by vegetation, and (5) microbiological treatment; the majority of which occur in the media zone (Cizek et al., 2016; Cizek et al., 2017; Davis et al., 2010; Flores et al., 2012; Koryto et al., 2018; Koryto et al., 2017).

Table 4-13. RSC Pollutant Reductions

Location & Source	*ELR TSS (%)*	*ELR TN (%)*	*ELR TP (%)*
Durham Co., NC (Koryto et al., 2017)	10	4.4	6.8
Wake Co.-A, NC (Koryto et al., 2018)	NA	20 – 46	33 – 68
Alamance Co., NC (Cizek et al., 2016)	92*	76*	77*
Wake Co.-C, NC Chapter 3	69**	3**	26**
Anne Arundel Co.-C, MD (Palmer et al., 2014)	70***	27***	NA
Anna Arundel Co.-D, MD (Browning, 2008)	28***	35***	NA

* Median load reduction: entire system
** Storm flow Removal Efficiency (RE) (Chapter 3)
*** Estimated annual load reduction

4.4.2.1. TSS and Particulate Matter

Table 4-14. RSC Design for TSS Sequestration

Design Parameter	*Range*	*Guidance*	*Supporting Studies*
Design Storm (mm)	25-38	RSC storage volume is the sum of surface and media storage; media storage based on media at field capacity.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Loading Ratio (unitless)	80-120:1	RSC can provide effective treatment at 80-120:1; exact loading rate will depend on contributing, SCM areas and design volume.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Storage Ratio (unitless)	0.5-1.5:1	Lower storage ratios are more effective at surface-to-seepage conversion; if pool volume is too small bed will scour and sedimentation will be ineffective.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018) Cizek et al. (2017)
Media	Sand: >80% Fines: 8-15% Mulch: $d60/d10 < 15$	Coarse sand provides rapid infiltration; fines and OM improve sorption sites. Homogeneous in-media mulch will prevent dispersion and clogging. Surface mulching may preserve infiltration rate.	Hunt et al. (2006) Hsieh & Davis (2005a) Koryto et al. (2017) Cizek et al. (2016)
In-Stream Structures	Site Dependent	Larger in-stream geomorphic structures, riffles, steps, cascades, may improve hyporheic exchange	Hester & Doyle (2008)
Forebay	Yes, when no upstream flow control.	Dedicate pool 1 to "Forebay/Pretreatment;" follow existing forebay design guidance. Benefits seepage conversion by preserving subsequent pools.	Cizek et al. (2016) Koryto et al. (2017) (NC DEQ, 2017)
Vegetation	Yes, zonal planting areas	Vegetation may improve volume reductions by creating macro pores and increasing ET.	Li et al., (2009) Wardynski & Hunt, (2012) Hatt et al. (2009)

4.4.2.1.1. Design Storm & Storage Ratio

Effective (> 70%) TSS removal via RSC treatment is possible with an appropriate storage volume and a moderately low storage ratio. TSS load reductions ranged from 10-92% (Table 4-13). The Alamance Co. RSC had 72% of the required capture volume and a low storage ratio, 0.42:1; adequate total storage capacity with sufficient seepage capacity throughout the well-drained pools contributed to substantial TSS load reductions. Seventy-one percent of influent became seepage on an event basis in the well-drained pools 1-3 (Table 4-10)(Cizek et al., 2016);

TSS load reduction by these pools was commensurate (95%) (Cizek et al., 2016). TSS load reductions for the entire Alamance Co. RSC was slightly lower (92%) (Table 4-13), possibly from particle resuspension at the cascade entering the wetland pools.

The Wake Co.-C RSC achieved 69% TSS Removal Efficiency (RE); upstream flow control mitigated peak inflow rates, however, minimal total storage (3.2% WQV) and surface ponding likely inhibited effective sedimentation. Low storage ratio (0.01:1) and large boulder-weirs (0.3-m step) likely benefited hyporheic exchange and surface-to-seepage conversion, improving TSS filtration (Chapter 3).

Koryto et al. (2017) determined TSS reductions primarily occurred in the first 2 pools of the Durham Co. RSC, where 50% of influent volume was converted to seepage (Table 4-10). Significant pollutant concentration reductions were observed; however, TSS RE and Event Load Reductions (ELR) were low, 17% and 10%, respectively (Table 4-13). Under sizing (25% of the required capture volume) limited total storage capacity and 3.34:1 storage ratio, (Table 4-3), limited seepage capacity. Media scour or particle resuspension could also have contributed to TSS loads (Koryto et al., 2017). Sedimentation was, thus, the primary treatment mechanisms and subsurface filtration provided minimal additional treatment (Koryto et al., 2017).

High surface-to-media storage ratios will likely settle particles effectively but will limit surface-to-seepage conversion; alternatively, low surface-to-media storage ratios should convert surface flow to seepage but are likely limited by low surface infiltration rates. Ensuring sufficient volume to capture the design storm in the surface and media zones with a moderately low storage ratio (0.5-1.5:1) should support effective sedimentation in pools and media filtration of TSS (Table 4-14).

4.4.2.1.2. Media

TSS retention greater than 70% is common in media-based SCMs (Davis et al., 2006; Hunt et al., 2006; Page et al., 2015); high fractions of sand benefit rapid infiltration, while other components (fines, OM, and mulch) improve pollutant retention. The Alamance Co. RSC used sand media with surface mulching (Table 4-3); this design achieved good surface-to-seepage conversion and retention of TSS (92%) (Table 4-13). Sand media benefited rapid infiltration and surface mulching, likely prevented surface clogging (Hsieh & Davis, 2005a).

Koryto et al. (2017) observed a median saturated hydraulic conductivity of 18-cm/hr; localized clogging was caused by mulch and/or sedimentation/surface filtration of sediment (Koryto et al., 2017). Mulch with a high d60/d10 ratio (> 15.4) will disperse and clog media, reducing infiltration rates (Hsieh & Davis, 2005a). BRC media column studies with infiltration rates of 21-cm/hr during 6-hr constant head laboratory studies achieved moderate to excellent removal of TSS (Hsieh & Davis, 2005b). The Durham Co. RSC filtration was likely diminished by under-sizing and slow infiltration rates which produced excessive bypass and surface discharge. Media infiltration and saturated hydraulic conductivity rates can be adjusted with an increase or decrease in fines (silt and clay) (Hsieh & Davis, 2005a). The Wake Co.-C RSC used sand media mixed at an unknown ratio with an unknown mulch (Table 4-3); Loss on Ignition (LOI) analysis of bed substrate (i.e., a heterogeneous mix of sand, gravel, and OM) determined the mix contained less than 0.01% OM (Chapter 3). The Wake Co.-C RSC employed a coarse bed substrate (approximately 0.6-m) on top of the media bed which may have improved TSS capture (Chapter 3). The Ann Arundel Co.-C and -D RSCs employed unknown media and achieved estimated TSS reductions of 70% and 28%, respectively (Browning, 2008; Filoso, 2012).

Media with a high portion of sand will infiltration runoff rapidly; an increase or decrease in fines (silt and clay) allow media infiltration and saturated hydraulic conductivity rates to be adjusted (Hsieh & Davis, 2005a). Homogeneous in-media mulch (d60/d10 ratio < 15) will prevent dispersion and clogging; surface mulching may preserve surface infiltration rates (Table 4-14) (Cizek et al., 2016; Hsieh & Davis, 2005a; Hunt et al., 2006; Koryto et al., 2017).

4.4.2.1.3. In-Stream Structures

In-stream geomorphic structures will influence hyporheic exchange depending on structure size, slope, background groundwater discharge rates, and hydraulic conductivity (Hester & Doyle, 2008); increased interaction between surface flow and seepage will enhance surface filtration. The hydraulic gradient between riffles and pools will drive water into the media layer (Brown et al., 2010); however, once media pore space becomes saturated downwelling rates, residence times, and downwelling depths will likely behave similar to hyporheic exchange, versus sand filtration. The Durham Co. RSC used large boulder-cascades which created deep (approximately 1-m) ponding depths beneficial to settling; however, the large boulder-cascades created a steep grade (8.6% across pool 1) which contributed to media scour and particulate resuspension downslope (Koryto et al., 2017). The media bed was exposed downslope of the large cascade into pool 1 at the Durham Co. RSC; hardened bed material following forebay recommendations (Figure 4-10), cobble or otherwise, downslope of large, steep cascades would prevent scour and TSS resuspension.

The Alamance Co. RSC was constructed at a modest 4% slope (Table 4-3) with a series of 3 well-drained riffle-pool sequences, a cascade, and a series of 3 wetland riffle-pool sequences (Cizek et al., 2016). Modest ponding (approximately 0.5-m) depths and well-drained conditions provided effective TSS treatment; additional TSS removal occurred in the wetland pools. The

Wake Co.-C RSC achieved 69% TSS RE; upstream flow control mitigated inflow rates, however, minimal total storage (3.2% the WQV) and surface ponding (approximately 0.15-m) likely inhibited sedimentation. Low storage ratio (0.01:1) and large boulder-weirs (0.3-m steps) likely benefited hyporheic exchange and surface-to-seepage conversion improving TSS filtration (Chapter 3). Larger in-stream structures may enhance hyporheic exchange and surface-to-seepage conversion in saturated conditions, improving TSS filtration (provided downslope areas are protected from scour and degradation); structure size, type, and number will depend on engineering judgement and site constraints (Table 4-14).

4.4.2.1.4. Forebay

Most studies observe substantial TSS removal in the first pool (Cizek et al., 2016; Filoso, 2012; Koryto et al., 2017). Effective sedimentation requires reduction of flow velocities sufficient that particles settle via gravity (Stokes' Law) to the pool bottom (Figure 4-10); large, dense particles are readily settled compared to smaller, low density particles (Davis et al., 2010). Sufficient pool storage is necessary to achieve conditions suitable for sedimentation and to prevent pollutant resuspension (Davis et al., 2010; Miguntanna et al., 2013). RSC installation provides headcut stabilization which reduces TSS loads compared to untreated conditions (Koryto et al., 2017); the only study to quantify this is Palmer et al. (2014) Anne Arundel Co.-C RSC which observed a 70% reduction of TSS compared to an adjacent untreated gulley of similar condition (Table 4-13).

At the Alamance Co. RSC 71% of influent became seepage on an event basis in the well-drained pools 1-3 (Table 4-10)(Cizek et al., 2016); TSS load reduction by these pools was commensurate (95%)(Cizek et al., 2016), suggesting pools 1-3 functioned as a forebay. Cizek et al. (2016) determined 35% of the total TSS mass reduction was removed in pool 1 at the

Alamance Co. RSC, the remaining 70% was deposited in pools 2-3. Forebay design considerations in pool 1 at the Alamance Co. RSC would prevent sediment impacts on subsequent pools.

The Wake Co-C RSC achieved a TSS RE of 69% (Table 4-13) (Chapter 3); upstream flow control mitigated peak inflow rates and a low storage ratio (0.01:1) likely contributed to effective filtration. Therefore, minimal surface ponding and likely ineffective settling did not substantially negatively impact TSS removal. When RSCs are in-series with hydrologic control devices, forebay design considerations may be less important to cumulative TSS treatment (Table 4-14) (Chapter 3).

Koryto et al. (2017) determined TSS reductions primarily occurred in the first two pools of the Durham Co. RSC where 50% influent volume was converted to seepage (Table 4-10). Koryto et al. (2017) identified sedimentation as the primary TSS treatment mechanism with subsurface filtration providing minimal additional treatment (Koryto et al., 2017). The Durham Co. RSC had low TSS REs and ELRs, 17% and 10%, respectively (Table 4-13). A 1.6% volume increase for water quality events at the Durham Co. RSC, prevented mass reductions from equaling or exceeding EMC reductions.

Pool 1 at the Durham Co. RSC had twice the ponding depth and surface storage volume, but the large boulder-cascade produced a steep grade (8.6%) across pool 1 may have contributed to media scour and TSS resuspension (Table 4-15) (Davis et al., 2010; Koryto et al., 2017; Miguntanna et al., 2013). The Anne Arundel Co.-D SWC observed TSS concentrations increasing over the monitoring period, possibly from media washout, producing a lower estimated annual TSS load reduction (28%) when compared to other studies (Table 4-13) (Browning, 2008). This loss could have been mitigated by incorporating a forebay. SCM forebay design recommendations suggest a hardened material for forebay bottoms if it fits within the design aesthetic (NC DEQ, 2017); neither the Durham or Alamance Co. RSCs employed gravel, cobble, or rip-rap bed substrate in pool bottoms present in other RSC designs (Chapter 3; Cizek et al., 2017; Koryto et al., 2017), but hardened material incorporation in pool 1 may prevent scour and pollutant resuspension.

Table 4-15. Pool 1 Comparison Chapter 3, Cizek et al. (2016), and Koryto et al. (2017)

Location & Source	*Pool Depth (m)*	*Surface Storage (m^3)*	*Pool Slope (%)*
Durham Co., NC (Koryto et al., 2017)	1.3	12.8	8.6
Alamance Co., NC (Cizek et al., 2016)	0.5	3.4	1.3
Wake Co.-C, NC Chapter 3	0.13	0.89	2.0

In the absences of upstream flow controls, pool 1 should be dedicated to pretreatment; effective sedimentation in pool 1 will minimize sediment impact on the remaining practice. Proper flow stilling will minimize scour and pollutant resuspension (Davis et al., 2010; Miguntanna et al., 2013); hardened bed substrate may be necessary with steep grades (Koryto et al., 2017). Designers should consider dedicating the first pool of an RSC to be a forebay, to optimize sedimentation and system maintenance (Table 4-14) (Cizek et al., 2017; Koryto et al., 2017; NC DEQ, 2017).

4.4.2.1.5. Vegetation

Vegetation benefits pollutant treatment via increased surface roughness that increases retention time, thus improving particulate settling (Davis et al., 2010; Flores et al., 2012; Hsieh & Davis, 2005a); however, new planting recommendations are needed for species which can persist in RSCs. Vegetation establishment was attempted via hydro-seeding with stabilization grasses and surface application of composted hardwood mulch (Cizek, 2014; Cizek et al., 2016, 2017, Koryto et al., 2018, 2017). This method of mulching and hydro-seeding along with compost-amending of disturbed surfaces is recommended for temporary soil stabilization (Flores et al., 2012); however, these species may not be adapted for conditions in RSCs (e.g., high flow velocities, prolonged media saturation, well-drained, temporary inundation) (Browning, 2008; Cizek et al., 2016, 2017; Filoso, 2012; Koryto et al., 2018, 2017).

Hatt et al. (2009) found vigorous vegetation growth significantly increased infiltrtion capacity; indicating that the vegetation root zone helped maintain media porosity (Hatt et al., 2009). Therefore, vegetation with comparatively larger root growth may better maintain RSC infiltration capacity and media porosity; larger rooting zones may also benefit a plant's ability to maintain rooted during high velocity flows.

Portions of the Durham, Alamance, Brunswick Co. RSCs and the entire Wake Co.-C RSC experienced saturated conditions, similar to stormwater wetlands (Cizek et al., 2016, 2017; Koryto et al., 2017, Chapter 3); however, well-drained conditions were also present in most RSCs. Designers must select plants appropriate for each cell, not just for an RSC. Flores et al. (2012) specifies the use of native vegetation in RSCs; however, in BRC some cultivars outperform native species' at concentration and load reductions (Bratieres et al., 2007; Read et al., 2008; Turk et al., 2017). RSC designers should adapt existing zonal planting (i.e., shallow water, temperay inundation, and deep water zones) recommendations for BRCs and stormwater wetlands to RSCs (NC DEQ, 2017); however, deep water plantings may not be practical if well-drained conditions or high velocities can occur (Table 4-14).

4.4.2.2. Phosphorous

Table 4-16. RSC Design for Phosphorous Removal

Design Parameter	*Range*	*Guidance*	*Supporting Studies*
Design Storm (mm)	25-38	RSC storage volume is the sum of surface and media storage; media storage based on media at field capacity.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Loading Ratio (unitless)	80-120:1	RSC can provide effective treatment at 80-120:1; exact loading rate will depend on contributing, SCM areas and design volume.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Storage Ratio (unitless)	0.5-1.5:1	Lower storage ratios are more effective at surface-to-seepage conversion; if pool volume is too small bed will scour and sedimentation will be ineffective.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018) Cizek et al. (2017)
Media	Sand: >80% Fines: 8-15% Mulch: $d60/_{d10} < 15$	Fines and organic matter will improve pollutant sorption; homogeneous in-media mulch will prevent dispersion and clogging. Surface mulch may preserve infiltration rate and improve pollutant sorption; P-index ≤ 30 in NSW* and P-index ≤ 50 elsewhere.	Hunt et al. (2006) Hsieh & Davis (2005a) Koryto et al. (2017) Cizek et al. (2016)
Exfiltration Trench	In Well-Drained Soils	Can provide additional media volume to support appropriate storage ratio. Rake final 0.3-m of exfiltration to preserve subsoil infiltration.	Cizek et al. (2016) Koryto et al. (2017) Brown & Hunt (2009) Chapter 3
Seasonal High-Water Table	≥ 0.6 m Above SHWT	Media saturation will render exfiltration trench ineffective; do not include saturated media in storage volume calculations.	Koryto et al. (2017) Cizek et al. (2016) Chapter 3
Vegetation	Yes, zonal planting areas	Vegetation may improve volume reductions by creating macro pores and increasing ET.	Li et al., (2009) Wardynski & Hunt, (2012) Hatt et al. (2009)

*Nutrient Sensitive Waters

4.4.2.2.1. Design Storm, Loading & Storage Ratios

Adequate capture volume and appropriate loading with a moderately-low storage ratio will effectively (>50%) remove TP via settling and surface filtration. The Wake Co.-A RSC was designed and loaded appropriately (100% WQV, a 72:1 loading ratio, and 0.87:1 storage ratio, Table 4-3); Total Phosphorous (TP) ELRs ranged from 33%-68% for large storms with short ADPs and small storms with long ADPs, respectively. Subsurface TP removal was excellent, 83% (Koryto et al., 2018). Large events overwhelmed the media storage capacity, producing overflow and surface discharge (Koryto et al., 2018). The Wake Co.-C RSC was severely undersized (3.2% WQV), loaded at 25,000:1 with upstream flow control, and a low (0.01:1) storage ratio; it produced a TP RE of 26%.

Modest design storm (72% WQV), loading (100:1) and storage (0.72:1) ratios (Table 4-3) allowed the Alamance Co. RSC to detain runoff, promote sedimentation, and convert surface flows to seepage, improving filtration of particulate-bound phosphorous (PBP) (Davis et al., 2010; Miguntanna et al., 2013). The Alamance Co. RSC produced a significant 28% reduction in TP concentration (Cizek et al., 2016).

The Durham Co. RSC's large loading ratio (160:1) and undersized (25% WQV) design storm (Table 4-3), diminished TSS and PBP treatment; however, large storage ratio (3.34:1) improved sedimentation. Koryto et al. (2017) saw a significant ELR of 6.8% for TP (Table 4-13) at the Durham Co. RSC; however, RE was low, 17%, despite significant reductions in pollutant concentrations.

TP treatment is predicated on capture and treatment of the design storm, adjusting the loading ratio to accommodate the WQV will ensure appropriate storage volumes. A low storage ratio will improve surface-to-seepage conversion and filtration and sorption treatment processes (Table 4-14).

4.4.2.2.2. Media

TP concentrations reductions ranging from 70-85% are common in field monitoring of media based SCMs (Davis et al., 2006; Hunt et al., 2006; Page et al., 2015); the P-Index is a reliable predictor of medias' ability to bind or leach phosphorus (Hardy et al., 2009; Hunt et al., 2006). Subsurface Ortho-P and TP RE were 87% and 83%, respectively, at the Wake Co.-A RSC (Koryto et al., 2018). The P-Index for the media, sand, used in the Wake Co.-A RSC was 17, considered low and within the range for Nutrient Sensitive Waters (NSW) in NC (Table 4-4)(NC DEQ, 2017). TP effluent concentration targets for good ambient water quality in the NC Piedmont of 0.11-mg/L (McNett et al, 2010) were achieved 92% of the time for small rainfall events (<25-mm) where all runoff was converted to seepage (Koryto et al., 2018); low media P-Index, surface mulching, and some established vegetation likely improved TP removal (Hsieh & Davis, 2005a; Hunt et al., 2006). No other P-Indices were reported for media used in RSC field studies.

Durham Co. RSC influent TP effluent concentrations frequently (84%) exceeded Piedmont targets for good water quality 0.11-mg/L (Koryto et al., 2017; McNett et al., 2010). A 1.6% volume increase for water quality events at the Durham Co. RSC, prevented mass reductions from equaling or exceeding EMC reductions (Koryto et al., 2017); TP and Ortho-P were significantly reduced (ELRs of 6.8% and 40%, respectively) (Koryto et al., 2017). The Durham Co. RSC employed media composition recommended Flores et al. (2012) (Table 4-4).

The Wake Co.-C RSC produced a 26% RE for TP but Ortho-P concentrations significantly increased during storm flow, likely from a high media P-Index or continuous saturation and desorption (Chapter 3; Clark & Pitt, 2012; Hardy et al., 2009; Hunt et al., 2006). The Wake Co.-C RSC used sand media mixed at an unknown ratio of an unknown mulch (Table 4-3).

Ortho-P leaching was not observed at the Durham or Alamance Co. RSCs which also maintained saturation in portions of their media zones (Cizek et al., 2016; Koryto et al., 2017). At the Durham RSC a greater portion of the RSC was aerobic while at the Alamance RSC the media, sand, did not include in-media OM and likely had a similarly low P-Index to the Wake

Co.-A RSC. Under aerobic conditions, phosphorus sorbed to media should not be released (Clark & Pitt, 2012; Koryto et al., 2017).

Ortho-P is removed via adsorption to soil and OM particles; increasing the fraction of media fines (silts and clays) can increase Ortho-P binding sites (Davis et al., 2010; Hsieh & Davis, 2005a). Increased media flow will increase sorption of Ortho-P provided sufficient binding sites exist; sorbed pollutants are available for subsequent inter-event biological treatment or assimilation (Davis et al., 2010). Media specifications for BRC (Table 4-4) require 8-15% fines (silts and clays) and low P-Index less than or equal to 30 in NSW and less than or equal to 50 elsewhere in NC (NC DEQ, 2017). Media specifications from Flores et al. (2012) do not specify media fines or a target P-Index; RSC designers should consider BRC media specifications going forward (Table 4-16).

4.4.2.2.3. Exfiltration Trench

When considering the use of an exfiltration trench, the impact of media saturation (from base flow or groundwater intrusion) can be detrimental to TP treatment. Under anaerobic conditions, media sorbed phosphorus can be released (Clark & Pitt, 2012; Koryto et al., 2017). Prolonged media saturation should be avoided to prevent phosphorous leaching; therefore, exfiltration trench use should only be considered in well-drained underlying soils where greater influent volume reductions are a design goal (Table 4-16).

4.4.2.2.4. Forebay

Approximately 67% of TP is particulate-bound, therefore, effective removal of TSS will remove large fractions of TP (Novotny, 2003; Vaze & Chiew, 2004). The first 2 pools at both Alamance and Durham Co. RSCs proportionately reduced TP the most (Cizek et al., 2016; Koryto et al., 2017). The steep slope entering pool 1 at the Durham Co. RSC, however, may have contributed to scour and pollutant resuspension (Davis et al., 2010; Koryto et al., 2017; Miguntanna et al., 2013); hardened bed substrate may be necessary in RSC forebays to prevent scour entering pools (Koryto et al., 2017). Designers should consider dedicating the first pool of an RSC to be a forebay, to optimize sedimentation and system maintenance.

4.4.2.2.5. Vegetation

Factors influencing vegetation selection for nutrient sequestration include aesthetics, maintenance, and cost. In BRC herbaceous perennials (*Panicum virgatum*, *Helianthus angustifolius*, and *Eupatorium purpureum*) outperformed woody trees and shrubs when nutrient uptake, gram per square meter, was considered (Turk et al., 2017); BRC media did not affect phosphorous uptake (Turk et al., 2017). However, when species unit cost was included ($g/m^2/\$$) woody trees (*Betula* and *Magnolia*) were the best planting option (Turk et al., 2017). Differences in species' biomass, root architecture, physiological uptake, and physiochemical effects on surronding soils and micrrobial communities affects individual species performance at nutrient reductions (Read et al., 2008). Monocots *Carex appressa* R.Br (Cyperaceae), and diocots, *Melaleuca ericifolia* Sm. (Myrtaceae), performed best at nutrient uptake in laboratory and field-scale experiemnts simulating planted BRC treatment (Bratieres et al., 2008; Read et al., 2008). If planting RSC to optimize nutrient uptake, avoiding species with large mature canopies is best (Turk et al., 2017); however, when considering cost in planting decisions, species with large growth performed best at cumulative nutrient uptake (Turk et al., 2017).

4.4.2.3. Nitrogen

Table 4-17. RSC Design Nitrogen Removal

Design Parameter	*Range*	*Guidance*	*Supporting Studies*
Design Storm (mm)	25-38	RSC storage volume is the sum of surface and media storage; media storage based on media at field capacity.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Loading Ratio (unitless)	80-120:1	RSC can provide effective treatment at 80-120:1; exact loading rate will depend on contributing, SCM areas and design volume.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Storage Ratio (unitless)	0.5-1.5:1	Lower storage ratios are more effective at surface-to-seepage conversion; if pool volume is too small bed will scour and sedimentation will be ineffective.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018) Cizek et al. (2017)
Media	Sand: 75-85% Fines: 8-15% OM: 0-5% Mulch: $d60/_{d10} < 15$	Coarse sand provides rapid infiltration; fines and OM improve sorption and adjust infiltration rate. Homogeneous in-media mulch will prevent dispersion and clogging. Surface mulching may preserve infiltration rate.	Hunt et al. (2006) Hsieh & Davis (2005a) Koryto et al. (2017) Cizek et al. (2016)
In-Stream Structures	Site Dependent	Larger in-stream geomorphic structures, riffles, steps, cascades, may improve hyporheic exchange.	Hester & Doyle (2008)
Seasonal High-Water Table	≥ 0.6 m Above SHWT	If media saturation is a design feature, do not include saturated media in storage volume calculations.	Koryto et al. (2017) Cizek et al. (2016) Chapter 3
Forebay	Yes, when no upstream flow control.	Dedicate pool 1 to "Forebay/Pretreatment;" follow existing forebay design guidance. Benefits seepage conversion by preserving subsequent pools.	Cizek et al. (2016) Koryto et al. (2017) (NC DEQ, 2017)
Exfiltration Trench	When Applicable	Can provide additional media volume to improve seepage capacity. Rake final 0.3-m of exfiltration to preserve subsoil infiltration.	Cizek et al. (2016) Koryto et al. (2017) Brown & Hunt (2009) Chapter 3
Vegetation	Yes	Vegetation may improve volume reductions by creating macro pores and increasing ET. Care taken for wet vs. dry RSC zones.	Li et al., (2009) Wardynski & Hunt, (2012) Hatt et al. (2009)

4.4.2.3.1. Design Storm, Loading & Storage Ratios

The majority (50-70%) of nitrogen is dissolved (Novotny, 2003; Vaze & Chiew, 2004) and effective treatment occurs in the media layer; adequate capture volume and appropriate loading with a moderately low storage ratio will facilitate delivery of dissolved TN to the media zone. The Alamance Co. RSC (loading and storage ratios 100:1 and 0.42:1, respectively) provided good physical treatment of nitrogen, achieving a TN ELR of 76%. The Wake Co.-A RSC had favorable loading (72:1) and storage (0.87:1) ratios (Table 4-3) which provided good capture volume and effective surface-to-seepage conversion (Koryto et al., 2018). Dissolved pollutant concentrations showed little improvement via surface flow treatment (Koryto et al., 2018). Highlighting the importance of surface-to-seepage conversion for treatment by media; the Wake Co.-A RSC had a TN ELR of 20-46% (Koryto et al., 2018).

A small design storm (25% WQV), overloading (loading ratio of 160:1) and a large storage ratio (3.34:1) limited the Durham Co. RSC's ability to capture and convert surface flows to seepage, suppressing organic nitrogen (ON), Total Kjeldahl Nitrogen (TKN), and ammonium (NH_4^+) treatment; the Durham Co. RSC yielded a TN ELR of 4.4%. The Wake Co.-C RSC was severely undersized (3.2% WQV), loaded at 25,000:1 with upstream flow control with a low (0.01:1) storage ratio which likely benefited surface-to-seepage conversion; however, inadequate capture volume and saturation eliminated available capture volume (Chapter 3). The Wake Co.-C RSC was too small to substantially impact TN concentrations, the RE was 3% (Chapter 3). Palmer et al. (2014) observed an increase in particulate nitrogen (PN) concentration as precipitation depth increased, possibly from a reduction in the fraction of total flow converted to seepage; similar reductions in particulate treatment with increased storm size are observed at other RSCs (Cizek, 2014; Cizek et al., 2016; Koryto et al., 2018, 2017). Designing for surface-to-seepage conversion and volume reduction will ensure effective nitrogen delivery to the treatment zone (Table 4-17).

4.4.2.3.2. Media

Nitrogen treatment is highly dependent on media composition and field conditions as illustrated by the range of TN ELR (0-76%, Table 4-13). In simulated events, Koryto et al. (2018) determined all analytes (except nitrite-nitrate) experienced significant EMC reductions in subsurface flow; nitrite-nitrate ($NO_{2,3}$-N) significantly increased. NH_4^+ sorption during event flow is available for inter-event biological treatment (Davis et al., 2010). The simulated events at the Wake Co.-A RSC produced drier, aerobic conditions (Koryto et al., 2018). Aerobic conditions in media-based SCMs are, in some cases, linked to subsequent NO3- leaching via nitrification of ammoniacal nitrogen (Davis et al., 2010; Dietz & Clausen, 2005; Hsieh & Davis, 2005b; Hunt et al., 2006; Hunt et al., 2008). Anoxic conditions can support denitrification of NO3- . Localized anoxic conditions are thought to occur in well-drained SCMs contributing to nitrate reduction when no engineered saturation exist (Hunt et al., 2006; Kim et al., 2003).

In simulated events, Cizek (2014) observed well nitrogen concentrations varied from storm to storm; however, by hour 6 significant reductions from influent nitrogen concentrations had occurred and continued through hours 10 – 14 when wells emptied (Cizek, 2014). Median NO3- and TKN reductions relative to surface flows were 30% and 20%, respectively. Significant progressive treatment of TKN well concentrations occurred between hours 2 – 12; NO3- reduction was weaker. Cool soil temperatures, 3.6 – 10.6^{o}C, during simulations can inhibit microbial metabolism (Cizek, 2014). TKN reductions were likely driven by physical processes of sedimentation, filtration, and adsorption; whereas, cool temperatures inhibited microbial NO3- reductions (Davis et al., 2010; Hunt et al., 2012).

The Alamance Co. RSC significantly reduced TN concentrations (by 30%, Cizek et al., 2016). The well-drain portion of the Alamance Co. RSC, pools 1-3, accounted for median TN load reductions of 86%; due to a 36% increase in surface outflow from pool 4, the overall TN load reduction was 77% (Table 4-13) (Cizek et al., 2016). Production of Total Ammoniacal Nitrogen (TAN) in pool 3 via aerobic and anaerobic breakdown of OM likely underwent oxidation and nitrification at the cascade to pool 4 with subsequent denitrification in pool 4 (Cizek et al., 2016).

Koryto et al. (2017) reported low (3%) RE (Table 4-13) for TN at the Durham Co. RSC, despite significant reductions in pollutant concentration. Durham Co. RSC TN effluent concentrations frequently (50%) exceeded Piedmont targets for good water quality 0.99-mg/L (Koryto et al., 2017; McNett et al., 2010).

The Wake Co.-C RSC did not significantly reduce median TN concentration (RE of 3%) during storm flow; median TN influent and effluent representative sample concentrations were unaffected by treatment (Chapter 3). Storm flows at the Wake Co.-C RSC increased $NO_{2,3}$-N by 77%; however, the $NO_{2,3}$-N increase was not significant and possibly was the result of confluence with the minimally treated second inlet or nitrification prior to effluent sampling (Chapter 3). All other nitrogen species experienced minor (<11%) concentration reductions that were not statistically significant (Chapter 3). The Wake Co.-C RSC used a sand media mixed at an unknown ratio with an unknown mulch (Table 4-3); Loss on Ignition (LOI) analysis of bed substrate (i.e., a heterogeneous mix of sand, gravel, and OM) determined the mix contained less than 0.01% OM (Chapter 3).

Koryto et al. (2017) observed a median saturated hydraulic conductivity of 18-cm/hr; localized clogging was caused by mulch dispersion and/or sedimentation (Koryto et al., 2017). Mulch with a high d60/d10 ratio (> 15.4) will disperse and clog media limiting infiltration rates (Hsieh & Davis, 2005a). BRC media column studies have observed optimized $NO_{2,3}$-N removal at infiltration rates between 4-21-cm/hr during laboratory studies (Hsieh & Davis, 2005b; Kim et al., 2003); higher infiltration rates reduce TN residence times and limit biological treatment opportunities (Davis et al., 2010). Media infiltration and saturated hydraulic conductivity rates can be adjusted with an increase or decrease in the fraction of fines (silt and clay) (Hsieh & Davis, 2005a). More fines in media will likely increase the AL increasing sorption binding sites (Hsieh & Davis, 2005a). K_{sat} values observed by Koryto et al. (2017) should have produced better TN treatment than observed meaning physical RSC design or media composition likely contributed to the lack of treatment.

Base flow dampened Total Load Reductions (TLR) vis-à-vis ELR at the Durham Co. RSC due to inter-event leaching. TKN, TAN, and TN concentrations were significantly greater in inter-event grab samples compared to those of storm EMCs (Koryto et al., 2017). Koryto et al. (2017) reported a TAN increase of 390% during inter-event base flow while $NO_{2,3}$-N concentrations were significantly lower during inter-event flow. In contrast, inter-event treatment by the Wake Co.-C RSC significantly reduced concentrations for all nitrogen species except TAN (Chapter 3); TN, $NO_{2,3}$-N, ON, and TKN concentrations were reduced by 22%, 44%, 21%, and 17%, respectively. Lower media OM at the Wake Co.-C RSC limited TAN and TKN leaching in inter-event flow.

High media C:N ratios can lead to TKN and TAN production in saturated conditions (Hodge et al., 2000; Kim et al., 2003; Tiedje et al., 1982). Decomposition of OM exceeding rates of assimilation or reduction processes will export TKN, total organic carbon, and turbidity in BRC media columns (Kim et al., 2003). Denitrification and Dissimilatory Nitrate Reduction to Ammonia (DNRA) are competing processes under anoxic conditions depending on available organic carbon and NO3- concentration; denitrification is favored when organic carbon is limited, while DNRA is favored when NO3- is limited (Davis et al., 2010; Tiedje et al., 1982). Microbial mineralization of organic nitrogen is dependent on the C:N ratio of growth substrate and the microbes' need for nitrogen; at C:N ratios less than 25:1 decomposers typically export ammoniacal nitrogen (Hodge et al., 2000). Denitrification results in nitrogen loss as dinitrogen and nitrous oxide gases (Tiedje et al., 1983); DNRA conserves nitrogen with the production of TAN (Tiedje et al., 1983).

Media leaching may be explained by highly reduced conditions during base flow and near complete reduction of available NO3- along with the production of TAN via DNRA and OM decomposition. The Durham Co. RSC's media OM (hardwood mulch) has a C:N ratio greater than 50:1 (Tiquia et al., 2002); however, the small particle size and quantity used (20% by volume) contributed to its bioavailability and thus nitrogen leaching (Koryto et al., 2017). The Wake Co.-C RSC's design resulted in a large amount of gravel and cobble substrate under anoxic conditions which may have benefited total C:N ratios in the saturated zone, thus resulting in less TAN production during base flow.

Both the Durham and Alamance Co. RSCs contained pools that were either well-drained or saturated, yet TN treatment performance was substantially different. Media composition influenced treatment; the Durham Co. RSC employed media composition recommended by (Flores et al., 2012, 80% sand, 20% woodchips) (Table 4-4), while the Alamance Co. RSC used sand media with surface mulching of composted hardwood (Table 4-3). The lower C:N ratio in the Alamance Co. RSC produced better conditions for denitrification at influent NO3- concentrations. Similarly, the Wake Co.-A RSC used sand media (Table 4-3) with surface mulching of composted hardwood which also yielded substantially better TN treatment than the Durham Co. RSC (Table 4-13). Prolonged saturation produced in the Alamance Co. RSC's exfiltration trench and absence of in-media OM improved NO3- treatment comparable to that of IWS zones used in BRC (Brown et al., 2011; Davis, 2008; Kim et al., 2003), while 20% in-media OM contributed to leaching in areas of saturation at the Durham Co. RSC. In short, the Alamance Co. and Wake Co.-A RSCs employed a good media for TN removal, while Durham Co. and Wake Co.-C did not.

RSC media OM recommendations by Flores et al. (2012) are likely too high for typical urban runoff TN concentrations (Chapter 3; Cizek et al., 2016; Koryto et al., 2017); in well-drained conditions BRC media specifications of 75-85% sand, 8-15% fines, and 0-5% OM are appropriate (Koryto et al., 2017). When managing high influent TN concentrations like those from agricultural fields, higher OM may improve denitrification potential (Ergas et al., 2010). Design-specific (i.e., well-drained, versus saturated) RSC media specifications may be appropriate, with the later having less OM, to limit potential TN leaching while supporting denitrification. Based upon current knowledge, RSC designers should apply BRC media specifications to RSCs (Table 4-17) (Chapter 3; Cizek et al., 2016; Koryto et al., 2018, 2017).

4.4.2.3.3. In-Stream Structures

Effective conversion of surface flows to seepage will ensure treatment of dissolved TN in the media zone and filtration of PN. In simulated events, Koryto et al. (2018) determined TKN was the only analyte that experienced significant EMC reduction in surface flow, likely resulting from surface—seepage exchange across riffles; dissolved pollutant concentrations showed little improvement via surface flow treatment (Koryto et al., 2018). Cizek (2014), in simulated events, saw no significant difference in surface flow pollutant concentrations (Table 4-3). At the Alamance and Durham Co. RSCs, local anoxic zones likely contributed to simultaneous nitrification-denitrification processes in well-drained cells where substantial portions of surface flow were converted to seepage (Table 4-10) (Cizek et al., 2016; Hunt et al., 2006; Kim et al., 2003; Koryto et al., 2017). Denitrification during storm flow is inhibited by downwelling of aerated influent into the media zone, observed by Koryto et al. (2017) with DO measurement; anaerobic conditions returned within 48 to 72 hours post-event flow (Koryto et al., 2017). Lautz & Fanelli (2008) determined anoxic conditions existed upstream of in-stream structures in low-velocity, pool zones while downstream of in-stream structures oxic conditions existed in turbulent flow areas; anoxic and oxic conditions corresponded to the reduction and production of nitrate, respectively, regardless of season (Lautz & Fanelli, 2008). Designing RSCs to improve downwelling rates (larger in-stream structures) should benefit TN treatment, provided residence times are sufficiently long.

Delivery of influent dissolved TN to the media zone is vital to treatment; therefore, RSCs should be designed to effectively convert surface flows to seepage. Seepage will increase filtration of ON and sorption of TKN and NH_4^+, provided sufficient binding sites exist (Hsieh & Davis, 2005a). Larger structures, steps and weirs, will improve hyporheic exchange and pollutant delivery to the media zone (Table 4-17) (Hester & Doyle, 2008; Lawrence et al., 2013).

4.4.2.3.4. Forebay

Forebay usage will benefit capture of TKN and ON and preserve sequential pools from clogging. Most TN reduction occurred in well-drained portions of the Alamance and Durham Co. RSCs, where substantial portions of flow was converted to seepage (Cizek et al., 2016; Koryto et al., 2017). Koryto et al. (2017) specified sedimentation as the primary treatment mechanism with subsurface filtration providing minimal additional treatment (Koryto et al., 2017). Designers should consider dedicating the first pool of an RSC to be a forebay, to optimize sedimentation and system maintenance (Table 4-17)(Cizek et al., 2017; Koryto et al., 2017; NC DEQ, 2017).

4.4.2.3.5. Exfiltration Trench

The decision to create an area of prolonged media saturation will influence other design elements (e.g., storage capacity, media selection, in-stream structures, and vegetation). An exfiltration trench can likely improve NO3- treatment comparable to IWS zones in BRC (Brown & Hunt, 2011; Cizek et al., 2016; Davis, 2008; Kim et al., 2003); however, incorrect usage will limit storage volume diminishing RSC treatment and possibly lead to leaching of nutrients (Chapter 3; Koryto et al., 2017).

4.4.2.3.6. Vegetation

Vegetation will likely uptake nutrients; however, vegetation establishment has likely been inhibited by conditions within RSCs. No RSCs had established vegetation during the monitoring period (Browning, 2008; Cizek et al., 2016, 2017; Filoso, 2012; Koryto et al., 2018, 2017), due to early successional monitoring. The Wake Co. -A and -B RSC was established with warm season grasses; it only received flow during simulation events (Cizek, 2014; Koryto et al., 2018). The Brunswick Co. RSC was armored with Class A rip-rap and did report the establishment of wetland vegetation in an area of prolonged saturation (Cizek et al., 2017). Vegetation will slow velocities, trap sediment, and assimilate pollutants (Flores et al., 2012; Turk et al., 2017). Vegetation improves infiltration rates in BRC, by creating macropores around plant roots and improves volume reductions via ET (Brown et al., 2013; Li et al., 2009; Wadzuk et al., 2015; Wardynski & Hunt, 2012). Plant selection can impact nutrient concentration and load reductions (Bratieres et al., 2007; Read et al., 2008; Turk et al., 2017). Flores et al. (2012) specifies use of native vegetation in RSCs; however, in BRC, some cultivars outperform native species' at nutrient concentration and load reductions (Bratieres et al., 2007; Read et al., 2008; Turk et al., 2017).

In BRCs, herbaceous perennials (*Panicum virgatum, Helianthus angustifolius, and Eupatorium purpureum*) outperformed wood trees and shrubs when nutrient uptake (measured as gram per square meter) was considered (Turk et al., 2017); however, when species unit cost was included ($g/m^2/\$$) woody trees (*Betula* and *Magnolia*) were the best planting option (Turk et al., 2017). Differences in species' biomass, root architecture, physiological uptake, and physiochemical effects on surronding soils and micrrobial communities affects individual species' ability to reduce nutrients (Read et al., 2008); monocots, *Carex appressa* R.Br (Cyperaceae), and diocot, *Melaleuca ericifolia* Sm. (Myrtaceae), were best at nutrient uptake in laboratory and field-scale experiments simulating planted BRC treatment (Bratieres et al., 2008; Read et al., 2008). Using BRC plant selection as a guide for RSCs, to optimize nutrient uptake, avoiding species with large mature canopies was best (Turk et al., 2017); however, when considering cost woody species were best at cumulative nutrient uptake (Turk et al., 2017).

Vegetation is used in constructed wastewater treatment and urban stormwater wetlands to assimilate nutrients and polluants reducing pollutant concentrations and loads (Kadlec, 1999; Rogers et al., 1991; Song et al., 2001). Wetland planting recommendations for constructed stormwater wetlands apply a zonal planting method based on ponding depths which depend on species tolerance to saturation (NC DEQ, 2017). Designers must understand each RSC cell's degree of saturation prior to selecting vegetation. Drier cells will likely employ vegetation adapted to BRCs, while wetter cells will draw upon wetland guidance (Table 4-17).

4.5. Regenerative Stormwater Conveyance Comprehensive Design

Table 4-18. RSC Comprehensive Design

Design Parameter	*Range*	*Guidance*	*Supporting Studies*
Design Storm (mm)	25-38	RSC storage volume is the sum of surface and media storage; media storage based on media at field capacity.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Loading Ratio (unitless)	80-120:1	RSC can provide effective treatment at 80-120:1; exact loading rate will depend on contributing, SCM areas and design volume.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018)
Storage Ratio (unitless)	0.5-1.5:1	Lower storage ratios are more effective at surface-to-seepage conversion; if pool volume is too small bed will scour and sedimentation will be ineffective.	Koryto et al. (2017) Cizek et al. (2016) Koryto et al. (2018) Cizek et al. (2017)
Media	Sand: 75-85% Fines: 8-15% OM: 0-5% P-Index: ≤ 30 in NSW*; ≤ 50 else Mulch: $d60/_{d10} < 15$	Coarse sand provides rapid infiltration; fines and OM improve sorption. Homogeneous in-media mulch prevents clogging. Surface mulching may preserve infiltration rate and improve pollutant sorption; P-index ≤ 30 in NSW* and P-index ≤ 50 elsewhere.	Hunt et al. (2006) Hsieh & Davis (2005a) Koryto et al. (2017) Cizek et al. (2016)
In-Stream Structures	Site Dependent	Larger in-stream geomorphic structures, riffles, steps, cascades, may improve hyporheic exchange.	Hester & Doyle (2008)
Slope (%)	1-10	Dependent on site conditions; steeper slopes will require larger grade control structures and produce stronger hydraulic gradients.	Flores et al. (2012) Koryto et al. (2017) Cizek et al. (2016)
Exfiltration Trench	When Appropriate	Can provide additional media volume to improve seepage capacity or saturated zone for denitrification. Rake final 0.3-m of exfiltration to preserve subsoil infiltration.	Cizek et al. (2016) Koryto et al. (2017) Brown & Hunt (2009) Chapter 3
Forebay	Yes, when no upstream flow control.	Dedicate pool 1 to "Forebay/Pretreatment;" follow existing forebay design guidance. Benefits seepage conversion by preserving subsequent pools.	Cizek et al. (2016) Koryto et al. (2017) (NC DEQ, 2017)
Vegetation	Yes	Vegetation may improve volume reductions by creating macro pores and increasing ET. Care taken for wet vs. dry RSC zones.	Li et al., (2009) Wardynski & Hunt, (2012) Hatt et al. (2009)

*Nutrient Sensitive Waters

Comprehensive RSC design in North Carolina can capture 38- or 25-mm of rainfall depending on site location in coastal counties or otherwise, respectively. Depending on site constraints (i.e., available disturbance, right-of-way, economics, etc.) loading ratios should be adjusted to accommodate the design storm; however, effective treatment is possible when loaded at 80-120:1 (Table 4-18), provided no upstream flow controls exist. If constructed in-series with (downstream of) of hydrologic controls, effective treatment may be possible at much higher loading rates. Storage ratios should be used to understand the treatment implications of design options. Conversion of surface flow to seepage is an important factor influencing RSC treatment; storage ratios should range from 0.5-1.5:1 (Table 4-18).

Media composition can vary depending on setting (well-drained vs. saturated conditions); but should generally follow specifications for BRCs (Table 4-18). Type, size, and number of in-stream structures will be site dependent and based on engineering judgment (Table 4-18); decisions must balance site characteristics and constraints (i.e., available material, slope, area, energy dissipation, storage creation, etc.). Designers should consider where saturation may occur and design to optimize surface-to-seepage conversion. RSC slopes will depend on site conditions; however, it is evident that effective treatment is possible up to 10% grade (Table 4-18). At this time, it is not discernable which structures perform best at a given slope; however, 4-5% may be a threshold above which riffles should not be used. Steep slopes require larger structures and produce higher hydraulic gradients, improving surface-to-seepage conversion (Table 4-18), but may require additional armoring to prevent bed scour and particulate resuspension.

Exfiltration trenches can provide additional media volume and create conditions to improve volume reduction and nutrient treatment (Table 4-18); exfiltration trench application should influence other design elements (i.e., storage capacity, in-stream structures, media, and vegetation). Exfiltration trench construction should follow methods described by Brown & Hunt (2009) to preserved the underlying soil's infiltration rate.

In the absence of upstream flow controls, pool 1 should be designated as a forebay; pretreatment will provide effective sedimentation, still influent, and minimize scour and pollutant resuspension. A hardened or cobble bed substrate may be necessary at step grades (Koryto et al., 2017). A forebay likely streamlines maintenance as well (Table 4-18) (Blecken et al., 2017). Wetland planting recommendations for constructed stormwater wetlands apply a zonal planting method based on ponding depths (NC DEQ, 2017). Designers must understand each RSC cell's degree of saturation prior to selecting vegetation. Drier cells will likely employ vegetation adapted to BRCs, while wetter cells will draw upon wetland guidance (Table 4-18).

The adoption of RSCs in retrofit opportunities for which conveyance and treatment is a project objective will continue as scientific consensus on design parameters is achieved; however, current research has highlight three major design parameters that, if considered in design, will improve the performance of retrofit RSCs.

1. Storage ratio: Retrofit RSCs are often highly constrained and it is desireable to construct the practice within the existing gulley or headcut to minimizing the impact to riparian vegeation. Maximizing media seepage while maintaining pool deminsions able to disipate energy and prevent bed scoure will achevie the best overall treatment and project success.
2. Forebay: The major of RSCs observe subtaintial portions of sediment capture in the first pools of a practice. Considering forebay characteristics and maintenance concerns during

RSC design will lessen the impact of sediment on subsequent pools while easing the burden of maintenance when needed.

3. Media: Media selection is integral to effective nutrient treatment. At this time there is no evidence to suggest that media other than standard BRC media is merited for typical urban stormwater pollutant concentrations. If higher influent nutrient concentrations are expected careful consideration of increasing media OM maybe merited; however, practices that employ large additions of OM under prolonged saturation or typical urban stormwater concentrations have been associated with nutrient leaching and poor treatment.

A major hinderance to RSC treatment is media saturation. It is not advicable to construct the RSC media layer or pools in areas that may intersect the SHWT. Media saturation from groundwater or baseflow will limit seepage reducing both hydrologic and nutrient treatment. Prolonged saturation has produced leaching in some RSCs. The exfiltration trench is attractive for its ability to provide additional media volume; however, if the trench remains saturated no discernable hydrologic benefits will be observed. The exfiltration trench should therefore only be used in areas where well drained conditions are predicted and media volume will dewater prior to subsequent rainfall. If baseflow saturation is a concern, additional pool volume or a larger storage ratio may provide better treatment.

4.6. Conclusions

RSC has the potential, when designed and maintained properly, to provide effective conveyance of storm flows and treatment of stormwater pollutants. RSC is an emerging SCM and additional research and evaluation are needed to improve these design recommendations. Additional research should explore the following questions:

1. Media selection: the influence of media OM, amount and type, on pollutant treatment with varying design configurations (i.e., well-drained and saturated conditions) and influent concentrations.
2. In-stream structure selection: how does slope influence structure selection and vice versa? How do structure types influence surface-to-seepage conversion? How can structure type, size, number and slope be optimized to deliver flows to the media zone in saturated conditions?
3. Exfiltration trench: When is an exfiltration trench appropriate and how can it be optimized for pollutant reductions?
4. Forebay: Should RSC forebays (i.e., depth, volume, profile) differ from those of other SCMs? Will RSC forebay maintenance differ from that of other SCMs?
5. Vegetation: Is vegetation appropriate in RSCs? Are existing planting recommendations adequate for RSCs? How will vegetation affect RSC hydraulics?
6. RSC life cycle: Is an RSC expected to maintain the same function and appearance over time, or will it evolve into a more “natural” condition? To what extent do RSCs provide additional habitat or ecosystem services beyond stormwater treatment?

4.7. References

Barrett, M. E. (2003). Performance, Cost, and Maintenance Requirements of Austin Sand Filters. *Journal of Water Resources Planning and Management*, *129*(3), 234–242. https://doi.org/10.1061/(ASCE)0733-9496(2003)129:3(234)

Bennett, S. J., Alonso, C. V, Prasad, S. N., & Romkens, M. J. (2000). Experiments on headcut growth in concentrated flows typical of upland areas. *Water Resources Research*, *36*(7), 1911–1922.

Blecken, G. T., Hunt, W. F., Al-Rubaei, A. M., Viklander, M., & Lord, W. G. (2017). Stormwater control measure (SCM) maintenance considerations to ensure designed functionality. *Urban Water Journal*, *14*(3), 278–290. https://doi.org/10.1080/1573062X.2015.1111913

Bratieres, K., Fletcher, T. D., Deletic, A., & Zinger, Y. (2008). Nutrient and sediment removal by stormwater biofilters: A large-scale design optimisation study. *Water Research*, *42*(14), 3930–3940. https://doi.org/10.1016/j.watres.2008.06.009

Brown, R. A., & Hunt, W. F. (2009). Effects of Media Depth on Bioretention Performance in the Upper Coastal Plain of North Carolina and Bioretention Construction Impacts Study. *World Enviromental and Water Resources Congress*.

Brown, R. A., & Hunt, W. F. (2011). Underdrain Configuration to Enhance Bioretention Exfiltration to Reduce Pollutant Loads. *Journal of Environmental Engineering*, *137*(11), 1082–1091. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000437

Brown, R. A., Skaggs, R. W., & Hunt, W. F. (2013). Calibration and validation of DRAINMOD to model bioretention hydrology. *Journal of Hydrology*, *486*, 430–442. https://doi.org/10.1016/j.jhydrol.2013.02.017

Brown, T., Berg, J., & Underwood, K. (2010). Replacing Incised Headwater Channels and Failing Stormwater Infrastructure with Regenerative Stormwater Conveyance. In *Low Impact Development 2010: Redefining Water in the City*. https://doi.org/10.1061/41099(367)4

Browning, M. O. (2008). The Efficacy of Urban Stream Restorations to Improve Water Quality Across a Spectrum of Design Approaches. *Thesis Submitted to George Mason University*, 118.

Burns, M. J., Fletcher, T. D., Walsh, C. J., Ladson, A. R., & Hatt, B. E. (2012). Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform. *Landscape and Urban Planning*, *105*(3), 230–240. https://doi.org/10.1016/j.landurbplan.2011.12.012

Chattahoochee Riverkeeper. (2015). Tri-State Water Conflict. Retrieved from https://chattahoochee.org/our-work/tri-state-water-conflict/

Cizek, A. R. (2014). *Quantifying the Stormwater Mitigation Performance and Ecosystem Service Provision in Regenerative Stormwater Conveyance (RSC)*. North Carolina State Univeristy.

Cizek, A. R., Hunt, W. F., Winston, R. J., & Lauffer, M. S. (2017). Hydrologic Performance of Regenerative Stormwater Conveyance in the North Carolina Coastal Plain. *Journal of Environmental Engineering*, *143*(9), 05017003. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001198

Cizek, A. R., Hunt, W. F., Winston, R. J., Waickowski, S., Narayanaswamy, K., & Lauffer, M. S. (2016). Water Quality and Hydrologic Performance of a Regenerative Stormwater Conveyance (RSC) in the Piedmont of North Carolina. *Journal of Environmental Engineering*, *Submitted*(8), 1–11. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001344

Clark, S. E., & Pitt, R. (2012). Targeting treatment technologies to address specific stormwater pollutants and numeric discharge limits. *Water Research*, *46*(20), 6715–6730. https://doi.org/10.1016/j.watres.2012.07.009

Davis, & , Robert G. Traver, Hurt, W. F. (2010). Improving Urban Stormwater Quality: Applying Fundamental Principles. *Journal of Contemporary Water Research & …*, *146*(1), 2–10. https://doi.org/10.1111/j.1936-704X.2010.00387.x

Davis, A. P. (2008). Field Performance of Bioretention: Hydrology Impacts. *Journal of Hydrologic Engineering*, *13*(2), 90–95. https://doi.org/10.1061/(ASCE)1084-0699(2008)13:2(90)

Davis, A. P., Shokouhian, M., Sharma, H., & Minami, C. (2006). Water Quality Improvement through Bioretention Media: Nitrogen and Phosphorus Removal. *Water Environment Research*, *78*(3), 284–293. https://doi.org/10.2175/106143005X94376

Davis, A. P., Shokouhian, M., Sharma, H., Minami, C., & Winogradoff, D. (2003). Water quality improvement through bioretention: lead, copper, and zinc removal. *Water Environment Research : A Research Publication of the Water Environment Federation*, *75*(1), 73–82. https://doi.org/10.2175/106143003X140854

Dietz, M. E., & Clausen, J. C. (2005). A field evaluation of rain garden flow and pollutant treatment. *Water, Air, and Soil Pollution*, *167*(1–4), 123–138. https://doi.org/10.1007/s11270-005-8266-8

Doll, B. a., Grabow, G. L., Hall, K. R., Halley, J., Harman, W. A., Jennings, G. D., & Dani E. Wise. (2003). Stream Restoration: A Natural Channel Design Handbook, 128 pp. https://doi.org/(11/19/2014)

EPA. (2016). Summary of the Clean Water Act. Retrieved October 14, 2016, from https://www.epa.gov/laws-regulations/summary-clean-water-act

Ergas, S. J., Sengupta, S., Siegel, R., Pandit, A., Yao, Y., & Yuan, X. (2010). Performance of Nitrogen-Removing Bioretention Systems for Control of Agricultural Runoff. *Journal of Environmental Engineering*, *136*(10), 1105–1112. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000243

Filoso, S. (2012). Assessing the Effectiveness of Regenerative Stormwater Conveyance in reducing Sediment and Nutrient Loads to Three Mid-River Creeks in the Severn River watershed, (410), 1–22.

Fletcher, T. D., Vietz, G., & Walsh, C. J. (2014). Protection of stream ecosystems from urban stormwater runoff: The multiple benefits of an ecohydrological approach. *Progress in Physical Geography*, *38*(5), 543–555. https://doi.org/10.1177/0309133314537671

Flores-Cervantes, J. H., Istanbulluoglu, E., & Bras, R. L. (2006). Development of gullies on the landscape: A model of headcut retreat resulting from plunge pool erosion. *Journal of Geophysical Research: Earth Surface*, *111*(1), 1–14. https://doi.org/10.1029/2004JF000226

Flores, H., Mcmonigle, D., & Underwood, K. (2012). *Design Guidelines for Regenerative Step Pool Storm Conveyance*.

Gallo, C., Moore, A., & Wywrot, J. (2012). Comparing the adaptability of infiltration based BMPs to various U.S. regions. *Landscape and Urban Planning*, *106*(4), 326–335. https://doi.org/10.1016/j.landurbplan.2012.04.004

Hardy, D. H., Tucker, M. R., & Stokes, C. E. (2009). Crop Fertilization Based on North Carolina prepared by, (1).

Hatt, B. E., Fletcher, T. D., & Deletic, A. (2009). Hydrologic and pollutant removal performance of stormwater biofiltration systems at the field scale. *Journal of Hydrology*, *365*, 310–321.

https://doi.org/10.1016/j.jhydrol.2008.12.001

Henderson, C., Greenway, M., & Phillips, I. (2007). Removal of dissolved nitrogen, phosphorous and carbon from stormwater by biofiltration mesocosms. *Water Science and Technology*, *55*(4), 183–191. https://doi.org/10.2166/wst.2007.108

Hester, E. T., & Doyle, M. W. (2008). In-stream geomorphic structures as drivers of hyporheic exchange. *Water Resources Research*, *44*(3). https://doi.org/10.1029/2006WR005810

Hodge, A., Robinson, D., & Fitter, A. (2000). Are microorganisms more effective than plants at competing for nitrogen? *Trends in Plant Science*, *5*(7), 304–308. https://doi.org/10.1016/S1360-1385(00)01656-3

Hood, G., & Sommer, L. (2016). High Demand, Low Supply: Colorado River Water Crisis hits Across the West. *National Public Radio*. Retrieved from http://www.npr.org/2016/12/30/507569514/high-demand-low-supply-colorado-river-water-crisis-hits-across-the-west

Hsieh, C., & Davis, A. P. (2005a). Evaluation and Optimization of Bioretention Media for Treatment of Urban Storm Water Runoff. *Journal of Environmental Engineering*, *131*, 1521–1531. https://doi.org/10.1061/(ASCE)0733-9372(2005)131:11(1521)

Hsieh, C., & Davis, A. P. (2005b). Multiple event study of bioretention for treatment of urban storm water runoff. *Water Science and Technology*, *51*, 177–181. Retrieved from http://link.aip.org/link/?ASC/118/281/1%5Cnhttp://dx.doi.org/10.1061/40685(2003)281

Hunt, W. F., Davis, A. P., & Traver, R. G. (2012). Meeting Hydrologic and Water Quality Goals through Targeted Bioretention Design. *Journal of Environmental Engineering*, *138*(6), 698–707. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000504

Hunt, W. F., Jarrett, A. R. R., Smith, J. T., & Sharkey, L. J. (2006). Evaluating Bioretention

Hydrology and Nutrient Removal at Three Field Sites in North Carolina. *Journal of Irrigation and Drainage Engineering*, *132*(6), 600–608. https://doi.org/10.1061/(ASCE)0733-9437(2006)132:6(600)

Hunt, W. F., Smith, J. T., Jadlocki, S. J., Hathaway, J. M., & Eubanks, P. R. (2008). Pollutant Removal and Peak Flow Mitigation by a Bioretention Cell in Urban Charlotte, N.C. *Journal of Environmental Engineering*, *134*(5), 403–408. https://doi.org/10.1061/(ASCE)0733-9372(2008)134:5(403)

Jones, M. P., Hunt, W. F., & Winston, R. J. (2012). Effect of Urban Catchment Composition on Runoff Temperature. *Journal of Environmental Engineering*, *138*(12), 1231–1236. https://doi.org/doi:10.1061/(ASCE)EE.1943-7870.0000577

Kadlec, R. H. (1999). Chemical, physical and biological cycles in treatment wetlands. *Water Science and Technology : A Journal of the International Association on Water Pollution Research*. https://doi.org/S0273-1223(99)00417-5

Keith R. Underwood. (2002). United States Patent Application Publication - Regnerative Stormwater Conveyance System and Method, *1*(60). Retrieved from https://www.google.com/patents/US20090290936

Kim, H., Seagren, E. A., & Davis, A. P. (2003). Engineered Bioretention for Removal of Nitrate from Stormwater Runoff. *Water Environment Research*, *75*(4), 355–367.

Koryto, K. M. (2016). *Evaluation of the hydrologic and water quality benefits of regenerative stormwater conveyance*. North Carolina State University.

Koryto, K. M., Asce, M., Hunt, W. F., Arellano, C., & Page, J. L. (2018). Performance of Regenerative Stormwater Conveyance on the Removal of Dissolved Pollutants: Field Scale Simulation Study, *144*(6), 1–13. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001374

Koryto, K. M., Hunt, W. F., & Page, J. L. (2017). Hydrologic and water quality performance of regenerative stormwater conveyance installed to stabilize an eroded outfall. *Ecological Engineering*, *108*, 263–276. https://doi.org/10.1016/j.ecoleng.2017.04.041

Ladson, A. R., Walsh, C. J., & Fletcher, T. D. (2006). Improving Stream Health in Urban Areas by Reducing Runoff Frequency from Impervious Surfaces. *Australian Journal of Water Resources*, *10*(1), 23–33.

Lautz, L. K., & Fanelli, R. M. (2008). Seasonal biogeochemical hotspots in the streambed around restoration structures. *Biogeochemistry*, *91*(1), 85–104. https://doi.org/10.1007/s10533-008-9235-2

Lawrence, J. E., Skold, M. E., Hussain, F. A., Silverman, D. R., Resh, V. H., Sedlak, D. L., … McCray, J. E. (2013). Hyporheic Zone in Urban Streams: A Review and Opportunities for Enhancing Water Quality and Improving Aquatic Habitat by Active Management. *Environmental Engineering Science*, *30*(8), 480–501. https://doi.org/10.1089/ees.2012.0235

Leopold, L. B. (1994). *A View of the River*. Cambridge, Massachusetts: Harvard University Press.

Li, H., Sharkey, L. J., Hunt, W. F., & Davis, A. P. (2009). Mitigation of Impervious Surface Hydrology Using Bioretention in North Carolina and Maryland. *Journal of Hydrologic Engineering*, *14*(4), 407–415. https://doi.org/10.1061/ASCE1084-0699200914:4407

McNett, J. K., Hunt, W. F., & Osborne, J. A. (2010). Establishing Storm-Water BMP Evaluation Metrics Based upon Ambient Water Quality Associated with Benthic Macroinvertebrate Populations. *Journal of Environmental Engineering*, *136*(5), 535–541. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000185

Meyer, J. L., Paul, M. J., & Taulbee, W. K. (2005). Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society*, *24*(3), 602–612. https://doi.org/10.1899/04-021.1

Miguntanna, N. P., Liu, A., Egodawatta, P., & Goonetilleke, A. (2013). Characterising nutrients wash-off for effective urban stormwater treatment design. *Journal of Environmental Management*, *120*, 61–67. https://doi.org/10.1016/j.jenvman.2013.02.027

N.C. DEQ. (2017). North Carolina Stormwater Control Measure Credit Document. *North Carolina Department of Environmental Quality*. Retrieved from https://files.nc.gov/ncdeq/Energy Mineral and Land Resources/Stormwater/BMP Manual/SSW-SCM-Credit-Doc-20170807.pdf

NC DEQ. (2017). Stormwater Design Manual.

NCDENR. (2007). Stormwater Best Management Practices Manual. North Carolina Division of Water Quality.

NOAA National Weather Service. (n.d.). NOAA Atlas 14: Precipitation Frequency Data Server. Retrieved March 12, 2018, from https://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_cont.html

North Carolina Department of Environmental Quality. (2017). Stormwater Design Manual. Retrieved April 27, 2018, from https://deq.nc.gov/sw-bmp-manual

Novotny, V. (2003). *Water Quality: Diffuse Pollution and Watershed Management* (2nd ed.). Hoboken, New Jersey: John Wiley & Sons, Inc.

Page, J. L., Winston, R. J., & Hunt, W. F. (2015). Soils beneath suspended pavements: An opportunity for stormwater control and treatment. *Ecological Engineering*, *82*, 40–48. https://doi.org/10.1016/j.ecoleng.2015.04.060

Page, J. L., Winston, R. J., Mayes, D. B., Perrin, C., & Hunt Iii, W. F. (2015). Retrofitting with innovative stormwater control measures: Hydrologic mitigation of impervious cover in the municipal right-of-way. *Journal of Hydrology*, *527*, 923–932. https://doi.org/10.1016/j.jhydrol.2015.04.046

Palmer, M. A., Filoso, S., & Fanelli, R. M. (2014). From ecosystems to ecosystem services: Stream restoration as ecological engineering. *Ecological Engineering*, *65*, 62–70. https://doi.org/10.1016/j.ecoleng.2013.07.059

Read, J., Wevill, T., Fletcher, T., & Deletic, A. (2008). Variation among plant species in pollutant removal from stormwater in biofiltration systems. *Water Research*, *42*(4–5), 893–902. https://doi.org/10.1016/j.watres.2007.08.036

Rogers, K. H., Breen, P. F., & Chick, A. J. (1991). Nitrogen Removal in Experimental Wetland Treatment Systems - Evidence for the Role of Aquatic Plants. *Res. J. Water Pollut. Control. F.*, *63*(7), 934–941.

Rosgen, D. L. (1994). A classification of natural rivers: Reply. *Catena*, *22*(3), 169–199. https://doi.org/10.1016/0341-8162(94)90001-9

Song, Y., Fitch, M., Burken, J., Ross, C., & Feeler, A. (2001). Lead and Zinc Removal by Lab-Scale Constructed Wetlands. *Water Environment Research*, *73*(1), 37–44.

Stein, O. R., & LaTray, D. A. (2002). Experiments and modeling of head cut migration in stratified soils. *Water Resources Research*, *38*(12), 20-1-20–12. https://doi.org/10.1029/2001WR001166

Temple, D. M., & Hanson, G. J. (1994). Headcut development in vegetated earth spillways. *Applied Engineering in Agriculture*, *10*(5), 677–682.

Tiedje, J. M., Sexstone, A. J., Myrold, D. D., & Robinson, J. A. (1982). Denitrification: ecological niches, competition and survival. *Antonie van Leeuwenhoek*, *48*(6), 569–583. https://doi.org/10.1007/BF00399542

Tippett, R. (2017). North Carolina population growth at highest levels since 2010. Retrieved from http://demography.cpc.unc.edu/2017/02/10/north-carolina-population-growth-at-highest-levels-since-2010/

Tiquia, S. M., Lloyd, J., Herms, D. A., Hoitink, H. A. J., & Michel, F. C. (2002). Effects of mulching and fertilization on soil nutrients, microbial activity and rhizosphere bacterial community structure determined by analysis of TRFLPs of PCR-amplified 16S rRNA genes. *Applied Soil Ecology*, *21*(1), 31–48. https://doi.org/10.1016/S0929-1393(02)00040-9

Turk, R. P., Kraus, H. T., Hunt, W. F., Carmen, N. B., & Bilderback, T. E. (2017). Nutrient Sequestration by Vegetation in Bioretention Cells Receiving High Nutrient Loads. *Journal of Environmental Engineering*, *143*(2), 06016009. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001158

United Nations. (2014). World's population increasingly urban with more than half living in urban areas. Retrieved October 12, 2016, from http://www.un.org/en/development/desa/news/population/world-urbanization-prospects-2014.html

Vaze, J., & Chiew, F. H. S. (2004). Nutrient loads associated with different sediment sizes in urban stormwater and surface pollutants. *Journal of Environmental Engineering*, *130*(4), 391–396. https://doi.org/10.1061/(ASCE)0733-9372(2004)130:4(391)

Wadzuk, B. M., Hickman, J. M., & Traver, R. G. (2015). Understanding the Role of Evapotranspiration in Bioretention: Mesocosm Study. *Journal of Sustainable Water in the Built Environment*, *1*(2), 04014002. https://doi.org/10.1061/JSWBAY.0000794

Walsh, C. J., Fletcher, T. D., & Ladson, A. R. (2009). Retention Capacity: A Metric to Link Stream Ecology and Storm-Water Management. *Journal of Hydrologic Engineering*, *14*(4), 399–406. https://doi.org/10.1061/ASCE1084-0699200914:4399

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan II, R. P. (2005). The urban stream syndrome : current knowledge and the search for a cure The urban stream syndrome : current knowledge and. *Journal of the North American Benthological Society*, *24*(3), 706–723. https://doi.org/10.1899/04-028.1

Wardynski, B. J., & Hunt, W. F. (2012). Are Bioretention Cells Being Installed Per Design Standards in North Carolina? A Field Study. *Journal of Environmental Engineering*, *138*(12), 1210–1217. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000575

West Virginia Department of Environmental Protection. (2012). West Virginia Stormwater Management & Design Guidance Manual, 1–22.

APPENDICES

Appendix A

At the confluence of two 10.2-cm PVC pipes the diameter was increased to 15.2-cm PVC. The conveyance was trenched and buried (maximum flow length of approximately 30.5-meters) with an inlet at the top of the RWH; an overflow conveyance exited the tank at the same elevation as the inlet and carried overflow to the nearby stream (Figure 2-4; Figure A-1). The tank was plumbed with a pump and 2.54-cm pipe to distribute water to TFC's teaching garden.

Figure A-1. The Free Clinic Site Layout

Appendix B

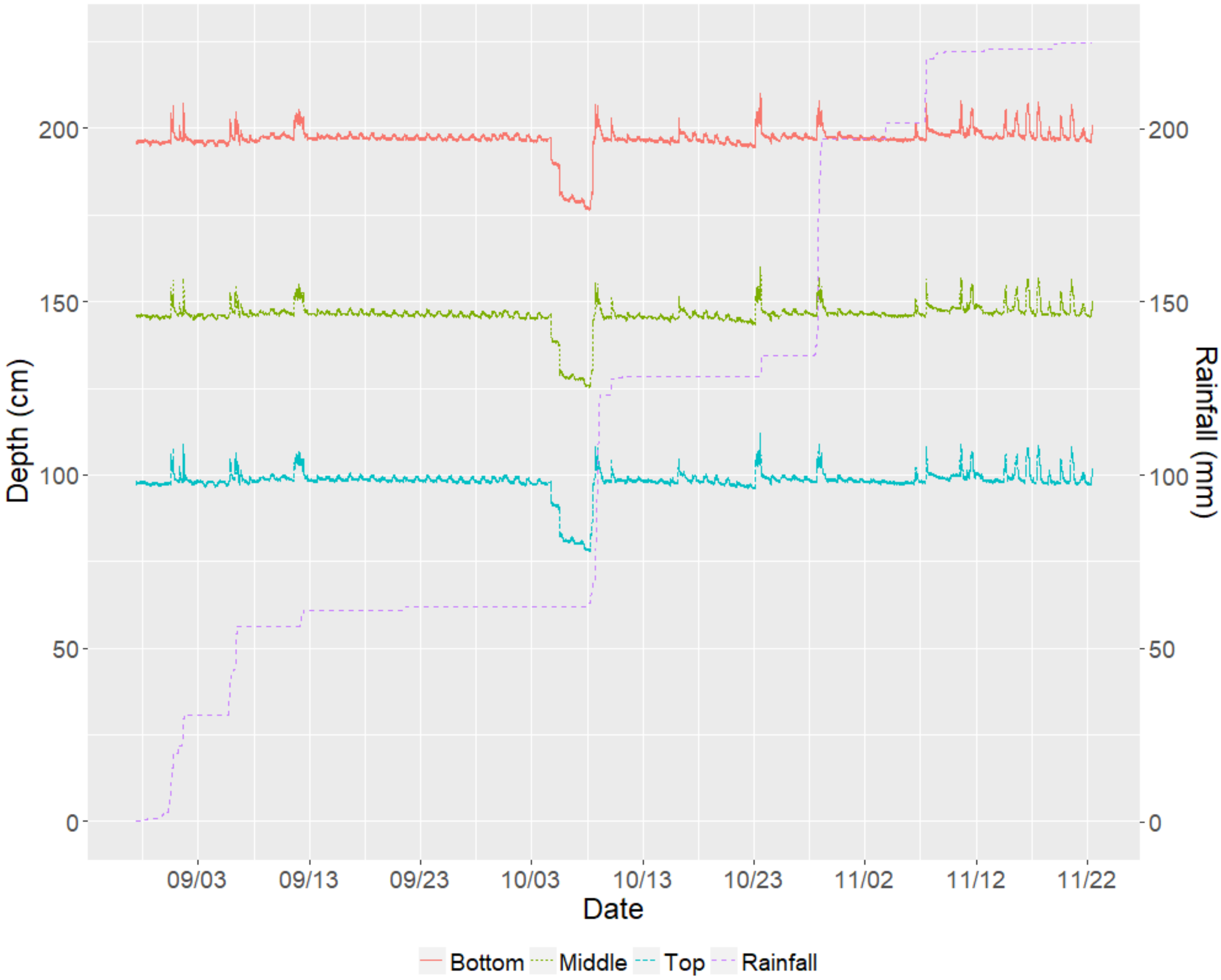


Figure A-2. Depth & Rainfall Plot

Appendix C

The invert height above the storage tank bottom was approximated using the median value of maximum bottom sensor depths for each dry (non-event) period to be 198.8-cm; bottom sensor depths greater than 198.8-cm represent system outflow.

Appendix D

All event flow data occurring between May 25, 2018, to July 7, 2018, (Figure A-3) produces a large gap in predictor values (Figure A-3). Removal of events less than or equal to 0.066 cms and averaging flows over 60-min intervals improved predictor normality, autocorrelation, and model fit (Figure 3-18).

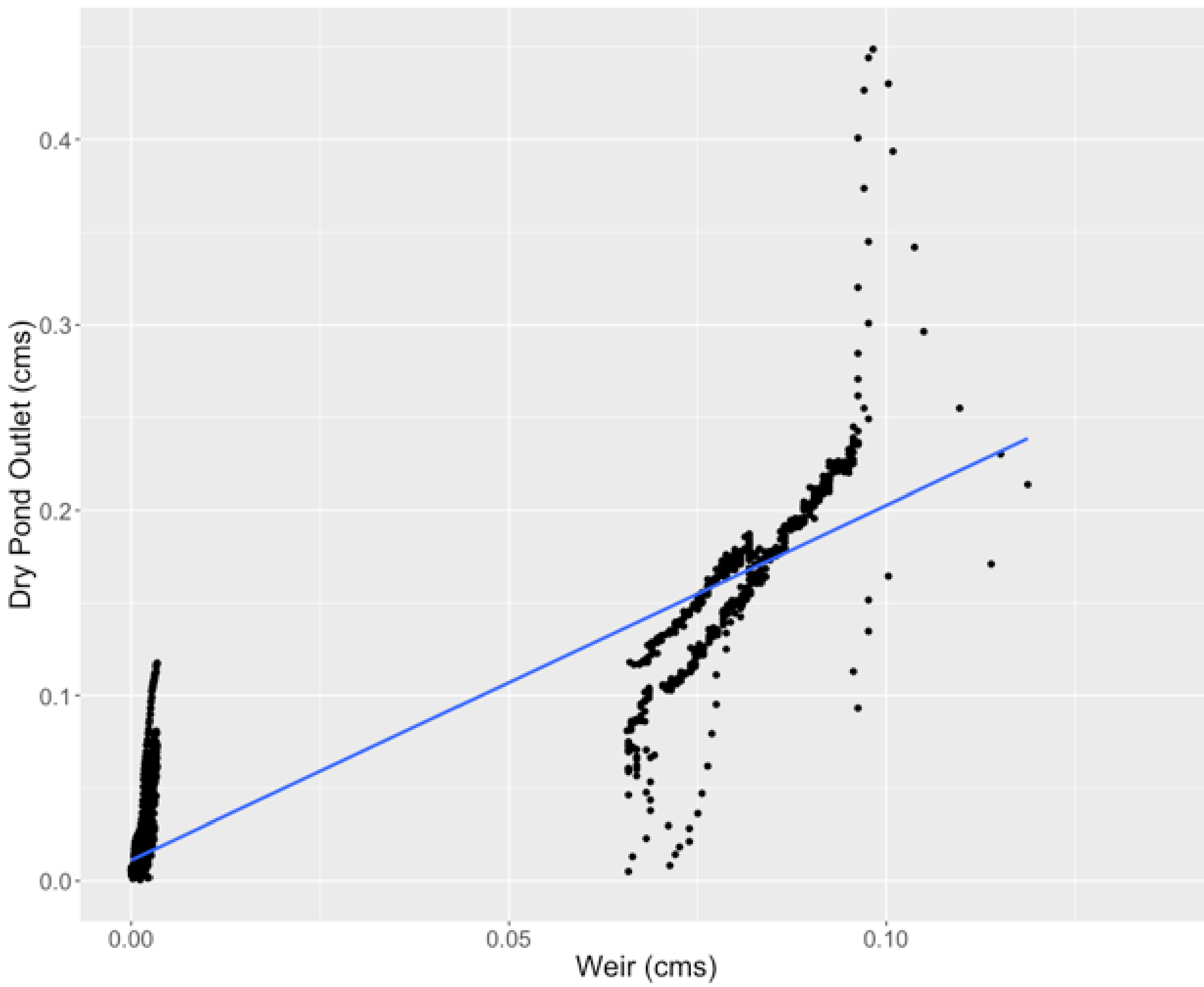
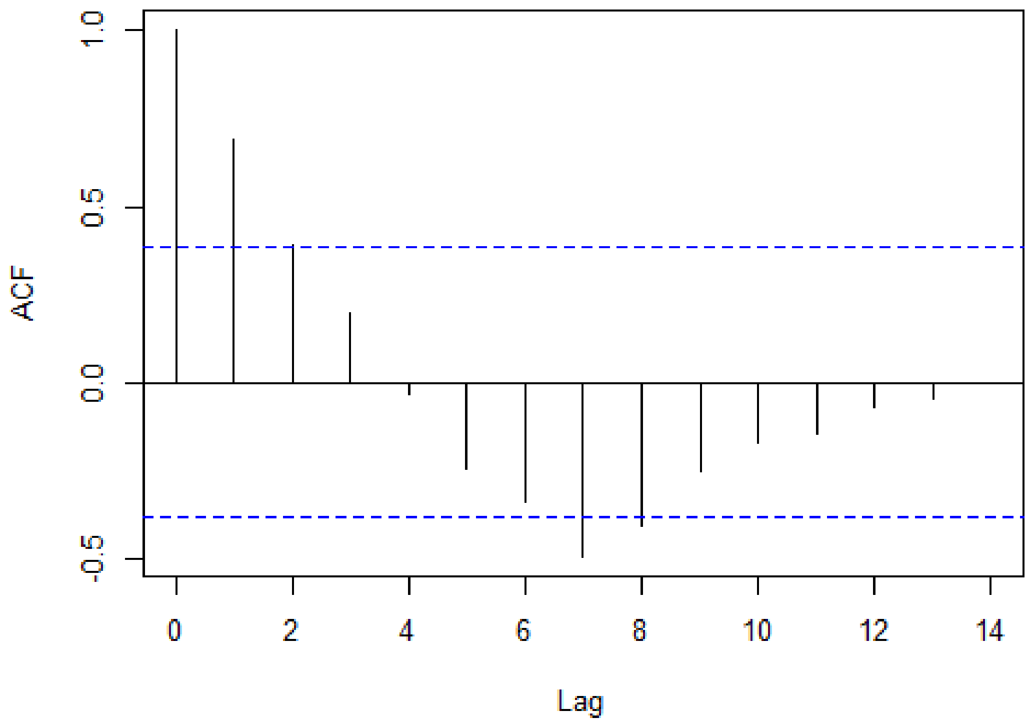


Figure A-3. IN1All Event Flow Data

Model autocorrelated was assessed using a correlogram; autocorrelation factor showed significance at lag 1, 2, 7, and 8 (Figure A-4 Top). Autocorrelation was then assessed using Runs Test and significant autocorrelation could not be confirmed (Figure A-4 Bottom).




```
        Runs Test - Two sided

data:  lm10.2$residuals
Standardized Runs Statistic = -1.6013, p-value = 0.1093
```

Figure A-4. IN1 Linear Correction Model Autocorrelation Factor (Top); Runs Test (Bottom)

IN1 model diagnostic plots allow visual assessment of model assumptions (Figure A-5). Model Residuals vs. Fitted values and Scale-Location assess model homoscedasticity. Generally, some patterns exist (residual and standardized residual decrease as fitted values increase); however, these trends are not extreme, so model homoscedasticity is accepted. The trends in model residuals are a result of more fitted values at low flows and decreasing number of fitted values as flow rate increases. Normal quantile-quantile plots assess model normality. The Normal Q-Q plot shows some deviation from the theoretical quantiles; however, some deviation is expected. Predictor and response values were assessed with the Shapiro-Wilks Normality test and normality was confirmed. Residuals vs. Leverage highlight residuals with strong leverage over model fit using Cook’s Distance, a measure of residual leverage. Outlier removed did not substantially improve model fit, so outliers were not removed.

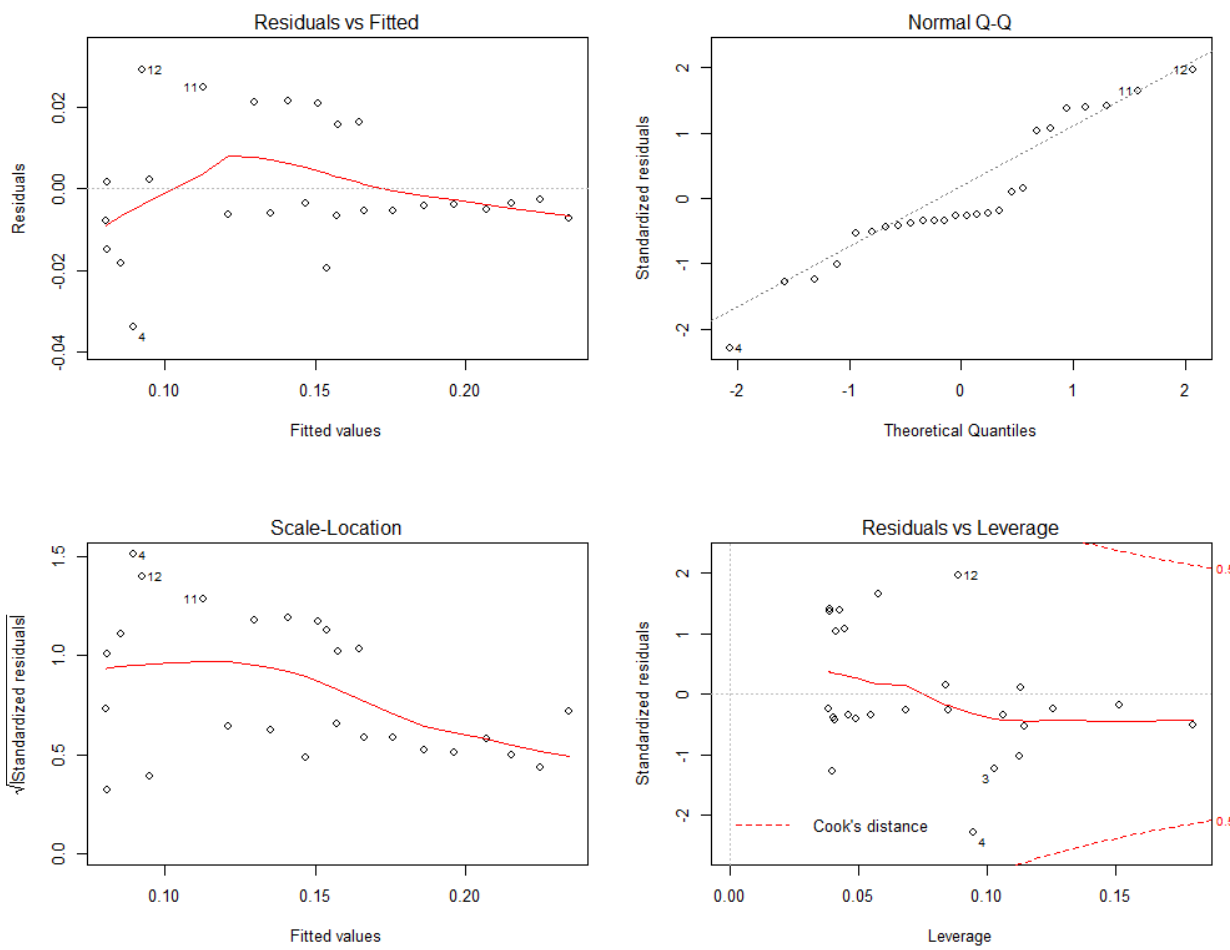


Figure A-5. IN Linear Correction Model Diagnostic Plots

The IN1 event flow linear correction model, $y = 5.31x - 0.27$ has a R^2 value of 0.91 and a residual standard error (RSE) of 0.016 (Figure A-6).

```
Call:
lm(formula = dryout ~ weir, data = flow.short1.corr)

Residuals:
      Min        1Q    Median        3Q       Max 
-0.034011 -0.006542 -0.004001  0.012509  0.029335 

Coefficients:
            Estimate Std. Error t value Pr(>|t|)    
(Intercept) -0.26997    0.02753  -9.806 7.20e-10 ***
weir         5.30783    0.34953  15.185 8.31e-14 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 0.01563 on 24 degrees of freedom
Multiple R-squared:  0.9057,	Adjusted R-squared:  0.9018 
F-statistic: 230.6 on 1 and 24 DF,  p-value: 8.306e-14
```

Figure A-6. IN1 Linear Correction Model Summary

Appendix E

All base flow data occurring between May 25, 2018, to July 7, 2018, produces substantial scatter and poor model fit (Figure A-7). Baseflow periods less than 1 day were removed from model creation to prevent higher flow rate outliers resulting from recent storm flows. Baseflow periods used in model creation varied in duration (1.4 – 9 days) and flow rate (0.0 – 0.0012 cms). Predictor and response values were averaged daily to improve model normality and minimize autocorrelation (Figure 3-21). Daily averaging of predictor and response values eliminated significant autocorrelation (Figure A-8).

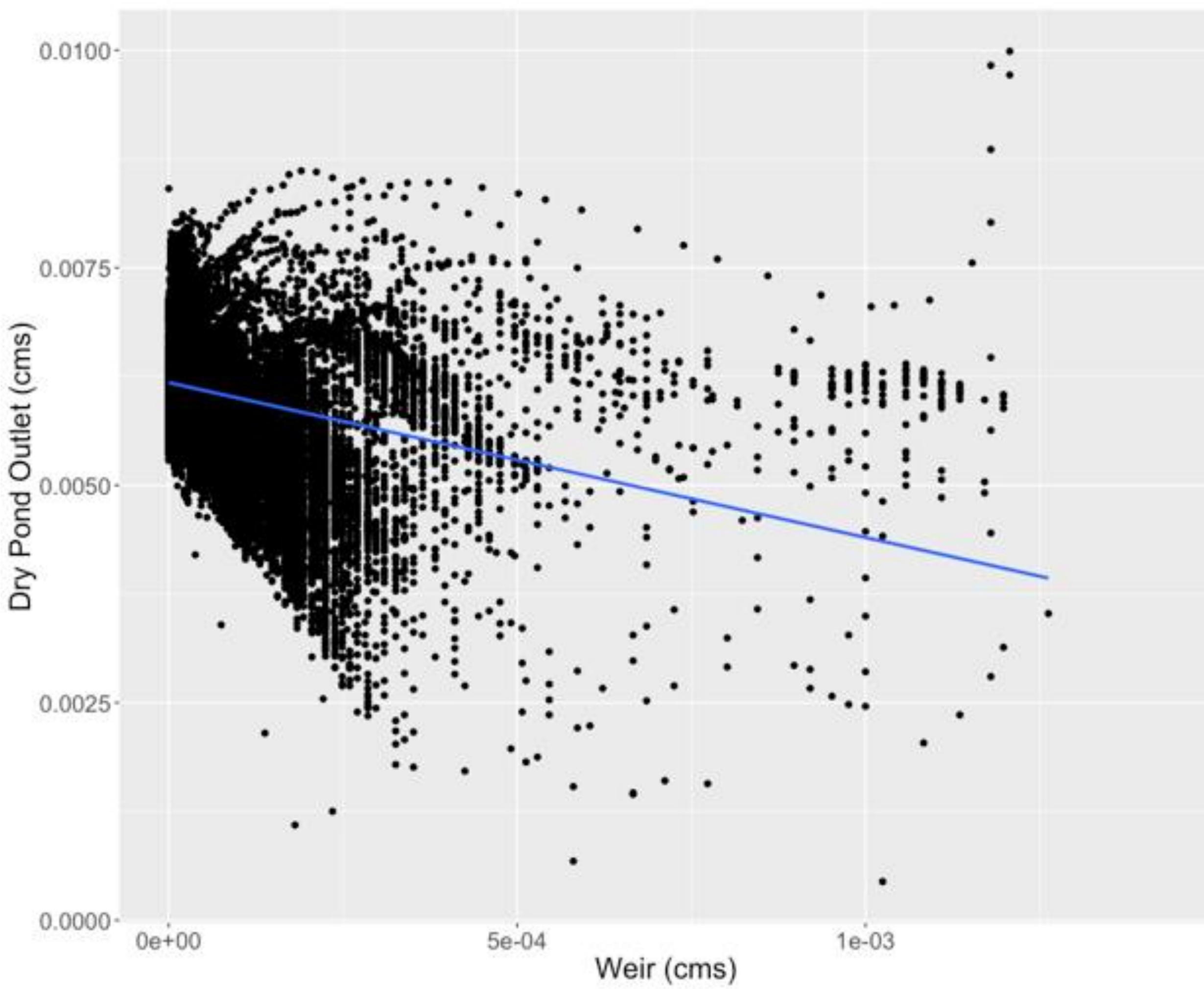


Figure A-7. Base Flow Correction Model

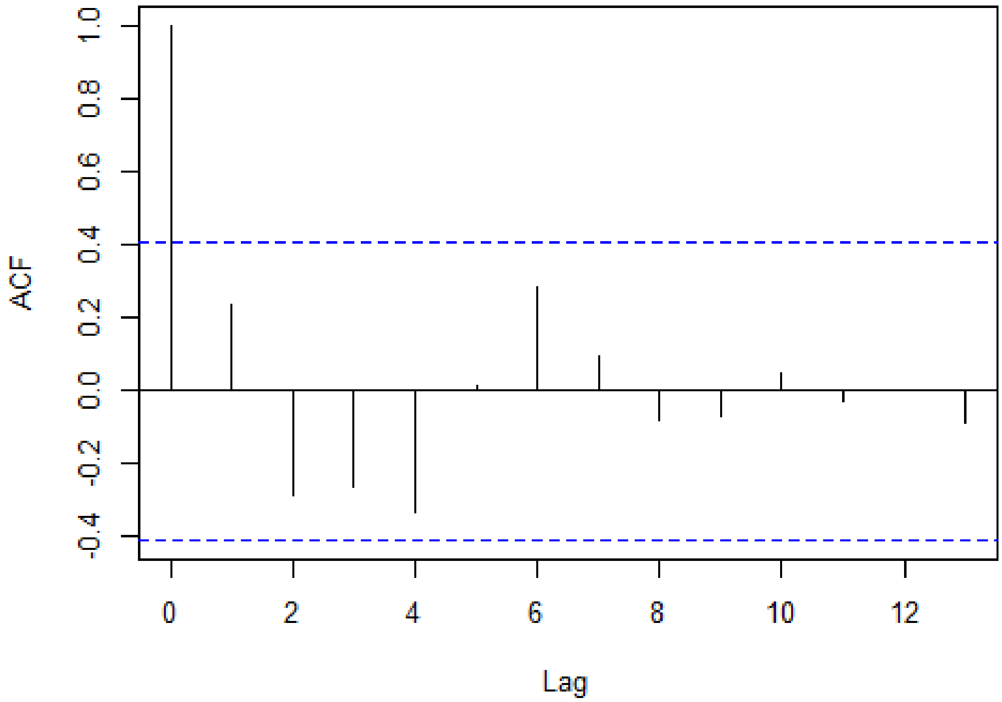


Figure A-8. Base Flow Correction Autocorrelation Factor

IN1 base flow model diagnostic plots allow visual assessment of model assumptions (Figure A-8; Figure A-9). Model Residuals vs. Fitted values and Scale-Location assess model homoscedasticity; generally, no strong patterns exist so model homoscedasticity is acceptable. The Normal Q-Q plots shows some deviation from the theoretical quantiles; however, some deviation is expected. One outlier was removed which sustainably improved model fit; the remaining outliers were not removed due to the low number of predictor values they were deemed more valuable to model fit.

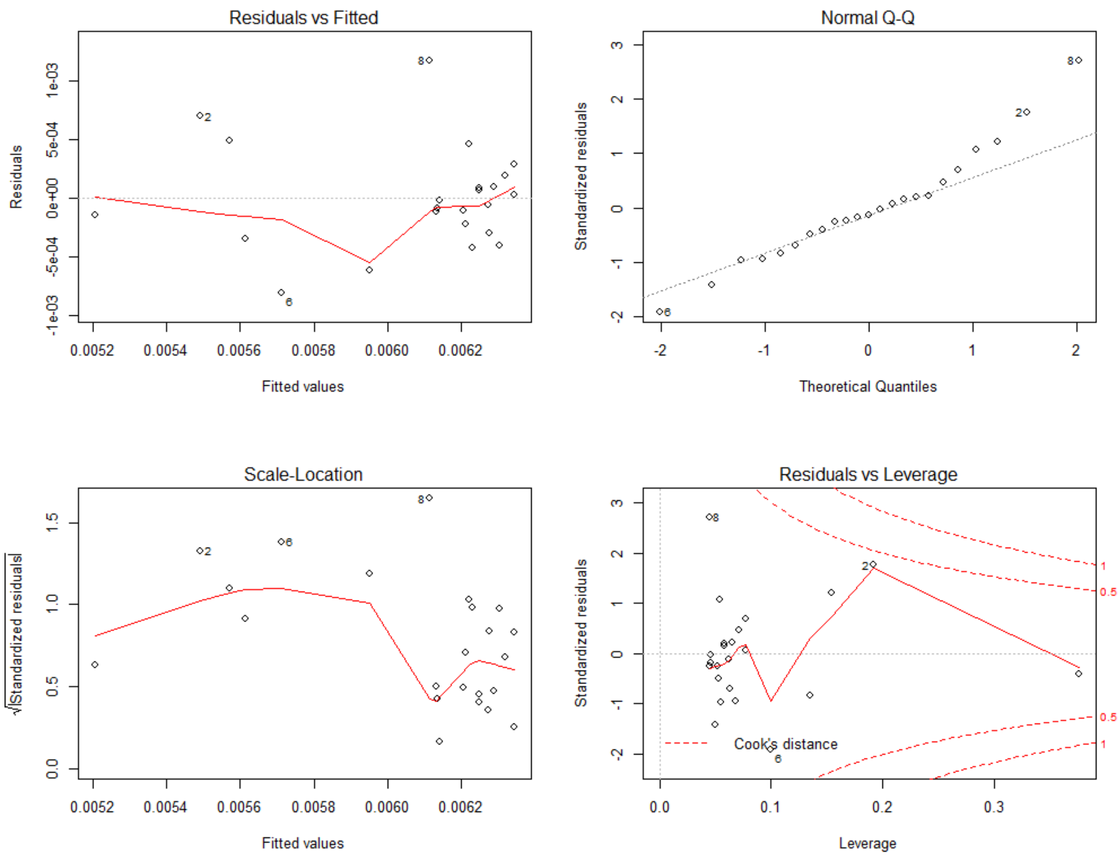


Figure A-9. Base Flow Model Diagnostic Plots

The IN1 base flow linear correction model, $y = 0.006 - 4.05x$ has a R^2 value of 0.35 and a RSE of 0.0004 (Figure A-10).

```
Call:
lm(formula = dryout ~ weir, data = DS2.baseflow.corr[-c(1), ])

Residuals:
       Min         1Q     Median         3Q        Max 
-8.049e-04 -2.567e-04 -5.417e-05  1.455e-04  1.174e-03 

Coefficients:
              Estimate Std. Error t value Pr(>|t|)    
(Intercept)  0.0063606  0.0001263  50.344  < 2e-16 ***
weir        -4.0539099  1.1996529  -3.379  0.00283 ** 
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 0.0004435 on 21 degrees of freedom
Multiple R-squared:  0.3522,	Adjusted R-squared:  0.3214 
F-statistic: 11.42 on 1 and 21 DF,  p-value: 0.002833
```

Figure A-10. IN 1 Base Flow Linear Correction Model

Appendix F

Hourly and daily well stage measurements and recession rates were neither normal or log-normal (Figure A-11); daily average well stage deviated from Normal quantile-quantile at the tails exhibiting some non-normality of residuals. Residual vs. Fitted, Scale-Location, and Residuals vs. Leverage for spring-summer (Figure A-11 Top) do not show strong trends in residual fit over the range of fitted values while autumn-winter (Figure A-11 Bottom) may show some evidence of "trumpeting" or increased residuals as fitted values increase. Several outliers exhibit substantial leverage; however, outlier removal did not improve model fit so no outliers were removed.

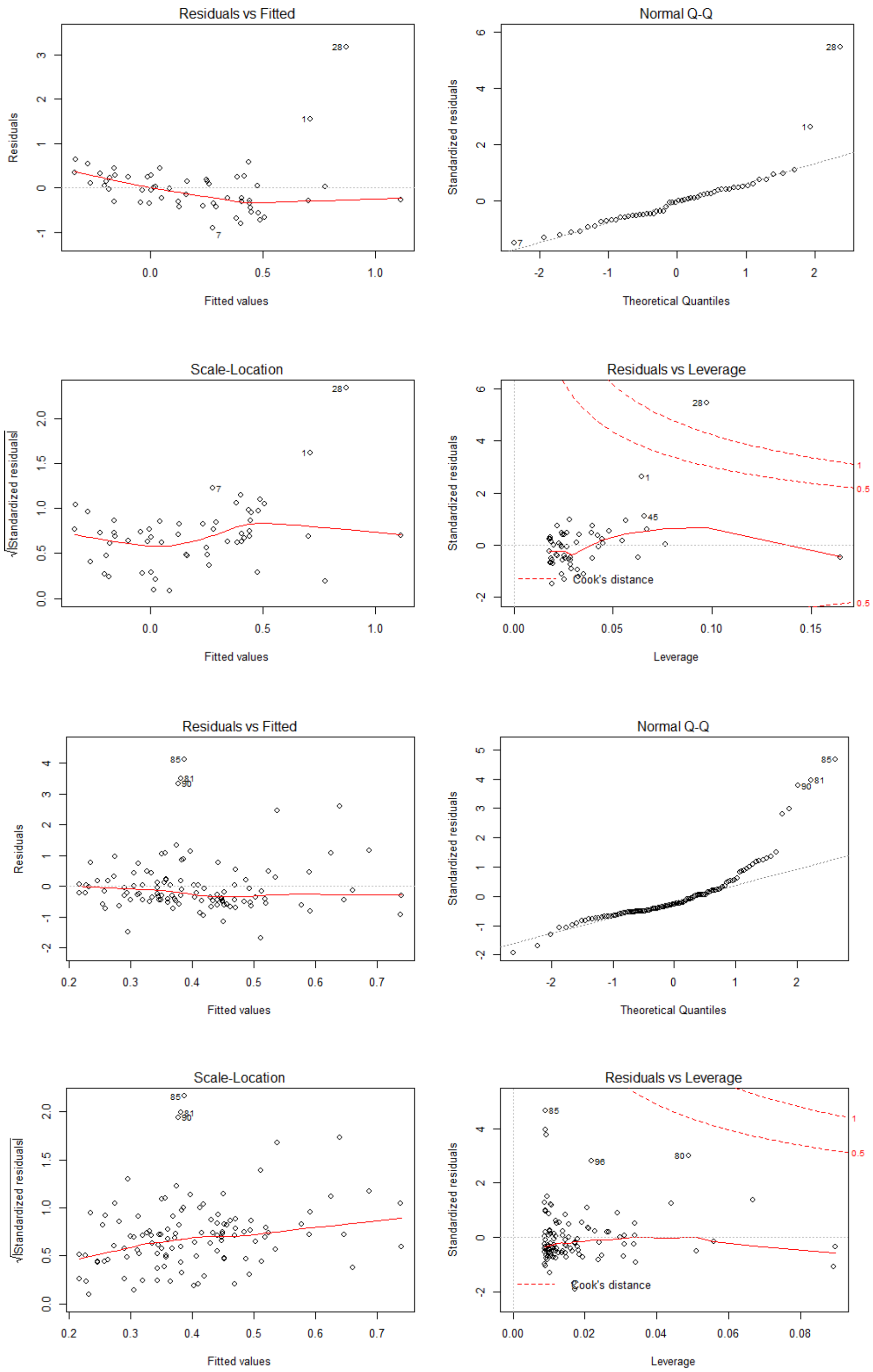


Figure A-11. Well Recession Linear Model Diagnostic Plots; Spring-Summer (Top), Autumn-Winter (Bottom)

Residual autocorrelation was not significant in well recession for spring-summer periods; however, significant autocorrelation was present in autumn-winter recession using plots of autocorrelation factor versus lag plots and confirmed with the Durbin-Watson test, and Run's test for autumn-winter well recession (Figure A-12; Figure A-13).

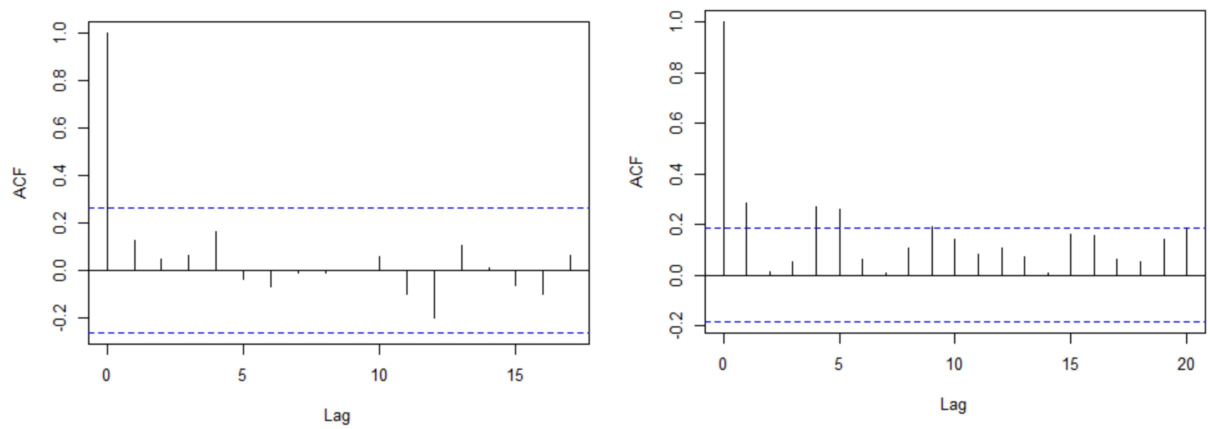
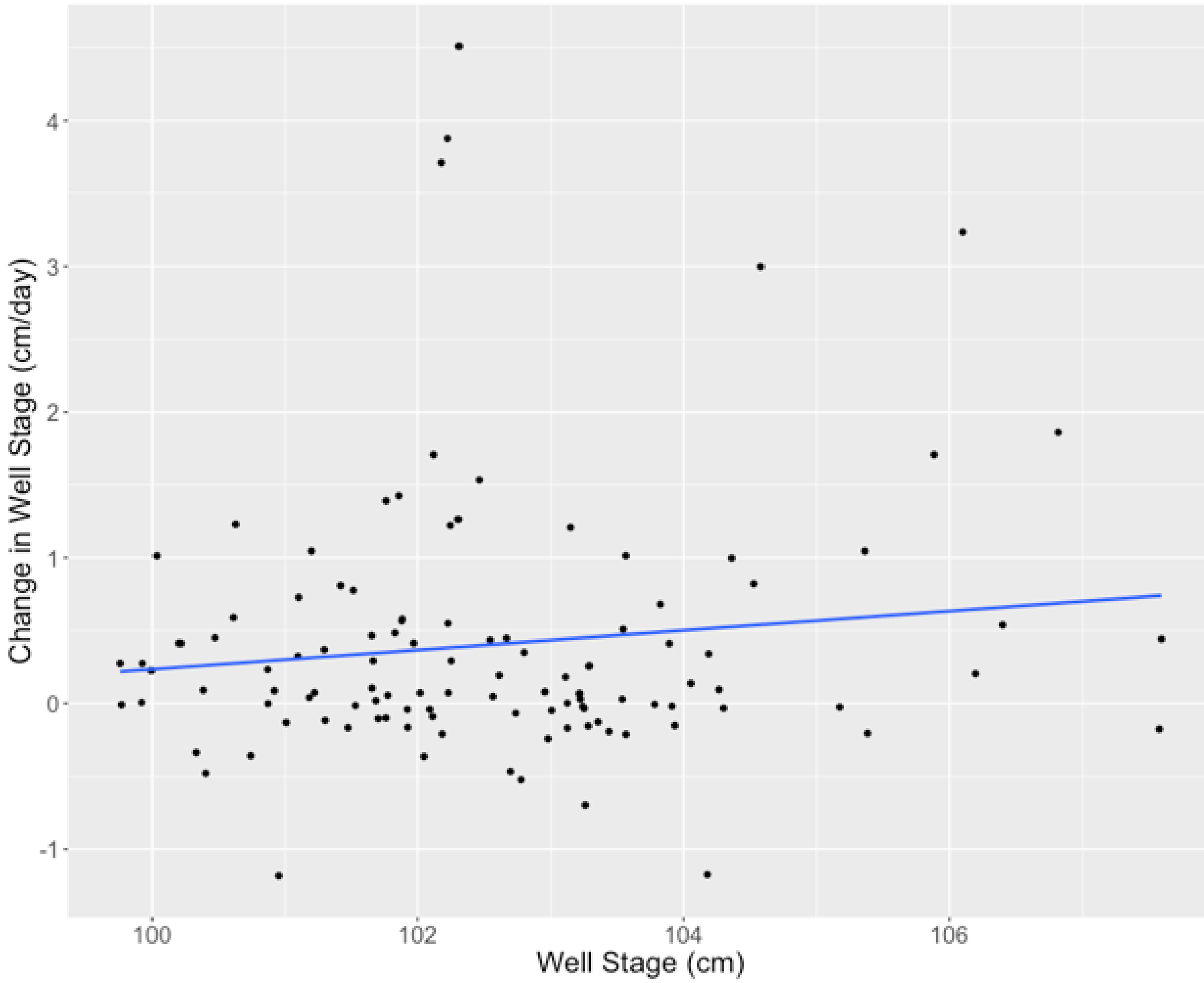


Figure A-12. Well Recession Autocorrelation Factor; Spring-Summer (Left), Autumn-Winter (Right)

```
        Durbin-Watson test

data:  lmbfaut
DW = 1.4151, p-value = 0.0006461
alternative hypothesis: true autocorrelation is greater than 0

        Runs Test - Two sided

data:  lmbfaut$residuals
Standardized Runs Statistic = -2.2578, p-value = 0.02396
```

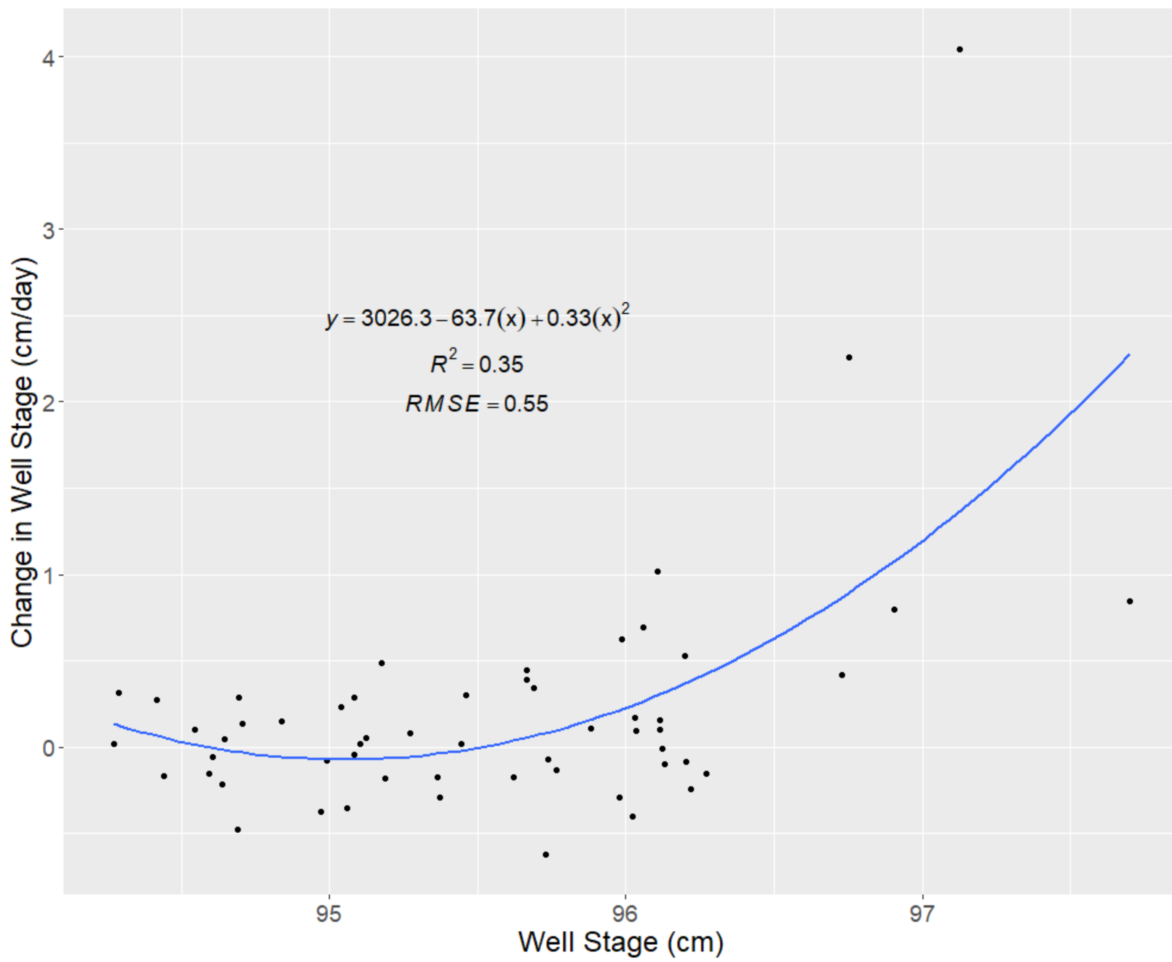
Figure A-13. Autumn-Winter Well Stage Recession Autocorrelation

Model fit of well stage recession for autumn-winter period is unacceptable (R^2=0.02) (Figure A-14); therefore, autocorrelation was not be corrected. Model fit of well stage recession for spring-summer period was best using a quadratic expression (R^2=0.35) (Figure A-15). This model provides an estimation of exfiltration to sub-soils.




```
Call:
lm(formula = delta.stage ~ well.cm, data = autwin[, ])

Residuals:
    Min      1Q  Median      3Q     Max 
-1.6868 -0.4700 -0.2333  0.1806  4.1239 

Coefficients:
            Estimate Std. Error t value Pr(>|t|)
(Intercept) -6.45270    5.14711  -1.254    0.213
well.cm      0.06685    0.05018   1.332    0.186

Residual standard error: 0.888 on 112 degrees of freedom
Multiple R-squared:  0.0156,    Adjusted R-squared:  0.006809 
F-statistic: 1.775 on 1 and 112 DF,  p-value: 0.1855
```

Figure A-14. Autumn-Winter Well Recession (Top); Model Summary (Bottom)




```
Call:
lm(formula = delta.stage ~ well.cm + I(well.cm^2), data = sprsum[,
    ])

Residuals:
     Min       1Q   Median       3Q      Max
-1.43235 -0.25757 -0.06632  0.20755  2.67594

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)  3026.2756   929.1180   3.257  0.00197 **
well.cm       -63.6757    19.4205  -3.279  0.00184 **
I(well.cm^2)    0.3349     0.1015   3.301  0.00173 **
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.5615 on 53 degrees of freedom
Multiple R-squared:  0.3546,    Adjusted R-squared:  0.3302
F-statistic: 14.56 on 2 and 53 DF,  p-value: 9.131e-06
```

Figure A-15. Spring-Summer Well Recession (Top); Model Summary (Bottom)

Appendix G

TSS RC ratios were determined to be log-normal; diagnostic plots of log accumulation compared to log TSS RC ratio show some variation in Residual vs. Fitted and Scale-Location plots, however, variation is not unacceptable (Figure A-16).

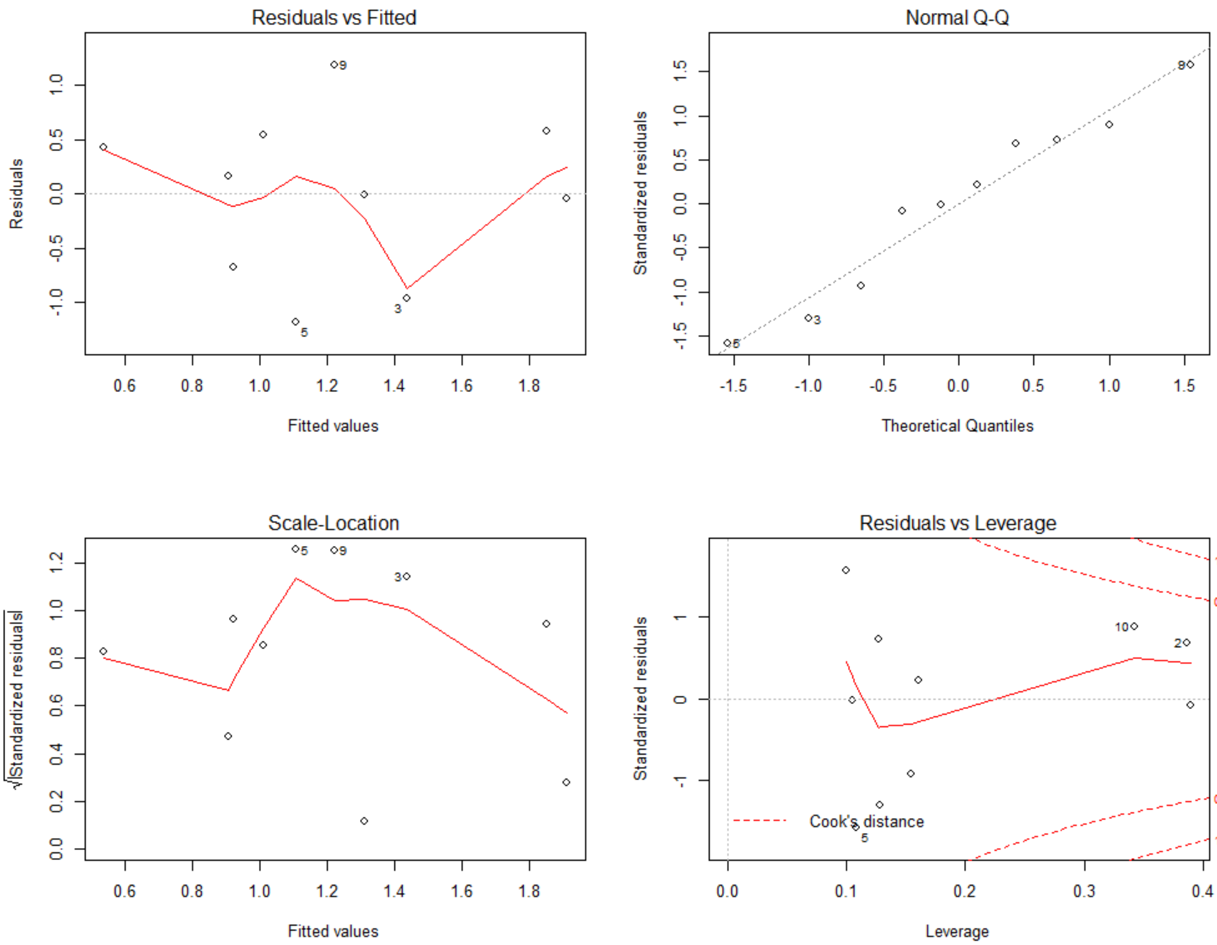


Figure A-16. Log Accumulation vs. TSS Log RC Ratio

Log accumulation versus log TSS RC ratio model summary (Figure A-17 and Figure A-18) highlights poor model fit (R^2= 0.25), insignificant intercept, and slope (1.1). The slope parameter suggests there is insufficient evidence to conclude that the proportion of influent TSS treated decreased as storm size increased; this could be the result of some up-stream flow control or small sample size (n=10).

```
Call:
lm(formula = log(TSS.x/TSS.y) ~ log(Accumulation), data = tot.wq)

Residuals:
     Min       1Q   Median       3Q      Max 
-1.18675 -0.52004  0.07562  0.51314  1.18810 

Coefficients:
                  Estimate Std. Error t value Pr(>|t|)
(Intercept)        -2.2762     2.1790  -1.045    0.327
log(Accumulation)   1.1058     0.6842   1.616    0.145

Residual standard error: 0.7952 on 8 degrees of freedom
Multiple R-squared:  0.2461,	Adjusted R-squared:  0.1519 
F-statistic: 2.612 on 1 and 8 DF,  p-value: 0.1447
```

Figure A-17. Log Accumulation vs. TSS Log RC Ratio Model Summary

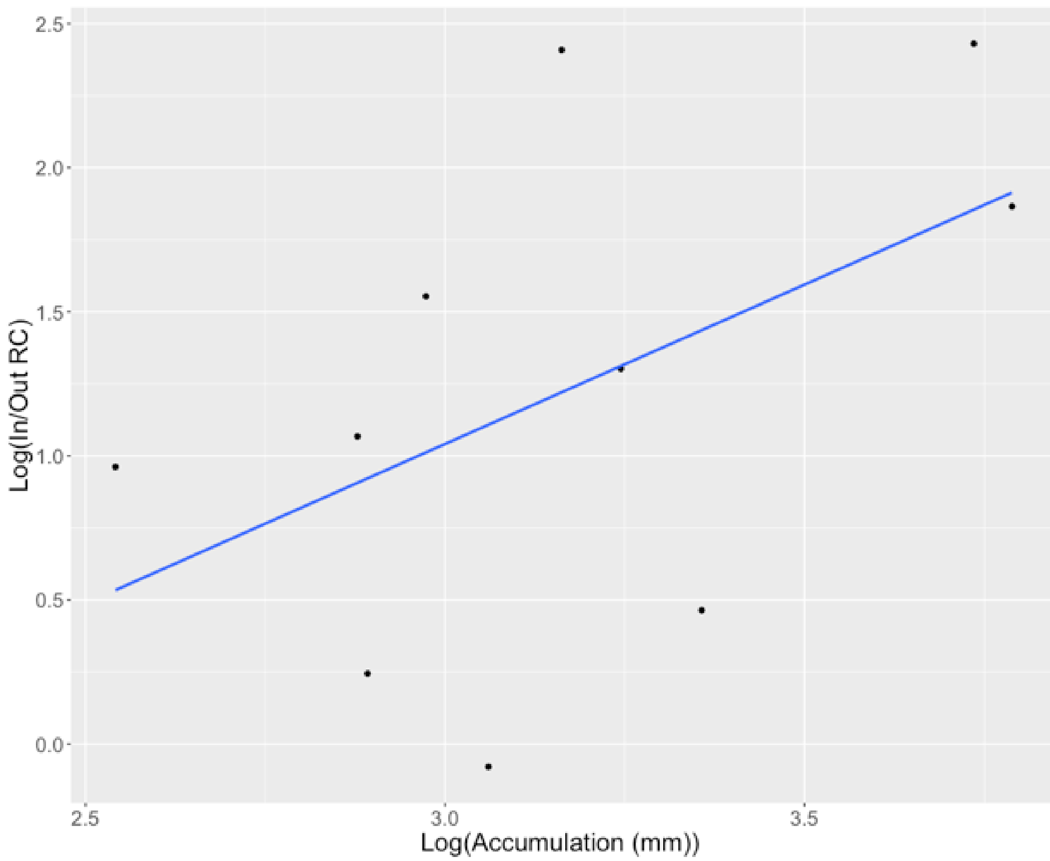


Figure A-18. Regression Analysis TSS; Comparing Log Accumulation (mm) to Log TSS RC Ratio

Appendix H

PBP RC ratios were determined to be log-normal; diagnostic plots of Log accumulation compared to Log PBP RC ratio show some variation in Residual vs. Fitted and Scale-Location plots, however, variation is not unacceptable (Figure A-19).

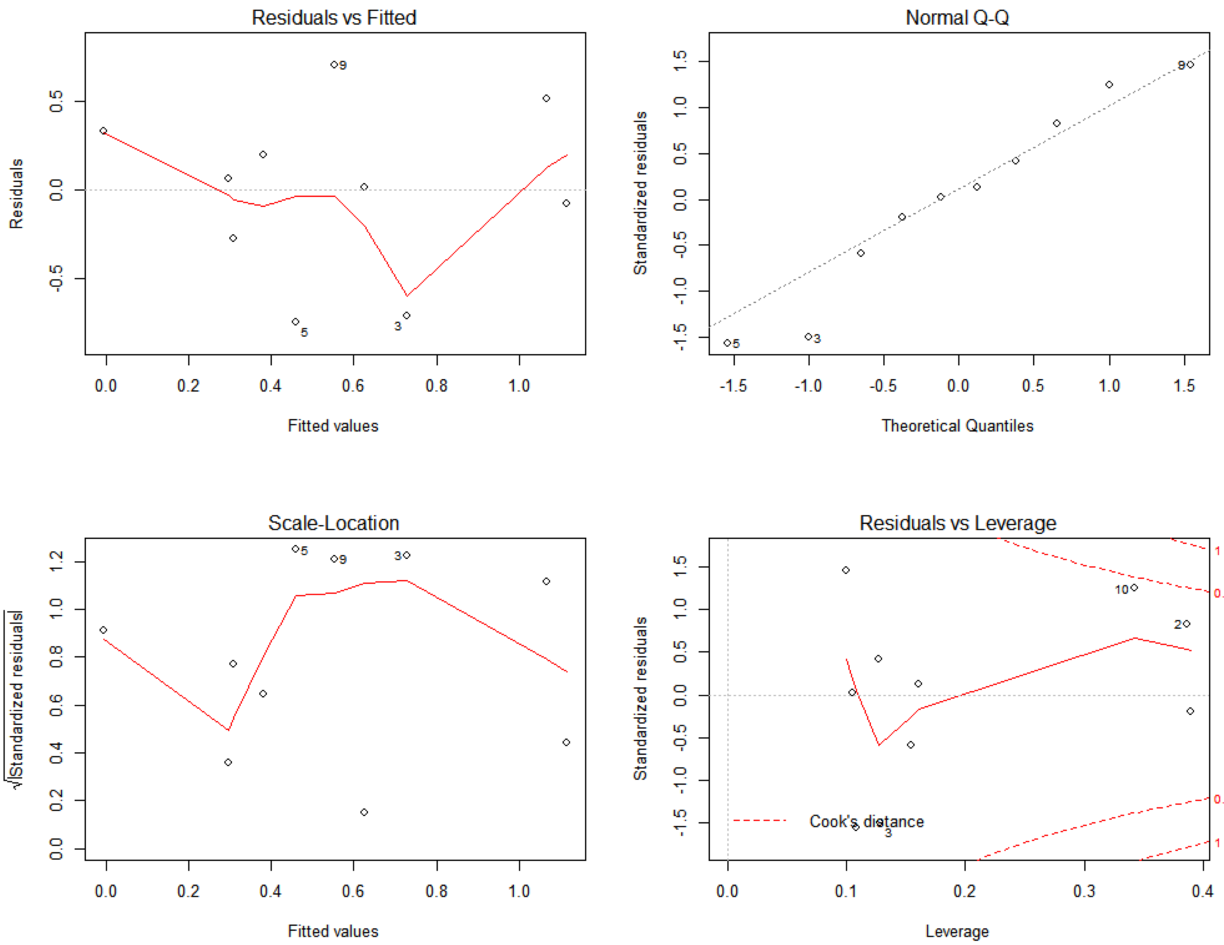


Figure A-19. Log Accumulation vs. PBP Log RC Ratio

Log PBP RC ratio versus Log accumulation model summary (Figure A-20 and Figure A-21) highlights poor model fit (R^2= 0.35), insignificant intercept and slope (0.9). The slope parameter suggests there is insufficient evidence to conclude that the proportion of influent TSS treated decreased as storm size increased; this could be the result of some up-stream flow control or small sample size (n=10).

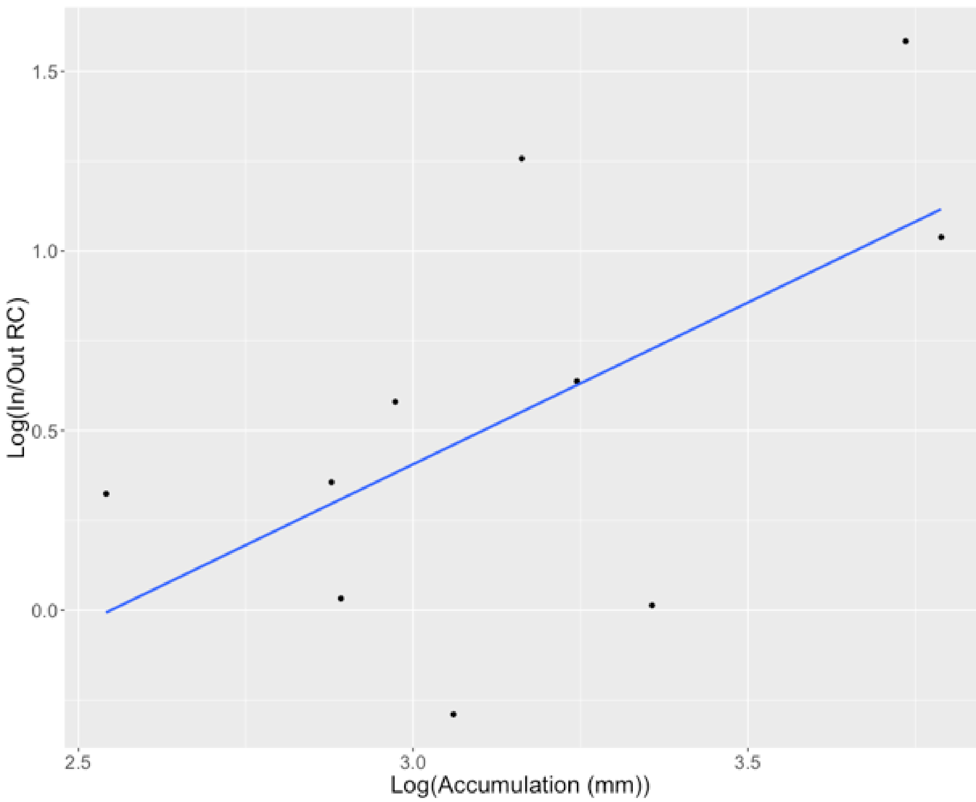


Figure A-20. Regression Analysis PBP; Comparing Log(Accumulation (mm)) to Log PBP RC Ratio

```
Call:
lm(formula = log(PBP.x/PBP.y) ~ log(Accumulation), data = tot.wq)

Residuals:
     Min       1Q   Median       3Q      Max 
-0.75044 -0.22709  0.03532  0.29735  0.70505 

Coefficients:
                  Estimate Std. Error t value Pr(>|t|)  
(Intercept)        -2.2949     1.3941  -1.646   0.1384  
log(Accumulation)   0.9003     0.4378   2.057   0.0737 .
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 0.5087 on 8 degrees of freedom
Multiple R-squared:  0.3459,    Adjusted R-squared:  0.2641 
F-statistic:  4.23 on 1 and 8 DF,  p-value: 0.07375
```

Figure A-21. Log Accumulation vs. PBP Log RC Ratio Model Summary