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Influence of Structural Disturbance on Stream Function and Macroinvertebrate Communities in Upper Coastal Plain Headwater Streams

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Richard Andrew Biemiller, Student

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Influence of structural disturbance on stream function and macroinvertebrate communities in Upper Coastal Plain headwater streams

DISSERTATION

A dissertation submitted in partial fulfillment of the
requirements for the degree of Doctor of Philosophy in the
College of Agriculture, Department of Entomology,
at the University of Kentucky

By
Richard A. Biemiller

Lexington, Kentucky

Director: Dr. Chris Barton, Professor of Hydrology and Watershed Management

Lexington, Kentucky

2015

ABSTRACT OF DISSERTATION

INFLUENCE OF STRUCTURAL DISTURBANCE ON STREAM FUNCTION AND MACROINVERTEBRATE COMMUNITIES IN UPPER COASTAL PLAIN HEADWATER STREAMS

Freshwater is a resource under threat due to anthropogenic actions. Stream restoration is a common method for mitigating disturbance. Inconsistent methodologies for evaluating restoration need have drawn criticism. Limited use of baseline data guiding stream restoration activities is of particular concern. This study was developed to elucidate metrics that differentiate reference and disturbed sites in Upper Coastal Plain streams. This information could improve resource use and success rates of restorations. Structural and functional variables were examined in 10 reference and 10 streams that meet the traditional definition of disturbance and would be restoration priorities. Disturbed streams were classified into two regimes, temporal, based on time since disturbance, and categorical, based on disturbance cause. Some metrics of geomorphology, water chemistry and macroinvertebrates differentiated reference from disturbed regimes and while other metrics separated streams within disturbance regimes. Surprisingly, leaf decay rate was not an effective metric for determining disturbance. However, macroinvertebrate leaf pack colonizers were found to be useful for differentiating reference sites and disturbance regimes. Of the 10 disturbed streams this study examined, my data suggests that only three are in immediate need of restoration. This study emphasizes the importance of baseline data and its potential benefits for guiding stream restoration.

KEYWORDS: Stream, Macroinvertebrates, Restoration, Reference, Disturbance, Leaf packs

Richard Andrew Biemiller
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Student's Signature

April 20, 2016
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Date

INFLUENCE OF STRUCTURAL DISTURBANCE ON STREAM FUNCTION AND MACROINVERTEBRATE COMMUNITIES IN UPPER COASTAL PLAIN HEADWATER STREAMS

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Dedicated to my family and friends especially Cletus Long who passed away during this study.

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Disclaimer

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Chapter 1 Overall Introduction

Freshwater has the potential to be the next major limiting resource for the human population (Vorosomarty et al 2000). Conservative estimates expect half of the world's population to be impacted by some type of water stress by 2025 (Dudgeon et al 2005). Beyond its importance to humans, freshwater is the home to 25% of described vertebrates and 40% of known fishes along with many juvenile invertebrates and a disproportionate number of endemic species (Vorosmarty et al 2010). While the importance of freshwater is clear, at least 80% of streams have been negatively impacted by human activity (Dudgeon et al 2005).

Recently, stream restoration has become an accepted method to deal with human disturbance of waterways. Billions of dollars have been spent on stream restoration projects in the United States alone (Jahnig et al 2011; Palmer et al 2005). In fact, by 2005 about 860 documented stream restoration projects had been completed in four Southeastern states (KY, GA, NC, SC) costing a total of over 860 million dollars U.S. (Suddeth 2007). As with many new sciences, stream restoration has encountered its share of challenges and problems. Questions have been raised regarding nearly all aspects of restoration projects including, but not limited to, the choice of project sites and scale of effort (Jahnig et al 2010), the paucity of pre-project baseline data (Downs et al 2011), misunderstandings of the relationship between stream structure (the patterns or organization of features within a system) and stream function (processes and rates of a system) (Fritz et al 2010), the lack of clearly defined goals (McMillan and Vidon 2014), the effectiveness of restorations (Palmer et al 2005), and the need for post project monitoring (Downs and Kondolf 2002). It has been suggested that an improved understanding of the state of the stream prior to implementation of any restoration activity could better inform site selection, choice of methods, allow for more site specific, biologically relevant and attainable goals and narrow the necessary post project monitoring (Downs et al 2011; Palmer et al 2007). A common way to evaluate the current condition of a potential restoration site is to compare it to an existing reference stream in a similar environment (Kosnicki et al 2014).

In order to test the relationships of reference and disturbed sites, flow charts were developed (modified from Royer and Minshall 2003) showing the expected relationships of variables in reference, and streams disturbed by either dams, channelization or run-off (Figures 1-4). Reference streams function in a way that allows them to compensate alteration for one or two factors. There are feedback loops that help maintain the stability of reference reaches (Figure 1). Streams affected by impoundment suffer from lower flow and mineral precipitates falling out of the water column, due to stagnant water, reducing macroinvertebrate feeding and diversity (Figure 2). Channelized stream are by their nature disconnected from the flood plain. This causes higher flashiness, which can reduce habitat and feeding opportunities for macroinvertebrates lowering their diversity (Figure 3). Streams impacted by excessive run-off can be degraded through inputs from adjacent agricultural areas or impervious surfaces. In addition, they can often be disconnected from flood plains due to erosion.

One or all of these factors can have negative consequences on macroinvertebrate feeding and diversity (Figure 4).

This work compares a group of 10 reference streams to 10 disturbed streams defined by evaluations made by walking the length of streams and noting any visible disturbances (i.e. dam remnants, channelization or evidence of run-off). This visual method of assessing streams is often the method employed to determine condition (Downs et al 2011; Palmer et al 2007). The disturbed streams were further classified into temporal regimes based on the time of disturbance and categorical regimes based on the cause of disturbance in order to better discern which streams should be made priorities for restoration. A large suite of geomorphic, water quality, and biotic variables along with structural and functional variables were compared between the reference and 3 disturbance regimes in the hope of finding a common variable that could help readily identify restoration priorities. The ability to quickly prioritize restoration priorities and the current condition of disturbed streams in comparison to references streams could enhance future projects by informing managers on more appropriate goals, which would help with choosing methods increase success rates and narrow the scope of post project monitoring. Spending more money in the pre-project stage should be offset by more efficient spending during the implementation and monitoring phase of restoration projects resulting in an overall savings. While there will be differences in results dependent on geography, the overarching idea of the comparison of reference to disturbed sites should be transferable to any area with a suitable reference system.

Chapter 1 Figures

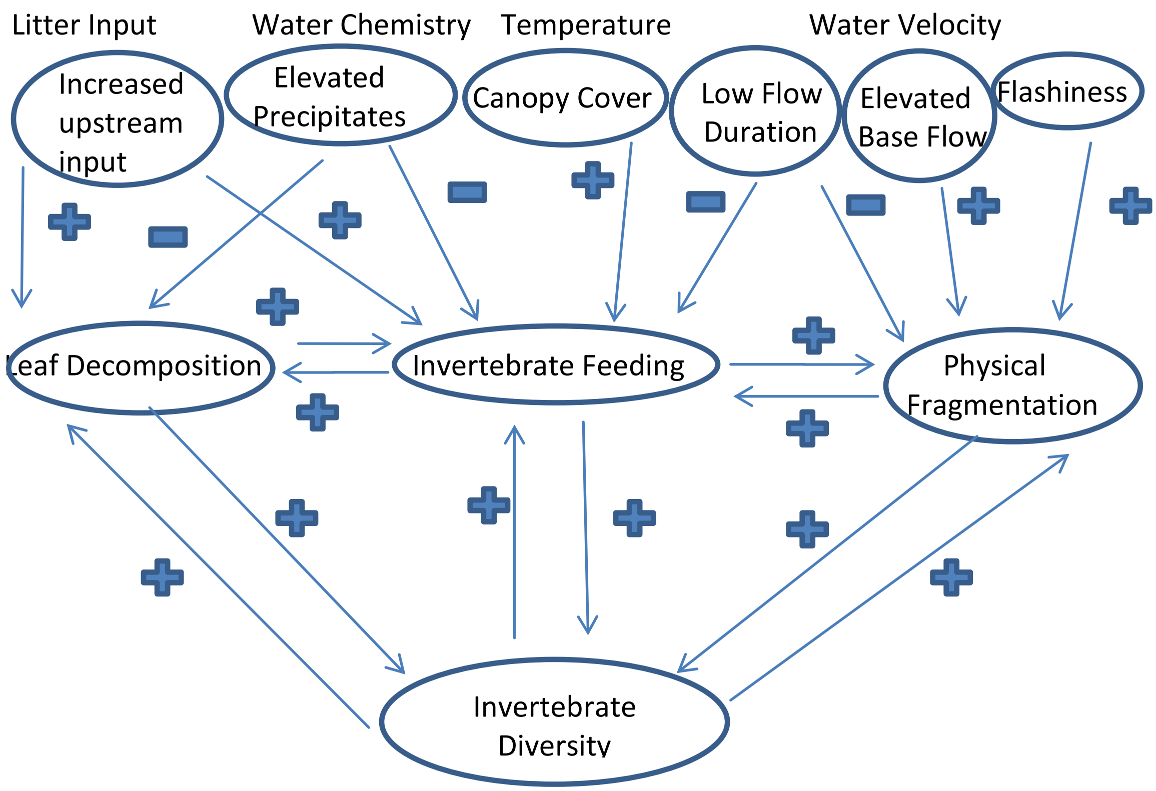

Figure 1) Reference site flow chart. Reference sites are those that have been least impacted by human activity. These streams would have overall higher invertebrate feeding opportunities due to leaf decomposition and fragmentation. Although low flow and elevated mineral precipitates could negatively impact feeding the other factors such as canopy cover and elevated base flow would override the negatives. In a reference system invertebrate diversity and feeding along with decomposition and physical fragmentation of leaves can create positive feedback loops in which the increase of one of the four can cause the increase of the others. These loops can become less evident or absent in disturbed streams.

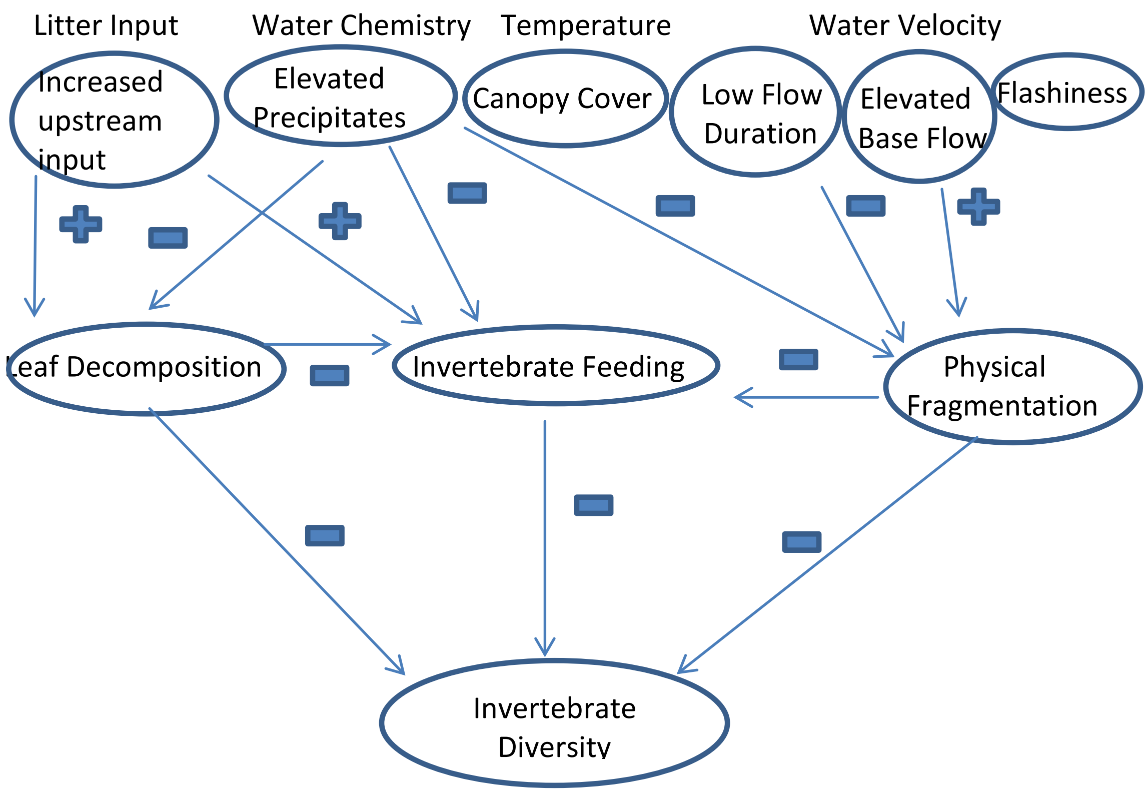

Figure 2) Areas affected by dams will often retain litter but the detritus is often buried in sediment caught behind the obstruction and are therefore inaccessible to macroinvertebrates. Mineral precipitates can fall out of the water column and gather, negatively affecting the macroinvertebrates and leaf decomposition. While base flow may be elevated due to the impoundment, the flow would remain low yielding very little change in physical fragmentation of detritus. Elevated precipitates and extended periods of low flow have a negative effect on both leaf decomposition and physical fragmentation. The combination of these factors decreases invertebrate feeding and diversity.

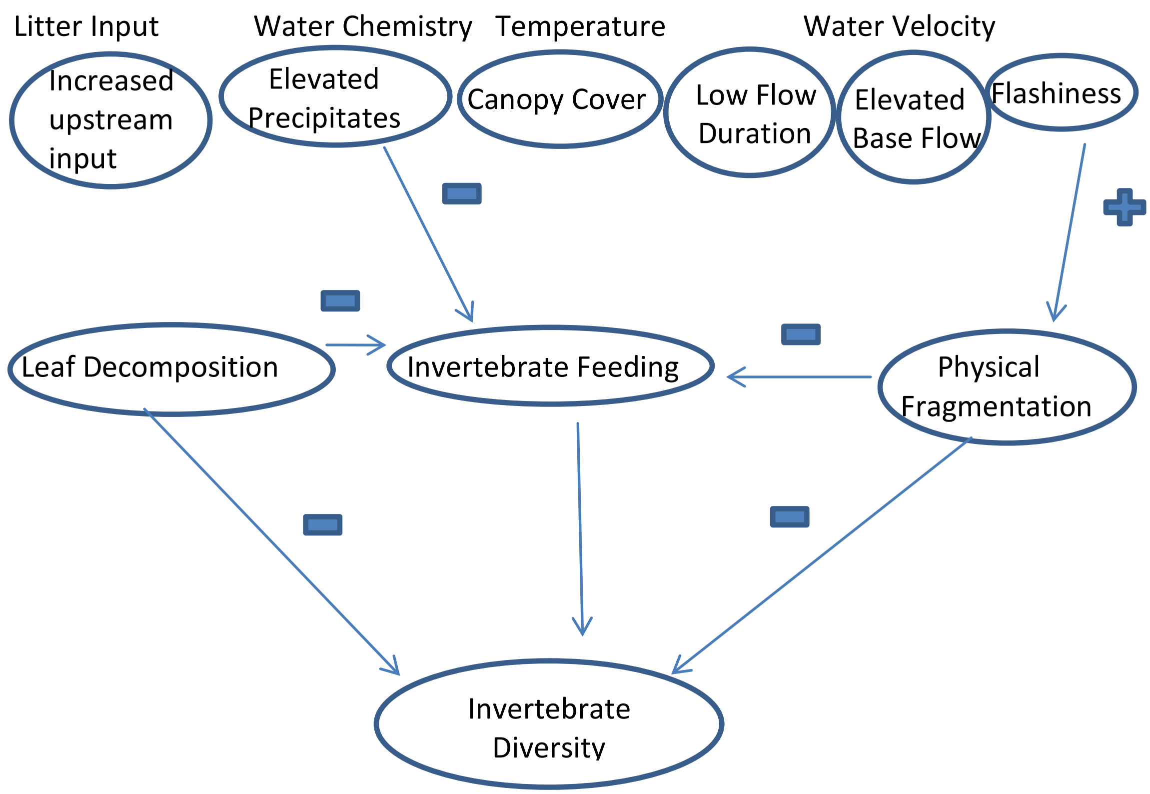

Figure 3) Channelizing or straightening streams can have several negative effects on streams. The most obvious effect is the increase in flashiness due the disconnection with the flood plain. This increase in flashiness can cause increased physical fragmentation of detritus and the flushing out of smaller leaf particles. This combination of these two elements can lead to reduced feeding opportunities and habitat for the invertebrates thereby decreasing the expected invertebrate diversity. Erosion of banks caused by the straightening of the channel and the increase in flashiness may lead to elevated precipitates causing decreased feeding again lowering invertebrate diversity.

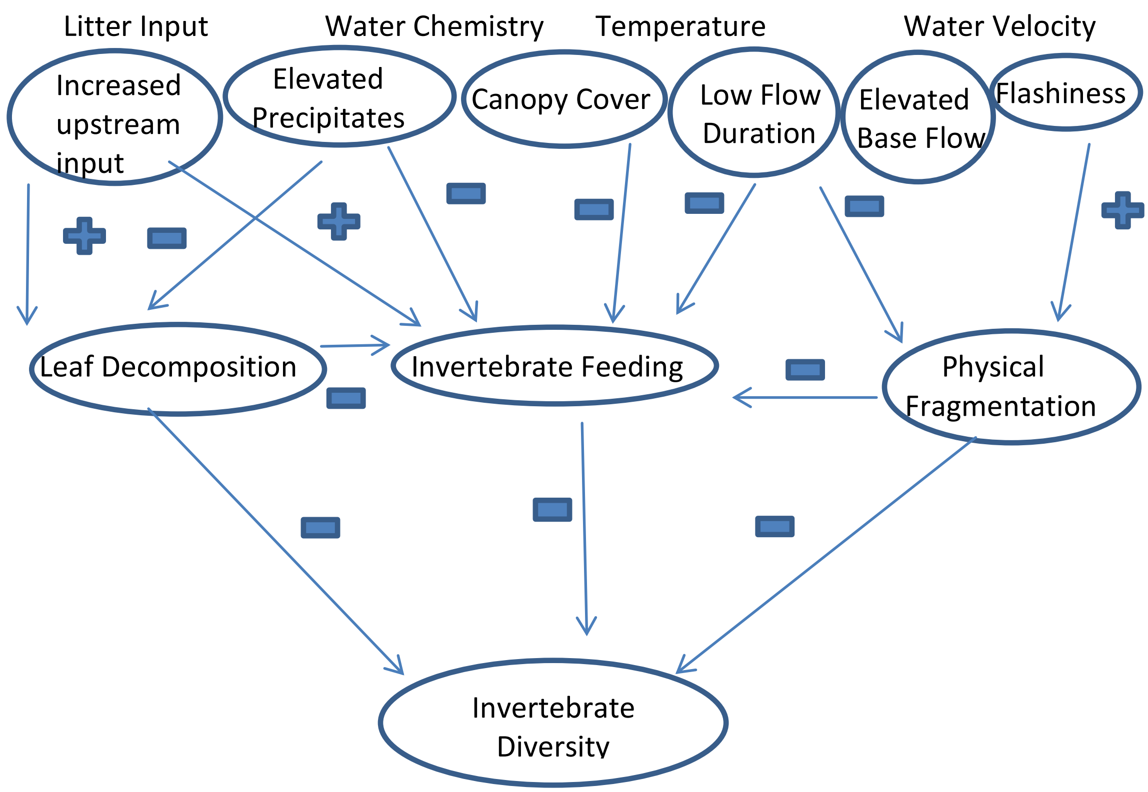

Figure 4) Streams impacted by run-off can receive increased upstream inputs from impervious surfaces. This can cause increases in precipitates and other harmful chemicals from urban areas or pesticides from agricultural areas. Excessive run-off can also erode stream banks which can lead to bank failure, sedimentation and loss of canopy cover. These streams are also vulnerable to higher flashiness as they lack the buffers that slow the input of precipitation into the stream. The input of water without adequate buffers in conjunction with loss of canopy cover can increase water temperature and lower invertebrate feeding. The higher temperature also limits the macroinvertebrates that can survive in the stream which leads to less diversity. As seen in the channelized stream diagram, the higher flashiness can also lead to lowered diversity of macroinvertebrates.

Chapter 2: Use of Leaf Packs to Evaluate Restoration Need in Disturbed Headwater Streams

Abstract

Fresh water is a vital resource for many biota, yet many of these ecosystems suffer high rates of anthropogenic disturbance. Offsetting stream disturbance through restoration is common but expensive. Improving the understanding of functional and structural characteristics of disturbed stream systems can increase resource use efficacy. This study examined variation in macroinvertebrate colonization of leaf packs in reference and three temporal disturbance regimes in Upper Coastal Plain headwater streams. Using Principal Component Analysis, relationships were established between disturbance type and richness, diversity, invertebrate density per gram detritus, and relative abundance of several important groups of macroinvertebrates. ANOVAs on four of the eight components differentiated reference sites from one or more disturbance categories ($p < 0.05$). Run-off influenced streams exhibited higher diverging macroinvertebrate colonization patterns compared to reference sites. Shredder and Trichoptera richness were important in differentiating run-off sites from references while Shredder relative abundance and density aided differentiation of these sites. Combining collector-gather relative abundance and density with Tricoptera and Ephemeroptera density differentiated previous and current temporal regimes and references from runoff sites. No differences in leaf decay rate among disturbance type were found. This was surprising given the large differences in shredder abundance across disturbances; suggesting that the examined disturbance categories did not influence decomposition, or that abiotic drivers of decomposition mask lower shredder presence in disturbed streams. Several habitat variables were examined in an effort to determine the drivers of the macroinvertebrate communities. Different sediment sizes categories were associated with temporal disturbance regimes. These findings could aid decision making regarding a stream's candidacy for restoration.

2.0 Introduction

Fresh water comprises about 0.01% of the Earth's water. It covers less than 1% of the planet's surface while being home to a disproportionate 40% of known fish species and 25% of described vertebrates (Dudgeon et al 2005; Abell 2002). The biota associated with these ecosystems includes some of the most endangered species in the world (Nel et al 2009). Freshwater ecosystems have been, and continue to be, altered by anthropogenic land use at a higher rate than any other ecosystem. In fact, nearly 80% of streams in the United States have been degraded by human activity (Palmer et al 2007; Revenga et al 2005).

In order to conserve these freshwater ecosystems, we must take action. Unfortunately, the resources needed to repair impacted streams are limited (Palmer et al 2005). This means we must prioritize streams that could benefit most from work and allow others to recover with less intervention. Currently, little or no consensus exists regarding methods needed to achieve this goal (Beechie et al 2008; Roni et al 2002). In order to develop a system of prioritization, the influence of two main factors on stream health, stream structure and stream function, need to be better understood. By increasing our knowledge of these two aspects of stream ecosystems we hope to develop a system for prioritizing the need for restoration and therefore expend resources more efficiently (Beechie et al 2008; Roni et al 2002).

Anthropogenic stream disturbances can take several forms. Among the most common are runoff, impoundments and channelization. In each case, one of the main consequences of disturbance is the loss of habitat heterogeneity. This simplification of the stream channel and alteration of flow regime can negatively affect retention and decomposition of detritus material (Gessner et al 2010). Loss of heterogeneity can reduce a stream's ability to cope with flashy hydrologic events (i.e. storms) which can lead to higher discharge rates that alter benthic communities (Boulton et al 1992). This could exacerbate the effect of scouring (erosion of banks and stream beds) often caused by high flow events. Such disturbances from excessive runoff and channel modification can also impact food webs by disrupting snag habitats used by macroinvertebrates, in turn removing a vital food source for fishes and birds which prey on them (Benke et al 2001). In addition to loss of snag habitats in the form of coarse wood, excessive runoff can dislodge organic matter that would otherwise accumulate in the snags or in sediment depositional zones. By influencing stream structure, flow regime, and community composition, both decomposition of litter and organic matter retention can be severely impacted. Disturbances in headwater streams can extend to negative downstream consequences by influencing the amounts and types of organic matter reaching downstream waters. These deviations from normal headwater function may alter community structure in larger streams and rivers (Lecerf and Richardson 2010; Vannote et al 1980). Stream restoration has become an accepted way to deal with severely disturbed streams and consequently has become a multibillion dollar enterprise (Suddeth et al 2007; Palmer et al 2005).

Assessment of the macroinvertebrate community in headwater streams has become a common measure for evaluating both stream health and restoration success.

Macroinvertebrates exhibit a wide range of pollution and disturbance sensitivities (Freitas et al 2012; Kazanci and Dugel 2010). This, along with their relatively short life spans and ease of capture has popularized them as bio-monitoring tools (Rosenberg et al 2008). Also, macroinvertebrates are integral to stream health, playing important roles in the breakdown of detritus, assimilation of biofilm, and as predators and prey (France 2011; Taylor 2005). As a group, macroinvertebrate communities respond negatively to physical channel alterations and riparian disturbances, such as road construction and deforestation (Paller et al 2014; Hedrick et al 2010; Davis et al 2003). Past studies have used a single or combinations of macroinvertebrate variables including taxa richness, Ephemeroptera, Plecoptera, and Trichoptera (EPT) relative abundance, and functional feeding group assemblages to assess the disturbance regime health (Maxted et al 2000; USEPA 1999).

Many macroinvertebrate groups are sensitive to different disturbance regimes. For instance, some EPT macroinvertebrates rely on availability of surface area of exposed rocky substrate, which would be absent in areas of severe erosion or sedimentation (Hamid and Rawi 2011). Other macroinvertebrates suffer deleterious sub-lethal effects in channelized areas that are prone to periodic high-flow events (Beveridge and Lancaster 2007). In fact, some macroinvertebrates with low tolerance values found in reference sites have been shown to be absent in nearby disturbed areas (Pond 2012). In other cases, communities could drastically shift after dam removal, going from a group comprised of species usually associated with lentic water back to those more often found in lotic water (Tszdel et al 2009).

Shifts in macroinvertebrates from headwaters to larger downstream rivers are well known and are associated with Vanote et al's 1980 River Continuum Concept. While specifics of the River Continuum Concept (Vanote et al 1980) have been challenged, the main thrust of the idea (the transfer of energy from lower to higher order streams) has generally been upheld (Greathouse and Pringle 2006; Jiang et al 2011). Temperate headwater streams receive energy input in the form of carbon from riparian zones, often from detritus leaf material (Anderson and Sedell 1979; Vanote et al 1980). This explains the wide use of litterbag studies to examine macroinvertebrate colonization and the detritus processing rates in headwater streams (e.g. Woodcock and Huryn 2005; Benefield et al 1977). Consequently, I used the leaf packs to examine the stream's functional ability to break down detritus material (Gessner and Chauvet 2002). Leaf packs also provide a food source and substrate for macroinvertebrate colonization, allowing examination of potential community differences across disturbance temporal and severity regimes, as well as disturbance types.

Information on a stream's function could prove valuable in assessing the condition of potential restoration sites (Heino 2005). However, restoration and mitigation projects in freshwater systems are often undertaken naively, without sufficient baseline data in attempts to reverse damage caused by direct or indirect disturbance from human activities (Lake et al 2007). As many restorations are implemented in haste, it is not surprising that detailed pre-restoration assessments of both structure and functional attributes of the stream ecosystem are rarely performed (Palmer et al 2007). The mere presence of a physical disturbance is often justification for undertaking a restoration project (Colas et al 2013; Steurer et al 2009). As a result, impaired functions are assumed to improve solely due to the act of providing a more natural stream structure or habitat heterogeneity (Sudduth et al 2011). There are many examples of restoration projects involving bank repair, impediment removal, addition of artificial structures and/or natural channel recovery that have been considered failures from a biological perspective (Palmer et al 2010). Though restoring a stream's structure may increase habitat heterogeneity and aesthetic appearance, it does not necessarily follow that stream functions will automatically return (Hilderbrand et al 2005).

Restoring biodiversity in streams and rivers that have been degraded by changes in land use such as, agriculture, or other environmental stressors has emerged over the last decade as a method for restoring entire stream ecosystems and the suite of services they provide (Palmer et al, 2007). However, growing evidence suggest that restoring physical attributes to a section of stream is not directly correlated to improved biodiversity (Palmer et al, 2010). Perhaps poor water chemistry, continued altered flow regime, or insufficient food sources overarch effects of the improved physical attributes on stream biodiversity (Roni et al 2008). Additionally, depending upon severity of disturbance and recovery time, a disturbed stream may recover its functional attributes while still exhibiting a degree of structural disturbance. It may also be possible that the physical disturbance provides a feature or function that improves species richness. Therefore, information on the severity of and time since disturbance, as well as the type of disturbance may be critical factors for understanding the potential stream impacts and subsequent restoration response.

Benthic macroinvertebrate communities have been widely used as indicators of stream health and restoration outcome (Karr and Chu, 2000; Allan and Castillo, 2007). Comparisons can be made to a reach before and after restoration and changes are evaluated as positive, negative or unchanged. Other criteria compare communities in a restored reach to those of a least disturbed reference reach as an evaluation of restoration effectiveness. However, few if any studies have compared the communities of the disturbed reach to those of the reference reach prior to restoration implementation. This could be a shortcoming if the benthic macroinvertebrate community has recovered from the physical disturbance or never suffered a serious impact by it. As such, improved methods for assessing the impact of physical disturbances on stream macroinvertebrate communities are needed to aid the decision making process on whether a stream is a good candidate for restoration. In this study I examined streams with a variety of disturbance types and range of severity and recovery time to determine if physical disturbance necessarily corresponds to a change in a stream functional attribute or biodiversity. I used leaf packs as a general investigative tool to provide information on both stream function (decomposition and organic matter retention) and macroinvertebrate diversity and abundance and richness to evaluate their effectiveness for discerning restoration need in headwater streams with documented structural disturbances. Habitat variables such as sediment size and macrophyte presence were evaluated across the same disturbance regimes in an attempt to determine drivers of the macroinvertebrate communities.

2.1 Methods

2.1.1 Study Sites

This study was conducted at the Savannah River Site (SRS), which encompasses parts of Aiken, Barnwell, and Allendale counties in South Carolina and borders the Savannah River. SRS exhibits an array of stream disturbances that have occurred over a long period of time and are typical in many developed nations across the globe. For example, some streams have been cleared of debris and dredged, and/or many contain dams, some of which remain intact while others have been reduced to remnants. Riparian areas show evidence of past logging activities. Roads, railroads and power line corridors have altered channels by changing their original configurations and through runoff and sedimentation. Livestock were allowed access to streams, and pesticides were used in agricultural areas. Runoff from impervious surfaces has altered stream channels and flow regimes. Some streams were thermally influenced by cooling water effluents from nuclear reactors (Kolka et al 2005; Lakly and McArthur 2000).

The primary function of the SRS, which occupies over 80,000 ha, is to process and store nuclear materials in support of defense and nuclear non-proliferation policies of the United States (Wyatt and Harris 2004). Prior to 1950, bottomland forests, agricultural production and several small towns comprised this area.

Construction of the SRS began in 1950 with the first reactors going critical three years later (US DOE 2014). In 1972 the SRS became the United States' first National

Environmental Research Park (Smith et al 2001). Currently, itis mostly comprised of forested land along with several industrial areas (Wyatt and Harris 2004).

Assessment of the potential SRS study area required examination of aerial photos (1938-2010), LiDAR imagery (2009), existing GIS data, and maps (1938 to current) to identify disturbances such as flow impediments, erosion, or channelization. I identified streams that spanned a broad temporal disturbance gradient from about the early 19th century (White 2004; Brooks et al 2000; White and Gaines 2000) to impacts from active industrial areas. This was supplemented with extensive ground surveys, during which study streams were walked from their confluences to near the drainage divides while mapping any active or historic disturbances. This included all valleys with perennial or intermittent channels, as well as significant ephemeral channels. In all, 20 stream reaches were selected for this study: 10 reference and 10 disturbed sites (Table 1).

Reference systems are tools often used to gauge stream health, identify disturbed areas and determine successes or failures of restoration (Kosnicki et al 2014). Reference sites (Ref) were chosen as examples of the least disturbed streams using the data sources above. In general, Ref streams exhibited little evidence of structural impediments and contained mostly intact riparian zones. The disturbed sites chosen for this study would be obvious candidates for restoration under current evaluation techniques and existing impairments noted in the visual survey.

Disturbed sites were assigned to temporal and categorical regimes. Due to constraints on site availability, neither of these two regimes had balanced designs. Both types of sites were identified using historical data and observation of each stream described by Fletcher et al (2011). In the temporal regime, sites disturbed before construction of SRS (Pre) (n=4) included channelized (MC5B) and previously dammed areas (TC2A, TC2C and MB7.5) (Table 1). Those disturbed during construction and operation (SRC) (n=2) included PB4 and MBHW. Sites of ongoing disturbance of varying degrees (Cur)(n=4) included sites impacted by runoff from industrial areas (U6, U8, MQHW, and MBHW) and one channelized site (U36C). The Cur sites all showed evidence of continuing disturbance such as obvious sites of runoff or in the case of U36C impact by roads.

The categorical regime included sites which exhibited disturbance from abandoned impediment structures (Dam) (n=3), one from a narrowly breached farm pond (MB7.5) and two narrowly breached mill dams (TC2A and TC2C) (Table 1). The pond dam on Meyers Branch was breached in the early 1950's, whereas those in Tinker Creek sites were breached prior to 1940. All of these streams had remnants of the dams within the stream and/or on the banks. Channelized sites (CH) MC5B and U36C were obviously straightened at some point (n=2) (Table 1). The Upper Three Runs tributary site (U36C), located along US-278, and was channelized sometime between 1943 and 1951. Although the Mill Creek tributary site (MC5B) is located below a low breached dam, it is isolated from current development and the date of channelization probably is much older. These sites show evidence of past incision but little active erosion. The other disturbed sites (n=5) were classified as receiving runoff (RO) from industrial areas: Upper Three Runs tributaries (U6, and U8), Pen Branch tributary (PB4), McQueen Branch drainage (MQHW) and one in Meyers Branch (MBHW). Again due the limitations of site selection there were several important crossovers between disturbance regimes. Three of the five RO sites were in the Cur temporal regime, the three Dam sites were in the Pre temporal regime.

Monitoring reaches were established at each selected site and marked with metal fence posts driven into the floodplain at 30m intervals on each side of the stream, approximately 1m behind bank-full level.

2.1.2 Study Organisms

Leaf packs were deployed in the week of March 5, 2012 then collected in the spring of 2012 (March 12 to May 4) and deployed again in the week of January 21, 2013 and collected in the winter of 2013 (January 28 to March 29).The leaf pack dimensions were33 x 43-cmmesh (J&M Industries, Ponchatoula,LA). The mesh size of the bags was 5mm. The individual strands of the bags were flexible so as not to preclude larger invertebrates from gaining access to the leaves. Each bag was filled with 5 grams dry weight (± 0.25g) of senesced white oak (*Quercus alba*) leaves (Cummins et al 1989), collected using a net positioned under several trees to prevent ground contact and air dried in the lab for two weeks before storing in large paper bags.

White oak is a ubiquitous species in the eastern U.S., found in 75% of the riparian areas of the streams included in this study, and often used in leaf pack studies (Cotton 2003, Nelson 2000; Meehan et al 1996; Rowe et al 1996). White oak has been shown to have a slower breakdown rate than some other species commonly found in the riparian area (i.e. red maple) and therefore the importance of shedders is increased for white oak decay (Wallace et al 1982). At each site, 5 bags were carried to the field and 4 were placed into a run habitat near the bottom of each reach and tied to a fence post on the stream bank to prevent them from washing away (Nelson 2000). The four bags to be deployed were tied together with strings on the corners of the bags making the spacing approximately 15cm between bags. The remaining bag was returned to the lab and weighed to determine handling loss for determining leaf decay rates following Hagen et al (2006). The other bags remained in the stream and were harvested1, 2, 4 and 8 weeks based on the dates of placement (Swan and Healy 2008) using a D-frame dipnet to retrieve the bag last bag in the string. The bags were frozen in stream water until processing (Nelson 2000). After thawing, macroinvertebrates were sorted from the leaf packs and identified to the lowest possible taxon as described by Merritt, Cummins and Berg (2008). Non-biting midges were identified only to family Chironomidae and aquatic worms only to Annelida.

After identification, all macroinvertebrates, except chironomids and non-insects, were placed into functional feeding groups following the taxonomy outlined in Merritt, Cummins and Berg (2008). A total of 11 richness/diversity metrics were assessed: total richness, EPT (Ephemeroptera, Plecoptera and Trichoptera) richness, richness of each order of EPT, functional feeding group richness. Shannon Diversity (H') and Simpson's Diversity (Simp) were calculated for each sample. Additionally, relative abundance and invertebrate densities per gram of the remaining leaf material were calculated for Ephemeroptera, Plecoptera, and Trichoptera (combined as EPT and by order), total macroinvertebrates, chironomids and each of the functional feeding groups for a total of 19 macroinvertebrate variables. Samples not containing macroinvertebrates were represented by zeros in all variable measures.

2.1.3 Habitat Variables

Canopy cover was measured at each cross-section of the study reaches using a model C spherical crown densitometer (Forestry Suppliers; Jackson, MS). The cross-section results were averaged for each study reach. Presence of macrophytes, rootmats, undercut banks and coarse woody debris (greater than 10cm diameter) were recorded every 10m across each study reach and the percentage of positive results was then calculated. Presence of stream bars and the type(s) of bars (sand, fine gravel, cobble and coarse gravel) were recorded at the same 10m stations. At each 30m interval semi-permanent cross sections were established for later geomorphic work. Streambed sediment samples (top 10 cm) were collected using a shovel from each cross-section and each sample was placed into gallon plastic bag (N'Guessan et al 2009; Amalfitano and Fazi 2008). Samples were returned to the lab and separated using standard sieves.

I used RiverMorph™ 4.3 to calculate the DB 84 and DB 50 for each cross section (the particle size of the 84th and 50th percentiles of the size, respectively), two common measures used in stream evaluations based on the size of a standard sieve set. RiverMorph is a software tool commonly in stream design. After entering cross section field measures many other measures are calculated through interpolation. These include geomorphic measures such as bankfull variables and provides the ability to compare cross sections on a year to year basis through overlaying the graphs. Cross-sectional results were averaged to determine values for the entire study reach.

2.1.4 Decomposition Rate

Decomposition rate for this study combined mechanical breakdown from flow and macroinvertebrates along with decay from fungi and bacteria. After the macroinvertebrates were removed, the remaining leaf material was separated, gently washed with deionized water, oven dried (40˚C for a minimum of 48 hours) and weighed on an analytical balance. After correcting for handling loss (Hagen et al 2006), the overall loss of mass was calculated by subtraction from the initial mass. Rate of decomposition/physical breakdown and material loss was calculated by dividing the mass loss by days exposed. This procedure was repeated for each sampling interval.

2.1.5 Statistical Analysis

Repeated measures statistical designs can be treated like univariate split-plot ANOVA designs (Wilkinson et al 1996). My design was modeled after the split-plot design presented in Wilkinson and Coward (2012). In the first step of analysis, decomposition rates, macroinvertebrate richness and diversity measures were compared in disturbed (n=10) versus reference reaches (n=10) using the model: disturbance regime, year, disturbance regime*year, week(year), disturbance regime*week(year). Disturbance regime refers to categorization of streams as either reference or disturbed. Habitat variables were compared in the disturbed reaches versus the reference sites using ANOVAs in the form of a generalized linear model. All comparisons were repeated using both temporal regime and disturbance category with those registering a p-value of 0.05 or less followed by Tukey's tests to further elucidate differences.

Due to the large number of variables derived from the macroinvertebrate data, Principal Component Analysis (PCA) with varimax rotation was used to reduce the dimensionality and improve the interpretation of patterns between disturbance times and types. Although density, relative abundance, richness and diversity were not independent, PCA was used to reduce the amount of variables to be tested. Macroinvertebrate relative abundance variables (proportional measures) were transformed ($\arcsin\sqrt{x}$) with macroinvertebrate densities log-transformed (ln(1+x)) to reduce skewness of the data. Principal component analysis was employed in order to compare macroinvertebrate metrics that co-varied among disturbance regimes.

One PCA analysis was run on the richness and diversity variables denoted by PC#R. A second PCA was performed on the relative abundance and diversity metrics. Useable principal components were determined by eigenvalues and scree plots. The amount of influence of specific variables on each of the principal components is indicated by the component loadings (CL). Magnitude and sign of the CL indicates the strength and direction of the influence. Component scores saved from the PCA and decomposition rates were used in the same split-plot ANOVA model used in the first step of analysis followed by Tukey's pair-wise comparisons to test differences based on disturbance class. An analogous ANOVA model and Tukey's tests were employed with decomposition rates. Statistical comparisons were conducted with SYSTAT® 13 statistical package (SYSTAT® Software Inc. 2009). Least square means acquired from the respective ANOVA models were used for graphic presentation (Figures 1 and 2). One-way ANOVAs were used to compare log transformed habitat variables across disturbance types and temporal regimes with Tukey's pair-wise test to clarify the differences.

2.2 Results

2.2.1 Richness and Diversity

Nearly 6,000 macroinvertebrates were collected from the leaf packs over the two study periods. One bag was found out of the stream in MB9 week 8 and therefore was not included in the analysis. Principal components analysis (PCA) of 9 taxa/functional feeding group richness metrics along with the Shannon (H') and Simpsons (Simp) diversity indices calculated from samples collected from 20 sites yielded 3 principal components that accounted for over 70% of total variation (Table 2). PC1R explained 32.6% total variation with relatively strong component loadings by shredder richness, Trichoptera richness, Total species richness, EPT richness, H', and week loadings of Plecoptera richness, and predator richness (all CL > 0.600) (Table 3). Higher PC1R factor scores indicate greater richness of these groups. PC1R was higher in reference than disturbed sites and indicated differences among collection weeks within a year [R^2 = 0.17, disturbance regime *p* = 0.02, year *p* = 0.23, disturbance regime*year *p* = 0.28, week(year) *p* = 0.03, disturbance regime*week(year) *p* = 0.39] (Figure 1a). ANOVA of PC1R showed difference in disturbance time regimes [R^2 = 0.28, disturbance time $p < 0.01$, year $p = 0.63$, disturbance time*year $p = 0.17$, week(year) $p = 0.99$, disturbance time*week(year) $p = 0.93$]. Ref and Pre site scores were higher than both SRC and Cur sites (post-hoc testing $p \leq 0.05$) (Figure 2a). The ANOVA of the same factor scores by disturbance type explained the most variance of the three comparisons [R^2= 0.41, disturbance type $p < 0.01$, year $p = 0.72$, disturbance type*year $p = 0.05$, week(year) $p = 0.85$, disturbance type*week(year) $p = 0.43$] and revealed differences among disturbance types (Figure 3a) and the potential effects of the interaction between disturbance and year. Tukey's post-hoc comparisons showed sites disturbed by on-going runoff (RO) had lower scores than those from all the other categories (Figure 3a). No other comparisons were statistically different. Overall these analyses indicated that shredder richness, Trichoptera richness, Total species richness, EPT richness, H', Plecoptera richness, and predator richness were reduced in disturbed sites. Further, within the temporal regime, these metrics were reduced in SRC and Cur sites. These differences appeared to be primarily driven by lower scores in sites receiving excessive runoff.

The second principal component (PC2R) explained only 15.4% of the total variation with predator richness and Simpson's diversity loading relatively strongly (CL > 0.640). Weaker loadings included Plecoptera richness (CL > 0.545) and total species richness, (CL > 0.354). No differences were apparent between Ref and Dist reaches (Figure 1b). ANOVA of PC2R for time regimes [R^2 = 0.31, disturbance time $p = 0.49$, year $p < 0.01$, disturbance*year $p = 0.99$, week(year) $p = 0.77$, disturbance time*week(year) $p = 0.97$] (Figure 2b), and disturbance types [R^2= 0.32, disturbance type $p = 0.59$, year p <

0.01, disturbance type*year p = 0.90, week(year) p = 0.96, disturbance type*week(year) p = 0.89], showed no differences except between years (Figures 2b and 3b). Predator richness and Simpson's diversity differed between years, but did not differ with respect to disturbance.

PC3R explained 25.4% of total variance with strong loadings (CL >0.750) by Ephemeroptera, collector and scraper richness and weaker loadings by total species, EPT and Plecoptera richness. No difference was found between Ref and Dist regimes (Figure 1c). ANOVA of PC3R and disturbance times was able to show differences in temporal regimes [R^2 = 0.22, disturbance time p < 0.01, year p = 0.84, disturbance*year p = 0.53, week(year) p = 0.69, disturbance*week(year) p = 0.93]. Tukey's post hoc comparisons indicated higher scores in Pre sites than the other three categories. (Figure 2c). The ANOVA of PC3R and disturbance type [R^2= 0.24, disturbance type p < 0.01, year = 0.53, disturbance type*year p = 0.57, week(year) p = 0.62, disturbance type*year(week) p = 0.77], revealed differences among disturbance types. Tukey's post hoc comparisons separated Dam sites from both Ref and Runoff (Figure 3c). Consequently, the increase in Ephemeroptera, collector and scraper richness, in Pre sites appears to be driven by higher richness in Dam sites.

2.2.2 Density and Relative Abundance

Relative abundance of the functional feeding groups is illustrated in Figure 4. PCA of 19 metrics related to the density and relative abundance of taxonomic or functional groups in the same 20 sites yielded 5 principal components accounting for

82% of the total variation. While ANOVAs were able to detect differences at all three levels (Ref vs. Dist; temporal regime and categorical regime) using several of these primary components, the results were similar to the richness and diversity primary components. The components created using density and relative abundance measures that were able to detect differences had similar loading components to those components made using richness and diversity (i.e. shredder richness vs. shredder density and relative abundance) (Table 3). In general, the few differences that were found separated Cur from Ref and Pre with RO being the most often separated disturbance type. All of the differences found through analysis of l principal components are summarized in Table 4.

2.2.3 Decomposition Rate

Decomposition rates were similar in all three disturbance regimes. ANOVA results included [R^2= 0.15, disturbance regime *p* = 0.76, year < 0.01, disturbance regime*year *p* = 0.58, week(year) *p* = 0.24, disturbance regime*week(year) *p* = 0.24] disturbance time [R^2= 0.10, disturbance time *p* = 0.90, year *p* < 0.01, disturbance time*year *p* = 0.92, week(year) *p* = 0.64, disturbance time*week(year) *p* = 0.98] or disturbance categories [R^2= 0.16, disturbance type *p* = 0.98, year *p* = 0.02, disturbance*year *p* = 0.96, week(year) *p* = 0.84, disturbance*week(year) *p* > 0.99]. However, all three analyses indicated decomposition rates were higher in 2012 than 2013. Significant differences among weeks within years were absent.

2.2.4 Habitat Variables

The only habitat measure that was able to differentiate between disturbed and reference sites was the percentage of areas with sediment bars present which can impede flow (sand, gravel etc.) (p = 0.01) (Table 5). Disturbed reaches had more sites with bars than reference reaches. However, several habitat values were useful in differentiating temporal regimes or disturbance types. Percentage of areas with undercut banks was higher in Ref compared to Pre and percentage of silt was lowest in Cur streams and higher in Ref and SRC. There were more areas with undercut banks in Ref sites than Pre sites. Presence of macrophytes and presence of bars were both able to distinguish disturbance categories. More areas with macrophytes were found in Dam sites compared to both Ref and RO sites while a higher percentage of areas with bars were found in RO than Ref sites. ANOVA results showed both measures of sediment size were different across disturbance time. Post-hoc testing showed larger DB 84 for Pre versus SRC sites for DB 84, but was unable to clarify differences for the DB 50 measure. Both sediment size measures were also able to differentiate disturbance types. ANOVAs of DB 84 and DB 50 showed RO sites to have larger sediment size than all other types while only DB 50 showed Dam sites to have smaller sediment size than Ref sites. In general, sediment size was larger in the Cur regime and in the RO disturbance type.

2.3 Discussion

2.3.1 Richness and Diversity

Richnesses of detritus shredding insects that colonized the leaf packs (often Trichoptera) were important in differentiating Reference from Disturbance streams. Generally, Trichoptera are known to be highly pollution sensitive and strong indicators of stream health (Pond 2012; Ruiz-Garcia et al 2012) while shredders are well known to be play an important role in carbon breakdown in headwater streams (Cummins et al 1989; Cummins 1973). This suggests that concentrating on the shredding leaf pack inhabitants may be an efficient way to differentiate runoff damaged areas from other sites in sandy coastal plain headwater streams. Further, these two groups of macroinvertebrates (Trichoptera and shredders) may be used to identify more recent disturbances from older or undisturbed reaches. Other studies have shown that Trichopteran assemblages were useful for identifying various types of pollution disturbances (Ruiz-Garcia et al 2012), so it is not surprising that they could be used to identify sites influenced by runoff. Trichoptera and shredders, in general, were not useful in differentiating channelization or abandoned dams from references sites which could be attributed to the low number of disturbed sites examined or that the streams have naturally recovered to a new equilibrium in the time since original disturbance. Greater structural differences, found in the DB 84 and DB 50 sediment size, between references sites and runoff sites corresponded to greater differences in Trichoptera and EPT in total richness. This could reflect the differing habitats created by larger sediment.

Simpson's diversity and predator richness were not useful in differentiating any disturbance regime from Ref. Predator richness may be a reflection of prey availability and therefore were not able to distinguish any disturbance regimes as there were midges found in all sites that could attract the predators. The difference in years is probably best explained by the different seasons in which the sampling was done due to the lifecycles of the macroinvertebrates their size and likelihood of capture varying seasonally (Biemiller 2011; Jaques and Pinto 1997).

Surprisingly, given their history as indicators of water health, lower richness of Ephemeroptera was not a good indicator of disturbance. In fact, sites in the Dam regime showed higher Ephemeroptera richness than found in Ref. This could be an indicator of higher habitat heterogeneity as the Dam sites are continuing their recovery from disturbance or another indication of the similarity between the sites thought to be disturbed and the reference streams (Tsydel et al 2009). Allowing more recovery time could permit the streams in the Dam regime to come closer to resembling the state of Ref streams. Given the time that has passed since breeching of the dams, in some cases over 50 years, however, this seems unlikely. It is, however, important to note that the dams were not entirely removed with remnants left both in the stream and on the banks. These remaining structures could influence the stream condition to a lesser extent. Another explanation could be that the Ephemeroptera are using macrophytes, which are abundant in 2 of the 3 Dam sites and only 2 of the 10 Ref sites, as an important habitat or food source (Casatti et al 2003). Further, the high richness of Ephemeroptera could be an artifact of the width of the streams in the Dam regime given that TC2A and TC2C were among the widest streams examined in this study. It is also possible that the differences between disturbance regimes and the reference were not great enough to have a measurable effect on the Ephemeroptera.

2.3.2 Density and Relative Abundance

Results from the density and relative abundance were very similar to those found in the examination of richness and diversity. Trichoptera would be the most important Order to examine to differentiate disturbance types in these streams. The ongoing disturbance of the RO sites explains the low numbers while the higher numbers in one of the abandoned Dam sites are likely due to habitat recovery, presence of macrophytes or the size of the stream as discussed earlier. Overall, it seems that the leaf pack inhabiting members of Trichoptera and shredders were the best indicators of both temporal regimes and disturbance types in this study based on their component loading factors. Concentrating sampling effort on these groups in the pre-restoration phase could yield a more effective evaluation of stream status, or macroinvertebrate community, which could be useful in making a determination on whether a site is a suitable candidate for restoration.

2.3.3 Decomposition Rates

Many studies have shown altered rates of leaf decay due to disturbance types similar to those reported here (Duarte et al 2008; Gulis and Suberkropp 2003; Gessner and Chauvet 2002). Surprisingly, decomposition rates of leaf pack detritus did not vary across disturbance types in this study. At first, decomposition rate may seem to be an inappropriate functional characteristic for examining disturbance in Upper Coastal Plain Streams, yet it is important to consider the drivers of this characteristic across disturbance categories. The relative abundance of shredders compared to total macroinvertebrates was higher in Ref than RO sites (*p's* < 0.01). Given this along with the high proportion of macroinvertebrates that were classified as shredders in the reference reaches, as compared to those in sites influenced by runoff (Figure 4), uniform decomposition rates between regimes were unexpected based on the invertebrate communities (Cummins et al 1989). However, other studies have noted similar results showing variations in macroinvertebrate communities across disturbance regimes while exhibiting similar leaf decomposition rates (Fritz et al 2010). Fritz et al (2010) noted the "tattered appearance" of leaf litter in disturbed reaches that were similar to observations of leaf structure from the Cur and RO regimes in my study. They suggested that other factors, such as flow regime related fragmentation or temperature, may be influencing decay rates in disturbed streams and masking the effect of shredder paucity in these areas. Other possible drivers for leaf decay rate in disturbed streams include: water chemistry (Duarte et al 2008), preconditioning or conditioning of the leaves (Dieter et al 2011; Webster and Benefield 1986), fungal activity (Gulis and Suberkropp et al 2003), and climate change (Schlief and Mutz 2011) to name a few. Possibly one, or a combination of the above factors, may be influencing rate of leaf litter loss in RO streams. Conversely, it is also plausible that there simply were no differences between decomposition of the examined reaches.

2.3.4 Habitat Variables

The comparison of sediment size across disturbance time and type may also help explain the differences I observed in macroinvertebrate assemblages. While it was not expected that groups such as shredders were more strongly associated with fine sediment areas than those with larger sediment, there are several possible explanations for this phenomenon. First, since sandy bottoms are the normal state of Upper Coastal Plain streams, it makes sense that the macroinvertebrates in the area are adapted to thrive in fine sediment. Second, the Cur disturbances are due to runoff and channelization. Both of these disturbance types, but particularly the runoff sites are characterized by excessively strong flows, unstable channels, and poor flood plain connectivity (Wetzel 2001). It may be the case that sediment and macroinvertebrates are being scoured out of the area by periodic high flow events that are mitigated in other streams by flooding. Therefore, sediment size composition along with flow stability or flashiness may be driving the macroinvertebrate composition. Sediment size could also be used to distinguish temporal regimes of disturbance. Further, the other variables that were able to distinguish any temporal or disturbance types (presence of bars, macrophytes, undercut banks, and silt percentage) all contribute to heterogeneity of habitat. It is possible that examining the stream flows and water quality variables in the Ref and various disturbance regimes may clarify these uncertainties.

2.4 Conclusions

Use of leaf packs provided valuable information on the status of streams that were impacted by a variety of disturbance types and times since impact, though not in the way I hypothesized. Although all disturbed reaches examined exhibited visible physical disturbances, not all differed from reference sites with regards to macroinvertebrate diversity or litter decomposition rate. This result directly refutes the flow charts from chapter 1 in that liter communities and macroinvertebrate diversity did not differ by disturbance type (Figures 1 -4).If a restoration goal for these disturbed reaches was to fix the impairment to improve habitat heterogeneity and biodiversity, then success may not be realized as several streams appeared to have already recovered some structural and functional characteristics. On the other hand, some metrics of the macroinvertebrate community that colonized leaf packs did vary between Ref and the disturbance regimes. As Palmer et al (2010) have discussed, restoration often does not result in increased biodiversity likely due to a variety of interactions among physical, chemical and habitat variables that control restoration response. Therefore, the drivers of the differences of the macroinvertebrate communities need to be examined. Streams are complex and dynamic systems even in an undisturbed state. The temporal effect on stream recovery in my study reaches seems to have had an overriding effect on the macroinvertebrate community such that few differences were detectable in all but those that experienced an active disturbance regime. This could be due to a number of factors, such as relatively intact riparian areas and closed canopy found in these forested sites compared to impaired streams in a more urban environment. As such, it may be naïve to think that restoration will provide a desired response in reaches with older disturbances given that differences in the parameters I examined were not obvious prior to restoration. This is not to say that restoration will not be effective on every older disturbance site. Comparisons to reference sites should be made whenever possible before any conclusions are drawn.

This study highlights the importance of pre-restoration comparisons of potential restoration sites to reference reaches. Given the tendency of streams to move towards equilibrium after the cause of disturbance is removed, it is vital that land managers determine what similarities and differences exist between potential restoration sites and reference reaches in order to increase efficacy of future projects. A more thorough understanding of a stream's status compared to reference streams prior to restoration is necessary to understand what changes need to occur in order to deem the project either a success or failure. As the data in this study makes clear, it is not unusual for macroinvertebrate communities from sites that have had time to recover to be similar to reference reaches. This could be due to either the resilience (ability to resist effects of disturbance) or recovery (ability to return to equilibrium after disturbance) of macroinvertebrate communities. Regardless, the similarities between macroinvertebrates of Ref and historically disturbed sites suggest that these areas should not be priorities for restoration efforts; whereas Cur sites should be considered high priority candidates for restoration. The differences found between reference sites and disturbances, temporal regimes and disturbance types prior to restoration can enhance managers' ability to set biologically relevant goals for restoration. This eliminates the potential to naively attribute similarities to reference reaches after restoration to the restoration action while at the same time highlighting areas where passive restoration may have been more appropriate. From a financial standpoint, one must also question whether restoring the historically disturbed sites is warranted. Finally, the most effective 'restoration method' may be implementing preventative measures in areas that have the potential to develop into problematic areas to stop habitat degradation before it occurs.

Chapter 2 Tables

Table 1.Stream study areas and their classificationsby disturbance, categorical and tmporial regimes. SRC= disturbed during construction or during operation of nuclear activities at Savannah River Site. Pre= disturbed prior to construction of Savannah River Site. Cur= sites undergoing active disturbance.

Stream	Reach Abbreviation	Disturbance Regime	Categorical Regime	Temporal Regime
Meyers Branch	MBHW	Disturbed	Runoff	SRC
Meyers Branch	MB6	Reference	Reference	
Meyers Branch	MB7.5	Disturbed	Dam Removal	Pre
Meyers Branch	MB9	Reference	Reference	
Mill Creek	MC5A	Reference	Reference	
Mill Creek	MC5B	Disturbed	Channelization	Pre
Mill Creek	MC6	Reference	Reference	
Tinker Creek	TC3	Reference	Reference	
Tinker Creek	TC5	Reference	Reference	
Tinker Creek	TC2A	Disturbed	Dam Removal	Pre
Tinker Creek	TC2C	Disturbed	Dam Removal	Pre
Pen Branch	PB3	Reference	Reference	
Pen Branch	PB4	Disturbed	Runoff	SRC
McQueen Branch	MQHW	Disturbed	Runoff	Cur
McQueen Branch	MQ8	Reference	Reference	
Upper Three Runs	U6	Disturbed	Runoff	Cur
Upper Three Runs	U8	Disturbed	Runoff	Cur
Upper Three Runs	U10	Reference	Reference	
Upper Three Runs	U36A	Reference	Reference	
Upper Three Runs	U36C	Disturbed	Channelization	Cur

Table 2. Loading components and strengths for principal components composed of richnesses and diversity are shown indicating the amount each variable influenced the corresponding principal component and the strength of the co-variation. Strong loaders, the most important metrics for each principal component, have loading strengths over 0.7. The amount of the total variance explained by each component is also listed.

	Principal Components		
Loading Components	*PC1R*	*PC2R*	*PC3R*
% of Total Variance Explained	*32.6*	*14.5*	*25.4*
Total Richness	0.703	0.354	0.524
Simpsons Diversity	-0.200	0.775	-0.022
Shannon Diversity	0.735	0.077	0.476
EPT Richness	0.687	0.278	0.589
Ephemeroptera Richness	0.076	0.126	0.910
Plecoptera Richness	0.505	0.545	0.112
Trichoptera Richness	0.850	-0.031	0.245
Shredder Richness	0.862	0.023	0.072
Scraper Richness	0.344	0.118	0.668
Collector Richness	0.189	0.024	0.734
Predator Richness	0.400	0.674	0.247

Table 3. Loading components and strength for principal components composed of relative abundances and densities are listed, indicating the amount each variable influenced the corresponding principal component. Strong loaders, the most important metrics, for each principal component, have loading strengths over 0.7. The amount of the total variance explained by each principal component is also listed.

	Principal Components				
Loading Components	*PC1D*	*PC2D*	*PC3D*	*PC4D*	*PC5D*
% of Total Variance Explained	35.9	17.0	13.3	9.4	6.6
EPT Density	0.511	0.382	0.344	0.527	0.391
Ephemeroptera Density	0.108	0.321	0.722	0.175	0.426
Plecoptera Density	0.376	0.324	0.073	0.778	0.076
Trichoptera Density	0.626	0.271	0.142	0.169	0.570
Shredder Density	0.775	0.330	0.025	0.349	0.123
Scraper Density	0.160	0.280	0.805	0.152	0.166
Collector Density	0.254	0.399	0.242	0.245	0.733
Predator Density	0.125	0.345	0.161	0.738	0.353
Total Density	0.242	0.834	0.117	0.297	0.237
Chironomidae Density	-0.020	0.926	-0.039	0.007	0.012
EPT Relative Abundance	0.406	-0.396	0.462	0.430	0.220
Ephemeroptera Relative Abundance	-0.120	-0.228	0.843	0.005	0.254
Plecoptera Relative Abundance	0.178	-0.120	0.024	0.877	-0.140
Trichoptera Relative Abundance	0.640	-0.252	0.062	-0.079	0.431
Shredder Relative Abundance	0.852	-0.217	-0.048	0.049	-0.180
Collector Relative Abundance	-0.046	-0.222	0.208	0.077	0.824
Scraper Relative Abundance	0.001	-0.152	0.913	-0.026	-0.080
Predator Relative Abundance	-0.239	-0.247	-0.011	0.761	0.226
Chironomidae Relative Abundance	-0.300	0.728	-0.103	-0.210	-0.188

Table 4.Macroinvertebrate principal components are listed along with their ability for identifying differences, in the form of p values, in the three regimes of disturbance. This illustrates that some components were useful for making differentiation between regimes while many were unable to make any differentiations. This is evidence for a large amount of similarity and by examining the loading components (listed in above tables) a better understanding of what metrics to focus on in this system can be determined.

	p-values of Principal Components by Regime		
Principal Components	*Disturbance Regime*	*Temporal Regime*	*Categorical Regime*
PC1R	<0.01	<0.01	<0.01
PC2R	0.44	0.50	0.59
PC3R	0.54	<0.01	<0.01
PC1D	0.02	<0.01	<0.01
PC2D	0.75	0.36	0.85
PC3D	0.55	0.60	0.90
PC4D	0.94	0.92	0.14
PC5D	0.80	0.02	<0.01

Table 5. R^2 and *p*-values for habitat variables in the three disturbance regimes are shown. Only the presence of sediment bars could distinguish the Ref and Dist regimes. Sediments sizes both DB 50 and 84 were smaller in both the temporal and categorical regimes compared to reference. This again makes clear the large amount of similarities among regimes and re-enforces the importance of examining the variable in different manners.

Variable	**Disturbance regime**		**Temporal Regime**		**Disturbance Type**	
	R^2	*P value*	*R^2*	*P value*	*R^2*	*P value*
Canopy Cover	0.089	0.2	0.106	0.6	0.14	0.48
Macrophytes	0.055	0.32	0.312	0.1	0.457	0.02
Undercut Banks	0.16	0.08	0.38	0.05	0.348	0.07
Root Mats	0.184	0.18	0.186	0.34	0.118	0.56
Silt %	0.179	0.06	0.553	< 0.01	0.233	0.22
Coarse Wood Presence	0.115	0.14	0.261	0.17	0.191	0.32
Overhanging Plants	0.011	0.66	0.014	0.97	0.057	0.81
Decomposition Rates	0.236	0.77	0.363	0.41	0.303	0.91
Presence of Bars	0.317	0.01	0.322	0.09	0.404	0.04
DB 84	0.016	0.6	0.379	0.05	0.543	< 0.01
DB 50	0.02	0.55	0.408	0.04	0.741	< 0.01

Chapter 2 Figures

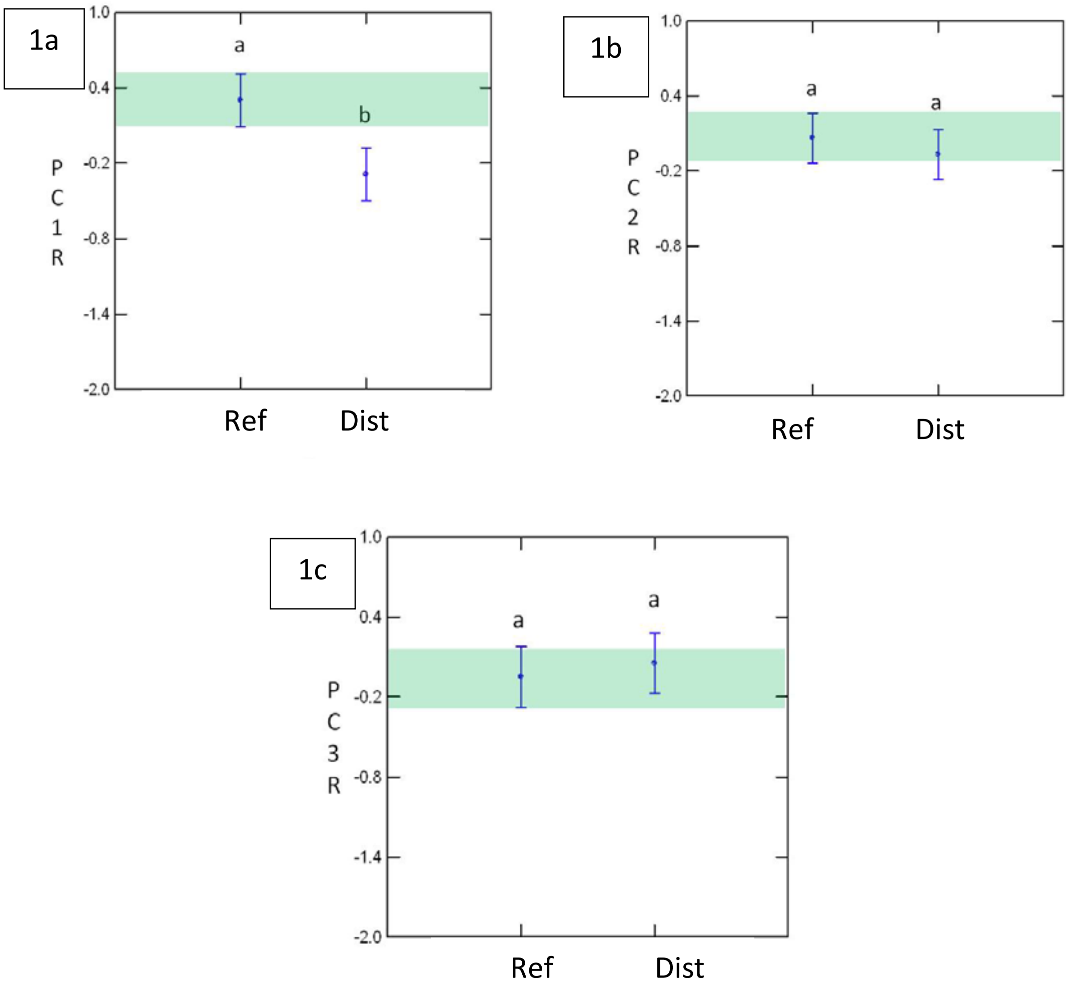

Figure 1) Relationship of co-varying metrics of macroinvertebrates that colonize leaf packs combined using PCA in the reference regime (Ref) and the disturbed regime (Dist) in the study sites using ANOVA results. The reference condition is denoted by the shaded area of the graphs. Differing letters above error bars denote statistically significant differences (*$p<0.05$*). Based on strength of loading components of PC1R, richnesses of shredders, Trichoptera, total macroinvertebrates, EPT, along with Shannon's diversity were able to distinguish Ref and Dist categories. The importance of these groups to headwater streams has been well-established (Cummins 1989) and biodiversity is used as a criterion to measure restoration success (Palmer et al 2005). These results suggest that concentrating on the strong loaders of PC1R can differentiate streams that are in good condition (Ref) and those that may be in need of some restoration action (Dist). The other richness and diversity co-varying metrics were unable to make differentiations on the health of disturbed Upper Coastal Plain headwater stream relative to that of Ref reaches. Figure 1a is the graph for Principal Component 1R, Figure 1b refers to Principal component 2R and so on.

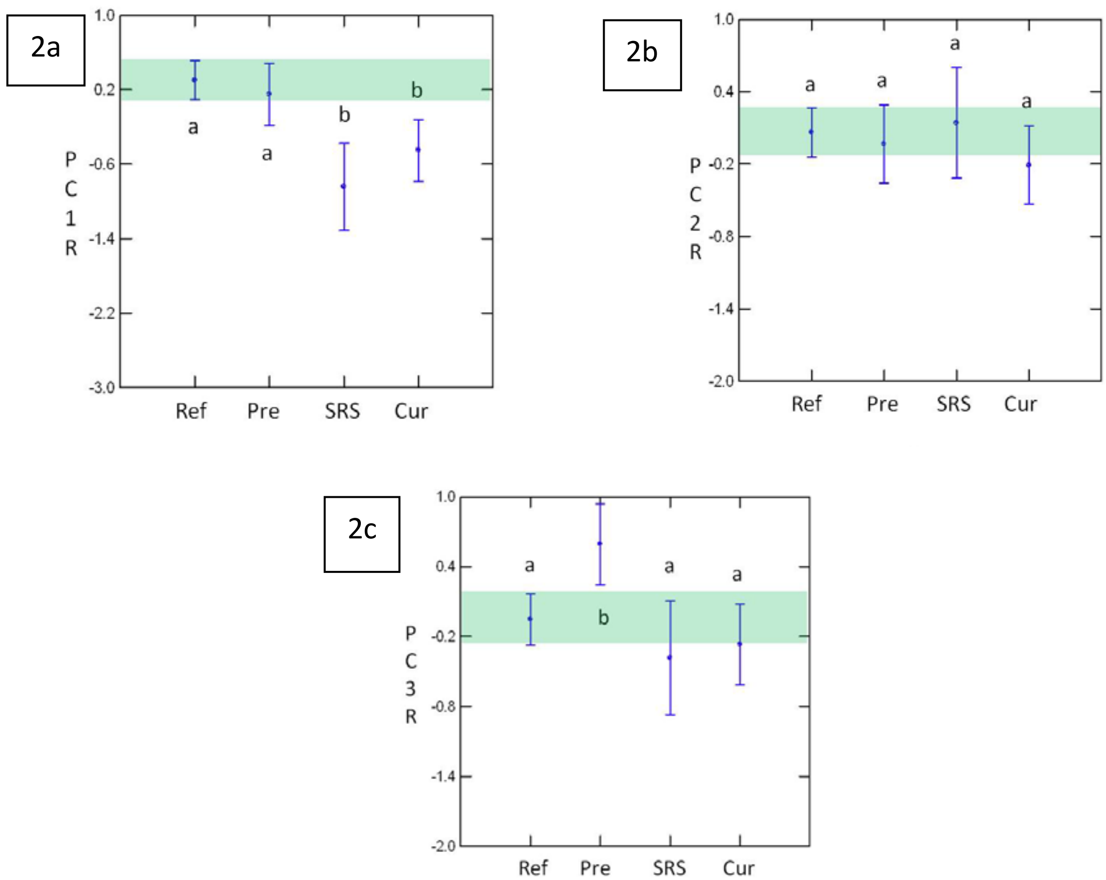

Figure2**)** Relationship of co-varying metrics of macroinvertebrates that colonize leaf packs combined using PCA in the reference regime (Ref) and the disturbance temporal regime in the study sites using ANOVA results. The reference condition is denoted by the shaded area. The results suggest the strong loaders of PC1R which include richnesses of shredders, Trichoptera, and the EPT group, along with Shannon's diversity are higher in reference and the oldest disturbed regime while those disturbed more recently exhibit lower scores of these metrics. This could indicate recovery of the Pre regime to a state more similar to Ref. The only other differences were established by the strong loaders of PC3R, which included richnesses of Ephemeroptera, collectors, scrapers and which were higher in the Pre than any other temporal regime. This could be due to an increase of habitat heterogeneity as the stream is returning to a state of equilibrium similar to the Ref. (x-axis Ref= reference sites, Pre= sites disturbed prior to SRS construction, SRC= sites disturbed during SRS construction and operation, Cur= sites with ongoing active disturbances). Differing letters above error bars denote differences ($p<0.05$). Figure 1a is the graph for Principal Component 1R, Figure 1b refers to Principal component 2R and so on.

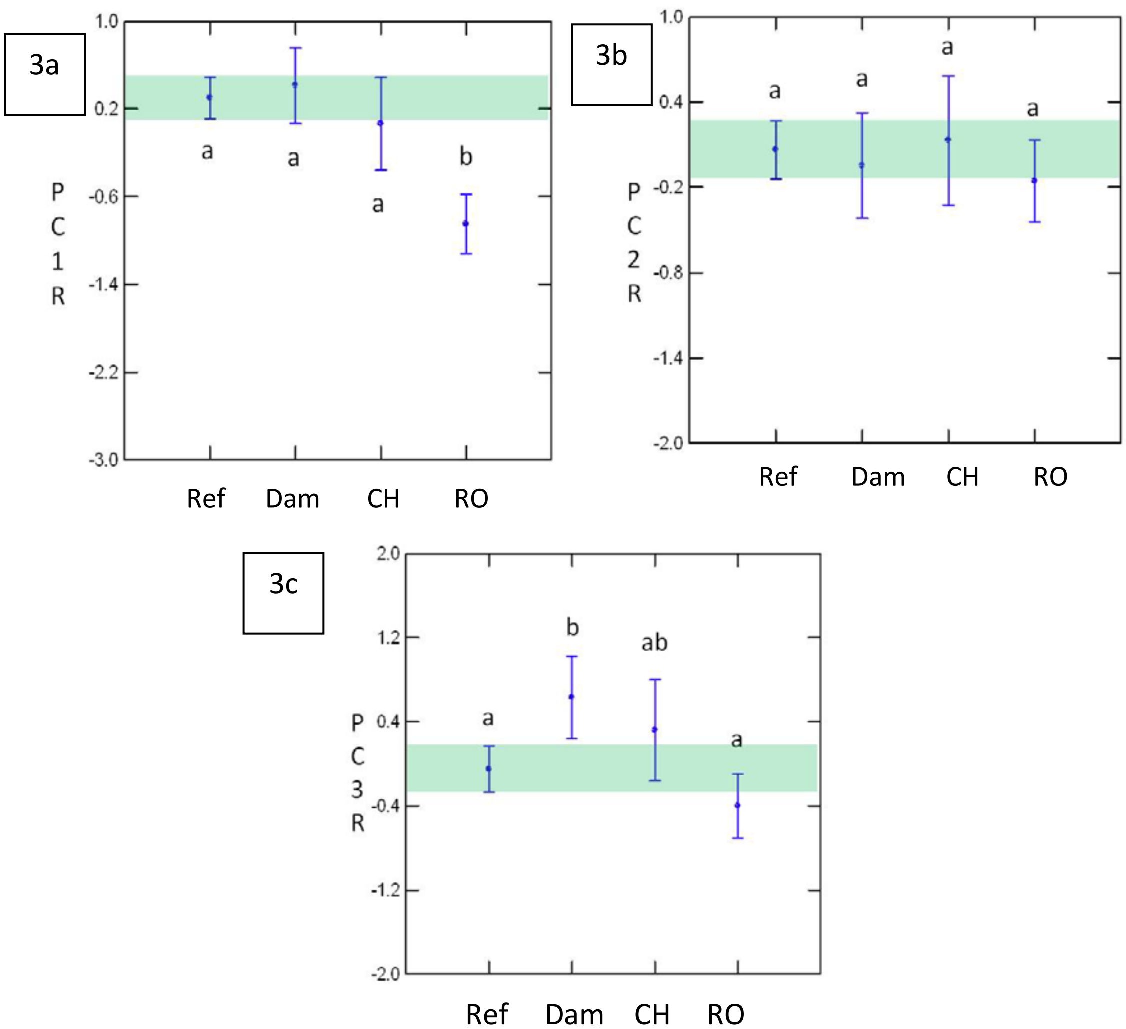

Figure 3) Relationship of co-varying metrics of macroinvertebrates that colonize leaf packs combined using PCA in the reference regime (Ref) and the disturbance categorical regime. The reference condition is denoted by the shaded area. The results suggest the strong loaders of PC1R which include richnesses of shredders, Trichoptera, and the EPT group, along with Shannon's diversity are higher in Ref, Dam and CH regimes while those in the RO regime exhibit lower scores of these metrics. This shows similarities between the Ref Dam and CH regimes. The only other differences were established by the strong loaders of PC3R, which included richnesses of Ephemeroptera, collectors, scrapers and which were higher in the Dam than any other temporal regime. This could be due to an increase of habitat heterogeneity as the stream is returning to a state of equilibrium similar to the Ref. (x-axis Ref= reference sites, Dam= Dam removal sites, CH= channelized sites, RO= runoff disturbed). Differing letters above error bars denote differences ($p<0.05$). The co-varying metrics comprising PC2R were unable to differentiate any categorical regime from each other or from the Ref regime. Figure 3a is the graph for Principal Component 1R, Figure 3b refers to Principal component 2R and so on.

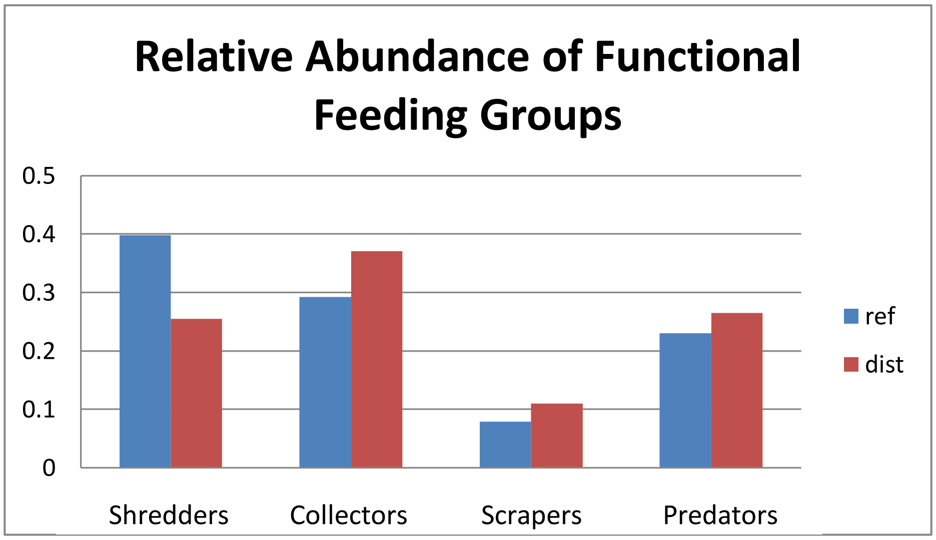

Figure 4) Relative Abundances of Functional Feeding Groups in reference sites (Ref) and disturbed (Dist) regimes. Note that the shedder fucntional feeding group which has been shown to be important in leaf breakdown in headwater streams (Cummins 1989) is nearly twice as great in the reference regime compared to the disturbed regime. The disturbed regime exihibits higher relative abundance of groups more commonly found in higher order streams (Cummins 1989). This futher illustrates the idea that the Ref regimes differs from the Dist regime important for function of healthy streams.

Chapter 3. Drivers of Macroinvertebrates and Restoration Priorities in Headwater Streams across Disturbance Regimes

Abstract

As human populations increase and technology continues to advance, our ability as a species to affect the environment (both positively and negatively) increases as well. About 80% of people live in areas of low water security or high biodiversity threat, due mainly to anthropogenic disturbance. Stream restoration is an accepted approach to manage disturbed freshwater systems and return biodiversity. Justification for undertaking restoration projects are largely based on fishery improvement, public opinion, mitigation, or aesthetics. Studies have questioned decision making procedures that drive restoration projects and subsequent methodologies employed. This study was developed to elucidate metrics that influence macroinvertebrate communities in reference and disturbed sites in Upper Coastal Plain streams. This information could improve resource use and success rates of restorations. Structural and functional variables were examined in 10 reference and 10 disturbed streams. Disturbed streams were classified into two regimes, temporal, based on time since disturbance, and categorical, based on disturbance cause. Some metrics of geomorphology, water chemistry and macroinvertebrates communities differentiated reference from disturbed treatments, while other metrics separated streams within disturbance regimes. It appears that the examined macroinvertebrate communities are influenced by a combination of geomorphology, hydrology and water chemistry. The information gained from this study shows the importance of pre-project study and can be used to inform restoration decisions in areas with available reference systems.

3.0 Introduction

The importance of biodiversity is well known (Dudgeon et al 2005). The high amount of endemic species found in freshwater ecosystems makes conservation of this ecosystem type paramount. Currently we are experiencing a worldwide decline in biodiversity at rates not seen in recorded history with freshwater ecosystems being more affected than their terrestrial counterparts on average (Abell 2002). Freshwater ecosystems are especially vulnerable to shifts in temperature and other forms of pollution (Strayer & Dudgeon 2010). Therefore, it is vital to accurately predict the future of these ecosystems and to improve the already impaired areas.

As human populations increase and technology continues to advance, our ability as a species to affect the environment (both positively and negatively) increases as well (Hilderbrand et al 2005). However, our ability to accurately predict the impact of humans on freshwater ecosystems has remained stagnant at worst or advanced little at best (Downes 2010). About 80% of people live in areas of low water security or high biodiversity threat, due mainly to anthropogenic disturbance (Vorosmarty et al 2010).There are many well-known methods to counteract the loss of freshwater including: water conservation, recycling of sewage water, use of gray water, limiting pollution and other categories of physical disturbance. While these techniques may lower the percentage of humans living in areas of low water security, they do little to address problems concerning biodiversity.

Biodiversity loss is most often due to five major factors: over-exploitation, water pollution, flow modification, species invasion and habitat degradation (Dudgeon et al 2005). A novel area of research known as hydroecology examines the interactions among the physical processes of water and biological aspects of ecology and provides a novel method for examining freshwater systems and biodiversity (Oki and Kanae 2006; Palmer and Bernhardt 2006).This includes aspects such as flashiness and the stream's interaction with soils and macroinvertebrates (Palmer and Bernhardt 2006). As such, the integration of disciplines opens the door for new areas of research that examine how changing hydrologic patterns may influence biological systems and vice versa. This may also be useful for providing a platform for evaluating the ecological response to physical disturbances in streams and for assessing the need for restoration (Dufour and Piegay 2009; Lake et al 2007; Palmer et al 1997).

Stream restoration is an accepted approach to manage disturbed freshwater systems and return biodiversity (Lake et al 2007). It is a billion dollar per year industry with most projects costing more than US 100,000 dollars (Bernhardt et al 2005). Physical alteration of channels, enhancing riparian vegetation, removing impediments to flow or fish movement (often dams), or limitation of livestock access through fencing are a few of the most commonly used restoration methods (Bernhardt et al 2007). Despite the variety of methods employed, justification for undertaking restoration projects are largely based on fishery improvement, public opinion, mitigation, or aesthetics (Kristensen et al 2012; Roni et al 2008; Clewell and Aronson 2006; Hildebrandt et al 2005; Karr 1999). Studies have questioned decision making procedures that drive restoration projects and subsequent methodologies employed (Kristensen et al 2011; Sudduth et al 2011; Jahnig et al 2010; Palmer et al 2010). Clear definitions of project goals or ideas of what goals are realistic and biologically meaningful are also often absent in many restoration projects (McMillian and Vidon 2014; Jahnig et al 2011;Palmer et al 2007; Sudduth et al 2007; Woolsey et al 2007; Giller 2005; Palmer et al 2005).

It has been suggested that a comprehensive collection and analysis of pre-restoration baseline data may be a way to address the deficiencies noted above (Downs et al 2011; Palmer et al 2005; Boon 1998). However, thorough collection of baseline data is most often confined to large restoration projects, while smaller projects tend to rely on basic visual surveying techniques (Downs et al 2011). By not fully understanding the influence of a disturbance on a streams' physical, biological and chemical state, development of quantifiable restoration goals are somewhat compromised (O'Donnell and Galat 2008). Baseline data can clarify existing differences between reference and disturbed streams in a system. This information can be used to decide which streams need restoration and then create biologically relevant goals for those streams (Kosnicki et al 2014). After goals are established, methods can be specifically tailored for success. This addresses the overused but under proven idea that restoring structure necessarily leads to improved function (i.e. "field of dreams hypothesis") (Palmer et al 2010; Hilderbrandt et al 2005) by allowing managers to choose more need based specific methods that may not involve structural alteration or at least not to the point of excluding other viable methods. Baseline information can also allow us to better define, or narrow, the post-project success criteria and subsequent monitoring. In summary, data gathered before restoration can be used to create objectives, aid in choosing restoration methods, and for assessing project performance (Tompkins and Kondolf 2007; Gillian et al 2005).

Current stream restoration techniques tend to be centered on the alteration of in-stream habitat structure (Bernhardt et al 2007; Bond and Lake 2003). Recently, questions have been posed regarding the effectiveness of structural restoration on water quality and stream biology (Hoellein et al 2012; Sudduth et al 2011; Jahnig et al 2010; Palmer 2010). Yet, more than 6,000 restoration projects in the US have incorporated in-stream habitat alteration this millennium with mixed results (Miller et al 2010). Baseline data, in the form of geomorphology, hydrology, water chemistry, and biota, could be useful in these instances as well, either by refining the choice of methods for the restoration or by elucidating the resilience of the stream to past disturbance (McCluney et al 2014; Downs et al 2011). Geomorphic data gathered over time can be used to distinguish those streams undergoing excessive aggradation or erosion from more stable streams. Altered hydrology can affect sediment transport, flooding regimes, and cause excessive alteration to stream geomorphology (Jones et al 2000). Water chemistry allows pollution levels of both stream water and storm runoff to be assessed. Biota in the form of macroinvertebrates and fishes are well known to be sensitive to disturbance and indicators of stream quality (Paller et al 2014; Jellyman et al 2013; Kenney et al 2009). By gathering pre-project data on these four stream aspects, decision makers can make more informed choices regarding: site choice, methodology, setting goals and monitoring needs (Zampella et al 2006).

Two aspects of disturbed stream ecology that are rarely addressed are resilience and recovery. Resilience refers to a streams ability to resist change due to disturbance while recovery refers to the ability of a stream to rebound to a new state of equilibrium after disturbance (Bogan et al 2014). Past work has shown that some streams with visible physical disturbances can recover some functions (litter decay, leaf pack colonization) to a state that is similar to relatively undisturbed reference streams (Biemiller et al *in review)*. In order to confirm the self-correcting ability of these disturbed areas, other variables including structural, water quality and composition of macroinvertebrates in other habitats need to be evaluated. Recovery ability in terms of water quality is rarely discussed but is probably dependent on the source of flow (i.e. spring or run-off), time since the disturbance occurred, and the nature of disturbance. Resilience of biota can depend on secondary factors such as the state of refugia after disturbance, while recovery may hinge on other factors such as population sinks and dispersal ability (Lake 2003). As such, a thorough examination of the existing condition of a disturbed stream should provide needed evidence to determine whether restoration is needed.

The focus of this study was to identify criteria that can be utilized in the decision making process for determining stream restoration need. Baseline data (water chemistry, hydrology, geomorphology, macroinvertebrate occupancy) will be compared between streams with visible structural disturbance and those exhibiting low disturbance characteristics to identify metrics that prove useful as indicators of functional, or active, disturbance. This data will also prove useful for evaluating stream recovery or resilience. Through these evaluations, metrics for prioritizing needs for restoration will be developed to aid land managers in assessing stream condition and need for restoration.

3.1 Methods

3.1.1 Study Sites

This study was conducted at the Savannah River Site (SRS) in conjunction with the University of Georgia's Savannah River Ecology Laboratory and the USDA Forest Service - Savannah River. SRS is located in South Carolina encompassing parts of Aiken, Barnwell, and Allendale counties and borders the Savannah River (Wyatt and Harris 2004). SRS covers over 80,000 ha and its primary function is to process and store nuclear materials to aid in defense and nuclear non-proliferation policies of the United States of America (Wyatt and Harris 2004). Prior to construction of the SRS, the land was used primarily for agricultural purposes and was home to several small towns including Ellenton. Bottomland forests were also prevalent on the site, primarily in the Savannah River floodplain. Construction of the SRS began in 1950 with the first reactors going critical two years later (Smith et al 2001). In 1972 the SRS became the United States' first National Environmental Research Park (Smith et al 2001). Currently, SRS is mostly comprised of forested land along with several industrial areas (Wyatt and Harris 2004).

Streams spanning a broad temporal disturbance gradient ranging from likely the early 19th century or earlier (White 2004; Brooks et al 2000; White and Gaines 2000) to active impacts from industrial areas were identified. Assessments required examination of aerial photos (1938-2010), LiDAR imagery (2009), existing GIS data and maps (1943 to current) to identify disturbances such as flow impediments, erosion, or channelization. A significant contribution of the assessment stemmed from extensive ground surveys. Study streams were walked from their confluence to near the drainage divide. This included all valleys with perennial or intermittent channels as well as significant ephemeral channels. The ephemeral channels were particularly important to determine where outfalls from industrial areas entered streams since they are often located at the head of ephemeral valleys. Disturbances were noted and waypoints saved. The ground survey located disturbances and stream features that would otherwise have gone undetected (Fletcher et al 2011). Twenty sites were chosen for this study with ten reaches designated as reference and ten as disturbed sites. These reference sites were chosen through examination of historical data and visual appraisals and represent examples of the least disturbed areas. The disturbed sites were similarly evaluated and all would likely be considered candidates for restoration based on current evaluation standards.

All sites chosen for this study (n = 20) represent headwater 1st to 3rd order streams (Fletcher et al 2011). They were separated into reference streams (Ref) (n = 10) and disturbed streams (Dist) (n = 10). The Dist sites were further separated into temporal regimes and disturbance categories. The temporal regimes included those disturbed prior to SRS construction (Pre) (n = 4). Other temporal regimes included those disturbed during construction and nuclear operation of SRS (SRC) (n = 2) and those exhibiting current, or active, disturbance (Cur) (n = 4). Disturbance categories included those formerly impaired by dams (Dam) (n = 3), sites that have been channelized (CH) (n = 2) and those affected by runoff (RO) (n = 5). While Ref and Dist sites are balanced, it was not possible to maintain a balance of the temporal regimes or disturbance category. There also exist several noteworthy crossovers between the temporal and categorical regimes. For example, three Cur streams happen to be in the Runoff disturbance type and streams containing evidence of removed dams were all impacted during the Pre temporal regime.

Reference sites were the least disturbed, containing intact riparian zones and generally low incision (Table 1). These included: two streams in the Tinker Creek watershed; TC3, a small 2nd order stream and TC5, again a 2nd order stream with a larger drainage area than TC3. Three Ref streams were located in the Upper Three Runs watershed; U10, a 2nd order stream, U36A a 1st order stream with a drainage area larger than 4km^2, and MQ8 a small stream located further from roads than any other site included in the study. Two Ref streams were located in the Meyers Branch watershed; MB6 a large tributary of Meyers Branch draining over 25% of the watershed and MB9 a 1st order stream located in a deep valley. Two Ref streams were also located in the Mill Creek watershed, MC5A a 2nd order stream and MC6 a 3rd order stream that eventually runs into beaver impacted areas. The Pen Branch watershed contained one Ref stream, PB3 a small 2nd order stream that drains a long, narrow basin.

Pre sites included channelized (MC5B) and previously dammed areas (TC2A, TC2C and MB7.5) (Table 1). Although no specific date could be assigned to the channelization of MC5B, evidence from the area strongly suggests that it took place prior to SRS construction (Fletcher et al 2012). Both sites on TC2 were previously impaired by mill dams and were flooded in aerial pictures from 1951 (Fletcher et al 2012). The breeched dam remains along with a stand pipe with indicators of more recent alteration than those found on TC2 (Fletcher et al 2012).SRC sites include two reaches(PB4 and MBHW) with evidence of (Fletcher et al 2011). PB4 contains a large scour bowl and received input from outside the watershed from six outfalls (Fletcher et al 2012). MBHW was altered during construction of railroads, now unused, on site (Fletcher et al 2012). Sites of ongoing disturbance (Cur) included active run-off sites from parking lots (U6 and MQHW), roads (U36C) and railroad structures (U8). U6 has a head cut over 7m created by runoff. Two check dams for sand entrapment are also locating on this 1st order stream (Fletcher et al 2012). MQHW was altered by railroad construction and continues to be impacted by a head cut filled with boulders and steel plating and areas that are armored with rip-rap (Fletcher et al 2012). Recent die off of riparian trees was also observed in MQHW. Current construction activities in the U8 watershed have resulted in significant run-off and streambed scouring even with the addition of a large retention basin constructed to buffer flows (Fletcher et al 2012). U36C was channelized during the construction of highway 278 in 1965 and continues to be impacted by the road today.

Sites categorized by disturbance type included three streams with abandoned dams, one from a narrowly breached farm pond (MB7.5) and two narrowly breached mill dams (TC2A and TC2C) (Table 1) all with dam remnants remaining in stream and along the banks. The pond dam on Meyers Branch was breached in the early 1950's while the Tinker Creek dams were breached prior to 1940 (Fletcher et al 2011). Channelized sites MC5B and U36C were obviously straightened at some point prior to construction of the SRS but an exact date is unavailable (Table 1). The Upper Three Runs site (U36C) is located downstream from U36A on the side of a major highway (US-278) while the Mill Creek site (MC5B) is isolated downstream from MC5A. These sites show evidence of past incision, but little active erosion (Fletcher et al 2011). Five sites were classified as receiving run-off from industrial areas: two in Upper Three Runs (U6, and U8), one in Pen Branch (PB4), one in McQueen Branch (MQHW) and one from Meyers Branch (MBHW).

3.1.2 Geomorphic Variables

<u>*3.1.2.1 Cross Section Measurements*</u>

Once a stream was designated for inclusion in the study, monitoring reaches were established at each site and marked with metal fence posts that were driven into the floodplain on each side of the stream, approximately 1m behind bank-full levels at

30m intervals. The length of the reach was determined by the width of the stream (150m for narrow streams and 210m for streams over 2m wide). Beginning in 2010, a level line was strung between the posts and measurements were recorded from the channel bottom to the string at 0.5m increments to determine channel cross-section dimensions. Readings were also taken at bankfull and edge of water. The data were then entered into RIVERMORPH™ 4.3 Software Package, a stream geomorphology and design software package, and graphs of each cross-section were constructed. This was repeated annually through 2013. The yearly cross-section graphs were then overlaid and the change in bankfull area was calculated. These changes were then averaged for each study reach in order to reduce the impact of any single cross-section aberrations.

3.1.2.2 Streambed Stability

Streambed stability was measured annually with a custom penetrometer for each reach. The penetrometer consisted of a graduated pole (approximately 2-m tall by 3-cm diameter) and a 12-kg hanging scale attached to its top. An initial water depth reading was taken and the scale pulled to the 12-kg mark causing the pole to be driven into the streambed. At this point, a second reading was recorded at the water surface. Penetration was calculated by subtracting the first reading from the second. The average penetrations per cross section and per reach were then calculated. This measurement was intended to clarify the role of exchange between stream water and ground flow (hyporheic exchange) and aid in understanding stream leakage as both are dependent on flow and the streambed substrate (Packman and Salehin 2003).

3.1.2.3 Other Geomorphic Variables

Upon entering cross section measurements into RIVERMORPH™ software, the program was able to use the values to draw cross sections and then calculate values for wetted width, wetted perimeter, hydraulic radius, entrenchment ratio and width-depth ratio for each cross section. Values were averaged yearly for each study reach. Yearly change was also calculated for all values and averaged in order to limit the influence of any single year aberrations.

3.1.3 Water Quality Variables

3.1.3.1 Grab Samples

Two grab samples (250ml) were taken at the zero meter cross section each year of the study. The samples were frozen and returned to the laboratory at the University of Kentucky for analysis. After thawing, samples were analyzed for chloride (Cl^-), sulfate (SO_4), magnesium (Mg^{2+}), calcium (Ca^{2+}), potassium (K^+), sodium (Na^+), nitrogen in the form nitrate (NO_3-N), nitrogen in the form of ammonia (NH_4-N), phosphate (PO_4), total organic carbon (TOC), alkalinity (Alk), specific conductivity, and pH. Alkalinity and pH were measured with an Orion 940/960 auto-titration combo. Samples were titrated to a 4.6 endpoint with 0.02 N HCl and analyzed using an ORION auto-titrater. Specific conductivity was determined using an YSI conductivity bridge (Scientific Division Yellow Springs Instrument Co., Inc, Yellow Springs OH, USA). Chloride, sulfate and phosphate were measured using a Dionex 2,500 ion chromatograph system (Dionex, UK). Cationic (Ca, Mg, Na, and K) solutes were analyzed with a GBC SDS-276 atomic absorption spectrophotometer (GBC Scientific Equipment Pty Ltd, Australia) using the Direct Air-Acetylene Flame Method. Presences of nitrogen containing compounds were measured using a Bran and Luebbe Auto-Analyzer (SEAL Analytical Inc. Mequon Wi USA) using the colormetric method. Finally, TOC was analyzed using a Shimadzu TOC 5000 A (Shimadzu Scientific Instruments, Columbia MD USA). All methods followed those outlined in APHA (1992).

3.1.3.2 Storm Water Sampling

Beginning in the summer of 2012, one ISCO GLS automated compact sampler (Teledyne Isco, Inc., USA) was employed at the downstream cross-section of each reach to collect storm samples. The sampler was positioned in the riparian zone and contained a tube that extended to the water column of the stream. The sampler also contained a liquid level actuator that initiated sample collection until the water level in the stream rose at least 1.5 inches indicating a storm event. Once activated, 80ml samples were taken at 15-minute intervals for a period of 24 hours, or until stream level decreased below the level of the actuator and composited in a single container located inside the sampler. The container was removed from the sampler, shaken for 1 minute and a 1 liter subsample was taken. The subsample was frozen and subsequently analyzed for total suspended solid concentration following the method outlined in APHA (1992). At least five storm events were collected from each study reach.

3.1.3.4 Stream Flashiness

Beginning in the summer of 2011, Sonlinst® Levellogger (Sonlinst, Canada) pressure transducers were installed in PVC well casings within each study reach. The transducers were set to record water level and temperature in fifteen-minute increments. The data were downloaded using Sonlinst® software version 3.4.1 and corrected using Sonlinst® baraloggers (Sonlinst, Canada) that were hung from trees within the study area. Flashiness (the amount of derivation from the mean flow) was calculated using the Richards-Baker index (Baker et al 2004) at each reach.

In 2012 and 2013, the graphs from the levelloggers and the depth of the corresponding cross section were used to determine the number of flooding events that occurred for each reach. Average floods per year were then calculated.

3.1.4 Macroinvertebrate Sampling

Beginning in the summer of 2011 and ending spring 2012, macroinvertebrates were sampled seasonally at each reach. Seasonal sampling was performed to account for shifts in taxonomic richness and diversity that occur throughout the year (Beche et al 2007). Summer samples were gathered July 30 to August 6 2011, fall samples were collected November 19 to November 25 2011, winter samples were gathered January 6 to January 12, 2012 with spring samples taken March 14 to March20, 2012. Samples were gathered using a standard D-frame kick net and kicking for 60 seconds following Lazorchak et al(1998). A standard kick was followed by two scrubbing motions with the kicking foot and then the process was repeated. Two habitats were sampled to account for microhabitat influences on community composition (Merten et al 2013; Costa and Melo 2008).One sample was taken from a mid-stream run, a second sample was taken from rootmats. The kick net was placed at the downstream end of the rootmat and the rootmat was then disturbed through kicking similar to the process for mid-stream sampling.

All samples were preserved using 70% EtOH. Sediment and detritus were separated from macroinvertebrates, which were subsequently identified to the lowest possible taxon (most often genus) and grouped into functional feeding groups (FFG) as outlined byMerritt, Cummins and Berg (2008). The exceptions to this include non-biting midges, bivalves, snails, worms, shrimp and crayfish that were also excluded from FFG. Total richness was calculated for the FFGs and select groups, Ephemeroptera, Plecoptera, Trichoptera by order and combined (EPT) by combining all samples by reach. Counts per unit effort and relative abundances were calculated for the above groups and Chrironomidae to be used in statistical analysis.

3.1.5 Statistical Analysis

3.1.5.1 Geomorphic Variables

Structural variables were log transformed (ln(1+x)) to reduce skewness. All structural variables were analyzed individually with a generalized linear model (GLM): disturbance; year; disturbance*year. Principal components analyses (PCA) were performed on two groups of the structural variables. The first group was comprised of the absolute values of yearly change in bankfull area, wetted width, wetted perimeter, width depth ratio, entrenchment ratio and hydraulic radius. The absolute values were used to examine the stability of each reach. The resulting principal components were labeled PCSC# for principal component structural changes. The second group was comprised of averages of overall changes per site in bankfull area, hydraulic radius, width to depth ratio, average entrenchment ratio, and wetted width, along with averages of streambed penetration, width depth ratio, entrenchment ratio, hydraulic radius.

The components created by this analysis were labeled PCSA# for principal component structural averages. The resulting principal components were compared to the three disturbance classifications (1: reference or disturbed; 2: disturbance temporal regimes; 3: categorical regimes) using the general linear model (GLM): disturbance regime; year; disturbance regime*year) for the components created from the first group of structural variables and using ANOVAs for those of the second group. Penetration values (used as an analog for stream bed stability) were analyzed using the same GLM as above while average penetration was compared to disturbances with ANOVAs. Tukey's post-hoc tests were used to differentiate regimes. All statistical tests were completed using SYSTAT® 13 statistical package (SYSTAT® Software Inc. 2009).

3.1.5.2 Water Quality

Variables measured from grab samples were log transformed to reduce skewness. The general linear model: disturbance; year; disturbance*year, were run for each of the 13 variables tested by temporal regimes and disturbance class. Again, Tukey's post-hoc tests were used to differentiate within regimes.

3.1.5.3 Storm Water Samples

Average TSS from storm samples were analyzed using ANOVAs and Kruski-Wallis testing. Samplers were activated when water exceeded a level that was approximately 4 to 10cm over minimum flows, depending on stream size, in order to ensure that the samples represented actual storm events (Harmel et al 2003).

3.1.5.4 Stream Flashiness

Flashiness results, in the form of the Richards-Baker index were analyzed yearly with ANOVAs and followed by Tukey's post-hoc tests to differentiate within regimes. Flooding events were analyzed using the GLM disturbance, year, disturbance*year and Tukey's post hoc testing to further differentiate between treatments.

3.1.6.5 Macroinvertebrate Variables

Macroinvertebrates per unit effort and richness were log transformed as above while relative abundance measures were arcsine transformed (arcsine($\sqrt{x}$)) to reduce skewness. PCAs were initially run using macroinvertebrates collected per unit effort of the following variables: shredders, collector/gatherers, scrapers, predators, Ephemeroptera, Plecoptera, Trichoptera, EPT, Chironomidae, and total macroinvertebrates. The resulting variables were labeled PCMC# for principal component of macroinvertebrate count.The second PCA used the 11arcsine transformed values: relative abundances of shredders, collector/gatherers, scrapers, predators, Ephemeroptera, Plecoptera, Trichoptera, EPT, and Chironomidae along with Simpson's and Shannon's diversity. The variables created from this analysis were labeled PCRA# for principal component of relative abundances. The final PCA was run on the richness of total macroinvertebrates, EPT, Ephemeroptera, Plecoptera, Trichoptera, and the same functional feeding groups and labeled PCR# for principal component of richness. All resulting components were compared to the three disturbance regimes using the model: disturbance; season; disturbance*season; habitat(disturbance).

3.2 Results

3.2.1 Geomorphic Variables

A summary of data from the geomorphic variables can be found in Table 2. Only one structural variable was able to differentiate Ref from Dist regimes when analyzed using the GLM (Table 2). The absolute value of change in bankfull area was higher in the disturbed regime compared to the reference (R^2 = 0.123; status p = 0.05; year p = 0.42; status*year p = 0.59).

Two principal components were developed using yearly change of structural variables and accounted for 54% of the total variance explained (Table 7). The first principal component of yearly change of structural variables (PCSC1) explained 33.94% of the total variance and was strongly loaded by changes in bankfull area and hydraulic radius and weakly influenced by changes in wetted perimeter and width to depth ratio.

PCSC1could not distinguish Ref sites from Dist sites [R^2 = 0.131; Ref vs. Dist p = 0.15; year p = 0.16; Ref vs. Dist*year p = 0.44].However, this component was significantly higher in sites disturbed in the SRC or Cur than the other temporal regimes [R^2 = 0.382, temporal regime p < 0.01; year p = 0.36; temporal regime*year p = 0.72] yet no differences were found in disturbance category [R^2 = 0.277; category p = 0.10].PCSC2 explained 18.47% of the total variance and was strongly influenced by change in wetted width and width to depth ratio with no weak loading components. No differences were found using PCSC2 between reference and disturbed areas [R^2 = 0.089; Ref vs. Dist p = 0.37; year p = 0.18; Ref vs. Dist*year p = 0.80], within temporal regimes [R^2 = 0.208; time p = 0.74; year p = 0.63; time*year p = 0.34], or within disturbance categories [R^2 = 0.166; category p = 0.53].

Three primary components of the average structural variables explained 77.38% of total variance. PCSA1 explained 30.84% and was negatively loaded by the width to depth ratio average and width to depth ratio average yearly change. Average hydraulic radius strongly loaded and average penetration weakly loaded PCSA1. GLM testing of PCSA1 could not distinguish reference from disturbed sites [R^2 = 0.102; Ref vs. Dist p = 0.17]. PCSA1 was shown to be higher in Cur than Pre [R^2 = 0.451; time p = 0.02] and Runoff was higher than Dam [R^2 = 0.443; category p = 0.02].PCSA2 explained 26.25% of total variance and was loaded by average changes in bankfull area, hydraulic radius and wetted width. Again no differences were observed between Dist and Ref [R^2 = < 0.001; Ref vs. Dist p = 0.94]temporal regime [R^2 = 0.12; time p = 0.53] or disturbance categories [R^2 = 0.233; categories p = 0.53]. Component 3 (PCSA3) was strongly influenced by average entrenchment ratio and the average change in entrenchment ratio, explaining 20.29% of the total variance. This final component revealed no differences between Ref and Dist [R^2 = 0.091; status p = 0.20]. PCSA3 was shown by GLM and Tukey's post-hoc testing to be higher in Cur than Ref [R^2 = 0.381; time p = 0.05] and higher in RO than Ref[R^2 = 0.417; category p = 0.03].

3.2.2 Water Quality Variables

3.2.2.1 Grab Samples

Several variables exhibited statistical differences between Ref and Dist sites when analyzed using the GLM. Specific conductance, calcium, sodium, potassium phosphate, and alkalinity were higher in Dist than Ref (Tables 3).In the temporal regime, alkalinity, potassium and sodium were higher in Cur than other regimes, while magnesium was higher only in Cur. Sulfate was higher in SRC than Pre and calcium was higher in Pre than Ref. By disturbance category, potassium, sodium and nitrate were higher in Runoff than any other category regime. Sulfate, magnesium and alkalinity were higher in RO than either Ref or Pre and specific conductance was higher in RO compared to Ref.

3.2.2.2 Storm Water

The average amount of TSS per storm sample by site ranged from 22.6mg/l to 1387.7mg/l with a mean of 186.5mg/l. Although no statistical difference was found for temporal regimes (p = 0.22) or disturbance category (p = 0.08), Cur and RO regimes tended to have higher levels of TSS (558.5mg/l and 489.7mg/l, respectively) than Ref (98.3mg/l). Ref sites, while not significantly different from any regime, (all *p's* > 0.22), tended to have higher baseline TSS (35.3mg/l) than Dist (27.7mg/l). Kruskal-Wallis testing of storm water sample TSS in specific sites showed several differences (Table4). The reaches U6, MQHW and U10 were shown to have higher TSS in storm samples than several Ref and Pre reaches (*p's*< 0.05) (Table 4).

3.2.2.3 Stream Flashiness

Analysis of the Richards-Baker index of flashiness scores showed no differences between Ref and Dist streams. However, in 2012 the Richards-Baker scores were significantly different in the temporal regime (*p*< 0.05). Tukey's post-hoc test showed the Cur regime scored higher than the Pre regime (*p* < 0.04). No other statistically differences were apparent. Surprisingly, the RO regime was statistically similar to Ref (*p* = 0.06). Many data sets were incomplete due to equipment issues. In several sites, the level loggers were removed from their wells by debris snagging on the lines anchoring the loggers to the bank. Yet, when their respective data sets were complete, U6, U8, U10, MQHW and PB4 all scored near the top end of the results.

While no differences were apparent between Ref and Dist regimes (p = 0.58) or temporal regimes (p = 0.14) in flooding frequency, there were differences with the categorical regime (p < 0.01). Tukey's post hoc testing showed flooding frequency to be lower in RO regimes than any other (all *p's*> 0.05)(Table 6).

3.2.3 Macroinvertebrate Variables

Similar numbers of total macroinvertebrates, chironomids and EPT were collected from reference (Ref) and the disturbed (Dist) reaches (Table 6). The number of shredders was higher in Ref regime than Dist (Table 6).PCA of the 10 macroinvertebrates collected per unit effort of time variables resulted in three components which accounted for 73.88% of the total variance explained. PCMC1 was strongly loaded by richness of EPT, shredders, Plecoptera, Trichoptera, collector, and total macroinvertebrates and was responsible for 53.1% of the total explained variance. Analysis of PCMC1 showed Ref scored higher than Dist except winter and root habitats were higher than mid-stream[R^2 = 0.262; Ref vs. Dist p = 0.02; season p = 0.41; Ref vs. Dist*season p = 0.10; habitat(Ref vs. Dist) p < 0.01].

The GLM of temporal regimes only showed root habitats scored higher than stream [R^2 = 0.230; time p = 0.11; season p = 0.81; time*season p = 0.23; habitat(time) p < 0.01] with similar results in disturbance categories [R^2 = 0.246; category p = 0.06; season p = 0.76; category*season p = 0.14; habitat(category) p < 0.01].PCMC2 accounted for 10.61% of total variance explained and was strongly loaded by both predators and Chironomidae and weakly by total macroinvertebrates. No differences were found between Ref and Dist PCMC2 [R^2 = 0.065; Ref vs. Dist p = 0.44; season p = 0.57; Ref vs. Dist*season p = 0.62; habitat (Ref vs. Dist) p = 0.82]. Analysis of temporal regimes showed higher scores in Pre and SRC compared to Cur[R^2 = 0.171; time p < 0.01; season p = 0.73; time*season p = 0.44; habitat(time) p = 0.26]. However, no differences were found across disturbance categories [R^2 = 0.115; category p =0.34; season p = 0.75; category*season p = 0.57; habitat(category) p = 0.28].The final richness component, PCMC3 explained 10.2% of the total variance was strongly loaded by Ephemeroptera and scrapers while being weakly influenced by total macroinvertebrates. No differences were found between Ref and Dist [R^2 = 0.065; Ref vs. Dist p = 0.44; season p = 0.57; Ref vs. Dist*season p = 0.62; habitat(Ref vs. Dist) p = 0.82]. While analyses of temporal regimes [R^2 = 0.250; time p = 0.12; season p < 0.01; time*season p = 0.40; habitat p = 0.11] and disturbance categories [R^2 = 0.227; category p = 0.38; season p < 0.01; category*season p = 0.37; habitat(category) p = 0.28] showed winter scores to be the highest in both cases.

Primary Component Analysis of 11 relative abundance and diversity measures resulted in four principal components explaining 66.45% of the total variance (Table 8). PCRA1 was loaded negatively by Simpson's diversity and the relative abundances of Chironomids and predators and explained 27.32% of the total variance. Testing of Ref versus Dist showed the Ref regime to be higher than Dist [R^2 = 0.142; Ref vs. Dist p < 0.01; season p = 0.77; Ref vs. Dist*season p = 0.36; habitat(Ref vs. Dist) p = 0.07]. ANOVAs also showed Ref to be higher than Pre sites and root habitats were higher than stream in all but SRC temporal regimes [R^2 = 0.164; time p = 0.03; season p = 0.67; time*season p = 0.72; habitat(time) p = 0.03] and the Dam regime [R^2 = 0.412; category p = 0.03; season p = 0.93; category*season p = 0.44; habitat(category) p = 0.07]. PCRA2 explained 15.66% of total variance and was strongly loaded by the relative abundances of collectors, Ephemeroptera, the EPT group and Shannon's Diversity. The only apparent difference in testing of Ref versus Dist was PCRA2which scored higher in winter than summer along with showing differences in habitats scoring higher in rootmats than streambed [R^2 = 0.181; Ref vs. Dist p = 0.27; season p.< 0.01; Ref vs. Dist*season p = 0.04; habitat(Ref vs. Dist) p = 0.52].Season of sampling was found to exhibit differences between temporal regimes [R^2 = 0.235; time p = 0.06; season p < 0.01; time*season p = 0.20; habitat(time) p = 0.24], and disturbance category [R^2 = 0.200; category p = 0.35; season p < 0.01; category*season p = 0.28; habitat(category) p = 0.44]. PCRA3 was loaded strongly by Plecoptera, shredder and EPT relative abundances and weakly loaded by Shannon's Diversity and accounted for 12.31% of the total variance explained. Analysis of Ref versus Dist regimes showed root habitats to be higher than stream [R^2 = 0.167; Ref vs. Dist p = 0.25; season p = 0.41; Ref vs. Dist*season p = 0.26; habitat(Ref vs. Dist) p < 0.01]. Similar results were found for temporal regimes [R^2 = 0.196; time p = 0.33; season p = 0.82; time*season p = 0.75; habitat(time) p < 0.01], while testing of disturbance categories not only showed difference in habitat but also showed higher scores in CH versus Dam [R^2 = 0.464; category p = 0.03; season p = 0.93; category*season p = 0.59; habitat(category) p < 0.01]. The final component from this group, PCRA4 explained 11.16% of the total variance explained and was loaded strongly by Trichoptera relative abundances and negatively by scraper relative abundances. No difference were found by analysis Ref vs. Dist regimes [R^2 = 0.059; Ref vs. Dist p = 0.57; season p = 0.40; Ref vs. Dist*season p = 0.72; habitat(Ref vs. Dist) p = 0.57], temporal regimes [R^2 = 0.074; time p = 0.91; season p = 0.72; time*season p = 0.81; habitat(time) p = 0.66] or disturbance categories [R^2 =

0.075; category p = 0.59; season p = 0.44; category*season p = 0.88; habitat(category) p = 0.70].

PCA of EPT richness, combined and by order, total macroinvertebrates and functional feeding groups yielded two components. When analyzed neither of the components were able to show differences in any of the three disturbance regimes (Table 6).

3.3 Discussion

3.3.1 Geomorphology

Flooding frequency is an important metric used to describe stream stability (Rosgen, 1994). Upper Coastal Plain streams have been shown to flood 7.5 times more often than streams in other geographic settings (0.19 years per flood in the Upper Coastal Plain vs. 1.5 years per flood nationwide) (Sweet and Geratz 2003). Consequently, bankfull metrics are very important in assessment. Bankfull area is a reflection of a stream's width and depth and while alone changes in width and depth were not significant, the overall change in bankfull area was the only geomorphic variable that differentiated reference from disturbed, it was almost twice as great in the Dist regime compared to the Ref regime (Table 2). Bankfull area along with a roughness coefficient, hydraulic radius and slope influence stream hydro-dynamics and dictate flooding frequency (Manning 1891).

In this study, streams in the RO regime flooded three times less frequently than Ref regimes and half as often as found in Sweet and Geratz (2003). Healthy Upper Coastal Plain streams require frequent flooding. A disconnect between the stream and its floodplain could indicate disturbance. Floodplain connectivity and flooding frequency allow input from riparian zones and allow sediment deposition on floodplains. Also, these streams must cope with storm events. Streams with poor connectivity have no release for increased flow from storms. This surge of water must then stay in the channel where it can flush small sediments downstream, increase the load of suspended solids in the water column and eroding stream banks (Batalla and Vericat 2009). Increases in bankfull area over time, in combination with increased entrenchment ratio, the ratio of bankfull height to flood prone area, can directly lead to lowered floodplain connectivity and flooding frequency making the changes in bankfull area and entrenchment ratio useful indicators of stream health. Disturbances such as runoff and channelization can lead to these changes though erosion of banks and deepening of the channel. This can mean the beginning of a self-reinforcing cycle where the erosion leads to lowered floodplain connectivity which leads to more power remaining in the channel during storm events which, in turn leads to more erosion.

No other geomorphic variables could be used to differentiate the Ref and Dist regimes (Figures 1 and 2). This could mean one of two things. First, solely examining geomorphic variables may not be sufficient to determine the condition of a stream (Northinton et al 2011). Second, these streams could be more similar than they appear upon visual examination. The 10 streams in the Dist regime probably would be considered good candidates for restoration based upon their visual condition and presence of physical impediments. The inability of the remaining geomorphic variables to differentiate the streams in the Ref versus Dist regime shows that differences in visual appearance do not necessarily correspond to measurable differences.

Similar to overall change in bankfull area, higher yearly change in bankfull area, translates into lowered bank stability. Increases in bankfull area and hydraulic radius lower the chance of flooding with all other conditions being stable. Both of these variables were important in differentiate the Cur regime from the Ref. Although patterns are difficult to discern, results suggest that there was greater incision in the more recently disturbed sites. Incised channels exhibit increased bank erosion which prevents sediment deposition in flood plains and increases water quality degradation.

Often used as an analog for floodplain connectivity, width to depth ratio has been shown to be important for biotic diversity, the increase of width to depth ratio can be another indicator of incision (Sullivan and Watzin 2009; Ward et al 1999). It is not surprising that this variable helped differentiate recently disturbed streams from Pre and Ref regimes. The similarity of Ref and Pre regimes has been noted earlier (Biemiller *in review*) and suggests that the sites with dams removed have had time to return to an equilibrium similar to the Ref. Least square results from generalized linear model of principal components of macroinvertebrate relative abundance variables from kick net samples between Ref and Dist categories supports this idea. It is also possible that the geomorphic variables examined in this study are poor indicators of historic disturbance by dams. Conversely, structural variables both alone and in combination via PCA, were able to distinguish RO and Cur from Ref sites. This also holds true with the differences found between runoff sites and both dams and reference sites. This similarity between the RO and Cur regimes was expected due to the crossovers in the disturbance breakdown. Past projects which have on altering physical channel structure in order to restore biota may use data like this and the established deleterious effect of scour on macroinvertebrates to justify their methods (Lepori and Hjerdt 2006; Bond and Lake 2003). Yet, tests on other measures of structural change were unable to detect differences in disturbance regimes.

3.3.2 Water Quality

In general, water quality characteristics for all study reaches were high. Most of the water quality variables examined were within or near those of Upper Three Runs located upstream of SRS (USGS 2000) (Table 3). This branch of Upper Three Runs has been proclaimed to exhibit the highest biodiversity of any stream in the western hemisphere and the 2nd most biodiverse in the world (SREL 2007; Voelz and Mcarthur 2000). Higher alkalinity level was one notable exception in both the Ref (19.16) and Dist (33.65) regimes as compared to the long term Upper Three Runs data set (maximum 8.55) (USGS 2000). Increased alkalinity levels in streams of the current study are likely due to differences in analytical methodology used. The USGS used a field titration method, while I used a laboratory titration method. It has been noted previously that field titration consistently provided lower results than laboratory analysis (USGS 2000).

Even though overall water quality was considered good, some differences between Ref and Dis were observed. The variables that did show differences between regimes were often associated with dissolved solids (or dissolved salts). Specific conductance (SC), an estimate of the total dissolved solids (minerals, salts, and ions) in water (Barton 2011), was higher in Dist than Ref (Table 3). Other variables found to be statistically different (Ca, K, Na and P) were likely derived from local soils or sediments and are likely responsible for the rise in specific conductance. The increased bankfull areas observed in Cur and Runoff streams clearly resulted from sediment movement which may be the source of the increased dissolved solids and associated elemental constituents. Although streambeds in this study were mostly comprised of sand, runoff and incision can cut into riparian soils and introduce other soil types (i.e. clay or silt) that easily dissolve in water and result in increased SC (Davies-Colley et al 1992). Sediment cores from several of the study streams were collected and are currently being analyzed to determine sediment origin (stream/riparian area or upland derived from erosion) and time since emplacement (recent versus historic). Those results are the focus of a separate MS thesis and will be presented elsewhere.

3.3.2.1 Stream Flashiness

Flashiness in headwater streams can have a profound effect on biota (Stanfield and Jackson 2011; Boulton et al 1992). A stream's response to storm events can limit intra-species competition by decreasing the population through reducing habitat availability (Gore et al 2001; Feminella and Resh 1990). On the other hand, these events can also be important for creating habitat heterogeneity and have a net positive effect on biodiversity (Lepori and Hjerdt 2006). The higher Richards-Baker index flashiness scores in Cur compared to Ref during 2012 is likely attributed to runoff as described elsewhere(Mudd 2006; Negishi et al 2002).Industrial sites often contain large areas with impervious surfaces (i.e. parking lots, roofs) that inhibit soil infiltration and promote surface runoff. The energy of flow in the runoff, and diminished lag-time between storm event start and stream response, often results in high erosion and sedimentation rates (Houser et al 2006). The 2012 water level data from this study suggests that Cur sites are flashier than Ref sites. Thus, increased flashiness could be a contributor to the erosion that ultimately is reflected in water chemistry. Higher flashiness may also preferentially remove streambed sediments and scour channels down to parent material, which are often sandy clays (Batalla and Vericat 2009). Higher flashiness could also lower the retention of leaves and coarse particulate organic matter (Koljionen et al 2012).Increased flashiness can also indicate lower flooding frequency. Flashiness is often associated with floodplain connectivity. A disconnect between a stream and its floodplain causes all of the power from a storm to remain within the channel which increases flashiness (Houser et al 2006).

Unfortunately, the elevated flashiness and effects on equipment posed limitations that may have prevented my ability to observe similar results in the other two years of the study. The pressure transducers, which measure changes in stream level, were placed into PVC wells and secured to fence posts that were driven into the streambed. In order to prevent loss of equipment, the pressure transducers were also tethered to structures on the stream bank. When major storm events occurred, debris that was transported downstream often got caught in the pressure transducer tethers and dislodged the equipment. On other occasions, large woody debris collided with the wells and resulted in their movement. These instances led to a significant amount of suspect data and rendered the storm-flow data useless for several precipitation events. Higher flashiness from incised streams (U6, U8, MQHW, and PB4) was observed during events where the loggers remained intact. Continued monitoring with a larger and more comprehensive data set is needed to fully understand these relationships.

3.3.2.2 Storm Water Samples

Analysis of TSS in storm water samples surprisingly showed no differences in any of the regimes (*p's* > 0.05). When analyzed individually, however, several of the reaches exhibited significantly higher TSS levels (Table 4). The sites with higher TSS were often surrounded by industrial areas, such as U6 and MQHW, and in both the Cur temporal disturbance regime and the Runoff disturbance class. One avenue that could yield more definitive results is a closer examination of TSS near the beginning of the storm event. Previous work has noted the importance of the "first flush" referring to the built up sediments early response to storm events (Li-qing et al 2006, Deletic and Maksimovic 1998). Concentrating sampling to early hours of a storm event and reducing time between samples would allow examination of the first flush. It is possible that this may reveal further differences in disturbance regimes.

3.2.3 Macroinvertebrate Variables

The similarity of Ref and Dist regimes with regards to the counts per unit effort of EPTs, and chironomids was surprising, at first, as many studies have shown these groups vary based on stream health (Gafner and Robinson 2007; Niyogi et al 2007; Barker et al 2006; Harding et al 1999). Disturbances, such as channelization, have been shown to result in lower macroinvertebrate density (Negishi et al 2002). Biodiversity has also been shown to vary based on stream condition (St. Pierre and Kovalenko 2014; Heino 2005). As such, the observed similarity in richness between Ref and Dist regimes was unexpected. The fact that richness principal components were unable to differentiate Ref regime in any of the disturbance regimes is in contrast to past studies (St. Pierre and Kovalenko 2014; Heino 2005). Even though there were statistically significant differences in water quality variables between regimes, as discussed earlier, none of the readings were at levels that I would expect to limit macroinvertebrate community composition.

Shredding insects, which are especially important to headwater stream function, (Cummins et al 1989) were collected in much great numbers in reference sites (Table 4).This could indicate a difference in stream function between reference and disturbed areas. Shredders are expected to make up 10% of the total macroinvertebrate abundance in woodland streams (Peterson et al 1989). Although the study by Peterson et al 1989 was done in a different ecosystem, and therefore may not be directly applicable to Upper Coastal Plain streams, there is evidence of a difference in relative abundance of shredders in the Ref (19%) versus Dist (8%).

Shredders have been shown to be important for detritus processing, consuming more than their own body weight daily, which is important to the headwater streams. In addition, they convert large detritus to finer material that can be consumed by organisms downstream, which in turn are important food sources to larger organisms (Cummins 1973). Leaf decay rate, which is often associated with shredder abundance, does not always vary based on disturbance due to possible masking effects (i.e. flashiness difference)in this system (Biemiller et al in review) or others (Hagen et al 2006). Therefore, the difference of shredder abundance becomes more important as an indicator of disturbance and may be explained by one or more of the following: differences in sediment as seen previously, by geomorphic differences in channel structure (i.e. flood plain connectivity) a response to differing water quality, response to altered flashiness, hydrologic regime all of which has been shown to affect available habitat for macroinvertebrates (Sawyer et al 2004; Benke 2001; Lenat 1988; Wood and Armitage 1997; Boulton et al 1992; Feminella and Resh 1990)

EPT, shredders, total macroinvertebrate, Trichoptera and Plecoptera were shown to co-vary. Together these variables were able to differentiate Ref and Dist regimes. This was somewhat intuitive as the orders of EPT are known to be sensitive to many categories of pollution. Also, shredders had been shown to be important in separating disturbed areas from reference in the previous chapter. This ties into the increased flashiness causing removal of detritus, which is vital for shredders, as well as downstream organisms that depend on the shredders to breakdown the organic material into finer particulates that they can use (Koljonen et al 2012).

While the currently disturbed streams may be so impaired as to minimize colonization, the sites disturbed before and during construction have had at least 30 years to recover. In fact, several studies have shown macroinvertebrate communities in some areas may recover one year after dam removal (Doyle et al 2005; Stanley et al 2002). Some of the macroinvertebrate variables were unable to differentiate the SRC temporal regime from others or the CH sites from other disturbance categories due to their loading components or the streams having recovered over time. It is also possible that these results are due to the small number of channelized sites examined in this study of both the SRC and CH reaches (n = 2). Although the results were not as clear with leaf packs in earlier work, as illustrated by the inability of either richness component to make any distinctions among regimes, the variables created by PCA of kicknet data were still valuable in distinguishing Ref from Dist streams.

3.4 Conclusions

A large number of variables were examined to evaluate their usefulness as effective indicators of disturbance. Most variables did not allow differentiation of streams in the Ref regime from those in the three regimes of disturbance. Either these variables were not good indicators of disturbance, or the disturbed streams in this study, chosen due to visual differences from the reference sites, were more similar to those in the Ref regime than they appeared from visual assessments and historical data.

While the discussion has focused on the differences found between reference and disturbed streams, within disturbance temporal regimes, and disturbance categories, the most crucial point may be their similarities. There is little doubt some of these sites are in need of restoration. However, there is also ample evidence that streams disturbed prior to SRS construction and those with dams removed have, or are currently, undergoing a transition to a new equilibrium, similar to findings described by Sawyer et al (2004). Sites from the Pre and Dam regimes exhibited similar scores to the Ref regime in many of the cases, including several water quality measures where Cur and RO regimes were not similar to Ref. In other instances where Ref was separated from either Cur or RO regimes, Pre or Dam regimes were indistinguishable from Ref. This could indicate an ongoing transition or return to an equilibrium similar to that of the Ref treatment. The similarities of the Pre and Dam regimes to the Ref regime suggest that money and effort may be more efficiently spent on other areas and allow those in the midst of recovery to continue unabated. Even the sites disturbed during SRS operation (the SRC treatment) have many parameters that tested similar to reference sites and were found to be different from the Cur regime indicating a shift towards a new equilibrium.

It would be over-reaching to interpret these findings as saying all streams with historic disturbances or removed dams could be left to recover on their own. Each potential restoration site should be examined individually before any conclusion is reached. Even after removal of the primary source of disturbance, an area may be vulnerable to secondary disturbances and continue to degrade. However, this does emphasize the importance of gathering baseline data before restoration because several streams that appear to be severely disturbed closely resemble reference streams in many important variables.

This study has identified several characteristics that provide guidance for land managers contemplating the need for stream restoration in Upper Coastal Plain landscapes. They include geomorphic components (e.g. changes in bankfull area, width to depth ratio or entrenchment ratio), water quality (e.g. specific conductivity or phosphate levels, among others) along with some biotic variables (e.g. shredder or EPT richness). These variables interact with each other to compound problems found in disturbed areas. For example, higher width to depth ratio, an example of floodplain connectivity, can cause changes in hydrology by increasing in channel flow during storms and reducing flooding frequency. This can lead to erosion, which can further alter geomorphology by increasing changes in bankfull area, increasing sedimentation leading to differences in water quality.

Changes in water quality and loss of habitat due to increased flow during storms, can have deleterious effects on biotic communities. Reduction in abundance and diversity of biotic communities can lower breakdown of detritus in streams which can lead to loss of function not only in the headwaters but further downstream.

Examining variables, such as those mentioned above, in each potential restoration site can help to determine specific areas of concern. By identifying the needs at the particular locations, specific biologically relevant goals could be set and restoration methods could be customized to suit the situation. This study, for example, began with ten streams that using existing protocol would be candidates for restoration. The data gathered throughout this work, suggests that only three of these sites (those that are both Cur and RO vary enough from reference sites to be in need of immediate restoration. Theoretically, this should lead to more efficient completion of future restoration projects, thereby saving money by choosing sites that are in need of restoration and tailoring methodology for each sites' needs. Setting and attaining biologically important goals is an important step in keeping public opinion favorable regarding stream restoration which is vital for continued funding of future projects (Dufour and Piegay 2009). Although the category of testing discussed herein is not free of cost, money spent prior to restoration could save expenses in the long run.

Overall, it seems that the data from this study supports the idea that hydroecology can be a good indicator of stream condition. In particular, out of the 10 streams examined in this study only those with on-going runoff disturbances were shown to vary from the Ref regime in enough categories that would warrant restoration activities. This is supported by the literature's long standing call for more of an integration of theory and restoration (Lake et al 2007; Lake 2001; Palmer et al 1997) along with the increasing trend of examining many potential stressors in potential restoration sites (Rasmussen et al 2013; Northington et al 2011; Tullos et al 2009; Zampella 2006).

No single method of sampling metric that managers could rely onto determine the function of streams and which sites are in need of restoration was identified in this study. However, many if not most of these metrics examined herein, can affect the macroinvertebrate communities. This suggests that examining these communities may be the best way to get an idea of the overall condition. After macroinvertebrate community variation from the Ref regime is detected, more elaborate data collection and analysis may be appropriate to better determine the drivers of these differences. Previous work has highlighted the interconnectedness in timescales, causal factors of disturbance, area, physical, chemical, and biological variables (Palmer and Bernhardt 2006; Wohl et al 2005), ideas which the results of this study support. Analyzing a group, in this case the macroinvertebrate community, that responds to many variables, could yield clues regarding the states of other interconnected variables and give managers an idea as to where to concentrate their data collection efforts.

It was not the place of this work to suggest restoration methodology that would be effective for the streams studied. Rather, this work aimed to give land managers guidance for determining appropriate assessment techniques for determining restoration need. To that end, it seems apparent that the use of a reference system and pre-restoration study is invaluable. This is a lesson that can be applied anywhere a suitable reference system exists. Also, this work would suggest that using a collection method (i.e. leaf packs) that stays in the stream can provide a better description of the stream than snap-shot sampling (i.e. kick nets) even when sampled seasonally.

Chapter 3 Tables

Table 1. Study sites and corresponding disturbance regimes located at the Savannah River Site, SC.

Stream	*Reach Abbreviation*	*Length (m)*	*Temporal Class*	*Disturbance Category*
Meyers Branch	MBHW	150	SRC	Runoff
Meyers Branch	MB6	150	Reference	Reference
Meyers Branch	MB7.5	150	Previous	Dam Removal
Meyers Branch	MB9	150	Reference	Reference
Mill Creek	MC5A	150	Reference	Reference
Mill Creek	MC5B	150	Previous	Channelization
Mill Creek	MC6	150	Reference	Reference
Tinker Creek	TC3	150	Reference	Reference
Tinker Creek	TC5	150	Reference	Reference
Tinker Creek	TC2A	210	Previous	Dam Removal
Tinker Creek	TC2C	210	Previous	Dam Removal
Pen Branch	PB3	150	Reference	Reference
Pen Branch	PB4	150	SRC	Runoff
McQueen Branch	MQHW	150	Current	Runoff
McQueen Branch	MQ8	150	Reference	Reference
Upper Three Runs	U6	110	Current	Runoff
Upper Three Runs	U8	150	Current	Runoff
Upper Three Runs	U10	150	Reference	Reference
Upper Three Runs	U36A	150	Reference	Reference
Upper Three Runs	U36C	150	Current	Channelization

Table 2. Changes in geomorphic variables showing the variability are used to show stability of study reaches. Lower change equates to greater stability.

GEOMORPHIC VARIABLES

	RANGE				
VARIABLE	*LOW*	*HIGH*	*MEAN*	*STD*	*SE*
Yearly Δ Bankfull area (m^2)	-4.56	3.03	-0.02	1.21	0.16
Yearly Δ in Wetted width (m)	-1.13	1.34	-0.05	0.37	0.05
Yearly Δ in Wetted perimeter (m)	-1.32	0.98	0.00	0.51	0.07
Yearly Δ in Width:depth ratio	-30.23	20.9	0.86	9.11	1.21
Yearly Δ in Entrenchment ratio	-0.71	0.47	-0.03	0.20	0.03
Yearly Δ in hydraulic radius (m)	-0.19	0.12	0.00	0.04	0.01
Avg. Hydraulic radius (m)	0.46	1.32	0.93	0.33	0.07
Avg. Streambed penetration (cm)	0.82	2.30	1.61	0.30	0.03
Avg. Entrenchment ratio	1.37	1.94	1.57	0.16	0.04
Avg. Width:depth ratio	5.78	23.67	11.37	5.8	1.3
Avg. overall Δ in Hydraulic radius (m)	-0.05	0.03	0.00	0.02	0.00
Avg. overall Δ in Bankfull area (m^2)	-0.69	0.63	0.00	0.39	0.09
Avg. overall Δ in Entrenchment ratio	-0.57	0	-0.23	0.13	0.03
Avg. overall Δ in Width:depth ratio	-8.54	2.57	0.1	2.57	0.58
Avg. overall Δ in Wetted width (m)	-0.34	0.19	-0.09	0.13	0.03

Table 3. Average values of water chemistry in Ref and Dist treatments along with the minimum and maximums of a 30year data set from a main branch of Upper Three Runs (UTR). Upper Three Runs is a known reference stream famed for its biodiversity. This shows that most scores were within the ranges found in Upper Three Runs.NA = not available

Variable	*Ref*	*Dist*	*UTR Min*	*UTR Max*
S.C.*(µS/cm)	27.15	35.22	10	22.0
Chloride (mg/l)	4.55	5.12	0.20	3.55
Sulfate (mg/l)	2.75	4.75	<0.02	5.29
Magnesium(mg/l)	0.23	0.27	<0.10	0.64
Calcium (mg/l)	1.91	2.88	0.10	2.40
Potassium(mg/l)	0.31	0.51	0.10	1.29
Sodium (mg/l)	0.88	2.31	0.20	3.45
Alkalinity (HCO_3 mg/l)	19.16	33.65	1.22	8.55
pH	5.65	5.88	4.5	7.3
Nitrite (mg/l)	0.065	0.14	0.37	1.14
Ammonium(mg/l)	0.04	0.05	<0.01	0.17
Total Organic Carbon (mg/l)	6.51	5.26	NA	NA
Phosphate (mg/l)	0.73	1.33	NA	NA

*= Specific Conductance

Table 4. Lists the reaches with significantly different ($p < 0.05$) storm sample total suspended solids (TSS) values. Reaches with higher TSS values are in the right hand column. Higher TSS equates to higher input from storm events.

Reach	*Reach*	*p value*
MQHW	MC6	0.03
MQHW	PB3	0.01
U10	PB3	0.04
U10	TC2C	0.01
U6	TC2C	0.02
U6	MC5A	0.02
U6	MC5B	0.02
U6	PB3	0.04
U6	TC5	0.02
Kruskal-Wallis Test Statistic = 36.346 *p value* = 0.01 with 19 degrees of freedom		

Table 5. Flooding frequencies are listed by treatment.
Healthy Upper Coastal Plain streams flood at a rate of more than 5 times per year or 0.19 years per flood.

Regime	*Floods per year*	*Years per Flood*
Reference	8.4	0.12
Previous	12.6	0.08
SRC	4	0.25
Current	7.8	0.13
Dam	12.8	0.08
Channelized	13.5	0.07
Runoff	2.6	0.38

Table 6. Total number collected of selected groups of Macroinvertebrates are listed. Differing letters denote significant differences (p> 0.05). Shredders are well known to be important for carbon breakdown in headwater streams.

Stream Class	*SHREDDERS*	*EPT*	*MIDGES*	*TOTAL*
REFERENCE	906a	1239a	2688a	4791a
DISTURBED	381b	942a	2605a	4627a

Table 7. Loading factors of yearly structural change are listed showing the influence of the variables on the components

	PCSC1	*PCSC2*
% Total Variance Explained	30.8	21.6
Bankfull Area Δ	0.818	0.247
Wetted Width Δ	0.14	0.875
Wetted Perimeter Δ	0.442	0.153
Width:Depth Δ	0.431	0.55
Entrenchment Ratio Δ	0.261	0.351
Hydraulic Radius Δ	0.845	-0.125

Table 8. Loading factors of macroinvertebrate relative abundancesshowing the influence of the variables on the omponents.

	PCRA1	*PCRA2*	*PCRA3*	*PCRA4*
% Total Variance Explained	27.3	15.7	12.3	11.2
EPT relative abundance	0.782	0.297	0.234	0.156
Trichoptera relative abundance	0.634	-0.176	0.062	0.455
Chironomidae relative abundance	-0.579	0.444	0.195	0.326
Plecoptera relative abundance	0.528	-0.028	0.495	-0.394
Shredder relative abundance	0.520	-0.250	0.543	0.057
Shannons Diversity	0.499	0.557	0.127	-0.139
Ephemeroptera relative abundance	0.500	0.544	-0.334	-0.172
Simpsons Diversity	-0.496	0.501	0.367	0.343
Scraper relative abundance	0.010	0.361	-0.288	-0.550
Collector relative abundance	0.415	0.402	-0.407	0.463
Predator relative abundance	-0.435	0.440	0.454	-0.219

Chapter 3 Figures

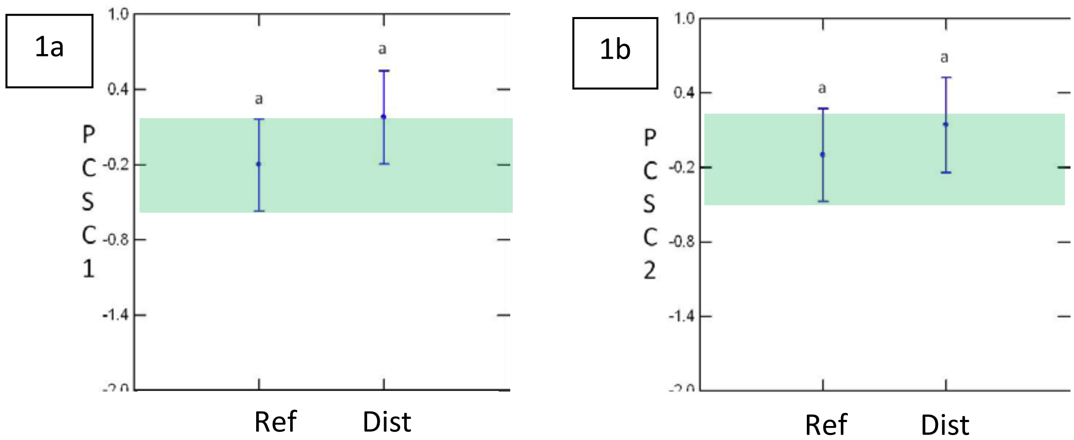

Figure 1) The results of GLM analysis of the principal components comprised of co-varying metrics of geomorphic yearly change. No principal component made up of any combination of geomorphic variables was able to differentiate the Ref and Dist regimes. This illustrates the weakness of relying soley on structural variables to evaluate the needs of streams. Reference (Ref) and Disturbed (Dist) sites (x-axis). Figure 1a shows scores for PCSC1 and 1b shows scores for PCSC2. Shaded denotes range of scores in reference sites.

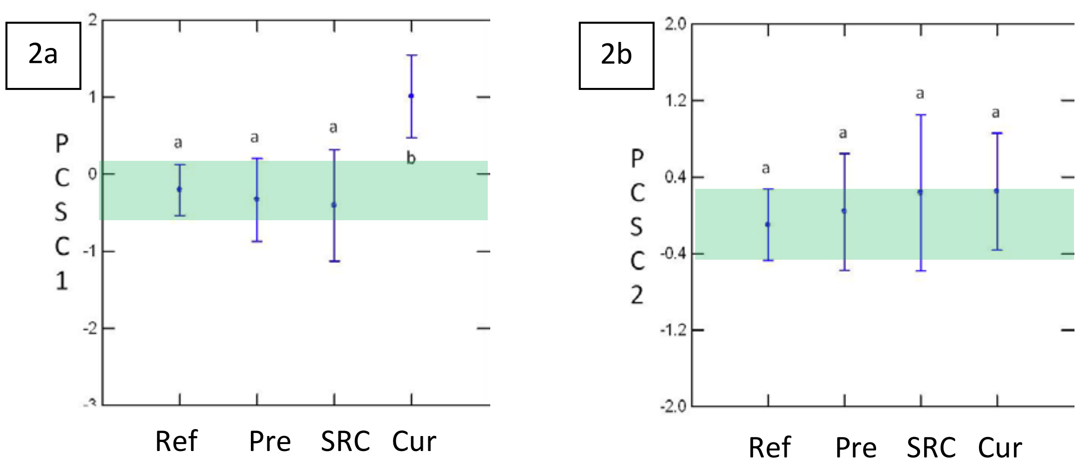

Figure 2) These graphs show the GLM results from principal components of geomorphic yearly change variables between Reference (Ref) and Disturbance Temporal Regimes: Previous to SRS Construction (Pre), During construction (SRC) and Currently disturbed (Cur) (x-axis). The Cur regime was differentiated from others showing that streams with on-going disturbances are suffering higher rates of yearly change in the strong loaders of PCSC1 (yearly changes in bankfull area and hydraulic radius) than other regimes. Shaded denotes range of scores in reference sites. Figure 2a is PCSC1 and PCSC2 results are shown in figure 2b.

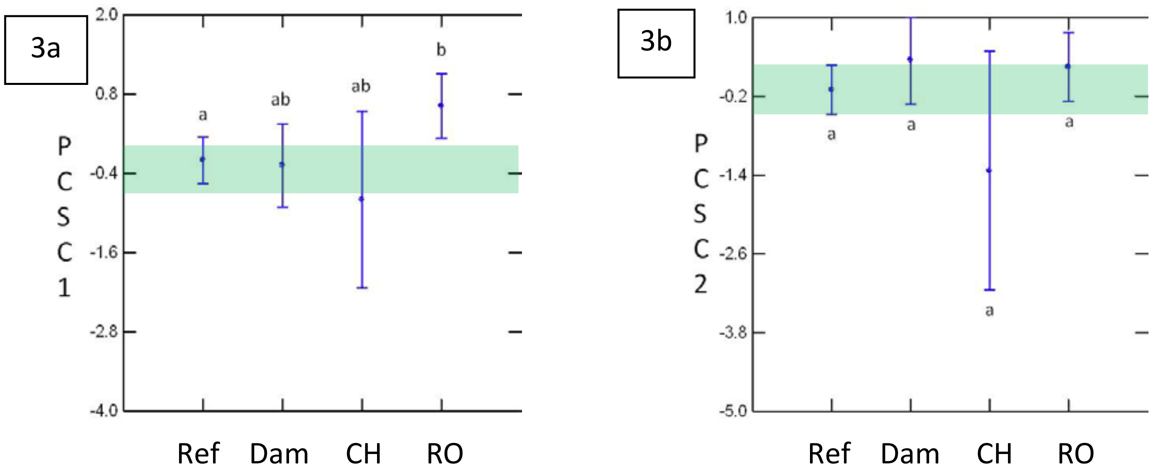

Figure 3) Principal components of structural yearly change variables GLM results between Reference (Ref) and Disturbance Categories: Sites that had dams (Dam), channelized sites (CH) and runoff disturbed (RO) (x-axis) are illustrated here.3a shows the differences between the Cur and the other regimes regarding yearly change in structural variables (especially bankfull area and hydraulic radius) with RO sites exhibiting more yearly variation. Shaded denotes range of scores in reference sites.

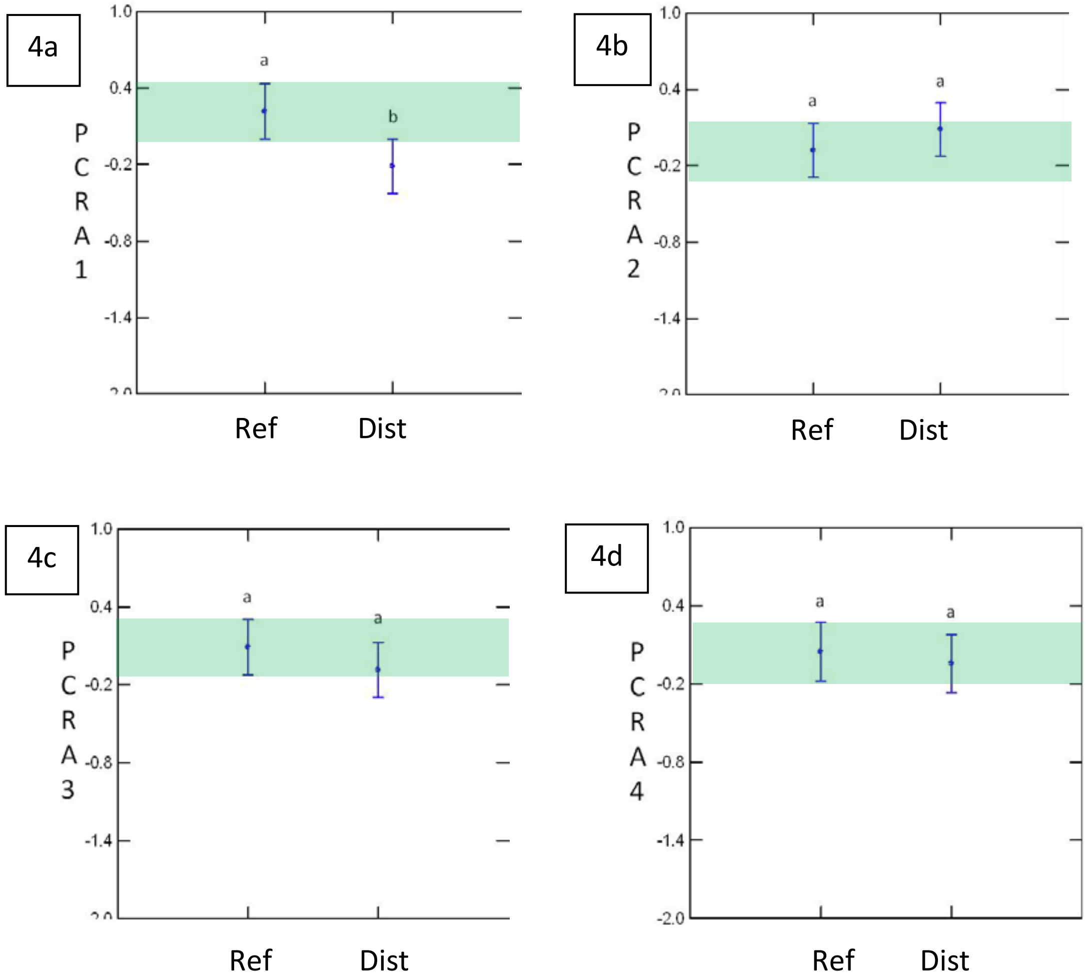

Figure 4) Principal components of macroinvertebrate relative abundance variables from kick net samples in Reference (Ref) and Disturbed (Dist) sites (x-axis) GLM results are illustrated here. Figure 4a shows the ability of relative abundances of EPT, Trichoptera, Plecoptera, shredders and negative loading of the abundance of Chironomids (the strong loaders of PCRA1) to distinguish Ref from Dist regimes. These variables co-varied and scored higher in the Ref regime. The groups that scored higher have been shown to be important in headwater streams and the negative loading relative abundance of Chironomids in Ref means they were more abundant in the Dist regime. Figure 4a corresponds to PCRA1 and 4b to PCRA2 and so on. Shaded denotes range of scores in reference sites.

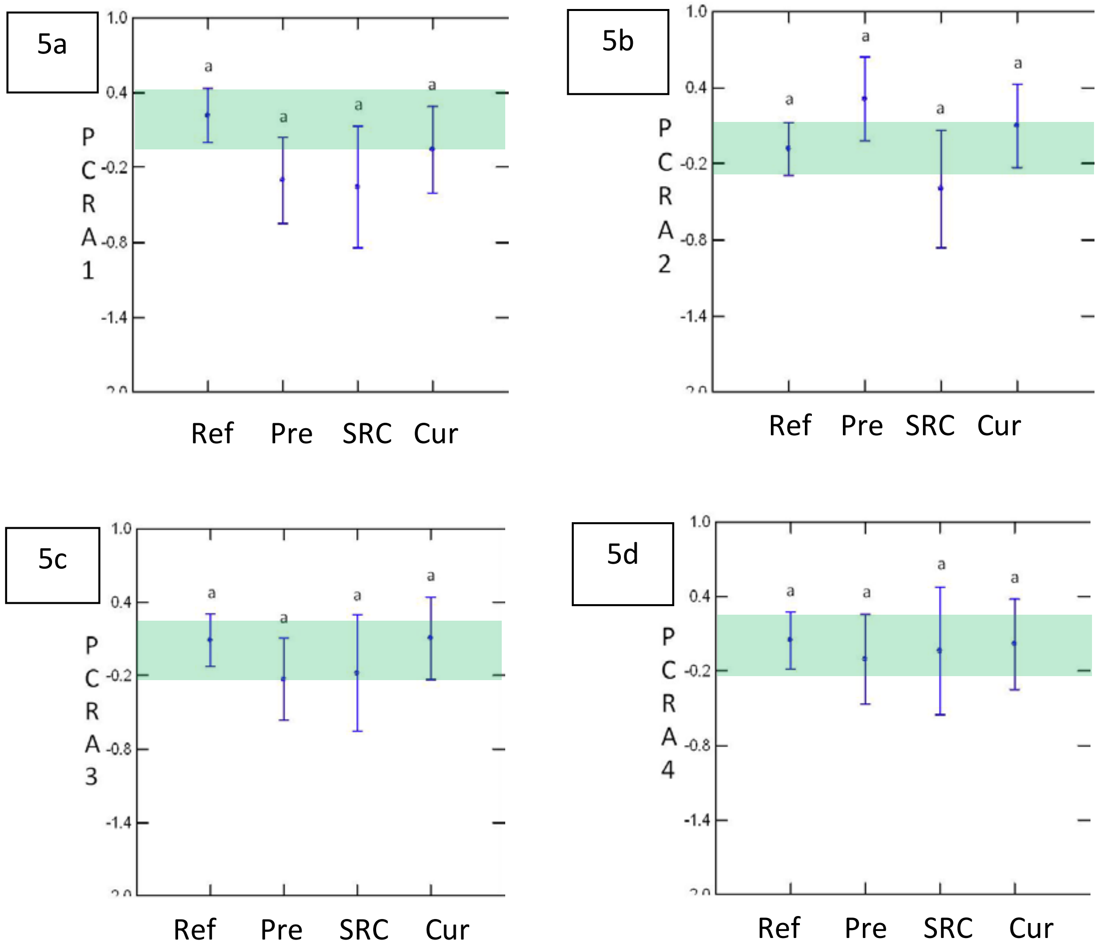

Figure 5) Illustrates the inability of the principal components comprised of macroinvertebrate relative abundance variables to separate any of the temporal regimes from either Ref or each other. Previous to SRS Construction (Pre), During construction (SRC) and Currently disturbed (Cur) (x-axis).This not only shows similarity across disturbances but also how ineffectual it can be to rely on a single type of metric to determine the condition of streams. Shaded denotes range of scores in reference sites.

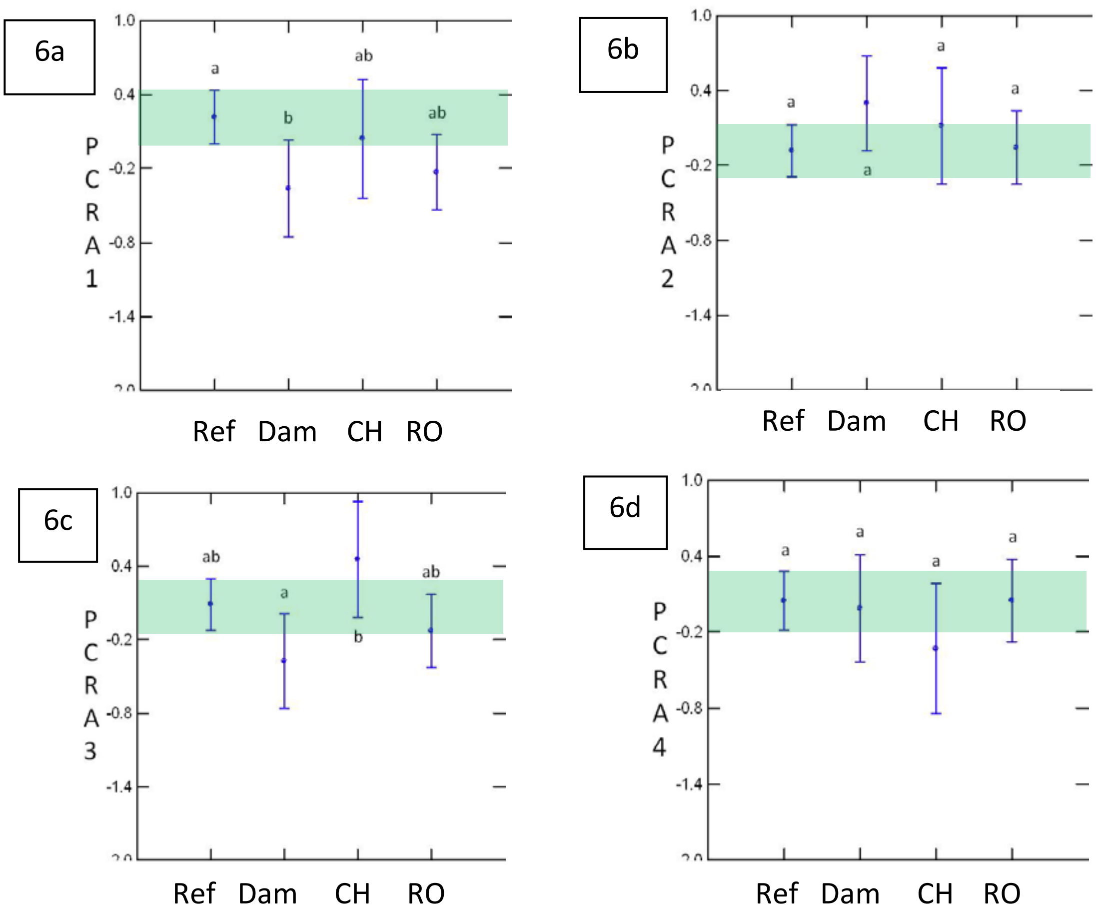

Figure 6) Macroinvertebrate relative abundance variables from kick net samples were only able to distinguish between the Dam and Ref in 6a and Dam and CH in 6c as shown above. In 6a PCRA1 the most important variables were the relative abundances of EPT, Trichoptera, Plecoptera, shredders and negative relative abundance of Chironomids. In 6c (PCRA3) CH relative abundance of shredders were higher in the CH regime. While some differences were evident, these macroinvertebrate variables were unable to consistently distinguish categorical regimes from the Ref. Reference (Ref) and Disturbance Categories: Sites that had dams breeched (Dam), channelized sites (CH) and runoff disturbed (RO) (x-axis).Figure 6a corresponds to PCRA1 and 6b to PCRA2 and so on. Shaded denotes range of scores in reference sites.

Chapter 4. Overall Conclusions

4.0. Conclusions

This project addressed the paucity of baseline and pre-project data in stream restorations. Restoration sites are often times chosen by obvious presence of disturbance such as the appearance of incision, dams or obvious run off. Other common methods for choosing sites for restoration include using systems like the Environmental Protection Agency's rapid bioassessment index (RBI) or the Natural Resources Conservation Service's stream visual assessment protocol (SVAP) to determine the condition of the potential restorations sites. However, these protocols rely on heavily on structural variables and visual appearance. In contrast, this project assessed a myriad of parameters, structural and functional, in a more comprehensive manner to better prioritize the stream's need of restoration.

Variables of many types including: habitat, geomorphic, water quality, storm water, hydrology, biotic, flooding frequency and leaf decay rate were examined over the course of this study. In the process of searching for one metric to define the condition of potential restoration sites, it became apparent that each variable examined was highly interconnected with one another. For example, flooding frequency controls habitat complexity, geomorphic variables and flow pattern which have an influence on water quality and biotic communities. This interconnectedness made it difficult to discern a single metric to define stream restoration priority based on variation from the Ref regime. However, the composition of the macroinvertebrate communities that colonized the leaf packs proved to be a good metric for distinguishing Ref and Dist regimes and showing differences within the temporal and categorical regimes. This was interesting given the similarities in leaf decay rate across the regimes, which is often used as a measure of stream function. Using a single aspect of the macroinvertebrate community, (i.e. EPT density) may not be sufficient to determine the condition of a stream and determine its restoration priority. Yet, examining a multitude of macroinvertebrate community aspects allowed similarities and differences of Ref and Dist regimes along with the temporal and categorical regimes. Macroinvertebrate communities are known to be sensitive to a variety of factors. The combination of their responses to other stream factors (geomorphic, hydrologic, water quality) and the effect that they can have on stream function made these communities good indicators of stream condition. The responses of the macroinvertebrates to these factors may have contributed to the inaccuracy of the flow charts from chapter 1. Most studies focus on specific aspects of macroinvertebrate communities such as diversity, percent EPT, shredders or chironomidae. In this study, the community aspects of macroinvertebrates were combined and examined together. This allowed for more efficient differentiation between several types of disturbance. It is possible that by replacing macroinvertebrate diversity with the overall macroinvertebrate community response would improve the flow charts designed in chapter 1. This could be applied to other studies using macroinvertebrates as indicators of similarity to reference reaches. While examining leaf pack colonizing macroinvertebrate does require time and money, those communities comprised the variables most able to distinguish the Ref regime from other disturbance types and therefore can be recommended as a good screening tool for evaluating stream condition.

Under the prevailing paradigm of assessment of stream condition, all 10 of the streams in the Dist regime would be considered high priority sites for stream restoration. The data gathered for this work suggests that, based on comparison to the Ref regime, only three streams are actually high priority for restoration. The historically disturbed sites, those in the Pre and SRC exhibit many similarities to the Ref regime and therefore can be described as having returned to, or found a new, equilibrium. Their recovery could have been aided by existence of the mostly intact riparian systems found along the stream sides. Using data from Suddeth et al (2007), the average cost of a single restoration in the Southeast is over US 500,000 dollars. By allowing the seven streams that this study identified as lower priority for restoration to continue their natural recovery would not only theoretically save US 3,500,000 dollars but also eliminates the risk of reversing ecological gains made during their natural recovery. This monetary savings would more than offset the increased initial investment that would accompany more thorough pre-project data collection and analysis. The pre-project assessment also better informed us as to the points of variation between high priority restoration targets and their reference sites. This could lead to better-defined goals and more appropriate choices of methodology, thereby increasing the efficacy of resource use and in the end increase the percentages of success.

4.1 Applications to Other Systems

While this research took place in the Upper Coastal Plain, I believe the lessons provided herein have the potential to be valuable in a myriad of settings where a viable reference system exists. It is true that a majority of streams have been negatively impacted by human activity (Vorosmarty et al 2010 & 2000) however, sometimes a reference stream can refer to the least disturbed stream (Kosinski et al 2014). In many cases, the improvement from a severely disturbed stream to one more similar to the condition of stream with minor disturbance could be enough to label the restoration effort a success and return many ecosystem functions to the highly disturbed areas. Given the possibility of questionable conditions of reference streams in some areas, it is even more important that decisions made regarding potential restoration sites are made with the best possible baseline data. This will increase the probability that the maximum improvement can be achieved using the most efficient methods. Hopefully, the potential monetary savings along with the increased chance of success will convince more managers to implement more intense pre-project study.

4.2 Future Work

Near the end of this study soil cores were taken from the flood plains of many of the sites using a vibra- core technique. The analysis of vibra-soil cores from the flood plains could add insight to historical changes to the sediment composition and will be performed as part of a separate MS thesis. Also, a hester-dendy (a macroinvertebrate collection device that is analogous to coarse wood) comparison to changes in coarse woody debris volume ran concurrently with this study. The data gathered from this work could add another dimension to the similarities and differences found in the disturbance regimes. Several other avenues are available for future work including more stream to stream comparison rather than comparing regimes and a canonical correspondence analysis to look at all types of data (macroinvertebrate, geomorphic and water quality) together and more extensive storm flow monitoring using the level loggers.

Appendix i *General data for all study sites*

Table i. Canopy cover of all study sites.

Canopy Cover	
REACH	*AVG % OPEN*
MBHW	16.2
MB6	11.7
MB75B	15.0
MB9	17.4
MQHW	7.5
MQ8	8.3
MC5A	9.5
MC5B	8.9
MC6	11.1
PB 3	9.3
PB 4	9.3
TC 2A	12.7
TC 2C	14.5
TC 3	8.7
TC 5	11.2
U6	14.8
U8	11.4
U10	10.9
U36A	8.3
U36C	13.7

Table ii. Percentages of presence of macrophytes, undercut banks, rootmats and overhanging shrubs in all study sites are listed. These indicate habitat heterogeneity.

Percentage of Habitat Sites within Reach with the Presence of:				
Reach	*Macrophytes*	*Undercut banks*	*Root mats*	*Overhanging Shrubs*
MBHW	6.3	18.8	0	25
MB6	0	0	6.3	0
MB75B	6.3	6.3	6.3	12.5
MB9	0	12.6	0	18.8
MQHW	0	68.8	0	0
MQ8	0	50	0	6.3
PB3	0	31.3	0	0
PB4	0	6.3	12.5	0
TC2A	95.5	0	0	0
TC2C	22.7	0	0	18.2
TC3	0	31.3	6.3	100
TC5	93.8	31.3	6.3	31.3
U6	0	0	0	0
U8	0	37.5	0	6.3
U10	0	31.3	0	0
U36A	0	0	0	0
U36C	0	6.3	0	100
MC5A	0	12.5	18.8	6.3
MC5B	0	0	0	0
MC6	0	31.3	12.5	50

Table iii. The percentages of habitat sites within each study site with presence of riffles, runs, pools and coarse woody debris are listed which are more indicators of habitat heterogeneity.

Percentage of Habitat Sites within Reach with the Presence of:				
Reach	*Riffles*	*Runs*	*Pools*	*Coarse Woody Debris*
MBHW	0	43.8	56.3	31.3
MB 6	0	43.8	56.3	43.8
MB75B	0	93.4	6.3	25
MB 9	0	87.8	12.5	25
MQHW	0	81.3	18.8	50
MQ 8	0	43.8	56.3	12.5
MC 5A	0	75	25	18.8
MC 5B	0	87.5	12.5	31.3
MC 6	6.3	56.3	37.5	43.8
PB 3	0	56.3	37.5	31.3
PB 4	0	31.3	68.8	50
TC 2A	0	90.9	9.1	13.6
TC 2C	0	65.5	4.5	18.2
TC 3	0	75	25	37.5
TC 5	0	81.3	18.8	50
U 6	8.3	75	12.5	25
U 8	0	81.3	18.8	25
U 10	0	68.8	31.3	18.8
U36A	0	81.3	18.8	37.5
U36C	0	93.8	6.3	6.3

Table iv. Presence and absence of sediment bars and types by study site which contribute to habitat heterogeneity but also can indicate unstable hydrology.

Percentage of Habitat Sites within Reaches with Presence Sediment Bars by Type					
Reach	*Sand*	*Fine Gravel*	*Gravel-Pebble*	*Cobble*	*Total*
MBHW	0	0	0	6.3	6.3
MB 6	0	0	0	0	0
MB75B	12.5	0	0	0	12.5
MB 9	6.3	0	6.3	0	6.3
MQHW	6.3	6.3	0	0	18.8
MQ 8	6.3	0	25	0	6.3
MC 5A	81.3	0	12.5	0	100
MC 5B	6.3	0	0	0	18.8
MC 6	31.3	0	0	0	31.3
PB 3	0	0	0	0	0
PB 4	25	6.3	0	0	31.3
TC 2A	13.6	0	0	0	163.6
TC 2C	18.2	0	0	0	18.2
TC 3	0	0	0	0	0
TC 5	0	0	0	0	0
U 6	50	0	0	0	50
U 8	25	0	6.3	0	31.3
U 10	0	0	0	0	0
U36A	0	0	0	0	0
U36C	0	0	0	0	0

Table v. Streambed penetration across study sites showing connectivity of ground water to the streams.

Average Streambed Penetration by Reach					
Reach	*2013*	*2012*	*2011*	*2010*	*Overall*
MBHW	2	1.7	2.2	1.2	1.8
MB 6	2.5	2.2	3.5	2.9	2.8
MB75B	3.6	4.2	3.7	3.5	3.7
MB 9	2.8	2.9	3.1	2.5	2.8
MQHW	7.3	6.9	9.3	7.9	7.9
MQ 8	3.7	3.2	4.3	4	3.8
MC 5A	3.3	3.4	3.7	4.7	3.8
MC 5B	4.7	3.5	3.8	NA	4
MC 6	5.4	5.9	6.5	3.6	5.3
PB 3	3.5	6	2.8	3.6	4
PB 4	7.5	6.8	5.3	7	6.6
TC 2A	2.9	3.1	4.3	5.4	3.9
TC 2C	4.9	5	5.4	5.7	5.3
TC 3	4.2	4.2	5.5	NA	4.6
TC 5	3.9	4.4	4	4.5	4.2
U 6	7.9	1.7	2.3	2.4	3.6
U 8	5.6	3.1	7.8	4.7	5.3
U 10	4.2	4.7	4.2	3.8	4.2
U36A	4.8	3.5	4	3.3	3.9
U36C	3.7	3.2	3.7	NA	3.5

Table vi. Average percentage of silt coverage of each 10m section of each study reach.

Average Percentage of study site's Silt Coverage	
Reach	*% Silt Coverage*
MBHW	43.4
MB 6	56.1
MB75B	41.3
MB 9	66.9
MQHW	14.1
MQ 8	41.6
MC 5A	32.8
MC 5B	44.7
MC 6	46.8
PB 3	53.4
PB 4	68.1
TC 2A	19.5
TC 2C	20
TC 3	56.6
TC 5	22.5
U 6	2.9
U 8	50.9
U 10	17.3
U36A	47.5
U36C	24.1

Table vii. Total macroinvertebrates collected from leaf packs. Bold indicates order abundance and underline indicates family abundance.

List of Macroinvertebrates Collected from Leaf Packs			
ORDER	*FAMILY*	*GENUS*	*#*
Diptera			4334
Diptera	<u>Chironomidae</u>		4027
Trichoptera			895
Plecoptera			741
Ephemeroptera			563
Trichoptera	<u>Lepidostomatidae</u>		387
Trichoptera	Lepidostomatidae	*Lepidostoma*	386
Trichoptera	<u>Hydropsychidae</u>		363
Plecoptera	<u>Perlidae</u>		312
Plecoptera	<u>Leuctridae</u>		307
Plecoptera	Perlidae	*Claasenia*	286
Ephemeroptera	<u>Heptageniidae</u>		275
Plecoptera	Leuctridae	*Leuctra*	236
Trichoptera	Hydropsychidae	*Hydropsyche*	229
Ephemeroptera	Heptageniidae	*Maccaffertium*	204
Coleoptera			153
Ephemeroptera	<u>Leptophlebiidae</u>		128
Diptera	<u>Ceratopogonidae</u>		123
Coleoptera	<u>Elmidae</u>		121
Coleoptera	Elmidae	*Stenelmis*	112
Trichoptera	Hydropsychidae	*Diplectrona*	106
Diptera	<u>Tipulidae</u>		99
Ephemeroptera	<u>Ephemerellidae</u>		94
Annelid			80
Annelid	<u>Oligochaeta</u>		80
Ephemeroptera	Ephemerellidae	*Serratella*	79
Plecoptera	Leuctridae	*Paraleuctra*	70
Diptera	Tipulidae	*Tipula*	69
Diptera	Ceratopogonidae	*Bezzia*	65
Diptera	Ceratopogonidae	*Forcipomyia*	58
Odonota			57
Plecoptera	<u>Chloroperlidae</u>		56
Ephemeroptera	Leptophlebiidae	*Paraleptaphlebia*	50
Megaloptera			48
Megaloptera	<u>Corydalidae</u>		48
Ephemeroptera	Heptageniidae	*Stenonema*	47
Diptera	<u>Simuliidae</u>		45
Megaloptera	Corydalidae	*Nigronia*	44
Plecoptera	<u>Perlodidae</u>		42

List of Macroinvertebrates Collected from Leaf Packs part2			
ORDER	*FAMILY*	*GENUS*	*#*
Plecoptera	Chloroperlidae	*Haploperla*	42
Diptera	Dixidae		38
Diptera	Dixidae	*Dixella*	38
Ephemeroptera	Baetidae		36
Bivalvia			33
Bivalvia	Sphaeriidae		33
Plecoptera	Perlodidae	*Isoperla*	32
Diptera	Simuliidae	*Simulium*	29
Odonota	Gomphidae		29
Trichoptera	Limnephilidae		26
Trichoptera	Limnephilidae	*Pycnopsyche*	25
Ephemeroptera	Baetidae	*Acerpenna*	23
Odonota	Gomphidae	*Erpetogomphus*	23
Trichoptera	Calamoceratidae		21
Trichoptera	Calmoceratidae	*Anisocentropus*	21
Coleoptera	Ptlodactylidae		20
Coleoptera	Ptlodactylidae	*Anchytarsus*	20
Plecoptera	Nemouridae		19
Ephemeroptera	Caenidae		19
Ephemeroptera	Caenidae	*Caenis*	19
Plecoptera	Nemouridae	*Amphinemura*	19
Ephemeroptera	Leptophlebiidae	*Habrophlebia*	18
Diptera	Tipulidae	*Hexatoma*	18
Gastropoda			16
Trichoptera	Dipseudopsidae		15
Trichoptera	Dipseudopsidae	*Phylocentropus*	15
Ephemeroptera	Leptophlebiidae	*Leptophlebia*	15
Plecoptera	Perlidae	*Hansonoperla*	14
Trichoptera	Odontoceridae		13
Diptera	Simulidae	*Ectemnia*	13
Trichoptera	Brachycentridae		12
Trichoptera	Odontoceridae	*Psilotreta*	12
Diptera	Tipulidae	*Polymera*	12
Gastropod	Viviparidae		12
Plecoptera	Chloroperlidae	*Plumiperla*	12
Trichoptera	Brachycentridae	*Brachycentrus*	11
Trichoptera	Molannidae		11
Trichoptera	Molannidae	*Molanna*	11
Ephemeroptera	Ephemerellidae	*Timpagona*	11
Odonota	Calopterygidae		11
Odonota	Coenigrionidae		11

List of Macroinvertebrates Collected from Leaf Packs part3			
ORDER	*FAMILY*	*GENUS*	*#*
Odonota	Coenagrionidae	*Amphiagrion*	11
Odonota	Calopterygidae	*Calopteryx*	11
Trichoptera	Sericostomatidae		10
Trichoptera	Sericostomatidae	*Fattiga*	10
Trichoptera	Beraeidae		9
Trichoptera	Beraeidae	*Beraea*	9
Plecoptera	Perlodidae	*Clioperla*	9
Hemiptera			9
Hemiptera	Saldidae		9
Trichoptera	Leptoceridae		7
Trichoptera	Leptoceridae	*Nectopsyche*	7
Coleoptera	Gyrinidae		7
Coleoptera	Gyrinidae	*Dineutus*	7
Decapod			7
Decapod	Atyidae		7
Plecoptera	Peltoperlidae		6
Plecoptera	Peltoperlidae	*Tallaperla*	6
Ephemeroptera	Neoephemeridae		5
Ephemeroptera	Lephtophlebiidae	*Habrophlebiodes*	5
Ephemeroptera	Neoephemeridae	*Neoephemera*	5
Trichoptera	Polycentropodiae		5
Ephemeroptera	Baetidae	*Fallceon*	4
Ephemeroptera	Ephemeridae		4
Plecoptera	Perlidae	*Eccoptura*	4
Ephemeroptera	Ephemeridae	*Hexagina*	4
Odonota	Gomphidae	*Arigomphus*	4
Odonota	Aeshnidae		4
Odonota	Aehnidae	*Boyeria*	4
Trichoptera	Philopotamidae		3
Trichoptera	Philopotamidae	*Chimarra*	3
Gastropod	Physidae		3
Trichoptera	Polycentropodidae	*Polycentropus*	3
Ephemeroptera	Baetiscidae		2
Ephemeroptera	Baetiscidae	*Baetisca*	2
Ephemeroptera	Heptageniidae	*Stenocron*	2
Coleoptera	Dryopidae		2
Coleoptera	Hydrophilidae		2
Coleoptera	Hydrophilidae	*Sperchopsis*	2
Coleoptera	Elmidae	*Macronychus*	2
Coleoptera	Dryopidae	*Helichus*	2
Coleoptera	ELMIDAE	*Dubriaphia*	2

List of Macroinvertebrates Collected from Leaf Packs part4			
ORDER	*FAMILY*	*GENUS*	*#*
Coleoptera	Elmidae	*Ancyronyx*	2
Trichoptera	Hydropsychidae	*Potamyia*	1
Trichoptera	Lepidostomatidae	*Theliopsyche*	1
Plecoptera	Perlodidae	*Cultus*	1
Ephemeroptera	Baetidae	*Baetis*	1
Coleoptera	<u>Dytiscidae</u>		1
Coleoptera	Dytiscidae	*Hydrocolus*	1
Odonota	<u>Cordulegastridae</u>		1
Odonota	<u>Corduliidae</u>		1
Odonota	Cordulegastiidae	*Cordulegaster*	1
Odonota	Corduliidae	*Epitheca*	1
Megaloptera	Corydalidae	*Chauliodes*	1
Diptera	<u>Culicidae</u>		1
Diptera	<u>Ptychopteridae</u>		1
Diptera	Culicidae	*Culex*	1
Diptera	PTYCHOPTERIDAE	*Ptychoptera*	1
Gastropod	<u>Planorbidae</u>		1

Table viii. Total macroinvertebrates collected from leaf packs. Bold indicates order abundance and underline indicates family abundance.

Macroinvertebrates collected from kick nets			
ORDER	*FAMILY*	*GENUS*	*#*
Diptera			5768
Diptera	<u>Chironomidae</u>		5249
Coleoptera			1380
Trichoptera			1043
Coleoptera	<u>Ptilodactylidae</u>		680
Coleoptera	Ptilodactylidae	*Anchytarsus*	680
Coleoptera	<u>Elmidae</u>		619
Plecoptera			583
Ephemeroptera			553
Coleoptera	Elmidae	*Stenelmis*	547
Odonota			399
Trichoptera	<u>Hydropsychidae</u>		395
Trichoptera	<u>Philopotamidae</u>		369
Trichoptera	Philopotamidae	*Chimarra*	369
Plecoptera	<u>Leuctridae</u>		365
Plecoptera	Leuctridae	*Leuctra*	362
Trichoptera	Hydropsychidae	*Diplectrona*	305
Odonota	<u>Calopterygidae</u>		195
Odonota	Calopterygidae	*Calopteryx*	195
Ephemeroptera	<u>Leptophlebiidae</u>		189
Diptera	<u>Simuliidae</u>		151
Diptera	Simuliidae	*Simuliium*	151
Odonota	<u>Gomphidae</u>		151
Diptera	<u>Dixidae</u>		139
Diptera	Dixidae	*Dixella*	139
Ephemeroptera	<u>Baetidae</u>		129
Ephemeroptera	Baetidae	*Procleon*	129
Gastropod			92
Gastropod	<u>Vivaparidae</u>		92
Diptera	<u>Tipulidae</u>		88
Ephemeroptera	<u>Heptageniidae</u>		88
Diptera	Tipulidae	*Tipula*	87
Plecoptera	<u>Perlodidae</u>		87
Trichoptera	<u>Polycentropodidae</u>		87
Trichoptera	Hydropsychidae	*Hydropsyche*	84
Diptera	<u>Ceratopogonidae</u>		82
Ephemeroptera	Leptophlebiidae	*Harbophlebia*	80
Ephemeroptera	Heptageniidae	*Maccaffertium*	79

Macroinvertebrates collected from kick nets part 2			
ORDER	*FAMILY*	*GENUS*	*#*
Decopoda			78
Plecoptera	Perlidae		74
Ephemeroptera	Ephemerellidae		73
Trichoptera	Polycentropodidae	*Polycentropus*	72
Diptera	Ceratopogonidae	*Probezzia*	68
Coleoptera	Elmidae	*Ancyronyx*	66
Trichoptera	Limnephilidae		66
Annelida			59
Annelida	Oliogochaeta		59
Ephemeroptera	Leptophlebiidae	*Leptophlebia*	59
Megaloptera			59
Megaloptera	Corydalidae		59
Megaloptera	Corydalidae	*Nigronia*	58
Plecoptera	Perlidae	*Claasenia*	56
Diptera	Limoniidae		54
Odonota	Gomphidae	*Progomphus*	54
Decopoda	Aytidae		52
Diptera	Limoniidae	*Hexatoma*	51
Plecoptera	Perlodidae	*Isoperla*	51
Odonota	Gomphidae	*Arigomphus*	46
Odonota	Gomphidae	*Erpetogomphus*	45
Trichoptera	Limnephilidae	*Pycnopspyche*	44
Ephemeroptera	Ephemerellidae	*Ephemerella*	41
Hemiptera			37
Trichoptera	Calamoceratidae		34
Trichoptera	Calamoceratidae	*Anisocentropus*	34
Trichoptera	Lepidostomatidae	*Lepidostoma*	34
Trichoptera	Lepidostomatidae		34
Hemiptera	Vellidae		31
Ephemeroptera	Ephemerellidae	*Euryophella*	30
Odonota	Cordulegastridae		30
Odonota	Cordulagstidae	*Cordulegaster*	30
Ephemeroptera	Ephemeridae		29
Ephemeroptera	Ephemeridae	*Hexagenia*	29
Ephemeroptera	Caenidae		28
Decopoda	Cambaridae		26
Ephemeroptera	Leptophlebiidae	*Unknown*	24
Trichoptera	Leptoceridae		23
Plecoptera	Chloroperlidae		22
Hemiptera	Vellidae	*Microvelia*	21
Trichoptera	Limnephilidae	*Unknown*	21

Macroinvertebrates Collected from Kick nets part3			
ORDER	*FAMILY*	*GENUS*	*#*
Plecoptera	Peltoperlidea		20
Plecoptera	Peltoperlidae	*Yoraperla*	20
Trichoptera	Leptoceridae	*Nectopsyche*	20
Trichoptera	Dipseudopsidae		19
Trichoptera	Dipseudopsidae	*Phylocentropodus*	19
Plecoptera	Chloroperlidae	*Suwallia*	18
Bivalvia			16
Bivalvia	Sphaeridae		16
Ephemeroptera	Leptohyphidae		16
Plecoptera	Perlodidae	*Cultus*	16
Ephemeroptera	Leptophlebiidae	*Habrophlbiodea*	15
Plecoptera	Nemouridae		15
Plecoptera	Nemouridae	*Amphinemura*	15
Ephemeroptera	Caenidae	*Americaenis*	14
Ephemeroptera	Caenidae	*Caenis*	13
Plecoptera	Perlidae	*Beloneuria*	13
Ephemeroptera	Leptohyphidae	*Tricorythodes*	11
Ephemeroptera	Lepthophlebiidae	*Paraleptophlebia*	11
Trichoptera	Polycentropodidae	*Unknown*	11
Plecoptera	Perlodidae	*Clioperla*	10
Plecoptera	Perlodidae	*Diura*	10
Hemiptera	Vellidae	*Rhagovelia*	9
Odonota	Aeshnidae		9
Odonota	Aeshnidea	*Boyeria*	9
Odonota	Cordullidae		9
Coleoptera	Dytiscidae		7
Diptera	Ceratopogonidae	*Culicoides*	7
Diptera	Ceratopogonidae	*Bezzia*	7
Ephemeroptera	Heptageniidae	*Unknown*	7
Trichoptera	Brachycentridae		6
Trichoptera	Brachycentridae	*Brachycentrus*	6
Trichoptera	Hydropsychidae	*Macrosternum*	6
Trichoptera	Psychomiidae		6
Trichoptera	Psychomiidae	*Lype*	6
Ephemeroptera	Leptohyphidae	*Unknown*	5
Hemiptera	Saldidae		5
Odonota	Coenagrionidae		5
Plecoptera	Perlidae	*Unknown*	5
Collembola			4
Collembola	Dicytromidae		4

Macroinvertebrates Collected from Kick nets part4			
ORDER	*FAMILY*	*GENUS*	*#*
Collembola	Dicytromidae	*Dicytroma*	4
Diptera	<u>Empididae</u>		4
Odonota	Cordulidae	*Eipitheca*	4
Odonota	Gomphidae	*Hagenius*	4
Plecoptera	Chloroperlidae	*Unknown*	4
Trichoptera	<u>Molannidae</u>		4
Trichoptera	Molannidae	*Molanna*	4
Coleoptera	Dytiscidae	*Unknown*	3
Diptera	Empididae	*Unknown*	3
Odonota	Coenagrionidae	*Unknown*	3
Plecoptera	Leuctridae	*Unknown*	3
Trichoptera	Leptoceridae	*Leptocerus*	3
Trichoptera	Polycentropodidae	*Crynellus*	3
Coleoptera	Dytiscidae	*Dytiscus*	2
Coleoptera	Elmidae	*Dubiraphia*	2
Coleoptera	Elmidae	*Unknown*	2
Coleoptera	<u>Hydrophilidae</u>		2
Diptera	Limoniidea	*Antocha*	2
Ephemeroptera	Heptageniidae	*Stenacron*	2
Odonota	Coenigrionidae	*Chromagrion*	2
Odonota	Cordulidae	*Heliocordulia*	2
Coleoptera	Dytiscidae	*Desmopachria*	1
Coleoptera	Dystiscidae	*Cybister*	1
Coleoptera	Elmidae	*Oulinius*	1
Coleoptera	Elmidae	*Macronychus*	1
Coleoptera	Hydrophilidae	*Unknown*	1
Coleoptera	Hydrophilidae	*Sperchopis*	1
Diptera	<u>Culicidae</u>		1
Diptera	Culicidae	*Anopheles*	1
Diptera	Empididae	*Hemerodromia*	1
Diptera	Limoniidae	*Pilaria*	1
Diptera	Tipulidae	*Leptotarsus*	1
Ephemeroptera	<u>Baetiscidae</u>		1
Ephemeroptera	Baetiscidae	*Baetisca*	1
Ephemeroptera	Caeniidae	*Unknown*	1
Ephemeroptera	Ephemerellidae	*Unknown*	1
Ephemeroptera	Eephemerellidae	*Seratella*	1

Macroinvertebrates collected from kick nets part 5

ORDER	*FAMILY*	*GENUS*	*#*
Hemiptera	Gerridae		1
Hemiptera	Gerridae	*Trepobates*	1
Hemiptera	Vellidae	*Unknown*	1
Megaloptera	Corydalidae	*Unknown*	1
Odonota	Corduliidae	*Unknown*	1
Odonota	Corduliidae	*Neurocordulia*	1
Odonota	Corduliidae	*Cordulia*	1
Odonota	Gomphidae	*Stylurus*	1
Odonota	Gomphidae	*Gomphus*	1
Trichoptera	Limnephilidae	*Limnephilud*	1
Trichoptera	Polycentropodus	*Neureclipsis*	1

Appendix ii Meyers Branch Data Tables

Table ix. Widths of the stream from the left edge of water to the right edge of water measured yearly at each cross section.

		Wetted Width (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MB HW	0	8.04	6.89	10.99	8.2
MB HW	30	8.53	11.15	6.82	6.56
MB HW	60	6.85	5.58	4.59	9.84
MB HW	90	5.25	4.93	4.27	4.27
MB HW	120	19.03	19.03	15.74	18.38
MB HW	150	14.77	12.8	12.47	12.8
MB 6	0	10.17	6.94	8.53	8.2
MB 6	30	8.69	8.75	7.22	7.87
MB 6	60	5.58	6.4	4.92	5.25
MB 6	90	5.25	6.39	4.26	2.62
MB 6	120	4.27	3.94	2.62	3.28
MB 6	150	6.56	5.25	3.6	3.28
MB 75B	0	7.87	3.94	2.29	4.92
MB 75B	30	3.28	5.74	4.27	6.23
MB 75B	60	3.61	2.95	2.95	2.96
MB 75B	90	3.6	3.61	3.61	4.26
MB 75B	120	5.51	5.9	5.9	5.9
MB 75B	150	5.57	5.25	4.59	4.92
MB 9	0	3.44	1.97	1.97	2.3
MB 9	30	6.89	4.92	3.93	4.92
MB 9	60	4.26	4.8	2.62	2.95
MB 9	90	7.05	6.56	2.62	3.28
MB 9	120	3.61	3.11	1.96	2.95
MB 9	150	5.58	4.26	3.93	1.64

Table x. Wetted perimeters of Meyers Branch by year.

		Wetted Perimeter (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MB HW	0	13.53	14.37	12.66	14.14
MB HW	30	16.32	15.68	16.81	14.65
MB HW	60	15.25	23.06	15.81	13.88
MB HW	90	10.44	9.5	10.1	7.06
MB HW	120	28.15	26.12	19.18	18.38
MB HW	150	21.84	19.69	18.97	12.8
MB 6	0	16.07	15.52	11.35	16.58
MB 6	30	11.36	15.35	13.5	11.06
MB 6	60	11.42	10.56	8.7	9.11
MB 6	90	10.56	10.56	8.06	7.96
MB 6	120	7.76	11.77	7.18	6.8
MB 6	150	12.4	9.94	10.5	11.05
MB 75B	0	9.06	9.24	6.82	8.95
MB 75B	30	3.59	7.7	7.54	7.53
MB 75B	60	4.56	6.43	9.69	6.11
MB 75B	90	7.31	7.06	7.53	9.2
MB 75B	120	13.96	10	8.63	9.8
MB 75B	150	8.75	9.94	9.27	11.63
MB 9	0	7.57	7.32	8.81	6.73
MB 9	30	10.24	9.84	7.08	11.12
MB 9	60	7.21	7.27	6.93	8.14
MB 9	90	9.89	9.42	11.79	11.03
MB 9	120	5.35	5.54	6.31	7.46
MB 9	150	10.19	9.92	10.43	9.38

Table xi. Maximum depths of Meyers Branch sites by year.

		Maximum Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MB HW	0	1.22	1.48	1.21	1.37
MB HW	30	2.71	2.08	2.33	2.05
MB HW	60	2.89	4.46	2.79	1.8
MB HW	90	2.27	1.81	1.95	1.22
MB HW	120	2.31	2.3	1.74	2.3
MB HW	150	2.27	1.87	1.86	1.21
MB 6	0	1.79	1.75	1.12	1.63
MB 6	30	1.44	1.47	1.64	1.39
MB 6	60	1.36	1.2	1.12	1.13
MB 6	90	1.76	1.2	1.22	1.21
MB 6	120	1.51	NA	1.35	1.44
MB 6	150	1.67	1.51	1.64	1.67
MB 75B	0	0.27	0.26	0.23	0.32
MB 75B	30	0.48	0.86	0.53	0.62
MB 75B	60	0.57	0.73	0.96	0.63
MB 75B	90	0.9	0.87	0.92	1
MB 75B	120	1.43	1.16	0.98	1.09
MB 75B	150	1.3	1.37	0.89	1.41
MB 9	0	0.71	0.74	1.11	0.56
MB 9	30	0.5	0.94	0.76	0.93
MB 9	60	0.85	0.92	0.83	0.95
MB 9	90	1	0.73	1.37	1.23
MB 9	120	0.61	0.69	0.78	0.69
MB 9	150	1	1.02	0.85	0.89

Table xii. Mean depths of Meyers Branch sites by year.

		Mean Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MB HW	0	0.86	1.09	0.87	0.97
MB HW	30	2.17	1.73	1.8	1.58
MB HW	60	1.9	3.1	1.73	1.06
MB HW	90	1.75	1.41	1.43	0.99
MB HW	120	1.58	1.72	1.38	1.67
MB HW	150	1.67	1.4	1.48	0.93
MB 6	0	1.26	1.17	0.89	1.06
MB 6	30	1.23	1.06	1.18	1.21
MB 6	60	0.81	0.78	0.85	0.84
MB 6	90	1.3	0.78	0.86	0.86
MB 6	120	0.86	NA	0.82	0.89
MB 6	150	1.17	1.51	1.12	1.1
MB 75B	0	0.13	0.19	0.1	0.16
MB 75B	30	0.13	0.51	0.35	0.3
MB 75B	60	0.48	0.46	0.44	0.3
MB 75B	90	0.58	0.63	0.57	0.41
MB 75B	120	0.69	0.79	0.72	0.79
MB 75B	150	0.94	0.99	0.55	1.03
MB 9	0	0.35	0.36	0.55	0.32
MB 9	30	0.85	0.53	0.5	0.46
MB 9	60	0.58	0.7	0.6	0.63
MB 9	90	0.73	0.57	0.94	0.89
MB 9	120	0.41	0.46	0.49	0.42
MB 9	150	0.77	0.79	0.65	0.65

Table xiii. Width to Depth ratios of Meyers Branch sites by year.

		Width to Depth Ratio			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MB HW	0	14.86	11.78	13.53	13.41
MB HW	30	6.41	7.75	8.58	8.28
MB HW	60	6.35	3.83	7.49	8.99
MB HW	90	4.58	5.38	6.02	5.93
MB HW	120	14.12	11.85	12.56	11.86
MB HW	150	9.7	10.67	11.68	15.22
MB 6	0	11.71	12.36	11.9	14.77
MB 6	30	7.49	11.42	10.57	7.84
MB 6	60	11.2	12.23	9.47	9.62
MB 6	90	5.67	12.23	8.56	8.35
MB 6	120	7.69	1.39	7.94	6.57
MB 6	150	7.79	7.54	8.65	8.96
MB 75B	0	69.23	48.47	67.8	55.13
MB 75B	30	25	14.43	21.17	22.77
MB 75B	60	8.27	12.8	20.75	15
MB 75B	90	11.64	10.41	12.6	18.76
MB 75B	120	15.74	10.11	10.53	9.63
MB 75B	150	8.05	8.44	15.6	10.48
MB 9	0	20.4	19.06	14.91	19.34
MB 9	30	16.7	16.11	13.1	20.85
MB 9	60	11.21	8.94	10.87	11.9
MB 9	90	12.21	15.51	11.96	11.54
MB 9	120	12.24	10.63	12.27	16.83
MB 9	150	11.75	11.3	15.14	13.29

Table xiv. Entrenchment Ratios of Meyers Branch sites by year.

		Entrenchment Ratio			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MB HW	0	1.43	1.42	1.47	1.44
MB HW	30	1.42	1.48	1.32	1.4
MB HW	60	1.39	1.41	1.37	1.34
MB HW	90	1.52	1.61	1.37	1.7
MB HW	120	1.09	1.2	1.23	1.29
MB HW	150	1.22	1.33	1.18	1.28
MB 6	0	1.14	1.16	1.64	1.13
MB 6	30	1.49	1.25	1.13	1.52
MB 6	60	1.51	1.44	1.71	1.71
MB 6	90	1.66	1.44	1.6	1.64
MB 6	120	1.38	1.86	1.51	1.68
MB 6	150	1.51	1.57	1.42	1.36
MB 75B	0	1.63	1.66	1.74	1.77
MB 75B	30	3.75	1.66	1.39	1.81
MB 75B	60	3.05	2.07	1.29	2.22
MB 75B	90	1.81	2.09	1.87	1.75
MB 75B	120	1.12	1.53	1.78	1.77
MB 75B	150	1.81	1.46	1.34	1.22
MB 9	0	1.49	1.55	1.32	1.8
MB 9	30	1.64	1.61	2.2	1.51
MB 9	60	1.87	1.95	1.96	1.71
MB 9	90	1.54	1.55	1.31	1.44
MB 9	120	1.65	1.79	1.69	1.39
MB 9	150	1.14	1.16	1.1	1.2

Table xv. Bankfull areas of Meyers Branch sites by year.

		Bankfull Area (ft^2)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MB HW	0	10.96	13.94	10.28	12.60
MB HW	30	30.21	22.23	27.83	20.71
MB HW	60	22.95	36.82	22.41	15.15
MB HW	90	14.06	10.68	12.29	5.8
MB HW	120	35.28	35.14	23.94	33.07
MB HW	150	27.07	20.9	25.6	13.16
MB 6	0	18.6	16.87	9.46	16.63
MB 6	30	11.31	12.84	14.77	11.51
MB 6	60	7.38	7.41	6.84	6.79
MB 6	90	9.57	7.41	6.31	6.18
MB 6	120	5.65	17.24	5.33	5.21
MB 6	150	10.67	10.11	10.85	10.89
MB 75B	0	1.15	1.72	0.66	1.45
MB 75B	30	0.41	3.78	2.61	2.05
MB 75B	60	1.92	2.7	4.06	1.34
MB 75B	90	3.89	4.16	4.09	3.16
MB 75B	120	7.49	6.33	5.44	5.98
MB 75B	150	7.14	8.24	4.71	11.15
MB 9	0	2.53	2.45	4.49	1.96
MB 9	30	4.13	4.5	3.3	4.44
MB 9	60	3.78	4.36	3.93	4.74
MB 9	90	6.47	5.01	10.62	9.13
MB 9	120	2.06	2.23	2.95	2.94
MB 9	150	6.96	7.02	6.44	5.61

Table xvi. Hydraulic Radii of Meyers Branch sites by year.

		Hydraulic Radius			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MB HW	0	0.81	0.89	0.81	0.89
MB HW	30	1.85	1.48	1.66	1.41
MB HW	60	1.5	1.6	1.48	1.09
MB HW	90	1.35	1.12	1.22	0.82
MB HW	120	1.25	1.35	1.25	1.45
MB HW	150	1.24	1.06	1.35	0.78
MB 6	0	1.16	1.09	0.83	1
MB 6	30	1	0.84	1.09	1.04
MB 6	60	0.65	0.7	0.79	0.75
MB 6	90	0.91	0.7	0.78	0.78
MB 6	120	0.73	0.61	0.74	0.77
MB 6	150	0.86	1.02	1.03	0.99
MB 75B	0	0.13	0.19	0.1	0.16
MB 75B	30	0.11	0.49	0.35	0.27
MB 75B	60	0.42	0.42	0.42	0.22
MB 75B	90	0.53	0.59	0.54	0.34
MB 75B	120	0.54	0.63	0.63	0.61
MB 75B	150	0.82	0.83	0.51	0.96
MB 9	0	0.33	0.33	0.51	0.29
MB 9	30	0.4	0.46	0.47	0.4
MB 9	60	0.52	0.6	0.57	0.58
MB 9	90	0.65	0.53	0.9	0.83
MB 9	120	0.39	0.4	0.47	0.39
MB 9	150	0.68	0.71	0.62	0.6

Table xvii. Sediment sizes found in Meyers Branch sites using a standard sieve set. DB 84 = size at the 84th percentile and DB 50 = size at the 50th percentile.

Sediment Sizes (mm)			
Stream	*X Section (m)*	*DB 84*	*DB50*
MB HW	0	60.06	17.69
MB HW	30	35.6	3.71
MB HW	60	33.99	42.5
MB HW	90	54.36	19.35
MB HW	120	54.16	6.38
MB HW	150	65.84	2.27
MB 6	0	0	10.5
MB 6	30	0	7.5
MB 6	60	4.81	20.5
MB 6	90	3.92	13
MB 6	120	0	15
MB 6	150	61.03	20.18
MB 75B	0	0	1
MB 75B	30	0	1
MB 75B	60	0	1
MB 75B	90	0	1
MB 75B	120	0	1
MB 75B	150	0	1
MB 9	0	0	1
MB 9	30	0	1
MB 9	60	0	1
MB 9	90	0	6.5
MB 9	120	0	12.5
MB 9	150	0	9

Table xviii. Canopy cover of Meyers Branch sites showing the percentage of open canopy.

Canopy Cover		
Reach	*X Section*	*% Open*
MBHW	0	20.93
MBHW	30	17.16
MBHW	60	13.78
MBHW	90	12.22
MBHW	120	16.12
MBHW	150	17.03
MB6	0	13.91
MB6	30	10.27
MB6	60	12.87
MB6	90	13.52
MB6	120	10.4
MB6	150	9.36
MB75B	0	16.77
MB75B	30	19.76
MB75B	60	16.64
MB75B	90	10.27
MB75B	120	11.44
MB75B	150	14.82
MB9	0	18.07
MB9	30	20.54
MB9	60	10.92
MB9	90	23.4
MB9	120	17.68
MB9	150	14.04

Table xix. Streambed penetration across Meyers Branch sites showing connectivity of ground water to the streams.

		Streambed Penetration (cm)			
Stream	*X Section (m)*	*2013*	*2012*	*2011*	*2010*
MB HW	0	1.3	1.9	0.7	0.5
MB HW	30	1.6	0.6	4.5	1.2
MB HW	60	2.5	1.8	1.9	1.3
MB HW	90	2.6	1.1	1.1	1
MB HW	120	1.3	1.3	1.8	1.1
MB HW	150	2.9	3.3	3.2	2.3
MB 6	0	3	1	2.5	3.2
MB 6	30	2	3	3.1	1.6
MB 6	60	2.3	2.4	5.5	3.3
MB 6	90	2	1.9	2	1
MB 6	120	2.3	2.2	3.8	4.3
MB 6	150	3.2	2.8	3.8	4.2
MB 75B	0	4	3.5	3.4	3.2
MB 75B	30	3	6.4	5.1	4.2
MB 75B	60	3	4.8	3.2	2.3
MB 75B	90	3	4.1	3.8	3.5
MB 75B	120	3.8	3.5	3.4	4.8
MB 75B	150	4.9	2.9	3	2.7
MB 9	0	3.4	3.8	3.9	1.8
MB 9	30	2.5	3	2.1	4
MB 9	60	2.3	3.7	2.8	2.3
MB 9	90	2	1.5	5.1	2.3
MB 9	120	3.2	3.4	1.7	1.8
MB 9	150	3.4	2.2	3	2.8

Appendix iii *Mill Creek Data Tables*

Table xx. Widths of the stream from the left edge of water to the right edge of water measured yearly at each cross section

		Wetted Width (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MC 5A	0	4.59	3.78	4.1	4.26
MC 5A	30	7.71	7.54	7.54	8.53
MC 5A	60	4.92	4.26	4.59	4.59
MC 5A	90	3.93	2.8	2.62	3.93
MC 5A	120	5.58	4.92	2.3	4.92
MC 5A	150	3.61	3.28	3.28	2.95
MC 5B	0	NA	5.09	5.9	9.85
MC 5B	30	NA	4.59	4.76	6.24
MC 5B	60	NA	6.72	4.92	5.25
MC 5B	90	NA	6.07	5.58	5.24
MC 5B	120	NA	2.91	2.95	5.9
MC 5B	150	NA	3.77	5.25	4.27
MC 6	0	4.92	3.93	3.28	4.92
MC 6	30	4.92	2.31	3.61	1.97
MC 6	60	5.9	6.23	5.9	5.9
MC 6	90	6.24	5.91	4.92	6.89
MC 6	120	4.59	4.92	3.93	4.92
MC 6	150	3.61	3.28	1.97	3.61

Table xxi. Wetted perimeter at Mill Creek sites by year.

		Wetted Perimeter (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MC 5A	0	9.09	7.88	9.68	8.41
MC 5A	30	11.9	12.21	12.76	14.15
MC 5A	60	10.1	11.93	7.85	11.3
MC 5A	90	6.13	7.21	7.78	11.57
MC 5A	120	11.11	10.57	8.83	9.79
MC 5A	150	8.94	8.1	7.92	8.9
MC 5B	0	NA	14.13	14.71	14.28
MC 5B	30	NA	10	10.43	11.02
MC 5B	60	NA	12.39	12.73	14.09
MC 5B	90	NA	9.84	9.2	12.08
MC 5B	120	NA	9.1	8.63	9.77
MC 5B	150	NA	11.28	10.25	9.76
MC 6	0	11.46	11.19	10.66	10.64
MC 6	30	11.46	11.53	11.22	8.81
MC 6	60	11.49	11.73	12.8	14.93
MC 6	90	13.56	11.72	10.92	14.07
MC 6	120	11.58	12.37	9.65	9.97
MC 6	150	7.75	7.05	9.39	8.48

Table xxii. Maximum depths of Mill Creek sites by year.

		Maximum Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MC 5A	0	1.12	1.12	1.34	0.99
MC 5A	30	1.58	1.69	1.97	2.2
MC 5A	60	0.95	1.08	0.69	1.21
MC 5A	90	0.43	0.72	0.59	1.45
MC 5A	120	1.7	1.35	0.75	1.12
MC 5A	150	2.79	1.63	1.77	1.69
MC 5B	0	NA	1.62	1.61	1.39
MC 5B	30	NA	1.49	1.3	1.39
MC 5B	60	NA	1.38	1.38	1.71
MC 5B	90	NA	1.22	1.45	1.84
MC 5B	120	NA	1.58	1.41	2.13
MC 5B	150	NA	1.87	1.67	1.51
MC 6	0	1.61	2.53	2.73	2.55
MC 6	30	1.61	1.03	1.12	0.81
MC 6	60	2.44	2.77	2.82	1.57
MC 6	90	2.66	2.85	2.76	3.58
MC 6	120	1.48	1.35	2.33	2.19
MC 6	150	2.06	1.65	2.36	2.33

Table xxiii. Mean depths of Mill Creek sites by year.

		Mean Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MC 5A	0	0.87	0.73	0.87	0.69
MC 5A	30	1.35	1.39	1.44	1.71
MC 5A	60	0.57	0.55	0.4	0.77
MC 5A	90	0.3	0.44	0.36	1
MC 5A	120	1.27	0.99	0.5	0.76
MC 5A	150	2.14	0.96	1.15	1
MC 5B	0	NA	1.14	1.18	1.03
MC 5B	30	NA	1.08	0.88	1.04
MC 5B	60	NA	1.06	1	1.15
MC 5B	90	NA	0.85	1.15	1.28
MC 5B	120	NA	1.12	1.05	1.53
MC 5B	150	NA	1.33	1.26	1.2
MC 6	0	1.15	1.4	1.83	1.96
MC 6	30	1.15	0.65	0.78	0.56
MC 6	60	1.7	2.37	2.09	1.46
MC 6	90	1.77	2.16	1.92	2.49
MC 6	120	1.22	1.13	1.67	1.71
MC 6	150	1.58	1.26	1.25	1.52

Table xxiv. Width to depth ratios of Mill Creek sites by year.

		Width to Depth Ratio			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MC 5A	0	9.54	9.9	10.41	11.26
MC 5A	30	7.1	7.24	7.8	6.68
MC 5A	60	17.12	20.42	19.07	13.78
MC 5A	90	19.73	15.55	20.69	10.15
MC 5A	120	7.53	9.43	17.2	11.67
MC 5A	150	2.5	5.85	5.63	6.65
MC 5B	0	NA	11.54	11.37	12.78
MC 5B	30	NA	9.61	11.28	9.49
MC 5B	60	NA	10.75	12.09	11.25
MC 5B	90	NA	8.89	6.99	8.2
MC 5B	120	NA	6.47	7.25	5.18
MC 5B	150	NA	6.81	7.17	7.11
MC 6	0	8.81	5.97	4.4	3.77
MC 6	30	8.81	12.48	13.56	13.27
MC 6	60	2.93	3.21	4.6	5.99
MC 6	90	4.3	3.61	4.17	3.63
MC 6	120	4.77	5.13	4.42	3.87
MC 6	150	3.24	4.01	6.02	3.6

Table xxv. Entrenchment ratios of Mill Creek sites by year.

		Entrenchment Ratio			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MC 5A	0	1.65	1.69	1.32	1.56
MC 5A	30	1.59	1.51	1.39	1.38
MC 5A	60	1.41	1.22	1.69	1.27
MC 5A	90	1.45	1.52	1.34	1.36
MC 5A	120	1.28	1.31	1.24	1.4
MC 5A	150	1.99	1.9	1.9	1.68
MC 5B	0	NA	1.27	1.3	1.35
MC 5B	30	NA	1.54	1.35	1.4
MC 5B	60	NA	1.34	1.33	1.27
MC 5B	90	NA	1.61	1.59	1.41
MC 5B	120	NA	1.89	1.72	1.7
MC 5B	150	NA	1.35	1.31	
MC 6	0	1.5	1.64	1.63	1.86
MC 6	30	1.5	1.71	1.52	1.48
MC 6	60	1.7	1.81	1.47	1.46
MC 6	90	1.6	1.56	1.56	1.35
MC 6	120	1.63	1.55	1.64	1.81
MC 6	150	2.09	2.72	1.4	1.92

Table xxvi. Bankfull areas of Mill Creek sites by year.

		Bankfull Area (ft^2)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MC 5A	0	7.19	5.26	7.9	5.37
MC 5A	30	12.94	13.98	16.19	19.49
MC 5A	60	5.6	6.15	3.03	8.16
MC 5A	90	1.75	3	2.72	10.1
MC 5A	120	12.16	9.26	4.33	6.75
MC 5A	150	11.48	6.62	7.46	6.63
MC 5B	0	NA	14.98	15.86	13.49
MC 5B	30	NA	9.61	8.77	10.3
MC 5B	60	NA	12.05	12.09	14.92
MC 5B	90	NA	6.42	9.28	13.37
MC 5B	120	NA	8.13	8	12.15
MC 5B	150	NA	12	11.4	10.22
MC 6	0	11.66	14.03	14.76	14.5
MC 6	30	11.66	9.71	8.27	4.15
MC 6	60	17.49	18.01	20.14	14.77
MC 6	90	13.48	16.85	15.38	22.49
MC 6	120	14.46	15.51	12.34	11.29
MC 6	150	8.08	6.36	9.37	8.3

Table xxvii. Hydraulic radii of Mill Creek sites by year.

		Hydraulic Radius			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MC 5A	0	0.79	0.67	0.82	0.64
MC 5A	30	1.09	1.14	1.27	1.38
MC 5A	60	0.55	0.52	0.39	0.72
MC 5A	90	0.29	0.42	0.35	0.87
MC 5A	120	1.09	0.88	0.49	0.69
MC 5A	150	1.28	0.82	0.94	0.8
MC 5B	0	NA	1.06	1.12	0.95
MC 5B	30	NA	1.57	0.84	0.93
MC 5B	60	NA	0.97	0.95	1.06
MC 5B	90	NA	0.65	1.01	1.11
MC 5B	120	NA	0.89	0.93	1.24
MC 5B	150	NA	1.06	1.64	1.05
MC 6	0	1.02	1.25	1.38	1.36
MC 6	30	1.02	0.84	0.74	0.47
MC 6	60	1.52	1.54	1.57	1.03
MC 6	90	0.99	1.44	1.41	1.6
MC 6	120	1.25	1.25	1.28	1.13
MC 6	150	1.04	0.9	1	0.98

Table xxviii. Sediment sizes found in Mill Creek sites using a standard sieve set. DB 84 = size at the 84th percentile and DB 50 = size at the 50th percentile.

Sediment Sizes (mm)			
Stream	*X Section (m)*	*DB 84*	*DB 50*
MC 5A	0	0	7
MC 5A	30	0	15
MC 5A	60	0	15
MC 5A	90	2.5	9.5
MC 5A	120	0	8.5
MC 5A	150	0	11.5
MC 5B	0	2.37	1
MC 5B	30	0	7.5
MC 5B	60	5.66	6.5
MC 5B	90	2.72	6
MC 5B	120	0	11
MC 5B	150	0	8.5
MC 6	0	2.75	11
MC 6	30	3.61	6
MC 6	60	0	8.5
MC 6	90	6.68	7.5
MC 6	120	0	9.5
MC 6	150	0	8

Table xix. Canopy cover of Mill Creek sites showing the percentage of open canopy.

Canopy Cover		
Reach	*X Section*	*% Open*
MC5A	0	5.33
MC5A	30	11.57
MC5A	60	6.37
MC5A	90	12.74
MC5A	120	8.97
MC5A	150	11.83
MC5B	0	7.41
MC5B	30	10.01
MC5B	60	7.93
MC5B	90	10.01
MC5B	120	7.54
MC5B	150	10.66
MC6	0	14.3
MC6	30	14.95
MC6	60	10.4
MC6	90	5.85
MC6	120	9.75
MC6	150	11.18

Table xxx. Streambed penetration across Mill Creek sites showing connectivity of ground water to the streams.

		Streambed Penetration (cm)			
Stream	*X Section (m)*	*2013*	*2012*	*2011*	*2010*
MC 5A	0	2.7	3.9	3.3	5.4
MC 5A	30	2	3	3.9	5.8
MC 5A	60	3.2	4.8	2.3	5
MC 5A	90	2.3	3.4	3.8	3.4
MC 5A	120	4	2	5	3.7
MC 5A	150	5.8	3.5	3.6	4.6
MC 5B	0	7.5	3.5	2.4	NA
MC 5B	30	6.4	4	3.8	NA
MC 5B	60	4.6	3.6	3.9	NA
MC 5B	90	3	2.3	5.2	NA
MC 5B	120	4	5.6	4.2	NA
MC 5B	150	2.7	2.3	3	NA
MC 6	0	3.5	4.9	4.9	3.5
MC 6	30	2	3.2	2.4	4
MC 6	60	4.2	3.9	3.3	4
MC 6	90	14.6	11.3	13.4	1.3
MC 6	120	4.4	5.7	7	4.8
MC 6	150	3.4	6.3	8.2	3.9

Appendix iv *Pen Branch Data Tables*

Table xxxi. Widths of the stream from the left edge of water to the right edge of water measured yearly at each cross section.

Wetted Width (ft)					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
PB 3	0	9.51	3.94	4.59	5.25
PB 3	30	8.2	5.9	5.9	4.92
PB 3	60	4.98	4.15	3.93	2.63
PB 3	90	7.39	4.01	3.45	3.94
PB 3	120	5.58	4.57	3.28	2.32
PB 3	150	5.9	3.61	3.28	2.29
PB 4	0	4.43	6.89	1.64	9.19
PB 4	30	6.07	4.25	5.25	6.89
PB 4	60	5.57	2.95	4.27	4.27
PB 4	90	6.56	6.73	6.56	4.26
PB 4	120	7.22	6.56	5.91	3.23
PB 4	150	7.22	5.57	0	0.99

Table xxxii. Wetted perimeters of Pen Branch sites by year.

Wetted Perimeter (ft)					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
PB 3	0	17.37	13.48	17.58	12.34
PB 3	30	14.6	14.08	11.55	11.59
PB 3	60	12.99	11.53	13.79	11.23
PB 3	90	12.99	11.02	14.53	11.63
PB 3	120	11.65	9.79	9.85	2.62
PB 3	150	9.67	10.1	13.07	11.1
PB 4	0	11.86	12.68	13.73	17.1
PB 4	30	14.46	14.29	14.45	16.5
PB 4	60	10.38	10.46	11.92	11.47
PB 4	90	13.09	14.19	12.6	10.98
PB 4	120	11.42	11.33	12.02	12.02
PB 4	150	13.22	12.92	14.56	12.12

Table xxxiii. Maximum depths of Pen Branch sites by year.

		Maximum Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
PB 3	0	2.2	1.39	2.46	1.58
PB 3	30	2.09	2.2	1.74	1.85
PB 3	60	1.61	1.79	1.78	1.38
PB 3	90	2.31	1.68	2.72	1.95
PB 3	120	1.64	1.51	1.58	1.74
PB 3	150	1.88	1.93	2.29	1.95
PB 4	0	1.69	2.06	2.3	1.61
PB 4	30	1.53	1.71	2.03	2.59
PB 4	60	0.93	0.97	1.15	1.48
PB 4	90	2.58	2.57	3.05	2.18
PB 4	120	2.56	2.84	2.2	2.27
PB 4	150	1.88	2.05	2.26	2.44

Table xxxiv. Mean depths of Pen Branch sites by year.

		Mean Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
PB 3	0	1.58	0.49	1.48	1.13
PB 3	30	1.29	1.24	1.28	1.24
PB 3	60	0.93	1.2	1.2	0.89
PB 3	90	1.36	0.98	1.77	1.36
PB 3	120	0.98	0.83	1.06	1.01
PB 3	150	1.23	1.24	1.34	1.25
PB 4	0	1.13	1.36	1.39	0.99
PB 4	30	0.9	1.06	1.04	1.28
PB 4	60	0.67	0.7	0.77	0.9
PB 4	90	1.78	1.84	2.01	1.3
PB 4	120	1.54	1.71	1.57	1.44
PB 4	150	1.13	1.29	1.59	1.38

Table xxxv. Width to depth ratios of Pen Branch sites by year.

Width to Depth Ratio					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
PB 3	0	7.74	15.22	11.08	10.19
PB 3	30	9.04	9.93	8.19	8.23
PB 3	60	15.84	11.55	10.65	11.8
PB 3	90	15.28	8.04	7.15	7.04
PB 3	120	10.47	8.58	8.35	10.34
PB 3	150	6.51	7.02	8.7	7.42
PB 4	0	9.47	9.47	8.99	16.39
PB 4	30	14.26	12.81	13.03	10.54
PB 4	60	14.84	14.31	15.03	11.77
PB 4	90	5.74	5.73	4.84	6.95
PB 4	120	8.36	7.88	6.8	7.35
PB 4	150	9.51	8.73	6.41	7.41

Table xxxvi. Entrenchment ratios of Pen Branch sites by year.

Entrenchment Ratio					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
PB 3	0	1.5	1.79	1.2	1.71
PB 3	30	1.31	1.24	1.55	1.61
PB 3	60	1.14	1.21	1.4	1.58
PB 3	90	1.35	1.82	1.3	1.75
PB 3	120	1.34	1.86	1.56	1.32
PB 3	150	1.71	1.58	1.13	1.45
PB 4	0	1.64	1.42	1.47	1.37
PB 4	30	1.43	1.35	1.4	1.44
PB 4	60	1.39	1.52	1.35	1.42
PB 4	90	1.49	1.45	1.65	1.66
PB 4	120	1.3	1.24	1.63	1.67
PB 4	150	1.56	1.49	1.83	1.72

Table xxxvii. Bankfull areas of Pen Branch sites by year.

		Bankfull area ft^2			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
PB3	0	20.57	14.45	24.24	12.99
PB3	30	15.65	15.31	13.38	12.64
PB3	60	6.72	8.53	15.34	9.39
PB3	90	14.71	11.72	22.47	13
PB3	120	10.02	8.82	9.36	10.6
PB3	150	9.83	10.74	15.64	11.63
PB4	0	12.07	14.03	17.38	16.05
PB4	30	11.56	14.38	14.16	17.28
PB4	60	6.64	7.03	8.95	9.52
PB4	90	18.2	19.43	19.54	11.73
PB4	120	12.18	11.61	16.71	15.28
PB4	150	12.1	14.57	16.74	14.12

Table xxxviii. Hydraulic radii of Pen Branch sites by year.

		Hydraulic Radius			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
PB 3	0	1.18	1.07	1.38	1.05
PB 3	30	1.07	1.09	1.16	1.09
PB 3	60	0.52	0.74	1.11	0.84
PB 3	90	1.13	1.06	1.55	1.12
PB 3	120	0.86	0.84	0.95	0.9
PB 3	150	1.02	1.06	1.2	1.05
PB 4	0	1.02	1.11	1.27	0.94
PB 4	30	0.8	1.01	0.98	1.05
PB 4	60	0.64	0.67	0.75	0.83
PB 4	90	1.39	1.37	1.55	1.07
PB 4	120	1.07	1.02	1.39	1.27
PB 4	150	0.92	1.13	1.15	1.17

Table xxxix. Sediment sizes found in Pen Branch sites using a standard sieve set. DB 84 = size at the 84th percentile and DB 50 = size at the 50th percentile

Sediment Size (mm)			
Stream	*X Section (m)*	*DB 84*	*DB 50*
PB 3	0	27.67	40.5
PB 3	30	7.82	17
PB 3	60	26.14	3.66
PB 3	90	27.59	31.5
PB 3	120	65.36	5.54
PB 3	150	54.95	36
PB 4	0	0	12.5
PB 4	30	2.63	16.5
PB 4	60	2.28	21.5
PB 4	90	0	1
PB 4	120	53.84	3.72
PB 4	150	46.32	40

Table xl. Canopy cover of Pen Branch sites showing the percentage of open canopy.

Canopy Cover		
Reach	*X Section*	*% Open*
PB 3	0	7.54
PB 3	30	6.11
PB 3	60	5.85
PB 3	90	7.41
PB 3	120	19.24
PB 3	150	9.62
PB 4	0	8.06
PB 4	30	9.62
PB 4	60	12.22
PB 4	90	14.82
PB 4	120	5.07
PB 4	150	6.11

Table xli. Streambed penetration across Pen Branch sites showing connectivity of ground water to the streams.

		Streambed Penetration (cm)			
Stream	*X Section (m)*	*2013*	*2012*	*2011*	*2010*
PB 3	0	2.6	7	3.7	4.2
PB 3	30	7.8	5.3	2.4	6.7
PB 3	60	2.2	5.6	2.1	1.4
PB 3	90	2.2	10.2	2.9	3.7
PB 3	120	2.5	6.2	2.4	1.3
PB 3	150	3.9	1.4	3.1	4.4
PB 4	0	9.2	9.4	5.9	6.3
PB 4	30	3.7	4.6	6.3	8.4
PB 4	60	9.2	8.8	5.5	4.9
PB 4	90	13.3	7.6	8.2	10.2
PB 4	120	7.4	3.8	4	9.1
PB 4	150	2	NA	1.8	3.3

Appendix v. *Tinker Creek Data Tables*

Table xlii. Widths of the stream from the left edge of water to the right edge of water measured yearly at each cross section

		Wetted Width (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
TC 2A	0	12.13	9.18	10.5	6.89
TC 2A	30	10.63	11.18	9.85	11.83
TC 2A	60	10.82	10.49	6.24	8.53
TC 2A	90	6.89	5.91	8.86	5.91
TC 2A	120	9.19	9.19	7.55	7.55
TC 2A	150	9.84	9.03	7.21	NA
TC 2A	180	11.49	6.24	7.87	10.84
TC 2A	210	9.2	8.86	9.2	9.2
TC 2C	0	10.83	10.5	8.53	9.19
TC 2C	30	13.29	13.29	13.45	9.19
TC 2C	60	8.69	8.69	8.2	9.51
TC 2C	90	11.32	11.32	10.18	11.48
TC 2C	120	6.23	6.23	5.25	8.53
TC 2C	150	10.83	10.83	6.56	6.89
TC 2C	180	12.13	7.87	7.55	8.53
TC 2C	210	10.17	10.34	8.2	9.19
TC 3	0	NA	3.61	3.28	3.94
TC 3	30	NA	4.1	3.94	3.94
TC 3	60	NA	5.9	4.26	5.24
TC 3	90	NA	4.26	4.59	4.92
TC 3	120	NA	5.74	5.25	6.23
TC 3	150	NA	3.77	3.61	3.93
TC 5	0	5.41	5.25	5.32	4.6
TC 5	30	6.39	6.07	5.57	6.23
TC 5	60	5.58	5.9	4.92	4.92
TC 5	90	4.26	4.1	4.26	4.26
TC 5	120	5.9	4.92	4.59	4.26
TC 5	150	5.24	4.26	4.92	4.59

Table xliii. Wetted perimeter of Tinker Creek sites by year.

		Wetted Perimeter (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
TC 2A	0	17.3	16.72	15.57	15.72
TC 2A	30	17.74	17.19	12.87	14.25
TC 2A	60	16.96	14.8	11.76	15.21
TC 2A	90	11.37	11.41	13.16	12.58
TC 2A	120	14.42	12.7	12.85	12
TC 2A	150	13.15	14.45	11.74	NA
TC 2A	180	16.48	14.55	16.53	20.83
TC 2A	210	15.08	15.51	11.92	15.37
TC 2C	0	18.51	14.97	13.8	15.49
TC 2C	30	18.74	18.74	18.82	16.96
TC 2C	60	16.99	16.99	16.85	16.65
TC 2C	90	14.17	14.17	14.82	16.32
TC 2C	120	13.93	13.93	14.19	16.45
TC 2C	150	19.69	15.9	16.95	15.94
TC 2C	180	16.2	17.07	15.36	19.54
TC 2C	210	15.37	16.24	17.42	20.6
TC 3	0	NA	5.27	8.54	8.62
TC 3	30	NA	11.06	11.21	9.31
TC 3	60	NA	10.53	10.65	11.99
TC 3	90	NA	6.4	7.91	10.4
TC 3	120	NA	11.33	12.78	10.64
TC 3	150	NA	11.51	8.6	9.18
TC 5	0	10.18	10.55	11.8	8.73
TC 5	30	9.99	10.27	8.25	11.59
TC 5	60	9.86	8.6	8.62	9.49
TC 5	90	10.58	11.32	6.62	11.72
TC 5	120	10.86	9.62	12.73	18.59
TC 5	150	11.41	9.98	10.39	13.02

Table xliv. Maximum depths of Tinker creek sites by year.

		Maximum Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
TC 2A	0	1.17	1.31	1.05	1.15
TC 2A	30	1.18	1.23	1.12	1.15
TC 2A	60	1.48	1.21	1.55	1.51
TC 2A	90	1.18	1.41	1.18	1.4
TC 2A	120	1.36	1.02	1.41	1.58
TC 2A	150	1.11	1.18	1.77	na
TC 2A	180	1.82	1.79	1.11	2.07
TC 2A	210	1.42	1.32	1.08	1.33
TC 2C	0	1.48	1.31	1.44	1.64
TC 2C	30	1.27	1.27	1.28	1.1
TC 2C	60	1.25	1.25	1.22	1.43
TC 2C	90	0.99	0.99	1.21	1.18
TC 2C	120	1.67	1.67	1.64	1.7
TC 2C	150	1.14	1.14	0.85	0.95
TC 2C	180	0.75	0.95	0.66	0.96
TC 2C	210	1.28	1.24	1.58	2.25
TC 3	0	NA	0.63	1.08	0.89
TC 3	30	NA	2.9	3.25	2.59
TC 3	60	NA	1.59	1.86	2.61
TC 3	90	NA	1.03	1.31	1.9
TC 3	120	NA	3.1	3.53	3.18
TC 3	150	NA	2.77	2.19	1.89
TC 5	0	1.12	1.08	1.18	0.81
TC 5	30	1.12	1.13	0.88	1.17
TC 5	60	1.3	1.14	1.15	1.43
TC 5	90	1.97	2.15	1.15	2.21
TC 5	120	1.73	1.64	2.06	2.21
TC 5	150	1.99	2.13	1.84	2.69

Table xlv. Mean depths of Tinker Creek sites by year.

		Mean Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
TC 2A	0	0.7	0.64	0.7	0.63
TC 2A	30	0.71	0.77	0.7	0.84
TC 2A	60	0.97	0.87	0.93	1.04
TC 2A	90	0.75	0.94	0.83	0.91
TC 2A	120	0.86	0.79	0.83	0.9
TC 2A	150	0.81	0.7	1.11	NA
TC 2A	180	1.13	0.86	0.67	1.27
TC 2A	210	0.96	0.76	0.79	0.97
TC 2C	0	0.91	0.94	1	1
TC 2C	30	0.94	0.94	0.99	0.81
TC 2C	60	0.54	0.54	0.61	0.65
TC 2C	90	0.7	0.7	0.79	0.75
TC 2C	120	0.81	0.81	0.79	0.77
TC 2C	150	0.47	0.58	0.47	0.43
TC 2C	180	0.34	0.59	0.28	0.6
TC 2C	210	0.86	0.69	0.73	1.46
TC 3	0	NA	0.48	0.65	0.32
TC 3	30	NA	1.9	1.82	1.52
TC 3	60	NA	1.1	0.93	1.73
TC 3	90	NA	0.64	0.8	1.13
TC 3	120	NA	2.76	2.26	2.52
TC 3	150	NA	1.79	1.4	1.24
TC 5	0	0.63	0.63	0.75	0.5
TC 5	30	0.74	0.71	0.67	0.71
TC 5	60	0.89	0.86	0.85	0.91
TC 5	90	1.25	1.26	0.99	1.23
TC 5	120	1.12	1.15	1.26	1.22
TC 5	150	1.3	1.28	1.28	1.4

Table xlvi. Width to depth ratios of Tinker Creek sites by year.

		Width to Depth Ratio			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
TC 2A	0	23.41	23.8	21.69	23.87
TC 2A	30	22.87	20.42	15.81	15.63
TC 2A	60	16.29	15.98	11.52	13.68
TC 2A	90	14.15	10.74	15.18	12.37
TC 2A	120	14.29	13.82	14.82	12.53
TC 2A	150	14.77	19.1	9.3	NA
TC 2A	180	12.5	15.14	24.19	14.21
TC 2A	210	14.69	17.63	14.32	15.15
TC 2C	0	18.15	14.04	13.08	13.61
TC 2C	30	18.33	18.33	18.29	19.77
TC 2C	60	28.11	28.11	26.57	23.43
TC 2C	90	19.23	19.23	18.23	20.2
TC 2C	120	15.7	15.7	16.46	19.79
TC 2C	150	39.57	25.46	35.68	36
TC 2C	180	46.44	28.15	54.25	31.77
TC 2C	210	16.53	22.57	23.37	13.09
TC 3	0	NA	9.56	9.37	23.41
TC 3	30	NA	3.8	4.34	4.45
TC 3	60	NA	8.2	9.55	4.68
TC 3	90	NA	9.13	9.14	8.09
TC 3	120	NA	2.29	4.13	2.54
TC 3	150	NA	3.3	4.82	6.08
TC 5	0	15.13	15.48	14.87	15.7
TC 5	30	10.89	11.49	11.64	14.9
TC 5	60	9.82	8.85	9.46	9.36
TC 5	90	5.84	5.67	5.11	6.84
TC 5	120	8.64	7.18	9.29	10.5
TC 5	150	4.48	4.55	5.06	6.09

Table xlvii. Entrenchment Ratios of Tinker Creek sites by year.

		Entrenchment Ratio			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
TC 2A	0	1.39	1.46	1.45	1.52
TC 2A	30	1.31	1.36	1.62	1.7
TC 2A	60	1.25	1.43	1.5	1.41
TC 2A	90	1.44	1.51	1.41	1.35
TC 2A	120	1.34	1.39	1.47	1.49
TC 2A	150	1.53	1.37	1.91	na
TC 2A	180	1.4	1.52	1.5	1.11
TC 2A	210	1.3	1.37	1.65	1.29
TC 2C	0	1.11	1.39	1.48	1.42
TC 2C	30	1.24	1.24	1.21	1.39
TC 2C	60	1.21	1.21	1.11	1.21
TC 2C	90	1.36	1.36	1.32	1.28
TC 2C	120	1.56	1.56	1.51	1.31
TC 2C	150	1.23	1.55	1.39	1.5
TC 2C	180	1.45	1.38	1.38	1.21
TC 2C	210	1.35	1.32	1.25	1.2
TC 3	0	NA	1.66	1.72	1.49
TC 3	30	NA	1.48	1.37	1.57
TC 3	60	NA	1.35	1.4	1.58
TC 3	90	NA	1.99	1.62	1.33
TC 3	120	NA	1.93	1.23	1.9
TC 3	150	NA	1.81	1.61	1.41
TC 5	0	1.6	1.48	1.29	1.75
TC 5	30	1.89	1.87	1.74	1.32
TC 5	60	1.74	1.77	1.65	1.54
TC 5	90	1.88	1.92	1.82	1.68
TC 5	120	1.73	2.03	1.46	1.33
TC 5	150	2.35	2.35	2.03	1.58

Table xlviii. Bankfull areas of Tinker Creek sites by year.

		Bankfull Area (ft^2)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
TC 2A	0	11.4	9.78	10.65	9.42
TC 2A	30	11.51	12.17	8.16	11.06
TC 2A	60	15.31	12.11	9.94	14.8
TC 2A	90	7.94	9.47	10.3	10
TC 2A	120	10.54	8.59	10.2	10.16
TC 2A	150	14.77	9.35	11.41	na
TC 2A	180	15.91	11.19	10.79	23.01
TC 2A	210	13.48	10.14	8.97	14.25
TC 2C	0	15.08	12.35	13.11	13.57
TC 2C	30	16.24	16.24	18.02	12.94
TC 2C	60	8.14	8.14	9.84	9.95
TC 2C	90	9.45	9.45	11.42	11.43
TC 2C	120	10.29	10.29	10.32	11.69
TC 2C	150	8.82	8.5	7.89	6.64
TC 2C	180	5.35	9.82	4.26	11.38
TC 2C	210	12.24	10.81	12.39	27.82
TC 3	0	NA	2.19	5.98	2.4
TC 3	30	NA	13.72	14.37	10.3
TC 3	60	NA	9.96	8.29	14.03
TC 3	90	NA	3.73	5.88	10.36
TC 3	120	NA	17.48	21.09	16.16
TC 3	150	NA	10.56	9.46	9.34
TC 5	0	5.96	6.16	8.34	3.93
TC 5	30	5.99	5.82	5.2	7.54
TC 5	60	7.79	6.57	6.85	7.72
TC 5	90	9.16	9.01	5.01	10.36
TC 5	120	10.84	9.54	14.72	15.63
TC 5	150	7.57	6.62	8.27	11.96

Table xlix. Hydraulic Radii of Tinker Creek sites by year.

Hydraulic Radius					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
TC 2A	0	0.66	0.58	0.68	0.6
TC 2A	30	0.65	0.71	0.63	0.78
TC 2A	60	0.9	0.82	0.85	0.97
TC 2A	90	0.7	0.83	0.8	0.82
TC 2A	120	0.73	0.68	0.79	0.85
TC 2A	150	0.74	0.65	0.97	na
TC 2A	180	0.97	0.77	0.65	1.1
TC 2A	210	0.89	0.65	0.75	0.93
TC 2C	0	0.81	0.83	0.95	0.88
TC 2C	30	0.87	0.87	0.96	0.76
TC 2C	60	0.48	0.48	0.58	0.6
TC 2C	90	0.67	0.67	0.77	0.7
TC 2C	120	0.74	0.74	0.73	0.71
TC 2C	150	0.45	0.53	0.47	0.42
TC 2C	180	0.33	0.58	0.28	0.58
TC 2C	210	0.8	0.67	0.71	1.35
TC 3	0	NA	0.42	0.7	0.28
TC 3	30	NA	1.24	1.64	1.11
TC 3	60	NA	0.95	0.91	1.17
TC 3	90	NA	6.4	0.74	1
TC 3	120	NA	0.92	1.65	1.52
TC 3	150	NA	0.92	1.1	1.02
TC 5	0	0.59	0.58	0.71	0.45
TC 5	30	0.6	0.57	0.63	0.65
TC 5	60	0.79	0.76	0.79	0.81
TC 5	90	0.87	0.8	0.76	0.88
TC 5	120	1	0.99	1.16	0.84
TC 5	150	0.66	0.66	0.8	0.92

Table L. Sediment sizes found in Tinker Creek sites using a standard sieve set. DB 84 = size at the 84th percentile and DB 50 = size at the 50th percentile

	Sediment Size (mm)		
Stream	*X Section (m)*	*DB 84*	*DB50*
TC 2A	0	0	2.5
TC 2A	30	0	0.5
TC 2A	60	0	1
TC 2A	90	0	6.5
TC 2A	120	0	1
TC 2A	150	0	1.5
TC 2A	180	0	1
TC 2A	210	0	1
TC 2C	0	0	1
TC 2C	30	0	1
TC 2C	60	0	3.5
TC 2C	90	0	1
TC 2C	120	0	1
TC 2C	150	0	1
TC 2C	180	0	1
TC 2C	210	0	1
TC 3	0	2.98	9
TC 3	30	3.42	6.5
TC 3	60	0	12
TC 3	90	3.27	10
TC 3	120	0	7
TC 3	150	2.79	5
TC 5	0	0	3
TC 5	30	0	5
TC 5	60	0	6.5
TC 5	90	0	6
TC 5	120	5.18	7.5
TC 5	150	0	4

Table Li. Canopy cover of Tinker Creek sites showing the percentage of open canopy.

Canopy Cover		
Reach	*X Section*	*% Open*
TC2A	0	6.63
TC2A	30	16.25
TC2A	60	10.01
TC2A	90	6.11
TC2A	120	17.55
TC2A	150	15.08
TC2A	180	22.49
TC2A	210	7.8
TC2C	0	5.85
TC2C	30	9.1
TC2C	60	9.23
TC2C	90	41.73
TC2C	120	8.84
TC2C	150	18.33
TC2C	180	10.27
TC2C	210	12.74
TC 3	0	9.75
TC 3	30	8.06
TC 3	60	7.02
TC 3	90	11.05
TC 3	120	5.46
TC 3	150	11.05
TC5	0	13.91
TC5	30	11.05
TC5	60	7.02
TC5	90	16.77
TC5	120	9.75
TC5	150	8.19

Table Lii. Streambed penetration across Tinker Creek sites showing connectivity of ground water to the streams.

		Streambed Penetration (cm)			
Stream	*X Section (m)*	*2013*	*2012*	*2011*	*2010*
TC 2A	0	2.8	4.2	5.3	NA
TC 2A	30	3.9	3.5	5.2	NA
TC 2A	60	3.3	1.4	4.8	4
TC 2A	90	1.8	2.3	2.5	11.7
TC 2A	120	2.2	3.7	3.2	4
TC 2A	150	NA	3.7	6.4	6
TC 2A	180	5	3.1	3.2	3.3
TC 2A	210	1.4	2.6	3.7	3.6
TC 2C	0	2.7	4.9	3.7	2.8
TC 2C	30	2.6	3.3	2.2	2.8
TC 2C	60	5.6	4.1	4.9	3.7
TC 2C	90	4.7	3.7	8.2	6.8
TC 2C	120	3.3	5.9	8.8	5.8
TC 2C	150	13.2	9.4	6.8	11.2
TC 2C	180	3.4	3.6	4.1	7
TC 2C	210	3.7	5	4.8	5.8
TC 3	0	2.2	2.8	4.8	NA
TC 3	30	2.4	1.4	4.2	NA
TC 3	60	3	4.8	3.5	NA
TC 3	90	3.8	3	5.8	NA
TC 3	120	4.2	4.8	5.3	NA
TC 3	150	9.8	8.2	9.6	NA
TC 5	0	2.8	4.1	2.5	2.8
TC 5	30	2.2	3	3.2	5
TC 5	60	2.6	6.3	1.7	2.3
TC 5	90	8.2	6.8	11.2	3.5
TC 5	120	2	2.9	3.1	5
TC 5	150	5.5	3.5	2.2	8.6

Appendix vi *Upper Three Runs Data Tables*

Table Liii. Widths of the stream from the left edge of water to the right edge of water measured yearly at each cross section

Wetted Width (ft)					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
U 6	0	3.9	3.44	4.26	8.53
U 6	30	2.46	1.15	3.28	7.62
U 6	50	2.62	2.95	4.59	5.52
U 6	70	2.94	4.59	0.99	1.53
U 6	90	6.07	6.23	2.63	3.36
U 6	110	2.95	2.14	2.96	3.97
U 8	0	6.23	2.46	2.3	4.92
U 8	30	9.19	4.92	3.28	5.91
U 8	60	6.23	4.76	4.26	5.91
U 8	90	8.86	6.56	3.28	7.54
U 8	120	12.47	5.25	1.31	9.68
U 8	150	10.17	6.89	3.94	10.84
U 10	0	2.3	1.97	1.64	1.64
U 10	30	3.54	2.63	2.62	4.27
U 10	60	3.28	1.97	0.98	1.97
U 10	90	5.91	3.25	2.3	3.28
U 10	120	2.29	1.13	0.65	1.64
U 10	150	2.95	2.12	2.3	3.28
U 36A	0	2.46	2.96	4.59	4.59
U 36A	30	3.94	2.62	3.94	2.95
U 36A	60	5.58	3.46	3.93	4.59
U 36A	90	5.25	3.28	6.89	6.56
U 36A	120	4.59	4.76	2.3	2.62
U 36A	150	9.68	3.61	3.93	3.28
U 36C	0	NA	2.46	4.27	4.92
U 36C	30	NA	6.23	6.89	3.94
U 36C	60	NA	3.12	2.95	3.61
U 36C	90	NA	3.28	2.95	2.3
U 36C	120	NA	2.96	3.28	2.95
U 36C	150	NA	2.63	3.28	3.29

Table Liv. Wetted bankfull perimeters of Upper Three Runs sites by year.

		Wetted Perimeter (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
U 6	0	11.58	11.64	12.16	13.91
U 6	30	8.85	8.33	13.13	12.63
U 6	50	14.41	13.31	13.46	10.02
U 6	70	15.77	14.19	18.21	10.43
U 6	90	16.54	12.39	16.16	10.34
U 6	110	7.46	7.23	9.91	6.16
U 8	0	14.62	17.36	13	11.61
U 8	30	22.35	21.26	17.47	20.27
U 8	60	21.99	11.33	12.17	22.32
U 8	90	19.17	12.7	16.8	21.15
U 8	120	22.43	18.19	16.67	23.29
U 8	150	19.68	13.38	15.34	18.84
U 10	0	7.52	7.15	5.02	3.69
U 10	30	5.31	6.9	5.64	7.69
U 10	60	11.63	7.39	11.32	6.71
U 10	90	11.25	11.76	12.18	11.2
U 10	120	5.76	6.88	6.7	7.1
U 10	150	8.38	8.65	8.92	10.86
U 36A	0	6.87	7.3	7.58	15.66
U 36A	30	11.91	7.24	9.32	9.01
U 36A	60	10.46	13.44	7.51	12
U 36A	90	16.1	10.83	11.67	14.61
U 36A	120	14.26	9.75	8.6	9.56
U 36A	150	17.77	14.41	12.41	11.55
U 36C	0	NA	7.98	6.5	6.57
U 36C	30	NA	11.44	10.85	11.09
U 36C	60	NA	11.38	11.88	7.94
U 36C	90	NA	6.31	6.34	6.11
U 36C	120	NA	5.15	5.2	8.18
U 36C	150	NA	7.75	10.51	6

Table Lv. Maximum depths of Upper Three Runs sites by year.

		Maximum Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
U 6	0	2.09	*2.26*	2.49	0.31
U 6	30	1.82	1.32	2.48	0.89
U 6	50	2.01	2.43	2.36	0.56
U 6	70	1.41	2	2.13	0.8
U 6	90	3.42	2.14	3.51	0.77
U 6	110	2.26	2.33	3.97	1.17
U 8	0	3.54	2.05	3.24	2.82
U 8	30	3.87	3.36	2.95	3.36
U 8	60	4.92	2.88	3.38	4.1
U 8	90	3.9	2.1	3.68	4.63
U 8	120	3.04	2.68	2.98	4.69
U 8	150	2.93	1.85	2.82	3.39
U 10	0	1.2	1.23	1.15	0.85
U 10	30	1.36	1.29	1.11	1.27
U 10	60	2.4	1.05	2.75	1.66
U 10	90	1.89	1.78	1.93	1.64
U 10	120	0.82	0.95	1.27	1.08
U 10	150	2.05	2.02	2.46	2.4
U 36A	0	0.54	0.66	1.04	1.64
U 36A	30	0.71	0.74	1.31	1.41
U 36A	60	0.36	0.53	1.05	1.78
U 36A	90	0.64	0.49	2	2.25
U 36A	120	0.99	0.62	1.31	1.7
U 36A	150	0.64	0.47	1.71	1.82
U 36C	0	NA	1.15	0.98	0.83
U 36C	30	NA	1.46	1.48	1.44
U 36C	60	NA	1.37	1.28	1.08
U 36C	90	NA	0.78	0.95	0.91
U 36C	120	NA	0.73	0.69	1.09
U 36C	150	NA	0.82	0.92	0.95

Table Lvi. Mean depths of Upper Three Runs sites by year.

		Mean Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
U 6	0	1.35	1.54	1.58	0.17
U 6	30	1.35	0.9	1.64	0.77
U 6	50	1.32	1.74	1.62	0.4
U 6	70	0.89	1.3	1.13	0.38
U 6	90	2.41	1.4	2.53	0.57
U 6	110	1.82	1.69	2.73	0.85
U 8	0	2.3	1.21	2.42	2.29
U 8	30	2.49	1.66	1.68	2.19
U 8	60	3.09	2.35	2.68	2.57
U 8	90	2.7	1.59	2.58	3.12
U 8	120	2.29	1.85	2.18	3.02
U 8	150	1.91	1.42	2.08	2.71
U 10	0	0.75	0.68	0.79	0.68
U 10	30	0.93	0.87	0.77	0.98
U 10	60	1.4	0.84	1.67	0.99
U 10	90	1.39	1.22	1.3	1.21
U 10	120	0.51	0.41	0.74	0.73
U 10	150	1.38	1.42	1.57	1.24
U 36A	0	0.39	0.41	0.82	0.97
U 36A	30	0.29	0.32	0.92	0.84
U 36A	60	0.21	0.22	0.76	0.84
U 36A	90	0.35	0.28	1.5	1.29
U 36A	120	0.56	0.38	0.81	0.9
U 36A	150	0.34	0.2	1.2	1.12
U 36C	0	NA	0.63	0.75	0.6
U 36C	30	NA	0.96	1.13	0.89
U 36C	60	NA	0.78	0.75	0.69
U 36C	90	NA	0.47	0.63	0.34
U 36C	120	NA	0.48	0.57	0.63
U 36C	150	NA	0.36	0.45	0.63

Table Lvii. Width to depth ratios of Upper Three Runs sites by year.

		Width to Depth Ratio			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
U 6	0	7.47	6.29	6.75	79.82
U 6	30	4.93	8.14	7.09	15.6
U 6	50	7.95	6.47	7.41	24.52
U 6	70	16.4	8.86	14.67	25
U 6	90	4.95	6.98	5.13	17.21
U 6	110	2.29	2.41	1.59	6.14
U 8	0	4.32	5.89	3.9	3.41
U 8	30	6.82	8.56	9.15	7.27
U 8	60	3.86	2.91	2.94	3.76
U 8	90	5.26	6.42	5.12	5.22
U 8	120	7.18	7.19	6.71	6.08
U 8	150	8.38	8.22	6.45	5.35
U 10	0	7.89	8.47	5.18	4
U 10	30	6.12	6.09	5.95	6.08
U 10	60	6.03	4.73	5.49	5.06
U 10	90	6.64	8.39	8.49	8.22
U 10	120	9.78	12.17	8.18	8.84
U 10	150	3.54	3.85	4.29	5.77
U 36A	0	16.79	17	8.15	12.38
U 36A	30	38.31	21.88	8.82	9.31
U 36A	60	48.62	60.5	9.09	12.42
U 36A	90	42.09	37.93	6.43	10.1
U 36A	120	21.77	23.87	9.17	9.08
U 36A	150	48.71	71.25	9.39	9.23
U 36C	0	NA	8.87	7.75	9.47
U 36C	30	NA	10.7	8.6	9.62
U 36C	60	NA	13.53	15.08	10.36
U 36C	90	NA	11.53	9.27	13.91
U 36C	120	NA	9.1	8.12	9.65
U 36C	150	NA	18.39	21.89	7.71

Table Lviii. Entrenchment ratios of Upper Three Runs sites by year.

		Entrenchment Ratio			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
U 6	0	1.36	1.42	1.38	1.01
U 6	30	2.26	1.62	1.33	1.18
U 6	50	1.35	1.61	1.45	1.34
U 6	70	1.31	1.78	1.22	1.16
U 6	90	1.41	1.4	1.32	1.22
U 6	110	1.98	1.98	2.38	1.38
U 8	0	1.23	1.56	1.36	1.56
U 8	30	1.35	1.61	1.52	1.34
U 8	60	1.15	1.77	1.75	1.42
U 8	90	1.29	1.49	1.47	1.17
U 8	120	1.36	1.6	1.91	1.23
U 8	150	1.14	1.43	1.44	1.33
U 10	0	1.2	1.24	1.6	4
U 10	30	1.87	2.01	2.29	1.82
U 10	60	1.44	1.68	1.29	2.19
U 10	90	1.49	1.47	1.34	1.46
U 10	120	1.83	1.83	1.52	1.47
U 10	150	2.19	1.95	1.51	1.42
U 36A	0	1.85	1.97	1.8	1.04
U 36A	30	1.1	1.74	1.48	1.55
U 36A	60	1.49	1.23	1.9	1.29
U 36A	90	1.03	1.43	1.63	1.23
U 36A	120	1	1.34	1.77	1.49
U 36A	150	1.1	1.28	1.37	1.49
U 36C	0	NA	2.18	2.12	1.85
U 36C	30	NA	1.33	1.18	1.3
U 36C	60	NA	1.59	1.42	2.25
U 36C	90	NA	2.25	2.02	2.5
U 36C	120	NA	1.66	1.62	1.75
U 36C	150	NA	1.38	1.1	1.89

Table Lix. Bankfull areas of Upper Three Runs sites by year.

		Bankfull Area (ft^2)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
U 6	0	13.59	14.95	16.82	2.26
U 6	30	9.01	6.58	19.03	9.28
U 6	50	18.69	19.59	19.46	3.97
U 6	70	13.06	14.99	18.8	3.62
U 6	90	28.71	13.72	32.86	5.59
U 6	110	7.58	6.91	11.86	4.43
U 8	0	22.87	20.26	22.88	17.88
U 8	30	42.34	23.58	25.85	34.91
U 8	60	33.09	16.06	21.13	24.86
U 8	90	38.38	16.25	34.08	50.92
U 8	120	37.64	24.56	31.88	55.47
U 8	150	30.58	16.54	27.91	39.38
U 10	0	4.45	3.92	3.25	1.83
U 10	30	7.18	4.64	3.51	5.82
U 10	60	11.8	6.38	15.29	4.98
U 10	90	12.83	12.51	14.34	12.01
U 10	120	2.54	2.02	4.45	4.71
U 10	150	6.74	7.78	10.56	8.89
U 36A	0	2.57	2.86	5.49	11.65
U 36A	30	3.27	2.21	7.49	6.56
U 36A	60	2.12	2.96	5.25	8.77
U 36A	90	5.12	2.95	14.48	16.79
U 36A	120	6.85	3.41	6	7.32
U 36A	150	5.61	2.81	13.5	11.53
U 36C	0	NA	3.54	4.37	3.39
U 36C	30	NA	9.9	10.99	7.62
U 36C	60	NA	8.2	8.45	4.91
U 36C	90	NA	2.57	3.66	1.59
U 36C	120	NA	2.12	2.65	3.85
U 36C	150	NA	2.37	4.39	3.04

Table Lx. Hydraulic radii of Upper Three Runs sites by year.

		Hydraulic Radius			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
U 6	0	1.17	1.29	1.38	0.16
U 6	30	1.02	0.79	1.45	0.73
U 6	50	1.3	1.47	1.45	0.4
U 6	70	0.83	1.06	1.03	0.35
U 6	90	1.74	1.11	2.03	0.54
U 6	110	1.02	0.96	1.2	0.72
U 8	0	1.56	1.17	1.76	1.54
U 8	30	1.89	1.11	1.48	1.72
U 8	60	1.5	1.42	1.74	1.11
U 8	90	2	1.28	2.03	2.41
U 8	120	1.68	1.35	1.91	2.38
U 8	150	1.55	1.24	1.82	2.09
U 10	0	0.59	0.55	0.65	0.5
U 10	30	0.74	0.67	0.62	0.76
U 10	60	1.01	0.86	1.35	0.74
U 10	90	1.14	1.06	1.18	1.07
U 10	120	0.44	0.29	0.67	0.66
U 10	150	0.8	0.9	1.18	0.82
U 36A	0	0.37	0.39	0.72	0.74
U 36A	30	0.27	0.3	0.8	0.73
U 36A	60	0.2	0.22	0.7	0.73
U 36A	90	0.32	0.27	1.24	1.15
U 36A	120	0.48	0.35	0.7	0.77
U 36A	150	0.32	0.19	1.09	1
U 36C	0	NA	0.44	0.67	0.52
U 36C	30	NA	0.87	1.01	0.69
U 36C	60	NA	0.72	0.71	0.62
U 36C	90	NA	0.41	0.58	0.26
U 36C	120	NA	0.41	0.51	0.63
U 36C	150	NA	0.31	0.42	0.51

Table Lxi. Sediment sizes found in Upper Three Runs sites using a standard sieve set. DB 84 = size at the 84th percentile and DB 50 = size at the 50th percentile

Sediment Sizes (mm)			
Stream	*X Section (m)*	*DB 84*	*DB50*
U 6	0	89.12	23.49
U 6	30	0	1
U 6	50	30.3	31
U 6	70	59	29.24
U 6	90	77.03	47.34
U 6	110	0	1
U 8	0	47.9	9.51
U 8	30	26.7	36
U 8	60	48.07	44.5
U 8	90	51.36	43
U 8	120	129.28	53.38
U 8	150	105.71	54.09
U 10	0	0	10
U 10	30	0	1
U 10	60	0	1
U 10	90	65.73	17.36
U 10	120	0	4.5
U 10	150	2.5	5
U 36A	0	0	2.5
U 36A	30	0	8
U 36A	60	0	1
U 36A	90	0	9.5
U 36A	120	0	1
U 36A	150	0	6
U 36C	0	0	1
U 36C	30	0	1
U 36C	60	0	1
U 36C	90	0	1
U 36C	120	0	1
U 36C	150	0	1

Table Lxii. Canopy cover of Upper Three Runs sites showing the percentage of open canopy

Canopy Cover		
Reach	*X Section*	*% Open*
U6	0	10.66
U6	30	9.1
U6	60	12.09
U6	90	30.03
U6	120	14.95
U6	150	12.09
U8	0	10.14
U8	30	14.95
U8	60	8.32
U8	90	17.94
U8	120	8.97
U8	150	7.93
U10	0	7.8
U10	30	13
U10	60	8.71
U10	90	10.14
U10	120	16.64
U10	150	9.125
U36A	0	11.05
U36A	30	7.15
U36A	60	7.15
U36A	90	7.54
U36A	120	9.88
U36A	150	6.89
U36C	0	19.63
U36C	30	19.37
U36C	60	15.86
U36C	90	7.28
U36C	120	9.62
U36C	150	10.27

Table Lxiii. Streambed penetration across Upper Three Runs sites showing connectivity of ground water to the streams.

		Streambed Penetration (cm)			
Reach	*X Section*	*2013*	*2012*	*2011*	*2010*
U6	0	8.4	2.2	1.5	2.8
U6	30	7.4	4.9	7	2.8
U6	50	9.9	1.2	2.2	2.2
U6	70	8.4	0	0.2	5.1
U6	90	7.3	1.1	2.1	0.1
U6	110	5.8	1	0.8	1.4
U8	0	4.9	4.3	9	8.6
U8	30	4.8	4.1	9.8	6.7
U8	60	4	2.5	12.2	5
U8	90	9.4	4.1	1.9	5.8
U8	120	7.8	1.7	8	2
U8	150	2.9	1.6	5.7	0.3
U10	0	4.3	3.3	2.9	2.3
U10	30	3.8	3.6	3.7	3.8
U10	60	6.4	6.3	1.4	3.3
U10	90	2.7	2.7	2.1	1.3
U10	120	4.3	6.7	4.7	4.5
U10	150	3.7	5.7	10.4	7.3
U36A	0	4.2	4.4	2.8	2.2
U36A	30	2.8	3.1	2.8	2
U36A	60	7.1	2.3	4.2	5.3
U36A	90	6.8	2.2	4.4	2.7
U36A	120	4	4.5	5.2	3.8
U36A	150	3.6	4.2	4.3	3.7
U36C	0	6	4.2	6	NA
U36C	30	3.8	2.8	4.2	NA
U36C	60	3.6	3.7	3.5	NA
U36C	90	2.4	3	4.3	NA
U36C	120	3.2	2.3	2	NA
U36C	150	3.3	3.2	2.2	NA

Appendix vii *McQueen Branch Data Tables*

Table Lxiv. Widths of the stream from the left edge of water to the right edge of water measured yearly at each cross section.

Wetted Width (ft)					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MQ HW	0	6.56	2.62	6.23	2.95
MQ HW	30	3.05	2.13	3.61	2.63
MQ HW	60	4.92	4.26	3.94	4.27
MQ HW	90	5.53	2.7	3.61	5.91
MQ HW	120	7.71	7.01	1.64	2.62
MQ HW	150	4.59	4.27	4.92	3.28
MQ 8	0	5.15	1.31	3.93	1.64
MQ 8	30	4.26	2.95	2.63	2.95
MQ 8	60	3.61	3.94	1.32	2.95
MQ 8	90	3.94	1.28	3.28	3.61
MQ 8	120	5.9	4.75	2.95	3.61
MQ 8	150	5.24	3.28	2.95	1.64

Table Lxv. Wetted bankfull perimeters of McQueen Branch sites by year.

Wetted Perimeter (ft)					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MQ HW	0	20.77	16.24	16.48	16.48
MQ HW	30	23.52	11.71	12.82	14.24
MQ HW	60	10.11	14.07	10.18	19.05
MQ HW	90	13.4	8.6	13.09	15.19
MQ HW	120	14.31	12.66	13.1	15.69
MQ HW	150	12.59	11.08	13.6	11.61
MQ 8	0	10.02	10.3	12.15	9.75
MQ 8	30	8.08	9.57	9.16	8.05
MQ 8	60	7.04	8.5	9.14	10.96
MQ 8	90	7.86	5.93	6.16	8.36
MQ 8	120	8.78	12.3	9.42	11.01
MQ 8	150	14.4	11.14	9.26	9.68

Table Lxvi. Maximum depths of McQueen Branch sites by year.

		Maximum Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MQ HW	0	3.74	3.71	4.14	3.22
MQ HW	30	4.18	3.87	3.94	4.3
MQ HW	60	2.32	2.92	2.99	3.93
MQ HW	90	3.24	2.13	2.62	2.93
MQ HW	120	3.74	3.09	3.45	3.99
MQ HW	150	3.72	3.28	3.9	3.07
MQ 8	0	1.44	1.44	1.84	1.5
MQ 8	30	1.54	1.54	1.64	1.36
MQ 8	60	2.11	1.67	1.94	1.69
MQ 8	90	1.18	0.88	0.85	1.21
MQ 8	120	1.45	2.59	1.64	1.7
MQ 8	150	2.33	1.91	2.17	2.15

Table Lxvii. Mean depths of McQueen branch sites by year.

		Mean Depth (ft)			
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MQ HW	0	2.48	2.97	2.98	2.02
MQ HW	30	3.36	2.88	2.94	2.46
MQ HW	60	2.92	2.02	2.29	1.96
MQ HW	90	2.03	1.59	2.62	2.21
MQ HW	120	3	2.57	2.5	2.68
MQ HW	150	3.34	2.66	2.97	2.27
MQ 8	0	0.88	0.78	0.91	0.91
MQ 8	30	1.13	0.95	1.17	0.91
MQ 8	60	1.78	1.09	1.24	0.9
MQ 8	90	0.81	0.64	0.57	0.74
MQ 8	120	1.21	1.77	1.28	0.94
MQ 8	150	1.27	1.22	1.61	1.53

Table Lxviii. Width to depth ratios of McQueen Branch sites by year.

Width to Depth Ratio					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MQ HW	0	4.23	3.54	3.85	5.12
MQ HW	30	1.99	2.26	2.73	3.43
MQ HW	60	2.71	2.73	2.81	3.7
MQ HW	90	4.87	3.67	5.91	5.11
MQ HW	120	2.76	3.17	3.63	3.84
MQ HW	150	1.86	2.27	2.9	3.24
MQ 8	0	10.53	12.37	12.46	9.63
MQ 8	30	5.81	8.89	6.97	7.71
MQ 8	60	2.33	5.72	6.44	8.11
MQ 8	90	8.6	7.84	9.74	9.35
MQ 8	120	5.77	4.86	6.16	9.83
MQ 8	150	7.03	5.04	4.14	4.58

Table Lxix. Entrenchment ratios of McQueen Branch sites by year.

Entrenchment Ratio					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MQ HW	0	1.31	1.31	1.2	1.33
MQ HW	30	1.97	1.87	1.55	1.48
MQ HW	60	1.94	2.21	1.73	1.58
MQ HW	90	1.39	1.88	1.26	1.22
MQ HW	120	1.66	1.68	1.52	1.37
MQ HW	150	1.72	2.02	1.33	1.56
MQ 8	0	1.48	1.42	1.19	1.57
MQ 8	30	5.81	1.44	1.41	1.66
MQ 8	60	2.58	1.71	1.32	1.44
MQ 8	90	1.75	2.43	2.09	1.71
MQ 8	120	2.18	1.77	1.91	1.63
MQ 8	150	1.54	2.33	2.07	1.97

Table Lxx. Bankfull areas of McQueen Branch sites by year.

Bankfull Area (ft^2)					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MQ HW	0	26.01	31.21	34.2	20.88
MQ HW	30	22.37	18.78	23.62	20.76
MQ HW	60	14.6	11.12	14.72	14.21
MQ HW	90	20.04	9.28	20.18	25
MQ HW	120	24.79	20.92	22.71	27.61
MQ HW	150	20.7	16.01	25.58	16.71
MQ 8	0	8.15	7.57	10.36	7.98
MQ 8	30	7.43	8.06	9.55	6.4
MQ 8	60	7.36	6.79	9.86	6.55
MQ 8	90	5.64	3.21	3.17	5.1
MQ 8	120	8.44	15.2	10.12	8.67
MQ 8	150	11.33	7.49	10.69	10.72

Table Lxxi. Hydraulic radii of McQueen Branch sites by year.

Hydraulic Radius					
Stream	*X Section (m)*	*2010*	*2011*	*2012*	*2013*
MQ HW	0	1.25	1.92	2.08	1.27
MQ HW	30	1.79	1.6	1.84	1.46
MQ HW	60	1.44	0.79	1.44	0.75
MQ HW	90	1.5	1.08	1.54	1.65
MQ HW	120	1.73	1.65	1.73	1.76
MQ HW	150	1.64	1.45	1.88	1.44
MQ 8	0	0.81	0.73	0.85	0.82
MQ 8	30	0.92	0.84	1.04	0.8
MQ 8	60	1.05	0.8	1.08	0.6
MQ 8	90	0.72	0.54	0.51	0.61
MQ 8	120	0.96	1.24	1.07	0.79
MQ 8	150	0.79	0.67	1.15	1.11

Table Lxxii. Sediment sizes found in McQueen Branch sites using a standard sieve set. DB 84 = size at the 84th percentile and DB 50 = size at the 50th percentile.

Sediment Size (mm)			
Stream	*X Section (m)*	*DB 84*	*DB50*
MQ HW	0	0	10.5
MQ HW	30	0	5
MQ HW	60	0	8.5
MQ HW	90	7.61	21.5
MQ HW	120	0	18.5
MQ HW	150	5.45	16
MQ 8	0	3.54	13
MQ 8	30	5.74	10.5
MQ 8	60	0	8.5
MQ 8	90	4.45	11.5
MQ 8	120	6.97	17.5
MQ 8	150	0	11.5

Table Lxxiii. Canopy cover of Upper Three Runs sites showing the percentage of open canopy.

Canopy Cover		
Reach	*X Section*	*% Open*
MQHW	0	7.67
MQHW	30	6.37
MQHW	60	6.89
MQHW	90	7.41
MQHW	120	7.54
MQHW	150	9.36
MQ8	0	4.55
MQ8	30	4.42
MQ8	60	7.93
MQ8	90	15.21
MQ8	120	14.3
MQ8	150	3.38

Table Lxxiii. Streambed penetration across McQueen Branch sites showing connectivity of ground water to the streams.

		Streambed Penetration (cm)			
Reach	*X Section (m)*	*2013*	*2012*	*2011*	*2010*
MQ HW	0	10.3	8.1	10.7	7.3
MQ HW	30	6.4	9.5	6.7	7.5
MQ HW	60	6.4	8.8	10.1	10.0
MQ HW	90	5.7	4.8	6.8	6.5
MQ HW	120	8.1	8.7	14.3	12.7
MQ HW	150	7.2	1.6	7.3	3.4
MQ 8	0	2.7	2.5	3.8	2.0
MQ 8	30	2.2	2.9	2.6	2.0
MQ 8	60	8.1	5.6	3.7	3.8
MQ 8	90	3.1	3.0	4.5	4.0
MQ 8	120	2.9	2.9	4.8	8.8
MQ 8	150	3.5	2.4	4.6	5.3

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