Riparian Habitat Health Evaluation Following Stream Restoration

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ABSTRACT

RIPARIAN HABITAT HEALTH EVALUATION
FOLLOWING STREAM RESTORATION

Michele June Sapundjieff

Stream restoration has been widely used as a crisis response in situations where severe stream erosion occurs. After an impacted system is restored the riparian buffer is also expected to improve, but little is known about the actual effect of stream restoration on downstream riparian habitat condition. The D'Olive watershed in southeastern Alabama is a watershed with severe erosion that discharges into Mobile Bay, AL. The EPA's National Estuary Program is restoring 12 stream reaches within the watershed in an attempt to reduce sediment loading to the bay. This study monitored stream stability and riparian habitat over a two year study period to quantify changes in habitat health following restoration. The study included the development of the Riparian Habitat Health Level Evaluation (RipHLE) specifically for use in riparian forests in urban watersheds. Erosion potentials and RipHLE values generally decreased following restoration activity. Observed changes in vegetation were attributed to seasonal growth patterns rather than restoration. No cumulative effects were observed downstream most likely because the two year monitoring period is not sufficient to capture these changes. These results will be of use to the management agencies for establishing baseline criteria on vegetative response and stream stability following restoration.

INTRODUCTION

Stream Restoration

Ecological restoration refers to the recovery of a degraded, damaged or destroyed ecosystem with human intervention (Clewell & Aronson, 2013). Ecological restoration includes projects of different scales ranging from local tree plantings to large ecosystem restoration projects such as the reversion of the Florida Everglades to its wetland state. Historically, the primary focus has been on plant ecology but the science has expanded into many different system types including streams, meadows and even hilltops (Palmer et al., 2005). Generally, the purposes of restoration projects are to increase ecosystem goods and services while protecting the surrounding habitat (Palmer et al., 2005). Failures in restoration can be used as a test of ecological understanding and may reveal knowledge gaps. These gaps can reveal opportunities for advancement in restoration science since the success of restoration efforts is dependent on the knowledge of the system and its functions.

Restored systems that require active and constant management are considered ecologically engineered. In these systems, the ecosystem is redesigned using engineering principles to reestablish biophysical processes to improve societal and environmental benefits (Palmer, Filoso, & Fanelli, 2014). This is different from ecological restoration which is less invasive and requires more passive management and non-interference. Either way, both ecological engineering and restoration focus on ecosystem resilience defined as the ability to recover or withstand most disturbances (Palmer, Filoso, & Fanelli, 2014).

Specifically, stream restoration refers to the recovery of the present riverine system to a pre-disturbance condition (Berger, 1990). These restorations are generally in response to crisis situations where something, such as the removal of the riparian buffer or downstream headcuts, imbalances the system and can cause accelerated morphological changes such as stream incision, head cutting, and undercutting of the stream banks (Berger, 1990). It is important to note that stream channels are dynamic by nature as their position naturally moves over a decadal time scale. A natural and healthy stream channel should show changes in geomorphology over time as the stream reacts to environmental stressors. It is when those position changes alter morphologic characteristics, such as dimension, pattern or profile, significantly that a channel is no longer considered in equilibrium (Miller & Kochel, 2010). The most effective restorations are multifaceted to achieve multiple goals like reducing streambank erosion, improving water quality, improving floodplain connectivity, increasing diversity, and decreasing storm runoff velocity in the channel (Downs & Kondolf, 2002).

D'Olive Watershed Restoration

The specific work sites in this study are located within the D'Olive Creek watershed in southeastern Alabama. The watershed includes three main tributaries of D'Olive Creek- Joe's Branch, D'Olive Branch, and Tiawassee Creek (Figure 1). These impacted systems are in a highly urbanized setting and erosive potential threatens infrastructure such as interstates and housing developments. Large volumes of sediment, possibly resulting from these eroding banks, are transported through the watershed into Mobile Bay (Collini, 2015). This negatively impacts water quality in both the streams and the bay, by disturbing the sediment. Any disturbance can cause pollutants adhered to the sediment to become mobilized in the water column (Namour et al., 2015). As the sediment moves through the system it is deposited along the streams and into Mobile Bay which can reduce the total area of critical habitats, decrease water quality, increase turbidity causing vegetation lines to recede, and lead to further erosion.

In response to the increased erosion, a collaboration between the Mobile Bay National Estuary Program, representatives from the City of Daphne, Alabama Department of Environmental Management, Geologic Survey of Alabama, Northern Gulf Institute, University of South Alabama, and Dauphin Island Sea Lab, identified twelve stream reaches in need of restoration (Figure 1).

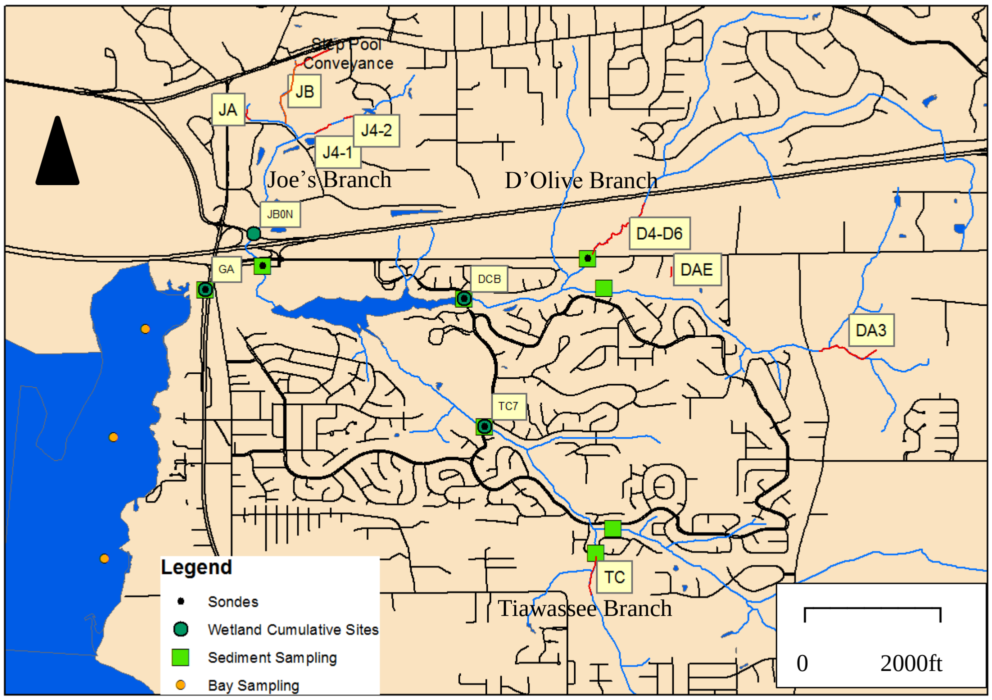


Figure 1: Mobile Bay National Estuary Program restoration and monitoring locations in the D'Olive Creek Watershed. The restoration reaches are noted in red and labeled in the text boxes (Collini, 2015).

The plans include monitoring for sedimentation and flow, water quality, submerged aquatic vegetation, wetland habitat, riparian habitat, and biology as shown in Figure 1. The present research will determine the effects of restoration on downstream riparian habitat of the D'Olive Creek watershed and creates an index for riparian habitat monitoring in the southeastern United States.

LITERATURE REVIEW

Riparian Habitat

A riparian buffer, as defined by the USDA Forest Service, is the aquatic and adjacent terrestrial ecosystem directly affected by the aquatic environment including streams, lakes, bays, floodplains and wetlands (Gillian, 1996). Others define a riparian buffer as a complex assemblage of organisms in an area adjacent to laterally flowing water that rises and falls at least once within a growing period (Gillian, 1996). Regardless, the riparian buffer supports the fluvial ecosystem and integrates many interactions between the aquatic and terrestrial ecosystems (Gonzalez del Tanago & Garcia de Jalon, 2010). To protect these ecosystems best management practices, like riparian forest buffer systems, have been established as water quality controls in forestry and other operations (Gore, Bryant & Crawford, 1994). Best management practices are techniques, measures, or structural controls that help manage the quantity and quality of storm water runoff (Loperfido, Noe, Jarnagin & Hogan, 2014).

Riparian buffers provide many ecosystem services (ES) including water infiltration, aquifer recharge, soil carbon sequestration, flood attenuation, reduction of hydrological risks and serve as nurseries for riverine and oceanic fisheries (Gonzalez del Tanago & Garcia de Jalon, 2010; Meli, Rey Benayas, Balvanera, Martinez Ramos, 2014). The riparian zone can influence biodiversity through its buffering ability to prevent the spread of disturbance related ecological issues such as invasive species, insects and biological diseases (Osbourne & Kovacic, 1993). Riparian zones can influence avian migratory patterns by altering food availability and biogeochemical pathways and rates by diluting, concentrating, modifying, or incorporating pollutants or chemicals as they travel throughout the channel system (Osborne & Kovacic, 1993). The presence of riparian forests impacts geomorphology, concentrations of bioavailable nutrients, and algal biomass independently of urban effects (Walsh, Roy, Feminella, Cottingham, Groffman, & Morgan, 2005).

Riparian buffers support fluvial ecosystems and integrate many interactions between the aquatic and terrestrial ecosystems (Gonzalez del Tanago & Garcia de Jalon, 2010). It is important to protect these riparian buffers because they serve as crucial habitat for aquatic and terrestrial organisms. The buffers can vary in size from a mowed strip of grass between a stream and a housing foundation, to an intact forest surrounding a stream. Increases in stream biota have been linked to vegetation type, decreased erosion hazard, and increased forest cover (Simpson & Norris, 2000; Stewart, Wang, Lyons, Horwatich & Bannerman, 2001). Dominant vegetation in another study was correlated to in-stream nutrient concentrations, physical characteristics of the environment and energy balance (Tanaka, Teixeira de Souza, Moschini, & Kannebley, 2016).

Several studies have linked riparian habitat quality to stream quality (Osborne & Kovacic, 1993). Riparian buffers can alter water chemistry before substances enter a lotic system by adding, removing or amplifying substance concentrations, moderating temperature, reducing sediment input, stabilizing stream banks, and providing organic matter into streams (Osborne & Kovacic, 1993). These buffers can help control nonpoint source pollution (Williams et al., 2013) and the implementation of low environmental impact architectural design principles can improve that control. Low impact design reduces anthropogenic impacts on the environment by reducing connectivity between impervious surfaces and water systems (Walsh et al., 2005).

To effectively utilize riparian buffers it is important to not only understand their function but also their structure. Changes in the physical habitat are impacted by the interaction of sediment supply, sediment transport capacity, and vegetation, which illustrates the importance of riparian zone composition (Segura & Booth, 2010). Riparian buffers have different vegetation zones with increased distance from the water's edge (Naiman, Décamps, & McClain, 2005). These zones are broadly classified into a lower, or inner, floodplain that is frequently flooded each year, and a higher, or outer floodplain, that is flooded less frequently. The vegetation in riparian buffers is influenced by flow magnitude, inundation area, and frequency of inundation events (Winward, 2000). These hydrologic factors can alter the water availability and changes in the hydrology may begin to favor more flood or drought tolerant species over existing species. Tree dominated riparian zones provide several ES including streambank protection, structural diversity, species diversity, stream temperature control, and habitat value (foraging, thermal cover, nesting sites, etc…) (Winward, 2000). Woody species are important in riparian areas to increase substrate cohesion and modification of bed roughness which are erosion controls both in the channel and floodplain (Gonzalez del Tanago & Garcia de Jalon, 2010).

Land use and riparian vegetation condition have also been found to impact fluvial processes such as stream bank erosion and deposition which alter channel morphology (Gurnell, 2014). Riparian vegetation can reduce streambank erosion by using the roots to hold sediment intact and varying land uses can reduce infiltration by reducing soil porosity, lowering surface roughness and rates of evapotranspiration which impact the stream (Gurnell, 2014). Forest buffers with large woody species, while not the only stream bank vegetation, were found more effective in reducing stream bank erosion than grass banks (Zaimes, Schultz, & Isenhart, 2006). This highlights the importance of restoring woody stream bank vegetation and not just grassy vegetation as vegetation is a driver in shaping channel morphology (Segura & Booth, 2010). This relationship between woody species and erosion is also evident in seasonal changes along forested stream banks where most erosion occurs in early spring. This is when the only protection is mechanical reinforcement (Zaimes & Schultz, 2015).

Environmental controls of streamside forests within the riparian buffer have been well studied but more insight into other environmental controls is needed (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015). As described by Pielech, Anioł-Kwiatkowska, & Szczęśniak (2015) there are many factors that impact forest composition such as stream order, water chemistry, flood duration, specific landform, soil texture, and landscape variables (forest continuity, forest cover). Stream order is often associated with water quantity and therefore may have a hydrologic control on streamside forest composition, much like flood duration. Available nutrients and toxins in the stream may be more favorable for species suited to that resource level. Landform characteristics such as slope and aspect may have shading implications which reduce light availability or solar radiation, thereby decreasing stream temperatures. Soil texture is one of many components that impact the establishment of plant communities by affecting the root dissemination though the soil and nutrient retention ability. Soils high in sand will likely have lower available nutrients compared to clayey soils due to differences in cation exchange capacity. Other landscape variables will impact the ability of forests to naturally regenerate.

Proximity and elevation above the stream can impact vegetative growth because lower elevations are associated with periods of greater flooding and can lead to anoxia. Certain plant species (*Athyrium filixfemina*, *Dryopteris spinulosa*, *F. excelsior, Carex sylvatica and Oxalis acetosella*) are more tolerant to these anoxic events (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015). Elevation has consistently been found to influence species richness and composition (Newton et al., 2012). A study in Wisconsin found that flood tolerance caused 29.6% of tree species variation. In the same study, relative elevation also explained variation in tree species and abundance (Turner, Gergel, Dixon, & Miller, 2004). In Veracruz and Oaxaca, Mexico, elevation was significantly positively correlated with species density with r values at 0.85 and

0.62, respectively ($p<0.05$), but there was no significant relationship in a study from Central Chile (Newton et al., 2012). In Veracruz elevation was significantly correlated to species richness ($r=0.83$) whereas Oaxaca and Central Chile showed no significant relationship (Newton et al., 2012). A study using wetland indicator species predictive modelling found that elevation above the channel explained the greatest deviance for herbs (19%) and shrubs (37%) which indicates that terrain elevation drives understory composition (Shoutis, Patten & McGlynn, 2010).

Studies on the response of individual plant species to environmental variables have shown that tree species can have opposite responses along environmental gradients (elevation, altitude, distance to the stream) (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015). As some species increase in biomass as altitudes increase, other species decrease in biomass following the same increase. This illustrates that plant species have different tolerances to environmental factors. This concept is important when considering riparian buffers. Management decisions can be based solely on the width of a riparian buffer, but as the research suggests, other variables should also be considered (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015). The width, distance to the stream, altitude and elevation are all factors that need to be considered when creating and protecting riparian buffers. Urbanization and upstream restoration can alter downstream conditions, by affecting the contributing water source quality and quantity (through impervious surfaces which increase stormflow in streams through overland flow rather than groundwater infiltration) and forest continuity (through fragmentation resulting from urbanization) (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015).

Another aspect of riparian habitat health is indicated by species diversity, which is impacted by both species abundance and species richness (the number of species present in a sample) (McGinley, 2014). Riparian plant diversity has been linked to greater regional biodiversity, wildlife and ecosystem function (Knops, Wedin, & Tilman, 2001; Balvanera et al., 2006; Meli et al., 2014). Although high diversity is not critical to maintain ecosystem processes in stable conditions, it is important for resiliency under changing conditions by enabling species to respond and adapt (Cleland, 2011).

Diversity loss at regional scales can reduce the diversity of colonists in disturbed or degraded systems, which can reduce ecosystem resilience (Knops et al., 2001). This will limit the potential for compositional adjustments in response to changing environmental conditions. The rate of leaf litter decomposition is not consistent across all plant species. The decomposition rate also impacts the diversity of producer macroinvertebrates (Knops et al., 2001), which is important because as diversity is impacted at lower trophic levels it may lead to a variety of responses at higher trophic levels (Loreau, 2001).

The relationship between ES and plant diversity is significant and can be used to quantify floodplain biodiversity. Riparian forests were found to provide seven of eight ecosystem services studied (soil formation, gas regulation, nutrient regulation, habitat provision, food provision and raw materials production, education and recreation riparian forests). The strongest correlations were habitat provision, education, recreation, nutrient regulation and soil formation (Felipe-Lucia & Comín, 2015). In the study, plant diversity was determined through three diversity indexes: species richness, species diversity and total abundance. Most correlations between ES and plant diversity were positive and significant, and many of them (39.58%) were strong ($|0.7| > r > |0.5|$) or very strong ($r > |0.7|$). Provision of habitat ($0.80 > r > 0.50$) was correlated with all three indices allowing plant diversity to serve as a proxy for habitat provision (Felipe-Lucia & Comín, 2015). Plant type, herbaceous versus woody also provides varying levels of habitat and stability. As an example, a conversion of herbaceous to woody stream bank cover reduced the erodibility of streams by as much as 39% in the Blacksburg, Virginia area (Wynn, 2004). Some evidence suggests that the individual plant species is of greater importance than the overall diversity especially in soil processes. Legumes, for example, have a stronger effect on plant biomass than other plant types (Loreau, 2001).

Factors Impacting Riparian Habitat Health.

The forested buffer width, the size of the intact riparian forest surrounding a stream, has been linked to stream quality (Stewart et al., 2001). These riparian buffers have also been shown to maintain species habitat through provison of organic matter and debris to the system and regulation of stream temperatures (Wenger, 1999). Best management practices have been specifically created to protect water bodies by keeping at least some width of riparian buffer intact (Lowrance & Sheridan, 2005). In many places this buffer is protected by legislation since the forest has been shown to reduce runoff entering water bodies therefore reducing nutrient loading and sediment input from surrounding urban or agricultural land (Lowrance & Sheridan, 2005; Wenger, 1999). Legislation in the southeastern United States, where the present study is located, protects an average riparian buffer width of 12.1m for intermittent streams and 19.4 m for large streams (Lee, Smyth & Boutin, 2004). A study in Western Lake Michigan and the Upper Illinois River Basins in eastern Wisconsin showed that buffer widths of 15-30 m can provide sufficient protection depending on local hydrology, soil factors and slope (Stewart et al., 2001). Larger buffer widths (30 m) were correlated with higher fish species diversity whereas smaller forest buffers (0-10 m, 10-20 m) had higher abundances of pollution tolerant fish species and fewer intolerant species (Stewart et al., 2001). This shows that larger buffers maintain the necessary in-stream habitat needed to allow intolerant species to prevail, indicating a healthier system. Percent of forest land within the watershed was also related to increased fish diversity (Stewart et al., 2001). The size of an adequate forested buffer varies with the type of species in question. As an example, larger buffers are needed to protect terrestrial fauna than aquatic and vegetation organisms. Aquatic macroinvertebreates were more strongly correlated with 10-30m buffer widths (Stewart et al., 2001) compared to songbirds that were more correlated with larger sized buffer widths. Specifically, population diversity and density of songbirds increase with forested buffer width in several studies (from 25 m to 800 m in width) (Shirley & Smith, 2005). Nearly ninety percent of all bird species were located within 150-175 m of the streams compared to 90% of plant diversity located within 15 m of the stream (Spackman & Hughes, 1995).

Bank Erosion Hazard Index (BEHI) is a model that measures several geomorphic and erosion indicators that can help define stream health by estimating the risk of bank erosion. BEHI was related to riparian health through studies correlating lower BEHI scores to higher macroinvertebrate abundances and diversity counts (Rosgen, 2001; Simpson, Turner, Brantley & Helms, 2014). Organic matter retention levels were also higher at low BEHI sites, indicating that areas of less risk to erosion provide better aquatic habitat (Simpson et al., 2014). Habitat complexity (i.e., increased diversity) was important in community stability as stable streams (low BEHI implying low risk of erosion) were linked with higher macroinvertebrate abundance (Brown, 2003; Mykra, Heino, Oksanen, & Muotka, 2011; Simpson et al., 2014). Taxon richness in the sediment substrate was significantly negatively correlated with erosion and that taxon diversity was greatest during intermediate rates of sedimentation/ deposition (Miyake & Nakano, 2002).

Higher tree densities and lower percent canopy cover in forest regeneration stands following timber harvest have been correlated with higher macroinvertebrate and fish abundances (Nislow & Lowe, 2006). This suggests that moderate light attenuation through the canopy (moderate meaning an intermediate between full and no light reaching the forest floor) may be most efficient in excellent habitat conditions compared to high canopy cover percentages. Intermediate canopy cover will increase percent ground cover because the canopy will not shade out the understory. It allows for both shade and sun tolerant species to establish within the riparian zone. This provides an increase in food sources available to terrestrial wildlife. Canopy cover also varies with land use. Highly impacted agricultural systems in the coastal plains of Alabama have on average 5% canopy coverage, urban impacted streams have 50% canopy cover and less impacted interior streams generally have 80-90% cover (Shaneyfelt & Metcalf, 2014). This illustrates that higher canopy cover (meaning less light reaching the forest floor) is generally related to lower levels of impact (urban, agricultural, intact) (Shaneyfelt & Metcalf, 2014). In other words, higher canopy cover is ideal. However, there is competing evidence that forests with extremely high canopy cover (>95%) were associated with lower macroinvertebrate abundances (Townsend, Scarsbrook, & Dolédec, 1997). In this same study, areas with intermediate canopy cover, between 70-80%, were associated with high taxon richness which suggests that intermediate basal canopy cover is ideal for both light attenuation and macroinvertebrate abundance.

Tree basal area is a measure of the average area occupied by tree stems. Basal area provides an estimate of biomass per acre and has been shown to have impacts on other ecosystem conditions such as ground cover and understory growth (Sagar & Singh, 2006). Optimal tree basal areas noted by the Mississippi Wildlife, Fisheries and Parks department are around 13.7-16 square meters per hectare to balance wildlife and timber objectives (Elledge & Barlow, 2012). Optimal wildlife habitat basal area is less than 13.7 square meters per hectare whereas forests with basal areas higher than 18 square meters per hectare show negative impacts in overcrowding, resource competition, and disease outbreak (Elledge & Barlow, 2012). Basal area is variable with less direct impact on species richness.

Bank root density, organic matter and presence of nonnative invasive species are all linked in quantifying habitat quality. Higher leaf biomass of nonnative invasive species is characteristic of degraded sites, specifically *Acacia macracantha, Citrus aurantium* L., *Ligustrum lucidum,Gleditsia triacanthos* L., *Morus alba* L., *Pinus taeda,* L., *Pyracantha angustifolia* and *Ricinus communis* L. This directly relates to differing biochemistry between native and exotic species which can introduce changes in habitat quality by changing nutrient input resulting from decomposition. These exotic species leaves have higher cellulose, lower nitrogen and lower chemical inhibitors which negatively impacted microbial decomposition of the organic matter (Mesa, Reynaga, del Correa, & Sirombra, 2013). This significantly higher leaf biomass therefore is related to poor condition riparian habitats. Bank root density is also impacted by exotic species composition. Some authors suggest using the diversity of native species rather than presence of nonnative invasive species at a pristine location to determine habitat condition (Casatti, Ferreira, & Langeani, 2009), due to the anthropogenic influence on exotic species encroachment. In other words, non-native species can be spread naturally using phylogenic adaptation for increased reproductive efficiency (seed shape, dispersal, tubers, etc) but they can also be spread through seeds on tractor tires, accidental species introductions and other human interference. Therefore, in urbanized areas where the spread of non-native species is also likely contributed to humans it is not suggested to use non-natives to study habitat intactness because is in a non-naturalized manner of invasiveness.

Structural complexity (vertical structure) is another factor impacting riparian health via biodiversity. Terrestrial wildlife abundance, as an example, increases with light attenuation to the forest floor. This increases photosynthetic activity which increases the amount of leaf litter in the floodplain, which when decomposes reintroduces nutrients back into the forest soils, improving growing conditions (Townsend, de Lange, Duffy, Miskelly, Molloy, & Norton, 2008).

Stream Restoration

As previously noted, streams are dynamic systems that under natural conditions will continually alter their channels. In response to stress, streams have natural resilience which allows a channel in disequilibrium to re-establish that equilibrium in five stages: I) stable, II) incising (degradation), III) widening, IV) aggrading and V) quasi-equilibrium (Zaimes & Schultz, 2015). A force that will shift a stream out of equilibrium (stable state) causes a stream incision, where the stream begins to deepen so that the bank height ratio is greater than 1.0. After streams incise the forces of the water can erode the stream bank causing the stream to widen which can reduce the velocity of the flowing water and introduce excess sediment into the riverine system which is generally following by stream bank/ channel aggradation (or deposition of sediment to increase the stream gradient). Following the deposition of the eroded sediment the stream will reach a quasi-equilibrium or steady state once more until another force (such as a large rainfall or runoff event) shifts the system out of equilibrium. Thus, in response to changing environmental conditions, stream channels can migrate or shift. This has implications in channel restoration since restoration plans need to compensate for the dynamic nature of streams.

As an example, after channel reconstruction 60% of sites resulted in at least a 20% change in channel capacity (the ability of a stream channel to transport its water and sediment inputs without changing its dimensions) (Miller & Kochel, 2010). This is important because large changes in the reconstructed channel were associated with high transport capacity, increased sediment supply, and easily eroded bank materials, which were not anticipated in the restoration designs. Other studies show that over time restored reaches will begin to degrade and continually erode, supporting the claim that changes in channel stability occur over large time scales. At one particular restored streamside meadow, the channel was in need of a second restoration nine years after the initial. Seven years after the completion of the second restoration, indicators of channel instability, namely channel incision, were present (Pope, Lisle, Montoya, Brownlee, & Dierks, 2015). Another study showed that restoration can improve channel heterogeneity by decreasing the amount of fine-particles directly downstream from restoration structures by 25%. This study also showed that sediment distribution was significantly different at all study sites after channel reconstruction (Collins Flotemersch, Swecker, & Jones, 2015).

Causes for Stream Restoration.

Healthy riparian forests are important because of their high levels of biodiversity, flood mitigation applications, and stream bank erosion prevention (Segura & Booth, 2010). Urbanizing landscapes which cause a loss of floodplain and riparian buffer continue to be a main cause of stream restoration. This is primarily because prime candidates for stream restoration are those of economic or intrinsic value to the people in those urban landscapes. As urbanization continues to fragment the landscape, riparian forests serve as both ecological and genetic corridors between green spaces (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015). These crucial corridors are susceptible to many disturbances that may create a need for restoration of the stream and riparian zone. A disturbed stream where channel erosion threatens urban infrastructure is a primary candidate for restoration with economic and ecologic implications.

Urbanization leads to an increase in impervious surfaces which increases storm water runoff directly to streams. This exacerbates peak flows, with smaller lag times, higher pollutant loads, potential for channel erosion and decreased water quality (Booth & Jackson, 1997; Walsh et al., 2005; Palmer, Filoso, & Fanelli, 2014). Studies show that bank-height ratio, which is a ratio to describe channel incision, mean channel grain size, and cross-sectional area are greater for urban streams than rural streams indicating sedimentation in downstream reaches and upstream channel instabilities. Urbanized channels showed 3.4 times the maximum capacity of water within streambanks than rural streams (O'Driscoll, Soban, & Lecce, 2009).

Other effects often associated with urbanization include reduced base flow and increased suspended solids (Walsh et al., 2005). The flow regime within a river itself can also change drastically with increased urbanization. Overland flow increases as streams are urbanized which increases the total volume of water within a stream leading to smaller lag times before peak flow. The lag time refers to the time lapse between initial rainfall and peal flow within the channel (a decreased lag time indicates less water is being absorbed or diverted before reaching the channel which may cause the system to become flashier meaning quicker to reach peak flow). This can increase the erosive potential of the waters leading to increased channel widths and scour (Walsh et al., 2005). These issues at the local or regional level can amplify downstream as fine sediment is transported which can result in a flux of nitrogen in coastal waters (Palmer, Filoso, & Fanelli, 2014).

Urbanization can also lead to simplified channel morphologies with uniform beds and fewer, deeper pools (Walsh et al., 2005; Segura & Booth, 2010). Urban streams can become incised, or disconnected from the floodplain, causing lateral constraint which modifies natural floodplain development and geomorphological processes (Schwartz, Neff, Dworak, & Woockman, 2015).

Along with changes in geomorphology, urbanization may result in tree removal in riparian forests. Canopy loss in urban stream eliminates overhead shade as a temperature control and limits leaf litter into the system (Booth & Jackson, 1997). Leaf litter becomes part of the aquatic food chain and thus its removal can negatively impact biodiversity. Overall, urbanization can reduce biotic richness and increase dominance of pollution tolerant species (Paul & Meyer, 2001; Meyer, Paul, & Taulbee, 2005).

In areas where grazing is common along stream banks, livestock can destroy bank cover, which provides natural erosion protection, and remove natural riparian vegetation. However, the effects of grazing on streambanks are more commonly studied in native grassland ecosystems rather than forested buffers. A study by the University of Iowa found that even though there was decreased vegetative cover where livestock had unlimited access to streams, either continually stocked or rotationally stocked, there was no net change in erosion compared to riparian buffer with grazing exclusion (Haan, Russell, Kovar, Nellesen, Morrical, & Strohbehn, 2007). On the contrary, parcels in central Iowa with different land cover showed changes in net erosion from 1998-2002: forest buffer (75 tonnes/km), row crops (484 tonnes/km), and pastures (grazing) (557 tonnes/km) (Zaimes et al., 2006). In this case, grazing did directly impact the net erosion on stream banks. This is further supported by studies in Alberta, Canada which show that the removal of cattle significantly increased bank stability and riparian vegetation biomass (Scrimgeour & Kendall, 2003). Despite the discussion regarding the extent of impact that grazing has on streambank erosion; authors agree that it is a contributing factor to decreasing riparian vegetation. The reduced vegetation cover can result in issues such as reduced stream shade, increasing water temperatures, increased turbidity, and alternating depositional and erosional patterns strongly influencing channel morphology (Hough-Snee, Roper, Wheaton, Budy, & Lokteff, 2013).

Stream degradation results in increased sediment yields, which as a non-point source pollutant, causes degradation of the physical and biological stream function (Rosgen, 2001). As a result, the loss of biodiversity in running water systems currently exceeds that of terrestrial and marine systems indicating a need for improved restoration science (Palmer, Filoso, & Fanelli, 2014).

Stream Restoration Practices.

Effective ecological restoration may include several different practices that are often confused with the term restoration. These practices include rehabilitation, reclamation, revegetation and remediation. Rehabilitation refers to the reparation of ecosystem processes, productivity, and services rendered without regard to achieving the fullest possible reestablishment of preexisting biota in terms of its species composition and community. Reclamation involves the conversion of land from an economically worthless condition to a productive condition (agriculture, aquaculture, or silviculture). Replanting degraded or reclaimed land is termed revegetation or reforestation depending on the nature of the plant species. Remediation refers to pollutant removal or reduction (Clewell & Aronson, 2013). These methods are used during restoration projects but each method alone does not constitute a restored ecosystem.

Many different ecosystems, at different levels of degradation- ranging from conversion of grassland habitats to superfund site contamination, may be in need of restoration. Each restoration is regional or site specific. There are many approaches to stream restoration. Rosgen

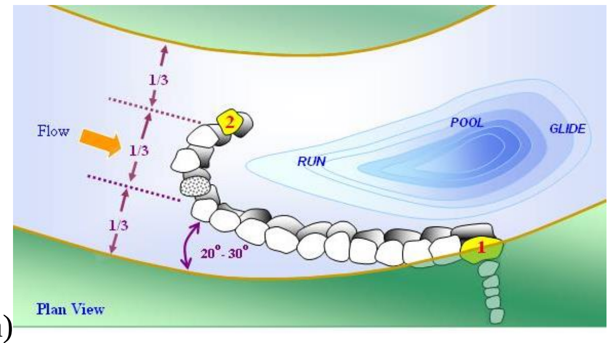
(2007) introduces the idea of natural channel design which uses local reference reaches, along with hydraulic relationships and sediment transport models as a restoration goal. Natural channel design groups similar streams with similar characteristics (morphology, sedimentology, hydrology, and biology) together as opposed to treating the system the same as those in different regions (Rosgen, 2007).

Rosgen’s (2007) natural channel design method is not without its critics. The method follows a form based approach so that the design of the reconstructed channel, or its form, creates function, or provides ecological and biological benefits, without analytical assessment of existing conditions (Simon et al., 2007). There is much debate within the literature over the academic validity of the basic assumption within the natural channel design method due to regional variability in geomorphic process-response regimes (Juracek & Fitzpatrick, 2003) and identifying bankfull stage across unstable stream banks (Hey, 2006; Simon et al, 2007). Other critiques of the method apply to its lack of analysis of existing geomorphic conditions, such as bedload transport (Hey, 2006). There is also concern that if the stream being restored is not accurately described by the reference reach conditions (undergoing different rates of geomorphic processes), then it will continue to be unstable after the attempted restoration (Juracek & Fitzpatrick, 2003). Other discussions of limitations to the natural channel design method are prevalent in the literature, which further critique subjectivity in analysis (Roper, Buffington, Archer, Moyer, & Ward, 2008) and the concept of channel classification without extensive geomorphic and process based analysis of components like bedload, sheer stress, gradient, and other factors (Gillian, 1996; Savery, Belt & Higgins, 2001). Despite its criticism, natural channel design is the most widely used restoration approach for many government agencies (Simon et al., 2007).

The techniques employed in stream restoration such as natural channel design are intended to reduce erosive forces upon the stream banks. Some examples of design choices include 1) cross vane structures, which are designed to reduce near-bank stream velocity and simultaneously increase main channel energy, and 2) j-hooks (Figure 2a), typically placed on the outside of stream bends designed to reduce bank erosion and decrease near bank velocity. Wing deflectors (Figure 2b) are designed to decrease the width of over-widened streams (Rosgen, 2001). These can be created using natural materials such as woody materials or aggregate material foreign to coastal plain systems such as boulders (Figure 3). Other methods of restoring resilience use hard stabilization techniques such as textiles, rip rap, and stream channelization, which are less ecologically friendly and often constrain the channel (Palmer et al., 2005). These stabilization techniques are employed when erosion might be a threat to infrastructure or when accelerated erosion in present.

Channel reconfiguration, such as Rosgen's (2007) method, is often a response to infrastructure threat and provides immediate risk reduction. Studies suggest that restoration involving natural processes like plantings is more likely to succeed over the long term. Drawbacks to enhanced natural recovery however include longer intervals before risk reduction is achieved (Miller & Kochel, 2010).

Hard stabilization techniques used within the stream channel, like rip rap and textiles, can hinder fish and other organism migration, become safety hazards, impact recreational activities such as canoeing, and can be composed of non-natural materials (Miller & Kochel, 2010). Other river restoration practices such as localized bed and bank treatments, habitat improvement devices, and stream channel reconfiguration may be used to meet restoration goals (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015).

a) 

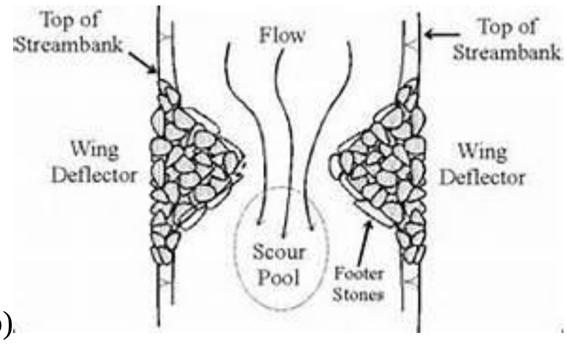
b) 

Figure 2: Diagrams showing two different hard stabilization geomorphic structures created using stones installed in restored streams: a) j-hook (left)[i] and b) wing deflector (right).

Some hard stabilization techniques involve placing woody material, which is natural habitat in streams, using machinery. Suggested changes in forest management to allow riparian forests to mature would create a renewable source of instream woody material that would not prevent natural channel migration (Palmer et al., 2005). Current criticisms suggest that wetland restoration methods do not allow for the full recovery of biogeochemical function and biotic structure (Meli et al., 2014). This might be attributed to an improper application of restoration techniques like the inclusion of large woody material in areas where the woody material needs are higher or lower than the referenced studies. As an example, the impact of large woody material is well studied on salmon habitat in the Northwestern United States but research on large woody material in the southeastern United States is not as thoroughly documented (Opperman, Merenlender, & Lewis, 2006).

a) b)

Figure 3: Examples of hard stabilization techniques used in the study area at a) Joe's Branch step pool conveyance (completed 2013) and b) JB1R (completed 2015).

Vegetation Restoration.

One of the difficulties in wetland restoration is the reestablishment of woody species (Clewell & Aronson, 2013). Even under natural conditions stable plant communities can be short lived. Long term self-perpetuating communities in one study were only found in areas with stable enough environments for the community types to reach equilibrium (Winward, 2000). Therefore, geographic location is vital to the establishment of stream side vegetation. In the southeastern United States woody species are more common than graminoid species which can spread by seed or by vegetative expansion. The high root mass in woody species reduce erosive potential by increasing the flow resistance on the banks, and provide food source habitat (Zaimes & Schultz, 2015). Woody species are more difficult to restore using passive approaches such as natural regeneration (Hough-Snee et al., 2013) and are insufficient as the only restoration activity (Walsh et al., 2005). Riparian woody expansion is much slower than graminoid expansion which decreases the zone's ability to quickly reach restoration objectives without active management. Since it is necessary to restore vegetation as a method of bank protection, monitoring and management after the restoration may be required.

Issues with Restoration.

Although restoration may seem beneficial, oftentimes the biodiversity and ES do not reach predegradation levels. A meta-analysis of restoration and biodiversity studies showed that non-recovered ES, soil amendment and revegetation can lead to a decrease in biodiversity of 24% as compared to natural wetlands (Meli et al., 2014). Direct impacts can also reduce habitat provision thus affecting species regeneration and reestablishment. As an example, stream channel modifications can result in homogenous instream habitat patterns reducing habitat availability for many species (Eekhout, Hoitink, de Brouwer, & Verdonschot, 2015). Even though ES may not reach pre-degradation levels, there is a possibility for ES recovery. A meta-analysis of studies on ES and biodiversity throughout the world showed that ecological restoration overall increased biodiversity by 44% and ES by 25% compared to pre-restoration levels (Newton et al., 2012). These numbers are not higher than pre-degradation levels but are an improvement to the biodiversity within system compared to a non-restored system.

Ecosystem recovery is dependent on how much biodiversity is present, whether ecosystems service levels can be recovered through restoration, if ES and biodiversity correlate, and whether the recovery is dependent on the specific context (ecosystem type, main agent of degradation, restoration action and restoration age) (Meli et al., 2014). In order to maintain and regulate the ecosystem services that do rebound after restoration, active management could halt biodiversity losses. Even more beneficial in ecosystem recovery, land use planning concurrent with restoration goals is crucial (Felipe-Lucia & Comín, 2015).

Specific to a stream restoration scenario, changes in stress or disturbance affect the ability of a system to rebound. Stress can result from soil compaction from equipment access roads, altered streamflow from diversion, or altered resource availability from cleared vegetation. While there are not many studies regarding the disturbance directly resulting from restoration, a study on dam related disturbances showed increased vegetative colonization of resource limited species in low flow areas and channel narrowing during managed flows (Shafroth, Stromberg, & Patten, 2002). The vegetative responses in the first few years were transient, which led to a new equilibrium state of dominant tree canopies.

This initial transient response is also reported by Richardson et al. (2007) who explain that riparian vegetation intactness is negatively altered by invasive species establishment, though vegetation can return to a more intact state following a transient decline (Appendix 1). In other words, much like a stream's geomorphic responding to erosion or incision, a decline in vegetation intactness can be overcome after a short period where the system responds and reaches the same equilibrium. However, should those effects surpass a threshold then cumulative invasive species effects can send the system to a new equilibrium state. This will prevent the system from reaching the original level of intactness, where the invasive species outcompete natural riparian vegetation that is already established (Richardson et al., 2007). When invasive species effects are coupled with anthropogenic disturbance such as road or bridge construction, the riparian vegetation intactness can plummet. In these cases, the invasive species are not competing with natural riparian vegetation, since that vegetation is not present, and have the ability to overtake the system (Richardson et al., 2007). This can be problematic in channel reconstruction restoration if invasive species are not controlled.

Disturbance, such as the re-meandering of streams, can also impact habitat diversity in a positive manner (Barral, Rey Benayas, Meli, & Maceira, 2015; Eekhout, 2015; Meli et al., 2014). Several meta-analyses found that restoration significantly increased vertebrate species in riparian wetlands, (+53%, Meli et al., 2014) and in cropland and pasture sites, (+54%, Barral et al., 2015). Land use affected diversity following restoration because wetland terrestrial invertebrates only increased by 17% compared to pre-restoration levels whereas cropland invertebrate species increased by 79% (Barral et al., 2015). Wetland macroinvertebrates were not significantly affected (Meli et al., 2014). Soil microfauna and vascular plants increased between 54-79% in cropland and increased by 15-45% in wetland areas (Barral et al., 2015; Meli et al., 2014). Conversely, the diversity of non-native vascular plants were 44% lower in restored wetlands than natural wetlands, suggesting that restored wetlands had less invasive species diversity, although changes in ES function could also contribute to changes in biodiversity. The invertebrate diversity was 37% greater in restored wetlands (Meli et al., 2014). Meli et al. (2014) also showed that ES were positively correlated with biodiversity in multiple ecosystem types, supporting the idea that biodiversity is a driver for ecosystem services.

Stakeholders.

Restoration projects generally involve collaborations among various stakeholders including professional restoration organizations, funding sources, management companies, contracted labor, designers/engineers, specialists/consultants and landowners. Companies may be legally obligated to mitigate land in order to build in specific areas. Decision makers, government agencies like the United States Environmental Protection Agency, the United States Fish and Wildlife Service, state agencies and local municipalities, may also be invested in restoring degraded lands to protect public infrastructure. In the present study, the Mobile Bay

National Estuary Program teamed up with the cities of Spanish Fort and Daphne, the US Fish and Wildlife Service, the Alabama Department of Environmental Management, and Thompson's engineering to design and implement the several restoration projects to protect federally listed species and improve critical habitats and connectivity.

Stream Restoration Monitoring

Importance of Monitoring.

Once a stream has been restored it is important to determine the effectiveness of the project on both economic and ecological scales to determine if the project protected infrastructure, accomplished its purpose, increased recreation, advanced restoration science, or reached the goals identified during project development (Palmer et al., 2005). To answer these questions, monitoring programs are put in place. The most effective restoration project is one that accomplishes stakeholder goals (aesthetics, economic benefits, recreation, and education), ecological goals (improvement, self-sustainable, complete assessment) and learning goals (scientific contribution, improved methods) (Palmer et al., 2005).

In creating these monitoring regimes, it is important to ensure that monitoring length is sufficient to answer the driving questions and that there is room for feedback to account for unanticipated changes (Downs & Kondolf, 2002). In a study observing changes in cross-sectional channel diameter in a disequilibrium system, the time required to reach equilibrium, exceeded the three to six year monitoring period that was in place (Miller & Kochel, 2010). This is one example of an unsuccessful monitoring regime as the monitoring period was not suitable to the restoration goals.

Since monitoring is so specific to the restoration goals, there are no well accepted criteria for ecological success that drive monitoring programs. This has hampered the progress of restoration science. There is little incentive for practitioners to assess and report outcomes without industry standards and agency funding which may affect the implementation of monitoring programs (Palmer et al., 2005).

Specific difficulties in monitoring riparian zones include the many land management activities that impact and influence the resources in a specific area (Winward, 2000). In other words, changes that may be noted during a monitoring program might be caused by other activities within the riparian zone such as a change from forest land to grazing animals.

Stream Monitoring Methods.

Stability monitoring.

Although streams are dynamic by nature, stream stability is commonly monitored to assess the capability of a channel to accommodate or resist change from inputs of sediment, water, organic matter, or alterations of the riparian vegetation. These are monitored through indicators like channel pattern, bank conditions, gravel bars, and riffle-pool dispersal (Segura & Booth, 2010). Sediment is primarily moved in high flow conditions, where the waters velocity is greatest as illustrated in more than 100 studied streams in Virginia and Maryland (Hack, 1957). Studies on stream bank stability generally monitor bank retreat, grain size, deposition patterns and sediment transport to determine how these interact with changes over time (Daly, Miller, & Fox, 2015; Collins et al., 2015; Levell & Chang, 2008).

Bank Erosion Hazard Index (BEHI).

The BEHI model is used to estimate risk of bank erosion by examining geomorphic and erosion indicators (Appendix II). This method measures percent surface cover, root depth, stream bank soil, percent root density, and bank angle. The BEHI method considers these physical streambank conditions and scores them with values from very low to extreme erosion hazard (Rosgen, 2001). These values are then graphed along an index rating curve created by Rosgen (2001) to determine their indexed values, since the relationships are non-linear. These indexed values are combined to obtain a final BEHI rating which corresponds with a categorical risk ranging from very low to extreme erosion hazard. The root depth and root density are both determined through visual estimates. As it is improbable to accurately guess the percent density below the stream bank surface, this method is limited in accuracy. Different researchers could produce vastly different BEHI results (Roper et al., 2008).

Riffle Cross Section.

Riffle and pool instream characteristics both provide unique habitat for fish and benthic macroinvertebrates that have adapted to the specific environment (Keck et al., 2014). Therefore, the loss of riffle-pool sequences degrades habitat quality and function. Because riffles provide habitat and food via leaf pack for primary consumers they are essential in maintaining life within the stream (Schwartz et al., 2015).

Monitoring of riffle geomorphology using riffle cross sectional data allows for modeling of changes in elevation. These changes should be compared to verify whether the channel is in a state of equilibrium or aggradation/degradation (Zaimes & Schultz, 2015). The Alabama Department of Environmental Management has used riffle cross-sections, longitudinal profiles, bed material and stream classification to characterize stream geomorphology (Shaneyfelt & Metcalf, 2014).

Longitudinal Profile.

Channel geomorphology is impacted by both sediment supply and sediment transport capacity (Segura & Booth, 2010). If the restoration serves its purpose in reducing streambank erosion then the reduced sediment input into the system would be reflected in changing downstream geomorphology through decreases in new instream sandbar deposition and reduced fine sediment load through the system (Hack, 1957). To quantify this change, a longitudinal profile is can be used to measure instream topography (Hack, 1957) which can be compared to riffle cross sections to determine changes in stream bed and water surface elevation.

Near Bank Stress.

Near bank stress (NBS) uses disproportionate energy measurements as an estimate of streambank erosion potential (Appendix III). Changes in the disproportionate energy can accelerate erosion. According to Rosgen (2001) NBS can be determined using seven different methods which vary based on the level of monitoring completed. These seven assessments are: channel pattern, transverse bar or split channel/ central bar influences, ratio of radius of curvature to bankfull width, ratio of pool slope to average water surface slope, ratio of pool slope to riffle slope, ratio of near-bank maximum depth to bankfull mean depth, ratio of near-bank shear stress to bankfull shear stress, and velocity profiles (Rosgen, 2001; Sass & Keane, 2012).

Near bank maximum depth to mean depth will be used to determine NBS along the riffle cross section in the present study. This method was chosen because it uses quantitative in place of qualitative data to predict bank stress. The maximum riffle depth for a reconstructed channel is the product of the ratio of max depth to the mean depth for the reference reach by the calculated mean riffle depth for the restoration site (Hey, 2006).

***Pfankuch Stability, modified for sand bed stream*.**

Several methods can be used to assess stream stability including digital rock marking using repeat photography, hydrological regime indices, shields number and pfankuch qualitative index. In a study comparing these four methods, the authors suggest using the Pfankuch Stability Index as it relates disturbance to benthic organism's habitat (Peckarsky et al., 2014). The Pfankuch Stability Index (Appendix IV) has also been correlated with erosion in other studies (Schnackenberg & MacDonald, 1998; Harmel, Haan, & Dutnell, 1999; Magner, Vondracek, & Brooks, 2008; Schwendel, Death, Fuller, & Joy, 2011).

The Pfankuch stability index is a multi-metric index adjusted for stream type (Rosgen, 2007). The index categorizes stream stability as excellent, good, fair or poor using visual and quantitative measures including width to depth ratios and evidence of mass wasting events (Pfankuch, 1975).

Habitat Monitoring Methods.

Habitat monitoring.

Habitat monitoring is vital to diagnosing and repairing the riparian buffer and can be used to design and implement restoration activity in response to human activities (Gonzalez del Tanago & Garcia de Jalon, 2010). Since riparian zones have been linked to ecological function of rivers, the structure of the zone, the river, and the hydrological regime represent the main elements supporting the biological communities (Gonzalez del Tanago & Garcia de Jalon, 2010). This suggests that riparian monitoring can be used as a proxy for habitat condition.

Although there are several existing indices to measure riparian habitat quality, these indices need to be calibrated to the specific region being studied and no index has yet been calibrated to Southern Alabama, where the present study will be conducted. Mobile Bay Natural Estuary Program is currently calibrating an Integrated Biological Index but was not completed by the end of this study. For this reason, this study will develop a riparian habitat assessment index to assess habitat quality in place of using the existing methods reviewed below.

Riparian Quality Index.

This method is a standardized multi-metric index that collects quantitative information on the provision of habitat within the riparian zone. It includes river dynamics, natural vegetation, flow regime, land use and channel management (Gonzalez del Tanago & Garcia de Jalon, 2010). Specifically, the riparian quality index evaluates: (i) dimensions of land with riparian vegetation (average width of riparian corridor); (ii) longitudinal continuity, coverage and distribution pattern of riparian corridor (woody vegetation); (iii) composition and structure of riparian vegetation;(iv) age diversity and natural regeneration of woody species; (v) bank conditions; (vi) floods and lateral connectivity; and (vii) substratum and vertical connectivity to provide a score between 10 and 120 (Felipe-Lucia & Comín, 2015).

Other methods, like the index of biological integrity, focus primarily on vegetation structure, land use, macroinvertebrates, and habitat quality (Munné & Prat, 1998; Winward, 2000; Munné, Prat, Solà, Bonada, & Rieradevall, 2003). Simpson et al. (2014) found that a decrease in habitat quality can alter the fish and macroinvertebrate assemblages in streams suggesting that these species could be used as a proxy for habitat quality whereas Meli et al.

(2014) showed that diversity of macroinvertebrates was not significantly impacted by restoration. This difference in these studies can be attributed to spatial scales. Simpson et al. (2014) used a single location compared to Meli et al. (2014) which was a meta-analysis of multiple studies in varying geographic locations. Simpson and Norris (2000) linked geomorphological features with biota to determine the ability of the aquatic habitat to support optimal biological conditions.

Index of Biological Integrity.

Multi-metric indices like the index of biological integrity measure end response variables of biological degradation and synthesize the cumulative effects of environmental impacts (Morley & Karr, 2002). The index of biological integrity utilizes well tested attributes of stream biota, namely fish, invertebrates and algae to produce a single number. Higher values indicate healthier systems. Typically information on pollution tolerant taxa, taxa composition and population attributes are included in the index although that is variable depending on the number of metrics within the index (Karr, 1996). There is a calibrated benthic index of biological integrity in the Pacific Northwest but no existing calibration for the Southeastern United States although other macroinvertebrate based assessments such as the Florida DEP bio assessment (Fore et al., 2007) and te (Karr & Chu, 1998; Morely & Karr, 2002).

Vegetation Sampling.

There are several methods for quantifying vegetation whether by density, basal area, stems per acre or other volume measurement which can be accomplished through transect lines, vegetative plots or a combination thereof (Reinecke, Brown, Esler, King, Kleynhans, & Kidd, 2015). The method is dependent on the type of vegetation present whether it’s a zone of graminoids or a higher zone of woody shrub and tree species.

Combined methods use transect lines with evenly spaced contiguous plots sampled along both sides of the transect line midpoint (Reinecke et al., 2015). Transect methods can be randomized using the line intercept method, also known as the line-point intercept method, which collects data at set intervals along the transect line, or by measuring every species along the transect line. This is used for obtaining community type cover and composition (Winward, 2000). The grid-point intercept method uses the intersection of parallel gridlines as the sampling location. The point quarter method, on the other hand, assumes that vegetation follows a random spatial pattern and it only measure the plants closest to predetermined points in each 90 degree quadrat surrounding the point (Pilliod & Arkle, 2013).

To quantify stream bank vegetation the measurements must occur above the greenline which typically is located near bankfull stage (Winward, 2000). The greenline, or the elevation at which vegetation becomes established, may be several feet above bankfull stage in eroding or entrenched streams. The greenline indicates the height along the bank where it is typically above the waterline. When determining bank cover using a line intercept method, it must be used in reference to the greenline in place of a fluctuating water height to ensure comparable data (Winward, 2000).

Other streambank vegetation sampling methods include quadrat methods where a set plot is placed and data is collected along the greenline. Examples might include a 50cm by 20cm plot starting at the greenline where every species within the grid is measured or identified. The plot size would be variable depending on the entire area being sampled (Hough-Snee et al., 2013).

Stream Identification.

While vegetation is an easily observable trait to identify habitat health, other component have be associated with diversity and species richness in riparian areas. Stream type and flow regime, as an example, have been related to total species richness (r=0.76) (Morley & Karr, 2002). The flow regime can be specifically measured during rainfall events using gauge data for accurate temporal changes in peak flow and flashiness, but in more general terms can also be classified into ephemeral, intermittent and perennial streams. The North Carolina Division of Water Quality (2010) published a worksheet that uses visual surveys and macroinvertebrate sampling to determine the stream type (Appendix V). This is the accepted method as identified by the Alabama Department of Environmental Management which uses this classification system in their published documents (Shaneyfelt & Metcalf, 2014).

RESEARCH GAP

Stream restoration can greatly improve a riparian ecosystem but may not accomplish all restoration goals. Restored ecosystems, while improving upon non-restored densities of vascular plants, still result in lower than natural densities and lesser ecosystem function. This demonstrates a need for improved research into species composition, community structure and functional ecology to improve restoration practices (Meli et al., 2014) which is needed at the landscape scale (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015). Since restoration success is driven by available ecosystem knowledge, the restoration of an ecosystem to its predegradation condition is nearly impossible based on current knowledge of pre-degradation conditions. While reference systems can provide invaluable information, the knowledge on what biodiversity levels were to inform target restoration goals is often lacking. A better understanding of the interaction between physical features of the environment and vegetative controls would enhance restoration science, potentially leading to improved ecological success (Pielech, Anioł-Kwiatkowska, & Szczęśniak, 2015). Regionally specific vegetative control studies would also lead to improved restoration science as riverine forests are less studied than montane and grassy bank streams (Peckarsky et al., 2014).

The question arises if there are unrealized consequences of not sufficiently restoring a stream to support the full suite of species in riparian buffers (Palmer, Filoso, & Fanelli, 2014). Additionally, as many influences at the landscape scale become cumulative downstream (Winward, 2000), how do those influences impact vegetation composition at considerable distances from the restoration site? It is not evident in the peer reviewed literature that vegetative responses have been documented beyond the immediate area of the active restoration site. In response, the present study will identify if restoration activities impact downstream vegetative and stability responses at various distances from the restoration activity and quantify those responses.

In the D’Olive Creek watershed, potential infrastructure damage to roads, interstates, and housing foundations was a large factor in identifying at-risk reaches for restoration. Therefore, monitoring for stream stability is essential in meeting restoration objectives. As the influence of the stream stabilization and upstream restoration has not been identified on riparian vegetation, the present study will quantify the resulting changes in habitat condition using a novel index developed for forested riparian systems in southern Alabama.

RESEARCH OBJECTIVES

The goal of the present study is to quantify the changes in habitat condition and geomorphology that occur upstream, within and downstream from several restored reaches in the D'Olive Creek watershed in southern Alabama. The study areas were monitored every six months for two years to quantify changes. The study addresses the following objectives:

- Develop an index using biological indicators to evaluate riparian habitat condition (Riparian Habitat Health Level Evaluation (RipHLE) Index).
- Calculate RipHLE values for study sites.
- Identify localized stream restoration impacts upon the downstream riparian habitat condition.
- Determine if multiple upstream restoration sites affect the RipHLE values at the lower reaches of the stream in any predictable manner (cumulative impacts).
- Identify any relationship between habitat condition and stream stability metrics.

STUDY AREA

Geologic Description

The study sites are located near Daphne and Spanish Fort in southern Alabama, and are located within the East Gulf Coastal Plain (Figure 4). The soils in southern Alabama are mostly Ultisols which often support productive forests with low native fertility. The soils are characterized by a subsurface horizon of accumulated clays and are strongly leached and acidic. Southern Alabama has mostly udults, found in humid climates with well distributed rainfall on surfaces that range from Pleistocene to Pliocene in age (McDaniel, 1999). While the subsurface layers may be characterized by loamy or clayey subsoils, the surface layers of Smithdale, Luverne, Savannah, Dothan and Orangeburg soils are sandy loam, loam or loamy sand. The elevation of the southern Alabama region ranges from sea level to 152 m (Mitchell, 2008).

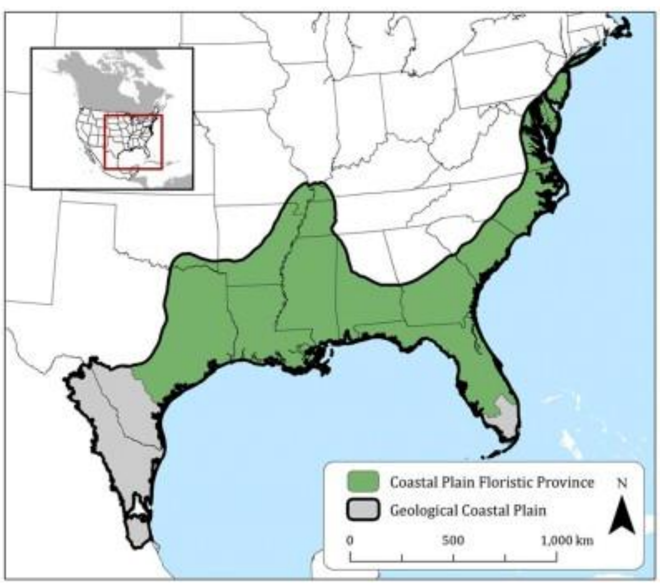


Figure 4: Map of the Gulf of Mexico Coastal Plain.

Dominant land use in the study area includes urban, mixed forest, evergreen forest, and agriculture (Figure 5). In the forests, dominant vegetation includes yellow poplar (*Liriodendron tulipifera*), black cherry (*Prunus serotina),* black birch *(Betula lenta),* eastern hemlock *(Tsuga canadensis),* white pine *(Pinus strobus),* maple species *(Acer sp.)* and oak species *(Quercus sp.),* among others (Hedman & Van Lear, 1995).

Southern Alabama is characterized by numerous habitat areas and drainages including: (1) the Mobile-Tensaw River Delta, (2) Mobile Bay, (3) the Escatawpa River, (4) the Perdido River and (5) barrier islands. The present study will focus on the D'Olive Creek Watershed that drains into Mobile Bay. Mobile Bay is the fourth largest estuary in the nation encompassing 1070 km^2, 50 km in length and 39 km maximum width (Shaneyfelt & Metcalf, 2014). Alabama coastal lowlands consist of coastal streams, wetlands, delta, lagoons, islands and bays. A saline and/or fresh high water table creates an abundance of wetland types (i.e. tidal marsh, bay-gum, cypress swamp) found within the study area (Shaneyfelt & Metcalf, 2014).

Daphne and Spanish Fort receive 1680 mm of precipitation annually with temperatures ranging from 10.5 °C during the winter months to 28 °C in the summer months. Precipitation per month varies from 15.5 mm, during the driest month, to 67 mm during the wettest month (Herbert, 2012). These varying amounts of precipitation per season in concert with unconsolidated alluvial sand, gravelly sands, and clays in south Alabama affect the turbidity in the shallow Mobile Bay (Shaneyfelt & Metcalf, 2014).

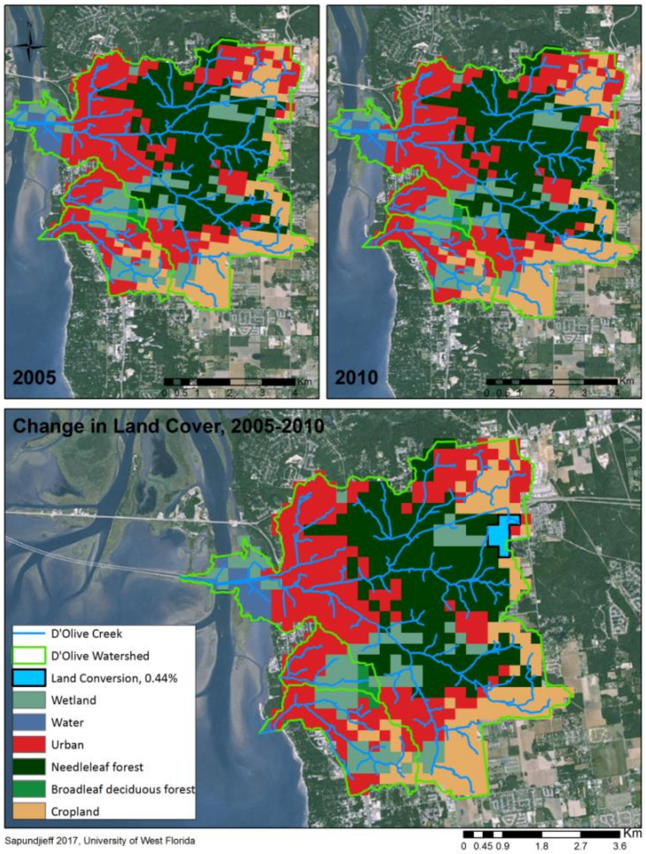


Figure 5: Dominant land use in D'Olive watershed from 2005-2010.

METHODS

Sampling Locations

At the time of study design, six restoration sites were expected to be completed by June 2016. Those completed in that timeline were included within the study. Construction delays resulted in only three restorations completed by the deadline: Tiawassee Creek (T0), Joe's Branch 1 (JB1R), and Joe's Branch 2 (JB2R). Around those restoration sites, six different types of sites were monitored for habitat quality and stream stability (defined below) in this two-year study: restoration sites, upstream sites, downstream sites, cumulative impact sites, an overall cumulative site, and a reference site. A naming convention.() was created to represent the relationship between each tributary and site type. The restoration sites were the actively engineered channel locations. The up and downstream sites were located within 300 meters of the active restoration site along the same channel. The cumulative impact sites were located at the conjunction of each major tributary (Tiawasee Creek, Joe's Branch, and D'Olive Creek). The overall cumulative site (A0C) was located at the conjunction of the three tributaries before they flow into Mobile Bay. The sites were all located within the D'Olive Creek watershed along its three major tributaries (Figure 6). The reference site (Y1) was located in a bordering subwatershed due to the lack of reference reaches within D'Olive Creek watershed.

The sites in the D'Olive Creek watershed restoration that were completed before Fall 2016 and were included in this study are: JB2 (Upstream, Downstream), JB1 (restoration, downstream), and T0C (cumulative), JB0C (cumulative) and A0C (cumulative).

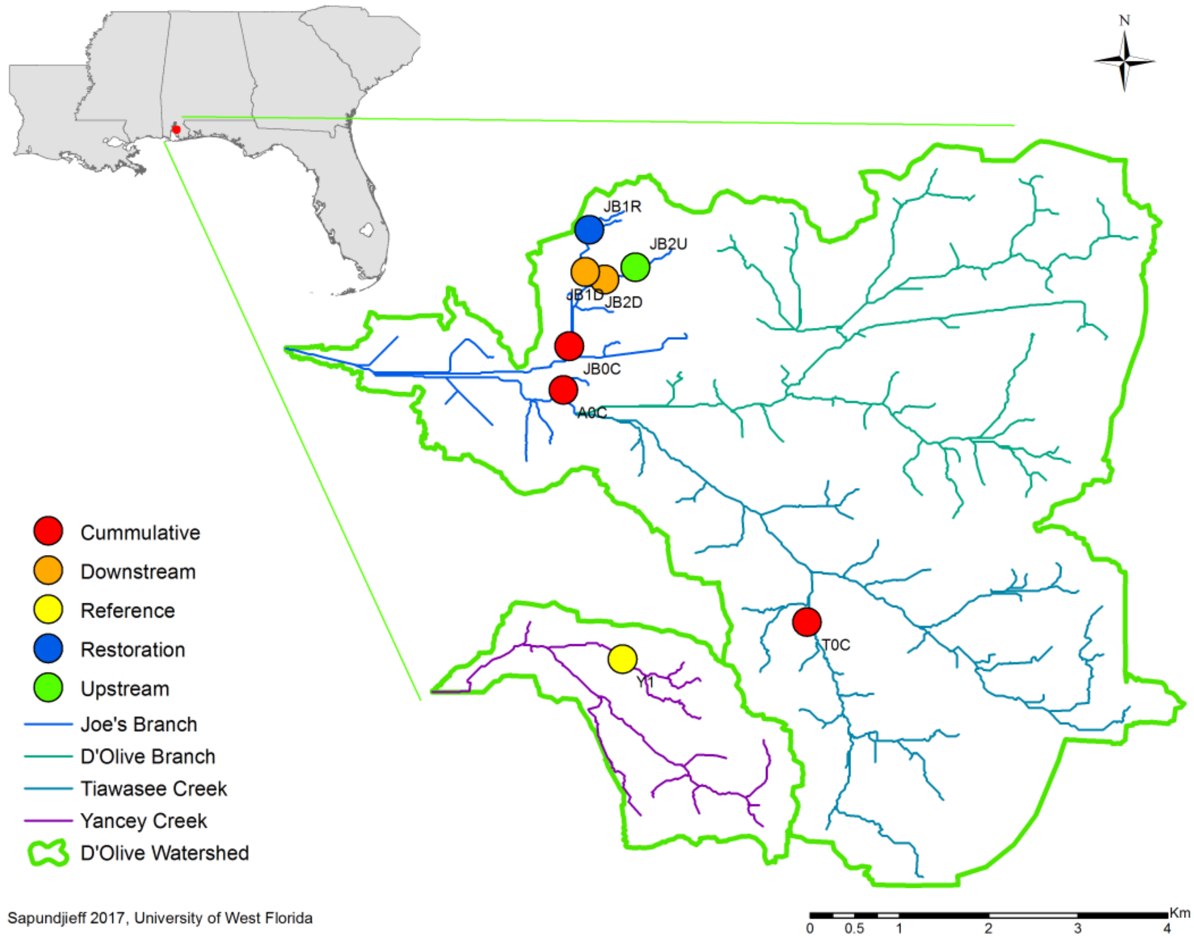


Figure 6: Map of sites monitored including cumulative impact and reference sites.

The restoration, upstream, downstream and cumulative impact sites were measured twice per year for two years, ideally once before the restoration construction and every six months thereafter. However, this was not obtainable due to construction complications. Table 2 shows the data collected during each sampling period while Figure 7 shows the date of each visit relative to the site construction.

Table 1: The naming convention used to identify each site.

First Letter (Tributary Name)	Base (Specific Restoration)	Final Letter (location relative to restoration)
JB- Joe's Branch	#- site	C- cumulative
T- Tiawassee Creek	0- cumulative site	D- downstream
A- Overall		U- upstream
Y- Yancey Branch		R- restoration
		No suffix- control site
Example: JB1D	1	D

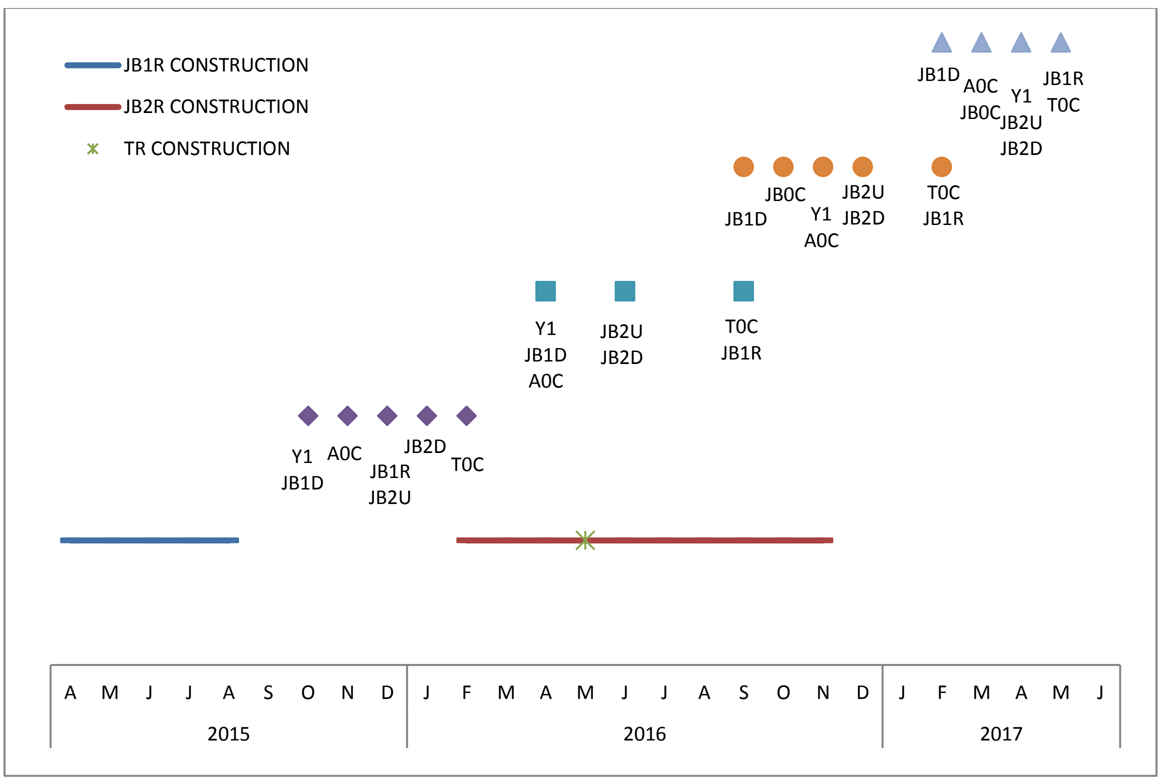


Figure 7: Timeline of restoration construction and data collection.

Table 2: Data collection timeline by data and site type.

	Downstream		Upstream	Restoration	Cumulative			Reference
	JB1D	JB2D	JB2U	JB1R	T0C	JB0C	A0C	Y1
Overstory Transect	10/15 2/17	1/16 4/17	12/15 4/17	12/15 5/17	2/16 5/17	10/16 2/17	11/15 2/17	10/15 4/17
Understory Transect	10/15 4/16 9/16 2/17	1/16 6/16 12/16 4/17	12/15 6/16 12/16 4/17	12/15 4/16 9/16 5/17	2/16 9/16 2/17 5/17	10/16 2/17	11/15 4/16 11/16 2/17	10/15 4/16 11/16 4/17
Stream Bank Vegetation	10/15 4/16 9/16 2/17	1/16 6/16 12/16 4/17	12/15 6/16 12/16 4/17	12/15 4/16 9/16 5/17	2/16 9/16 2/17 5/17	10/16 2/17	11/15 4/16 11/16 2/17	10/15 4/16 11/16 4/17
Canopy Cover	10/15 2/17	1/16 4/17	12/15 4/17	12/15 5/17	2/16 5/17	10/16 2/17	11/15 2/17	10/15 4/17
Stream Identification	9/16	12/16	12/16		2/16 5/17	10/16	11/16	
NBS	10/15 4/16 9/16 2/17	1/16 6/16 12/16 4/17	12/15 6/16 12/16 4/17		2/16 9/16 2/17 5/17	10/16 2/17	11/15 4/16 11/16 2/17	
BEHI	10/15 4/16 9/16 2/17	1/16 6/16 12/16 4/17	12/15 6/16 12/16 4/17		2/16 9/16 2/17 5/17	10/16 2/17	11/15 4/16 11/16 2/17	
Pfankuch	9/16	1/16 4/17	12/16 4/17		2/16 9/16 2/17 5/17	10/16 2/17	11/16 2/17	
Riffle Cross Section	10/15 4/16 9/16 2/17	1/16 6/16 12/16 4/17	12/15 6/16 12/16 4/17		2/16 9/16 2/17 5/17	10/16 2/17	11/15 4/16 11/16 2/17	
Longitudinal Profile	10/15 2/17	1/16 4/17	12/15 4/17		2/16 5/17	10/16	11/15 2/17	

Channel Stability Metrics

Riffle Cross Sections.

At each site there were two riffle cross sections that extended 15m from each bank (Figure 8). These riffle cross sections were placed perpendicular to the streamflow through two different riffles within the study reach. The cross sections helped determine channel stability by comparing changes in floodplain geomorphology and floodplain connectivity. The riffle cross sections were marked for repeated visits by placing 1m start (right bank) and end (left bank) rebar pins 15m back from each bank. The topography of the floodplain was measured using a stadia rod read through an automatic survey level placed on a tripod located where the entire cross section was visible (Figure 9). The elevation was measured at 0.5m intervals along the transect and 0.25m intervals within the stream to identify stream bed changes. The data was recorded to the half centimeter.

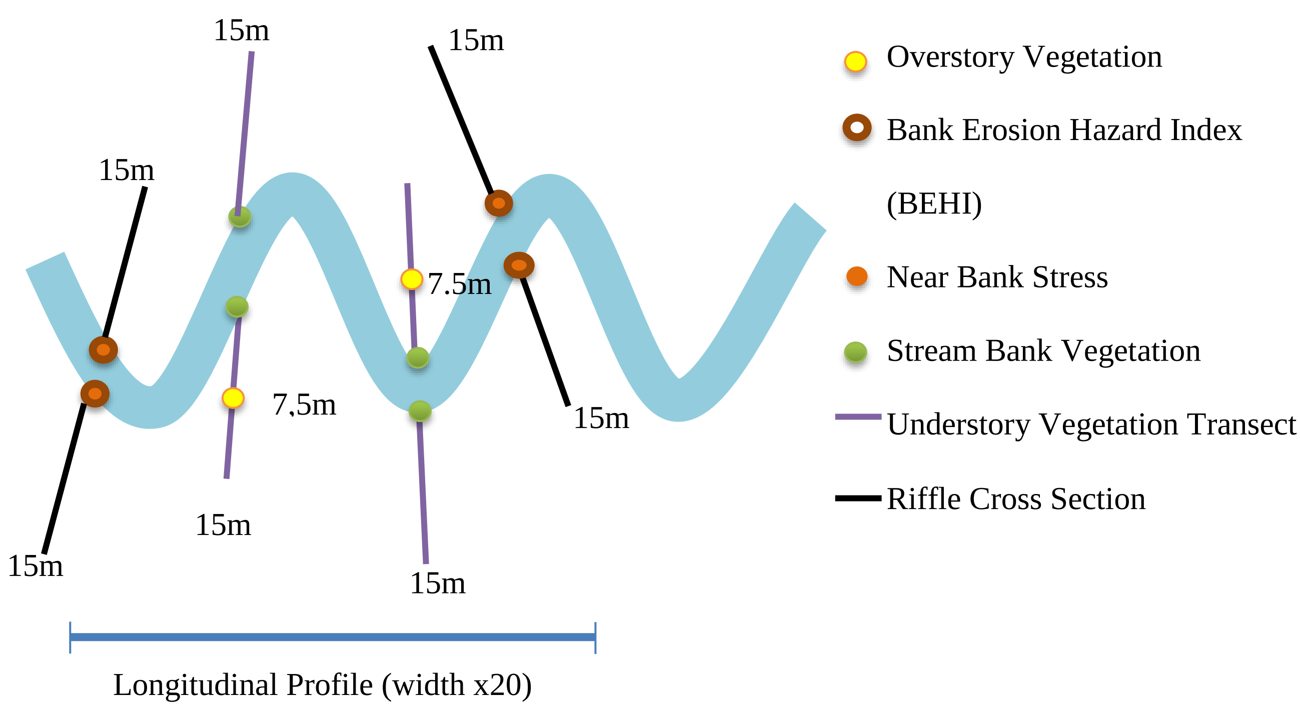


Figure 8: Schematic of site layout.

a)  b) 

Figure 9: Riffle cross section setup at a) JB2U Dec 2015 and b) T0C Feb 2016.

Bank Erosion Hazard Index.

BEHI was determined according to Rosgen (2001) (Appendix II) on both banks at each riffle cross section (Figure 8). The BEHI component variables were then scored to identify BEHI component scores (Appendix II A) and graphed according to (Appendix II B). The surface cover estimate needed for BEHI included root protection, leaf litter and vegetation cover. The surface cover was estimated by visually dissecting the bank into quadrants on the transect line to better estimate percent cover. The root depth was measured and reported as percentage depth that the roots extend through the vertical bank height. The stream bed sediment type (sand, silt, or clay) was characterized through texture using the feel method (Leopold, Clarke, Hanshaw, & Balsley, 1971). The feel method is a tactile method of identifying soil composition by creating soil ribbons; longer ribbons signify higher clay content, whereas an inability to form a ribbon signifies high sand content. The % root density was determined through visual estimate but additionally using a gridded intercept method. This method involved measuring the presence or absence of roots by inserting a thin metal rod into the gridded area. The grid was centered along the riffle cross section transect and extended one meter to both sides along the bank starting at the bank toe and being measured at 20 centimeter vertical intervals. The bank angle was determined using an electronic clinometer. The survey equipment as set up for the riffle cross-section was also used to measure bankfull and the height of the bank (variables in the BEHI). Because the restoration sites were engineered using hard stabilization techniques, BEHI was not measured at those sites.

Near Bank Stress.

NBS was measured along the riffle cross section transects using the ratio of near-bank maximum depth to bankfull mean depth as described in Appendix III (Rosgen, 2001). NBS was determined on the right and left bank, at both riffle cross section transects (Figure 8) using the survey equipment to measure the maximum near-bank water depth within a distance equivalent to one third of the stream width from the study bank and the bankfull mean depth. The mean depth was determined by averaging the stream bed depth measured at 0.25m intervals along the riffle cross section.

Longitudinal Cross Section.

In addition to two riffle cross sections, one longitudinal profile was measured at each site to quantify changes in elevation within the stream. The longitudinal cross sections started at the riffle cross section transect upstream and extended 20 times the width of the stream at bankfull or the distance between the two riffle cross sections, whichever was greatest. This cross section was used to quantify elevation and bed features within the stream. Readings were taken at slope breaks because it reflects greater variability than a set interval method. The distance between each measurement was measured using a tape measure. The topographic changes were measured with the survey equipment and stadia rod with measurements taken along the deepest portion, or thalweg, of the stream (Hack, 1957). The thalweg of the stream is the line connecting the deepest portion of consecutive cross sections (Figure 10).

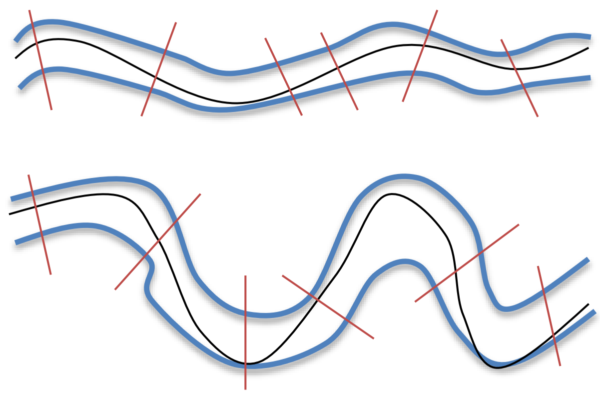


Figure 10: Graphical depiction of a thalweg (thin black line) connecting the deepest portion of consecutive cross sections (red lines).

Modified Pfankuch Stability Index.

Given the purpose of the present study, the use of a stability index that incorporates species response is most relevant. The present study used the USFWS modified pfankuch method (Appendix IV) to measure channel stability at each riffle cross section location (Pfankuch, 1975). The method was originally created in rock bed streams but has been revised for sand bed streams (Metcalf, 2015), thus the modified version is used in this study. This method measures several indicators on a numerically weighted cumulative scale from excellent to poor. The bank slope was determined using a laser range finder to determine the height of bank divided by the distance to the bank from the bank toe (rise/run). The present study averages pfankuch ratings at both riffle cross sections as opposed to generating a rating over the entire stream reach. The two pfankuch ratings are averaged to obtain the final rating. Evidence of mass movement and slope failure, obstructions, and scouring were visually surveyed for presence and extent. The debris jam potential was determined using a visual survey along the riffle cross section to identify large woody material, stream constraints and other geomorphic conditions that could produce debris jams in the upper banks. The vegetation bank protection was determined using the bank protection survey habitat metric explained in the following section. The channel capacity, width to depth ratio, and channel cuts (depth of the erosion under overhanging banks) were measured with a measuring tape or stadia rod to the nearest centimeter. The root density was measured as a ratio of root depth to bank height. The deposition rate was determined with the assumption that bare deposits are more recent than vegetated deposits with attention paid to size of particles. Sand particle size was determined using the feel method (Leopold et al., 1971) and were recorded as sand, sandy loam or loamy sand. Sand dune presence and large woody material were visually estimated as percent cover along the cross section. The presence and influence of scour (low to high) on aquatic vegetation and organic material within the channel was visually estimated along the riffle cross section transect.

Riparian Habitat Metrics

Stream Type Identification.

The streams were categorized as intermittent, ephemeral or perennial according to North Carolina Division of Water Quality –Methodology for Identification of Intermittent and Perennial Streams and Their Origins v. 4.11 (NC Division of Water Quality, 2010). The present study uses this procedure as it is the accepted procedure within the Alabama Department of Environmental Management, one of the primary stakeholders in the study.

The method classifies streams into ephemeral, intermittent, and perennial streams using geomorphological, hydrological and biological indicators. Using a form (Appendix V), geomorphic, hydrologic and biologic indicators are each ranked into one of four categories; absent (0), weak (1), moderate (2), and strong (3). If the cumulative score is between 19 and 30 the stream is classified as intermittent, if it is greater than 30 the stream is perennial according to the method. The stream type was identified within the first year of visiting the site.

Appendix V shows the geomorphic, hydrologic and biologic indicators considered in the method although the filamentous algae and iron oxidizing material was not included and fungi were not identified to species but noted for presence.

Understory Surveys.

Because vegetation on the riffle cross section transects had to be partly removed to ensure visibility through the automatic level, additional vegetation transects were needed to monitor vegetative growth. These new transects could not have any disturbance or bias that would impact the habitat metrics. Therefore, two vegetation transects were evenly placed between the riffle cross sections end pins (Figure 8).

The vegetation transect lines extended 15m back from the stream banks (Figure 8). The point-intercept method (Goodall, 1952) was used every 0.5m to estimate percent cover of leaf litter, bare soil, woody material, live vegetation and other material up to one meter in height. Only vegetation less than one meter in height, determined using a meter stick, was considered understory vegetation (Figure 11). When the meter stick touched vegetation at any height and another variable (woody material, bare soil, or leaf litter), all variables that touched the meter stick were marked present. The percent cover was determined as points with each variable present (leaf litter, bare soil, live vegetation, woody material, other) divided by the total number of points sampled. The only species that were noted were common non-native species. Common non-native species in the riparian forest include chinese privet (*Ligustrum sinense*) chinese tallowtree (*Triadica sebifera*), camphor tree (*Cinnamomum camphora*), japanese climbing fern (*Lygodium japonicum*) and kudzu (*Pueraria lobata*). Other non-native species were added to the list if they were observed more than three times at one site. In this particular study, coral ardisia (*Ardisia crenata*) was a common non-native species.

Figure 11: Understory survey point intercept method along the vegetation cross sections at T0C in February 2017.

Overstory Surveys.

Overstory composition was quantified using two variable radius vegetation plots 7.5m back from the stream bank along the understory vegetation transects. One plot was located on each side of the stream bank along each vegetation transect totaling two plots (Figure 8). Using a 10 BAF wedge prism the applicable trees were measured for size (diameter at breast height), genus, and compositional structure. All of the “in” trees and half the borderline trees were measured. See 12 to differentiate between “in”, “out”, and “borderline” trees.

Figure 12: Image showing the difference between in, borderline, and out trees for use within the non-fixed radius vegetation plots (Hemery, 2011)

Diameter at breast height (DBH) was measured at a height of 1.37 m using a diameter tape (Figure 13). The trees were identified to genus using reference materials. The trees were categorized into emergent, canopy, subcanopy, and midstory. At each plot the canopy cover was measured using a densiometer (Figure 14). The data was analyzed to determine percent nativity, genus diversity and abundance.

Overstory vegetation was only measured at the first and final data collection dates to show changes over the study period instead of changes every six months as woody vegetation grows much slower than herbaceous vegetation.

Figure 13: Volunteer using the DBH tape at A0C in April 2017.

Figure 14: Image of the densiometer instrument. Canopy cover is determined using the gridded mirror and presence/ absence within the grid.[ii]

Stream Bank Vegetation.

Vegetation along the stream bank provides habitat for invertebrates and some vertebrate species. The point intercept method (Goodall, 1952) was used to determine percent stream bank vegetation relative to the size of the bank. In order to achieve this goal, ten measurements were taken on each bank with varying intervals (i.e. number of contacts relative to the total number of points sampled). This was accomplished using a vertical stadia rod, stabilized using a bubble level, placed at the bank toe. A meter stick held horizontal with a line level was used on the right side of the stadia rod (placed directly under the transect line) to determine whether vegetation was present. If the meter stick touched vegetation then it was marked as present, if it did not touch vegetation it was marked as absent. The stadia rod was held on the upstream side of the transect tape. The bank height above the greenline (greenline to the top of bank) was divided by 10 to find the interval for the point intercept method. The first measurement was taken at the greenline, which is the lowest section of the bank with growing vegetation.

Riparian Habitat Health Level Evaluation

Many indices to establish riparian health include hydrological, geomorphological and biological assays of macroinvertebrates, fish, and vegetation (Felipe-Lucia & Comín, 2015; Gonzalez del Tanago & Garcia de Jalon, 2010; Morley & Karr, 2002). Included within some of these indices are indirect measures of water quality using macroinvertebrate species diversity or abundance counts (Krishnamoorthi & Sarkar, 1979). Due to the established connections between macroinvertebrates, geomorphology and habitat strata, this study proposes the creation of the Riparian Habitat Health Level Evaluation (RipHLE) to rapidly quantify general habitat health focusing primarily on the vegetative biology. By avoiding an exhaustive biological assay of all indicator macroinvertebrate and fish species and instead using already established associations, this index will reduce operating time and budget in the field. A traditional fauna assay would include costs for field operation and lab identification with significant delays while awaiting results. Creating a rapid assessment has important implications in optimizing monitoring efforts to reduce costs which enable monitoring to more cost effective. When costs of monitoring are too high, it is often left out of restoration projects (Palmer et al., 2005), which will not allow the end users to identify successful strategies in the long term. This prioritization for rapid assessment, with scientific basis for evaluation may provide a useful annual monitoring tool that should be able to indicate a need for more intensive evaluation. This tool is not intended to assess stability success of a restoration project on its own, but can be evaluated in conjunction with other stability measures to identify changes in habitat quality which may necessitate additional in depth monitoring (where the quality decreases).

The Riparian Habitat Health Level Evaluation (RipHLE) was created using factors that have been shown in the literature to be associated with species diversity or abundance in either the stream channel or riparian/ floodplain forest and correlated to regional macroinvertebrate data (abundance and/or diversity). The proposed index contains eight variables able to distinguish between poor, moderate, and good habitat conditions: riparian buffer width, bank erosion hazard index (BEHI), canopy cover, tree basal area, bank root density, leaf litter (% cover), structural complexity and metrics of non-native species (details below). The eight variables were selected based on a literature review as explained below and presented to a panel of experts for review. This panel included Renee Collini and Jason Kudulis, Mobile Bay National Estuary Program; Randy Shaneyfelt and Lisa Huff, Alabama Department of

Environmental Management; Don Blanchard, Sustainable Ecosystem Restoration, LLC; Patrick Harper, United States Fish and Wildlife Services; and Johan Liebens, University of West Florida.

Buffer Widths.

Larger wildlife species require larger buffer widths than smaller species (songbirds compared to macroinvertebrates), therefore macroinvertebrates, as indicators of stream health, were considered when generating parameter limits. Many researchers agree that the benefit to biodiversity increases overall with buffer width (Lowrance & Sheridan, 2005; Wenger 1999, Lee, Smyth & Boutin, 2004). Larger buffers widths (>30m) correspond to improved habitat conditions as designated by increased indicator fish species diversity and local hydrology, soil factors and slope (Stewart et al., 2001) and avian species diversity and range (Spackman & Hughes, 1995). Therefore, good conditions are described by large buffer widths (>20m), moderate conditions (10-20) and poor conditions (0-10m).

BEHI.

Geomorphic and habitat stability, determined through low BEHI scores, are linked to higher diversity (macroinvertebrate abundances) which are known indicators of water quality and food sources (Simpson et al., 2014). BEHI also includes a measure of bank height to bankfull ratio which is a driver in floodplain connectivity; connected floodplains are able to pass resources between the stream and riparian buffer. Therefore, good conditions are described by very low to low erosion potential, which equates to 5-19.5 on the BEHI index (Appendix II), moderate conditions by moderate BEHI (19.6-29.5), and poor conditions described by high to extreme erosion potential (29.6-50).

Canopy Cover.

Canopy cover affects light attenuation to the forest floor and regulates stream temperatures. Moderate canopy cover (60-80%) is associated with high taxon richness (Townsend et al., 1997) compared to higher canopy cover (>95%) which is correlated to lower macroinvertebrate abundances (Nislow & Lowe, 2006). Due to a lack of regionally specific canopy cover data in the literature, the parameter limits (high: >95%, moderate: 60-80%, and low: 30-50%) were presented to the expert panel. Shaneyfelt and Metcalf (2014) in their Coastal Alabama pilot headwater stream survey study, found that ideal conditions in similar riparian forests typically have moderate canopy cover, suggesting that moderate cover is representative of a good canopy cover condition, although typical values are lower than presented in the literature (51- 88%). Therefore, with the inclusion of expert opinion, canopy cover parameter limits were altered to high: 89-100%, moderate: 51-88%, and low: 30-50%, extremely low: <30%. These value are categorized into good (cover 51-88%), moderate (30-50%) and poor (89-100, <30%) habitat condition. The RipHLE components do not all contribute to the resulting index equally. The index is more sensitive to extreme values in the variables, especially for canopy cover and leaf litter where both high and low extremes are rated poor. Furthermore, good condition parameters generally have a narrower range of values to reflect only the idealized conditions, whereas moderate and poor parameters are broader in range. This is not consistent across all variables however, because bank root densities have the lowest range in the moderate condition category. Due to the larger index values in the poor categories, decreasing BEHI values have an overall larger effect on the cumulative RipHLE value than other indicators. The range of BEHI values (5-50) is smaller than the possibilities for other variables (0-100), which mean smaller numerical shifts in BEHI scores, will have larger effects on RipHLE values. This is to reflect the relationship between stream stability and riparian buffer health (Table 3).

Basal Area.

Basal area affects the ability of wildlife to meet their basic needs: find appropriate food and shelter. Low basal area (<13.7 square meters per hectare) is optimal for wildlife although it provides trade offs for other ecosystem services (Forest and Wildlife Research Center, 2017). After seeking expert opinion from the panel the optimal basal areas for urbanized watershed (species dependent) ranged from 13.7-16 square meters per hectare. Areas with basal areas greater than 18 square meters per hectare begin to see negative impacts on vegetation growth (Elledge & Barlow, 2012). Following this input, the parameters for basal area were altered to: poor, >18 square meters per hectare; moderate: 13.7-18 square meters per hectare; and good: <13.7 square meters per hectare.

Root Density.

Root density, as a vegetative component of stream stability, is included in the index to correct for BEHI visual estimates of surface protection, root depth and root density. Cavities between bank roots are critical habitat for some macroinvertebrates, provide stabilization for streams, and allow riparian vegetation to access stream resources (Rhodes & Hubert, 1991). As this additional variable is intended as a correction factor for BEHI estimates, the density parameters reflect BEHI density categorization: good, 55-100%; moderate, 30-55%; and poor, 0-29%.

Leaf Litter.

Leaf litter decomposition introduces nutrients, such as carbon, back into the ecosystem and affects diversity at the trophic level. Leaf litter can be an indicator of decomposition processes, and therefore microbial complexity within ecosystems. Leaf litter parameters were based on studies analyzing organic matter by dry ash weight and converted to percentages. This method is flawed in that it assumes that all organic matter is composed of leaf litter. For this reason this data was presented to the panel of experts from various government agencies for review. The leaf litter percentages are consistent with visual accounts of ideal riparian habitat, with one exception, in that very low litter was considered poor rather than moderate habitat condition. Therefore the parameter limits are: good, 40-89%; moderate, 11-39%; and poor, 0-10 and >90%.

Structural Complexity.

Structural complexity is important to carrying capacity of a forest. Structural complexity represents spatial and multi-age distribution of dominant vegetative species which impact photosynthetic activity and riparian forest productivity (Townsend et al., 2008). To maintain a simplistic representation of forest strata to maintain low operating costs, the parameter limits follow: good, multi strata (>2); moderate, dual strata (2); and poor, single strata or lacking dominant vegetation.

Non-native Species.

Non-native species are used as an indication of the level of disturbance. The study area consists of urban streams, but not all are alike in their level of development. Non-native species are problematic when found in natural forests for many reasons, some of which include proclivity toward establishment either by enhanced reproduction (reproduces in multiple ways- seed pods and rhizomes) or disruption of native plants (acidification of soils resulting from biogeochemical processes involved in leaf litter decomposition) (Funk & Vitousek, 2007). In a draft design of the RipHLE index, non-native plants were included so that any presence meant a poor habitat condition. This method may be useful in an intact forest, but per the expert panel input, is not useful in urban streams because the likelihood of invasive species is greater. The panel suggested the use of a gradient of invasiveness (by relative distance along the understory transect) to determine the invasion of the interior and theoretically non-impacted portion of the riparian buffer. Therefore, this segment of the RipHLE index was edited to account for presence/absence and spatial distribution of non-native species. This prevents the index from reflecting poor habitat quality due to the presence of small amounts of non-native species that are likely in urban riparian buffers. The parameter limits were based on discussions that non-native species venturing more than 1/3 of the width of the riparian forest were established colonies. With this in mind, the transects are 15m in length so any presence of non-native species in the outer, external, third (5m) or internal third will dictate habitat quality. If the non-natives are only present at the outer edges of the transect (away from the stream) then the index scores a moderate condition, (external only). Where there are non-natives throughout the transect (beyond the external), the index scores a 1 indicating a poor habitat condition for high presence of non-native species.

Reporting RipHLE.

For the final RipHLE for a site, each poor category receives a 1, moderate a 2, and good a score of 3. In order to obtain an overall good rating on the assessment, 5 of the 8 categories must be rated at least a 3 on the index resulting in a score range of 21-24 for good, 13-20 for moderate, and 7-12 for poor. This index allows for an overall scoring of habitat condition based on associations between habitat variables and macroinvertebrate abundance/diversity (used as a proxy for stream quality) as a less time intensive and scientifically valid method of determining riparian habitat quality.

The RipHLE components do not all contribute to the resulting index equally. The index is more sensitive to extreme values in the variables, especially for canopy cover and leaf litter where both high and low extremes are rated poor. Furthermore, good condition parameters generally have a narrower range of values to reflect only the idealized conditions, whereas moderate and poor parameters are broader in range. This is not consistent across all variables however, because bank root densities have the lowest range in the moderate condition category. Due to the larger index values in the poor categories, decreasing BEHI values have an overall larger effect on the cumulative RipHLE value than other indicators. The range of BEHI values (5-50) is smaller than the possibilities for other variables (0-100), which mean smaller numerical shifts in BEHI scores, will have larger effects on RipHLE values. This is to reflect the relationship between stream stability and riparian buffer health.

Table 3: Riparian Habitat Health Level Evaluation Index

Index Rating	**Poor (1)**	**Moderate (2)**	**Good (3)**
Buffer Width	low (0-10m)	moderate (10-20)	high (>20m)
BEHI	high- extreme (29.6-50)	moderate (19.6-29.5)	very low –low (5-19.5)
Canopy Cover	high (89-100%, <30%)	low (30-50%)	moderate (51-88%)
Tree Basal Area	very high (>18)	moderate (13.7-16)	low (<13.7)
Bank Root Density	low % (0-29%)	moderate (30-55%)	high percent (55-100%)
Leaf Litter (% ground cover)	very low or very high (0-10, >90)	low (11-39)	moderate-high (40-89)
Structural Complexity	low (single strata)	moderate (dual strata)	high (multistrata)
Non-Native Species	throughout	external only	none
	Score 7-12	Score 13-20	Score 21-24

RESULTS AND DISCUSSION

Stream Stability Indicators

Stream Type Identification.

Three sites were classified as intermittent streams and three sites as perennial streams based on the NC Division of Water Quality (Appendix V). Of these, the first order streams were classified as intermittent, with third order and above streams classified as perennial (Table 4).

Table 4: Stream identification values and corresponding classifications.

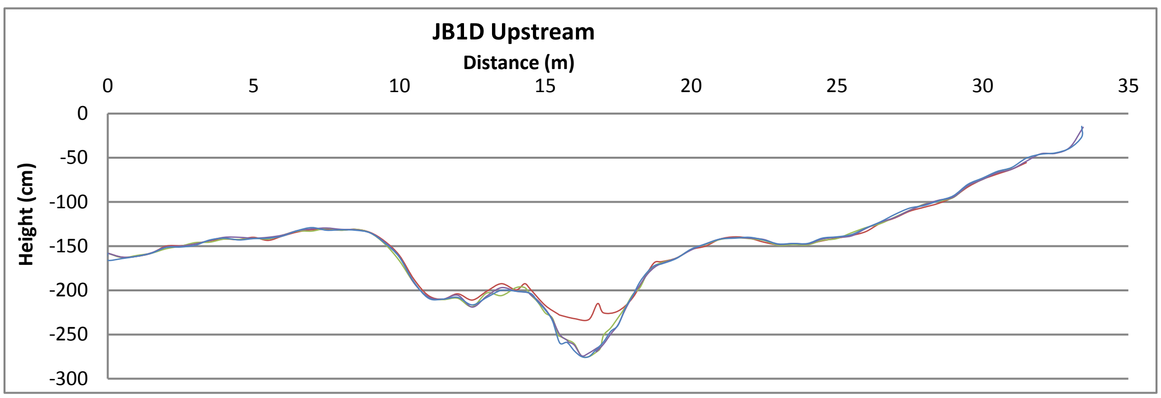
Site	Score[1]	Classification
A0C	32.5	Perennial
JB2D	26	Intermittent
JB2U	20.5	Intermittent
JB0C	32.5	Perennial
JB1D	23.5	Intermittent
T0C	43.5	Perennial

[1]scores <19 are ephemeral, 19≥score>30: intermittent; score ≥30: perennial.

Floodplain and Channel Stability.

The channel and floodplain morphology, much like the BEHI and Pfankuch values, are variable with site location. Many of the sites show stable floodplain morphology over the study period which can indicate a disconnected or inactive floodplain, such as JB1D and T0C (Figure 15 and Figure 16). Other sites show minor variability, 0.5-2cm vertical changes in the floodplain which indicate an active floodplain where overbank flow occurs within a six month period

Figure 17: JB0C, Figure 18: A0C, Figure 19: JB2U). JB2D is a special case that shows more major changes (Figure 20), sometimes an increase up to seven centimeters, between visits. This site specifically was primarily sandy deposition resulting from large gully erosion upstream (at the restoration site). The restoration occurred during the second sampling period (June 2016) and was completed just one month prior to the third sampling period, which may have continually contributed construction materials to the downstream floodplain. JB2D shows deepening in both the floodplain and channel after the start of the JB2 restoration construction in February 2016 (Figure 20). This site does show response to the restoration activity upstream, not only in the floodplain, but also in the changing channel morphology.



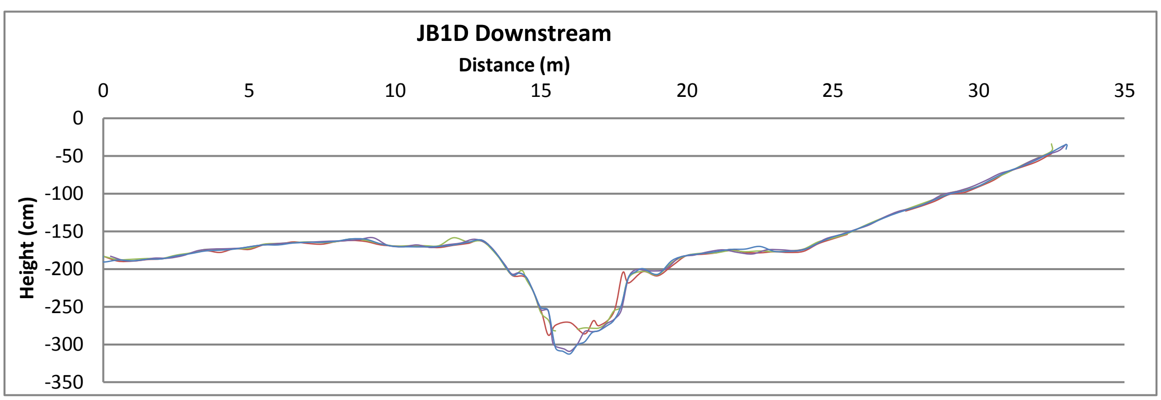


Figure 15: JB1D Riffle Cross Section profile. Visits in chronological order are colored: red (October 2015), green (April 2016), purple (September 2016), and blue (February 2017).

Even though the floodplains may have been stable between the sampling periods, both JB1D and T0C show increased depth in the stream channel following the respective restorations. The JB1 restoration was completed just two months prior to the first sampling period, which may have been too temporally close to notice any changes at the JB1D site. At the second sampling period (April 2016) the stream channel deepened by an average of 29.6cm in the upstream transect (only an average scour of 9.13cm at the downstream transect). By the third sampling period (September 2016) the upstream transect was not incised but the downstream transect was deepened by 21.7cm compared to sampling period 1 in October 2015 (Figure 20). The T0 restoration occurred in May 2016, two months after the first sampling period, and three months prior to the second. By the second sampling period (September 2016) portions of the channel had changed by an average of 19.6cm elevation (Figure 16).   JB0C also shows some increased depth in the stream channel by an average of 7.4cm (

Figure 17 17).

A0C is another site that showed major changes in channel elevation, although more extreme at the downstream riffle cross section (Figure 18). However, unlike the prior sites, A0C showed major filling of the channel by up to 18.6cm at the upstream transect and up to an average of 47cm at the downstream transect (Figure 18). While this could indicate that the sediment from the channel bottom is not being deposited into Mobile Bay but rather in the stream before reaching the bay, this site is within 150m of the mouth of the stream and tidal influences are more likely a cause of the changing channel bed. While tides have not changed significantly throughout the 2 year study period, of all the sites, A0C is the only site with tidal oscillations each day.

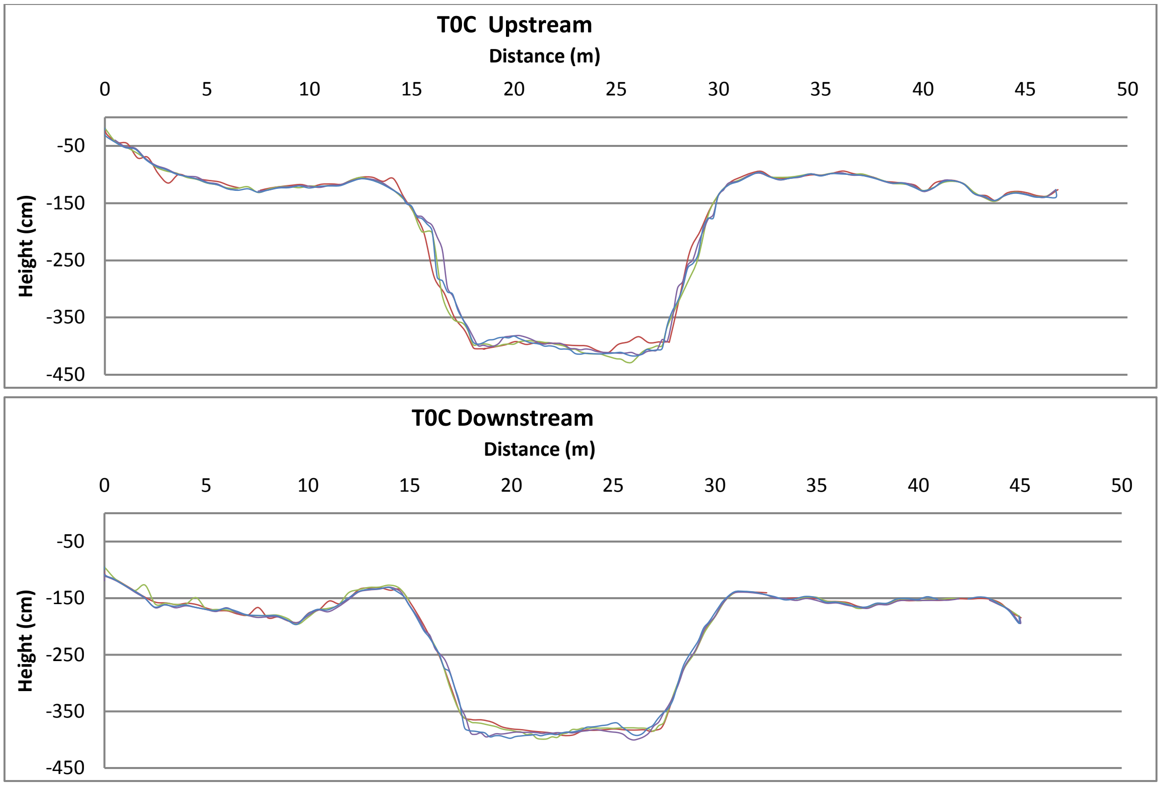


Figure 16: T0C Riffle Cross Section profile. Visits in chronological order are colored: red (February 2016), green (September 2016), purple (February 2016), and blue (May 2017).

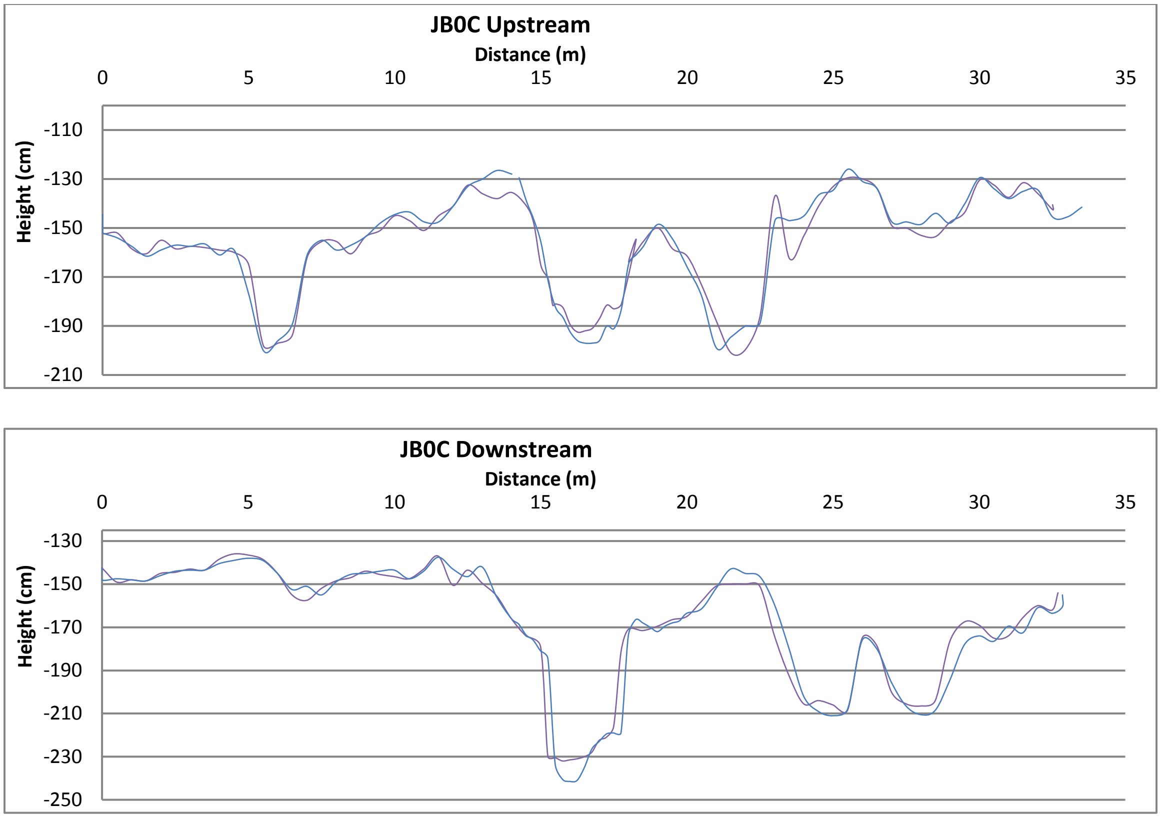


Figure 17: JB0C Riffle Cross Section profile. Visits in chronological order are colored: purple (October 2016), and blue (March 2017).

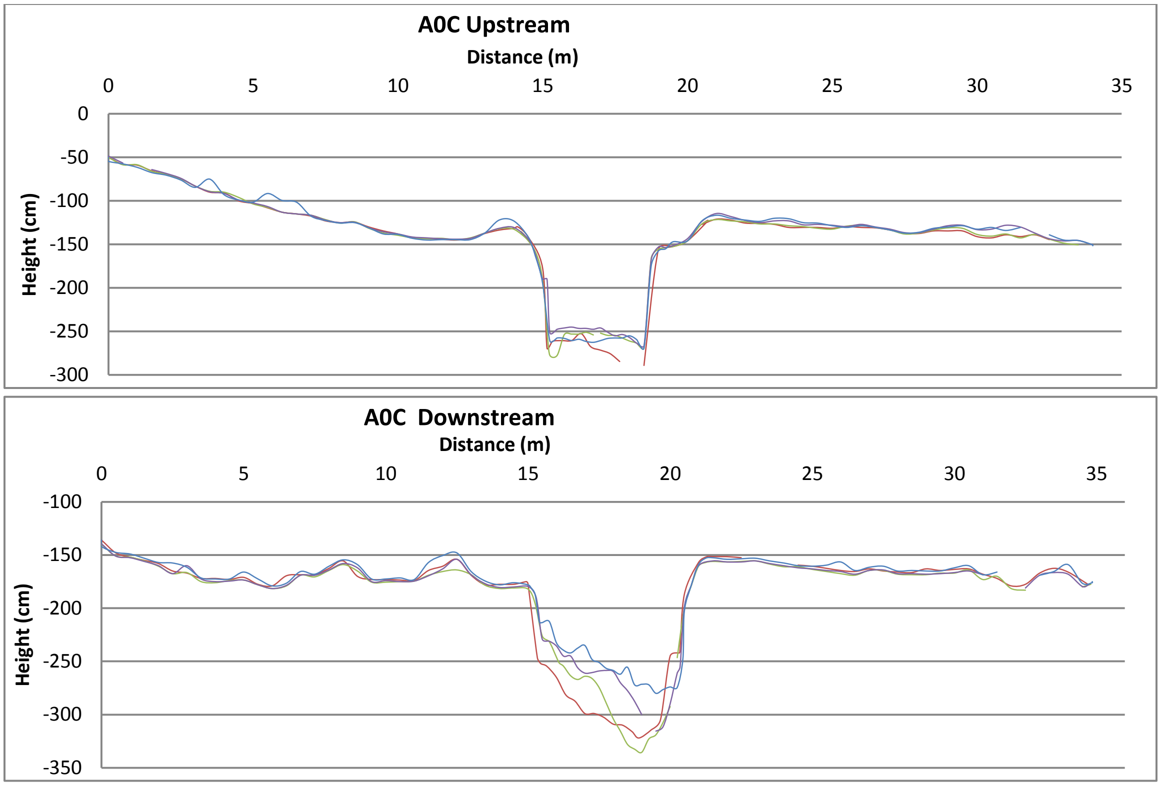
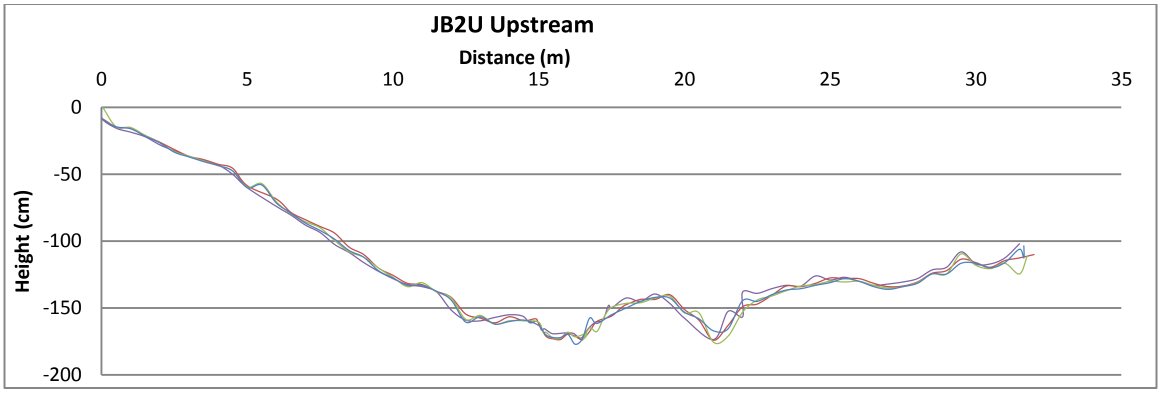


Figure 18: A0C Riffle cross section profile. Visits in chronological order are colored: red (November 2015), green (April 2016), purple (November 2016), and blue (March 2017). Erroneous data is noted as a break in the line series.

Unlike the other sites, JB2U (Figure 19) has minor fluctuations throughout the entire transect, including the channel bottom. This site does not exhibit any major incision or filling which is to be expected from a site upstream of the restoration construction. JB2D has major changes in morphology through both the floodplain and the stream channel (Figure 20 and Figure 21). The channel bottom at the upstream transect was mostly consistent with a deepening just after the restoration construction (December 2016) that was filled in by the final sampling period (April 2017) (Figure 20). The downstream transect showed a continually deepening channel, which although channel is deepening, is not incised.



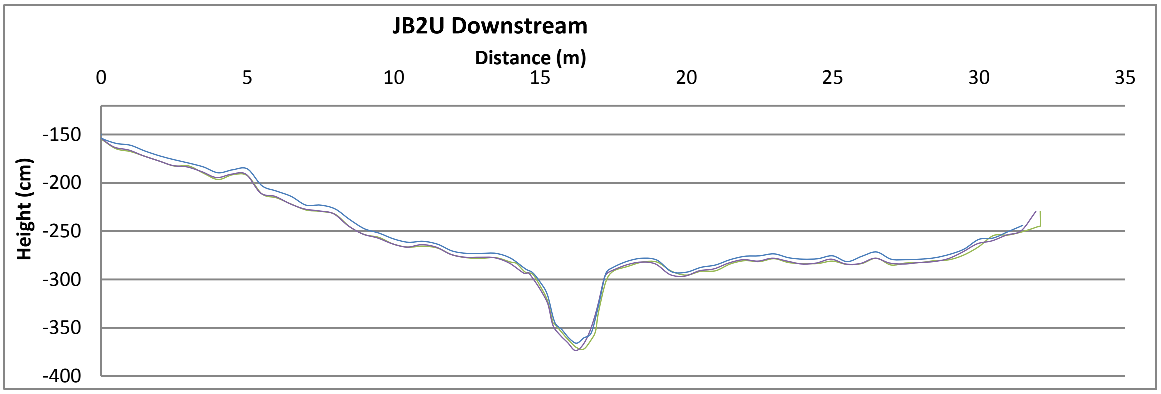
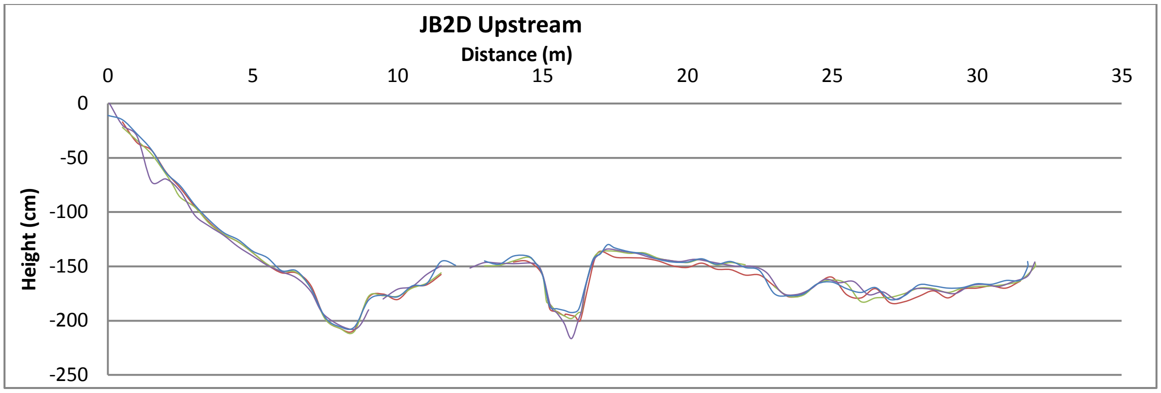


Figure 19: JB2U Riffle Cross Section profile. Visits in chronological order are colored: red (Jan 2016), green (June 2016), purple (December 2016), and blue (April 2017).



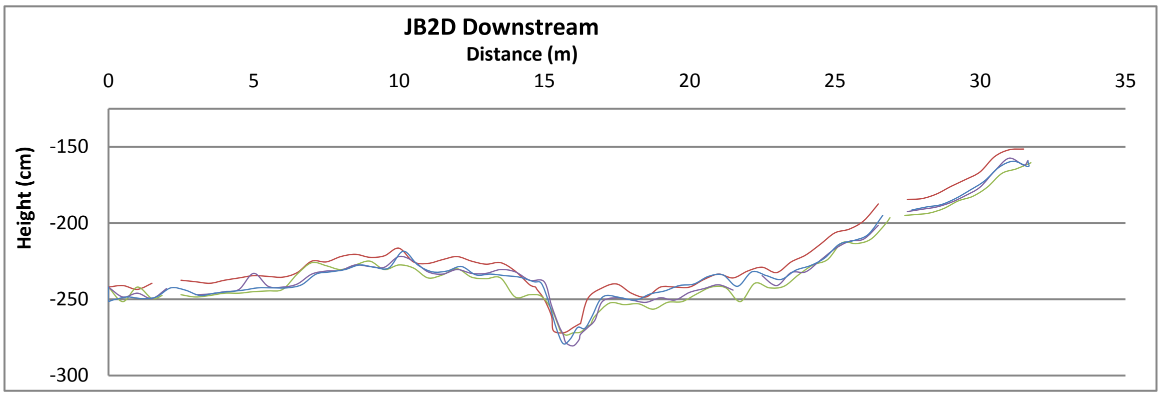


Figure 20: JB2D Riffle Cross Section profile. Visits in chronological order are colored: red (December 2015), green (June 2016), purple (December 2016), and blue (April 2017).

Figure 21: Image showing the sandy deposition at the downstream riffle-cross section at JB2D in April 2017.

Bank Cross Sectional Area

Cross sectional area is the product of channel width and mean channel depth which generally increases along a river downstream as more tributaries feed into the main branch (Ritter, 2017). Following this general pattern, the cumulative sites would have larger cross sectional areas than the upstream sites. JB0C however has a cross sectional area that is comparable to JB2U and JB2D even though it is further downstream from all JB sites. JB0C unlike the upstream JB sites is a braided channel which means that not all of the flowing water through the singular channel sites flows through the measured JB0C channel. T0C has the largest cross sectional area ranging from 52 to 68 m^2 which is at least three times the cross sectional area of A0C. The water height at T0C, while not specifically measured in this study, was consistently lower than the height of wader boots as seen in Figure 22. The low water level, large cross sectional area suggests that T0C is entrenched, meaning that the channel waters are contained within its banks and otherwise unable to overflow its banks.

a)  b) 

Figure 22: T0C water level was consistently shallower than boot height in a) February 2016 and b) September 2016.

The percent change between the first and final study periods suggest that the largest percent change in cross sectional area occurs at JB0C with an increase in 28% capacity which is comparable to changes found by Nichols and Ketcheson (2013) in Finney Creek, Washington, although this study looked at changes in inner channel area. They report findings of up to +26% changes at locations 10m downstream of log jams after a six year study period. However, the present study suggests that three sites experienced a decrease in cross sectional area over the two year study period. Studies like Nichols and Ketcheson (2013) also show some negative percent changes (-2%) but not similar to the -22% change observed at A0C or the -23% observed at JB2U. When comparing this data to the riffle cross section profiles, both A0C and JB2U indicate some amount of deposition occurred between SP1and SP4 (Figure 18: A0C, Figure 19: JB2U). This would cause the mean depth to decrease which decreases the cross sectional area.

Table 5: Average cross sectional area of each site where stability metrics were measured over the two year study period.

Site	SP1	SP2	SP3	SP4	% change (entire study period)
A0C	17.39	14.71	13.75	13.57	-22%
JB0C			6.61	8.48	+28%
T0C	58.36	67.88	64.11	52.55	-10%
JB1D	11.71	10.81	14.66	11.75	0%
JB2D	4.79	5.06	5.85	5.33	+11%
JB2U	8.90	7.83	7.14	6.85	-23%

Longitudinal Profiles.

In addition to observed scour within the channels, several of the stream reaches exhibited gradient changes. As an example, JB1D had a 45.5 cm change in elevation from the starting pin of the longitudinal profile to the final pin in April 2016 and a 51.5 cm change in February 2017, showing an overall difference of +6 cm between the two visits. Figure 23 shows the two longitudinal profiles from JB1D where the thalweg is noticeably deeper in 2017. There was no longitudinal profile measured in October 2015 at the first sampling period directly following restorations.

The longitudinal profile of A0C was not measurable due to incoming tides throughout the data collection period and safety precautions. Other sites showed changes in gradients as well; T0C, 4.5cm and JB2D, 8cm. The thalweg elevates in the JB2D longitudinal transects (Figure 24) compared to the JB1D transect (Figure 23) shows much more variability. Thus, the JB2D site, which showed variability in the riffle cross sections, also shows that the thalweg of the stream was in a continual state of flux over the two year study period.

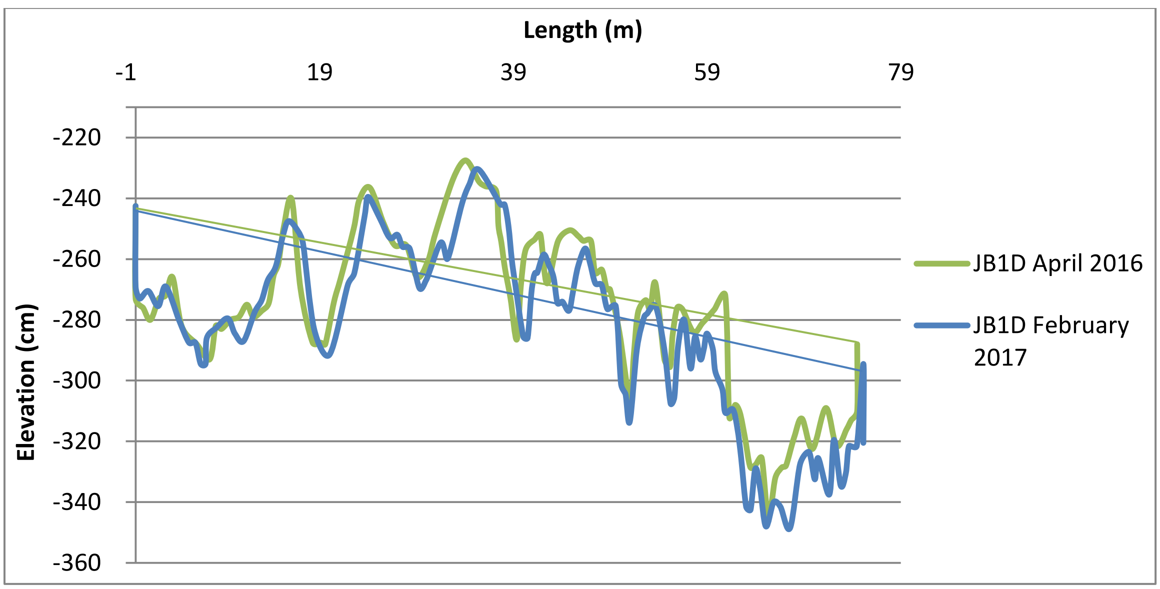


Figure 23: Longitudinal profile of JB1D showing thalweg of the measured stream in April 2016 and February 2017. Change in gradient is illustrated with straight line segments.

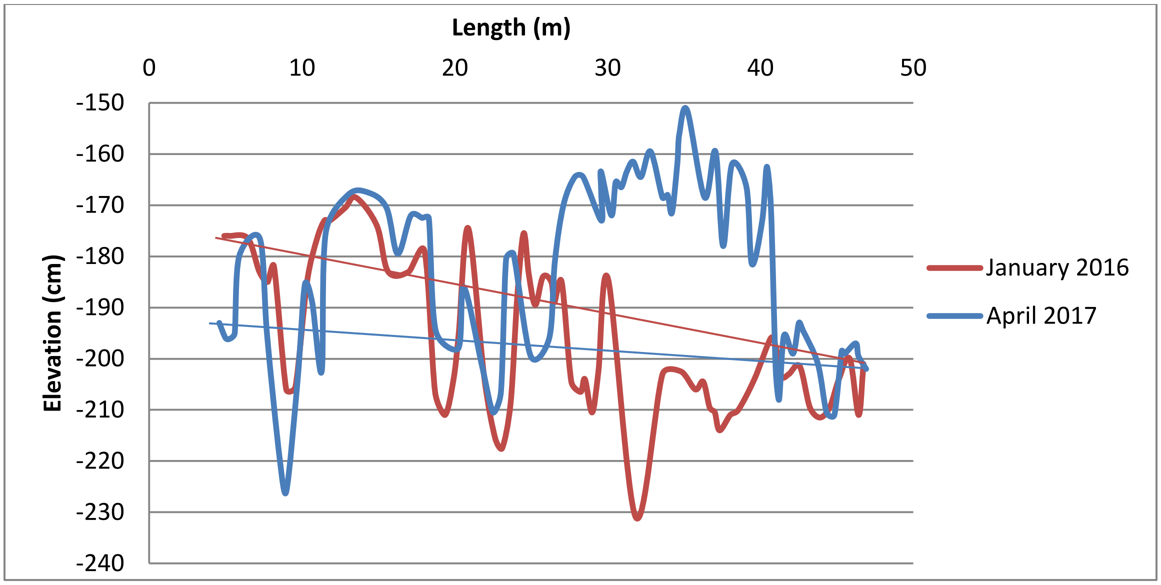


Figure 24: Longitudinal profile of JB2D showing thalweg of the measured stream in January 2016 and April 2017. Change in gradient is illustrated with straight line segments.

Streambank Vegetation.

The streambank vegetation cover also shows variability between sampling periods. It was expected there would be increased vegetation during the sampling periods that occurred in late spring and early fall but the results show that the vegetation coverage did not consistently change with seasonal variation (Figure 25). External factors can influence stream bank vegetation including fallen trees (as occurred at T0C), rapid sedimentation (such is the case at JB2D where the channel and floodplain continually changed between each sampling period Figure 20) and tidal influence (as seen in A0C). Rapid sedimentation could be caused by increased suspended sediment caused by erosion or even stream restoration.

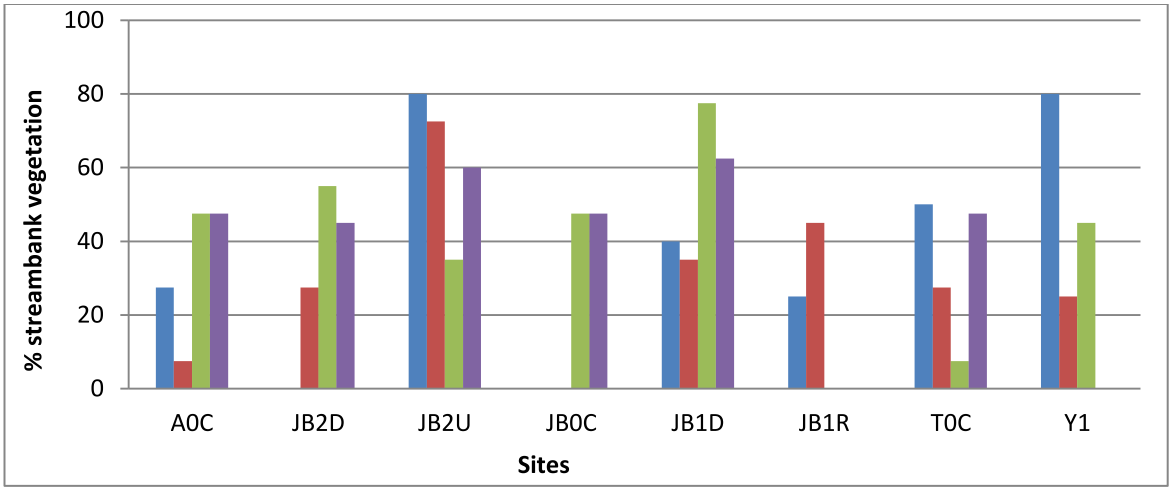


Figure 25: Streambank vegetation cover at each site per site sampling period. Sampling period 1Fall/Winter (red), SP2 Spring/Summer (green), SP3 Fall/Winter (purple), and SP4 Spring/Summer (blue).

Riparian Habitat Indicators - RipHLE Components

Buffer Width.

Buffer widths remained constant throughout the study period to at least 20m (largest distance reflected in RipHLE), although some sites, such as Y1 did have lumber thinning occur in the upland areas surrounding the riparian buffer. This has several implications that are important within the context of this study. By not showing any changes in riparian buffer width, the study can exclude urban development of the buffer as a reason for any results during the study period. This is also important because it allows us to reasonably compare the study sites. While larger buffers widths are associated with healthier systems, we can determine the other factors that might improve the riparian health if increasing RipHLE scores are indicated.

BEHI.

The BEHI at each site only showed significant changes ($p<0.05$) between sampling periods at JB2D (Table 6). At each study site, four BEHI measurements were taken: at the up and downstream riffle cross section transects on both the right and left bank, which when excluding data errors totals 83 individual BEHI measurements taken over the two year study period. The overall changes between the four BEHI measurements at each site were not significantly different between collection periods. This means the internal variation between the four measurements from each data collection period are similar. This verifies that the average of the four BEHI values at each time period can be used a descriptor to describe the BEHI state, which supports the inclusion of the BEHI variable, in terms of a site average in the Index. When using the average BEHI to determine categorical risk at each data collection period (totaling 22 over the two year study period), there are seven high erosion hazard sites, 13 moderate erosion hazard sites, and two low erosion hazard sites. (Table 6 6).

Of these sites, all of the downstream sites (JB1D, JB2D) and the upstream site (JB2U) underwent a decline in erosion risk over the two year study period; changes which correspond with restoration construction. For example, the JB1 restoration was completed two months before the first sampling period (October 2015), which may not have exhibited any changes, due to an effect lag at JB1D, until April 2016 (sampling period 2) when the risk shifts from high to moderate and maintains a moderate risk through the end of the study.(Table 6). This pattern follows the concept that the restoration improved the downstream stream stability according to the BEHI metric. The second sampling period for JB2D (June 2016) took place during active restoration of JB2; so the erosion potential declined from high (first sampling period- January 2016) to moderate risk (second sampling period- June 2016) before the restoration was completed. The restoration was completed one month prior to the third (December 2016) sampling period, so the BEHI shows an increase back to High (December 2016), before declining to a low at the final sampling period (April 2017) following the restoration. This indicates that the sites may exhibit increased erosion hazard before reaching equilibrium at a lower erosion potential.

Table 6: Mean values (+SD) of bank erosion hazard total and corresponding categorical hazard from 6 sites in D'Olive Creek watershed, Alabama.

Site	Time of Visit								Between	Within
	SP1 n=4		SP2 n=4		SP3 n=4		SP4 n=4		$p1$[3]	p
A0C	25.9 (0.3)[1]	M[2]	28.5 (6.4)[1]	M	27.9 (3.8)	M	27.9 (9.0)	M	0.821	**<0.05**
JB2D	30.5 (12.3)	H	28.4 (7.8)	M	37.8 (8.6)	H	18.4 (13.4)	L	**<0.01**	**<0.01**
JB2U	22.0 (6.4)	M	21.3 (3.2)	M	21.5 (4.5)	M	17.0 (1.1)	L	0.062	0.077
JB0C					30.0 (5.9)	H	30.7 (1.6)	H	0.274	0.816
JB1D	34.6 (6.2)	H	23.4 (4.7)	M	29.2 (9.5)	M	25.1 (6.8)	M	0.083	**<0.05**
T0C	28.8 (3.0)	M	31.0 (3.8)	H	31.0 (4.9)	H	28.1 (6.3)	M	0.109	**<0.01**

[1]n=2
[2]H= high, M=medium, L=low
[3]p-values are from analysis of variance (ANOVA) comparing variable means across the sampling periods (SP1-4). The significant ($p<0.05$) values are noted in bold.

JB2U on the other hand, maintained a moderate erosion risk throughout the restoration and also declined to a very low erosion hazard. The upstream site is considered a control as the effect of the restoration is not expected to be seen upstream of the restoration site, although there are no studies to corroborate this point. There may be an external factor in this system that caused the decline in BEHI values but that is not included within the scope of this study.

The cumulative sites did not undergo a change in the two year study period, with the exception that T0C does shift to a high risk during the two intermediate sampling periods, but returns back to a moderate risk by the fourth (final) sampling period, even though the T0C construction occurred before the two intermediate sampling periods (Table 6). This suggests that the erosion hazard at T0C may have been naturally increasing and following the restoration began declining, but it may also show changing external factors.

When analyzed individually, there was significant variation among the 83 BEHI measurements over the two year study period. Only two of the measured banks were classified with an extreme hazard (BEHI >46), three banks with a very high hazard (BEHI 40-45), 16 banks with low hazard (BEHI 10-19.9), with the majority of banks being either high or moderate risk: 22 banks were high hazard (BEHI 30-39.5) and 41 banks were moderate (BEHI 20-29.5) (Table 7).

Table 7: Mean values (+SD) of index value from bank erosion hazard variables from 83 measured banks in D'Olive Creek Watershed, Alabama.

Variable	BEHI Category					p^1
	Extreme	Very high	High	Moderate	Low	
Behi Components	n=2	n=3	n=22	n=41	n=16	
Bank:Height ratio	8.9 (0.9)	6.5 (4.8)	6.9 (3.0)	4.1 (2.9)	1.4 (1.4)	<**0.01**
Root:Depth ratio	9.5 (0.7)	3.4 (4.3)	2.7 (2.7)	0.5 (1.0)	0 (0)	<**0.01**
Root Density	8.15 (1.6)	8.9 (0.75)	6.0 (1.6)	4.3 (1.4)	2.2 (1.3)	<**0.01**
Slope Steepness	2.4 (.5)	2.4 (0.1)	3.2 (0.6)	3.3 (0.6)	2.9 (0.8)	0.057
Surface protection	10 (0)	10 (0)	5.3 (2.8)	3.6 (1.8)	1.4 (1.0)	<**0.01**

BEHI Score	48.9 (1.3)	41.5 (0.5)	34.0 (2.8)	25.9 (2.5)	17.9 (1.6)	<**0.01**

[1]p-values are from analysis of variance (ANOVA) comparing variable means across bank condition categories. Significant (< 0.05) differences between sampling periods are labeled in bold.

The Pfankuch stability ratings indicate that the reach condition at the initial sampling period was fair at three sites and good at three other sites (Table 8). At the next sampling period, all sites were classified as good (Table 8). This indicates improvement in the stream stability according to the Pfankuch method. These results also provide an alternative indication of stability than the BEHI method erosion potential results which indicate that only the JB2 sites improved in stability metrics (Table 6). For the use of this study, the more sensitive index was included within the RipHLE index in order to use stringent stability metrics to identify habitat health.

Table 8: Results of the Pfankuch index modified for sand bed streams and corresponding categorical classifications.

	Measure 1[1]	Reach Condition	Measure 2	Reach Condition
A0C	77	F[2]	66	G
JB2D	58.5	G	61.5	G
JB2U	60	G	62.5	G
JB0C	83.75	F	68.5	G
JB1D	94.5	F	65	G
T0C	70	G	55	G

[1]Note that Pfankuch index was measured twice at each site, however not corresponding to sampling period.
[2]Good (G): 50-75, fair (F): 76-96 and poor (P): >97.

Canopy Cover.

Percent canopy cover decreased at JB1D, JB0C, Y and A0C and increased at JB2U, JB2D, JB1R and T0C (Figure 26) during the study period. These changes are not consistent over site type (cumulative, upstream or downstream). The changes in both the upstream (JB2U) and reference site (Y) indicate that seasonal influences are contributing to canopy cover rather than any restoration effects. Sampling period 1 occurred between October 2015-January 2016 and sampling period 4 between February-May 2017. Those sites sampled in winter for sampling period 1 (JB2U, JB2D, JB1R and T0C) showed the increase in canopy cover (from SP1 in winter-December/January/ February to SP2 in spring- April/May). The sites sampled in fall for sampling period one show an opposite response (SP1 in fall- October/ November to SP2 in spring- February/April), indicating that total leaf senescence for the system occurred between

November and December 2015. For future use of the index, I recommend identifying RipHLE values during the same season (Fall before leaf senescence) to minimize seasonal vegetation effects.

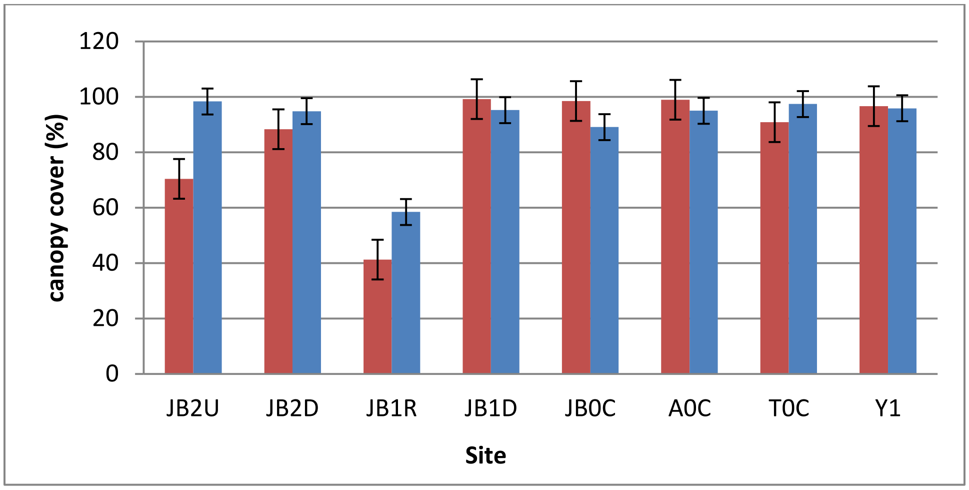


Figure 26: Changes in percent canopy cover between sampling period 1 (red) and sampling period 4 (blue) with standard error.

Basal Area.

Basal area, which is not affected by seasonal change, shows an increase over time at all sites (Figure 27). JB1R shows minimal change in basal area, which is attributed to a lack of overstory vegetation at the restoration site. A0C and T0C both show a much larger increase in basal area during the study period. While the cause is unknown, these are two cumulative sites that have not shown impacts from the restoration activity. It is possible that the rate of biomass increase (a large increase over the same time period) is due to the sites being unaffected by the restoration activity. It is possible that the stress related to the restoration impacts stunted the tree basal area growth at the other sites as shown in other studies (Shafroth, Stromberg & Patten, 2002). JB0C is the only site showing a decrease in biomass which to some extent is explained by fallen overstory species at sampling period 4.

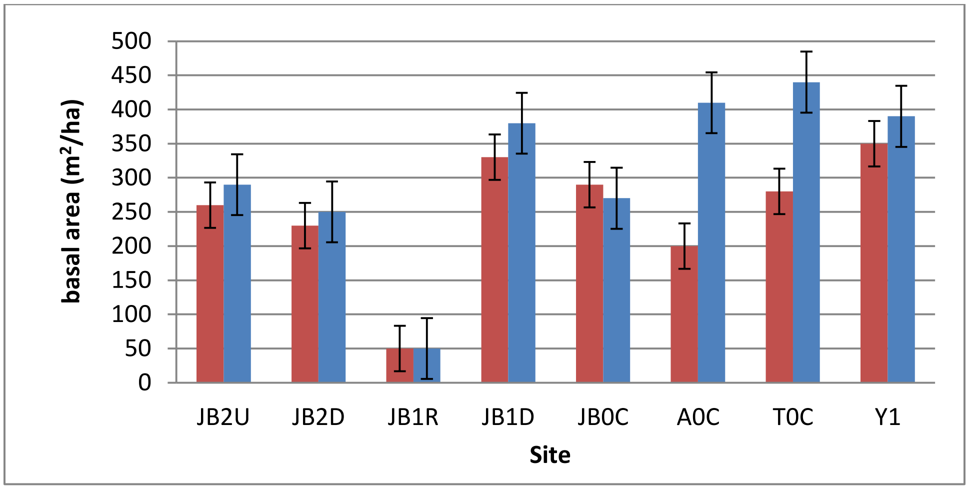


Figure 27: Changes in basal area between sampling period 1 (red) and sampling period 4 (blue) with standard error.

Bank Root Density.

Bank root densities generally decreased over the four year study period. Root density is a representation of the above ground biomass, so one would expect to see an increase in density to mirror an increase in basal area. However, these results depict the opposite (Figure 28). JB1D showed much less variability than the other sites in root density over the entire study period.

The largest decrease in root density occurred at T0C which is attributed to a fallen tree and debris after sampling period 1 that prevented measure of the left portion of the right bank. The bank density was calculated using the same method but with fewer sampling points. This could illustrate a flaw in the method since the ecosystem services provided by tree roots were still provided. The fallen tree and debris could have provided stream stability and habitat, among other ES, for macroinvertebrates, snakes and other wildlife although that was not explicitly researched within the present study.

JB2D with its consistently changing floodplains and channel as noted by the riffle cross section data showed less root density at sampling period 4, perhaps due to changing streambanks (Figure 20). JB2U also showed a decrease in bank root density which suggests that there may external factors (perhaps different flow regimes) impacting the bank root density. The literature reports that root structure is closely related to the ability to obtain moisture in times of drought, waterlogging, and scouring (Richardson et al., 2007). It is possible that this change in root density is a biological response to altered hydrology during the study period. Many of the channels were diverted and several pools infilled, thus altering the amount of water moving through the system.

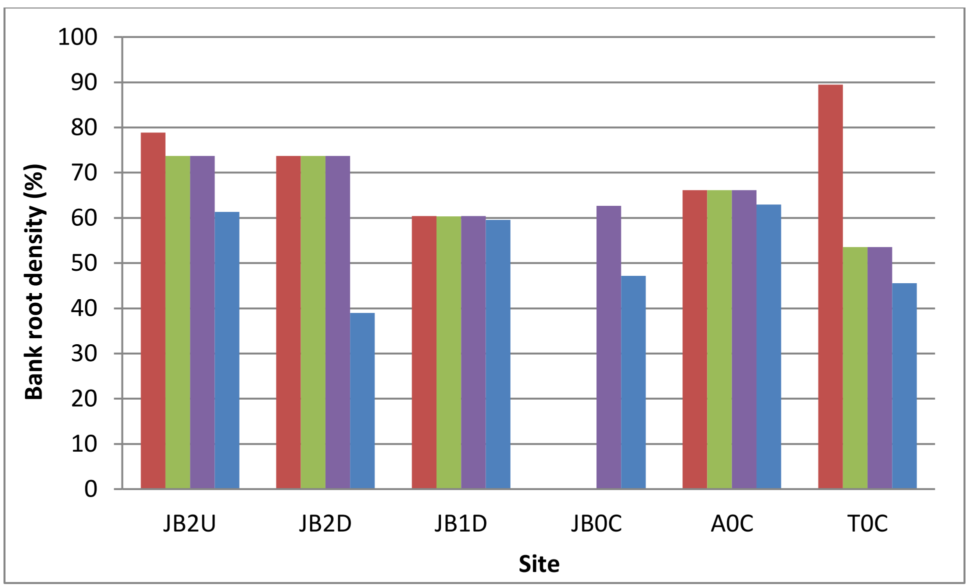


Figure 28: Changes in bank root density between sampling period 1 (red), sampling period 3 (purple) and sampling period 4 (blue). Bank root density was not consistently measured during sampling period 2 so the data is excluded.

Understory Survey- Leaf Litter.

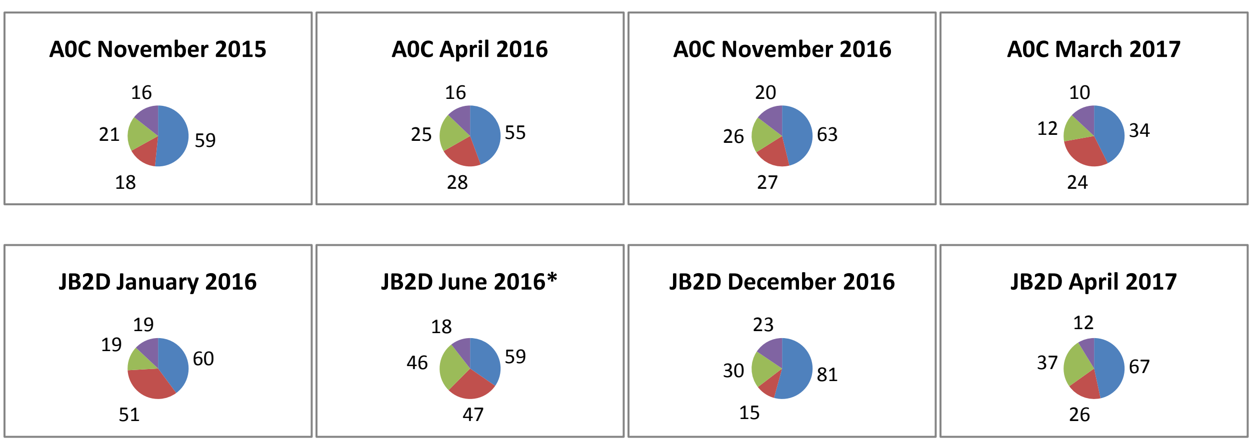
The changes among riparian habitat variables such as understory composition, streambank vegetation, and overstory composition show variability between sampling periods, much like the changing channels. The understory transects (Figure 29) show that leaf litter (LL) is the predominant ground cover at each site; LL cover ranges from 31 to 86 percent. The restoration site, JB1R, has a dominance of LL during the first two sampling periods (54%) until the vegetation becomes dominant (sampling period 3 and 4 respectively: 68 and 63% for vegetation compared to 31 and 52% for LL). The restoration sites were supplemented with plantings in order to boost the ecological transition to riparian forest, which explains the vegetation understory dominance. All other sites show at least three vertical strata (understory, midstory and overstory) meaning that the understory would receive fewer resources explaining the lower percentages of understory vegetation. Each site consistently has greater LL percentages, followed by vegetation, and then nearly equal proportions of woody material and exposed soil at every visit regardless of sampling period. However, JB0C has consistent LL percentages, but decreasing vegetation between sampling period 1 and 2 (October 2016 and March 2017) which is consistent with seasonal vegetation biomass increases and spring blooming. JB2D and T0C also show increasing in vegetation from a sampling period during Fall-Winter time period (October- January) to a spring-summer sampling period (April-June). The other variables along the understory transect did not systematically reflect seasonal changes between sampling periods.

When restoration activity is assessed, LL percentages at JB2D increase following the restoration completed in November 2016 (January 2016: 60%; June 2016: 59%; December 2016: 81%; April 2017: 67%) and vegetation decreases (January 2016: 19%; June 2016: 46%;

December 2016: 30%; April 2017: 37%) (Figure 26 and Figure 29). At the site just upstream of the JB2R restoration, JB2U, the LL average remained similar to the initial sampling period (December 2015:86%; December 2016; 83%), although the LL increased from June 2016 (66%) to December 2016 (83%). This change at JB2U reflects seasonal biomass changes due to senescence. JB0C also shows vegetation decrease just following restoration activity from both JB1 (April 2015) and JB2 (November 2016), however, the change occurs between sampling period 3 (October 2016: 51%) and sampling period 4 (March 2017: 36%) where an increase in vegetation is expected from seasonal biomass changes. This indicates that the restoration activity may have an impact on riparian vegetation by decreasing understory biomass. JB1D, unlike the other two downstream sites, showed minimal fluctuation in vegetation cover between sampling period 1: 30; SP 2:25; SP 3: 21; and SP 4: 28. Since this decline in vegetation was not systematically observed at every downstream site following restoration activity, the decline is most likely due to external factors and is not conclusive.

Sampling Period

1 2 3 4



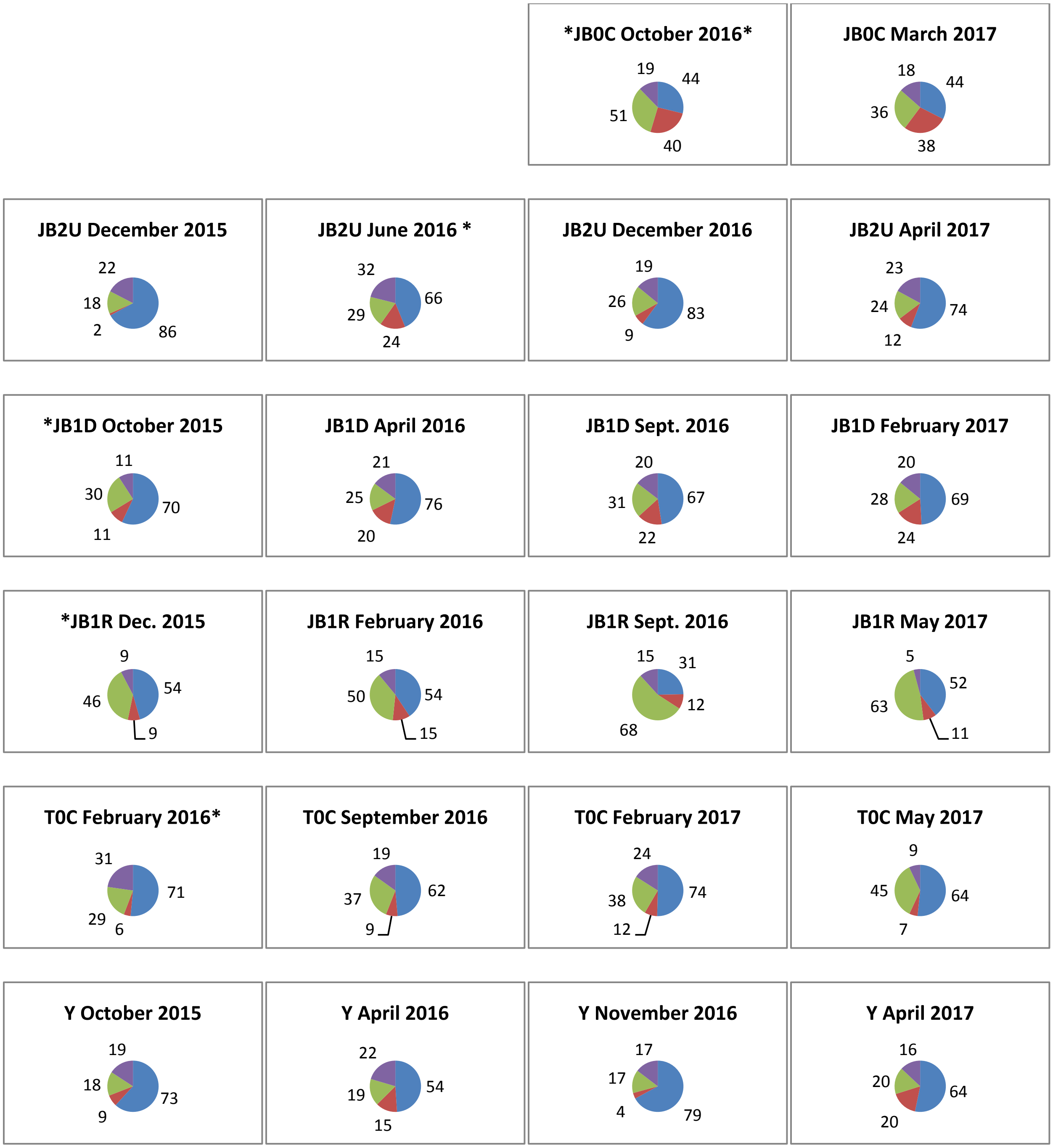


Figure 29: Relative changes in understory composition. Data labels represent percent of the transect covered by each: leaf litter (blue), exposed soil (red), vegetation (green) and woody material (purple). The asterisks denote the restoration completion relative to the sampling period date. Asterisks before title indicates before that date, after title indicates during that date. Cumulative sites will have two. Each chart may exceed 100%.

Structural Complexity.

The structural complexity was not sensitive to changes at the majority of the sites (Figure 30). The strata increased at JB2D, JB1D, Y1, A0C but decreased at T0C, JB2U and JB1R (likely due to seasonal changes in understory vegetation along the vegetation transect). The changes in the reference site (Y1) suggest that external factors contributed to the changes in strata recorded over the study period. This variable will be most useful in monitoring severely disturbed sites, such as the actual restoration sites where the overstory is removed and the strata will develop during the monitoring period. To remedy this change, the data should be collected during the same season on an annual, rather than semi-annual basis.

Figure 30: Average number of strata at each site.

Comparison of RipHLE indicators

After discussion with the expert panel there was concern that several variables within the RipHLE index were covariates. The two variables of primary concern were canopy cover and leaf litter which when compared at the first and final sampling period, did not show any correlation. The R^2 values for both sampling periods were less than 0.1 (Figure 31), which shows a lack of covariation between canopy cover and leaf litter.

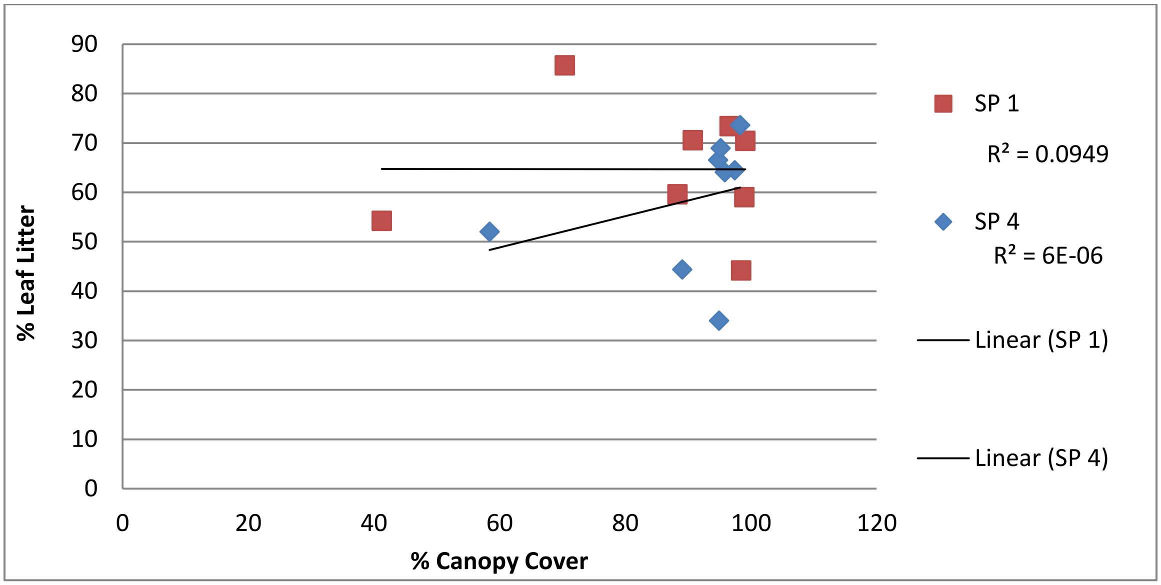


Figure 31: Comparison of percent leaf litter to percent canopy cover showing a lack of predictive power.

Even though canopy cover and leaf litter were not related, there was a clear relationship between basal area and canopy cover with R^2 values of 0.6-0.7. These were the largest R^2 values present among the RipHLE variables (Figure 32). This is likely due to the larger trees having more resources to allocate to increased leaf production, thereby increasing canopy diameter which is reflected in the densitometer reading.

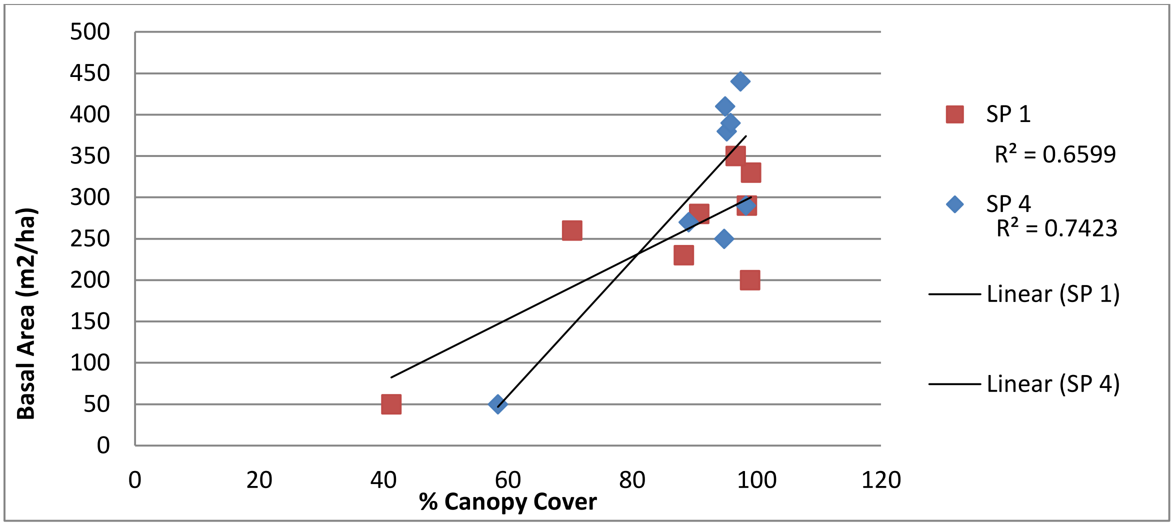


Figure 32: Comparison of basal area to percent canopy cover with intermediate R^2 values.

RipHLE Values

Although individually there is much variation among the components of the Riparian Habitat Health Level Evaluation Index (RipHLE), the patterns reflected by the values when considered with the timing of restoration events do show some correlation. Of the 22 total RipHLE measurements, 13 were classified as moderate habitat health level, and nine as poor health levels (Figure 33). JB2U and JB2D December/January 2015 (SP1) were measured one month before the start of restoration which then took ten months to complete, so the June 2016 visit (sampling period 2) occurred during active restoration. The December 2016 visit (sampling period 3) occurred within one month of the completion of the JB2R restoration. Therefore, the April 2017 visit (sampling period 4) for both JB2U and JB2D was the only visit with significant time lapse following the restoration, which at both sites show a decrease in RipHLE values. JB1D Fall 2015 occurred within two months of the restoration and the following sampling periods each had significant time lapse after the restoration, and all sampling periods starting in Spring 2016 show low moderate RipHLE values.

This decline in habitat level is shown primarily through the habitat metrics used within the RipHLE since the floodplain is inactive according to the riffle cross section data. Both JB0C measures were taken following the JB restoration construction but the Fall 2016 visit (sampling period 3) occurred during the active restoration of JB2R, so the Spring 2017 visit (sampling period 4) which shows a poor RipHLE value occurred after both restorations occurred. T0C was first visited before the restoration (sampling period 1), and then the three following sampling periods occurred after the restoration by at least four months. The two final sampling periods occurred at least 8 months after the restoration of Tiawasee Creek, and show declining RipHLE values. Along Tiawasee creek, there are many man-made culverts (Figure 34) and historical restorations disrupting stream flow from the TCR restoration which may interfere with RipHLE values.

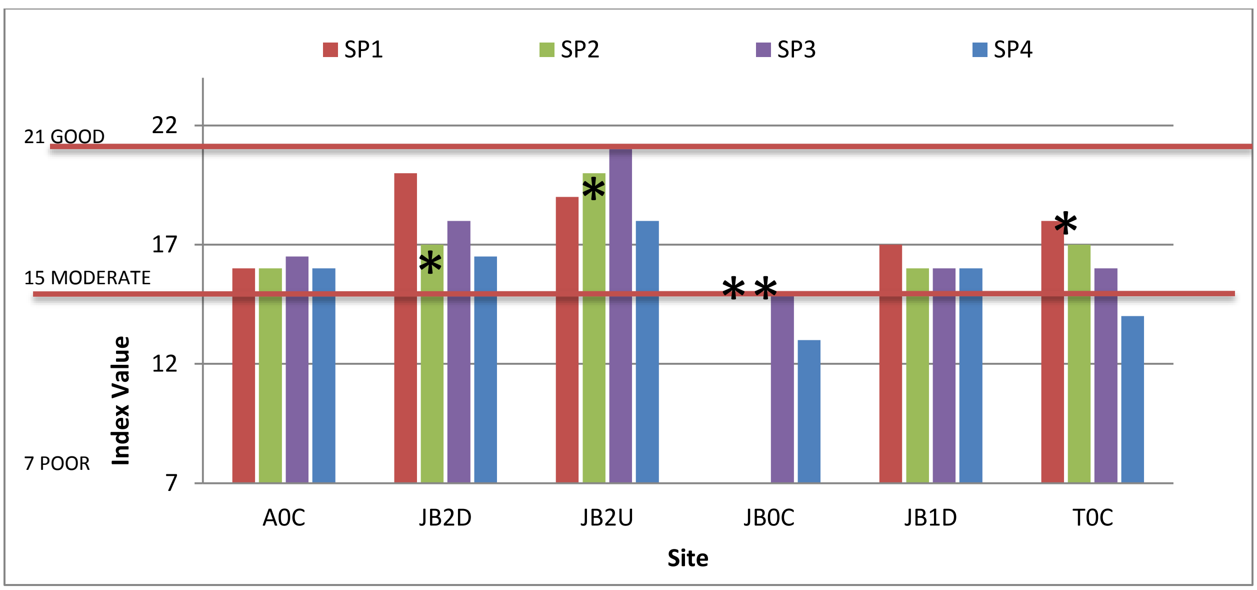


Figure 33: Riparian Habitat Level Evaluation (RipHLE) method where scores > 21 are good, 15-21 moderate, and ≤15 poor. The asterisks indicate the timing of the corresponding restorations (to the left meaning before, on top meaning during and between two bars indicating in between visits).

The A0C site shows low moderate RipHLE values at each sampling period which occurred during and after active restoration on all three restoration channels in addition to restoration on channels not included in the present study. This site was found to be very tidally influenced so these low RipHLE values may be connected to tidal variation, saltwater mixing and other variables rather than to restoration activity. This site is also located at significantly larger distances downstream from the restoration activity, impeded by culverts, historic restorations, road crossings, diversions and many other anthropogenic effects. It is likely that there have not been sufficient sampling periods to observe any changes at this site.

Figure 34: An example of the culvert and rip rap upstream from T0C in September 2015.

Overall the RipHLE values show a transient decrease in riparian intactness as visualized in Appendix I. This is also supported by Shafroth, Stromberg and Patten (2002) who found that plant community composition changed immediately following disturbance or stress related to damning activities. While understory species composition was not explicitly examined in the current study, changes in vegetation cover along the understory transect following restoration activities did reduce in many cases (Figure 29). Plant community composition is reflected in the RipHLE index through three variables: canopy cover, tree basal area and structural complexity. Overall the RipHLE values reflect what could be a transient decrease following at least a six month lag time. Note that JB2D and JB2U both seem to increase in RipHLE values following restoration, however sampling period 2 occurred within one month of the restoration. A boost in habitat level would be expected following restoration assuming that invasive species are controlled and that supplemental seeding or planting of desirable species are maintained (Richardson et al., 2007).

CONCLUSIONS

Overall, variation in the floodplain and channel geomorphology suggests that downstream sites are impacted by changes in upstream channel reconfiguration (i.e., restoration) after at least a two month lag period. The changes that are temporally consistent with the observed geomorphological changes in relation to the restoration activity include BEHI, pfankuch index values, and leaf litter percentages. Other variables such as canopy cover, vegetation, bank root density and basal area are explained by biological response (continual growth, seasonal fluctuations, or water resource efficiency). Structural complexity and buffer width remained consistent over the study period.

However, while there are many factors that can influence habitat level, the RipHLE index, which combines eight variables, did show response to restoration activity. The RipHLE values at the downstream sites show a decrease in habitat health level following restoration, when accounting for at least a six month time lag following completed channel reconstruction. These declining habitat levels are explained by increased channel depth at the downstream sites, so localized channel reconstruction does impact the immediate downstream riparian habitat level. When cumulative sites were analyzed for habitat level, there are too many external factors to conclude that any observed changes were directly resulting from channel reconstruction, although a decline in habitat level was consistently measured at all but the A0C site. This is most likely due to the distance of the site from the restoration activity; a two year study period may not be sufficient to capture any changes. This two year study period may therefore also be insufficient to show any positive improvement following channel restoration so a longer monitoring period is recommended to demonstrate any considerable ecological lift following streambank restoration (Richardson et al., 2007).

Although the study period is insufficient, patterns of covariation among the components, specifically basal area and canopy cover, emerged. Therefore in future implementation of the index, further research should be conducted on whether these patterns remain over time. In the event that these patterns continue over the longer study, I would recommend removing basal area from the index to reduce redundancy within the index. Basal area could potentially be replaced with plant diversity counts or another biological indicator though further research is needed to verify the correlation between these variables and riparian buffer health in the literature.

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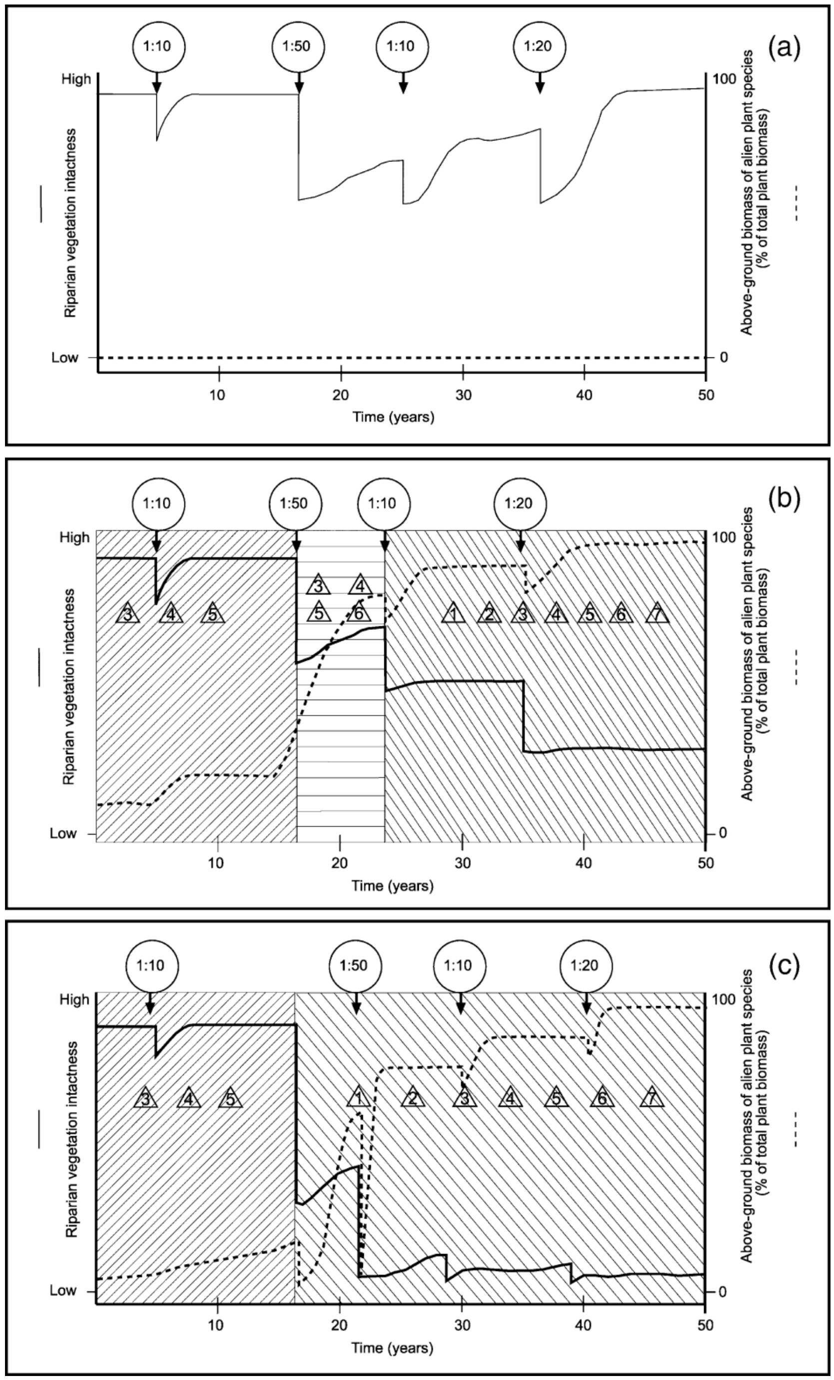
APPENDICES

Appendix I

Riparian Vegetation Response Schematic (Richardson et al., 2007)

Box 1 Schematic representation of changes in vegetation intactness in a hypothetical riparian ecosystem over 50 years. In the three scenarios (a, b, c) the system experiences four flood events: 1-in-10-years floods at years 5 and 25; a 1-in-20-years flood at year 36; and a 1-in-50-years flood at year 17. Panel (a) shows a riparian ecosystem unaffected by invasive alien plants. The system shows a high degree of resilience, with a quick return to pre-flood intactness of structure and functioning following the 1-in-50-years flood. Panel (b) depicts an ecosystem with a low representation of invasive alien species at time 0. Each successive flood event promotes further establishment and proliferation of alien plants, with an escalating effect on system intactness and resilience. After 50 years, the riparian community comprises only invasive alien plants and is severely compromised in terms of resilience and functioning. A biotic threshold induced by the invasive species occurs at year 20, and an abiotic threshold is induced at year 25. Panel (c) shows the combined effects of an engineering intervention (e.g. road or bridge construction) and invasion of alien plants. The massive human-induced disturbance at year 11 causes a substantial reduction in biomass of native species and impairs functioning; it also stimulates rapid proliferation of invasive species which benefit further from each ensuing flood event. The human-induced abiotic threshold caused by the engineering event and the biotic threshold caused by the rapid expansion of invasive species typify the rapid changes of many riparian systems driven by invasion together with other forms of stress or disturbance.

Shadings in Panels (b) and (c) indicate where fundamentally different management options are available — potential interventions for the different zones are indicated by triangles (numbers denote options described in Table 2).



Appendix II

Bank Erosion Hazard Index Worksheet

Bank Erosion Hazard Rating Guide

Stream | Reach | Date | Crew

Bank Erosion Potential	Bank Height (ft): Bankfull Height (ft):		Bank Height/ Bankfull Ht	Root Depth/ Bank Height	Root Density %	Bank Angle (Degrees)	Surface Protection%
VERY LOW		Value	1.0-1.1	1.0-0.9	100-80	0-20	100-80
		Index	1.0-1.9	1.0-1.9	1.0-1.9	1.0-1.9	1.0-1.9
		Choice	V: I:	V: I:	V: I:	V: I:	V: I:
LOW		Value	1.11-1.19	0.89-0.5	79-55	21-60	79-55
		Index	2.0-3.9	2.0-3.9	2.0-3.9	2.0-3.9	2.0-3.9
		Choice	V: I:	V: I:	V: I:	V: I:	V: I:
MODERATE		Value	1.2-1.5	0.49-0.3	54-30	61-80	54-30
		Index	4.0-5.9	4.0-5.9	4.0-5.9	4.0-5.9	4.0-5.9
		Choice	V: I:	V: I:	V: I:	V: I:	V: I:
HIGH		Value	1.6-2.0	0.29-0.15	29-15	81-90	29-15
		Index	6.0-7.9	6.0-7.9	6.0-7.9	6.0-7.9	6.0-7.9
		Choice	V: I:	V: I:	V: I:	V: I:	V: I:
VERY HIGH		Value	2.1-2.8	0.14-0.05	14-5.0	91-119	14-10
		Index	8.0-9.0	8.0-9.0	8.0-9.0	8.0-9.0	8.0-9.0
		Choice	V: I:	V: I:	V: I:	V: I:	V: I:
EXTREME		Value	>2.8	<0.05	<5	>119	<10
		Index	10	10	10	10	10
		Choice	V: I:	V: I:	V: I:	V: I:	V: I:
	V = value, I = index			**SUB-TOTAL** (Sum one index from each column)			

Bank Material Description:

Bank Materials

- **Bedrock** (Bedrock banks have very low bank erosion potential)
- **Boulders** (Banks composed of boulders have low bank erosion potential)
- **Cobble** (Subtract 10 points. If sand/gravel matrix greater than 50% of bank material, then do not adjust)
- **Gravel** (Add 5-10 points depending percentage of bank material that is composed of sand)
- **Sand** (Add 10 points)
- **Silt Clay** (+ 0: no adjustment)

BANK MATERIAL ADJUSTMENT

Stratification Comments:

Stratification

Add 5-10 points depending on position of unstable layers in relation to bankfull stage

STRATIFICATION ADJUSTMENT

VERY LOW	LOW	MODERATE	HIGH	VERY HIGH	EXTREME
5-9.5	10-19.5	20-29.5	30-39.5	40-45	46-50

Bank location description (circle one)

Straight Reach   Outside of Bend

GRAND TOTAL
BEHI RATING

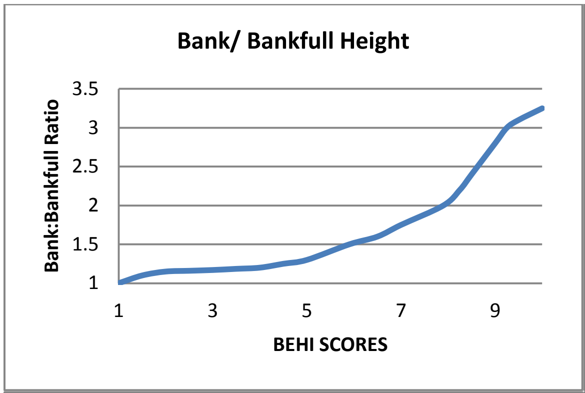
Appendix II A

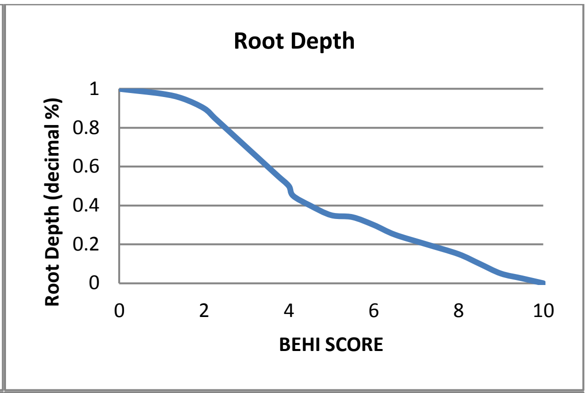
Table used to identify BEHI Component Scores

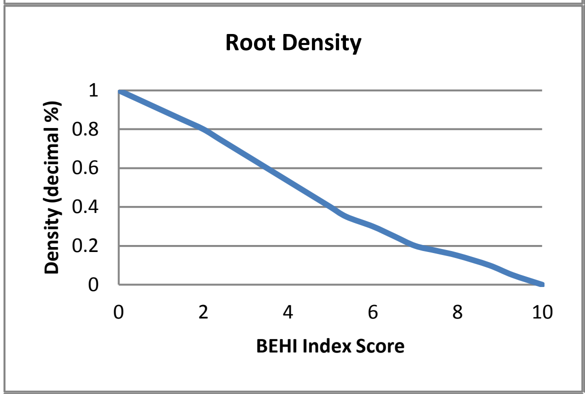
BEHI SCORES									
Bank/ Bankfull Height	Score	Root Depth	Score	Root Density	Score	Bank Angle	Score	Surface Protection	Score
1	1	1	0	1	0	120	10	1	0
1.1	1.5	0.975	1	0.95	0.5	120	9	0.95	0.5
1.15	2	0.95	1.5	0.9	1	100	8.5	0.9	1
1.16	2.5	0.9	2	0.85	1.5	90	8	0.85	1.5
1.17	3	0.85	2.25	0.8	2	85	7	0.8	2
1.185	3.5	0.8	2.5	0.75	2.375	80	6	0.75	2.375
1.2	4	0.75	2.75	0.7	2.75	75	5.25	0.7	2.75
1.25	4.5	0.7	3	0.65	3.125	72	5	0.65	3.125
1.3	5	0.65	3.25	0.6	3.5	65	4.3	0.6	3.5
1.5	5.9	0.6	3.5	0.55	3.875	60	4	0.57	3.75
1.6	6.5	0.55	3.75	0.5	4.25	55	3.7	0.55	4
1.75	7	0.5	4	0.45	4.625	50	3.5	0.5	4.3
2	7.9	0.45	4.1	0.4	5	45	3.25	0.46	4.5
2.2	8.25	0.4	4.5	0.35	5.375	40	3	0.4	5
2.4	8.5	0.35	5	0.3	6	35	2.75	0.35	5.5
2.8	9	0.34	5.5	0.25	6.5	30	2.5	0.3	6
3	9.25	0.3	6	0.2	7	25	2.25	0.25	6.5
3.1	9.5	0.25	6.5	0.175	7.5	20	2	0.2	7
3.25	10	0.2	7.25	0.15	8	15	1.75	0.175	7.5
		0.15	8	0.1	8.75	10	1.5	0.15	8
		0.1	8.5	0.05	9.3	5	1.25	0.125	8.5
		0.05	9	0	10	0	1	0.1	9
		0.025	9.5					0.1	10
		0	10						

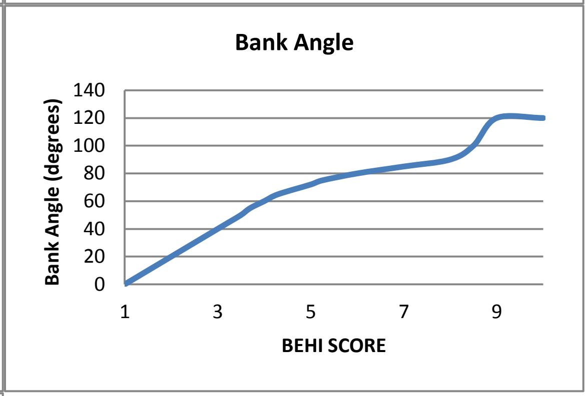
Appendix II B

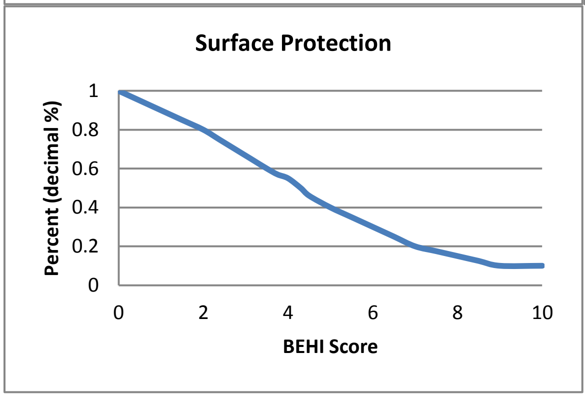
Graphical depiction of BEHI component scores


Bank/ Bankfull Height
Bank:Bankfull Ratio
3.5
3
2.5
2
1.5
1
1
3
5
7
9
BEHI SCORES


Root Depth
Root Depth (decimal %)
1
0.8
0.6
0.4
0.2
0
0
2
4
6
8
10
BEHI SCORE


Root Density
Density (decimal %)
1
0.8
0.6
0.4
0.2
0
0
2
4
6
8
10
BEHI Index Score


Bank Angle
Bank Angle (degrees)
140
120
100
80
60
40
20
0
1
3
5
7
9
BEHI SCORE


Surface Protection
Percent (decimal %)
1
0.8
0.6
0.4
0.2
0
0
2
4
6
8
10
BEHI Score

Appendix III

Near Bank Stress Methods Worksheet

Estimating Near-Bank Stress (NBS)

Stream:		Location:		Date:		Crew:	

Methods for Estimating Near-Bank Stress

(1) Transverse bar or split channel/central bar creating NBS/high velocity gradient: **Level I - Reconnaissance.**

(2) Channel pattern (Rc/W): **Level II - General Prediction.**

(3) Ratio of pool slope to average water surface slope (S_p/S): **Level II - General Prediction.**

(4) Ratio of pool slope to riffle slope (S_p/S_{rif}): **Level II - General Prediction.**

(5) Ratio of near-bank maximum depth to bankfull mean depth (d_{nb}/d_{bkf}): **Level III - Detailed Prediction.**

(6) Ratio of near-bank shear stress to bankfull shear stress (τ_{nb}/τ_{bkf}): **Level III - Detailed Prediction.**

(7) Velocity profiles/Isovels/Velocity gradient: **Level IV - Validation.**

Level I

(1)	Transverse and/or central bars - short and/or discontinuous. NBS = High/Very High
	Extensive deposition (continuous, cross channel). NBS = Extreme
	Chute cutoffs, down-valley meander migration, converging flow (Figure X). NBS = Extreme

Level II

(2)	Radius of Curvature	Bankfull Width	Ratio	Near-Bank Stress
	Rc (feet)	W_{bkf} (feet)	Rc/W	

(3)	Pool Slope	Average Slope	Ratio	Near-Bank Stress
	S_p	S	S_p/S	

Dominant Near-Bank Stress

(4)	Pool Slope	Riffle Slope	Ratio	Near-Bank Stress
	S_p	S_{rif}	S_p/S_{rif}	

Level III

(5)	Near-Bank Max Depth	Mean Depth	Ratio	Near-Bank Stress
	d_{nb} (feet)	d (feet)	d_{nb}/d	

(6)	Near-Bank Max Depth	Near-Bank Slope	Near-Bank Shear Stress	Mean Depth	Average Slope	Shear Stress	Ratio	Near-Bank Stress
	d_{nb} (feet)	S_{nb}	τ_{nb} (lb/ft^2)	d (feet)	S	τ (lb/ft^2)	τ_{nb}/τ	

Level IV

(7)	Velocity Gradient (ft/s/ft)	Near-Bank Stress

Converting Values to a Near-Bank Stress Rating

Near-Bank Stress Rating	Method Number						
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Very Low	N/A	>3.0	< 0.20	< 0.4	<1.0	<0.8	<1.0
Low		2.21 - 3.0	0.20 - 0.40	0.41 - 0.60	1.0 - 1.5	0.8 - 1.05	1.0 - 1.2
Moderate		2.01 - 2.2	0.41 - 0.60	0.61 - 0.80	1.51 - 1.8	1.06 - 1.14	1.21 - 1.6
High	See (1) Above	1.81 - 2.0	0.61 - 0.80	0.81 - 1.0	1.81 - 2.5	1.15 - 1.19	1.61 - 2.0
Very High		1.5 - 1.8	0.81 - 1.0	1.01 - 1.2	2.51 - 3.0	1.20 - 1.60	2.01 - 2.3
Extreme		< 1.5	> 1.0	> 1.2	> 3.0	> 1.6	> 2.3

Overall Near-Bank Stress Rating

Appendix IV

Pfankuch Modified Stability Index

Stream: Location: Valley Type: Observers: Date:

Location	Key	Category	Excellent Description	Excellent Rating	Good Description	Good Rating	Fair Description	Fair Rating	Poor Description	Poor Rating
Upper Banks	1	Landform slope	Bank slope gradient <30%.	2	Bank slope gradient 30–40%.	4	Bank slope gradient 40–60%.	6	Bank slope gradient > 60%.	8
	2	Mass erosion	No evidence of past or future mass erosion.	3	Infrequent. Mostly healed over. Low future potential.	6	Frequent or large, causing sediment nearly yearlong.	9	Frequent or large, causing sediment nearly yearlong OR imminent danger of same.	12
	3	Debris jam potential	Essentially absent from immediate channel area.	2	Present, but mostly small twigs and limbs.	4	Moderate to heavy amounts, mostly larger sizes.	6	Moderate to heavy amounts, predominantly larger sizes.	8
	4	Vegetative bank protection	> 90% plant density. Vigor and variety suggest a deep, dense soil-binding root mass.	3	70–90% density. Fewer species or less vigor suggest less dense or deep root mass.	6	50–70% density. Lower vigor and fewer species from a shallow, discontinuous root mass.	9	<50% density plus fewer species and less vigor indicating poor, discontinuous and shallow root mass.	12
Lower Banks	5	Channel capacity	Bank heights sufficient to contain the bankfull stage. Width/depth ratio departure from reference width/depth ratio = 10. Bank-Height Ratio (BHR) = 10.	1	Bankfull stage is contained within banks. Width/depth ratio departure from reference width/depth ratio = 10–12. Bank-Height Ratio (BHR) = 10–11.	2	Bankfull stage is not contained. Width/depth ratio departure from reference width/depth ratio = 12–14. Bank-Height Ratio (BHR) = 11–13.	3	Bankfull stage is not contained; over-bank flows are common with flows less than bankfull. Width/depth ratio departure from reference width/depth ratio > 14. Bank-Height Ratio (BHR) > 13.	4
	6	Root density[1]	Weighted root density > 60%	2	Weighted root density 40-60%.	4	Weighted root density 20-40%.	6	Weighted root density <20%.	8
	7	Obstructions to flow	Roots and logs firmly imbedded. Flow pattern w/o cutting or deposition. Stable bed.	2	Some present causing erosive cross currents and minor pool filling. Obstructions fewer and less firm.	4	Moderately frequent, unstable obstructions move with high flows causing bank cutting and pool filling.	6	Frequent obstructions and deflectors cause bank erosion yearlong. Sediment traps full, channel migration occurring.	8
	8	Cutting	Little or none. Infrequent raw banks <6".	4	Some, intermittently at outcurves and constrictions. Raw banks may be up to 12".	6	Significant. Cuts 12–24" high. Root mat overhangs and sloughing evident.	12	Almost continuous cuts, some over 24" high. Failure of overhangs frequent.	16
	9	Deposition	Little or no enlargement of channel or side channel bars.	4	Some new bar increase, mostly from coarse gravel.	8	Moderate deposition of new gravel and coarse sand on old and some new bars.	12	Extensive deposit of predominantly fine particles. Accelerated bar development.	16
Bottom	10	Sand Particle Size	Even distribution of sand particle sizes.	2	Sand particle sizes medium.	4	Sand particle sizes trending to fine sand and some silts.	6	Invasion of silt and clay particles.	8
	11	Presence of Sand Dunes	Sand dunes occupying >75% of the active bed.	2	Sand dunes occupying 50-75% of the active bed.	4	Sand dunes occupying 25-50% of the active bed.	6	Sand dunes absent. Plain bed.	8
	12	Large Woody Material	Large woody material present on 25-40% of the channel bed.	4	Large woody material present on 10-25% of the channel bed.	8	Large woody material present on 4-10% of the channel bed.	12	Large woody material sparse, < 3% on the channel bed or in excess of 40%.	16
	13	Scouring and deposition	<5% of bottom affected by scour or deposition.	6	5–30% affected. Scour at constrictions and where grades steepen. Some deposition in pools.	12	30–50% affected. Deposits and scour at obstructions, constrictions and bends. Some filling of pools.	18	More than 50% of the bottom in a state of flux or change nearly yearlong.	24
	14	Vegetation and/or Organic Material	Aquatic vegetation growth abundant; Leaf pack numerous; Limited bed scour influence on organic material.	1	Aquatic vegetation growth common; Leaf pack frequent; Low influence of bed scour on organic material.	2	Aquatic vegetation growth present; Leaf pack infrequent; Moderate influence of bed scour on organic material.	3	Aquatic vegetation growth and leaf pack absent; High influence of bed scour on organic material.	4
			Excellent Total =		Good Total =		Fair Total =		Poor Total =	

Stream Type	A1	A2	A3	A4	A5	A6	B1	B2	B3	B4	B5	B6	C1	C2	C3	C4	C5	C6	D3	D4	D5	D6
Good (Stable)	>43	>43	>90	>95	>95	>80	>45	>45	>60	>64	>68	>60	>50	>50	>85	>90	>90	>85	>107	>107	>107	>98
Fair (Mod. Unsta	44-47	44-47	91-129	96-132	96-142	81-110	46-58	46-58	61-78	65-84	69-88	61-78	51-61	51-61	86-105	91-110	91-110	86-105	108-132	108-132	108-132	99-125
Poor (Unstable)	48<	48<	130<	133<	143<	111<	59<	59<	79<	85<	89<	79<	62<	62<	106<	111<	111<	106<	133<	133<	133<	126<

Stream Type	DA3	DA4	DA5	DA6	E3	E4	E5	E6	F1	F2	F3	F4	F5	F6	G1	G2	G3	G4	G5	G6
Good (Stable)	>63	>63	>63	>63	>63	>75	>75	>63	>85	>85	>110	>110	>115	>95	>60	>60	>107	>107	>112	>107
Fair (Mod. Unsta	64-86	64-86	64-86	64-86	64-86	76-96	76-96	64-86	86-105	86-105	111-125	111-125	116-130	96-110	61-78	61-78	108-120	108-120	113-125	108-120
Poor (Unstable)	87<	87<	87<	87<	87<	97<	97<	87<	106<	106<	126<	126<	131<	111<	79<	79<	121<	121<	126<	121<

Grand Total =	
Existing Stream Type =	
*Potential Stream Type =	
Modified Channel Stability Rating =	

[1]If BHR is >1.0, weighted root density into the upper banks should be considered.

*Rating is adjusted to potential stream type, not existing.

Appendix V

NC Division of Water Quality Stream Identification Worksheet

Date:	**Project:**	**Latitude:**
Evaluator:	**Site:**	**Longitude:**
Total Points: *Stream is at least intermittent if ≥ 19 or perennial if ≥ 30**	**County:**	**Other** *e.g. Quad Name:*

A. Geomorphology (Subtotal = ________)	**Absent**	**Weak**	**Moderate**	**Strong**
1[a]. Continuous bed and bank	0	1	2	3
2. Sinuosity	0	1	2	3
3. In-channel structure: ex. riffle-pool, step-pool sequence	0	1	2	3
4. Soil texture	0	1	2	3
5. Stream sediment sorting	0	1	2	3
6. Active/relic floodplain	0	1	2	3
7. Depositional bars or benches	0	1	2	3
8. Braided channel	0	1	2	3
9. Recent alluvial deposits	0	1	2	3
10. Headcuts	0	1	2	3
11. Grade controls	0	0.5	1	1.5
12. Natural valley or drainageway	0	0.5	1	1.5
13. Second or greater order channel on existing USGS or NRCS map or other documented evidence.	No = 0		Yes = 3	

[a] Man-made ditches are not rated; see discussions in manual

B. Hydrology (Subtotal = ________)				
14. Groundwater flow or discharge	0	1	2	3
15. Water in channel and > than 48 hrs since rain	0	1	2	3
16. Leaflitter	1.5	1	0.5	0
17. Sediment on plants or debris	0	0.5	1	1.5
18. Organic debris lines or piles (Wrack lines)	0	0.5	1	1.5
19. Soil-based Evidence of seasonal high water table?	0	1	2	3

C. Biology (Subtotal = ________)				
20[b]. Fibrous roots in channel	3	2	1	0
21[b]. Rooted plants in channel	3	2	1	0
22. Crayfish	0	0.5	1	1.5
23. Bivalves/mollusks	0	1	2	3
24. Fish	0	0.5	1	1.5
25. Amphibians	0	0.5	1	1.5
26. Macrobenthos (note diversity and abundance)	0	1	2	3
27. Filamentous algae; periphyton	0	1	2	3
28. Iron oxidizing bacteria/fungus.	0	0.5	1	1.5
29[b]. Wetland plants in streambed	FAC = 0.5; FACW = 0.75; OBL = 1.5 SAV = 2.0; Other = 0			

[b] Items 20 and 21 focus on the presence of upland plants, Item 29 focuses on the presence of aquatic or wetland plants.
*perennial streams may also be identified using other methods. See p. 30 of manual.

Notes: (use back side of this form for additional notes.)

Sketch:

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[i] Image retrieved from: https://www.fws.gov/northeast/njfieldoffice/landowners.html
[ii] Image retrieved from: https://s-media-cache-ak0.pinimg.com /736x/12/79/b1/1279b10aebb848b02ebdc239eb5c2d12.jpg