Monitoring the Effectiveness of Streambank Stabilization Projects in Northeast Kansas.

by

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Abstract

Sedimentation of Federal reservoirs in Kansas has been identified as a critical issue affecting municipal and industrial water supplies, flood control, recreation, and aquatic life. Eroding streambanks are major sources of sediment. Many streambank stabilization projects have been installed over the past 20 years, but there has been very little follow-up monitoring of the effectiveness of these practices. The project goal is to quantify the environmental benefits of government-sponsored streambank stabilization and restoration projects in northeastern Kansas, with a focus on six sites in which tree ad rock revetments were installed. Several of the sites had stabilized reaches and similar un-stabilized reaches as controls. Macroinvertebrate bioassessments were conducted at two sites, on the Delaware River and Plum Creek on the Kickapoo reservation, to compare eroding and stabilized stream reaches. Biotic Index, Biological Monitoring Working Party (BMWP), Average Score per Taxon (ASPT), and Elmidae – Plecoptera – Trichoptera (EIPT) were calculated to compare the stabilized sites performance for water quality and aquatic habitat. The biological indices showed habitat quality on stabilized reaches compared to control reaches. Alfa diversity Shannon-Wiener and Simpson indices were calculated and improve in habitat quality and macroinvertebrate diversity was shown in stabilized reaches. Two new cedar revetments were established in 2017 on Little Grasshopper and Wolfley creeks. These cedar revetment installations resulted in heavy sediment deposits after high flow events with the revetments retaining 121 and 48 cubic meters, respectively. A novel method of using exposed roots was used successfully to quantify erosion on Axtell-Schmidt Dairy farm creek and Wolfley creek, where we found an average yearly erosion of 3.39 and 10.26 cm respectively. Other sites also showed reduced erosion on stabilized reaches and a development of vegetation cover along the riparian areas near the streams. Cedar revetments are

shown to be a cost-effective stabilization method on smaller streams. Also, these practices and evaluation methods are a good opportunity for community and stakeholder involvement because it is possible to train community members in the monitoring practices. It is recommended to continue monitoring these sites to compare them with the designated control in order to document long-term effects.

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Acknowledgments

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Dedication

I dedicate this to my Dad and Mom, Enrique Benitez and Victoria Nassar de Benitez.

Chapter 1 - Introduction

Rivers are dynamic components of the landscape. Over time, environmental factors such as heavy rain and geomorphological features cause changes on a river's course. Under extreme events, such as a flood, the movements of the river over the landscape increases the erosion of the streambanks. Erosion from streams may cause loss of land, influences agricultural activities, and destruction in urban areas. Also, the river's sediments may contain contaminants that potentially affect water quality and sediments may eventually reach water reservoirs reducing water availability and causing a detrimental effect in the aquatic habitat.

An excessive amount of sediment can be harmful in streams and can cause biological impairment of rivers. A report by the EPA (2018) lists 1,264 rivers that have been declared impaired by different causes such as pathogens, nutrients, metals, organic contaminant, and sediment.

In Kansas, there are 24 federal reservoirs, and residents depend on reservoirs for drinking, domestic and recreational water use. Reservoir water storage capacity is decreasing $14.3\ \mathrm{Mm}^3/$ year and on average these reservoirs have lost an average of 17% of their original capacity, some reservoirs have lost much more, such as Peery (20%) and Tuttle Creek (45%) (Rahmani et al., 2018).

Among the indicators used for national rivers and streams assessment is biological benthic macroinvertebrates, physical streambed sediments, and riparian vegetative cover. The indicators provide insight to ecological conditions that may negatively affect the health of stream biological communities (EPA, 2014).

The main idea of a Stream Restoration Project (SRP) is to modify the river so it will reach a steadier state, in which the amount of water and sediment coming in and out from the stream reaches its correct balance. As a result, all the components of the river (bankfull width, bankfull depth ratio, the sinuosity, the slope of the channel and the dominant bed material) are controlled allowing the river to develop over time without degradation or aggregation (Dutnell, 1998). “Bankfull discharge is the maximum peak flow that occurs momentarily several days in a year” (Rosgen, 1994). The project funds differ from state to state, but the main effort of the SRP is to protect water quality and availability for residents, industry, and farmers.

During the restoration process different type of stream bank stabilization practices can be done to help the stream to correct their thalweg, the stabilization practices is defined as the engineering design that changes the physical components of the river to protect the bank from erosion. Some stabilization practices may or may not help the stream restoration process, therefore, it importance on evaluating the stabilization projects.

Stabilization projects have been practiced since 1997 in the state of Oklahoma and Mississippi to reduce stream bank erosion (Dutnell, 1998). Rivers in these states have been altered by the construction of dams and straightening of channels, without the full knowledge of the effects that these changes could cause to stream habitat and the quality of water. Currently, Barden has developed over 20 streambank stabilization projects during the last 19 years in northeast Kansas (personal communication, 2018).

However, the effectiveness of river stabilization projects needs monitoring and evaluation. More comprehensive and technical knowledge will help improve the restoration projects. This literature review addresses the practices of using Eastern redcedar (*Juniperus virginiana*) trees in the revetment projects for bank protection on smaller creeks and rock vanes projects for flow deflection and bank protection on river in northeast Kansas. The review aims to assess the feasibility for using surveys of macroinvertebrates as an estimator of the effectiveness of different types of restoration practices comparing stabilized sites to non-stabilized sites.

Stream Restoration Projects (SRP)

There are many different restoration practices such as rootwad (cedar revetment) or rock weir to be applicable in many restoration projects alone or simultaneously (Brown, 2000). In the application of SRP, each stream is expected to experience problems such as bank erosion, grade control, and flow deflection or concentration. Designers engineers must design projects to encounter these specific issues. Fluvial geomorphology is used by stream restoration specialists to provide a rational understanding of stream issues through the study of the tendencies of the natural stream system and incorporating those into the implementation design. In 1994 Rosgen introduced the methodology of natural-channel-design (NCD). This methodology tries to mimic natural biological conditions of streams by significantly increasing abundance and biomass habitat for aquatic life (Ernst, Warren, & Baldigo, 2012). NCD and fluvial geomorphology knowledge are used in revetment practices to target the problem of bank erosion. Designers are constantly modifying practices and experimenting with techniques to adapt restoration challenges to the stream environment (Brown, 2000). When implementing revetment practices is important to reduce the cost of the revetment with materials based on the availability in the area. Examples in Kansas include the use of native Kansas rock and redcedar trees.

Redcedar Revetment

Redcedar revetments work by decreasing the water velocity during high flows and deflecting the current at low flows. Both functions reduce the force in which the water hits the bank. The effect of a cedar revetment is observed over time when the vertical bank will attain a more gradual slope as the sediment material is caught and held by the revetment (Goard, 2006). An advantage of using redcedar trees is their singular characteristic of having many multiple branches and fine twigs that serve as a filter that can capture sediments; Also, redcedar trees have decay resistant wood and are found in abundance throughout Kansas.

Recommendation for Installation of Redcedar revetments

The ideal time for a revetment to be installed is in spring before a heavy rain event. Larger trees are better at covering the bank but harder to get into the position. Trees from 4.6 to 6 meters is an appropriate size. Trees should be cut close to the day of the establishment in order to maintain the density of foliage on the twigs. The revetment should start from downstream of the selected outer meander bend. It is necessary to fill gaps with small cedar trees cabled to the larger ones already in place. The revetment should be stablished as shown in Figure 1. It is also recommended to plan a buffer zone (Goard, 2006).

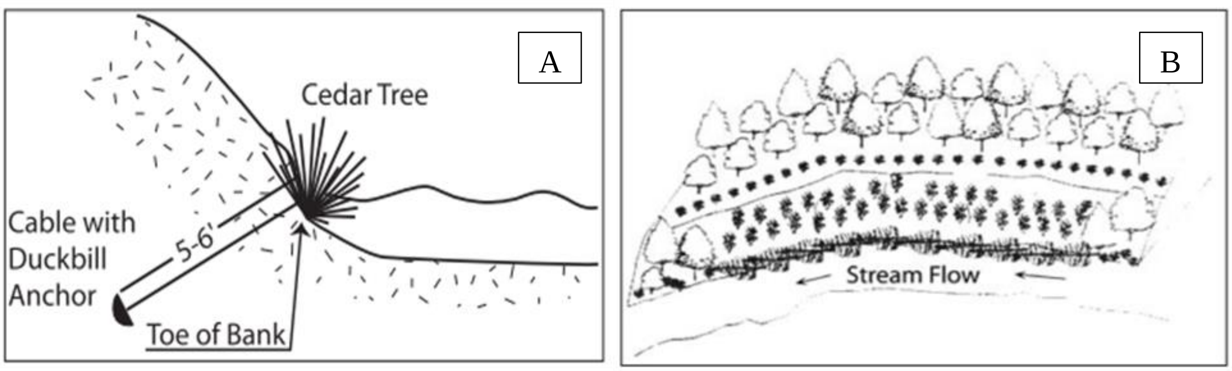

Figure 1 A. Cross section of redcedar revetment drawing the placement of tree in the toe of the bank. **B.** Diagram showing the plans for a final revetment project with vegetation cover and a riparian buffer. Taken from Goard (2006).

Rock Revetment

In rock revetments, rocks are placed as vanes or weirs located downstream of the point where the stream flow encounters the streambank at acute angles (Figure 2). Vanes protect the streambank by redirecting the stream flow away from the streambank and towards the center of the channel helping to improve in-stream habitat through scouring, oxygenation, and cover (Harman & Smith, 2017). According to the USDA (2013) rock revetments are more effective in gravel and cobble-bed stream with slopes less than 3%, and they may be not adequate for sand bed streams, but Kansas has many rock revetments working well on sand bed stream on the Big Blue River and Little Blue River (Balch, 2007).

More Stream stabilization structures.

There are many different types of structures according to the need of the rivers for flow resistance and energy dissipation. These structures include Stream Barbs, Vanes, Bendway weirs, Spur Dikes, Toe Wood, Log Jams, Rock Walls and Riprap. Stream Barbs, Vanes, and Bendways weirs shift the helicoidal flow patterns away from the banks by forcing overtopping flow perpendicular to the structure alignment, this way diminishing flow velocity near the bank (USDA-NRCS, 2012). These structures are built using rock and are low structures that get completely overtop the during a channel forming flow event (Figure 2). They can be built in spaces in between the exposed rock near the middle of the channel for fish passages, called Porous weirs and Solid weirs are continuous, both are planned in a “V” or “U” with the orientation upstream (US Department of Agriculture, 2013). The stream has a path with the maximum depth and velocity, referred to as, thalweg. This path can move along the streambed near to the edges of the bank, so the main idea of these structures is to direct the thalweg in to the center of the stream.

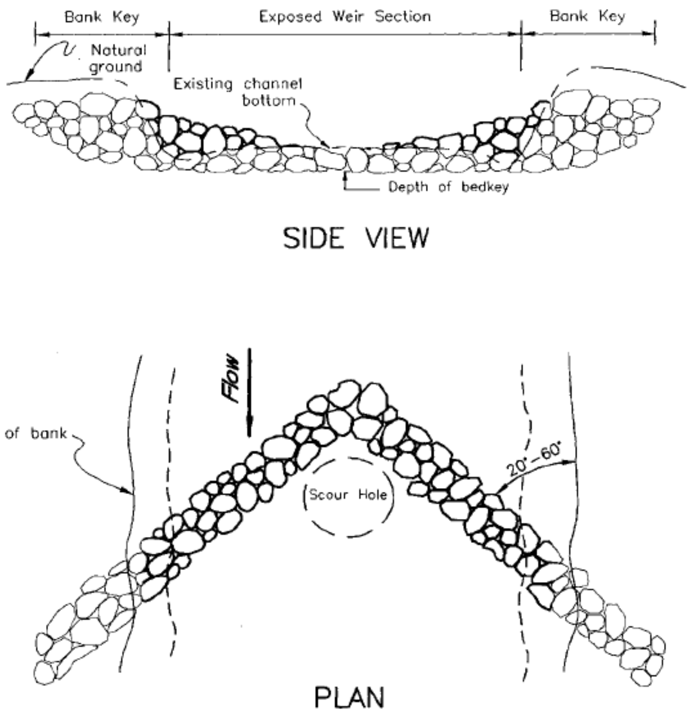

Figure 2. Plan for building a rock weir. The figure is taken from the US Department of Agriculture, (2013).

A spur dike (Figure 3A) are a salient feature from the bank into the channel, with a horizontal surface that is above the high flow water level. Toe wood is the construction of a bankfull bench using primarily un-milled woods for structure, soil lifts to create bankfull surface and vegetation. The logjam (Figure 3C) is log structures that deflect flows, give flow resistance to the bank and increase deposition. Rock wall and rip rap (Figures 3B and 3D) are basic bank protection tools used where infrastructure structure protection is required near buildings or roads (USDA-NRCS, 2012). Some structures are better for promoting vegetation, such as, log jams and cedar revetment that leave aside space for vegetation growth, or structure like porous weirs that protect aquatic habitat by leaving available space for them. Other structures like riprap can have negative ecological consequences on streams because they stop the sediment and wood input and reduce vegetation growth (Reid & Church, 2015).

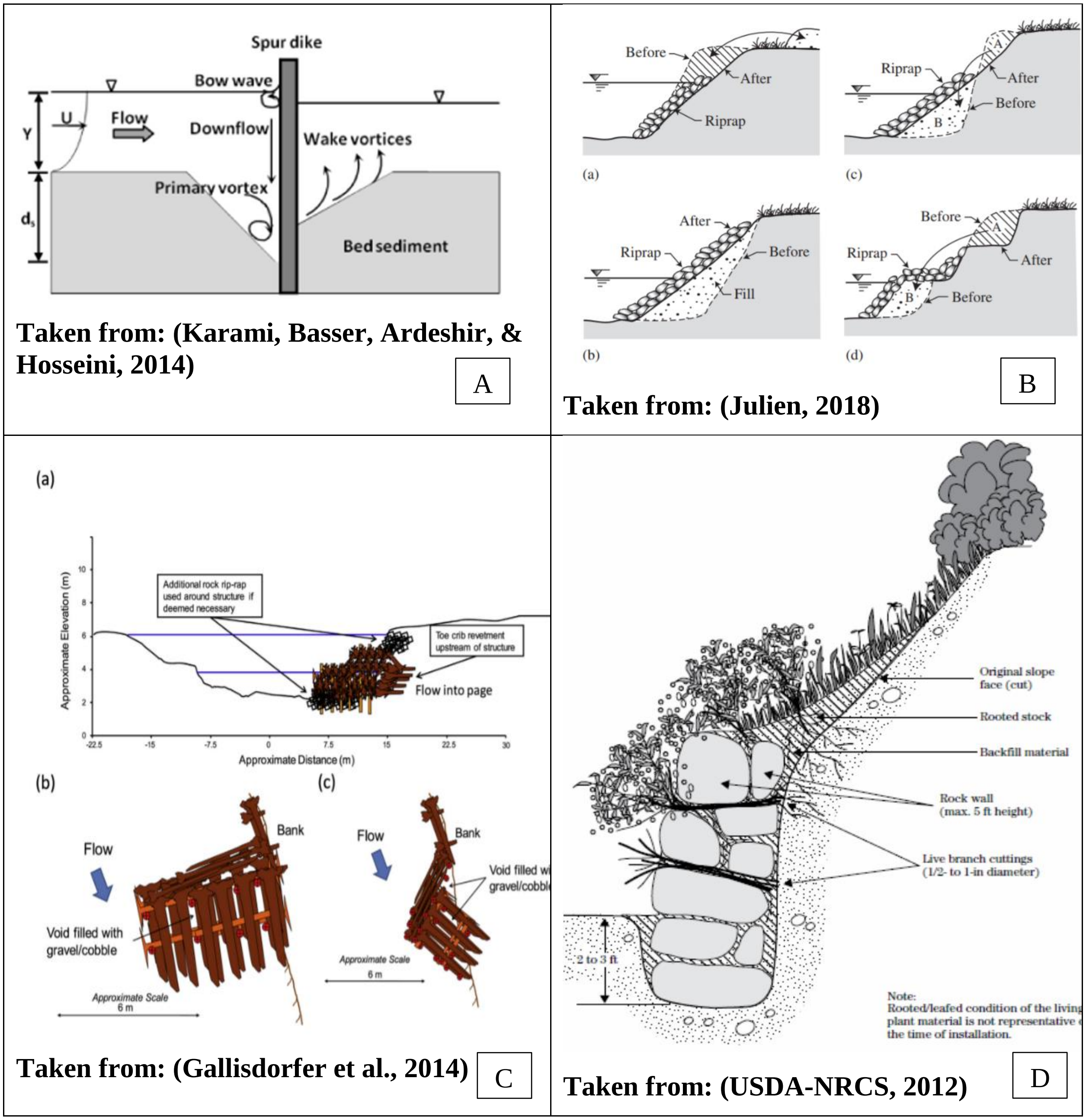

Figure 3. Different streambank stabilization practices. **A.** Spur dikes, **B.** Riprap, **C.** Logjam **D.** Rock wall.

Monitoring of Restoration Practices

A previous study on restoration practices by Kondolf and Micheli (1995) established the general objectives that a stream restoration project should accomplish. Their evaluation criteria required that the channel bankfull and stability should be maintained, there should be an improvement of aquatic habitat, riparian habitat, and water quality, and there should be opportunities for recreation and community involvement. To evaluate channel stability and bankfull geomorphology the Rosgen (1994) classification system is generally used for monitoring. This classification is based on the assumption that dynamically stable streams have a morphology that provides appropriate distribution of flow energy during storm events. The Rosgen classification identifies eight dependent variables that affect the stability of a stream: channel width, channel depth, flow velocity, discharge, channel slope, the roughness of channel materials, sediment load and sediment particle size distribution. The measuring of these variables helps to monitor and evaluate stream health, because it gives the parameter for a healthy steady stream. However, Brown, (2000) after evaluating 24 past restoration projects concludes that a study of the aquatic communities' response to the stream restoration is also a good measurement of success.

Little updated monitoring of success or failure of streambank stabilization practices has been reported until 2017 when Dave and Mittelstet (2017) quantified the effectiveness of several practices such as jetties, cedar tree revetments, rock vane, rock toe, and retaining wall and gravel banks on the Cedar River in Nebraska. They quantified erosion and deposition from 1993-2016 using National Agricultural Imagery Program (NAIP) aerial photographs in the program ArcGIS 10.3. The median streambank erosion rate was significantly different between stabilized and control reaches, with control sites showing more erosion. A flood that occurred in 2010 caused significantly more erosion on control sites. Jetties successfully increased deposition upstream as compared with the control sites. Even though tree revetment was least successful post stabilization, other practices presented more erosion during and post-flood, like rock vanes.

Comparing the most cost-effective practice jetties were the most cost-effective practice with a reduction of 2.93% per dollar spent per meter compared to Tree revetment that only had a reduction in average 0.50%. They found that the most cost-effective practices where jetties, rock toe and tree revetment.

Uses of Macroinvertebrates to Assess Water Quality.

Benthic macroinvertebrates are commonly used to quantify water quality because many macroinvertebrates start to develop their early formation stages in the stream before migration to terrestrial zones. The use of macroinvertebrates to evaluate water quality is a type of bioassessment (Barbour, 1996). The main idea of macroinvertebrate biotic indices is that the assessment of stream integrity could be achieved by evaluating the aquatic invertebrate community structure (Dos Santos, et al. 2011).

Using biotic indices for bio-assessing streams started in Europe prior to use in the United States by the Biological Monitoring Working Party (BMWP) (Armitage, Moss, Wright, & Furse, 1983). These indices are based on the absence or presence of species that give information about pollution status. Bio-indices have been recognized as suitable criteria for understanding the quality of the aquatic environment. The BMWP system considers the sensitivity of invertebrates to pollution: macroinvertebrates families are assigned a score between 1to10, accordingly. The score is the sum of the values for all families and values higher than 100 are associated with clean streams. Another index is the Average Score Per Taxon (ASPT) and is determined by dividing the BMWP by the number of taxa present. A high ASPT is considered a clean reach. Two more commonly used indices are, Ephemeroptera-Plecoptera-Trichoptera (EPT) and Elmidae-Plecoptera-Trichoptera (EIPT), which are similar to each other but differ on the specific pollution sensitive taxa used to measure water quality.

Specifically, ASPT is a useful tool for monitoring areas impacted by diffuse factors such as, patchy landscapes, pastures, and crops. The EIPT is more suitable for areas that receive significant sediment loads (Fossati, Wasson, Héry, Salinas, & Marin, 2001), and BMWP performs the best in highly affected areas of sediment (Dos Santos et al., 2011). The BMWP is mainly used for establishing a baseline for rivers, and it needs to be adjusted to the specific area that is being used to achieve higher versatility than the other indices.

Habitat Changes and Macroinvertebrates.

The dynamic nature of streams sustains ecosystem habitat, since some bank erosion is natural and provides a sediment source that creates riparian habitat, maintains diverse structure and habitat functions. Riparian vegetation promotes bank stability and contributes large woody debris, while bank erosion modulates changes in channel morphology and pattern (Florsheim, Mount, & Chin, 2008).

Physical stressors like poor riparian vegetative cover and high levels of riparian disturbance are widespread stressors, contributing to erosion and allowing more pollutants to enter the waterway. The sediments can smother the habitat gradually where many aquatic organisms live or breed. Poor biological condition is twice as likely in rivers and streams with excessive levels of streambed sediments (EPA, 2014).

Mažeika, et al. (2004) suggested that changes in the physical structure and condition of the stream influence the biological communities and processes. If the physical structure of the stream affects the biological communities, it will reflect poor water quality and reduce the biodiversity of benthic insects. However, 60% of the sites assessed by Brown (2000) did not achieve all objectives for habitat enhancement successfully.

When erosion is occurring in a stream, an erosional zone and a depositional zone form along the meander of the stream. As a result, the sediment will move from the erosion zone to the depositional zone. The interaction between erosional zones and depositional zones and nutrients (soluble and particulate) and sediment can negatively affect aquatic life. Riffles zones are typified by the contents at the bottom of the stream (stones, gravel, and sand) and because the water flow faster, sediment particles move on the streambed. This type of zone is often expected to have diverse insect fauna. On the other hand, the depositional zones are characterized by slower water and small sediment particles. In addition, these zones have a fewer number of insect than those found in erosional zones but may have many individuals of dominant species (McCafferty, 1998).

Ernst et al., (2012) investigated the influence of the stream-reach geomorphic state on in-stream habitat and aquatic macroinvertebrate communities and compared measures of habitat conditions and macroinvertebrate community composition in stable and unstable stream reaches. Their results suggest that the use of this macroinvertebrate bioassessment is not effective since it does not necessarily respond to physical changes in the rivers. On the other hand, Sudduth and Meyer, (2006) indicated the results always tend to be beneficial for restoration projects that imitate the macroinvertebrate habitat.

On the overall sites studied by Sudduth et al. (2006) and Mažeika et al., (2004), sites that used rock revetment had smaller number of macroinvertebrates compare to sites that used tree revetment; however, the sites using rock revetment had higher numbers of macroinvertebrates compare to the control sites. These reaches had an length for 250 to 330 meters on a basin water catchment of 18 kilometers (Sudduth & Meyer, 2006). This result may suggest that tree revetments tend to have more organic habitat and diversity compared to rock revetment. Mažeika et al., (2004) found no difference in macroinvertebrates abundance between stabilized and non-stabilized sites but the insect communities on stabilized sites tended to be dominated by higher percentages EPT.

Quantifying erosion rates

The erosion of streams and river is a natural process in landscapes, so the quantifying of erosion rates is more commonly done by researchers unlike the monitoring of streambank stabilization projects. There are different methods to quantify erosion rates, this section describes two such methods. The first method is a classical technique while the second is novel to this research.

Reinforcement bar for bank erosion analysis

Besides the new techniques of quantifying or monitoring stream bank erosion, several researchers have used erosion pins to date historical erosion rates (Couper, Stott, & Maddock, 2002). Erosion pins are a classic and useful tool to collect data of the spatial and temporal variations of streambank since erosion is not evenly distributed along the bank. These pins are inserted horizontally into the eroding bank (Thorne, 1981).

Erosion pins are used to measure on-site erosion, unlike the other techniques that use spatial imaginary to collect streambank erosion data. The erosion pins provide a reference point to interpret from its exposed length the bank erosion. The first pins were made from wood however, they decayed, so plastic or metal is more commonly used, although this last one may suffer from severe corrosion (Haigh, 1977). Concrete reinforcement bar (rebar) is commonly used as erosion pins.

Pope & Odhiambo (2014) used pins to analyze erosion in the Ni reservoir located in, Virginia, USA. They discovered that severity of bank erosion increases with proximity to the dam of Cool spring lake and that the results varied in respect to the different heights that the pins were established. The use of pins can have some disadvantages as well, such as, disturbance of the bank when inserting the pins, something that is almost impossible to avoid when putting a rebar on a slope (Haigh, 1977). Hupp et al. (2009) documented bank erosion along the Roanoke River in North Carolina. This large-scale study used more than 700 bank erosion pins installed along 66 bank transects, concluding that erosion rates are greater in the middle reaches with a mean of 63 mm/yr. Erosion pins have been used successfully to document erosion by several researchers.

Dendrogeomorphic approach to estimate erosion rates

Dendrogeomorphology is the science that combines dendrology (the study of trees) with geomorphology (the study of the surface of the earth) to estimate with accuracy soil erosion or deposition (Ballesteros-Cánovas et al., 2013). Dendrogeomorphology uses the growth anomalies in tree-ring records from roots instead of stem samples to infer about past geomorphic conditions. In 1960’s exposed roots were used as an indicator of ground surface at the time of germination of *Pinus aristata* in California (Stoffel, et al. 2017), now the method has evolved for the analysis of sheet-erosion (Rubiales, et al. 2008) and to estimate erosion rate of streambanks (Stotts, et al 2014) .

Dendrogeomorphology is a useful tool to analyze long-term historical erosion, as this method can obtain more information on the distribution and timing of past flood events, which could help communities prepare for severe climate events (Stoffel et al., 2017). For example, Spain had been using dendrogeomorphic research for successfully applied flood risk analysis (Díez-Herrero, Ballesteros, Ruiz-Villanueva, & Bodoque, 2013).

When geomorphic changes happen in the landscape, it can leave tree roots exposed. This exposure causes the loss of the soil cover of the root and cellular anatomical changes occur in the growth rings. The anatomical changes of the growth rings are explained by the variation in soil temperature and humidity, reduction in pressure of soil cover and mechanical stress (Corona et al., 2011). In more simple words, without the soil cover, the roots produce protective bark, and scars can be seen in cross sections cut of the root (Figure 4).

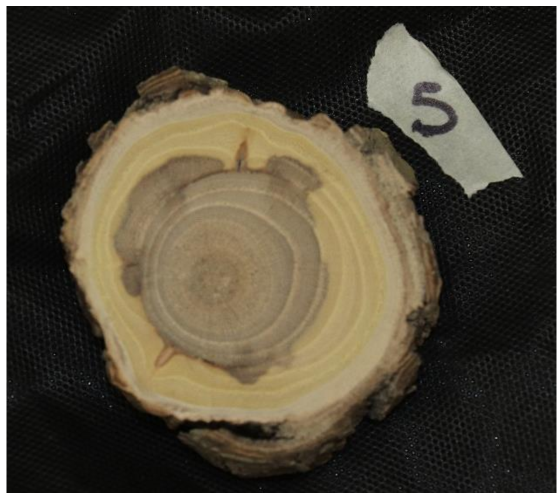

Figure 4. Photo showing cross section cut of a root and it scars.

Analysis can be done from a macroscopic or a microscopic level. The microscopic technique can yield more information about the floods but is much more expensive and requires special equipment to prepare the examining slides. For microscopic analysis it is necessary to cut microsections from the root sample and follow the procedures from Schweingruber (2006) for microsection slide preparation, a highly equipped laboratory is necessary to obtain the perfect image of the root cells in order to and find the scars. Although, both macroscopic and microscopic analysis may be used together in soil erosion analysis (Rubiales et al., 2008; Corona et al., 2011; Sun, Wang, & Hong, 2014) on a limited budget macroscopic analysis can be used alone to calculate annual erosion rates.

This literature review supports the use of macroinvertebrates to evaluate sediment sensitivity in streams subject to human influences like pastures and crops. Although, macroinvertebrates do not show a strong correlation with habitat changes, the literature expresses the importance of monitoring water quality and aquatic habitat conditions in streams that have been stabilized. Monitoring the sites is important because biological recovery is expected to occur at a slow pace over the years. This literature review also supports the use of the erosion pin method and the newer technique, using exposed roots to measure and quantify streambank erosion.

The use of restoration practices is becoming more popular over time and helps to mitigate the effects of extreme water events (Brown, 2000). They might not directly improve the quality of the water, but these practices help to ensure the availability of water by reducing sediment loads entering reservoirs. Even when conclusions concerning the best practice for achieving a good ecological status are unclear, the literature provides information about which practices are most likely to succeed and how to manage further monitoring of these specific streams. Continuing research and assessment of past restoration practices will improve the design of restoration practices, allowing for improved decisions on implementation and assessment of the practices.

Purpose of study

Kansas water supply reservoirs have an annual depletion rate range from 0.02% to 0.84% and an estimated range of 0.5 to 1% of capacity loss (Rahmani et al., 2018). A sediment-filled reservoir is not able to support municipal and industrial water use and will not be able to meet demands during drought events (Juracek, 2015). Therefore, it is necessary to validate strategies and actions to provide a reliable water supply. Achieving a reliable water supply will include actions to conserve and extend the useful life of reservoir storage. The Kansas Water Office stated in 2012 “Kansas needs plans to improve water quality; reduce vulnerability to extreme events, such as from floods and drought; develop and maintain water infrastructure; loss of arable land; and improve recreational opportunities available to our citizens" (KWO, 2013).

Despite the increasing commitment of resources to stream restoration, post-monitoring of stream restoration projects has generally been neglected and few project have undergone assessments to determine which perform best under various conditions (Brown, 2000). Evaluation is useful for determining whether project objectives have been satisfied, these evaluations can also help stakeholders to improve restoration practices (Kondolf & Micheli, 1995). If the bioassessment is conducted, usually macroinvertebrates are used to determine the environmental benefit. Just a few studies especially focus on quantification of streambank erosion as a part of monitoring the stabilization (Dave & Mittelstet, 2017). Lenhart (2018) established that farming is the most important reason for protecting water quality according to population surveys, it also establishes that farmers need the evidence that practices will work in the long-term for cost-benefit motivated decisions. Lenhart’s surveys also suggest that stakeholders need the demonstration projects and the opportunity to have an open information exchange with researchers and the government. This project aimed to have an open relationship with landowners and used newly stabilized sites as successful demonstration projects for future extension education.

The purpose of this study was to quantify the environmental benefits of government-sponsored streambank stabilization and restoration projects in northeast Kansas, with a focus on sites within the Kickapoo Tribe in Kansas and Prairie Band Potawatomi Nation Indian

Reservations. Specific objectives were to document the erosion and deposition rate of existing streambank stabilization sites and conduct bio-assessment surveys to document aquatic organism presence at the stabilized sites compared to nearby un-stabilized reaches.

Chapter 2 - Materials and Methods

The following chapter will review the materials and study methods used in the assessment of streambank stabilization projects in northeast Kansas. Two main evaluations were conducted: the evaluation of capability of the stream for supporting aquatic life using bioassessment of macroinvertebrates and measurements of erosion rates using exposed tree roots and erosion pins. The evaluation analysis methods correspond to the restoration projects objectives; channel stability, improve aquatic and riparian habitat while encouraging community involvement, as suggested by Kondolf & Micheli, 1995. This chapter will also give a brief description of the selected sites and locations.

Description of sites.

Seven sites were selected in five main locations distributed in the northeast Kansas: Potawatomi Reservation, Kickapoo Reservation, Atchison County, Marshall County, Jackson County, and Nemaha County (Table 1). These streams have been stabilized with redcedar revetment or rock revetment in the past years. Several sites have a stabilized reach and a non-stabilized reach that serves as a control for the study.

Figure 5 shows the study site of Axtell Dairy farm; the creek is located Marshall County at 30°49.695' North and 96°16.397'West. This creek has a drainage area of 14.09 square kilometers and a mean annual precipitation of 84.84 centimeters. A cedar revetment was installed in 2007, the revetment was placed to prevent further incision into the farmstead and the street. This site is being monitored with root exposure methodology. There is a narrow area of tree and vegetation cover around the creek and dairy cow pasture around the creek.

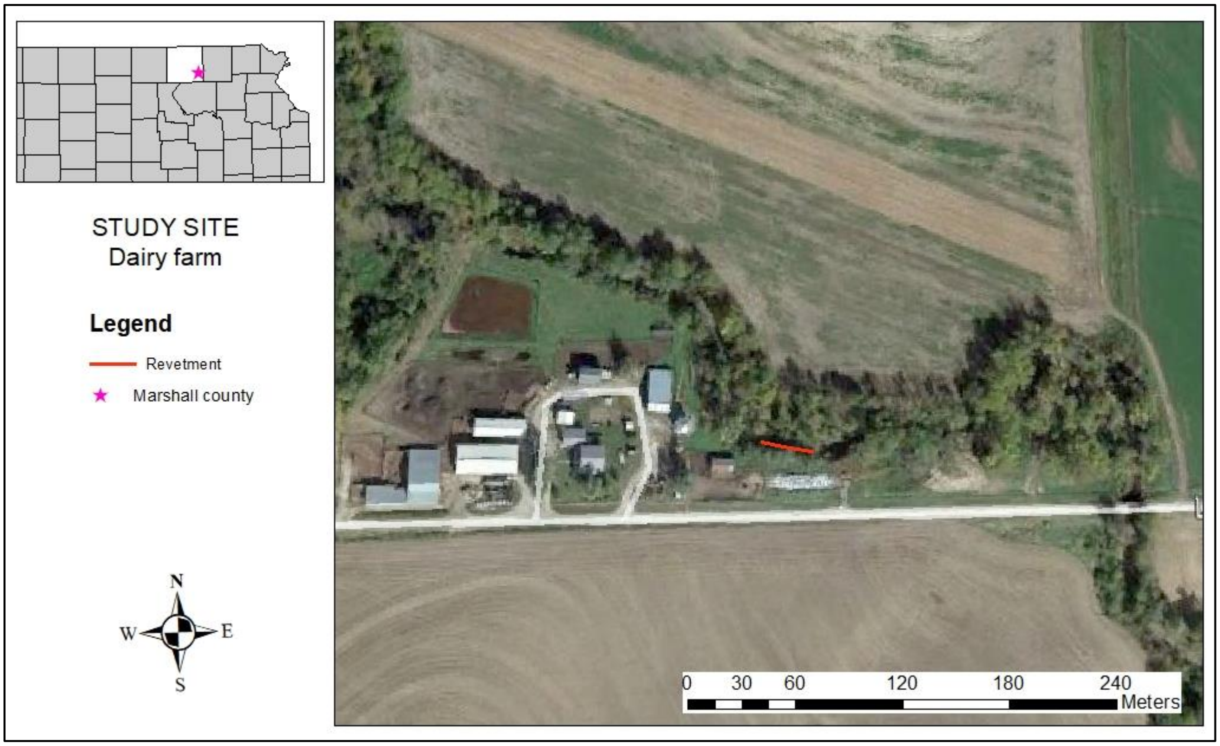

Figure 5. Map of location with stabilized did not have control site in the Axtell Dairy farm.

Figure 6. Map of location with stabilized and control sites in the Little Soldier creek.

Little Soldier creek (Figure 6) is located in Jackson County inside the Potawatomi Reservation at 39°21.275’North and 95°48.107’West. This creek has a drainage area of 25.25 square kilometers and a mean annual precipitation of 90.93 centimeters. A cedar revetment was installed in March 2000, it has a length of 60.96 meters. This site was monitored using reinforcement bars.

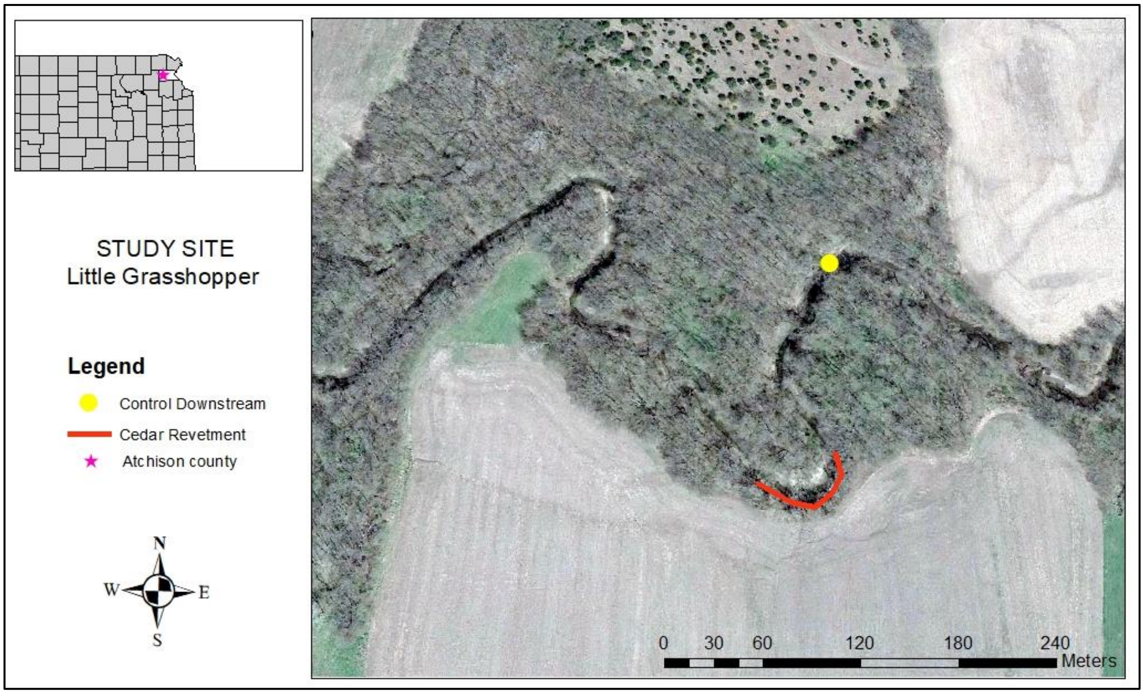

Figure 7. Map of location with stabilized and control sites in the Little Grasshopper Creek.

Little Grasshopper Creek (Figure 7) is located in Atchison County at 39°34.935’ North and 95°26.337’West. This creek has a drainage area of 58.20 square kilometers and a mean annual precipitation of 92.96 centimeters. The texture of the site soil is silt loam. This site was monitored using reinforcement bars. We installed a new cedar revetment in March, 2017. This revetment is 115.82 meters long and the cut bank is 6.10 meters high. We placed the first tree at the base of the eroding bank, with the cut end of the tree pointing upstream. An anchor was used to attach a 3.6 meters cable where the top of the first tree was located. A tractor was used to move the trees to the bank. Once, the tree is located at the toe of the bank, we held it against the bank and attached the cable to the top of the tree using a cable clamp. Another anchor was placed where the butt of the second tree was located. Then we anchored the tree tightly against the bank moving the next tree into place with its top overlapped by 1/3 the butt of the first tree. Secure cable was used to anchor the butt of the first tree to the top of the second tree. This process continued upstream until the entire base of the streambank was covered with trees. *Salix sp.* stakes were placed above the cedar tree revetment in order to promote vegetation cover (Figure 8).

Figure 8. Photo of cedar revetment establishment on Wolfley Creek.

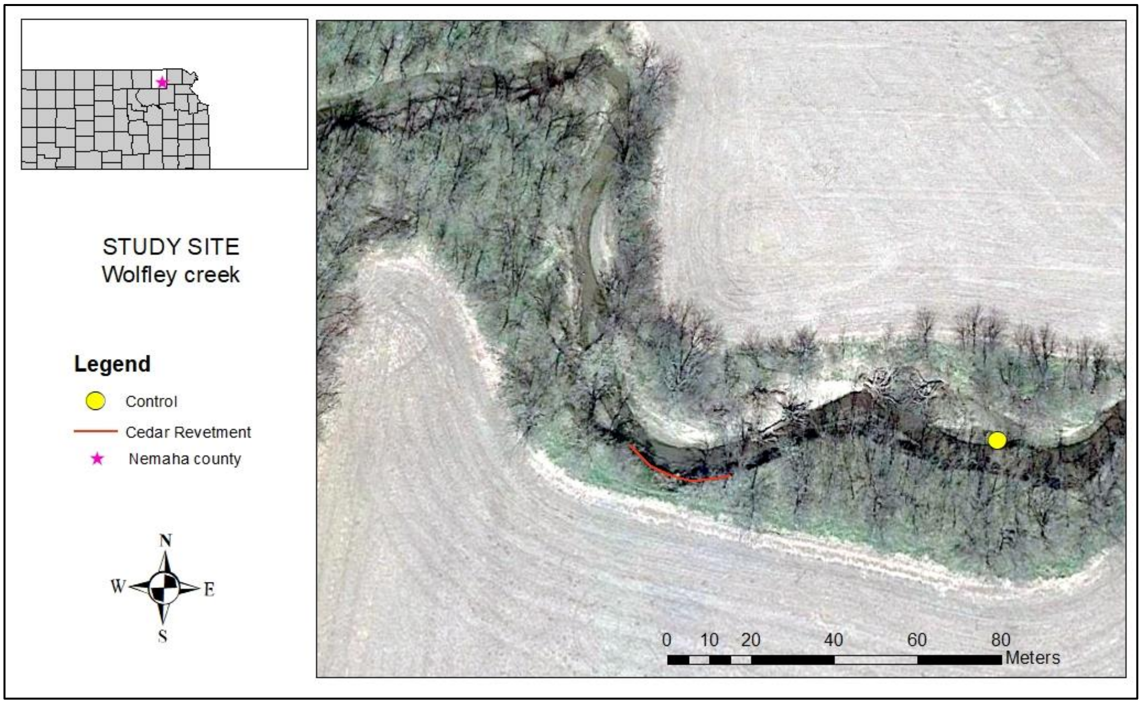

Figure 9. Map of location with stabilized and control sites in the Wolfley creek.

Wolfley Creek (Figure 9) is located in Nemaha County at 39°43.040’North and 95°52.604’West. This creek has a drainage area of 43.98 square kilometers and a mean annual precipitation of 87.38 centimeters. We also installed a new revetment in April, 2017. This revetment is 88.4 meters long and 9.14 meters in depth. The texture of the soils is silt loam. This site was being monitored by reinforcement bars and macro analysis of exposed roots.

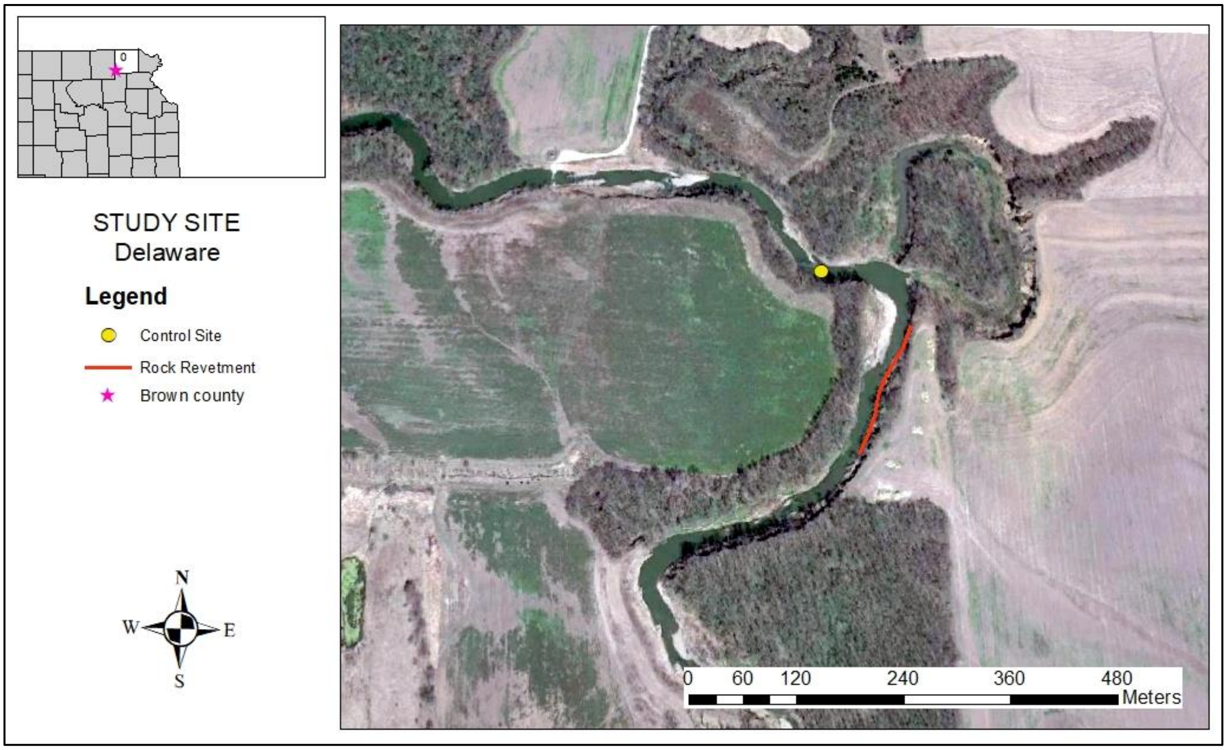

Figure 10. Map of location for Bioassessment in Delaware River section.

Delaware (River Figure 10) in the Kickapoo reservation located at 39°39.287' North and 95°38.510'West. This river has a drainage area of 381.14 square kilometers and a mean annual precipitation of 87.63 centimeters. A rock revetment was installed in 2015. The method of monitoring on this site is Macroinvertebrate Bioassessment.

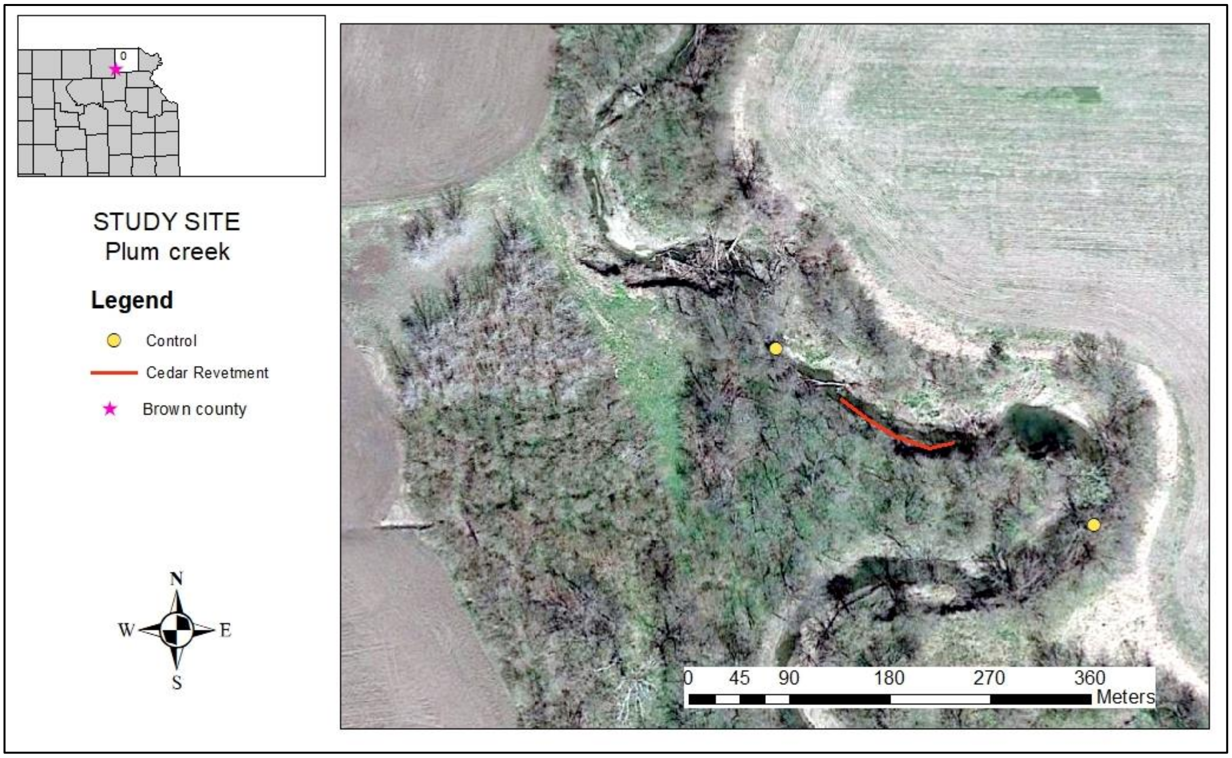

Figure 11. Map of location of Bioassessment in Plum Creek.

Plum Creek (Figure 11) in the Kickapoo reservation is located at 39°41.423’ North and 95°41.694’ West. This creek has a drainage area of 48.90 square kilometers and a mean annual precipitation of 87.88 centimeters. A cedar revetment was installed in 2010. The method of monitoring this site is, Macroinvertebrate Bioassessment. The estimated length of the reach in the study is of 40 meters with a bank height of 7.5 meters. The land is used for crops and has a riparian zone containing 12 meters wide of grass and cottonwood trees and 5 meters of vegetation before the crop field. One more site is represented here as a future possible site where we took roots samples, this site named Cross Creek (Figure 12) located in Jackson county 39°14.854’ North and 95°59.784’ West. This creek has a drainage area of 306.89 square kilometers and a mean annual precipitation of 88.40 centimeters.

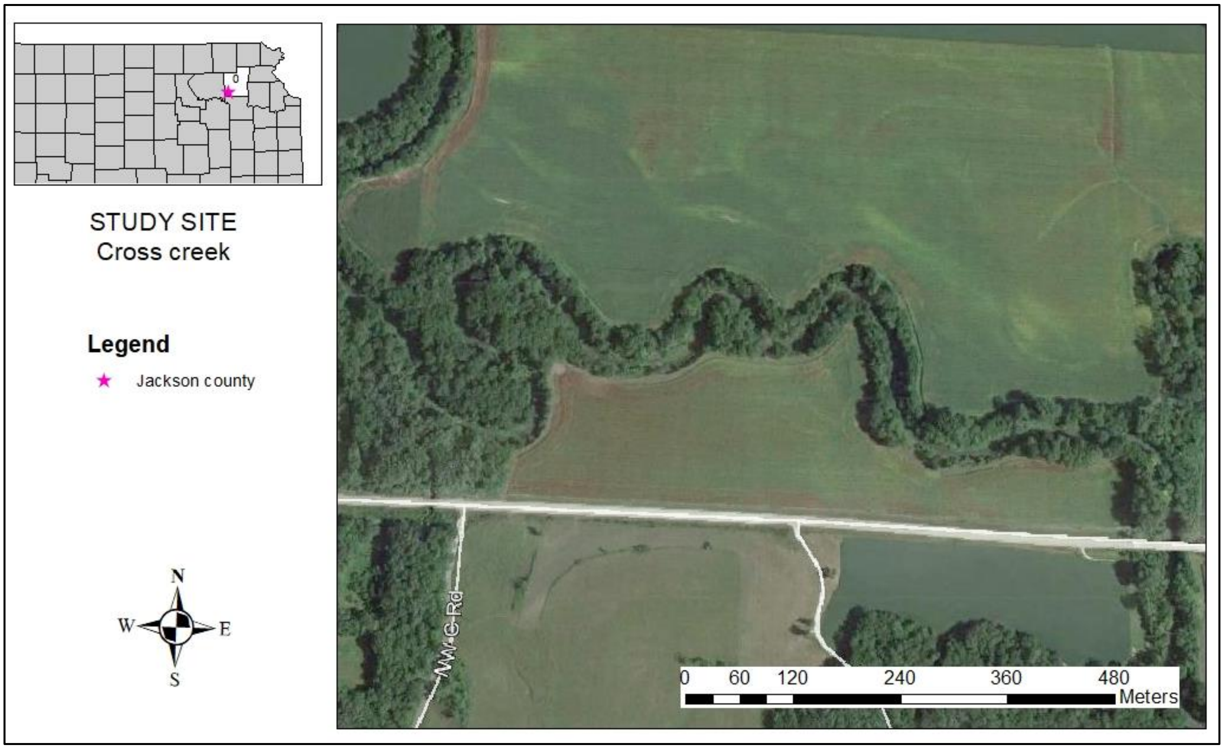

Figure 12. Map of location of Cross Creek.

Table 1. Summary of Study sites and monitoring methods used.

Location	Name	Revetment	Year installed	Method
Potawatomi Reservation	Little Soldier Creek	Cedar	2000	Rebar
Kickapoo Reservation	Delaware River	Rock	2015	Bioassessment
	Plum Creek	Cedar	2010	Bioassessment
Atchison County	Little Grasshopper Creek	Cedar	2017	Rebar
Nemaha County	Wolfley Creek	Cedar	2017	Rebar + Roots
Marshall County	Axtell-Schmidt Creek	Cedar	2007	Roots
Jackson County	Cross Creek	None	Possible	Roots

Macroinvertebrates Bioassessment

Macroinvertebrates Bioassessment was conducted in two locations in the Kickapoo Reservation: Delaware River and Plum Creek. The bioassessment of water quality used macroinvertebrates, as designated at three different in-stream habitats to take a collection of macroinvertebrates, bank edges, riffles, and pool. Macroinvertebrates were collected using two types of nets a bottom kick net for riffles and a D-net for banks edges and pools with 500 µm mesh size, following the adapted procedure in the Volunteer Stream Monitoring manual (Hoosier Riverwatch, 1997). The collection was conducted during 10 to 15 minutes, also on riffle habitat ten rocks were examined and macroinvertebrates collected. Macroinvertebrates were put into bottles with a 50/50 ratio of water and ethyl alcohol. The samples were taken to the laboratory and identified to family through a stereoscope examination using the Bioindicators of Water Quality Quick-Reference guide by Purdue University (Speelman & Carroll, 2012). Temperature, pH, and dissolved oxygen were taken *in situ* with using a Multiparameter Water Quality Sonde 6600 V2 manufactured YSI ®. The macroinvertebrate bioassessments was performed during late spring - early summer (May – June) and were collected twice a year for two years.

Data Analysis

The macroinvertebrate individuals collected were identified by Order and Family to be used as the data to calculate the following biotic and diversity indices.

Biological Monitoring Working Party (BMWP): The BMWP, for each site, score was calculated by simply summing the individual scores of all the families found (Table 2). Pollution intolerant taxa have a high score; therefore, a higher score would indicate a better biological condition for macroinvertebrates.

Average Score Per Taxon (ASPT): ASPT is calculated by dividing the BMWP score by the total number of the scoring taxa. A high ASPT value is characteristic of a clean site, meaning that has more number of higher scoring taxa (Armitage et al., 1983).

Table 2. Table taken from Armitage, Moss, Wright, & Furse, (1993) containing the BMWP tolerance.

FAMILIES	SCORE
Siphlonuridae, Heptageniidae, Leptophlebiidae, Ephemerellidae, Potamanthidae, Ephemeridae, Taeniopterygidae, Leuctridae, Caprniidae, Perlodidae, Perlidae, Chloroperlidae, Aphelocheridae, Phryganeidae, Molannidae, Beraeidae, Odontoceridae, Leptoceridae, Goeridae, Lepidostomatidae, Brachycentridae, Sericostomatidae.	10
Astacidae, Lestidae, Agriidae, Gomphidae, Cordulegasteridae, Aeshnidae, Corduliidae, Libelluiidae	8
Caenidae, Nemouridae, Rhyacophilidae, Polycentropidae, Limnephilidae	7
Neritidae, Viviparidae, Ancylidae, Hydroptilidae, Unionidae, Corophiidae, Gammaridae, Platycnemididae, Coenagriidae	6
Mesoveliidae, Hydrometridae, Gerridae, Nepidae, Naucoridae, Notonectidae, Pleidae, Corixidae, Haliplidae, Hygrobiidae, Dytiscidae, Gyrinidae, Hydrophilidae, Clambidae, Helodidae, Dryopidae, Elmidae, Chrysomelidae, Curculionidae, Hydropsychidae,Tipulidae, Simuliidae, Planariidae, Dendrocoelidae	5
Baetidae, Sialidae, Piscicolidae	4

Table 2. Continued.

FAMILIES	SCORE
Valvatidae, Hydrobiidae, Lymnaeidae, Physidae, Planorbidae, Sphaeriidae, Glossiphoniidae, Hirudidae, Erpobdellidae, Asellidae	3
Chironomidae	2
Oligochaeta (whole class)	1

Elmidae – Plecoptera - Trichoptera: This score is abbreviated as EIPT, the index is calculated by identifying the number of species within these taxa *Elmidae*, *Plecoptera* and *Trichoptera* at the site of study (Dos Santos et al., 2011).

Biotic Index: The number found from each family was multiplied by the tolerance value of the family established by Speelman & Carroll from the Quick-Reference guide of Purdue University (2012). The sum of the family tolerance score is divided by the grand total of the number of families founded.

Table 3. Water quality rating for the biotic index (Speelman & Carroll, 2012).

Biotic Index	Water Quality Rating
0.00-3.75	Excellent
3.76-4.25	Very Good
4.26-5.00	Good
5.01-5.75	Fair
5.76-6.50	Fairly poor
6.51-7.25	Poor
7.26-10.00	Very poor

Alpha diversity Shannon-Wiener and Simpson scores were calculated using R® program for diversity indices to determine water quality (Moreno, 2001).

Shannon-Wiener: $\boldsymbol{H'} = \sum_{i=1}^{S} \boldsymbol{p_i} \ln(\boldsymbol{p_i})$

Simpson: $\boldsymbol{\lambda} = \sum \boldsymbol{p_i^2}$

Where:

p_i= Proportion of abundance of the species.

S= Total number of species.

Erosion Estimation Techniques

To measure short-term erosion reinforcement bars 1 cm in diameter and approximated 60.5 to 121 cm were placed approximately 92 cm vertically apart. Reinforcement bars were inserted in the stabilized reach and a non-stabilized reach of each site following the methods of Pope et al. (2014) and Couper et al. (2002). Three to four reinforcement bars were placed in vertical transect along the eroded bank, depending on the cut bank length, perpendicular to the bank face using a hammer. All reinforcement bars were inserted all the way, and a length of orange flagging attached to the end, to aid in finding them in the future. We measured the exposure or deposition of the reinforcement bar by uncovering the soil o top. Three sites were measured using erosion pins: Little Soldier, Little Grasshopper and Wolfley Creek once each year. In locations where it was difficult to locate previously placed reinforcement bars, a metal detector, ACE 300, manufactured by GARRET ™ was used.

To measure long-term erosion, a macro analysis of exposed tree root was used. The samples were taken from three sites Axtell-Schmidt, Cross Creek and Wolfley Creek. Samples were 5-10 cm thick, to avoided disturbances by the stem cross sections were cut 50 – 100 cm from the trunk and still living. Before the cross section was cut in-situ details of the root were recorded such as the species, location in the bank and a photo. The length of soil loss at each location of roots was recorded using a meter stick oriented perpendicular to the flow and the bank (Figure 12), four labeled measurements to record relative bank position were marked as A, B, C and D in the cross section with a permanent ink marker following Stotts et al., 2011 specifications (Figure 13).

Figure 13. Length of the eroded bank E_X was obtained by averaging the perpendicular distance from the riverside edge of the root to the current bank position. (A) Downstream bottom, (B) Upstream bottom, (C) Downstream top, and (D) Upstream top.

Samples were air dried for 1-2 months, cut samples in 2 cm thick sections, and sanded smooth using sequentially finer sanding papers 80, 150, P220, P320 and P400 (Corona et al., 2011; Sun, Wang, & Hong, 2014; Stotts, O'Neal, Pizzuto, & Hupp, 2014).

Data Analysis

Erosion rates were calculated using the following equation modified by Stotts, O'Neal, Pizzuto, & Hupp (2014) from Corona et al. (2011):

$$E_r = E_x - (Gr1) + \frac{(B1 + B2)}{2}$$

$$E_{ra} = E_r / NRex$$

Where:

E_r: Corrected length of the eroded bank.

$Gr1$: Root growth after exposure (Figure 14).

E_x: Average distance between the riverside edge of the root and the current bank position.

$\frac{(B1+B2)}{2}$: Average bark width.

E_{ra}: Annual erosion.

NRex: number of years the root has been eroded (Figure 14).

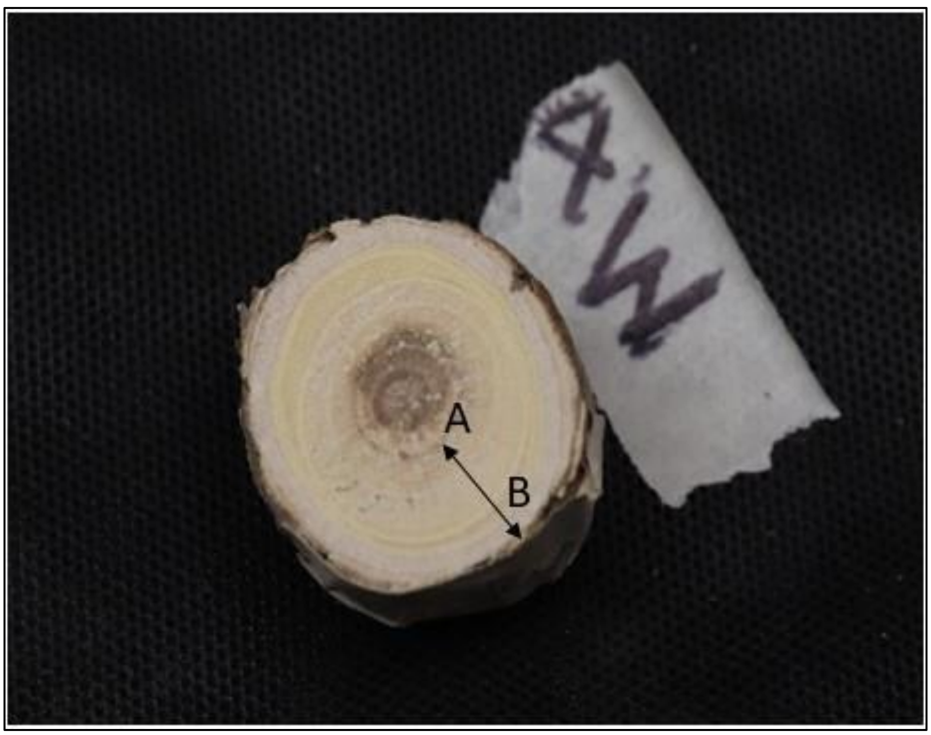

Figure 14. Several years since root exposure, annual rings were counted between (A) the exposure indication this case a scar and (B) the outside edge of the root. Gr1 is the measurement in centimeter from A to B.

Chapter 3 - Results and Discussion

The following chapter will present the result of the data collected over the course of two years. The data presented is described as case studies for each site, starting with the bioassessment and finishing with the quantification of erosion rates. Unfortunately, there was insufficient data to establish true statistical comparisons between the stabilized and un stabilized stream reaches erosion or bioassessment. Several other studies indicate at least five years of repeated measures are needed for statistical analysis (Arslan et al., 2016; Selvakumar, O'Connor, & Struck, 2010).

The cost estimated for the cedar revetment in these projects is $4,000 for a 76.20 meters long revetment, or approximately $52.49 per meter. This estimate is similar to the calculation per Dave (2017) of a tree revetment $72 per meter. To calculate the estimated cost for a rock revetment, the Delaware stabilization project total cost of $563,295 with 11 sites was used. This project had a length between 152.5 meters to 76.2 meters. Delaware River rock revetment was estimated to cost $335.8 per meter with a length of 152.5 meters similar to the $205 per meter calculated by Dave (2017). The cost of a rock revetment is higher because this technique requires an engineer to design the project, cost of material and the use of heavy equipment.

Macroinvertebrate bioassessment

The streams are of the warm water type. The water parameters for Delaware river averaged a temperature of 25.01C°, 10.15 ppm of dissolved oxygen, 18.85 ppm NO_{3^-} , a conductivity of 0.79 ms /cm with a turbidity of 8.55 NTU and pH 8.22. For both rivers, more species found were in the riffle habitat while the least number of species in the pool habitat, this is expected because of the suitability for reproduction of the sessile aquatic insects, due to the increase water flow in the riffle (McCafferty, 1998).

The most dominant taxa were *Hydropsychidae* followed by *Chironomidae* and *Leptohyphidae*, representing 54.26% and 50.4% of riffle species on stabilized and control reaches respectively (Table 4). The substrate created by restoration is suitable for the attachment of net-spinning *Hydropsychidae* and *Chironomidae* and are tolerant of silt and sand (Selvakumar et al., 2010). The least represented families for the riffle on stabilized reach were *Corydalidae, Veliidae,* and *Calopterygidae*, these were just 0.48% of the total taxa. On the control reach the least represented families where *Potamanthidae, Stratiomyidae, Lepidostomatidae, Corydalidae,* and *Mesoveliidae* these were 0.57%.

Table 4. Delaware river macroinvertebrates communities recorded from data in Stabilized and Control reaches. Average over two years and 4 collections.

Delaware		**Stabilized**			**Control**			
Order	**Family**	**Riffle**	**Pool**	**Cut bank**	**Riffle**	**Pool**	**Cut bank**	**Total**
Ephemeroptera	*Baetidae*	29	2	5	62	1	33	132
	Heptageniidae	28	8	1	55	2	0	94
	Ephemerellidae	34	0	0	22	0	4	60
	Potamanthidae	8	0	3	1	3	0	15
	Isonychiidae	30	10	0	63	0	4	107
	Baetiscidae	4	2	0	2	0	0	8
	Siplhonuridae	4	0	0	6	0	8	18
	Leptophlebiidae	4	0	7	2	0	1	14
	Caenidae	4	2	0	8	0	0	14
	Leptohyphidae	114	4	0	14	1	3	136
Diptera	*Culicidae*	8	0	0	0	0	0	8
	Stratiomyidae	0	0	0	1	0	1	2
	Chironomidae	60	5	7	101	15	57	245
	Empididae	0	0	0	5	0	2	7
	Muscidae	0	0	0	6	0	0	6
	Simuliidae	0	0	0	34	0	7	41
	Culicidae	8	0	0	0	0	0	8
	Psychodidae	0	1	0	0	0	0	1

Table 4. Continued.

Delaware		Stabilized			Control			
Order	**Family**	**Riffle**	**Pool**	**Cut bank**	**Riffle**	**Pool**	**Cut bank**	**Total**
Diptera	*Stratiomyidae*	0	0	0	1	0	1	2
	Chironomidae	60	5	7	101	15	57	245
	Empididae	0	0	0	5	0	2	7
	Muscidae	0	0	0	6	0	0	6
	Simuliidae	0	0	0	34	0	7	41
	Psychodidae	0	1	0	0	0	0	1
	Tipulidae	6	0	0	0	0	0	6
	Sciomyzidae	0	0	0	0	0	2	2
	Atherecidae	0	0	0	2	0	0	2
	Ceratopogonidae	0	0	0	6	0	0	6
	Dixidae	8	0	0	2	0	0	10
Plecoptera	*Perlidae*	0	0	0	4	0	0	4
	Capniidae	4	0	0	2	0	0	6
Trichoptera	*Philopotamidae*	38	0	0	24	0	0	62
	Lepidostomatidae	0	0	0	1	1	0	1
	Brachycentridae	5	0	0	0	0	0	5
	Glossosomatidae	6	0	0	0	0	0	6
	Hydroptilidae	6	0	0	10	0	6	22
	Leptoceridae	4	0	0	0	0	0	4
	Hydropsychidae	160	0	2	321	0	62	545
Coleoptera	*Dryopidae*	37	0	0	23	1	13	74
	Haliplidae	0	0	3	0	0	0	3
	Dytiscidae	0	0	0	0	1	0	1
	Hydrophilidae	0	0	1	2	1	1	5
	Elmidae	10	0	0	73	1	6	90
Odonata	*Coenagrionidae*	0	2	0	0	0	0	2
	Calopterygidae	2	1	0	6	0	0	9
Megaloptera	*Corydalidae*	1	0	0	1	0	0	2
Hemiptera	*Corixidae*	0	13	13	0	32	19	77
	Gerridae	0	0	3	0	0	6	9
	Mesoveliidae	0	0	0	1	0	0	1
	Notonectidae	0	0	0	4	0	0	4
	Veliidae	1	0	12	0	0	0	13
Total specimens		615	50	57	864	59	235	1879
Total families		26	11	11	30	11	19	108

Plum Creek was found with a good diversity of tree species in the riparian area like *Celtis occidentalis*, *Morus rubra*, *Parthenocissus quinquefolia*, *Ribes missouriense,* and others. For Plum Creek the average water parameters were: temperature 25.11C°, 8.39 ppm of dissolved oxygen, 20.57 NO_3- content, 4.4 NTU of turbidity, with 0.71 ms/cm of conductivity and pH 8.12. The most dominant taxa found in Plum Creek were *Chironomidae*, *Baetidae,* and *Heptageniidae* which represented 50.34% and 56.03% of species found in the riffle habitat on stabilized and control reaches respectively (Table 5). *Chironomidae* and *Baetidae* are often found to be the dominant species in other similar macroinvertebrate bioassessments studies, because they are resistant to habitat changes and contamination (Arslan et al., 2016). The least dominant families in the riffle on Plum Creek where *Mesoveliidae, Capniidae, Coenagrionidae, Ephemeridae, Caenidae, Empididae, Philopotamidae* and *Leptoceridae* representing 2.83% of the stabilized reach compare to control reach the least dominant were *Culicidae, Perlidae, Capniidae* and *Philopotamidae* representing 2.4%.

Table 5. Plum Creek macroinvertebrates communities recorded from data in Stabilized and Control reaches.

Plum		Stabilized			Control			
Order	**Family**	**Riffle**	**Pool**	**Cut bank**	**Riffle**	**Pool**	**Cut bank**	**Total**
Ephemeroptera	*Baetidae*	61	16	29	13	9	13	141
	Baetiscidae	0	0	1	0	0	0	1
	Heptageniidae	32	0	5	24	4	6	71
	Ephemerellidae	19	9	3	0	6	6	43
	Potamanthidae	10	0	2	6	0	0	18
	Isonychiidae	11	10	0	9	4	1	35
	Ephemeridae	1	0	0	0	0	0	1
	Siphlonuridae	12	0	2	12	4	0	30
	Lepthophlebiidae	4	0	0	0	0	1	5
	Caenidae	1	8	0	0	4	1	14
	Leptohyphidae	1	0	0	5	3	0	9
Diptera	*Chironomidae*	67	126	45	56	30	19	343
	Psychodidae	0	0	1	0	0	0	1
	Culicidae	0	0	0	1	0	0	1

Table 5. Continued.

Plum		Stabilized			Control			
Order	**Family**	**Riffle**	**Pool**	**Cut bank**	**Riffle**	**Pool**	**Cut bank**	**Total**
Diptera	*Stratiomydae*	0	0	2	0	0	0	2
	Empididae	1	0	0	2	2	0	5
	Dixidae	2	6	0	0	2	0	10
	Ephydridae	0	0	0	1	0	0	1
	Tipulidae	0	2	0	0	0	0	2
	Simuliidae	17	0	1	2	0	0	20
Plecoptera	*Perlidae*	6	0	0	1	0	0	7
	Perlodidae	2	0	0	0	0	0	2
	Capniidae	1	0	0	1	0	0	2
Odonata	*Calopterygidae*	4	1	6	0	0	0	11
	Aeshnidae	0	0	1	0	1	0	2
	Gomphidae	0	1	0	0	0	0	1
	Coenagrionide	1	0	0	0	0	0	1
Lepicloptera	*Pyralidae*	2	0	0	0	0	0	2
Trichoptera	*Philopotamidae*	1	0	0	1	0	0	2
	Leptoceridae	1	0	0	0	0	0	1
	Hydroptilidae	0	0	1	0	0	0	1
	Hydropsychidae	30	0	1	18	0	0	49
Coleoptera	*Dryopidae*	10	0	0	11	0	0	21
	Haliplidae	0	1	0	0	0	0	1
	Gyrinidae	0	0	0	0	0	1	1
	Elmidae	18	10	1	3	0	0	32
	Hydrophilidae	2	0	0	0	0	0	2
Hemiptera	*Corixidae*	0	12	3	0	0	0	15
	Gerridae	0	1	4	0	0	0	5
	Nepidae	0	1	0	0	0	0	1
	Mesoveliidae	1	0	0	0	0	0	1
Total specimens		318	204	108	166	69	48	913
Total families		27	14	18	17	11	8	95

On the Delaware River Macroinvertebrate Biotic index indicated good habitat quality for stream aquatic species on the riffle and poor habitat quality for the cut bank and pool habitat, thus the overall quality quality is fair for stabilized reach to compare to control reach that resulted in good habitat quality for riffle, fairly poor for cut bank and very poor for pool habitat (Table 6). Stabilized reach shows higher habitat quality than the control reach meaning that stabilized reach provided better substrate for creating a suitable macroinvertebrate habitat, although statistical significance of these observed differences cannot be determined.

When measuring BMWP and ASPT index stabilized and control reaches have similar overall water quality although, within the habitat riffle and cut bank the stabilized reach had indication of higher water quality than control. EIPT were more highly represented in stabilized reach on the riffle habitat and was double the value of the control. Simpson index for riffle habitat on the stabilized reach resulted higher than control 0.86 and 0.80, respectively. Simpson index produces values from 0 to 1, meaning the closer to 1, there is more diversity species. So, if two more macroinvertebrates individuals were randomly selected on the stabilized site, the probability of these two different species is 3% higher than in the control site. The results show that there is slightly more species diversity presented in the stabilized reach than the control site.

Table 6. Average Macroinvertebrate Indices, BMWP, ASPT, EIPT, Shannon-Wiener and Simpson Biodiversity Indices from Delaware river site in stabilize and control reaches.

	Stabilized			Control		
Channel Unit	Riffle	Cut Bank	Pool	Riffle	Cut Bank	Pool
Biotic Index	4.50	5.78	5.95	4.28	5.14	7.45
BMWP	127.00	61.00	34.00	132.00	77.00	61.00
ASPT	4.88	5.55	3.09	4.40	4.05	5.55
EIPT	12.00	1	0	6	3	2
Shannon-Wiener (H’)	2.41	2.02	2.16	2.21	2.25	1.40
Simpson	0.86	0.83	0.84	0.80	0.86	0.61

The Biotic Index on Plum Creek indicated good habitat quality for riffle and cut bank, fair for the pool on stabilized reach, on control reach all habitat resulted in good habitat quality the overall both reach presented good water quality (Table 7). Using the BMWP and ASPT score the stabilized reach resulted in good quality and the control was of moderate quality. EIPT were more highly represented on the stabilized reach while on the control reach resulted in only five taxa on riffle habitat. Simpson index for riffle habitat on stabilized reach resulted in higher than control index of 0.88 and 0.81. Meaning that, if two more macroinvertebrate were randomly taken in the stabilized site, the probability of these being different species is 4% higher than to the control reach. Similar to the Delaware River stabilized reaches present more evenness in the distribution of macroinvetebrate community.

Table 7. Average Macroinvertebrate Indices, BMWP, ASPT, EIPT, Shannon-Wiener and Simpson Biodiversity Indices from Plum Creek site in stabilize and control reaches.

	Stabilized			Control		
Channel Unit	Riffle	Cut Bank	Pool	Riffle	Cut Bank	Pool
Biotic Index	4.40	4.99	5.65	4.91	4.44	4.81
BMWP	165.00	85.00	61.00	84.00	38.00	51.00
ASPT	6.11	4.72	4.36	4.94	4.75	4.64
EIPT	7	3	1	5	0	0
Shannon-Wiener (H')	2.51	1.87	1.40	1.98	1.56	1.89
Simpson	0.88	0.74	0.55	0.81	0.74	0.76

In this particular case study we are comparing how the stabilization practices affect the sediment balance for suitable habitat for the development of macroinvertebrate as an indicator of good water quality, EIPT measures the most taxa of sediment sensitive macroinvertebrates (Dos Santos et al., 2011). Figure 15 shows the overall average EIPT from both sites comparing the stabilized and control reaches. Stabilized reach results were higher for riffle and cut bank except for the pool habitat, which, is not a common habitat for macroinvertebrates. At both sites, the stabilized reach pool was considerably shallower than the control reach pool.

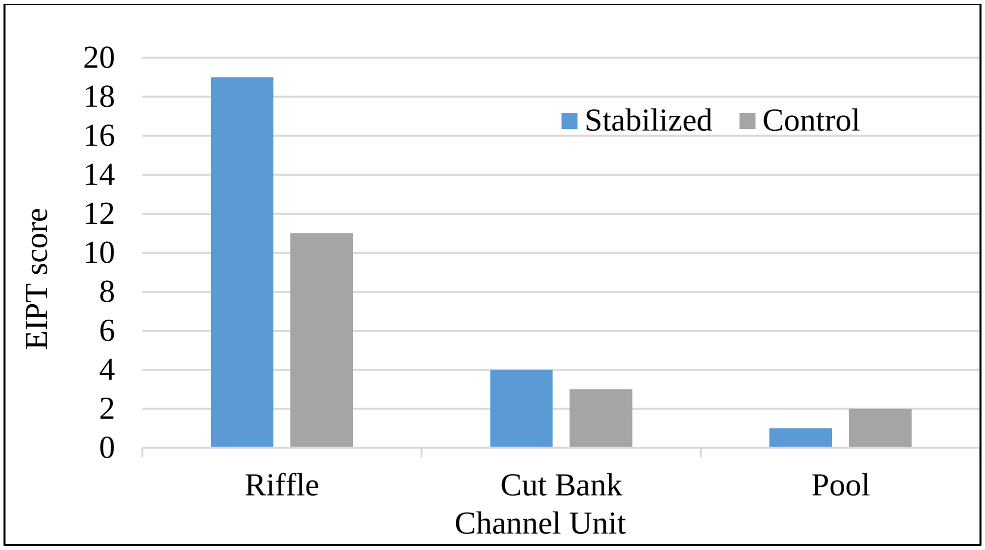

Figure 15. Overall average EIPT comparing control reaches and stabilized reaches on the Delaware River and Plum Creek.

The abundance and equity of a community can tell us about the health of the environment in which they are found. In order to compare the stabilized and control reaches the average Shannon-Wiener was calculated and resulted in higher abundance and distribution for the stabilized compared to the control. Even though the Shannon-Wiener (H') was 2.5 in the riffle, which is lower than the maximum diversity expected for that community (H' max) 3.3, it still represents more abundance and equity of species than the control site that has a Shannon-Wiener (H') of 2.09 with a maximun diversity expected for the controls community (H' max) of 3.2. The Shannon-Wiener index clearly shows the difference between all three habitats.

This studies' results show a slight improvement in habitat conditions on stabilized sites compared to control sites, these are similar results of the Selvakumar et al., (2010), study which found statistically significant change using VASCI, HBI and EPT indices between before and after restoration at α=0.1.

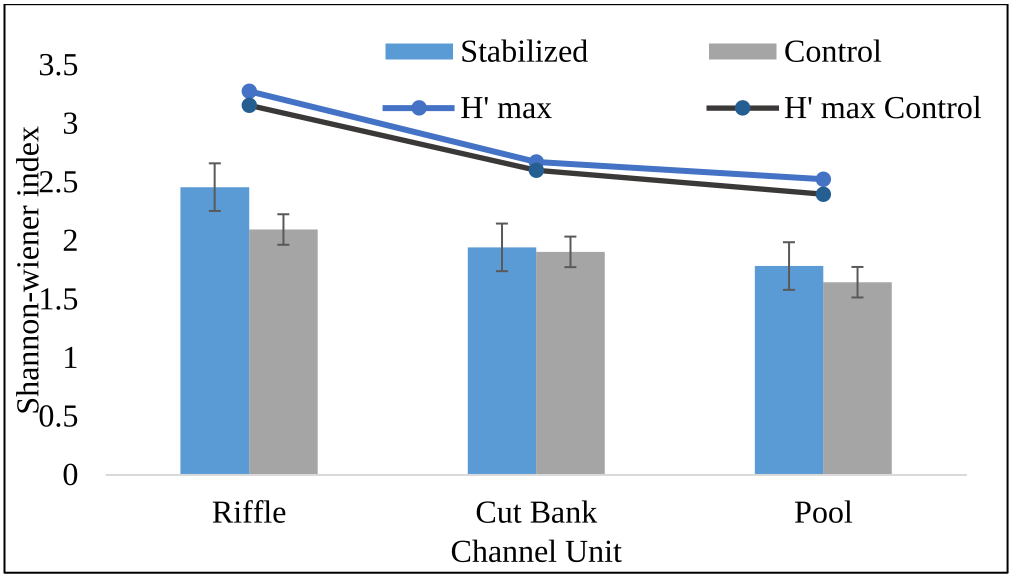

Figure 16. Overall average Shannon-Wiener index comparing control and stabilized reaches from Delaware and Plum Creek.

Erosion Estimates Case Studies

Case study Little Grasshopper Creek

After the installation on March 2017, within a month a high flow event occured, with a water discharge of approximately of 254 cubic meters per second and a depth approximately of 6 meters which exceed bank full flows. The cedar revetment was successful in capturing sediment by retaining and causing deposition of 121 cubic meters of sediment, showing the efficiency of cedar revetment on capturing sediment. *Salix sp*. stakes sprouted and became established as a vegetation cover similar to Šlezing, Jana, & Lenka (2017). According to the reinforcement bars placed in August 2017, the overall erosion that occurred during the last year is similar for both the control and stabilized reach (Figure 17), however, the difference between them is where the erosion is occurring. On the stabilized reach, the erosion occurred in the upper bank and deposition on the lower bank, creating a more gently sloping bank appropriate for vegetation establishment. The erosion on the control reach is occurring in the middle bank position, this happened from undercutting the upper bank which will cause slumping and more erosion in the future.

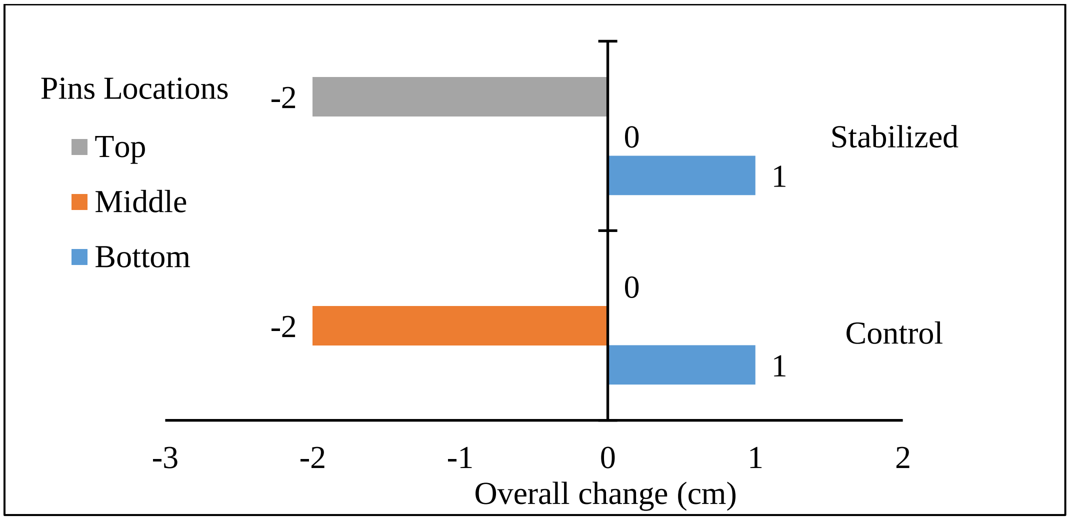

Figure 17. Streambank erosion estimates from reinforcement bars at two control reaches and a stabilized reach. From Little Grasshopper Creek after one year. Negative values indicate deposition.

Case Study Wolfley Creek

After the installation in April 2017, the cedar revetment was successful in capturing sediment by retaining and causing deposition of approximately 48 cubic meters of sediment. On this case study, the difference between the stabilized and control reach is notable after one year of monitoring. The stabilized site did not result in any erosion, and a minor amount deposition of 0.5 cm on the top reinforcement bar (Figure 18). Wolfley Creek control sites resulted with more erosion a total of 6 cm and 3.3 cm on transects one and two, respectively. Transect one has an erosion similar to the control of the Little Grasshopper, which had erosion in the middle of the bank position. This erosion is dangerous for the bank because it will cause future extreme erosion during a high flow event. Control two had more severe erosion than control one because erosion is occurring on the bottom of the bank and this is caused by the helical flow of the water under bankfull events (Whipple, 2004).

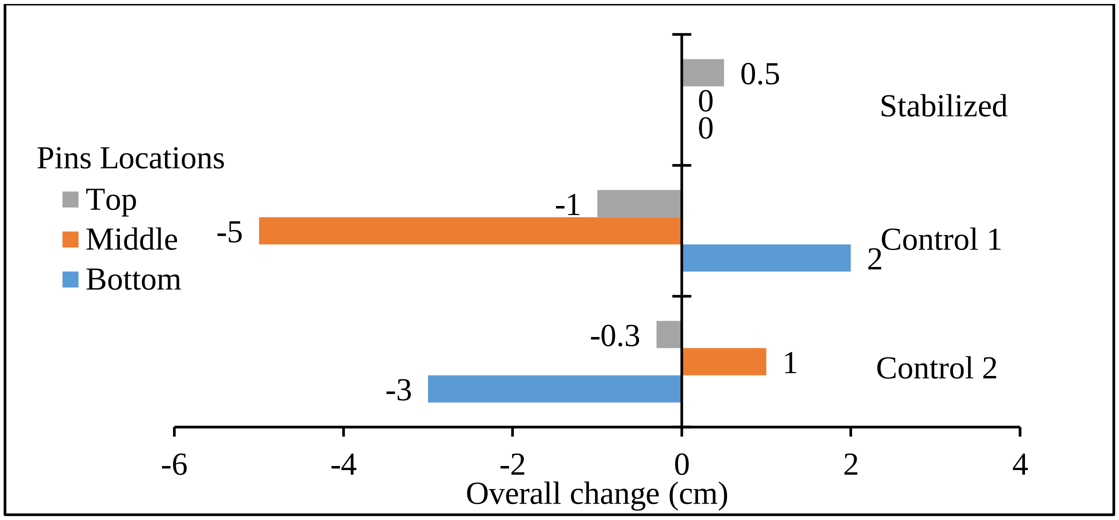

Figure 18. Streambank erosion estimates from Reinforcement bars at two control transects and stabilized reach from Wolfley creek after one year. Negative values indicate deposition.

Case Study Little Soldier Creek.

The cedar revetment was established on March 2000 (Figure 19 A and B) with the main objective to stop erosion that is causing the creek to move towards the family house in the Potawatomi reservation. After five years the revetment shows recovery of vegetation and growth of riparian buffer (Figure 19 C). During the next years sediment is trapped by the cedar branches and vegetation continues to grow. During 2007, captured sediment was measured within the 60.96-meter-long cedar revetment resulted on 65.75 cubic meters (Figure 19 D). Over the past 17 years of documentation archive, erosion resulted on 140 cm, meaning approximately 8.24 cm per year of sediment was lost and the revetment is still in a functioning state (Figure 19 D). As indicated in Figure 19 E, erosion occurred between March-April 2017 floods and Figure 19 F shows the recent exposure of roots denoted by the change in root color.

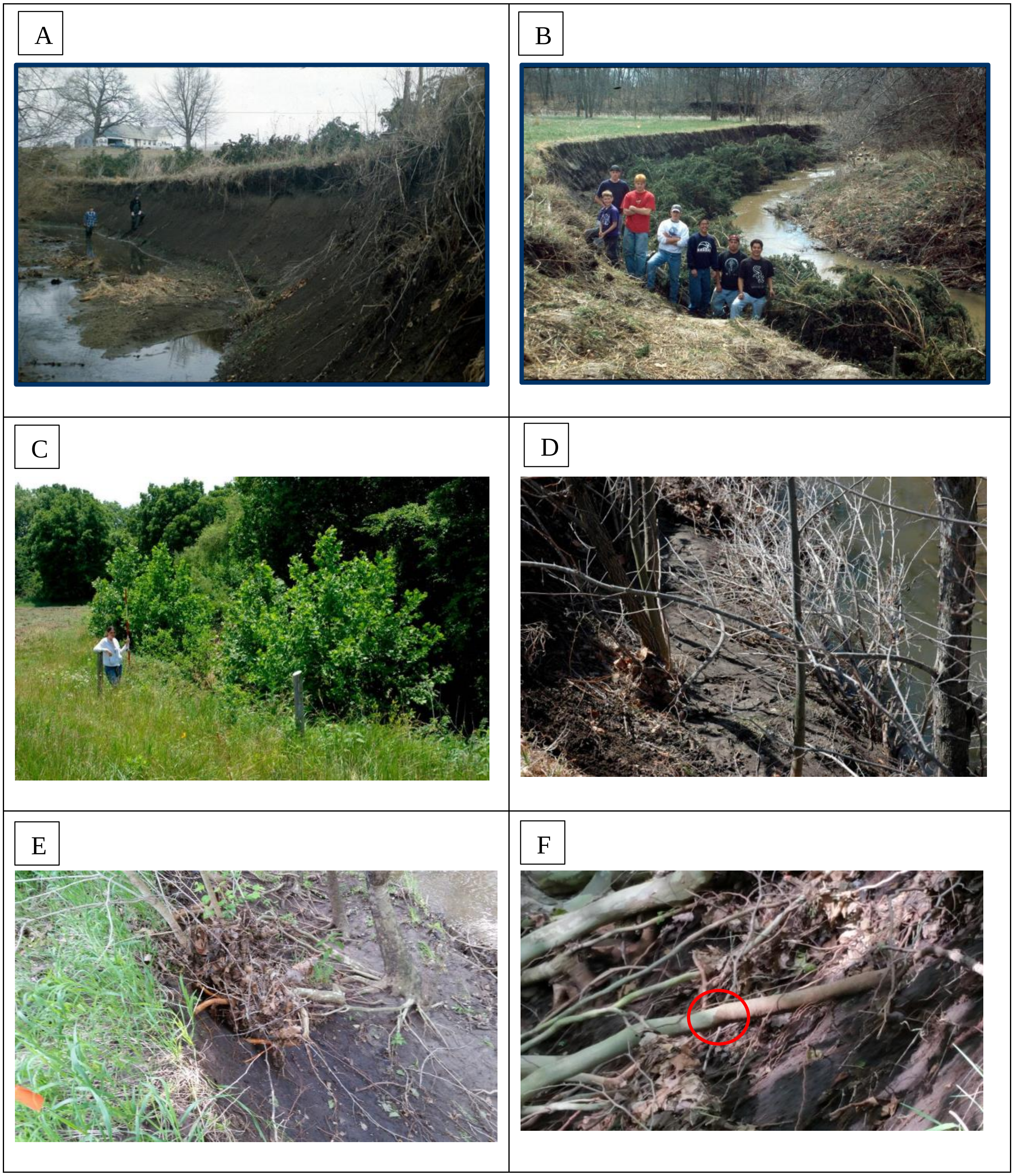

Figure 19. Photographic documentation of Little Soldier Creek. A. Before revetment, B. After revetment, C. After 5 years, D. 2007 sediment deposition, E. Bank erosion from 2017 and F. Root exposure 2017.

Case Study using exposed tree roots.

The novel methodology was ideally used to evaluate the Axtell dairy farm because no reinforcement bars installed in 2008 were found. Exposed tree roots methodology was used as an alternative. Ten exposed roots samples were collected from the stabilized reach (Table 8). Erosion on an average was 3.39 cm/year around the stabilized reach, since this site does not have an established control, it was compared to the control transect found on Wolfley Creek site. Transect one resulted in an average erosion of 10.26 cm/year, similar to the erosion found using reinforcement bars from the past year (6 cm). Erosion will change according to the flow events of each year. The overall results from this project are similar to Dave & Mittelstet (2017), where they found a reduction on erosion rates after streambank stabilization of 45 cm/year and 16 cm/year, respectively. They also found more erosion on control sites as compared to stable sites after flood event 0 cm/year and 17 cm/year, respectively. Cross Creek has no stabilization practice and presented the most average erosion rate of 14.84 cm/year. It is important to consider this site because it represents an ideal site for establishment of a cedar revetment practice as we will have pre-establishment erosion rates documented, this site will continue to be monitored for future reference if the landowner is willing to pay for the project in the future.

The tree roots found on Axtell dairy farm, Wolfley Creek and Cross Creek were from deciduous species, other studies have also used deciduous tree root successfully to quantify erosion (Stotts et al., 2014; Scuderi, 2017).

Table 8. Estimates of yearly erosion according to exposed roots on Axtell dairy farm, Wofley Creek study sites and Cross creek posible study site.

Site / Creek name	Sample Name	Tree Sp.	Years exposed NREX	ERA (cm/y)	Average
Axtell-Schmidt	1	*Ulmus pumila*	8	6.03	
Axtell-Schmidt	2	*Ulmus pumila*	7	0.37	
Axtell-Schmidt	3	*Ulmus americana*	7	7.15	
Axtell-Schmidt	4	*Ulmus americana*	5	0.08	
Axtell-Schmidt	5	*Ulmus americana*	8	5.60	
Axtell-Schmidt	6	*Ulmus americana*	7	0.31	
Axtell-Schmidt	7	*Ulmus americana*	7	4.34	
Axtell-Schmidt	8	*Ulmus americana*	6	0.38	
Axtell-Schmidt	9	*Ulmus americana*	10	5.18	
Axtell-Schmidt	10	*Ulmus americana*	8	4.53	**3.39**
Wolfley	1W	*Fraxinus sp.*	4	10.56	
Wolfley	2W	*Celtis occidentalis*	4	12.61	
Wolfley	3W	*Ulmus americana*	10	7.49	
Wolfley	4W	*Ulmus americana*	7	10.36	**10.26**
Cross	1D	*Ulmus americana*	18	2.51	
Cross	2D	*Ulmus americana*	11	4.6	
Cross	3D	*Quercus macrocarpa*	1	37.4	**14.84**

Chapter 4 - Conclusions and Recommendations

Our assessment of stream restoration practices found that cedar revetment is effective at capturing sediment and both rock vanes and cedar revetments are effective at stabilizing stream reaches. Restoration practices were successful in stabilizing the stream bank and preventing further erosion into property. Macroinvertebrates results show a positive influence on improving habitat for benthic insects. It will depend upon the main objectives established for the restoration to include the evaluation of environmental benefits that the restoration project may provide. Therefore, it is recommended to take the time when planning future assessment projects and establishment of the monitoring protocol from the beginning, according to the main objectives.

Shannon-Wiener and Simpson successfully complemented the information provided by the biological indices, and they provided stronger indicators of the aquatic community. The alfa diversity index explained that the macroinvertebrate communities on the stabilized sites have slightly more evenness and abundance of species. The increase of environmental benefits such as improve macroinvertebrate habitat is expected to continue in the coming years of monitoring.

Cedar revetment and establishment of willows (*Salix sp*.) along the streambanks allow the process of vegetation recovery by protecting the streambank. On these case studies, it is shown that the establishment of vegetation is a key point for the stream to develop a bank with a gentle slope.

The use of exposed tree roots is an effective alternative to reinforcement bars and photo image methods to quantify yearly and historical erosion. Reinforcement bars can present challenges in the search for them when collecting data, hence, the use of macro analysis of the exposed roots is a useful method that can help to be prepared for the yearly erosion of the streams and monitor the possible increase or decrease of the average estimated erosion. To use this to document stabilization effects roots should be sampled before stabilization, and at least 5-years after.

It is recommended to do a thorough photographic documentation to monitored through the years the changes of the stream. In this study, the photo documentation helped to show the results on sedimentation in Little Soldier, Little Grasshopper and Wolfley creek sites. Photographs, when practical would be helpful when using exposed root methodology to know how the location of the roots sampled on the streambank and where erosion is occurring.

In order to collect enough data for statistical between stabilized sites and control sites, repeated assessment of macroinvertebrates, erosion, and deposition is necessary. Also, this information will help to inform the improvement of future “Best Management Practice” and for future preparation of landowners for extreme weather events. Also, these case studies showed that it is possible to work and train students and stakeholders in cedar revetment installation and basic monitoring techniques to increase documentation in the future.

The lack of sufficient data to make valid statistical comparisons is a serious shortcoming of this case study approach. This project has laid the foundation for further long-term data collection which will enable future statistical analysis. These case studies where almost all already established before the author began their graduate program.

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