Riparian Functional Assessment: Choosing Metrics that Quantify Restoration Success in Austin, Texas

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Abstract

In an effort to understand how various levels of management have impacted the ecological function of urban riparian zones, the City of Austin performed Riparian Functional Assessments (RFA) at 28 site locations in the spring of 2012. Sites were categorized into degraded (history of vegetative control and disturbance) and reference (minimal vegetation management and anthropogenic disturbance) in order to determine which of the 15 measured RFA parameters could be used to monitor improvements to riparian zone function as a result of vegetative restoration over time. Results suggest that monitoring for changes in soil compaction and moisture, riparian zone width, in-stream canopy cover, plant cover and structural diversity, hardwood demography, and seedling recruitment over time will allow managers to accurately assess if ecological function is being improved following restoration activities. Being able to prove restoration project success is vital to maintaining public support and funding for future riparian restoration projects.

Introduction

Riparian zones are widely recognized as functionally unique and dynamic systems that provide a suite of essential ecosystem services (Fischer and Fischenich 2000). Healthy riparian buffers can function to provide pollutant removal, protection from stream bank erosion, slowing of floodwaters, increased groundwater infiltration, temperature buffering, carbon sequestration, and plant and animal habitat (Fischer and Fischenich 2000, Stacey *et al.* 2006, Richardson *et al.* 2007, Woolsey *et al.* 2007). In general, the more degraded an ecosystem, the more fundamentally altered the basic ecosystem services it can provide (Hobbs and Cramer 2008). Therefore, riparian zone restoration is a commonly applied method for improving the ecological function of a degraded site. A vast majority of current restoration endeavors involve either removal of vegetation, planting, or both, without measuring essential ecosystem processes that may be affected. As there is currently no quantitative measure of how these restoration projects strengthen the environmental functionality of the riparian zone at the City of Austin, it is difficult to show success of a restoration project. Without the ability to prove success in restoration projects there is a great risk that public support will diminish (Woosley *et al*. 2007). Clearly defined goals and methods will ensure effective use of resources and increase the likelihood of achieving success (Woosley *et al*. 2007; Hobbs and Prach 2008). In order to guide future restoration in the City of Austin there is a need to clearly define methods that allow managers to measure improvements in ecosystem function following restoration activities.

Previous studies by the City of Austin have identified a methodology for diagnosing and later monitoring the improved ecological function of urban riparian systems following restoration activities (Duncan 2012). This Riparian Functional Assessment (RFA) consists of 15 field parameters identified from the literature to respond to changes in management and have a direct link to ecological function (Duncan 2012). The objective of this study is to identify which of the RFA metrics can detect differences between functionality of degraded sites and healthy reference sites. All metrics should be scientifically based but should be time and cost effective. Thus the first portion of this study will be to compare sites that have a history of vegetation control and disturbance (degraded) to sites containing minimal management disturbances (reference). Metrics will be chosen from this initial study to use in the second phase of the project. Once the appropriate metrics are chosen, specific degraded sites will be actively or passively restored and monitored in order to discern if degraded sites improve in functional metric scores. This study will help guide future restoration efforts for the City of Austin.

Methods

In an effort to understand how various levels of management have impacted the functionality of riparian zones, the City of Austin chose 28 riparian site locations to examine. One goal of this project was to distinguish any differences that existed between highly managed riparian zones and minimally managed riparian zones, thus the sites were split into two groups. The degraded category consisted of 16 sites that had a history of vegetative control and disturbance while the reference category consisted of 12 sites that have experienced little management (Table 1). All degraded sites were selected according to the City of Austin Riparian Zone Restoration Site Prioritization methodology (Duncan *et al*. 2012). Reference sites were selected based on geographic location (ecoregion), hydrology (drainage acreage), and existing riparian vegetation. Sites were selected that had similar drainage acreage, were in the same ecoregion, and where possible the same watershed, and had an existing mature riparian forest buffer of at least 100 feet. Only 12 reference locations fit the above criteria.

There exist two Ecoregions in Austin, TX, with different soil characteristics. The Edwards Plateau contains shallow soils overlaying limestone. The deeper soils in this region are loamy and calcareous. Blackland Prairie soils were formed from shale parent material and are highly clayey with shrink-swell properties. They are typically deep overlaying marl or chalk (USDA 2008a). These differences in soil characteristics have the potential to affect the parameters collected in this project. Thus sites were selected from both the Edwards Plateau and Blackland Prairie regions so that any differences that existed due to Ecoregion could be distinguished.

The time of day also has the potential to affect several parameters collected in this project. In order to compensate for this fact, the time of day was split into three categories. Early morning samples were collected between 9:00 – 11:00 AM, late morning samples were collected between 11:00 AM – 1:00 PM, and afternoon samples were collected between 1:00 – 3:00 PM. The reference sites were divided up so that 4 reference samples would be collected in each of the three time categories. Degraded sites were divided so that 5 sites would be collected in each of the three time categories with one additional sample collected in any of the time categories.

At each site, a 100 m transect was run along the center-line of the associated creek. The length was chosen so that an extensive range of site conditions present at a site could be captured during sampling. Transect starting points were marked by tree tags, or associated permanent marker, and the direction of the transect (upstream or downstream) was denoted on the data sheet as reference for future evaluations. Photographs were taken at the upstream and downstream ends of the study reach, at 50 m looking downstream and upstream, as well as any other location that would be valuable for future comparisons. Reach parameters were collected along the entire transect, while creek parameters were collected at 5, 50, and 95 m on the transect. Riparian parameters were collected in 10m by 10m quadrats along both the left and right banks. The quadrats began at bankfull, ran 10m away from the creek, and were centered around 5, 50, and 95 m on the in-stream transect for a total of 6 quadrats at each site.

Table 1: Site list classified by site type, drainage area, and ecoregion (BP = Blackland Prairie, EP = Edwards Plateau).

Site Type	Site #	Site Name	Drainage Area (acres)	Ecoregion
Degraded	5556	East Bouldin @ Gabion in Gillis Park	320	EP
	5580	Barton Creek Trib @ Lund and Robert E. Lee	64	EP
	5584	Buttermilk Creek @ Buttermilk Park	320	BP
	5585	Boggy Creek @ 10th St	1280	BP
	5586	Bull Creek 1600ft upstream Loop 360	1280	EP
	5588	Common Ford Trib ds xing in Common Ford Ranch	1280	EP
	5591	Johnson Creek in Tarrytown Park	320	EP
	5592	Little Walnut Creek @ Dottie Jordan Park	1280	BP
	5593	South Boggy @ Dittmar Park near Strickland	640	EP
	5594	Shoal Creek @ Shady Oak Court	1280	EP
	5595	Tannehill Creek @ Bartholomew Park near Berkman Dr	640	BP
	5596	Tannehill Creek upstream storm pipe in Givens Park	1280	BP
	5598	Taylor Slough South in Reed Park @ Footbridge	120	EP
	5601	Walnut Trib @ North Star Greenbelt	64	EP
	5606	Williamson Creek in Battle Bend Park	64	BP
	5582	Blunn Creek @ Rosedale	640	BP
Reference	633	Barton Creek Ephemeral 3	0	EP
	5581	Bee Creek Downstream Loop 360	320	EP
	5583	Blunn Creek Upstream of Cow Trough Spring	320	EP
	5589	Common Ford Trib us Bridge @ Common Ford Ranch	1280	EP
	5590	Fort Branch downstream Tura Ln	1280	BP
	5597	Taylor Slough North waterfall pool @ Mayfield Park	320	EP
	5599	Little Walnut Trib @ Gus Garcia Park	640	BP
	5600	Walnut Trib @ Lincolnshire and Garnaas	128	EP
	5602	Walnut Creek downstream Old Manor Rd	1280	BP
	5603	West Bouldin Creek @ Audrey Court	320	EP
	5604	West Bouldin Creek in West Bouldin Greenbelt	1280	EP
	5605	Williamson Creek @ Wagon Bend Trail	0	BP

Parameters collected were selected based on current literature and ecological theory. Every parameter was thought to be directly or indirectly related to some aspect of how a riparian zone functions properly to protect water quality and erosion in adjacent water bodies. A list of the functional parameters can be seen in Table 2. Duncan (2012) discussed these parameters in greater detail.

Table 2: Parameter list for evaluation of riparian zone function.

Parameter Type	Functional Parameter	Rational
Reach	Macroalgae Cover	The percentage of macroalgae cover has been linked to the amount of nutrients in the water column, type of substrate, and amount of available light at a site (Mabe 2007).
	Diatoms	Specific species of diatoms have been linked to increased nutrients in the water column while other species have been linked to low concentrations of nutrients (King and Winemiller 2009). Community composition metrics will be calculated to help determine aquatic function in the study reach.
	Gap Frequency	Using heuristic models Weller *et. al.* (1998) showed that the best predictor of nutrient discharge into a stream was the frequency of gaps along stream buffers when the buffers were assumed to be highly retentive.
	Bank Stability	Unstable vertical banks increase erosion and sediment loading to streams (Stacey *et al.* 2006).
	Large Woody Debris	Streams with adequate LWD generally have greater habitat diversity, a more natural stream shape, and greater resistance to flood events (Stacey *et al.* 2006).
Creek	In-stream Canopy Cover	The amount of solar shading provided by adjacent and in stream riparian vegetation is critical for maintenance of temperature refugia. Decreased streambank vegetation cover, increased channel width, and reduced stream depth increases exposure, raises water temperatures and impacts aquatic life (Stacey *et al.* 2006).
	Entrenchment Ratio	Channel entrenchment (incision) is an indication of floodplain connection and overbank flow (Rosgen 1994; Stacey *et al.* 2006). The absence of floodplain connectivity lowers the water table, reduces nutrient availability, decreases plant germination, growth, and survivorship, and may lead to riparian vegetation loss and invasion of upland species (Stacey *et al.* 2006).
Riparian	Soil Compaction	Increasing soil compaction can reduce the soil's ability to function for structural support, water and solute movement, and restrict root growth (USDA 2008b). Compaction can result in shallow rooted plants and poor plant growth, reduced vegetative cover, increased erosion, and reduction in water infiltration (USDA 2008b).
	Soil Moisture	Hydrologic changes associated with urbanization often result in lower water tables and drier more aerobic soil conditions (Gift *et al.* 2010). These changes can result in reduced denitrification and altered plant species composition (Gift *et al.* 2010; Sung *et al.* 2011).
	Soil pH	Soil pH influences the solubility of nutrients, microbial decomposition, and most chemical transformations in the soil (USDA 1998).
	Plant Cover and Structural Diversity	High cover and structural diversity of vegetation (groundcover, understory, and canopy) indicates a productive plant community, high species diversity, adequate food resources and habitat for wildlife, and reduced flood impacts along banks (Stacey *et al.* 2006).

Table 2 (cont.): Parameter list for evaluation of riparian zone function.

Parameter Type	Functional Parameter	Rational
Riparian	Hardwood Demography	Size and age class distribution of the dominant tree species indicates recruitment success and disturbance intervals. Missing age classes is often a result of disruptions to natural ecosystem processes and can result in successional changes and species loss (Stacey *et al.* 2006). Dominant species exert the most influence, and thus the greatest functional changes will occur if the abundance of these species is altered (Richardson *et al.* 2007).
	Recruitment/Succession	The understory (sapling) community reflects current ecological condition of habitat; while overstory (tree) communities are reminders of past environmental condition (Woosley *et al.* 2005).
	Riparian Zone Width	A wide riparian buffer has been shown to filter pollutants, control erosion, prevent flooding, and provide habitat and nutrient inputs into the stream (Barbour *et al.* 1999; Fischer and Fischenich 2000).
	Ratio (Riparian Zone Width to SPTH)	The site potential tree height (SPTH) is defined as the average maximum height to which a dominant tree will grow if left undisturbed (Sedell *et. al.* 1993). FEMAT found that factors including root strength, litter fall, shading, and coarse material input were protected with a buffer of 1 SPTH while factors that affected the microclimate would be protected with a buffer of 3 SPTH (FEMAT 1993).

Macroalgae data was collected using the City of Austin zig-zag method (WRE SOP 2010). A total of 50 sample points were spaced evenly through the 100 m transect. Each sample point consisted of recording a categorical percent cover of filamentous algae in a 1 ft^2 area and the dominant substrate type within that same area. The categories of percent cover and substrate can be found in Table 3. Macroalgae sampling was not conducted in dry or non-flowing regions.

Table 3: Macroalgae cover and substrate categories.

Percent Cover Category	Percent Cover	Substrate Type No.	Substrate
0	No Cover	0	Bedrock
1	< 5%	1	Boulder (> 256 mm)
2	5 – 24%	2	Large Cobble (128 – 256 mm)
3	25 – 49%	3	Cobble (64 – 128 mm)
4	50 – 75%	4	Gravel (32 - 64 mm)
5	>75%	5	Unstable Substrate

Macroalgae cover data was analyzed using the Kruskal-Wallis procedure to detect differences between substrate types. The minimum p-value method was used to detect which substrates grew a different percentage of filamentous algae when the Kruskal-Wallis test showed significant differences. Mann-Whitney tests were then conducted on percent algae cover data to distinguish any differences between degraded and reference sites.

Diatoms were collected from periphyton containing rocks (epilithon) of riffle habitat within the 100 m in stream transect. Three rocks were taken randomly from appropriate riffle habitat if available. A small Petri dish (47 cm^2) and a sharp scoring object were used to mark the area of each rock to be sampled. This area was scraped with a wire brush, and the particulate matter deposited into a shallow collecting pan. A sufficient quantity of ambient creek water was used to flush epilithon from each rock. After each scraping, the wire brush was also thoroughly rinsed for finer plant material into the collection pan. After the rocks were scraped and rinsed, the contents of the collection pan were poured into a darkened or opaque bottle (approximately 125 mL) and more water was used to flush all remaining particulate matter from the pan (WRE SOP 2010). Samples were kept on ice until they arrived at the lab where they were preserved with 10% buffered formalin.

Diatom samples were analyzed by Winsborough Inc. Five hundred diatom "cells" were identified and enumerated in each sample. Counts were log transformed and used in a non-metric Multidimensional Scaling (nMDS) ordination to explore the difference in community structure between sites with a reference riparian zone and a degraded riparian zone. NMDS is a distance based procedure that ordinates data by dissimilarity (Minchin 1987, Clarke 1993). The Bray-Curtis dissimilarity was used as the distance measure in the ordination as this has been demonstrated as a robust measure for ecological communities (Faith and Norris 1989). Three dimensions were used for the ordination as the stress value decreased from 0.198 in a two dimension ordination to 0.103 in the three dimension ordination but did not greatly decrease by adding more dimensions. The minimum stress value is obtained when there are adequate dimensions in the ordination to fully display distances between units. Two hundred iterations were used in the ordination although the minimum stress was reached after roughly twenty iterations. Diatom samples were then grouped by site type/ecoregion and used in an Analysis of Similarity (ANOSIM) and a similarity percentage (SIMPER) analysis to statistically investigate community structure differences. An ANOSIM tests whether the samples within a group are more similar in composition than samples in other groups (Clarke and Warwick 1994). The null hypothesis defined in an ANOSIM is that there is no difference between samples from various groups, thus a p-value less than $\alpha=0.05$ implicates that there is evidence that the samples within a group are more similar than would be possible by random chance. The sample statistic can range from -1 to 1 with -1 indicating that samples are outside the defined groups, 0 representing random patterns of similarity, and 1 representing tight clustering within each group. The SIMPER procedure defines the contribution of each species to the similarity within groups and the dissimilarity between groups (Clarke 1993).

To further investigate diatom community structure at reference sites and degraded sites, diatom species were classified into metrics discussed in literature (Table 4) (Porter 2008). Wilcoxon rank sum tests were used to test significant differences between diatom metrics at reference and degraded sites.

Table 4: List of diatom metrics used in this project with a description.

Metric Label	Metric Description
NF_YS	Abundance of nitrogen-fixing diatoms
NF_NO	Abundance of non nitrogen-fixing diatoms
TR_OL	Abundance of oligotrophic diatoms
TR_OM	Abundance of oligo/mesotrophic diatoms
TR_MT	Abundance of mesotrophic diatoms
TR_ME	Abundance of meso/eutrophic diatoms
TR_ET	Abundance of eutrophic diatoms
TR_EY	Abundance of indifferent diatoms
DTN_HI	Abundance of high TN indicator species
DTN_LO	Abundance of low TN indicator species
DTP_HI	Abundance of high TP indicator species
DTP_LO	Abundance of low TP indicator species
SP_OL	Abundance of diatoms that prefer O2 saturation >85%; BOD <2 mg/L
SP_BM	Abundance of diatoms that prefer O2 saturation 70-80%; BOD 2-4 mg/L
SP_AM	Abundance of diatoms that prefer O2 saturation 25-70%; BOD 4-13 mg/L
SP_AP	Abundance of diatoms that prefer O2 saturation 10-25%; BOD 13-22 mg/L
ON_AL	Abundance of taxa intolerant to organic N
ON_AH	Abundance of taxa tolerant to organic N
ON_HF	Abundance of taxa requiring periodic elevated organic N
PC_MT	Abundance of the most tolerant diatoms
PC_LT	Abundance of less tolerant diatoms
PC_SN	Abundance of sensitive diatoms
OT_AH	Diatoms that require nearly 100% DO saturation
OT_PH	Diatoms that require >75% DO saturation
OT_MD	Diatoms that require >50% DO saturation
OT_LW	Diatoms that require >30% DO saturation
OT_VL	Diatoms that require 10% DO saturation or less
MT_YS	Abundance of motile algae
MT_NO	Abundance of non-motile algae

Throughout the entire 100 m in-stream transect, the number of gaps in riparian vegetation along both banks of the creek were counted. A riparian gap was defined as a void in vegetation 1 m in length where surface runoff had an unimpeded path to the stream channel. As both sides of the creek were sampled, there was a total of 200 possible gaps along the transect. The tallied number of riparian gaps was divided by 2 to obtain a site percentage for gap frequency. In a similar manner the number of 1 m sections along both banks where there was active erosion was counted and divided by 2 to obtain a site percentage for bank stability. Large Woody Debris pieces (LWD) were counted along the in-stream transect as well. LWD was defined as wood that was partially exposed to the water or located within the active stream channel and was at least 6" in diameter and 3' long.

In-stream canopy cover was taken at the center point of the creek at 5, 50, 95 m on the in-stream transect. This was done by holding a densiometer level, 12" – 18" in front of the body so the operators head was just outside of the grids. The number of quarter squares occupied by vegetation was counted at each location along the transect and recorded as percent cover. The entrenchment ratio was determined by dividing the width of the flood prone area by the bankfull width at 5, 50, and 95 m on the in-stream transect. The flood prone area was defined by measuring the width of the channel at twice bankfull depth. Bankfull corresponds to the start of the floodplain and is indicated by a break in slope from the channel bank, a change in vegetation from bare surfaces or annual wetland species to perennial water-tolerant or upland species, and from a change in the size distribution of surface sediments.

Soil compaction, soil moisture, and soil pH were measured as close to the center of each 10 m by 10 m quadrat as possible. Soil compaction was recorded as the pounds per square inch on the penetrometer at a depth of 3 inches of soil. Soil moisture and soil pH were recorded from a soil probe tester. The probe was chemically cleaned prior to each quadrat sampling and inserted into the ground for approximately two minutes prior to taking a reading to allow the probe to stabilize. The soil probe would not activate if there was no moisture in the soil. In such instances, a zero was recorded for soil moisture and DI water was poured over the soil so that a pH reading could be obtained. A total of three measurements were taken within each quadrat for soil compaction, soil moisture and soil pH.

The percent cover of vegetation in the canopy, understory, and groundcover layers for each 10 m by 10 m riparian quadrat was collected for the plant cover and structural diversity functional parameter. The canopy layer was greater than 5m high, the understory was 0.5 to 5 m high, and the ground cover was less than 0.5m high. The dominant hardwood species was noted for each quadrat and the presence or absence of this particular species in multiple age classes was recorded (seedlings, immature, mature, snags). The dominant species was determined to be the hardwood species that had the highest percent cover in the quadrat. Seedlings were defined as less than 12 inches and having sprouted within the last year, immatures were greater than 12 inches but had yet to reach half of its potential mature height, matures were approaching their maximum height and displayed full developed canopy, and snags were dead trees with little to no vegetation and a reduced canopy. For the recruitment/succession parameter the hardwood species with the highest number of seedlings in each quadrat was recorded. For each riparian quadrat, a measuring tape was run perpendicular to the in-stream transect starting at bankfull and ending at the edge of the riparian zone buffer. This distance was recorded in meters. Finally each riparian zone width was divided by the site potential tree height (SPTH) of each associated riparian quadrat. The SPTH was defined as the average maximum height to which a dominant tree would grow if left undisturbed and can be found on the USDA plant database.

Functional Parameter Scores

A second objective of this project was to be able to score sites as having a healthy or a dysfunctional riparian zone. Each functional parameter was transformed into a 0 to 100 scale with 0 implicating lack of function and 100 being equivalent to the best possible riparian functionality. Correlation and principal component analysis (PCA) was performed on the parameter scores to determine redundancy of parameters. A final site score was computed as the average of all functional parameter scores deemed necessary after analysis. The functional parameters were scored in the following ways.

Macroalgae Cover
Data was collected in such a way that did not allow for a simple average to be used in scoring. Instead scores were calculated in two separate parts. The first portion was simply the sum of all categorical cover measurements at a site without including measurements taken on unstable substrate (Cover Sum). Unstable substrate was not included because there was statistically less algal growth on unstable substrates than any other substrate. The higher this portion of the score the more algae covered the reach. It is natural for some algae to grow in a creek and it was thought that these counts should be weighted on reference sites. To do this, the count of each cover classification was averaged for reference sites only and changed into a percentage of 100. Reference sites on average showed no filamentous algae cover in 51.96% of the sites, less than 5% algae cover in 2.34% of the sites, 5-25% algae cover in 9.56% of the sites, 25-50% algae cover in 10.10% of the sites, 50-75% algae cover in 6.38% of the sites, and greater than 75% cover in 2.65% of the sites. If a site was missing a particular cover classification then the percentage associated with that cover (Missing Class Sum) was added to the overall site score. The overall sum was divided by the number of cells that were not unstable substrate. Finally the 5^{th} (p5) and 95^{th} (p95) percentile was found. The equation was as follows:

Algae Metric = (Cover Sum + Missing Class Sum)/percent not unstable substrate
Algae Score = 100 – (100*(Algae Metric – p5)/(p95 – p5))

As the data set consisted of a range of functional and dysfunctional riparian zone sites, the score was constructed to range from 0 to 100. The algae scores was subtracted from 100 as lower numbers are related to higher riparian zone function. The 5^{th} and 95^{th} percentile were used in the equations so that extreme outliers would not be involved in the formulation of the scores.

Diatoms
No score was calculated for diatoms as the raw data did not show significant differences between reference and degraded sites for diatom community structure or diatom metrics.

Gap Frequency, Bank Stability, Large Woody Debris
Once each functional parameter was calculated, the 5^{th} (p5) and 95^{th} (p95) percentile was found for each parameter data set. These numbers were used in the equations below to create the parameter scores:

Gap Frequency Score = 100 – (100*((Data – p5) / (p95 – p5)))
Bank Stability Score = 100 – (100*((Data – p5) / (p95 – p5)))
Large Woody Debris Score = 100*((Data – p5) / (p95 – p5))

As the data set consisted of a range of functional and dysfunctional riparian zone sites, the scores were constructed to range from 0 to 100. The gap frequency and bank stability scores were subtracted from 100 as lower numbers are related to higher riparian zone function (Weller *et. al.* 1998, Stacey *et. al.* 2006). The 5th and 95th percentile were used in the equations so that extreme outliers would not be involved in the formulation of the scores.

In-stream Canopy Cover and Entrenchment Ratio
Once each functional parameter was calculated, the average parameter was found for each site. The 5th (p5) and 95th (p95) percentile was found for each parameter data set. These numbers were used in the equations below to create the parameter scores:

Canopy Cover Score = 100*((Data – p5) / (p95 – p5))
Entrenchment Score = 100*((Data – p5) / (p95 – p5))

As the data set consisted of a range of functional and dysfunctional riparian zone sites, the scores were constructed to range from 0 to 100. The 5th and 95th percentile were used in the equations so that extreme outliers would not be involved in the formulation of the scores.

Soil Compaction, Soil Moisture, Soil pH
The average of each functional parameter at a site was calculated. The 5th (p5) and 95th (p95) percentile was found for each parameter data set. These numbers were used in the equations below to create the parameter scores:

Soil Compaction Score = 100 – (100*((Data – p5) / (p95 – p5)))
Soil Moisture Score = 100*((Data – p5) / (p95 – p5))
Soil pH Score = 100 – (100*((Data – p5) / (p95 – p5))

As the data set consisted of a range of functional and dysfunctional riparian zone sites, the scores were constructed to range from 0 to 100. The soil compaction and soil pH scores were subtracted from 100 as lower numbers are related to higher riparian zone function (USDA 2008b, USDA 1998). The 5th and 95th percentile were used in the equations so that extreme outliers would not be involved in the formulation of the scores.

Plant Cover and Structural Diversity
This functional parameter was collected in three different parts; canopy cover, understory cover, and groundcover. The average of each part was calculated at each site. The 5th (p5) and 95th (p95) percentile was found for canopy cover, understory, and ground cover. These numbers were used in the equations below:

Canopy = 100*((Data – p5) / (p95 – p5))
Understory = 100*((Data – p5) / (p95 – p5))
Groundcover = 100*((Data – p5) / (p95 – p5))

While increasing the amount of canopy cover, understory cover, and groundcover should all increase riparian function, eventually a limit is reached where adding more canopy cover will negatively affect understory cover or groundcover. This phenomenon is similar for adding understory and groundcover. Thus the optimum percent cover of each part needed to be determined. In order to find the optimum percentages, the average canopy cover, understory cover, and groundcover were calculated using reference sites only. These averages were used as weights for each score. The scores were multiplied by the weights and added together to achieve a plant cover and structural diversity (PCSD) site score:

$$\text{PCSD Score} = 0.465*\text{Canopy} + 0.311*\text{Understory} + 0.224*\text{Groundcover}$$

Hardwood Demography

There are many aspects that could be considered in the scoring of this parameter because the dominant species greatly affects the ecology present within a riparian zone. A few of these will be discussed later in this report. Currently the hardwood demography score was based on the Wetland Indicator Status of the dominant species in each quadrat. The Wetland Indicator Status of a species was first made official by the US Fish and Wildlife in 1988; however, the US Army Corps of Engineers have recently released an updated list of plants and their corresponding Wetland Indicator category (Lichvar and Kartesz 2009). Wetland Indicator Status categories include OBL (almost always a hydrophyte, rarely in uplands), FACW (usually a hydrophyte but occasionally in uplands), FAC (commonly occurs as either a hydrophyte or non-hydrophyte), FACU (occasionally a hydrophyte but usually in uplands), and UPL (rarely a hydrophyte, almost always in uplands). A species was placed into one of these categories based on frequency and abundance in wetlands versus uplands and taking into consideration the landscape percentage of wetlands to uplands in a region (Lichvar and Minkin 2008). If the dominant species in an area is considered to be almost always a hydrophyte than the area that species is located in is thought to be well saturated. This would imply a high water table within the riparian zone and functional infiltration, lowering the risk of floods and raising the sustainability of base flow in adjacent creeks.

Trees within each Wetland Indicator category were ranked on a 0 to 100 scale using the following ranks: OBL = 100, FACW = 80, FAC = 60, FACU = 40, UPL = 20, and no dominant species = 0. Each quadrat of a site was given a Wetland Indicator score based on the dominant species present. The number and type of age class present within the quadrat was also important and scored on a 0 to 100 scale. The age class score was calculated as the sum of the age class values within a quadrat. The seedling, mature tree, and immature tree age classes were given an age class value of 28.57 while the snag age class was given 14.29 as a value. This was done to put emphasis on the seedling, mature tree, and immature tree age classes as they are more closely related to the current conditions of a site while maintaining a maximum score of 100. The average Wetland Indicator score and average age class was calculated for each site. The final hardwood demography score was defined as the average of the Wetland Indicator site score and the site age class score.

Recruitment/Succession

Similar to the hardwood demography functional metric, there are multiple aspects of the riparian health that could be used in the recruitment score. The current score only contained information regarding the Wetland Indicator Status of the overall dominant species and the dominant species in the seedling age classification. The Wetland Indicator categorical scores used in this score were the same as for the hardwood demography score (see above). At each quadrat, the Wetland Indicator score for the dominant seedling was multiplied by two and the Wetland Indicator score of the overall dominant species was subtracted from the product. This was done to put emphasis on the current Wetland Indicator Status of the seedling age class and not just score the difference between the old dominant species and current dominant species status. The 5th (p5) and 95th (p95) percentile was found and used in the equation below:

Change in Wetland Status = 2*(Dominant seedling status) – (overall dominant status)
Recruitment = 100*((Change in Wetland Status – p5) / (p95 – p5))

Riparian Zone Width
The maximum measured riparian zone width was 100 m. If the riparian zone extended further than this the width was just noted as > 100 m. Thus the riparian zone width score was the exact same value as the average riparian zone width at a site. This is equivalent to dividing the average riparian zone width by the maximum (100 m) and multiplying by 100 in order to scale the score.

Ratio (Riparian Zone Width to SPTH)
After the riparian zone width to Site Potential Tree Height (SPTH) ratio had been calculated, the average at each site was divided by 3, the optimal riparian zone width to SPTH ratio suggested by the Federal Ecosystem Management Team (FEMAT), and multiplied by 100 to scale the score. FEMAT found that a buffer width of 3 SPTH would protect not only root strength, litter fall, shading, and course material but the microclimate within the riparian zone as well (FEMAT 1993). Benefits of riparian buffers that were wider than 3 times the SPTH were miniscule thus the maximum for this parameter was set to 3.

Statistical analysis
Statistical analysis for raw data involved with the macroalgae and diatom functional parameters has been described above. Wilcoxon rank sum tests were used on raw data for the other functional parameters and on all functional parameter metric scores to test significant differences between diatom metrics at reference and degraded sites. Correlation and Principal Component Analysis (PCA) was performed on the functional parameter scores in order to determine redundancy in data collection and scoring capability. An overall riparian functional score was given to each site as the average of the non-redundant functional parameter scores. SAS9.2 was used for all statistical analysis and $\alpha = 0.05$ was set as the alpha level for all tests.

Results

Macroalgae cover data collected from each site was coupled with substrate type in order to determine if the available substrate had any affect on the amount of filamentous algae cover. The percentage of points classified as a particular type of substrate for reference and degraded site are in Table 5. There was significantly less algae cover on unstable substrate than any other type of substrate ($p<0.0001$, Figure 1), with no significant difference between other types of substrate.

Table 5: Total number of macroalgae sample points in each substrate.

Substrate #	Substrate	Counts		Percentage	
		Degraded	Reference	Degraded	Reference
0	Bedrock	169	67	24	17
1	Boulder (> 256 mm)	2	6	0	2
2	Large Cobble (128 - 256 mm)	164	54	23	14
3	Cobble (64 - 128 mm)	95	43	14	11
4	Gravel (32 - 64 mm)	137	124	20	31
5	Unstable Substrate	133	106	19	27

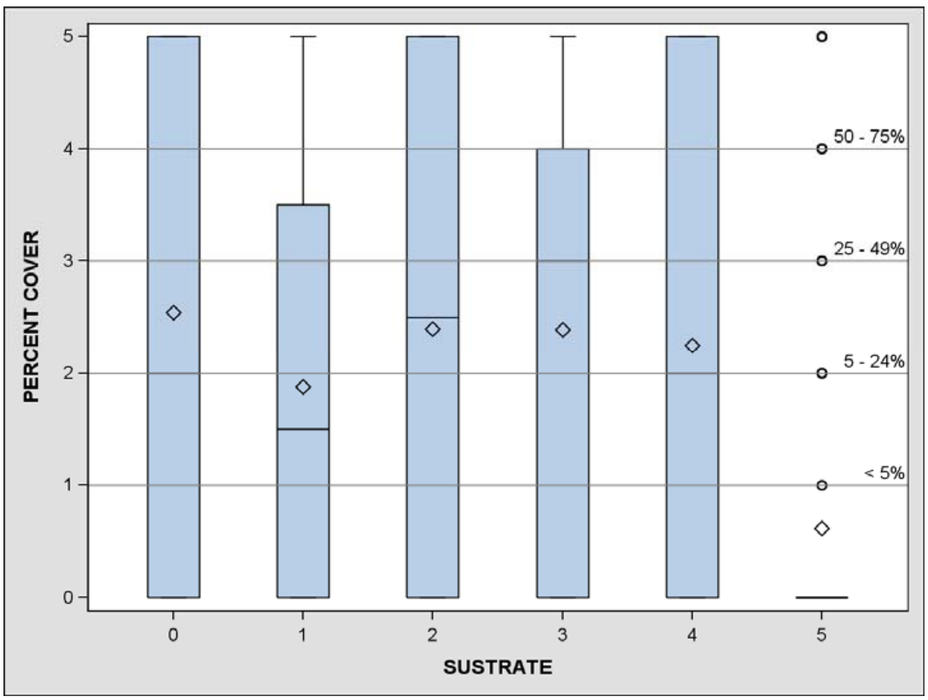

Figure 1: Boxplot of macroalgae cover for various types of substrate within the study. Percent cover classifications include 0= none, 1= <5%, 2= 5-24%, 3= 25-49%, 4= 50-75%, 5= >75%. Diamonds are means and lines are medians.

The percentage of unstable substrate data points sampled at the reference sites was higher than the percentage sampled in degraded sites. As the algal cover is less on unstable substrates, further analysis of the macroalgae cover data excluded unstable substrate data points in order to keep from displaying an artificial difference in filamentous algae growth between reference and degraded sites. Filamentous algae cover was significantly less at reference sites compared to degraded sites ($p<0.0001$). According to the median, algal cover at reference sites was less than 5% while algal cover at degraded sites was between 25 and 49% (Figure 2A). The macroalgae cover score was significantly lower at degraded sites than reference sites ($p < 0.05$) (Figure 2B). This was expected as the score should accurately represent the raw data. If the raw data was significantly different then the score should be as well. The mean score for degraded sites was 50 while the average score for reference sites was 80.

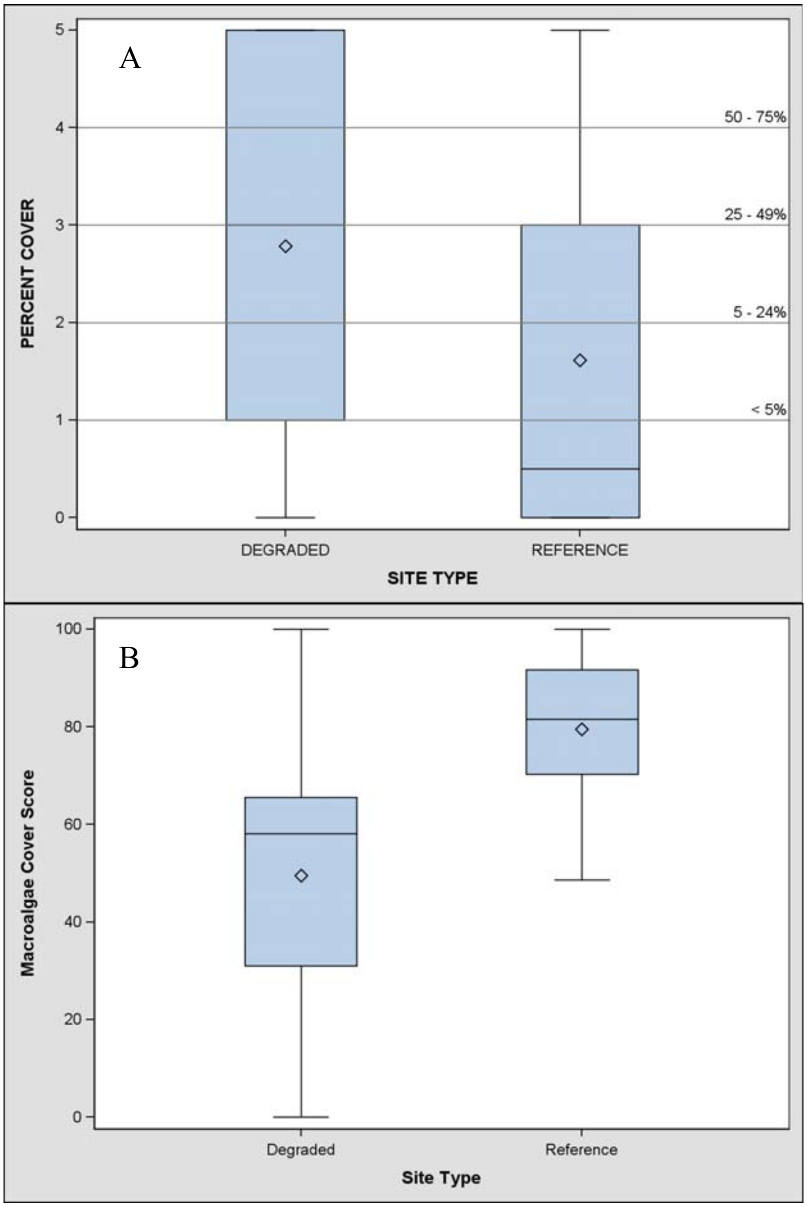

Figure 2: Boxplot of A) categorical macroalgae percent cover based on the zig-zag collection method B) macroalgae cover scores for degraded and reference sites. Percent cover classifications include 0= none, 1= <5%, 2= 5-24%, 3= 25-49%, 4= 50-75%, 5= >75%. Diamonds are means and lines are medians.

Diatoms

Non-metric Multidimensional Scaling (nMDS) ordination did not show differences between diatom communities in reference sites and degraded sites, as there is no clear distinction between light grey/blue samples and yellow/red samples (Figure 3). It should be noted that there could be some difference between diatom communities in the Blackland Prairie reference samples (blue) and the communities in the Edward's Plateau reference samples (light grey). As sites become degraded in either ecoregion the diatom communities tend to be more centralized in the ordination so that you cannot distinguish between them. Overall ANOSIM results showed that samples within a group were not more similar than samples from the other groups (p-value = 0.105), but the pairwise comparison of groups showed that the diatom communities in the Blackland Prairie degraded sites were significantly different than communities in the Edward's Plateau reference sites (Table 6). Diatom communities in Blackland Prairie reference sites were almost significantly different from communities in Blackland Prairie degraded sites and Edward's Plateau reference sites (Table 6). *Achnanthidium minutissimum* was present in almost all samples and in many cases was the dominant species, which could imply that many of these sites recently experienced disturbance. SIMPER analysis showed that Edward's Plateau reference samples contained large populations of *Amphora pediculus*, *Amphora inariensis*, *Achnanthes lanceolata*, and *Nitzschia amphibia*; while Blackland Prairie degraded samples contained populations of *Encyonema silesiacum*, *Nitzschia amphibia*, and *Nitzshia amphibioides*; and Blackland Prairie reference samples contained large populations of *Eolimna minima*, *Cocconeis placentula*, *Achnanthes lanceolata*, and *Nitzshia amphibioides*.

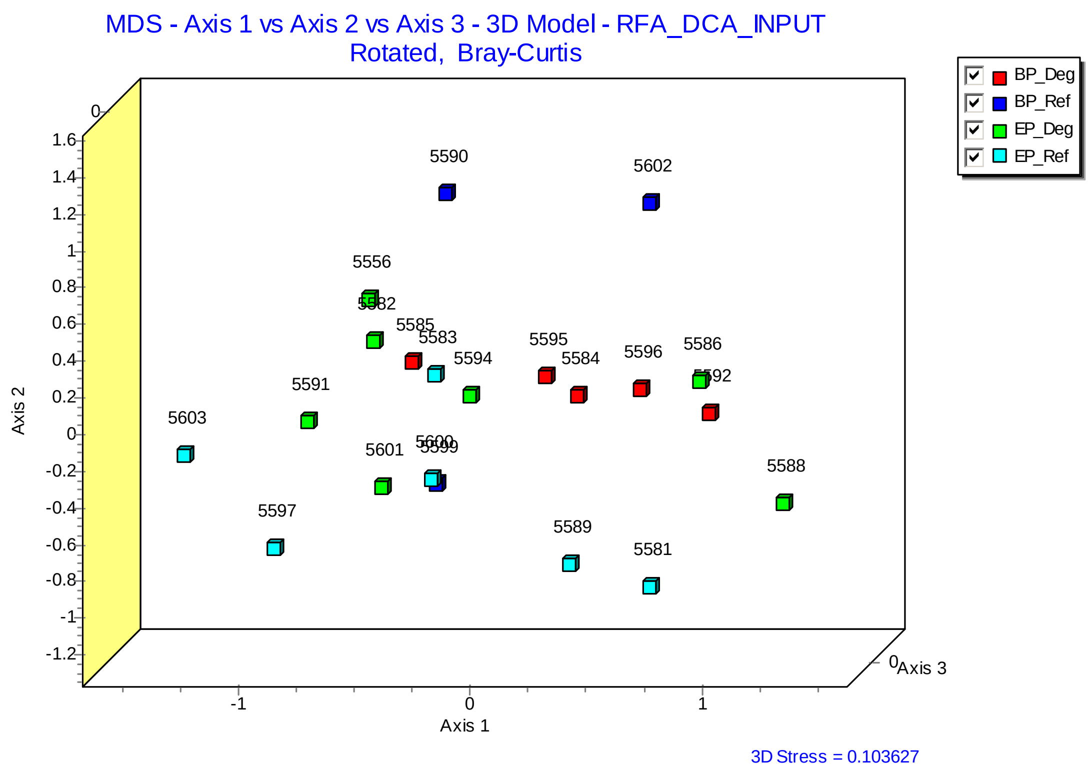

Figure 3: Non-metric Multidimensional Scaling (nMDS) ordination of diatom community structures. Legend labels are as follows: BP_Deg = Blackland Prairie Degraded, BP_Ref = Blackland Prairie Reference, EP_Deg = Edward's Plateau Degraded, EP_Ref = Edward's Plateau Reference.

Table 6: Pairwise ANOSIM results for diatom community structure.

1st Group	2nd Group	P Value	Sample Stat.
Blackland Prairie Degraded	Blackland Prairie Reference	0.053571	0.415385
Blackland Prairie Degraded	Edward's Plateau Degraded	0.406566	0.0138249
Blackland Prairie Degraded	Edward's Plateau Reference	0.034632	0.328
Blackland Prairie Reference	Edward's Plateau Degraded	0.433333	-0.00793652
Blackland Prairie Reference	Edward's Plateau Reference	0.059524	0.407407
Edward's Plateau Degraded	Edward's Plateau Reference	0.663	-0.0568783

With no significant differences between diatom communities in reference and degraded sites it was not surprising that the analysis showed no significant differences between reference and degraded sites for the majority of the diatom metrics (Table 7). In most of the metrics the standard deviation in both site types was high which made it difficult to detect any differences. It may be that the riparian zone functionality is not a major contributing factor to the health of the diatom community. The only diatom metric that showed a significant difference was the number of motile individuals in each site type. Reference sites contained significantly less motile individuals than degraded sites (p-value = 0.0235). Higher abundance of motile individuals implies that the organisms present can move towards the surface as the habitat is silted over and has been used as a measure of siltation (Barbour et. al. 1999). Thus the degraded riparian sites seemed to have more siltation problems than the reference sites.

Table 7: Wilcoxon rank sum results for diatom metrics between reference and degraded sites. P-values and median ± standard deviation are presented.

Metric Label	P-VALUE	Reference	Degraded
NF_NO	0.1176	482 ± 95	490 ± 54
TR_OL	0.8339	47 ± 49	25 ± 22
TR_OM	1.0000	6 ± 4	3.5 ± 16
TR_MT	0.2266	9 ± 25	2 ± 9
TR_ME	0.6834	13.5 ± 23	10 ± 29
TR_ET	0.8221	133 ± 145	165 ± 106
TR_EY	0.7900	132 ± 104	130 ± 97
DTN_HI	0.8333	54 ± 155	80.5 ± 119
DTN_LO	0.8608	116 ± 129	123 ± 100
DTP_HI	0.5294	38 ± 152	44 ± 93
DTP_LO	0.7791	65 ± 131	99 ± 101
SP_OL	0.3541	53 ± 43	14 ± 44
SP_BM	0.7524	180 ± 129	194.5 ± 95
SP_AM	0.7791	26 ± 146	47 ± 66
SP_AP	0.2117	19 ± 15	34 ± 41
ON_AL	0.4188	51.5 ± 45	18 ± 44
ON_AH	0.5521	189 ± 130	248.5 ± 63
ON_HF	0.9701	36 ± 146	51 ± 49
PC_MT	0.1234	6 ± 6	23 ± 46
PC_LT	0.3664	110 ± 136	76 ± 72
PC_SN	0.6997	197 ± 126	212 ± 101
OT_AH	0.9161	121 ± 131	119 ± 115
OT_PH	0.9091	68.5 ± 63	42.5 ± 84
OT_MD	0.6241	41 ± 141	60 ± 64
OT_LW	0.2218	7 ± 6	23 ± 48
OT_VL	0.2192	12 ± 6	4 ± 4
MT_YS	**0.0235**	**59 ± 75**	**144 ± 85**
MT_NO	0.3849	349 ± 102	298 ± 86

It was thought that the substrate difference between Edward’s Plateau sites and Blackland Prairie locations may be a contributing factor to the variation in diatom community structures, so a Kruskal-Wallis test was used to compare any differences that may occur between the Ecoregions and site type. Again the only metric to show a significant difference was the number of motile individuals present. There were significantly less motile individuals in the Blackland Prairie reference sites than in at the Blackland Prairie Degraded locations (Table 8). It appears that even with the explanatory factor of ecoregion, the riparian health of a site had little to do with diatom community metrics. These metrics will probably be more responsive to direct water quality measurements such as total phosphorus, total nitrogen, or dissolved oxygen levels. While it is not recommended that diatom community structure be continued in this study as there is no clear difference between diatom communities in reference sites and degraded sites, further research should be done in understanding the diatom community structures around Austin. SIMPER analysis showed some community structure differences between sites but it is unclear if a functional difference exists based on the presence or absence of the different species.

Table 8: Kruskal-Wallis results for diatom metrics between reference and degraded sites within the Edward's Plateau and Blackland Prairie Ecoregions. Medians ± standard deviations are presented.

Metric Label	P-VALUE	Edward's Plateau		Blackland Prairie	
		Reference	Degraded	Reference	Degraded
NF_NO	0.1557	480 ± 88	487 ± 67	487 ± 127	498 ± 19
TR_OL	0.9570	47 ± 49	25 ± 25		25 ± 0
TR_OM	0.5647	6 ± 4	4 ± 18		2 ± 0
TR_MT	0.4059	9 ± 24	3.5 ± 13	29 ± 38	2 ± 1
TR_ME	0.7107	13.5 ± 27	10.5 ± 37	13.5 ± 8	6 ± 7
TR_ET	0.5604	127 ± 115	194.5 ± 118	174 ± 217	93 ± 69
TR_EY	0.7763	104 ± 111	150 ± 84	199 ± 115	110 ± 114
DTN_HI	0.9803	80.5 ± 116	93 ± 141	54 ± 245	68 ± 81
DTN_LO	0.5910	66.5 ± 131	108 ± 86	199 ± 143	131 ± 111
DTP_HI	0.8175	38 ± 95	37 ± 122	53 ± 246	68 ± 39
DTP_LO	0.7262	40 ± 133	93 ± 83	199 ± 144	105 ± 126
SP_OL	0.3918	83.5 ± 40	17 ± 56	10 ± 4	6 ± 11
SP_BM	0.9470	179 ± 108	201 ± 101	198 ± 192	173 ± 96
SP_AM	0.9738	33.5 ± 97	34 ± 83	21 ± 236	53 ± 39
SP_AP	0.5762	16.5 ± 17	29.5 ± 56	19 ± 16	34 ± 14
ON_AL	0.4204	86.5 ± 44	25 ± 55	13.5 ± 6	8 ± 17
ON_AH	0.7859	183.5 ± 115	252 ± 59	203 ± 184	178 ± 72
ON_HP	0.9602	31.5 ± 99	44 ± 57	52 ± 229	58 ± 42
PC_MT	0.1231	5 ± 3	7 ± 69	16 ± 9	35 ± 13
PC_LT	0.6306	125.5 ± 94	86 ± 90	63 ± 227	66 ± 28
PC_SN	0.9384	201.5 ± 98	232 ± 123	197 ± 198	183 ± 72
OT_AH	0.9739	114 ± 143	105 ± 110	198 ± 132	133 ± 133
OT_PH	0.7207	68.5 ± 66	31 ± 102	57.5 ± 74	54 ± 48
OT_MD	0.7502	74.5 ± 95	63 ± 75	41 ± 231	54 ± 44
OT_LW	0.1884	5.5 ± 3	7 ± 72	16 ± 8	35 ± 13
OT_VL	0.2178	8 ± 0	2 ± 3	16 ± 0	8 ± 4
MT_YS	**0.0492**	**63.5 ± 88**	**112 ± 90**	**35 ± 18***	**181 ± 78***
MT_NO	0.6931	328 ± 88	360 ± 97	453 ± 145	282 ± 78

* Diatom metric significantly different between these site types.

Creek Parameters

The biology within the stream at the riparian site locations was only one portion of the parameters examined. Several physical characteristics in and along the creek were also compared between the reference and degraded sites. Three out of five of these parameters were shown to be significantly different for the raw data and the calculated scores (Table 9).

Table 9: Wilcoxon rank sum results for in-stream parameters and scores of the analysis. Medians and standard deviation are presented for raw data while means are presented for scores.

PARAMETER	DEGRADED	REFERENCE
GAP FREQUENCY**	49 ± 32	1.0 ± 5.2
BANK INSTABILITY	34 ± 23	10.5 ± 30
LARGE WOODY DEBRIS**	0.0 ± 1.6	3.0 ± 3.3
ENTRENCHMENT RATIO	1.9 ± 1.1	2.1 ± 3.0
INSTREAM COVER (%)**	60 ± 40	94.5 ± 24
GAP FREQUENCY SCORE**	47	97
BANK INSTABILITY SCORE	53	72
LARGE WOODY DEBRIS SCORE**	12	43
ENTRENCHMENT SCORE	19	32
INSTREAM COVER SCORE**	50	82

* Significantly different with $p < 0.05$.
** Significantly different with $p < 0.01$.

The gap frequency was significantly higher at degraded sites rather than reference sites ($p<0.0001$, Figure 4A). A highly connected riparian zone is an area with very few gaps in vegetation. Gaps can create flow pathways for surface runoff which allow for water to directly enter a water body without being slowed down and filtered by the riparian vegetation. The value of nutrient retention is greatly dependent on the vegetation present in the riparian zone; however, even in highly retentive riparian zones increasing the number of gaps can increase the nutrient load to the adjacent water body (Weller *et. al.* 1998, Baker *et. al.* 2006). An effective riparian buffer is one that is both connected and retentive. The degraded sites have a high gap frequency and are thus not well connected. These sites could be prone to increased nutrient loading into the creek from surface runoff, which could be detrimental to the biology within the creek. The gap frequency scores were also significantly different between degraded and reference sites ($p<0.0001$, Figure 4B). Scores close to 100 indicate sites with a lower gap frequency. Most reference sites scored well for gap frequency, while the degraded sites had a wide range of scores. The scores represented the raw data well.

Bank instability was not significantly different between degraded and reference sites ($p = 0.1089$, Figure 5A). Thus there would be no difference in active erosion or sediment loading to the streams. While Figure 5A showed a slightly higher amount of active erosion in the degraded sites, the variation within degraded sites and reference sites indicated that the functionality of the immediate riparian zone was not a good explanatory variable for active erosion at a site. The bank stability scores were not significantly different between degraded and reference sites either ($p = 0.0772$, Figure 5B). This should not be continued to be collected for the Riparian Functional Assessment as there is no clear distinction between reference and degraded sites.

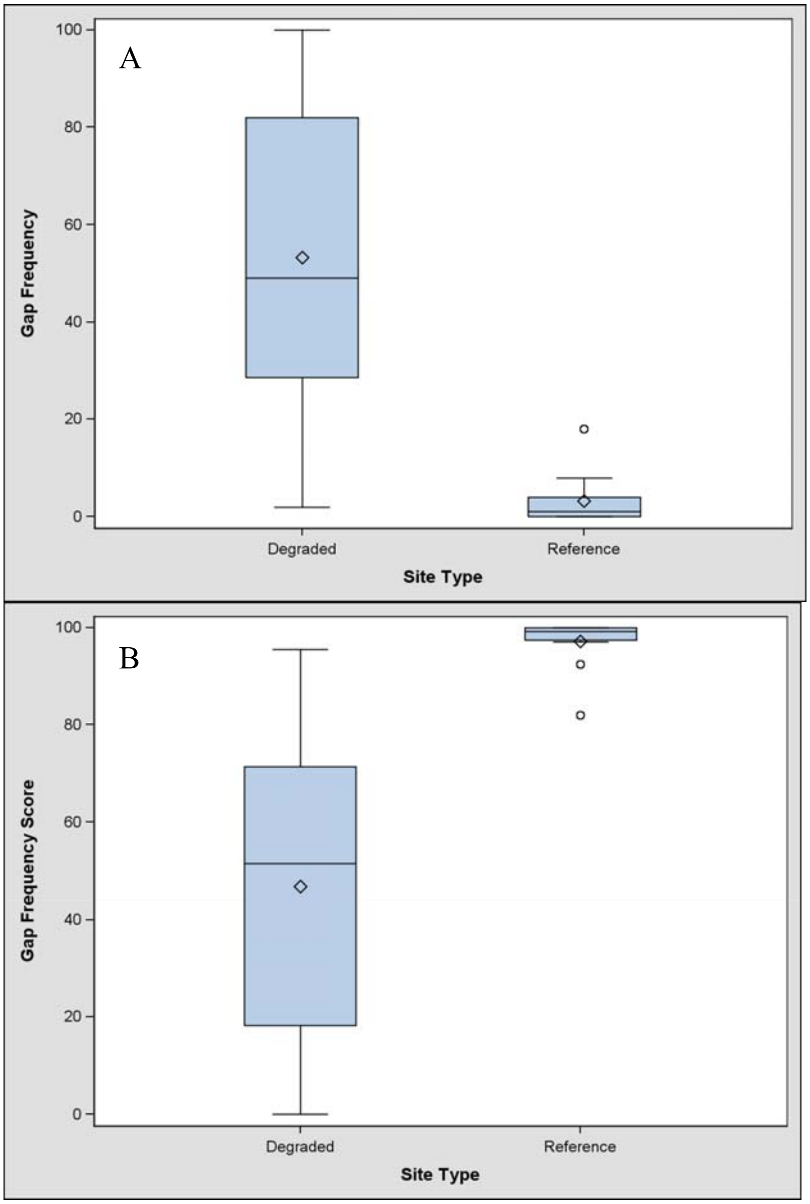

Figure 4: Boxplot of A) raw gap frequency and B) gap frequency scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

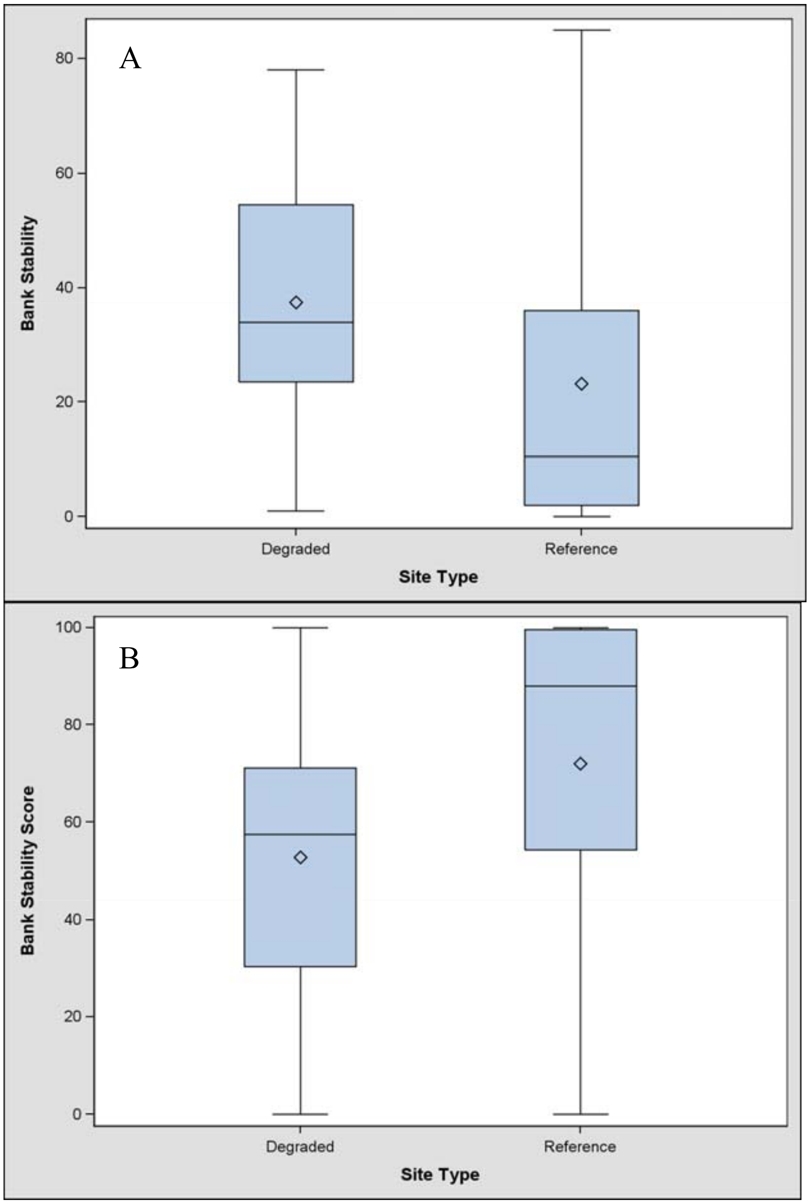

Figure 5: Boxplot of A) raw bank instability and B) bank instability scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

The amount of large woody debris was significantly greater in reference sites ($p = 0.0007$, Figure 6A). Thus the reference sites are more likely to have higher habitat diversity. This includes important nursery habitat, protective cover, and feeding zones for fish as well as protective zones for certain macroinvertebrates. Reference sites also are more likely to have a natural channel shape with higher resistance to high water events because they have more large woody debris in the channel (Stacey *et al.* 2006). The large woody debris scores were significantly higher in reference sites ($p = 0.0076$, Figure 6B). However, the scores for reference sites varied greatly with a large number of scores falling below 50.

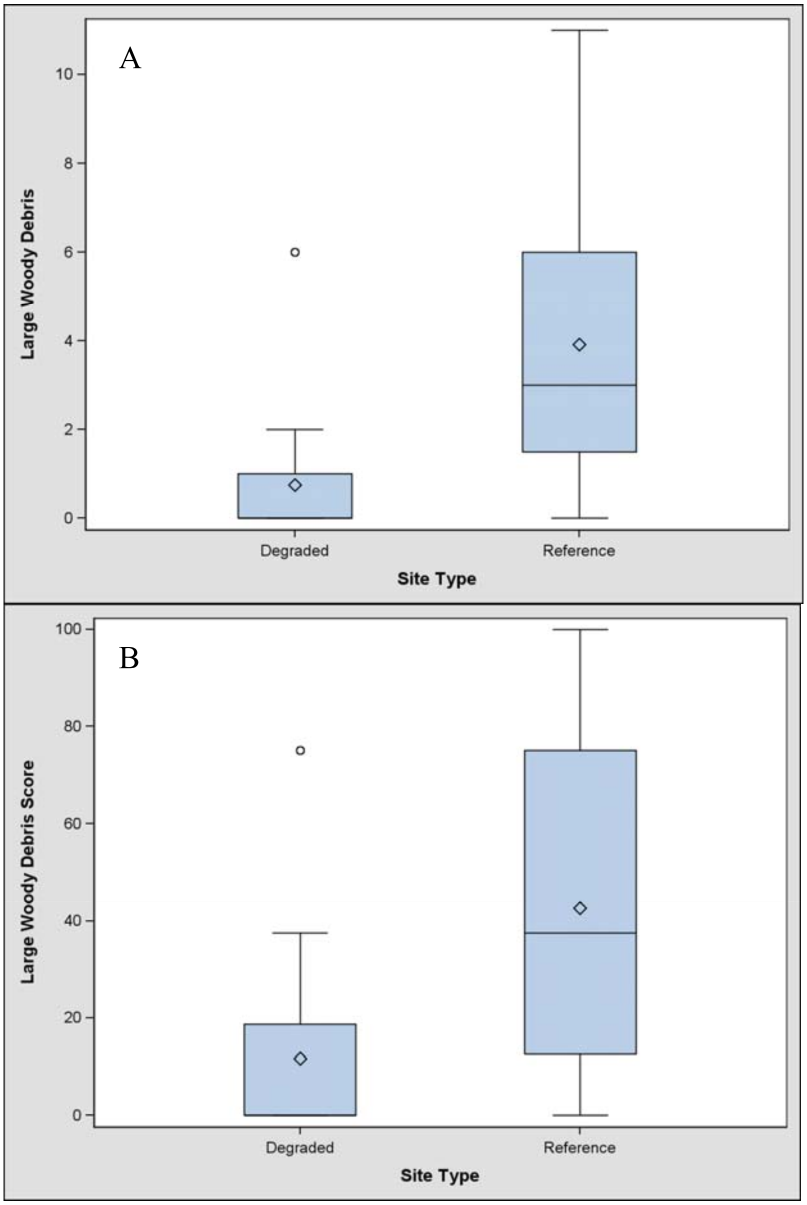

Figure 6: Boxplot of A) raw large woody debris counts and B) large woody debris scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

The entrenchment ratio was not significantly different between degraded and reference sites ($p = 0.2009$, Figure 7A). The higher the entrenchment ratio the more floodplain connection will occur. Flood waters disperse across the floodplain instead of remaining strictly in the channel in low entrenchment scenarios. Increased entrenchment, or lack of floodplain connection, can lead to a lower water table, reduced nutrients to the floodplain vegetation, reduced plant germination, and possibly the loss of riparian vegetation (Stacey *et al.* 2006). However, an entrenched system has an entrenchment ratio of < 1.4 and a slightly entrenched system has an entrenchment ratio of 2.2 or greater (Rosgen 1994). While some of these creek locations were entrenched, with the minimum entrenchment ratio in degraded sites of 1.12 and the minimum ratio in reference sites of

1.21, many were only slightly entrenched. The median ratios for both degraded and reference sites were close to being classified as only slightly entrenched. Similar to bank stability and diatom community, the functionality of the immediate surrounding riparian zone does not seem to greatly impact the entrenchment. A cumulative riparian buffer zone along the watershed may help to better explain why an area is entrenched or this could be a highly hydrologic parameter. The entrenchment score was also not significantly different between degraded and reference sites (p = 0.7276, Figure 7B).

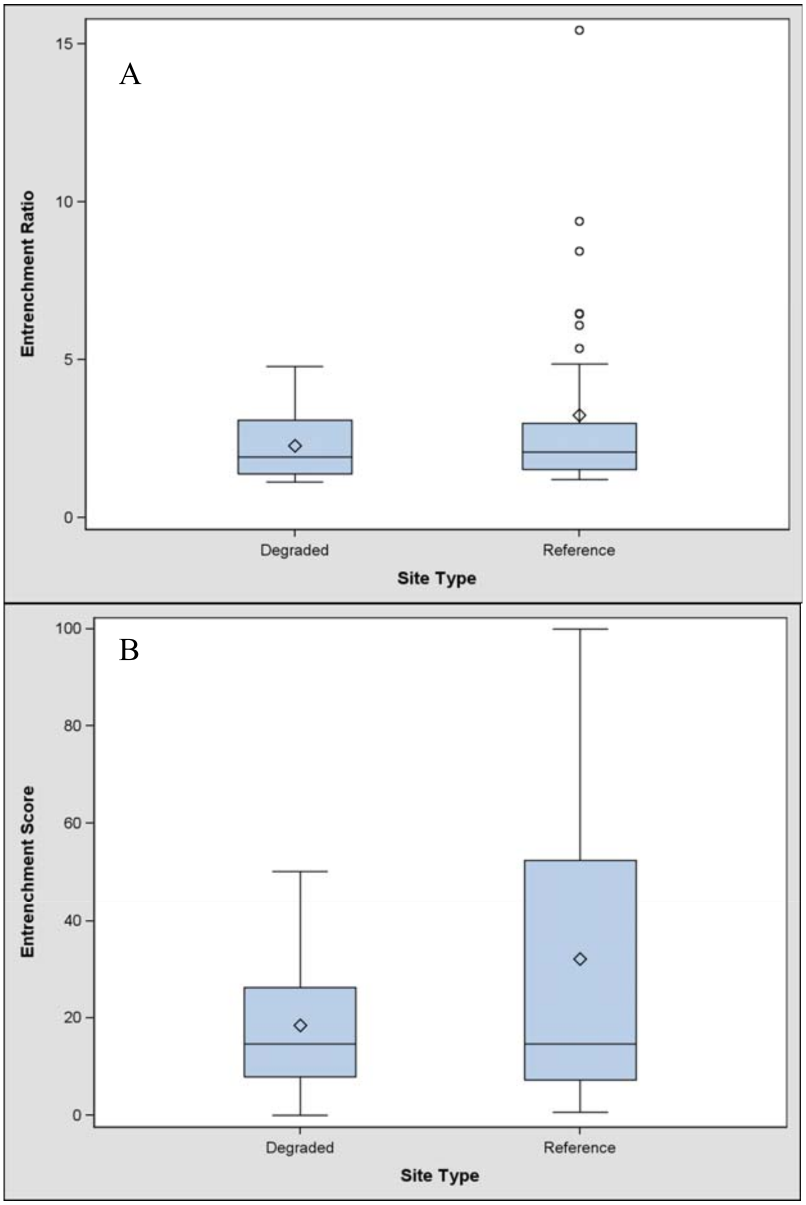

Figure 7: Boxplot of A) the raw entrenchment ratio and B) entrenchment scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

The instream canopy cover from adjacent and aquatic vegetation was significantly higher in reference sites (p = 0.0002, Figure 8A). Canopy cover over the stream acts as a buffer from solar radiation. Water temperatures can vary widely with increased solar exposure and can become very warm in the afternoon. A few of the degraded sites had a vegetative buffer, but many were prone to high temperature swings throughout the day which could cause increased productivity in the form of algal blooms, alter decomposition rates of organic material, disrupt macroinvertebrate life-cycles, alter metabolic rates of organisms in the creeks, or in extreme cases be lethal to certain individuals in a creek (Stacey *et al.* 2006, Woolsey *et al.* 2005, Petts 2000). Thus, degraded sites are at risk of a variety of negative environmental impacts. The instream canopy cover score was significantly higher at reference sites (p = 0.0075, Figure 8B).

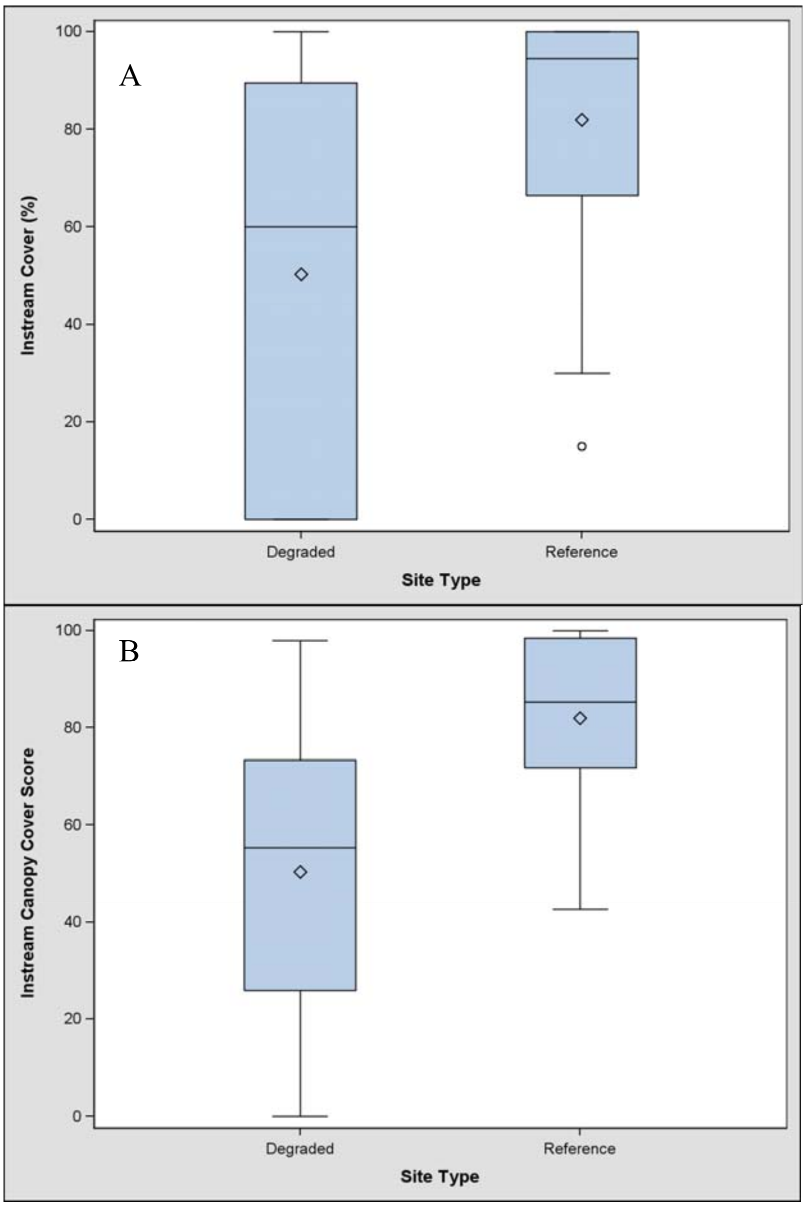

Figure 8: Boxplot of A) instream canopy cover and B) instream canopy cover scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

Riparian Parameters

Parameters collected in the actual riparian zone included soil compaction, soil moisture, soil pH, canopy cover, understory cover, groundcover, riparian zone width, the ratio of the riparian zone width to the site potential tree height (SPTH), dominant hardwood species, and the dominant hardwood recruitment species. The dominant hardwood species and dominant hardwood recruitment species could not be analyzed statistically. The other riparian parameters and all riparian parameter scores showed significant differences between degraded and reference sites (Table 10). While the soils present in the Edward's Plateau and Blackland Prairie Ecoregions are considerably different, there were no significant differences noted between the Ecoregions in any of the riparian parameters.

Table 10: Wilcoxon rank sum results for riparian parameters and scores of the analysis. Medians and standard deviation are presented for raw data while means are presented for scores.

PARAMETER	DEGRADED	REFERENCE
SOIL COMPACTION (psi)**	350 ± 292	200 ± 92
SOIL MOISTURE (%)**	65 ± 32	82 ± 23
SOIL pH (standard unit)**	6.9 ± 0.3	6.8 ± 0.3
CANOPY COVER (%)**	28 ± 27	70 ± 21
UNDERSTORY COVER (%)**	10 ± 21	40 ± 23
GROUNDCOVER (%)**	70 ± 27	25 ± 28
RIPARIAN ZONE WIDTH (m)**	0.5 ± 21	100 ± 28
RATIO (RIPARIAN ZONE WIDTH/SPTH)**	0.01 ± 1.79	1.67 ± 2.77
SOIL COMPACTION SCORE**	37	89
SOIL MOISTURE SCORE*	43	78
SOIL pH SCORE*	25	48
PLANT COVER AND STRUCTURAL DIVERSITY SCORE**	45	72
RIPARIAN ZONE WIDTH SCORE**	7	83
RATIO SCORE**	10	73
HARDWOOD DEMOGRAPHY SCORE*	49	63
RECRUITMENT/SUCCESSION SCORE*	43	71

* Significantly different with $p < 0.05$.
** Significantly different with $p < 0.01$.

Soil compaction was significantly higher at the degraded sites ($p < 0.0001$, Figure 9A). While some soil compaction can be beneficial by slowing the transport of water through the soil, high soil compaction can be very detrimental to soil functionality (USDA 2008). The soil compaction values in the degraded sites have the potential to cause plants to develop poorly with shallow roots. This could cause increased erosion from lack of vegetative cover. Surface runoff could also be increased at these sites as the water may no longer be able to infiltrate the soil (USDA 2008). As the scores were flipped, soil compaction scores were significantly higher at the reference sites ($p < 0.0001$, Figure 9B).

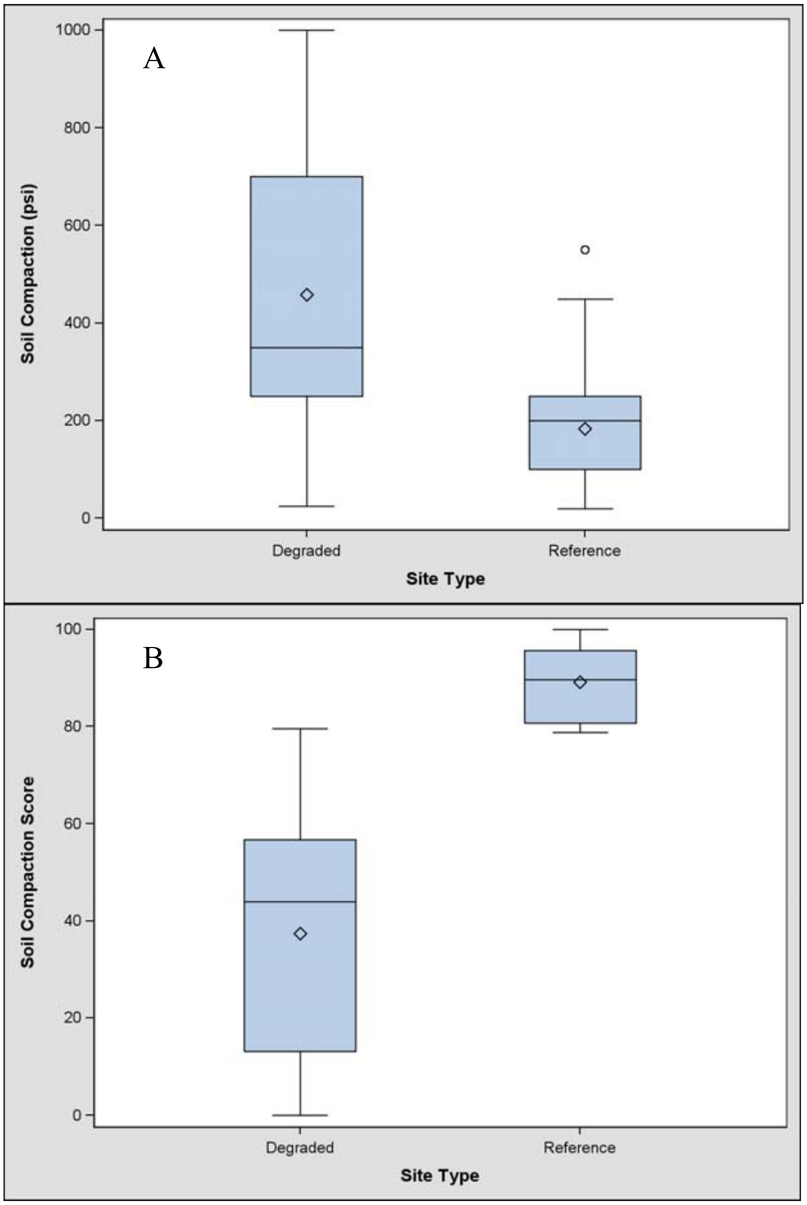

Figure 9: Boxplot of A) soil compaction and B) soil compaction scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

Soil moisture was significantly higher at reference sites ($p < 0.0001$, Figure 10A). One of the hydrologic effects of urbanization is that the water table can be lowered which would create dry, aerobic conditions (Gift *et al.* 2010). The soil moisture data collected showed that a degraded riparian zone does not buffer the water loss in the soil as well as a fully functional riparian zone. Thus the soil at the degraded sites was most likely under more aerobic conditions than that of the soil in the reference sites. The significance of this difference in soil moisture is that increased aerobic conditions have the net effect of increasing nitrate production and decreasing nitrate consumption in the riparian zone (Gift *et al.* 2010, Groffman *et al.* 2002). Functional riparian zones are natural sinks for nitrate in urban watersheds, but the increased aerobic conditions cause riparian zones to no longer act as sinks. This would allow for increased nitrate loading into creeks. Thus the degraded sites are at a higher risk of increased nitrate loading and the environmental impacts that are related to increased nitrate values in the water column. The soil moisture scores were also significantly higher in reference sites (p = 0.0114, Figure 10B).

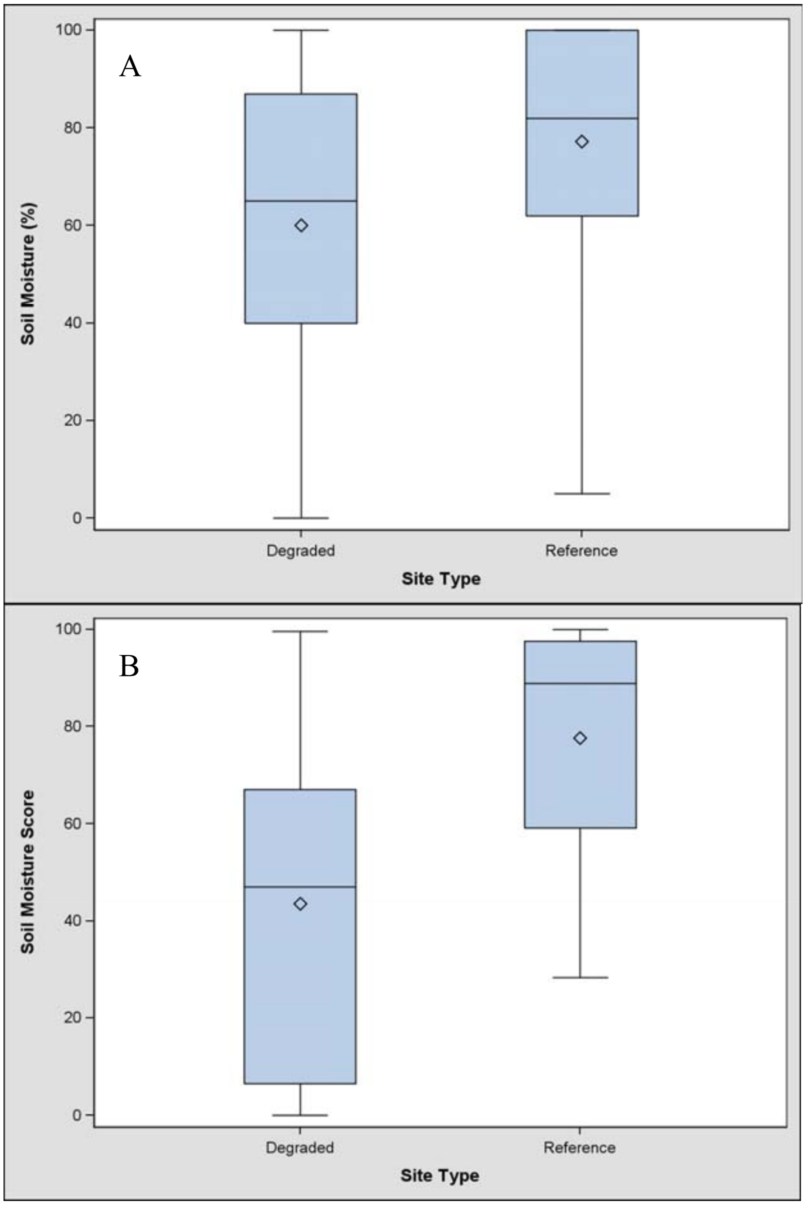

Figure 10: Boxplot of A) soil moisture and B) soil moisture scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

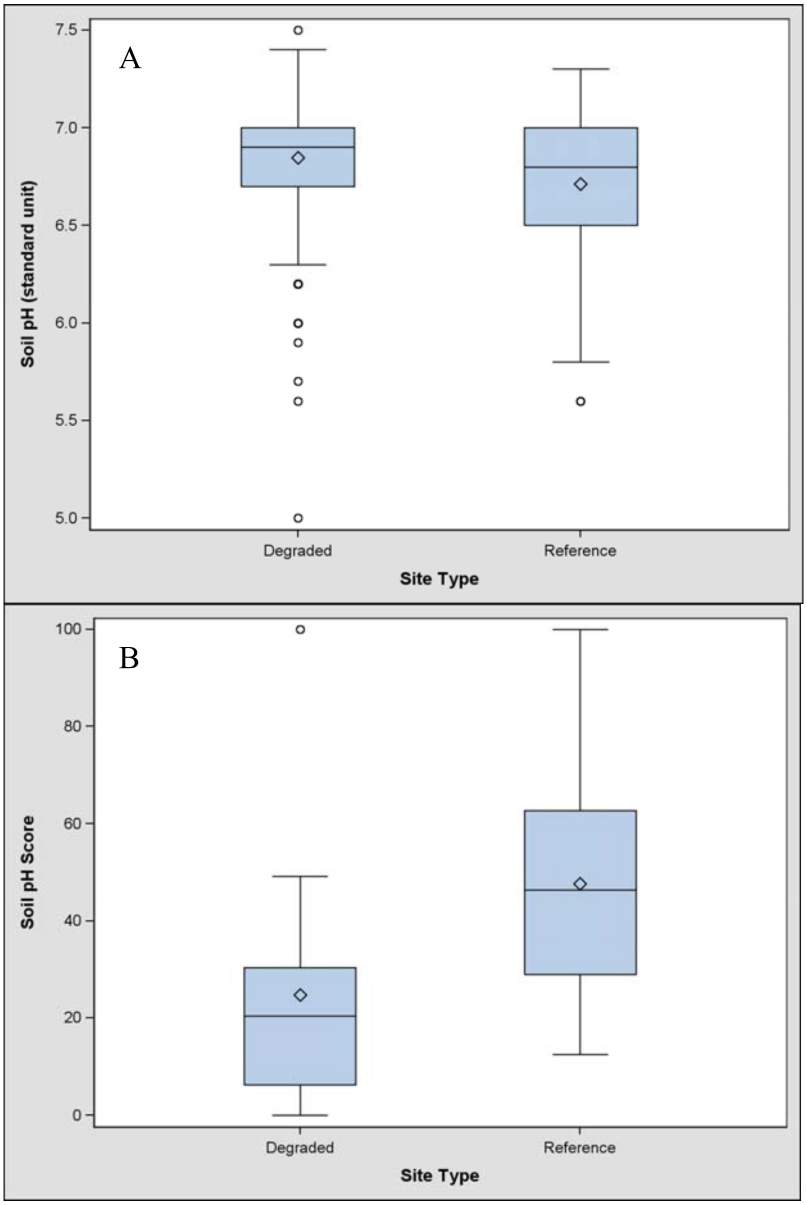

Figure 11: Boxplot of A) soil pH and B) soil pH scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

Soil pH was significantly lower at reference sites ($p < 0.0001$, Figure 11A). While differences in soil pH can lead to different levels of nitrogen and phosphorus retention, phosphatase activity, denitrification rates, or microbial decomposition; the difference in pH levels between reference and degraded sites may not be large enough to alter any of these biological processes (Lyons *et al.* 1998, Amador *et al.* 1997, Ashby *et al.* 1988, USDA 1998). In fact, phosphatase activity has been found to be optimal at a soil pH range of 6.5 to 6.9 which is the range of soil pH for most of the sites in this study (Amador *et al.* 1997). The soil pH scores were significantly higher in reference sites ($p = 0.0113$, Figure 11B). Given that the pH difference was probably not biologically significant, this functional parameter may not be optimal for use in distinguishing degraded and reference riparian zone for this project.

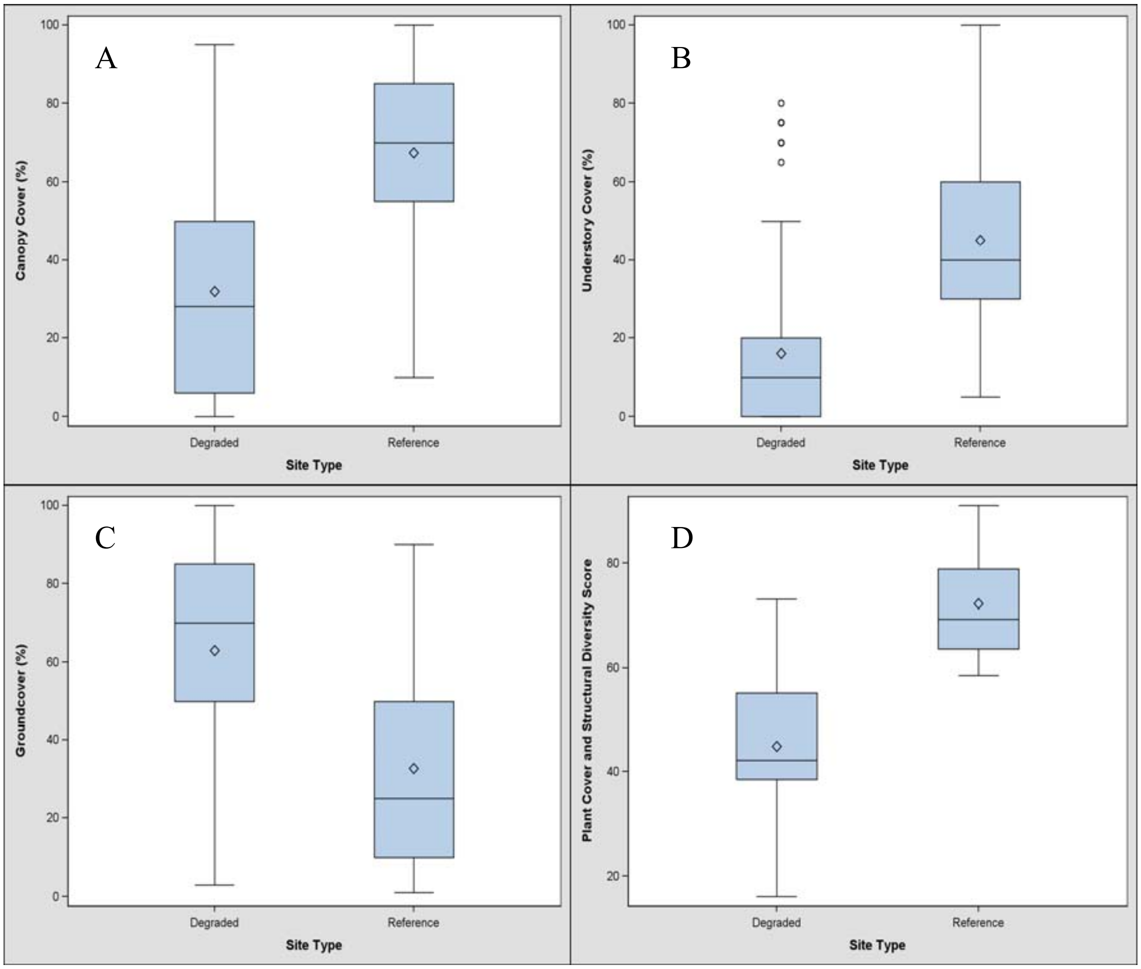

Figure 12: Boxplots of A) percent canopy cover, B) percent understory cover, C) percent groundcover, and D) plant cover and structural diversity scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

Percent canopy cover, understory cover, groundcover, and the plant cover and structural diversity score were all significantly different between degraded and reference sites ($p < 0.0001$). Reference sites had more canopy cover and understory cover than did the degraded sites (Figure 12A, 12B). Woody vegetation such as trees and shrubs in the canopy and understory layers are important to the health and function of the riparian zone. Woody Vegetation is better at stabilizing severely eroded banks, contributes more large woody debris to the stream, has increased nitrogen uptake, is better at reducing temperature extremes, provides greater organic matter to the stream, replenishes the soil in the riparian zone, and provides greater habitat for sensitive macroinvertebrate taxa than riparian locations dominated by grassy vegetation (Lyons *et al.* 2000). Canopy and understory plants also slow surface runoff greatly and help to reduce peak flows in the creeks and filter out the nutrients in this water before it reaches the creek (USDA 2002). While the groundcover does help in these processes, the layer has less of an impact than does the canopy or the understory. The degraded sites had a larger amount of groundcover than the reference sites, but that was to the detriment of the canopy and understory layers (Figure 12C). Thus the reference sites had higher plant cover and structural diversity scores because they had more functional percent covers in all three categories and not just the ground cover (Figure 12D).

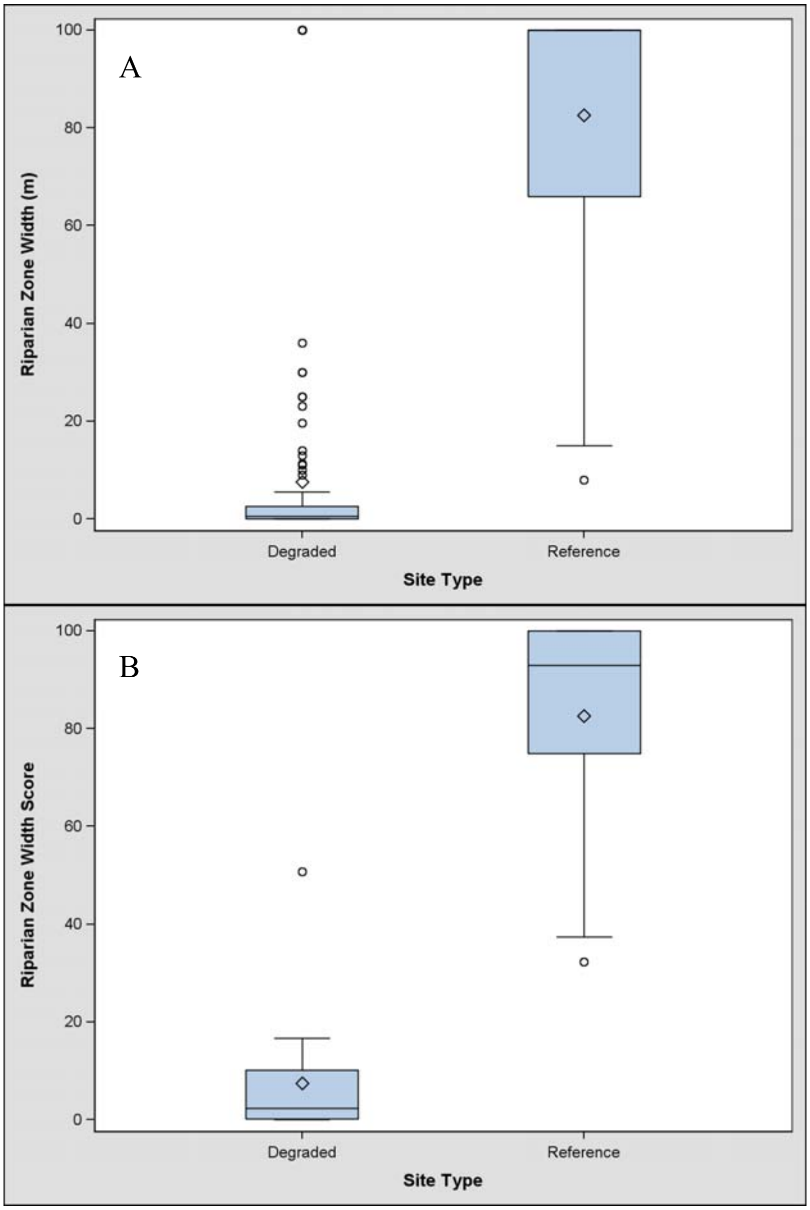

Figure 13: Boxplots of A) riparian zone width and B) riparian zone width scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

The riparian width and the riparian width scores were both significantly higher in the reference sites ($p < 0.0001$, Figure 13). Wider riparian buffers filter more pollutants, control erosion more, prevent more flooding, and provide better habitat for wildlife than riparian with a narrower width (Barbour 1999, Fischer and Fischenich 2000). Some of the degraded sites had little to no riparian buffer which will prevent these sites from protecting against erosion and degradation to water quality.

The ratio of the riparian zone width to the site potential tree height was significantly higher in reference sites ($p < 0.0001$, Figure 13A). This functional parameter can be used as a proxy to riparian width as the idea behind the parameter is that riparian forests profoundly impact the stream habitat up to and exceeding a lateral distance of one tree height (FEMAT 1993). Functions protected by riparian width are also protected by this functional parameter. The ratio scores were also significantly greater in reference sites ($p < 0.0001$, Figure 13B).

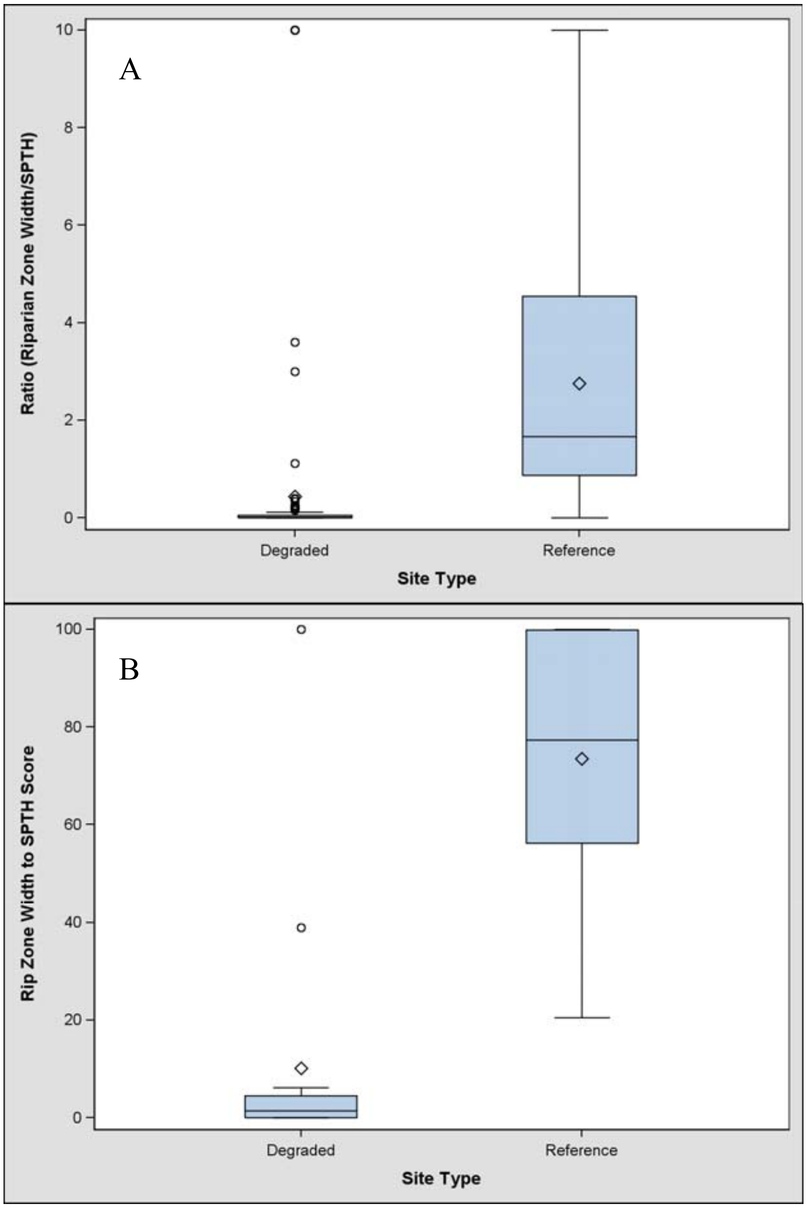

Figure 13: Boxplots of A) riparian zone width to SPTH and B) riparian zone width to SPTH scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

The hardwood demography score was significantly higher in reference sites than in degraded sites ($p = 0.0434$, Figure 14). This means that the dominant species in the reference sites were more hydrophilic and present in a higher number of age classes than the dominant species at the degraded sites. From this we can deduce that the water table at reference sites was probably higher and that the floodplain may be recharged more steadily than at degraded sites. The higher number of age classes implies that the reference sites are sustaining themselves naturally and going through less disturbance than the degraded sites. As the dominant species of tree has great impact on a site, this metric could tell us much more about a site than aspects of the hydrology. There have been efforts to relate tree species to the amount of erosion control they provide for a system (Jones-Lewey 2011). Unfortunately there is no national tree list for how much stability a particular species adds to a system. There has also been study on how much a species of tree effects nutrient cycling in riparian zones (Scott and Binkley 1997, Schimel *et al.* 1998, Lovett and Rueth 1999, Lawrence *et al.* 2000). It has been shown that the species of tree present in a plot could affect how much nutrients leach out of the riparian zone, especially nitrogen (Lovett *et al.* 2004, Templer *et al.* 2003, 2005). However, there are limited results as to what characteristic of a tree species accounts for this difference in nutrient leaching. The best correlation to date was the ratio of soil nitrogen to soil carbon at a site (Gundersen *et al.* 1998, Dise and Wright 1995, Goodale and Aber 2001, Lovett *et al.* 2002, 2004, Christenson *et al.* 2009). In the future the City of Austin should attempt to categorize trees found in the area into stability categories and further study the difference in nutrient leaching under different dominant canopies to incorporate into this functional score.

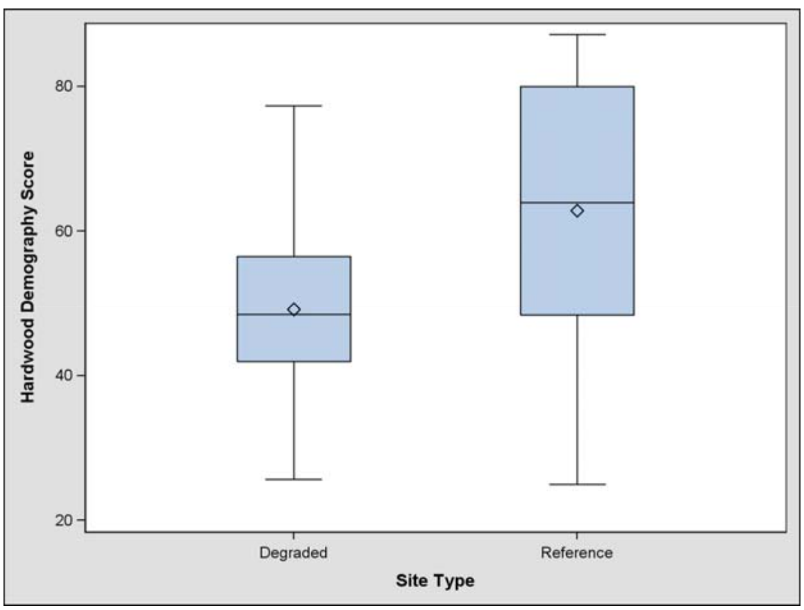

Figure 14: Boxplot of hardwood demography scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

The recruitment/succession score was significantly higher in reference sites (p = 0.0105, Figure 15). This means that the reference site plant communities were being replenished with more hydrophilic species than the degraded site plant communities. Because seedlings of hydrophilic species dominated the riparian floor, the systems must continually be hydrologically recharged by floods to maintain an elevated water table at these sites. The recruitment of less hydrophilic species in the degraded sites means that water tables were lower and will probably stay lower until some sort of restoration is done (passive or active). Like the demography score, further information should be considered for this score so that stability and nutrient retention could be measured in addition to water availability at a site.

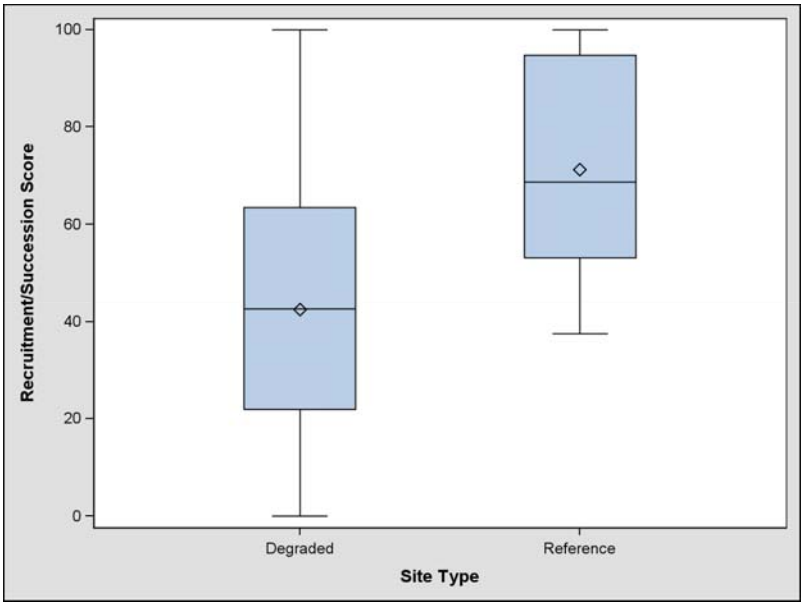

Figure 15: Boxplot of recruitment/succession scores at degraded and reference sites. Means are diamonds, medians are horizontal lines, and outliers are circles.

Correlation and Principal Component Analysis (PCA) was done on the functional parameter scores in order to identify redundancy. In the correlation analysis, the riparian width score and the ratio of riparian width to site potential tree height showed a strong positive correlation (r = 0.8916). Due to this high correlation and the fact that the ratio of riparian zone width to SPTH is based on the riparian zone width, it was decided that the scores were redundant and the ratio score was dropped from further analysis. The stability score and the entrenchment score were not shown to be statistically different between degraded and reference sites, so they were not prime candidates for PCA and were discarded prior to running the analysis. Lastly, the macroalgae scores were temporarily taken out of analysis due to the missing data at the dry site locations. The initial PCA for component 1 and component 2 showed that the plant cover and structural diversity score and the gap score could be redundant due to their close proximity (Figure 16). Similar trends in proximity for the two scores were seen when looking at graphs of the first 4 principal components (not shown). Thus the scores were determined to be redundant. As the gap frequency functional parameter was thought to increase rapidly during site restoration and plateau almost immediately while the plant cover and structural diversity score was thought to slowly increase over time as a site improved, the gap frequency was removed from further analysis. While the soil pH score was not redundant and was significantly different between degraded and reference sites, the range of actual soil pH was low. Biological activity may not be altered by the soil pH in degraded sites versus reference sites, so the soil pH score was dropped from further analysis. The large woody debris score was also not redundant and was significantly different between degraded and reference sites; however, the range of scores was very low in both degraded and reference sites with only a few sites having extreme scores. Because most of the scores spanned from 0 to 50, the large woody debris score was dropped from further analysis. A secondary PCA was run without the dropped functional parameters which showed no further parameter being redundant. The average of the compaction score, moisture score, riparian zone width score, instream canopy cover score, plant cover and structural diversity score, demography score, and recruitment score was calculated at each site as the Riparian Functional Assessment of that site (Table 11). The macroalgae score was added back in to the assessment; however, the score did not change any of the averages. Because there could be complications with collecting this functional parameter due to intermittent flow at sites and the lack of change that occurred in the scores, the macroalgae score was dropped from further analysis and left out of the Riparian Functional Assessment (RFA). The RFA was significantly higher in reference sites (mean = 77) than at degraded sites (mean = 39) ($p < 0.0001$). This was not surprising as all components of the RFA were significantly higher in reference sites.

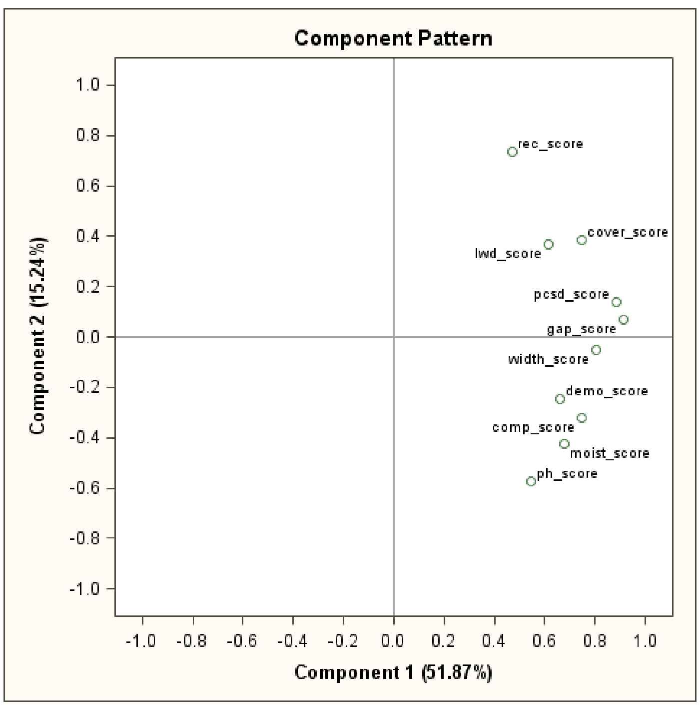

Figure 16: Principal Component Analysis of the functional parameters collected for the Riparian Functional Assessment.

Table 11: Riparian Functional Assessment score at each site.

Site Type	Site #	Site Name	Riparian Functional Assessment
Degraded	5556	East Bouldin @ Gabion in Gillis Park	34
	5580	Barton Creek Trib @ Lund and Robert E. Lee	13
	5584	Buttermilk Creek @ Buttermilk Park	29
	5585	Boggy Creek @ 10th St	42
	5586	Bull Creek 1600ft upstream Loop 360	34
	5588	Common Ford Trib ds xing in Common Ford Ranch	56
	5591	Johnson Creek in Tarrytown Park	40
	5592	Little Walnut Creek @ Dottie Jordan Park	40
	5593	South Boggy @ Dittmar Park near Strickland	60
	5594	Shoal Creek @ Shady Oak Court	45
	5595	Tannehill Creek @ Bartholomew Park near Berkman Dr	27
	5596	Tannehill Creek upstream storm pipe in Givens Park	32
	5598	Taylor Slough South in Reed Park @ Footbridge	39
	5601	Walnut Trib @ North Star Greenbelt	51
	5606	Williamson Creek in Battle Bend Park	32
	5582	Blunn Creek @ Rosedale	55
Reference	633	Barton Creek Ephemeral 3	71
	5581	Bee Creek Downstream Loop 360	69
	5583	Blunn Creek Upstream of Cow Trough Spring	78
	5589	Common Ford Trib us Bridge @ Common Ford Ranch	80
	5590	Fort Branch downstream Tura Ln	75
	5597	Taylor Slough North waterfall pool @ Mayfield Park	65
	5599	Little Walnut Trib @ Gus Garcia Park	82
	5600	Walnut Trib @ Lincolnshire and Garnaas	77
	5602	Walnut Creek downstream Old Manor Rd	84
	5603	West Bouldin Creek @ Audrey Court	80
	5604	West Bouldin Creek in West Bouldin Greenbelt	81
	5605	Williamson Creek @ Wagon Bend Trail	80

Conclusions

By testing and comparing all Riparian Functional Assessment (RFA) parameters between reference (minimal vegetation management and anthropogenic disturbance) and degraded (history of vegetative control and disturbance) locations, the City of Austin was able to identify which measures of ecological function are accurate for proving success of riparian restoration projects. Overall, monitoring for changes in soil compaction and moisture, riparian zone width, in-stream canopy cover, plant cover and structural diversity, hardwood demography, and seedling recruitment over time will allow managers to accurately assess if ecological function is being improved following restoration activities. Removing parameters that were non-significant, redundant, and biologically irrelevant helped to strengthen the overall scoring metrics and will reduce future sampling effort. No significant differences were found between reference and degraded sites for the bank stability and entrenchment ratio parameters. Although these parameters are important for understanding stream stability and water flow they are not useful in showing improvements in riparian zone function at the scale of this study. Site Potential Tree Height (SPTH) and gap frequency, although significant, were found to be redundant parameters. SPTH was similar to riparian zone width while the gap frequency was similar to the plant cover and structural diversity parameter. The significant differences observed with the soil pH and large woody debris (LWD) parameters were deemed to be biologically meaningless for the purposes of our analysis. Differences in soil pH were slight (6.8- 6.9) and not likely to disrupt biological activity (Lyons *et al.* 1998, Amador *et al.* 1997, Ashby *et al.* 1988, USDA 1998). Differences in LWD between reference and degraded sites were very low with only a few extreme scores suggesting minimal biological significance. Finally the macroalgae score was removed due to missing data as a result of dry sites. Although significantly different and biologically meaningful the difficulties in sampling intermittent and ephemeral streams preclude the use of macroalgae cover from this and future analysis. All of the above RFA parameters, that were removed from overall site calculation, which showed significant differences between reference and degraded sites, should be considered for future analysis or special study.

Overall RFA scores, based on the remaining 7 parameters, were significantly higher in reference than at degraded sites (Table 11). These differences are directly related to the vegetation management activities occurring in and around the riparian zone. Currently, all of the degraded locations have been incorporated into the Watershed Protection Departments Riparian Zone Restoration program (http://www.austintexas.gov/department/riparian-restoration). This program primarily attempts to restore the ecological function of a degraded site by allowing passive vegetation growth in protected buffers adjacent to the creek. These passive restoration buffers or “grow zones” will receive annual Riparian Functional Assessments in order to determine if the successional trajectory of vegetation is improving ecological function. Over time the calculated RFA scores of the degraded sites should mimic that of the reference locations. Parameters such as soil moisture and compaction and riparian zone width can change relatively rapidly and positive changes are expected after a few growing seasons. Overall plant cover and structural diversity along with in-stream canopy cover are slower to respond with changes not expected for at least 5-10 years. Hardwood demography and seedling recruitment can also change rapidly but are more interpretive and allow for managers to adaptively manage a site over time. For example, if undesirable species such as exotic, upland or annual species dominate the recruitment class after the first few growing seasons than active seeding, planting, or vegetation management may be necessary, especially if other variables such as compaction and moisture have improved. Adjacent site conditions, approximately 100 meter distance, are crucial when relying on passive vegetation growth (Hobbs and Prach 2008). The understory community reflects a habitats current ecological condition and can be an indication of future succession trends (Woosley *et al.* 2005). Being able to adaptively manage the early stages of succession is vital for improving overall ecological function.

Recommendations

Continue to monitor all City of Austin Riparian Zone Restoration sites using the seven selected Riparian Functional Assessment parameters which include soil compaction, soil moisture, riparian zone width, in-stream canopy cover, plant cover and structural diversity, hardwood demography, and recruitment. Annual sampling events will allow mangers to track restoration success and utilize adaptive management approaches to maximize improvements to ecological function over time. Further expansion of the hardwood demography and succession/recruitment metrics is in need. Additional literature review and/or field investigation is necessary to establish stability rankings, nutrient leaching rates, water usage, and other species specific characteristics that could be added to expand our understanding of successional changes and better inform management decisions. Additional studies on sapling growth and survival in urban riparian zones is also needed to help guide future planting/seeding efforts for restoration projects.

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