EVALUATION OF ECOLOGICAL FUNCTION OF URBAN RIPARIAN AND STREAM SYSTEMS: GUIDING ECOLOGICAL RESTORATION IN AUSTIN, TEXAS, USA

THESIS

Presented to the Graduate Council of
Texas State University-San Marcos
in Partial Fulfillment
of the Requirements

for the Degree

Master of SCIENCE

by

Leilani F. Williams, B.S.

San Marcos, Texas
May 2013

EVALUATION OF ECOLOGICAL FUNCTION OF URBAN RIPARIAN

AND STREAM SYSTEMS: GUIDING ECOLOGICAL

RESTORATION IN AUSTIN, TEXAS, USA

Committee Members Approved:

____
Yixin Zhang

____
Paula S. Williamson

____
Walter Rast

Approved:

____
J. Michael Willoughby
Dean of the Graduate College

ACKNOWLEDGEMENTS

I would like to thank my major advisor Dr. Yixin Zhang for his help over the last two years while serving as my advisor and chair of my graduate committee. Dr. Zhang gave me advice throughout my time with him and has been an essential link, connecting me with other individuals who are also passionate about stream ecology. I would also like to thank my fellow committee members Dr. Paula Williamson and Dr. Walter Rast for their insightful comments and guidance.

Many thanks go to current and former members of the Stream Ecology lab: Monika Henn, Mario Sullivan, and Trey Nobles for all of their support, statistical advice and for keeping me sane throughout this process. Monika especially was a great friend and fellow connoisseur of local sandwiches.

Special thanks go to members of the Austin Watershed Protection Department Surface Water Team. Alex Duncan has been influential in the design and implementation of this project. Without his vast store of knowledge and field help, this project would not have been possible. I also thank him for the splendid conversations and for not pointing out my inexperience but taking it in stride and utilizing teaching opportunities. Alex's patience and professional guidance has been an extremely beneficial and necessary part of my professional development. Mateo Scoggins has also been instrumental in helping this project since the beginning stages, and providing funding throughout most of the project. Todd Jackson's help with entomological identification was priceless, as was his playful banter during fieldwork. Aaron Richter's statistical advice was outstanding, and more than once he tactfully steered me back onto the right course. Mary Gilroy and Andrew Clamann were also helpful and supportive, always suggesting ideas that come from their years of environmental experience. Joan Brook was also a dependable fieldwork assistant as well as a great photographer. Many of the photographs for this project were provided by her and Alex.

Finally, I would like to especially thank my family and friends for their continued love and encouragement that has made this excursion possible. I thank my mother and father, Ann and Ron Wight, for their continual encouragement and for being my biggest supporters. I also thank my siblings and their families: Michiko, Brandon, and Cosette; Andrew, Teresa, Darius, and Abraham; Heather, Rob, Jordan, Jemma, and Liam; and Kristina, for letting my love of biology invade their lives. I also want to thank my family-in-law for their support. Clark and Carol Williams have been so encouraging and Bailey, Sam, James, and Ryan helped me have a life besides school and fieldwork.

I thank my husband, Chad, for being an amazing individual, possessing an endless supply of patience, and always being understanding throughout this process, not to mention helping with fieldwork occasionally. His support and encouragement made this experience possible. Last but certainly not least, my major debt is to the maker of heaven and earth.

This manuscript was submitted on March 22, 2013

TABLE OF CONTENTS

LIST OF TABLES

Table **Page**

LIST OF FIGURES

Figure **Page**

ABSTRACT

EVALUATION OF ECOLOGICAL FUNCTION OF URBAN RIPARIAN

AND STREAM SYSTEMS: GUIDING ECOLOGICAL

RESTORATION IN AUSTIN, TEXAS, USA

by

Leilani F. Williams, B.S.

Texas State University-San Marcos

May 2013

SUPERVISING PROFESSOR: YIXIN ZHANG

Restoration ecology seeks to restore ecosystem function and biodiversity in natural systems impacted by human activities. Restoration of riparian areas is a common recommendation of water management plans today and often deemed necessary to maintain ecosystem sustainability. Assessment of the condition of ecosystems is a critical prerequisite for alleviating effects of the multiple anthropogenic stresses imposed on them. To best determine ecosystem function of a group of urban streams, leaf-litter decomposition was used as an integrated metric for assessing anthropogenic impacts. I measured leaf-litter decomposition rate with two species, Texas Red Oak (*Quercus texana*) and American Sycamore (*Platanus occidentalis*), as a response variable between reference and degraded riparian sites. Other measured variables were macroinvertebrate colonization in leaf bags, riparian soil composition including metals and nutrients, water quality, and water temperature. Results include no significant differences in water quality variables and leaf-litter decomposition between different riparian sites. Soil composition variables do demonstrate regional patterns, including higher nutrient and metal concentrations at sites farthest south, but irrespective of riparian site status. Similar leaf pack macroinvertebrate colonization patterns and biomass values were observed for both leaf species, irrespective of riparian site status. Using these results to compare sites that have a history of riparian disturbance to sites with fewer disturbances will have a potential to help guide future riparian restoration activities.

CHAPTER 1

EVALUATION OF ECOLOGICAL FUNCTION OF URBAN RIPARIAN AND STREAM SYSTEMS: GUIDING ECOLOGICAL RESTORATION IN AUSTIN, TEXAS, USA

Introduction

The estimated average value of the world's ecosystem functions and services is US $33 trillion per year, approximately double the global gross national product total of US $18 trillion per year (Costanza 1997). Resources and processes supplied by natural ecosystems that benefit humans are collectively called ecosystem services. Ecosystem functions are collective activities of plants, animals, and microbes and the effect these activities have on their surroundings (Huston 1997, Naeem et al. 1999). Categories of ecosystem functions include human provisioning, cultural, and regulating ecosystem functions. General regulating ecosystem functions include leaf litter breakdown, community respiration, transformation of organic matter, and removal of water-column nutrients from point and nonpoint sources (Meyer et al. 2005).

Freshwater systems occupy 0.8% of the Earth's surface (McAllister et al. 1997) and contain about 2.4% of all Earth's plant and animal species, approximately 44,000 species (Reaka-Kudla 1997). Freshwater ecosystems provide many vital ecosystem services, including disturbance regulation, water regulation, storage and retention of water supply, food production, erosion control, waste treatment, refugia, recreation, cultural uses, nutrient cycling, and climate regulation (Costanza et al. 1997). Additionally, ecosystems are directly utilized by countless organisms. For example, 25% of birds and 11% of mammals in Europe use freshwater wetlands as their main breeding and feeding areas (Kristensen 1994). Ecosystem function varies with condition of the system, and many types of disturbances affect function, including mining, deforestation, overexploitation, and rural and urban land use (Costanza et al. 1997).

The 20th century has been marked by rural to urban migration. In 1900, 16% of the global population lived in urban areas (Goldewijk et al. 2010), while in 2010, 52.1% of the global population lived in urban areas (United Nations 2012), with population increases in the next half century projected to occur specifically in urban areas in the developing world (Cohen 2003). In the United States of America, more than 75% of the population resides in urban areas (Paul and Meyer 2001).

With increasing urbanization comes an increase in humans living close to freshwater. Approximately 50% of the world population lives within 3 km distance from surface freshwater (Kummu et al. 2011). This proximity to freshwater results in ease of transportation, opportunities for recreation, increases in development, supplies drinking water, and altogether sustains society (Daily 1997, Palmer et al. 2004); therefore humans place enormous importance on these functions and services.

However, catchment urbanization severely alters physical characteristics of streams, including hydrology and channel geomorphology, chemical features such as nutrient cycling, and biological and trophic resources of stream ecosystems (Chadwick et al. 2006). These altered processes caused by urbanization affect the land-water interactions between the stream and riparian zone, including changes in shading, nutrients, pollutants, coarse river debris, predation rate, and riparian zone movement of aquatic macroinvertebrates (Wiens 2002). Extensive and widespread urbanization, including metropolitan-area sprawl (Meyer et al. 2005), is a threat to stream ecosystems, with over 130,000 km of streams and rivers in the US impaired by urbanization (USEPA 2000).

Altered stream ecological structure and function often indicate urban streams (Meyer et al. 2005, Walsh et al. 2005). Interestingly, the extent of urbanization impacts on aquatic ecosystems is growing faster than the rate of urban population growth because of present attitudes promoting decentralization and urban sprawl (McGranahan and Satterthwaite 2003).

Human activities impact stream ecosystems (Benke 1990, Zwick 1992, Allan and Flecker 1993, Dynesius and Nilsson 1994, Boon 2000). An urban stream is a formerly natural waterway which flows through a heavily populated area. Urban streams are often significantly polluted, due to urban runoff and combined sewer outflows. The “Urban Stream Syndrome,” the consistently observed ecological degradation of streams draining urban land, serves as a template for understanding urban streams (Paul and Meyer 2001, Walsh et al. 2005). Symptoms of the syndrome include a flashier hydrograph, elevated concentrations of nutrients and contaminants, altered channel morphology and stability, and reduced biotic richness. Many symptoms can also signify signs of nonpoint source pollution, whether from urban or rural areas. An increase in tolerant species in dominance is another characteristic. Other characteristics of urban streams include high-magnitude storm flows, disconnected riparian zones, and homogeneous habitats.

A general correlation is that urban catchments become increasingly degraded as the catchment becomes increasingly urbanized (Sudduth et al. 2011). Both point and nonpoint sources impact urban stream water quality (Paul and Meyer 2001) and even in areas where it does not dominate, urbanization has major influences on the environment (Alberti et al. 2003). Damage to streams, lakes, and estuaries from nonpoint source pollution was estimated to be about $7 to $9 billion a year in the mid-1980s in the U.S. (Ribaudo 1986).

Riparian zones, the interface between rivers or streams and land, were first considered to be well-defined landscape features that warranted special consideration in the 1970's (Odum 1978, Johnson and McCormick 1978). Since then, riparian systems have been a main topic of research and scientific meetings (e.g., Warner and Hendrix 1984, Johnson et al. 1985, Naiman and Décamps 1990, Malanson 1993, Wigington and Beschta 2000) and are now often incorporated in watershed management (e.g., Naiman 1992, Doppelt et al. 1993, Naiman and Bilby 1998).

Today, riparian systems are included in nearly all aspects of water management (García de Jalón and Vizcaíno 2004, European Declaration for a New Water Culture 2005) and restoration of riparian areas is considered essential for ecosystem sustainability (Gonzalez del Tanago and García de Jalón 2006). Riparian buffers have been shown to be effective in controlling nonpoint source pollution by removing nutrients, especially nitrogen, and reducing sediment input to aquatic ecosystems (Correll 1996). Assessment of the condition of ecosystems is a critical prerequisite for alleviating effects of the multiple anthropogenic stresses imposed on them (Gessner and Chauvet 2002, Zhang 2013).

Urbanized riparian zones often consist of altered soil composition (Moffatt et al. 2004), and vegetation composition of urban environments is commonly dominated by multiple exotic species, whose spread is closely linked to anthropogenic modification (Pennington et al. 2010). Absent riparian zones can often be re-implemented with a passive ecological restoration approach. Prach and Hobbs (2008) extol a passive ecological restoration approach, because these approaches are associated with a more resilient biological community, reduced management effort, and lower costs than when compared to active approaches.

However, passive restoration often takes longer and may not always progress an ecosystem towards the target community composition, especially if the vegetation is altered by exotics (Prach and Hobbs 2008), as often is the case in Austin. Management practices consistent with guiding principles are likely to lead to ecological, economic, and social wellbeing, while those practices that are not consistent with the guiding principles risk species loss, degraded environments, and long term social problems (Kaufmann et al. 1994).

For my project, I observed the effects of catchment urbanization and riparian zone condition on ecosystem function in streams in Austin, Texas, USA. The Austin-Round Rock-San Marcos, TX metropolitan statistical area (MSA) is the 8th fastest growing MSA in the U.S.A., with a 37.3% increase in population from 2000 to 2010 (U.S. Census Bureau 2010). Such a high rate of growth and urbanization is undoubtedly affecting the natural areas in Austin.

To assess riparian zone disturbance on stream ecosystem function (Paul et al. 2006), leaf-litter decomposition was used as an integrated metric for assessing anthropogenic impacts to stream ecosystem function and since allochthonous litter plays a crucial role in streams, the demonstrated effects of anthropogenic perturbations are evident on litter breakdown (Gessner and Chauvet 2002). I measured leaf litter breakdown rate, aquatic macroinvertebrate leaf bag colonization, water chemistry and riparian soil composition, which are all analytical measurements intended to quantify the state of a system.

Overall, understanding which metrics are most closely linked to ecological function will allow managers in the future to better streamline monitoring efforts and allow for more focused restoration activities. This information will be used to identify the extent of the urban-stream syndrome in Austin (Meyer et al. 2005, Walsh et al. 2005).

As population growth in Austin and surrounding areas is projected to continue, putting additional strain on the already dwindling water resources, understanding the links between urbanization and stream health becomes increasingly important. My results can be used to provide guidance for management practices and inform decision-makers. This understanding is necessary prior to implementing broad restoration plans.

Methods

Study Area

The study sites (Table 1 and Figure 1) were selected as part of an ongoing project with the City of Austin's Watershed Protection Department. The paired sites were chosen because they shared watersheds or were in adjacent watersheds, drained similarly sized watersheds, and were least likely to dry over the sampling period based on historical data, since leaf litter decomposition and water quality would be impossible to sample without flowing water. All maps created for the project were prepared with ArcGIS (ESRI 2011).

Leaf Litter Decomposition

Using leaf-litter decomposition as a response variable is a quantitative measure of assessing stream ecosystem health condition (Gessner and Chauvet 2002). I sampled leaf bags in two week increments (Gessner and Chauvet 2002). Newly abscised American Sycamore (*Platanus occidentalis*), and Texas Red Oak (*Quercus texana*) leaves were collected in January 2012, the last leaf dropping incident of the year, from several trees located on the Texas State University-San Marcos campus in San Marcos, Hays County, Texas, USA. Care was taken to remove leaves from a single location to avoid complications of varying leaf-litter quality that can occur among catchments (Chadwick and Huryn 2003, Chadwick et al. 2006).

Leaves were dried in a forced-air oven (~60°C) for 24-48 hours to constant mass. Approximately 5 g (±0.1g) of forced-air oven dried leaves were placed in each litter bag. Litter bags were approximately 1’ in length, with black hardware nets approximately 6” × 6” placed inside to hold yellow bag open and avoid variations in leaf bag dimension over time. Bags were secured to a small brick with a zip tie and laid in on the stream bed. Bags were deployed in June 2012 and collected on three periods after: two week, four week, and six week samples. On each date, bags were removed from the streams, placed in individuals plastic bags, and returned to the laboratory on ice. By incubating leaves *in situ*, they are exposed to the normal fluctuations in temperature and moisture. Mesh bags allow macroinvertebrates access to leaves.

Laboratory Methods for Leaf Analysis

Litter-bag contents were washed with running water to remove any non-leaf material. Whole leaves and fragments were removed by hand and placed into paper bags for oven drying. Remaining material was washed through a 250-µm sieve and material remaining in the sieve preserved in ~95% ethanol. All remaining leaf material was dried to constant mass in a forced-air oven (60° C) and weighed to determine mass loss. Difference in mass after one day from the leaching bags was used to calculate a leaching loss coefficient, which was applied to each two, four, and six week leaf bags to calculate an adjusted initial mass for each bag (Gessner and Schwoerbel 1989).

Macroinvertebrate Diversity from Leaf Litter

Benthic diversity from leaf litter bags is necessary for functional feeding group analysis of macroinvertebrates. Macroinvertebrates are affected by disturbance conditions (Walters and Post 2011), which characteristically lead to an overabundance of tolerant taxa (Chironomidae and Oligocheatea) in urban streams and decline of sensitive taxa, including Ephemeroptera, Plecoptera, and Trichoptera.

For macroinvertebrates, richness and biomass were analyzed from leaf-litter bags collected. Leaf material was washed through a 250-µm sieve and animals retained were identified to lowest practical taxonomic level, preferably genus (Merritt and Cummins 1996), length measured to nearest 1 mm and preserved in 95% ethanol for future reference with labels. Individual and total biomass were estimated using taxon-specific, length-mass relationships (Benke et al. 1999). Length-mass relationships not provided by Benke et al.(1999) were found for oligocheate, physella, ferissia, and *Helisoma anceps*

(Miserendino 2001), *Melanoides tuberculatus* (Carvalho Silva et al. 2010), and Hirudinea (Edwards et al. 2009).

Macroinvertebrates were also grouped into functional feeding groups for further ecosystem function analysis (Cummins et al. 2005, Merritt and Cummins 1996, Smith 2001, Thorp and Covich 2001). The groups included taxa that primarily consumed either fine particulate organic matter, coarse particulate organic matter, or other animals. Consumers of fine particulate organic matter were divided into filtering and nonfiltering taxa. This resulted in five distinct functional feeding groups: predators (PR), collector-gatherers (CG), filterers (FL), snails (SN), and shredders (SH) (TCEQ 2007, Barbour et al. 1999).

Water Quality

Water temperature, conductivity, dissolved oxygen, and pH were measured *in situ* with hand-held meters at 0, 2, 4, and 6 week periods during the study with a Hydro Tech Hydrolab MiniSonde 4a v2.06. All City of Austin sondes receive routine maintenance and are pre/post calibrated at every sampling event (COA 2010). Densitometer reading using a convex forest densiometer at leaf pack site was also taken at the beginning and end of the study period. Densiometer readings were taken at week 0 from the location of the leaf packs. Three readings were taken, facing left, center, and right. These three readings were averaged for the mean densiometer readings. Depth of stream channel was also recorded each time.

Flow data were collected using a Flow-Mate Model 2000 Water Current and Flow Meter (Flow-Tronic, Welkenraedt, Belgium), depth using a standard USGS wading staff, wetted and bankfull width using a 50-meter tape at both the beginning and end of study period. All surface water quality monitoring was done in accordance with the City of Austin, Water Resource Evaluation Standard Operating Procedures Manual (COA 2010).

Soil Composition

Soil samples were collected at each sampling location, approximately five meters from the stream, to characterize and compare sites. A hand shovel was used to collect a cylinder of soil at one spot at each site along the 100 m transect, noting location of soil taken, making a grab sample. Samples on ice were sent to DHL for analysis (DHL Analytical 2012). Analysis included: metals, nitrite, nitrate, orthophosphate, total phosphorus, ammonia, and percent moisture.

Soil composition results from DHL were used to create interpolated nutrient and metal composition maps for all the sites using ArcGIS (ESRI 2011). I created interpolations of each of the 18 variables. Variables were separated into two groups: heavy metals and nutrients. For each group, a raster calculator was used for resulting interpolation compilations. Compiled metal composition across Austin is shown in Figure 2. The same protocol was followed for the nutrient data, shown in Figure 3.

Data Analysis

The decomposition rate constant, k, was estimated for all treatments using a negative exponential decay model with the formula $Wt/Wo = e^{-kt}$, where Wo is the initial mass, Wt is the mass remaining on day t, and k is the breakdown rate constant (Petersen and Cummins 1974, Webster and Benfield 1986).

Leaf-litter decomposition data were analyzed first with a Welch two sample T-test for the spatial data, and a Pearson correlation test for the temporal variable. Decomposition was then analyzed with a linear mixed effect model used to find significant differences between groups after field collection of the data, comparing degraded to reference sites.

Pearson correlation analysis was used to determine correlation of soil composition variables. Pearson correlation analysis was also used to analyze invertebrate biomass and taxa richness. I used a non-parametric statistical Friedman test to check for overall site differences in macroinvertebrate data, because I had a complete block design using leaf bags as blocks. If Friedman test resulted in significant site differences, then I used the Wilcoxon, Nemenyi, McDonald-Thompson multiple comparison test. That test, which is a distribution free two-sided all-treatment multiple comparison test, controls for an experimentwise error term; thus, no bonferroni correction was needed.

I analyzed soil metrics with a principal components analysis (PCA) (Manly 1986). I then used a MANOVA test, Hotelling's two sample T square test, to determine treatment differences in both nutrient and metal data. Statistical analyses were performed using the R package version 2.15.0 (www.r-project.org, 11/1/12) or SAS (Version 9.1.3, SAS Institute Inc.).

Results

Water Quality and Site Characterization

Site characterization consisted of visual assessment at each site at each sampling event. Water temperature, DO, pH, and conductivity were also measured as a grab sample each sampling event. The data are therefore instantaneous grabs, with four samples per site (weeks 0, 2, 4, and 6). Since several of these measurements shift throughout the day due to biological activity, especially DO, sites are described with means and standard deviations.

All means and standard deviations for water quality data are found in Table 2. The site with the highest mean DO was site 6, Blunn at Cow Trough Spring, with a mean DO of 8.73 mg/L and also the highest standard deviation for that category, while the site with the lowest DO was site 3, Walnut Trib at Northstar, with a mean of just 2.81 mg/L, which could be characterized as hypoxic.

The site with the highest mean temperature was site 4, Lil Walnut at Dottie Jordan, with a mean of 28.28 °C, while the site with the lowest mean temperature was site 7, Bee at Loop 360, with a mean of 23.09°C. Site 8, Walnut at Old Manor had the highest standard deviation for temperature, at 1.25.

Site 1, Taylor Slough South in Reed was the site with the highest mean pH, at 7.96, while site 3, Walnut Trib at Northstar had the lowest mean pH, at 6.94. All standard deviations for pH were below 0.20.

Site 7, Bee at Loop 360 had the highest mean conductivity, 912.28 µS/cm, while site 4, Little Walnut at Dottie Jordan had the lowest mean conductivity, at 404.28 µS/cm. Site 3, Walnut Trib at Northstar had the highest conductivity standard deviation, with a value of 205.88.

The deepest stream channel where leaf bags were placed was site 2, Blunn at Rosedale, with a mean depth of 3.1 feet, while the shallowest channel was site 3, Walnut

Trib at Northstar. Site 1, Taylor Sough South in Reed had the least stream depth standard deviation, 0.21, remaining the most constant depth throughout the sampling period.

The results for the water quality mixed effects model are shown in Table 5. There were no significant differences between reference and degraded groups, indicating similar water quality environments for both groups. However, there were significant differences for within-site p-values, which indicate highly variable character of sites over time. Site 1, TSS in Reed also had the highest mean densiometer readings, with a mean of 97.67 cover, while site 6, Blunn at Cow Trough had the lowest, with a mean reading of 33.33 canopy cover.

Leaf-Litter Breakdown

Leaf-litter decomposition data were analyzed first with a Welch two sample T-test for the spatial data, and a Pearson correlation test for the temporal variable. Welch two sample T-tests were performed to assess differences among the streams for both leaf species. There were no significant differences between sites for the Red Oak leaf-litter breakdown ($p=0.62$), or for Sycamore leaf-litter breakdown ($p=0.21$). Pearson correlation coefficients show leaf breakdown was highly correlated based on temporal variables, with correlation values of 0.82 for Red Oak and 0.73 for Sycamore.

Decomposition was then analyzed with a linear mixed effect model used to find significant differences between groups after field collection of the data, comparing degraded to reference sites. The results of the linear mixed effect model for both species are in Table 6. For Red Oak, there was a significant difference for the temporal variable, with $F=81.60$ and $p<0.0001$. For Sycamore, there was a significant difference for the temporal variable, with F=63.07 and p<0.0001. However, there was not a significant difference in leaf breakdown when compared to site for Red Oak, with F=0.50 and p=0.51, or for Sycamore, with F=0.88 and p=0.38. There was also not a significant difference in the interaction term between site and time for either leaf species.

Macroinvertebrates

The total number of macroinvertebrate taxa identified from each stream ranged from 3 to 10 taxa for Red Oak and 3 to 12 taxa for Sycamore; values are shown in Table 4. The functional feeding group structure of macroinvertebrates associated with both Red Oak and Sycamore were similar. Collector gatherers (CG) were mainly ostrocoda, ceratopogonidae, cambaridae, and callibaetis. Predators (PR) most commonly found were tanypodidae, rhagovelia, argia, and cetaropogonidae. The only shredders (SH) found colonizing the leaf bags were from the genus hyalella, and shredders were only found at two of the eight sites. Consumers of coarse particulate organic matter, scrapers (SH), were dominated by physella and ferrissia, with some Helisoma. The filterer/collector group (FC) was dominated by chironomidae.

Length-mass relationships were obtained for each taxa except for Ostrocods, were not available in the literature (Wynn, personal communication, 2013). Therefore, ostrocoda contribute to overall richness and FFG richness, but not biomass measurements.

Using a Pearson correlation test, invertebrate biomass and taxa richness were not significantly correlated with each other, with P>0.05 for both leaf species. Using a Friedman test, there was no difference in biomass between sites, with P>0.05.

There was a difference in richness between sites. Using a Wilcoxon, Nemenyi, McDonald-Thompson multiple comparison test, I found Site 3 and 7 were not significantly different from each other in regards to total richness. Sites 7, 1, and 4 were not significantly different. Sites 1, 4, 6, and 2 were not significantly different in total richness. Site 5 was significantly different than all other sites in total richness.

There was no difference in biomass or richness for individual functional feeding groups, per the Friedman test ($P>0.05$). Number of predators and CG/SH came close to being different. Also using the Friedman test, there was no difference in biomass or richness between site types, degraded or reference ($P>0.05$). There was no difference in biomass or richness between site types for individual functional feeding groups although the biomass of predators came close to being different, but $P>0.05$. There was no correlation between biomass and leaf-litter decomposition. There was also no correlation between richness and leaf-litter decomposition.

Soil Composition

Soil compilation maps were made in ArcGIS from resulting DHL data, seen in Table 3. Metal composition is seen in Figure 2. Nutrient composition is seen in Figure 3. For both maps, references sites are shown in green and degraded sites are shown in red. Patterns are demonstrated regionally, but not according to riparian zone condition.

A Pearson correlation analysis was run on all the metals. Chromium was highly correlated to iron ($r=0.99$), magnesium ($r=0.91$), and nickel (0.96). They were excluded in the PCA, and any inference made for chromium can relate to them as well. Calcium was discounted as well because it correlates to copper ($r=0.83$). Metal PCA results can be seen in Figure 4. Grouping sites into reference and degraded groups yielded no definite explanations, even with component #1 explaining 38.5% of the metal variation and 45.4% of the nutrient variation.

A correlation analysis was used to analyze all soil nutrients. Nitrite and percent moisture were highly correlated (r=0.99). However, all nitrite values were below the detectable limits, and were therefore omitted from the PCA. Also, percent moisture was variable across sampling diel period, and even though there is a correlation between percent moisture and site status (r=0.67), it was also omitted from the PCA. Site 6 seems to be high in nitrate, ammonia, and total phosphorus. Site 7 is high in ammonia but low in nitrate, phosphorus and orthophosphorus. Sites 1 and 5 are high in orthophosphorus but low in the other three nutrients. Site 3 is high in all measured nutrients except ammonia. Additionally, sites 2, 4, and 8 are sites with moderate to low nutrient levels. Nutrient PCA results can be seen in Figure 5.

MANOVA was performed for the soil composition using a Hotelling's two sample T square test. For the nutrient data, the F statistic= 1.33, p-value= 0.42. For soil metal data, F=1.53 and p=0.38. Both tests show no significant differences in riparian soil nutrient or metal data based on treatment type, reference or degraded.

Discussion

Leaf-litter decomposition showed no significant spatial differences, but did demonstrate temporal differences, indicating stream function over a temporal scale. Invertebrate similarities were found for both richness and biomass, regardless of riparian condition. Water quality measurements showed no significant differences. This indicated that leaf-litter decomposition may not show differences either, irrespective of riparian zone condition. Indeed, water quality correlations for both leaf species were not highly correlated to riparian zone condition.

Possible reasons for leaf-litter not indicating significant differences based on riparian zone condition are: all sampled sites are urban streams, and as such, are subject to altered hydrology upstream of chosen sampling locations; current extreme drought conditions in Austin could also be influencing leaf-litter breakdown, as well as macroinvertebrate community assemblage with streams used in the study were small, and though each site was chosen based on historical flow records, some were likely to dry, at least partially, during the study period (Woodcock and Huryn 2005).

Additional factors affecting leaf-litter signals include a lack of shredders, evident in the macroinvertebrate analysis, which feed on exclusively or largely on leaf detritus (Chadwick et al. 2006). Texas in general has a dearth of shredders. Also, there is often a synergistic effect of multiple leaves in decomposition (Gulis 2003), and this experiment only tested single species leaf bags. Additional important structural ecosystem components could also be influencing results, including hydrology upstream (Seybold et al. 1999), as well as possible toxic effects of metals on microbial community may reduce leaf-litter decomposition (Chadwick et al. 2006).

The role that macroinvertebrates play in processing stream detritus is well known, as is the response of macroinvertebrates to urbanization, such as the simplification of stream communities due to loss of sensitive taxa (Morse et al. 2003, Gray 2004, Woodcock and Huryn 2005). Since the only significant differences detected were in total richness, and the sites that were significantly different were not necessarily in the same riparian classification, reference or degraded, it cannot truly be said that the macroinvertebrate communities differ in different riparian zone conditions. This is an unexpected finding, because previous leaf-litter studies found differences in both biomass (Chadwick et al. 2006) and richness (Thorp and Covich 2001) based on status of riparian zone.

Riparian soil composition was shown to be highly variable across riparian zone condition. For compiled soil composition maps, both nutrient and metal concentrations show patterns. The higher nutrient densities occur in the south part of the city, with a small high density at site 3, Northstar park. The lowest nutrient sites were on the east side, which was surprising because many of the streams on the east side of the city have been draining agricultural lands for over 100 years. There seemed to be little correlation between the riparian status, reference or degraded, and the distribution of metal density.

For example, the two reference zones in the south, site 5, Audrey Court and site 6, Cow Trough Spring, have some of the highest values of metal concentration. Both of these sites have relatively large, healthy riparian zones, and are not downstream of agricultural land. This implies that a well-defined and healthy riparian zone is not the only component structural ecosystem component important in soil nutrient buffering (Seybold et al. 1999), and that hydrology upstream, including flood events, is key to understanding riparian soils. Future sampling could include riparian areas in the broad “belt” of highest metal concentration, the inverted V-shape on Figure 2, to determine if the values in this region are indeed the highest in the city.

The most important factor controlling effectiveness of riparian buffers is hydrology: how the water moves through or over the buffer. For example, removal of contaminants from surface runoff requires that runoff water be sufficiently slowed to allow sediment to settle out. Often, channelized water moves almost as quickly through a buffer as it does from the field, thereby making the buffer ineffective at pollutant removal (Dillaha et al. 1989).

To quantitatively predict nutrient removal in riparian buffers, it is necessary to understand the hydrology of each site (Hill 1996). For future study, I would incorporate both impervious cover as well as analysis of all upstream hydrological alterations. This would demonstrate whether hydrology is the abiotic driver of unexplained biotic patterns, such as the lack of significant difference in leaf-litter decomposition in varied riparian zone condition sites.

Historically, urban development transformed streams into drains or sewers. This often reduced potential precious natural resources to humans who lived near them and devastates ecosystems (Gregory et al. 1991, Meyer et al. 2005). In many cities of the developed world streams remain in poor ecological conditions (Walsh et al. 2005).

However, societal awareness is also growing, and both scientists and decision makers are seeking measures to alleviate the resulting negative effects (e.g., Convention on Biological Diversity 1992, Christensen et al. 1996, Stanford and Poole 1996, Ward 1998, Blöch 1999, Petts 1999). New urban design and waterway management show great potential for achieving all public safety and amenity goals, and improved ecological condition of streams in urban areas (Lloyd et al. 2002).

Beginning this experiment, I expected to see changes in stream ecosystem functioning and benthic diversity along a gradient of riparian zone conditions, which are response variables to riparian environmental changes. The null hypothesis that riparian zone condition treatment in this experiment had no significant effect on the functioning urban streams should be accepted.

Passive restoration could be recommended for sites where temporary or easily modified human disturbance has taken place (COA 2012). Diminished public support is a great risk if researchers lack the ability to prove success in restoration projects (Woolsey et al. 2007). Management of riparian zones will likely be unpopular with the public (Duncan 2012), and almost certainly, catchment-scale solutions are required to reverse symptoms of the urban stream syndrome (Walsh et al. 2005).

As population growth in Austin and its surrounding areas is expected to continue, putting additional strain on the already dwindling water resources, understanding the links between urbanization and stream health becomes increasingly important. Overall, understanding which environmental factors are most closely linked to ecosystem function will allow managers to better streamline monitoring efforts and allow for more focused restoration activities. My results are likely to instruct future monitoring strategy in urban areas, and could show where a more holistic approach, including incorporating impervious cover and hydrological analysis, would be most effective.

Table 1: Initial site list for all sites used in Chapter 1. Sites dropped from experiment due to drought conditions include Tannehill Creek @ Bartholomew Park, East Bouldin @ Gillis Park, Little Walnut @ Gus Garcia Park, and Walnut Trib @ Lincolnshire and Garnaas.

Degraded			
Site No.	**Site Name**	**Drainage (Acres)**	**Watershed**
0	Tannehill Creek @ Bartholomew Park	640	Tannehill Branch
0	East Bouldin @ Gillis Park	320	East Bouldin
1	TSS in Reed Park @ Footbridge	120	Taylor Slough South
2	Blunn Creek @ Rosedale	640	Blunn
3	Walnut Trib @ North Star Greenbelt	64	Walnut
4	Little Walnut Creek @ Dottie Jordan Park	1280	Little Walnut
Reference			
0	Little Walnut Trib @ Gus Garcia Park	640	Little Walnut
0	Walnut Trib @ Lincolnshire and Garnaas	128	Walnut
5	West Bouldin Creek @ Audrey Court	320	WestBouldin
6	Blunn @ Cow Trough Spring	320	Blunn
7	Bee @ Loop 360	320	Bee
8	Walnut Creek downstream Old Manor Rd	1280	Walnut

Table 2: Water quality and physical data for each site averaged across study period.

Site	DO	Temperature	pH	Conductivity	Depth	Densiometer Total
1	7.22±1.60	25.71±0.54	7.96±0.13	630.15±112.01	1.20±0.21	97.67
2	6.39±1.22	26.58±0.65	7.92±.015	718.70±236.84	3.10±1.39	81.67
3	2.81±1.10	24.02±1.07	6.94±0.08	818.63±205.88	0.98±0.67	68.33
4	6.87±1.66	28.28±1.23	7.93±0.19	404.28±120.75	1.63±1.02	84.00
5	5.19±1.12	25.13±0.63	7.47±0.08	823.15±11.26	1.53±0.36	76.67
6	8.73±2.06	26.33±0.97	7.83±0.15	518.13±157.18	1.28±0.57	33.33
7	7.66±0.96	23.09±0.55	7.60±0.20	912.28±29.53	1.05±0.46	80.33
8	4.91±0.69	25.97±1.25	7.46±0.13	560.63±31.76	1.70±0.62	88.67

Table 3: Results from soil composition analysis. Table is split into metal above and nutrients below.

Site	As	Cd	Ca	Cr	Cu	Fe	Pb	Mg	Ni	K	Na	Zi
1	5.81	0.353	130000	17.2	13.4	12000	157	5170	10.6	1850	158	77.7
2	6.97	0.367	224000	44.4	11.7	24300	23.9	11600	63.4	2040	148	48.3
3	2.91	0.246	262000	10.7	5.92	8560	8.44	4760	7.97	2170	139	46.3
4	6.03	0.281	267000	14.3	7.4	11400	15.4	4810	9.2	2320	246	49.1
5	4.52	0.408	178000	32.2	11.7	18100	31.5	7220	30.6	2000	263	71.5
6	3.32	0.429	118000	31.3	16.5	18600	22.2	6460	48.4	1860	85.5	78.6
7	6.14	0.321	193000	12.4	6.5	11600	14.4	5550	9.17	3560	75.3	31.8
8	7.26	0.392	229000	17.3	8.9	13700	15.2	4570	12.4	3300	184	38.6

Table 3-Continued: Results from soil composition analysis. Table is split into metal above and nutrients below.

Site	Soluble Ammonia	Total Phosphorus	Total Orthophosphate	Nitrate	Nitrite	Percent Moisture
1	1.44	11.8	3.29	5.81	5.81	15.59
2	1.76	14.6	1.93	23.4	6.38	22.26
3	1.13	28.8	5.05	20.3	6.2	20.29
4	1.39	13.2	1.39	19.8	5.59	11.65
5	1.73	14.4	4.7	11.4	6.98	29.25
6	2.72	33.9	1.66	28.4	6.72	26.46
7	3.26	11.9	1.28	6.41	6.41	23.2
8	1.25	24.1	0.848	6.18	6.18	19.15

Table 4: Macroinvertebrate biomass (mg/litter bag) and richness (no. taxa/functional feeding group), by detrital leaf type, at the Austin stream sites.

Biomass (mg/litter bag), by feeding group							**Richness (no. taxa), by feeding group**					
Site	**CG**	**SC**	**P**	**CG/SH**	**FC/P**	**Total**	**CG**	**SC**	**P**	**CG/SH**	**FC/P**	**Total**
Red Oak												
1	0.0	19.2	2.6	0.0	0.1	21.9	0	3	5	0	1	9
2	0.0	40.5	0.3	0.0	2.4	43.2	0	3	2	0	2	7
3	6.3	12.9	151.8	0.0	0.0	171.0	3	3	3	0	1	10
4	0.0	2.6	1.0	0.4	0.2	4.2	1	2	2	1	1	7
5	0.0	34.2	0.1	0.0	0.0	34.3	0	2	1	0	0	3
6	1.0	6.2	0.5	0.0	0.2	7.9	1	2	2	0	1	6
7	0.7	3.1	0.3	0.2	0.1	4.4	1	3	3	1	1	9
8												
Sycamore												
1	0.0	1.8	20.2	0.0	0.1	22.1	0	3	4	0	1	8
2	0.0	6.6	0.5	0.0	0.1	7.2	0	2	2	0	1	5
3	0.0	10.4	83.1	0.0	0.1	93.6	2	4	4	0	2	12
4	0.0	10.2	1.4	0.3	0.5	12.4	0	2	2	1	1	6
5	0.0	13.6	0.0	0.0	0.1	13.7	0	2		0	1	3
6	307.6	5.8	0.4	0.0	0.2	314.0	2	2	2	0	1	7
7	0.0	5.8	1.2	0.4	0.1	7.5	1	2	3	1	1	8
8												

Notes: Functional feeding group key: CG, Collector-gatherer; SC, Scraper; P, Predator; FC, Filterer/ Collector

Table 5: Water Quality Mixed Effects Model Results. Note that there are no significant differences between reference and degraded groups, indicating similar water quality environments for both groups. Also note significant within site p-values, which indicate highly variable character of sites over time.

Water Quality Parameter	Sample Group	F Statistic	P value
Dissolved Oxygen	Within Site	84.81	<.0001
	Between Group	0.75	>0.05
Water Temperature	Within Site	2446.10	<.0001
	Between Group	0.85	>0.05
pH	Within Site	4220.18	<.0001
	Between Group	0.11	>0.05
Conductivity	Within Site	55.33	<.0001
	Between Group	0.94	>0.05

Table 6: Leaf-litter linear mixed effect model between degraded and reference sites. No significant differences between reference and degraded sites ("Habitat" in table), showing that breakdown differentiation according to group did not occur. Significant differences over time ("Week" in table), showing that breakdown over time occurred as expected.

Species	Variable	F Statistic	P-value
Sycamore	Riparian Condition	0.88	0.38
Sycamore	Temporal Variable	63.07	<0.0001
Red Oak	Riparian Condition	0.50	0.51
Red Oak	Temporal Variable	81.60	<0.0001

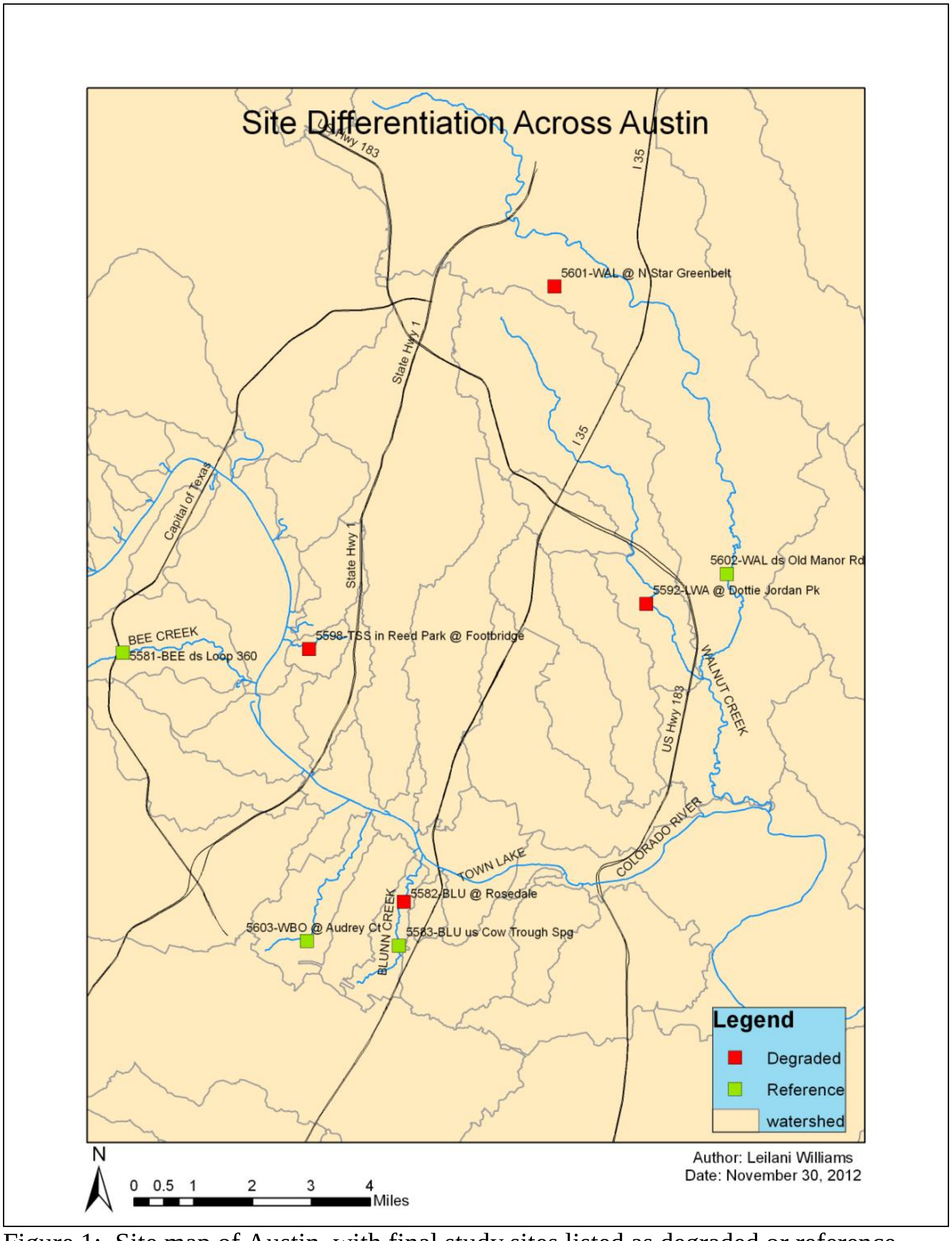

Figure 1: Site map of Austin, with final study sites listed as degraded or reference. Streams of interest, major roads and major reservoirs are also shown and labeled, along with unlabeled watersheds.

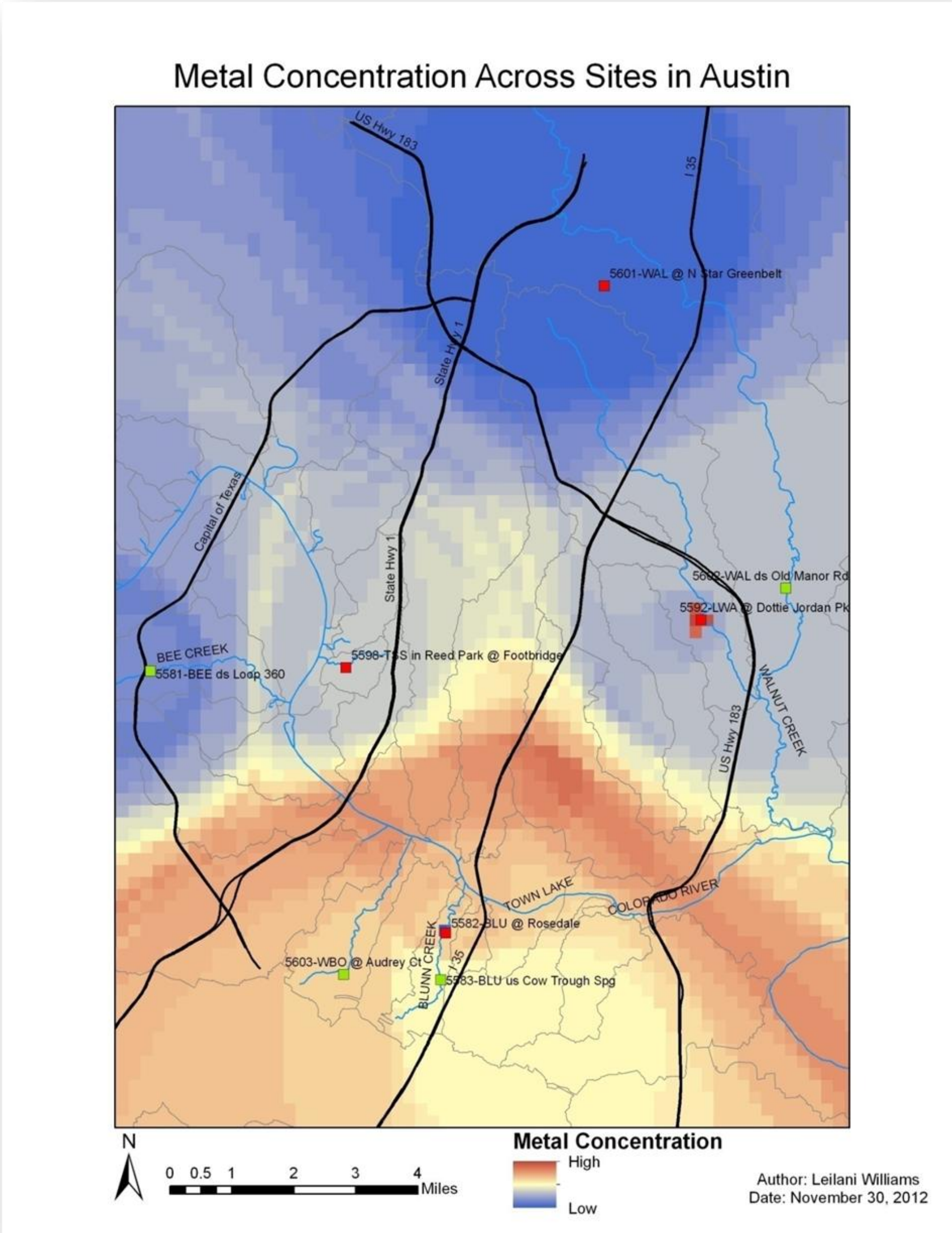

Figure 2: Resulting interpolation map of riparian soil metal variables across all sites. References sites are shown in green and degraded sites are shown in red.

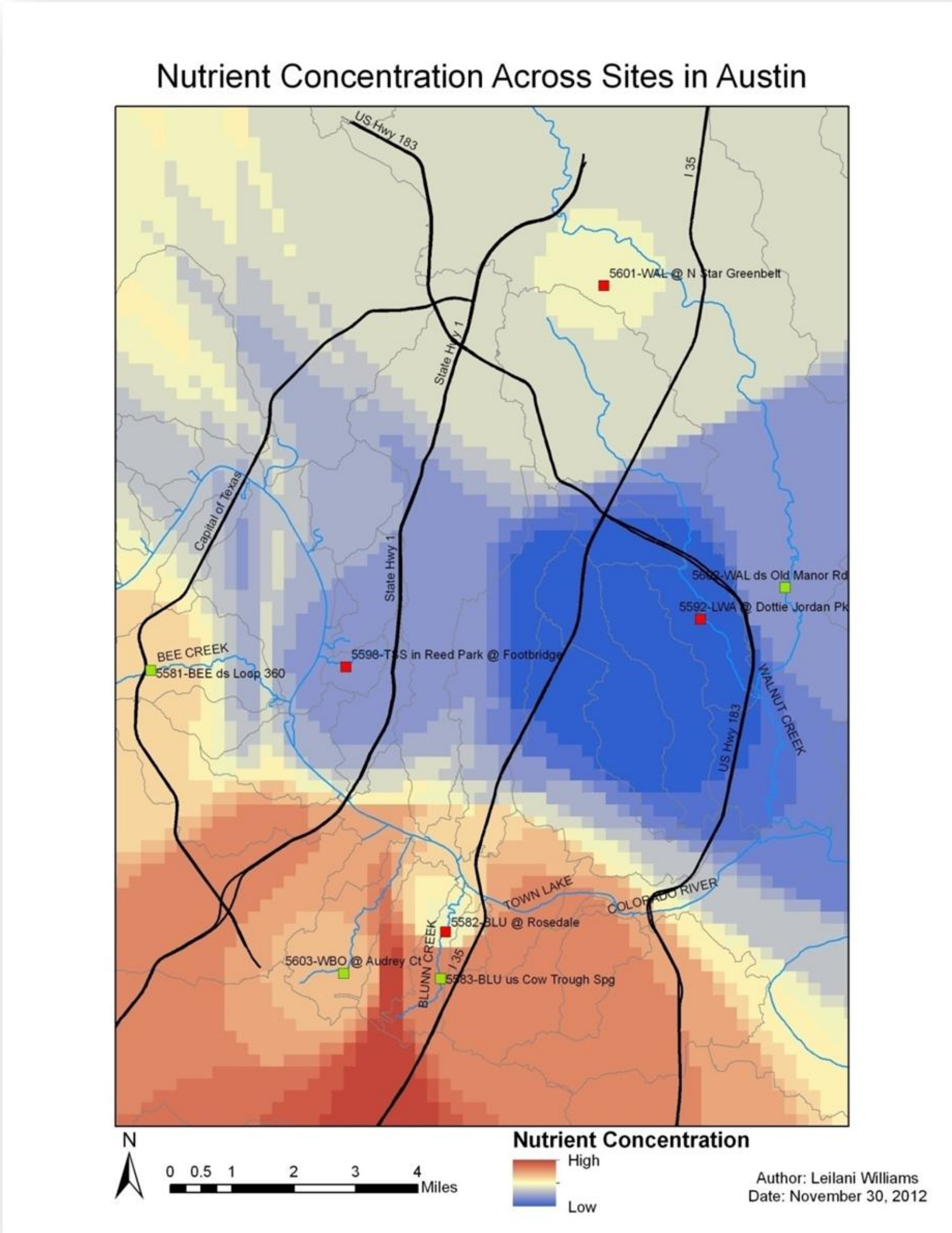

Figure 3: Resulting interpolation map of riparian soil nutrient variables across all sites. References sites are shown in green and degraded sites are shown in red.

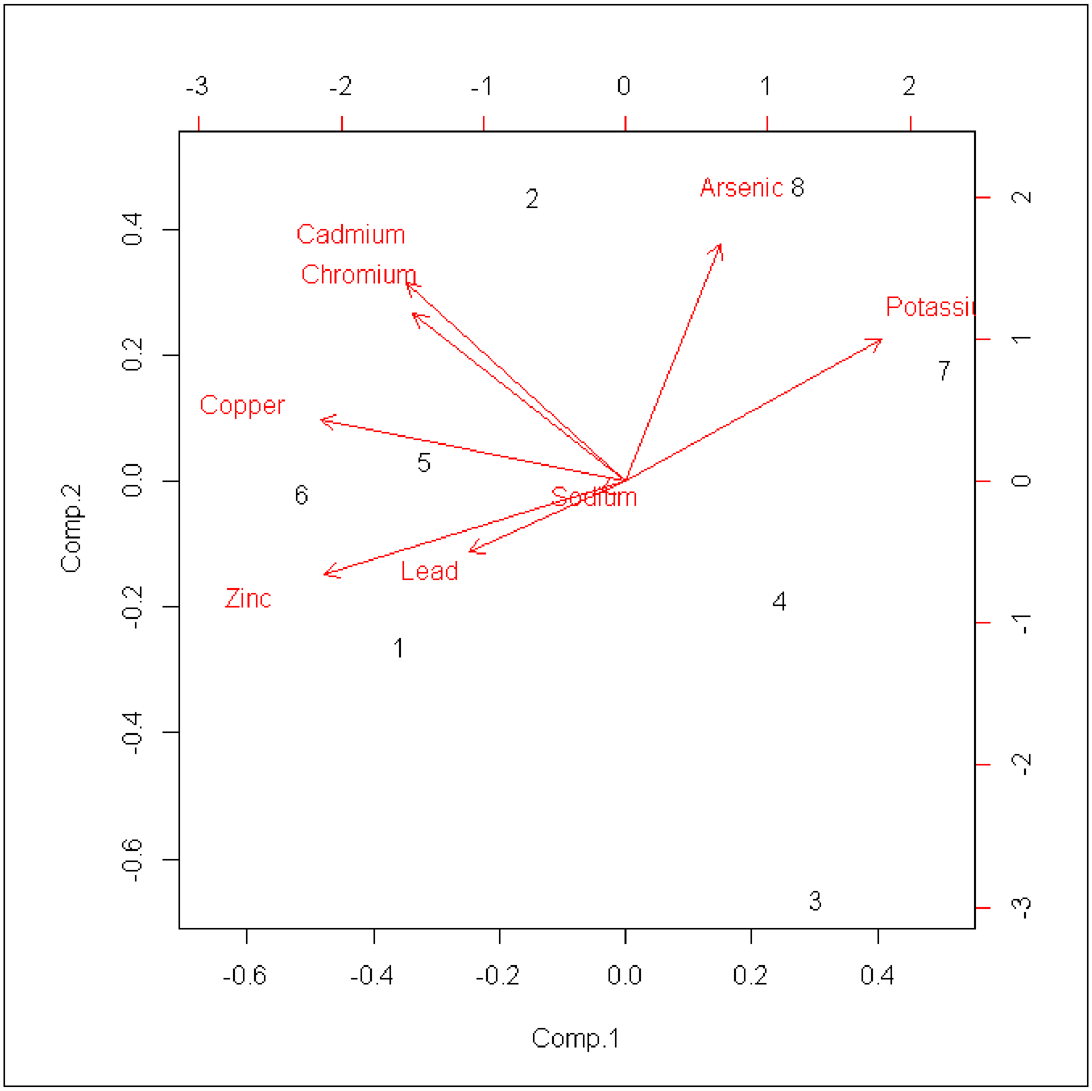

Figure 4: Summary of metal PCA for soil analysis. Component 1 = 45.4% of the overall variation, and component 2 = 19.4% of the variation, and component 3 = 14.9% of the variation. Those three components account for 80% of the variation in the samples.

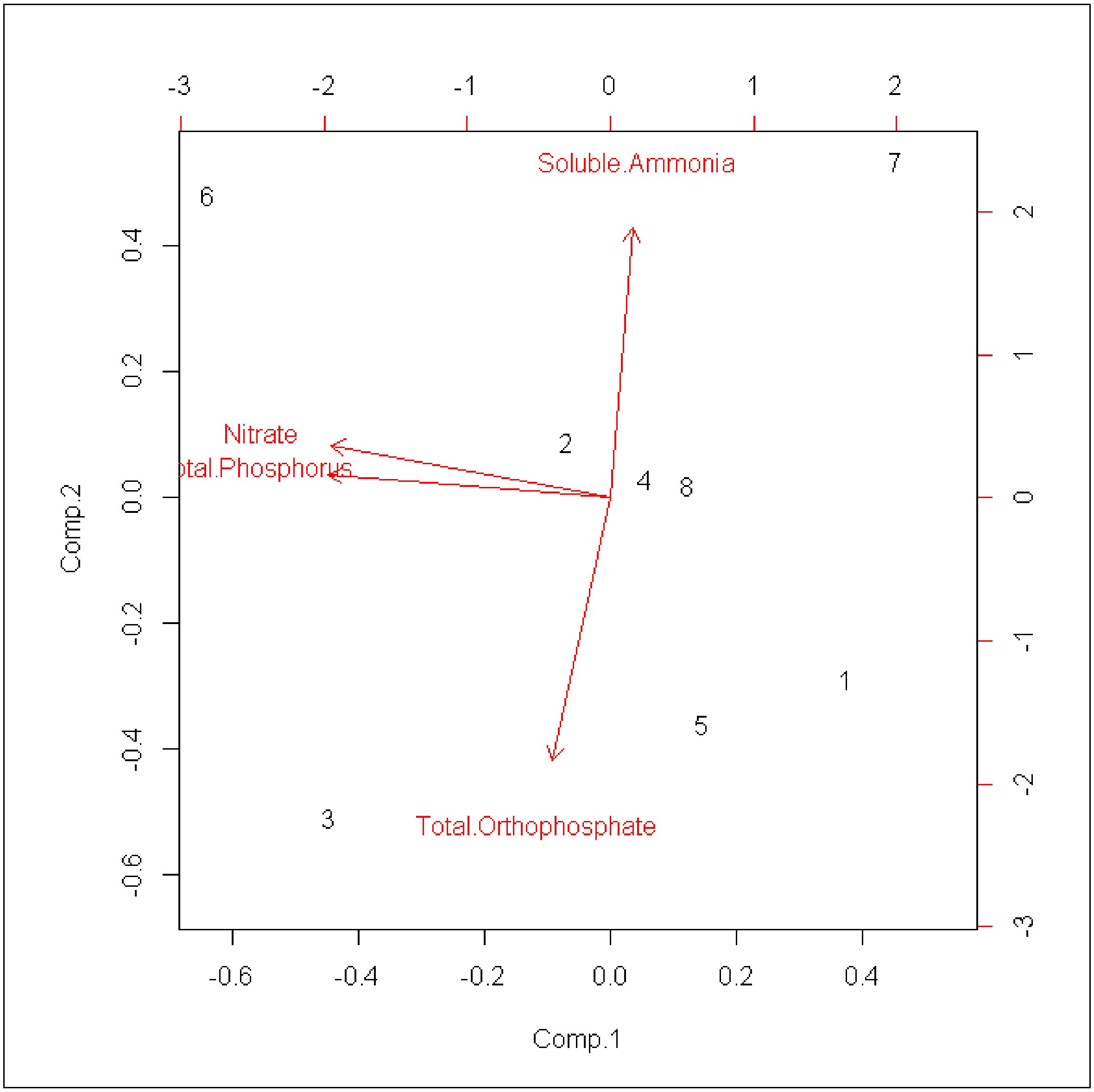

Figure 5: Summary of nutrient PCA. Component 1= 38.5% of the variation, and component 2 = 34.6% of the overall variation, making those two components worth 73% of the total variation in the samples.

WORKS CITED

Abramovitz, J. N. 1996. Imperiled Waters, Impoverished Future: The Decline of Freshwater Ecosystems. Worldwatch Paper 128. Worldwatch Institute, Washington, DC.

Alberti, M., J. M. Marzluff, E. Shulenberger, G. Bradley, C. Ryan, and C. Zumbrunnen. 2003. Integrating Humans into Ecology: Opportunities and Challenges for Studying Urban Ecosystems. Bioscience 53:1169-1179.

Allan, J. D., and A. S. Flecker. 1993. Biodiversity Conservation in Running Waters. Bioscience 43:32-43.

Arnell, N., B. Bates, H. Lang, J. Magnuson, and P. Mulholland. 1996. Hydrology and freshwater ecology. Pages 325- 363 in R. T. Watson, M. C. Zinyowera, and R. H. Moss, editors. Climate change 1995: impacts, adaptations and mitigation of climate change: Scientific-Technical Analyses. Cambridge University Press, Cambridge, UK.

Baker, M., D. Weller, and T. Jordan. 2006. Improved methods for quantifying potential nutrient interception by riparian buffers. Landscape Ecology 21:1327-1345.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. U.S. Environmental Protection Agency; Office of Water, Washington, D.C. [Appendix B: Regional Tolerance Values, Functional Feeding Groups and Habitat/Behavior Assignments for Benthic Macroinvertebrates].

Benke, A.C., A.D. Huryn, A.S. Leonard, and J.B. Wallace. 1999. Length-mass relationships for freshwater macroinvertebrates in North America with particular reference to the southeastern United States. Journal of the North American Benthological Society 18: 308-343.

Benke, A. C. 1990. A Perspective on America's Vanishing Streams. Journal of the North American Benthological Society 9:77-88.

Blöch, H. 1999. The European Union water framework directive: taking European water policy into the next millennium. Water Science and Technology 40:67-71.

Boon, P. 2000. Global perspectives on river conservation. John Wiley and Sons, New York, New York, USA.

Carvalho Silva, E., J. Molozzi, and M. Callisto. 2010. Size-mass relationships of *Melanoides tuberculatus* (Thiaridae: Gastropoda) in a eutrophic reservoir. Zoologia 27:691-695.

Chadwick, M. A., D. R. Dobberfuhl, A. C. Benke, A. D. Huryn, K. Suberkropp, and J. E. Thiele. 2006. Urbanization affects stream ecosystem function by altering hydrology, chemistry, and biotic richness. Ecological Applications 16:1796-1807.

Chadwick, M. A., and A. D. Huryn. 2003. Effect of a whole-catchment N addition on stream detritus processing. Journal of the North American Benthological Society 22:194-206.

Chin, A., F. Gelwick, D. Laurencio, L. R. Laurencio, M. S. Byers, and M. Scoggins. 2010. Linking Geomorphological and Ecological Responses in Restored Urban Pool-Riffle Streams. Ecological Restoration 28:460-475.

Christensen, N. L., A. M. Bartuska, J. H. Brown, S. Carpenter, C. D'Antonio, R. Francis, J. F. Franklin, J. A. MacMahon, R. F. Noss, D. J. Parsons, C. H. Peterson, M. G. Turner, and R. G. Woodmansee. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological Applications. 6:665-691.

City of Austin. 2010. Standard Operating Procedures Chapters 3, 5 & 6.

City of Austin Watershed Protection Department. 2002. Environmental Integrity Index Methodology. City of Austin Environmental Resource Management 1999-01.

Cohen, J. E. 2003. Human Population: The Next Half Century. Science 302:1172-1175.

Convention on Biological Diversity. 1992. United Nations Conference on Environment and Development, Rio de Janero, Brazil. 5 June 1992.

Correll, D. L. 1996. Buffer zones and water quality protection: general principles. Pages 7 - 20 In: Buffer Zones: Their Processes and Potential in Water Protection. The Proceedings of the International Conference on Buffer Zones. September 1996. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding, and G. Pinay., Harpenden, Hertfordshire, AL5 5LJ, UK.

Costanza, R., R. d'Arge, R. de Groot, S. Farberk, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253-260.

Costanza, R. C. 1997. The Development of Ecological Economics. Edward Elgar Press, Cheltenham, UK.

Cummins, K. W., R. W. Merritt, and P. C. N. Andrade. 2005. The use of invertebrate functional groups to characterize ecosystem attributes in selected streams and rivers in south Brazil. Studies on Neotropical Fauna and Environment 40:69-89.

Daily, G. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington DC.

Dillaha, T. A., J. H. Sherrard, and D. Lee. 1989. Long-term effectiveness of vegetative filter strips. Water Environment and Society 1:419-421.

Doppelt, B., M. Scurlock, C. Frissell, and J. Karr. 1993. Entering the watershed. Island Press, Washington. 462 pp.

Duncan, A. 2012. Riparian Functional Assessment:Choosing metrics that quantify restoration success in Austin, Texas. Short Report 12-12.

Dynesius, M., and C. Nilsson. 1994. Fragmentation and flow regulation of river systems in the northern third of the world. Science 266:753-762.

Edwards, F. K., R. B. Lauridsen, L. Armand, H. M. Vincent, and J. I. Jones. 2009. The relationship between length, mass and preservation time for three species of freshwater leeches (Hirudinea). Fundamental and Applied Limnology 173:321-327.

Elser, J. J., M. E. S. Bracken, E. E. Cleland, D. S. Gruner, W. S. Harpole, H. Hillebrand, J. T. Ngai, E. W. Seabloom, J. B. Shurin, and J. E. Smith. 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecology Letters 10:1135-1142.

ESRI. 2011. ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute.

European Declaration for a New Water Culture. 2005. Fundación Nueva Cultura del Agua, Zaragoza. Pg 63.

FEMAT. 1993. Forest ecosystem management: an ecological, economic, and social assessment. U.S. Government Printing Office, Washington, DC.

Fischer, R. A. and J. C. Fischenich. 2000. Design recommendations for riparian corridors and vegetated buffer strips. U.S. Army Engineer Research and Development Center, Environmental Laboratory.

Flecker, A. S. 1996. Ecosystem engineering by a dominant detritivore in a diverse tropical stream. Ecology 77:1845-1854.

García de Jalón, D., and P. Vizcaíno (Eds). 2004. Aquatic habitats: Analysis and resroration. Fifth Internacional Symposium on Ecohydraulics, IAHR Congress Proceedings, Madrid. 1453 pp.

Gessner, M. O., and E. Chauvet. 2002. A case for using litter breakdown to assess functional stream integrity. Ecological Applications 12:498-510.

Gessner, M. O., and E. Chauvet. 1993. Ergosterol-to-Biomass Conversion Factors for Aquatic Hyphomycetes. Applied and Environmental Microbiology 59:502-507.

Gessner, M.O. & J. Schwoerbel. 1989. Leaching kinetics of fresh leaf-litter with implications for the current concept of leaf-processing in streams. Archiv für Hydrobiologie 115: 81-90.

Gift, D. M., P. M. Groffman, S. S. Kaushal, and P. M. Mayer. 2010. Denitrification Potential, Root Biomass, and Organic Matter in Degraded and Restored Urban Riparian Zones. Restoration Ecology 18:113-120.

Goldewijk, K. K., A. Beusen, and P. Janssen. 2010. Long-term dynamic modeling of global population and built-up area in a spatially explicit way: HYDE 3.1. The Holocene 20:565-573.

Gonzalez del Tanago, M., and D. García de Jalón. 2006. Attributes for assessing the environmental quality of riparian zones. Limnetica 25:389-402.

Gray, L. 2004. Changes in water quality and macroinvertebrate communities resulting from urban stormflows in the Provo River, Utah, U.S.A. Hydrobiologia 518:33-46.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. BioScience 41:540-551.

Hill, A. R. 1996. Nitrate removal in stream riparian zones. Journal of Environmental Quality 25:743-755.

Huston, M. 1997. Hidden treatments in ecological experiments: re-evaluating the ecosystem function of biodiversity. Oecologia 110:449-460.

Johnson, R. R., C. D. Ziebell, D. R. Patton, P. F. Flolliott, and R. H. Hamre (Eds.). 1985. Riparian ecosystems and their management: reconciling conflicting uses. USDA Forest Service, GTR RM-120, Fort Collins, Colorado. 523 pp.

Johnson, R. R., and J. F. McCormick. 1978. Strategies for Protection and Management of Floodplain Wetlands and other Riparian Ecosystems. GTR WO-12, USDA Forest Service,Washington D.C. 410 pp.

Kaufmann, M. R., R. T. Graham, D. A. Boyce, W. H. Moir, L. Perry, R. T. Reynolds, R. L. Bassett, P. Mehlhop, C. B. Edminster, W. M. Block, and P. S. Corn. 1994. An ecological basis for ecosystem management. Page Gen. Tech. Rep. RM 246. Fort Collins, CO.

King, R. S., and K. O. Winemiller. 2009. Development of Biological Indicators of Nutrient Enrichment for Application in Texas Streams. Texas Commission of Environmental Quality.

Kristensen, P. 1994. European Rivers and Lakes: Assessment of Their Environmental State. EEA. Page Environmental Monographs 1. Copenhagen, Denmark.

Kummu M., H. de Moel, P. J. Ward, and O. Varis. 2011. How Close Do We Live to Water? A Global Analysis of Population Distance to Freshwater Bodies. PLoS ONE 6: e20578.

Lloyd, S. D., T. H. F. Wong, and B. Porter. 2002. The planning and construction of an urban storm-water management scheme. Water Science and Technology 45:1-10.

Lowrance, R., L. S. Altier, J. D. Newbold, R. R. Schnabel, P. M. Groffman, J. M. Denver, D. L. Correll, J. W. Gilliam, J. L. Robinson and R. B. Brinsfield, K. W. Staver, W. Lucas, A. H. Todd. 1997. Water Quality Functions of Riparian Forest Buffers in Chesapeake Bay Watersheds. Environmental Management 21:687-712.

Mabe, J. A. 2007. Nutrient and biological conditions of selected small streams in the Edwards Plateau, Central Texas, 2005-06, and implications for development of nutrient criteria: U.S. Geological Survey Scientific Investigations Report 2007-5195.

Malanson, G. P. 1993. Riparian Landscapes. Cambridge Studies in Ecology, Cambridge University Press. 296 pp.

Manly, B. F. J. 1986. Multivariate statistical methods: a primer. Chapman and Hall, London, UK.

McAllister D. E, A. L. Hamilton, B. Harvey. 1997. Global freshwater biodiversity: striving for the integrity of freshwater ecosystems. Sea Wind Bull. Ocean Voice Int. 11:1-140.

McGranahan, G., and D. Satterthwaite. 2003. Urban centers: an assessment of sustainability. Annual Review of Environment and Resources 28:243-274.

Merritt, R. W., and K. W. Cummins, editors. 1996. An introduction to the aquatic insects of North America. Third edition. Kendal/Hunt Publishing, Dubuque, Iowa, USA.

Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society 24:602-612.

Miserendino, M. L. 2001. Length-mass relationships for macroinvertebrates in freshwater environments of Patagonia (Argentina). Asociación Argentina de Ecología. Ecología Austral 11:3-8.

Moffatt, S. F., S. M. McLachlan, and N. C. Kenkel. 2004. Impacts of land use on riparian forest along an urban-rural gradient in southern Manitoba. Vegetatio 174:119-135.

Morse, C. C., A. D. Huryn, and C. Cronan. 2003. Impervious surface area as a perdictor of the effects of urbanization on stream insect communities in Maine, U.S.A. Environmental Monitoring and Assessment 89:95-127.

Naeem, S., F. S. Chapin III, R. Costanza, P. R. Ehrlich, F. B. Golley, D. U. Hooper, J. H. Lawton, R. V. O'Neill, H. A. Mooney, O. E. Sala, A. J. Symstad, and D. Tilman. 1999. Biodiversity and Ecosystem Functioning:Maintaining Natural Life Support Processes. Issues in Ecology 4.

Naiman, R. J., and R. E. Bilby (Eds.). 1998. River Ecology and Management. Springer, New York. 705 pp.

Naiman, R. J. and H. Decamps. 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics 28:621-658.

Naiman, R. J. (Ed.). 1992. Watershed Management. Springer-Verlag, New York. 542 pp.

Naiman, R. J., and H. Decamps (Eds.). 1990. The Ecology and Management of Aquatic-Terrestrial Ecotones. Man and the Biosphere Series, UNESCO, París. 316 pp.

Nations, U. 2008. World Urbanization Prospects: The 2009 Revision. Population Division of the Department of Economic and Social Affairs of the United Nations: http://esa.un.org/unpd/wup/.

Nicholls, S., and J. L. Crompton. 2005. The Impact of Greenways on Property Values: Evidence from Austin, Texas. Journal of Leisure Research 37:321-341.

Norris, R. H., and M. C. Thoms. 1999. What is river health? Freshwater Biology 41:197-209.

Odum, E. P. 1978. Ecological Importance of the Riparian Zone. In: Strategies for Protection and Management of Floodplain Wetlands and other Riparian Ecosystems. R. R. Johnson & J. F. McCormick (eds.).: 2-5. GTR WO-12, USDA Forest Service, Washington D.C.

Palmer, M., E. Bernhardt, E. Chornesky, S. Collins, A. Dobson, C. Duke, B. Gold, R. Jacobson, S. Kingsland, R. Kranz, M. Mappin, M. L. Martinez, F. Micheli, J. Morse, M. Pace, M. Pascual, S. Palumbi, O. J. Reichman, A. Simons, A. Townsend, and M. Turner. 2004. Ecology for a Crowded Planet. Science 304:1251-1252.

Paul, M. J., J. L. Meyer, and C. A. Couch. 2006. Leaf breakdown in streams differing in catchment land use. Freshwater Biology 51:1684-1695.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333-365.

Pavao-Zuckerman, M. A. 2008. The Nature of Urban Soils and Their Role in Ecological Restoration in Cities. Restoration Ecology 16:642-649.

Pennington, R. T., M. Lavin, T. Särkinen, G. P. Lewis, B. B. Klitgaard, and C. E. Hughes. 2010. Contrasting plant diversification histories within the Andean biodiversity hotspot Proceedings of the National Academy of Sciences 107:13783-13787.

Petersen, R. C., and K. W. Cummins. 1974. Leaf processing in a woodland stream. Freshwater Biology 4:343-368.

Petts, J. 1999. Handbook of environmental impact assessment. Two volumes. Blackwell Science, Oxford, UK.

Pouyat, R. V., I. D. Yesilonis, J. Russell-Anelli, and N. K. Neerchal. 2007. Soil chemical and physical properties that differentiate urban land-use and cover types. Soil Science of America Journal 71:1010-1019.

Prach, K., and R. J. Hobbs. 2008. Spontaneous Succession versus Technical Reclamation in the Restoration of Disturbed Sites. Restoration Ecology 16:363-366.

Reaka-Kudla, M. L. 1997. The Global Biodiversity of Coral Reefs: A Comparison with Rain Forests. Pages 83-108 in D. E. W. M. L. Reaka-Kudla, and E. O. Wilson, editor. Biodiversity II: Understanding and Protecting Our Biological Resources. Joseph Henry Press, Washington, DC.

Ribaudo, M. O. 1986. Regional estimates of off-site damages from soil erosion. Pages 29-46. In: Waddell, T.E. (editor). The Off-Site Costs of Soil Erosion. Proceedings of a symposium held May 1985. Conservation Foundation, Washington, D.C. 284 pages.

Richardson, D. M., P. M. Holmes, K. J. Esler, S. M. Galatowitsch, J. C. Stromberg, S. P. Kirkman, P. Pyšek, and R. J. Hobbs. 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. Diversity and Distributions 13:126-139.

Rosgen, D. L. 1997. A geomorphological approach to restoration of incised rivers in Proceedings of the Conference on Management of Landscapes Disturbed by Channel Incision.

SAS Institute Inc., SAS 9.1.3 Help and Documentation, Cary, NC: SAS Institute Inc., 2000-2004.

Sedell, J. R., P. A. Bisson, F. J. Swanson, and S. V. Gregory. 1993. What we know about large trees that fall into streams and rivers. Pacific Northwest Research Station, U.S. Dept. of Agriculture, Forest Service, Portland, OR.

SER. 2002. The SER Primer on Ecological Restoration. Society for Ecological Restoration Science and Policy Working Group. Available at www.ser.org.

Seybold, C. A., J. E Herrick, J. J. Brejda. 1999. Soil Resilience: A Fundamental Component of Soil Quality. Soil Science. 164(4):224-234.

Smith, D. G. 2001. Pennak's freshwater invertebrates of the United States: Porifera to Crustacea. Fourth edition. John Wiley and Sons, New York, New York, USA.

Stacey, P. B., A. Jones, J. C. Catlin, D. A. Duff, L. E. Stevens, and C. Gourley. 2006. User's Guide for the Rapid Assessment of the Functional Condition of Stream. Riparian Ecosystems in the American Southwest. Wild Utah Project.

Stanford, J. A., and G. C. Poole. 1996. A protocol for ecosystem management. Ecological Applications 6:741-744.

Sung, C. Y., M.-H. Li, G. O. Rogers, A. Volder, and Z. Wang. 2011. Investigating alien plant invasion in urban riparian forests in a hot and semi-arid region. Landscape and Urban Planning 100:278-286.

Texas Commission on Environmental Quality [TCEQ]. 2007. Surface Water Quality Monitoring Procedures, Volume 2: Methods for Collecting and Analyzing Biological Assemblage and Habitat Data. Prepared by Surface Water Quality Monitoring Program, Monitoring Operations Division, TCEQ.

Thorp, J. W., and A. P. Covich, editors. 2001. Ecology and classification of North American freshwater invertebrates. Second edition. Academic Press, San Diego, California, USA.

Tongway, D. J., J. A. Ludwig. 2011. Restoring Disturbed Landscapes: Putting Principles Into Practice. Island Press. Washington, D.C.

Vannote, R. L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The River Continuum Concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Vorosmarty, C. J., P. Green, J. Salisbury, and R. B. Lammers. 2000. Global water resources: vulnerability from climate change and population growth. Science 289:284-288.

United Nations 2012. World Urbanization Prospects, the 2011 Revision. Official Record. New York. http://esa.un.org/unpd/wup/CD-ROM/Urban-Rural-Population.htm.

U.S. Census Bureau: State and County QuickFacts. 2013. Data derived from Population Estimates, American Community Survey, Census of Population and Housing, County Business Patterns, Economic Census, Survey of Business Owners, Building Permits, Consolidated Federal Funds Report, Census of Governments.

USDA. 1997. Riparian Forest Buffer Handbook for the Chesapeake Bay Watershed. United States Department of Agriculture, Environmental Protection Agency, Forest Service and Northeastern Area State & Private Forestry, NA-TT-02-97. Washington, DC.

USEPA. 2000. The quality of our nation's waters. Environmental Protection Agency.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706-723.

Walters, A. W. 2011. Resistance of aquatic insects to a low flow disturbance: exploring a trait-based approach. Journal of the North American Benthological Society 30:346-356.

Walters, A. W., and D. M. Post. 2011. How low can you go? Impacts of a low-flow disturbance on aquatic insect communities. Ecological Applications 21:163-174.

Ward, J. V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation 83:269-278.

Warner, R. E., and K. M. Hendrix (Eds.). 1984. California Riparian Systems. California University Press, Berkeley. 1035 pp.

Webster, J. R., and E. F. Benfield. 1986. Vascular Plant Breakdown in Freshwater Ecosystems. Annual Review of Ecology and Systematics 17:567-594.

Weller, D. E., T. E. Jordan, and D. L. Correll. 1998. Heuristic models for material discharge from landscapes with riparian buffers. Ecological Applications 8:1156-1169.

Wenger, S. J., A. H. Roy, C. R. Jackson, E. S. Bernhardt, T. L. Carter, S. Filoso, C. A. Gibson, W. C. Hession, S. S. Kaushal, E. Martí, J. L. Meyer, M. A. Palmer, M. J. Paul, A. H. Purcell, A. Ramírez, A. D. Rosemond, K. A. Schofield, E. B. Sudduth, and C. J. Walsh. 2009. Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. Journal of the North American Benthological Society 28:1080-1098.

Wiens, J. A. 2002. Riverine landscapes: taking landscape ecology into the water. Freshwater Biology 47:501-515.

Wigington, P. J., and R. L. Beschta. 2000. Riparian Ecology and Management in Multi-Land Use Watersheds. Procs. AWRA´s 2000 Summer Specialty Conference, Portland, Oregon. 616 pp.

World Health Organization. 2010. Progress on Sanitation and Drinking Water: 2010 update. World Health Organization (WHO) and UNICEFF: Joint Monitoring Programme for Water Supply and Sanitation.

Woodcock, T. S., and A. D. Huryn. 2005. Leaf litter processing and invertebrate assemblages along a pollution gradient in a Maine (USA) headwater stream. Environmental pollution 135:363-375.

Woolsey, S., F. Capelli, T. O. M. Gonser, E. Hoehn, M. Hostmann, B. Junker, A. Paetlod, C. Roulier, S. Sshweizer, S. D. Tiegs, K. Tockner, C. Weber, and A. Peter. 2007. A strategy to assess river restoration success. Freshwater Biology 52:752-769.

Woolsey, S., C. Weber, T. Gonser, E. Hoehn, M. Hostmann, B. Junker, C. Roulier, S. Schweizer, S. Tiegs, K. Tockner, and A. Peter. 2005. Handbook for evaluating rehabilitation projects in rivers and streams. Rhone-Thur project. Eawag, WSL, LCH-EPFL, VAW ETHZ.

Wynn, T. 2013. Personal Communictaion. University of Alabama Department of Biological Sciences.

Zhang, Y. 2013. Riparian Ecology and Conservation: The Home of Cutting-edge Research on Riparian Biophysical Processes, Biodiversity, Ecosystem Functions and Services. Riparian Ecology and Conservation 1:1-2.

Zwick, P. 1992. Stream habitat fragmentation - a threat to biodiversity. Biodiversity and Conservation 1:80-97.

VITA

Leilani Faith Williams was born Leilani Faith Wight in Lompoc, California on July 3, 1989, the fourth child of Ann Lorraine Milne Wight and Ronald Hansen Wight. She moved to Benton City, Washington in 1991. After graduating from Kiona-Benton City High School in Benton City, Washington, in June 2007, she drove a tractor in the summer cherry harvest, toured Japan, and then entered Brigham Young University in Provo, Utah in August 2007 on several academic scholarships, including the Saundra Wolkenhauer Memorial Scholarship, Benton-Franklin Horse Council Scholarship, and a full tuition Flour Corporation Academic Scholarship. While at BYU, she completed two internships with Dr. Russell B. Rader and Bio-West involving the ecological restoration of the Lower Hobble Creek Wildlife Management Area in Springville, Utah and the corresponding survivorship of the endangered June Sucker, *Chasmistes liorus*. She published her first journal article with Dr. Jerry Johnson of the BYU Ecology and Evolution laboratory in the journal Ecosphere on the differences in media reporting of ecological topics. She received the degree of Bachelor of Science in Biology from BYU in August 2011. Two weeks later, in August 2011, she entered the Graduate College at Texas State University-San Marcos to pursue a Master of Science in Aquatic Resources.

Permanent Address:

40603 Highland Road

Benton City, WA 99320

This thesis was typed by Leilani F. Williams.