Using Leaf-Litter to Determine Ecological Function of Urban Riparian and Stream Systems in Austin, Texas

SR-13-11, December 2013

Leilani Williams

City of Austin
Watershed Protection Department
Environmental Resource Management Division

Abstract

Riparian restoration is a common water management practice used to reestablish ecosystem functionality. Ecosystem assessments may determine the loss in functionality of a system and are a prerequisite for identifying and alleviating effects of anthropogenic stresses. Leaf-litter decomposition was proposed as an integrated metric for assessing anthropogenic impacts and ecosystem functionality in urban streams. Texas Red Oak (*Quercus texana*) and American Sycamore (*Platanus occidentalis*) litter decomposition rates were measured in response to levels of riparian integrity. Macroinvertebrate colonization, riparian soil composition, and water quality characteristics were collected in addition to leaf-litter decomposition rates. No difference was found in leaf-litter decomposition rates or macroinvertebrate colonization between levels of riparian integrity. Leaf-litter decomposition is not recommended for future urban riparian ecosystem assessments as other metrics generate more useful data with less time and resources.

Introduction

Riparian systems are included in aspects of water management (García de Jalón and Vizcaíno, 2004) and riparian restoration is considered essential for ecosystem sustainability (Gonzalez del Tanago and García de Jalón, 2006). Urban riparian systems often consist of altered soil composition (Moffatt et al., 2004), and the vegetation composition is commonly dominated by exotic species, whose spread is closely linked to anthropogenic modification (Pennington et al., 2010). Urbanization also alters stream hydrology, channel geomorphology, chemical features such as nutrient cycling, and biological and trophic resources (Chadwick et al., 2006).

These altered processes affect 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). Urban riparian zones often exhibit symptoms of the Urban Stream Syndrome (Walsh et al., 2005), which include:

- Flashier hydrographs
- Elevated concentrations of nutrients and contaminants
- Altered channel morphology and stability
- Reduced biotic richness
- Increase in nonpoint source pollution
- Increase in tolerant species in dominance
- Disconnected riparian zones
- Homogeneous habitats

Ecosystem assessments are critical for determining and alleviating stresses caused by urbanization (Gessner and Chauvet, 2002; Zhang 2013). Leaf-litter decomposition was used as an integrated metric for assessing riparian zone disturbance on stream ecosystem function (Paul et al., 2006). Allochthonous litter plays a crucial role in streams and effects of anthropogenic perturbations are often evident on litter decomposition (Gessner and Chauvet, 2002). We measured leaf litter decomposition rate, aquatic macroinvertebrate leaf bag colonization, water chemistry, and riparian soil composition. Overall, understanding which metrics are most closely linked to ecological function will allow managers to better streamline monitoring efforts and allow for more focused restoration activities.

Methods and Results

Study Area

The study sites located in Austin, Travis County, Texas, USA (Table 1 and Figure 1), were selected based on the City of Austin Riparian Functional Assessment (Richter and Duncan, 2012). Sites were chosen based on the following criteria:

- Reference and degraded sites were generally paired.
- Sites have similar drainage areas
- Sites were not likely to dry over the sampling period based on historical data as leaf litter decomposition and water quality would be impossible to sample without flowing water, although some study sites did go dry during the study.

Table 1: Initial site list for all sites. Sites dropped from experiment (numbered 0) 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

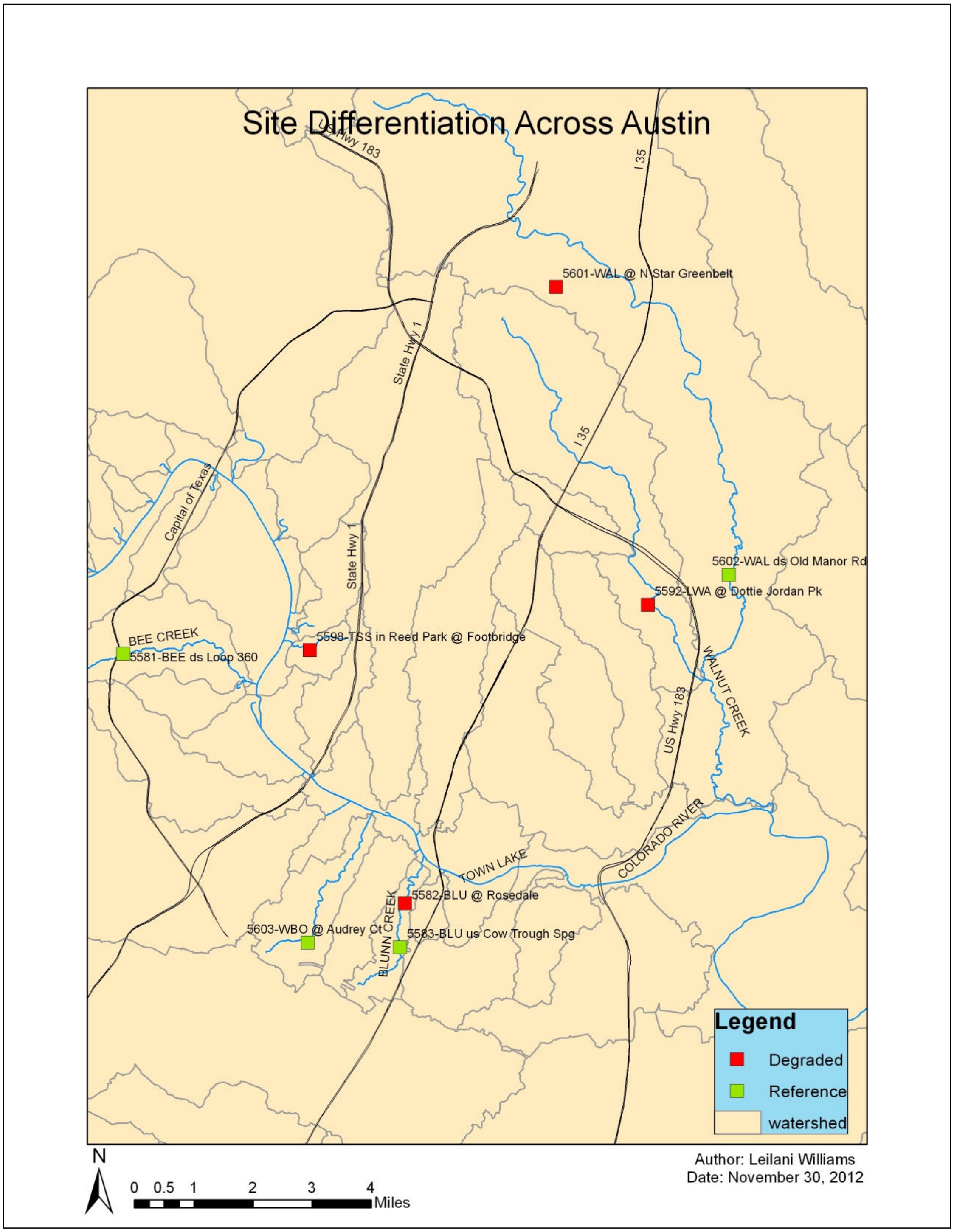

Figure 1: Site map of Austin, with final study sites listed as degraded or reference. Streams of interest, major roads and major reservoirs are also shown and labeled, along with unlabeled watersheds.

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). We sampled leaf bags in two week increments. 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 (±0.1 g) of forced-air oven dried leaves were placed in each litter bag. Litter bags were approximately 1 ft in length, with black hardware nets approximately 6 in × 6 in placed inside to hold yellow bag open and avoid variations in leaf bag dimension over time (Gessner and Chauvet, 2002) (Figure 2).

Figure 2: Leaf-litter bag with 5 g dried Red Oak leaves

Bags were secured to a small brick with a zip tie and laid in on the stream bed (Figure 3). Bags were deployed in June 2012 and collected after two weeks, four weeks, and six weeks. On each date, bags were removed from the streams, placed in individual 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 also allow macroinvertebrates access to leaves.

Figure 3: Securing leaf-litter bags into stream

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 of deployment from the leaching bags was used to calculate a leaching loss coefficient, which was subtracted from each two, four, and six week leaf bags to calculate an adjusted initial mass for each bag (Gessner and Schwoerbel, 1989).

Leaf-litter decomposition data were analyzed first with a Welch two sample T-test for difference in riparian condition (spatial), and a Pearson correlation test for change over time (temporal). 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 ($p=0.62$), or for Sycamore leaf-litter decomposition ($p=0.21$). Pearson correlation coefficients show leaf decomposition was highly correlated with length of deployment, with correlation values of 0.82 for Red Oak and 0.73 for Sycamore leaf litter. 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.).

Decomposition was 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 (Table 2). For Red Oak and Sycamore, there was a significant difference for the temporal variable ($p<0.0001$). However, there was not a significant difference in leaf decomposition when compared to riparian condition (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 riparian condition and time for either leaf species.

Table 2: Leaf-litter linear mixed effect model between degraded and reference riparian condition sites. No significant differences between reference and degraded sites ("Riparian Condition" in table), showing that decomposition differentiation according to group did not occur. Significant differences over time ("Temporal Variable" in table), showing that decomposition over time occurred, as expected.

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

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 (.

Macroinvertebrate 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, usually 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. This resulted in five distinct functional feeding groups: predators (PR), collector-gatherers (CG), filterers (FL), snails (SN), and shredders (SH) (TCEQ, 2007).

The functional feeding group structure of macroinvertebrates associated with both Red Oak and Sycamore were statistically similar (Table 3). 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.

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

There were some differences in richness among groups of sites. Using a Wilcoxon, Nemenyi, McDonald-Thompson multiple comparison test, Site 3 and 7 were grouped (not significantly different), sites 7, 1, and 4 were grouped, and sites 1, 4, 6, and 2 were grouped. 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 by the Friedman test ($p>0.05$). 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. There was no correlation between biomass and leaf-litter decomposition. There was also no correlation between richness and leaf-litter decomposition.

Table 3: Macroinvertebrate biomass (mg/litter bag) and richness (number 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 and SH**	**FC**	**Total**	**CG**	**SC**	**P**	**CG and SH**	**FC**	**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	[redacted]											
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	[redacted]											

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

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 Hydrolab MiniSonde 4a v2.06 (Table 4). All City of Austin multiprobes receive routine maintenance and are calibrated prior to deployment and accuracy is validated post-deployment (CoA 2010). Densiometer reading using a concave forest densiometer at leaf pack sites was also taken at the beginning and end of the study period. Flow data were collected using a Flow-Mate Model 2000 Water Current and Flow Meter (Flow-Tronic, Welkenraedt, Belgium), depth using a standard United States Geological Survey wading staff, wetted and bankfull channel width using a 50 m 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).

Table 4: Water quality and physical data for each site averaged across study period. Means are presented ± one standard deviation.

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

There were no significant differences between reference and degraded riparian condition site groups for the water quality mixed effects model, indicating similar water quality environments for both groups (Table 5). There were significant differences within sites over time, which indicates highly variable site measures.

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

Soil Composition

Soil samples were collected at each sampling location, approximately 5 m 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 Analytical Lab for analysis. 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). Soil composition maps (Figure 4) suggest regional patterns but do not appear to be related to riparian condition.

A Pearson correlation analysis was used on all the metals. Chromium was highly correlated to iron (r=0.99), magnesium (r=0.91), and nickel (0.96). Calcium was highly correlated as to copper (r=0.83). 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 site/riparian condition. We used a MANOVA test, Hotelling’s two sample T square test, to determine treatment differences in both nutrient and metal data.

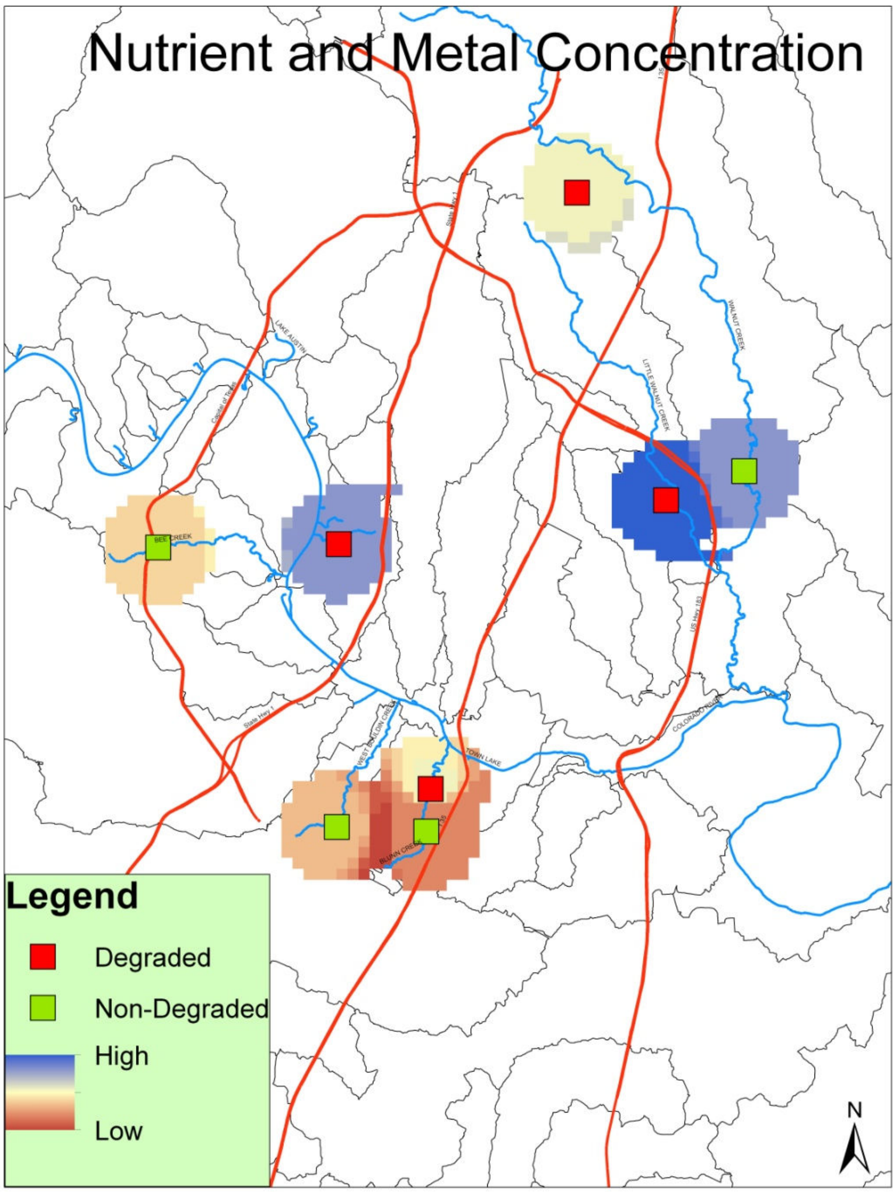

Figure 4: Compiled riparian soil data. Map prepared with ArcGIS (ESRI 2011).

Conclusions

Leaf-litter decomposition showed no significant spatial differences between reference and degraded riparian condition, but did demonstrate temporal differences indicating stream function over a temporal scale, with rates of decomposition consistent with other values from the literature. Similarities were found for both benthic macroinvertebrate richness and biomass regardless of riparian condition. Water quality measurements showed no significant differences between riparian condition sites. Water quality correlations were not highly correlated to riparian zone condition.

There are multiple possible explanations for these leaf-litter decomposition results. Despite a gradient of riparian conditions, all sampled sites are urban streams and as such, are subject to some degree of altered hydrology upstream of chosen sampling locations. Current extreme drought conditions in Austin could be influencing the variability of leaf-litter decomposition, as well as macroinvertebrate community assemblages. There was a lack of shredders observed in the macroinvertebrate analysis, which feed largely on leaf detritus (Chadwick et al., 2006) and whose presence would have altered tolerance metric results for study sites. Possible synergistic effect of multiple leaves in decomposition could affect benthic macroinvertebrate colonization potential (Gulis, 2003), although 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).

Recommendations

For many of the degraded riparian areas in Austin, Texas, passive or active restoration has been recommended for sites where temporary or easily-modified human disturbance has taken place (Richter and Duncan, 2012). Therefore, it is important to have reliable metrics that can measure differences in riparian zones and ultimately assess success of riparian restoration over time.

Based on the cost in time and labor resources without clear indications of differences between sites with degraded versus reference riparian conditions, we suggest leaf-litter decomposition and macroinvertebrate leaf bag colonization not be used repeatedly as a metric by the City of Austin for riparian ecosystem assessments. 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.

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. Our results, though not successful in providing another metric for assessing riparian ecosystem function, do provide guidance for management practices and inform decision-makers. This understanding is necessary prior to implementing broad restoration plans and identifying metrics to measure restoration success over time.

References

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.

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.

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

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.

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.

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

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 decomposition to assess functional stream integrity. Ecological Applications 12:498-510.

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.

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

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.

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.

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

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.

Richter, A., and A. Duncan. 2012. Riparian Functional Assessment: Choosing Metrics that Quantify Restoration Success in Austin, Texas. City of Austin Watershed Protection Department. SR-12-12.

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

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.

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.

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., 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.

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

Zhang , Yixin. 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.