Final Report

The ecological response of small streams to stormwater and stormwater controls in Austin, Texas USA

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Preface and Acknowledgements

This project was funded by the United States Environmental Protection Agency (USEPA) under a cooperative agreement with the Watershed Management Institute (WMI). J. Maxted, E. Livingston, and E. Shaver of the WMI developed the project work plan. The WMI contracted with the City of Austin (COA), Texas to conduct this research. Sites were selected, sampled, and the data compiled and managed by COA staff, including M. Scoggins, R. Hansen, and O. Hernandez. Macroinvertebrate sample processing was conducted by COA staff. Taxonomic identifications were conducted by O. Hernandez and A. Leopold, with support provided by Dr. R. Nelson of the University of Texas. Chironomidae identifications were contracted to S. Thompson and S. Stringer. Quality control (QC) analyses were conducted by CEO staff, and the results are presented in a report entitled "Quality Assurance report" dated 29 July 1998. Additional QC testing of the macroinvertebrate data was performed by J. Maxted of WMI and M. Winnell of Freshwater Benthic Services, with results presented in a report entitled "Final Quality Assurance Report on Biological Data Prepared by the City of Austin", dated 3 December 1998. All data presented passed QC checks.

This is a summary report of this research, with supporting data contained in Appendix A-F. More detailed data are also contained in the project binder in hard copy and on disk. Preliminary results were presented at a United States (US) (Hansen 1997) and an international conference (Maxted 1999).

Appendices:

- Appendix A – site names, identification numbers, sample dates
- Appendix B – catchment area, land use, and % impervious cover data
- Appendix C – stormwater control data and descriptions
- Appendix D1 – physical habitat data
- Appendix D2 – stream channel, hydrologic, and water quality data
- Appendix E1 – macroinvertebrate metric data
- Appendix E2 – macroinvetebrate taxonomic data
- Appendix F - 35 mm slide log sheet

Executive Summary

Macroinvertebrate, flow, and physical characteristics of 43 wadeable streams were sampled during the summer of 1997 in central Texas, in and around the City of Austin (COA). Over the preceding 15 years, the COA's stormwater program has required the construction of on-line retention ponds in first and second order streams to treat stormwater. 20 sites in catchments with no stormwater controls were compared to 18 sites with 60-95% of their catchments treated by these ponds. Stormwater ponds significantly reduced the adverse effects of urbanization in fully developed catchments (> 40 %IC). The mean EPT richness was significantly higher in streams with ponds (3.4 ± 2.2) (mean ± 1 standard deviation) compared to streams with no ponds (1.8 ± 1.3). The mean %EPT abundance of was also significantly higher in streams with ponds (16 ± 17) compared to streams without ponds (6 ± 4). There was no significant effect in streams with lower urban densities (< 40 %IC), and those with and without stormwater controls ponds were similar. Similar results were found using multidimensional scaling of the relative abundance data.

The lack of biological effects at moderate and low levels of urbanization (10-40 %IC) may have been due to two primary factors. First, urban land use in the catchments of the reference sites (5.6-7.7 %IC) may have eliminated or substantially reduced the ability to detects effects. Second, the invertebrate community of the study streams may be different from those in temperate regions, requiring a different set of metrics and interpretive thresholds. The narrow range of HBI scores (5.1-6.3) over a wide urban density gradient (5-70 %IC) indicated that the community responded differently to urbanization than published studies in temperate regions. Further evidence included low EPT richness (< 10 taxa), absence of Plecoptera, and dominance of Chironomidae (Diptera) at reference sites. The mechanisms supporting a different community in Texas streams compared to those further north may include more variable flow conditions, including periodic desiccation, and higher temperatures. The communities sampled may have been in a state of recovery from partial or full desiccation during the drier than average winter and spring of 1997. Water temperatures are normally > 25°C in central Texas streams, and is above the lethal threshold for many pollution sensitive EPT taxa. It may be more difficult to detect adverse effects of urbanization using the invertebrate community because of the absence of pollution sensitive taxa. Additional research is needed on the ecology of these streams before the invertebrate community can be fully utilized to assess the effects of urbanization and the benefits of stormwater controls.

Introduction

Austin is the fifth largest city in Texas and the twenty-seventh largest in the United States (US) with a 1997 census population of 567,566. This study was conducted within the Austin Metropolitan Statistical Area which includes Travis, Williamson, and Hays counties, covering a total land area of 586 km^2 and a 1997 population of 1,026,299. Austin is located in the Texas hill country approximately 230 miles from the border with Mexico. Austin has a moderate climate with averages of 34 inches of rainfall and 300 days of sunshine per year. It rarely snows in Austin.

While the climate and total rainfall would be classified as moderate, the distribution of rainfall is substantially different than temperate regions (Baker, 1977). A greater proportion of the rainfall is associated with severe storm events, and long periods (weeks to months) with no rainfall are common. The lower relative humidity and higher temperatures also lead to high rates of evaporation and greater reductions in surface water between rainfall events than in temperate regions. Small streams are prone to periodic desiccation.

Stream flows are supported extensively by underground aquifers. The Edwards Aquifer is approximately 160 miles long measuring from Brackettville to Kyle and varies in width from 5 to 40 miles. It traverses several streams in three major river basins including the Nueces, San Antonio and Guadalupe. Karst characteristics of the Edwards Aquifer make it one of the most productive aquifers in the US, and ground water flow within the aquifer is complex and difficult to predict. The Edwards Aquifer is designated by the EPA as a "sole source" drinking water supply for the 1.5 million people of San Antonio and the Austin-San Antonio corridor. The aquifer is also vital to the agricultural and light industrial economy of the region. Spring flows from the Comal and San Marcos Springs provide water for the tourist and recreation industry, critical habitat of several endangered species, and appropriated water use for regions to the south as far as the San Antonio Bay area.

The region west of Interstate 35 from Waco to San Antonio is underlain by the Edwards Plateau, and the dominant land surface forms are tablelands with relief, plains, and open high hills. Natural vegetation includes juniper and oak savanna, mesquite and oak savanna, and tall and short grasslands. The vegetation is supported by a thin layer of reddish-brown, gravelly sandy loam prairie soils. Much of the region is open woodland, forest, or native grassland that has been

grazed by cattle or sheep. Creeks and streams in this region tend to exhibit clear upland flow with alternating runs and riffles and intermittent glides and pools. Riffle and run areas with clear flow and appropriate cobble substrate provide excellent habitat for algae, periphyton, and benthic macroinvertebrate communities.

Sampling sites were within the Central Texas Plateau the Texas Blackland Prairie ecoregions (Omernick 1987) (Fig. 1). Most of the sites were in the Central Texas Plateau ecoregion characterized by bedrock and cobble substrates, moderate gradients (slope 1-2 degrees), clear water and low nutrient concentrations (COA 1996). The remaining sites were in the Blackland Prairie ecoregion characterized by a lower gradient (slope < 1 degree), finer sediment, lower water clarity and higher nutrient concentrations (Fig. 1). Riffles were common at all sites and were dominated by cobble and gravel substrata (Cummins 1962).

Stream base flows are sustained primarily by spring and artesian groundwater discharges where the Edwards Limestone is overlain by the low permeability Del Rio clay. In areas of Edwards Limestone outcrops (e.g., the Barton Springs Recharge Zone), streams can become losing streams where they intersect the high permeability and highly fractured Edwards karst limestone. During periods of extended low rainfall, many of the smaller streams can become intermittent resulting in the dewatering of riffle and run areas. Larger streams can also become dry during extended period of low rainfall. Under moderate to extreme drought conditions, aquatic habitats in small streams are often reduced to isolated pools.

At the other extreme, stream channels are also subjected to high velocities and flows during intense storms, resulting in scouring of the channel. High flow regimes can also persist for extended periods of time during extremely wet years (e.g. 1991-1992) resulting in widening of stream channels and deposition of gravel and sediment downstream. Additionally, these storm events can deliver increased nutrient loads. Studies conducted by the Texas Water Commission on Barton Creek in 1985 and the James River in 1987 reported nitrate-nitrite and orthophosphorus levels of 0.01-1.21 mg/l and 0.01-0.02 mg/l, respectively (TWC, 1989). During storm events of the same period, the COA Environmental and Conservation Services Department reported nitrate-nitrite and orthophosphorus levels of 2.48 mg/l and 0.90 mg/l, respectively (COA, 1995).

The naturally low levels of nutrients in these stream systems are an important factor in the level of primary productivity and the abundance and diversity of benthic macroinvertebrates. However, when nutrient concentrations increase and optimal growth conditions exist, naturally

low levels of algae growth can increase dramatically and produce nuisance levels of algae cover, primarily composed of filamentous green algae of the orders Cladophorales and Zygnematales. Drought conditions following storm events exacerbate these problems in pools and stormwater ponds. Algae blooms dominated by *Cladophora*, *Rhizoclonium*, and *Spirogyra* genera are often episodic in response to climatic variations, resulting in impacts on the physical, chemical, and biological conditions in streams. The periodic cycling of high flow and desiccation shape the structure and function of the aquatic communities in these streams.

In response to relatively common flash flooding and the city's rapid growth, erosion and sediment control regulations, floodplain management and stormwater control regulations were implemented by the COA in the mid-1970s. In the mid-1980s stormwater control regulations required new developments to use appropriate BMPs to treat stormwater quantity and quality. Design criteria for erosion and sediment control BMPs and stormwater BMPs are set forth in the City's Environmental Criteria Manual (COA, 2003).

The principal BMP for stormwater treatment is a sedimentation/filtration system that is combined with a dry detention basin for peak discharge rate control. These systems are designed to treat a minimum of one-half inch of runoff plus 0.1" for each 10% increase in impervious area above 20 percent. The dry detention system is designed so that the post-development peak discharge rate does not exceed the pre-development rate for a 2, 10, 25, and 100 year, 24 hour storm. Additionally, the city implemented three ordinances (Comprehensive Watershed Ordinance-1986, Urban Watersheds Ordinance – 1991, and the Barton Springs Ordinance – 1992) that require source controls and special land use management.

Study Objectives, Study Design, and Site Selection

The Austin metropolitan area was selected for study because it contained numerous small streams with similar physical characteristics, and a range of urban development densities that provided a disturbance gradient for assessing the impacts of urbanization. Sites were distributed across the Central Texas Plateau and Texas Blackland Plateau ecoregions (Omernik, 1987). The stormwater control ponds put in place since the mid-1980s provided the basis for assessing the effectiveness of this control device in mitigating or reducing the adverse effects of urbanization.

The macroinvertebrate community was selected as the primary assessment endpoint for two primary reasons. First, the macroinvertebrate community provides a holistic measure of ecological condition at a high level of biological organization. The community reflects the cumulative effects of physical, chemical and biological conditions over wide spatial and temporal scales. Second, sampling and data analysis methods and thresholds are well documented. A variety of physical and water quality variables were used to interpret the biological results. The study addressed two primary questions:

1. What is the impact of urbanization on small streams in central Texas?
2. To what degree do stormwater ponds mitigate adverse effects?

Research has shown that the ecological quality of urban streams, as measured using macroinvertebrate community, is inversely related to urban density (Booth et al., 1994, Weaver and Garman 1994, Limburg and Schmidt 1990, Jones and Clark 1987, Garie and McIntosh 1986, Pedersen and Perkins 1986, Klein 1979). This research has recently been summarized (Schueler 2003, 1994). However, relatively little is known about the impacts of urbanization on small streams in central Texas. The implementation of stormwater controls in Austin over the last 15 years also provided the opportunity to assess their effectiveness in reducing adverse effects. The assessment of the ecological benefits of stormwater controls was the primary objective for this research, and for similar studies under this Cooperative Agreement in Maryland, Washington, and Colorado.

Stream sites were selected in two groups across the full range of urban density. 20 sites had no stormwater control ponds in the catchment, and were referred to as “no pond” (NP) sites (Fig. 1). 18 sites were selected with the majority (60-95%) of the catchment draining to stormwater control ponds, and were referred to as “with pond” (WP) sites (Fig.1). Comparisons were made between these site groups across the range of urban density to assess (1) the effects of development density on the stream biota, and (2) the effectiveness of stormwater ponds to reduce or mitigate the adverse effects. 5 non-urban “reference” (R) sites were also sampled to allow for comparisons to sites with a higher degree of land use protection and fewer stressors (Fig.1).

Stormwater control best management practices (BMP) were selected and classified using COE records, staff knowledge, and field inspections (Table 1). Controls were both “ponds” with permanent open water and dry detention basins with no permanent open water (Table 1) It was

difficult to obtain design specifications, and their design characteristics were assessed from a combination of engineering records and visual inspections using a tape measure and a staff gauge. Drainage areas to each site and to each BMP were estimated using topographic maps and GIS software.

Urban density was measured using the percentage of the contributing catchment with impervious surfaces such as buildings, roads, and parking lots. The catchment area draining to each site was delineated, and the area of 10 land use categories was estimated using digital land use records. The percent impervious cover (%IC) for the catchment was calculated by multiplying the area of each land use category by a set of land use specific estimates of %IC determined from COA geographic and aerial photographic analysis (Table 2).

A similar number of sites were selected within 4 range classes to ensure that sites covered the full range of urban density; "low" (< 13 % IC), "medium-low" (14-23 %IC), "medium-high" (24-40 %IC), and "high" (> 40 %IC) densities. Reference sites and sites with no stormwater controls were selected that had similar physical characteristics to the sites with stormwater controls. The sampling period and the preceding 3 months were relatively wet, with mean monthly flows averaging 2-times the average monthly flows. This allowed sampling to take place in the riffles of the small streams affected by stormwater controls.

Methods

43 sites were sampled between 20 June and 16 July 1997. Measurements along a 100 meter reach included macroinvertebrates, physical habitat (channel, banks, and riparian zone), channel dimensions, flow, velocity, and water quality (pH, temperature, conductivity, and dissolved oxygen). Habitat quality was measured using a visual assessment of 13 variables, and scored on a relative scale of 1-20 points (EPA, 1989, Table 3). Total habitat scores were determined for each site by summing the scores of the individual measures; the total score range was 13-260 points. Methods were taken from procedures developed by the USEPA (Barbour et al., 1999). Photographs were taken at each site.

Channel dimensions and physical habitat variables were taken according to EMAP Field Operations and Methods Manual for Streams (EPA, 1994). Discharge was measured with a Marsh-McBirney Flowmate following standard procedures (USGS 1984). Physicochemical

measurements were taken using a Hydrolab Datasonde following standard procedures (COA, 2002).

The number of sites within the 4 density classes were similar between the NP and WP groups (Table 4). The catchment areas were also similar between NP, WP, and R groups, and ranged 0.6-6.8 km^2 for the NP group, 0.1-6.0 km^2 for the WP group, and 1.1–10.5 km^2 for the R group (Table 3). Habitat scores, flows, channel dimensions, embeddedness, and % shade were not significantly different between the 3 groups of sites (Table 5).

Macroinvertebrate samples were collected in riffles substrates using a 600 um kick-net following procedures developed by the USEPA (Barbour et al., 1999). Samples were collected from 2 m^2 of riffle substrate and preserved in 80% ethanol for later processing in the laboratory. A single 200-organism subsample was removed from each sample using a Caton subsampling tray (Caton 1995) following methods developed by the USEPA (Barbour et al., 1999).

Sample size was reduced to 200-count by rarefaction and the data adjusted to the genus level before calculating biological metrics and conducting multivariate ordination using non-metric multi-dimensional scaling (MDS). Metrics included total richness (TR), Ephemeroptera, Plecoptera, Trichoptera richness (EPT), % EPT abundance (%EPT), % dominant taxon (%DT), and the Hilsenhoff Biotic Index (HBI). Quality assurance tests were applied to the data (COA 1997, Maxted 1998).

Taxa with 1-2 individuals in the entire dataset were removed to simplify the MDS analysis. The similarity in species composition and relative abundance between sample pairs was calculated using Bray-Curtis distance (Primer 2001). The technique arranges samples in two-dimensional space that place similar samples close together and dissimilar samples further apart. The "goodness of fit" of the two-dimensional solution was given by the 'stress' value. A stress value < 0.15 was selected to indicate a "good fit"; i.e., a good separation between sites. Spearman rank correlation coefficients between MDS axis scores and the relative abundance data were used to identify taxa significantly ($r_s > 0.80$, or $r_s < -0.80$) affecting the ordination (Jandel Scientific 1995).

Mean and standard deviation (± 1 SD) estimates were calculated for sample groups, and significant differences between groups were determined using the Mann-Whitney Rank Sum Test

at $p < 0.05$ (Jandel Scientific 1995). %IC was used as the principle independent valuable for plotting macroinvertebrate metric values because it provided a visual comparison of sites groups across the full urban density gradient. Significance of the linear regressions were assessed at $p < 0.05$; regressions with $r^2 < 0.25$ were considered "weak" (Jandel Scientific 1995).

Results

WP sites in the highest density class (> 40 %IC) had significantly higher mean values for EPT and %EPT compared to NP sites (Table 6). There were no significant differences in mean metric values between sites with (WP) and without (NP) stormwater ponds for the low (< 15 %IC) and medium (15-40 %IC) density classes. In the high-density class, the mean EPT richness for WP sites was 3.4 (± 2.2) (mean ± 1 standard deviation) compared to 1.8 (± 1.3) for NP sites, and the mean %EPT abundance for WP sites was 16 (± 17) compared to 6 (± 4) for NP sites. A similar pattern was found when mean metric values for each class were compared to the reference (R) class. Mean values for %EPT were significantly lower for the medium-high (NP) and high (NP, WP) classes, and for all (NP) classes combined, as compared to the R class (Table 6).

Metric values were plotted against %IC to illustrate differences between site groups, and to assess the strength of the associations with %IC, including richness (TR and EPT; Fig. 2), composition (%DT and %EPT; Fig. 3), and pollution tolerance (HBI; Fig. 4) metrics. The linear regression was significant and "weak" ($r^2 = 0.22$) for %EPT and insignificant for all other metrics. Statistically significant differences between NP, WP, and R groups at sites with > 40 %IC are illustrated in the plots for EPT (Fig. 2) and %EPT (Fig. 3) metrics. Generally, NP and WP sites were more variable than R sites for all metrics.

The lack of a strong association between groups and classes was also indicated by the MDS ordination of the relative abundance data (Fig. 5). The stress value of 0.23 indicated a "weak" association between sites. The R group sites were generally spread in the bottom right corner of the ordination, and were clustered with the low-density (NP) sites. The one reference site (R5) in the lower right-hand corner of the ordination had the largest catchment area (10.5 km^2), the highest total richness (33), the highest EPT richness (11), and the lowest (best) HBI (5.0) metric values in the study. The high-density NP sites had the largest separation from the R-group along axis 1.

The Diptera *Rheotanytarsus* was the only taxa with a high Spearman rank correlation (r_S = 0.86), providing further evidence of a weak separation between sites. This taxon was positively correlated with axis-1, indicating that it was associated with low-density sites (R-group and low-density classes) in the lower-right corner of the ordination. *Polypedilum* (r_S = -0.50) was negatively correlated with axis-1, indicating that it was associated with the high-density NP group on the left-side of the ordination (Fig. 5). The mayflies *Fallceon* (r_S = -0.63) and *Camelobaetidius* (r_S = -0.66) were negatively correlated with axis 2, indicating that these taxa were associated with low-density sites (R-group and low-density class) in the lower-right corner of the ordination.

Five major groups were rare in the data-set, representing < 2% of total abundance for all sites, and included Oligochaeta, Hirudinea, Pelecepoda, Crustacea, and Hemiptera; Plecoptera were absent (Table 7). Diptera dominated abundance and taxonomic richness, comprising 51% of the total abundance of 10,127 individuals and 41 % of the total richness of 142 taxa; 2 genera (*Rheotanytarsus* and *Polypedilum*) made up 58% of the Diptera abundance. The next most abundant groups were Trichoptera (18 % of total abundance) and Odonata (13 % of total abundance); 2 genera made up 80% of the Trichoptera abundance (*Cheumatopsyche* and *Chimarra*), and 1 genera made up 92% of the Odonata abundance (*Argia*). Coleoptera had the highest richness (12 taxa) for a group that comprised < 3 % of total abundance (290 individuals) (Table 7).

Indicative distributions of selected taxa by site type and urban class are summarized in Table 8. *Polypedilum* were common at all sites, and abundant at high-density sites with no ponds (NP). *Rheotanytarsus and Similium* were also common at most sites but rare at all high-density sites. *Dugesia* (Platyhelminthes, flatworm) abundance was inversely related to urban density because it was rare at reference and low-density sites and common at medium and high-density sites.

Discussion

Stormwater controls mitigated the adverse effects of urbanization in high-density urban catchments (> 40 % IC) as indicated by two biological metrics (EPT and %EPT) and the MDS ordination of the relative abundance data. We found no significant effect of urbanization at lower densities of urbanization (< 40 % IC), and sites with and without stormwater controls ponds had

similar invertebrate communities. The ability to detect biological effects was confounded by the weak and poorly defined relationship between the invertebrate community and urban density.

The comparisons between urban sites and non-urban (reference) sites further supported the conclusion that there were no significant adverse effects of urbanization at low and moderate urban densities. However, urban land uses in the catchments of the reference sites (5.6-7.7 %IC) may have affected the ability to define differences between site groups. In addition, the variability in metric values was often higher in urban sites, and may have contributed to the inability to detect effects at low and moderate urban densities.

Both EPT metrics (EPT and %EPT) were sensitive to the effects of urbanization, although the effects were detected at a higher density of urbanization than in temperate regions of the US. Other studies have documented adverse effects at lower urban densities (10-15 %IC) (Booth et al., 1994, Weaver and Garman 1994, Limburg and Schmidt 1990, Jones and Clark 1987, Garie and McIntosh 1986, Pedersen and Perkins 1986, Klein 1979). The lack of sensitivity in HBI scores across a wide urban density range (5-70 %IC) provided further evidence that the streams in this region have invertebrate communities different from those in temperate regions. The MDS ordination further supported this conclusion.

Similar studies in temperate regions in the eastern (Montgomery County, Maryland) and western (Puget Sound, Washington) United States highlighted the different communities we found in central Texas. Metric values in non-urban (reference) sites were comparable to values in high-density urban sites in Maryland (Van Ness 1997) and Washington (Horner 1997). This indicated that the invertebrate communities in central Texas were different than these temperate regions. Separate metrics and quality thresholds may be needed to fully assess the effects of urbanization and the benefits of stormwater controls. Other measures such as algae, fish, and water chemistry may be more sensitive to adverse effects at low to moderate levels of urban density in central Texas streams.

We suspect that the weak relationship we found between the invertebrate community and urban density may have been due to desiccation of riffles in the months prior to the sampling period. 1997 was drier than average year overall, but the 3 months prior to the sampling period (June/July) was wetter than average. The biological communities in the riffles sampled may have been in a state of recovery following desiccation in late winter and spring. Previous studies in

1993-1994 revealed that benthic macroinvertebrate communities can be lost during extended hot, dry periods, even in larger streams, and that sampling during subsequent wet periods document recolonization and recovery of the community following desiccation (Hansen, 1997). The streams sampled in the present study had very low flows (median 0.33 cfs) and were very shallow (mean depth 0.2 ft. ± 0.1), and would have been susceptible to desiccation during the preceding seasons.

These streams are also subjected to high peak flows during frequent extreme storm events that scour the channel , mobilize bed material, and physically remove the invertebrate communities. The communities of these small streams may be in a constant state of recovery from extremes of desiccation and high flows. The site with the largest catchment area was reference site R5 (see Appendices), and had the highest total richness (33), highest EPT richness (11), lowest HBI score (5.0), and largest separation in the MDS ordination. This site may have been less affected by hydrologic extremes due to a relatively large catchment area and high proportion of undeveloped land uses.

The invertebrate community dominated by Chironomidae (Diptera) was indicative of a community in a state of recovery. Many chironomidae have short aquatic stages (1-2 weeks) and are able to recolonize areas quickly after adverse events (Hynes 1972, Williams 1987, Merritt and Cummins 1996). The two Trichoptera genera (C*heumatopsyche* and *Chimarra*) common at many sites are free-living generalists that are tolerant of both nutrient enrichment and hydrologic modification (Merritt and Cummins 1996, Rosenberg and Resh 1993). The taxa, overall, were tolerant of organic pollution (HBI >5), and may have contributed to the weak association with the various stressors in urban streams.

Stream temperatures during this survey were normal for a summer period. Many pollution sensitive invertebrate taxa, particularly in the EPT groups, are sensitive to the temperatures >25°C common during the sampling period (Hynes 1972, Rosenberg and Resh 1993). The lack of these pollution sensitive taxa in streams in this part of Texas may have also contributed to the inability to detect adverse effects at low urban densities (< 40 %IC).

Physiographic factors such as geology and soils may have also contributed to data variability because sites were spread over 2 ecoregions. However, as noted previously, there were no sites that were typical of the Blackland Prairie ecoregion and there were only minor transitional effects

along the buffer between the two ecoregions. Further research is needed to determine whether stream assessments in these 2 ecoregions should be assessed separately or assessed together as was done in this study.

The percentage of the catchment treated by stormwater controls was not likely to have had an effect on the results because the percentage range was high (60-95%) across all sites, was > 80% for 11 of the 18 sites, and similar between urban classes and groups (NP, WP). Water quality may have played an important role, but the single measurements taken were not sufficient to assess their contributions. Further research is needed on the ecology of these streams before the invertebrate community can be fully utilized to assess the effects of urbanization and the benefits of stormwater controls.

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Table 1. Number, age range, and physical characteristics of stormwater ponds within the catchments of the 18 stream sites sampled, and the proportion (%) of catchment treated.

Stream ID	Stream name	No. ponds	Age range (yrs)	% catchment treated	Total pond area (m^2)	Total pond volume (m^3)
T1	Motorola	5	3-6	95	5,941	5,974
T2	Copperfield	4	3-15	85	9,295	18,677
T3	Wells Branch	1	10-12	90	51,592	103,194
T4	Boulder	5	6	75	5,376	6,381
T5	Fireoak	1	10	95	4,877	8,920
T6	Boggy Creek	8	4-15	73	57,529	68,379
T7	Cutting	4	5-17	70	6,476	7,829
T8	Bridge Point	2	8-12	85	1,589	2,124
T9	W. Howard	2	15	80	100,333	96,841
T10	Tanglewood	3	5-10	65	2,251	5,019
T11	Rain Creek	1	10	90	2,508	4,587
T12	Tar Branch	3	12-15	80	44,802	54,622
T13	Park Bend	2	10	90	42,502	62,649
T14	Graceland	1	8	95	3,600	7,879
T15	Carson	5	6-10	60	26,066	42,238
T16	Loop 360	3	7-12	80	4,447	8,825
T17	Jollyville	5	7-10	85	2,539	2,568
T18	Mearns	11	10-15	60	67,586	91,056
	Total	66				

Table 2. Land use specific estimates of percent impervious cover (%IC) for areas within (urban) and outside (non-urban) the metropolitan urban area; used to calculate area weighted estimates of %IC for catchments draining to stream sites.

Land Use	Urban (%)	Non-urban (%)
Single Family	40	30
Multi Family	80	45
Commercial	95	60
Industrial	95	60
Civic	70	30
Park	15	5
Transportation	100	85
Undeveloped	15	5
Office	95	60

Table 3. Description of physical habitat measures used to produce habitat quality scores for each site.

Variable	Points	Description
Anaerobic condition	1-20	Frequency and severity of H_2S smells
Bank condition	1-20	Stability of streams banks and evidence of erosion
Bank vegetation protection	1-20	Proportion of the bank covered in vegetation
Channel alteration	1-20	Degree of human alteration of the stream channel
Channel flow status	1-20	Percent of the active channel that is wetted
Disruptive pressure	1-20	Degree of human disturbance along the streams channel
Embeddedness	1-20	% embeddedness of cobbles in fine sediment in riffles
Epifaunal substrate	1-20	% of substrate suitable for epifaunal colonization
Frequency of riffles	1-20	% of channel length composed of riffle substrata
Instream cover	1-20	% of channel containing habitat for fish and aqautic life
Riparian zone width	1-20	Width of the riparian zone with no human disturbance
Sediment deposition	1-20	Evidence of recent deposition of sediment
Velocity depth regime	1-20	Variety of velocity and depths
Total habitat score	13-260	Sum of all variable scores

Table 4. Catchment area range and % impervious cover range for three groups of sites sampled; 5 non-urban sites (reference, R), 20 urban sites with no stormwater control ponds (no ponds, NP), and 18 urban sites with stormwater ponds (with ponds, WP).

Group	Development density	N	Impervious cover range (%)	Catchment area range (km^2)
Reference (R)	Low	5	5.6 - 7.7	1.1 – 10.5
No ponds (NP)	Low	5	5.4 – 9.5	0.6 – 5.7
	Med-low	5	13.1 – 22.5	0.6 – 4.2
	Med-high	4	24.8 – 32.9	0.9 – 4.4
	High	6	43.7 – 70.2	1.43 – 6.8
	Subtotal	**20**	**5.4 – 70.2**	**0.6 – 6.8**
With ponds (WP)	Low	3	5.4 – 12.4	0.9 – 2.6
	Med-low	7	15.3 – 21.8	0.4 – 4.6
	Med-high	5	24.2 – 39.2	0.8 – 6.0
	High	3	40.8 – 59.9	0.1 – 5.1
	Subtotal	**18**	**5.4 – 59.9**	**0.1 – 6.0**
	Total	43		

Table 5. Comparison of mean (± 1 SD) values for 6 physical characteristics between streams with (WP, in bold) and without (NP) stormwater control ponds, by urban density class low (<13 %IC), medium low (14-23 %IC), medium-high (24-40 %IC), and high (> 40 %IC). Comparison to non-urban reference sites (R) also provided; embeddedness (embed), habitat scores (260 points possible); percent impervious cover (%IC).

Urban Class	Group	N	Habitat	Flow (cfs)	Width (feet)	Depth (feet)	Embed (%)	Shade (%)
	R	5	226 (05)	1.7 (2.0)	8.9 (4.0)	0.3 (0.1)	14 (4)	52 (11)
Low	NP	5	213 (15)	1.1 (1.0)	6.5 (1.7)	0.2 (0.1)	18 (6)	46 (18)
	WP	**3**	**200 (24)**	**0.2 (0.2)**	**4.8 (2.1)**	**0.2 (0.1)**	**18 (6)**	**70 (17)**
Med-low	NP	5	202 (33)	1.1 (1.1)	11.4 (8.9)	0.3 (0.1)	22 (4)	62 (15)
	WP	**7**	**203 (28)**	**0.4 (0.4)**	**6.5 (4.4)**	**0.2 (0.1)**	**17 (6)**	**59 (12)**
Med-high	NP	4	186 (08)	0.4 (0.1)	3.7 (1.3)	0.2 (0.1)	19 (5)	46 (18)
	WP	**5**	**201 (15)**	**0.7 (0.2)**	**10.6 (13.5)**	**0.3 (0.1)**	**17 (8)**	**63 (17)**
High	NP	6	190 (22)	0.5 (0.3)	8.8 (3.0)	0.2 (0.1)	16 (5)	62 (14)
	WP	**3**	**195 (27)**	**0.4 (0.5)**	**4.6 (1.4)**	**0.2 (0.1)**	**19 (4)**	**43 (15)**
All	NP	20	198 (23)	0.8 (0.8)	7.8 (5.3)	0.2 (0.1)	19 (5)	55 (17)
	WP	**18**	**202 (22)**	**0.4 (0.4)**	**7.0 (7.5)**	**0.2 (0.1)**	**18 (6)**	**59 (16)**

Table 6. Comparison of mean (± 1 SD) values for 5 biological metrics between streams with (WP, in bold) and without (NP) stormwater control ponds, by urban density class low (<13 %IC), medium low (14-23 %IC), medium-high (24-40 %IC), and high (> 40 %IC). Comparison to non-urban reference sites (R) also provided. Total richness (TR), Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness, percent EPT abundance (%EPT), percent dominant taxon (%DT), and the Hilenhoff biotic index (HBI), percent impervious cover (%IC). Underlined values significantly different between NP and WP groups for each urban class, * values significantly different from R group.

Urban Class	Group	N	TR	EPT	%EPT	%DT	HBI
	R	5	21.6 (6.7)	6.6 (2.5)	42 (07)	33 (11)	5.5 (0.4)
Low	NP	5	23.4 (5.0)	5.8 (1.5)	29 (16)	37 (14)	5.8 (0.2)
	WP	**3**	**24.3 (2.1)**	**5.0 (2.6)**	**39 (22)**	**40 (16)**	**5.7 (0.2)**
Med-low	NP	5	21.4 (3.0)	6.0 (2.8)	31 (12)	26 (04)	5.6 (0.3)
	WP	**7**	**21.9 (5.4)**	**3.7 (2.8)**	**24 (21)**	**35 (12)**	**5.7 (0.2)**
Med-high	NP	4	25.0 (4.9)	6.0 (2.2)	* 22 (14)	31 (07)	5.8 (0.3)
	WP	**5**	**22.2 (4.1)**	**4.2 (1.1)**	**28 (10)**	**34 (05)**	**5.7 (0.1)**
High	NP	6	19.2 (4.1)	* 1.8 (1.3)	* 6 (04)	* 51 (13)	6.1 (0.1)
	WP	**3**	**22.3 (0.6)**	**4.7 (2.1)**	* **23 (19)**	**38 (12)**	**5.8 (0.6)**
all	NP	20	21.9 (4.5)	4.7 (2.7)	* 21 (15)	38 (14)	5.8 (0.3)
	WP	**18**	**22.4 (4.0)**	**4.2 (2.2)**	**27 (17)**	**36 (10)**	**5.7 (0.3)**

Table 7. Number of individuals, number of taxa, and proportion (%) of both by major taxonomic groups for the entire data set of 10,127 individuals and 142 taxa (genus level).

			Abundance		Richness	
Phylum	Class	Order	No.	%	No.	%
Platyhelminthes	Turbellaria	Tricladida	257	2.5	1	0.7
Annelida	Oligochaeta	all	28	0.3	6	4.2
	Hirudinea	all	7	0.1	3	2.1
Mollusca	Gastropoda	Lymnophila	272	2.7	5	3.5
	Pelecypoda	Eulamellibranchia	29	0.3	3	2.1
Arthropoda	Insecta	Crustacea	25	0.2	4	2.8
		Hemiptera	43	0.4	8	5.6
		Coleoptera	290	2.9	12	8.5
		Plecoptera	0	0.0	0	0.0
		Trichoptera	1804	17.8	16	11.3
		Ephemeroptera	766	7.6	10	7.0
		Odonata	1325	13.1	8	5.6
		Diptera	5152	50.9	59	41.5
		other	129	1.3	7	4.9
		totals	10127	100	142	100

Table 8. Indicative distributions of absent (or rare) and common taxonomic orders and genera; percentages (out of 10,127 individuals) for 8 common taxa in parenthesis; without stormwater control ponds (NP), reference (R).

Distribution Type	Order	Genera (%)
Common at all sites, abundant at NP high-density sites	Diptera	*Polypedilum* (14)
Common at all sites, except high-density sites	Diptera Diptera	*Rheotanytarsus* (16) *Simulium* (6)
Common at many sites	Ephemeroptera Odonata Ephemeroptera	*Fallceon* (5) *Argia* (12) *Cheumatopsyche* (10)
Common at R, low-density, and med-low-density sites	Trichoptera	*Chimarra* (4)
Absent or rare at R and low density sites	Platyhelminthes	*Dugesia* (3)
Absent or rare at most sites	Oligochaeta, Hirudinea, Plecoptera, Gastropoda Pelecepoda, Crustacea, Hemiptera,	

Figure 1. Locations of non-urban reference sites (REF), and sites with (WP) and without (NP) stormwater controls by urban density class low (< 13 %IC), medium-low (14-23 %IC), medium-high (24-40 %IC), and high (> 40 %IC).

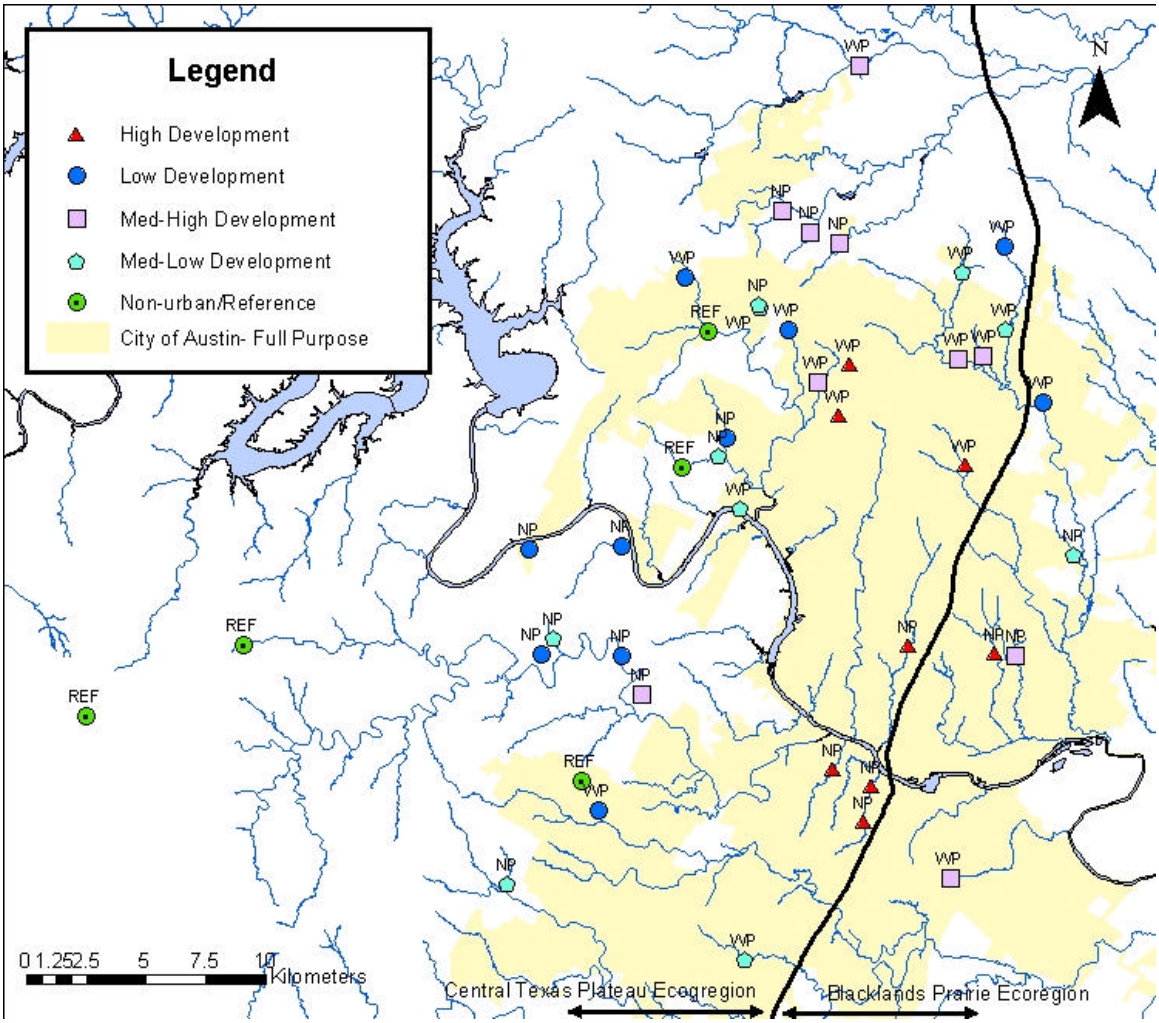

Figure 2. Comparison of the macroinvertebrate communities between streams with (WP) and without (NP) stormwater controls ponds plotted against % impervious area of the contributing catchment using 2 richness metrics; total richness (A), and Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness (B). Non-urban reference sites shown for comparison.

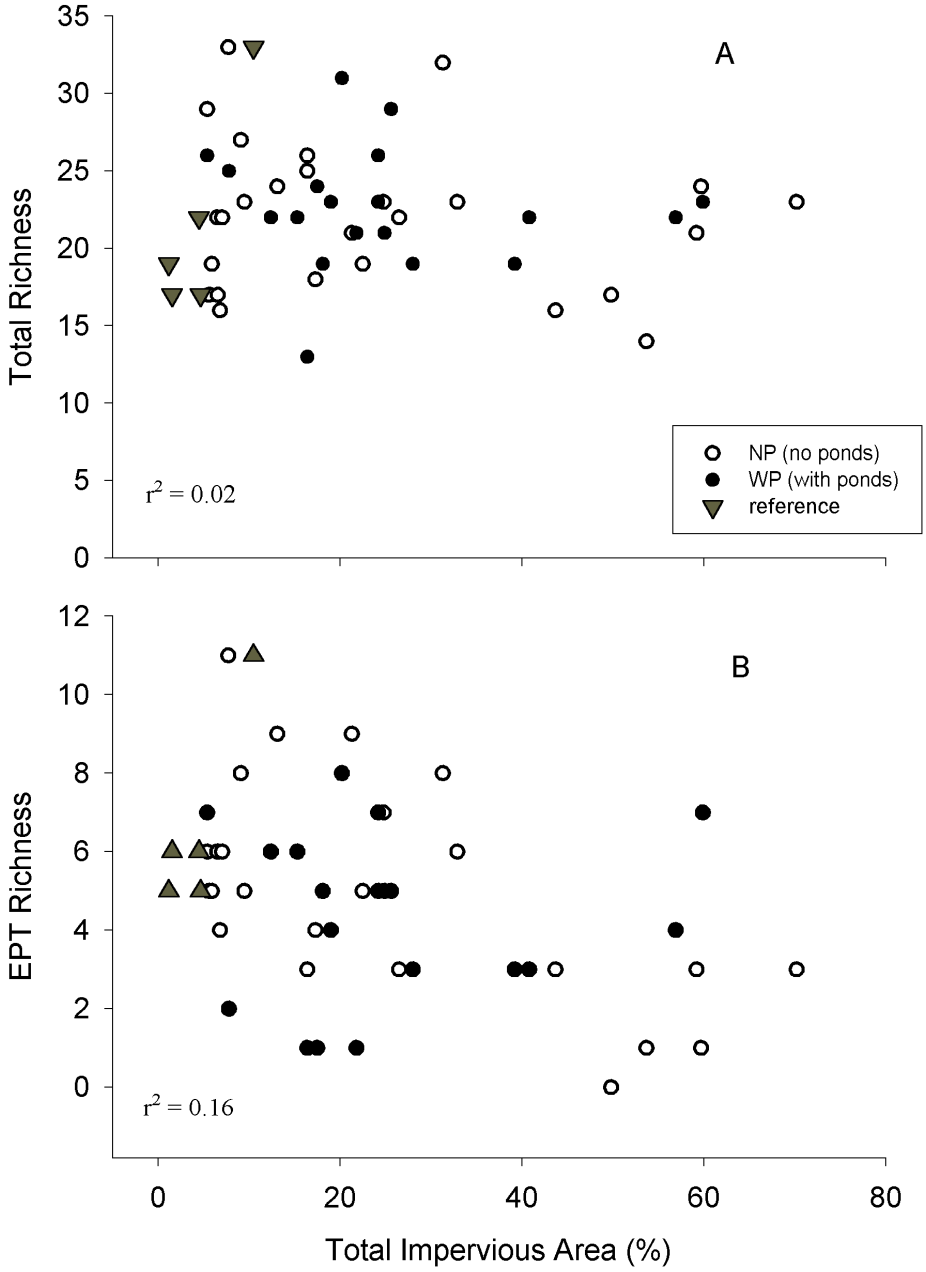

Figure 3. Comparison of the macroinvertebrate communities between streams with (WP) and without (NP) stormwater controls ponds plotted against % impervious area of the contributing catchment using 2 composition metrics, % Ephemeroptera, Plecoptera, and Trichoptera (% EPT) abundance (A), and % dominant taxon (B). Non-urban reference sites shown for comparison.

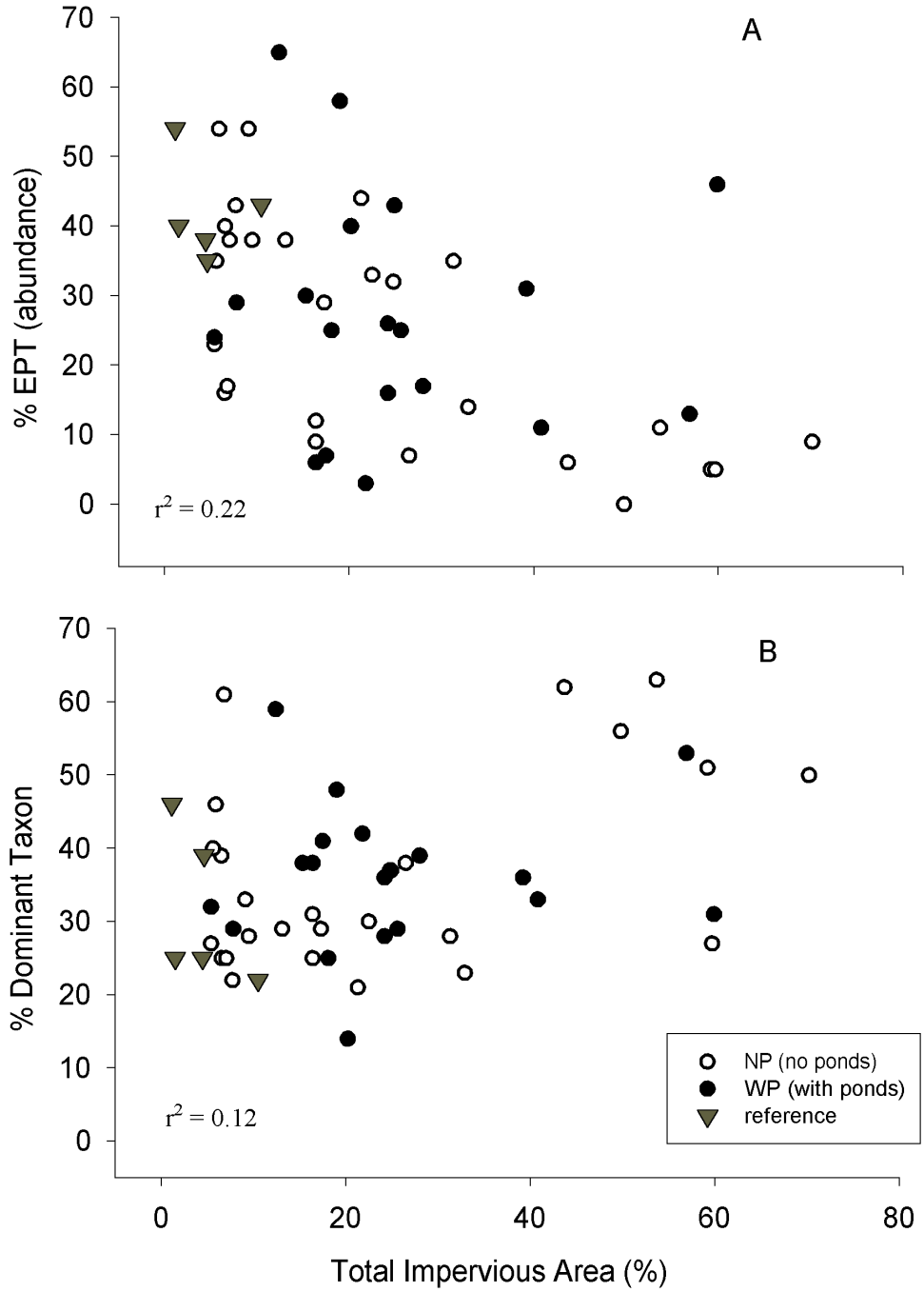

Figure 4. Comparison of the macroinvertebrate communities between streams with (WP) and without (NP) stormwater controls ponds plotted against % impervious area of the contributing catchment using the Hilsenhoff Biotic Index. Non-urban reference sites shown for comparison.

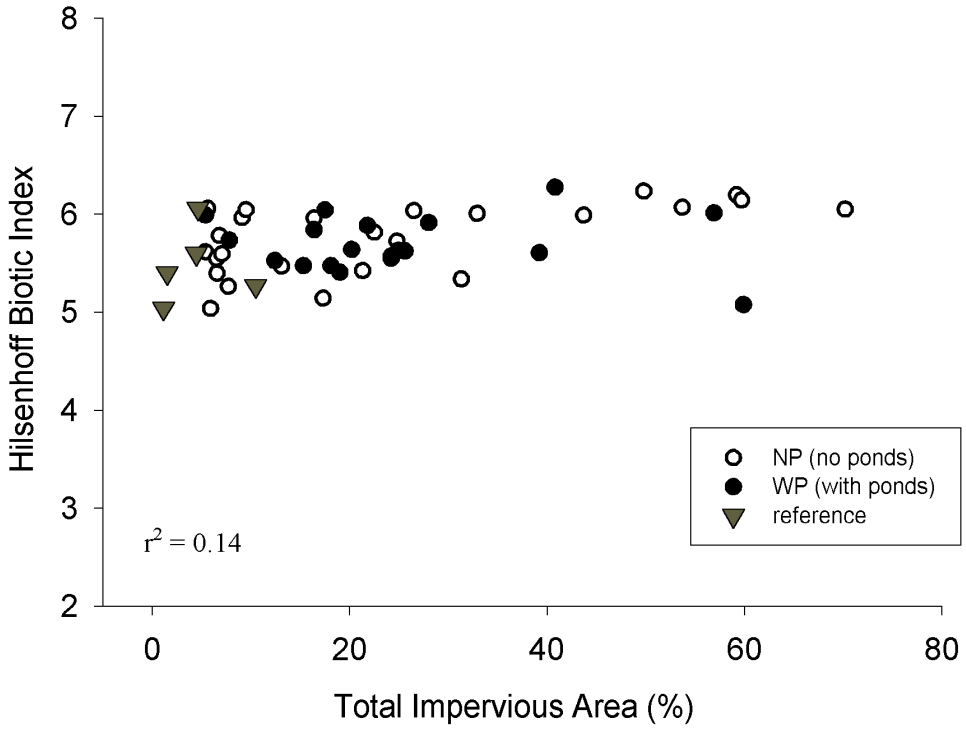

Figure 5. Multidimensional scaling of relative abundance data for streams with (WP) and without (NP) stormwater control ponds by urban density class low (<13 %IC), medium-low (14-23 %IC), medium-high (24-40 %IC), and high (> 40 %IC). Non-urban reference sites shown for comparison.

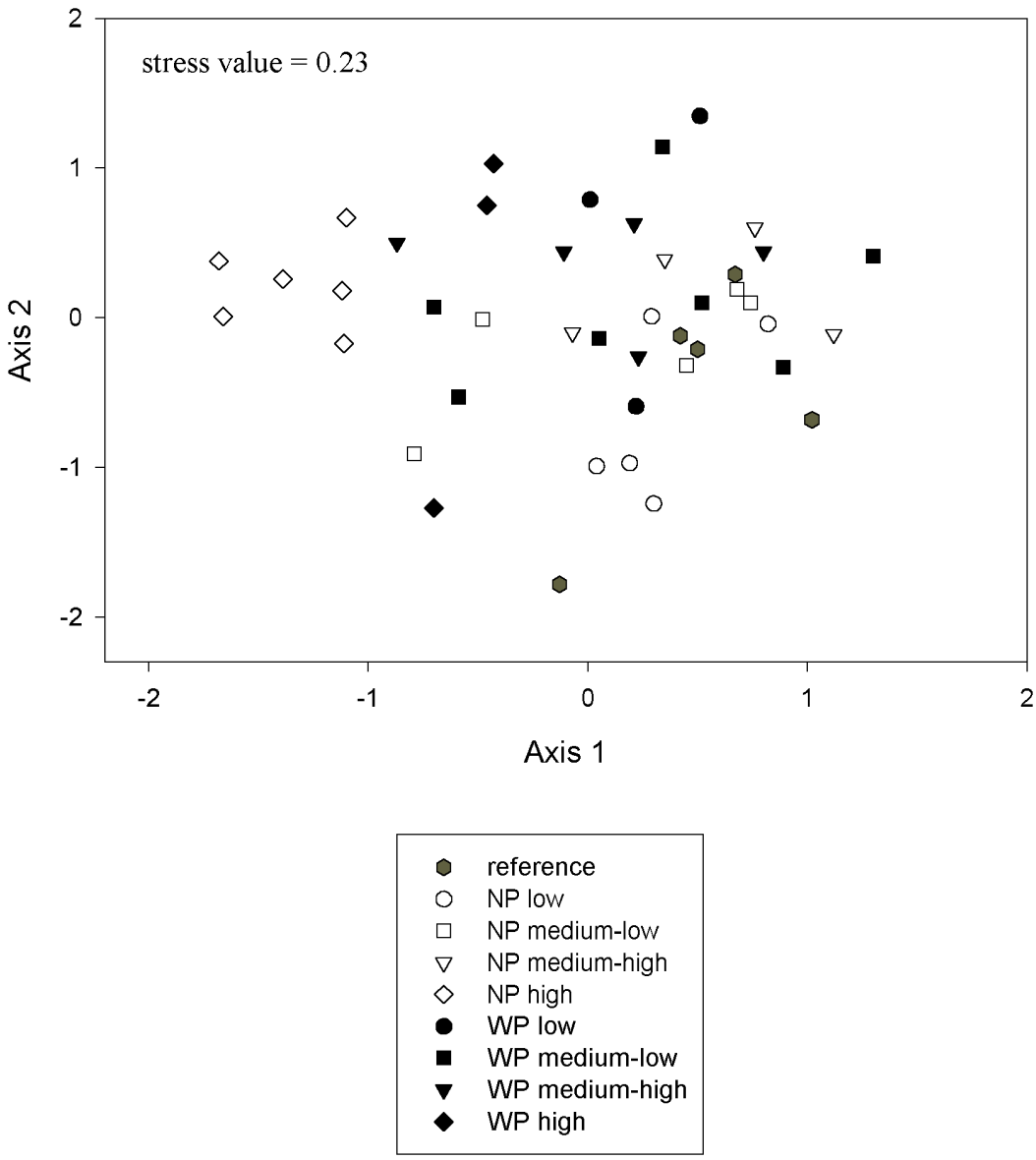
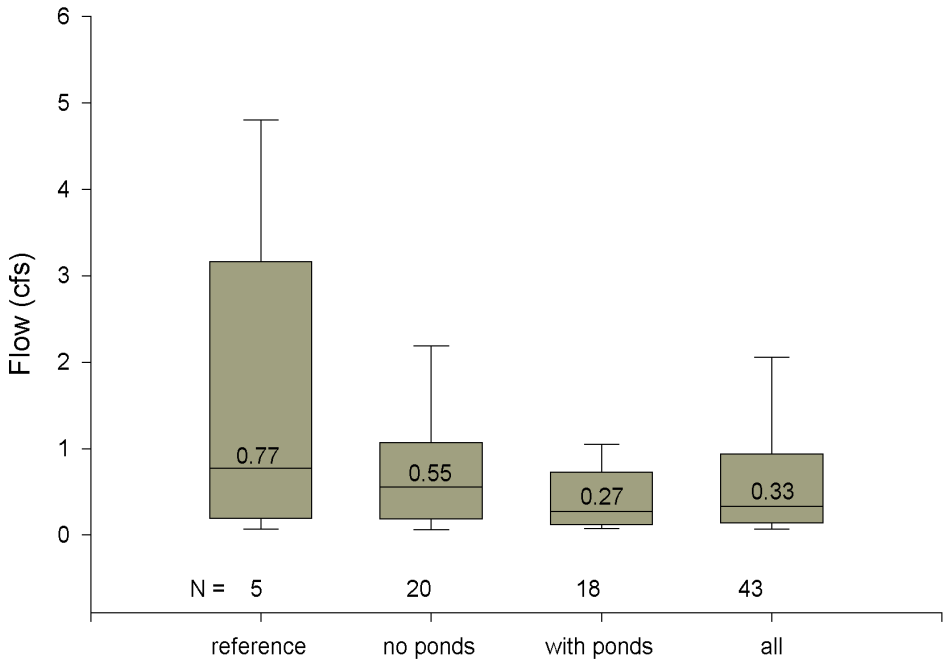
Figure 6. Box (median, 25th, 75th percentiles) and wisker (5th and 95 percentiles) plot distribution of flow (cfs) data for all 43 stream sites, stream sites with (WP) and without (NP) stormwater control ponds, and non-urban reference sites; median values presented.