EFFECTS OF HYDROLOGIC VARIABILITY ON MACROINVERTEBRATE-BASED BIOLOGICAL ASSESSMENTS OF STREAMS IN AUSTIN, TX

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ABSTRACT

EFFECTS OF HYDROLOGIC VARIABILITY ON MACROINVERTEBRATE-BASED

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The City of Austin, Texas, has been using Rapid Biological Assessments (RBA's) in urban and sub-urban streams for the last seven years in an effort to expand and improve its environmental monitoring programs. Central Texas weather is characterized by flashy spates, long dry periods and is distinct from the temperate climates where the RBA's were developed. In addition to these naturally dramatic hydrological cycles, urbanization and its high levels of impervious cover further exaggerate stream flow patterns, producing greater runoff volumes, higher peak flows and less baseflow. Eleven hydrologic statistics were calculated using historical USGS flow data for 11 study streams and compared to available City of Austin benthic macroinvertebrate data using analysis of variance and multiple regression. A field study was also conducted to evaluate the effects of antecedent hydrologic conditions on biological assessments of three streams of differing development condition (high, medium, low) during a 6-month spring flow season. Results show that both long-term hydrologic character of streams in this area as well as immediately antecedent hydrologic conditions have a significant affect on the results of RBA's, which is compounded in urbanized streams. Hydrologic variability should be utilized as a template in interpreting biological assessments using RBA metrics.

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INTRODUCTION:

Rapid Biological Assessments (RBA's, Barbour et al. 1998, Plafkin et al. 1989) are used across the United States to evaluate stream health, particularly as a means to assess the effects of non point source pollution that may not be apparent in traditional water chemistry analysis (Karr and Chu 1999, Barbour et al. 1998, Merrit and Cummins 1996, Resh and McElravy 1993, Plafkin et al. 1989). Benthic macroinvertebrates, the community used most often in RBA's, provide a sensitive measure of cumulative or low-level chronic contamination and also may reflect the physical or structural degradation of aquatic habitats which can occur in urbanized watersheds (Karr and Chu 1999, Barbour et al. 1998, Rosenberg and Resh 1996, Plafkin et al. 1989, Hynes 1970). Benthic macroinvertebrate data are transformed into metrics and compared to reference conditions to establish a qualitative scoring gradient (Karr and Chu 1999, Hughes 1994, Barbour et al. 1994) that reflects the main aspects of community structure (taxonomic richness, composition, tolerance). Assessments using these metrics and indices are often used as water quality management tools and recently as regulatory criteria (most explicit in the states of Maine, Florida and Ohio) in water quality monitoring programs throughout the U.S (Davis et al. 1996, Southerland and Stribling 1995).

Although RBA's are intended to assess the effects of point and non-point source pollution, they have not generally been used to distinguish between the effects of disturbed and undisturbed hydrologic regimes. Instead, every effort is made during the RBA process to minimize variability in conditions outside of the changes caused by pollution sources. For example, habitat quality assessment has become an integral part of

RBA's (Barbour and Stribling 1993), normalizing the physical variability attributed to habitat, such as substrate size, riffle development, habitat heterogeneity, and embeddedness. Seasonal and ecoregion variation are also important considerations in most biological sampling programs. Stream hydrology, although recognized by ecologists as integral in defining ecosystem structure and function (Clausen and Biggs 1997, Gordon 1992, Poff and Ward 1989, Resh et al. 1988, Hynes 1970) is only superficially considered in the interpretation of RBA scores. However, the amount of variability introduced by the hydrologic regime and the unique preceding hydrologic conditions may be significant, especially in urbanized streams where impervious cover has greatly altered natural flow patterns.

The biological response to hydrologic disturbance has been well documented in studies of relatively pristine systems (Angradi 1997, Clausen and Biggs 1997, Dole-Olivier et al. 1997, Death and Winterbourn 1995, Flecker and Feifarek 1994, Quinn and Hickey 1994, Poff and Ward 1989, Scrimgeour and Winterbourn 1989,), as has the biological, physical and chemical response of streams to urbanization (Elliot et al. 1997, Poff et al. 1997, USEPA 1997, Lenat and Crawford 1994, Tikkanen et al. 1994, Britton et al. 1993, Gordon et al. 1992, Pratt et al. 1981, Klein 1979, Baker 1977). However, the effects of hydrologic variability and antecedent hydrologic conditions on biological assessments and their interpretation in monitoring programs are not well known.

Urban streams in Austin, Texas present the typical hydrological problems encountered in densely developed areas all over the world. Impervious cover reduces baseflow by limiting the amount of infiltration in a watershed. Flow volumes and velocities in streams generally increase during storms due to the higher quantity of water that runs off impervious cover and into the stream channel. This creates an unstable system that goes from destructive floods to total de-watering in very short time intervals (Fig. 1a). The resulting biological communities are under constant stress and adjustment. Due to the short duration, high intensity nature of rainfall events in central Texas, hydrologic regimes of streams tend to be more variable and dramatic than in more temperate regions (Baker 1977), where bioassessment protocols were developed (Fig. 1b, 1c). Consequently, understanding the effects of hydrology on the biological communities of streams in this region is critical to the interpretation of bioassessment data.

The City of Austin, Texas has been using RBA's in area streams for the last seven years in an effort to expand and improve its environmental monitoring programs. Biological surveys of streams are completed every three years in most of the watersheds within the City's jurisdiction. However, documentation of hydrologic character is not a common practice for any municipal, state or national agency that conducts RBA's because long-term hydrology data is generally available only at select USGS gaging stations and is rarely collected or reviewed prior to an RBA analysis. Here, I hypothesize that general hydrologic stream character and antecedent hydrologic conditions will have a significant effect on the benthic macroinvertebrate community (as expressed by univariate metrics). Specifically, a significant amount of variability in metric scores at bioassessment sites in the Austin area will be explained by general and antecedent hydrological variables. If so, the results of bioassessments may not necessarily reflect degradation due to non-point source pollution.

I will examine this hypothesis in two ways. First, I will review and analyze historical City of Austin benthic macroinvertebrate data collected in area streams from 1993 – 1999 in relation to the associated hydrologic condition of these streams over the past 10-20 years (general hydrologic conditions). Second, I will conduct a field comparative study of spatial and temporal variation of biological indicators and antecedent hydrological conditions at three sites in the Austin, Texas area, during the spring flow season of 2000. This represents a "natural trajectory experiment" in which flow variation provides the natural perturbations on three streams with varying levels of watershed development (Dole-Olivier et al. 1997, Diamond 1986). Both of these analyses have the goal of documenting the relationship between Austin area hydrology and its role in understanding the results of benthic macroinvertebrate bioassessments.

METHODS:

A. City of Austin historical data analysis of general hydrologic condition:

Study sites

The City of Austin monitors benthic macroinvertebrate communities in 45 watersheds in the Austin area, which occur in two ecoregions, the Central Texas Plateau to the west and the Texas Blackland Prairies to the east (Omernick 1987). Although there are a few watersheds where characteristics of the Blackland Prairie ecoregion predominate (low gradient, silt/soil sediments, low water clarity and higher nutrient concentrations) most of the streams the City of Austin monitors are in the Central Texas Plateau and are characterized by bedrock substrate, moderate gradients (1-2%), clear water and low nutrient concentrations (COA 1996). Riffles are relatively common and are dominated by cobble and gravel, based on the modified Wentworth scale (Cummins 1962).

The data for this study was limited to only those watersheds which had USGS gaging stations with a daily mean flow record and only those sites which were hydrologically associated with the mainstem of the stream being gauged. These restrictions resulted in a total biological data set consisting of 333 data points (unique date/site combination) at 78 sites in 11 watersheds over a period of 7 years with a total macroinvertebrate taxa count of 242. Density of organisms at study sites varied from 75 per square meter to over 13,000, with from 2 to 50 taxa per sample and with the sample median containing 223 organisms and 17 taxa. Undeveloped sites tend to be dominated by Ephemeroptera, Trichoptera and/or Coleopterans while developed sites generally have more Diptera. Plecoptera are relatively rare, although under certain conditions they can be prevalent at undeveloped and moderately developed locations.

Bioassessment

Standard rapid bioassessment methods, Level III (EPA, 1989) were followed in the collection and processing of City of Austin benthic macroinvertebrate samples with the following exceptions:

- All organisms were sorted and preserved in the field, as opposed to the laboratory.

- All samples were sorted in their entirety, as opposed to sub-sampling to a fixed-count.
- Surber samplers (0.1 m^2, 500um mesh net) were used instead of 1m^2 kick nets.
- 200 organisms were used as the target sample size instead of 100.

Generally three randomly selected replicates were collected from each riffle. Organisms from each sample were sorted, enumerated and identified to the lowest possible taxonomic unit possible, usually genus, using the keys of Merritt and Cummins (1996), Wiggins (1996), Epler (1996), Thorp and Covich (1991), Pennak (1989) and Berner & Pescador (1988) and by City of Austin taxonomists. The following groups were not identified to genus: Chironomidae, Ostrocoda, Hydracarina, Hirudinea, and Oligochaeta. Due to naturally low densities of benthic macroinvertebrates in this area it is generally unnecessary to sub-sample down to a fixed count of organisms. Surbers were collected and picked in their entirety until the target number of organisms was achieved (200 + or – 20%).

The metrics used in this analysis (Table 1) were selected *a priori*, based on the traditional categories of benthic macroinvertebrate community structure (taxonomic richness, composition and tolerance; Karr and Chu 1999, Barbour, et al. 1998, Merritt and Cummings 1996, Resh and Jackson 1993, Hynes 1970) and input from local experts. Metric scores for each site were used as the individual dependent variables in this analysis since they are the most direct measurement of community variation used in bioassessment techniques. Total Taxa is simply the number of taxa that are present in a sample and is a measurement of taxonomic richness. EPT Taxa is the number of taxa in each of the orders, Ephemeroptera, Plecoptera and Trichoptera, which are all generally sensitive to pollution while number of Dipteran Taxa quantifies the richness of the dipteran order, which is generally a tolerant group. The Percent Dominance metric is the percent of the entire sample that is made up by the single most numerous taxa. The higher the percent dominance the more degraded a site may be. Percent EPT measures the percent of the whole sample that is made up by the three most sensitive orders, Ephemeroptera, Plecoptera and Trichoptera. The higher the percent EPT the less degraded a site should be. The Percent Chironomidae measures the percent of the whole sample that is made up of the Family Chironomidae, which is generally a tolerant group. The EPT to EPT plus Chironomidae ratio is the total number of EPT organisms divided by the EPT organisms plus the total number of Chironomidae organisms, which is a regional adaptation to a traditional metric (EPT/Chironomidae) that deals with naturally low densities of Chironomidae in Central Texas streams (COA 1996). The Hilsenhoff Biotic Index (Hilsenhoff 1987) is a measure of community tolerance which assigns tolerance values to each taxon and then sums them for the entire sample, resulting in an overall site tolerance score between 1 and 10, with 1 being sensitive and 10 being very tolerant. The metric is calculated as follows:

$$HBI = \sum \frac{X_i T_i}{n}$$

where X_i = # of individuals within a taxon
and T_i = Tolerance value of a taxon
and n = Total number of organisms in sample.

Hydrology

Daily mean flow measurements, taken from the 11 USGS gaging stations in the Austin area were converted to ecologically relevant parameters (Table 2) as established by Poff and Ward (1989) and by Richter et al. (1996, 1997). The Indicators of

Hydrological Alteration (IHA) software package (Richter et al. 1996) was used to calculate 33 statistics from the raw USGS daily mean flows. Correlation analysis was then used to determine which variables were highly correlated (correlation coefficient >0.80). The independent variable list was subsequently reduced from 28 to 19. The latter were then divided into two main groups, the summary statistics (6 variables from Poff and Ward 1989) and the main IHA statistics (13 variables from Richter et al. 1996). Analysis of variance was then used to evaluate the ability of the 13 IHA variables to separate Austin-area streams. Considering the wide variation of development conditions and watershed sizes, if a hydrology parameter was the same for all watersheds, it was not measuring variation that would be important for this analysis. The hydrology parameters that were not able to distinguish at least three groups among the 11 USGS gaged watersheds were excluded from the analysis. This reduced the number of IHA variables down to five (# of Zero Flow Days, Number of Low Pulse Counts, Number of High Pulse Counts, Fall-Rate and Number of Reversals) which are combined with Poff's Summary Statistics (6) for a total of 11 hydrology variables. These parameters were used to characterize the flow regimes in the selected watersheds so that their relationship with the biological variables could be evaluated. The hydrology variables were used as multiple independent variables in these analyses.

Analysis

Graphical analysis (Karr and Chu 1999, Statsoft 1995, Microsoft Excel 1993) was used initially to evaluate spatial and temporal relationships for all of the variables in this analysis. "Often the most effective way to describe, explore and summarize a set of numbers it to look at pictures of those numbers..." (Tufte 1983). Line graphs or time-series were used to evaluate temporal data, while box plots (Tukey 1977) were used to compare variance and spatial patterns. Correlation matrices (Gravetter and Wallnau 1996, Statsoft 1995) were used to evaluate intercorrelation among both dependent and independent variables. A variable was excluded from analysis if it was significantly correlated with another variable with a correlation coefficient greater than 0.80 (selection between correlated variables was based on experience and or the variable more commonly used in the literature). Analysis of variance (Gravetter and Wallnau 1996, Kirk 1995, Statsoft 1995) was used to evaluate the efficacy of the hydrological parameters in distinguishing between different development levels, regional effects on hydrology and the resolution of the biological and hydrological measures in dividing Austin streams. ANOVA empirically evaluates mean differences between two or more treatments (Gravetter and Wallnau 1996) and in this analysis the treatments were generally "natural" spatial perturbations (land use, region). Multiple linear regression (forward stepwise, Kirk 1995, Statsoft 1995, Snedecor and Cochran 1989) was used to evaluate the relationship between the biological data (metric scores) and the hydrological characterization of the study streams (summary and IHA parameters). Regression analysis is generally used to explore empirical relationships between variables (SAS 1990), in this case between the biological response variable and the single or multiple regressor, or hydrologic variables. The forward stepwise model was selected based on its ability produce the best regression models with the study variables. Standard regression requires that the independent variables be ordered in the regression based on an *a priori* designation of importance to the dependent variable. In this case, there was no a priori hypothesis as to the relative importance of each hydrologic variable. Forward stepwise regression adds or deletes each independent variable until the "best" regression model is made (SAS 1990). This proved to be the best method for this data set, providing the strongest models for each of the dependent variables.

B. Spring 2000 study/Antecedent hydrologic conditions.

Study Sites

Although precipitation is fairly evenly spread throughout the year in the Austin area, spring (March – June) is the wet season and generally exhibits the highest and longest flow periods and the larger rain events. Many smaller drainages in the Austin area only flow during wet seasons and spring is the most likely period to encounter baseflow conditions in area streams. The intention of this 6 month survey was to document variation in biological condition in three watersheds over a normal Spring flow regime. The Austin area was in a drought from summer 1998 through fall 2000. Spring, 2000 was preceded by 6 months of relatively dry conditions, followed by rain in February which brought surface flow to the three selected study streams for about 6 months, until late July. The watersheds selected varied in their development levels but were relatively similar in their geology, morphology and drainage areas. All three are underlain by cretaceous age limestone (Glen Rose, Edwards or Austin Chalk formations) and have low to moderate stream gradients (0.6 to 0.9%). Selection of sampling sites was based on proximity to USGS gaging stations (<300m downstream) and habitat comparability. The Bear Creek @ 1826 USGS gage site has a drainage area of 31.6 square km, a total impervious area of 2.5%, a stream length of 8.8 km and lies in the Southwest corner of the City of Austin jurisdiction. The Bull Creek @ 360 USGS gage site has a drainage area of 57.5 square km, a total impervious area of 15.6%, a stream length of 19.3 km and drains directly into the Colorado river in the northwest corner of the Austin jurisdiction. The Shoal Creek @ 12th Street USGS gage site has a drainage area of 31.6 square km, a total impervious area of 56%, a stream length of 13.9 km and drains into Town Lake (Colorado River impoundment) in central Austin. All sites had well-developed riffles, dominated by gravel and cobble substrate. Flow variability at each site significantly affected the quantity of viable substrate (wetted area) and physical conditions outside the immediate channel differed for all three streams, depending on development. All three sites are on the eastern edge of the Central Texas Plateau Ecoregion (Omernick 1987).

Bioassessment

Benthic macroinvertebrate sampling was conducted on a monthly basis from February through July, 2000 at one site on each of three study streams in the Austin Area (Fig. 2). Dewatering of these sites due to drought conditions limited the number of sampling events on each creek. Beginning in February, all three sites were visited, but only one (Bull Creek) was sampled due to dry conditions at the other sites. Sites were sampled each month after that through July, when the sites again went dry and the study was terminated. These surveys resulted in 5 visits to Shoal Creek, 6 Visits to Bull Creek and 4 visits to Bear Creek (Appendix 1). The June survey was not completed on Bear Creek due to a sampling error.

Each survey consisted of the following:

- 3 Surber samples collected from the bottom, middle and top of the study riffle. Each surber was sorted in its entirety and preserved in the field in 70% isopropyl alcohol.

- Flow velocity was measured at each location a surber was collected and total discharge was measured (Harrelson et al. 1994) at each site using a Marsh McBirney flow meter.
- Physicochemical parameters (pH, Conductivity, Dissolved Oxygen, Temperature) were taken at each site using a Hydrolab Minisonde 4a.
- Photos were taken of the study riffle and immediate channel to document variation in flow and substrate conditions.
- Following each survey, benthic macroinvertebrate samples were processed and identified to the lowest possible taxonomic level, usually genus, using a 40X dissecting scope and the appropriate dichotomous keys and references (Epler 1996, Merritt and Cummins 1996, Wiggins 1996, Berner & Pescador 1988).

Univariate metric calculations were performed using the SAS statistical software package (SAS 1990). The same metrics selected for the historical analysis were used for this analysis with the exception of number of individuals, which was added (Table 1). As opposed to the historical data analysis, which incorporated data from various studies utilizing slightly variable field and sorting methodologies, the data from the antecedent study was carried out by the same field staff and used the same quantitative methods throughout, allowing for comparison of densities between sites and dates.

Hydrology

Hydrological data analysis was carried out using hourly mean discharge values from the three USGS gaging stations associated with each sampling site. The parameters selected to characterize the hydrology for the 30 days previous to the sampling event were adapted from Poff and Ward (1989) and Richter et al. (1997,1996) to fit the different time scale of this analysis (30 days vs. 10+ years and hourly vs. daily mean discharge values) (Table 3). Correlation analysis was used to remove redundant variables (correlation coefficient >0.80) and a final list of 11 independent variables was selected for use in the analysis.

Graphical analysis and ANOVA (Statsoft, 1995) were used to compare variation in both hydrological and biological variables among the three sites (spatially and temporally). Multiple regression analysis (Statsoft, 1995) was used to evaluate the relationship between biological variables and hydrological variables. Metric scores from each site, over the 6-month period were regressed against the hydrological variables to evaluate the effects of antecedent hydrology on macroinvertebrate community structure as indicated by the selected metrics. In addition to the above analysis, regression was also used to evaluate the relationship of biological variables to physicochemical data.

RESULTS AND DISCUSSION:

A. City of Austin historical data analysis:

Evaluation of the hydrology variables

Three streams were compared which had similar geological settings and similar drainage areas, but differing levels of development (See previous discussion of Shoal, Bull and Bear creeks). The 33 IHA hydrology parameters were calculated for each of these streams for each year that USGS gage data was available and compared using box plots (Statsoft, 1995). Many of these plots showed distinct patterns relating hydrologic variability of streams to degree of impervious cover in their watersheds. For example, the

Number of High Pulse Counts in any given flow year (discharge values above the 75th percentile of the entire period of record) was lower in the undeveloped stream than the moderately developed stream and both of these were much lower than the Number of High Pulse Counts in the highly developed stream (Fig. 3). Similarly, the Rise Rate in these three streams was much faster in the more developed watersheds (Fig. 4), which corresponds to the expectation for higher peak flows and faster velocities in urban streams (Finkenbine et al. 2000, USEPA 1997, Klein 1979). The Number of Zero Flow Days statistic is a measure of how often a stream goes completely dry at the gage station and should be positively correlated to impervious cover (Barrett et al 1998, USEPA 1997, Klein 1979). As percent impervious cover increases, infiltration into groundwater decreases and the number of zero flow days in a watershed should go up. Although there were more zero flow days in the developed watershed (Shoal), the moderately developed watershed (Bull), had fewer zero flow days than the undeveloped system (Fig. 5). This could be due to the larger drainage areas in the moderately (57.5 sq. km.) than the undeveloped stream (31.6 sq. km.) or due to the geology in the Bull Creek watershed, which exhibits an almost perennial flow regime (Gandara et al. 1997), probably due to strong groundwater influences.

For all 33 Hydrology parameters evaluated using the three different watershed development conditions, 19 showed a significant difference (ANOVA, $P<0.05$) among the three streams. Since the primary difference among these three streams is development level, the 19 hydrologic statistics should reflect variation due to impervious cover. This finding agrees with many authors who have documented the general hydrologic effects of impervious cover on stream systems (Finkenbine et al. 2000, USEPA 1997, Richter et al. 1996, Booth 1990, Klein 1979). However, the ability of these parameters to quantify specific physical responses to development is important if the relationship of hydrology to biological communities is to be determined. The fact that the hydrological characterization tools in this analysis separated streams based on level of development indicates that even with the naturally high level of hydrologic variability in this area, impervious cover still significantly alters flow regimes.

The IHA parameters were also used to compare differences between hydrologic variability in Austin and the temperate Southeast (Frederick County, Virginia). Watersheds of similar drainage area (31.6 and 38.9 sq. km.), similar development level (<5% impervious cover) and the same period of record (1979 - 1998) were compared based on each IHA parameter and 26 out of the 33 were significantly different ($P<0.05$) between the two regions. This underscores the distinct differences between hydrological regimes in the southeast, where bioassessment techniques were primarily developed, and our semi-arid region of the southwest. This conclusion is not surprising, considering the large differences in climate, but it does demonstrate how these differences may influence the structure and function of benthic macroinvertebrate communities in the Austin area. Many researchers have documented the need for regionally specific bioassessment techniques (Feminella 2000, Davies et al. 1999, Hornig et al. 1999, Spindler 1996, Allen 1995), but hydrologic variability is generally not considered a primary environmental variable in the development of these methods, or in the interpretation of results. Central Texas "probably has the most intense stream flooding regimen in the conterminous US"

(Baker 1977) and has significantly different hydrologic regimes than the Virginia example. It is logical then, that hydrology could play a different and possibly more important role in structuring benthic community structure in area streams than in the temperate climate where bioassessment techniques were developed.

Biology vs. Hydrology

The hydrological data set used in the regression analysis combined the six summary statistics proposed by Poff and Ward (1989) and the five IHA variables (Richter et al. 1996) that were selected based on their ability to distinguish among Austin area watersheds (Table 4). This group was regressed against a single biological metric data set to evaluate the relationship between the biological indicator and the characterization of hydrological variation for each watershed. Analysis of these two data sets showed a significant relationship ($p<0.05$) between all 8 biological metrics and some combination of the eleven hydrologic variables, with coefficients of determination (R^2) ranging from 0.12 to 0.46 (Table 5). This is a significant portion of the variability in the biological data. Clausen and Biggs (1997) in a similar study on New Zealand streams, found that their model of hydrologic variability explained 14 to 55% of the variability in their 7 benthic community measures. Although these studies had diverse data sets and distinct analysis approaches, these similar results support the hypothesis that some measure of hydrologic variability is important in structuring benthic macroinvertebrate communities in streams in Austin area watersheds.

Although all 8 metrics were significantly related to the hydrology variables, <u>EPT/Chironomidae</u>, <u>Percent Chironomidae</u>, <u>EPT Taxa</u> and the <u>HBI</u> had the most variation in their scores explained by the hydrology variables, with R^2's above 0.30 (Table 5). The two metrics with the highest R^2 were significantly correlated (EPT/Chironomidae and % Chironomidae, r = -0.78), but clearly Chironomidae are responding to the hydrology variables. Several studies have documented the ability of many dipterans, and particularly Chironomidae to quickly colonize disturbed habitats and to dominate in relatively harsh or degraded environments (Miller and Golladay 1996, Johnson et al. 1993, Jowett and Duncan 1990, Iversen et al. 1978, Hynes 1960). Similarly, in the stochastic and "flashy" flow events of the Central Texas area, and particularly in degraded urban environments, Chironomidae disturbance resilience would be closely tied to hydrologic variability. One problem with this analysis is that the City of Austin only identifies this group to the family level. Since not all Chironomidae are exclusively opportunistic and tolerant of lotic disturbance (Merritt and Cummins 1993, Hynes 1972) this generalization may result in a loss of discrimination ability in bioassessments. Townsend and Scarsbrook (1997) emphasized in their study of disturbance in streams that any given measure of disturbance may relate differently to even closely related taxa and that understanding species level traits may be necessary in understanding these interactions. It is likely that better taxonomic resolution of the Chironomidae could result in improved detection of human induced degradation, both physical and chemical.

EPT Taxa and Taxa Richness were also auto-correlated (r = 0.78) but the former was more closely related to the hydrology variables (R^2 of 0.37 vs. 0.29). Several studies on flow variability found that overall Taxa Richness was related to disturbance, showing both positive and negative relationships (Clausen and Biggs 1997, Death and Winterbourn 1995, Jowett and Duncan 1990). Feminella (1996) in contrast, found that EPT taxa showed the strongest relationship to flow permanence and that it was always positive in that the more permanent a stream, the higher the EPT taxa richness. The EPT organisms are generally considered the most sensitive to chemical degradation (Karr and Chu 1999, Barbour et al. 1998, Merritt and Cummins 1996, Johnson et al. 1993), but their response to physical/hydrological degradation is less documented. Apparently, in addition to the permanence gradient that Feminella studied, the EPT group also negatively responds to the measures of flow variability examined in this study. Generally, as the hydrologic variables increased in magnitude, the number of EPT taxa decreased. Although more detailed analysis would probably show that some taxa within the EPT are more resistant to hydrologic disturbance, this group appears to be sensitive to hydrologic variability and absence of, or a reduction in, EPT organisms could be an early indicator of hydrologic alteration. However, separating this physical effect from chemical effects due to non-point pollution on RBA's will be problematic.

Each metric had a distinct "best" regression model (including from 6 to 9 hydrologic variables). Based on the beta scores[1], <u>% of Floods in 60 day period</u>, <u>Annual Coefficient of Variation</u>, and <u>Constancy Predictability</u> were the best independent variables as they were present in more models than the other hydrologic variables (8, 7,

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[1] Beta scores are regression coefficients that can be used to compare the relative contribution of each independent variable in the prediction of the dependent variable.

and 6, respectively) and generally had higher beta values in each model (Table 5). Although there was some similarity, these results do not completely match findings of the effects of hydrology on benthic biota in studies in other parts of the world. Clausen and Biggs (1997) found that Flood Frequency (frequency of flows over 3 times the median flow) was the best predictor of benthic community structure. Although the Flood Frequency and % of Floods in 60 day period hydrology parameters are both measures of flooding disturbance, the former looks at frequency of disturbance events, while the latter and Constancy Predictability look at the predictability or "seasonality" of disturbance events. It is likely that due to the naturally high intensity and high frequency of spates in this area, the species that have evolved under these conditions have developed survival mechanisms such as flexible life cycle, high rates of migration and dispersal and diapausing eggs to deal with frequent disturbance events (Williams 1996, Delucchi 1987). Several studies of benthic macroinvertebrate communities in areas of naturally high disturbance show they recover quickly from disturbance events due to evolutionary adaptations (Dole-Olivier et al 1997, Miller and Golladay 1996, Poff and Ward 1989, Scrimgeour and Winterbourn 1989), but that time since last disturbance is more important than frequency of disturbances (Death and Winterbourn 1995, Lake et al. 1989). Since the measure of flood predictability (% of floods in 60 day period) was a better predictor of metric scores than the measure of flood frequency (high pulse count) according to the regression models in this analysis, it would appear that the timing and /or lack of seasonal predictability of spates in the Austin area are more important to benthic community structure than frequency of events.

Annual Coefficient of Variation (ACV) was the best overall flow variable in predicting benthic macroinvertebrate community metrics based on higher beta values and rank compared to other variables in each model (Table 5). Contrary to intuition, the benthic macroinvertebrate metrics in this analysis responded positively to increasing ACV. As ACV went up, the metrics tended show improvement. Poff and Ward (1989) in their regional analysis of streamflow patterns predicted that in streams subjected to both intermittency and frequent flooding, life history adaptations to this disturbance regime will result in high community persistence, with organisms responding quickly and opportunistically to these disturbance events. However, the most common response to natural disturbance in the literature has been recovery to "equilibrium" community structure, by way of adaptive mechanisms and behavior (Miller and Golladay 1996, Scrimgeour and Winterbourn 1989, Delucchi 1988) and not necessarily an increase in richness or diversity with increasing environmental variation. This phenomenon requires further evaluation, and may fall under the intermediate disturbance hypothesis (Ward and Stanford 1983, Connell 1978), where diversity is greatest at intermediate levels of disturbance. Perhaps hydrologic variation in Austin area streams provides intermediate levels of disturbance, thereby improving metric scores. Streams with very little surface flow would have lower variation, as would streams with perennial, ground water influenced flows. The "intermediate" or average stream, with the normally wide range of flow variation would have the most resistant and resilient communities with the highest diversity. It is important to note that when impervious cover at the selected gaging stations was plotted against ACV, there was a significant negative relationship (R^2 = 0.51). As impervious cover went up, ACV went down (Fig. 6), showing that developed streams, probably due to a loss of baseflow, actually have lower variation than undeveloped streams.

B. Spring 2000 study/Antecedent hydrologic conditions.

Evaluation of the hydrology variables

The effectiveness of the selected hydrological parameters (Table 5) at distinguishing among the development levels at the three study streams (Bull, Bear and Shoal) used in this 6 month study was also tested using graphical plots and one-way ANOVA (Statsoft, 1995). Of the 11 hydrology parameters selected, 5 showed significant differences ($p < 0.05$) between at least two of the three study sites (Table 6). In most cases, and as will be noted in later analysis, Shoal Creek (highly developed) was significantly different than one or both of the other study streams (Bull and Bear, moderately and minimally developed, respectively), but Bull and Bear creeks were not different from each other. For example, the Coefficient of Variation, Discharge of High Pulse and the Rate of Rise for Shoal Creek were much higher than for Bull or Bear creeks, which were very similar to each other (Fig. 7). This shows the effects of impervious cover on antecedent watershed hydrology and confirms the predictions of Finkenbine et al. (2000), Horner et al. (1997), and Klein (1979) that increasing urbanization will have an effect on the physical stream environment.

The difference in impervious cover between Bull and Bear creeks (15.6 and 2.5 %, respectively) did not result in significant differences between the hydrology of the two streams. It is likely that the size differences between the two drainages (57.5 sq. km. for

Bull and 31.6 sq. km for Bear) along with other un-quantified hydrogeologic variation between the two watersheds is obscuring hydrologic differences as measured by the variables in this study. Alternately, a “threshold” phenomenon (Schueler and Claytor 1997, Schueler 1994, Klein 1979) proposes that the negative effects of impervious cover are difficult to detect and less dramatic up to a certain point (10%), but after that point, sensitive stream elements are lost and the physical environment is more quickly degraded. It is possible that Bull Creek is near this threshold, and its response to impervious cover is still small enough that the measures in this study were not yet able to detect it. Further studies on the Bull Creek watershed are warranted to determine changes in hydrology and benthic communities as development increases.

Evaluation of the biological variables

Graphical and analysis of variance were again used to evaluate the temporal and spatial variation of the biological variables (univariate metrics) at the three sites during the study period. There was temporal variation between monthly biological surveys at all three sites, although the least developed site (Bear) and the most developed site (Shoal) both had more variation (as measured by the coefficient of variation) than Bull Creek, which had significantly less variation in the biological metrics than either of the other two study sites (Fig. 8). This could be a product of drainage area differences among these watersheds, as discussed previously. In smaller-scale (1-4th order streams), within the range of those used in this analysis, three studies have found a consistent positive relationships between drainage area and stability of the flow regime (Death and Winterbourn 1995, Poff and Ward 1989, Resh et al. 1988) which should translate into more temporally variable benthic macroinvertebrate communities in streams with smaller drainage areas. Since Bull has a larger drainage area (57.5 sq km) than Bear and Shoal (31.6 sq km), this could explain lower variation in metric scores for Bull Creek. In addition to the differences in drainage area, Bull Creek has much more perennial baseflow than other streams of similar size, probably due to a strong groundwater influence in this watershed (see previous discussion, Gandera et al. 1995), which also would contribute to more temporally stable benthic communities.

In several of the metrics evaluated, there was a distinctive increase in scores as the study progressed, indicating that the benthic macroinvertebrate community was changing during the entire study period. This was particularly true for the Number of EPT Taxa metric, which went from 2 to 8 on Bear Creek, from 3 to 7 on Bull Creek and from 1 to 3 on Shoal Creek (Fig. 9). Due to the monthly increases in these metrics, it is likely that all of these streams were in adjustment during the entire study period and didn't necessarily reach a persistent, "equilibrium" state, as has been proposed by many papers on macroinvertebrate structure in stream systems (Lake 2000, Death and Winterbourn 1994, Townsend et al. 1987, Minshall and Peterson 1985). The drought period leading up to this study could be considered a reset mechanism (Resh et al. 1988), in which all of the communities in these streams had to recover from a severe disturbance. Lake, (2000) found that response of benthic communities to flooding is much better understood than the response to drought, and that drought probably requires different levels of resilience and resistance than flooding in order to return to an equilibrium state. Although many studies have found the recovery period following disturbance in stream ecosystems to be much shorter than the 6 month length of this study (Angradi 1997, Miller and Golladay

1996, Iverson et al. 1978), it is possible that it was not long enough to observe a leveling off of EPT taxa richness. This has implications to the usefulness of bioassessment techniques in this area. If it takes more than 6 months after a drought to reach a relatively stable, "equilibrium" community, any assessment done in the interim period will reflect transitional communities. However, due to the comparative nature of most bioassessment programs, in which a reference, or control condition is established during the same time period, exposed theoretically to the same antecedent hydrological conditions, this problem may be minimized. By comparing transitional control streams to transitional test streams, even though the communities may not be in equilibrium, investigators should still be able to detect degradation if the sites maintain different benthic communities, as was demonstrated during this study. However, the resolution of these methods would be expected to be lower, considering all the variability introduced by using transitional communities. These results suggest that reference conditions sampled at the same time as your test surveys should be an integral part of bioassessment in this area, particularly following prolonged drought conditions.

Four out of the 9 metrics used showed significant among site differences (Table 7). In two of the metrics (Taxa Richness, EPT Taxa) a similar site-separation pattern to previous results was observed, in which the densely developed site (Shoal) had significantly different metric scores than the moderate (Bull) and undeveloped (Bear) site, but that the moderate and undeveloped site were not different (Fig. 10). In the other two metrics (Number of organisms, HBI) Shoal was different than one of the other streams, but Bull and Bear were never different (Fig. 11). The uncontrolled variables in this study (groundwater influence, drainage area) could result in too much background noise to be able to detect differences in benthic communities at these two sites (Bear and Bull), the metrics themselves may not be sensitive enough to detect differences, or there may not be any differences. It is possible that the hydrological template and high level of physical disturbance for streams in this area reduces the resolution of bioassessment techniques, such that the relatively large difference in development level between these two sites (2.5 and 15.6% impervious cover for Bear and Bull, respectively) was not detectable. This hypothesis was supported by previous work done by City of Austin staff (1996) documenting the difficulty of detecting non-point source pollution in Austin streams using bioassessment techniques.

Physico-chemical variables (Conductivity, Dissolved Oxygen, pH, Temperature, Velocity and Discharge) were measured during each survey and associated with the biological data set for that sample. Multiple regression analysis was performed using the array of physico-chemical variables vs. each biological variable (metric). Discharge and pH were found to be inter-correlated with other variables and were excluded from the regression analysis. The physico-chemical group (Conductivity, Dissolved Oxygen, Temperature and Velocity) was significantly related to each of the biological variables except Percent Dominance (Table 8), explaining a relatively large percentage of the variation in the metric scores (R^2 values from 0.29 to 0.68). Velocity was the best independent variable, with the highest beta values in 5 out of 9 of the metrics evaluated, while Taxa Richness and EPT Taxa were the metrics most closely related to the physico-chemical variables ($R^2 = 0.68$ for both models).

Variation in Conductivity, Dissolved Oxygen and pH could potentially be attributed to development level in the watershed (Lenat and Crawford 1994, Klein 1979, Hynes 1972). However, velocity, as measured at the riffle location in each stream where the biological data was collected, is not necessarily related to level of urbanization, but to discharge and stream gradient, which are elements of the habitat template. Inherent physical differences among these three sites could describe a significant proportion of the variability among their respective benthic communities, as has been documented in studies of the habitat template theory (Townsend et al. 1997, Parson and Norris 1996, Scarsbrook and Townsend 1993, Minshall and Peterson 1985), where stream community structure is driven by spatial and temporal habitat variables. Although every effort was made to select riffle habitats as similar as possible among these three sites, there is obviously variation that was not accounted for in this study (riffle size, canopy cover, woody debris). It is likely that in addition to velocity, there are other habitat variables that would account for significant variation in these biological communities. Since quantitative habitat evaluation is not practical and not part of most bioassessment programs, this is another confounding factor for biological methods in general and which reduces their ability to distinguish accurately between levels of human degradation.

Biology vs. Hydrology

The selected hydrologic variables (Table 3) were used as independent variables in a forward stepwise multiple regression analysis to evaluate the ability of the hydrology variables to predict the metric scores at the three study streams. Due to the limited data points at each site (4, 5 and 6 for Bear, Shoal and Bull, respectively), all three streams were combined for a total of 15 biological data sets vs. the specific antecedent hydrology variables from each watershed, calculated for the 30 days preceding each sampling event. The results of that analysis showed a significant relationship ($P<0.05$) between some combination of independent variables and 7 of the 9 dependent variables (Table 9). Only Diptera Taxa and Percent Dominance were not significantly related to the hydrology variables. Taxa Richness and EPT taxa were the two dependent variables best predicted by the group of hydrology variables ($R^2 = 0.96$ and 0.92 respectively). This suggests that these variables are most influenced by antecedent hydrologic conditions. Feminella (1996) found that Total Taxa and EPT Taxa were significantly correlated to stream permanence, and that drought and/or stream de-watering had strong effects on benthic communities, even several months after flow resumed. These results, along with the results from this study, support the hypothesis that antecedent hydrological conditions strongly influence certain bioassessment metrics. This is particularly important because both Total Taxa and EPT Taxa, are central to RBA protocols worldwide (Karr & Chu, 1999, Barbour et al. 1998, Rosenberg and Resh 1993). If they are as closely tied to antecedent hydrology as my data indicates, RBA's in this region, and probably other regions as well, need to incorporate hydrological condition, not just non-point source pollution, into their interpretation of the results of these assessments.

Of the independent variables, the Coefficient of Variation and Rate of Rise were the best at describing the biological variables, based on higher beta values (Table 9). However, a combination of five independent variables in this analysis were relatively evenly represented, all having important roles in predicting metric scores (Coefficient of variation, Duration of High Pulse, Number of High Pulses, Rate of Rise and Time Since Max Discharge). There has been little work quantifying antecedent hydrologic conditions and none based on 1-hour gage data, possibly due to the difficulty of obtaining this data or the expense of setting up gaging stations. Although Clausen and Biggs (1997) found measurements of flood frequency to be the most important to benthic communities in their study, it is difficult to compare to this study due to the differences in temporal scale. They used daily average discharge (cubic meters per second) over a 7-year study period. No significant floods occurred during the 6-month period of the Spring 2000 study and the 1-hour data allowed for a much finer grained analysis of flow variability. Whereas predictability and flow variability were important factors in the longer-term historic analysis of City of Austin benthic macroinvertebrate data, the results from the Spring 2000 study showed that frequency of disturbance events, magnitude of events and flow variability are all important at temporal scales based on 1-hour hydrographs. For example, using daily average flows on a typical Austin watershed, a four-hour storm event with 2 inches of rain would produce a single, slightly higher, average flow discharge point for that day, possibly returning to a baseflow condition within 24 hours. Using 1-hour average discharge, the same steam hydrograph would show the rate of increase as the stream rose, the actual peak flow, and the rate at which it came back to baseflow condition. This difference in resolution is more dramatic the smaller a watershed is, generating smaller hydrographs that are reduced in importance over longer averaging periods. Due to the short-duration, high-intensity rain events in Central Texas, longer-term averaging periods are potentially problematic in producing accurate hydrologic statistics, which could explain why the regression results for the Spring 2000 study are so much better than for the long-term historic analysis. Clearly, this finer scale data is better for estimating hydrologic changes as they pertain to bioassessment scores in the streams in this area, as is evidenced by the results of this regression analysis.

Since the Spring 2000 study was pseudo-replicated (sensu Hurlbert 1984, one site on each of three streams) these results need to be considered cautiously and technically apply to that specific location and not necessarily to the entire stream. However, since sampling was done over a 6-month period it is unlikely that a problematic location effect would produce such consistent and significant results. In stream assessment methods, it is generally unnecessary to sample multiple sites in the same reach to characterize the condition of the drainage (Karr and Chu 1999, Rabeni et al. 1999, Barbour et al. 1998). Many RBA's are based on one sample from one site that represents the entire watershed (Karr and Chu 1999, Barbour et al. 1998, Barbour et al. 1994). Substantial intercorrelation among the independent variables is another issue which could produce exaggerated relationships in Forward Stepwise Regression (Snedecor, 1989). Although all variables that were highly correlated ($r > 0.80$) were removed from the analysis, there were some significant correlations that were left in (e.g.<u>Taxa Richness</u> was significantly correlated to <u>Percent Dominance</u>, $r = 0.58$ and <u># of High Pulses</u> was significantly correlated to <u># of Reversals</u>, $r = 0.67$). This decision was based on the belief that the benefit provided by including as many potentially important hydrology variables as possible outweighed the potential problems with intercorrelation.

CONCLUSIONS:

I examined how biological assessments of streams in Central Texas may be influenced by general and antecedent hydrological conditions. The results of this analysis are based on analysis of 7 years of biological monitoring in the Austin area in addition to a small, more controlled and homogenous data set from three distinct streams during one flow season (6 months). The results show that not only do the measures of hydrologic variability selected for this work differentiate between development condition, but that they are also ecologically relevant, showing strong relationships to the biological communities in this area in both developed and undeveloped streams. However, this study also showed that variation due to hydrogeological influences, small differences in drainage area, and other variables un-quantified in this study cause some reduction in the ability to detect development condition differences in these watersheds. Clearly the ecological constraints on benthic macroinvertebrate communities in the Central Texas area are a complex mix of macro-scale climatological influences, natural watershed variation and anthropogenic influences. By isolating hydrology, as this study has attempted, we are closer to understanding the anthropogenic influences, which is the goal of bioassessment in general.

Of the biological and hydrological variables selected for this study, some specific conclusions can be made about their relationships. Measures of taxonomic richness, particularly EPT Taxa, as well as the metrics involving the Chironomidae family (Percent Chironomidae and EPT/EPT+Chironomidae), appear to be highly influenced by hydrological variability. This was supported by both the long and short term studies.

These are critical biological metrics, used in benthic macroinvertebrate-based biological assessments in Central Texas by a wide range of water quality monitoring agencies (Texas Natural Resource Conservation Commission, Lower Colorado River Authority, City of Austin) and it is important that people applying these methods are aware of this relationship. Generally, as the magnitude of hydrologic variability increases, these metrics are negatively affected. In urban streams, this translates into poor bioassessment scores due to the altered hydrology from impervious cover, but not necessarily from non-point source pollution. In undeveloped streams, which function as reference conditions, the effects of hydrology are probably less important, but still represent an important predictor of benthic community structure. Geology, geomorphology and specific habitat variables introduce a high level of site variability, reducing the ability of bioassessment techniques to detect differences in development level in the Austin area. This was shown in the Spring 2000 study in the inability to separate Bull (moderately developed) from Bear Creek (undeveloped) using the selected biological metrics or the hydrology variables. Resolution of bioassessment methods is probably reduced in this area compared to the more predictable, stable flow regimes in temperate areas where these methods were developed. Apparently, we can distinguish between gross differences in development condition (Shoal and Bear creeks), but we still don't know to what extent that difference is due to hydrology or non-point source pollution.

Lack of predictability and the coefficient of variation were the best hydrological variables in predicting benthic macroinvertebrate community structure in the 7-year historic analysis. Lack of predictability tended to lower biological metric scores, while increasing variation improved richness scores. However, in the Spring 2000 study, there was no single hydrology variable that performed significantly better than the others. The results of both of these analyses showed that a combination of multiple hydrology variables work much better at predicting macroinvertebrate metrics than a single, "master" hydrology variable, as has been proposed by several studies (Clausen and Biggs 1997, Townsend and Scarsbrook 1997, Jowett and Duncan 1990). The work done by Poff and Ward (1989) and Richter et al. (1996) in developing the "hydrological indices" approach utilized in this analysis proved to be ecologically relevant and with application to the streams in this area.

Drought conditions prior to the Spring 2000 study appear to have strongly influenced the community structure at the three study streams. De-watering, especially for extended period of times, is a much stronger disturbance mechanism than pulse flood events, as was demonstrated by the slow (6-month) recovery of Taxa Richness in the three study streams. Although several studies have shown that intermittent stream organisms respond quickly to the return of flow to a stream (Miller and Golladay 1996, Williams 1996, Delucchi 1989, Delucchi 1988), the length of drought, and how that influences stream recovery is not well documented or understood. It is possible that extended droughts experienced in the Central Texas area require longer recovery periods than has been documented in the literature for more temperate streams, and that the users of bioassessment in this area should factor long recovery periods following drought into their monitoring schedules.

The Spring 2000 study, which used 1-hour mean discharge from the gage sites instead of the daily (24-hour) means used for the historic analysis, resulted in much better predictive abilities for the biological variables. Due to the size of the watersheds that are assessed in the City of Austin monitoring programs, and the high intensity, short duration rain patterns common here, daily mean flows probably do not characterize flow regimes adequately. In the Spring 2000 study there was a wider range of flow variables (5) which significantly influenced the biological variables than in the historic analysis (3). This is probably due to the loss of the majority of the hydrograph when hydrology variables were based on daily mean flows. Since hourly mean flows are difficult to obtain and the USGS will not certify their accuracy, it is unlikely that hydrology indices can be based on finer scaled data for many state and federal monitoring groups. However, agencies working on the regional scale that can obtain 1 hour and even 15-minute interval flow data from USGS contacts may be able to generate accurate hydrology indices that can predict both physical and biological variation due to development condition.

This study evaluated the effects of impervious cover on hydrology and the effects of hydrology on biological communities. However, I have not separated the differential effects of undeveloped and urbanized hydrology regimes on biological assessment. It is very likely that with further study, it can be shown that urbanized, altered hydrology has a stronger relationship to benthic macroinvertebrate community structure than the natural flow regime found in undeveloped watersheds. This would not provide all the answers to biological monitoring problems in this area, but would go far in defining what we are measuring when we detect water quality degradation in urbanizing streams. Much of what we determine to be a "water quality" problem in urban streams in Central Texas may be physical in nature and not related to water chemistry or non-point source pollution. This is an important consideration in managing urban streams since decision-makers must be able to develop and implement solutions to water quality problems based on an understanding of what the source of the problem is. If non-point pollution is not the only problem with urban streams, than understanding benthic macroinvertebrate communities in urban streams and utilizing biological indicators of stream health is a much more appropriate measure of "water quality" than collecting water samples for chemical analysis. Research into finer grained hydrological indices, improved metrics and larger biological data sets should provide better resolution of bioassessment methods in this area and separate the specific effects of hydrology vs. non-point source chemical degradation in urban watersheds.

Table 1. Selected biological metrics used in the analysis of City of Austin historic benthic macroinvertebrate data, their calculation and measure of community structure (1993-1999).

	Metric Name	Community Structure	Calculation
1	Total Taxa	Taxonomic Richness	# of taxa
2	Diptera Taxa	Taxonomic Richness	# of taxa in Diptera order
3	EPT Taxa	Taxonomic Richness	# of taxa in three EPT orders
4	% Dominant Taxa	Community Composition	% of largest taxa in total sample
5	% EPT Abundance	Community Composition	% of EPT organisms in total sample
6	% Chironomidae Abundance	Community Composition	% of Chironomidae family in total sample
7	EPT/EPT+Chironomidae	Community Composition	Total # EPT orgs/EPT orgs plus # of Chironomidae orgs
8	Hilsenhoff Biotic Index	Community Tolerance	HBI=sum (X_iT_i)/n where: X_i=# of indiv. In each species, Ti=tolerance value of each species, n=total organims in sample

Table 2. Hydrologic Parameters used to characterize flow regime in City of Austin historic analysis. Group 1 was taken from Poff and Ward (1989). Group 2 was taken from Richter et al. (1996). Shaded parameters were highly inter-correlated (r >0.80) with another parameter and not used as independent variables in the regression analysis.

	Group 1 - Summary statistics (Poff and Ward 1989)	Units
1	Mean Annual Flow	cfs
2	Annual Coefficient of Variation	n/a
3	Flow Predictability	n/a
4	Constancy Predictability	n/a
5	% of Floods in 60 day period	%
6	Flood Free Season	# days
	Group 2 - Index of Hydrologic Alteration statistics (Richter et al. 1996)	
7	Annual minima, 1-day means	cfs
8	Annual minima, 3-day means	cfs
9	Annual minima, 7-day means	cfs
10	Annual minima, 30-day means	cfs
11	Annual minima, 90-day means	cfs
12	Annual maxima, 1-day means	cfs
13	Annual maxima, 3-day means	cfs
14	Annual maxima, 7-day means	cfs
15	Annual maxima, 30-day means	cfs
16	Annual maxima, 90-day means	cfs
17	Number of Zero-Flow days	# days
18	Baseflow (7-day minimum flow/mean for year)	cfs
19	Julian date of each annual 1 day maximim	date
20	Julian date of each annual 1 day minimum	date
21	Number of low pulses within each year	#/year
22	Mean duration of low pulses within each year	# days
23	Number of high pulses within each year	#/year
24	Mean Duration of high pulses within each year	# days
25	The low pulse level	cfs
26	The high pulse level	cfs
27	Rate of rise (mean of + differences between daily means)	cfs/year
28	Rate of fall (mean of - differences between daily means)	cfs/year
29	Number of reversals	#/year

Table 3. Selected hydrological variables used as independent variables in Spring 2000 study. They are adapted from the works of Poff and Ward (1989) and Richter et al. (1996). Shaded variables were found to be highly correlated with at least one other variable and were eliminated.

Parameter Name	Description	Units
Mean Flow	Mean for previous 30d	cfs
Coefficient of Variation	CV for previous 30d	n/a
% of Flood Flows	% of time discharge was over 75th%	hours
% of Low Flow	% of time discharge was below 25th%	hours
Duration of High-pulse	Avg # of hours a pulse remained above the 75th %	hours
Duration of Low-pulse	Avg # of hours a pulse remained below the 25th %	hours
Avg. discharge of High pulses	Avg of discharge of all pulses above 75th%	cfs
Avg discharge of low pulses	Avg of discharge of all pulses below 25th%	cfs
# of high pulses	# of times distinct pulses went above 75th%	n/a
# of low pulses	# of times distinct pulses went below 25th%	n/a
# of reversals	# of sign changes (+ or -) during 30d period	n/a
Rate of rise	Mean of all + diff. between consecutive hrly values	cfs/hour
Time since max discharge	# of hours since max discharge in 30d period	hours
Time since min discharge	# of hours since min discharge in 30d period	hours

Table 4. Hydrology variables used in regression analysis of City of Austin historical biological data. The first six are summary statistics proposed by Poff and Ward (1989), and the remaining 5 were selected from the hydrological statistics proposed by Richter et al. (1996).

Hydrology Statistic	Description	Source
Mean Annual Flow	Average of daily means for each year of record.	Poff & Ward 1989
Annual Coefficient of Variation	Standard deviation/mean for each year of record.	Poff & Ward 1989
Flow Predictability	Colwells (1974) predictibility for periodic phenomena	Poff & Ward 1989
Constancy Predictability	Predictibility as described by constancy (colwell 1974).	Poff & Ward 1989
% of Floods in 60 day period	Max proportion of floods that occur in any 60-d period.	Poff & Ward 1989
Flood Free Season	Max number of days in each year in which no floods occured.	Poff & Ward 1989
Number of Zero Flow Days	Number of days with no flow in each flow year.	Richter et al. 1996
High Pulse Count	Number of high pulses in each flow year.	Richter et al. 1996
Low Pulse Count	Number of Low pulses in each flow year.	Richter et al. 1996
Fall Rate	Average negative rate of change in flow for each flow year.	Richter et al. 1996
Number of Reversals	Number of changes in flow direction (increasing/decreasing) in each flow year.	Richter et al. 1996

Table 5. Multiple regression analysis of City of Austin historic biological data vs. 11 hydrologic variables. Table provides Beta values and R^2 at the bottom for each model. NR variables are not related to the model. Highest two beta values in each model are shaded.

Model Components (Hydro-variables)	Metric 1 (HBI)	Metric 2 (EPT/Chir)	Metric 3 (Taxa Richness)	Metric 4 (EPT Taxa)	Metric 5 (Diptera Taxa)	Metric 6 (Percent Dom.)	Metric 7 (Percent Chir.)	Metric 8 (Percent EPT)
Mean Annual Flow	NR	0.183	NR	NR	-0.727	NR	-0.197	-0.418
Annual Coefficient of Variation	-1.84	1.069	NR	0.309	0.325	-0.642	-0.636	1.21
Flow Predictability	0.735	NR	-0.158	-0.287	NR	-0.054	NR	0.399
Constancy Predictability	-1.13	1.066	-2.082	-1.93	-0.515	NR	-0.75	NR
% of Floods in 60 day period	0.833	-0.483	-1.07	-1.069	-0.847	0.523	0.189	-0.468
Flood Free Season	-1.63	0.899	-0.924	-0.763	NR	NR	-0.419	NR
Number of Zero Flow Days	NR	-0.888	-0.549	-0.422	NR	NR	NR	NR
High Pulse Count	-0.332	0.471	0.976	0.876	-0.387	NR	0.48	NR
Low Pulse Count	NR	0.195	-0.09	NR	-0.216	-0.101	-0.522	0.366
Fall Rate	0.492	NR	NR	NR	-0.52	0.276	NR	-0.766
Number of Reversals	NR	-0.639	-1.356	-1.281	NR	-0.027	NR	-0.358
Variability explained (R^2)	**0.35**	**0.46**	**0.29**	**0.37**	**0.20**	**0.12**	**0.41**	**0.29**

Table 6. ANOVA comparing site effects of 11 hydrologic variables (from Shoal, Bull and Bear creeks) in the Spring 2000 study. Significant ANOVA results indicate that at least two of the three streams were different from each other.

Hydrology Variable	P-Value	Difference at P<0.05
Coefficient of Variation	0.0002	Significant
Mean Flow	0.0666	Not Significant
% of Flood Flows	0.5404	Not Significant
Duration of High-pulse	0.2624	Not Significant
Avg. discharge of High pulses	0.0006	Significant
# of high pulses	0.0713	Not Significant
# of low pulses	0.0459	Significant
# of reversals	0.0002	Significant
Rate of rise	0.0042	Significant
Time since max discharge	0.7462	Not Significant
Time since min discharge	0.0797	Not Significant

Table 7. ANOVA evaluating differences in overall metrics scores among Shoal, Bull and Bear creeks during Spring 2000 study. Shaded cells (P<0.05) indicate no significant difference between sites.

Metrics	p-value
# of Organisms (n)	0.036
Hilsenhoff Biotic Index	0.004
EPT/EPT+Chironomid	**0.231**
Total Taxa	0.000
EPT Taxa	0.005
Diptera Taxa	**0.486**
% Dominant Taxa	**0.060**
% Chironomidae Abundance	**0.057**
% EPT Abundance	**0.322**

Table 8. Multiple regression analysis of physico-chemical variables vs. each biological metric from the Spring 2000 study. Table provides Beta values and R2 values at the bottom for each significant model (P<0.05). NR variables are not related to the model. Highest beta value in each model is shaded.

Model Components (Physico-Chem. Var.)	Metric 1 (HBI)	Metric 2 (EPT/Chir)	Metric 3 (Taxa Richness)	Metric 4 (EPT Taxa)	Metric 5 (Diptera Taxa)	Metric 6 (Percent Dom.)	Metric 7 (Percent Chir.)	Metric 8 (Percent EPT)	Metric 9 (# of Orgs)
Velocity (f/s)	NR	0.65	0.74	0.74	NR	NS	-0.58	0.42	0.78
Conductivity (uS/cm)	0.54	NR	-0.25	NR	-0.59	NS	NR	NR	0.19
Dissolved Oxygen (mg/l)	NR	-0.36	-0.22	-0.32	0.23	NS	NR	-0.36	NR
Temperature (°C)	NR	0.34	NR	0.37	NR	NS	-0.25	0.50	-0.47
Variability explained (R^2	**0.29**	**0.54**	**0.68**	**0.68**	**0.39**	**NS**	**0.44**	**0.52**	**0.66**

Table 9. Multiple regression analysis of antecedent hydrological variables vs. each biological metric from the Spring 2000 study. Table provides Beta values and R^2 values at the bottom for each significant model (P<0.05). NR variables are not related to the model. The highest two beta value in each model are shaded. Metric 5 and 6 (Diptera Taxa and Percent Dominance) were not significantly (NS) related to the hydrologic model.

Model Components (Hydro-variables)	Metric 1 (HBI)	Metric 2 (EPT/Chir)	Metric 3 (Taxa Richness)	Metric 4 (EPT Taxa)	Metric 5 (Diptera Taxa)	Metric 6 (Percent Dom)	Metric 7 (Percent Chir.)	Metric 8 (Percent EPT)	Metric 9 (# of Orgs)
Mean Flow	NR	NR	NR	NR	NS	NS	-0.71	NR	NR
Coefficient of Variation	0.75	NR	-0.65	-0.92	NS	NS	NR	NR	0.30
% of Flood Flows	NR	NR	NR	NR	NS	NS	NR	NR	NR
Duration of High-pulse	NR	0.30	0.43	0.61	NS	NS	NR	0.27	0.54
Avg. discharge of High pulses	NR	0.57	NR	NR	NS	NS	NR	0.55	NR
# of high pulses	NR	NR	-0.48	-0.56	NS	NS	0.30	-0.42	0.58
# of low pulses	NR	NR	-0.25	NR	NS	NS	0.37	NR	0.56
# of reversals	0.80	NR	0.75	0.48	NS	NS	NR	NR	0.31
Rate of rise	NR	-1.11	-0.13	-0.21	NS	NS	1.12	-1.04	NR
Time since max discharge	-0.41	-0.51	-0.18	-0.51	NS	NS	0.61	-0.70	NR
Time since min discharge	0.35	NR	-0.48	-0.70	NS	NS	NR	NR	NR
Variability explained (R^2	**0.71**	**0.64**	**0.96**	**0.92**	**NS**	**NS**	**0.80**	**0.66**	**0.70**

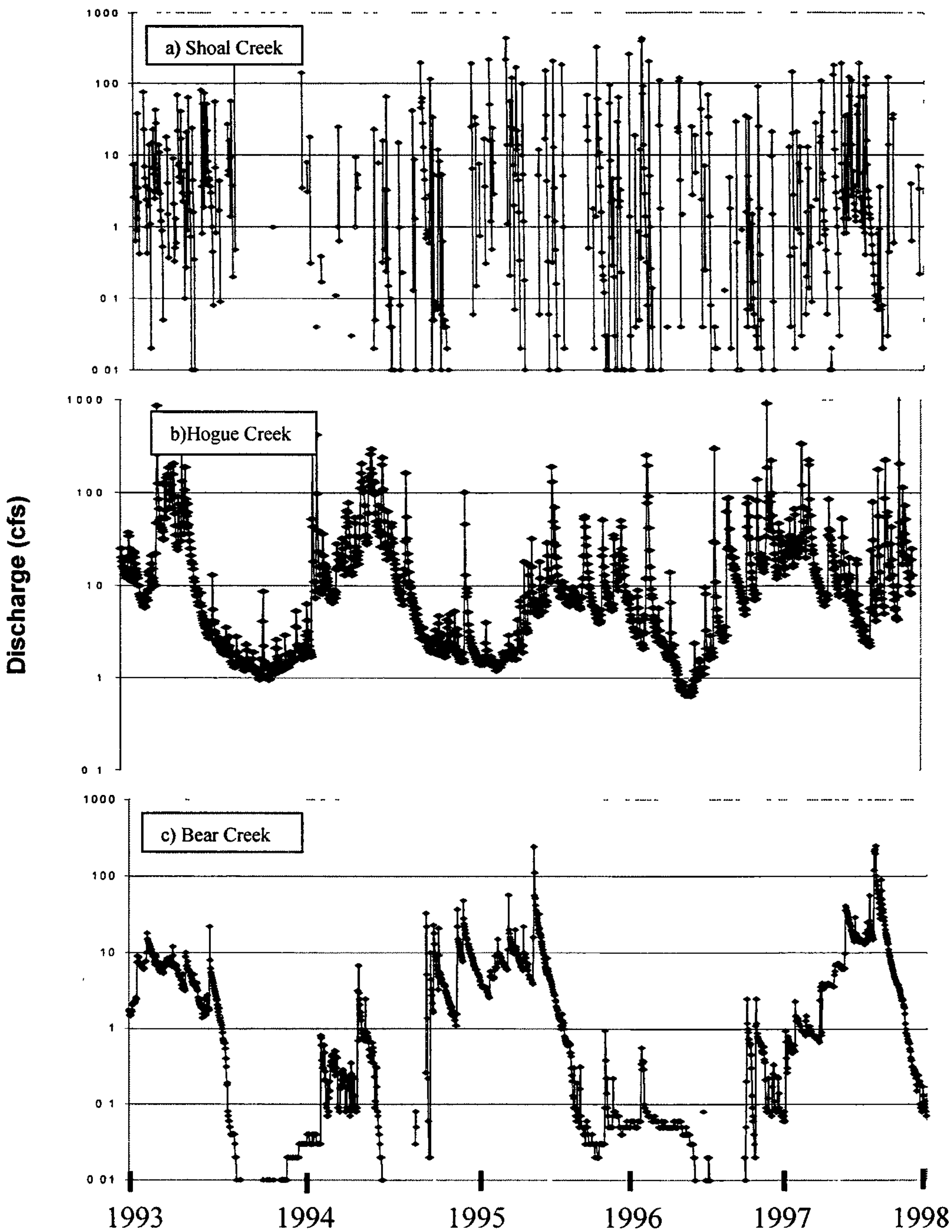


Figure 1 Hydrograph of mean daily discharge on Shoal (a), Hogue (b) and Bear (c) creeks from 1993 – 1998 Shoal (a), is an urban drainage (31.6 sq km drainage area) in Austin, TX and shows short storms (vertical lines) followed by periods of no flow Hogue creek is an undeveloped drainage (38 9 sq km) in Northern Virginia showing typical temperate hydrology. Bear Creek is an undeveloped drainage (31.6 sq km) in the Austin, TX area.

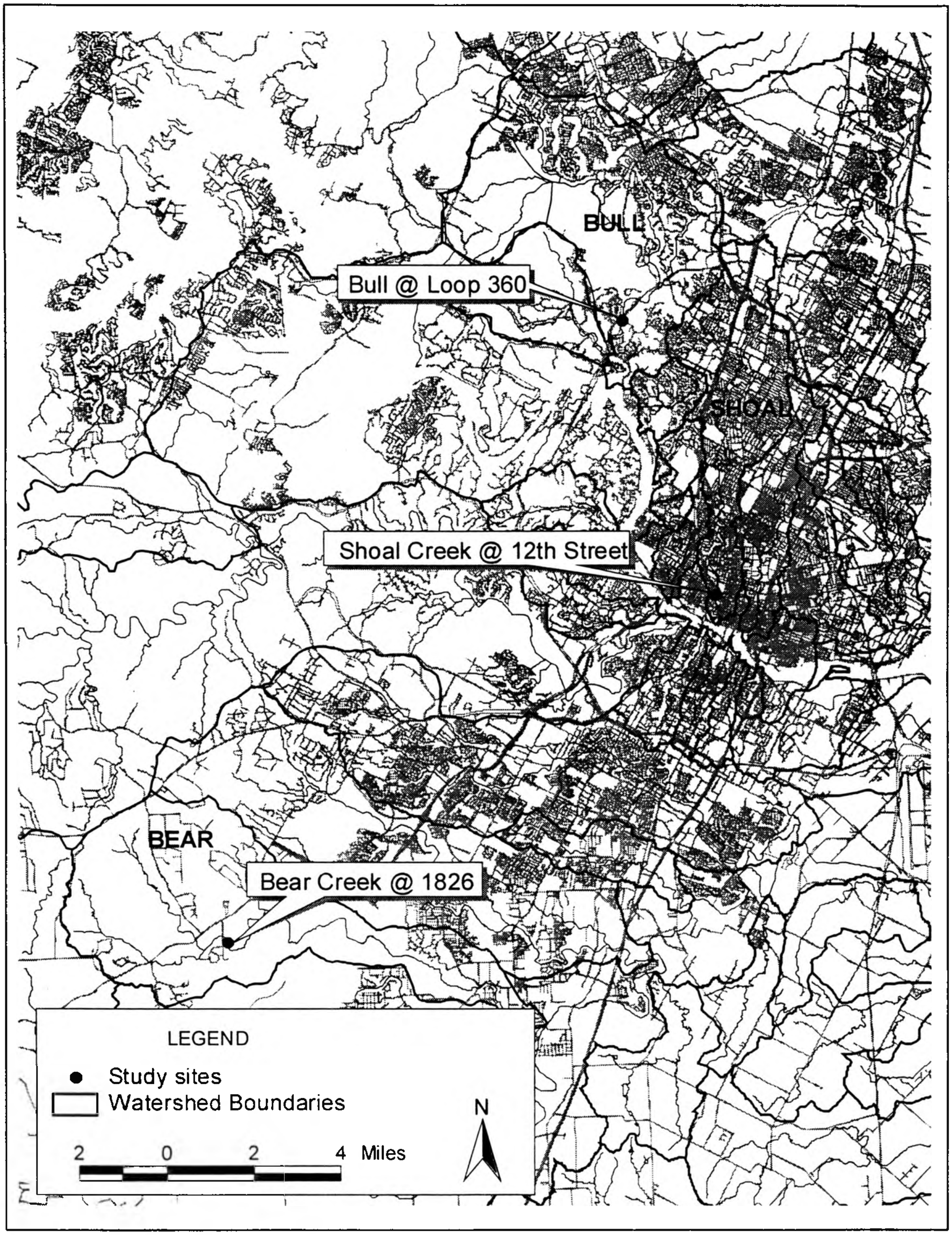


Figure 2. Map showing general location of three study watersheds and sampling sites used for the Spring 2000 study. USGS gaging stations are immediately (<300m) upstream of study sites.

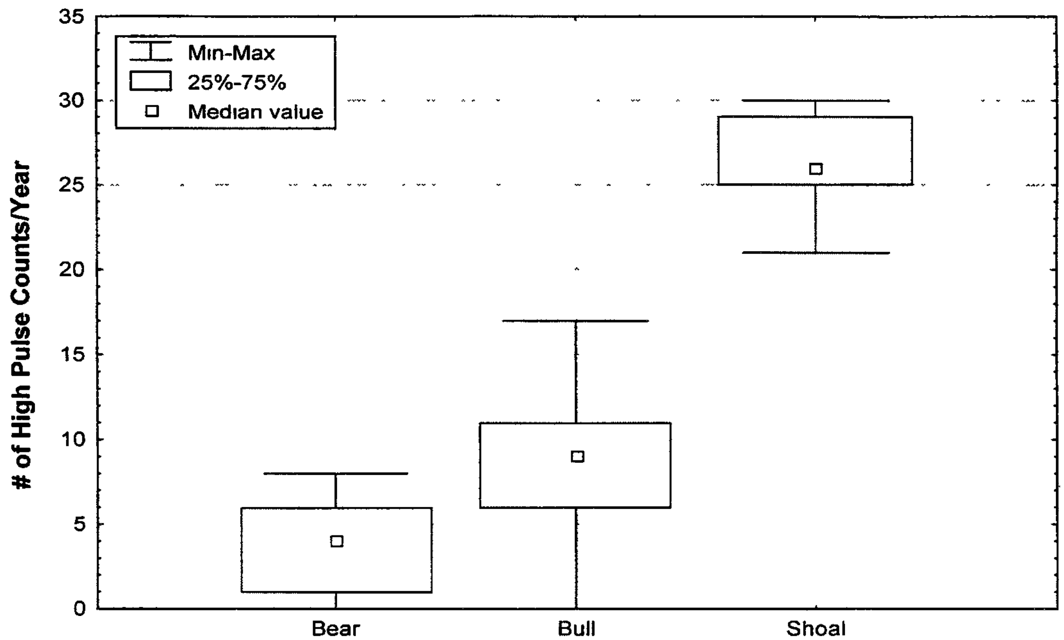


Figure 3. Box plots of annual number of High Pulse Counts at gage stations of three streams during period of record (1993-1998).

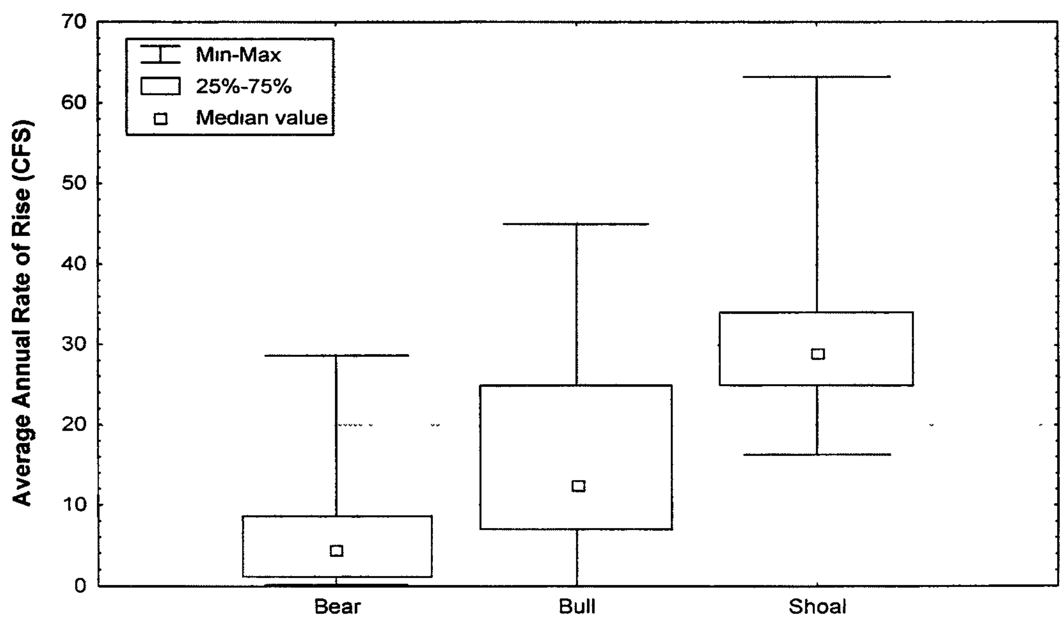


Figure 4 Box plots of average annual Rate of Rise (CFS) at gage stations of three streams during period of record (1993-1998).

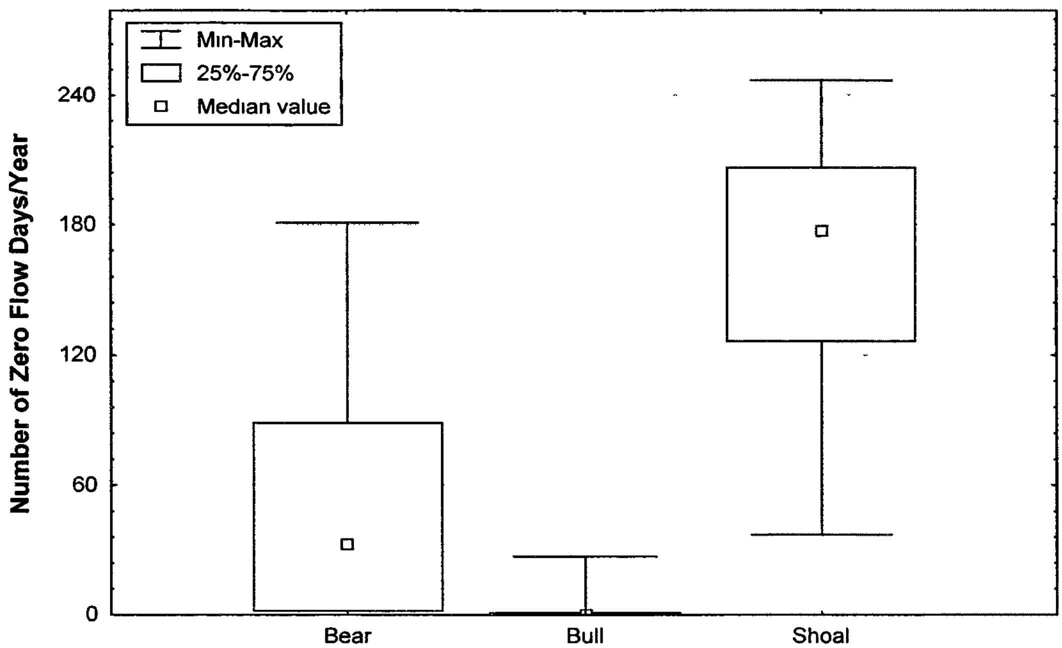


Figure 5. Box plots of annual number of Zero Flow Days at gage stations of three streams during period of record (1993-1998).

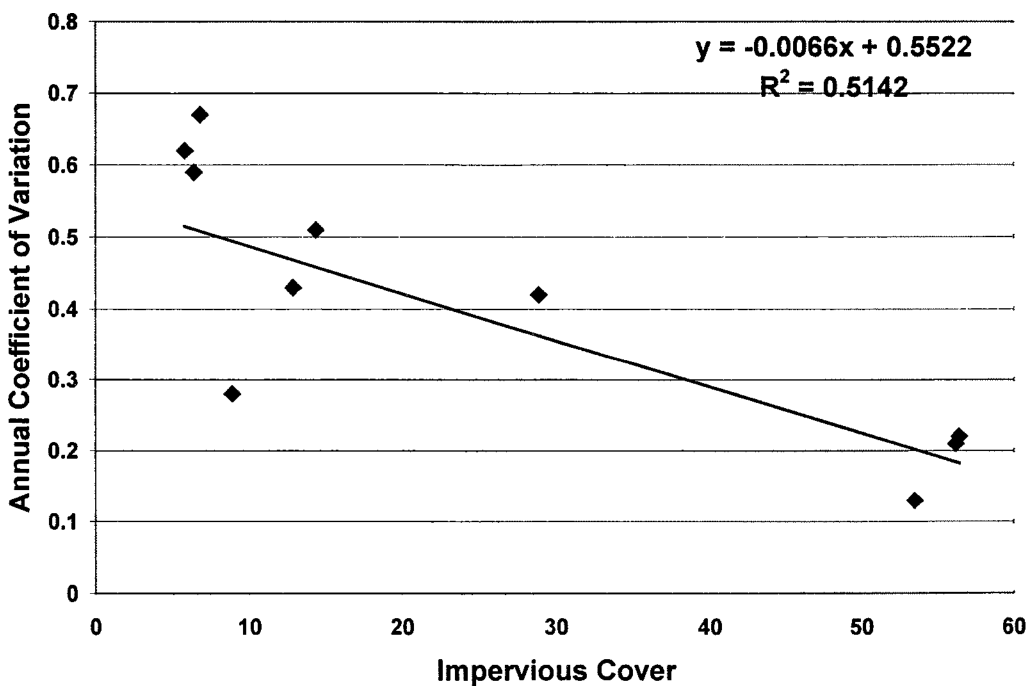


Figure 6. Linear regression of annual Coefficient of Variation (ACV) vs. Impervious Cover at the 11 USGS gaging stations where City of Austin historic (7-year) hydrology variables were measured.

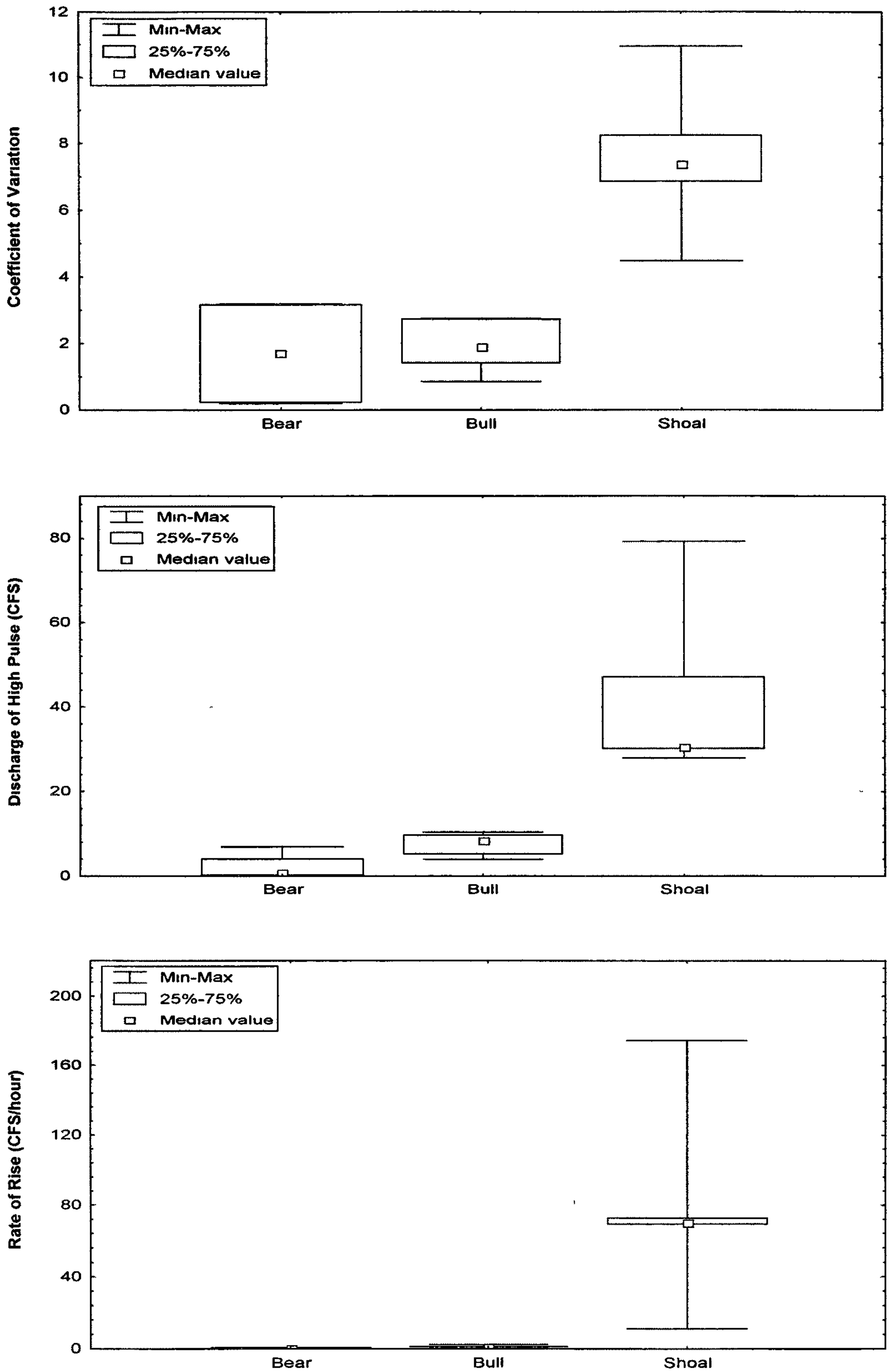


Figure 7 Box plots of the hydrology variables Coefficient of Variation, Discharge of High Pulse and Rate of Rise at three study streams (Feb – July, 2000)

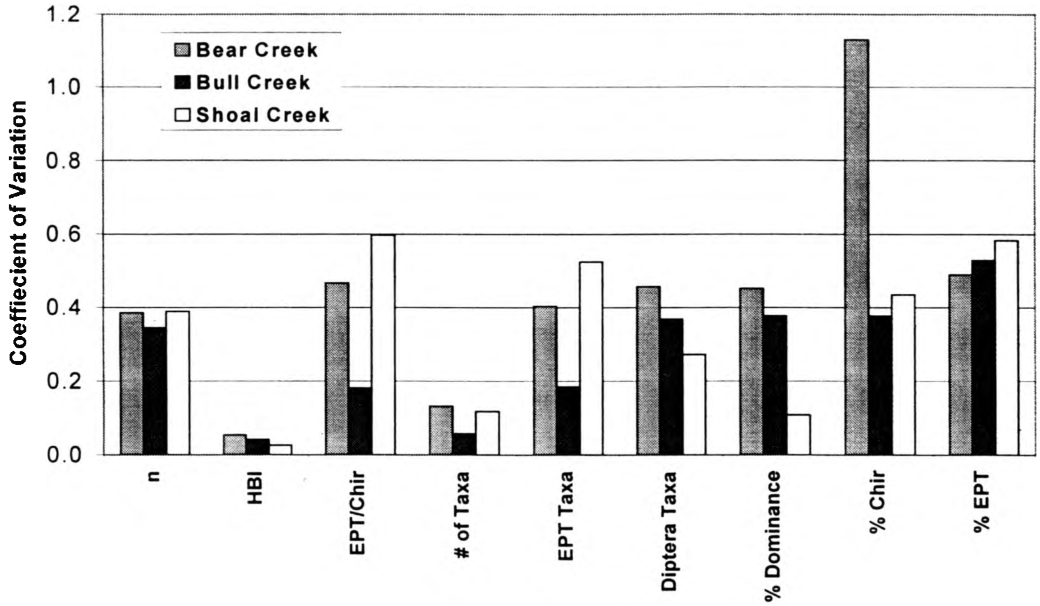


Figure 8. Chart of the Coefficient of Variation (CV) of the metrics used in the Spring 2000 study analysis. Each bar represents one of the three study streams in order of increasing impervious cover (Bear, Bull and Shoal). Bear and Shoal had the highest CV's in the most metrics (5 and 4, respectively), Bull consistently had the lowest.

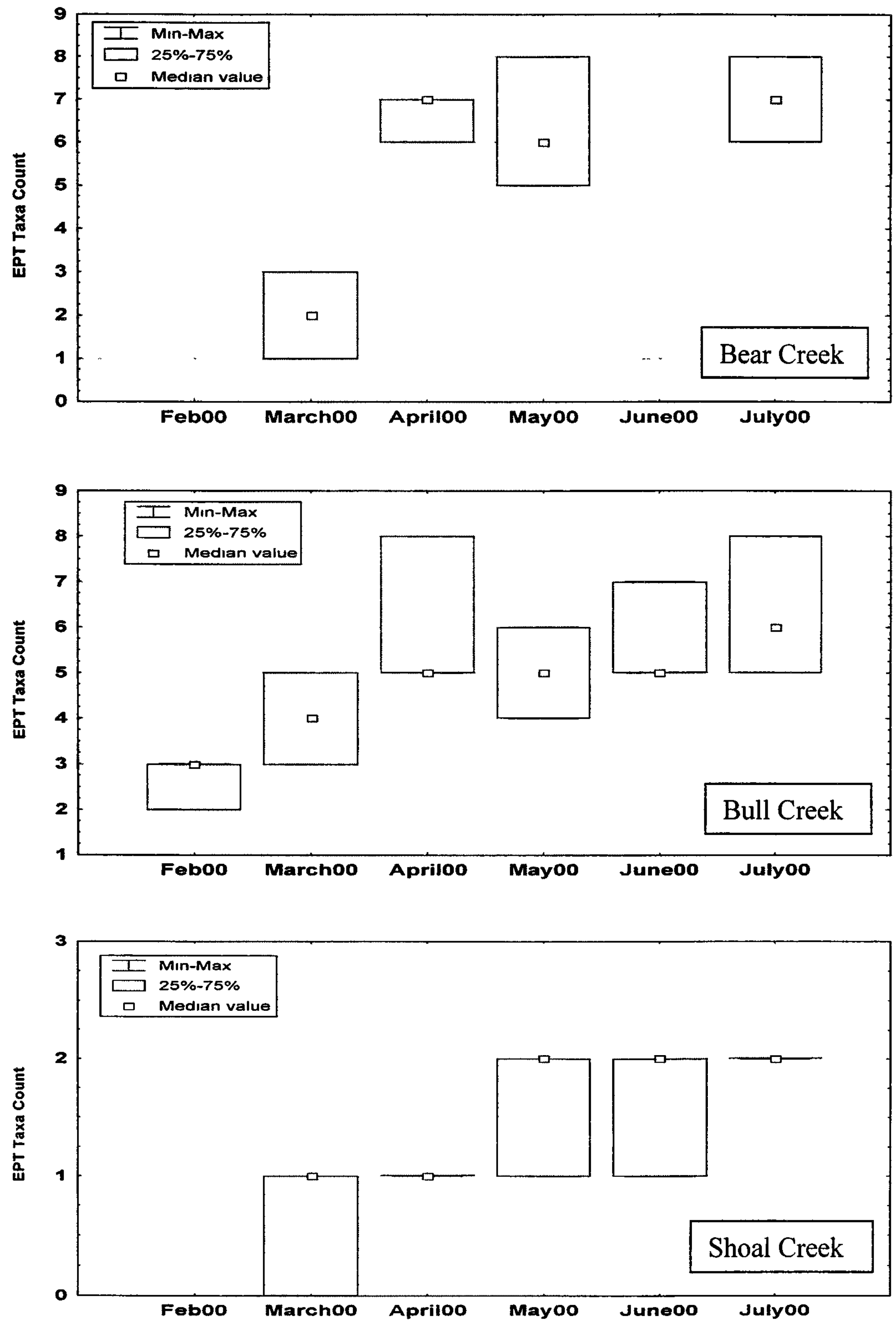
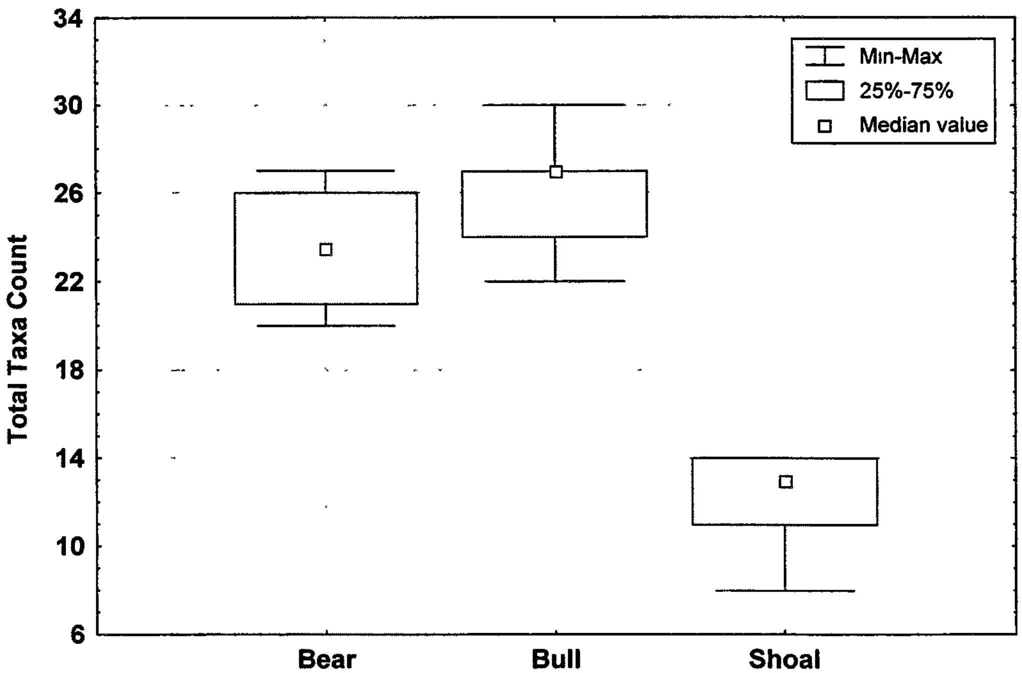


Figure 9. Monthly range of EPT Taxa metric on Bear, Bull and Shoal creeks during Spring 2000 study. At all three streams there was a temporal trend toward increasing number of EPT taxa. Bear and Shoal were dry during February survey June survey on Bear was not completed due to field error.



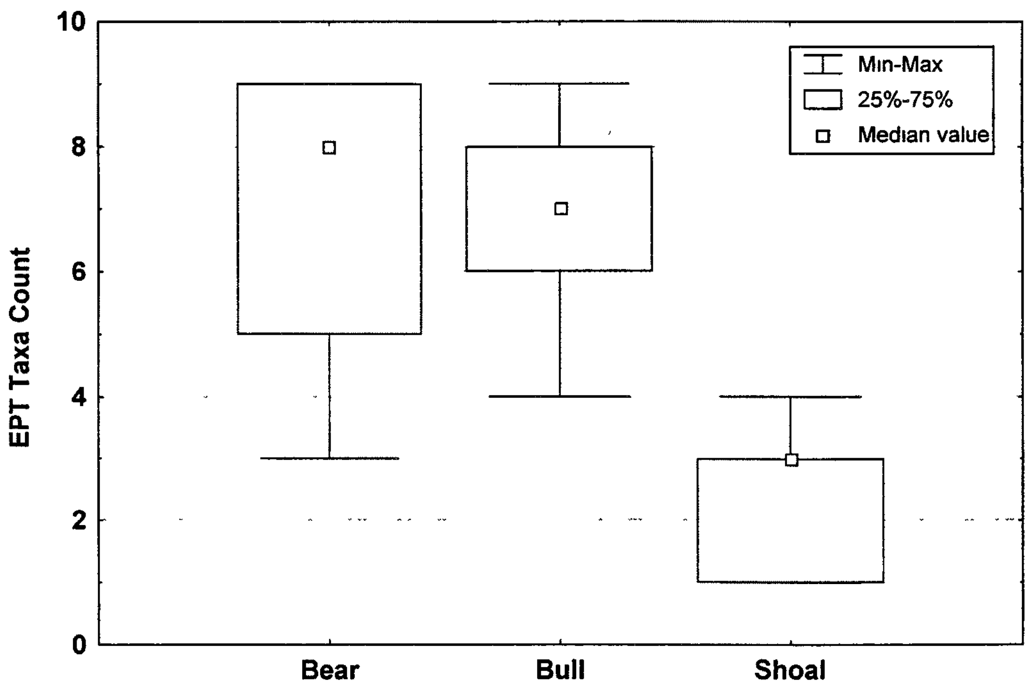
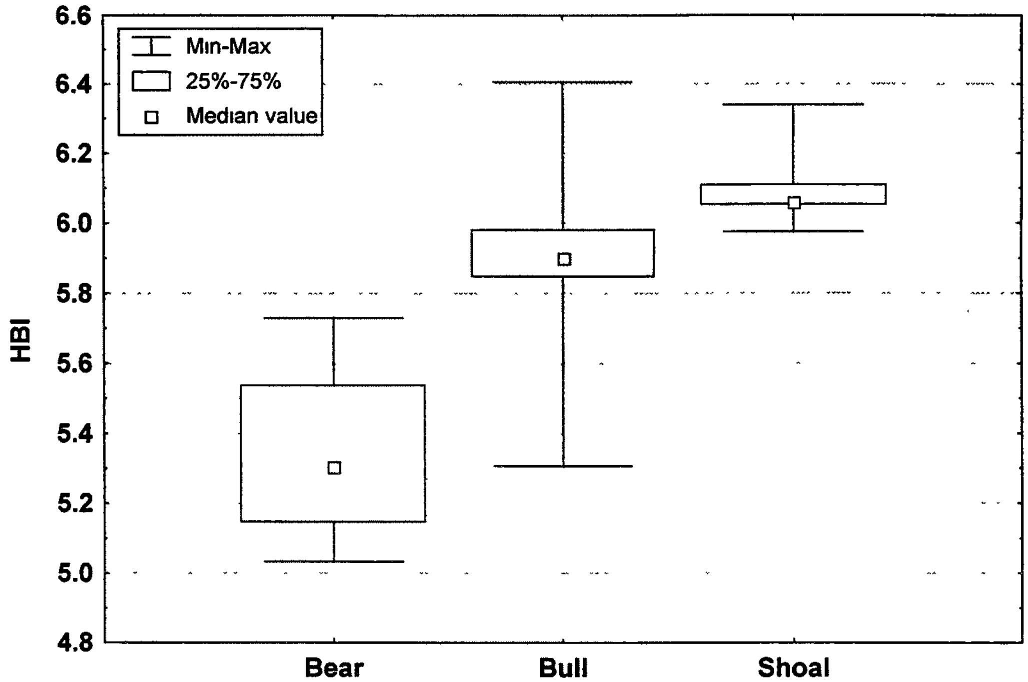


Figure 10 Total Taxa and EPT Taxa metric scores during 6-month Spring 2000 study. Box plots show range of scores during study period at three streams. In both cases, Shoal was significantly different than Bull or Bear, but Bull and Bear were not different from each other.



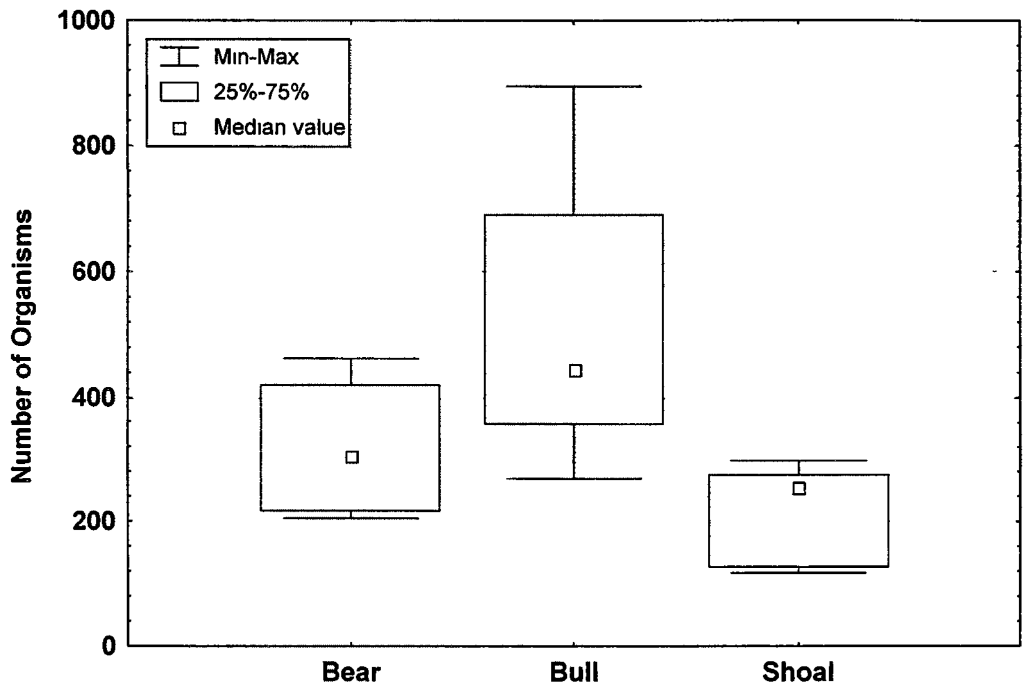


Figure 11. Hilsenhoff Biotic Index (HBI) and Number or Organisms box plots showing range of scores during study period at three streams. The higher the HBI score the lower the metric score (more tolerant organisms). Shoal Creek was different than Bear for the HBI metric and different than Bull for the Number of Organisms metric.

APPENDIX 1

Taxa						Shoal Creek @ 12th St. (USGS)				
Phylum	Class	Order	Family	Genus/Species	Tolerance Value	3/23/00	4/20/00	5/17/00	6/8/00	6/30/00
Arthropoda	Insecta	Odonota	Coenagrionidae	*Argia*	7			1	2	7
Arthropoda	Insecta	Ephemoroptera	Caenidae	*Caenis*	7			1		1
Arthropoda	Insecta	Diptera	Stratiomyiidae	*Caloparyphus*	7			1	4	
Arthropoda	Insecta	Trichoptera	Hydropsychidae	*Cheumatopsyche*	6			2	2	
Arthropoda	Insecta	Trichoptera	Philopotamidae	*Chimarra*	4					1
Arthropoda	Insecta	Diptera	Chironomidae		6	258	118	144	29	16
Arthropoda	Insecta	Copopoda			4	2				
Arthropoda	Insecta	Diptera	Culicidae		6	7	4	3		7
Platyhelminthes	Turbellaria	Tricladida	Planariidae	*Dugesia tigrina*	7		4	10	2	8
Arthropoda	Insecta	Diptera	Stratiomyiidae	*Euparyphus*	7				1	
Arthropoda	Insecta	Ephemoroptera	Baetidae	*Fallceon quillen*	6	3	79	43	68	56
Annelida	Hirudinea	Gnathobdellida			7		1		2	
Arthropoda	Crustacea	Amphipoda	Talitridae	*Hyallela azteca*	8		13			1
Arthropoda	Insecta	Coleoptera	Dytiscidae	*Hydaticus*	5		1			
Arthropoda	Arachnoidea	Hydracarina			4					1
Arthropoda	Insecta	Coleoptera	Hydrophilidae	*Hydrophilus*	8	1				
Arthropoda	Insecta	Trichoptera	Hydroptilidae	*Hydroptila*	6				1	1
Arthropoda	Insecta	Hemiptera	Veliidae	*Microvelia*	6		1			
Arthropoda	Insecta	Diptera	Muscidae		6		1			
Annelida	Oligochaeta				8	9	14	2	6	
Arthropoda	Crustacea	Ostracoda			4			1		
Mollusca	Gastropoda	Limnophila	Physidae	*Physella*	8	1	18	2		
Mollusca	Gastropoda	Limnophila	Planorbidae		6		2			1
Arthropoda	Insecta	Coleoptera	Psephenidae	*Psephenus*	4			1		1
Arthropoda	Insecta	Hemiptera	Veliidae	*Rhagoelia*	6					1
Arthropoda	Insecta	Diptera	Simuliidae	*Simulium*	6	17	20	41		
Arthropoda	Insecta	Coleoptera	Elmidae	*Stenelmis*	5			2	9	15
				Total # of organisms		298	276	254	126	117
				Total # of Taxa		8	13	14	11	14

Taxa						Bull Creek @ Loop 360					
Phylum	Class	Order	Family	Genus/Species	Tolerance Value	2/11/00	3/24/00	4/28/00	5/18/00	6/8/00	7/3/00
Mollusca	Gastropoda	Lymnophila	Ancylidae	*Hebetancylus*	6					1	
Arthropoda	Insecta	Odonota	Coenagrionidae	*Argia*	7	13	14	13	25	127	59
Arthropoda	Insecta	Coleoptera	Hydrophilidae	*Berosus*	5	10	11	6	5	4	5
Arthropoda	Insecta	Odonota	Libellulidae	*Brechmorhoga mendax*	6				4	13	16
Arthropoda	Insecta	Ephemoroptera	Caenidae	*Caenis*	7		1		3		11
Arthropoda	Insecta	Ephemoroptera	Baetidae	*Callibaetis*	4			1			
Arthropoda	Insecta	Diptera	Stratiomyiidae	*Caloparyphus*	7	2			2	1	
Arthropoda	Insecta	Ephemoroptera	Baetidae	*Camelobaetidius*	4		1	9	10	8	3
Arthropoda	Insecta	Diptera	Ceratopogonidae		6						1
Arthropoda	Insecta	Trichoptera	Hydropsychidae	*Cheumatopsyche*	6		61	38	256	273	18
Arthropoda	Insecta	Trichoptera	Philopotamidae	*Chimarra*	4					10	2
Arthropoda	Insecta	Diptera	Chironomidae		6	125	14	60	74	67	51
Arthropoda	Insecta	Collembola			6			1			
Mollusca	Gastropoda	Pelecypoda	Corbiculidae	*Corbicula fluminea*	5	1	1		5		2
Arthropoda	Insecta	Diptera	Culicidae		6	5	1		5		9
Platyhelminthes	Turbellaria	Tricladida	Planariidae	*Dugesia tigrina*	7	4	21	24	11	30	18
Arthropoda	Insecta	Coleoptera	Hydrophilidae	*Enochrus*	9		1				
Arthropoda	Insecta	Odonota	Gomphidae	*Erpetogomphus*	4						4
Arthropoda	Insecta	Ephemoroptera	Baetidae	*Fallceon quilleri*	6	61	20	18	24	13	6
Arthropoda	Insecta	Trichoptera	Helicopsychidae	*Helicopsyche*	5	1		1	2	5	4
Arthropoda	Insecta	Diptera	Emipidae	*Hemerodromia*	6				1		1
Arthropoda	Insecta	Odonota	Calopterygidae		6			1			
Arthropoda	Crustacea	Amphipoda	Talitridae	*Hyallela azteca*	8		2	19	18	14	98
Arthropoda	Arachnoidea	Hydracarina			4	7	11	16	10	7	1
Arthropoda	Insecta	Hemiptera	Hydrometridae	*Hydrometra*	4						1
Arthropoda	Insecta	Trichoptera	Hydroptilidae	*Hydroptila*	6	3	2	17	5	7	19
Mollusca	Gastropoda	Limnophila	Lymnaidae		6	2	4		1	2	1
Arthropoda	Insecta	Coleoptera	Elmidae	*Macrelmis*	4			3		3	
Arthropoda	Insecta	Coleoptera	Elmidae	*Microcylloepus pusillus*	4	18	16	6	4	43	5
Arthropoda	Insecta	Hemiptera	Veliidae	*Microvelia*	6						1
Arthropoda	Insecta	Coleoptera	Elmidae	*Neoelmis caesa*	4	7	24	9	3	6	
Arthropoda	Insecta	Trichoptera	Leptoceridae	*Oecetis*	5					1	

Taxa						Bull Creek @ Loop 360					
Phylum	Class	Order	Family	Genus/Species	Tolerance Value	2/11/00	3/24/00	4/28/00	5/18/00	6/8/00	7/3/00
Annelida	Oligochaeta				8	8	7		3	4	1
Arthropoda	Crustacea	Ostracoda			4	2	4	3	1	1	
Arthropoda	Insecta	Plecoptera	Perlidae	*Perlesta*	3	32	39	2			
Arthropoda	Insecta	Lepidoptera	Pyralidae	*Petrophila*	5				3	3	9
Mollusca	Gastropoda	Limnophila	Physidae	*Physella*	8	16	8	4			
Mollusca	Gastropoda	Limnophila	Planorbidae		6	11	10	4	4		3
Arthropoda	Insecta	Coleoptera	Psephenidae	*Psephenus*	4	2		3	6	17	23
Arthropoda	Insecta	Coleoptera	Scirtidae	*Scirtes*	7					1	
Arthropoda	Insecta	Diptera	Simuliidae	*Simulium*	6	552	5	1			
Arthropoda	Insecta	Trichoptera	Hydropsychidae	*Smicridea*	4			2			
Mollusca	Gastropoda	Pelecypoda	Sphaeriidae		8				1		
Arthropoda	Insecta	Coleoptera	Elmidae	*Stenelmis*	5	13	79	7	12	27	10
Arthropoda	Insecta	Diptera	Tabanidae	*Tabanus*	5						1
Arthropoda	Insecta	Ephemeroptera	Tricorythidae	*Tricorythodes*	4			1		2	9
				Total # of organisms		895	357	269	498	690	392
				Total # of Taxa		22	24	26	27	27	30

Taxa						Bear @ 1826			
Phylum	Class	Order	Family	Genus/Species	Tolerance Value	3/23/00	4/20/00	5/18/00	7/3/00
Arthropoda	Insecta	Coleoptera	Dytiscidae	*Agabus*	5	2			
Arthropoda	Insecta	Hemiptera	Naucoridae	*Ambrysus*	5		2	3	
Arthropoda	Insecta	Odonota	Coenagrionidae	*Argia*	7		9	37	11
Arthropoda	Insecta	Coleoptera	Hydrophilidae	*Berosus*	5	1	6	6	
Arthropoda	Insecta	Odonota	Libellulidae	*Brechmorhoga mendax*	6			1	5
Arthropoda	Insecta	Ephemoroptera	Caenidae	*Caenis*	7			1	
Arthropoda	Insecta	Ephemoroptera	Baetidae	*Callibaetis*	4	1			
Arthropoda	Insecta	Diptera	Stratiomyiidae	*Caloparyphus*	7	4			
Arthropoda	Insecta	Ephemoroptera	Baetidae	*Camelobaetidius*	4				5
Arthropoda	Insecta	Diptera	Ceratopogonidae		6	1	1		1
Arthropoda	Insecta	Trichoptera	Hydropsychidae	*Cheumatopsyche*	6		28	80	48
Arthropoda	Insecta	Trichoptera	Philopotamidae	*Chimarra*	4		39	130	84
Arthropoda	Insecta	Diptera	Chironomidae		6	116	17	35	33
Arthropoda	Insecta	Megaloptera	Corydalidae	*Corydalus*	5				2
Arthropoda	Insecta	Diptera	Culicidae		6	14		4	4
Platyhelminthes	Turbellaria	Tricladida	Planariidae	*Dugesia tigrina*	7				4
Arthropoda	Insecta	Ephemoroptera	Baetidae	*Fallceon quilleri*	6	18	11	56	85
Arthropoda	Insecta	Trichoptera	Helicopsychidae	*Helicopsyche*	5			1	
Arthropoda	Insecta	Odonota	Calopterygidae		6			1	1
Arthropoda	Insecta	Coleoptera	Dytiscidae	*Heterosternuta*	6	10	1	9	
Annelida	Hirudinea	Gnathobdellida			7			1	1
Arthropoda	Crustacea	Amphipoda	Talitridae	*Hyallela azteca*	8	3			
Arthropoda	Arachnoidea	Hydracarina			4	5	2		
Arthropoda	Insecta	Hemiptera	Hydrometridae	*Hydrometra*	4		1		
Arthropoda	Insecta	Trichoptera	Hydroptilidae	*Hydroptila*	6		7		
Arthropoda	Insecta	Coleoptera	Elmidae	*Macrelmis*	4	2			
Arthropoda	Insecta	Coleoptera	Elmidae	*Microcylloepus pusillus*	4	4	2		
Arthropoda	Insecta	Coleoptera	Elmidae	*Neoelmis caesa*	4	1		3	4
Arthropoda	Insecta	Trichoptera	Hydroptilidae	*Neotrichia*	3		4	3	10
Arthropoda	Insecta	Trichoptera	Leptoceridae	*Oecetis*	5			1	
Annelida	Oligochaeta				8	2	1	3	4

Taxa						Bear @ 1826			
Phylum	**Class**	**Order**	**Family**	**Genus/Species**	**Tolerance Value**	**3/23/00**	**4/20/00**	**5/18/00**	**7/3/00**
Arthropoda	Crustacea	Ostracoda			4	1			
Arthropoda	Insecta	Plecoptera	Perlidae	*Perlesta*	3	13	43	19	
Mollusca	Gastropoda	Limnophila	Physidae	*Physella*	8	5	18	6	2
Mollusca	Gastropoda	Limnophila	Planorbidae		6		3	3	2
Arthropoda	Insecta	Hemiptera	Veliidae	*Rhagoelia*	6			2	
Arthropoda	Insecta	Hemiptera	Gerridae	*Rheumatobates*	(n/a)			3	
Arthropoda	Insecta	Diptera	Simuliidae	*Simulium*	6	5			25
Arthropoda	Insecta	Coleoptera	Elmidae	*Stenelmis*	5	15	7	49	29
Arthropoda	Insecta	Ephemoroptera	Heptageniidae	*Stenonema*	6		3	4	4
Arthropoda	Insecta	Diptera	Tabanidae	*Tabanus*	5	4		1	
Arthropoda	Insecta	Ephemoroptera	Leptophlebiidae	*Thraulodes gonzalesi*	4				12
Arthropoda	Insecta	Diptera	Tipulidae	*Tipula*	4				1
Arthropoda	Insecta	Ephemoroptera	Leptophlebiidae	*Traverella presidana*	2				2
Arthropoda	Insecta	Ephemoroptera	Tricorythidae	*Tricorythodes*	4				1
Arthropoda	Insecta	Coleoptera	Hydrophilidae	*Tropisternus (larva)*	8	2		1	
				Total # of organisms		229	205	463	380
				Total # of Taxa		22	20	27	25

BIBILIOGRAPHY:

Allen, J.D. 1995. Stream Ecology: Structure and function of running waters. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Angradi, T. R. 1997. Hydrologic context and macroinvertebrate community response to floods in an Appalachian headwater stream. American Midland Naturalist 138:371-386.

Baker, V.R. 1977. Stream channel response to floods, with examples from Central Texas. Geological Society of America Bulletin 88:1057-1071.

Barbour, M.T. and J.B. Stribling. 1993. A technique for assessing stream habitat structure. Pages 156-178 *in* Conference Proceedings: Riparian ecosystems in the humid U.S., March 15-18, 1993, Atlanta, Georgia.

Barbour, M.T., J.B. Stribling and J.R. Karr. 1994. Multimetric approach for establishing biocriteria and measuring biological condition. Pages 63-77 *in* W.S. Davis and T.P. Simon (editors). Biological Assessment and Criteria. Tools for Water Resource Planning and Decision-Making. Lewis (CRC) Publishers, Boca Raton, Florida.

Barbour, M.T., J. Gerritson, B.D. Snyder and J.B. Stribling. 1998. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. 2nd Edition. EPA/841/B/98-010. Office of Water, US Environmental Protection Agency, Washington, D.C.

Barrett, M.E., A.M. Quenzer and D.R. Maidment. 1998. Water quality and quantity inputs for the urban creeks future needs assessment. Center for Research in Water Resources Bureau of Engineering Research. University of Texas at Austin.

Berner, L. and M.L. Pescador.1988. The Mayflies of Florida. Revised Edition. University Presses of Florida. Tallahassee/Gainesville, USA.

Britton, D.L., J.A. Day and M.P. Henshall-Howard. 1993. Hydrochemical response during storm events in a South African mountain catchment: the influence of antecedent conditions. Hydrobiologia 250:143-157.

Booth, D.B. 1990. Stream channel incision following drainage-basin urbanization. Water Resources Bulletin 26:407-417.

City of Austin, Environmental Resource Management Division. 1993. Bull Creek Watershed Study: Cumulative impacts of development on water quality and endangered species in the Bull and West Bull Creek watersheds. Water Quality Report Series COA-ERM/WRE 1993-03. Austin, TX.

City of Austin, Environmental Resource Management Division. 1996. Bioassessment strategies for non-point source polluted creeks. Water Quality Report Series COA-ERM/WRE 1996-01. Austin, TX.

Clausen, B. and B.J.F. Biggs. 1997. Relationships between benthic biota and hydrological indices in New Zealand Streams. Freshwater Biology 38:327-342.

Colwell, R.K. 1974. Predictability, constancy and contingency of periodic phenomena. Ecology 55:1148-1153.

Cummins, K.W. 1962. An evaluation of some techniques for the collection and analysis of benthic samples with a special emphasis on lotic waters. American Midland Naturalist 67:477-504.

Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. Science 199:1302-1310

Davies, S.P., L. Tsomides, J.L. DiFranco and D.L. Courtemanch. 1999. Biomonitoring retrospective: Fifteen year summary for Maine rivers and streams. DEPLW1999-26. Department of Environmental Protection, State of Maine. Augusta, Maine.

Davis, W.S., B.D. Snyder, J.B. Stribling and C. Stoughton. 1996. Summary of state biological assessment programs for streams and wadeable rivers. EPA 230-R-96-007. U.S. Environmental Protection Agency; Office of Policy, Planning and Evaluation; Washington, DC.

Death, R.G. and M.J. Winterbourn. 1994. Environmental stability and community persistance: a multi-variate perspective. Journal of the North American Benthological Society 13:125-139.

Death, R.G. and M.J. Winterbourn. 1995. Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. Ecology 76:1446-1460.

Delucchi, C.M. 1988. Comparison of community structure among streams with different temporal flow regimes. Canadian Journal of Zoology 66:579-586.

Diamond, J. 1986. Overview:laboratory experiments, field experiments and natural experiments. Pages 3-22 *in* J. Diamond and T.J. Case (editors). Community Ecology. Harper and Row, New York.

Dole-Olivier, M.J., P. Marmonier and J.L. Beffy. 1997. Response of invertebrates to lotic disturbance: is the hyporheic zone a patchy refugium? Freshwater Biology 37:257-276.

Elliot A.G., W.A. Hubert, and S.H. Anderson. 1997. Habitat associations and effects of urbanization on macroinvertebrates of a small, high-plains stream. Journal of Freshwater Ecology 12:61-73.

Epler, J.H. 1996, Identification manual for the water beetles of Florida. Florida Department of Environmental Protection, Tallahassee, Florida.

Feminella, J.W. 1996. Comparison of benthic macroinvertebrate assemblages in small streams along a gradient of flow permanence. Journal of the North American Benthological Society 15:651-669.

Feminella, J. W. 2000. Correspondence between stream macroinvertebrate assemblages and 4 ecoregions of the southeastern US. Journal of the North American Benthological Society, 19:442 – 459.

Finkenbine, J.K., J.W. Atwater and D.S. Mavinic. 2000. Stream health after urbanization. Journal of the American Water Resources Association 36:1149-1160.

Flecker, A.S. and B. Feifarek. 1994. Disturbance and the temporal variability of invertebrate assemblages in two Andean streams. Freshwater Biology 31:131-142.

Gandara, S.C., W.J. Gibbons, F.L. Andrews, J.C. Fisher, B.A. Hinds and R.E. Jones. 1995. Water Resources Data – Texas, Water Year 1995, Volume 3, USGS-WDR-TX-95-3. United States Geologic Survey, Water Resources Division, Austin TX.

Gordon, N.D., T.A. McMahon and B.L. Finlayson. 1992. Stream Hydrology: An Introduction for Ecologists. John Wiley & Sons, N.Y., N.Y.

Harrelson, C.C., C.L. Rawlins and J.P. Potyondy. 1994. Stream channel reference sites: An illustrated guide to field technique. USDA Forest Service, General Technical Report RM-245. Fort Collins, CO.

Hornig, C.E., C.W. Bayer, S.R. Twidwell, J.R. Davis, R.J. Kleinsasser, G.W. Linam and K.B. Mayes 1995. Development of regionally based biological criteria in Texas. Pages 145-152 *in* S.W Davis and T.P. Simon (editors). Biological Assessment and Criteria: Tools for Water Resource Planning and Decision-Making. Lewis (CRC) Publishers, Boca Raton, Florida.

Hughes, R.M. 1994. Defining acceptable biological status by comparing with reference conditions. Pages 31-47 *in* S.W Davis and T.P. Simon (editors). Biological Assessment and Criteria: Tools for Water Resource Planning and Decision-Making. Lewis (CRC) Publishers, Boca Raton, Florida.

Hurlbut, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Mongraphs, 54:187-211.

Hynes, H.B.N. 1972. The Ecology of Running Waters. University of Toronto Press, Toronto, Canada.

Hynes, H.B.N. 1960. The Biology of Polluted Waters. University of Toronto Press, Toronto, Canada.

Iverson, T.M., P. Wiberg-Larson, S. B. Hansen and F.S. Hansen. 1978. The effects of partial and total drought on the macroinvertebrate communities of three small Danish streams. Hydrobiologia 60:235-242.

Jowett I.G. and M.J. Duncan. 1990. Flow variability in New Zealand rivers and its relationship to in-stream habitat and biota. New Zealand Journal of Marine and Freshwater Research 24:305-317.

Johnson, J.K, T. Wiederholm, and D.M. Rosenberg. 1993. Freshwater biomonitoring using individual organisms, populations, and species assemblages of benthic macroinvertebrates. Pages 40-158 *in* D. M Rosenberg and V.H. Resh (editors). Freshwater biomonitoring and benthic macroinvertebrates. Chapman & Hall. N.Y, N.Y.

Karr, J.R and E.W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington, D.C.

Klein, R.D. 1979. Urbanization and stream quality impairment. Water Resources Bulletin 14:948-963.

Lake, P.S, T.J. Doeg and R. Marchant. 1989. Effects of multiple disturbance on macroinvertebrate communities in the Acheron River, Victoria. Australian Journal of Ecology 14:507-514.

Lake, P.S. 2000. Disturbance, patchiness, and diversity in streams. Journal of the North American Benthological Society 19:573-592.

Lenat, D.R. and J.K. Crawford. 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia 294:185-199.

Merritt, R.W. and K.W. Cummins. 1996. An Introduction to the Aquatic Insects of North America . Kendall/Hunt Publishing Co., Dubuque, Iowa.

Miller, A.M and S.W. Golladay. 1996. Effects of spates and drying on macroinvertebrate assemblages of an intermittent and a perennial prairie stream. Journal of the North American Benthological Society 15:670-689.

Minshall G.W. and R.C. Peterson Jr. 1985. Towards a theory of macroinvertebrate community structure in stream ecosystems. Archiv fur Hydrobiologie 104:49-76.

Omernick, J.M. 1987. Ecoregions of the conterminous United States. Supplement to the Annals of the Association of American Geographers, 77:118-125.

Pennak, R.W. 1989. Freshwater invertebrates of the United States: Protozoa to Mollusca. John Wiley & Sons, Inc. NY, NY.

Parsons, M. and R.H. Norris. 1996. The effect of habitat specific sampling on biological assessment of water quality using a predictive model. Freshwater Biology 36:419-434.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, R.M. Hughes. 1989. Rapid Bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. U. S. Environmental Protection Agency, Office of Water. EPA/44/4-89-001.

Poff, L.N., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime: a paradigm for river conservation and restoration. Bioscience 47:769-784.

Poff, L.N. and J.D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76:606-627.

Poff, L.N. and J.V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: A regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46:1805-1817.

Pratt, J.M., R.A. Coler and P.J. Godfrey. 1981. Ecological effects of urban stormwater runoff on benthic macroinvertebrates inhabiting the Green River, Massachusetts. Hydrobiologia 83:29-42.

Quinn, J.M. and C.W. Hickey. 1994. Hydraulic parameters and benthic invertebrate distributions in two gravel-bed New Zealand rivers. Freshwater Biology 32:489-500.

Rabeni, C.F., N. Wang and R.J. Sarver. 1999. Evaluating adequacy of the representative stream reach used in invertebrate monitoring programs. Journal of the North American Benthological Society 18:284-291.

Resh, V.H. and J.K. Jackson. 1993. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. Pages 195-233 *in* D. M Rosenberg and V.H. Resh (editors). Freshwater biomonitoring and benthic macroinvertebrates. Chapman & Hall. N.Y, N.Y.

Resh, V.H. and E.P. McElravy. 1993. Contemporary quantitative approaches to biomonitoring using benthic macroinvertebrates. Pages 159-194 *in* D. M Rosenberg and V.H. Resh (editors). Freshwater biomonitoring and benthic macroinvertebrates. Chapman & Hall. N.Y, N.Y.

Resh, V.H., A.V. Brown, A.P. Covich, M.E. Gurtz, H.W. Li, G.W. Minshall, S.R. Reice, A.L. Sheldon, J.B. Wallace and R.C. Wissmar. 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society, 7(4): 433-455.

Richter, B. D., J.V. Baumgartner, J. Powell and D.P. Braun. 1996. A Method for assessing hydrologic alteration within ecosystems. Conservation Biology 10:1163-1174.

Richter, B. D., J.V. Baumgartner, R. Wigington and D.P. Braun. 1997. How much water does a river need? Freshwater Biology 37:231-249.

Rosenberg, D. M. and V.H. Resh. 1996. Use of aquatic insects in biomonitoring. Pages 87-97 *in* R.W Merrit and K.W. Cummins (editors). An Introduction to the Aquatic Insects of North America . Kendall/Hunt Publishing Co., Dubuque, Iowa.

SAS. 1990. The SAS system for Windows. SAS Institute Inc. Cary, North Carolina.

Scarsbrook, M.R. and C.R. Townsend. 1993. Stream community structure in relation to spatial and temporal variation: a habitat template study of two contrasting New Zealand streams. Freshwater Biology 29:395 –410.

Schueler, T. and R. Claytor. 1997. Impervious cover as an urban stream indicator and a watershed management tool. Pages 513-529 *in* L. Roesner (Editor). Effects of Watershed Development and Management on Aquatic Ecosystems. American Society of Civil Engineers, NY, NY

Schueler,T. 1994. The importance of imperviousness. Watershed Protection Techniques 1 :100-111.

Scrimgeour, G.J and M.J. Winterbourn. 1989. Effects of floods on epilithon and benthic macroinvertebrate populations in an unstable New Zealand river. Hydrobiologia 171:33-34.

Snedecor, G.W. and W.G. Cochran. 1989. Statistical Methods. 8th Edition. Iowa State University Press. Ames, Iowa.

Southerland, M.T., and J.B. Stribling. 1995. Status of biological criteria development and implementation. Pages 81-96 *in* W.S. Davis and T.P. Simon (editors).

Biological assessment and criteria: tools for water resource planning and decision making. Lewis Publishers/CRC Press. Boca Raton, FL.

Spindler, P. 1996. Using ecoregions for explaining macroinvertebrate community distribution among reference site in Arizona, 1992. OFR 95-7, Water Quality Division, Arizona Department of Environmental Quality. Phoenix, Arizona.

Statsoft, Inc. 1995. STATISTICA for Windows. 2300 East 14th Street, Tulsa, OK 74104.

Thorp, J.H. and A.P. Covich. 1991. Ecology and classification of North American freshwater invertebrates. Academic Press, Inc. San Diego, CA.

Tikkanen, P., P. Laasonen, T. Muotka, A. Huhta and K. Kuusela. 1994. Short-term recovery of benthos following disturbance from stream habitat rehabilitation. Hydrobiologia 273:121-130.

Townsend, C.R., S. Doledec and M.R. Scarsbrook. 1997. Species traits in relation to temporal and spatial heterogeneity in streams: a test of habitat templet theory. Freshwater Biology 37:367-387.

Townsend, C.R. and M.R. Scarsbrook. 1997. Quantifying disturbance in streams: alternative measures of disturbance in relation to macroinvertebrate species traits and species richness. Journal of the North American Benthological Society 16:531-544.

Tufte, E.R. 1983. The visual display of quantitative information. Graphics Press. Cheshire, CT.

Tukey, J.W. 1977. Exploratory data analysis. Addison-Wesley Inc. Reading, MA.

U.S. Environmental Protection Agency. 1997. Urbanization and Streams: Studies of hydrologic impacts. U.S. Environmental Protection Agency, Office of Water. 841-R-97-009. Washington, D.C.

Ward, J.V. and J.A. Stanford. 1983. The intermediate-disturbance hypothesis: an explanation for biotic diversity patterns in lotic ecosystems. Pages 347-355 *in* T.D Fontaine and S.M. Bartell (editors). Dynamics of Lotic Ecosystems. Ann Arbor Science Publishers, Ann Arbor, Michigan.

Wiggins, G.B. 1996. Larvae of the North American Caddisfly genera (Trichoptera). 2nd Edition. University of Toronto Press Inc. Toronto, Canada.

Williams, D.D. 1996. Environmental constraints in temporary fresh waters and their consequences for the insect fauna. Journal of the North American Benthological Society 15:634-650.

VITA

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