

## ABSTRACT

### Coupling Changes in Physical Habitat and Fish Community Structure in Central Texas Streams with Interannual Variability in Stream Discharge

Charles E. Stanley, M.S.

Committee Chairperson: Ryan S. King, Ph.D.

Hydrological variability and instream habitat connectivity play predominant roles in governing fish communities in lotic ecosystems. Hydrologic extremes of drought (2006) and flood (2007) events were the backdrop for fish assemblage and physical habitat data collected during summer in 28 central Texas streams. I evaluated the correspondence between the magnitude of physical habitat and fish community composition change in stream reaches sampled in these two contrasting years using multivariate statistics and ordination techniques. Streams characterized by disconnected pools had different fish community structure and different habitat characteristics than streams that had habitats connected by flowing water. The amount of interannual change in both fish community structure and habitat characteristics was greatest between streams that had disconnected pools in 2006 and their paired samples in 2007. Indicator species analysis revealed indicator species of disconnected and connected habitat types whose life histories give us information about the biotic interactions within these habitats.

Coupling Changes in Physical Habitat and Fish Community Structure in Central Texas  
Streams with Interannual Variability in Stream Discharge

by

Charles E. Stanley, B.S.

A Thesis

Approved by the Department of Biology

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Robert D. Doyle, Ph.D., Chairperson

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of  
Master of Science

Approved by the Thesis Committee

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Ryan S. King, Ph.D., Chairperson

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Robert D. Doyle, Ph.D.

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Joseph D. White, Ph.D.

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Stephen I. Dworkin, Ph.D.

Accepted by the Graduate School  
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J. Larry Lyon, Ph.D., Dean

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## CHAPTER ONE

### Introduction

Variation in discharge, the amount of surface water passing a given point in a river or stream within a given interval of time, has been called the “master variable” (Richter 2003) in freshwater lotic systems. It affects nearly all aspects of river ecosystem functions including aquatic life and its related habitat, nutrient cycling, sediment transport, water temperature, riverbank stability, groundwater recharge, and many others (Allan 1995). There is a strong association between both functional and taxonomic composition of fish assemblages and hydrologic regimes at a regional scale (Poff 1995) because variability in discharge can lead to rapid changes in environmental conditions, such as temperature, turbidity, and salinity (Ostrand and Wilde 2002), which can greatly limit the amount and type of biota that can persist in these environments.

Fish species composition in freshwater streams is regulated by the effects of abiotic (density-independent) factors, such as discharge and habitat connectivity, and local biotic (density-dependent) factors, such as predation and competition, on the instream biota (Taylor 1997), and the relative importance of each of these two factors in structuring fish assemblages varies longitudinally (Ostrand and Wilde 2002). Environmental conditions and fish assemblages in lower-order, headwater streams are considered highly variable. This variability can be accounted for by the typical hydrology of headwater streams; smaller, low-volume, high gradient, and prone to disturbances and intermittence. In these streams, fish assemblage structure is thought to

be controlled primarily by abiotic factors and frequency of disturbances such as flood and drought (Echelle et al 1972; Matthews and Styron 1981; Meador and Matthews 1992; Taylor et al 1993; Taylor 1997; Herbert and Gelwick 2003). As a result, predominant fish species in this area are physiologically tolerant species capable of surviving stressful conditions, are able to move in response to changing conditions, and/or able to rapidly recolonize areas of local extinction caused by intermittence, some moving long distances in relatively short time periods into previously dry reaches (Winemiller 1989, Fausch and Bramblett 1991).

In contrast, environmental conditions and fish assemblages in higher-order, lowland streams are considered more stable and more resistant to hydrological extremes. A greater number of species are able to coexist in this more stable habitat including larger insectivorous and piscivorous species of fish. In these streams, biotic factors such as predator-prey interactions and competition may be relatively more important in structuring fish assemblages than in lower order streams (Horwitz 1978; Matthews and Styron 1981). Seasonal variability in physiochemical parameters is generally reported to be a strong correlate to fish assemblages; however, this may not be the case in central Texas where summers are warm and winters are mild, in contrast to temperate regions of northern USA where seasonal conditions are more discrete and have a more profound influence on fish assemblages. Here, local instream variables and disturbance frequency outweigh that which is uniquely caused by seasonal variation (Ostrand and Wilde 2002, Herbert and Gelwick 2003, Li and Gelwick 2005).

While many previous studies have examined the longitudinal change of abiotic and biotic variables along the river continuum (*sensu* Vannote et al. 1980),



disproportionately fewer studies examine the effect of drying within streams leading to differing assemblages in intermittent perennial pools during drought disturbance events. Tolerances of individual species to increasing abiotic environmental stressors, such as high temperature and low dissolved oxygen, as perennial pools are formed vary. In addition, during these intermittent conditions each pool is subject to local extinctions due to biotic interactions between organisms in the pool at the time that habitat connectivity was lost. This lack of connectivity prevents immigration or emigration of fishes and the continuing reduction of pool volume intensifies the biotic interactions as population density increases (Taylor 1997). In a similar ecosystem, Zeug et al (2005) found that periodic connection between the main river channel of the upper Brazos River, Texas and oxbow lakes of varying depth and permanence was one of the main factors controlling the species assemblage and diversity within these two environments. Oxbow lakes in this instance are a parallel of the dynamic of summer drying of streams resulting in intermittent pools with connectivity to source populations being an important determinant of assemblage structure in aquatic systems.

My study is concerned with the relative importance of abiotic and biotic controls on fish assemblages as well as temporal variation in physiochemical habitat variables and fish assemblages of 28 central Texas stream reaches between sampling events, which differed greatly in their discharges and subsequent physiochemical habitat variables due to interannual difference in precipitation. Temperatures and precipitation amounts immediately before and during our index sampling period differed a great deal between 2006 and 2007 sampling events; The summer of 2006 was the hottest summer on record

as well as a time of severe drought, whereas 2007 was the wettest summer on record for the state of Texas resulting in devastating flooding (<<http://www.ncdc.noaa.gov/oa/ncdc.html> >) Historically, as flow recedes during summer months, streams in the middle Brazos River and upper Trinity River basins become intermittent often forming isolated pools within the streambed (Echelle et al. 1972; Wilde and Ostrand 1999). This effect was observed during the 2006 collection period in 16 of the 28 stream reaches sampled, but in none of the sites sampled in 2007. Stream reaches were classified into *a priori* groups based upon this habitat connectivity criterion, determined by discharge measurements (Appendix A-1) and photographic evidence. This variation presents a unique opportunity to examine the relationship between extremes of hydrologic variability and fish assemblages in central Texas streams. Thus, using this unique data set that spans two sequential years of contrasting hydrological conditions, I address the following five questions:

1. Is there a difference in quantifiable habitat characteristics of stream reaches between 2006 and 2007 within and among groups of streams?
2. Is there a difference in fish community composition between 2006 and 2007 within and among groups of streams?
3. Is there a different pattern of change in fish community structure between years for sites that were intermittent in 2006 when compared sites that had measurable discharge in 2006?
4. Are there differences in the predictability of composition, measured as dispersion in ordination space, due to contrasting hydrological conditions?
5. Are there species of fish associated with groups that can be used as indicators of these groups?

## CHAPTER TWO

### Methods

#### *Site Selection*

The Trinity River and Brazos River drainage basins are two of the main drainage basins in central Texas. These drainage basins span several ecoregions, including the Cross Timbers ecoregion, which encompasses the entirety of our study area. This ecoregion is a transitional zone between the once prairie, now winter wheat-growing, regions to the west and the forested low mountains or hills of eastern Texas. A mosaic of forest, woodland, savanna, and prairie characterizes this ecoregion (Griffith et al 2004).

Stream reaches were selected from wadeable streams and river tributaries throughout this ecoregion (Figure 1) based on their accessibility, aesthetics, dissolved nutrient content, stream order, and sampling ability. Streams were sampled once in 2006 and once in 2007 during an index sampling period (1 May – 15 October) defined by TCEQ (2003). Thirty-eight stream reaches were sampled in 2006; however, due to high discharges during the index sampling period only 28 streams could be sampled during the 2007 field season. Because this study is concerned with interannual variation, it is limited to only the 28 sites that were sampled during both years of data collection. Lengths of stream reaches were determined by multiplying the estimated mean wetted width within the stream reaches by 40 with a maximum stream reach length of 500 m. Each stream reach was divided into 5 or 6 equidistantly spaced transects throughout the reach based upon the width ( $X \leq 300\text{m}$  = five transects,  $300 < X \leq 500$  = six transects).

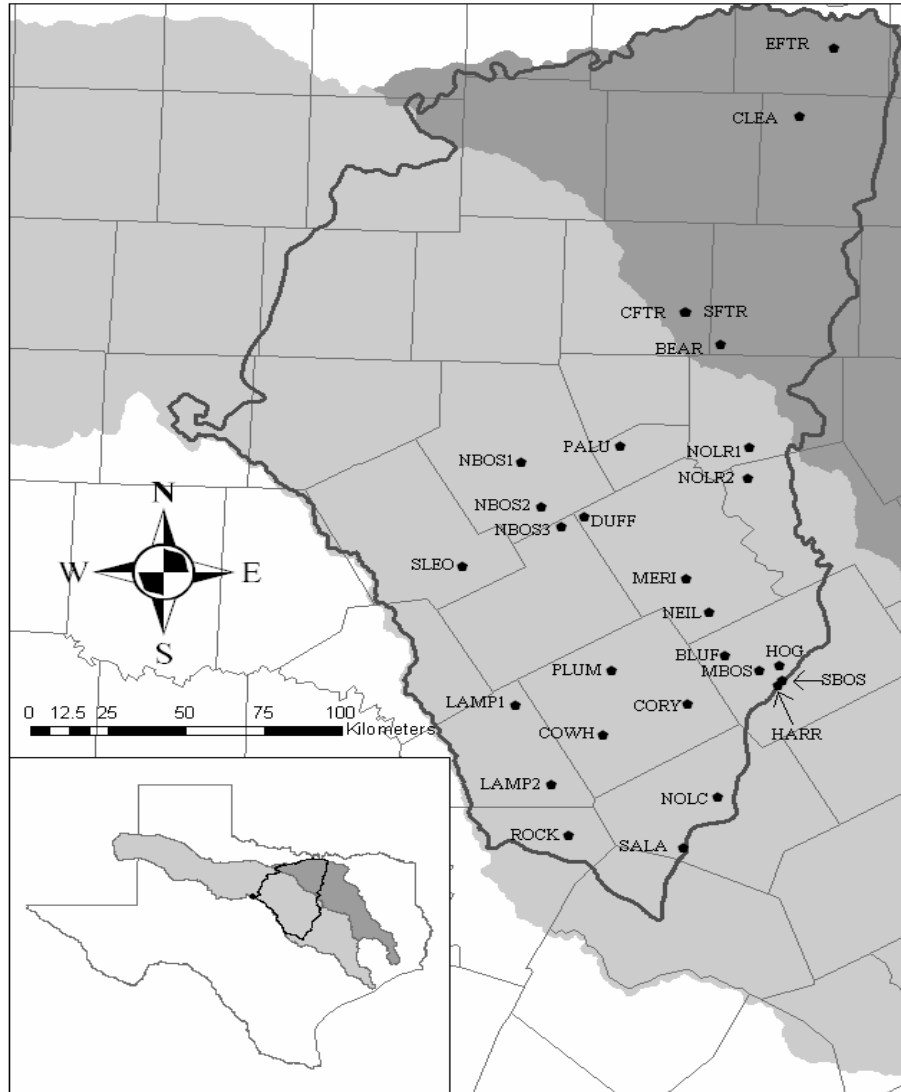


Figure 1: Map of study area. The Cross Timbers ecoregion is outlined. The shaded areas represent the Brazos (lighter) and Trinity (darker) drainage basins. Map resolution was unable to separate the spatially close CFTR and SFTR site locations resulting in one point on the map for these two sites. For explanation of site codes, see Appendix A-1.

### *Habitat Data Collection*

A suite of physiochemical parameters were measured or quantified using a variety of techniques and instruments. Dissolved oxygen concentration/ percent saturation, salinity, specific conductivity, water temperature, and pH were measured using an YSI 556 Multi-Probe Meter with Barometer; Yellow Springs Instruments. Stream discharge

was measured using a Flomate™ Marsh McBirney Model 2000 portable electromagnetic current meter at a point within the stream reach that had optimal conditions for the instrument. Total reach length variables such as percent riparian cover classes, overall aesthetics, and total number of stream features such as riffles, pools, and stream bends were quantified based on sampling protocol criteria. Reach length variables such as percent riparian cover classes, overall aesthetics, and total number of stream features such as riffles, pools, and stream bends were quantified based on sampling protocol criteria. At each of the 5 or 6 transects, mean tree canopy cover was quantified using a Forest Densiometers' Model-A Convex Mirror Spherical Densiometer with a total of 4 measurements being taken at either bank and facing either direction at the approximate center of the transect. Bank angles for both stream banks at each transect were measured using a Suunto PM-5/360PC Clinometer. Depth was measured at 11 equidistant locations across each transect by meter stick, and mean depth for each transect was calculated from these measurements. Other transect level variables such as percent substrate types, percent in-stream habitat types, mean depth, bank erosion potential, and algal/ macrophyte abundances were measured. All transect variables were averaged for each stream reach for data analysis. For a more detailed description of habitat sampling methods, see TCEQ protocols (2003).

### *Fish Collection*

Specimens were collected by seine netting as well as electrofishing using 5 mm mesh size seines (9 m or 4.5 m)X 2 m with leads spaced every 0.15 m and a Smith Root Inc. LR-24 electrofisher (300-600 volts, 65hz, and 25% duty cycle in accordance with TCEQ protocol, manufacturer's recommendations, and stream conductivity),

respectively. Each stream reach was electrofished for at least 900 seconds of trigger time but with sufficient trigger time to collect around most submerged structures within the reach that were either unable to be seined or were deemed likely habitat for fish such as submerged logs, vegetation, undercut banks, and boulders (maximum trigger time never to exceed 3000 sec.).

After electrofishing was complete, the fish were quarantined in live wells while the reach was seined to prevent duplicate collections of individual fish. Stream reaches were seined six times with each haul was at least 10 m long or instead was a group of riffle-kicks (3-5 separate kicks depending on riffle size and complexity) where the netting was placed within a riffle and the substrate was disturbed upstream of the netting in order to force fish out of interstitial spaces in the substrate and into the seine for collection. Once completed, all fish specimens were identified and enumerated separately for the electrofishing and seining collections and most were released back into the stream. Voucher specimens, rare, and/or uneasily identifiable fishes were preserved in 10% buffered formalin solution and taken to lab for identification (TCEQ 2003). Individual fish were classified as either adult or juvenile based upon size classes given in regional fish identification keys (Trautman 1981; Mettee 1996; Ross et al 2001; Thomas 2007) and were counted separately. For analyses, total counts of individuals of each species were combined for both sampling methods and age classes to give a single count per species per stream reach per sampling date.

#### *Data Analyses*

Data analyses were conducted using PC-ORD 5.11 statistical analysis software package (MJM Software, Gleneden Beach, OR). Streams that had connected habitat,

allowing movement of fish throughout the reach during 2006 sampling and with total stream discharge greater than  $0.001 \text{ m}^3 \cdot \text{s}^{-1}$  were classified as having connected instream habitat (2006 C). Streams that had disconnected perennial pools, preventing movement of fish throughout the reach during 2006 sampling, with discharge less than  $0.001 \text{ m}^3 \cdot \text{s}^{-1}$  were classified as having disconnected instream habitat (2006 DC). All stream reaches sampled in 2007 had connected habitat throughout the stream reaches and discharge greater than  $0.001 \text{ m}^3 \cdot \text{s}^{-1}$ . Stream reaches sampled in 2007 were classified by whether habitat was previously connected (2007 PC) or previously disconnected (2007 PDC) during the 2006 sampling period.

PCA (Principle Components Analysis; Pearson 1901 and McCune and Grace 2002) conducted on a stream reach (28 sites X 2 years = 56 sites/ rows) and habitat variable (49 habitat variables/ columns) correlation matrix was used to ordinate stream reaches in 2-dimensionsal habitat variable space. PCA is an appropriate statistical technique for reducing a multivariate data set to few key, synthetic variables that describe most of the shared variance among variables. However, this is only true so long as there are approximate linear relationships existing between the variables or by using data transformation to achieve an approximate linear relationship. These statistical requirements were met with square root data transformations of environmental variables and using the correlation matrix as the basis for PCA, which relativizes variables to standard deviate for comparison. McCune and Grace (2002) state that although PCA is often misused in describing heterogeneous ecological assemblage data, like the fish species composition data of this study, it is particularly useful for the analysis of the

combination of climatic, hydrologic and environmental variables for multivariate data sets.

NMS (Non-metric Multidimensional Scaling; Mather 1976, Kruskal 1964, and McCune and Grace 2002) was conducted on a stream reach (28 sites X 2 years = 56 sites/ rows) and species composition (34 species/ columns) dissimilarity matrix to ordinate stream reaches in 2-dimensional fish community space. Prior to analysis, species abundances were  $\log_{10}(x + 1)$  transformed in order to reduce the influence of highly abundant, small-bodied species on the ordination. NMS is the most generally effective ordination for ecological species composition data and has two key advantages. First, NMS avoids the assumption of linear relationships among variables necessary for PCA. Second, it allows the investigator to choose the distance measure that is most relevant to the ecological question (Clarke 1993, McCune and Grace 2002).

Once PCA and NMS ordinations were completed, I tested for differences in composition among the four hydrological groups and the dispersion of each group using two statistical techniques. First, MRPP (Multivariate Response Permutation Procedure; Mielke 1984) was used to describe differences in habitat and community structure using Euclidean distances among sites in PCA and NMS ordination space, respectively. This technique is a multivariate analogue to a Student's T-test of difference between groups. MRPP is used to determine the significance of separations of groups in ordination space and thus significant differences in either fish community structure or habitat characteristics between groups of streams. Coordinates from both PCA and NMS ordination plots were exported separately into a data matrix (28 sites X 2 years = 56 sites/ rows by 3 dimensions' coordinates/ columns) on which MRPP could be performed.



Groups were weighted using a natural weighting as recommended by Mielke (1984). Pairwise comparisons were made between the locations of group centroids in both PCA and NMS ordination space to determine the significance of differences in-group habitat or community composition.

Second, the statistical program PERMDISP (PERMutational analysis of multivariate DISPersion; Anderson 2006a) was used to determine if the dispersions (degree of variability among sites) of each group around their group centroid were significantly different from one another. PERMDISP is useful in determining if the separations between groups indicated by MRPP are affected by differences in the dispersions of groups. This technique is a multivariate analogue to Levene's test as an ANOVA on the Euclidean distances of individual observations to their group centroid (Anderson 2006a). Anderson (2006b) also suggests that the overall dispersion of a group measured as average distance (or dissimilarity) from an individual sampling unit to the group centroid may be used as a multivariate measure of community beta diversity. I hypothesized that DC sites would have greater dispersion (beta diversity) due to increased biotic interactions, particularly predation, within disconnected perennial pools causing a decoupling of fish species composition from habitat variables.

It was also necessary to examine the dispersions of groups in ordination space and using PERMDISP because MRPP distance comparisons can be sensitive to between-group differences in dispersions. Larger group dispersions can be interpreted as having more variation in either habitat or species composition than smaller groups. Larger groups are also more likely to overlap with other groups due to their size resulting in reduced separation among groups.

Successional vector diagrams were used to visualize patterns in change of habitat variables and species composition between annual sampling events for both PCA and NMS ordinations, respectively. These diagrams show the direction and magnitude of change in both fish community composition and habitat characteristics between interannual pairs of sites. Each interannual pair of sites were translated to origin, meaning that all 28 sites sampled in 2006 are located at (0, 0) of the 2-dimensional successional vector diagram (McCune and Grace 2002). The corresponding paired 2007 sites were graphed in 2-dimensional space around the origin and connected with vector lines so that direction and magnitude of interannual change in ordination space were visualized. The Pythagorean Theorem was used to calculate vector lengths between interannual pairs of sites from successional vector diagram coordinates exported to Microsoft Excel. A one-tailed Student's T-test comparing group means was used to determine if vector lengths for interannually paired connected and disconnected sites were significantly different. In addition, coordinates for successional vector diagrams were exported into a PC-ORD data matrix and used as the basis for MRPP comparisons. This MRPP analysis addresses a different question than the previous in that this MRPP is concerned with the difference in magnitude and direction of interannual change in habitat condition and species composition.

Finally, ISA (Indicator Species Analysis; Dufrêne and Legendre 1997) was used to contrast the affinity of different species to groups of varying environmental condition. ISA assigns an indicator value (IV) to each taxon by calculating the product of the relative frequency (% occurrence of a taxon among sample units in each group) and relative abundance (% of the total abundance of a taxon in each group) of each species to

a group. The probability of achieving an equal or larger IV value among groups ( $p$ ) was estimated using 999 random permutations of the original data (Dufrêne and Legendre 1997). Species with significantly ( $p \leq 0.05$ ) high IVs for a given group likely found in other regional streams of similar environmental conditions and suggest a hydrological effect on that species. ISA was used determine group indicator values for all species included in analyses.

## CHAPTER THREE

### Results

#### *Physical Habitat Analyses*

Principal components 1 and 2 accounted for 32.7 % of the cumulative variance explained by PCA for the 49 habitat variables (Table 1). PC1 and PC2 are synthetic axes, such that the ordination of points along these axes describe hydrologic, geologic and biotic aspects of habitat variation in our study area simultaneously. Points that are spatially near one another in PCA ordination space have more similar habitat condition than distant points. PC1 and PC2 were labeled with the two strongest eigenvectors for the positive and negative direction for each (Figure 2). Greater values of PC1 were indicative of lower discharge streams with bedrock substrates and heavier macrophyte and filamentous algae growth. Lower values of PC1 were indicative of flowing streams with greater mean depths and sandy substrates. PC2 was primarily a hydrologic variable axis with higher values corresponding to flowing streams with deeper pools and more riffles. Lower values corresponded to habitats that had less surface water that was slower moving with erosional banks resulting in muddy substrates.

The 2006 DC group centroid was significantly separated from all other groups in PCA space according to MRPP pairwise comparisons (Figure 2, Table 4) with the greatest separation from 2007 PDC. The location of this group in PCA space suggests that habitat conditions among streams indicative of this group are characterized by comparatively higher macrophyte and filamentous algae abundance, bedrock substrates,

and, by group definition, no surface flow between disconnected perennial pools. The 2006 C group was not significantly different from either 2007 group. These remaining groups' position in PCA space suggest conditions of higher discharge, mean depth, and pool depths than 2006 DC sites along with smaller substrate particle sizes (sand, mud, and silt).

PERMDISP analysis of dispersion showed no significant differences between any pairwise comparison dispersion of groups around their centroid in PCA habitat space, suggesting that differences in MRPP were due to differences in habitat structure among groups alone. PERMDISP and MRPP analyses conducted upon the location of stream sites in the PCA successional vector plot revealed no difference in dispersion between interannual paired hydrologic groupings but a significant separation between streams with disconnected and streams with connected in stream habitats. This separation can be interpreted as disconnected sites having a greater amount of change in habitat condition from 2006 to 2007 than connected sites. Additionally, one-tailed T-test comparing group mean successional vector lengths between interannually paired connected and disconnected habitats confirmed that the vector lengths were significantly longer for the disconnected habitat pairs of sites than the connected (Figure 3, Table 2) indicating a greater amount of interannual change among disconnected habitats.

### *Fish Community Analyses*

Fish that were included in analyses totaled 38,897 individuals from 34 species, 23 genera, 12 families, and 7 orders for 2006 and 2007 collections (Table 1, Appendix A-2). A small number of individuals of *Ameiurus melas*, *Astyanax mexicanus*, *Ictiobus*

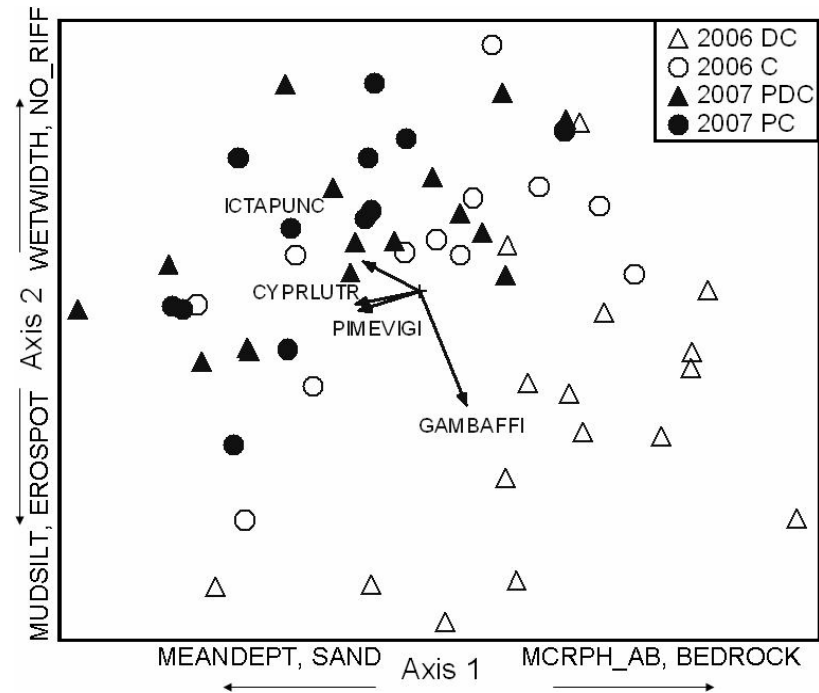


Figure 2: Configuration of stream reaches in 2-dimensional PCA representation of habitat variable correlation matrix. Joint bi-plot overlay shows direction and magnitude of correlation of four most correlated fish species (cutoff  $r^2 = 0.175$ , vector scaling 100%)

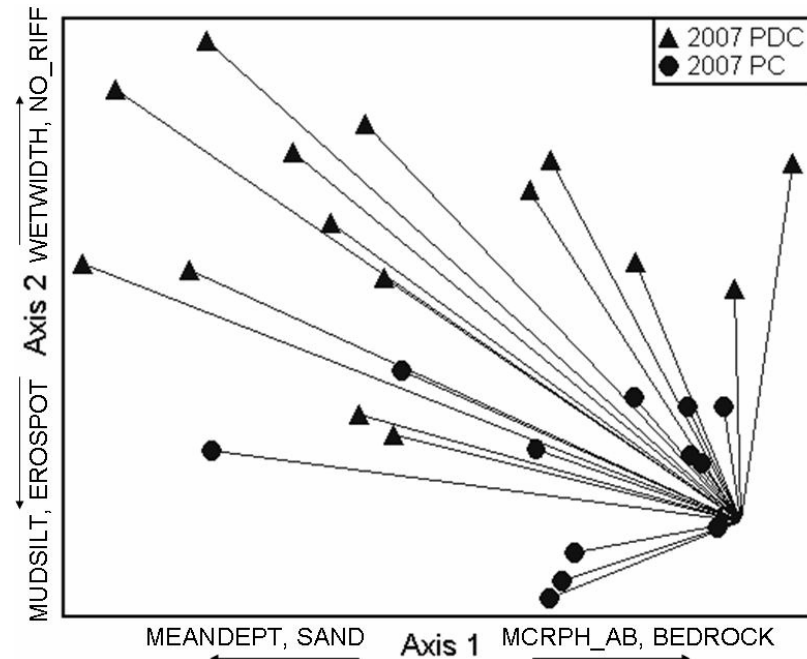


Figure 3: Diagram of successional vectors translated to origin (2006) describing magnitude and direction of change in 2-dimensional PCA space of sites between 2006 and 2007.

*bubalus*, *Labidesthes sicculus*, *Lepomis microlophus*, *Lythrurus fumeus*, *Minytrema melanops*, *Notemigonus crysoleucas*, *Noturus gyrinus*, *Noturus nocturnus*, and *Percina carbonaria* were also collected, but these species were excluded from analyses (Appendix A-3) because they were collected from less than five percent of the 56 sampling units (McCune and Grace 2002).

Qualitatively, there was a large decline in the total number of fishes collected from 2006 to 2007 associated with increased discharges due to heavy precipitation in 2007 (Table 1). Flood disturbance level discharges immediately prior to our sampling index period resulted in considerable instream habitat alteration, the likely displacement and/or death of fishes, as well as a reduced sampling efficiency associated with collecting in deeper habitats. This decline was most notable in many of the species collected from the family Cyprinidae and Centrarchidae including *Campostoma anomalum* (central stoneroller) and the two bass species *Micropterus salmoides* (large-mouthed bass) and *Micropterus punctulatus* (spotted bass). There was also a pronounced reduction in the numbers of *Fundulus notatus* (blackstripe topminnow) and *Gambusia affinis* (western mosquitofish). Few species increased in abundance from 2006 to 2007 most notably including *Lepomis cyanellus* (green sunfish), *Ictalurus punctatus* (channel catfish), and *Pylodictus olivarius* (flathead catfish). These increases in abundance are likely due to fishes, primarily juveniles, from higher order streams moving into smaller tributaries in order to colonize post-flood disturbed instream habitats.

Ordination of NMS analysis of fish communities (Figure 4) showed poorer separation between hydrologic groups and interannual group pairs than PCA. MRPP pairwise comparisons indicated differences in species composition were present between

Table 1: Change in number of individuals and relative abundance for fish species included in data analyses (occurred at > 5% of the sampling locations).

Family	Species	Number of Individuals			Relative Abundance (%)		
		2006	2007	2006→2007	2006	2007	2006→2007
Atherinidae	<i>Menidia beryllina</i>	29	54	25	0.10	0.53	0.43
Catostomidae	<i>Carpiodes carpio</i>	58	4	-54	0.20	0.04	-0.16
	<i>Moxostoma congestum</i>	63	38	-25	0.22	0.37	0.15
Centrarchidae	<i>Lepomis auritus</i>	30	0	-30	0.10	0	-0.1
	<i>Lepomis cyanellus</i>	805	1592	787	2.81	15.58	12.77
	<i>Lepomis gulosus</i>	44	72	28	0.15	0.70	0.55
	<i>Lepomis humilus</i>	24	11	-13	0.08	0.11	0.03
	<i>Lepomis macrochirus</i>	553	715	162	1.93	7.00	5.07
	<i>Lepomis megalotis</i>	3540	1037	-2503	12.34	10.15	-2.19
	<i>Lepomis</i> spp.	1013	169	-844	3.53	1.65	-1.88
	<i>Micropterus punctulatus</i>	343	39	-304	1.20	0.38	-0.82
	<i>Micropterus salmoides</i>	481	239	-242	1.68	2.34	0.66
	<i>Pomoxis annularis</i>	3	1	-2	0.01	0.01	0
	<i>Pomoxis nigromaculatus</i>	0	88	88	0	0.86	0.86
Clupeidae	<i>Dorosoma cepedianum</i>	286	3	-283	1.00	0.03	-0.97
	<i>Dorosoma petenense</i>	6	131	125	0.02	1.28	1.26
Cyprinidae	<i>Campostoma anomalum</i>	2082	473	-1609	7.26	4.63	-2.63
	<i>Cyprinella lutrensis</i>	2818	1276	-1542	9.83	12.49	2.66
	<i>Cyprinella venusta</i>	6486	2175	-4311	22.62	21.28	-1.34
	<i>Cyprinus carpio</i>	80	47	-33	0.28	0.46	0.18
	<i>Lythurus umbratilis</i>	3	29	26	0.01	0.28	0.27
	<i>Notropis volucellus</i>	475	139	-336	1.66	1.36	-0.3
	<i>Pimephales vigilax</i>	1279	400	-879	4.46	3.91	-0.55
Fundulidae	<i>Fundulus notatus</i>	439	36	-403	1.53	0.35	-1.18
Ictaluridae	<i>Ameiurus natalis</i>	199	179	-20	0.69	1.75	1.06
	<i>Ictalurus punctatus</i>	176	595	419	0.61	5.82	5.21
	<i>Pylodictis olivaris</i>	9	64	55	0.03	0.63	0.6
Lepisosteidae	<i>Lepisosteus osseus</i>	50	8	-42	0.17	0.08	-0.09
Moronidae	<i>Morone chrysops</i>	17	13	-4	0.06	0.13	0.07
Percidae	<i>Etheostoma spectabile</i>	228	167	-61	0.80	1.63	0.83
	<i>Percina macrolepida</i>	2	14	12	0.01	0.14	0.13
	<i>Percina sciera</i>	12	19	7	0.04	0.19	0.15
Poeciliidae	<i>Gambusia affinis</i>	7040	387	-6653	24.55	3.79	-20.76
Sciaenidae	<i>Aplodinotus grunniens</i>	4	6	2	0.01	0.06	0.05
Sum=		28677	10220	-18457			

2006 and 2007 but not between habitat connectivity groupings for either year (Table 4).

However, successional vectors derived from the NMS ordination (Figure 5) implied that, consistent with the relative differences in habitat between years, fish communities in



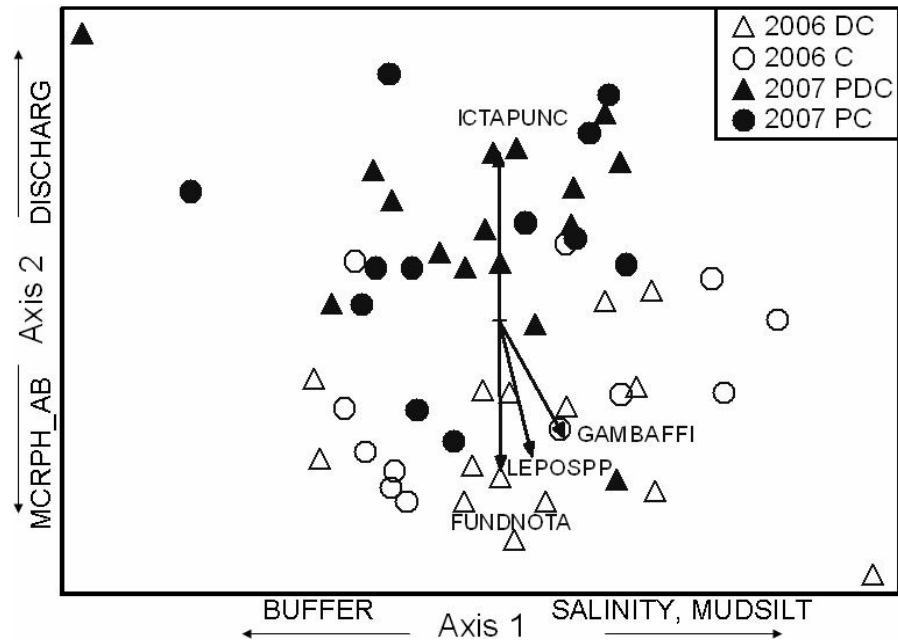


Figure 4: Configuration of stream reaches in 2-dimensional NMS representation of fish species composition dissimilarity matrix. Joint bi-plot overlay shows direction and magnitude of correlation of four most correlated fish species (cutoff  $r^2 = 0.325$ , vector scaling 100%)

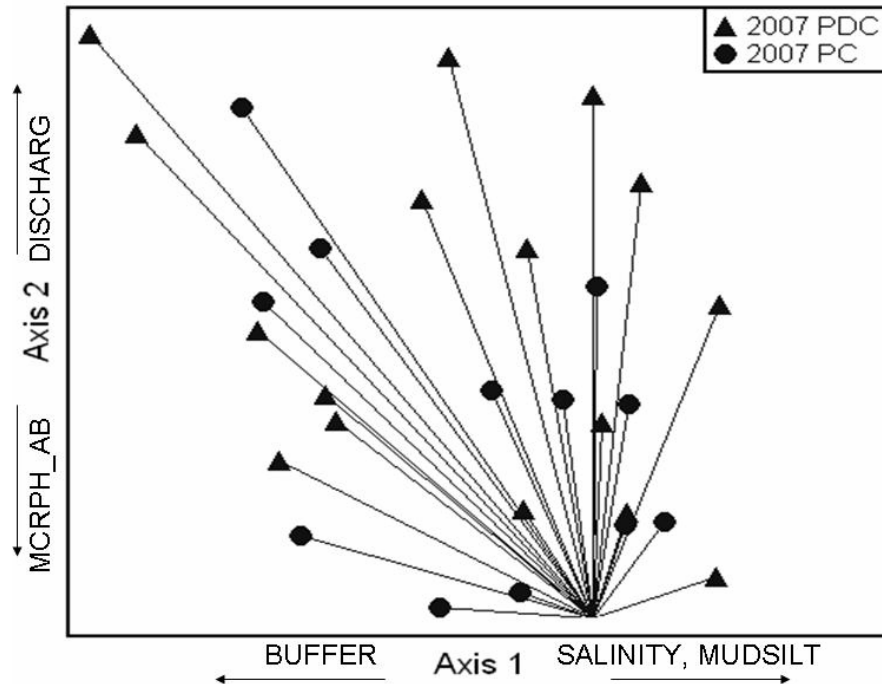


Figure 5: Diagram of successional vectors translated to origin (2006) describing magnitude and direction of change in 2-dimensional NMS space of sites between 2006 and 2007.

Table 2: PCA results showing the loadings (standardized correlation coefficients) the 20 most interpretable habitat variables for principle components (PC) 1 and 2.

	Eigenvalue	Cumulative Variance
PC1	9.346	19.074
PC2	6.660	32.666

Variable code	PC1	PC2
ALGAE_AB	0.516	0.067
BEDROCK	0.642	0.449
DO	-0.548	0.333
EMBEDDED	-0.4917	-0.4342
EROSPOT	-0.543	-0.514
FILA_ALG	0.526	-0.154
HAB_TYPE	-0.338	0.514
MACRPHYT	0.638	-0.209
MCRPH_AB	0.698	-0.154
MEANDEPT	-0.672	0.298
MUDSILT	-0.458	-0.537
NO_RIFF	-0.342	0.686
POOL_DEP	-0.511	-0.117
SAND	-0.662	-0.249
SM_BLDR	0.272	0.296
SOIL	-0.589	-0.478
STDVDEPT	-0.609	0.212
STRM_COV	0.509	-0.012
VELDEPTH	-0.391	0.646
WETWIDTH	-0.315	0.734

Table 3: One-tailed T-test comparing group mean successional vector lengths

	Interannual pairs comparison	Mean	s	$\sigma$	P
PCA	2006 DC → 2007 PDC	5.66	2.42	0.60	0.00022
	2006 C → 2007 PC	2.34	1.73	0.50	
NMS	2006 DC → 2007 PDC	1.03	0.51	0.13	0.05817
	2006 C → 2007 PC	0.74	0.45	0.13	

2006 DC sites had a greater magnitude of change from 2006 to 2007 than 2006 C sites.

A one-way t-test (Table 3) also revealed a low probability ( $p=0.054$ ) of obtaining an

equal or greater effect size by chance alone in the amount of interannual change between hydrologic groups, although the comparison of overall change in direction and magnitude in community structure suggested no predictable effect on pattern of community succession between years ( $p=0.399$ , Table 4). PERMDISP analysis of NMS ordination as well as successional vector plot coordinates showed no significant differences in the dispersion of hydrologic groups in ordination space.

Despite the apparent lack of separation of habitat connectivity groupings in NMS space, indicating low dissimilarity between these groups, ISA conducted on both habitat connectivity and interannual groupings (Table 5) revealed interesting patterns in fish community composition among these groups. Both PCA and NMS joint bi-plot overlays

Table 4: MRPP pairwise comparisons between groups and mean distance of observations to group centroid for PCA and NMS analyses.

	Pairwise comparison	T	A	P	Mean distance to centroid	
<u>PCA</u> A= 0.136 P< 0.0005	2006DC vs. 2007PDC	-11.535	0.168	<0.005	2006 DC	5.91
	2006DC vs. 2006C	-5.922	0.096	<0.005	2006 C	4.88
	2006DC vs. 2007PC	-11.092	0.19	<0.005	2007 PDC	4.66
	2007PDC vs. 2006C	-0.142	0.003	0.325	2007 PC	4.74
	2007PDC vs. 2007PC	1.097	-0.019	0.988		
	2006C vs. 2007PC	-1.081	0.023	0.13		
Vectors	DC/PDC vs. C/PC	-6.430	0.117	<0.005		
<u>NMS</u> A= 0.073 P< 0.0005	2006DC vs. 2007PDC	-7.924	0.111	<0.005	2006 DC	1.17
	2006DC vs. 2006C	0.76	-0.013	0.766	2006 C	1.21
	2006DC vs. 2007PC	-5.234	0.082	<0.005	2007 PDC	1.22
	2007PDC vs. 2006C	-4.238	0.067	<0.005	2007 PC	1.29
	2007PDC vs. 2007PC	1.242	-0.02	0.99		
	2006C vs. 2007PC	-2.121	0.041	0.039		
Vectors	DC/PDC vs. C/PC	-0.042	0.001	0.399		

Table 5: Results of indicator species analysis showing significant indicator fish species associated with *a priori* habitat connectivity groups (top) and interannual groups (bottom). MaxGrp indicates the group in which the maximum indicator value (IV) for a species was found.

Species	MaxGrp	Observed	Randomized		
		IV	Mean IV	s	p
<i>Gambusia affinis</i>	2006 DC	70.6	35.5	7.41	0.0001
<i>Campostoma anomalum</i>	2006 C	64.8	31.8	7.32	0.001
<i>Lepomis</i> spp.	2006 DC	55.5	29.9	7.95	0.0055
<i>Fundulus notatus</i>	2006 DC	40.5	19.9	6.25	0.0088
<i>Micropterus punctalatus</i>	2006 C	40.8	20.7	6.56	0.0127
<i>Pylodictis olivaris</i>	2007 PDC	37	21.6	6.36	0.0258
<i>Micropterus salmoides</i>	2006 DC	46.3	30.1	6.84	0.0258
<i>Pimephales vigilax</i>	2006 C	51.9	34.4	8.62	0.0326
<i>Lepomis megalotis</i>	2006 DC	42.4	33.5	4.32	0.0372
<i>Ictalurus punctatus</i>	2007 PDC	39.2	31.2	5.08	0.0785*
<i>Gambusia affinis</i>	2006	91.4	52	7.6	0.0001
<i>Ictalurus punctatus</i>	2007	74.4	49.9	5.48	0.0001
<i>Lepomis megalotis</i>	2006	74.6	55.6	4.6	0.0002
<i>Pylodictis olivaris</i>	2007	56.4	28.7	5.86	0.0003
<i>Fundulus notatus</i>	2006	49.5	25.6	5.66	0.0013
<i>Cyprinella venusta</i>	2006	66.9	52.7	4.75	0.0085
<i>Lepomis cyanellus</i>	2007	66.4	55.3	3.95	0.0112
<i>Campostoma anomalum</i>	2006	66.9	46.3	7.55	0.0146
<i>Micropterus punctalatus</i>	2006	41.7	26.9	5.93	0.0235
<i>Lepomis</i> spp.	2006	55.1	39.6	7.37	0.0353
<i>Dorosoma cepedianum</i>	2006	21.2	13.1	5.01	0.0495

\*does not meet  $p \leq 0.05$  significance criterion

(Figure 2 and 4) as well as ISA reveal that *G. affinis* is a strong indicator species for the group 2006 DC. ISA also indicated *M. salmoides*, *L. megalotis*, *F. notatus* and *Lepomis* spp. juveniles were also significant indicator species for this group. *G. affinis* and *F. notatus* are two species tolerant of high temperature, low dissolved oxygen habitats, which are habitat condition characteristics of the group 2006 DC. *M. salmoides* is a

larger bodied predatory fish that is known to utilize smaller fishes, like the ones previously discussed, as prey items. *M. punctalatus* and *C. anomalum* were significant indicator species of the 2006 C group, which may indicate habitat connectivity and flowing water as important determinants of fish community composition in these streams. Indicator values for the catfish species *P. olivarius* and *I. punctatus* were greater when ISA was conducted on interannual groupings than habitat connectivity groupings (Table 5). This result suggests that habitat conditions in 2007 were possibly greater determinants of fish community composition than prior instream habitat connectivity classification.

## CHAPTER FOUR

### Discussion

Differences in hydrology and instream habitat connectivity had a significant impact of the fish species compositions of my study sites. Extended lack of precipitation in 2006 resulted in the drying of streams across central Texas as the summer progressed. As perennial pools were formed, preventing movement of fishes throughout a study reach, biotic interactions within these pools increased with increases in population density. Separation of the group 2006 DC in PCA ordination space from all other groups suggests that the general habitat associated with this group significantly differs from the habitat conditions of all other study sites. This group exhibited the greatest amount of interannual change in habitat as shown by the t-test comparison of successional vector lengths. This difference in habitat resulted in a unique species composition, driven largely by the significant indicator species *M. salmoides*, *G. affinis*, and *F. notatus*. *G. affinis* has the ability to survive in water with very little dissolved oxygen. They generally forage at the waters surface where contact with the atmosphere increases dissolved oxygen slightly. Mosquitofish are known to gulp air to supplement oxygen intake during times of low dissolved oxygen, which is facilitated by the dorsally oriented mouth and dorso-ventrally flattened head (Pyke 2005). *F. notatus* has similar body morphology, body size, habitat preference, and has also been observed at our study sites gulping air. These physical and behavioral adaptations of these species allow them to persist in perennial pools such as the ones present in the 2006 DC group due to the presence of conditions which they are better adapted to tolerate. In addition, the small

size of both these fishes allows them to stay in very shallow waters where they may use cobble substrates and vegetation as cover to potentially avoid predation from larger bodied fishes that may be unable to access these shallow areas. *M. salmoides* is a large-bodied predatory fish, which may potentially prey upon the smaller bodied indicator species *F. notatus* and *G. affinis*. It is typically found in deeper pools within stream reaches where it may use deeper water as cover from which it may ambush its prey. The high indicator values of both predator and prey species in this group may suggest habitat partitioning among the fish community of this group, with predators occupying the deeper area of pools and prey utilizing shallow habitats as cover to avoid predation.

Another interesting relationship observed in 2006 streams was the significant association of *M. punctulatus* and *C. anomalum* with the 2006 C group. *C. anomalum* is a small-bodied herbivorous fish that feeds on filamentous algae, diatoms, and attached periphyton by scraping these from submerged objects along the stream bottom. Food habits of spotted bass change with increase in body length with larger individuals consuming macroinvertebrates as well as small fish species, such as *Camptostoma*. The significant indication of these species that have a documented predator-prey interaction to the group 2006 C suggests the potential for habitat partitioning within these stream sites. Power and Matthews (1983) showed that the presence of *Micropterus salmoides* or *Micropterus punctulatus* greater than 70 mm standard length in perennial pools in an Oklahoma stream resulted in spatial habitat segregation between these species and *Camptostoma anomalum*. In the presence of bass, *Camptostoma* occupied shallower regions of pools or were completely absent due to either pre-habitat disconnection avoidance of pools with *M. punctulatus* or extend periods of predation after habitat

separation had occurred. Subjectively, many of the *Campostoma* collected from our study sites were collected from shallow, flowing riffle habitats whereas bass were typically collected from deep pools around submerged structures such as logs, boulders, and undercuts. During 2006 as flows decreased and riffles dried separating the stream into disconnected pools, the niche previously occupied by the *Campostoma* was no longer available. In the absence of riffle habitat, *Campostoma* were forced into pools where they became potential prey items for basses and other large predatory fishes. This may account for the low numbers of *Campostoma* in the group 2006 DC relative to the abundance of this species in the group 2006 C.

Habitat conditions in our study streams in 2007 were markedly different from those in 2006. Extended periods of precipitation immediately prior to our sampling resulted in flood disturbance conditions. These floods caused massive debris flows, rearrangement of large instream substrates, instream algal scouring, loss of riparian vegetation, alteration of stream channel morphology, and displacement and/or death of resident fishes and macroinvertebrate prey items. Resulting post disturbance habitats had low biomass and newly created habitats that rapidly colonizing species of fish could readily utilize. In addition, once sampling had begun, the amount of surface water present within our study reaches may have reduced sampling efficiency of both collection methods. The combination of these two factors resulted in a nearly two-thirds reduction in number of individuals captured compared to 2006. *P. olivarius* was a significant indicator species for the group 2007 PDC, the only significant indicator species for either 2007 groups. *I. punctatus* had an indicator value higher than *P. olivarius*, but did not meet the Monte Carlo randomization test significance criterion of  $P > 0.05$ . However,



when ISA was performed using annual classifications as groups, indicator values for these two species increased significantly (Table 5). This may suggest that the post-flood disturbance hydrologic conditions of the streams in 2007 was the primary determining factor of fish community composition independent of instream connectivity classification. These fishes are primarily lentic or large-river species found in large lakes and reservoirs. *I. punctatus* is aquacultured in large earthen ponds in many areas of the southern U.S.A. as a food item for human consumption in addition to wild populations. In lotic habitats, it occupies deeper habitats in higher order streams for the majority of its adult life. In the wild, this species is primarily an illiophagus (mud eating) omnivore deriving nutrition from detritus and benthic macroinvertebrates found in muddy substrates (Jackson 2004, Shephard and Jackson 2006). Adult *P. olivarius* also prefer deeper habitats, but are primarily piscivorous as adults, and this species has been used as a bio-control for species undesirable to anglers, such as *Cyprinus carpio* and *Dorosoma cepedianum*, in lentic habitats (Haas 2001). These two catfish species have been described as fast colonizers that move from more typical, higher-order stream home ranges into these lower-order streams immediately following flood disturbance conditions in search of newly accessible resources such as food and spawning habitats (Ross et al 2001). The adaptations of these catfishes that allow them to move into and colonize newly created habitats following the flood disturbances such as the one observed in 2007 may be the reason for the high indication of these species to 2007 stream sites.

Historically, emphasis has been placed upon biological monitoring of stream ecosystems as an indicator stream ecosystem function since the enactment of the Water

Quality Act Amendments of 1972, which called for new emphasis on biological monitoring in government agencies' assessment of freshwater resources. Karr (1981) suggested the use of multivariate approach examining the "health" of freshwater communities, such as his proposed index of biotic integrity (IBI). However, the IBI he proposed relies on the *a priori* assignment of tolerance values to taxa. The tolerances of organisms as well as potential biotic and abiotic interactions differ regionally. Species that may be considered "tolerant" in the context of one type or condition of a stream ecosystem may not be in another context. Recent technological, computational, and statistical advancements allow us to examine ecosystem processes in a much more data-driven manner. By examining the interaction between habitat condition and fish species composition using multivariate ordination techniques discussed in this study, we allow the variables to determine the foundation on which comparisons can be made, and not pre-assigned tolerance values. This allows us to make a richer assessment of the biotic and abiotic interactions occurring within streams independent of preconceived notions about the life histories of fish species.

In conclusion, habitat connectivity and stream discharge are major influences on instream habitat as well as species composition. Extremes of discharge values can result in varied balance between abiotic and biotic controls on fish communities. Streams with disconnected pools had species compositions resulting from increased biotic interactions (such as predatory top-down control) whereas streams following a high flow disturbance event are regulated primarily by abiotic habitat controls. Knowledge about the life histories of fishes may be useful in elucidating meaning behind patterns in species composition. The use of multivariate ordination techniques allows us to make more

ecologically meaningful inferences about the relationship between habitat and communities than previous methods of evaluating biotic integrity of streams. I believe that the techniques discussed in this study are viable for use in the long-term monitoring of stream ecosystems to examine climatological, anthropogenic, or disturbance driven changes within these environments over time.

## APPENDIX

Supplementary tables referenced in thesis

Table A-1: Site locations, sampling dates, and discharges ( $\text{m}^3 \cdot \text{s}^{-1}$ ) for 28 stream reaches sampled 2006 and 2007.

Stream Name	SITE_ID	Latitude	Longitude	2006 Date and Discharge		2007 Date and Discharge	
Bear Creek	BEAR	32.59442	97.51018	7/6/2006	<0.001	8/15/2007	0.377
Bluff Creek	BLUF	31.55536	97.47570	8/22/2006	<0.001	8/29/2007	0.048
Clear Fork Trinity	CFTR	32.70082	97.62979	7/6/2006	0.014	8/15/2007	0.07
Clear Creek	CLEA	33.35909	97.25029	8/10/2006	<0.001	8/17/2007	0.312
Coryell Creek	CORY	31.39070	97.59826	7/5/2006	<0.001	7/24/2007	2.118
Cowhouse Creek	COWH	31.28327	97.88241	6/14/2006	0.002	10/12/2007	0.699
Duffau Creek	DUFF	32.01341	97.96521	8/7/2006	<0.001	9/18/2007	0.101
Elm Fork Trinity	EFTR	33.58631	97.13076	7/13/2006	0.041	8/17/2007	0.365
Harris Creek	HARR	31.45960	97.29253	8/24/2006	<0.001	8/23/2007	0.102
Hog Creek	HOG	31.52264	97.28924	6/7/2006	<0.001	8/22/2007	0.206
Lampasas River Site 1	LAMP1	31.37802	98.18063	7/11/2006	0.006	10/5/2007	0.481
Lampasas River Site 2	LAMP2	31.11558	98.05432	8/17/2006	0.006	10/26/2007	1.112
Middle Bosque River	MBOS	31.50748	97.35624	7/10/2006	<0.001	8/31/2007	0.275
Meridian Creek	MERI	31.81095	97.60911	6/13/2006	<0.001	7/23/2007	3.818
*North Bosque River Site 1	NBOS1	32.18898	98.18258	7/19/2006	0.504	9/14/2007	0.0342
North Bosque River Site 2	NBOS2	32.04166	98.11330	7/12/2006	<0.001	8/28/2007	0.276
North Bosque River Site 3	NBOS3	31.97692	98.03974	6/8/2006	<0.001	9/20/2007	0.621
Neils Creek	NEIL	31.69952	97.53088	8/13/2006	<0.001	7/23/2007	2.995
Nolan Creek	NOLC	31.08916	97.48826	6/28/2006	0.735	8/31/2007	0.821
Nolan River Site 1	NOLR1	32.25064	97.40433	7/18/2006	0.14	8/14/2007	0.326
Nolan River Site 2	NOLR2	32.14660	97.40600	6/20/2006	0.139	8/14/2007	0.639
Paluxy River	PALU	32.24960	97.84615	6/15/2006	0.043	9/11/2007	1.517
Plum Creek	PLUM	31.49967	97.85686	7/5/2006	<0.001	8/13/2007	0.231
Rocky Creek	ROCK	30.94494	97.99117	6/21/2006	0.029	8/16/2007	0.904
Salado Creek	SALA	30.91275	97.60105	8/17/2006	0.023	9/21/2007	0.216
South Bosque River	SBOS	31.47359	97.27638	8/24/2006	<0.001	11/2/2007	0.057
South Fork Trinity	SFTR	32.69943	97.63202	8/9/2006	<0.001	8/15/2007	0.231
South Leon River	SLEO	31.83891	98.37763	8/18/2006	<0.001	8/13/2007	0.057

\*North Bosque River Site 1 is the only site whose discharge decreased from 2006-2007. This is most likely due to the fact that discharge is primarily controlled by rate of effluent discharge rate from Stephenville Wastewater System Treatment facility located <1 km upstream of our stream reach.

Table A-2: Taxonomy and assigned species codes for the 34 fish species collected in 2006 and 2007 included in analyses. LEPOSPP species code is the designation for *Lepomis spp.* juveniles that were identifiable only to genus due to small size.

Order	Family	Genus	Species	Species Code
Siluriformes	Ictaluridae	Ameiurus	<i>Ameiurus natalis</i>	AMEINATA
Perciformes	Sciaenidae	Aplodinotus	<i>Aplodinotus grunniens</i>	APLOGRUN
Cypriniformes	Cyprinidae	Campostoma	<i>Campostoma anomalum</i>	CAMPANOM
Cypriniformes	Catostomidae	Carpiodes	<i>Carpiodes carpio</i>	CARPCARP
Cypriniformes	Cyprinidae	Cyprinus	<i>Cyprinus carpio</i>	CYPRCARP
Cypriniformes	Cyprinidae	Cyprinella	<i>Cyprinella lutrensis</i>	CYPRLUTR
Cypriniformes	Cyprinidae	Cyprinella	<i>Cyprinella venusta</i>	CYPRVENU
Clupeiformes	Clupeidae	Dorosoma	<i>Dorosoma cepedianum</i>	DOROCEPE
Clupeiformes	Clupeidae	Dorosoma	<i>Dorosoma petenense</i>	DOROPETE
Perciformes	Percidae	Etheostoma	<i>Etheostoma spectabile</i>	ETHESPEC
Cyprinodontiformes	Fundulidae	Fundulus	<i>Fundulus notatus</i>	FUNDNOTA
Cyprinodontiformes	Poeciliidae	Gambusia	<i>Gambusia affinis</i>	GAMBAFFI
Siluriformes	Ictaluridae	Ictalurus	<i>Ictalurus punctatus</i>	ICTAPUNC
Semionotiformes	Lepisosteidae	Lepisosteus	<i>Lepisosteus osseus</i>	LEPIOSSE
Perciformes	Centrarchidae	Lepomis	<i>Lepomis auritus</i>	LEPOAURI
Perciformes	Centrarchidae	Lepomis	<i>Lepomis cyanellus</i>	LEPOCYAN
Perciformes	Centrarchidae	Lepomis	<i>Lepomis gulosus</i>	LEPOGULO
Perciformes	Centrarchidae	Lepomis	<i>Lepomis humilus</i>	LEPOHUMI
Perciformes	Centrarchidae	Lepomis	<i>Lepomis macrochirus</i>	LEPOMACR
Perciformes	Centrarchidae	Lepomis	<i>Lepomis megalotis</i>	LEPOMEGA
Perciformes	Centrarchidae	Lepomis	<i>Lepomis spp.</i>	LEPOSPP
Cypriniformes	Cyprinidae	Lythrurus	<i>Lythrurus umbratilis</i>	LYTHUMBR
Atheriniformes	Atherinidae	Menidia	<i>Menidia beryllina</i>	MENIBERY
Perciformes	Centrarchidae	Micropterus	<i>Micropterus punctulatus</i>	MICRPUNC
Perciformes	Centrarchidae	Micropterus	<i>Micropterus salmoides</i>	MICRSALM
Cypriniformes	Catostomidae	Minytrema	<i>Minytrema melanops</i>	MINYMELA
Perciformes	Moronidae	Morone	<i>Morone chrysops</i>	MOROCHRY
Cypriniformes	Catostomidae	Moxostoma	<i>Moxostoma congestum</i>	MOXOCONG
Cypriniformes	Cyprinidae	Notropis	<i>Notropis volucellus</i>	NOTRVOLU
Perciformes	Percidae	Percina	<i>Percina macrolepida</i>	PERCMACR
Perciformes	Percidae	Percina	<i>Percina sciera</i>	PERCSCIE
Cypriniformes	Cyprinidae	Pimephales	<i>Pimephales vigilax</i>	PIMEVIGI
Perciformes	Centrarchidae	Pomoxis	<i>Pomoxis annularis</i>	POMOANNU
Perciformes	Centrarchidae	Pomoxis	<i>Pomoxis nigromaculatus</i>	POMONIGR
Siluriformes	Ictaluridae	Pylodictis	<i>Pylodictis olivaris</i>	PYLOOLIV

Table A-3: Taxonomy and assigned species codes for the 11 fish species collected in 2006 and 2007 that were present in < 5% of sites and therefore not included in analyses.

Order	Family	Genus	Species	Species Code
Siluriformes	Ictaluridae	Ameiurus	<i>Ameiurus melas</i>	AMEIMELA
Cypriniformes	Characidae	Asyanax	<i>Astyanax mexicanus</i>	ASTYMEXI
Cypriniformes	Catostomidae	Ictiobus	<i>Ictiobus bubalus</i>	ICTIBUBA
Atheriniformes	Atherinidae	Labidesthes	<i>Labidesthes sicculus</i>	LABISICC
Perciformes	Centrarchidae	Lepomis	<i>Lepomis microlophus</i>	LEPOMICR
Cypriniformes	Cyprinidae	Lythrurus	<i>Lythrurus fumeus</i>	LYTHFUME
Cypriniformes	Catostomidae	Minytrema	<i>Minytrema melanops</i>	MINYMELA
Cypriniformes	Cyprinidae	Notemigonus	<i>Notemigonus crysoleucas</i>	NOTECRYS
Siluriformes	Ictaluridae	Noturus	<i>Noturus gyrinus</i>	NOTUGYRI
Siluriformes	Ictaluridae	Noturus	<i>Noturus nocturnus</i>	NOTUNOCT
Perciformes	Percidae	Percina	<i>Percina carbonaria</i>	PERCCARB

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