SPATIAL AND TEMPORAL PATTERNS IN THE FISH ASSEMBLAGE OF THE BLANCO RIVER, TEXAS, AND REPRODUCTIVE ECOLOGY AND DIET OF THE GRAY REDHORSE, *MOXOSTOMA CONGESTUM*

THESIS

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CHAPTER I

INTRODUCTION

Burr and Mayden (1992) and Warren et al. (1997) noted that the southern United States possesses the greatest fish diversity and number of endemic species in North America, north of Mexico. Although the southern United States has such a diverse fish fauna, 28% are listed as extinct, endangered, threatened, or of special concern/vulnerable (Warren et al. 2000). Habitat degradation in the form of altered hydrologic regimes is among the greatest threats to the freshwater fauna of the United States (Richter et al. 1997) and Texas leads the nation with over 6,000 dams constructed within its waters (Shuman 1995).

In the following studies, the spatial and temporal trends of Blanco River fishes and the diet and reproduction of *Moxostoma congestum* are examined. Specifically, factors structuring the fish assemblage are identified as are the effects of low-head dams on the assemblage. *Moxostoma congestum* has persisted in the Blanco River and throughout much of its historical range despite habitat degradation such as low-head dam construction. Diet and reproduction of *M. congestum* are examined in a low-head dam impoundment as well as in a large reservoir and are compared between the two habitat types as well as to life history characteristics often associated with adaptability to reservoirs.

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CHAPTER II

SPATIAL AND TEMPORAL PATTERNS IN THE FISH ASSEMBLAGE OF THE BLANCO RIVER, TEXAS

ABSTRACT

The fish assemblage of the Blanco River watershed was sampled quarterly at ten sites for two years. Eight sites were on the mainstem of the river, and two were on major tributaries - one on the Little Blanco River, and one on Cypress Creek. Cyprinids made up 78% of the overall assemblage, with *Cyprinella venusta* (41%), *Pimephales vigilax* (14%), and *Notropis amabilis* (11%) being the most abundant species. Variation in the fish assemblage was examined using Canonical Correspondence Analysis. Physical habitat parameters explained 15.3%, followed by site (11.2%), and season (2.3%). The low-head dam impoundment assemblage was markedly different from riverine mainstem sites (Analysis of Similarities, $P < 0.01$) in that the former was dominated by more lentic species and generally lacked species normally associated with higher velocity runs and riffles.

INTRODUCTION

In an ecological context, identification of patterns in species diversity and abundance and their causal mechanisms have received much attention (Shmida and Wilson 1985; Brown and Maurer 1989). The causal mechanisms generally are subdivided into abiotic and biotic factors, and are evaluated on recent and localized scales (Brown and Maurer 1989; Matthews 1998). Abiotic factors include both physical and chemical characteristics of a stream (i.e., depth, current velocity, substrate, temperature, pH, dissolved oxygen, and turbidity) and can affect assemblages based on the autecology of species (Whiteside and McNatt 1972; Matthews 1998). For example, elevated water temperature can negatively affect fecundity, larval production, and growth rates of stream fishes (Hubbs 1964; Bonner et al. 1998; Strange et al. 2002). Additionally, some abiotic factors (i.e., current velocity and temperature) may exhibit greater temporal variability in upstream sites than downstream sites (Harrel and Dorris 1968; Hynes 1970; Whiteside and McNatt 1972; Horwitz 1978; Vannote et al. 1980). Of the many abiotic factors, gradients in current velocity, depth, and substrate often most strongly associated with variation in fish assemblages at a local scale (Gorman and Karr 1978; Schlosser 1982; Cantu and Winemiller 1997; Walters et al. 2003; Williams et al. 2005). Biotic factors thought to affect fish assemblages include: intra- and interspecific competition, food availability, and predation (Matthews 1998). These biotic interactions are often difficult to quantify, and thus are less often used in attempts to explain patterns in fish assemblages. Understanding which factors are most strongly associated with the distributional patterns of stream fishes can reduce the error in predicting how fish assemblages might change as habitats are impacted by anthropogenic effects (Harding et al. 1998).

Although the southern United States has a highly diverse fish fauna (Burr and Mayden 1992, Warren et al. 1997), 28% are listed as extinct, endangered, threatened, or of special concern/vulnerable (Warren et al. 2000). Williams et al. (1989) lists five factors contributing to the demise of North American species that are endangered, threatened, or of special concern. They are, briefly: 1) habitat degradation, 2) overexploitation, 3) disease, 4) natural or anthropogenic-induced biotic factors, and 5) restricted range. Warren et al. (2000), however, attributes the decline of native fishes of the southern United States primarily to habitat degradation.

Anthropogenic disturbance in the form of increased sediment and nutrient loads, introduced species, and altered hydrologic regimes are among the greatest threats to the freshwater fauna of the United States (Richter et al. 1997) and are cited as the reason for species declines across the country (Warren et al. 2000). Impoundment of streams reduces the connectivity of upstream and downstream segments (Edwards 1978) and of streams within a drainage (Herbert and Gelwick 2003), decreases the discharge (Bonner and Wilde 2000) and magnitude of floods (Adams 1985) downstream from impoundments, and creates a more lentic habitat within the impounded segment (Taylor et al. 2001). These disturbances can have many effects on fish assemblages. Dams present a barrier to the longitudinal movement of fishes, especially upstream (Porto et al. 1999). This restricted movement of fishes may lead to decreased upstream diversity and extirpation of obligate riverine species (Winston et al. 1991; Porto et al. 1999). This decrease in obligate riverine species is generally associated with dominance of the assemblage by habitat generalists (Winston et al. 1991; Taylor et al. 2001). When variable upstream reaches experience harsh conditions that cause a local extirpation of all fish from a stream reach, these reaches are subsequently repopulated by species surviving in stable, downstream habitats (Whiteside and McNatt 1972). Impoundments serve as a stable source of generalist species from which upstream habitats are opportunistically colonized (Herbert and Gelwick 2003). With obligate riverine species being absent from reservoirs, even temporary cessation of flow upstream from an impoundment might lead to the permanent extirpation of fluvial specialists and an assemblage dominated by habitat generalists. Downstream from dams, changes in habitat caused by scouring of substrate (Gillette et al. 2005) and reduced peak discharge (Bonner and Wilde 2000) contribute to changes in fish assemblages.

The purpose of this study was to identify factors important in structuring the Blanco River fish assemblage and determine the effects of low-head dams within the watershed. The Blanco River is a stream system typical of the Texas hill country and Edwards Plateau characterized by low turbidity, and high dissolved solids. Additionally, these streams posses many endemic taxa, about which, little is known. These stream systems face several threats (Bowles and Arsuffi 1993) including low-head dams as Texas leads the nation with over 6,000 dams constructed in its waters (Shuman 1995). The effects of low-head dams on the fish assemblages of streams of the Texas Hill Country are not known. Description of the current fish assemblage and identification of factors structuring the assemblage will allow for determination of future changes in the Blanco River fish assemblage and prediction of impacts of anthropogenic disturbance within the watershed on regional and drainage endemic species (e.g. *Dionda* *nigrotaeniata*, *Macrhybopsis marconis*, and *Micropterus treculii*) as well as the overall assemblage. Specifically, our objectives were to determine current habitat and fish assemblage structure and identify habitat associations, longitudinal and seasonal patterns, and effects of low-head dams on the Blanco River fish assemblage.

MATERIALS AND METHODS

The Blanco River drains an area of 1,067 km^2 in Kendall, Comal, Blanco, and Hays counties, Texas, before its confluence with the San Marcos River (Appendix 1.1). Little Blanco River and Cypress Creek are the two largest tributaries of the Blanco River. Both tributaries are spring-fed although baseflow in the Little Blanco River is subterranean about 5 km before reaching the Blanco River.

Ten sites in the Blanco River watershed were sampled quarterly from October 2003 through July 2005. Eight sites were located on the mainstem with two upper (sites 1 and 2), two middle (sites 3 and 4), and four lower reach sites (sites 5, 6, 7, and 8). Two sites were established on major tributaries with one on the Little Blanco River (Site 9) and one on Cypress Creek (Site 10).

At each site, fish were collected from available geomorphic units (*i.e.*, run, riffles, pools, backwaters, reservoirs, and plunge pools; Arend 1999) by a combination of seining (9.5 mm mesh), backpack electrofishing (Smith-Root Model 12-B POW), and experimental gill nets (3 nets set for 2 hours). Seines were used at all sites, backpack electrofishing was used in areas not conducive for seining (i.e., around cover, large woody debris, boulders, and shallow riffles) and gill nets were used at Site 2 (Reservoir Site) in deepwater (>2 m) habitats. Fish were collected from each geomorphic unit until fish were depleted from the geomorphic unit or until only a few individuals were captured and no new species were was collected. Fish from each geomorphic unit were isolated in buckets until sampling was completed in all geomorphic units. Fish were identified to species (classification following Nelson et al. 2004), measured to the nearest millimeter total length (up to 30 specimens per species per site), and released or retained as voucher specimens. Voucher specimens were anesthetized with a lethal dose of tricaine methanesulfonate and preserved in 10% formalin.

Habitat parameters recorded include geomorphic unit type, length, stream width, percent substrate (silt, sand, gravel, cobble, boulder, and bedrock), percent woody debris, percent vegetation, percent detritus, mean current velocity (m/s), maximum current velocity (m/s), mean depth (m), maximum depth (m), temperature (°C), pH, conductivity (μS/cm), dissolved oxygen (mg/l), and turbidity (NTU). Geomorphic unit length and width were measured to the nearest meter. Percent substrate, woody debris, vegetation, and detritus were visually estimated for each geomorphic unit (Williams et al. 2005). Depth was recorded to the nearest 0.01 m and current velocity was measured using a Marsh-McBirney FLOW-MATE™ model 2000 flow meter. Temperature and chemical data were measured using a YSI-Model 85 and YSI-Model 650 MDS. Additionally, site estimates of physical and chemical data were calculated by weighted averaging by geomorphic unit area, and stream discharge data were obtained from USGS stations No. 08171000 (Wimberley, TX) and No. 08171300 (Kyle, TX).

Principal Components Analysis (PCA) was performed using site means of physical habitat data. Qualitative data (i.e. geomorphic units) were represented with dummy variables whereas quantitative data were z-score transformed (Krebs, 1999). The resulting loadings and plots were used to describe habitat present at each site. Fish abundance and habitat data were analyzed using canonical correspondence analysis (CCA; Canoco 4.5, ter Braak 1986) to determine habitat associations as well as seasonal, site, and habitat effects in structuring the Blanco River fish assemblage. A variance partitioning method (Borcard et al. 1992) was used to determine pure site, season, and habitat effects as well as shared (two- and three-way) effects by producing a reduced CCA model for each effect with the additional two effects as covariates. Species richness, Shanon-Wiener diversity indices, and Pielou's evenness indices were calculated in PRIMER (version 5; Primer-E, Ltd., Plymouth, United Kingdom) for each site per each quarter. Bray-Curtis similarity indices were calculated for species abundance data pooled by season for each site. Species abundance data were standardized as relative abundances because sampling effort (i.e. area sampled) was not equal among sites. The resulting similarity matrix was used in analysis of similarities (ANOSIM; Clarke and Green 1988; Clarke 1993) to test for differences in fish assemblage structure among sites within, adjacent to, and distant from impoundments. Sites adjacent to impoundments were defined as those within 1 km of an impoundment whereas sites distant from impoundments were greater than 1 km from an impoundment. Distance to impoundment was determined along the thalweg by examination of aerial photographs and topographical maps. Determination of which species were contributing the greatest amount to the dissimilarity between the impoundment and the other categories was accomplished using the SIMPER function in PRIMER.

RESULTS

Stream Characteristics

The Blanco River, Little Blanco River, and Cypress Creek were generally shallow to moderate depth wadeable streams with substrate dominated by bedrock with some coarse gravel (Appendix 1.2). The tributaries, Little Blanco River and Cypress Creek, were deeper, had slower current velocities, greater percentages of gravel substrate, and more aquatic vegetation and detritus than mainstem sites. Mainstem sites were wider and shallower with swifter current velocities and greater percentages of bedrock substrate. Sand and boulder substrates were uncommon across all sites but were both highest at Site 5. Among mainstem sites, the Reservoir Site (Site 2) and Site 8 had the greatest amount of vegetation (filamentous algae and emergent macrophytes: 22%), whereas no vegetation was present at Site 5. During the duration of the study, median discharge was 4.25 cms between sites 4 and 5 (USGS Station No. 08171000) and 3.96 cms between sites 7 and 8 (USGS Station No. 08171300). Across sites and dates, mean temperature was 20°C, pH was 9.18, conductivity was 441 μS/cm, dissolved oxygen was 8.77 mg/l, and turbidity was 2.9 NTU. Seasonal and diel water quality and geochemistry measurements were taken concurrently with this study and reported by Cave (2006).

Principal Components Analysis of Habitat Parameters

Principal Component axes I and II explained 37 % of the variation in habitat among sites within the Blanco River watershed. PC I represented a substrate gradient whereas PC II represented a velocity, depth, and substrate gradient (Appendix 1.3). Strongest positive loadings for PC I were gravel (0.52), woody debris (0.34), and vegetation (0.32), while the strongest negative loadings were bedrock (-0.52), boulder (-0.16), and sand (-0.16). Strongest positive loadings for PC II were silt (0.56), depth (0.38), and vegetation (0.36), while the strongest negative loadings were current velocity (-0.58), bedrock (-0.18), and gravel (-0.16). Site 1 and the Reservoir Site had higher percentages of bedrock and silt substrate. The Reservoir Site differed from Site 1 by having greater vegetation and depth. Site 4 had a high percentage of bedrock and gravel, and had the lowest percentage of silt. Sites 5 and 6 had greater percentages of bedrock and boulder substrates and higher current velocities whereas sites 3 and 8 and Little Blanco River had greater percentages of gravel substrate with sites 3 and 8 having greater percentages of cobble and sites 8 and Little Blanco River having greater amounts of woody debris. Cypress Creek had a greater percentage of bedrock and cobble substrate as well as detritus.

Fish Abundance and Structure

We collected 29,265 fishes representing 10 families and 33 species from October 2003 through July 2005 within the Blanco River watershed (Appendix 1.4). Overall fish abundance was highest at Site 2 ($N = 6{,}586$) and Site 8 ($N = 4{,}318$) and lowest at Site 4 ($N = 1{,}213$) and Cypress Creek ($N = 1{,}273$). The most abundant families were Cyprinidae (78%), Centrarchidae (10%), and Poeciliidae (8%). Lepisosteidae, Catostomidae, Characidae, Ictaluridae, Fundulidae, Percidae, and Cichlidae each comprised less than 2% of the overall assemblage. *Cyprinella venusta* (41%), *Pimephales vigilax* (14%), *Notropis amabilis* (11%), *Gambusia affinis* (8%), and *Notropis volucellus* (5%) were the most abundant species. Cyprinidae (N of species =

11) and Centrarchidae (N of species = 10) were the most species rich families. Species richness ranged from 3 to 15 among samples with the lowest richness occurring at Site 4 in January 2004 and the highest richness occurring at Site 8 in October 2003 and at Site 7 in July 2005. Mean Shannon Diversity and Pielou's evenness were highest at the Little Blanco River (1.87 and 0.80) and Cypress Creek (1.51 and 0.68) and lowest at Site 5 (1.07 and 0.49, respectively). Highest individual sample diversities occurred at the Little Blanco River in July 2005 (2.37) and July 2004 (2.09). Lowest individual sample diversities occurred at Site 1 (0.33, July 2004) and Site 4 (0.54, January 2005).

Cyprinids generally were more abundant in the mainstem and Cypress Creek, systems with more persistent flows, whereas centrarchids and poeciliids generally were more abundant in Little Blanco River, a stream with intermittent flows near its confluence with the Blanco River mainstem. Cyprinidae (81%) was the most abundant family in the Blanco River, followed by Poeciliidae (8%) and Centrarchidae (7%). Likewise, Cyprinidae (69%) was the most abundant family in Cypress Creek, followed by Centrarchidae (20%), Poeciliidae (4%), Percidae (4%), and Cichlidae (3%). In the Little Blanco River where pool habitats were common, Centrarchidae (42%) was the most abundant family, followed by Cyprinidae (38%), Poeciliidae (14%), Percidae (4%), and Catostomidae (2%).

Among fishes with a relatively small geographic range, *Dionda nigrotaeniata* was only present in the tributaries with its greatest abundance occurring in the Little Blanco River (1.8%). Three individuals of *Macrhybopsis marconis* were collected at Site 8. *Notropis amabilis* occurred at all sites and was abundant (> 2%) at six sites. *Moxostoma congestum* occurred at six sites and was most abundant at the Little Blanco River (2.2%)

and the Reservoir (0.9%). Fish initially identified as *Micropterus treculii* based on morphology (Hubbs 1991) were present at four sites with a relative abundance < 0.3% at all sites and 0.04% overall. However, subsequent genetic analyses conducted on the Blanco River population failed to detect pure *M. treculii* in the population, which indicates that only *Micropterus dolomieu* x *Micropterus treculii* hybrids exist in the Blanco River drainage (Littrell et al. in press).

Canonical Correspondence Analysis

Physical parameters, site, and season accounted for 40% ($P < 0.01$) of the variation in the Blanco River drainage fish assemblage. Pure effects of physical parameters accounted for 15.3% ($P < 0.01$), site accounted for 11.2% ($P < 0.01$), and season accounted for 2.3% ($P < 0.01$) of fish assemblage variation. Two- and three-way shared effects among physical parameters, site, and season accounted for 10.7% of fish assemblage variation. Physical parameters with the strongest positive centroids for the first canonical axis (CA I) were riffle (1.57), side channel (0.64), and maximum velocity (0.56). Physical parameters with the strongest negative centroids for CA I were reservoir (-1.28), pool (-0.78), and silt (-0.54). Within the mainstem of the Blanco River, CA I centroids were negative for sites 1 through 3 and positive for sites 4 through 8. Among the tributaries, Little Blanco River had a negative centroid whereas Cypress Creek had a positive centroid. CA I expressed a gradient from upstream sites with slow current velocities, greater depths, silt substrate, and detritus to downstream sites with faster current velocities, shallower depths, and cobble substrate. Physical parameters with the strongest positive centroids for the second canonical axis (CA II) were riffle (1.00), backwater (0.54), and detritus (0.27). Physical parameters with the strongest negative centroids for CA II were sand (-0.72), plunge pool (-0.70), run (-0.52), and boulder (-0.48). CA II expressed a weaker habitat gradient from shallow backwaters to deeper runs with sand substrate. The strongest negative loadings of CA II described the habitat at Site 5. Summer and fall had negative centroids for CA I and winter and spring had positive centroids for CA I, however, these centroids were generally weak.

Species with the strongest positive associations with CA I include *Percina sciera*, *Pimephales promelas*, *Percina carbonaria*, *Etheostoma spectabile*, and *Ameiurus natalis* (Appendix 1.5). Species with the strongest negative associations with CA I include *Lepomis microlophus*, *Micropterus salmoides*, *Cyprinus carpio*, *Cyprinella lutrensis*, and *Lepomis gulosus*. Along the habitat gradients expressed by CA I and CA II, *Percina carbonaria* ($N = 20$) and *Etheostoma spectabile* ($N = 540$) were strongly associated with riffles having high current velocities, shallow depths, and intermediate-size substrate such as gravel and cobble. Cobble and gravel substrates were dominant at Site 8 where *Etheostoma spectabile* relative abundance was highest among mainstem sites (Appendix 1.4). *Cyprinella venusta* ($N = 11{,}918$) and *Gambusia affinis* ($N = 2{,}419$) were associated with intermediate currents and showed no strong substrate affinities. *Campostoma anomalum* ($N = 1{,}160$) and *Ictalurus punctatus* ($N = 110$) were associated with intermediate current velocities, shallow depths, and cobble substrate. *Notropis amabilis* ($N = 3{,}308$) and *Micropterus dolomieu* ($N = 33$) were associated with intermediate current velocities and coarse substrates. Fish species associated with deep, low-velocity habitats with greater amounts of vegetation and detritus included *Cyprinella lutrensis* ($N = 5$), *Cyprinus carpio* ($N = 3$), *Dionda nigrotaeniata* ($N = 36$), *Pimephales vigilax* ($N =$

4,136), *Moxostoma congestum* ($N = 115$), *Lepomis cyanellus* ($N = 165$), *Lepomis gulosus* ($N = 6$), *Lepomis macrochirus* ($N = 795$), *Lepomis megalotis* ($N = 511$), *Lepomis microlophus* ($N = 10$), and *Micropterus salmoides* ($N = 196$). Species strongly associated with pool and reservoir habitats included *Cyprinus carpio*, *Cyprinella lutrensis*, *Pimephales vigilax*, *Moxostoma congestum*, *Lepomis gulosus*, *Lepomis macrochirus*, and *Lepomis megalotis*.

Although pure site effects explained only 11.2% of fish assemblage variation, several species showed strong site affinities. *Macrhybopsis marconis* ($N = 4$), *Astyanax mexicanus* ($N = 15$), *Poecilia latipinna* ($N = 1$), *Lepomis punctatus* ($N = 16$), and *Percina sciera* ($N = 1$) occurred exclusively at sites 7 and 8 (Appendix 1.4) downstream from the lowermost low-head dam on the Blanco River and nearest to the confluence with the San Marcos River. *Cyprinus carpio* was collected only from the Reservoir Site and in a deep pool at Site 4. *Etheostoma spectabile*, *Percina carbonaria*, and *Percina sciera* were present downstream of the falls at Site 4 but were absent above the falls. *Dionda nigrotaeniata* was present in the tributaries (Little Blanco River and Cypress Creek) but was absent from mainstem sites.

Analysis of Similarities

Sites were grouped into four categories to test for influence of low-head dams on fish assemblages. Site categories were "adjacent to low-head dams (<1 km)" (sites 1 and 7); "impounded by a low-head dam" (Reservoir Site); "distant from a low-head dam (>1 km)" (sites 3, 4, 5, 6, and 8), and "tributaries" (Little Blanco River and Cypress Creek). Analysis of similarities (Appendix 1.6) indicated that differences existed ($P < 0.01$)

among fish assemblages relative to low-head dams. Pair-wise tests indicated significant differences between impounded and adjacent site assemblages ($P < 0.01$), impounded and distant site assemblages ($P = 0.04$), adjacent and tributary site assemblages ($P < 0.01$), and distant and tributary site assemblages ($P < 0.01$). Fish assemblage differences were not detected between impounded and tributary site assemblages ($P = 0.07$) or between adjacent and distant site assemblages ($P = 0.70$). Fish assemblage differences between impounded and adjacent and between impounded and distant assemblages were attributed to the large number of *Pimephales vigilax* (61%) and low number of *Cyprinella venusta* (20%) in the Reservoir Site.

DISCUSSION

Physical habitat parameters, site effects, and seasonal effects explained significant amounts of variation within the Blanco River fish assemblage. However, the amount of variation explained by season was small whereas physical habitat parameters explained a relatively large amount of variation. Few species (i.e. *Notropis volucellus*, *Pimephales vigilax*, *Gambusia affinis*, and *Micropterus salmoides*) showed strong seasonal trends in relative abundances. Relative abundance was highest for *G. affinis* and *N. volucellus* in the Fall. Mean length was lowest for both species in fall, thus the higher relative abundances may represent an abundance of juveniles prior to significant seasonal mortality (Matthews 1998). The peak in *P. vigilax* relative abundance occurred in Winter and resulted from a single seine haul at the Reservoir Site. *Micropterus salmoides* abundance was highest in Summer when age-0 fish were small and unable to escape from seines. This presumably resulted in a greater capture rate for smaller individuals similar to that reported by Weinstein and Davis (1980).

The extent of site associations was highly variable among species. *Dionda nigrotaeniata* typically inhabits spring-influenced headwaters (Hubbs et al. 1991) and showed a strong site affinity for the Little Blanco River and Cypress Creek. *Ameiurus natalis* was collected exclusively in Cypress Creek and is often associated with small, clear, rock- or gravel-bottomed streams (Robison and Buchanan 1988). All percids (i.e., *Etheostoma spectabile*, *Percina carbonaria*, and *Percina sciera*) were absent above Site 4. This observation is consistent with reports by the Texas Game and Fish Commission (now Texas Parks and Wildlife; 1957), indicating that the falls at Site 4 present an apparent barrier to the upstream movement of fishes, and contributes to an abrupt discontinuity in assemblage variation. *Macrhybopsis marconis*, *Astyanax mexicanus*, *Poecilia latipinna*, and *Percina sciera* occurred only at sites 7 and 8. These species were rare, and likely represent vagrants from the San Marcos River, as larger, more stable water bodies serve as species pools from which less stable upstream habitats are colonized (Whiteside and McNatt 1972). Spatial variation within the Blanco River fish assemblage across geographically distant sites (i.e. tributary vs. mainstem and upstream vs. downstream) likely represents differences in stream processes (Wilkinson and Edds 2001). The relatively small amounts of spatial and seasonal variation suggest adequately stable habitats with assemblages primarily structured by local habitat parameters (Meador and Matthews 1992).

Physical habitat parameters were found to be the primary factors structuring fish assemblages. The first two canonical axes of CCA both represented gradients best described as velocity and substrate gradients. Along these gradients, centrarchids were generally more abundant in lentic type habitats with lower velocities and greater percentages of silt substrate and vegetation whereas percids were more abundant in shallow lentic habitats such as riffles dominated by cobble and gravel substrate. Cyprinids, however, exhibited a much wider range of habitat associations. For example, *Campostoma anomalum* was strongly associated with shallow riffles whereas *Notropis amabilis* was associated with runs and *Cyprinella venusta* did not exhibit any strong associations. Gorman and Karr (1978) noted that stream depth, current velocity, and substrate are important in structuring stream fish assemblages. Greater variability in these components results in increased habitat complexity which regulates local assemblages of fish as stream fish are commonly habitat specialists (Mendelson 1975).

The low-head dam at the Reservoir Site created a distinctly lentic habitat in which centrarchids were abundant, as were the ubiquitous *Cyprinella venusta* and *Pimephales vigilax*. Santucci et al. (2005) reported major differences in habitat quality between impounded and free flowing reaches such as higher turbidity, lower dissolved oxygen minima (as low as 2.5 mg/L), and homogenization of habitats. Homogenization of habitats can lead to increased abundance of generalist native species (Scott and Helfman 2001) and loss of native stream specialists (Boet et al. 1999). In addition to habitat alterations, fish assemblages within reservoirs may be altered by purposeful introductions of sportfish species as well as incidental introductions via bait-buckets (Taylor et al. 2001). The Reservoir Site ranked sixth in species richness with 17 species collected, while the mean species richness across all sites was 17.3 species. Although species richness at the Reservoir Site was nearly average for the sites, the structure of the assemblage at this site was significantly different from that of sites both adjacent to and distant from reservoirs. *Campostoma anomalum*, *Micropterus dolomieu*, and *Micropterus treculii*, which typically inhabit swifter waters, were rare or absent from the Reservoir Site, but were present and more abundant at Site 3. Gillette et al. (2005) and Taylor et al. (2001) reported similar shifts in assemblages with lower abundances of fishes normally associated with higher current velocities, such as percids and stream dwelling micropterids. Additionally, the absence of percids at the Reservoir Site is attributed to the natural barrier present at Site 4 and is likely not the result of the low-head dam.

The Little Blanco River consisted mostly of a series of pools due to low-flow conditions during the study, and was dominated by lentic species (i.e., centrarchids). In contrast, the consistently flowing Blanco River and Cypress Creek were dominated by cyprinids. Periods of low flow and increased centrarchid abundance likely resulted in the relatively high degree of similarity between tributaries and the Reservoir and a lack of a significant difference between the two treatment groups in analysis of similarities. Although no differences were detected among assemblages at sites either adjacent to or distant from impoundments, it cannot be concluded that low-head dams do not impact the fish assemblage as there is no comparison to the pre-impoundment assemblage structure.

Although no pre-impoundment assemblage data are available, several changes in occurrence have taken place since the Texas Parks and Wildlife Department survey in 1957. *Dorosoma cepedianum*, *Carpiodes carpio*, *Notemigonus crysoleucas*, *Notropis buchanani*, *Ameiurus melas*, *Pomoxis annularis*, and *Etheostoma lepidum* were not collected in our sampling efforts, but were present, though relatively rare, in the 1957 collection. Species present in our collections but absent from the 1957 collection include the introduced species *Pimephales promelas*, *Poecilia latipinna*, and *Micropterus dolomieu*, as well as *Percina sciera* which is likely a vagrant from the San Marcos River.

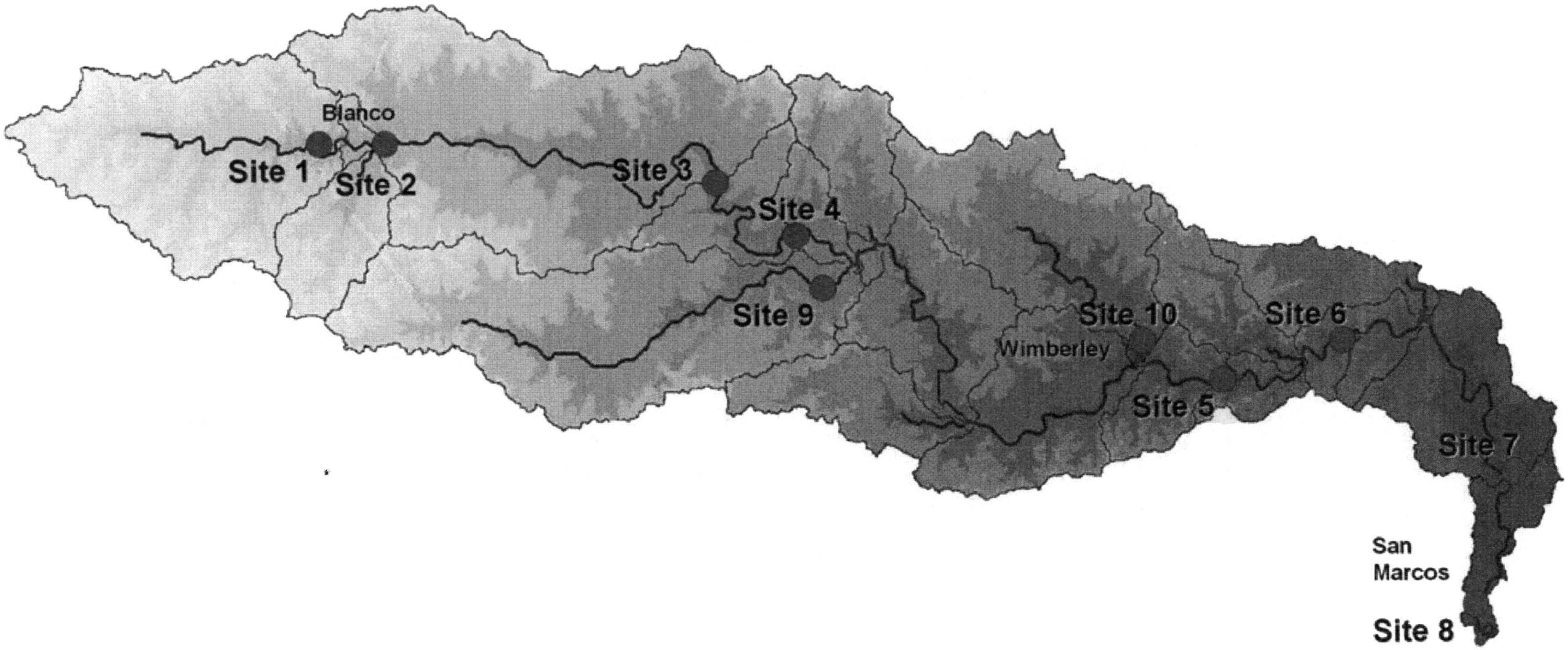
Physical habitat parameters, site, and season all influenced Blanco River fish assemblage. However, there is a substantial amount of variability in the assemblage that is not explained by the present model. This variability may reflect unmeasured or inestimable factors such as land use, riparian vegetation, or biological interactions that act to structure the fish assemblage at the local and watershed level. Although these unmeasured factors are likely to influence the structure of the fish assemblage, current velocity, depth, and substrate often adequately account for variation in fish assemblages at a local scale (Gorman and Karr 1978; Schlosser 1982; Cantu and Winemiller 1997;

Walters et al. 2003; Williams et al. 2005). Such factors, comprising major aspects of stream morphology, are likely to shift in response to anthropogenic disturbance (Odemerho 1984; Golladay et al. 1987), and the fish assemblage can be expected to track with such shifts.

Among the threats to the Blanco River and other Texas Hill Country streams, excessive groundwater pumping is the greatest (Bowles and Arsuffi 1993). Continued excessive pumping, especially during drought, will likely result in a loss of spring associated headwater specialists such as *Dionda nigrotaeniata.* As rapid urbanization continues, changes in geomorphology and hydrology will likely include increased impervious cover resulting in higher but shorter-duration hydrograph peaks and changes in substrate composition resulting from siltation, similar to changes in nearby and urbanized Waller Creek in Austin, TX (Swezey 1991). Such changes in stream characteristics will likely influence changes in fish assemblage structure as several species in the Blanco River are strongly associated with particular substrate types (e.g., percids).

APPENDICES

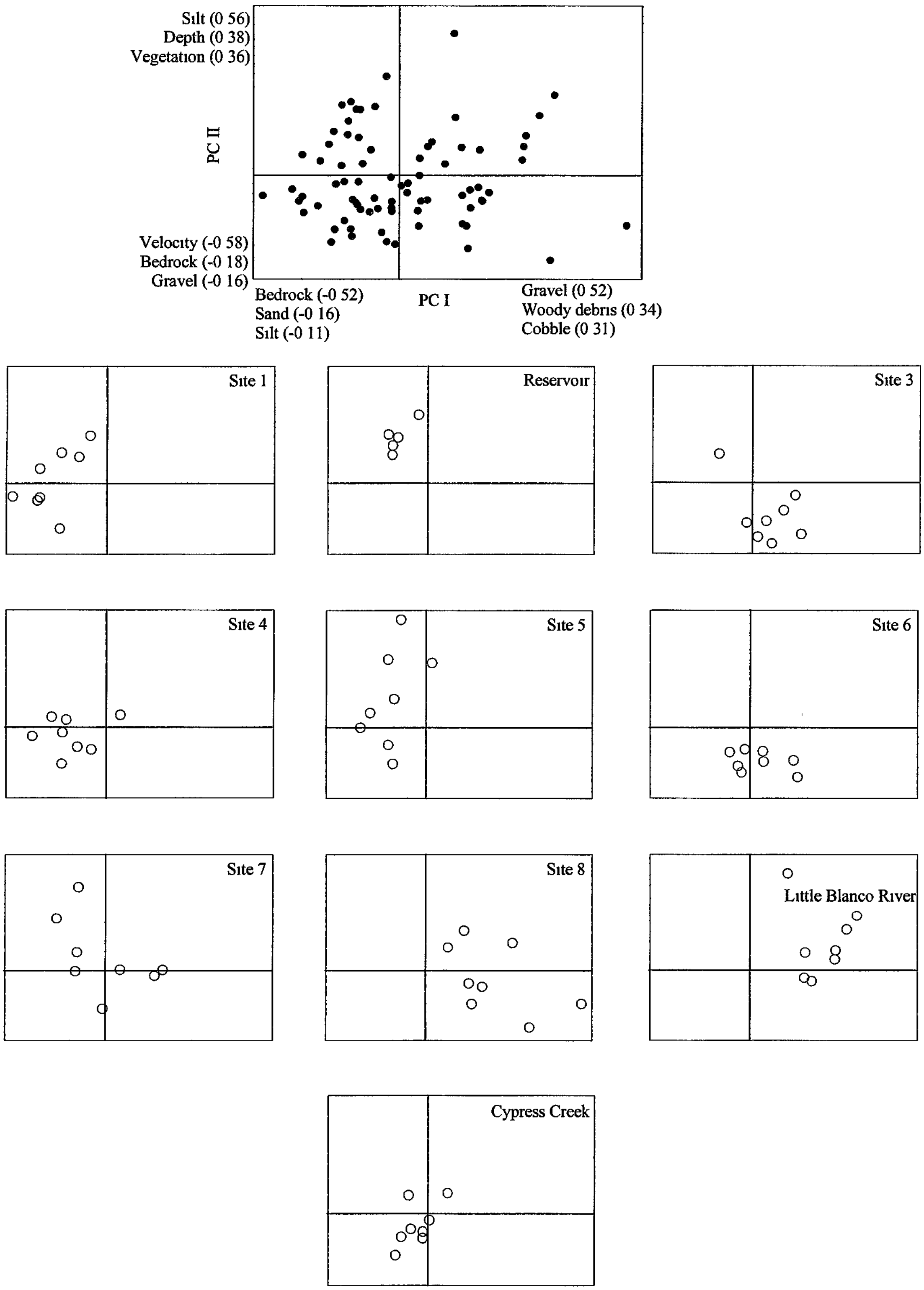
Appendix 1.1: Map of sampling sites within Blanco River watershed.



Appendix 1.2: Mean (± SD) physical habitat parameters across sampling dates for sites sampled between October 2003 and July 2005.

			(Reservoir)						Main Stem								Little Blanco		Cypress Creek	
	1		2		3		4		5		6		7		8		9		10	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Width (m)	15 1	4 8	35 0	0 0	11 9	3 3	13 9	2 9	44 1	3.2	19 6	5.6	16 1	4.7	12.5	8 6	11 5	2 3	11 3	3.0
Depth (m)	0 4	0.3	0 8	0 0	0.5	0 1	0 6	0.2	0.5	0.2	0 4	0 1	0 4	0 1	0 5	0 1	0.9	0 3	0 3	0 0
Current Velocity (m/s)	0.2	0 3	0 0	0.0	0.2	0.1	0.2	0 2	0 3	0 2	0 6	0.2	0 4	0 2	0 4	0 4	0 1	0 1	0 2	0 2
Substrate Type (%)																				
Silt	25.9	31 9	43 3	6 2	25 0	23 0	0.9	2 4	2 8	3 6	3 3	3 2	20 9	29 4	17 8	24 3	11.6	27.7	13.3	17 1
Sand	3.8	7 4	0.0	0 0	1.4	2.6	0.0	0 0	6 3	10 3	0.0	0 0	0 5	1 4	0 4	0 5	0.1	0 4	0 5	1 4
Gravel	2.8	4 4	1 7	1 5	24.9	20 6	12 1	12 6	15 9	11.3	45.4	11.4	31 2	30.2	47.9	31 4	77 8	26 4	9 5	13 6
Cobble	1.3	2 3	1 7	1.5	31 5	23.9	1.3	2 3	8.1	6.9	8 3	7 9	9 1	8 8	33 2	30 2	8.9	12.8	29.6	20.4
Bedrock	64.9	37.2	51 7	1 5	13 8	12 7	83.1	16 4	53.4	22 9	42 2	18 0	38.2	27 3	0 2	0 4	0.0	0 0	38 2	21 3
Boulder	1 3	2 3	1 7	1 5	3.5	3 5	2.7	5 7	13.4	14 6	0 9	1 6	0.0	0 0	0 5	1.1	1 5	1 8	9 0	14 0
Detritus (%)	12.5	35.4	7 5	9.8	0 8	2 3	0 0	0 0	0 0	0.0	3.2	8 3	0.0	0.0	0.0	0.0	10.8	20.3	18 8	24 7
Woody Debris (%)	0 8	0 7	0.8	0 6	1.8	1 3	2 4	4 8	1 7	1 5	2 8	5 8	1.7	1 8	9 3	13 3	3.5	1.6	4 9	2 3
Vegetation (%)	2 6	3 5	22 3	21 2	6 7	8.0	11.1	26.9	0 0	0 0	1.6	2.6	5 8	4 6	21.6	29 4	40 6	44.4	5 1	13 1

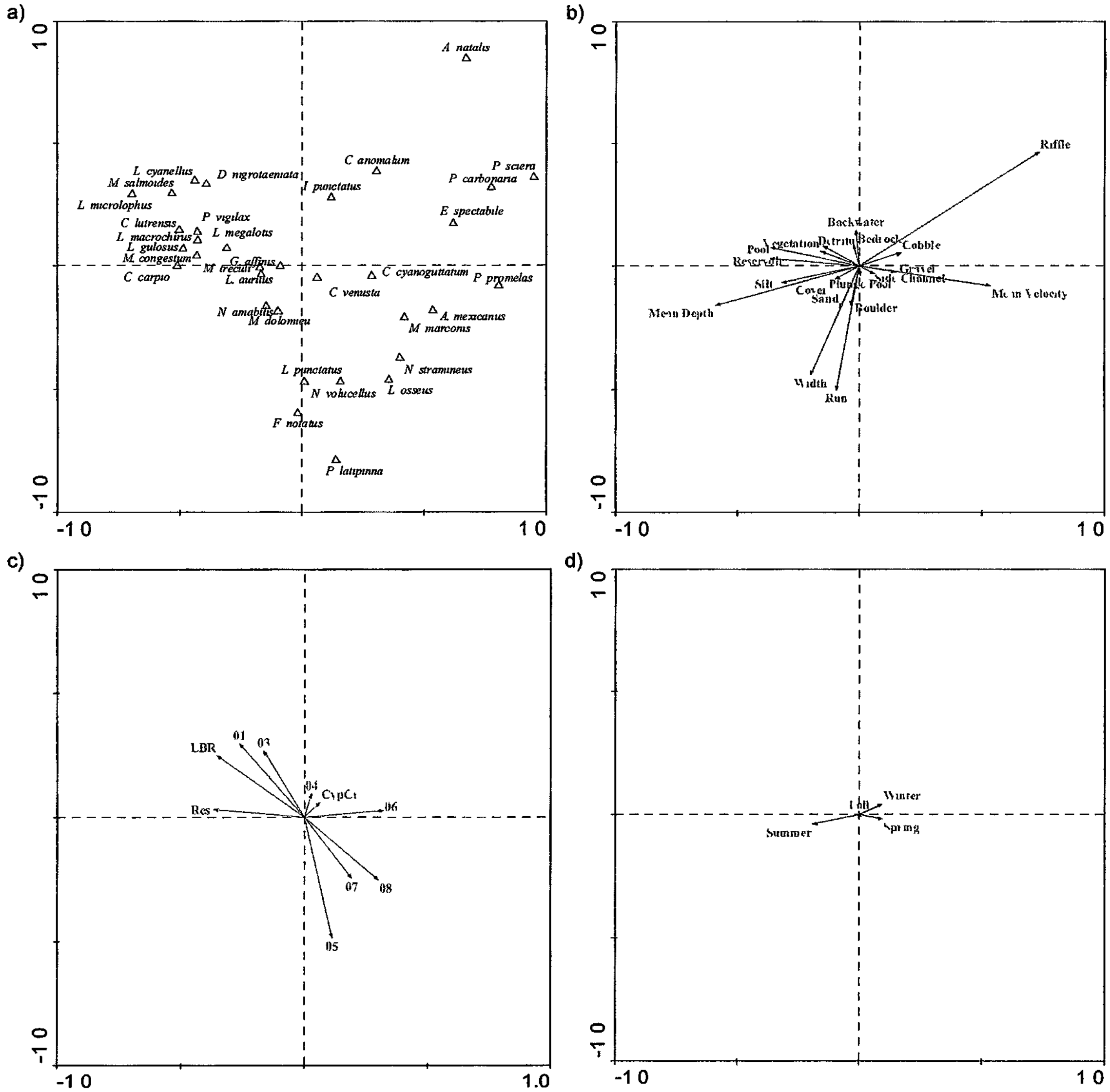
Appendix 1.3: PCA habitat plots for the Blanco River, Little Blanco River, and Cypress Creek.



Appendix 1.4: Relative abundance across dates for the Blanco River, Little Blanco River, and Cypress Creek.

	Site									
	1	Res.	3	4	5	6	7	8	LBR	CypCr
Lepisosteus osseus	-	-	-	-	-	0.1	-	-	-	-
Campostoma anomalum	7.8	0.1	4 4	13.4	0 2	5.2	5.0	2.2	4.9	12.3
Cyprinella lutrensis	-	0.1	0.1	-	-	-	-	-	-	-
Cyprinella venusta	70.0	20 2	41 1	37.6	61.6	36.2	65.2	39 7	5 8	33.9
Cyprinus carpio	-	0.03	-	0 1	-	-	-	-	-	-
Dionda nigrotaeniata	-	-	-	-	-	-	-	-	1.8	0.4
Macrhybopsis marconis	-	-	-	-	-	-	-	0 1	-	-
Notropis amabilis	1.7	11 5	28.3	19 7	8.4	1.4	5.2	19 5	11 3	21 8
Notropis stramineus	-	-	-	1.4	16.9	4 2	0.3	0.2	-	-
Notropis volucellus	-	-	-	3.0	6.1	1 3	7 7	20.8	9 3	0 2
Pimephales promelas	-	-	-	-	-	0 03	-	0.02	-	-
Pimephales vigilax	0.03	61.3	0.3	0.5	-	-	-	0.3	4.5	-
Moxostoma congestum	-	0.9	0 2	0 1	-	0 03	0.1	0.1	2 2	0 4
Astyanax mexicanus	-	-	-	-	-	-	0 3	0 2	-	-
Ameiurus natalis	-	-	-	-	-	-	-	-	-	0 2
Ictalurus punctatus	0.6	0.1	1.5	2 1	0.1	0 4	0.2	0 2	-	0.1
Fundulus notatus	-	-	-	-	-	-	0.9	0 1	-	0.4
Gambusia affinis	3.9	1 2	2 4	13 5	3.0	38.6	2 6	6.1	14.3	3 8
Poecilia latipinna	-	-	-	-	-	-	-	0.02	-	-
Lepomis auritus	3.0	2 0	9.9	2 6	1 1	1 3	5.0	2 1	11 4	16 0
Lepomis cyanellus	0.8	0 03	3 5	-	0 03	-	0.1	-	4 0	0 1
Lepomis gulosus	0.1	0 03	-	-	-	-	-	-	-	0 2
Lepomis macrochirus	4.5	2.0	0 2	-	0.4	3.2	0 8	1 3	17.9	0.2
Lepomis megalotis	4.0	0.3	6.1	4.5	0.9	0.6	0 3	0.2	6 0	2.7
Lepomis microlophus	0 1	0.02	0.2	-	-	-	-	-	0.2	-
Lepomis punctatus	-	-	-	-	-	-	0.6	-	-	-
Micropterus dolomieu	-	-	0 2	0.6	0.3	0.1	0.04	-	0.2	0.5
Micropterus salmoides	3.5	0.3	0.5	0.5	-	0 1	0.04	-	2.4	0.2
Micropterus treculi	-	-	0 3	-	0 1	-	-	0.1	0.1	-
Etheostoma spectabile	-	-	-	-	0 6	5.4	3 3	3.3	3.8	3.6
Percina carbonaria	-	-	-	0.6	-	-	0.04	0.3	0.1	-
Percina sciera	-	-	-	-	-	-	-	0.02	-	-
Cichlasoma cyanoguttatum	-	0.1	1 0	-	0.4	2.0	2.3	3.2	-	3 1
N =	3,303	6,586	1,766	1,213	2,928	3,326	2,820	4,319	1,731	1,273

Appendix 1.5: CCA ordination plots of (a) species, (b) physical habitat parameters, (c) site, and (d) season for the Blanco River October 2003 through July 2005.



Appendix 1.6: ANOSIM global and pair-wise tests.

			R	P value
Global Test			0.264	< 0.01
Pairwise Tests				
Near	vs.	Impounded	0.654	< 0.01
Near	vs.	Distant	-0.063	0.70
Near	vs.	Tributary	0.585	< 0.01
Impounded	vs.	Distant	0.346	0.04
Impounded	vs.	Tributary	0.267	0.07
Distant	vs.	Tributary	0.386	< 0.01

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CHAPTER II

REPRODUCTIVE ECOLOGY AND DIET OF THE GRAY REDHORSE, *MOXOSTOMA CONGESTUM*

ABSTRACT

Reproductive ecology and diet of gray redhorse *Moxostoma congestum* were examined in a Texas Hill Country stream and a central Texas reservoir from September 2004 through August 2005. Temporal patterns in gonadosomatic index and oocyte diameter frequency indicated that *M. congestum* spawns over a very brief period in late March or early April. *Moxostoma congestum* was found to be an opportunistic benthic invertivore and diets differed between stream and reservoir habitats and among seasons. Though habitat degradation is of concern for *M. congestum*, it has persisted in habitats disturbed by low-head dam and mainstem reservoir construction as its opportunistic feeding strategy allows it to be adaptable to lentic systems.

INTRODUCTION

Moxostoma congestum is part of the autochthonous element of the ichthyofauna of the western Gulf Slope from the Brazos River drainage to the Nueces River drainage in the western Gulf Slope (Conner and Suttkus 1986). Additionally, its natural range extends into the Rio Grande drainage of Texas, New Mexico, and eastern Mexico (Jenkins 1980). Although *M. congestum* is listed as stable by Warren et al. (2000), Williams et al. (1989) listed it as of special concern due to habitat degradation. The streams in which *M. congestum* is found face several threats (Bowles and Arsuffi 1993) including impoundment. Despite over 6,000 dams in Texas's waters (Shuman 1995), *M. congestum* has persisted in many of these impounded systems.

Moxostoma congestum is typically found in upland and lowland rivers and streams (Conner and Suttkus 1986) and is often associated with deep, low current velocity, low turbidity habitats (Rose and Echelle 1981; Cantu and Winemiller 1997). The description of diet by Cowley and Sublette (1987) is consistent with the diet predicted by Eastman (1977) for most *Moxostoma* species based on pharyngeal teeth type, consisting of primarily benthic insects and thin shelled mollusks. However, the diet description of Cowley and Sublette (1987) is based on 5 individuals ranging from 100-250 mm and might not reflect adult diets or seasonal changes in diet. Reproductive behavior of *M. congestum* was described by Martin (1986) as occurring in March and April with spawning occurring over a 2 day period and was similar to that of *M. lachneri* (Burr 1979). However, information on spawning preparedness and minimum size at first reproduction is lacking.

The purpose of this study was to provide information on diet and reproduction of *M. congestum* in a central Texas stream and in a mainstem reservoir. Specific objectives were to describe overall food habits as well as seasonal and habitat associated differences in diet and to describe trends in gonadosomatic index (GSI) and oocyte maturation. Additionally, comparisons among stream and mainstem impoundment populations were made as *M. congestum* has persisted in several impounded systems.

MATERIALS AND METHODS

The Blanco River drains an area of 1,067 km^2 in Kendall, Comal, Blanco, and Hays counties, Texas, before its confluence with the San Marcos River. It is generally a shallow to moderate depth wadeable stream with substrate dominated by bedrock and gravel. Several low-head dams have been constructed on the Blanco River from the headwaters to its confluence with the San Marcos River. Canyon Lake (Comal County, Texas) is a 3,331 ha reservoir on the Guadalupe River with a mean depth of 13 m and a maximum depth of 48 m.

Fish were collected monthly from September 2004 through August 2005 from the pool of a low-head dam on the Blanco River near the town of Blanco, and from Canyon Lake. In the Blanco River, individuals were collected with experimental gill nets. In Canyon Lake, individuals were collected by boat electrofishing. Each individual was measured for total length to the nearest millimeter and weighed to the nearest 25 g. All individuals were immediately killed by pithing. The gut tract and gonads were removed and preserved in a 10% formalin solution for diet and reproductive analysis, respectively.

In the laboratory, gonads were weighed to the nearest 0.01 g, and diameters of 50 oocytes from each mature female were measured to the nearest 0.01 mm using a Nikon SMZ 1500 microscope with a Nikon Digital Sight DS-L1 digital camera. A gonadosomatic index [GSI; (gonad weight/total weight)*100] was calculated for each individual. Mean GSI was calculated for males and females for each month. Frequency histograms of oocyte diameter were constructed to determine spawning periodicity and frequency. Gut contents, from the esophagus through the first loop of the gut, were counted, weighed and identified to order, when possible, for each fish. Unweighted means for each food type category were calculated for each month. Bray-Curtis similarity indices were calculated for diets of each pairwise combination of individuals. Total weight of food type was used because several food types are not meaningfully countable (e.g., crushed mollusk shells). Diet data were standardized as proportions of total weight of gut contents. Analysis of similarities (ANOSIM; Clarke and Green 1988; Clarke 1993) was used to test for differences in diet between the Blanco River and Canyon Lake and for seasonal effects.

RESULTS

Gray Redhorse Reproduction

Sixty *Moxostoma congestum* were collected for reproductive analysis. Fish were not collected in November 2004 due to flooding. Among individuals collected, six (204-337 mm TL) were immature, and their sex could not be determined by macroscopic gonadal examination. Mature females ($N = 34$) ranged from 260 to 497 mm TL. Mature males ($N = 20$) ranged from 285 to 468 mm TL. There was no difference in mean TL between males and females ($t_{0.05, 52} = 1.91$, $P = 0.06$). Sex ratio was not different from 1:1 ($\chi^2_{0.05,1} = 3.63$, $P = 0.06$).

Temporal patterns in GSI (Appendix 2.1) and oocyte diameter frequency (Appendix 2.2) indicate female gonadal quiescence from May to August. Recrudescence began in October with 100% of females possessing developing oocytes. Mean GSI was greatest in February (12%) and diminished in March (5.4%) and April (1.9%) before quiescence. Spent ovaries occurred in 75% of females in April and 100% were spent in May. Mean GSI was below 1% from May to September. Two size classes of oocytes were present from January to March with the larger size class absent in April.

Gray Redhorse Diet

Sixty two *Moxostoma congestum* were collected for diet analysis. Mean weight of gut contents was 0.31 g. Food items consisted of aquatic insects (47%, unweighted mean across months), mollusks (42%), amphipods (5%), ostracods (3%), fish (1%), Hydrachnida (< 0.1%), algae (< 0.1%), plant seeds (< 0.1%), and unidentifiable insect parts (2%). Mean percent occurrence of detritus by month was 91% and mean occurrence of substrate was 27%. Mean weight of detritus was 0.3 g and mean weight of substrate was 0.04 g. Mean percent of empty stomachs was 6% across months (unweighted mean).

Across months, aquatic insects comprised the majority of *Moxostoma congestum* diets from October to May ranging from 42% in January to 71% in February (Appendix 2.3). From June to September diets were primarily comprised of mollusks ranging from 56% in August to 84% in June. Most common aquatic insects were Diptera and Ephemeroptera larvae followed by Tricoptera, Odonata, Coleoptera, and Megaloptera. The most common mollusk was *Corbicula fluminea*. Other mollusks, including gastropods and bivalves, were not identifiable below class because their shells were too finely crushed. Amphipods were relatively abundant in *M. congestum* stomachs in October (17%) and December (28%) while ostracods were abundant in December (14%) and March (19%).

Diets were significantly different between the Blanco River and Canyon Lake (Global R = 0.103, $P < 0.01$; Appendix 2.4) and between seasons (Global R = 0.067, $P = 0.04$). Summer *M. congestum* diets differed significantly from both Winter (R = 0.183, $P < 0.01$) and Spring (R = 0.117, $P = 0.03$) diets. No other pairwise comparisons among seasons demonstrated significant differences in diet.

DISCUSSION

Temporal patterns in GSI and oocyte diameter frequency suggested that *Moxostoma congestum* spawns a single clutch of eggs or multiple clutches of the same egg cohort over a brief spawning period in late March or early April. Resolution of data at the monthly level did not allow distinction between the two patterns of reproduction (i.e., release of a single clutch or multiple clutches). However, Martin (1986) reported spawning in Walnut Creek, Texas, to occur within a two-day period with repeated spawning events within schools, lending support to the pattern of multiple clutches spawned over a brief spawning period. Several species of *Moxostoma* spawn over a short 1 to 3 day period (Burr 1979; Jenkins and Jenkins 1980; Kwak and Skelly 1992) whereas *M. valenciennesi* spawns over a period extending up to 8 days (Cooke and Bunt 1999). In April, large vitellogenic oocytes were present in *M. congestum*, but these likely represented atretic oocytes undergoing reabsorption (Khanna and Pant 1967). Although sex could be identified in individuals as small as 260 mm TL, developing oocytes were not found in individuals <392 mm. This suggested *M. congestum* is a late maturing species. Minimum age of mature *M. valenciennesi* (Cooke and Bunt 1999) and *M. erythrurum* (Kwak and Skelly 1992) of similar size is estimated at 5 to 6 years.

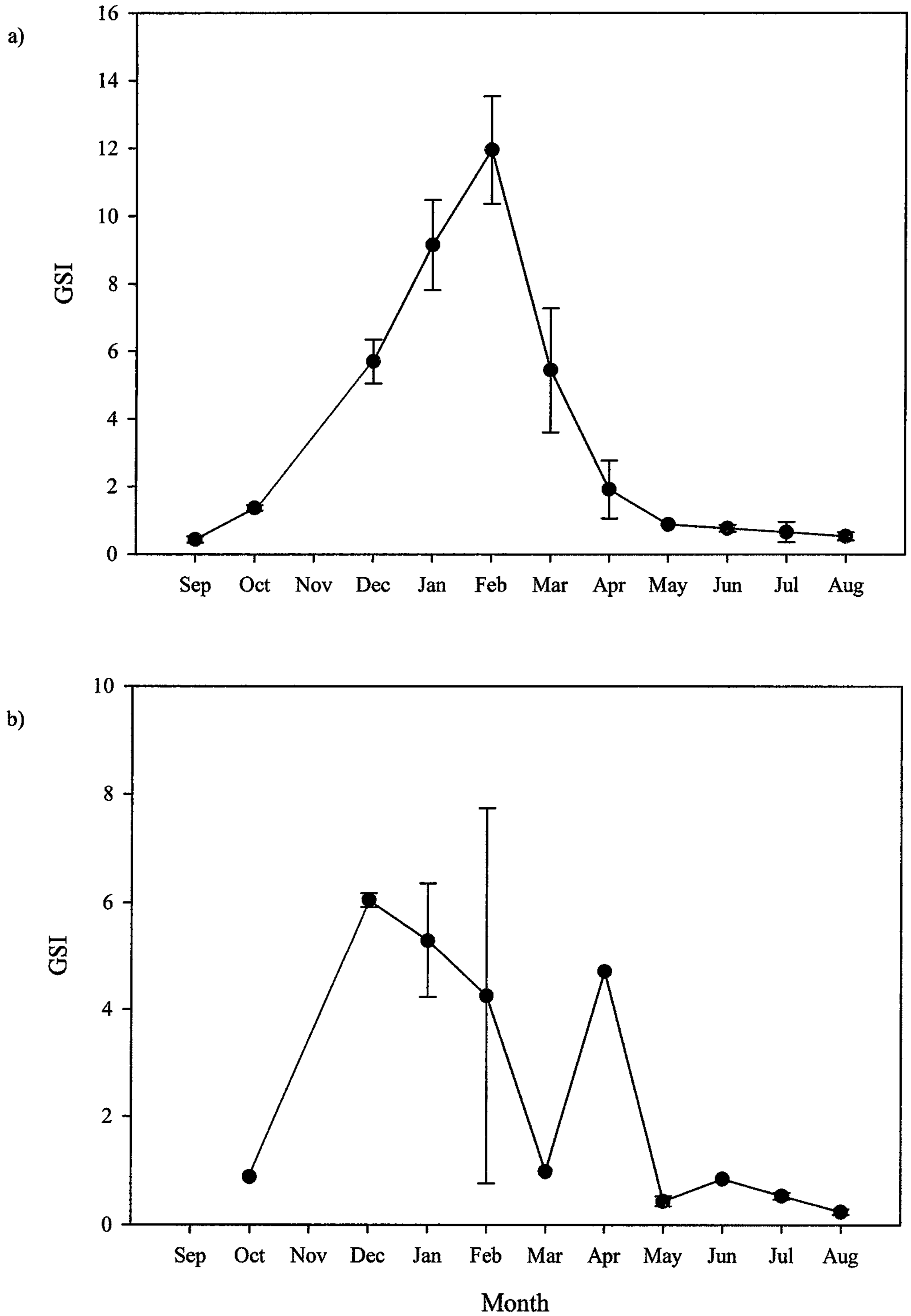
Analysis of diet suggested that *Moxostoma congestum* is a benthic invertivore and confirms the prediction of Eastman (1977) of a diet consisting primarily of benthic insects and thin shelled mollusks. The presence of algae, plant seeds, and fish in gut contents also suggested that *M. congestum* is somewhat opportunistic. Among aquatic insects dipterans, particularly chironomids, were the most common food item suggesting that feeding takes place in calm waters of pools or stream edges (Timmons and Ramsey

1983). Ephemeropterans comprised a large portion of the aquatic insects consumed by fish in this study. However, Cowley and Sublette (1987) did not report Ephemeroptera in *M. congestum* diets. Differences in diet between *M. congestum* from the Blanco River and Canyon Lake likely reflected differences in available foods and subsequent diet shifts in a novel environment (Loureiro-Crippa and Hahn 2006). Mollusks were more abundant in fish from Canyon Lake whereas ephemeropterans were more abundant in fish from the Blanco River. Across seasons, significant differences in diet only occurred between winter and summer, and between spring and summer. Corresponding to these differences, winter and spring diets were dominated by aquatic insects whereas summer diets were dominated by mollusks. It is not clear whether these seasonal differences in diet reflect differences in availability or consumption.

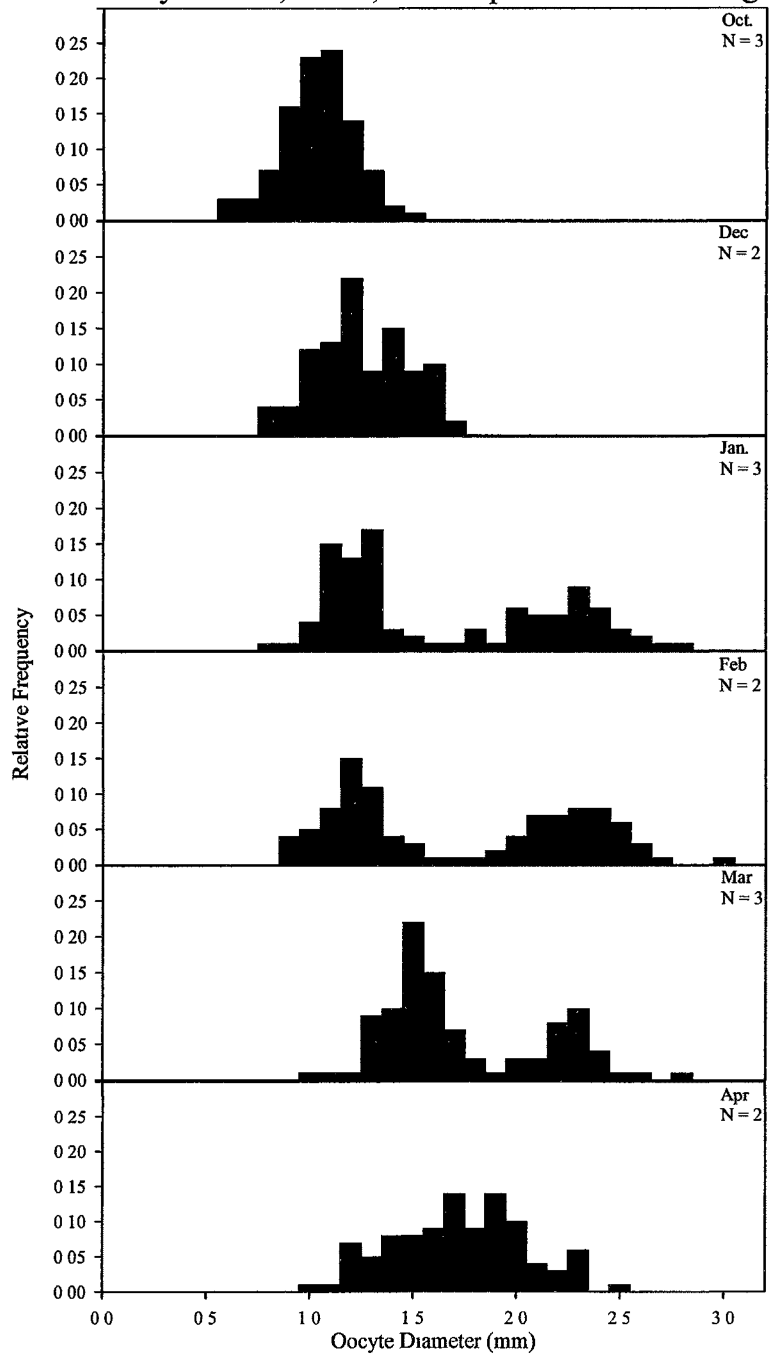
Despite being listed as a species of special concern by Williams et al. (1989) due to habitat degradation, *Moxostoma congestum* has persisted in systems disturbed by low-head and mainstem dams. For example, *M. congestum* has persisted in the Blanco River and was most abundant in deep pools of tributaries and in the impoundment above a low-head dam. The opportunistic-invertivorous feeding strategy of *M. congestum* has been associated with adaptability to reservoir habitats (Loureiro-Crippa and Hahn 2006). Although *M. congestum* has persisted in impounded streams, it has declined in areas where reduced discharge have diminished water quality (Hoagstrom, 2001). Among the greatest threats to the ichthyofauna of Texas Hill Country streams is excessive groundwater pumping (Bowles and Arsuffi 1993). If excessive groundwater pumping and drought lead to local extirpations of *M. congestum* due to reduced discharge or reduced water quality, the presence of low-head dams may inhibit recolonization.

APPENDICES

Appendix 2.1. Monthly GSI ± SE of female (a) and male (b) *Moxostoma congestum* from the Blanco River and Canyon Lake, Texas, from September 2004 through August 2005.



Appendix 2.2. Oocyte diameter frequencies from *Moxostoma congestum* collected from the Blanco River and Canyon Lake, Texas, from September 2004 through August 2005.



Appendix 2.3. Percent occurrence of empty stomachs, detritus, and substrate, and percent of food items in the gut contents of *Moxostoma congestum* collected from the Blanco River and Canyon Lake, Texas, from September 2004 through August 2005.

		% Occurrence			% Abundance								
Month	N	Empty Stomachs	Detritus	Substrate	Aquatic Insects	Mollusks	Amphipods	Ostracods	Hydrachnids	Fish	Algae	Plant Seeds	Unidentifiable Insects
Sep	6	16 7	80 0	-	33 6	65 2	-	-	-	-	0 2	-	1 0
Oct	5	-	100 0	60 0	53 9	19 6	16 5	-	-	-	-	-	10 0
Dec	5	-	80 0	20 0	52 4	-	27 7	13 6	-	-	-	0 4	5 9
Jan	7	-	100 0	14 3	41 9	41 3	3 8	-	< 0 1	11 0	< 0 1	-	2 0
Feb	4	-	100 0	50 0	71 2	28 6	0 1	-	-	-	< 0 1	-	-
Mar	5	-	80.0	20 0	45 7	35 2	-	19 2	-	-	-	-	-
Apr	8	-	100 0	37 5	64.6	32 4	0 9	2 1	< 0 1	-	-	-	-
May	5	-	80 0	40 0	68 9	27 7	2 7	-	< 0 1	-	< 0 1	-	0 6
Jun	6	33 3	100 0	-	13 5	83 6	2 8	-	-	-	-	-	-
Jul	5	-	100 0	40 0	30 7	67 1	-	-	-	-	0 3	-	1.9
Aug	6	16 7	80 0	20 0	42 8	56 2	-	1 0	-	-	-	-	-

Appendix 2.4. Results of ANOSIM tests for differences in diet between the Blanco River and Canyon Lake, Texas, and between seasons.

	R	P
Site:		
Global Test	0.103	**<0.01**
Season:		
Global Test	0.067	**0.04**
Pairwise Tests:		
Fall vs. Winter	0.023	0.32
Fall vs. Spring	0.034	0.27
Fall vs. Summer	0.062	0.14
Winter vs. Spring	-0.012	0.55
Winter vs. Summer	0.183	**<0.01**
Spring vs. Summer	0.117	**0.03**

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VITA

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