

Historical Zoogeography and Abundance of Fishes in Two Texas River Basins with an annotated species list

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Project Overview

Purposes of this project were to evaluate existing biological data and to develop an annotated species lists for fishes in the Trinity River, downstream of Dallas/Ft. Worth area to Galveston Bay, and Guadalupe River, downstream of Canyon Lake to confluence with San Antonio River. Deliverables for Task 1 (Evaluate Existing Biological Data) include 1) a summary of gaps in data coverage, 2) a summary detailing the temporal and spatial trends in existing data, and 3) recommendations for sampling strategies and locations to ensure critical gaps in data are addressed. Deliverables for Task 2: (Develop Annotated Species List) include 1) historical trends in occurrence and abundance within each study area, 2) available life history information, species-specific conceptual models linking life history traits with flow regimes (i.e., subsistence flows, base flows, high flow pulses, and overbanking flows), physical habitat (e.g., flow sensitivity and habitat utilization) and other environmental requirements (e.g., water quality, connectivity, and others), 3) an overall synthesis of these conceptual models into a model of fish assemblage dynamics, and 4) identification of key species and/or habitat guilds in each basin.

Sufficient biological data existed to enable a comprehensive assessment of fish abundance changes through time for upper and lower Guadalupe River and the San Marcos

River. Section I of this report, along with the annotated species list available at www.bio.txstate.edu/~tbonner/txfishes/Guadalupe.htm, provides a detailed assessment of temporal and spatial trends in fish assemblages from the existing data, historical trends in fish occurrence and abundance within each study area, summary of available life history information, species-specific conceptual models linking life history traits with flow regimes (i.e., high flow pulses) and other environmental requirements (i.e., water quality, connectivity), an overall synthesis of these conceptual models into a model of fish assemblage dynamics, and identification of key species and guilds in each river.

Availability of biological data in the Trinity River basin was not sufficient to enable a comprehensive assessment of historical composition before large-scale in stream modification. Changes in assemblage composition and water quality in the Trinity River between 1975 and 1997 immediately downstream of Dallas are reported by Land et al. (1998); however, downstream reaches were not considered and described changes in water quality were limited to ammonia and dissolved oxygen. Spatial distribution and geographic locations for existing fish collections in the Trinity River are reported in Kiesling and Flowers (2002); however temporal trends among historical collections were not presented. Although temporally constrained, existing biological data in the Trinity River provides opportunity to document fish assemblage responses to improved water quality, advance our understanding of ecosystem response to conservation efforts (e.g., Eklov et al. 1998), and determining the feasibility of restoring ecosystem function to extensively degraded environments (e.g., Kinsolving and Bain 1993; Doyle et al. 2005). Accordingly, Section II of this report summarizes the available information regarding historical trends in fish assemblage composition and water quality 1970-2008, species-specific conceptual models linking life history traits to environmental requirements, and

identification of key species in the Trinity River between Dallas-Fort Worth and Galveston Bay. Annotated species list is available at www.bio.txstate.edu/~tbonner/txfishes/Trinity.htm.

Section I: Historical Fish Assemblage Changes in the Guadalupe River Drainage

Study Objectives

Objectives of this study were to describe changes in mean annual flow and frequency of small and large flood events in the upper and lower Guadalupe River and San Marcos River for the period of record of multiple gauging stations, to assess fish assemblage occurrence and abundance, reproductive guild, and trophic guild changes between time periods, which represent pre- and post modifications of instream flows, to determine population status of native and introduced fish taxa, and to quantify collective changes in fish populations related to pre- and post modifications of instream flow.

Study Area

The Guadalupe River originates at the confluence of the North Fork Guadalupe River and South Fork Guadalupe River near the City of Hunt, Kerr County, Texas (Figure 1). The total drainage area is 15,700 km² as it flows about 370 km southeast toward the Gulf of Mexico. Among the seven mainstem impoundments on the Guadalupe River, Canyon Lake Reservoir was constructed in 1964. With a maximum depth of about 40 m and with a surface area of 3,300 ha, Canyon Lake Reservoir is the only deep storage reservoir within the Guadalupe River basin, representing the most significant alteration of mainstem discharge (Young et al. 1972, Edwards

1978). Remaining mainstem reservoirs are impounded by low-head dams, constructed from 1928 through 1931 in the lower Guadalupe River (Young et al. 1972).

The San Marcos River, among the largest tributaries of the Guadalupe River, originates from artesian springs in the City of San Marcos, Hays County, Texas and flows about 120 km before reaching its confluence with the Guadalupe River near Gonzales, Gonzales County, Texas. San Marcos River has seven low-head dams and numerous low water crossings constructed between 1849 and 1901 (Taylor 1904). Several low-head dams were constructed in the upper Blanco River, a tributary of the San Marcos River, by Civilian Conservation Corps in mid 1930s and by private landowners through the 1950s (pers. comm. H. Hammond, Blanco County landowner). Flood retarding structures were constructed by the Natural Resource Conservation Service (NRCS) in the Plum Creek and York Creek drainages of the lower San Marcos River in the mid 1960s and 1970s (pers. comm. I. Morales, District Conservationist, NRCS, Lockhart, Texas). Flood retention structures were constructed by NRCS in the upper San Marcos River watershed in the 1980s (Woods and Earl 2002).

Methods

Daily discharges were obtained from three locations on the Guadalupe River (US Geological Survey Station 08168500, New Braunfels, Texas; USGS Station 08176500, Victoria, Texas; USGS Station 08167500, Spring Branch, Texas) and one location on the San Marcos River (USGS Station 08172000, Luling, Texas; Figure 1). These locations encompass the largest available spatiotemporal range in drainage discharge (350 km mainstem Guadalupe River; 1927 - 2007 for Station 08168500; 1934 - 2007 for Station 08176500; 1922 - 2007 for Station 08167500, and 1938 - 2007 for Station 08172000). For each Guadalupe mainstem site,

discharge data were divided into Period I (earliest record – 1963) and Period II (1965 – 2007), using the completion of Canyon Reservoir (1964) as the environmental impact. For the San Marcos River, a break in ichthyological data ranging 1963 to 1976 was used to define Period I (1927-1963) and Period II (1976-2006), which generally corresponds with pre- and post construction of water retarding and retention dams developed in the upper and lower watershed for flood control. Changes in frequency of small and large floods and mean annual discharge between periods were assessed with Indicators of Hydrologic Alteration, v. 7.0.3 (IHA). Flood frequency and mean annual discharge were used because of strong interrelationship among these parameters, fish habitat availability, and stream morphology (Richter et al. 1996, Runyan 2007). Small floods were defined as high flow events (i.e., exceeding 75% of discharge in Period I) with recurrences of at least 2 years. Large floods were defined as high flow events with recurrences of at least 10 years.

Historical ichthyofaunal collections from the Guadalupe River and San Marcos River drainages were obtained from museum collections, agency reports, unpublished records, and published documents. Museum collections were obtained from the Texas Natural History Museum (University of Texas), Texas Cooperative Wildlife Collection (Texas A&M University), Tulane Museum of Natural History, University of Kansas Natural History Museum, University of Michigan Museum of Zoology, Field Museum of Natural History (Chicago, Illinois), San Noble Oklahoma Museum of Natural History (University of Oklahoma), and the National Museum of Natural History (Smithsonian). Agency reports and published data included Texas Game and Fish Commission (TGFC, now Texas Parks and Wildlife, 1956), TGFC (1958), TGFC (1962), TGCF (1973), Underwood and Dronen (1984), Longley et al. (1996), Terre and Magnilia (1996), Kelsey (1997) and Longley et al. (1998). Unpublished data

were obtained from K. Mayes (Texas Parks and Wildlife Department), B. Beard (Texas Parks and Wildlife Department), B. Moring (US Geological Survey) and B. Whiteside, T. Bonner and P. Bean (Texas State University). Species occurrences and abundances, date and location of collection, principal collector, and methods of collection were obtained from all collections (Appendix 1).

Historical ichthyofaunal collections were filtered before assessing assemblage occurrence and abundance analyses (Runyan 2007). Species list was compared to expected ichthyofaunal list for the Guadalupe River drainage (Conner and Suttkus 1986, Thomas et al. 2007, Hubbs et al. 2008). Questionable identifications were confirmed or refuted with voucher specimens.

Others were noted and removed when voucher specimens were not taken or available.

Tributaries lacking sufficient temporal collections to infer assemblage changes were removed from abundance analyses. Consequently, abundance analyses were performed with assemblage data only from mainstem Guadalupe River and San Marcos River. Upper and lower Guadalupe River sections were analyzed independently because of the potential for longitudinal differences in fish assemblages. Seining and electroshocking collections within each river were retained if the collection had >5% of the total taxa found in the drainage and if the collection had >0.1% of the total number of individuals collected to improve the likelihood of the collection being a representative sample of the fish assemblage on a given date.

Fish relative abundance was calculated for each collection retained for analyses. Among collections, relative abundances of a species were $\log_{10}(N+1)$ -transformed and plotted through time. Time represented the number of days from the first collection (June 23, 1938). Simple linear regression was used to test if slope of relative abundance differed ($\alpha = 0.05$) through time. Populations were classified as increasing ($b_1 > 0$) or decreasing ($b_1 < 0$) in abundance.

Populations were classified as stable if slope did not differ from zero ($b_1 \neq 0$). Population status of rare species (i.e., occurring in <10% of total collections) and populations of species reported only once were classified as indeterminable. Native status of each species was determined using Conner and Suttkus (1986), Thomas et al. (2007) and Hubbs et al. (2008). Primary and secondary reproductive guilds were determined for each species using the classification scheme of Simon (1999), and trophic guilds after Goldstein and Simon (1999). Mean relative abundance of each species, excluding rare species, for Period I and Period II was determined (sum of relative abundance in each collection/number of collections) to facilitate direct comparison of species abundance between periods. Reproductive and trophic guild changes were described among decreasing and increasing taxa for assemblages demonstrating significant changes through time.

For each period, taxa richness (S) and Simpson's Index of Diversity ($1 - D$) were calculated along with similarity matrices. Bray-Curtis similarity matrices (Bray and Curtis 1957) created in Primer 6.1.6 were tested with analysis of similarity (ANOSIM; $\alpha = 0.05$) using 9,999 permutations to assess average rank dissimilarity between periods (Runyan 2007). Data were fourth-root transformed to standardize the contribution of high and low abundance species and illustrated using a multi-dimensional scaling (MDS) plot. Mean relative abundance of MDS axis I and II were averaged for 5-year intervals to assess trajectory of fish assemblage change. To compare collective trends in increasing or decreasing species through time, relative abundances of increasing and decreasing populations were z-scored transformed to standardized relative abundance distributions (mean = 0; SD = 1) of each species. Z-scored transformed abundances were averaged across all increasing or decreasing species by year (dependent variable) and regressed against time (independent variable) with piecewise regression model. Least-squares

regression and joinpoint analyses to detect significant changes in rate through time (i.e., test for appropriate piecewise models) were performed with the program JOINPOINT (Joinpoint Regression Program, Version 3.0, National Cancer Institute, 2005), a program designed to use grid-search methods for optimizing model parameters (Brendon and Bence 2008). Parsimonious joinpoint models were selected following permutation testing ($N = 5,000$; default) rather than BIC selection approach (Brendon and Bence 2008).

Results

Mean annual flows increased between periods in the Guadalupe River and San Marcos River with frequency of small and large flood events increasing only in the upper Guadalupe River and decreasing in the lower Guadalupe River and San Marcos River. Mean annual flow in the upper Guadalupe River (Spring Branch, Texas) increased from $7.27 \text{ m}^3/\text{s}$ in Period I (1927-1964) to $14.04 \text{ m}^3/\text{s}$ in Period II (1965-2007) with annual frequency of small ($95 \text{ m}^3/\text{s}$) and large ($837 \text{ m}^3/\text{s}$) flood events increasing from 0.81 to 1.07 between periods (Figure 2). In the lower Guadalupe River, mean annual flow at New Braunfels, Texas, increased from $9.70 \text{ m}^3/\text{s}$ in Period I (1927-1964) to $17.70 \text{ m}^3/\text{s}$ in Period II (1965-2007) with annual frequency of small ($120 \text{ m}^3/\text{s}$) and large ($949 \text{ m}^3/\text{s}$) floods decreasing from 0.84 to 0.42. Also in the lower Guadalupe River, mean annual flow at Victoria, Texas, increased from $48.10 \text{ m}^3/\text{s}$ in Period I (1938-1964) to $64.01 \text{ m}^3/\text{s}$ in Period II (1965-2007) with annual frequency of small ($569 \text{ m}^3/\text{s}$) and large ($1,461 \text{ m}^3/\text{s}$) floods decreasing from 0.56 to 0.42. In the San Marcos River, mean annual flow at Luling, Texas increased from $9.55 \text{ m}^3/\text{s}$ in Period I (1938-1963) to $13.31 \text{ m}^3/\text{s}$ in Period II (1976-2007) with annual frequency of small ($143 \text{ m}^3/\text{s}$) and large ($490 \text{ m}^3/\text{s}$) floods decreasing from 0.87 to 0.70 (Figure 3).

Fish Assemblage Changes-Guadalupe River—A total of 78 species was reported in the Guadalupe River mainstem (Table 1). Among the 190 collections obtained for this study, 69 species and 41,869 individuals were taken from the Guadalupe River mainstem from 1938 to 2000 (Appendix 1). Cyprinidae was most abundant (69% in relative abundance), followed by Centrarchidae (11%), Poeciliidae (6%), Percidae (5%), Catostomidae (2%) and Ictaluridae (2%). Among marine-derived taxa, *Mugil cephalus*, *Mugil curema* and *Achirus lineatus* were not considered significant freshwater components of the assemblage. Guadalupe River mainstem assemblage consisted of two basin endemics (*Dionda nigrotaeniata* and *Percina apristis*), disjunct populations of two fishes (*Erimyzon sucetta* and *Percina shumardi*), southwestern natural distributional extent, along with the adjacent and connected San Antonio River, of seven species (*Macrhybopsis marconis*, *Fundulus notatus*, *Lepomis humilis*, *Micropterus punctulatus*, *Micropterus treculii*, *Etheostoma chlorosoma*, and *Etheostoma spectabile*), and 15 introduced species (or 22 introduced species of $N = 78$ fishes reported in the drainage). Relative abundance of introduced fishes was <6% of the total fish assemblage.

Within the upper Guadalupe River, Cyprinidae was most abundant (73% in relative abundance), followed by Percidae (8.1%), Poeciliidae (7.3%), and Centrarchidae (7.1%). Assemblages were similar between Period I (1938 – 1963) and Period II (1965 – 1997). Taxa richness declined from Period I ($S = 42$) to Period II ($S = 41$), and diversity decreased from Period I ($1 - D = 0.86$) to Period II ($1 - D = 0.74$). However, assemblage similarity did not differ between periods (Bray-Curtis index = 37.4%; ANOSIM global $R = 0.079$, $P = 0.08$). Multi-dimensional scaling (MDS) plot and trajectory plot indicated Period II collections were nested within Period I collections (Figure 4). Despite overall assemblage similarities between periods, population changes were found in 10 taxa. Three cyprinids (*Cyprinella venusta*, *Notropis*

amabilis, *Notropis volucellus*), two centrarchids (*Micropterus treculii*, *Lepomis auritus*) and one catostomid (*Moxostoma congestum*) increased in relative abundance, collectively increasing from 30% in Period I to >75% in Period II. Three cyprinids (*Cyprinella lutrensis*, *Macrhybopsis marconis*, and *Pimephales vigilax*) and one percid (*Etheostoma spectabile*) decreased in relative abundance, collectively decreasing from 41% in Period I to <5% in Period II. Two joinpoints were the most parsimonious models for increasing ($P < 0.01$) and decreasing ($P < 0.01$) populations. For increasing taxa, two joinpoints in 1961 denoted two distinct regression models with independent variables ranging from 1938 – 1961 and 1961 – 1997. Relative abundances were not associated with either time interval ($b_1 \neq 0$, $P > 0.75$). For decreasing taxa, a joinpoint in 1950 and one in 1961 denoted three distinct regression models with independent variables ranging from 1938 – 1950, 1950 – 1961, and 1961 – 1997. Relative abundance was negatively associated ($\beta_0 = -0.000259$, $P < 0.01$) with time period 1950 – 1961.

Within the lower Guadalupe River, Cyprinidae was most abundant (68%), followed by Centrarchidae (13%), Poeciliidae (4.8%), and Catostomidae (3.0%). Differences were found in the fish assemblage between periods. Taxa richness increased between Period I ($S = 40$) and Period II ($S = 62$), and diversity decreased between Period I ($1 - D = 0.92$) and Period II ($1 - D = 0.82$). Fish assemblage similarity differed between periods (Bray-Curtis index = 25%; ANOSIM global $R = 0.409$, $P < 0.01$). Multi-dimensional scaling plot and trajectory plot indicated that assemblages in Period I were segregated from those of Period II (Figure 4). Differences in the fish assemblages were attributed, in part, to increases in non-native fish occurrences ($N = 13$) between periods and to changes in relative abundances. Three centrarchids (*Lepomis macrochirus*, *L. megalotis*, and *Micropterus salmoides*), one catostomid (*Ictiobus bubalus*) and one clupeid (*Dorosoma cepedianum*) increased in relative abundance, collectively increasing

from <4% in Period I to >10% in Period II. Two poeciliids (*Gambusia affinis* and *Poecilia latipinna*), two percids (*Percina carbonaria* and *P. apristis*), one cyprinid (*Notropis buchmanii*), and one fundulid (*Fundulus notatus*) decreased in relative abundance, collectively decreasing from 27% in Period I to 6% in Period II. Two joinpoints was the most parsimonious model for increasing populations ($P < 0.01$), whereas one linear regression model (i.e., no joinpoint) was the most parsimonious model for decreasing populations ($P = 0.04$) (Figure 5). For increasing taxa, one joinpoint in 1995 and another in 1997 denoted three distinct regression models with independent variables ranging from 1950 – 1995, 1995 – 1997, and 1997 – 2000. Relative abundance was positively associated ($\beta_0 = 0.000715$, $P < 0.01$) with time only during 1995 – 1997. For decreasing taxa without a joinpoint, relative abundance was negatively associated ($\beta_0 = -0.000063$, $P < 0.01$) with time during 1950 – 2000.

Fish Assemblage Changes-San Marcos River—Sixty-six species and 58,727 individuals were taken in 94 collections from the San Marcos River from 1938 to 2006 (Table 2; Appendix 1). Poeciliidae were most abundant (66%), followed by Cyprinidae (17%), Centrarchidae (10%) and Percidae (<7%). San Marcos River fish assemblage consisted of one endemic (*Gambusia georgei*), three basin endemics (*Dionda nigrotaeniata*, *Percina apristis*, and *Etheostoma fonticola*), disjunct populations of two fishes (*Percina shumardi*, sympatric with those in the Guadalupe River, and *Notropis chalybaeus*), five fishes with southwestern natural distributional extent in the Guadalupe River drainage (*Macrhybopsis marconis*, *Fundulus notatus*, *Micropterus punctulatus*, *Micropterus treculii*, and *Etheostoma spectabile*), and 16 introduced species of fish. Relative abundance of introduced fishes represented <7% of the total fish assemblage. Currently, one species (*Ictalurus lupus*) is reported as extirpated (Kelsch and Hendricks 1990) and another (*Gambusia georgei*) is considered extinct (Miller et al. 1989).

Differences in the San Marcos River fish assemblage were found between Period I (1938 – 1963) and Period II (1975 – 2006). Taxa richness increased between Period I ($S = 48$) to Period II ($S = 58$), and diversity decreased between Period I ($1 - D = 0.91$) and Period II ($1 - D = 0.80$). Fish assemblage similarity differed between periods (Bray-Curtis index = 27%; ANOSIM global $R = 0.19$, $P < 0.01$). Multi-dimensional scaling plot and trajectory plot indicated that assemblages in Period I were segregated from those of Period II (Figure 4). Fish assemblage differences were attributed, in part, to increases in non-native fish occurrences ($N = 8$) between periods and to changes in relative abundances. A total of 19 taxa, including four cyprinids (*Campostoma anomalum*, *Notropis amabilis*, *N. chalybaeus*, *N. volucellus*), five non-native taxa (*Astyanax mexicanus*, *Hypostomus*, *Ambloplites rupestris*, *Lepomis auritus*, *Micropterus dolomieu*), and six native centrarchids (*Lepomis gulosus*, *L. macrochirus*, *L. miniatus*, *Micropterus punctulatus*, *M. salmoides*, *M. treculii*) increased in abundance, collectively increasing from 13% in Period I to 15% in Period II. Three cyprinids (*Cyprinella lutrensis*, *Macrhybopsis marconis*, and *Pimephales vigilax*), two percids (*Percina carbonaria* and *Etheostoma fonticola*), one ictalurid (*Noturus gyrinus*), and one introduced cichlid (*Cichlasoma cyanoguttatum*) decreased in abundance, collectively decreasing from >48% in Period I to <5% in Period II. Two joinpoints were the most parsimonious model for increasing populations ($P < 0.01$) and one joinpoint was the most parsimonious model for decreasing populations ($P < 0.01$; Figure 6). For increasing taxa, two joinpoints in 1993 denoted two distinct regression models with independent variables ranging from 1938 – 1993 and 1993 – 2006. Relative abundance was positively associated ($\beta_0 = 0.000021$, $P < 0.01$) with time during 1938 – 1993 and negatively associated ($\beta_0 = -0.000040$, $P < 0.01$) with time during 1993 – 2006. For decreasing taxa, one joinpoint in 1950 denoted two distinct regression models with independent variables ranging

1938 – 1950 and 1951 – 2006. Relative abundance was negatively associated ($\beta_0 = -0.000029$, $P < 0.01$) with time only during 1951 – 2006.

Reproductive and trophic guilds changes—Changes in reproductive guilds and trophic guilds were assessed for the lower Guadalupe River and San Marcos River, two fish assemblages with significant assemblage differences between periods. Among the 24 fishes with increasing populations through time, reproductive guilds consisted of 58% nest builders, 33% open substrate spawners, and 8% brood hidiers; trophic guilds consisted of 54% invertivores, 17% predators, 13% omnivores, 13% herbivores, and 4% detritivores. Among the 13 fishes with decreasing populations through time, reproductive guilds consisted of 30% brood hidiers, 23% open substrate spawners, 15% nest builders, 15% internal bearers, and 15% substrate choosers; trophic guilds consisted of 76% invertivores, 15% omnivores, and 7% herbivores.

Discussion

Occurrence and abundance of fishes changed in the Guadalupe River and San Marcos River during a span of about 70 years. During this same time period, characteristics of river discharge were modified; specifically, mean annual flow increased and frequency of small and large flood events generally decreased. Increases in mean annual flows among all three reaches of this study were attributed to computational effects of low water years in 1950s, often described as the drought of record (1949 – 1959; Loaiciga et al. 2000), and to the effects Canyon Lake Reservoir. Discharge during the drought of record represented 25% to 35% of daily discharge records in Period I, lowering mean annual flow estimates. Consequently, we suspect that mean annual flows have not meaningfully increased in the upper Guadalupe River or San Marcos River. In contrast, detected increases in mean annual flows are meaningful in the lower

Guadalupe River because of water releases at Canyon Lake Reservoir. Canyon Lake Reservoir, operated by US Army Corp of Engineers, regulates discharge releases as part of the reservoir management plan for flood control and recreational activities (Gillig et al. 2001). Likewise, decreases in frequency of small and large flood events were attributed to Canyon Lake Reservoir in the lower Guadalupe River, with effects more noticeable at the nearest downstream Station (08168500). In the San Marcos River, decreases in frequency of small and large flood events were attributed to flow retarding and retention structures in the San Marcos watershed (Woods and Earl 2002). Interestingly, significant differences in fish assemblage similarities are associated with river reaches where frequencies of small and large flood events decreased (i.e., lower Guadalupe River and San Marcos River).

Fish assemblage changes associated with reductions in frequency of small and large flood events are well documented in temperate and tropical rivers, streams, and small tributaries (Gehrke et al. 1999, Bunn and Arthington 2002, Agostinho et al. 2004, Roy et al. 2005, Mercano-Silva et al. 2006). Reduction in flood frequency affects stream geomorphology, causing a shift toward lentic-type habitat (Poff et al. 1997), contributing to a replacement effect of fluvial specialist with lentic-type, generalist species (Scott and Helfman 2001, Haxton and Findlay 2008). Decreasing abundance of fluvial specialist species occurs through numerous mechanisms, including reduced reproductive success (Durham and Wilde 2006), loss of spawning cues (Bunn and Arthington 2002), barriers to dispersion (Luttrell et al. 1999) and competitive exclusion from resources (Higgins and Strauss 2008). Subsequent replacement by generalist species occurs through numerous mechanisms as well, including refugia from flood displacement (Steven et al. 2007), fulfillment of void niches (Winston et al. 1991) and increased sedimentation (Poff et al. 1997, Scott and Helfman 2001).

Assemblage changes in the lower Guadalupe River and San Marcos River are consistent with general trends in generalist fish replacements. Within and outside of western gulf slope drainages, generalist fishes are those typically becoming more abundant in areas of flow alterations and include clupeids, some cyprinids, catostomids, poeciliids and centrarchids, whereas fluvial species tend to become less abundant, such as several species of cyprinids, percids and catostomids (Winston et al. 1991, Kingsoving and Bain 1993, Travnichek et al. 1995, Scott and Helfman 2001, Li and Gelwick 2005, Mercado-Silva et al. 2006, Runyan 2007). In this study, abundances of generalist species, including one clupeid, four cyprinids, one catostomid, and 10 centrarchids, increased through time in the lower Guadalupe River and San Marcos River. Correspondingly, abundances of fluvial specialist (four percids and two cyprinids) decreased through time in the lower Guadalupe River and San Marcos River. Exact mechanisms of these replacements are not known, but likely related to changes in fluvial specialist habitats and reductions in displacement floods as reported in other studies (Valdez et al. 2001, Herbert and Gelwick 2003, Holden et al. 2005, Watson 2006). In fact, effects of displacement floods on generalist fishes were demonstrated by the results of this study. In the San Marcos River, taxa considered increasing from 1938 to 1993, specifically generalist fishes (i.e., *Lepomis auritus*, *Lepomis gulosus*, *Lepomis macrochirus*, *Micropterus salmoides* and *Ictalurus punctatus*), abruptly decreased from 1993 to 2006. These abundance declines occurred during a period of six large flood events including a catastrophic flood in 1998.

There were inconsistencies with general trends in generalist fish replacements in the lower Guadalupe River and San Marcos River. Poeciliid abundances are expected to increase with decreases in frequency of flood events in western gulf slope drainages (Ward et al. 2003), but they actually decreased through time in the lower Guadalupe River. Similar trends of

decreasing abundance through time were observed for *Gambusia affinis* in the San Antonio River (Runyan 2007) and for *Poecilia latipinna* in the upper Guadalupe River (Stevens et al. 2007 and this study). Stevens et al. (2007) suggested that a single flood was responsible for population declines, if not extirpation, of *P. latipinna* in the upper Guadalupe River. As the authors noted, a similar response occurred with another poeciliid during a flood in a Sonoran Desert stream (Collins, et al. 1981). Based on this information, it is likely that high flow pulses in the lower Guadalupe River, though altered, are sufficient to regulate the abundance of at least some of the generalist taxa. Another inconsistency in general trends in generalist fish replacements is that abundances of several fluvial specialists (*N. amabilis*, *N. volucellus*, and *N. chalybaeus*) increased through time, whereas abundances of taxa typically associated with flow altered systems, generalist species with broad tolerances (*Cyprinella lutrensis*, *Pimephales vigilax* and *Etheostoma spectabile*; Matthews 1985, Greenburg 1989, Li and Gelwick 2005, Runyan 2007), decreased through time in the San Marcos River. Similar results were observed in the upper Guadalupe River; *C. lutrensis*, *P. vigilax*, and *E. spectabile* decreased through time, specifically from 1950 through 1961. Based the timing of abundance declines in these more tolerant taxa in the upper Guadalupe River, we propose that fish collections during the drought of record reflected a stressed system with an abundance of tolerant taxa. With the return of average precipitation and consequently stream discharge post 1959, fluvial specialists again proliferated while tolerant taxa declined. Proliferation of the natural fish assemblage does occur in streams once the natural environment returns or is restored, assuming source populations exist and recolonization is not impeded by instream structures (Kinsolving and Bain 1993, Doyle et al. 2005).

Despite alteration of flow regime and some changes in fish populations, overall fish assemblages within all three reaches of this study remained relatively intact; assemblages were dominated by native taxa, many of the endemic taxa, taxa on the periphery of range, and taxa with disjunct populations have stable populations, and most non-native taxa comprise a small portion of the overall assemblage. In comparison, flow alterations in other river systems have led to large-scale changes in taxa diversity, number of exotic taxa, and trophic guilds (de Merona et al. 2005, Mercado-Silva et al. 2006) with native taxa ultimately replaced by generalist nonnative taxa (Holden et al. 2005). Though relatively intact, extinction and extirpation of fishes have occurred within Guadalupe River basin but likely are independent of flow alteration. Presumed extirpated *Ictalurus lupus* and extinct *Gambusia georgei* of the San Marcos River were associated with the introduction of exotic species or sportfish (Miller et al. 1989, Kelsch and Hendricks 1990). Future extirpations likely will be among taxa with limited geographic range (i.e., *Macrhybopsis marconis*, *Dionda nigrotaeniata*) as well as species of special concern ($N = 7$ Guadalupe River, $N = 7$ San Marcos River; Hubbs et al. 2008).

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Table 1. Native status, primary and secondary reproductive guild, trophic guild, mean relative abundance per time period, population trend and P-value for species collected in the upper and lower Guadalupe River. Native status was determined by Hubbs et al. (2008) as native (N) or introduced (I). Time period I (1938-1963) represents ichthyological collections leading up to the impoundment of Canyon Reservoir and time period II (1965-2000) after impoundment; 'X' indicates rarely reported species. Population trends are increasing (↑), decreasing (↓), stable (S) and indeterminable (-); P-values are reported only for species indicating significant population change. Reproductive guilds follow Simon (1999) and trophic guilds are detritivore (D), herbivore (H), invertivore (IF), omnivore (O), piscivore (P) and planktivore (PL; Goldstein and Simon 1999).

Species	Status	Upper Guadalupe River				Lower Guadalupe River				Primary Reproductive Guild	Secondary Reproductive Guild	Trophic Guild
		Period I	Period II	Population Trend	P-value	Period I	Period II	Population Trend	P-value			
<i>Atractosteus spatula</i> ¹	N			-			-			Open Substratum	Phytophil	P
<i>Lepisosteus oculatus</i>	N	0.01		-		0.20	0.34	S		Open Substratum	Phytophil	P
<i>Lepisosteus osseus</i>	N	0.62	0.10	-		0.08	0.03	-		Open Substratum	Phytolithophil	P
<i>Anguilla rostrata</i>	N			-		0.10	0.02	-		Catadromous	Catadromous	P
<i>Dorosoma cepedianum</i>	N	1.38	1.86	S		0.34	3.27	↑	0.013	Open Substratum	Lithopelagophil	H
<i>Dorosoma petenense</i>	N		0.10	-			<0.01	-		Open Substratum	Phytophil	PL
<i>Camptostoma anomalum</i>	N	2.39	2.53	S		0.10	0.73	S		Brood Hiders	Lithophil	H
<i>Carassius auratus</i> ¹	I			-				-		Open Substratum	Phytophil	IF
<i>Ctenopharyngodon idella</i> ¹	I			-				-		Open Substratum	Pelagophil	H
<i>Cyprinella lutrensis</i>	N	21.53	1.10	↓	<0.001	22.16	35.15	S		Brood Hiders	Speleophil	IF
<i>Cyprinella venusta</i>	N	23.12	47.41	↑	<0.001	0.24	3.86	S		Brood Hiders	Speleophil	IF
<i>Cyprinus carpio</i>	I		0.67	-			0.87	-		Open Substratum	Phytolithophil	O
<i>Dionda nigrotaeniata</i>	N	0.02	0.12	-				-		Open Substratum	Lithophil	H
<i>Macrhybopsis marconis</i>	N	1.80	0.27	↓	0.025	3.87	1.51	S		Open Substratum	Pelagophil	IF
<i>Notemigonus crysoleucas</i> ²	N		X	-			X	-		Open Substratum	Phytophil	IF
<i>Notropis amabilis</i>	N	3.24	10.48	↑	<0.001	3.54	7.39	S		Open Substratum	Pelagophil	IF
<i>Notropis buchani</i>	N	0.07		-		5.34	0.68	↓	0.001	Open Substratum	Pelagophil	IF
<i>Notropis stramineus</i>	N	1.42	0.14	S		0.10	0.15	-		Open Substratum	Lithophil	IF
<i>Notropis volucellus</i>	N	2.71	12.20	↑	<0.001	4.74	3.39	S		Open Substratum	Phytophil	O
<i>Opsopoeodus emiliae</i>	N			-		0.29	0.72	-		Nest	Speleophil	DT
<i>Pimephales promelas</i>	I	0.01		-				-		Nest	Speleophil	O
<i>Pimephales vigilax</i>	N	3.16	0.40	↓	0.027	7.86	3.43	S		Nest	Speleophil	O
<i>Carpionodes carpio</i>	N	0.43	0.04	-				-		Open Substratum	Lithopelagophil	DT
<i>Cycleptus elongatus</i> ¹	N			-				-		Open Substratum	Lithopelagophil	IF
<i>Erimyzon sucetta</i>	N	0.02		-				-		Open Substratum	Phytolithophil	IF
<i>Ictiobus bubalus</i>	N			-			1.02	↑	0.017	Open Substratum	Lithopelagophil	O
<i>Moxostoma congestum</i>	N	1.22	1.29	↑	0.001	0.57	3.73	S		Open Substratum	Lithophil	IF
<i>Astyanax mexicanus</i>	I	0.33	0.05	-			1.34	S		Open Substratum	Pelagophil	IF
<i>Ameiurus melas</i>	N	0.01		-				-		Nest	Speleophil	IF

Table 1 continued.

Species	Status	Upper Guadalupe River				Lower Guadalupe River				Primary Reproductive Guild	Secondary Reproductive Guild	Trophic Guild	
		Period I	Period II	Population	Trend	P-value	Period I	Period II	Population				Trend
<i>Ameiurus natalis</i>	N	0.19	0.03	-		0.03	0.40	-			Nest	Speleophil	IF
<i>Ameiurus nebulosus</i> ¹	I			-				-			Nest	Speleophil	IF
<i>Ictalurus furcatus</i> ²	N	X	X	-		X	X	-			Nest	Speleophil	P
<i>Ictalurus lupus</i>	N	0.06	0.02	-				-			Nest	Speleophil	O
<i>Ictalurus punctatus</i>	N	1.91	1.16	S		2.59	1.88	S			Nest	Speleophil	O
<i>Noturus gyrinus</i>	N			-		0.25	<0.01	-			Nest	Speleophil	IF
<i>Pylodictis olivaris</i>	N	0.34	0.04	S		0.16	0.23	S			Nest	Speleophil	IF
<i>Hypostomus sp.</i> ¹	I			-				-			Nest	Speleophil	DT
<i>Oncorhynchus mykiss</i>	I			-			0.04	-			Brood Hiders	Lithophil	IF
<i>Salmo trutta</i>	I			-			<0.01	-			Brood Hiders	Lithophil	IF
<i>Fundulus grandis</i> ¹	N			-				-			Open Substratum	Phytophil	O
<i>Fundulus notatus</i>	N			-		2.18	0.21	↓	0.011		Open Substratum	Phytophil	H
<i>Gambusia affinis</i>	N	8.81	3.81	S		6.90	3.07	↓	0.026		Internal Bearer	Viviparous	IF
<i>Poecilia latipinna</i> ³	N	0.05		-		5.29	0.60	↓	0.013		Internal Bearer	Viviparous	O
<i>Membras martinica</i> ¹	I			-				-			Open Substratum	Phytophil	O
<i>Menidia beryllina</i>	I		0.27	-			0.97	S			Open Substratum	Phytophil	IF
<i>Morone chrysops</i> ²	I		X	-			X	-			Open Substratum	Phytolithophil	P
<i>Morone saxatilis</i>	I			-			0.02	-			Open Substratum	Phytolithophil	P
<i>Ambloplites rupestris</i>	I	0.01		-			0.72	S			Nest	Polyphil	IF
<i>Lepomis auritus</i>	I	1.12	2.19	↑	<0.001		2.90	S			Nest	Polyphil	IF
<i>Lepomis cyanellus</i>	N	0.70	0.60	S		0.13	1.28	S			Nest	Polyphil	IF
<i>Lepomis gulosus</i>	N	0.50	0.12	S		1.12	0.25	S			Nest	Lithophil	IF
<i>Lepomis humilis</i>	N			-			0.07	-			Nest	Lithophil	IF
<i>Lepomis macrochirus</i>	N	0.66	1.73	S		0.74	4.52	↑	0.001		Nest	Polyphil	IF
<i>Lepomis megalotis</i>	N	1.97	1.84	S		1.95	5.98	↑	<0.001		Nest	Polyphil	IF
<i>Lepomis microlophus</i>	I	0.20	0.35	S			0.52	S			Nest	Polyphil	IF
<i>Lepomis miniatus</i>	N	0.51	0.07	S			0.38	S			Nest	Polyphil	IF
<i>Micropterus dolomieu</i>	I			-			0.69	S			Nest	Polyphil	P
<i>Micropterus punctulatus</i>	N			-		0.10	0.78	S			Nest	Polyphil	IF

Table 1 Continued

Species	Status	Upper Guadalupe River				Lower Guadalupe River				Primary Reproductive Guild	Secondary Reproductive Guild	Trophic Guild
		Period I	Period II	Population Trend	P-value	Period I	Period II	Population Trend	P-value			
<i>Micropterus salmoides</i>	N	0.71	1.39	S		0.25	2.24	↑	<0.001	Nest	Polyphil	P
<i>Micropterus treculii</i>	N	0.54	1.44	↑	<0.001	1.25	0.82	S		Nest	Polyphil	P
<i>Pomoxis annularis</i>	N		0.05	-			0.07	-		Nest	Phytophil	P
<i>Pomoxis nigromaculatus</i>	I			-			0.02	-		Nest	Phytophil	IF
<i>Etheostoma chlorosoma</i>	N			-		0.63	0.03	-		Substratum Chooser	Phytophil	IF
<i>Etheostoma gracile</i>	N			-		0.51		-		Substratum Chooser	Phytophil	IF
<i>Etheostoma lepidum</i>	N	2.08	0.92	S		0.52	0.17	S		Brood Hiders	Lithophil	IF
<i>Etheostoma spectabile</i>	N	11.03	2.91	↓	0.028	1.58	0.37	S		Brood Hiders	Lithophil	IF
<i>Percina carbonaria</i>	N	2.05	1.32	S		4.60	0.09	↓	<0.001	Brood Hiders	Lithophil	IF
<i>Percina macrolepida</i>	N			-		0.46	0.04	-		Brood Hiders	Lithophil	IF
<i>Percina apristis</i>	N	0.42		-		2.12	0.16	↓	<0.001	Brood Hiders	Lithophil	IF
<i>Percina shumardi</i>	N	0.07		-		2.71	0.13	-		Brood Hiders	Lithophil	IF
<i>Stizostedion vitreum</i> ²	I		X	-			X	-		Substratum Chooser	Lithopelagophil	P
<i>Cichlasoma cyanoguttatum</i>	I	0.48	0.28	S		1.39	1.52	S		Substratum Chooser	Lithophil	IF
<i>Oreochromis aureus</i> ²	I		X	-			X	-		Bearer	Mouth Brooder	O
<i>Oreochromis mossambicus</i> ¹	I			-				-		Bearer	Mouth Brooder	O
<i>Agonostomus monticola</i>	N			-			0.01	-		Catadromous	Catadromous	O
<i>Mugil cephalus</i>	N			-		0.56	0.08	-		Catadromous	Catadromous	DT
<i>Mugil curema</i>	N			-			<0.01	-		Catadromous	Catadromous	O
<i>Achirus lineatus</i>	N			-			<0.01	-		Catadromous	Catadromous	O
Collections During Period:		86	24			12	68					
Individuals Collected:		12,266	6,626			1,390	21,587					
Taxa Richness:		42	41			40	62					
Diversity:		0.86	0.74			0.92	0.82					

¹Occurs in Drainage (USFWS 1973, Conner and Suttkus 1986, Prentice et al. 1998); not recorded in historical collections and not used in richness nor diversity calculations

²Species reported in Canyon Lake Reservoir (Whiteside 1983-2000, Unpublished Data); included in richness and diversity calculations

³Native to lower Guadalupe River (Conner and Suttkus 1986); introduced to upper Guadalupe River (Stevens et al. 2007)

Table 2. Native status, primary and secondary reproductive guild, trophic guild, mean relative abundance per time period, population trend and P-value for species collected in the San Marcos River. Native status was determined by Hubbs et al. (2008) as native (N) or introduced (I). Time period I (1938-1963) represents ichthyological collections leading up to a break in data collection (1963-1976) and time period II (1976-2006) after; ‘X’ indicates rarely reported species. Population trends are increasing (↑), decreasing (↓), stable (S) and indeterminable (-); P-values are reported only for species indicating significant population change. Reproductive guilds follow Simon (1999) and trophic guilds are detritivore (D), herbivore (H), invertivore (IF), omnivore (O), piscivore (P) and planktivore (PL; Goldstein and Simon 1999).

Species	Status	Period		Population		Primary Reproductive Guild	Secondary Reproductive Guild	Trophic Guild
		I	II	Trend	P-value			
<i>Lepisosteus oculatus</i>	N		0.10	-		Open Substratum	Phytophil	P
<i>Lepisosteus osseus</i>	N		0.09	-		Open Substratum	Phytolithophil	P
<i>Anguilla rostrata</i>	N		0.02	-		Catadromous	Catadromous	P
<i>Dorosoma cepedianum</i>	N	0.17	2.20	↑	0.01	Open Substratum	Lithopelagophil	H
<i>Camptostoma anomalum</i>	N	0.13	1.33	↑	0.01	Brood Hiders	Lithophil	H
<i>Carassius auratus</i>	I		0.06	-		Open Substratum	Phytophil	IF
<i>Cyprinella lutrensis</i>	N	14.33	1.45	↓	<0.01	Brood Hiders	Speleophil	IF
<i>Cyprinella venusta</i>	N	5.34	6.50	S		Brood Hiders	Speleophil	IF
<i>Cyprinus carpio</i>	I		0.30	-		Open Substratum	Phytolithophil	O
<i>Dionda nigrotaeniata</i>	N	0.61	1.35	S		Open Substratum	Lithophil	H
<i>Hybopsis amnis</i>	N	0.24		-		Open Substratum	Lithophil	IF
<i>Macrhybopsis marconis</i>	N	2.43	0.37	↓	<0.01	Open Substratum	Pelagophil	IF
<i>Notemigonus crysoleucas</i>	N	0.05	0.27	-		Open Substratum	Phytophil	IF
<i>Notropis amabilis</i>	N	3.73	7.82	↑	0.03	Open Substratum	Pelagophil	IF
<i>Notropis buchani</i>	N	0.43		-		Open Substratum	Pelagophil	IF
<i>Notropis chalybaeus</i>	N	0.09	0.50	↑	0.01	Open Substratum	Lithopelagophil	IF
<i>Notropis stramineus</i>	N	0.08	0.27	-		Open Substratum	Lithophil	IF
<i>Notropis volucellus</i>	N	1.02	6.89	↑	0.02	Open Substratum	Phytophil	O
<i>Opsopoeodus emiliae</i>	N	<0.01	<0.01	-		Nest	Speleophil	DT
<i>Pimephales promelas</i>	I	0.03		-		Nest	Speleophil	O
<i>Pimephales vigilax</i>	N	4.68	1.49	↓	0.02	Nest	Speleophil	O
<i>Carpodes carpio</i>	N	0.13		-		Open Substratum	Lithopelagophil	DT
<i>Ictiobus bubalus</i>	N		0.29	-		Open Substratum	Lithopelagophil	O
<i>Moxostoma congestum</i>	N	0.04	1.63	↑	<0.01	Open Substratum	Lithophil	IF
<i>Astyanax mexicanus</i>	I	0.48	2.01	↑	0.03	Open Substratum	Pelagophil	IF
<i>Ameiurus melas</i>	N	0.10	0.06	-		Nest	Speleophil	IF
<i>Ameiurus natalis</i>	N	0.71	0.36	S		Nest	Speleophil	IF
<i>Ictalurus furcatus</i>	N		0.02	-		Nest	Speleophil	P
<i>Ictalurus lupus</i> ¹	N	0.16		-		Nest	Speleophil	O
<i>Ictalurus punctatus</i>	N	0.54	1.08	↑	0.04	Nest	Speleophil	O
<i>Noturus gyrinus</i>	N	2.95		↓	<0.01	Nest	Speleophil	IF
<i>noturus nocturnus</i>	N	0.06		-		Nest	Speleophil	IF
<i>Pylodictis olivaris</i>	N		0.12	-		Nest	Speleophil	IF
<i>Hypostomus sp.</i>	I		0.02	↑	<0.01	Nest	Speleophil	DT
<i>Fundulus notatus</i>	N	0.61	0.30	S		Open Substratum	Phytophil	H

Table 2 continued

Species	Status	Period		Population		Primary Reproductive Guild	Secondary Reproductive Guild	Trophic Guild
		I	II	Trend	P-value			
<i>Gambusia affinis</i>	N	6.82	6.75	S		Internal Bearer	Viviparous	IF
<i>Gambusia geiseri</i>	N	11.77	9.73	S		Internal Bearer	Viviparous	IF
<i>Gambusia georgei</i> ²	N	0.79		-		Internal Bearer	Viviparous	IF
<i>Poecilia formosa</i>	I	2.36	0.98	S		Internal Bearer	Viviparous	IF
<i>Poecilia latipinna</i>	I	3.94	2.46	S		Internal Bearer	Viviparous	O
<i>Cyprinodon variegatus</i>	N	0.03		-		Nest	Polyphil	O
<i>Ambloplites rupestris</i>	I	0.27	1.50	↑	<0.01	Nest	Polyphil	IF
<i>Lepomis auritus</i>	I	0.39	5.48	↑	<0.01	Nest	Polyphil	IF
<i>Lepomis cyanellus</i>	N	0.62	1.16	S		Nest	Polyphil	IF
<i>Lepomis gulosus</i>	N	0.21	0.66	↑	0.02	Nest	Lithophil	IF
<i>Lepomis macrochirus</i>	N	2.64	5.19	↑	<0.01	Nest	Polyphil	IF
<i>Lepomis megalotis</i>	N	0.76	3.14	S		Nest	Polyphil	IF
<i>Lepomis microlophus</i>	N	0.51	0.51	S		Nest	Polyphil	IF
<i>Lepomis miniatus</i>	N	3.78	5.37	↑	0.01	Nest	Polyphil	IF
<i>Micropterus dolomieu</i>	I		0.21	↑	0.01	Nest	Polyphil	P
<i>Micropterus punctulatus</i>	N		0.31	↑	<0.01	Nest	Polyphil	IF
<i>Micropterus salmoides</i>	N	1.50	2.12	↑	0.01	Nest	Polyphil	P
<i>Micropterus treculii</i>	N	0.16	0.57	↑	0.04	Nest	Polyphil	P
<i>Pomoxis annularis</i>	I		0.05	-		Nest	Phytophil	P
<i>Pomoxis nigromaculatus</i>	I		0.04	-		Nest	Phytophil	IF
<i>Etheostoma fonticola</i>	N	15.45	2.33	↓	0.03	Substratum Chooser	Phytophil	IF
<i>Etheostoma lepidum</i>	N			-		Brood Hiders	Lithophil	IF
<i>Etheostoma spectabile</i>	N	0.13	0.71	↑	0.01	Brood Hiders	Lithophil	IF
<i>Percina carbonaria</i>	N	0.39	0.09	↓	0.03	Brood Hiders	Lithophil	IF
<i>Percina macrolepida</i>	N		0.09	-		Brood Hiders	Lithophil	IF
<i>Percina apristis</i>	N	2.25	2.28	S		Brood Hiders	Lithophil	IF
<i>Percina shumardi</i>	N	0.05	0.01	-		Brood Hiders	Lithophil	IF
<i>Cichlasoma cyanoguttatum</i>	I	3.83	1.47	↓	0.03	Substratum Chooser	Lithophil	IF
<i>Cichlasoma nigrofasciatum</i>	I		0.03	-		Substratum Chooser	Lithophil	IF
<i>Oreochromis aureus</i>	I		0.19	-		Bearer	Mouth Brooder	O
<i>Oreochromis mossambicus</i>	I		0.03	-		Bearer	Mouth Brooder	O
Collections During Period:		47	47					
Individuals Collected:		10,695	48,032					
Taxa Richness:		48	58					
Diversity:		0.91	0.80					

¹Species presumed extirpated (Kelsch and Hendricks 1990)²Species presumed extinct (Miller et al. 1989)

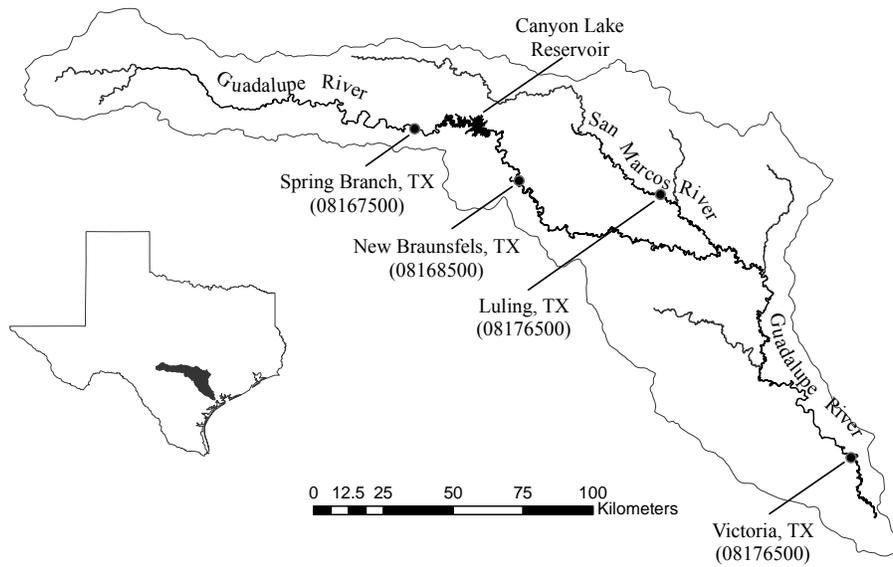


Figure 1. Guadalupe River basin in south central Texas and location of USGS stream flow gauging stations used in analysis of flow alterations.

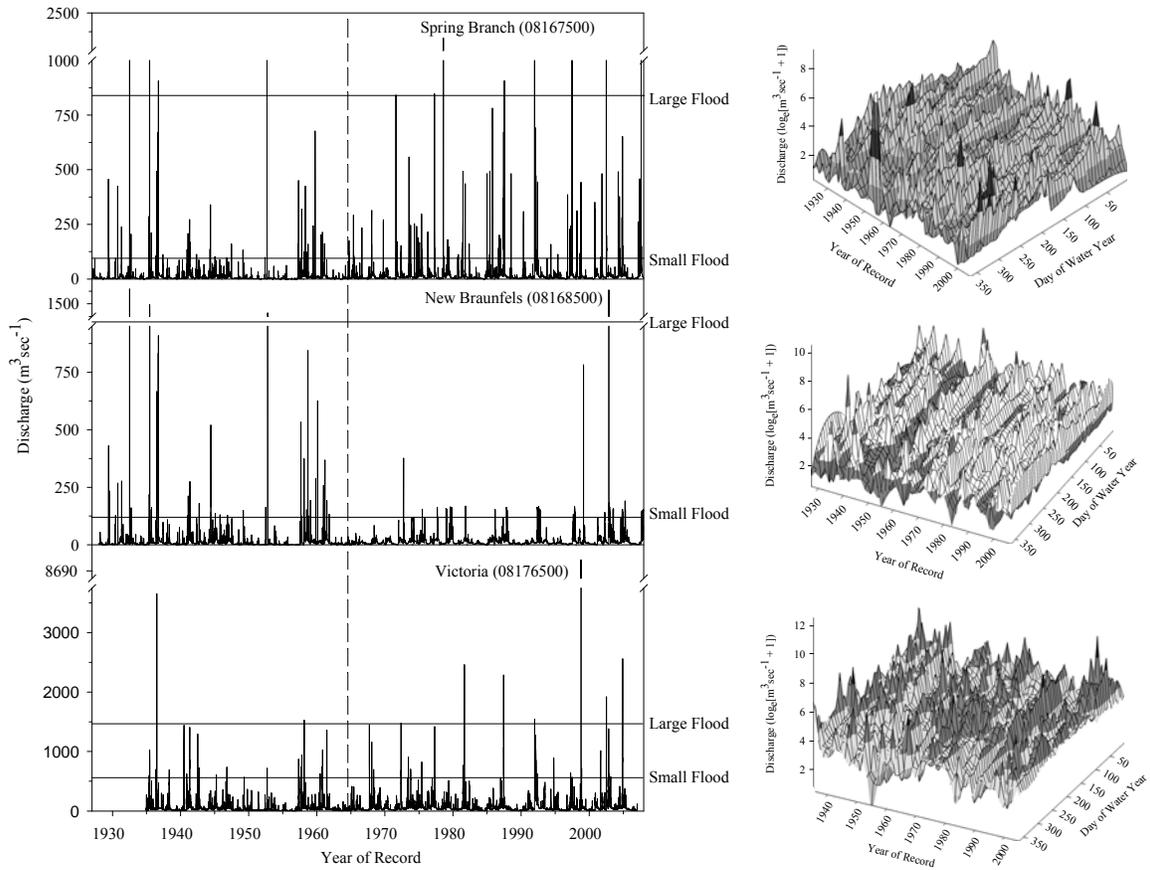


Figure 2. Hydrographs and flow histories of the Guadalupe River at Spring Branch, New Braunfels and Victoria, Texas USGS gauging stations. Indicators of Hydrologic Alteration was used to calculate thresholds and occurrence of large and small floods. Vertical dashed line indicates the completion of Canyon Lake Reservoir in 1964.

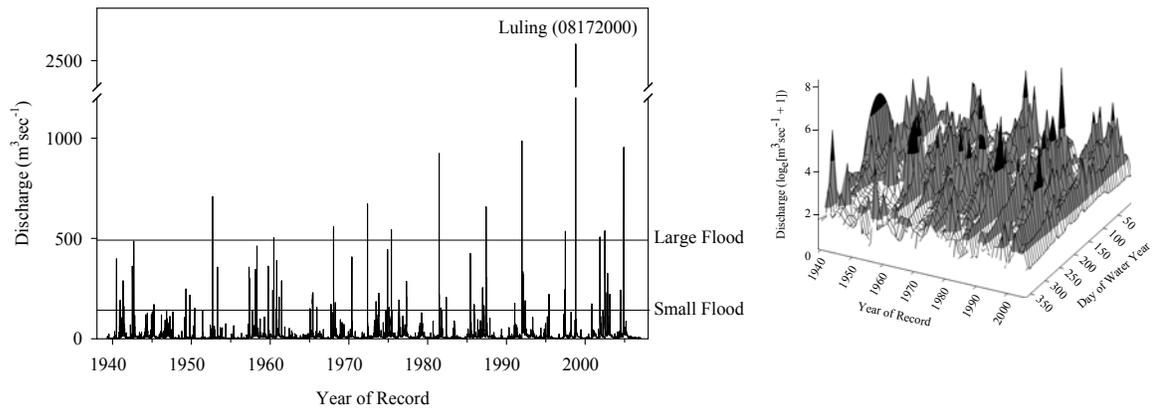


Figure 3. Hydrograph and flow history of the San Marcos River at Luling, Texas USGS gauging station. Indicators of Hydrologic Alteration was used to calculate thresholds and occurrence of large and small floods.

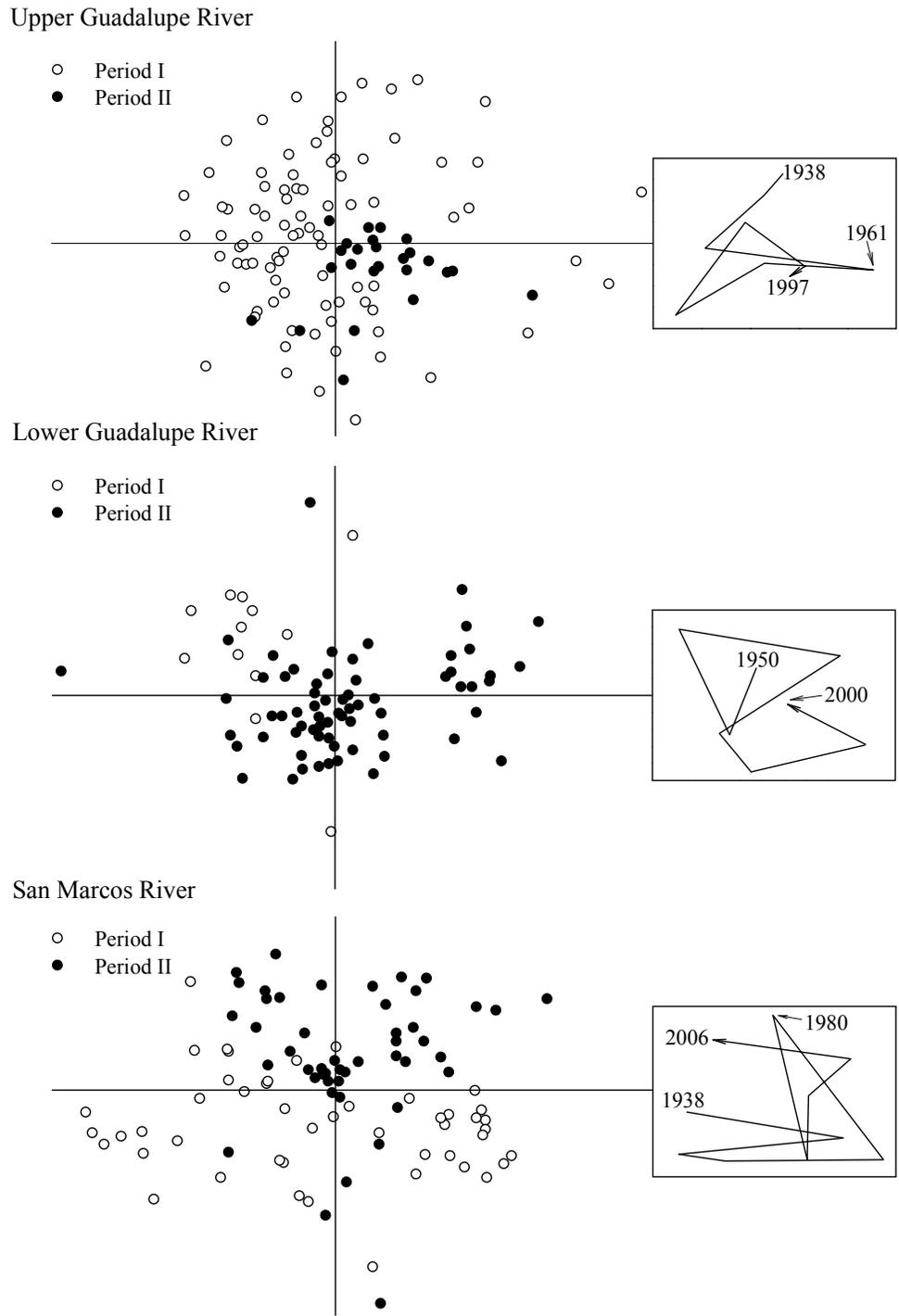


Figure 4. Multi-dimensional scaling (MDS) plots and trajectories for upper Guadalupe, lower Guadalupe and San Marcos River fish assemblages. Points represent ichthyological collections during Period I (open circles) and Period II (black circles) and are plotted following fourth root transformation of relative abundances. Trajectory plots represent 5-year running averages of MDS coordinates.

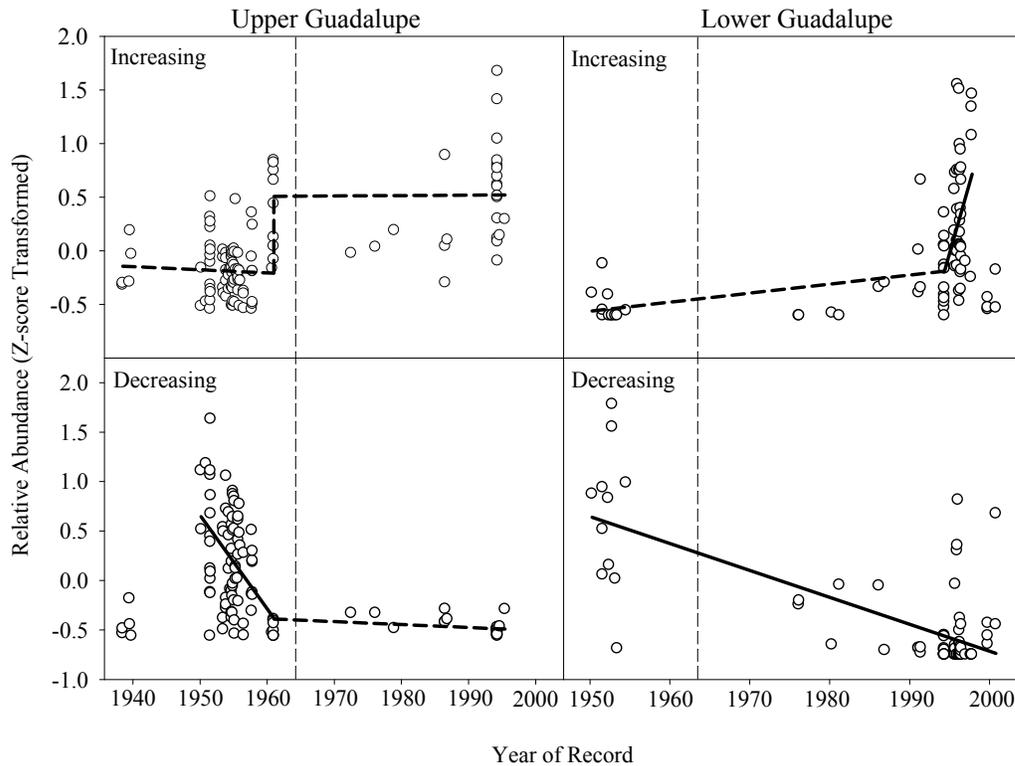


Figure 5. Joinpoint analysis of mainstem Guadalupe River fish assemblages upstream and downstream of Canyon Lake Reservoir. Open circles represent Z-score transformed mean relative abundance for all species demonstrating significant increasing or decreasing populations. Solid lines indicate significant slopes ($\beta_1 \neq 0$), dashed lines indicate non-significant slopes ($\beta_1 = 0$), non-significant slopes generated with < 5 observations not shown. Vertical dashed line illustrates impoundment of Canyon Lake Reservoir in 1964.

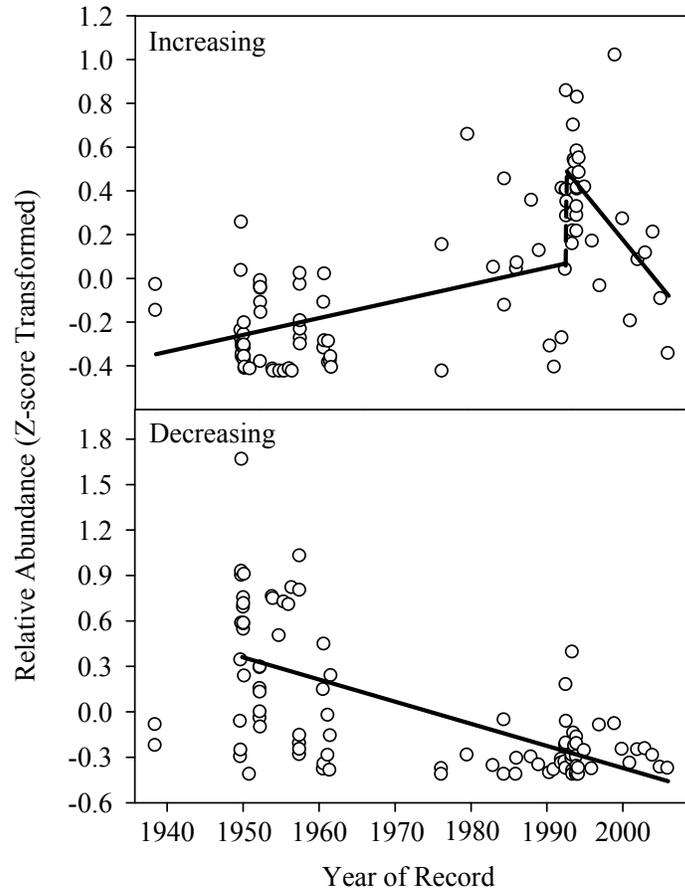


Figure 6. Joinpoint analysis of San Marcos River fish assemblage. Open circles represent Z-score transformed mean relative abundance for all species demonstrating significant increasing or decreasing populations. Solid lines indicate significant slopes ($\beta \neq 0$), dashed lines indicate non-significant slopes ($\beta = 0$), non-significant slopes generated with < 5 observations not shown.

Section II: Current Status and Historical Changes in the Trinity River Fish Assemblage

Study Objectives

Objectives of this study were to quantify changes in fish species abundance, to describe trends in fish assemblage composition and water quality parameters during 1970-2008, and to assess current status and conservation concerns for freshwater fishes in the Trinity River between Dallas-Fort Worth and Galveston Bay.

Study Area

Trinity River mainstem originates at the confluence of West Fork and Elm Fork in Dallas County, Texas, and flows southeast to Trinity Bay in Chambers County, Texas, encompassing a total drainage basin area of 46,500 km². Upper reaches of the drainage basin traverse hilly topography and cretaceous limestone of the Cross Timbers and Blackland Prairie ecoregions. Sub-watersheds include heavily urbanized Dallas-Fort Worth metropolitan area (Dahm et al. 2005). Lower reaches traverse gently rolling hills and sedimentary rocks of the East Central Texas Plain and Western Gulf Coastal Plain ecoregions. In all, 21 major reservoirs were constructed within the drainage basin, with the highest densities located within Cross Timbers and Blackland Prairie ecoregions. The largest reservoir, Lake Livingston, is located on the Trinity River mainstem about 150 km upstream from Trinity Bay. For the purposes of this study, Trinity River downstream of the Dallas-Fort Worth metropolitan area was divided into two sections: (1) the middle

Trinity River between Dallas-Fort Worth and Lake Livingston, and (2) the lower Trinity River between Lake Livingston and Trinity Bay.

Methods

Daily water chemistry data for ammonia (mg/L), total phosphorous (mg/L), sulfate (mg/L), nitrate (mg/L), biological oxygen demand (BOD; mg/L), and dissolved oxygen (DO; mg/L) were obtained from US Geological Survey (USGS) monitoring station 08057410 downstream of Dallas-Fort Worth. Water chemistry samples reviewed during this study were processed according to USGS standard field and laboratory methods (USGS 2009). Available data ranged 1967-2008 for sulfate, 1968-1997 for BOD, 1968-2008 for DO, and 1969-2008 for nitrate, ammonium, and phosphate. Water chemistry collection methods changed little through time, with the exception of filtering processes for nitrate and ammonia which were noted during analyses. Changes in concentrations through time were tested by regressing log-transformed concentrations against time, and tested for appropriate piecewise regression models with the program JOINPOINT (Joinpoint Regression Program, Version 3.0, National Cancer Institute 2005; Brendon and Bence 2008). Within JOINPOINT, grid search technique was used to optimize model parameters and parsimonious joinpoint models were selected according to permutation testing ($N = 5,000$; Brendon and Bence 2008). Analysis of water quality was limited to data collected immediately downstream of Dallas-Fort Worth; given this area is likely the source of many pollutants in downstream reaches (Dickson et al. 1989).

To test for potential dilution effects due to increased flows during 1967-2008, daily discharge data were obtained from three sites on the mainstem Trinity River (USGS

station 08057410, downstream of Dallas, Texas; USGS station 08065000, near Oakwood, Texas; USGS station 08066500, at Romayor, Texas). Linear regression in Indicators of Hydrologic Alteration (IHA, v. 7.0.3, Nature Conservancy 2002) was used to test for changes in base flow index, number of low flow pulse counts, number of high flow pulse counts, frequency of small floods, and frequency of large floods between 1967 and 2008. Within IHA, small floods were defined as high flow events (i.e., exceeding 75% of discharge) with recurrences of at least 2 years, and large floods were defined as high flow events with recurrences of at least 10 years (Runyan 2007).

Existing biological data for freshwater fishes collected in the Trinity River was reviewed by examining peer-reviewed publications, federal and state agency reports, unpublished data, private consulting firm reports, and museum collections. Queries were run at numerous museums within and outside of the state of Texas, including the Texas Natural History Museum (TNHC; University of Texas), Texas Cooperative Wildlife Collection (TCWC; Texas A&M University), Tulane Museum of Natural History (Tulane), University of Kansas Natural History Museum (KU), University of Michigan Museum of Zoology (UMMZ), Field Museum of Natural History (FMNH; Chicago, Illinois), San Noble Oklahoma Museum of Natural History (OMNH; University of Oklahoma), and the National Museum of Natural History (NMNH; Smithsonian). Agency reports and published literature reviewed for this study included Texas Parks and Wildlife (TPWD; 1974), Conner (1977), Dickson et al (1989), Kleinsasser and Linam (1989), Gelwick et al. (2000), Gelwick et al. (2001), and PCRA (2008). Unpublished data included in this study were obtained from Bruce Moring and John Rosendale of the USGS (unpubl. data, 1992-2006). Collections were divided into middle and lower

Trinity River reaches as well as tributary streams, and collection techniques (i.e., seining and electrofishing) were summed within collections. Collections made with unknown techniques or for which sampling effort could not be determined, were noted and removed from analyses, which generally included many museum records. Consequently, only collections consisting of combined electrofishing and seining were used in analyses, and many museum collection records were used for presence or absence data only. Relative abundance of each species was figured as $([\text{number of each species}/\text{total number of individuals}] * 100)$ and compared across collection periods and river reaches. Tributary collections were included in analyses because water quality degradation occurred to a lesser extent within tributary streams, which may have served as source populations during recolonization of the mainstem for some species (e.g., Gammon 1995; Ward 1998).

Assemblage compositions were compared among collection periods and river reaches using analysis of similarity (ANOSIM) and Bray-Curtis similarity matrices (Bray and Curtis 1957) created in Primer 6.1.6 (Plymouth Marine Laboratory 2006). Relative abundance data were fourth-root transformed to standardize the contribution of high and low abundance species and compared with ANOSIM ($\alpha = 0.05$; 9,999 permutations) using permutations to assess average rank dissimilarity between periods and reaches (Runyan, 2007). Multi-dimensional scaling (MDS) was used to illustrate spatiotemporal variability in assemblage composition among periods and reaches. Similarity indices among collection periods and river reaches were created by averaging relative abundances of all seine and electrofishing efforts within each collection (e.g., all species collected by TPWD 1974) and plotted as a dendrogram of hierarchical clustering to illustrate

relationships among periods and reaches. Occurrence of native and introduced species was determined according to Conner and Suttkus (1986) and Thomas et al. (2007) and conservation status of each species was determined according to Hubbs et al. (2008).

Results

Temporal changes in fish assemblage composition

Among 99 species collected or reported within the Trinity River basin (Conner and Suttkus 1986; Thomas et al. 2007; Hubbs et al. 2008), 86 occurred within collections reviewed during this study (TPWD 1974; Dickson et al. 1989; Kleinsasser and Linam 1989; USGS, unpubl. data; Gelwick et al. 2000; Gelwick et al. 2001; PCRA 2008).

Twenty-four collections taken from the middle Trinity River by the TPWD during 1972-1974 consisted of 3,187 individuals, representing 12 families and 34 species (Table 1). Clupeidae was the most abundant family (53% relative abundance), followed by Centrarchidae (24%), Cyprinidae (8%), Moronidae (5%), and Sciaenidae (4%). Fifty-nine collections taken from the Middle Trinity River by Dickson et al. (1989) and Kleinsasser and Linam (1989) during 1987-1988 consisted of 77,423 individuals, representing 12 families and 47 species. Cyprinidae was the most abundant family (89%), followed by Poeciliidae (7%), Clupeidae (1%), and Ictaluridae (1%). Twenty-one collections taken from the middle Trinity River by the USGS during 1992-2006 consisted of 1,692 individuals, representing 13 families and 38 species. Cyprinidae was the most abundant family (32%), followed by Ictaluridae (23%), Clupeidae (21%), Centrarchidae (10%), Lepisosteidae (6%), and Catostomidae (5%). Forty-five collections taken from Trinity River tributaries between Dallas-Fort Worth and Galveston Bay during 1970-

2006 (TCWC $n = 3$; TNHC $n = 19$; USGS $n = 11$; Gelwick et al. 2000 $n = 8$; Gelwick et al. 2001 $n = 4$) consisted of 8,635 individuals, representing 14 families and 65 species. Cyprinidae was the most abundant family (44%), followed by Centrarchidae (22%), Poeciliidae (16%), Clupeidae (7%), Ictaluridae (3%), and Percidae (3%). Twenty-one collections taken from the lower Trinity River by Conner (1977) during 1970-1972 consisted of 53,628 individuals, representing 14 families and 47 species. Cyprinidae was the most abundant family (95%), followed by Poeciliidae (3%), and Atherinidae (1%). Twenty-two collections taken from the lower Trinity River by the USGS and PCRA (2008) during 1992-2008 consisted of 24,443 individuals, representing 20 families and 53 species. Clupeidae was the most abundant species (57%), followed by Cyprinidae (24%), Atherinidae (9%), Centrarchidae (4%), and Poeciliidae (2%). Among collections, Simpson's diversity index and species richness were highest for tributary collections (1-D = 0.91, $S = 65$), and middle and lower mainstem reaches collections taken during 1992-2008 had highest species diversity (middle: 1-D = 0.89; lower: 1-D = 0.67).

Fish assemblage composition differed among collection periods and river reaches (ANOSIM global $R = 0.624$, $P < 0.01$). Bray-Curtis Similarity results suggested collections taken during 1970-1972 in the lower Trinity River were least similar (Bray-Curtis similarity index = 45%) to other collections, followed by collections taken from the middle Trinity River during 1972-1974 (53%; Figure 4). Within the middle Trinity River, collections taken during 1972-1974 were least similar to tributary collections, whereas collections taken during 1987-1988 were increasingly similar to tributary collections as well as lower Trinity River collections taken during 1970-1972 (Figure 4a). Among reaches and collection periods, middle and lower Trinity River collections taken

during 1993-2006 (middle) and 1993-2008 (upper) were most similar (64%; Figure 4b) and indicated the highest degree of overlap in multi-dimensional space.

Eight imperiled species listed by Hubbs et al (2008) were reported in collections reviewed in this study. Imperiled species reported in a single collection included *Polyodon spathula* (PCRA 2008), *Anguilla rostrata* (PCRA 2008), *Notropis potteri* (Conner 1977), and *Erimyzon oblongus* (TNHC #30808). *Polyodon spathula* and *Anguilla rostrata* were reported in low abundances among recent collections downstream of Lake Livingston (PCRA 2008); however, *Notropis potteri* was not reported after the early 1970s (Conner 1977). Contemporary relative abundances increased since the 1970s in the middle Trinity River for *Atractosteus spatula* (0.06% to 0.18%) and in the lower Trinity River for *Notropis sabinae* (0.00% to 0.58%) and *Notropis shumardi* (0.61% to 4.13%). Imperiled species reported only within tributary collections included *Erimyzon oblongus* (0.01%) and *Notropis atrocaudalis* (0.28%). *Cycleptus elongatus* is listed as native to the Trinity River drainage (Conner and Suttkus 1986), but was not reported among collections reviewed during this study.

Temporal changes in water quality and quantity

Detectable changes in water quality were found between 1967 and 2008. Concentrations of total phosphorus (mg/L), ammonia (mg/L), sulfate (mg/L), and biological oxygen demand (BOD; mg/L) generally decreased through time, whereas concentrations of nitrate (mg/L) and dissolved oxygen (mg/L) increased (Figure 2). One joinpoint was the most parsimonious model for total phosphorous, ammonia, sulfate, and BOD ($P < 0.02$), whereas two joinpoints was the most parsimonious model for nitrate

and dissolved oxygen ($P < 0.02$). For total phosphorous, one joinpoint in 1995 denoted two distinct regression models with independent variables ranging 1969-1995 and 1995-2008, and phosphorous concentration was negatively associated ($b_1 = -0.000076$, $P < 0.01$) with time during 1969-1995 and not associated ($b_1 = 0.000003$, $P = 0.9$) with time during 1995-2008. For ammonia, one joinpoint in 1993 denoted two distinct regression models with independent variables ranging 1969-1993 and 1993-2008, and ammonia concentration was negatively associated ($b_1 = -0.000258$, $P < 0.01$) with time during 1969-1993 and not associated ($b_1 = 0.000039$, $P = 0.1$) with time during 1993-2008. For sulfate, one joinpoint in 1978 denoted two distinct regression models with independent variables ranging 1967-1978 and 1978-2008, and sulfate concentration was not associated ($b_1 = 0.000019$, $P = 0.08$) with time during 1967-1978 and negatively associated ($b_1 = -0.000017$, $P < 0.01$) with time during 1978-2008. For BOD, one joinpoint in 1980 denoted two distinct regression models with independent variables ranging 1968-1980 and 1980-1997, and BOD concentration was not associated ($b_1 = 0.00033$, $P = 0.09$) with time during 1968-1980 and negatively associated ($b_1 = -0.000107$, $P < 0.01$) with time during 1980-1997. For nitrate, two joinpoints during 1977 and 1988 denoted three distinct regression models with independent variables ranging 1967-1977, 1977-1988, and 1988-2008, and nitrate concentration was negatively associated ($b_1 = -0.000312$, $P < 0.01$) with time during 1967-1977, positively associated ($b_1 = 0.000249$, $P < 0.01$) with time during 1977-1988, and not associated ($b_1 = 0.000002$, $P = 0.8$) with time during 1988-2008. For dissolved oxygen, two joinpoints during 1976 and 1982 denoted three distinct regression models with independent variables ranging 1968-1976, 1976-1982, and 1982-2008, and dissolved oxygen

concentration was negatively associated ($b_I = -0.000099$, $P = 0.03$) with time during 1968-1976, positively associated ($b_I = 0.000302$, $P < 0.01$) with time during 1976-1982, and not associated ($b_I = 0.00001$, $P = 0.2$) with time during 1982-2008.

Few changes in flow were detected for the mainstem Trinity River between 1967 and 2008. Base flow indices did not differ through time downstream of Dallas-Fort Worth ($b_I = -0.0009$, $P = 0.5$), near Oakwood, Texas ($b_I = 0.0032$, $P = 0.05$), or at Romayor, Texas ($b_I = 0.0028$, $P = 0.25$). Number of low pulse flow counts decreased through time downstream of Dallas-Fort Worth ($b_I = -0.4681$, $P < 0.01$), but did not differ through time near Oakwood, Texas ($b_I = -0.0967$, $P = 0.25$) or at Romayor, Texas ($b_I = -0.0393$, $P = 0.25$). Number of high flow pulse counts increased through time downstream from Dallas-Fort Worth ($b_I = 0.1541$, $P < 0.01$), but did not differ near Oakwood, Texas ($b_I = 0.0012$, $P = 0.5$) or at Romayor, Texas ($b_I = 0.0095$, $P = 0.5$). Between 1967 and 2008, annual frequency of small (threshold = $576 \text{ m}^3 \text{ sec}^{-1}$) and large ($1,269 \text{ m}^3 \text{ sec}^{-1}$) floods did not differ downstream from Dallas-Fort Worth (small flood $b_I = 0.0059$, $P = 0.5$; large flood $b_I = 0.0019$, $P = 0.5$), frequency of small ($942 \text{ m}^3 \text{ sec}^{-1}$) and large ($2,145 \text{ m}^3 \text{ sec}^{-1}$) floods did not differ near Oakwood, Texas (small flood $b_I = 0.0012$, $P = 0.5$; large flood $b_I = -0.0011$, $P = 0.5$), and frequency of small ($1,424 \text{ m}^3 \text{ sec}^{-1}$) and large ($2,580 \text{ m}^3 \text{ sec}^{-1}$) floods did not differ at Romayor, Texas (small flood $b_I = 0.0139$, $P = 0.25$; large flood $b_I = 0.0005$, $P = 0.25$; Figure 3).

Discussion

During the 40 years of information assessed here, aspects of fish assemblage composition, water quality, and water quantity differed through time in the Trinity River

between Dallas-Fort Worth and Galveston Bay. Specifically, species richness and diversity generally increased in middle and lower mainstem reaches, becoming increasingly similar to tributary assemblage composition. During this time period, discharges from the Dallas-Fort Worth Metroplex changed little in terms quantity (i.e., no detectable change in base flow index), whereas quality of discharges became less polluted through time. Magnitude of eutrophication and biological oxygen demand reduced, while concentration of dissolved oxygen increased between 1967 and 2008. These factors collectively contributed to contemporary fish assemblage composition that is likely more characteristic of naturally occurring assemblage composition, although a paucity of historical records exists for the Trinity River before large-scale alteration.

Prior to the 1970s, extensive water quality degradation in the Trinity River basin was confined largely to the mainstem channel, leaving assemblage composition of many tributary streams downstream of Dallas-Fort Worth relatively unaltered (Dickson et al. 1989; Kleinsasser and Linam 1989). During the 1970s and 1980s, middle and lower mainstem reaches of the Trinity River were numerically dominated by cosmopolitan species with broad physiological tolerances, including *Cyprinella lutrensis*, *Pimephales vigilax*, *Dorosoma cepedianum*, and *D. petenense* (Miller 1960; Matthews 1985; Rutledge and Beiting 1989). Species not reported in middle or lower mainstem reaches during the 1970s, which subsequently indicated increasing relative abundance, generally occurred in adjacent tributary streams, including *Cyprinella venusta*, *Lythrurus fumeus*, *Notropis volucellus*, *Ictiobus bubalus*, *Ictalurus furcatus*, *Noturus gyrinus*, *Lepomis humilis*, *Percina macrolepida*, and *Percina sciera*. Meta-communities are known to exist between mainstem and tributary assemblages, especially for fish species capable of

inhabiting a variety of stream sizes (Fausch et al. 2002; Wiens 2002). Furthermore, tributary streams are considered essential refugia for stream-dwelling fishes during disturbances (Detenbeck et al. 1992), including disturbances that eliminate some species altogether (Sedell et al. 1990). Accordingly, recolonization of the mainstem Trinity River by many species after improvement of water quality was likely facilitated by persistence of refuge populations in tributary streams (Gammon 1995), as illustrated by increasing similarities between mainstream reaches and tributaries.

Efforts initiated during the early 1970s to improve water quality in the Trinity River downstream of Dallas-Fort Worth were successful. Concentration of total phosphorus and ammonia declined significantly during 1967-1995 and indicated no significant change in composition 1995-2008. Similarly, sulfate and BOD concentrations began to significantly decline in the late 1970s and early 1980s and concentrations of nitrate and dissolved oxygen declined initially, but began to increase in 1976 before stabilizing in during the 1980s. The Upper Trinity River Basin Comprehensive Sewage Plan of 1971 eliminated many industrial and municipal wastewater discharges from Dallas-Fort Worth and marked the beginning of modern secondary wastewater treatment (Dickson et al. 1989). Secondary wastewater treatment in Dallas-Fort Worth likely attenuated eutrophication within the mainstem Trinity River and reduced discharges of ammonia and nitrate (Eklov et al. 1998), as illustrated by reductions in ammonia and nitrate during the early 1970s. Biological oxygen demand levels likely reduced during this time period because of reduced oxidation of ammonia to nitrate, but measurable concentrations remained high within the river until 1980 likely as an artifact of high microbial activity within Trinity River sediments (Dickson et al. 1989; Kleinsasser and

Linam 1989). High remnant levels of BOD likely resulted in the lag associated with increased dissolved oxygen, which increased as BOD declined until the early 1980s. Declining concentrations of sulfate were likely related to elimination of untreated industrial runoff from Dallas-Fort Worth in 1971 (Dickson et al. 1989) and possibly to reduced domestic oil production and oilfield runoff in the Trinity River watershed (Van Metre and Callender 1996). Based on the results of this study, the goal of restoring fishable and swimmable waters by 1983, set forth by the Federal Clean Water act of 1972 (CWA 1972) was successful within the mainstem Trinity River and contemporary water quality does support a productive and diverse fish assemblage. However, taxonomic concern exists for rare and imperiled species within the Trinity River.

Declines in abundance and distribution have occurred for imperiled Trinity River fishes in and outside of the drainage, including *Polydon spathula* (Graham 1997), *Atractosteus spatula* (Jelks et al. 2008), *Anguilla rostrata* (Haro et al. 2000), *Notropis potteri* (Perkin et al. 2009), *Notropis sabinae* (Williams and Bonner 2006), and *Notropis shumardi* (Runyan 2007; Hubbs et al. 2008). Results of this study suggest these species exist in rare abundances in the lower Trinity River or might be extirpated from the drainage. Destruction or degradation of habitat is a primary factor contributing to the decline of many of these species, including alteration of flow regime, riverscape fragmentation, and water quality degradation (Jelks et al. 2008; Hubbs et al. 2008). *Polydon spathula* have declined within the Trinity River of Texas (Blackwell et al. 1995) as have *Anguilla rostrata* (Haro et al. 2000), whereas contemporary abundances of *Atractosteus spatula*, *Notropis sabinae*, and *Notropis shumardi* appear to be greater than abundances reported during the 1970s. Each of the latter species maintain populations in

lower portions of the Trinity River and persistence might be related to relatively little change in flow regime and stream channel morphology downstream of Lake Livingston (Wellmeyer et al. 2004; Phillips et al. 2005); which are reported as primary causes of decline in other drainages (Williams and Bonner 2006; Runyan 2007; Hubbs et al. 2008). *Notropis potteri* has not been reported in the Trinity River since the early 1970s, when the species was apparently confined to the lower reaches downstream of Lake Livingston (Conner 1977). Given population declines throughout *Notropis potteri* range (Perkin et al. 2009); localized extirpation might have occurred in the lower Trinity River and should be investigated further. Lastly, given the apparent narrow distribution of imperiled species in the lower Trinity River, future anthropogenic stream alteration including installation of hydroelectric power generators and further manipulation of flows at Lake Livingston (PCRA 2008) should carefully consider negative impacts upon native fish populations.

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Table 1 continued

Scientific Name	Common Name	Status	Middle Mainstem			Tributaries	Lower Mainstem	
			1972-1974	1987-1988	1992-2006	1970-2006	1970-1972	1992-2008
<i>Notropis atrocaudalis</i>	blackspot shiner	NE	0.00	0.00	0.00	0.28	0.00	0.00
<i>Notropis buchanani</i>	ghost shiner	N	0.00	1.48	0.00	0.00	3.64	0.00
<i>Notropis potteri</i>	chub shiner	NE	0.00	0.00	0.00	0.00	0.06	0.00
<i>Notropis sabinae</i>	Sabine shiner	NE	0.00	0.00	0.00	0.00	0.00	0.58
<i>Notropis shumardi</i>	silverband shiner	NE	0.00	0.05	0.00	0.00	0.61	4.13
<i>Notropis stramineus</i>	sand shiner	N	0.00	0.00	0.06	0.00	0.00	0.02
<i>Notropis texanus</i>	weed shiner	N	0.00	0.02	0.00	3.20	< 0.01	0.00
<i>Notropis volucellus</i>	mimic shiner	N	0.00	0.06	1.42	0.37	< 0.01	2.84
<i>Opsopoeodus emiliae</i>	pugnose minnow	N	0.06	< 0.01	0.00	0.71	0.07	0.00
<i>Phenacobius mirabilis</i>	suckermouth minnow	N	0.00	0.00	0.00	0.00	0.16	0.00
<i>Pimephales promelas</i>	fathead minnow	N	0.00	0.00	0.00	0.02	0.00	0.00
<i>Pimephales vigilax</i>	bullhead minnow	N	0.06	13.53	4.20	5.04	29.42	2.36
<i>Semotilus atromaculatus</i> *	creek chub	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Carpionodes carpio</i>	river carpsucker	N	0.50	0.01	0.06	0.03	0.13	0.12
<i>Cycleptus elongatus</i> *	blue sucker	NE	0.00	0.00	0.00	0.00	0.00	0.00
<i>Erimyzon oblongus</i>	creek chubsucker	NE	0.00	0.00	0.00	0.01	0.00	0.00
<i>Erimyzon sucetta</i>	lake chubsucker	N	0.00	0.00	0.00	0.15	0.00	0.00
<i>Ictiobus bubalus</i>	smallmouth buffalo	N	0.50	0.20	4.43	1.66	0.00	0.67
<i>Minytrema melanops</i>	spotted sucker	N	0.03	0.00	0.00	0.09	0.02	0.00
<i>Moxostoma poecilurum</i>	blacktail redhorse	N	0.00	0.00	0.00	0.03	< 0.01	0.01
<i>Astyanax mexicanus</i> *	Mexican tetra	I	0.00	0.00	0.00	0.00	0.00	0.00
<i>Ameiurus melas</i>	black bullhead	N	0.06	< 0.01	0.00	0.20	0.00	0.00

Table 1 continued

Scientific Name	Common Name	Status	Middle Mainstem			Tributaries	Lower Mainstem	
			1972-1974	1987-1988	1992-2006	1970-2006	1970-1972	1992-2008
<i>Ameiurus natalis</i>	yellow bullhead	N	0.09	0.01	0.00	0.57	0.00	< 0.01
<i>Ictalurus furcatus</i>	blue catfish	N	0.00	0.41	8.98	0.14	0.00	0.60
<i>Ictalurus punctatus</i>	channel catfish	N	0.50	0.07	1.95	0.45	0.42	0.32
<i>Noturus gyrinus</i>	tadpole madtom	N	0.00	0.00	0.30	0.61	0.00	0.00
<i>Noturus nocturnus</i>	freckled madtom	N	0.00	0.04	0.12	0.65	0.08	0.00
<i>Pylodictis olivaris</i>	flathead catfish	N	0.09	0.15	11.94	0.05	0.00	0.05
<i>Esox americanus</i>	grass pickerel	N	0.00	0.00	0.00	0.13	0.00	0.00
<i>Aphredoderus sayanus</i>	pirate perch	N	0.00	0.00	0.00	0.43	0.00	0.00
<i>Cyprinodon variegatus</i>	sheepshead minnow	N	0.00	0.00	0.00	0.00	0.00	< 0.01
<i>Fundulus chrysotus</i> *	golden topminnow	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Fundulus blairae</i> *	western starhead topminnow	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Fundulus dispar</i>	starhead topminnow	N	0.00	0.00	0.00	0.01	0.00	0.00
<i>Fundulus grandis</i> *	Gulf killifish	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Fundulus notatus</i>	blackstripe topminnow	N	0.13	0.01	0.12	1.23	< 0.01	< 0.01
<i>Fundulus olivaceus</i>	blackspotted topminnow	N	0.00	0.00	0.00	1.09	0.00	0.00
<i>Fundulus pulvereus</i> *	bayou killifish	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Fundulus zebrinus</i> *	plains killifish	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Lucania parva</i> *	rainwater killifish	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Gambusia affinis</i>	western mosquitofish	N	0.06	6.80	0.65	15.70	2.53	1.89
<i>Poecilia latipinna</i> *	sailfin molly	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Labidesthes sicculus</i>	brook silverside	N	1.00	0.00	0.06	0.30	< 0.01	0.00
<i>Membras martinica</i>	rough silverside	N	0.00	0.00	0.00	0.00	< 0.01	0.00

Table 1 continued

Scientific Name	Common Name	Status	Middle Mainstem			Tributaries	Lower Mainstem	
			1972-1974	1987-1988	1992-2006	1970-2006	1970-1972	1992-2008
<i>Menidia beryllina</i>	inland silverside	N	0.06	0.44	0.30	0.51	0.62	8.68
<i>Morone chrysops</i>	white bass	I	4.58	0.10	0.18	0.03	< 0.01	0.12
<i>Morone mississippiensis</i>	yellow bass	N	0.60	0.01	0.00	0.25	< 0.01	0.23
<i>Morone saxatilis</i>	striped bass	I	0.00	< 0.01	0.06	0.01	0.00	0.37
<i>Elassoma zonatum</i> *	banded pygmy sunfish	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Centrarchus macropterus</i>	flier	N	0.00	0.00	0.00	0.16	0.00	0.00
<i>Lepomis auritus</i>	redbreast sunfish	I	0.00	< 0.01	0.00	0.02	0.00	0.00
<i>Lepomis cyanellus</i>	green sunfish	N	0.22	0.09	0.95	1.56	< 0.01	0.02
<i>Lepomis gulosus</i>	warmouth	N	0.56	0.03	1.60	2.04	0.01	0.04
<i>Lepomis humilis</i>	orangespotted sunfish	N	0.00	0.13	0.12	2.68	0.00	0.26
<i>Lepomis macrochirus</i>	bluegill	N	12.49	0.08	1.48	5.32	0.07	0.60
<i>Lepomis marginatus</i> *	dollar sunfish	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Lepomis megalotis</i>	longear sunfish	N	6.90	0.26	2.25	4.81	0.31	1.31
<i>Lepomis microlophus</i>	redecor sunfish	N	0.72	0.00	0.00	0.44	0.00	< 0.01
<i>Lepomis miniatus</i>	redspotted sunfish	N	0.00	< 0.01	0.00	0.54	0.00	0.01
<i>Lepomis symmetricus</i>	bantam sunfish	N	0.00	0.00	0.00	0.20	0.00	0.00
<i>Micropterus punctulatus</i>	spotted bass	N	0.03	< 0.01	0.65	0.38	0.03	0.39
<i>Micropterus salmoides</i>	largemouth bass	N	2.13	< 0.01	2.30	0.98	0.04	0.82
<i>Pomoxis annularis</i>	white crappie	N	0.56	0.03	0.77	1.66	0.01	0.03
<i>Pomoxis nigromaculatus</i>	black crappie	N	0.66	0.00	0.00	0.90	0.02	0.15
<i>Ammocrypta vivax</i>	scaly sand darter	N	0.00	0.00	0.00	0.00	0.08	0.00
<i>Etheostoma chlorosoma</i>	bluntnose darter	N	0.00	< 0.01	0.00	0.15	< 0.01	0.00

Table 1 continued

Scientific Name	Common Name	Status	Middle Mainstem			Tributaries	Lower Mainstem	
			1972-1974	1987-1988	1992-2006	1970-2006	1970-1972	1992-2008
<i>Etheostoma gracile</i>	slough darter	N	0.00	0.01	0.00	0.59	< 0.01	0.00
<i>Etheostoma parvipinne</i>	goldstripe darter	N	0.00	0.00	0.00	0.02	0.00	0.00
<i>Etheostoma proeliare</i>	cypress darter	N	0.00	< 0.01	0.00	0.00	0.00	0.00
<i>Etheostoma spectabile</i> *	orangethroat darter	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Etheostoma whipplei</i> *	redfin darter	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Percina macrolepida</i>	bigscale logperch	N	0.00	< 0.01	0.06	0.37	0.00	< 0.01
<i>Percina sciera</i>	dusky darter	N	0.00	0.01	0.30	1.41	0.11	0.05
<i>Aplodinotus grunniens</i>	freshwater drum	N	3.70	0.06	0.59	0.09	0.00	0.29
<i>Mugil cephalus</i>	striped mullet	N	0.00	0.00	0.00	0.00	0.10	1.57
<i>Mugil curema</i> *	white mullet	N	0.00	0.00	0.00	0.00	0.00	0.00
<i>Alosa chrysochloris</i>	skipjack herring	N	0.00	0.00	0.00	0.00	0.00	0.02
<i>Anchoa mitchilli</i>	bay anchovy	N	0.00	0.00	0.00	0.00	0.03	0.00
<i>Elop saurus</i>	ladyfish	N	0.00	0.00	0.00	0.00	0.00	< 0.01
<i>Strongylura marina</i>	Atlantic needlefish	N	0.00	0.00	0.00	0.00	< 0.01	0.00
<i>Trinectes maculatus</i>	hogchocker	N	0.00	0.00	0.00	0.00	0.00	< 0.01
<i>Kathetostoma giganteum</i>	giant stargazer	I	0.00	0.00	1.24	0.00	0.00	0.00
Total			3,187	77,423	1,692	8,635	53,628	24,443
Richness			34	47	38	65	47	53
Diversity			0.80	0.48	0.89	0.91	0.56	0.67

*Species reported in drainage, but not collected (Conner and Suttkus 1986; Thomas et al. 2007)

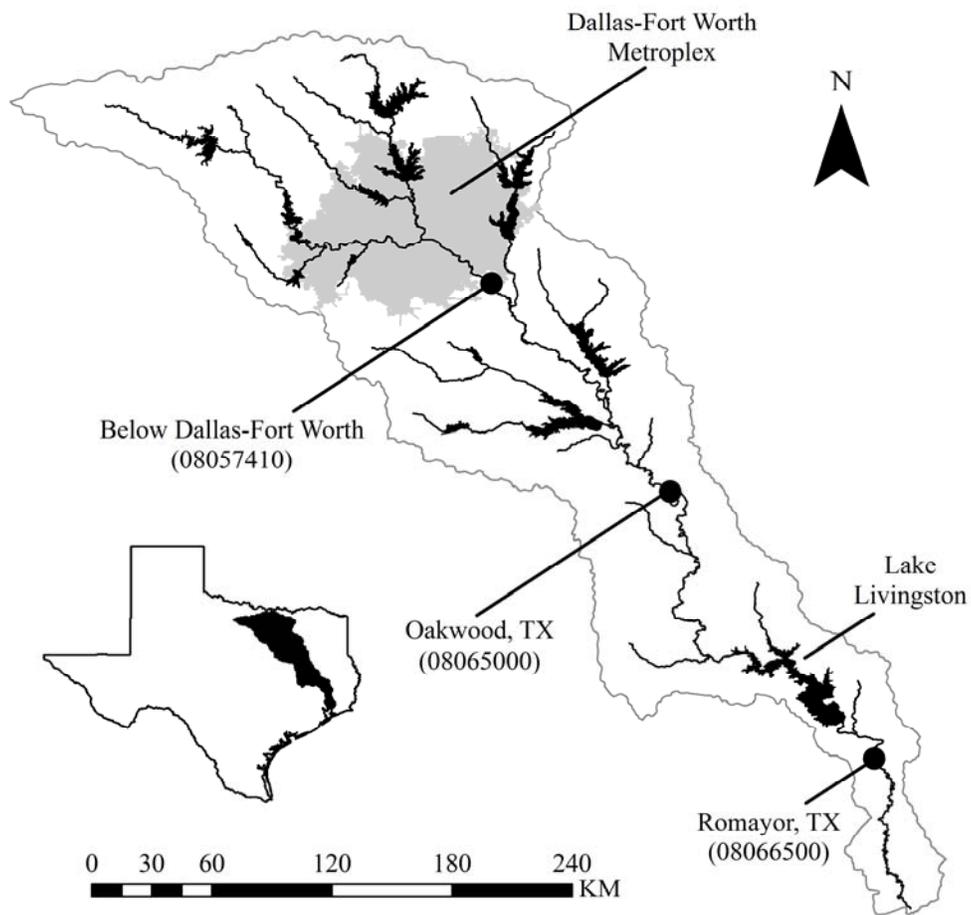


Figure 1. Trinity River Basin of Texas. Dots represent USGS gauging stations (station I.D. number) from which water quality or stream flow data were obtained.

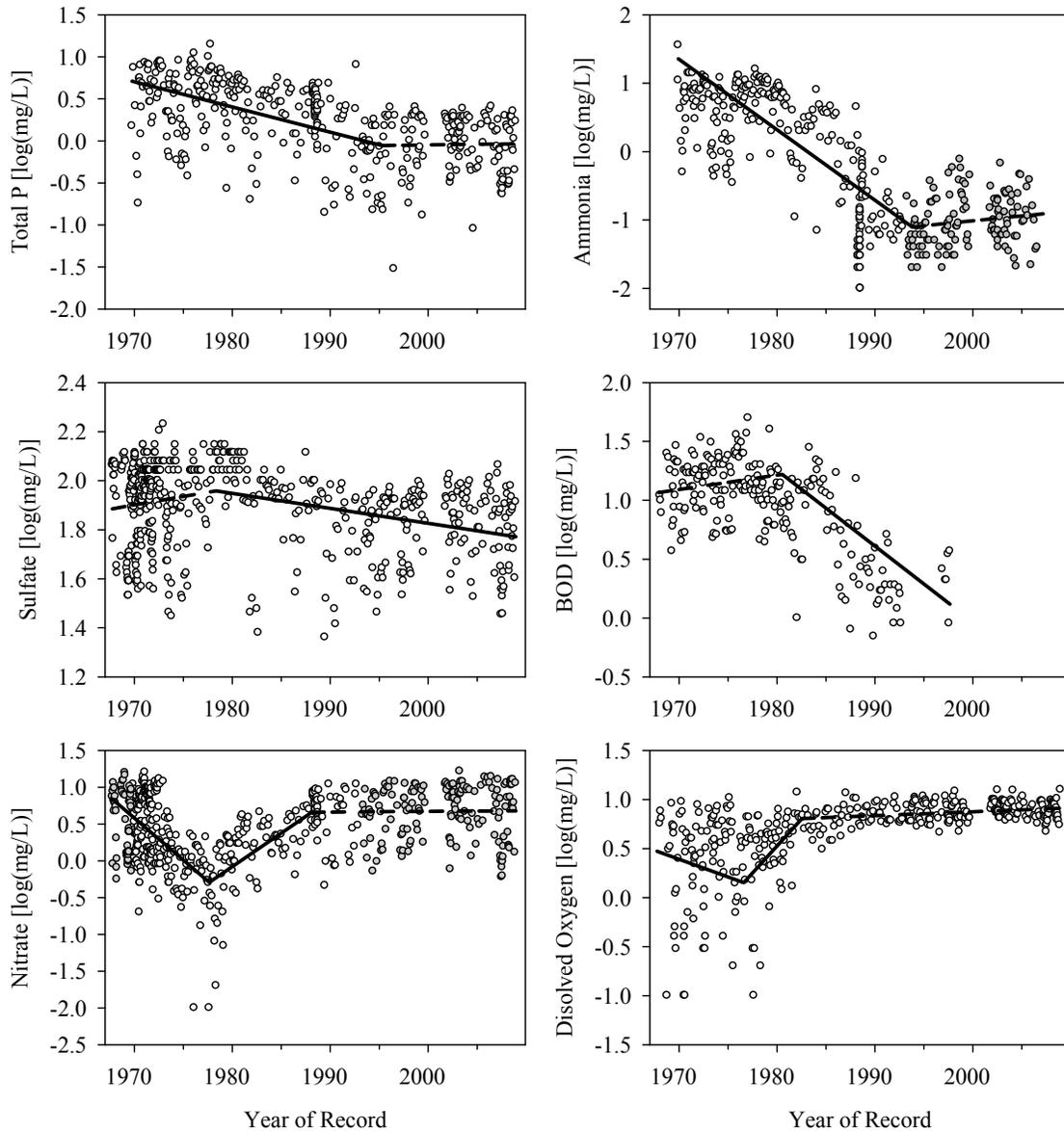


Figure 2. Joinpoint regressions for water quality parameters in the Trinity River 1967-2008 measured downstream of Dallas-Fort Worth (USGS gauge 08057410). Solid lines indicate slopes differ from zero ($b_1 \neq 0$, $P < 0.05$), dashed lines indicate slopes do not differ from zero ($b_1 = 0$, $P > 0.05$), white circles indicate filtered water samples, and grey circles indicate unfiltered water samples.

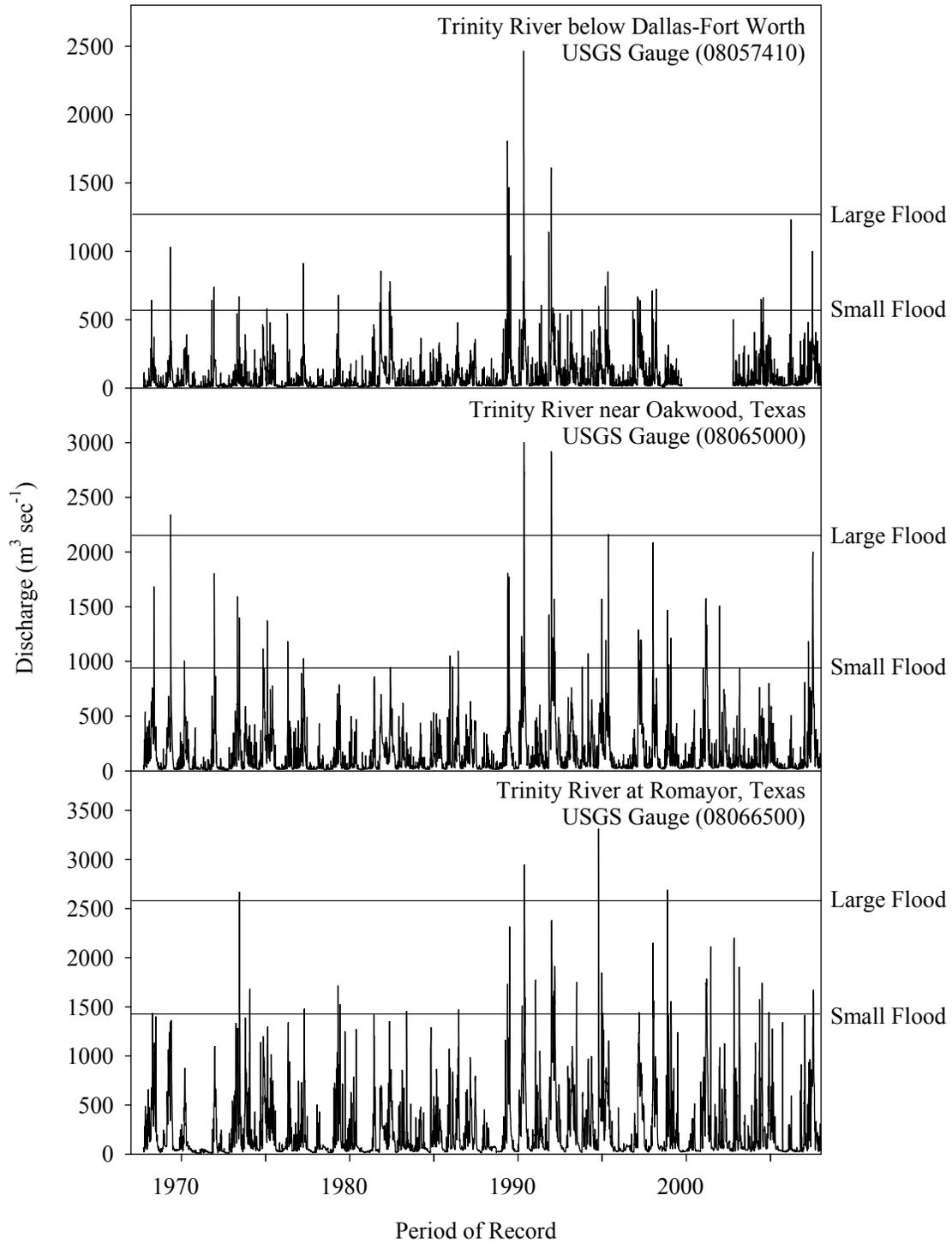


Figure 3. Hydrographs and thresholds for small and large floods in mainstem Trinity River downstream of Dallas-Fort Worth (USGS # 08057410), near Oakwood, Texas (USGS # 08065000), and at Romayor, Texas (USGS # 08066500) 1967-2008. Flood thresholds were calculated using Indicators of Hydrologic Alteration.

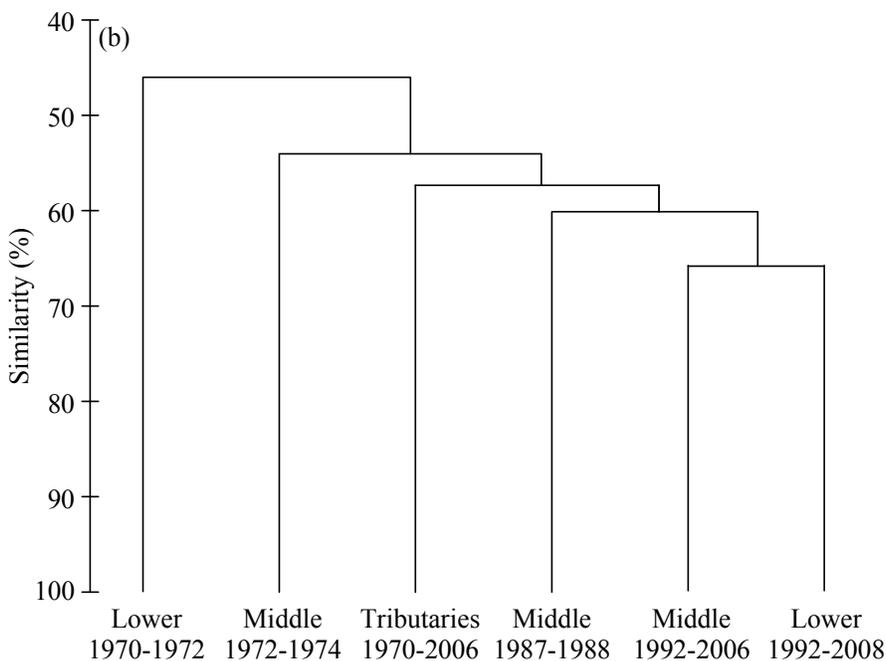
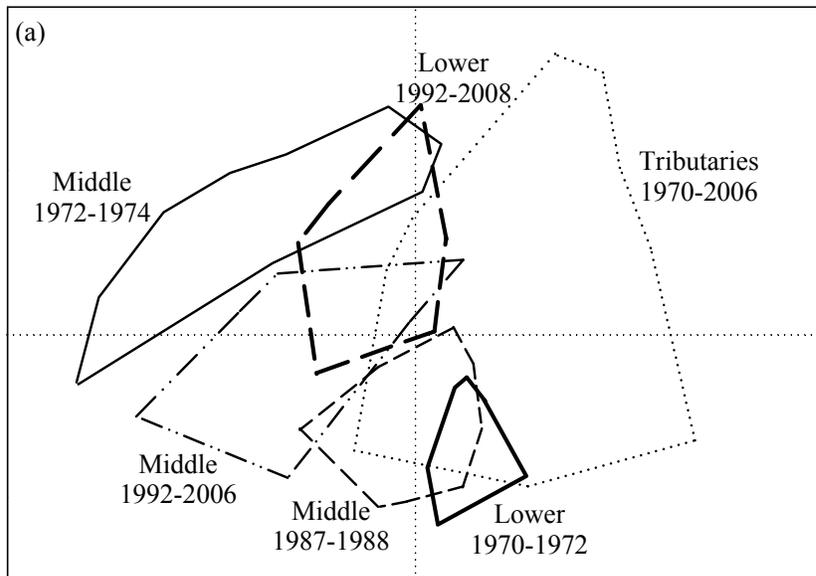


Figure 4. Spatiotemporal similarity among Trinity River fish assemblage collections taken from the middle (Dallas-Fort Worth to Lake Livingston) and lower (downstream of Lake Livingston) Trinity River between 1970 and 2008, illustrated as (a) multi-dimensional scaling plot with polygons denoting sampling scores among collections and (b) dendrogram for hierarchical clustering of six collections. Relative abundance data were fourth root transformed and used to construct Bray-Curtis similarity matrices in Primer 6.1.6.