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HISTORICAL CHANGES IN FISH ASSEMBLAGE COMPOSITION FOLLOWING WATER QUALITY IMPROVEMENT IN THE MAINSTEM TRINITY RIVER OF TEXAS

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ABSTRACT

The Clean Water Act of 1972 is credited with improving water quality across the USA, although few long-term studies tracking hydrologic, chemical, and biological responses to cleanup efforts exist. The Trinity River of Texas was plagued by poor water quality for more than a century before passage of legislation to reduce point source pollution from the Dallas–Fort Worth (DFW) Metroplex. We tracked changes in components of flow regime; concentrations of ammonia, nitrate, phosphorus, and biochemical oxygen demand (BOD); and fish assemblage composition in three mainstem reaches during a 40-year period (1968–2008) following implementation of a large-scale cleanup initiative. Results suggest little change in flow regime components such as magnitude, timing, and rate of change among the three reaches during 1968–2008. Concentrations of water quality parameters declined through time and with greater distance from DFW, including the lowest concentrations in the reach downstream of a mainstem reservoir (Lake Livingston). Fish assemblage composition shifts correlated with attenuated nutrient and BOD concentrations, and species richness generally increased among all reaches. Native and intolerant fishes consistently increased through time among all three reaches, although lentic and non-native species also increased downstream of Lake Livingston. Our findings suggest a revitalization of the Trinity River fish assemblage associated with reduced nutrient pollution in DFW (even among distant reaches) and also illustrate potential confounding factors such as stream impoundment and continued nutrient deposition that likely preclude complete recovery. Copyright © 2014 John Wiley & Sons, Ltd.

KEY WORDS: flow regime; water quality; nutrient pollution; fish assemblage

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INTRODUCTION

Streams of North America have endured historical periods of degradation caused by excessive nutrient loading (i.e. nitrogen and phosphorus). Examples of excessive nitrogen and phosphorus loadings include the historical elimination of aquatic plants in the Potomac River near Washington, DC (Carter and Rybicki, 1986), and the complete depletion of dissolved oxygen and consequently salmon runs in reaches of the Willamette River of Oregon (Huff and Klingeman, 1976). Perhaps the most widely appreciated example is the Cuyahoga River of Ohio, which caught fire for the 13th time in 1969 and stimulated national concern for the condition of US waters. This eventually led to the development of the Federal Water Pollution Control Act Amendments of 1972, also known as the Clean Water Act (CWA; Knopman and Smith, 1993). The CWA of 1972 was specifically designed 'to restore and maintain the physical, chemical, and biological integrity of the Nation's waters' through regulation of discharges entering navigable streams (see review by Carey and Migliaccio, 2009). Since the passage of the CWA of 1972, aquatic plants have returned to the Potomac River, and the Willamette River was once considered the largest river in the USA with restored water quality (Carter and Rybicki, 1986; Hughes and Gammon, 1987). However, despite these examples, the success of the CWA has been difficult to evaluate because of the lack of long-term observational studies combining hydrology, water quality, and biology (Wolman, 1971).

The Trinity River of north-central Texas has a long history of nutrient contamination stemming from the rapidly growing and dense human population in the Dallas–Fort Worth (DFW) metropolitan area (Figure 1). As early as the 1880s, conditions in the stream near DFW were described as unfavourable to fish life because of sewage discharge and the associated effects on aquatic communities (Jordan and Gilbert, 1886). By 1925, the stream was labelled a 'mythological river of death' by the Texas Department of Health because of fish kills, obnoxious odours, and generally poor water quality (Land *et al*., 1998). Consequently, attempts to treat wastewater before discharge into the Trinity River were initiated during the 1920s and 1930s but proved to be unsuccessful in terms of notably improving water quality in the river (Dickson *et al*., 1989). In 1970, the National

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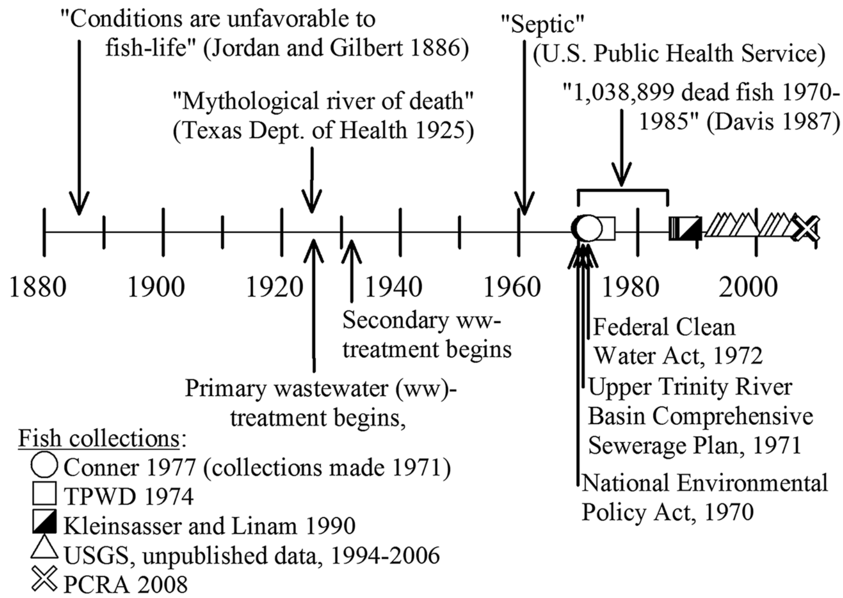

Figure 1. Timeline of Trinity River of Texas water quality and fish sampling. Water quality notes are after Land *et al.* (1998), and fish collection data were compiled from published literature, state, federal, and private sources

Environmental Policy Act was implemented, followed by the Upper Trinity River Basin Comprehensive Sewerage plan in 1971 and the CWA of 1972. All legislation targeted improved water quality in the Trinity River downstream of DFW and aimed to produce notably cleaner water by the mid-1980s through regulation of effluent discharges within DFW.

Despite the passage of legislation in 1972, 13 major fish kills resulted in the loss of more than one million fish in the Trinity River downstream from DFW during 1970–1985 (Davis, 1987). These fish kills prompted detailed investigations by the Texas Natural Resource Conservation Commission (now the Texas Commission on Environmental Quality, TCEQ), Texas Parks and Wildlife Department, and US Environmental Protection Agency (Davis, 1997). Investigations revealed that flow pulses known as 'black rises' originated in DFW and caused elevated biochemical oxygen demand (BOD), dissolved oxygen depletion, and fish kills in downstream reaches (Mirochna, 1988; Davis, 1997). Land *et al.* (1998) suggested that improvements to water quality associated with legislation passed in the early 1970s allowed for improvement in the Trinity River fish assemblage immediately downstream of DFW, but resuspension of residual benthic organic material during black rises caused fish kills of unprecedented magnitude. This conclusion was supported by local improvements in the fish assemblage immediately downstream of DFW during 1972–1993 (Land *et al.*, 1998). However, little additional attention has been devoted to the long-term patterns in nutrient loads and BOD throughout the Trinity River. This is surprising given major fish kills occurred far downstream of DFW, including the Trinity River between Trinidad, Texas, and Lake Livingston (>200 river km downstream; Davis, 1997). Retrospective analysis of hydrology, water quality, and fish assemblage change is necessary to evaluate whether mitigation initiatives implemented at the scale of DFW produced responses throughout the Trinity River mainstem. Furthermore, the extensive attention devoted to the Trinity River during the 40 years following passage of the CWA allows for addressing a critical science need related to assessing long-term trends in hydrology, water quality, and biology in a specific river where a large-scale control initiative has been implemented (Wolman, 1971).

The goal of this study was to assess long-term patterns in hydrological, chemical, and biological elements of the Trinity River following implementation of cleanup initiatives in DFW. In particular, our objectives were as follows: (i) to document changes in ecologically relevant components of flow regime during a 40-year period (1968–2008); (ii) to assess patterns in concentrations of water quality parameters including ammonia, nitrate, phosphorus, and BOD during the same period; and (iii) to evaluate long-term responses in fish assemblage composition and species richness to changes in hydrology and water quality. Whereas our approach is specific to the Trinity River of Texas, poorly treated wastewater discharge entering streams is a global problem (Eklov *et al.*, 1998; Siligato and Bohmer, 2001; Dudgeon *et al.*, 2006). Improved ability to predict long-term and broad-scale fish assemblage responses to mitigation implemented at finer spatial scales will ultimately aid in guiding future biodiversity and ecosystem restoration initiatives (Bohn and Kershner, 2002).

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STUDY AREA

The Trinity River originates as four forks (Clear, East, Elm, and West) in north-east Texas and drains ~46 500 km^2 as it flows south-east to Trinity and Galveston bays and then into the Gulf of Mexico. The upper portions of the Trinity River are encompassed by the rapidly growing DFW Metroplex characterized by high population densities (i.e. 98 people km^{-2}; Dahm *et al*., 2005). Twenty-one major reservoirs now exist in the Trinity River basin, including the mainstem impoundment Lake Livingston, which is the primary water supplier for the Greater Houston area (Figure 2). For the purposes of this study, the Trinity River was divided into three reaches: (1) the upper Trinity River mainstem beginning just downstream of DFW and extending to Oakwood, Texas (229 stream km in length); (2) the middle Trinity River between Oakwood, Texas, and the upper reaches of Lake Livingston (175 stream km); and (3) the lower Trinity River between the dam at Lake Livingston and the Highway 162 crossing near Moss Hill, Texas (80 stream km) where the influence of marine fishes on assemblage composition increases (Conner, 1977). The boundary between Reaches 1 and 2 represents the border of the Texan (Reach 1) and Austroriparian (Reach 2) biologic provinces, which is associated with changes in fish assemblage composition (Hubbs, 1957). Connectivity between Reaches 2 and 3 was severed by the completion of Lake Livingston in 1968 (Wellmeyer *et al*., 2005).

METHODS

Flow regime

We obtained daily streamflow data from US Geological Survey (USGS) gauges during 1968–2008 to quantify changes in flow regime that might have contributed to changes in water quality parameter concentrations (e.g. because of increased human appropriation of water; Postel, 2003) or fish assemblage composition (e.g. because of human suppression of flood pulses; Perkin and Bonner, 2011). Streamflow gauges were chosen on the basis of distribution among our three reaches and availability of historical data (i.e. at least back

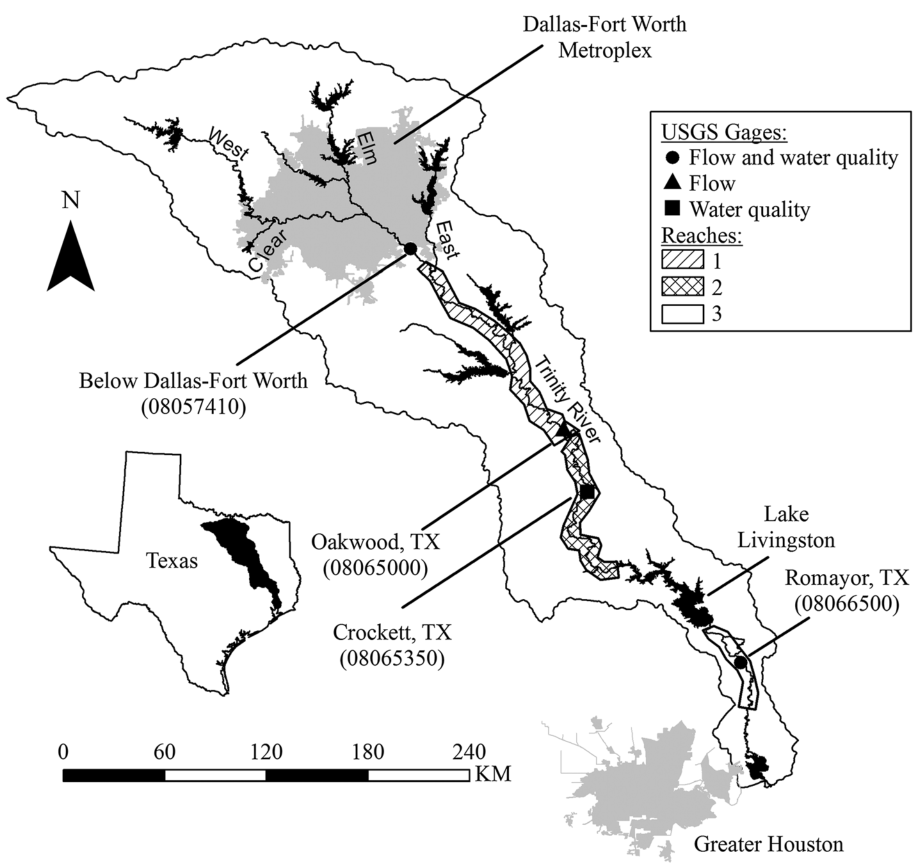

Figure 2. Trinity River of Texas illustrating reaches included in the study (polygons) and location of US Geological Survey (USGS) gauging stations (station ID number) from which flow and water quality data were obtained. Reach 1 is from Dallas–Fort Worth to Oakwood, Texas (USGS gauge 08057410); Reach 2 is from Oakwood, Texas, to Lake Livingston (USGS gauges 0806500 and 08065350); and Reach 3 is from Lake Livingston to Highway 162 bridge near Moss Hill, Texas (USGS gauge 08066500)

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to 1968) with limited breaks in continuous data logging during the targeted period of record (i.e. 1968–2008). We first assessed changes in annual streamflow magnitude because of the strong relationship between this parameter and fish assemblage composition (Poff and Zimmerman, 2010). We used a generalized additive model (GAM) approach (Zuur *et al*., 2009) to assess changes in mean annual streamflow through time by plotting mean annual discharge (dependent variable) against time (independent variable) using the [mgcv] package with generalized cross-validation smoothing parameters in Program R version 2.15.2 (Wood, 2004). This approach is robust to assumptions regarding independence of observations, temporal and serial autocorrelations, and standardization of variances (Zuur *et al*., 2009). GAMs are a class of spline function capable of capturing non-linear relationships among variables, in which the significance of the smoothing function is used to evaluate change in the response variable (here, a significant smoothing function indicates significant change in flow magnitude through time). In addition to this analysis, we quantified five major characteristics of streamflow regime that are commonly related to stream fish assemblages: magnitude, frequency, duration, timing, and rate of change (Poff *et al*., 1997). To facilitate comparison with studies in other systems (Mathews and Richter, 2007; Pracheil *et al*., 2009), we quantified flow regime attributes using Indicators of Hydrologic Alteration (IHA), version 7.0.3 (Richter *et al*., 1996), to assess significance of regression slopes ($b_1 \neq 0$, $p < 0.05$) for the period 1968–2008. We selected one to three parameters that best represented each of the five flow regime characteristics according to Olden and Poff (2003). We estimated streamflow magnitude using the IHA parameter for base flow index (i.e. low-flow conditions) as well as mean monthly flow during the month of December (i.e. average-flow conditions), streamflow frequency using number of low-flow and high-flow pulses, streamflow duration using number of zero-flow days as well as low-flow and high-flow pulse durations, streamflow timing using the Julian date of minimum flow, and streamflow rate of change using number of streamflow reversals (see Olden and Poff, 2003, for detailed descriptions of IHA parameters). Mean monthly discharge for the month of December was chosen on the basis of previous analyses in the region, which included characterizing the flow regime in the runoff-dominated Trinity River (Poff, 1996; Olden and Poff, 2003).

Water quality

We assessed spatiotemporal variability in concentrations of water quality parameters closely tied to fish assemblage composition and fish kills in the Trinity River (Davis, 1987, 1997). These parameters included ammonia (NH_3-N), nitrate (NO_3-N), total phosphorus, and BOD. We obtained water quality data from USGS gauging stations in each section of our study and evaluated changes in monthly concentrations through time during 1968–2008 using the same GAM approach as for mean annual discharge. Here, water quality concentration ($\mathrm{mg\,l^{-1}}$) data (dependent variable) were plotted against time (independent variable). We conducted GAM regression analysis for each water quality parameter in each of the three stream reaches to assess longitudinal changes in concentrations associated with increased distance from DFW. For both flow regime and water quality GAM approaches, we present the residual deviance adjusted correlation coefficient and the associated percentage of deviance explained by the models, as well as the estimated degrees of freedom, *F*-values, and *p*-values associated with smoothing terms (Zuur *et al*., 2009). All GAM analyses were conducted using the [mgcv] package in Program R version 2.15.2 (Wood, 2004). Finally, we compared long-term trends in water quality parameter concentrations with standards proposed by Nemerow (1974) and TCEQ (2012).

Fish assemblage composition

We obtained historical fish assemblage data from a diversity of agencies that sampled the mainstem Trinity River during 1968–2008. In each case, historical samples were selected only if the primary goal of the study included assessment of the entire fish assemblage, and studies targeting particular species or subsets of the assemblage were excluded. Although a comprehensive survey of the Trinity River basin was completed in 1957, 36 of the 39 samples were taken from tributary streams or impoundments, and the remaining collections covered only a small extent of the mainstem (TGFC, 1957). Consequently, available historical fish assemblage surveys included those by the Texas Parks and Wildlife Department (1974), Conner (1977), Kleinsasser and Linam (1990), the USGS (unpublished data 1994–2006), and Paul C. Rizzo Associates, Inc. (2008). These collections were obtained using a variety of gear types deployed across multiple years but generally involved the combined use of gill nets, seining, and electrofishing during multiple seasons (Table I). For these studies, the term ‘collection’ is used to describe the occurrence of any fishes documented using all gear types across all seasons within a year at each site. Thus, for the purposes of our study, a fish sample represents all fish species encountered at a site during a year. This approach is useful for producing a fish collection database that is robust with regard to bias caused by sampling gear or season. Because all studies utilized gears useful for collecting a range of size classes from a diversity of habitats and were likely to encounter a variety of species (but with varying relative abundances), we only used presence/absence data for statistical analyses of spatiotemporal variability in assemblage composition (Gido *et al*., 2010). Based on the timing and availability of samples, we divided collections into three periods for each

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Table I. Sources of data for fish community data, sampling gear, frequency of collection, and reaches and periods for historical fish community data in the mainstem Trinity River of Texas

Collector	Collection gears	Collection frequency	Reaches	Timing
Texas Parks and Wildlife Department (1974)	Gill net, electrofishing	Seasonally for two years	1, 2	Period I
John Van Conner (1977)	Gill net, seines, rotenone	Seasonally for multiple years	3	Period I
Kleinsasser and Linam (1990)	Gill net, seines, electrofishing	Seasonally for two years	1, 2	Period II
USGS (unpublished, 1994–2006)	Gill net, seines, electrofishing	May–Sept for multiple years	1, 2, 3	Period III
PCRA (2008)	Gill net, seines, electrofishing	Seasonally for two years	3	Period III

study reach: Period I (1971–1974), Period II (1987–1988), and Period III (1994–2008). These study periods corresponded with temporal patterns in water quality (see water quality results) and produced 62 collections for Reach 1 (11, 35, and 16 by period, chronologically), 39 samples for Reach 2 (10, 24, and 5), and 42 samples for Reach 3 (20, 0, and 22).

To document changes in the fish assemblage through time, we assessed patterns in assemblage composition and species richness. For assemblage composition, we constructed Bray–Curtis distance matrices (Bray and Curtis, 1957) based on presence/absence data and used permutational multivariate analysis of variance (PMANOVA, $N = 10\,000$ iterations) on the distance matrices to test for differences in assemblage composition among periods within each reach. We used non-metric multi-dimensional scaling (NMDS) plots with minimum convex polygons based on distance matrices to illustrate differences in assemblage composition among periods using the [Vegan] Package in Program R (Oksanen, 2009). Additionally, we included environmental parameter vectors for all water quality parameters as well as discharge (cubic metres per second) recorded for the day of the fish collection to illustrate environmental correlates for clusters identified in the NMDS plots. Before plotting, we used the [envfit] function in Program R (Oksanen, 2009) to estimate correlation coefficients and significance of individual environmental parameters using permutation tests and only plotted environmental variables that explained significant levels of variation in the assemblage ordination. This approach allowed for assessing relationships between community structure and the measured environmental variables. For species richness, we used a rarefaction approach to estimate species accumulation curves as a function of the number of samples taken during each period and from each reach (Gotelli and Graves, 1996). Within a reach, we used the program EcoSim (Gotelli and Entsminger, 2000) to conduct Monte Carlo simulations ($N = 10\,000$ iterations) in which collections were added in randomized order and used to estimate mean richness and 95% confidence intervals by period. We then made comparisons among periods using a standardized number of collections to avoid sampling effort bias in our assessment of temporal changes in fish species richness (Gido *et al*., 2010). Statistical analyses for PMANOVA and NMDS were conducted in Program R version 2.15.2.

Initial results for assemblage composition and species richness suggested shifts in the assemblage occurred among periods, and these shifts were associated with a general increase in species richness. We then tested for changes in species riches through time for guilds related to native status, habitat associations, and tolerance level. For these guilds, we classified species as native or non-native using a guide for the freshwater fishes of Texas (Hubbs *et al*., 2008), lentic (lacustrine) or lotic (stream adapted) using trait data compiled by Frimpong and Angermeier (2009), and tolerant or intolerant using a regionalized index of biotic integrity (Linam *et al*., 2002). Although alternative methods for classifying fishes into guilds exist, we used these data sources because they are widely available and represent syntheses of existing literature. We then used a random subset resampling approach similar to that of Gido *et al*. (2010) to avoid bias caused by uneven sampling effort among periods as well as lack of independence among collections taken during the same period. Specifically, we randomly sampled 80% of the collections for the reach and period with the minimum number of reported collections (i.e. Reach 2, Period III = 5 collections) from all reaches and periods (i.e. random subsamples contained four collections). We calculated mean species richness for each category of the three guilds (i.e. a total of six analyses) among the four subsampled collections and repeated the subsampling 1000 times. We then calculated mean and 95% confidence intervals and made comparison among periods within each reach and guild, assuming differences were significant if 95% confidence intervals did not overlap (Gido *et al*., 2010). Randomized subsampling and permutation tests were conducted in Program R version 2.15.2.

RESULTS

Flow regime

The Trinity River flow regime changed little during 1968–2008 (Figure 3). Modelled magnitude of streamflow measured as

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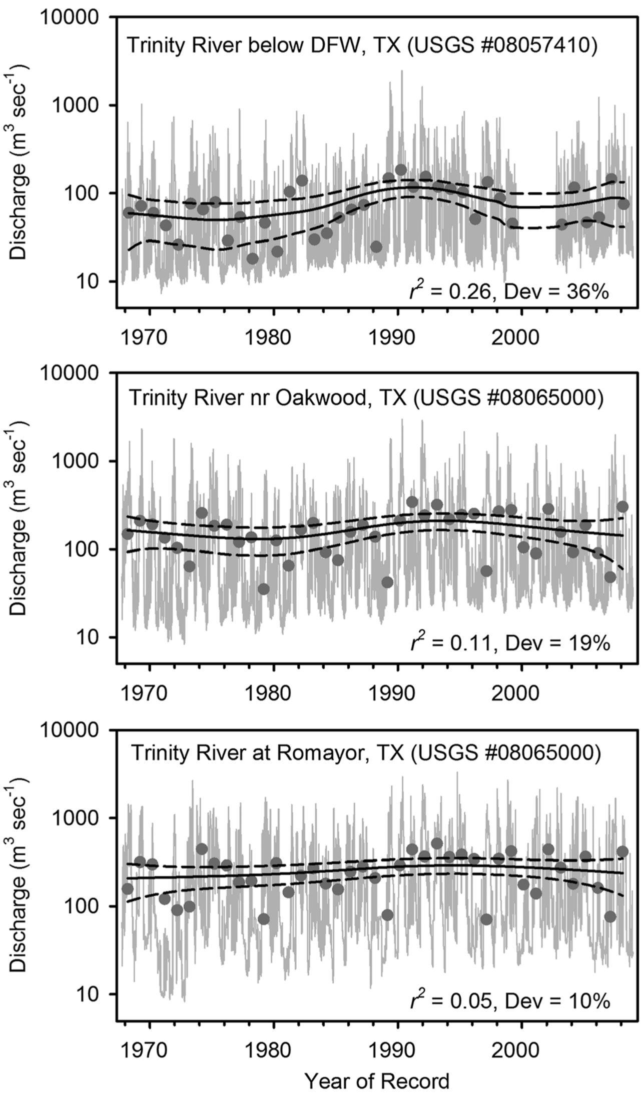

Figure 3. Hydrographs for the Trinity River of Texas below Dallas–Fort Worth (DFW), near Oakwood, and at Romayor, Texas, USGS gauging stations. Grey lines are daily flow data, points represent mean annual discharge values, and lines represent generalized additive models (solid: mean; dashed: 95% confidence interval) for time (independent variable) versus mean annual discharge (dependent variable). Correlation coefficients and percentage of deviance explained are given for each model

mean annual discharge was characterized by a significant GAM smoothing function in Reach 1 (r^2 = 0.26, deviation explained = 36%, estimated df = 5.1, F = 2.55, p = 0.04), but not in Reach 2 (r^2 = 0.11, deviation = 18.3%, df = 3.5, F = 1.44, p = 0.24) or Reach 3 (r^2 = 0.05, deviation = 10.4%, df = 2.4, F = 1.11, p = 0.36). Greater mean annual flow in Reach 1 was related to increases in high-flow pulses (IHA, slope = 0.154, p = 0.005) and to decreases in low-flow frequency (slope = −0.468, $p < 0.001$) and duration (slope = −0.223, p = 0.001). This change occurred during the early 1990s based on visual inspection of the hydrograph and associated GAM smoothing function. Among Reaches 2 and 3, only low-flow pulse duration declined in Reach 2 (slope = −0.229, p = 0.001); no other flow regime components changed through time (Table II).

Water quality

Water quality parameters varied spatially and temporally, with a general pattern of decline through time and with longitudinal distance from DFW (Figure 4). All GAMs were characterized by significant smoothing functions with correlation coefficient values ranging 0.07 to 0.62 (Table III). In Reach 1, concentrations of ammonia averaged 8 mg l^{-1} in the early 1970s and exceeded the TCEQ (2012) limit of 3 mg l^{-1} until 1983. Similarly, BOD concentrations exceeded the general upper threshold of unpolluted streams (8 mg l^{-1}; Nemerow, 1974) until 1986, whereas phosphorus concentrations exceeded the TCEQ limit of 0.5 mg l^{-1} throughout the period of record. Modelled nitrate did not exceed the TCEQ limit of 10 mg l^{-1} during the period of record, although sampling points occasionally exceeded this value during 1968–1973 and then again during 1988–2008. In Reach 2, modelled ammonia and nitrate did not exceed TCEQ (2012) limits during the period of record. BOD spiked in 1973 and 1978 and in both cases exceeded 8 mg l^{-1} for a relatively short period before falling well below this value after 1985. Phosphorus declined steadily through time, and modelled values were less than 0.5 mg l^{-1} after 1997. In Reach 3, modelled values for nutrients and BOD did not exceed any limits during the period of record, although sampled concentrations of BOD occasionally exceeded 8 mg l^{-1} during the late 1970s and late 1980s, as did phosphorus during the early 1970s.

Fish assemblage composition

Significant shifts in fish assemblage composition occurred in Reach 1 (PMANOVA, pseudo-$F_{2,59}$ = 18.07, $p < 0.01$), Reach 2 ($F_{2,36}$ = 11.44, $p < 0.01$), and Reach 3 ($F_{1,40}$ = 27.44, $p < 0.01$). NMDS plots illustrated clustering among collections by period with two-dimensional stress values of 0.20, 0.18, and 0.16 among Reaches 1, 2, and 3, respectively (Figure 5). In Reach 1, environmental correlation coefficients were significant for ammonia (r^2 = 0.65, $p < 0.01$), BOD (r^2 = 0.53, $p < 0.01$), phosphorus (r^2 = 0.49, $p < 0.01$), and nitrate (r^2 = 0.39, $p < 0.01$), and Period I collections were correlated with higher concentrations of ammonia, BOD, and phosphorus. In Reach 2, environmental correlation coefficients were significant for ammonia (r^2 = 0.61, $p < 0.01$), BOD (r^2 = 0.70, $p < 0.01$), phosphorus (r^2 = 0.72, $p < 0.01$), and nitrate (r^2 = 0.44, $p < 0.01$), all of which were positively correlated with Period I collections. In Reach 3,

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Table II. Flow regime components for three reaches of the Trinity River during 1968–2008

	Reach 1[a]		Reach 2[b]		Reach 3[c]	
	IHA parameter		IHA parameter		IHA parameter	
Flow characteristic	Slope	Significance	Slope	Significance	Slope	Significance
Magnitude						
Mean December flow	1.272	0.500	1.146	0.500	1.721	0.50
Base flow index	−0.001	0.500	0.003	0.050	0.003	0.25
Frequency						
Low-flow pulses	**−0.468**	**0.001**	−0.097	0.250	−0.039	0.25
High-flow pulses	**0.154**	**0.005**	0.001	0.500	0.010	0.50
Duration						
Zero-flow days	0.000	0.500	0.000	0.500	0.000	0.50
Low-flow pulse duration	**−0.223**	**0.001**	**−0.229**	**0.001**	0.497	0.25
High-flow pulse duration	0.014	0.500	0.091	0.500	0.063	0.50
Timing						
Julian date of minimum	0.929	0.500	0.319	0.500	0.814	0.25
Rate of change						
Reversals	0.132	0.500	−0.107	0.500	−0.087	0.50

Streamflow data were downloaded from US Geological Survey (USGS) gauges and analysed using linear regression in Indicators of Hydrologic Alteration (IHA; Richter *et al*., 1996). Bolded parameters indicate significant changes in slope through time ($b_1 \neq 0$; $p < 0.05$). Flow regime characteristics and associated IHA parameters were selected following Olden and Poff (2003).
[a]USGS gauge 08057410.
[b]USGS gauge 08065000.
[c]USGS gauge 08066500.

environmental correlation coefficients were significant for ammonia ($r^2 = 0.85$, $p < 0.01$), phosphorus ($r^2 = 0.17$, $p = 0.02$), and daily streamflow ($r^2 = 0.23$, $p < 0.01$), and ammonia and phosphorus were positively correlated with Period I. Euclidean distances (i.e. scale of x-axis and y-axis) among sampling periods were greatest in Reach 1, followed by Reaches 2 and 3. The average number of fish species per collection and rarefied richness estimates generally increased through time in all three reaches (Table IV). In Reach 1, the number of species detected within 10 samples was 13.7, 28.9, and 27.8 during Periods I, II, and III, respectively. Rarefied richness values were 23.3, 26.1, and 21.7, among periods in Reach 2 (measured at four samples) and 43.1 and 50.1 among periods in Reach 3 (measured at 19 samples).

Mean richness for species belonging to various guilds differed among reaches and periods. Native and intolerant species richness consistently increased through time in all three reaches, and richness values were generally greater in downstream reaches (Figure 6). Non-native species richness was low and never exceeded an average of two species, although there was an increase in Reach 3 between Periods I and III. Tolerant and lentic species increased in Reaches 1 and 3, although this pattern was not evident in Reach 2. Lotic species increased initially from Periods I to II in Reaches 1 and 2, followed by intermediate values during Period III, whereas lotic species richness declined through time in Reach 3.

DISCUSSION

Our findings empirically show reduced nutrient concentrations and increased fish species richness occurring during the 40-year period following a large-scale cleanup initiative in the mainstem Trinity River of Texas downstream of DFW. These patterns were not explained by dilution of concentrations caused by increased streamflow magnitudes and were not attributed to increased fish richness caused by introductions of non-native species. Instead, our findings suggest that increases in native and intolerant species richness occurred as the fish assemblage shifted through time coincident with attenuated concentrations of nutrients and BOD. These results support previous conclusions regarding increased control over contaminants originating in DFW following legislation (Mirochna, 1988; Van Metre and Callender, 1996), which ultimately allowed for revitalization of aquatic assemblages in close proximity to DFW (Davis, 1997; Land *et al*., 1998). Our findings expand upon these previous conclusions by documenting declining nutrient and BOD concentrations in the mainstem Trinity River far downstream (up to 400 km) of DFW, which coincided with fish community changes characterized by increased species richness through time and with distance from DFW.

Spatial and temporal changes in the Trinity River mainstem fish assemblage were associated with improved water chemistry. Longitudinal improvement in fish assemblage composition with increased distance from sewage

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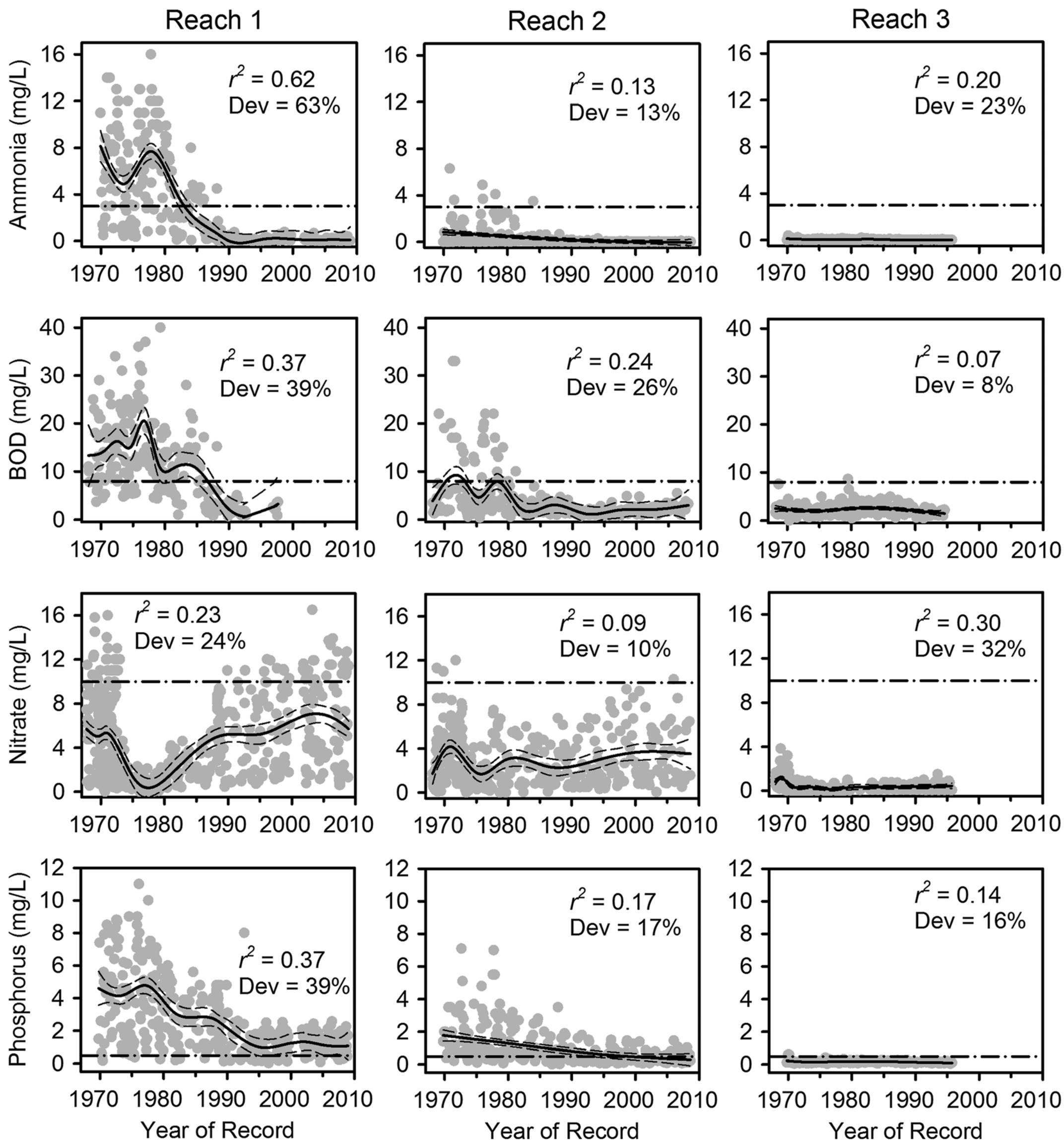

Figure 4. Generalized additive models for concentrations (mg l^{-1}) of ammonia nitrogen, biochemical oxygen demand (BOD), nitrate nitrogen, and phosphorus measured through time in three reaches of the mainstem Trinity River of Texas. Fitted models (solid lines) and 95% confidence intervals (dashed lines) are plotted on observations (gray points), and correlation coefficients and percentage of deviance explained are given. Dash-dotted lines represent water quality standards proposed by the Texas Commission on Environmental Quality (ammonia, nitrate, and phosphorus) and the general upper threshold for BOD concentrations in unpolluted streams according to Nemerow (1974)

outflows is well documented and is generally related to the effects of dilution by non-polluted water sources such as tributary streams (Katz and Gaufin, 1953; Hughes and Gammon, 1987; Ryon, 2011). In the case of the Trinity River, dilution caused by tributary inflows downstream of DFW was a likely driver of downstream reduction in nutrient and BOD concentrations, but these tributaries may have also contributed to increases in species richness. Relative to the mainstem Trinity River, tributary streams generally contained lower concentrations of nutrients (Davis, 1997) and higher levels of fish species richness prior to cleanup initiatives. The most comprehensive survey of the Trinity River drainage prior to the early 1970s included 36 tributary samples and three mainstem samples (TGFC, 1957). Based on these data, tributary sampling sites contained twice as many species as the mainstem sites, and each of the mainstem sites occurred within the three reaches we defined for the current study. In Reach 1, the assemblage consisted of six native species, of which four were tolerant (two intolerant) and five lentic (one lotic). In Reach 2, the assemblage consisted of five natives, of which three were tolerant (two intolerant) and four were lentic (one lotic). In Reach 3, the assemblage consisted

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Table III. Generalized additive model results for water quality parameters in three reaches of the Trinity River measured for the period 1968–2008 (except for Reach 3, for which data ended in 1995)

Water quality parameter	Adjusted r^2	Deviance explained	Estimated *df*	*F*-value	*p*-value
Reach 1[a]					
Ammonia	0.62	63.0	8.6	77.08	<0.001
Biochemical oxygen demand	0.37	39.2	8.6	15.47	<0.001
Nitrate	0.23	23.7	8.8	21.40	<0.001
Phosphorus	0.37	38.6	7.9	29.11	<0.001
Reach 2[b]					
Ammonia	0.13	13.3	2.0	16.40	<0.001
Biochemical oxygen demand	0.24	25.9	8.9	10.16	<0.001
Nitrate	0.09	10.3	7.7	4.98	<0.001
Phosphorus	0.17	17.2	1.9	22.18	<0.001
Reach 3[c]					
Ammonia	0.20	23.4	7.8	6.73	<0.001
Biochemical oxygen demand	0.07	8.4	4.0	3.35	0.007
Nitrate	0.3	31.7	8.8	14.86	<0.001
Phosphorus	0.14	16.0	6.3	5.28	<0.001

Models describe relationships between time (independent variable) and parameter concentration ($mg\,l^{-1}$; dependent variable). Results include residual deviance adjusted correlation coefficient, percentage of deviance explained, estimated degrees of freedom (*df*) used in smoothing, and the associated estimated *F*-value and *p*-value.
[a]USGS gauge 08057410.
[b]USGS gauge 08065350.
[c]USGS gauge 08066500.

of four natives, of which three were tolerant (one intolerant) and two were lentic (two lotic). Comparison of these values with the modelled richness values given in Figure 6 suggests the assemblage during the 1950s, prior to cleanup initiatives, was similar to the assemblage during the early 1970s in terms of guild composition. These data also illustrate a general increase in species diversity in the mainstem Trinity River through time, suggesting refuge populations that served as sources for recolonization of the mainstem must have existed. Refuge populations are known to enhance temperate fish assemblage recovery following press (i.e. constant and long-term) disturbances (Detenbeck *et al*., 1992), particularly among stream fishes capable of inhabiting both mainstems and adjacent tributary streams (Sedell *et al*., 1990). Given established relationships between mainstem fishes and tributary streams (Pracheil *et al*., 2009, 2013), we hypothesize that tributaries to the mainstem Trinity River contributed to longitudinal dilution of nutrient concentrations and spatiotemporal increases in fish species richness.

Studies tracking fish assemblage response to water quality parameters have demonstrated increased fish diversity associated with attenuated nutrient concentrations and improved oxygen regimes in streams (Eklov *et al*., 1998; Porter and Janz, 2003). Nitrogen and phosphorus are generally limiting in freshwater ecosystems but can increase to harmful levels because of anthropogenic contributions such as sewage outflows (Allan and Flecker, 1993; Wang *et al*., 2007). A direct effect of ammonia on freshwater fishes includes toxicity when concentrations exceed $2.79\,mg\,l^{-1}$ (Randall and Tsui, 2002). In Reach 1 of our study, ammonia levels were likely toxic to fishes prior to at least 1985 on the basis of concentrations exceeding $3\,mg\,l^{-1}$ (Land *et al*., 1998). Indirect effects of nitrogen (in the form of ammonia and nitrate) and phosphorus in aquatic ecosystems include eutrophication, algal blooms, spikes in microbial activity that lead to anoxic conditions, and increases in BOD (Dodds, 2006; Heisler *et al*., 2008). Phosphorus levels averaged $>3\,mg\,l^{-1}$ in Reach 1 until 1987, whereas BOD concentrations were elevated in Reach 1 until 1987 and declined precipitously thereafter. Based on the timing of declines in ammonia and phosphorus in the Trinity River mainstem, elevated nutrient concentrations were likely primary contributors to elevated BOD and consequently fish kills (Volkmar and Dahlgren, 2006; Wang *et al*., 2007). In fact, the decline of BOD concentrations below $8\,mg\,l^{-1}$ in 1987 corresponded with the last reported fish kill in the Trinity River and the onset of aquatic invertebrate revitalization (Davis, 1997). This suggests control of nutrient pollution sufficient to avoid major fish kills was achieved by the 1980s but does not indicate complete recovery of the system. In fact, modelled phosphorus concentrations in Reach 1 exceeded the acceptable threshold of $0.5\,mg\,l^{-1}$ throughout the period of observation, and concentrations in Reach 2 did not decline to acceptable levels until approximately 2007. Similarly, although nitrate concentrations declined from 1968 until 1978 in Reaches 1 and 2, levels increased again after 1978. Other pollutants including endocrine disruptors, surfactants, carbohydrates, and uric acids are still discharged

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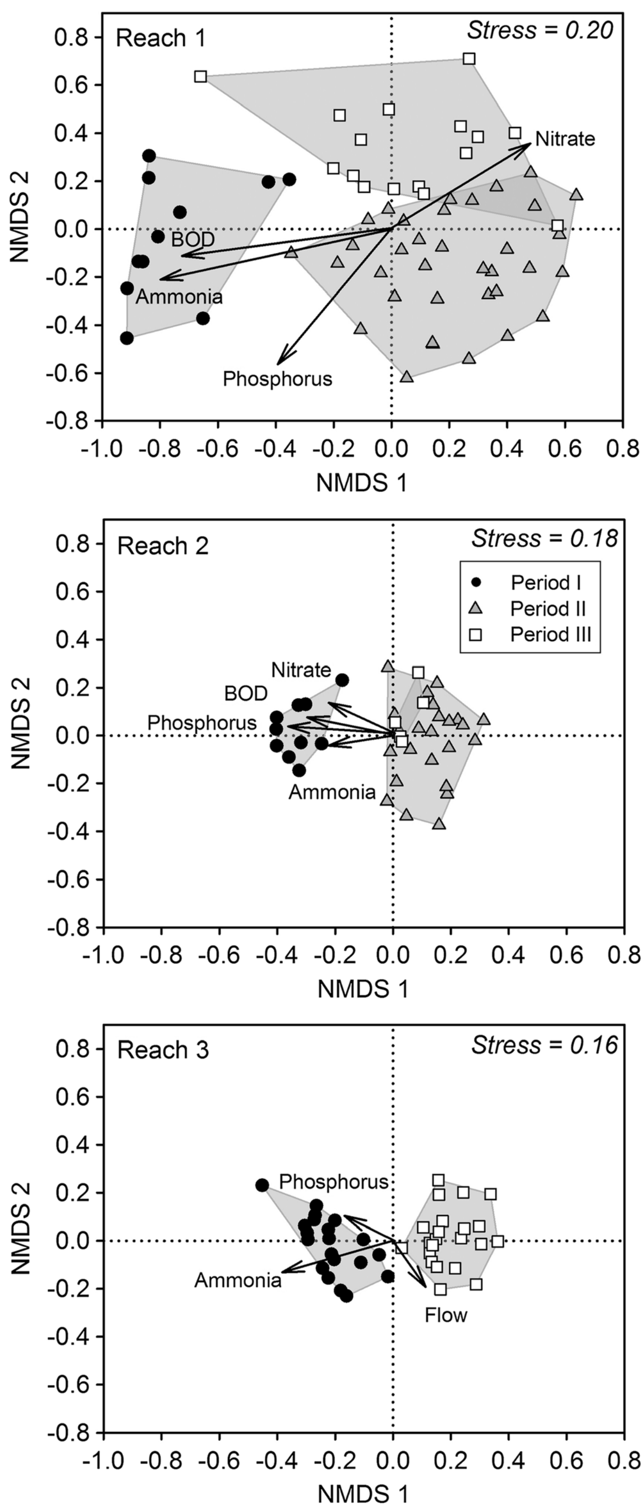

Figure 5. Non-metric multi-dimensional scaling plots and associated environmental vectors for Trinity River of Texas fish assemblage samples taken from three reaches during periods I (1971–1974, black circles), II (1987–1988, grey triangles), and III (1994–2008, white boxes). Only environmental variables with significant correlation coefficients are shown, and polygons envelope collections by period

into the Trinity River system in DFW (Hung *et al*., 2005; Atkinson *et al*., 2009). Non-point sources of nutrients exist throughout the watershed, including increasing predominance of agricultural landscapes in areas downstream of DFW (Chen *et al*., 2000). These continued threats to aquatic biota exemplify the complexity of landscape alterations that plague aquatic environments (Dudgeon *et al*., 2006) and illustrate the historically overriding effect of pollution from DFW.

Spatiotemporal patterns in water quality and fish assemblage composition were related to the effects of Lake Livingston. Strong shifts in nutrient and BOD concentrations downstream of Lake Livingston suggest the reservoir acted as a nutrient sink. Reservoirs are known to interrupt nutrient spiralling in riverine systems through the framework of the serial discontinuity concept (Ward and Stanford, 1983). Under this framework, the creation of a large lentic reservoir upstream of the dam at Lake Livingston acted as a resetting mechanism, so that nutrients from the inflowing Trinity River became assimilated in the lake and outflows trended towards lower nutrient concentrations (Reddy *et al*., 1982; Groeger and Kimmel, 1984). Other pollutants such as heavy metals (lead) and dichlorodiphenyltrichloroethane have settled in the sediments of Lake Livingston since construction in 1969 (Van Metre and Callender, 1996). Interestingly, whereas Lake Livingston acted as a sink for nutrients, the reservoir likely served as a source for some fish populations. Changes in assemblage composition in Reach 3 were related to increases in non-native or lentic species such as goldfish *Carassius auratus*, grass carp *Ctenopharyngodon idella*, common carp *Cyprinus carpio*, white bass *Morone chrysops*, and striped bass *Morone saxatilis*. Impoundments such as Lake Livingston generally facilitate the invasion of such species through aquatic corridors in upstream and downstream directions (Havel *et al*., 2005; Heard *et al*., 2012). This process likely explains the large number of lentic species in Reach 2 immediately upstream of Lake Livingston, especially during Period I when fish kills were common upstream of the impoundment. Downstream of Lake Livingston, the flow regime has been stable through time and is reflective of pre-impoundment conditions according to Wellmeyer *et al*. (2005). Still, we detected a decline of species susceptible to habitat fragmentation and destruction caused by dams, such as stream-obligate ghost shiner *Notropis buchanani*, chub shiner *Notropis potteri*, and suckermouth minnow *Phenacobius mirabilis* (Bestgen and Compton, 2007; Perkin *et al*., 2009).

Our ability to assess the potential for full recovery of the Trinity River mainstem fish assemblage is further confounded by lack of historical data pertaining to the natural state of the assemblage. The earliest known records from the Trinity River describe the fish assemblage as already degraded (Jordan and Gilbert, 1886), and recent discovery of the invasive zebra

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Table IV. Fish species native status (native, N; non-native, I), habitat association (lotic, Lot; lentic, Len), and tolerance level (tolerant, T; intolerant, I), and number of times collected by reach and period from the Trinity River mainstem

Genus and species	Native status[a]	Habitat association[b]	Tolerance level[c]	Reach 1			Reach 2			Reach 3	
				I	II	III	I	II	III	I	III
Polyodon spathula	N	Len	I	0	0	0	0	0	0	0	1
Atractosteus spatula	N	Len	T	1	2	1	0	5	2	0	1
Lepisosteus oculatus	N	Len	T	9	12	10	1	11	5	3	11
Lepisosteus osseus	N	Len	T	10	24	9	2	14	5	3	4
Lepisosteus platostomus[d]	N	Len	T	0	0	0	0	0	1	0	1
Amia calva	N	Len	T	0	0	0	1	0	0	0	1
Anguilla rostrata	N	Lot	I	0	0	0	0	0	0	0	2
Dorosoma cepedianum	N	Len	T	3	16	12	10	11	5	3	16
Dorosoma petenense	N	Len	I	0	5	5	7	10	5	11	21
Campostoma anomalum	N	Lot	I	0	2	0	0	0	0	0	0
Carassius auratus	I	Len	T	0	0	0	0	0	0	0	1
Ctenopharyngodon idella	I	Len	T	0	0	1	0	0	0	0	2
Cyprinella lutrensis	N	Len	T	1	35	15	4	24	5	19	20
Cyprinella venusta	N	Lot	I	0	6	0	0	13	3	19	17
Cyprinus carpio	I	Len	T	10	9	5	9	7	1	0	11
Hybognathus nuchalis	N	Lot	T	0	0	0	0	0	0	3	0
Lythrurus fumeus	N	Lot	I	0	0	0	0	1	0	1	2
Lythrurus umbratilis	N	Lot	I	0	3	0	0	4	0	0	0
Notemigonus crysoleucas	N	Len	T	0	0	0	2	0	0	11	4
Notropis buchanani	N	Lot	I	0	13	0	0	17	0	20	0
Notropis potteri	N	Lot	I	0	0	0	0	0	0	6	0
Notropis sabinae	N	Lot	I	0	0	0	0	0	0	0	8
Notropis shumardi	N	Lot	I	0	0	0	0	5	0	19	10
Notropis stramineus	N	Lot	I	0	0	0	0	0	1	0	2
Notropis texanus	N	Lot	I	0	0	0	0	3	0	2	0
Notropis volucellus	N	Lot	I	0	7	0	0	5	3	1	13
Opsopoeodus emiliae	N	Len	I	0	0	0	1	1	0	9	0
Phenacobius mirabilis	N	Lot	I	0	0	0	0	0	0	10	0
Pimephales vigilax	N	Lot	I	0	35	11	2	24	4	20	18
Carpiodes carpio	N	Len	T	4	0	0	4	4	1	12	2
Ictiobus bubalus	N	Len	I	6	22	14	0	13	3	0	17
Minytrema melanops	N	Len	I	0	0	0	1	0	0	2	0
Moxostoma poecilurum	N	Lot	I	0	0	0	0	0	0	1	3
Ameiurus melas	N	Len	T	0	1	0	2	0	0	0	0
Ameiurus natalis	N	Len	I	0	3	0	3	0	0	0	1
Ictalurus furcatus	N	Lot	I	0	14	10	0	17	5	0	18
Ictalurus punctatus	N	Len	T	5	10	9	4	9	0	16	17
Noturus gyrinus	N	Len	I	0	0	4	0	0	0	0	0
Noturus nocturnus	N	Lot	I	0	2	2	0	7	0	7	0
Pylodictis olivaris	N	Len	I	0	12	14	2	14	5	0	7
Cyprinodon variegatus	N	Len	T	0	0	0	0	0	0	0	1
Fundulus notatus	N	Len	I	0	3	2	2	2	0	2	1
Gambusia affinis	N	Len	T	0	34	4	1	21	2	19	12
Labidesthes sicculus	N	Len	I	0	0	1	3	0	0	3	0
Membras martinica	N	Len	I	0	0	0	0	0	0	1	0
Menidia audens	N	Len	I	0	0	0	0	0	0	11	0
Menidia beryllina	N	Len	I	0	7	1	2	7	2	12	19
Morone chrysops	I	Len	I	5	0	1	7	6	1	2	13
Morone mississippiensis	N	Len	I	0	0	0	3	1	0	1	5
Morone saxatilis	I	Len	I	0	1	1	0	2	0	0	11
Lepomis auritus	I	Len	I	0	1	0	0	0	0	0	0
Lepomis cyanellus	N	Len	T	0	18	9	2	5	1	2	2
Lepomis gulosus	N	Len	T	2	10	6	6	4	0	3	5
Lepomis humilis	N	Len	I	0	19	1	0	5	0	0	8
Lepomis macrochirus	N	Len	T	2	17	8	8	6	1	8	17

(*Continues*)

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Table IV. (Continued)

Genus and species	Native status[a]	Habitat association[b]	Tolerance level[c]	Reach 1			Reach 2			Reach 3	
				I	II	III	I	II	III	I	III
Lepomis megalotis	N	Len	I	3	25	10	10	13	3	18	21
Lepomis microlophus	N	Len	I	0	0	0	6	0	0	0	1
Lepomis miniatus	N	Lot	I	0	0	0	0	1	0	0	3
Micropterus punctulatus	N	Len	I	0	2	2	1	0	1	4	19
Micropterus salmoides	N	Len	I	0	1	8	9	0	2	5	18
Pomoxis annularis	N	Len	I	2	11	4	5	6	2	4	4
Pomoxis nigromaculatus	N	Len	I	0	0	0	5	0	0	5	7
Ammocrypta vivax	N	Lot	I	0	0	0	0	0	0	9	0
Etheostoma chlorosoma	N	Lot	I	0	2	0	0	0	0	1	0
Etheostoma gracile	N	Lot	I	0	6	0	0	0	0	1	0
Etheostoma proeliare	N	Len	I	0	1	0	0	0	0	0	0
Percina caprodes[d]	N	Len	I	0	1	1	0	0	0	0	1
Percina sciera	N	Lot	I	0	0	4	0	4	0	9	4
Aplodinotus grunniens	N	Len	T	3	4	4	3	9	4	0	12
Mugil cephalus	N	—	—	0	0	0	0	0	0	13	19
Alosa chrysochloris	N	—	—	0	0	0	0	0	0	0	2
Anchoa mitchilli	N	—	—	0	0	0	0	0	0	3	0
Elops saurus	N	—	—	0	0	0	0	0	0	0	1
Strongylura marina	N	—	—	0	0	0	0	0	0	4	0
Trinectes maculatus fasciatus	N	—	—	0	0	0	0	0	0	0	1
Kathetostoma giganteum	I	—	—	0	0	2	0	0	0	0	0
Number of samples				11	35	16	10	24	5	20	22
Mean richness				6	11	12	13	13	15	17	20
Rarefied richness				13.7	28.9	27.8	23.3	26.1	23.3	43.1	50.1

Mean and rarefied richness values are given for each reach and period.
[a]Native status from Hubbs *et al*. (2008).
[b]Habitat association from Frimpong and Angermeier (2009).
[c]Tolerance level from Linam *et al*. (2002).
[d]Possible misidentification.

mussel *Dreissena polymorpha* suggests new forms of degradation are still occurring. Historical periods of nutrient contamination likely plagued many large-order streams that flow through agricultural or urbanized reaches (Hoagstrom *et al*., 2011), but data-logging infrastructure was not in place before implementation of clean-water initiatives for most of the USA (Wolman, 1971; Smith *et al*., 1987). This lack of empirical support is regarded as a major limiting factor hindering evaluation of the CWA and its long-term success in preserving or restoring the biological, chemical, and physical conditions of US rivers (Knopman and Smith, 1993). We addressed this limitation by compiling existing biological, chemical, and hydrologic data sources into a single database, but this approach suffers from potential confounding effects. Changes in gear type and sampling effort through time can cause artificial changes in biological community structure (Patton *et al*., 1998) and might have contributed to the observed shift in assemblage composition between Periods 1 and 2. We addressed this issue by using only collections that targeted the entire assemblage (Perkin and Bonner, 2011), by reducing the data to species occurrence, and by accounting for unequal effort through rarefaction (Gido *et al*., 2010). We believe the observed shifts in community structure represent actual change rather than sampling artefacts for two reasons. First, our results closely match those reported by Land *et al*. (1998) in terms of increasing fish species and guild compositions in the mainstem Trinity River between the early 1970s and early 1990s. Second, the timing of observed changes in fish community structure in our study matches the timing of invertebrate assemblage revitalization and the cessation of major fish kills in the river (Davis, 1997). The natural state of water chemistry in the Trinity River is also largely unknown, and quantitative records obtained by the USGS only date back to the period immediately prior to cleanup initiatives. We addressed this issue by developing a qualitative history dating back to the 19th century when conditions were described as degraded. After implementation of cleanup, data indicated strong shifts in water quality during the quantified period of history (1968–2008). Although hydrologic data were abundant for the study area, there was little change in the Trinity River flow regime during recent history (Wellmeyer *et al*., 2005) when the fish assemblage and quality of water underwent dynamic change.

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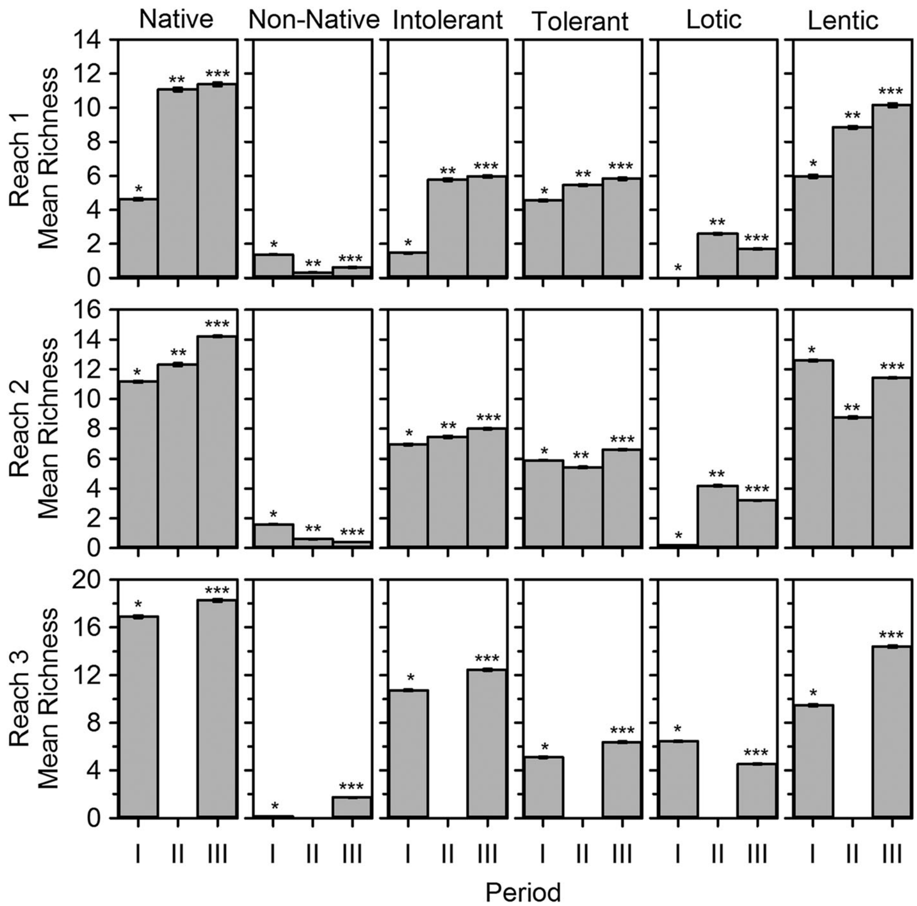

Figure 6. Temporal changes in mean species richness for native status, tolerance level, and habitat association guilds for Trinity River of Texas fish assemblage collections taken from three reaches during three periods (Period I: 19710–1974; Period II: 1987–1988; and Period III: 1994–2008). Values are means (±95% confidence intervals) based on randomized repeated subsamples (see text for details); asterisks represent significant differences based on non-overlapping confidence intervals

CONCLUSIONS

Lack of long-term, broad-scale data evaluating hydrological, chemical, and biological responses to restoration initiatives limits our understanding of what constitutes effective environmental management in rivers (Wolman, 1971). This has been especially true in the context of the CWA of 1972 and rivers in the USA (Knopman and Smith, 1993). Our results suggest attempts to reduce point source pollution in the mainstem Trinity River downstream of DFW caused declines in nutrient contamination and increases in fish species richness and guild composition during a 40-year period. Our results also reveal that manipulations within DFW caused changes in water quality and fish assemblage composition among distant downstream reaches greater than 200 km away. Together, these findings signify the potential long-term and broad-scale effectiveness of aquatic environmental management approaches, even among river ecosystems characterized by extensive damage from anthropogenic alterations (Allan and Flecker, 1993).

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