Peaking hydropower and fish assemblages: an example from the Tallapoosa River, AL

Elijah B. Lamb[1], Dennis R. DeVries[1], and Russell A. Wright[1]

[1]Auburn University School of Fisheries Aquaculture and Aquatic Sciences

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Abstract

Dams alter many aspects of riverine environments and can have broad effects on aquatic organisms and habitats both upstream and downstream. While dams and the associated reservoirs can provide many services to people (hydropower, recreation, flood control, navigation, etc.), they can negatively affect riverine ecosystems. In particular, hydropeaking dams affect downstream fish habitats by increasing variability in discharge and temperature. To assess the effects of Harris Dam on the Tallapoosa River, AL, operating under an adaptive management plan implemented in 2005, we sampled fish for community and diet analyses from four sites on the river: three in the regulated region downstream of the dam, and one unregulated site upstream. Fish were collected every other month using boat/barge electrofishing. We used Shannon's H, nonmetric multidimensional scaling (NMDS), a multiresponse permutation procedure (MRPP), and indicator species analysis to quantify patterns in fish assemblage structure and determine how assemblages varied among sites. NMDS and MRPP indicated significant fish assemblage differences among sites with the tailrace fish assemblage being distinct from the other downstream sites, and sites becoming more similar to the upstream, unregulated site (relative to fish assemblages) with distance downstream of the tailrace. The tailrace fish assemblage included higher proportions of rheophilic species that may be better suited for variable and/or high flows. Altered fish assemblages demonstrated continued effects of Harris Dam on the downstream aquatic systems, particularly close to the dam. These effects may indicate further mitigation should be considered depending on conservation and management goals.

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Elijah B. Lamb, Dennis R. DeVries, Russell A. Wright

Auburn University School of Fisheries, Aquaculture, and Aquatic Sciences

203 Swingle Hall,

Auburn, AL 36849

Corresponding author: Elijah Lamb ebl0020@auburn.edu

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Abstract

Dams alter many aspects of riverine environments and can have broad effects on aquatic organisms and habitats both upstream and downstream. While dams and the associated reservoirs can provide many services to people (hydropower, recreation, flood control, navigation, etc.), they can negatively affect riverine ecosystems. In particular, hydropeaking dams affect downstream fish habitats by increasing variability in discharge and temperature. To assess the effects of Harris Dam on the Tallapoosa River, AL, operating under an adaptive management plan implemented in 2005, we sampled fish for community and diet analyses from four sites on the river: three in the regulated region downstream of the dam, and one unregulated site upstream. Fish were collected every other month using boat/barge electrofishing. We used Shannon's H, nonmetric multidimensional scaling (NMDS), a multiresponse permutation procedure (MRPP), and indicator species analysis to quantify patterns in fish assemblage structure and determine how assemblages varied among sites. NMDS and MRPP indicated significant fish assemblage differences among sites with the tailrace fish assemblage being distinct from the other downstream sites, and sites becoming more similar to the upstream, unregulated site (relative to fish assemblages) with distance downstream of the tailrace. The tailrace fish assemblage included higher proportions of rheophilic species that may be better suited for variable and/or high flows. Altered fish assemblages demonstrated continued effects of Harris Dam on the downstream aquatic systems, particularly close to the dam. These effects may indicate further mitigation should be considered depending on conservation and management goals.

Keywords: fish assemblage, hydropower, flow regulation, community ecology

Introduction

Dams cause physical and ecological impacts to aquatic ecosystems disrupting over half of the world's major river systems (Nilsson et al., 2005). Although dams vary in their characteristics (e.g., size, function), they have some common effects on habitat, discharge, and energy inputs. All dams interrupt the movement of water and aquatic organisms and can disrupt energy and nutrient flows. Organisms in river systems rely on energy inputs from both upstream and downstream habitats, and structures interrupting river connectivity generally have watershed-scale effects (Vannote et al., 1980; Ward, 1989; Stanford & Ward, 2001; Schindler et al., 2013). Upstream, inundation eliminates terrestrial habitats, increases water depth, and removes turbulent flow, significantly affecting lotic specialists (Nilsson et al., 2005). Downstream, releases from the dams alter habitats, create unnatural temperature variation, alter natural flow variation, limit channel development/maintenance, and reduce system productivity (Nilsson et al., 2005). The serial discontinuity concept (SDC) details theoretical ecosystem responses to altered conditions downstream of such disturbances, identifying how certain variables (ecological connectivity, biodiversity, particulate matter abundance) are expected to decrease in tailrace areas before gradually increasing along a longitudinal gradient (Ward & Stanford, 1989; Stanford & Ward, 2001).

Hydropeaking dams in particular can have effects on downstream waters, including artificial flow variation and alteration to the natural thermal regime (Irwin & Freeman, 2002; Twardek et al., 2022). R.L. Harris Dam on the Tallapoosa River in east-central Alabama is such a facility that operates on a hydropeaking schedule. Constructed in 1983, the dam initially operated with hypolimnetic release and no partial flow between generation periods which resulted in both thermal and flow modification of the downstream area. In 2005, stakeholders adopted an adaptive management plan, (called the "Green Plan") that established regulations on dam operations, including base flow requirements between generation periods (Irwin & Freeman, 2002; Kleinschmidt Associates, 2018). Although the Green Plan mitigated some of the effects of dam operation, water was still released from the hypolimnion, leading to pulses of higher flow with colder water temperatures during summer. Downstream effects of dam operations under such a modified hydropeak-

ing schedule on fishes have not been thoroughly described prior to this work. The subsequent change in downstream water temperature between generation versus non-generation periods could reach 10°C and the change remained measurable many kilometers downstream (Irwin & Freeman, 2002), although the majority of hourly temperature changes were less than 2°C (DeVries et al. 2022).

Alabama is a hotspot of aquatic biodiversity, and each river system in the state has unique aspects of its native fish assemblages (Lydeard & Mayden, 1995; Mettee et al., 1996; Freeman et al., 2005). Historically as many as 126 fish species have been identified in the Tallapoosa River, and the broader Mobile drainage is one of the most diverse freshwater fish assemblages in North America (Lydeard & Mayden, 1995; Travnichek & Maceina, 1994; Freeman et al., 2005). Protecting biodiversity in Alabama rivers is increasingly important as anthropogenic impacts increase. To protect aquatic resources, it is necessary to identify how various threats (such as a dam) affect the ecology and life history of resident organisms. The Tallapoosa River from the tailrace of Harris Dam downstream to the headwaters of Lake Martin is diverse in both habitat and resident fishes and is considered one of the highest quality river segments in the piedmont region of Alabama (Irwin & Freeman, 2002). Although little information about the Tallapoosa River fish assemblage is available from before Harris Dam was constructed, several studies of the fish assemblage have been conducted since 1983 (Kinsolving & Bain, 1993; Travnichek & Maceina, 1994; Irwin & Freeman, 2002; Freeman et al., 2005; Irwin et al., 2019). Beyond the effects of hydropeaking dam operation discussed earlier, Travnichek and Maceina (1994) showed that the diversity and richness of fish inhabiting shallow downstream waters was decreased in regulated portions of the Tallapoosa River versus upstream/unregulated areas. However, little work has been conducted after the implementation of the modified hydropeaking operations on the overall fish community; as such, we quantified the fish assemblage across a spatial gradient downstream from the dam under the current "Green Plan" to determine the magnitude of continued effects of Harris Dam.

Methods

Fish Collection

Fish were collected by pulsed-DC boat electrofishing (Midwest Lake Management, Inc. Missouri, USA) once every other month from April 2019 through February 2021 (a total of 12 bimonthly sampling events), with sampling at each site consisting of six, 600-sec transects. We sampled four sites (Figure 1)– three of these were located on the mainstem Tallapoosa River below Harris Dam (tailrace, Wadley, Horseshoe Bend), and one site was located upstream of Harris Reservoir to serve as an unregulated reference site (Lee's Bridge). Output voltage was standardized between 700-900 volts with 100-120 pulses per second, and GPS coordinates were recorded at the start and end of each transect. A tow-barge electrofisher was used at the tailrace site given that it is inaccessible by boat; barge sampling consisted of one individual with the anode and 1-2 dip-netters wading alongside, with another individual pushing the barge itself. Barge electrofishing followed the same procedures, although a slightly lower voltage (500-700 volts) was used for safety.

For the first six sampling events, collected fish were returned to the lab for processing. For the remaining six sampling events, all initial processing was done in the field and fish were returned live to their sampling locations. In the lab, all collected fish were identified to species and up to 10 individuals of each species were weighed and measured. If more than 10 individuals of a given species were present in a transect, the remaining individuals were counted, and the group was bulk weighed. For processing in the field, collected fish were identified, measured (nearest mm TL), and weighed (nearest g) on site..

Data Analysis

Shannon's diversity index (H) and total species richness were calculated for each site (with all samples combined) to allow comparison across sites and with previous studies (e.g. Travnichek & Maceina, 1994; Freeman et al., 2005). Given the potential for bias, estimates were generated both with non-native species included and omitted. Additionally, the proportion contribution by numbers for each fish family at each site was calculated. Catch per effort (fish/hour of electrofishing) of each species was calculated for each season where spring included March-May, summer included June-August, fall included September-November, and winter included December-February.

Overall assemblage structure was characterized using multivariate methods as in Kiraly et al. (2014), all of which were conducted using R statistical software using the *Vegan* and *labdsv* packages (R core team, 2020; Oksanen et al., 2020; Roberts, 2019). We fourth-root transformed CPE data to account for the extreme skew that was present in these data before using Bray-Curtis dissimilarity to conduct nonmetric multidimensional scaling on the transformed CPE data (NMDS; Goodsell & Connell, 2002; Kiraly et al., 2014). Bray-Curtis dissimilarity typically performs better than other measures of dissimilarity for ecological datasets (Kiraly et al., 2014; Orksanen et al., 2020). Function metaMDS in R was used to perform nonmetric multidimensional scaling. Several random starts were used with 50 iterations maximum and final dimensionality was determined by considering stress reduction and interpretability (Kiraly et al., 2014; Oksanen et al., 2020). Kendall's tau correlation coefficient (T) was calculated to determine the magnitude and direction of species correlations and MDS axes. Tau correlation coefficients were summed across species for each family to determine which family contributed the most to each MDS axis.

To quantify differences in fish assemblages across sites, we used a multiresponse permutation procedure (MRPP) based on the same Bray-Curtis dissimilarity (Kiraly et al., 2014). Only the three seasons (spring, summer, fall) during which sampling occurred at all sites were included. MRPP generates an A -statistic as well as a p-value, both of which must be considered to fully interpret results. The A statistic is a measure of effect size and describes within-group homogeneity compared to the random expectation; A= 1 if all units within groups are identical and A =0 if heterogeneity among groups equals the expected value by chance (McCune & Grace, 2002). If the null hypothesis is true, the p-value is the likelihood that the observed difference between groups is due to chance (McCune and Grace, 2002). Average dissimilarities both between and within groups were calculated to create a dendrogram describing the relationships between groups and to create group blocks. Additionally, MRPP generates a test statistic, δ, which is the overall weighted mean of group mean differences (Oksanen et al., 2020). MRPP serves as a hypothesis test of differences between groups of sampling units where p is the probability that δ is less than the observed value. A dendrogram was generated based on the Bray-Curtis dissimilarities where the vertical termination of each branch represented the within-group dissimilarity across seasons while the horizontal lines represented the dissimilarity between site blocks.

Indicator species values (IndVal) were calculated based on the formula given in Dufrene and Legendre (1997) and clarified by Roberts (2019) in the *labdsv* R package for each species given a significant overall MRPP result. This formula calculates the indicator values "d" of species as the product of the relative frequency and relative average abundance in clusters (Roberts, 2019) as follows:

$$d_{\mathrm{ic}} = f_{\mathrm{ic}} * a_{\mathrm{ic}}$$

$$f_{\mathrm{ic}} = \frac{\sum_{j \in c} P_{\mathrm{ij}}}{n_c}$$

$$a_{\mathrm{ic}} = \frac{\sum_{j \in c} x_{\mathrm{ij}}/n_c}{\sum_{k=1}^{K} (\sum_{j \in c} x_{\mathrm{ij}}/n_k)}$$

where: P_{ij} = presence/absence (1/0) of species i in sample j , x_{ij} = abundance of species i in sample j , and n_c = number of samples in cluster c , for cluster c *[?]* K.

IndVal analysis accounts for species site specificity and fidelity and ranges from 0-1 (Dufrene & Legendre, 1997). The index equals one when a species is found in all sampling units (seasons) of a group (site). P-values were calculated for each species' IndVal using a randomization procedure (Roberts, 2019). Species were randomly reassigned to sampling units and groups 1000 times and IndVals were recalculated to create a distribution of possible IndVal values from the given data. The p-value was the proportion of randomized IndVals that were greater than the observed value (i.e., the probability that the observed value was due to chance; Dufrene & Legendre, 1997).

Results

Fish species diversity was highest at Wadley (H=2.88) and lowest at Horseshoe Bend (H=2.49), although values were generally similar across sites (range among sites was 0.39; Table 1). Centrarchids made up the largest proportion of collected fish across all sites with their highest contribution being in the tailrace and at Horseshoe Bend (Figure 2). Cyprinids/leuciscids contributed the second largest proportion of fish collected from all sites except the tailrace (where they were third highest), where the second largest proportion was from percids (Figure 2). The proportion of the assemblage composed of percids was also relatively high at Wadley, with much lower values at Lee's Bridge and Horseshoe Bend (Figure 2). Catastomids made up larger proportions of the collected families at Lee's Bridge, Wadley, and Horseshoe Bend and a much lower proportion in the tailrace (Figure 2). At Lee's Bridge, 3 of the 10 most abundant species were centrarchids and at the downstream sites it was between 3 of 10 in the tailrace and 4 of 10 at Horseshoe Bend (Table 2).

NMDS yielded stable, two-dimensional ordinations (Figure 3a, final stress = 0.09) in 40 iterations. Ordination axis MDS-1 correlated most positively with Snail Bullhead (T= 0.98), Bandfin Shiner (T= 0.80), Black Madtom (T =0.80), and Rough Shiner (T= 0.79; Figure 4) and most negatively with White Bass (T=-0.96), Yellow Perch (T =-0.92), River Redhorse (T =-0.92), and Bullhead Minnow (T =-0.92). Axis MDS-2 correlated most positively with Blueback Herring (T =0.63), Skipjack Herring (T =0.63), Golden Shiner (T =0.63), and Black Bullhead (T =0.63); and most negatively with Grass Carp (T =-0.62), Pretty Shiner (T =-0.62), Snail Bullhead (T =-0.34), and Bandfin Shiner (T= -0.31; Figure 4). At the family level, MDS-1 correlated most positively with ictalurids and percids and most negatively with clupeids, catastomids, and moronids (Table 3).

Ordinations showed clear separation between sites as well as seasonal variation in the assemblages within sites (Figure 3a). Separation across sites in MDS-1 showed longitudinal variation in fish assemblage downstream of Harris Dam; in addition, as distance downstream of the tailrace increased, MDS-1 of the fish assemblage approached that of the upstream reference site, identifying a potential recovery gradient (Figure 3a). Differences between seasons were driven by generally negative summer and positive spring values for MDS-2 (Figure 3a). MDS-2 appeared to vary seasonally, being higher in the spring (March-May) and lowest in the summer (June-August; Figure 3a).

Multiresponse Permutation Procedure and Indicator Species Analysis

MRPP indicated a significant difference among site-specific fish assemblages (p = 0.001, A =0.298, δ=0.290; Figure 3b). Indicator species analysis identified 1 - 5 significant indicator species depending on site (Table 4). Lee's Bridge had the highest number of indicator species which included five separate families (Table 4). In the tailrace three indicator species were identified including one percid (an Alabama listed Priority 2 species; Mirarchi et al. 2004), one ictalurid, and one centrarchid. Wadley had three indicator species including one percid and two campostomids (Table 4). The single indicator species identified at Horseshoe Bend was Black Redhorse, a campostomid (Table 4). Average between- and within-group dissimilarities grouped Horseshoe Bend and Wadley as the site block with the least dissimilarity and Lee's Bridge as the most dissimilar (Figure 5). The Tailrace fell within a block containing the downstream sites although separated from Wadley and Horseshoe Bend (Figure 5).

Discussion

The serial discontinuity concept describes how riverine systems are expected to respond to major disruptions along the course of a river (Ward and Stanford, 1989; Ward & Stanford, 1995; Stanford & Ward, 2001). The expectations vary depending on the geomorphology of the river and the responses are described for three different reach types – constrained, braided, and meandering (Ward & Stanford, 1995). The portions of the Tallapoosa River sampled in our study fit between the constrained and braided reach types, characterized by a stable channel, a narrow riparian corridor, and lotic habitats throughout. As such, the biodiversity of the Tallapoosa was expected to increase with distance downstream (Ward & Stanford, 1995). Interestingly, we measured no such increase in Shannon's H or species/family richness within the river section from Harris Dam to the headwaters of Lake Martin. However, we documented variation in the assemblage structure that

is likely influenced by the presence and operation of Harris Dam.

Sunfishes and minnows were generally the most common fish families in this part of the Tallapoosa River, and variation in diversity from upstream to downstream was neither large nor systematic. Catostomids, centrarchids, and cyprinids were dominant in catches above Harris Dam, similar to the findings of Travnichek and Maceina (1994) who conducted a survey (prior to the implementation of the water release management Green Plan in 2005) of the broader Tallapoosa River from the upper reaches near Heflin, AL downstream to the Coastal Plain. Overall species diversity index values for this study area were slightly higher and more variable in 1994 (1.98 - 3.53) versus our study (2.49 - 2.88), though this difference may have been driven in part by differences in sampling techniques (prepositioned electrofishing grids in 1994 versus boat electrofishing in our study). Trends in fish diversity upstream to downstream were similar between our findings and those of Travnichek and Maceina (1994), who found some evidence that river regulation diminished the number of obligate fluvial specialist species in the Tallapoosa River. It is important to note that centrarchids were not historically dominant in the Tallapoosa River (Irwin & Hornsby, 1997). Considering that catch rates of centrarchids in both our study and in Travnichek and Maceina (1994) were high downstream of Harris Dam, this supports the idea that generalist species (such as many centrarchids) may be less affected by river regulation (Scott, 1951; Swingle, 1953; Kinsolving & Bain, 1993; Travnichek & Maceina, 1994), and that this trend had not been altered by implementation of the more recent "Green Plan" flow modification. While Travnichek and Maceina (1994) observed an increase in species richness from upstream to downstream across a much broader study area, their results from within the bounds of this study area did not yield such a trend, again consistent with our findings.

Irwin and Hornsby (1997) compared rotenone surveys conducted at Horseshoe Bend in 1951 (pre-Harris Dam) versus 1996 (post-Harris Dam) to assess the effects of river regulation due to Harris Dam on downstream fish assemblages. Differences in species composition between these rotenone studies also suggested that the pre- versus post-dam fish assemblage at Horseshoe Bend may have shifted from one dominated by cyprinids and ictalurids to one dominated by centrarchids (Irwin & Hornsby, 1997). Our results showed a larger relative proportion (by numbers) of centrarchids versus the 1951 rotenone sample (0.01 in 1951, 0.46 in our study), but a similar proportion to the 1996 sample (0.51 in 1996). In addition, the proportion of cyprinids and catostomids in our sample was higher than in the 1996 rotenone sample (0.11 in 1996, 0.43 in our study), but similar to the 1951 findings (0.50 in 1951; Irwin & Hornsby, 1997). Differences in sampling method (electrofishing versus rotenone), sampling frequency (bimonthly here versus a single sample historically), and seasonality likely limits direct comparisons. However, the continued prevalence of centrarchids in the Tallapoosa River downstream of Harris Dam in our study, Travnichek and Maceina (1994), and Irwin and Hornsby (1997) suggests that Harris Dam has affected and continues to affect the downstream fish assemblage.

The presence of significant differences in fish assemblage composition across sites in ordinal space and the formation of upstream versus downstream site blocks on the Tallapoosa River suggests that discharge regulation or habitat modification by Harris Dam was responsible for much of the variation in fish assemblage structure. Although riverine fish assemblages naturally vary longitudinally, regulation of the Tallapoosa River by Harris Dam changes the quality of habitat by altering temperatures and flow regimes, and by fragmenting the river, changing natural patterns of fish movement, persistence, and colonization (Vannote et al., 1980; Irwin & Freeman, 2002; Kiraly et al., 2014, Irwin et al., 2019). Our study describes patterns that are influenced by Harris Dam and the resulting flow regulation. The first multidimensional scaling axis (MDS-1) correlated strongly with the CPE of ictalurids, percids, and centrarchids, and the tailrace had the highest scores of any site for MDS-1. The species within those families that largely influenced this trend tended to be smaller bodied ictalurids and darters – species that tend to be more rheophilic. This suggests that the tailrace is primarily occupied by species that are able to persist in the variable flow conditions, potentially at the expense of other species including minnows and suckers, that might be less well-suited for conditions in the tailrace. The number of indicator species at a site can be considered a measure of how unique that site is relative to the other sites considered. The presence of more indicator species at the upstream site supports the conclusion that the Lee's Bridge fish assemblage was distinct from the sites downstream of the dam.

The MDS analysis indicated both spatial and temporal variation in fish assemblage structure throughout the mainstem Tallapoosa River. Several previous studies did not include seasonal variation when quantifying the Tallapoosa fish assemblage (Travnichek & Maceina, 1994; Irwin et al., 2019). Our results indicate that conclusions drawn from Travnichek and Maceina (1994) and Irwin et al. (2019) need to be limited to the time scales they encompassed. Our inability to sample at Lee's Bridge in winter unfortunately hampered even broader generalization given that a model of complete annual variation in fish assemblage might identify additional patterns.

It is difficult to determine exactly how much Harris Dam has affected the fish assemblage in the Tallapoosa River given the paucity of data predating its construction. Longitudinal variation in fish assemblage structure is inevitable in river systems as energy and nutrient inputs change (Vannote et al., 1980). Overinterpreting the single rotenone sample taken before Harris Dam began operation is also questionable given the habitat heterogeneity of the Tallapoosa River and the limited spatial and temporal coverage of rotenone sampling (Swingle, 1953). Outside of this study area, literature indicates that flow regulation, and especially peaking flow, negatively impacts fish recruitment and spawning (Weyers et al., 2003; Rolls et al., 2013). The persistent regulation of the Tallapoosa River by Harris Dam likely resulted in a changes in the fish assemblage driven by the inability of certain species to adapt their spawning and feeding habits to the rapid temperature and flow fluctuation (Rolls et al., 2013).

Releases of water from dams can strongly affect habitat conditions for fish and other aquatic organisms (Freeman et al., 2005; Young et al., 2011). Impacts that affect fish at the individual scale can also be manifested at the population and assemblage scales. Our sampling spanned a longitudinal gradient that included a site above Harris Dam and three sites at increasing distances downstream of the dam, allowing us to examine whether patterns in fish communities are consistent with expected effects of the dam, namely a recovery gradient in the diversity or assemblage composition (Travnichek & Maceina, 1994; Ward & Stanford, 1995; Ward & Stanford, 2001), as well as whether the implementation of the modified flow regime has had any effects on the downstream fish community. Previous studies (see below) have quantified assemblage structure and responses of particular fish populations across this same reach, allowing comparisons that span a range of temporal scales.

Across the sampled sites downstream of Harris Dam, we expected to observe an increase in biodiversity as hypothesized under the serial discontinuity concept (Ward & Stanford, 1995). Interestingly, while we did see a recovery gradient in multivariate space downstream of the dam (where site assemblages became increasingly similar to the upstream, unregulated site), there was no such gradient in species richness or Shannon's H. These species diversity results are similar to those from other previous studies of the Tallapoosa River fishes. Clearly the use of simple diversity indices did not capture the changes in fish assemblage structure that we observed downstream of the dam. Quantifying more than just simple diversity indices will be required to capture the recovery gradient in response to disturbance.

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Data Availability Statement

The data that support the findings of this study are available on request from the corresponding author.

Table 1. Total number of fish species, families, and Shannon's H diversity index values collected from four sites on the Tallapoosa River, Alabama. All estimates were calculated including non-native species; the total number of species and values of Shannon's H calculated with non-native species excluded are in parentheses.

Site	Total Species	Total Families	Shannon's H
Lee's Bridge (LB)	39 (37)	9	2.80 (2.78)
Tailrace (TR)	38 (38)	7	2.59 (2.59)
Wadley (WD)	35 (35)	7	2.88 (2.88)
Horseshoe Bend (HB)	33 (31)	7	2.49 (2.54)
All	55 (51)	9	3.06 (3.05)

Table 2. Scientific names, common names, species abbreviations, site distributions, and classifications used in this report. Site abbreviations as defined in Table 1. The ten most abundant species at each site are identified by superscripts (most abundant =1).

Scientific Name	**Common Name**	**Abbreviation**	**T&M 1994**	**LB**	**TR**	**WD**	**HB**
Amia calva	Bowfin	BOWF	HG	X			
Alosa aestivalis	Blueback Herring	BBHR	HG				X
Alosa chrysochloris	Skipjack Herring	SKJH	HG				X
Dorosoma cepedianum	Gizzard Shad	GIZS	HG	X^{6}		X	X
Dorosoma petenense	Threadfin Shad	THSH	HG	X		X	X^{10}
Campostoma oligolepis	Largescale Stoneroller	LSSR	FS	X	X^{7}	X^{10}	
Cyprinella callistia	Alabama Shiner	ALSH	FS		X^{3}	X^{9}	X
Cyprinella gibbsi	Tallapoosa Shiner	TPSH	FS	X	X	X	
Cyprinella venusta	Blacktail Shiner	BTSH	HG	X^{5}	X	X^{4}	X^{4}
Cyprinus carpio	Common Carp	CCAR	HG	X^{9}	X	X	X
Luxilus chrysocephalus	Striped Shiner	STSH	FS		X	X	
Luxilus zonistius	Bandfin Shiner	BAFS	FS		X		
Lythrurus bellus	Pretty Shiner	PRSH	FS	X			
Notemigonus crysoleucas	Golden Shiner	GLDA	HG				X
Ctenopharyngodon idella	Grass Carp	GCAR	HG	X			
Notropis baileyi	Rough Shiner	RSHN	FS		X		
Notropis stilbius	Silverstripe Shinner	SPSH	FS	X		X^{8}	X^{3}
Notropis texanus	Weed Shiner	WESH	HG	X	X		
Notropis xaenocephalus	Coosa Shiner	COOS	FS	X	X	X	X
Pimephales vigilax	Bullhead Minnow	BUMN	HG	X^{7}			
Semotilus thoreauianus	Dixie Chub	DXCB	FS		X		
Hypentelium nigricans	Alabama Hogsucker	AHOG	FS	X	X	X^{7}	X
Minytrema melanops	Spotted Sucker	SPSR	HG	X	X	X	X
Moxostoma carinatum	River Redhorse	RVRH	FS	X			
Moxostoma duquesnei	Black Redhorse	BREH	FS	X		X	X^{9}
Moxostoma poecilurum	Blacktail Redhorse	BTRH	HG	X^{1}	X	X^{2}	X^{5}
Ameiurus brunneus	Snail Bullhead	SNBL	FS		X		
Ameiurus melas	Black Bullhead	BLBH	HG				X
Ameiurus natalis	Yellow Bullhead	YBUL	HG	X	X^{10}	X	X
Ameiurus nebulosus	Brown Bullhead	BRBH	HG		X	X	
Ictalurus furcatus	Blue Catfish	BCAT	HG	X^{10}			X
Ictalurus punctatus	Channel Catfish	CCAT	HG	X^{3}	X^{9}	X	X^{7}
Noturus funebris	Black Madtom	BLMT	FS		X		
Noturus leptacanthus	Speckled Madtom	SPMT	FS			X	
Pylodictis olivaris	Flathead Catfish	FCAT	HG	X	X		X
Fundulus olicaceus	Blackspotted Topminnow	BLTM	HG	X	X	X	X
Morone chrysops	White Bass	WHBA	HG	X			
Morone saxatilis	Striped Bass	STBA	HG	X	X		
Ambloplites ariommus	Shadow Bass	SHBA	FS	X	X^{4}	X	X
Lepomis auritus	Redbreast Sunfish	RBSF	HG	X	X	X^{3}	X^{2}
Lepomis cyanellus	Green Sunfish	GSUN	HG		X	X	X
Lepomis gulosus	Warmouth	WARM	HG	X	X	X	X
Lepomis macrochirus	Bluegill	BLGL	HG	X^{2}	X^{1}	X^{6}	X^{6}
Lepomis microlophus	Redear Sunfish	REAR	HG	X^{8}	X	X	X
Lepomis spp.	Bluegill X Green Sunfish	BGGN	HG		X	X	X
Lepomis spp.	Hybrid Redbreast	RBSX	HG			X	X

Scientific Name	**Common Name**	**Abbreviation**	**T&M 1994**	**LB**	**TR**	**WD**	**HB**
Micropterus henshalli	Alabama Bass	ALAB	HG	X[4]	X[6]	X[1]	X[1]
Micropterus salmoides	Largemouth Bass	LGMB	HG	X	X	X	X
Micropterus tallapoosae	Tallapoosa Bass	TPBA	FS	X	X	X	X[8]
Pomoxis annularis	White Crappie	WHCP	HG	X	X	X	
Pomoxis nigromaculatus	Black Crappie	BLCP	HG	X	X	X	X
Etheostoma chuckwachatte	Lipstick Darter	LIPD	FS		X[5]	X	X
Etheostoma stigmaeum	Speckled Darter	SPDT	FS	X	X	X	
Etheostoma tallapoosae	Tallapoosa Darter	TPDA	FS		X	X	
Perca flavescens	Yellow Perch	YPER	HG	X			
Percina kathae	Mobile Logperch	MLOG	FS	X	X	X	X
Percina palmaris	Bronze Darter	BRDT	FS	X	X[2]	X[5]	X
Percina smithvanizi	Muscadine Darter	MBDT	FS	X	X[8]	X	X

Table 3: Nonmetric multidimensional scaling (MDS) correlation coefficients summed and averaged by family for fish collected from the Tallapoosa River, Alabama. Sorted from largest to smallest values for MDS-1.

Family	MDS-1	MDS-1 Mean	MDS-2	MDS-2 Mean
Ictaluridae	1.56	0.17	-0.32	-0.04
Percidae	1.10	0.16	1.00	0.14
Centrarchidae	0.18	0.02	0.41	0.04
Fundulidae	0.07	0.07	0.17	0.17
Cyprinidae/Leucisidae	-0.49	-0.03	-0.15	-0.01
Amiidae	-0.91	-0.91	-0.13	-0.13
Moronidae	-1.27	-0.63	-0.40	-0.20
Catastomidae	-1.37	-0.27	0.30	0.06
Clupeidae	-1.44	-0.36	1.27	0.32

Table 4: Indicator species values and p-values for sites on the Tallapoosa River, Alabama.

Site	Species	Indicator Value	P-value
LB	Bowfin	1.000	0.015
LB	Bullhead Minnow	1.000	0.023
LB	Gizzard Shad	0.703	0.015
LB	Blue Catfish	0.664	0.024
LB	Redear Sunfish	0.462	0.023
TR	Lipstick Darter	0.594	0.015
TR	Yellow Bullhead	0.540	0.031
TR	Bluegill	0.312	0.043
WD	Speckled Darter	0.659	0.032
WD	Alabama Hogsucker	0.415	0.016
WD	Blacktail Redhorse	0.364	0.016
HB	Black Redhorse	0.402	0.045

Figure Legends

Figure 1: A map of the study area on the Tallapoosa River, AL. Sampling locations are indicated with filled,

black circles.

Figure 2: Proportion by number of the sampled fish assemblage belonging to each family collected from four sites on the Tallapoosa River, AL. Site abbreviations are as defined in Table 1.

Figure 3: (a) Multidimensional scaling ordinations of CPE by site and season for fishes captured from four sites in the Tallapoosa River. Site abbreviations are as defined in Table 1. (b) The distribution of permuted delta values from a multiresponse permutation procedure. The dashed line to the left represents the observed delta.

Figure 4: Kendall's tau correlations with nonmetric multidimensional scaling ordination axes for individual species CPE data from the Tallapoosa River upstream and downstream of Harris Dam. Species abbreviations are as defined in Table 2 and dashed lines represent zero on the MDS axes.

Figure 5: Site clusters based on between – and within – site dissimilarity for sites on the Tallapoosa River, AL. Branches hang to within-group dissimilarity while horizontal lines are positioned at among-group dissimilarities. Site abbreviations are as defined in Table 1.