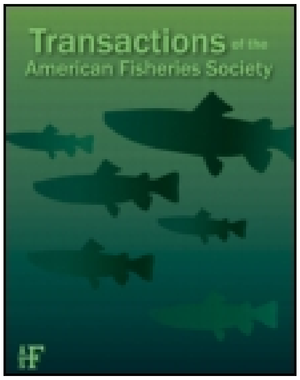

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ARTICLE

Development of a Multimetric Index for Fish Assemblages in a Cold Tailwater in Tennessee

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Abstract
Tailwaters downstream of hypolimnetic-release hydropeaking dams exhibit a unique combination of stressors that affects the structure and function of resident fish assemblages. We developed a statistically and biologically defensible multimetric index of fish assemblages for the Caney Fork River below Center Hill Dam, Tennessee. Fish assemblages were sampled at five sites using boat-mounted and backpack electrofishing gear from fall 2009 through summer 2011. A multivariate statistical approach was used to select metrics that best reflected the downstream gradients in abiotic variables. Five metrics derived from boat electrofishing samples and four metrics derived from backpack electrofishing samples were selected for incorporation into the index based on their high correlation with environmental data. The nine metrics demonstrated predictable patterns of increase or decrease with increasing distance downstream of the dam. The multimetric index generally exhibited a pattern of increasing scores with increasing distance from the dam, indicating a downstream recovery gradient in fish assemblage composition. The index can be used to monitor anticipated changes in the fish communities of the Caney Fork River when repairs to Center Hill Dam are completed later this decade, resulting in altered dam operations.

Tailwaters below hypolimnetic-release hydropeaking dams exhibit erratic flow patterns, unnatural thermal regimes, and reduced dissolved oxygen (DO) concentrations during periods of reservoir stratification (Baxter 1977; Ward and Stanford 1983; Olden and Naiman 2010). Such stressors affect the composition, structure, and function of various biotic assemblages in tailwaters (e.g., macroinvertebrates: Stevens et al. 1997, Johnson and Harp 2005; mussels: Vaughn and Taylor 1999, Layzer and Scott 2006; fish: Bain et al. 1988, Quinn and Kwak 2003). The disturbance influence of dams on abiotic factors is most pronounced near the dam and attenuates with increasing distance downstream. Correspondingly, biotic assemblages in tailwaters tend to exhibit longitudinal recovery gradients (Ward and Stanford 1983; Kinsolving and Bain 1993; Camargo and Voelz 1998).

The combination of stressors in flow-regulated rivers has been shown to affect the composition of resident fish assemblages through several mechanisms. High variability in discharge reduces the persistence of critical habitat for young-of-the-year (age-0) fishes, resulting in reductions in recruitment, especially among fishes that tend to spawn during periods when discharge is typically high (i.e., early spring; Scheidegger and Bain 1995; Freeman et al. 2001). Altered thermal regimes can affect the reproductive timing, egg development, growth, and other physiological processes of some species, thereby

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reducing survival and species persistence (Swink and Jacobs 1983; Donaldson et al. 2008). Hypoxic conditions have been linked to poor health and condition of fishes in tailwaters (Devlin and Bettoli 2001; Todd and Bly 2002). Because of the strong correlations between discharge, temperature, and DO, recent investigations have emphasized the importance of interactions between factors and have advocated for a focus on net ecological effects of disturbance in regulated rivers (Irwin and Freeman 2002; Bednarek and Hart 2005; Richter et al. 2006).

Multimetric indices comprise a number of individual metrics, which are biologically relevant and quantifiable measures of community responses to some disturbance (Karr et al. 1986). Multimetric indices are superior to single-component assemblage indices because they retain the simplicity of presenting complex data in the form of a single, easily interpretable number, but they are less sensitive to random process and observational errors, encompass a variety of relevant biocriteria, and can be dissected to the component metrics (Karr et al. 1986; Simon and Lyons 1995; Roset et al. 2007). Multimetric indices are most responsive to environmental disturbance if the component metrics are carefully chosen and calibrated to the system of interest (Simon and Lyons 1995; Whittier et al. 2001). Procedures for selecting metrics to incorporate into a multimetric index most typically involve using parametric statistical procedures to compare the response of biological metrics at one site to responses at a "reference site" that represents the least-impacted, or reference, conditions (Hughes 1995; Kennard et al. 2006; Stoddard et al. 2006). In instances where an appropriate reference site cannot be identified, multivariate statistics can be used to identify links between resident biotic assemblages and abiotic conditions (Manolakos et al. 2007; Collier 2009). Multivariate statistics have been applied as an exploratory tool for comparing trends in environmental disturbances to macroinvertebrate assemblages (Gerritsen 1995; Fore et al. 1996; Reynoldson et al. 1997) and fish assemblages (Angradi et al. 2009; Flinders et al. 2009; Doll 2011). Recently, multivariate procedures have also been used to directly identify sets of suitable metrics for multimetric indices (Hallett et al. 2012; Miranda et al. 2012). Multimetric indices that lack a benchmark reference condition can be useful for describing changes in assemblages, but they are not highly transferable to other systems and should not be used to define biotic integrity (Karr et al. 1986; Jennings et al. 1995; Hawkins et al. 2010).

Few indices have been specifically developed for radically altered systems, such as tailwaters downstream of large hydroelectric dams. Challenges associated with developing a multimetric index for radically altered systems may include reduced sampling efficiency (Flotemersch et al. 2011), atypical community responses to stressors (Schulz et al. 1999; Whittier et al. 2001), unique fish assemblages (Lyons et al. 1996; Quinn and Kwak 2003), and the lack of a reference condition (Jennings et al. 1995). We are aware of only two published multimetric indices that have been developed explicitly for regulated river ecosystems (Bowen et al. 1998; Scott 1999). The index developed for Tennessee Valley Authority (TVA) tailwaters (Scott 1999) is no longer used for monitoring because it lacked a statistical underpinning (it was based on expert opinion) and did not respond to long-term environmental changes in some of the tailwaters where it was applied (C. Saylor, TVA, personal communication). In some regulated systems, resident fish assemblages are not adaptive (Jennings et al. 1995). In systems with an adaptive fish assemblage, however, multimetric indices may be used to describe and monitor assemblage structures both spatially and temporally.

In 2008, the U.S. Army Corps of Engineers (USACE) initiated a Dam Seepage Control Program (DSCP) that changed the way in which water was stored by and released from Center Hill Dam on the Caney Fork River, central Tennessee. To relieve pressure on the dam and reduce leakage during the DSCP renovation, USACE has generally maintained water levels at 1.5–3.0 m—and as much as 5.5 m—below normal operating levels. Minimum flows (7 m^3/s) were provided for the first time ever, and peaking hydropower production was curtailed in frequency and magnitude. Changes in the physical, chemical, and biological characteristics of this cold tailwater are anticipated to occur after the multiyear project is completed and after normal (i.e., harsher) hydropower production and water level management have resumed. During September 2009, in cooperation with the U.S. Geological Survey and Tennessee Technological University, USACE launched an investigation into possible responses of the Caney Fork River ecosystem to the DSCP. Herein, we report on the development of a multimetric index relating the structure of fish assemblages in the Caney Fork River to disturbance effects from Center Hill Dam.

METHODS

Study area.—Center Hill Dam, located in Dekalb County, Tennessee, was constructed on the Caney Fork River in 1948. It is operated by USACE to provide hydropower, flood control, recreation, and other benefits. Center Hill Lake is a tributary storage impoundment that is stratified from late spring through fall and exhibits DO depletion in the metalimnion and hypolimnion during stratification. To boost DO concentrations in the discharge, the dam is equipped with aerating turbines and a sluice gate. Downstream of Center Hill Dam, the Caney Fork River flows approximately 42 km to its confluence with the Cumberland River (Figure 1). We sampled the fish community in the Caney Fork River at five sites located near access points at Lancaster (3.5 km downstream from Center Hill Dam), Happy Hollow (8.5 km), Betty's Island (13.8 km), Stonewall (24.3 km), and Carthage (31.5 km). Sites were designated A–E to reflect their sequential order in a downstream direction.

Fish sampling.—We sampled fish assemblages in the Caney Fork River by using methods adapted from those developed by the TVA (Scott 1999; Knight et al. 2008). Boat-mounted and backpack DC electrofishing gear were used to sample fish in all habitat types at each of the five sites. Sampling was conducted from fall 2009 through summer 2011. We attempted to sample all sites during each season (eight seasons in total) using both

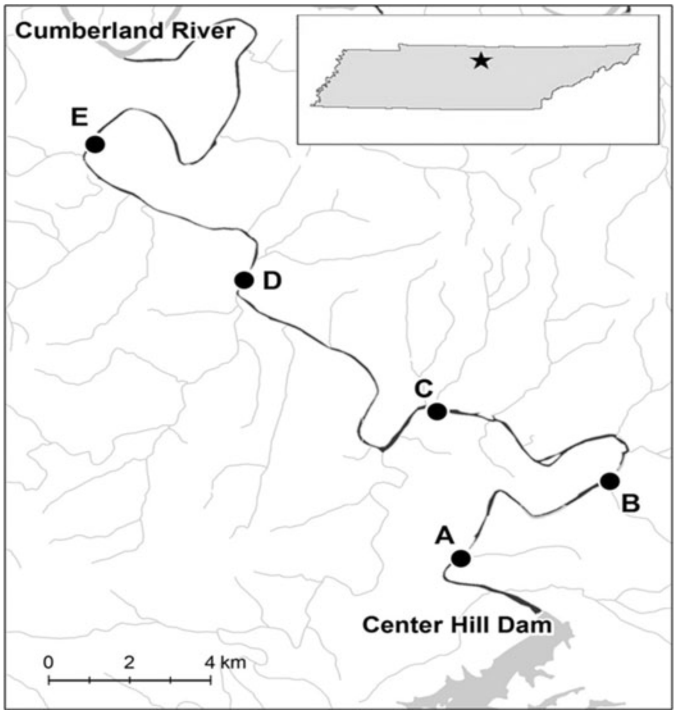

FIGURE 1. Map of the Caney Fork River downstream of Center Hill Dam, central Tennessee, showing five sites that were sampled with backpack electrofishing and boat-mounted electrofishing gear, 2009–2011.

types of electrofishing gear. Data collected during these 2 years of sampling were used to develop and calibrate the multimetric index.

Shorelines and channel sections consisting of mostly unwadeable habitat were sampled by using a Smith-Root 2.5 GPP (generator-powered pulsator) electrofishing unit (delivering 4 A at 60 Hz) in a jet-drive electrofishing boat that was positioned perpendicular to the shoreline and that traveled in a downstream direction. Boat electrofishing was conducted during the day and during periods of one turbine unit of hydropower generation (i.e., 100–117 m^3/s), with one netter positioned on the bow of the boat. Sampling at a site commenced at the access point and was confined to no more than the 2.5-km reach immediately downstream. A series of 10-min (pedal time) subsamples was conducted on alternating shorelines until two successive subsamples (each containing more than five individuals) failed to encounter any previously undetected species. One 10-min subsample encompassed roughly 0.4 km of shoreline. If the end of the sampling reach was arrived at before the stopping rule was fulfilled, additional runs were conducted on alternating shorelines and traveling downstream from the starting point until the stopping criterion was met.

Wadeable areas were sampled by using a backpack DC electrofishing unit and proceeding in a general upstream direction. The backpack operator was equipped with a net and was accompanied by a designated netter, who also carried a bucket for specimens. Backpack electrofishing was conducted during the day at base flow (i.e., river discharge was restricted to dam leakage, sluicing, and orifice gate operations). Sampling was conducted in all areas that were safely accessible by wading from the access point. A series of 10-min subsamples was conducted in shoreline habitat (<2 m from the bank) and channel habitat (>3 m from the bank). Sampling in each macrohabitat continued until two successive subsamples (each containing more than one individual) failed to encounter any species that were previously undetected in that macrohabitat.

For both sampling methods, we attempted to collect fish in the order that they were observed, regardless of species or size. However, preference was given to netting fish species that had not yet been encountered at a site. Additionally, we did not attempt to capture all clupeids (i.e., Gizzard Shad *Dorosoma cepedianum* and Threadfin Shad *D. petenense*) because they were usually encountered in large schools and any attempt to net all of the individuals would have compromised the catch rates of other species during the timed run. Fish were identified to species, measured for TL, counted, and released outside of the sampling reach. Any fish that could not be positively identified in the field were brought to the laboratory and identified to species based on the keys presented by Etnier and Starnes (1993).

Because of inherent differences among fishes in terms of their vulnerability to backpack and boat electrofishing, we expected that sample compositions would be dependent on gear type. Differences in the sizes of fish collected with the two sampling methods were investigated by using the Kolmogorov–Smirnov test to compare the length frequency distributions. Differences in the species collected were investigated by comparing sample composition (i.e., percent composition) using analysis of similarity (ANOSIM) and Bray–Curtis similarity matrices. Differences in the number of species collected using each sampling method were tested across sites and seasons by using mixed-model ANOVAs and multiple t-test comparisons. Microsoft Excel version 14.0 was used for data entry, manipulation, and arithmetic calculations. All univariate statistical procedures were performed using the Statistical Analysis System version 9.3 (SAS Institute 2010). Differences were considered significant at P-values less than 0.05.

Abiotic measurements.—It is widely understood that disturbances caused by hydropeaking dam operations include alteration of the hydrology, temperature, and DO concentration in the tailwater (Baxter 1977; Ward and Stanford 1983; Olden and Naiman 2010). We therefore quantified the extent of disturbance resulting from dam operations at each site by measuring variables associated with these three key habitat characteristics. Specific environmental measurements associated with these habitat characteristics were selected a priori to test for correlations with biotic data. The key measurement variables were (1) the change in water depth over 1 h resulting from one turbine unit of hydropower generation; (2) the average maximum daily temperature during the warmest month; and (3) the minimum daily DO concentration during fall 2011.

The change in depth over 1 h was measured on October 14, 2011, by placing weighted markers at the water's edge adjacent to the main channel at base flow and then measuring the water depth (cm) 1 h after the generation pulse reached the site. The average maximum daily temperature for the warmest month was determined by deploying temperature loggers (Onset HOBO, programmed to record water temperature every 0.5 h) at each of the five fish sampling sites during summer 2011. Dissolved oxygen concentrations were measured weekly from August 18, 2011, through November 1, 2011, at the tail-end of sustained generation with one turbine unit (i.e., when the impacts of hypolimnetic discharge were most pronounced); measurements were taken approximately 2 m from the shoreline in the main channel by using a hand-held YSI Model 55 DO meter. Because a suite of other (unmeasured) variables responds to impoundments in a predictable manner (Ward and Stanford 1983), the $\log_e$ transformed distance downstream of the dam was also considered a key environmental variable. A multivariate ordination relating sites to environmental variables was generated using principal components analysis.

Data pretreatment.—Data were pretreated to remove fish species or life stages that could disproportionately influence index scores, mask important relationships, or bias scores in a manner that would not reflect meaningful changes within the fish assemblages. Clupeids (Threadfin Shad and Gizzard Shad), stocked fishes (three species of trout [described below] and Striped Bass *Morone saxatilis*), and age-0 fishes represented potentially problematic groups and were all excluded from the data set. Clupeids are seasonally abundant in the Caney Fork River, tending to form large schools, and they were fairly common at all sites. Therefore, differences in clupeid catch rates are probably not indicative of assemblage composition but can potentially affect index scores (Jennings et al. 1995; Pearson et al. 2011). Brown Trout *Salmo trutta*, Rainbow Trout *Oncorhynchus mykiss*, and Brook Trout *Salvelinus fontinalis* are all stocked into the Caney Fork River, and none of those species reproduces in the river; likewise, Striped Bass that are stocked into the Cumberland River move into the Caney Fork River in certain seasons and do not reproduce. Stocking and perhaps angler harvest rates can differ among sites; therefore, the abundance of stocked sport fish can reflect fisheries management policy (Lyons et al. 1996; Langdon 2001). The reasons for excluding age-0 fish from the analyses include differences in catchability, habits, and life history traits between juveniles and adults (Angermeier and Karr 1986; Freeman et al. 2001). The size cutoffs for categorizing fish as age 0 were derived from length frequency histograms and from values reported by Etnier and Starnes (1993).

Metric selection.—Because of differences in fish sizes and species collected using boat electrofishing and backpack electrofishing, metric selection procedures were applied to the two data sets separately to produce two sets of gear-specific metrics. Candidate metrics evaluated here included some that were found to be informative in previous studies (Simon and Lyons 1995) as well as several unique metrics with the potential for responding to tailwater stressors. Fishes were classified into guilds a priori based on species accounts given in fish identification textbooks (e.g., Etnier and Starnes 1993), the literature (e.g., Coutant 1977), and reports of other multimetric indices (e.g., GDNR 2005; TVA 2004; NCDENR 2006). Instead of raw abundance data, percentage composition data were typically used to calculate metrics because the relative contribution of individuals to the total fauna better reflects community interactions, is less affected by sampling effort, and is generally more informative (Dauwalter and Pert 2003). Suites of 54 and 46 candidate metrics were developed, calculated, and tested for data collected using boat and backpack electrofishing methods, respectively (Table 1). Nineteen candidate metrics pertaining to proportional data collected with backpack electrofishing were recalculated by applying a $\log_e$ transformation to catch data before calculating the proportions. This transformation was tested because extremely common species can overwhelm the data set and reduce the sensitivity of certain metrics (Cao et al. 2011).

An automated multivariate stepwise procedure (BIOENV/BVSTEP; Clarke and Warwick 2001) was used to identify sets of suitable metrics for each sampling protocol from the respective lists of candidate metrics. The procedure was implemented using PRIMER version 6.1.13 (Clarke and Gorley 2006). The BIOENV/BVSTEP procedure uses similarity matrices (Euclidian distances) to categorize sites according to four key environmental variables (Table 2) and then uses an iterative process to select a subset of responsive candidate metrics that produce a site orientation (based on biotic attributes) that is most similar to the environmental site orientation (Clarke and Ainsworth 1993). The procedure maximizes Spearman's rank correlation coefficient (Spearman's ρ), a measure of similarity between the two multivariate data sets. Each candidate metric value was standardized by dividing by the largest observed response to that metric across all samples. Each environmental variable was normalized by subtracting the mean response from all sites and dividing by the SD. A random selection of 20 starting variables and 500 restarts (iterations) was specified, and sample similarity was evaluated using Euclidean distances. Redundant metrics were defined as highly correlated metrics ($|r| \geq 0.75$) that contained similar or opposing taxa and that described similar functional characteristics of the assemblage (Whittier et al. 2007; Angradi et al. 2009). If the procedure produced a set of metrics with redundancies, the BIOENV/BVSTEP procedure was repeated to assess which of the redundant variables, when entered independently, maximized the correlation between the biotic metrics and environmental data. That metric was then individually included in the multimetric index, and all redundant metrics were excluded. The metric set that contained at least four nonredundant metrics and maximized correlation was selected for incorporation into the multimetric index (Lyons et al. 1996; Langdon 2001; Southerland et al. 2007). Multivariate ordinations relating samples to the responses observed for each of the two sets of selected metrics were generated using principal components analysis.

Multimetric index scoring.—Metrics that were chosen for inclusion in the multimetric index were each scored on a

TABLE 1. List of all candidate metrics tested for inclusion in the multimetric index for the tailwater fish assemblage in the Caney Fork River downstream of Center Hill Dam (* = metrics that were only tested for the boat electrofishing data set; ** = metrics that were only tested for the backpack electrofishing data set; † = metrics [n = 19] pertaining to backpack electrofishing data that were also tested after applying a $\log_e$ transformation to underlying count data).

Diversity and abundance metrics
- Overall CPUE
- Number of species
- Number of benthic species
- Proportion dominant taxon
- Margalef’s diversity index
- Shannon’s diversity index
- Pielou’s evenness index

Tolerance metrics
- Proportion tolerant†
- Proportion intolerant†
- Proportion of fish with deformities, erosions, lesions, and tumors
- Proportion of species classified as tolerant
- Proportion of species classified as intolerant
- Proportion coolwater fishes†
- Proportion warmwater fishes†

Taxonomic composition metrics
- Proportion in the family Cyprinidae†
- Proportion in the family Centrarchidae†
- Percent of centrarchids in the genus *Micropterus* (black basses)*
- Proportion in the family Cottidae**
- Proportion in the family Catostomidae*
- Percent of catostomids in the genus *Ictiobus* (buffaloes)*
- Percent of catostomids in the genus *Moxostoma* (redhorses)*
- Percent of catostomids as *Ictiobus* or *Moxostoma**
- Proportion in the genus *Lepomis* (sunfishes)†
- Proportion in the genus *Campostoma* (stonerollers)**†
- Proportion darters[a]**†
- Number of species in the family Cyprinidae
- Number of species in the family Catostomidae
- Number of species in the family Centrarchidae
- Number of species in the genus *Lepomis*
- Number of species in the genus *Ictiobus**
- Number of species in the genus *Moxostoma**
- Number of species categorized as darters[a]

Life history metrics
- Proportion insectivores†
- Proportion omnivores†
- Proportion piscivores†
- Proportion insectivorous cyprinids†
- Proportion complex spawners†
- Proportion simple lithophilic spawners†
- Proportion benthic†
- Proportion of large fish[b] classified as benthic*
- Proportion of small fish[b] classified as benthic*
- Proportion benthic invertivores†
- Proportion pelagic insectivores†
- Proportion rheophilic†
- Proportion of species classified as insectivorous
- Proportion of species classified as omnivorous
- Proportion of species classified as piscivorous
- Proportion of species classified as insectivorous cyprinids
- Proportion of species classified as complex spawners
- Proportion of species classified as simple lithophilic spawners
- Proportion of species classified as benthic
- Proportion of large fish[b] species classified as benthic*
- Proportion of small fish[b] species classified as benthic*
- Proportion of species classified as benthic invertivores
- Proportion of species classified as pelagic insectivores
- Proportion of species classified as rheophilic

[a]Metric included all fish from the darter genera *Etheostoma* and *Percina*.
[b]Large = mean TL of adult individuals in samples was greater than 115 mm; small = mean TL of adult individuals was less than 115 mm.

continuous scale from 0 to 10 (Blocksom 2003; Stoddard et al. 2008; Pearson et al. 2011). Thresholds for defining floor and ceiling values, which were used to assign scores, were set by using all of the samples. Floor values for each metric were set 10% lower than the minimum value observed in the range of responses by that metric, and ceiling values were set 10% higher than the maximum value observed in the range of responses by the metric. Threshold values that were calculated outside of the possible range of responses by a metric were adjusted to reflect the minimum or maximum possible response value. For metrics that responded positively to improved conditions, the general formula for determining scores was

$$\text{Metric score} = (\text{metric value} - \text{floor})/(\text{ceiling} - \text{floor}) \times 10.$$

For metrics that responded negatively to improved conditions, floor and ceiling values were reversed in the calculation. Floor and ceiling thresholds for metric scoring were calculated by using all observations included in this study.

Multimetric index scores were then calculated as the sum of all metric scores for a specific site and season. To provide a more complete perspective of resident fish assemblages,

TABLE 2. Abiotic variables that were measured at five sites in the Caney Fork River downstream of Center Hill Dam (Figure 1) and incorporated in the BIOENV/BVSTEP procedure; DO = dissolved oxygen.

	Site				
Variable	A	B	C	D	E
Distance downstream of the dam (km)	3.2	8.5	13.8	24.3	31.5
Change in depth after 1 h (cm)	140	80	63	32	22
Minimum DO concentration (mg/L), September 2011	4.5	6.1	6.8	8.9	9.0
Mean maximum temperature (°C), August 2011	15.6	16.3	17	19.4	19.7

multimetric index scores represented the sum of all metric scores for paired boat and backpack electrofishing samples (i.e., samples that were collected with the different gear types at the same site and during the same season). To test the utility of relying solely on boat electrofishing or backpack electrofishing to index the fish assemblages in the Caney Fork River, the sums of metrics derived from each sampling method were examined in relation to the overall multimetric index scores by using linear regression. The correlations of boat electrofishing metrics, backpack electrofishing metrics, and the combination of all metrics with the environmental data (Table 2) were assessed by using Spearman's ρ.

RESULTS

Fish Sampling

We collected 36 site- and season-specific boat electrofishing samples (total pedal time = 77 h) and 38 site- and season-specific backpack electrofishing samples (total electrofishing time = 71 h). Unsuitable or unsafe river conditions precluded the collection of boat samples at sites B, C, and D during fall 2010 and at site D during spring 2011; backpack samples were not collected at site D in fall 2009 or spring 2011. With the two electrofishing methods, we collected a total of 67 fish species representing 16 families. Pretreating the data (removal of data from clupeids, stocked fishes, and age-0 fish) resulted in data sets that comprised 4,398 fish for all samples collected using boat electrofishing and 6,215 fish for all samples collected using backpack electrofishing.

As expected, each sampling method provided a different view of the Caney Fork River fish assemblage. Small fish dominated the backpack electrofishing samples in wadeable areas, whereas larger fish (and a greater range of fish sizes) characterized the boat electrofishing samples collected in deeper habitats (Figure 2; Kolmogorov–Smirnov test: $P < 0.0001$). The species composition also differed between the two sampling protocols (ANOSIM: $R = 0.982$, $P < 0.001$).

Excluding clupeids and trout species (all of which were abundant), the most common species collected with boat electrofishing gear across all sites and seasons were the Bluegill *Lepomis macrochirus*, Freshwater Drum *Aplodinotus grunniens*, and Spotted Sucker *Minytrema melanops* (Table 3). Backpack electrofishing samples across all sites and seasons were dominated by Banded Sculpins *Cottus carolinae*, which represented 67% of all fish collected with that gear; Central Stonerollers *Campostoma anomalum* and Bluntnose Minnow *Pimephales notatus* were also abundant in backpack electrofishing samples, and together they represented an additional 15% of all individuals collected (Table 4).

The average number of species encountered using boat electrofishing was related to site ($P = 0.0002$). Raw species richness (i.e., the average number of species encountered during a single sampling event) was lowest at site A (10.5 species/sample) and second lowest at site B (17.8 species/sample). The average number of species encountered did not significantly differ among the three downstream-most sites (i.e., site C: 21.8 species/sample; site D: 22.5 species/sample; site E: 22.6 species/sample). The average number of species collected was related to season ($P = 0.0009$). The number of species collected across all sites was highest during spring (23.2 species/sample) and summer (20.6 species/sample) and lowest during fall (16.3 species/sample) and winter (14.5 species/sample).

The average number of species encountered using backpack electrofishing gear was related to site ($P = 0.0003$). The average number of species was lowest at site A (4.1 species/sample) and site B (4.3 species/sample). The number of species encountered at site C (8.0 species/sample) was significantly higher than those at upstream sites but lower than those at the two downstream-most sites (site D: 14.1 species/sample; site E: 13.6 species/sample). The average number of species collected across all sites was not related to sampling season ($P = 0.1499$).

Abiotic Measurements

The effects of Center Hill Dam on key environmental attributes of the tailwater during 2011 were most pronounced near the dam and attenuated with increasing distance downstream. Site A, the site closest to the dam, experienced the most rapid increase in depth, the lowest mean temperature, and the lowest DO concentrations (Table 2). The warmest mean water temperatures were observed during August 2011, and minimum DO concentrations were observed on September 27, 2011. The three variables were highly correlated with each other: 97.5% of the variation was explained by the first principal component, and an additional 2.4% of the variation was explained by the second principal component (Figure 3).

Metric Selection

Five metrics derived from boat electrofishing samples were selected for incorporation into the index. This suite of metrics maximized the correlation with environmental data (Spearman's ρ = 0.656). Three metrics—Shannon's diversity index

TABLE 3. The five most abundant fish species in boat electrofishing samples at five sites on the Caney Fork River over all seasons. Percentages indicate the relative abundance of each species at each site.

Rank	Site A	Site B	Site C	Site D	Site E
1	Spotted Sucker (52%)	Spotted Sucker (19%)	Freshwater Drum (12%)	Freshwater Drum (15%)	Freshwater Drum (17%)
2	Bluegill (12%)	Freshwater Drum (13%)	Common Carp *Cyprinus carpio* (10%)	Golden Redhorse (11%)	Bluegill (13%)
3	Freshwater Drum (8%)	Bluegill (10%)	Bluegill (10%)	Black Redhorse *Moxostoma duquesnei* (7%)	Golden Redhorse (11%)
4	Largemouth Bass *Micropterus salmoides* (4%)	Rock Bass *Ambloplites rupestris* (7%)	Spotted Sucker (9%)	Black Buffalo *Ictiobus niger* (7%)	Black Buffalo (5%)
5	White Bass *Morone chrysops* (3%)	White Bass (6%)	Golden Redhorse *Moxostoma erythrurum* (8%)	Bluegill (6%)	Common Carp (5%)

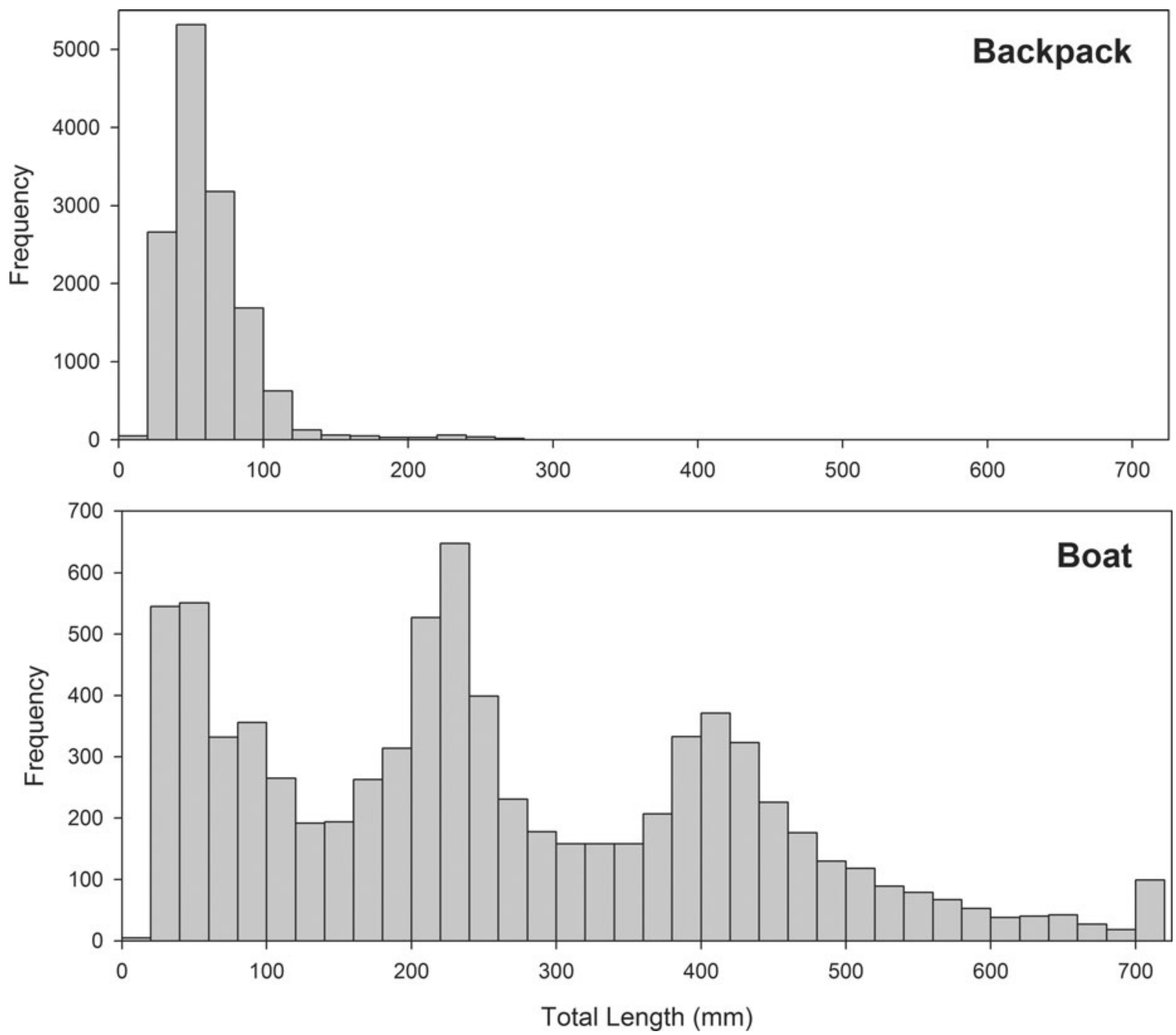

FIGURE 2. Length frequency distributions (mm TL) of all fish collected by using backpack electrofishing (top) and boat electrofishing (bottom) gear in the Caney Fork River below Center Hill Dam, 2009–2011.

TABLE 4. The five most abundant fish species in backpack electrofishing samples at five sites on the Caney Fork River over all seasons. Percentages indicate the relative abundance of each species at each site.

Rank	Site A	Site B	Site C	Site D	Site E
1	Banded Sculpin (79%)	Banded Sculpin (85%)	Banded Sculpin (63%)	Banded Sculpin (51%)	Banded Sculpin (60%)
2	Central Stoneroller (3%)	Central Stoneroller (2%)	Central Stoneroller (24%)	Central Stoneroller (21%)	Rainbow Darter (8%)
3	Striped Shiner *Luxilus chrysocephalus* (0.4%)	Bluntnose Minnow (1%)	Bluntnose Minnow (5%)	Rainbow Darter (8%)	Bluntnose Minnow (6%)
4	Bluntnose Minnow (0.4%)	Striped Shiner (1%)	Rainbow Darter *Etheostoma caeruleum* (1%)	Cumberland Snubnose Darter *Etheostoma atripinne* (3%)	Northern Studfish *Fundulus catenatus* (4%)
5	Green Sunfish *Lepomis cyanellus* (0.3%)	Whitetail Shiner *Cyprinella galactura* (0.2%)	Telescope Shiner *Notropis telescopus* (1%)	Bluntnose Minnow (2%)	Cumberland Snubnose Darter (4%)

(Shannon 1948), the percentage of catostomids as buffaloes *Ictiobus* spp. or redhorses *Moxostoma* spp., and the proportion warmwater fishes—increased along with vectors indicating decreased disturbance. Conversely, two metrics (the proportion in the family Catostomidae and the proportion of species classified as rheophilic) decreased along with vectors indicating decreased disturbance (Figure 3).

Four metrics derived from backpack electrofishing samples were selected for incorporation into the index based on their high correlation with environmental data (Spearman's $\rho = 0.655$). Metrics that were derived from $\log_e$ transformed data generally resulted in higher contrast among samples and better correlation with environmental data than metrics that were derived from untransformed data. Three metrics (Shannon's diversity index, proportion [$\log_e$] darters, and the number of darter species) increased along with vectors indicating decreased disturbance. One metric (proportion [$\log_e$] in the family Cottidae) decreased along with vectors indicating decreased disturbance (Figure 3). Due to the exceedingly high abundance of sculpins relative to other species, especially at upstream sites, Shannon's diversity index was inversely correlated with the proportion ($\log_e$) in the family Cottidae ($r = -0.88$); however, the metrics described two different assemblage characteristics that are not fundamentally coupled.

Multimetric Index Scoring

The directional trends and ranges of responses for the nine metrics were used to establish scoring thresholds (Table 5). The greatest range in responses among sites and across all samples was observed for the following metrics: proportion of the catch in the family Catostomidae (boat electrofishing samples), percentage of catostomids as *Ictiobus* spp. or *Moxostoma* spp. (boat electrofishing samples), proportion ($\log_e$) in the family Cottidae (backpack electrofishing samples), and number of darter species (backpack electrofishing samples). Selected metrics demonstrated predictable patterns of increase or decrease with increasing distance downstream of the dam, especially for the percentage of catostomids as *Ictiobus* spp. or *Moxostoma* spp. in boat electrofishing samples and for Shannon's diversity index in backpack electrofishing samples (Figure 4).

Multimetric index scores generally exhibited a pattern of increase with increasing distance from the dam within and across seasons. The average scores for the five sites in increasing distance from the dam were 15.5 for site A, 29.5 for site B, 40.7 for site C, 60.1 for site D, and 65.4 for site E (Table 6). Backpack

TABLE 5. Metrics that were incorporated into the multimetric index based on fish assemblage data from boat electrofishing and backpack electrofishing conducted in the Caney Fork River. The observed response (positive or negative) of the metric to improving environmental conditions is indicated. Floor and ceiling threshold values were calculated from the data and were used in the scoring procedure.

Metric	Response	Floor	Ceiling
Boat electrofishing metrics			
Shannon's diversity index	+	1.00	3.20
Proportion in the family Catostomidae	−	0.01	0.76
Percentage of catostomids as buffaloes *Ictiobus* spp. or redhorses *Moxostoma* spp.	+	0.00	100
Proportion warmwater fishes	+	0.23	0.94
Proportion of species classified as rheophilic	−	0.13	0.76
Backpack electrofishing metrics			
Shannon's diversity index	+	0.00	2.30
Proportion ($\log_e$) in the family Cottidae	−	0.04	1.00
Proportion ($\log_e$) darters	+	0.00	0.57
Number of darter species	+	0.00	6

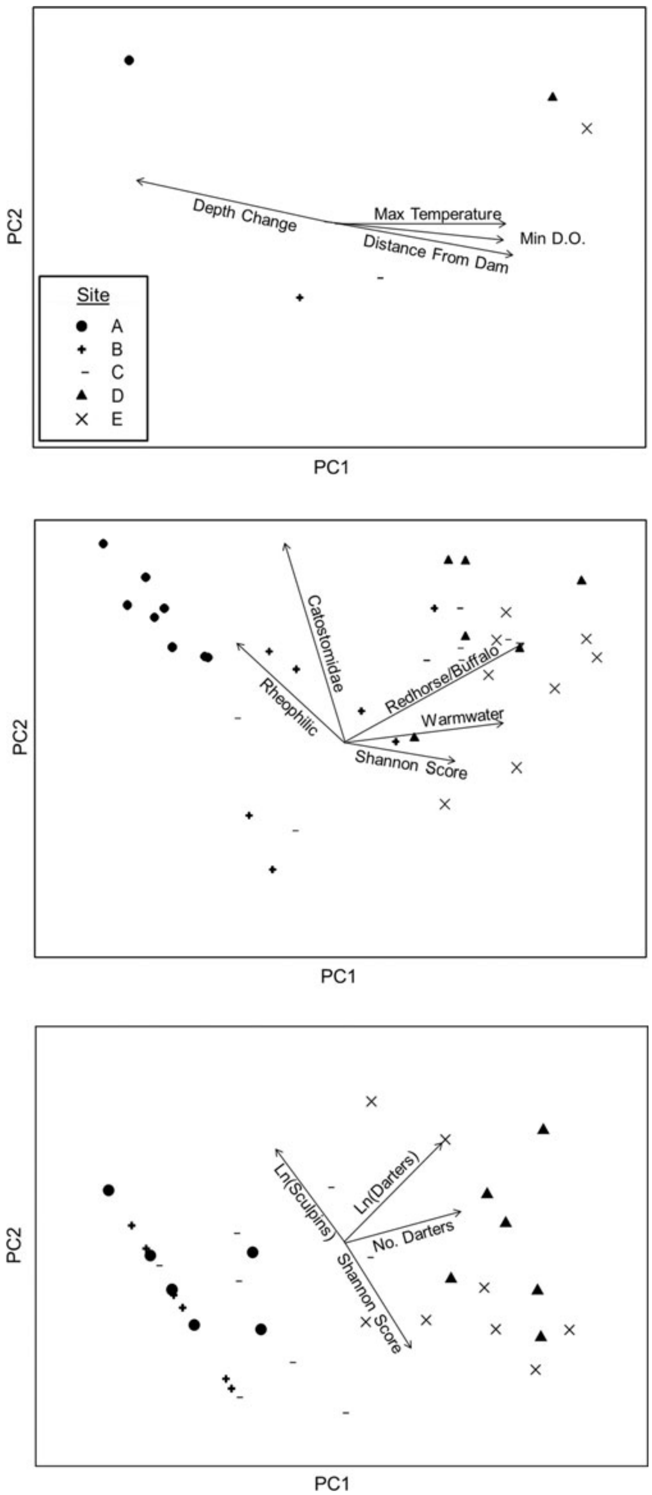

FIGURE 3. Principal components analysis ordinations of four abiotic variables measured in the Caney Fork River below Center Hill Dam (top panel), five metrics based on boat electrofishing samples (middle panel), and four metrics based on backpack electrofishing samples (bottom panel; PC = principal component; DO = dissolved oxygen concentration; see Table 5 for the full description of each metric). The lengths and directions of vectors indicate the magnitudes and directions of trends. Metrics for each gear type were independently selected using the BIOENV/BVSTEP procedure (see Methods).

TABLE 6. Multimetric index (MMI) scores for each of five sites on the Caney Fork River during eight seasons. Values for boat electrofishing and backpack electrofishing indicate the sums of all metrics associated with each sampling method.

Season and year	Score type	Site A	Site B	Site C	Site D	Site E
Fall 2009	MMI score	6.7	38.6	46.6		63.1
	Boat	6.7	27.1	31.1	27.8	39.4
	Backpack	0.0	11.5	15.5		23.6
Winter 2010	MMI score	10.6	26.2	22.2	62.9	53.2
	Boat	10.6	20.1	17.6	36.7	35.4
	Backpack	0.0	6.1	4.6	26.1	17.8
Spring 2010	MMI score	22.6	25.8	43.2	60.0	57.1
	Boat	14.5	18.7	31.2	30.8	33.7
	Backpack	8.1	7.0	12.0	29.3	23.4
Summer 2010	MMI score	19.8	31.4	53.3	63.7	64.9
	Boat	7.0	20.3	34.3	33.4	36.8
	Backpack	12.7	11.1	19.0	30.3	28.1
Fall 2010	MMI score	15.5				65.4
	Boat	9.4				32.5
	Backpack	6.0	4.0	19.0	28.0	32.9
Winter 2011	MMI score	17.5	24.9	32.3	53.0	59.6
	Boat	13.5	22.6	22.5	28.2	32.4
	Backpack	4.0	2.2	9.7	24.8	27.3
Spring 2011	MMI score	16.0	33.6	42.5		58.4
	Boat	10.0	30.0	32.1		38.5
	Backpack	6.1	3.6	10.4		19.9
Summer 2011	MMI score	15.3	26.3	44.6	61.0	65.2
	Boat	3.9	26.3	29.5	29.7	34.1
	Backpack	11.3	0.0	15.1	31.3	31.1

electrofishing and boat electrofishing scores were related to each other ($R^2 = 0.4667$; $P < 0.0001$) and to the overall multimetric index score ($R^2 \geq 0.8404$; $P < 0.0001$). The suite of metrics composing the multimetric index had a higher correlation with environmental variables (Spearman's $\rho = 0.785$) than the component metrics that were derived from either boat electrofishing or backpack electrofishing data.

DISCUSSION

The multimetric index for the Caney Fork River downstream of Center Hill Dam is statistically and biologically defensible. The index development procedure that we used adhered to general recommendations (e.g., Roset et al. 2007; Stoddard et al. 2008) but relied on a multivariate technique to select metrics from a large list of candidates. Benefits of using a multivariate approach such as the one employed here include objectivity, repeatability, and an explicit link between environmental stressors and biotic responses (Miranda et al. 2012). Perhaps the most important benefit of using a multivariate procedure, especially for a radically altered system, is the ability to develop a statistically

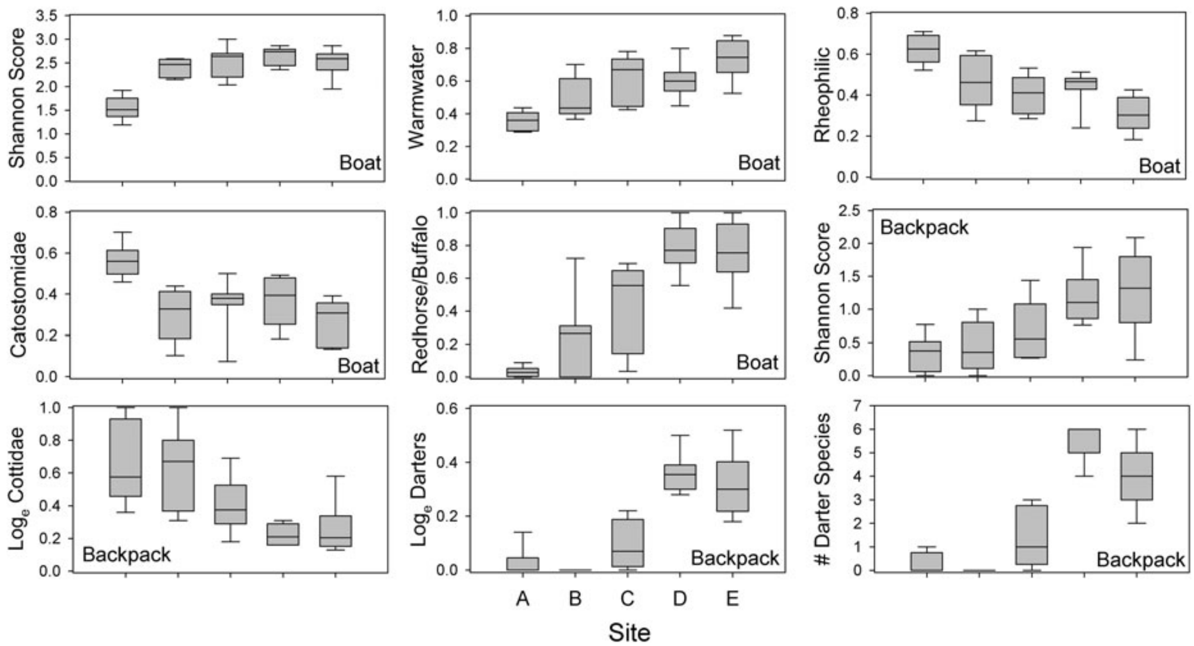

FIGURE 4. Responses observed across all samples at five sites for the nine metrics comprising the multimetric index developed for the Caney Fork River (horizontal line within box = median; box = central 75% of observations; whiskers = range of the response). The gear type (boat electrofishing or backpack electrofishing) used to collect the data for each metric is indicated at the bottom of each panel. See Table 5 for a full description of the metrics (*y*-axis labels).

sound multimetric index in the absence of an appropriate reference site (Hallett et al. 2012). Due to the persistence of Center Hill Dam and the artificial nature of the tailwater, we could not define or identify an appropriate reference condition. However, the multivariate approach enabled us to use data collected from relatively few sites located entirely within the studied system in order to infer the attributes of resident fish assemblages that were structured by abiotic conditions in the tailwater. This study provides ample evidence that the Caney Fork River contains adaptive fish assemblages that are structured in response to varying tailwater conditions.

We incorporated two sampling gear types because the varied habitat in the Caney Fork River prevented us from effectively sampling all available habitats with a single gear type. Samples that were collected with each gear type provided a different perspective on assemblage composition. Other researchers have noted the necessity of combining gear types to obtain suitable samples with which to develop indices for systems with varied habitat (Scott 1999; Drake and Pereira 2003; Pearson et al. 2011). Although the summed scores derived from backpack and boat electrofishing metrics were related, the higher correlation of combined metrics with the environmental data indicates that the overall assemblage structure can be best described using a combination of metrics that encompasses the different sizes and species of fish collected with each gear type. Aggregation of metrics derived from the two sampling protocols also makes the multimetric index more robust to variability due to condition-specific biases associated with the sampling efficiency of the two gear types and random effects related to fish behavior.

Patterns of increasing species diversity with increased distance from Center Hill Dam may also be influenced by migrants from tributaries to the Caney Fork River below the dam and from the Cumberland River. It is understood that dams interrupt the connectivity of river systems and affect the migration of fishes (Freeman et al. 2007). In a concurrent study (Brooke 2013), 53 fish species were identified from three tributaries to the Caney Fork River below Center Hill Dam, and many of those species also occurred in the Caney Fork River during our sampling. It is possible that populations of certain fishes in the tributaries and main stem are demonstrating source–sink metapopulation dynamics (Schlosser and Angermeier 1995). The lack of any seasonal effect on species richness estimates from backpack electrofishing indicates that any small-bodied migrants from tributaries are persistent in the Caney Fork River assemblage. The increase in species richness that was observed in spring and summer boat electrofishing samples may be evidence of temporary immigration into the Caney Fork River during those seasons. The species that were only encountered during those seasons were mainly large-river fishes (e.g., Bigmouth Buffalo *Ictiobus cyprinellus* and Paddlefish *Polyodon spathula*) and were probably migrating from the Cumberland River for spawning purposes. Because the metric selection procedure incorporated data collected during all four calendar seasons, most metrics should reflect persistent attributes of the community and the index should be only minimally affected by migrants.

However, variability among scores can be reduced by standardizing the time of year in which monitoring is conducted. Temporary and permanent migrants are important components of fish assemblages in tailwaters, and their presence reflects the suitability of local habitat conditions.

Because the disturbance effects of the dam on flow, temperature, and water chemistry are attenuated with increasing distance downstream of the dam, any spatial change in tailwater conditions should be reflected by a related shift in assemblage structure and composition (Ward and Stanford 1983; Kinsolving and Bain 1993; Olden and Naiman 2010), and metric-related trends observed in our study were consistent with this expectation. Longitudinal increases in organismal diversity (i.e., richness and evenness) below hydroelectric dams have been noted for other river systems (Kinsolving and Bain 1993; Hunter 2003). In the Caney Fork River, native fish assemblages at the upstream-most sites (i.e., those most impacted by hydroelectric operations) had relatively low scores for Shannon's diversity index, reflecting the presence of fewer species and the domination of the assemblages by Banded Sculpins and Spotted Suckers. Highly altered daily and annual thermal regimes are typical of tailwaters and affect the distribution of fishes by influencing movement, survival, and reproduction (Irwin and Freeman 2002; Olden and Naiman 2010). In the Caney Fork River, sites exhibiting higher average summer temperatures supported a greater proportion of warmwater fishes throughout the year. High water velocity can displace fish, relocating them to unsuitable habitat where they may be more susceptible to predation (Harvey 1987). In the Caney Fork River, water velocity during hydropower generation and the increase in velocity accompanying hydropower generation (i.e., the ramping rate) are highest at upstream sites, and correspondingly the proportion of the assemblage identified as rheophilic species was highest near the dam. Each of the five taxonomic metrics also reflected predictable impacts of the dam on specific groups of taxa (Tsai 1972; Gore et al. 1990; Emery et al. 1999; Scott 1999).

The multimetric index that we developed should be useful for monitoring fish assemblages in the Caney Fork River downstream of Center Hill Dam. This index does not provide an explicit measurement of biotic integrity because it was not developed based on a reference condition. However, we expect that changes in tailwater conditions over time will be reflected by a related shift in fish assemblages and a corresponding shift in index scores (i.e., if dam operations result in better tailwater conditions, then index scores across all sites should increase). Because our multimetric index was explicitly developed for the Caney Fork River by using data collected from this river, it may not be widely applicable to other systems. However, the analytical procedure used to derive this index is highly transferable and can be used to develop and test indices that are specific to any system. The procedure is particularly useful for specifically calibrating indices for radically altered systems where established indices are not applicable, an appropriate reference site does not exist, and traditional index development techniques are not appropriate.

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