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ELECTROFISHING IN BOATABLE RIVERS: DOES SAMPLING DESIGN AFFECT BIOASSESSMENT METRICS?

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Abstract. Data were collected from 60 boatable sites using an electrofishing design that permitted comparisons of the effects of designs and distances on fish assemblage metrics. Sites were classified *a priori* as Run-of-the-River (ROR) or Restricted Flow (RF). Data representing four different design options (i.e., 1000 and 2000 m for both single and paired banks) were extracted from the dataset and analyzed. Friedman tests comparing metric values among the designs detected significant differences for all richness metrics at both types of sites and for catch per unit effort and percent tolerant species at ROR sites. Richness metrics were generally higher for the two 2000-m designs than for the two 1000-m designs. When plotted against cumulative electrofishing distance, the percent change in metrics declined sharply within approximately 1000 m, after which metrics usually varied by less than 10%. These data demonstrate that designs electrofishing 1000 m of shoreline are sufficient for bioassessments on boatable rivers similar to those in this study, regardless of whether the shoreline is along a single bank or distributed equally among paired banks. However, at sites with depths greater than 4 m, it may be advisable to employ nighttime electrofishing or increase day electrofishing designs to 2000 m.

Keywords: bioassessment, biocriteria, biological criteria, boatable, electrofishing, fish surveys, large, monitoring, rivers

1. Introduction

Since the U.S. Environmental Protection Agency (EPA) endorsed the use of biological indicators to assess environmental conditions and ecological health (U.S. EPA, 1990a,b), there has been tremendous growth in their use among agencies that assess aquatic resources (Davis *et al.*, 1996). Fish assemblages are among the indicators frequently used in bioassessments (Barbour *et al.*, 1999; Simon, 1999; McCormick and Peck, 2000), and the advantages and disadvantages of using fish assemblages for bioassessments have been discussed extensively (Hocutt, 1981; Karr, 1981; Reynolds, 1983; Fausch *et al.*, 1990; Yoder and Rankin, 1995; Bayley and Dowling, 1993; Barbour *et al.*, 1999; Simon, 1999; McCormick and Peck, 2000). In addition, correlations have been successfully demonstrated between fish indices of biotic integrity (IBIs) and human activities that influence streams and rivers (e.g., Karr *et al*., 1985; Berkman *et al*., 1986; Leonard and Orth, 1986; Ohio EPA, 1987a, 1999; Steedman, 1988; Karr, 1991; Yoder and Rankin, 1995). Although IBIs have been widely applied in wadeable streams and are slowly gaining popularity for the assessment of large rivers, their application in large rivers has been relatively limited (Hughes and Gammon, 1987; Oberdorff and Hughes, 1992; Simon, 1999; Lyons *et al*., 2001).

Electrofishing is commonly used to collect fish for bioassessments because it is widely considered to be the single most comprehensive and effective method for collecting fishes in streams and rivers (Vincent, 1971; Gammon, 1973, 1976; Novotny and Priegel, 1974; Ohio EPA, 1987b; Davis *et al*., 1996; Barbour *et al*., 1999; Simon and Sanders, 1999). Although a wide variety of field electrofishing designs are currently in use, studies that compare these designs are limited. Variables that may be important in evaluating performance characteristics of a given field design include the spatial extent and relationship of habitat features, the spatial coherence of an assemblage, the local (alpha) diversity, and spatial and temporal distributions of fishes.

This study was undertaken to: (1) compare commonly used boat-based electrofishing designs; (2) determine the sampling distance at which the values of common bioassessment metrics begin to stabilize; and (3) study the influence of physical site characteristics on the designs. The compared designs are quantitative and serve the purpose of supporting bioassessment and monitoring activities. The primary goal of this study was to develop a Large River Bioassessment Protocol (LR-BP) that will provide states, regions, tribes, and other federal agencies needing methods with the ability to effectively use fish assemblages to evaluate the condition of large rivers, an integral part of achieving water quality for all surface waters.

2. Methods

2.1. Study area

We collected data during a single season (summer, 1999) from the Great Miami ($n = 20$), Scioto ($n = 20$), Kentucky ($n = 10$) and Green rivers ($n = 10$), each of which is a major tributary of the Ohio River (Figure 1). These sites were classified *a priori* into two general types of sites. The first type of sites were those that were either free flowing or associated with low-head dams that store rather than regulate waters. These sites were termed Run-of-the-River (ROR) sites. The second type of site sampled was that heavily influenced by navigational lock-and-dam structures built to support commercial traffic. These were termed Restricted Flow (RF) sites.

The Great Miami and Scioto rivers flow principally through agricultural and forested lands with some sections flowing through major urban and industrial corridors before reaching the Ohio River. Both rivers have sections with exposed riffles and rapids and sections with restricted flow, but are both generally shallower than the Kentucky and Green rivers and, therefore, largely ROR sites.

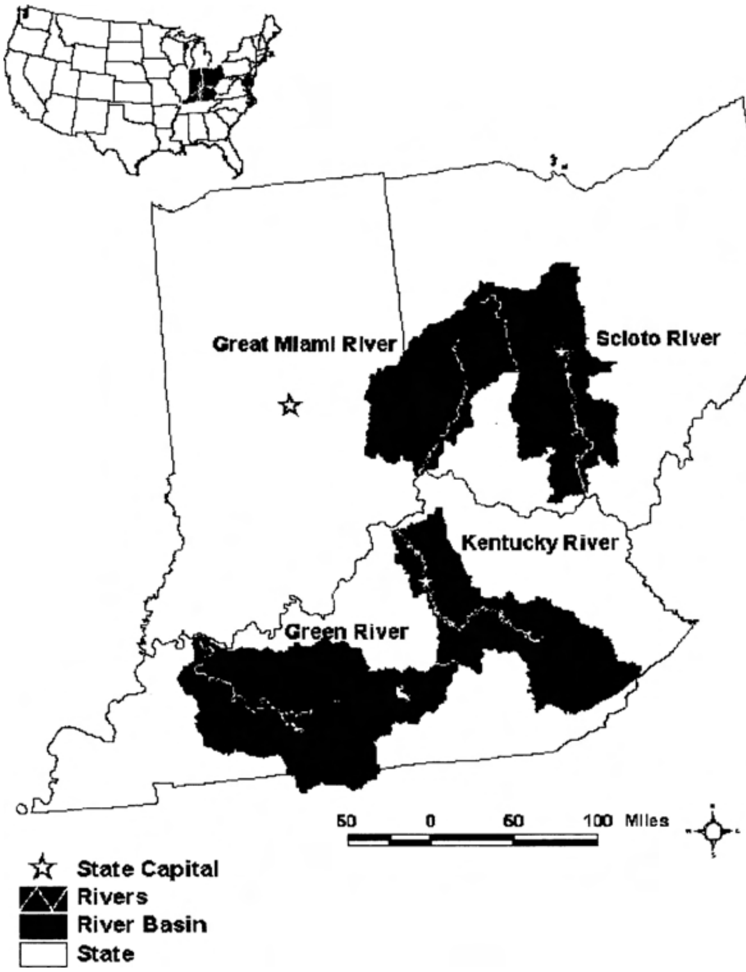

Figure 1. Sample sites on the Great Miami, Scioto, Kentucky and Green rivers, all major tributaries in the Ohio River basin.

The Kentucky River has a series of 14 lock-and-dam structures that span the length of the mainstem, rendering it completely impounded. The mainstem of the Green River has six lock-and-dam structures, the most upstream of which is at river kilometer (rkm) 292.5. Above the influence of this dam, the river is free flowing with significant areas of exposed riffles and rapids until rkm 330.1, where a dam for a large reservoir is located. As a result of impoundment, most sections of the Kentucky and Green rivers are much deeper than those of the Great Miami and Scioto rivers and therefore RF sites. However, those above rkm 292.5 on the Green River are ROR sites. Additional physical attributes of each basin and dominant land uses are summarized in Table I.

Sampling locations on the Great Miami and Scioto rivers were selected from existing Ohio EPA sampling sites. Sites for the Kentucky and Green rivers were

TABLE I
Descriptive characteristics and dominant land uses of basins

River	Length (km)	Drainage area (km^2)	Average stream gradient (cm/km)	Predominant land use and influences	Physiographic regions
Great Miami	274	13,947	73.9	*Upper*: agriculture *Middle*: urban, industrial, dams, channelization *Lower*: agriculture, gravel mining	Till plains and interior plateau (lowest portion)
Scioto	370	16,879	43.6	*Upper*: agriculture, some urban *Middle*: gravel and sand mining *Lower*: forested, limited agriculture	*Upper*: till plains *Middle*: glaciated and unglaciated Allegheny plateaus *Lower*: unglaciated Allegheny plateau
Kentucky	410	18,130	13.3	14 locks and dams *Upper*: forestry, coal mining, limited agriculture *Middle*: agriculture, urban *Lower*: forest and agriculture	*Upper*: Eastern Kentucky Coal Field *Middle and lower*: bluegrass
Green	532	23,040	NA	*Upper*: agriculture *Middle*: agriculture, locks and dams *Lower*: agriculture, locks and dams, strip mining	*Upper and middle*: Mississippian Plateau *Lower*: Western Kentucky Coal Field

chosen based on known boat ramp locations and a review of land-use maps. Sites were well distributed along the length of the main stem of each river and included a mixture of habitat types. For site-specific reach placement, we attempted to avoid obvious stressors, such as major outfalls, stream confluences, and bridges, because the effects of these features were not the focus of this study and their inclusion would influence comparisons among field designs.

2.2. Electrofishing methods

An electrofishing design was devised that permitted the concomitant collection of data to compare the effects of four designs and distance alternatives on metrics in a single pass of the study area (Figure 2). The design included electrofishing on both banks and consisted of 13 intermediate fish processing points.

On one bank, the distance electrofished was 40 times the wetted width (after McCormick and Hughes, 2000) to a maximum of 2000 m. Based on our experiences and personal communications with local, state, regional and national assessment communities, 2000 m was considered to be the longest logistically acceptable electrofishing distance a program could consider for rivers of this type. Reach lengths exceeding 2000 m may also have encompassed ranges of influences that were too broad to be synoptic. The total shore distance on this bank was divided into 10 zones (Figure 2) delineated by transects spanning the width of the stream and labeled "A" to "K" (after McCormick and Hughes, 2000). The downstream endpoint of the

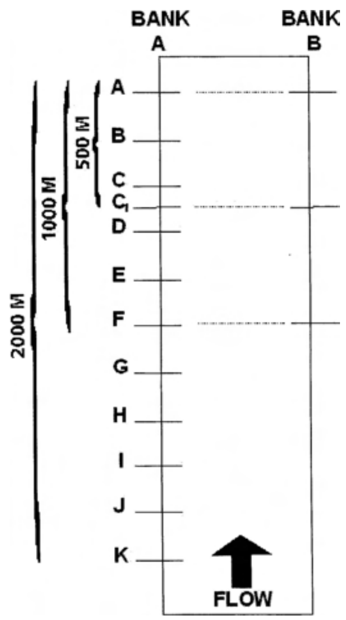

Figure 2. Electrofishing design used in study.

sample reach was transect “A”. From that point, each of the remaining transects was a distance equal to 1/10 of the designated reach length upstream of the previous transects. In most cases, this distance was 200 m. Electrofishing began at transect “K” and fish were processed at each transect “J” to “A” and at 500 m upstream of transect “A”. When the river was greater than 50 m wide, this additional processing point was designated as transect “C1”. On the opposite bank, 1000 m were electrofished with collected fish being processed at points that were 500 and 1000 m upstream of transect “A”.

Electrofishing was conducted following the methods of McCormick and Hughes (2000). Sampling proceeded in a downstream direction along the main-channel riparian habitat of each bank at a speed near or, if velocities were low, slightly exceeding the river velocity (Reynolds, 1983; Ohio EPA, 1989; McCormick and Hughes, 2000). At each of the processing points, all fish were identified and then retained in holding nets. After electrofishing had been completed on both banks, all fish were released with the exception of representative vouchers of specimens that needed to be identified in the laboratory.

All sampling was conducted during the low and stable-flow index period of mid-June to early October (Ohio EPA, 1989; Lazorchak *et al*., 2000; Moulton *et al*., 2002). This index period has been suggested and widely accepted based on the assumption that it increases the likelihood that samples throughout a study unit can be collected under similar flow conditions (Gilliom *et al*., 1995).

Data representing four different design options were extracted from the electrofishing dataset. The first design (SB-1000) used data collected along a single bank for 1000 m. The second design (PB-1000) used data collected along 500 m of paired banks (1000 m total shoreline). The third design (SB-2000) used data collected along a single bank for 2000 m, and the fourth design (PB-2000) used data collected along 1000 m of paired banks (2000 m total shoreline) (Figure 2).

All sample reaches with wetted widths less than 50 m were excluded from the analysis dataset. Consequently, all sites included in the dataset had reach lengths of 2000 m on one bank, 1000 m on the opposite bank and 13 processing points across the reach. This resulted in uniform design comparisons across all sites.

2.3. Physical habitat

To study the influence of physical site characteristics on the comparisons, habitat data were collected using the methods designed by Kaufmann (2000) for use in the EMAP-SW large river projects. Protocols of this approach are divided into channel and riparian/littoral measurements, and are integrated across 11 transects (A–K) for reach characterization. Transects used for electrofishing were used for the collection of these data. Habitat assessment techniques of these protocols are weighted toward quantitative measures. Physical habitat variables were calculated using descriptions and formulas in Kaufmann *et al*. (1999).

2.4. Analysis

To validate our *a priori* classification of sites as ROR or RF, we described natural variation in the physical habitat characteristics of sites using principal components analysis (PCA). Variables included in the analysis were mean shore depth, mean thalweg depth, range of thalweg depth, mean wetted width, bankfull height, mean temperature, mean width–depth ration, percent sand, percent gravel, percent cobble and larger substrate in thalweg, and number of substrates at a site. The first two principal components were plotted to look for separation of sites by impoundment class.

To compare the relative performance of the four-electrofishing designs tested in this study, we analyzed 12 fish metrics. These metrics were: (1) catch-per-unit-effort (CPUE); (2) number of taxa (excluding exotic species); (3) number of sunfish taxa; (4) number of sucker taxa; (5) number of intolerant taxa; (6) percent round-bodied suckers; (7) percent omnivores; (8) percent insectivores and invertivores; (9) percent carnivores; (10) percent tolerant individuals; (11) percent simple lithophils and (12) percent individuals with deformities, eroded fins, lesions, and tumors (DELT anomalies). These metrics were selected because of their wide use as effective metrics in the bioassessment of boatable rivers (Ohio EPA, 1987b; Simon, 1992, 1994). Multiple sources were consulted to determine the trophic status of collected species, and the designations used (Appendix) conformed largely to summaries in Barbour *et al*. (1999).

A nonparametric, repeated measures analysis of variance (i.e., the Friedman test) with associated multiple comparison procedures (Hollander and Wolfe, 1999) was used to compare electrofishing designs based on metric values. The Friedman test was used because most metric distributions were neither normal nor transformable to normality.

To examine the effect of electrofishing distance on metrics, we ran Monte Carlo simulations, which minimized the effect of influential sections within a sampling reach. In each simulation, the 10 individually processed, 200-m sections electrofished along a single bank within a site were randomly ordered. Then, each metric was calculated for progressively longer distances encompassing from 1 to 10 sections. This process was repeated 100 times for each site. For each metric, we calculated the percent change in metric value between successively longer sections of river. We plotted the mean percent change in metric value against the distance electrofished for each site as a way to identify patterns across sites. These analyses were run separately for the ROR and RF sites.

3. Results

Data were collected at 60 river sites. At each of these sites, fish were collected and processed at sub-sites to produce individual datasets for analysis. Seven sites were excluded because of anomalous or missing physical habitat or fish information. An additional four sites with wetted widths less than 50 m were excluded to allow for

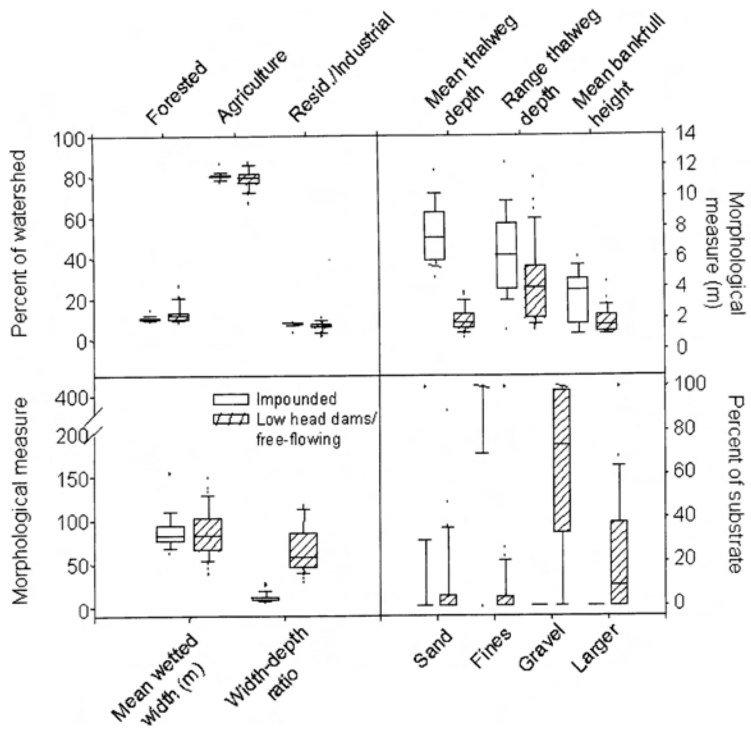

Figure 3. Physical site characteristics of sample sites used in analysis.

more straightforward statistical comparison of designs. For comparisons among designs, data from 49 sites and 637 individual datasets remained for analysis. Physical site characteristics included in analysis are summarized in Figure 3. Eighty-nine species in 15 families were identified from the 28,100 fish collected (Appendix).

The first axis of the PCA on physical habitat variables explained approximately 37% of the variation (Table II; Figure 4). The two variables with the highest loadings on the first axis were mean width–depth ratio and mean thalweg depth. Sites separated along the first PCA axis, corresponding to sites having a mean thalweg depth of more than 4 m (RF sites) or less than 4 m (ROR sites). These results validated our *a priori* separation of sites into ROR and RF sites and justified separate analyses by impoundment class.

Friedman tests comparing metric values among the four designs detected a significant difference for CPUE and percent tolerant species at ROR sites (Table III). Box plots comparing metric distributions among designs are presented in Figure 5. Significant differences were also detected among designs for all richness metrics at both ROR and RF sites, although the differences were not always detected in the multiple comparisons (e.g., number of sunfish taxa and number of intolerant species at RF sites). The only percentage metric with a significant difference among designs was percent tolerant individuals at ROR sites. However, the

TABLE II
Principal components analysis weights of physical habitat variables ($N = 48$; one site excluded because of missing substrate data point)

Variable	Axis 1[a]	Axis 2[b]
Mean wetted width	0.009	0.091
Bank full height	0.323	0.244
Mean water temperature	0.338	–0.003
Mean thalweg depth	0.490	–0.051
Mean width-depth ratio	–0.435	0.104
Range of thalweg depth	0.291	0.157
Number of substrates	–0.291	0.390
Percent sand in thalweg	–0.052	0.760
Percent gravel in thalweg	–0.381	–0.355
Percent cobble and larger in thalweg	–0.184	0.196

[a]Eigenvalues: $\lambda = 3.70$; % variance: 37.0%.
[b]Eigenvalues: $\lambda = 1.40$; % variance: 14.0%.

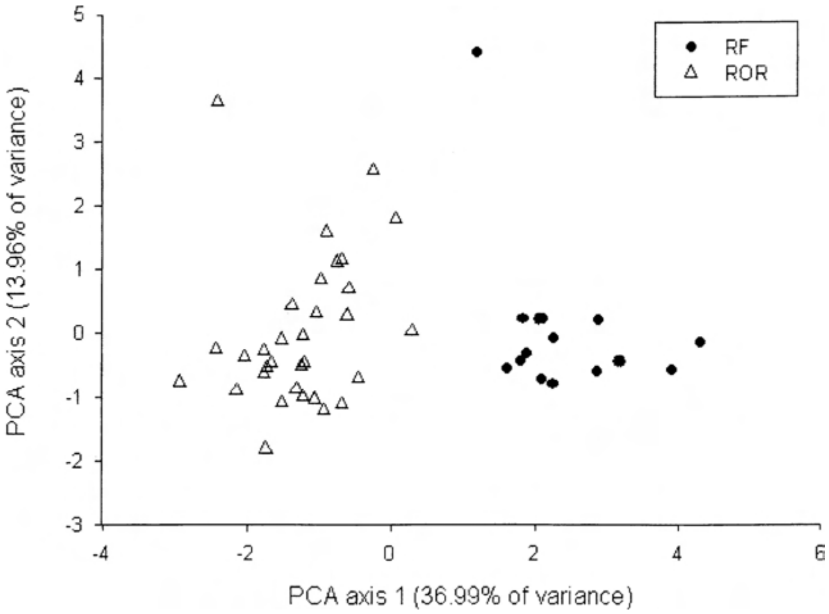

Figure 4. Principle component analysis showing the separation of sites along the first axis, which corresponded to grouping sites as having a mean thalweg depth of greater than 4 m (RF sites) or less than 4 m (ROR sites).

metric values were relatively low and likely have little interpretive value for this study.

In general, the richness metric values of the PB-2000 and SB-2000 designs were higher than those of the SB-1000 and PB-1000 designs. No significant differences

TABLE III
Comparison of metric values among four electrofishing designs (by river classification group) using Friedman tests (bolded if significant at 0.05) and multiple comparisons ($\alpha = 0.05$)

Metric	Group	S'	p-value	SB-1000	PB-1000	SB-2000	PB-2000
CPUE	**ROR**	**13.65**	**0.003**	**AB**	**B**	**A**	**AB**
	RF	5.67	0.129				
No. taxa	**ROR**	**71.77**	**<0.001**	**A**	**A**	**B**	**B**
	RF	**41.00**	**<0.001**	**A**	**A**	**B**	**B**
No. sunfish taxa	**ROR**	**24.56**	**<0.001**	**AB**	**A**	**CB**	**C**
	RF	**13.22**	**0.004**	A	A	A	A
No. sucker taxa	ROR	**40.41**	**<0.001**	**A**	**A**	**B**	**B**
	RF	**21.55**	**<0.001**	**A**	**A**	**B**	**B**
No. intolerant taxa	**ROR**	**42.22**	**<0.001**	**A**	**A**	**B**	**B**
	RF	**8.39**	**0.039**	A	A	A	A
% Round-bodied suckers	ROR	0.72	0.868				
	RF	1.69	0.639				
% Omnivores	ROR	4.39	0.222				
	RF	0.89	0.829				
% Insectivores + invertivores	ROR	3.93	0.269				
	RF	0.73	0.865				
% Carnivores	ROR	5.05	0.168				
	RF	1.00	0.801				
% Tolerant	**ROR**	**11.36**	**0.010**	**A**	**B**	**AB**	**AB**
	RF	1.81	0.613				
% Simple lithophils	ROR	3.12	0.374				
	RF	1.76	0.624				
% DELT anomalies	ROR	4.46	0.216				
	RF	7.57	0.056				

were detected between designs of equal shoreline distance electrofished for any of the richness metrics (i.e., SB-1000 *vs*. PB-1000 and SB-2000 *vs*. PB-2000).

For the examination of the effect of sampling distance on metrics, an additional five sites were excluded due to variance in transect delineation. These included sites where logistical constraints did not permit the delineation of transects at their assigned locations and some suffering from human error. Forty-four sites remained for inclusion in the analysis.

Plots of percent change in metrics by the distance electrofished along one bank demonstrated a sharp decline in changes in metrics within approximately 1000 m in ROR and RF sites (Figure 6). After 1000 m, the degree of variation in metric value was usually less than 10%.

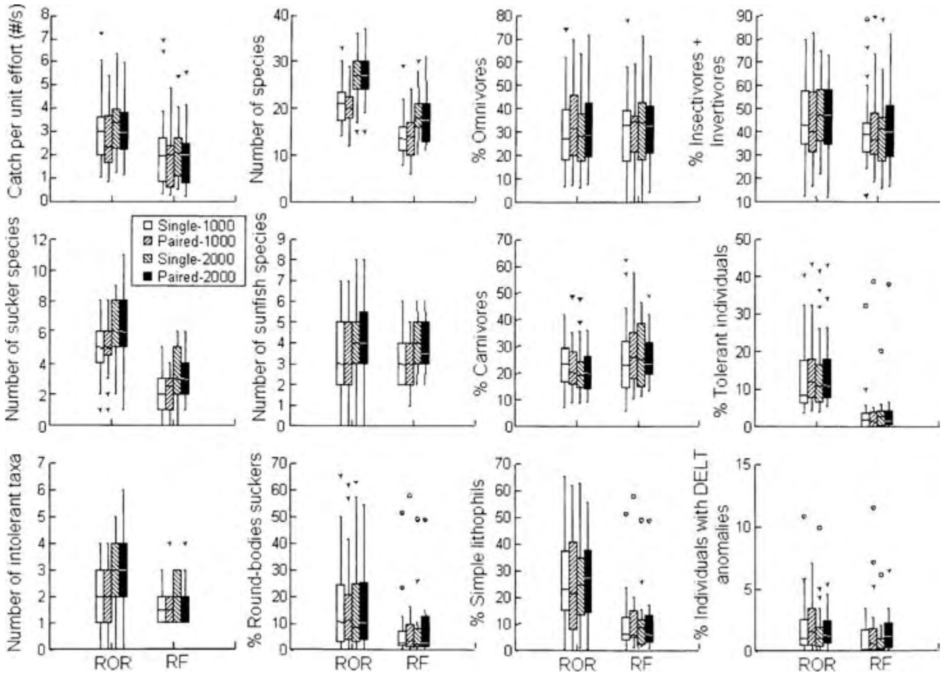

Figure 5. Box and whisker plots of mean metrics values compared across four electrofishing designs.

Percent change in the percent round-bodied suckers metric was slightly more variable with distance, especially in RF sites. However, the overall percent change was relatively low, usually below 15% for ROR and RF sites within 1000–1200 m, respectively. There was very little change in percent omnivores, percent carnivores, and percent insectivores and invertivores beyond 600 m for sites in either impoundment class. Plots for RF sites were more variable than those for ROR sites, particularly for number of sucker taxa.

4. Discussion

4.1. Design comparisons

The designs compared in this study are quantitative and have the purpose of supporting bioassessment and monitoring activities of states, regions, tribes and other agencies. They have been designed to collect samples that are as unbiased and representative as possible within the logistical realities of fieldwork and constraints of time and budget and are indicative of the ecological condition of a site when compared to sites of known condition. This sampling approach is not appropriate for qualitative studies that strive to maximize the number of species as a measure of local (alpha) diversity, although data collected using quantitative methods could be used to supplement qualitative investigations.

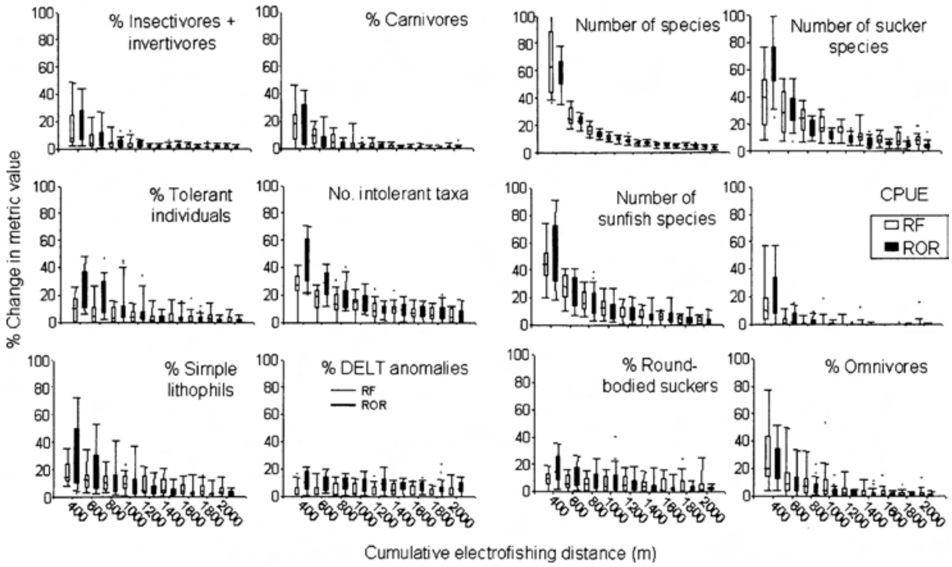

Figure 6. Plots of percent change in metrics by the number of sections electrofished along one bank.

A structured, quantitative sampling approach seeks to be as consistent as possible through time and space, and be scientifically sound. A sampling approach that is more qualitative could be considered to be consistent in that the field scientist seeks to collect as many species as possible as a measure of local diversity, but the ability to maximize species collection can vary greatly as a function of experience, enthusiasm, and attention to detail, as well as logistical constraints. Additionally, the structured and consistent nature of a quantitative sampling approach offers the feature of equal time allocation at sites, a desirable feature for planning and budgeting.

Most standardized electrofishing sampling designs for flowing waters are either fixed-distance or proportional-distance approaches (Barbour *et al*., 1999). The fixed distance selected may be arbitrary, based on features of an overall study design, or based on species accumulation curves. When species accumulation curves are used, the length of stream that must be electrofished before the curve of an encountered species reaches an asymptotic point, or approaches it so that the effort required to collect additional species is not justified, must first be determined at a pool of sites (Penczak and Zalewski, 1973, 1981; Angermeier and Karr, 1986; Angermeier and Schlosser, 1989; Yoder and Smith, 1999). Then, the fixed distance in which the consistently collected proportion of the population that is deemed necessary for bioassessment purposes can be determined. Fixed-distance designs have the logistical advantages of controlling for the total effort expended at a single reach and limiting the number of field-based decisions, because field personnel need only know a single point to establish the electrofishing zone.

Proportional-distance methods, as described by Lyons (1992), may be "established arbitrarily and based solely on physical features of the stream segment, such as a set number of riffle-pool sequences or a multiple of the mean stream width", or set based on species curves (e.g., Karr *et al.*, 1986; Lazorchak *et al.*, 2000). One example of this approach was demonstrated by Lyons (1992) where it was concluded that a stream reach of 35 times the mean stream width, or a length equal to three complete riffle-pool sequences, ensured that the cumulative number of species captured approached or exceeded an asymptotic level. Other examples recommend sampling for a distance equal to either 40 or 100 times the wetted width (McCormick and Hughes, 2000) or 85 times the wetted width (Hughes *et al.*, 2002). Although scientifically sound for their intended application, logistical issues arise when such designs are applied at sites differing from those for which they were intended (e.g., raftable streams; Hughes *et al.*, 2002) or where the river is excessively wide. This problem can be largely overcome by establishing a maximum sample reach distance (Moulton *et al.*, 2002).

Another issue encountered with proportional-distance methods is the variability associated with determination of the width of the river that will be used as the multiplier to establish site total reach length. Not only do individuals disagree on how and where this value should be determined, but fluctuations in flow status among repeat visits to a site also create discrepancies during analysis. While neither of these issues negates the validity or utility of this approach, they are issues that must be acknowledged.

We conducted this study to determine the electrofishing sampling distance required to produce robust measures of condition in boatable rivers of the study region. The electrofishing design we used for this study permitted the concomitant collection of data for two purposes in a single pass of the study area. This resulted in some datasets being subsets of others, but avoided the problem of observed differences being the result of differences among the river sections sampled for each design. Thus, when examining the results of the richness metrics, the significant differences detected between the PB-2000 and SB-2000 designs when compared to the SB-1000 and PB-1000 are logical. An increased electrofishing distance increases the likelihood of encountering species that occur less frequently or less randomly in the river. However, the importance of these results is that in both the ROR and RF sites, the richness metric results were not significantly different among electrofishing designs of equal shoreline distance (i.e., SB-1000 *vs*. PB-1000 and SB-2000 *vs*. PB-2000). This could lead to the conclusion that total shoreline distance electrofished has more bearing on results than whether a design is single- or paired-banked. However, this conclusion is not supported by the findings for CPUE.

The Friedman test of CPUE metric values at ROR sites detected significant differences among designs, but contrary to the richness metrics, shoreline distance does not explain these results. However, if the mean CPUE values by design are ordered by increasing magnitude (Table IV), we see the trend that as the total

TABLE IV
Mean CPUE metric values at ROR sites of tested electrofishing designs ordered in increasing mean magnitude

Design	SB-1000	PB-1000	PB-2000	SB-2000
Total shoreline electrofished (m)	1000	1000	2000	2000
Mean CPUE value	2.2	3.0	3.0	3.5
Linear river distance electrofished (m)	500	1000	1000	2000

number of linear river meters (not the total number of shore-line meters) sampled by the design increases, the CPUE increases. We explored the possibility that these findings could be explained by the increased likelihood of encountering shoaling species (e.g., gizzard shad *Dorosoma cepedianum* and emerald shiners *Notropis atherinoides*) that are often sporadically collected in large numbers (Simon and Sanders, 1999), but exclusion of these species from the analysis did not change the significance of results. Other possible explanations for this observation are still being explored.

The percentage metrics were very consistent across designs. The only significant difference detected was for percent tolerant species at ROR sites. No logical explanation for the detected differences has been determined. However, the metric values are relatively low and likely have little interpretive value. The consistent performance of the percentage metrics across designs does suggest that they may be of the highest utility when attempting to make future comparisons between different designs.

4.2. Distance effects

Examination of the effect of distance on metric values showed that at a reach span of approximately 1000 m along one bank, metrics changed relatively little with additional electrofishing. In addition, when only considering ROR sites, most metrics showed very little change between electrofishing 800 and 1000 m.

At the RF sites, some metrics (e.g., percent round-bodied suckers and number of sucker taxa) did not level off as well as they did for the ROR sites. This observation is likely a result of the diel movements of some fish species from near-shore during the night, to off-shore or deeper waters during the day (Sanders, 1991, and cited references). As a result, the daytime collection of such species may be sporadic and limited to individuals on exploratory forays. Our study used a daytime main-channel riparian habitat electrofishing design, and would, therefore, be susceptible to these realities. The sucker species seem to be especially prone to such movements (Sanders, 1991), which is evident in our results. Consequently, the daytime collection of species prone to diel movements at RF sites could be considered disruptive to analyses. At a minimum, metric values dependent on such species should be interpreted with caution.

Unfortunately, capturing this diel variation with night electrofishing is problematic. Night electrofishing can produce undue fatigue, pose possible safety risks, or be fiscally unfeasible (Graham, 1986) and is usually avoided if satisfactory results can be obtained through daytime sampling. Our data suggest that in these systems, at depths greater than 4 m, the diel movements of fish significantly impact the quality of daytime electrofishing results to the extent that the consideration of night electrofishing is justified. A depth criterion comparable to this is likely applicable to other river systems.

After electrofishing 180 km among four rivers, collecting 28,100 fish, and running 52,800 simulations, we arrived at the following conclusions.

1) Fixed-distance electrofishing designs of logistically practical and safe distances are sufficient for bioassessments on boatable river sites like those in this study.
2) Depth plays a critical role in the response of fish assemblages to electrofishing and the resulting metric values. For example, at sites less than 4 m, a daytime main-channel, border design that electrofishes 1000 m along a single bank or 500 m on paired bank is sufficient to characterize sites for bioassessment purposes. At sites greater than 4 m, results were more variable.
3) At sites greater than 4 m, we suggest that a switch from daytime to night electrofishing be considered. If night electrofishing is not feasible, we suggest increasing the electrofishing distance at these sites to a 1000-m paired-banks design or a 2000-m single-bank design. In addition, metrics based on fish species prone to diel movements should be interpreted with caution.

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Appendix: Fishes collected during the study "trophic status" and "special designation" classifications follow Barbour *et al.* (1999)

Latin name	Common name	Trophic status	Special designation
Petromyzondidae	Lampreys		
Lampetra appendix	American brook lamprey	Filter	
Ichthyomyzon bdellium	Ohio lamprey	Piscivore	
Ichthyomyzon unicuspis	Silver lamprey	Piscivore	
Lepisosteidae	Gars		
Lepisosteus osseus	Longnose gar	Piscivore	
Lepisosteus oculatus	Spotted gar	Piscivore	
Lepisosteus platostomus	Shortnose gar	Piscivore	
Amiidae	Bowfins		
Amia calva	Bowfin	Piscivore	
Clupeidae	Herrings		
Alosa chrysochloris	Skipjack herring	Piscivore	
Dorosoma cepedianum	Gizzard shad	Omnivore	
Hiodontidae	Mooneyes		
Hiodon tergisus	Mooneye	Insectivore	
Esocidae	Pikes		
Esox lucius	Northern pike	Piscivore	
Esox masquinongy	Muskellunge	Piscivore	
Cyprinidae	Minnows		
Cyprinus carpio	Common carp	Omnivore	Exotic
Carassius auratus	Goldfish	Omnivore	Exotic
Notemigonus crysoleucas	Golden shiner	Omnivore	
Semotilus atromaculatus	Creek chub	Generalist	
Nocomis micropogon	River chub	Insectivore	
Notropis rubellus	Rosyface shiner	Insectivore	
Notropis atherinoides	Emerald shiner	Insectivore	
Notropis stramineus	Sand shiner	Insectivore	
Notropis volucellus	Mimic shiner	Insectivore	
Notropis blennius	River shiner	Insectivore	
Notropis boops	Bigeye shiner	Insectivore	
Notropis photogenis	Silver shiner	Insectivore	
Phenacobius mirabilis	Suckermouth minnow	Insectivore	
Campostoma anomalum	Central stoneroller	Herbivore	
Pimephales notatus	Bluntnose minnow	Omnivore	
Pimephales vigilax	Bullhead minnow	Omnivore	
Cyprinella spiloptera	Spotfin shiner	Insectivore	
Cyprinella whipplei	Steelcolor shiner	Insectivore	

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Latin name	Common name	Trophic status	Special designation
Erimystax dissimilis	Streamline chub	Insectivore	
Erimystax x-punctatus	Gravel chub	Insectivore	
Luxilus chrysocephalus	Striped shiner	Insectivore	
Lythrurus ardens	Rosefin shiner	Insectivore	
Catostomidae	Suckers		
Catostomus commersoni	White sucker	Omnivore	Round-bodied
Carpiodes cyprinus	Quillback	Omnivore	
Carpiodes carpio	River carpsucker	Omnivore	
Carpiodes velifer	Highfin carpsucker	Omnivore	
Moxostoma macrolepidotum	Shorthead redhorse	Insectivore	Round-bodied
Moxostoma anisurum	Silver redhorse	Insectivore	Round-bodied
Moxostoma carinatum	River redhorse	Insectivore	Round-bodied
Moxostoma duquesnei	Black redhorse	Insectivore	Round-bodied
Moxostoma erythrurum	Golden redhorse	Insectivore	Round-bodied
Hypentelium nigricans	Northern hog sucker	Insectivore	Round-bodied
Cycleptus elongatus	Blue sucker	Insectivore	Round-bodied
Ictiobus bubalus	Smallmouth buffalo	Insectivore	
Ictiobus cyprinellus	Bigmouth buffalo	Insectivore	
Ictiobus niger	Black buffalo	Insectivore	
Minytrema melanops	Spotted sucker	Insectivore	Round-bodied
Ictaluridae	Catfishes		
Ictalurus punctatus	Channel catfish	Piscivore	
Noturus flavus	Stonecat	Insectivore	
Noturus miurus	Brindled madtom	Insectivore	
Pylodictis olivaris	Flathead catfish	Piscivore	
Ameiurus natalis	Yellow bullhead	Insectivore	
Ameiurus nebulosus	Brown bullhead	Insectivore	
Poecillidae	Mosquitofishes		
Gambusia affinis	Western mosquitofish	Insectivore	Exotic
Atherinidae	Silversides		
Labidesthes sicculus	Brook silverside	Insectivore	
Cottidae	Sculpins		
Cottus carolinae	Banded sculpin	Insectivore	
Percichthyidae	Temperate basses		
Morone saxatilis	Striped bass	Piscivore	Exotic
Morone chrysops	White bass	Piscivore	
Centrarchidae	Sunfishes		
Ambloplites rupestris	Rock bass	Piscivore	Blackbass
Lepomis cyanellus	Green sunfish	Insectivore	Sunfish
Lepomis gulosus	Warmouth	Piscivore	Sunfish

(*Continued on next page*)

Latin name	Common name	Trophic status	Special designation
Lepomis macrochirus	Bluegill	Insectivore	Sunfish
Lepomis gibbosus	Pumpkinseed	Insectivore	Sunfish
Lepomis humilis	Orangespotted sunfish	Insectivore	Sunfish
Lepomis megalotis	Longear sunfish	Insectivore	Sunfish
Lepomis microlophus	Redear sunfish	Insectivore	Sunfish
Micropterus dolomieu	Smallmouth bass	Piscivore	Blackbass
Micropterus punctulatus	Spotted bass	Piscivore	Blackbass
Micropterus salmoides	Largemouth bass	Piscivore	Blackbass
Pomoxis annularis	White crappie	Piscivore	Blackbass
Pomoxis nigromaculatus	Black crappie	Piscivore	Blackbass
Percidae	Perches		
Etheostoma nigrum	Johnny darter	Insectivore	
Etheostoma acuticeps	Sharphead darter	Insectivore	
Etheostoma blennioides	Greenside darter	Insectivore	
Etheostoma caeruleum	Rainbow darter	Insectivore	
Etheostoma camurum	Bluebreast darter	Insectivore	
Etheostoma tippecanoe	Tippecanoe darter	Insectivore	
Etheostoma zonale	Banded darter	Insectivore	
Perca flavescens	Yellow perch	Insectivore	
Percina caprodes	Logperch	Insectivore	
Percina sciera	Dusky darter	Insectivore	
Percina evides	Gilt darter	Insectivore	
Percina maculata	Blackside darter	Insectivore	
Percina phoxocephala	Slenderhead darter	Insectivore	
Stizostedion vitreum	Walleye	Piscivore	
Stizostedion canadense	Sauger	Piscivore	
Sciaenidae	Drums		
Aplodinotus grunniens	Freshwater drum	Invertivore	

(*Continued*)

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