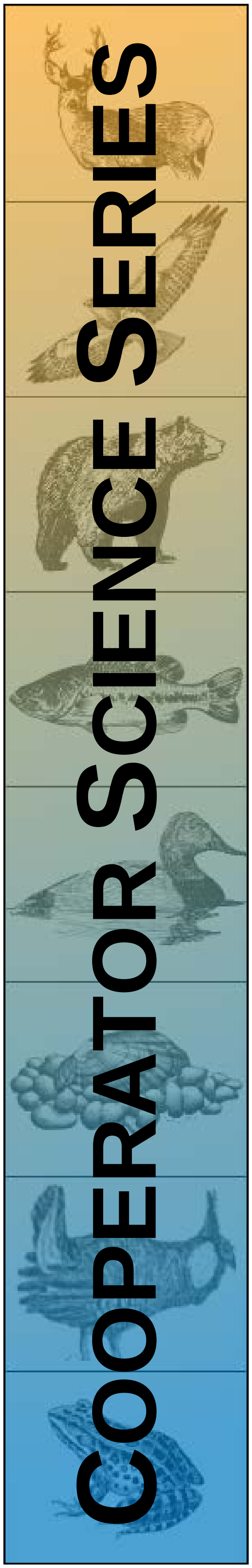

Standardized Methods for Sampling Oklahoma streams

***Shannon K.Brewer*[1]**
***Robert Mullenhauer*[2]**

[1]U.S. Geological Survey, Oklahoma Cooperative Fish and Wildlife Research Unit, Oklahoma State University, Stillwater, Oklahoma

[2]Oklahoma Cooperative Fish and Wildlife Research Unit, Oklahoma State University, Stillwater, Oklahoma

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For additional copies or information, contact:

Dr. Shannon Brewer
U.S. Geological Survey
Oklahoma Cooperative Fish and Wildlife Research Unit
Oklahoma State University
Stillwater, Oklahoma 74078
Phone: (405) 744-6342
E-mail: skbrewer@usgs.gov

FINAL REPORT

Standardized methods for sampling Oklahoma streams

Federal Aid Grant No. F13AF00192 (F-88-R)

S.K. Brewer[1] and R. Mollenhauer[2]

[1] U.S. Geological Survey, Oklahoma Cooperative Fish and Wildlife Research Unit; [2] Oklahoma Cooperative Fish and Wildlife Research Unit, Oklahoma State University

I. OBJECTIVES:

To develop a stream sampling protocol that would increase accuracy and precision to allow ODWC to make better sport-fish management decisions.

II. SUMMARY OF PROGRESS

A. APPROACH

Section 1: Review of sampling gears and their associated biases

We conducted a keyword search of three online databases (Taylor and Francis, Web of Science, and Water Resources Abstracts) using the queries Topic = fish AND sampling AND Topic = gear OR method AND Topic = efficiency OR comparison OR probability. Initially, we selected peer-reviewed studies that pertained to the evaluation of fish sampling gears or methods in freshwater ecosystems. Next, we categorized the studies as either lotic (streams and rivers) or lentic (lakes, ponds, and reservoirs).

For studies in the lotic category, we recorded basic information that included region, fish species, life stage(s), and gear type(s). We classified the system type for each study as either a river or a wadeable stream using the criteria described by Bonar et al. (2009). Additionally, each study was classified as coldwater, warmwater, or tropical. Systems were considered coldwater if they supported salmonid populations. We differentiated between warmwater and tropical by considering systems that supported families such as Cichlidae and Characidae (cichlids and tetras) as tropical.

We also assigned a “yes” or “no” to each lotic study for five questions, where questions were not mutually exclusive.

(1) Did the study quantitatively evaluate a sampling gear or method?

(2) Did the study quantitatively compare sampling gear types?

(3) Did the study identify fish-level or habitat-level sources of variability in capture probability among gear types or method?

(4) Did the study quantify capture probability (defined below)?

(5) Did the study quantify species detection (defined below)?

For question (4), we defined capture probability as the number of fish captured versus the "true" number present (Peterson and Paukert 2009). In order for a "yes" to be assigned to a study, the researchers needed to estimate a "known" population by releasing a known number of fish into a closed system, using an unbiased mark-recapture population estimate, or conducting a census using poisons or explosives. Species detection is a function of both capture probability and abundance. For question (5), we defined species' detection as the probability of detecting at least individuals of a fish species. For the final step of the sampling gear review, we conducted a meta-analysis of studies focused on capture probability. We have selected papers for the meta-analysis that disclose a "known" population estimate and measure either fish-level or environmental variables.

We reviewed each publication to ensure the study addressed capture efficiency. We focused on papers that used a known or estimated population size as the foundation for estimating efficiency. Population sizes were determined by: 1) mark-and-recapture or stocking to obtain a known population, 2) using or creating models to estimate the initial sample size and deriving an efficiency percentage based on the number of individuals recaptured, and 3) use of creel survey or public datasets to determine population size. Capture efficiency was derived using the known or estimated population size and recaptured number of individuals. Several articles clearly indicate the initial population size, recaptured number, and capture efficiency percent, whereas efficiency percent was derived from other articles using initial population size and the number of recaptures, if provided.

Articles relating to capture efficiency were sorted by the study system, thermal regime of the study site(s), and environmental biases. We grouped articles by aquatic system type: lentic (calm) or lotic (moving). We classified the thermal conditions of the study system as coldwater and warmwater. Coldwater systems support salmonids whereas warmwater systems are unable to support salmonids. We grouped fishes by family and quantified the number of studies that targeted each. Individual manuscripts were then interrogated for specific gear types, species captured, and estimates of efficiency.

The study objective(s) were recorded to clarify the focus of the manuscripts. Articles relating to management were classified as age and growth, length frequency, abundance, and fish presence/absence. Gear efficiency objectives were categorized as: 1) gear comparison with one gear having a known efficiency, 2) detection or capture probability, 3) standardized method comparison, and 4) creating a model based on statistical indices. Comparison objectives include: 1) gear comparison without known efficiency of either gear, 2) same gear comparison, 3) same gear comparison addressing temporal and/or spatial differences, or 4) same gear comparison addressing various physical and/or dynamic differences.

Section 2: Relate stream-fish abundance and species traits to streamflow and channel formation characteristics.

Organisms possess coevolved combinations of morphological, behavioral, and physiological traits that maximize fitness (Southwood 1977; Winemiller and Rose 1992; Frimpong and Angermeier 2010; Verberk et al. 2013). Relating measurable species traits to the environment can reveal underlying mechanisms responsible for adaptive responses; thereby elucidating complex scale-dependent ecological processes and improving the predictive ability of assemblage- and community-level management efforts (Poff 1997; Jackson et al. 2001; Frimpong and Angermeier 2010; Verberk et al. 2013). Linking stream fishes through common traits simplifies multispecies comparisons and provides insight into the ecology of both rare and invasive species. Associating traits with the stream environment also allows managers to assess the likelihood that individual fish species will persist with changing climate and land use, which provides essential information for identifying potential future species of conservation concern.

We examined relationships between streamflow and channel formation characteristics and both stream-fish abundance and fish traits. We intended to include a larger portion of Missouri Ozark Highland streams in the study; however, the delayed implementation of StreamStats in Missouri (anticipated completion date is now December 2017) limited the spatial coverage of available stream characteristics. Alternatively, we compiled environmental variables across reaches of the Ozark

Highlands in northeast Oklahoma and southwest Missouri using gage, spatial, and field data and examined relationships between both sunfish abundance and species traits and flow and channel formation characteristics (Figure 1).

Environmental variables - We used instream measurements to characterize variation in flow and channel formation among 40 stream reaches nested within 20 stream segments, where we defined a segment as a length of stream between two third-order or higher tributary confluences (Table 1; Figure 1). We calculated median particle size (D50) using pebble counts (Gordon et al. 2004). Fifty "rocks" were collected haphazardly, while peering over our right shoulder (the key), from three transitional zones at each reach (e.g., runs and downstream areas of pools). We measured the intermediate diameter (1.0 m) of each "rock" with the exception of sand and silt, which were distinguished using texture. We used 0.25 mm to represent sand and 0.05 mm to represent silt, which is consistent with average grain size on the Wentworth scale. We characterized pool habitat using residual pool depth. Residual pool depth quantifies relative depth among pools (i.e., independent of discharge; Lisle 1987) to characterize pool habitat. We calculated residual pool depth for three pools at each reach as the difference in depth between the deepest point of a pool and the crest of the downstream riffle (0.1 m), where pools were chosen haphazardly at reaches with > 3 riffle-run-pool sequences (hereafter referred to as channel unit sequences). We calculated bankfull width-depth ratio to characterize channel formation following Gordon et al. 2004). Bankfull for each reach was established using bank slope (i.e., a flat area adjacent to an abrupt slope towards the floodplain), point bars, deposition of fine sediment, and exposed root masses in undercut banks as indicators. We measured bankfull width (1.0 m) and bankfull depth (0.01 m), where depth was measured at 5.0-m intervals along a transect. We used the average D50, residual pool depth, and bankfull width-depth ratio to represent each reach. We also calculated the proportion of riffle habitat. GPS coordinates were recorded at both the upstream and downstream ends of each reach to estimate length (1.0 m). The proportion of riffle habitat was estimated as the summed length of riffles divided by reach length.

We calculated flow and channel formation characteristics among stream segments using spatial and gauge data. Stream reaches were georeferenced to 1:100,000 National

Hydrography Dataset (NHD) flowlines (http://nhd.usgs.gov/data.html) using ArcMap (version 10.2.1, ESRI, Red Lands, California) and assigned to segments. We calculated the sinuosity of each relevant segment (i.e., contained at least one reach) manually in ArcMap as stream length (1.0 m) divided by valley length (1.0 m). We also predicted the two-year flood magnitude (m^3/sec) of each segment based on the relationship with drainage area. We obtained drainage area estimates for all segments (i.e., we included segments that did not contain study reaches) that were both within the boundaries of our study area and had a drainage area size within the range of our study segments (i.e., hydrologically-similar segments). Drainage areas for gaged segments were obtained from USGS records. Drainage areas for ungaged streams were obtained from the NHD. We compared drainage area estimates for gaged segments between USGS records and the NHD to confirm consistency (confirmed). Two-year flood magnitude for gaged stream segments was estimated with the Bulletin 17B method (IACWD 1982) using the online program PeakFQ range of drainage (http://water.usgs.gov/software/PeakFQ/). We modeled the relationship between drainage area and two-year flood magnitude at gaged segments using ordinary-least-squares linear regression in the statistical software R (version 3.2.2, R Core Development Team, 2014), where two-year flood magnitude was log-10 transformed to improve linearity and promote homoscedastic variance. The resulting equation was used to predict two-year flood magnitude at ungaged stream segments (Table 2). We used the PeakFQ (i.e., more accurate) estimate of two-year flood magnitude to characterize gaged segments and our model estimates to characterize ungaged segments.

Sunfish data - Tow-barge electrofishing data were obtained via Section 3 of this report, and from additional stream reaches that were sampled in early autumn 2014. We focused on six common Ozark Highland sunfishes of similar size (Table 3). Catch data from two-pass electrofishing were adjusted using the multispecies electrofishing capture probability model developed in Section 3. We converted the sunfish abundance estimates to site-specific densities (fish/m) because reach length was variable.

Species traits - We used morphological and life-history characteristics to examine sunfish trait-environment relationships (Table 3). Each sunfish species was assigned to an ecomorphological group (trait group A and B) based on groups developed using a hierarchical cluster analysis with multiscale bootstrapping (Suzuki and Shimodaira 2006) for 92 Ozark Highland stream fishes, where we measured ecomorphological traits on 10-20 adult individuals of each fish species using a combination of calipers and digital methods (Table 4; Figure 2). We obtained samples from either Zoology Department at Oklahoma State University fish collection or field research. Longevity and age at maturation for sunfishes were acquired from “Fishes of Texas” (http://www.fishesoftexas.org/home/) to promote the regional accuracy of sunfish trait information. We used the batch spawner classification for each sunfish species (yes or no) from the “FishTraits database” (http://www.lib.vt.edu/find/databases/F/fishtraits-database.html).

Sunfish abundance model - We developed a linear mixed model using the package lme4 (Bates et al. 2014) in the statistical software R to both identify reach- and segment-scale factors related to variation in sunfish densities and examine trait-environment relationships. Sunfish densities were natural-log transformed due to evidence of heteroscedastic variance. The natural log-transformation of sunfish density was also justifiable ecologically because it diminished the influence of extremely high densities in the analysis. Median particle size (D50) and the two-year flood magnitude were also natural-log transformed due to right skewness. We included both stream reach and stream segment as random effects, where reaches were nested within segments. We also included both a year (2014 and 2015) and season random effect to account for temporal variability among the sampling events, where season was defined as late spring (May-June), summer (July-August), and early autumn (September-October). Species also was a random effect, where predictor variables had species-dependent terms (i.e., both slopes and intercepts varied among species). Sunfish trait-environment relationships were modeled as interaction terms (see Jamil et al. 2013).

We fitted models using a tiered forward model selection following Jamil et al. (2013). The tiered forward selection facilitated the identification of predictor variables related to heterogeneity in densities among sunfishes. Forward selection also allowed correlated predictor variables to be removed as model complexity increased rather than prior to model fitting. Thus, relative model fit can be assessed before removing correlated predictor variables from the analysis. We assessed models using SigAIC (Jamil et al. 2013). SigAIC is a variant of Akaike information criterion (Burnham and Anderson 2001) that introduces a higher penalty factor (*c*) for increased model complexity, where we used $\chi^2(0.05) = 3.84$. Our initial (null) model included only random reach, segment, year, and season intercepts and both random slopes and intercepts for species. For the first tier, we considered six reach-level predictor variables that were added to the model only as species-dependent random slopes (Table 5). The main effects of predictor variables were not considered here because the random model components partly accounted for them and it allowed us to better isolate species-dependent relationships (Jamil et al. 2013). We considered both individual predictor variables and two-way interactions for variables with $|r| < 0.28$, where r is the Pearson's product-moment correlation coefficient. At each step of the model-selection process, we considered models with SigAIC ≥ 1 and retained the variable or interaction term that most decreased SigAIC. Remaining variables with $|r| \geq 0.28$ were eliminated as each reach-level variable or interaction term was added to the model. We were conservative with respect to levels of correlation between predictor variables to minimize confoundedness and improve ecological interpretation of our final model. The first tier concluded by adding the main effects for selected reach-level predictor variables to complete the model. We considered trait-environment interaction terms during the second tier of the model-selection process, where we used the criteria described for the first tier to retain variables (Table 3). The batch-spawning designation for species was identical to the ecomorphological groupings; thus; we only included the trait groups. All continuous variables were standardized such that each had a mean of zero and a variance of one to improve interpretation of model coefficients and promote model convergence.

Lastly, we assessed the variation in sunfish densities explained by the final model. We calculated conditional R^2 (variation explained by both fixed and random effects; Nakagawa and Schielzeth 2013; Johnson 2014) using the MuMIn package (Bartoń 2016) in the statistical software R. The variation in sunfish densities was calculated by subtracting the conditional R^2 from the null model from the conditional R^2 for the final model. We did not consider marginal R^2, which assesses the variation explained only by fixed effects, because here we were only interested in species-specific relationships (i.e., random model components).

Section 3: Estimate tow-barge electrofishing capture probability and relative efficiency of snorkel surveys for stream fishes

A vital part of effective stream-fish sampling, and associated sound decision making, is adjusting catch data based on variation in capture probability (the proportion of fishes captured during sampling). Raw catch data provides only a relative abundance (i.e., an index or count), which results in minimum population estimate and may misrepresent variation in abundance within and among sites (Nichols 1992; Peterson and Paukert 2009). For example, biotic integrity indices (e.g., IBI), which use the relative abundance of certain fish species as a surrogate for water quality, do not adjust data for variable capture probability, often misrepresent assemblage structure and may provide false inferences about the condition of a stream system (Seegert 2000; Price and Peterson 2010). Conversely, adjusting catch data for variable capture probability provides a true population estimate useful for monitoring abundance across space and time (Gwinn et al. 2016). Because unadjusted sampling data are often uninformative, management decisions based on relative abundance may be incorrect and potentially costly. Price and Peterson (2010) showed that both current velocity and the amount of woody debris present in a stream increased the capture efficiency for some fish species, yet decreased the capture probability for others. Sampling that occurs at the same site under different streamflow conditions could result in differences in catch that are primarily a reflection of variable capture probability. Furthermore, results of stream-habitat improvement (i.e., the addition of woody debris as structure) can be confounded with changes in capture probability related to change in habitat.

All sampling gears are biased in some fashion and prone to variation in capture probability both across environmental conditions and among stream fishes. For example, water temperature, conductivity, depth, and turbidity influence the capture probability of stream-fish sampling gears (Burkhardt and Gutreuter 1995; Miranda and Dolan 2003; Peterson and Paukert 2009; Rabeni et al. 2009; Gwinn et al. 2016). Variable capture probability is also associated with fish behavior, habitat use, and fish morphology.

Gear calibration

Gear calibration is a preferred method to adjust stream-fish survey data for variable capture probability (Bayley and Austin 2002; Peterson and Paukert 2009). Gear calibration data collected across a range of sampling conditions can be used to develop a capture-probability that provides comparable abundance estimates across varying environmental conditions (e.g., Peterson and Rabeni 1995; Price and Peterson 2010) Capture probability estimates from the model can be used to estimate population size ($\hat{N}$) by dividing the number of individuals captured (C) by an unbiased estimate of capture probability ($\hat{q}$; Thompson and Seber 1994; Peterson and Paukert 2009). Marking or stocking a known population is the most common approach to gear calibration (Peterson and Paukert 2009). An alternative gear-calibration option is a dual-gear approach. A dual-gear approach requires that either (1) the second gear type has been previously calibrated (e.g., Peterson and Rabeni 2001; Thurow et al. 2006) or (2) a lethal method, such as poisoning, is used to obtain an accurate count (e.g., Bayley and Austin 2002; Penczak et al. 2003). Lethal methods are not a desirable option. Also, we are aware of only two instances of stream-fish sampling gear calibration in the Ozark Highlands and neither is practical for sampling centrarchid assemblages. Dauwalter and Fisher (2007) calibrated multiple types of electrofishing gear for Smallmouth Bass *Micropterus dolomieu* sampling. In addition to only modeling capture probability for one fish species, the authors sampled at only two streams, which limits the usefulness for stream-fish monitoring across a large study and a range of environmental conditions. Peterson and Rabeni (2001) calibrated a quadrant sampler in water depths ≤ 0.5 m and less for riffle-dwelling fishes that did not include any centrarchids. Thus, we developed a multispecies tow-barge electrofishing capture probability model and performed a rigorous model-validation procedure to assess model bias, precision, and accuracy.

Stream-fish sampling - We sampled stream fishes in 34 stream reaches nested in 20 stream segments in the Ozark Highlands ecoregion of northeast Oklahoma and southwest Missouri during summer 2014-2015 (Table 1; Figure 1). All reaches were wadeable (i.e., most habitat was < 1 m deep; Rabeni et al. 2009) and comprised three to five riffle-run-pool sequences to characterize stream habitat. The reaches provided both environmental variation and geographic diversity, with some streams located at the southern ecoregion boundary. We focused on nine species of Centrarchidae because they are both common and abundant in Ozark Highland streams, include popular sport fishes, and recover quickly from electrofishing with minimal mortality (Table 6; Bardygula-Nonn et al. 1995; Dolan et al. 2002; Dolan and Miranda 2004).

We installed two sets of block-off nets at both the upstream and downstream end of each reach to close the area to fish movement. Block-off nets were preferentially placed at shallow riffles to further inhibit fish movement (Peterson et al. 2004; Price and Peterson 2010). Either a low-water bridge at base flows or a dry riffle located at one end of the reach sometimes provided an adequate fish barrier and no block-off nets were installed.

On day one (marking day), we used both a tow-barge electrofisher (Midwest Lake Management, Polo, Missouri) and angling to establish marked populations of centrarchids. Marked fish were not released until the sampling crew had proceeded a minimum of one riffle-run-pool sequence upstream to minimize the probability of being recaptured (none were recaptured). All captured centrarchids were identified to species, measured (1.0-mm TL), and marked with a caudal fin clip. The minimum size for fish was 50-mm TL for *Lepomis* and Rock Bass *Ambloplites rupestris* and 80-mm TL for *Micropterus*. The size restrictions excluded most age-0 fishes not recruited to electrofishing (McClendon and Rabeni 1986) and was also based on observed mortalities for very small centrarchids (Dolan and Miranda 2004; personal observations). Marked fish were released throughout the site and allowed to recover and redistribute for ~48 h. The time between electrofishing events was nearly double the commonly accepted guideline for system recovery (Peterson and Cederholm 1984). Fish injured during sampling or that showed signs of excessive stress were released outside of the blocked-off area.

The recapture event (calibration day) consisted of two standardized removal electrofishing passes per riffle-run-pool sequence. There were four important assumptions associated with unbiased data.

1. No emigration or unknown mortality of marked fish (i.e., a closed system).
2. Marked fish are behaving similarly to unmarked fish.
3. The probability of recapture is equal for each marked fish and is constant throughout the calibration step (with the exception of fish size, which is accounted for in the model).
4. Unmarked fish are not influencing the proportion of marked fish that are recaptured.

The electrofishing crew comprised three people: one tow-barge operator (tow-barge Bob) armed with a hand net and two persons equipped with dip nets each operating one of the two anodes. We used pulsed direct current (DC), 60 Hz, and a 25% duty cycle for electrofishing. Voltage was adjusted to attempt to achieve a target power (W) that maintained a consistent electric field across levels of ambient water conductivity while minimizing electrofishing-induced injuries as described by Miranda (2009). We sampled areas $\geq$ 0.2-m deep, which excluded most riffle habitat, in an upstream direction with a zigzag pattern during each electrofishing pass. Logistic constraints of the tow-barge made very shallow habitats difficult to sample effectively, although use of these habitats are uncommon by centrarchids (Probst et al. 1984; Schlosser 1987; Brewer 2013). Care was taken to thoroughly electrofish areas of structure (e.g., instream large wood, rootwads, and boulders). All captured centrarchids were identified to species and measured (1.0-mm TL).

We used multiple methods to assess delayed fish mortality. The blocked-off area was routinely inspected from the surface for dead marked centrarchids, which were removed from the study (n = 69). The block-off nets and the area between them were inspected periodically for trapped (none were found) or dead (n = 1) centrarchids. We also performed a snorkel pass at a subset of sites (n = 22) prior to calibration day, where crew members were instructed to remove dead centrarchids (none were found).

Environmental measurements - We measured environmental variables hypothesized to influence capture probability of stream fishes. A conductivity pen (Myron L Company, Carlsbad, California; Model PT1) was used to measure water temperature (0.1ºC) and ambient water conductivity (μS/cm) at the downstream end of each reach. Wetted channel width (1.0 m) and thalweg depth (0.1 m) were measured at 50-m transects. Stream discharge (0.01 m3/s) was measured in a homogenous area of a run of each reach using the velocity-area method (Gordon et al. 2004). Water clarity (0.5 m) was measured as the horizontal distance an underwater observer could see a fish silhouette at the downstream end of each reach. We used the same fish silhouette at all sites to maintain consistency and it was designed to mimic the color, markings, and typical size (~200 mm) of Smallmouth Bass in the study streams. GPS coordinates were taken at both the upstream and downstream end of each reach to estimate reach length (1.0 m). We also measured the length of each riffle (1.0 m) to calculate the proportion of riffle habitat for each reach. Both instream large wood and emergent vegetation were estimated as the length (1.0 m) and width (1.0 m) of each patch to calculate proportional coverage of each reach.

We used spatial data to group stream segments (hereafter referred to as segment) into categories based on geology and soils (hereafter referred to "geosoil" groups) as cherty limestone, cherty alluvium, stony alluvium, and shale. The geosoil categories provided surrogates for substrate and streambank characteristics (e.g., lithological complexity and interstitial spaces). GIS layers were obtained for both rock fragment type (Miller and White 1998; Pennsylvania State University 2008) and geology type (USGS 2005). Using ArcMap (version 10.2.1, ESRI, Red Lands, California), a 50-m buffer was generated around each segment, where the dominant rock fragment type and geology type were used to characterize each segment.

Multispecies capture probability model - We modeled electrofishing capture probability among centrarchids across environmental conditions at multiple spatial scales using a generalized linear mixed model (GLMM). GLMMs are a flexible, powerful class of statistical models that allow for the inclusion of random effects, which broadens the scope of inference, accounts for a lack of independence among observations (i.e., pseudoreplication), and accommodates the multiscale (i.e., nested) structure of stream systems and unequal sample sizes common in ecological data (Wagner et al. 2006; Jamil et al. 2013). We implemented models using the package lme4 in the statistical software R. Capture probability was modeled as a Bernoulli process using a binomial error distribution, where recapture was a binary response variable (i.e., recaptured or not recaptured). We assigned recaptures by matching each recaptured fish to a marked fish for each species ± 5.0-mm TL to incorporate variation in fish size into the model. Individual recognition was not required from a modeling perspective (e.g., if 2 of 5 Smallmouth Bass ~200-mm TL were recaptured, the results of the model would be identical regardless of which two were assigned as recaptured); however, accurate species identification and measurements of fish length were critical for reliable results. The model included reach and segment random intercepts. The model also included random slopes and intercepts for species, where we allowed the slopes to vary as a function of reach-scale variables. Treating species as a random effect was both a superior and more statistically-appropriate approach than fitting separate generalized linear models (GLMs) for each species because (1) species-specific fit was presumably improved and estimated error was more likely more accurate due to shrinkage estimates (Pinheiro and Bates 2000), which provide model coefficient for each level of a random effect that are a reflection of all data collected and (2) a usable single-species model for species with smaller samples size would have been impossible, but as a random effect these species can be included and have a degree of robustness in the model (Jamil et al. 2013). A multispecies GLM that treated species as a fixed categorical effect would have been unlikely to converge if many species were included due to the additional parameters and unequal sample sizes. With a random species effect, model performance actually benefited from the inclusion of additional species as it improved the robustness of the variance component. Also, additional species can seamlessly be added to the model if management objectives change or the existing model is incorporated into a new research project.

We fitted models using the tiered forward model selection described in Section 1 using fish TL, 13 reach-scale predictor variables, and the geosoil categories (Table 7). However, we did not include trait-environment interaction terms and we set the threshold for the $|r|$ at 0.50. All continuous variables were natural-log transformed due to asymmetry and standardized such that each variable had a mean of zero and a variance of one to improve interpretation of model coefficients and promote model convergence. Reported estimates of detection probability for validation methods and model interpretation were back transformed. Standard error was first estimated with the delta method using the package msm (Jackson 2011) in the statistical software R and both the capture probability estimate and standard error were then back transformed from the logit scale.

Model validation. - We performed a leave-one-out cross validation to assess “on the ground” performance of our final model at the reach scale. For each cross-validation test (n = 34), we removed one stream reach from the dataset and modeled the remaining data using predictor variables included in our final model. Thus, the cross-validation allowed us to test the predictive ability of our final model at 34 “new” stream reaches (i.e., we tested the model, with less information, against data it had never seen 34 times). We used values of TL and reach-scale environmental characteristics and the segment-scale geosoil classification for the left-out stream reach to derive back transformed species capture probabilities for each stream reach based on linear combinations of model coefficients. Capture probability for each species was calculated using the median TL of 25-mm size classes and the weighted average was used to represent overall species capture probability. We compared model-predicted capture probabilities at the left-out stream reach to observed recapture proportions for species with both ≥ 1 recapture and ≥ 20 marked individuals. We assessed the bias and relative accuracy of our final model. Model bias was calculated as

(1) $$\text{Bias} = \frac{1}{n}\sum_{i=1}^{n}(\hat{y}_i - y_i),$$

where n is the number of species capture probabilities examined across all cross-validation tests, $\hat{y}_i$ is the model-predicted capture probability, and y_i is the observed recapture proportion.

We used root-mean-square deviation (RMSD) to assess relative accuracy. RMSD is a more conservative measure of model accuracy than mean absolute error because error in model-predicted estimates is inflated as it deviate further from the observed value (Freund et al. 2010). RMSD was calculated as

(2) RMSD = $\sqrt{\frac{1}{n}\sum_{i=1}^{n}(\hat{y}_i - y_i)^2}$.

Although the observed recapture proportions of species at stream reaches provided an informative reference to compare model estimates of capture probability, there of course was inherent uncertainty with respect to representing “true “capture probability. Thus, we also calculated 95% binomial probability confidence intervals for observed recapture proportions at left-out stream reaches using the exact Clopper-Pearson method (Freund et al. 2010), which provided a plausible range of “true” capture probabilities. We calculated a single binomial probability confidence interval for each species (not size bin) both for simplicity and to construct more restrictive intervals to test model predictions (i.e., confidence intervals would have been wider for each size bin due to smaller sample size). We evaluated how often model-predicted species capture probabilities were contained in the intervals. Because there was also uncertainty in the model estimates, we also assessed how often 95% confidence intervals around model-predicted capture probability overlapped with the binomial probability confidence intervals.

Alternate capture probability model for Smallmouth Bass

One notable advancement in addressing the challenges of variable capture probability is a class of models known as multinomial *N*-mixture (Royle 2004a; Dorazio et al. 2005; Royle and Dorazio 2006). Multinomial *N*-mixture models use a flexible hierarchical framework to independently estimate both abundance and capture probability of spatially-distinct subpopulations as a function of covariates, where capture probability can vary among both sites and surveys. The hierarchical structure of multinomial N-mixture models enables empirical Bayes calculations (Carlin and Louis 2000) to estimate abundance across spatially-distinct sites. Empirical Bayes estimates, unlike approaches that calculate abundance at each site separately, provide site-specific abundance estimates that are a reflection of data collected across all sites (i.e., the sites “borrow” information; Dorazio et al. 2005; Royle and Dorazio 2006). The dependency among datasets in the hierarchical multinomial *N*-mixture model framework not only improves the precision of confidence intervals but also reduces bias and improves the estimability of abundance at sites with sparse or insufficient data (e.g., low detection or sample size) given adequate data are available at some sites; therefore, all data are informative (i.e., no wasted resources).

We used a multinomial *N*-mixture capture-recapture model (hereafter referred to as multinomial capture-recapture model) to improve the applicability of electrofishing for estimating the abundance of Smallmouth Bass > age 1 in wadeable streams. Although Smallmouth Bass were included in the multispecies capture probability model, it was also important to use a single-species approach at a subset of characteristic Smallmouth Bass streams given their value as a sport fish, thus providing managers with an additional option to monitor Smallmouth Bass populations. We used multiple methods to validate Smallmouth Bass abundance estimates derived from the final model. See also Mollenhauer and Brewer (2017) for additional modeling details.

Smallmouth Bass sampling - We sampled > age-1 Smallmouth Bass in 25 stream reaches that each comprised three to five riffle-run-pool sequences to characterize habitat in the Ozark Highlands ecoregion of northeast Oklahoma and southwest Missouri (Table 8 and Figure 3). The mean ± SD water temperature and ambient water conductivity was 21.5± 2.7 ºC and 276 ± 68 µS/cm, respectively, among sites. We installed two sets of block-off nets using methods described for the multispecies capture probability model. The reaches represented spatially-distinct populations of Smallmouth Bass that were demographically closed during the sample event with mixing of individuals permitted over longer time periods, which is consistent with assumptions of multinomial *N*-mixture models (Royle 2004a; Dorazio et al. 2005). We sampled Smallmouth Bass populations over a three-day period at each reach.

On day one (hereafter referred to as capture day), we used tow-barge electrofishing to establish marked populations of Smallmouth Bass. Smallmouth Bass < 80 mm TL were excluded from the study, which excluded most age-0 fish and was also based on both observed mortalities and lack of recapture via electrofishing of Smallmouth Bass < 80 mm TL. In addition to size, age-0 Smallmouth Bass were easily recognizable due to prominent tri-colored tails. We used the electrofishing protocol described for the multispecies model. A minimum of two electrofishing passes were performed per channel unit sequence, although additional passes were performed at some sites to increase the marked population of Smallmouth Bass. Electrofishing time was recorded at each site to estimate variation in effort both among sites and between capture and recapture events, where electrofishing effort was calculated as electrofishing time divided by sampling area. We marked captured Smallmouth Bass with an upper caudal fin clip. Marked Smallmouth Bass were released throughout the site and allowed to recover and redistribute for ~24 h prior to snorkeling and ~48 h prior to the electrofishing recapture event to allow the system to fully recover (Peterson and Cederholm 1984). Smallmouth Bass injured during the sampling episode or that showed signs of excessive stress were released outside of the blocked-off area. We inspected the block-off nets and the area between them periodically for trapped or dead Smallmouth Bass (there were none). We also routinely inspected the blocked-off area for dead Smallmouth Bass and mortalities were removed from the marked population (only three were found).

On day two, we conducted Smallmouth Bass snorkel counts at 13 stream reaches where horizontal water clarity (see multispecies capture probability for method) was ≥ 3.0 m (Schill and Griffith 1984) to provide a coarse estimate of abundance. All crew members were trained in snorkeling protocols and participated in "practice" surveys with experienced snorkelers. Three persons were typically used for the snorkel surveys; however, only two snorkelers were used in stream areas where wetted channel width was < 10 m. We snorkeled areas ≥ 0.2-m deep in a slow upstream direction while avoiding sudden movements and carefully inspected areas of structure (e.g., searched for fish under logs and between boulders). Each snorkeler maintained a designated lane and stayed approximately in line laterally with other crew members. In general, snorkeling lanes with higher amounts of structure were narrower and snorkeling lanes with mostly open water were wider. Snorkelers maintained communication with each other to minimize double counting of individual fish. Smallmouth Bass estimated to be ≥ 80-mm TL with no prominent tri-colored tail were recorded on an underwater wrist cuff when they either passed or were passed by the snorkeler. We used fish silhouettes and rocks of known sizes to confirm crew member ability to recognize the fish-size cutoffs underwater. We also instructed snorkelers to collect dead Smallmouth Bass for an additional method to estimate delayed mortality (none were found).

On day three, we conducted the recapture event (hereafter referred to as recapture day). The electrofishing procedure was identical to capture day, with the exception that only two passes were performed for each channel unit sequence at each site. We recorded both marked and unmarked Smallmouth Bass ≥ 80 mm TL. Fish counts are reported as mean ± SD.

Environmental measurements - We measured and quantified wetted width, thalweg depth, discharge, water clarity, and water conductivity at stream reach using methods described for the multispecies capture probability model.

Multinomial capture-recapture model - We developed a Smallmouth Bass electrofishing multinomial capture-recapture model using the package unmarked (Fiske and Chandler 2011) in the statistical software R. Specifically, we used the function “gmultmix” with a single primary period, which fits a generalized form of the multinomial *N*-mixture model described by Royle (2004a) and assumes a closed system during the capture-recapture event at each site. In the multinomial *N*-mixture model framework, capture-recapture data is modeled to estimate both abundance and capture probability, where site-specific abundance *N* is treated as a latent variable with a discrete distribution (Chandler 2015). We specified a negative-binomial error distribution, which introduces a dispersion parameter to the model, due to evidence of overdispersion in the dataset. We did not include covariates to explain variation in abundance because the objective was to model Smallmouth Bass abundance (i.e., our focus was on “how many fish were there” rather than “why”); however, their inclusion is straight forward (see Fiske and Chandler 2011; Chandler 2015).

We fitted a candidate set of 12 multinomial capture-recapture models with varying complexity (Table 9). An “effort” capture probability covariate was included in every model to account for variation in electrofishing intensity among surveys. We used the capture probability covariates mean wetted width, mean water depth, discharge, and water clarity in the candidate models to characterize environmental variation in survey conditions, where discharge was natural-log transformed due to a right-skewed distribution. The *r* was > 0.51 between discharge and water depth, thus these covariates did not co-occur in any candidate models. The candidate models were ranked using Akaike information criterion corrected for small sample size (AICc; Burnham and Anderson 2001), where stream reach was the sample size. We did not anticipate that Smallmouth Bass capture probability would vary in relation to water conductivity because we standardized power and the range of conductivities among reaches was considerably less compared to the multispecies model; however, we used AICc to compare the top-ranked model to a model that also included a conductivity capture probability to confirm our expectation. Fish size is also an important consideration when estimating electrofishing capture probability (Peterson and Paukert 2009; Price and Peterson 2010). Currently, unmarked does not allow for individual-level covariates (Fiske and Chandler 2015). However, we did not anticipate that overall capture probability would differ among reaches in relation to Smallmouth Bass size because mean TL was not considerably variable (200 ± 28 mm). We used AICc to compare the top-ranked model to a model that also included a fish capture probability covariate to confirm our size assumption, where we used mean Smallmouth Bass TL to represent each reach. Lastly, we used AICc to compare the top-ranked model to a model that also included a categorical survey event capture probability covariate (i.e., capture and recapture) to confirm that included covariates adequately accounted for variation in detection between capture and recapture events. All capture probability covariates were scaled such that each had a mean of zero and a standard deviation of one to both promote model convergence and simplify interpretation of coefficients. We also developed a candidate set of models using a multinomial framework with the modeling procedures and model validations described for the capture-recapture model (Table 10).

We assessed fit of the top models using both a visual examination of residuals and a calculation of c-hat (an estimate of overdispersion where c-hat > 1 suggests overdispersion). We used a Chi-squared test as described by MacKenzie and Bailey (2004) with 10,000 bootstraps for the calculation of c-hat. We also calculated 95% confidence intervals for coefficients in the top-ranked model using a profile likelihood method (see Fiske and Chandler 2011) and derived cumulative capture probability for each reach. Cumulative capture probability was interpreted as the proportion of available individuals detected across the capture and recapture events.

Model validation - We compared reach-specific Smallmouth Bass abundance estimates derived from the top-ranked model using both a known reliable population estimation method (if assumptions were met for unbiased estimates) and snorkel counts (if available). Model abundance estimates and 95% confidence intervals were calculated using an Empirical Bayes calculation (Fiske and Chandler 2015). We calculated Petersen capture-recapture estimates with the Chapman (1954) bias correction (hereafter referred to as Petersen capture-recapture) using the library Rcapture (Baillargeon and Rivest 2007) in the statistical software R as

(5) $\hat{N} = \{(M + 1)\,(C + 1) / (R + 1)\} - 1,$

where $\hat{N}$ is the population estimate, M is the number of Smallmouth Bass marked during the capture event, C is the number of Smallmouth Bass captured during the recapture event, and R is the number of recaptured Smallmouth Bass that were marked. We only calculated Petersen capture-recapture at sites where assumptions were met for unbiased population estimates outlined by Lyons and Kanehl (1993): (1) At least 20 fish were marked, (2) At least 5 fish were recaptured, and (3) At least 15% of the number of fish captured during the recapture event were marked. We calculated 95% confidence intervals for site-specific Petersen capture-recapture estimates as $\hat{N} \pm z_{\alpha/2}$(SE), where we used a bias-corrected SE (Seber 1970).

Relative snorkeling efficiency

Smallmouth Bass - We assessed relative snorkeling efficiency for Smallmouth Bass at sites where data were available. We compared snorkel counts to Petersen capture-recapture abundance estimates at the four stream reaches where both were available from data collected for the multinomial capture-recapture model (Table 10; Figure 1). Logistical constraints and periodic rainfall (which resulted in extended periods of poor water clarity) during summer of both 2014-2015 prevented us from collecting adequate data to quantify Smallmouth Bass snorkeling capture probability. In addition to assessing relative snorkeling efficiency, we also examined species detection for Smallmouth Bass in Section 4.

Sunfishes - We also examined relative snorkeling efficiency for six sunfishes at 20 stream reaches in summer and early autumn 2014-2015 (Table 11; Figure 1). The snorkel surveys were performed prior to the tow-barge electrofishing surveys described in Section 1. We compared snorkel counts to electrofishing abundance estimates that were adjusted for variable capture probability using the multispecies model. We used the upper bound of a 95% confidence interval for model-estimated capture probabilities to promote more conservative abundance estimates.

Section 4: Estimate occurrence probability and examine species detection for stream fishes of conservation interest

We examined relationships between environmental characteristics and Ozark Highland stream-fish occurrence at multiple spatial scales and examined species detection using both tow-barge electrofishing and snorkeling. The list of target stream fishes was developed in conjunction with ODWC staff (Table 12). We also included species from the electrofishing gear calibration (Section 3) because data were available to broaden our question.

Environmental measurements - We characterized instream environmental variation among 80 channel unit sequences nested within 22 stream reaches of the Ozark Highlands. Stream reaches coincided with Section 1 field sampling in 2015 (Table1; Figure 1). Wetted channel width (1.0 m) and thalweg depth (0.1 m) were measured at 50-m transects unless the channel unit sequence was < 150 m (± 5 m GPS error). For shorter channel unit sequences, we measured wetted width and thalweg depth near the upstream end of the run, near the deepest area of the pool, and near the downstream end of the pool. Wetted channel width and thalweg depth were averaged for each channel unit sequence, where we also used the mean values to calculate a wetted width-depth ratio. GPS coordinates were recorded at both the upstream and downstream end of each channel unit sequence to estimate length (1.0 m). The area of each channel unit sequence

($1.0\ m^2$) was estimated as length multiplied by mean wetted channel width. We also measured the length of each riffle (1.0 m) to calculate the proportion of riffle habitat as riffle length divided by channel unit sequence area. The length (1.0 m) and width (1.0 m) of each patch of instream large wood and emergent vegetation were estimated to calculate area ($1.0\ m^2$). We calculated proportion of instream large wood, emergent vegetation, and total cover (i.e., instream large wood plus emergent vegetation) as the area of each divided by channel unit sequence area.

We characterized stream reaches using both instream measurements and spatial data. Instream large wood, emergent vegetation, and total cover were summed across channel unit sequences and divided by reach area to estimate proportional coverage. We used water temperature variation among reaches as a coarse surrogate for relative groundwater contribution. Two water temperature loggers (HOBO ProV2, Onset, Bourne, Massachusetts) were deployed at stream reaches in a pool area ~ 1.0-m deep for ~4 weeks. The same loggers were used to characterize each reach when reaches were located within 0.5 stream km. We used the mean water temperature and mean standard deviation between the two loggers to calculate a coefficient of variation (CV). The time period water temperature was recorded varied among reaches, where ~70% of the loggers were active during January-February and the remaining loggers were active during April-May. We scaled the CV for each stream reach by subtracting the mean CV of all stream reaches and dividing by the standard deviation to improve comparability, where the resulting *z*-scores were used to represent relative groundwater contribution. We calculated bankfull width-depth ratio, D50, proportion riffle, and residual pool depth following methodologies described in section 2.

We used the distance to the nearest downstream impoundment to characterize the spatial location of each stream reach. Reaches were georeferenced to 1:100,000 National Hydrography Dataset (NHD) flowlines (http://nhd.usgs.gov/data.html) using ArcMap (version 10.2.1, ESRI, Red Lands, California). Locations of impoundments were acquired from the National Anthropogenic Barrier Dataset (https://www.sciencebase.gov/catalog/item/56a7f9dce4b0b28f1184dabd/), the Oklahoma Dams Inventory (http://www.owrb.ok.gov/hazard/dam/dams.php), or the Missouri Dams Inventory (https://www.gisinventory.net/GISI-26268-MO-2014-Dams-SHP-Dam-

Inventory.html). The stream distance from the upstream end of the reach to the upstream edge of the nearest downstream impoundment (1.0 km) was calculated manually in ArcMap.

Stream-fish occurrence data - We used four-pass electrofishing removal to assess stream-fish occurrence at 80 channel unit sequences in summer 2015 (see Section 3 for tow-barge electrofishing protocol). A removal design was necessary due to a dependency among electrofishing passes (e.g., multiple individuals of a species were often shocked during a pass and the time between passes did not allow for system recovery). Species were nearly exclusively encountered during the first or second removal pass (i.e., there were few 001 or 0001 capture histories), which suggested high, constant species detection probability. Thus, modeling species detection probability was not necessary under the conditions we sampled. We also did not encounter additional species during snorkel surveys conducted prior to the electrofishing at a subset of reaches (n = 8). ODWC stream managers can reasonably assume a "true" absence at a channel unit sequence after four tow-barge electrofishing removal passes using the protocol described here for our target stream fishes under the sampling conditions that we encountered (Table 13). We considered stream fishes present in a channel unit sequence once encountered and absent if not encountered after the four electrofishing passes. Subadults and adults were considered separately for Smallmouth Bass and Largemouth Bass *Micropterus salmoides*, where we used 250-mm TL as the cutoff. Our size cut-off was consistent with both age-3 Smallmouth Bass in Ozark Highland streams (Brewer and Long 2015) and the minimum size for mature Largemouth Bass (Claussen 2015). We did not consider subadults and adults separately for Spotted Bass *Micropterus punctulatus* because their life history is poorly understood (Churchill and Bettoli 2015) and individuals large enough to be adults were only encountered at four channel unit sequences. Individuals < 80 mm for all black basses were considered age 0 and not included in the study because they are not readily recruited to electrofishing (see Section 3). Redhorses *Moxostoma* spp. were considered only at the genus level because accurate species identification is difficult in the field. We did not consider Redspot Chub *Nocomis asper* for channel unit sequences surveyed in

Turnback Creek and Little Sac River because these streams are located outside of their presumed native range (Pflieger 1997).

Stream-fish occurrence model - We fitted a GLMM using the package lme4 in the statistical software R to both identify channel unit sequence and reach-scale factors related to variation in stream-fish occurrence. We specified a logit link and a binomial error distribution for the model, where species presence was a binary variable. Water depth, wetted channel width, width-depth ratio, channel unit sequence area and length, D50, distance to dam, and proportion cover, instream large wood, and emergent vegetation (both channel unit sequence and reach) were natural-log transformed due to skewness. We included both channel unit sequence and reach as random effects. We also incorporated a season random effect to account for temporal variability as described in Section 1 (only data collected in 2015 were used so a random year effect was not necessary). Species also was a random effect, where variables included in the model had species-dependent terms (i.e., both slopes and intercepts varied among species). We fitted models using a tiered forward model selection described in Section 1 using 18 predictor variables (Table 13), with the exception that trait-environment interaction terms were not included. Our initial (null) only random reach, segment, and season intercepts and both random slopes and intercepts for species. All continuous variables were standardized such that each had a mean of zero and a variance of one to improve interpretation of model coefficients and promote model convergence. We calculated conditional R^2 (variation explained by both fixed and random effects). The variation in stream-fish occurrence was calculated by subtracting the conditional R^2 from the null model from the conditional R^2 for the final model (see Section 1 for more details).

Reach-scale stream-fish detection - We compared reach-scale species detection between single-pass tow-barge electrofishing and single-pass snorkeling. We sampled 23 stream reaches in late spring to early autumn 2014-2015, where data were collected in conjunction with other studies described here (Table 14; Figure 1; Appendix 1). We used electrofishing and snorkeling protocols described in Section 3. Our sampling in 2014 comprised only the stream fishes included in the electrofishing gear calibration (Section

3), while sampling in 2015 comprised the entire species pool (Table 14; Appendix 2). We assessed detection based on the sampling method's ability to detect a species known to be present at a reach (i.e., encountered using electrofishing, or snorkeling, or both). Logistic constraints prevented us from performing repeat snorkel surveys, where we could have included zero-capture histories to quantify snorkeling detection probability (e.g., occupancy modeling; MacKenzie et al. 2005) We established high electrofishing species detection probability for these stream fishes under 2015 conditions during the stream-fish occurrence sampling for this section.

B. RESULTS

Section 1: Review of sampling gears and associated biases

Our literature review yielded 556 stream-fish studies that evaluated sampling gears or methods in freshwater ecosystems. A total of 204 of published stream-fish studies was conducted in lotic systems, where 33% (*n* = 68) focused strictly on salmonids. Forty-six percent (*n* = 94) of the lotic stream-fish studies were conducted in warmwater systems. However, only 19% (*n* = 39) of warmwater studies focused on fish assemblages in wadeable streams, with only 7% (*n* = 14) conducted in the south-central United States. Most studies used quantitative metrics to evaluate stream-fish sampling gears or methods and half quantified differences between gear types (Table 15). About 70% of the studies attempted to identify sources of variability among gear types or methods, where 37% quantified capture probability. Only 8% of the stream-fish studies quantified species detection probability.

Ninety-two articles containing information on capture efficiency were reviewed intensively. The majority (53) of capture efficiency papers was published after 2000 (Figure 4). The earliest study identified was published in 1945. The majority of articles (59) were published in the North American Journal of Fisheries Management and (14) Transactions of the American Fisheries Society. The majority of publications resulted from studies completed in the Western (30), North Central (27), Southern (19), and in North Eastern (4) AFS regions. Seven publications resulted from studies in Canada, one in Denmark, one in Sweden, one in New Zealand, one in Norway, and one in Brazil.

Most studies were conducted in Georgia, Illinois, Montana, Missouri, North Carolina, South Carolina, Oregon, and Washington.

Most capture-efficiency studies were completed in lotic coldwater and warmwater systems (Figure 5) and the primary species of interest were salmonids. In lotic systems, nine studies were completed in water bodies of unknown depth, 43 in wadeable rivers, and 10 in non-wadeable rivers. Fewer studies were conducted in lentic ecosystems: two in ponds, two in marsh creeks, two in estuaries, 21 studies in lakes/reservoirs, and one study used both a lake and a pond as study systems. Two studies were conducted in both lentic and lotic environments. The most frequently targeted families of capture-efficiency studies were *Salmonidae* (41), *Centrarchidae* (37), *Cyprinidae (*28*)*, *Percidae* (25), *Catostomidae* (20), and *Ictaluridae* (19).

To determine capture efficiency for a gear, a population of known size must first be determined. Three approaches were used to determine a known or estimated population size. Eighty-two studies used mark-and-recapture or stocking to establish the initial population size, six studies used creel surveys or public data sets to obtain a population size, and nineteen studies used or created a statistical model to determine the population size. The efficiencies were then determined based on capture of the 'known' population estimates.

Average capture efficiencies of the most cited single gear type in coldwater and warmwater systems varied substantially. Capture efficiency for Salmonidae was greatest using snorkeling: snorkeling 63%, electroshocking 52%, seine 42%, angling 21%. Efficiency for sampling Percidae was highest using gill nets: gill net 31%, angling 21%, electroshocking 21%, and snorkel 17%. Snorkel efficiency was 27% for Ictaluridae and electroshocking efficiency was 25% for Centrarchidae. Average capture efficiencies of single gear types in warmwater ecosystems were also highly variable. Angling for centrarchids produced the lowest efficiency (6%) whereas efficiency was improved via other gear types: snorkeling 37%, electroshocking 32%, and seining 20%. Seining was more efficient for Cyprinidae (30%) compared to electrofishing (8%). Similar results were obtained using electrofishing or seining for Percidae (4%, 2%, respectively), Catostomidae (26% and 36%), or Ictaluridae (7% and 5%).

Section 2: Relate stream-fish abundance and species traits to streamflow and channel formation characteristics

Environmental measurements and sunfish data - We sampled sunfish populations of variable densities across a variety of habitats. Not surprising, Longear Sunfish *Lepomis megalotis* was found in the highest densities among stream fishes, with the most variability among stream reaches (Table 3). Bluegill *Lepomis macrochirus*, Green Sunfish *Lepomis cyanellus*, and Rock Bass were found in moderate densities and, as expected, Redear Sunfish *Lepomis microlophus* and Warmouth *Lepomis gulosus* were typically found in low densities. All instream variables, along with two-year flood magnitude, varied considerably among reaches (Table 5).

Species traits - Trait characteristics were somewhat variable among sunfishes (Table 3). Age at maturation and longevity among sunfishes ranged from 1-3 y and 3-7 y, respectively. Longear Sunfish and Rock Bass tend to live longer and reproduce at a later age than other sunfishes. Bluegill and Green Sunfish mature at a younger age, whereas Warmouth was the shortest-lived sunfish.

Sunfish abundance model - Bankfull width was the only reach-scale environmental characteristic associated with variation in sunfish densities and we did not identify any trait-environment relationships. With the exception of Rock Bass, sunfish densities were negatively associated with increasing bankfull width (Table 16). The negative relationship between density and bankfull width was strongest for Bluegill and Green Sunfish and weakest for the less common Redear Sunfish and Warmouth. Our top-ranked model explained 6% of the variation in sunfish densities among stream reaches.

Section 3: Estimate tow-barge electrofishing capture probability and relative efficiency of snorkel surveys for stream fishes

Gear calibration

Stream-fish sampling and environmental measurements - We marked 17,123 centrarchids across the 34 stream reaches (mean ± SD: 504 ± 302 fish; Table 6). *Micropterus* were larger and more variable than other centrarchids and Rock Bass tended to be larger than *Lepomis*. Longear Sunfish was the most abundant species and the number of marked individuals and recapture proportions varied among stream fishes. The overall recapture proportion was 26%. Reach-scale environmental characteristics, with the exception of instream large wood and water temperature, were considerably variable among reaches (Table 7). In particular, discharge was highly variable among reaches and we sampled under both low- and high-flow conditions. We classified approximately half of the 20 stream segments as cherty limestone.

Multispecies capture probability model - The predictor variables in our final model comprised fish TL, seven reach-level environmental characteristics, and the geosoil categories (Table 7 and Table 17). The reach-level main effects provided a comprehensive characterization of stream sampling conditions, which included a percent riffle-discharge interaction term. The final model also included random species slopes for water depth and discharge (Table 7 and Table 18). As expected, there was a strong positive relationship between capture probability and fish size (Table 16). Capture probability was higher for all geosoil categories relative to shale, but there was a stronger positive relationship with stony alluvium (Table 16). Capture probability decreased across species with increasing emergent vegetation, water clarity, and wetted channel width-depth ratio (Table 16). Although we adjusted electrofishing power across stream reaches, there was a moderate increase in capture probability with increasing ambient water conductivity across species (Table 16). There was no capture probability-riffle habitat relationship under higher flow conditions (Figure 6A). However, capture probability increased with increasing riffle habitat under low-flow conditions due to the discharge-percent riffle interaction (Figure 6B). Thus, the percent riffle-discharge interaction resulted in a sharp increase in estimated capture probability at low flows, even with a moderate amount of riffle habitat (Figure 6C). Although the positive relationship with stream discharge varied among species, the increase in estimated capture probability under low flows was similar across species (Table 16 and Table 17). For example, there was only a slight increase in Green Sunfish capture probability with decreasing discharge, and capture probability was slightly higher for Bluegill than Smallmouth Bass at moderate and high flows (Figure 7). However, capture probability was similar for Green Sunfish, Bluegill, and Smallmouth Bass at low flows. The relationship between capture probability and water depth also varied among stream fishes (Table 16 and Table 17). For example, there was no water depth-capture probability relationship for Longear Sunfish, but capture probability decreased with deeper conditions for other species (Figure 8). The negative relationship with water depth was most pronounced for Smallmouth Bass, where capture probability was much higher in shallower stream reaches compared to deeper reaches.

Model validation - The cross-validation test comparisons (*n* =124) indicated that the final model performed well with respect to predicting electrofishing capture probability among centrarchids at the reach scale across a range of sampling conditions. Capture probability estimates did not tend to either overpredict or underpredict when compared to observed recapture proportions (overall bias was 0.00; Table 19). Capture probability estimates tended to be lower than the observed recapture proportion for both Redear Sunfish and Warmouth. However, we were only able to make a small number of comparisons due to the relatively small size for these less-common centrarchids in our study area (Table 6). Overall RMSD across the cross-validation tests was 0.10 (Table 19). RMSD was lowest for Smallmouth Bass (0.13), which was not surprising due to the degree of heterogeneity in capture probability across levels of both discharge and water depth (Table 17). Model-estimated capture probability compared favorably to the observed recapture proportions when accounting for inherent uncertainty. Capture probability estimates were contained in 95% binomial probability confidence intervals 90% of the time, with similar trends among species (Table 19). Only one 95% confidence interval around model-estimated capture probability did not overlap with the binomial probability confidence intervals.

<u>Alternate capture probability models for Smallmouth Bass</u>

Smallmouth Bass sampling and environmental measurements - We sampled Smallmouth Bass populations across a range of environmental conditions with varying electrofishing effort. Mean wetted width varied among sites from 9-18 m (14 ± 3 m) and mean thalweg depth varied among sites from 0.5-1.1 m (0.8 ± 0.1 m). Discharge and water clarity varied the most among sites, ranging from 0.091-5.81 m^3/s (1.50 ± 1.43 m^3/s) and 1.5-7.0 m (3.5 ± 1.3 m), respectively. Electrofishing effort varied both among surveys, ranging from 0.013 -0.053 min/m^2 (0.033 ± 0.011 min/m^2), and between capture and recapture events (mean of 0.036 min/m^2 and 0.030 min/m^2, respectively). The number of marked Smallmouth Bass, the proportion of fish recaptured, and the number of fish encountered during snorkeling were highly variable among sites. The number of Smallmouth Bass marked at a site ranged from eight fish at Caney Creek2 to 120 fish at 14 Mile Creek6 (39 ± 30 fish; Table 8). The proportion of Smallmouth Bass recaptured at a site ranged from 0.00 at Flint Creek3 to 0.57 at 14 Mile Creek6 (0.24 ± 0.15). Baseline population estimates of Smallmouth Bass obtained using snorkel counts ranged from 18 fish at Caney Creek1 to 247 fish at Spring Creek5 (127 ± 75 fish).

Multinomial models - The top-ranked Smallmouth Bass capture-recapture model included the capture probability covariates electrofishing effort, water clarity, and a mean wetted channel width and mean water depth interaction (AICc = -7184.09; Table 9 and Table 20). Likely due to less variability in sampling conditions compared to the multispecies capture probability model, there was no evidence that water conductivity contributed to capture probability variation (AICc = -7179.95). We also found no evidence that variation in fish size among stream reaches influenced Smallmouth Bass capture probability (AICc = -7179.35) or that there was considerable remaining capture probability variation between capture and recapture events (i.e., included covariates adequately explained variation in Smallmouth Bass capture probability; AICc = -7180.41). Site-specific cumulative Smallmouth Bass capture probability varied from 0.23 at Big Sugar Creek2 to 0.84 at 14-mile Creek6 (0.45 ± 0.15; Table 8). Estimated capture probability at mean levels of covariates for a single survey was 0.25 ± 0.02 (Table 19). Smallmouth Bass capture probability increased with both increased electrofishing effort and increased water clarity (Table 20; Figure 9). The interaction term in the model indicated that the relationship between Smallmouth Bass detection and both wetted width and water depth varied at different levels of these covariates. Detection decreased sharply as mean water depth increased in narrower surveying conditions; however, the magnitude of the relationship diminished at higher levels of depth (Figure 10a). Conversely, there was only a slight negative relationship between detection and water depth in wider surveying conditions (Figure 10b). Similarly, detection increased with mean wetted channel width in shallower surveying conditions (Figure 10c), with virtually no relationship between detection and wetted width in deeper surveying conditions (Figure 10d). The interaction between water depth and channel width also indicated that the influence of each on Smallmouth Bass capture probability was more pronounced at lower levels of the alternate covariate, there was no influence of wetted width in deep conditions, and capture probability (although low) no longer decreased considerably at high levels of width and depth (i.e., very wide and very deep). The estimate of c-hat from the Chi-squared test (c-hat < 1) did not indicate overdispersion in the model. A plot of predicted versus fitted residuals (n = 75) also suggested adequate model fit (i.e., no evidence of heteroscedasticity).

The top-ranked multinomial removal model included the capture probability covariates wetted width and water depth (Table 10 and Table 21). Estimated cumulative Smallmouth Bass capture probability for the top-ranked removal model was higher (often by 20-30%) than the top-ranked capture-recapture model at 24 of 25 sites (Table 22).

Model validation - A comparison of the empirical Bayes calculations derived from the multinomial capture-recapture model to secondary methods increased confidence in the reliability of the model estimates. Although we only met assumptions for unbiased Petersen capture-recapture at 11 of 25 sites, the estimates were in general agreement with the empirical Bayes estimates and the 95% confidence intervals overlapped at every site (Table 8). However, the range of the confidence intervals for the empirical Bayes estimates were more precise at every site compared to the Petersen capture-recapture confidence intervals. The width of the confidence intervals for empirical Bayes and Petersen capture-recapture was 48 ± 16 fish and 109 ± 83 fish, respectively, at sites where both estimates were available. There was a similar level of precision for empirical

Bayes confidence intervals calculated across all sites, where the width of the interval ranged from 14 to 87 fish (49 ± 20 fish). The empirical Bayes confidence intervals contained the snorkel count at eight of 13 sites. The empirical Bayes confidence intervals exceeded the Smallmouth Bass snorkel count at four sites; however, the snorkel count was within nine fish of the lower bound of the confidence interval at two sites (Buffalo Creek3 and Caney Creek) and within 16 fish of the lower bound of the confidence interval at Buffalo Creek1. The Smallmouth Bass snorkel count exceeded the empirical Bayes confidence interval only at Spring Creek1. The three approaches compared favorably at the sites where we could compare the empirical Bayes abundance estimate to both snorkel counts and Petersen capture-recapture (Buffalo Creek1, Butler Creek, Spring Creek1, and Spring Creek2). We did not consider Smallmouth Bass snorkel counts at reaches of Evansville Creek or 14-mile Creek due to evidence of very low efficiency (e.g., snorkel counts were much lower than electrofishing counts) despite adequate water clarity. The low snorkeling efficiency at Evansville and 14-mile Creek was likely associated with different underlying lithology (these streams are located at the ecotone with the Boston Mountains ecoregion) and well-coordinated intermittent beaver attacks (i.e., the building of beaver dams in the middle of our study reach). We could not compare Smallmouth Bass abundance estimates derived from the multinomial capture-recapture model to either secondary method at only five sites.

Both snorkel counts and Petersen capture-recapture estimates supported that the multinomial removal model would consistently underestimate Smallmouth Bass abundance (Table 22). The snorkel counts were contained in the empirical Bayes confidence intervals derived from the multinomial removal model at only two of 13 sites where we were able to make the comparison. The snorkel count was more than double the empirical Bayes estimates at the other 11 sites, where the counts were > 50 fish higher than the upper bound of the confidence interval at seven sites and > 100 fish higher at four sites. The snorkel count was over double the value of the upper bound of the empirical Bayes confidence interval at five sites. The empirical Bayes confidence intervals derived from the multinomial removal model overlapped with Petersen confidence intervals at only three of 11 sites where we were able to make the comparison. The lower bound of the Petersen confidence interval was higher than the upper bound of the empirical Bayes confidence interval at the other eight sites, where the difference was > 50 fish at three sites. The consistent (sometimes severe) underestimates of Smallmouth Bass abundance from the multinomial model was likely a result of not meeting basic removal sampling assumptions (e.g., declining catch across passes and reasonable depletion of individuals; Lockwood and Schneider 2000) at a sufficient number of sites, which is essential for reliable multinomial model abundance estimates (Dorazio et al. 2005).

Relative snorkeling efficiency

Smallmouth Bass - Smallmouth Bass snorkel counts compared favorably to unbiased electrofishing estimates at stream reaches where we were able to make the comparison (Table 8). The snorkel count was contained in the Petersen capture-recapture 95% confidence interval at Spring Creek6 and slightly lower than the lower bound at Buffalo Creek5. Snorkel counts were somewhat higher than the upper bound of the Petersen confidence interval at both Butler Creek2 and Spring Creek5. The high snorkel counts at Butler Creek2 and Spring Creek5 could be attributed to double counting, although the counts were reasonable for providing coarse estimates of Smallmouth Bass abundance. The general agreement between multinomial capture-recapture estimates, Petersen estimates, and snorkel counts also supports relatively high Smallmouth Bass snorkeling efficiency under ideal conditions.

Sunfishes - Snorkel counts for sunfishes were unreliable and severely underestimated reach-scale abundance in some cases. Rock Bass, Green Sunfish, and Warmouth snorkel counts were nearly exclusively drastically lower than electrofishing abundance estimates (Table 11). Snorkel counts were reasonable for coarse abundance estimates at only one reach for Green Sunfish (Big Sugar Creek1) and Rock Bass (Evansville Creek2) and at no reaches for Warmouth. The low snorkeling efficiency for Rock Bass, Green Sunfish, and Warmouth could be attributed to their observed cryptic behavior. Longear Sunfish snorkel counts compared favorably to electrofishing estimates at Butler Creek1, Fivemile Creek1, Spavinaw Creek3, and Spavinaw Creek5, but were often low. Longear Sunfish snorkel counts were particularly low at high-density reaches (e.g., Big Sugar Creek1 and Flint Creek2), where grouping (and spastic behavior) likely resulted in difficulty obtaining accurate counts. Redear Sunfish counts were also much lower than electrofishing estimates at the only high-density site that we surveyed (Spavinaw Creek 4). Bluegill snorkel counts did often compare favorably to electrofishing estimates. However, Bluegill counts were somewhat low at a few reaches (e.g., Buffalo Creek1), including a high-density reach (Spavinaw Creek4). The general reliably of snorkeling for Bluegill could be attributed to their observed gregarious behavior, which made individuals relatively simpler to count and differentiate. Although snorkeling generally underestimated sunfish abundance, we did not observe extremely lower efficiency in Boston Mountain border streams (Evansville Creek and 14-mile Creek) similar to Smallmouth Bass (see multinomial model validation).

Section 4: Estimate occurrence probability and examine species detection for stream fishes of conservation interest

Environmental measurements and stream-fish occurrence data - All environmental variables used to characterize stream reaches and channel unit sequences were highly variable and channel unit sequence occurrence varied among stream fishes (Table 12 and Table 13). Redspot Chub, Longear Sunfish, and Green Sunfish were the most common stream fishes (encountered in ~ 90% of the channel unit sequences), while Spotted Sucker *Minytrema melanops* and White Crappie *Pomoxis annularis* were the least common species (encountered in ~10% of the channel unit sequences; Table 12). Both subadult Largemouth Bass and subadult Smallmouth Bass were more common than adults and were encountered in a similar proportion of channel unit sequences (~80%). Adult Smallmouth Bass were more common than adult Largemouth Bass (encountered in ~60% and ~30% of the channel unit sequences, respectively).

Stream-fish occurrence model - We identified one channel unit sequence-scale characteristic and three reach-scale characteristics associated with variation in the stream-fish occurrence (Table 23). Occurrence probability increased for all stream fishes with increasing channel unit sequence area; however, the magnitude of the relationship varied. Northern Studfish *Fundulus catenatus*, Spotted Sucker, and White Crappie were much more likely to occur in channel unit sequences with greater area relative to other stream fishes, where Redspot Chub occurrence had virtually no relationship with channel unit sequence area. The relationship between occurrence and reach-scale substrate size varied among stream fishes. Largemouth Bass (both subadult and adult), Spotted Bass, Northern Studfish, and Warmouth were more likely to occur in reaches with larger substrate relative to other stream fishes. Conversely, Rock Bass and Smallmouth Bass (both subadult and adult) were less likely to occur as reach-scale substrate size increased. Stream-fish occurrence was strongly associated with reach-scale water temperature variation. Redspot Chub, a very common species across reaches, decreased sharply with increasing reach-scale water temperature variation (thus decreasing reach-scale relative groundwater contribution). Although the relationship was weaker for Banded Sculpin *Cottus carolinae* and Northern Hogsucker *Hypentelium nigricans,* these stream fishes were also less likely to occur in reaches with greater water temperature variation. Among black basses, the occurrence probability of both subadult and adult Smallmouth Bass declined with increasing reach-scale water temperature variation, but increased for Spotted Bass and both subadult and adult Largemouth Bass. Green Sunfish, Longear Sunfish, Bluegill, and Redear Sunfish occurrence was also positively associated with greater water temperature variation, whereas Rock Bass and Warmouth had negative relationships. Green Sunfish and Longear Sunfish were more likely to occur in reaches with deeper residual pools. Conversely, occurrence probability for Northern Studfish, White Sucker, and Rock Bass was higher in reaches with shallower residual pools. Among black basses, increased occurrences of Spotted Bass and both subadult and adult Largemouth Bass were associated with stream reaches with deeper residual pools, whereas occurrence probability for both subadult and adult Smallmouth Bass decreased with increasing with residual pool depth. Our final model explained 29% of the variation in stream-fish occurrence among channel unit sequences.

Reach-scale stream-fish detection - Single-pass tow-barge electrofishing tended to detect more stream fish species than a single snorkel pass, although more species were sometimes encountered using both methods. Electrofishing detected more species than snorkeling at 17 of 23 stream reaches (Table 14). Electrofishing typically resulted in the detection of one or two additional stream fishes, but did result in the detection of three additional species at both Butler Creek1 and Caney Creek1 and four additional species at Buffalo Creek3. Snorkeling detected more species than electrofishing only at Evansville Creek2 (both White Crappie and Largemouth Bass). One additional species was encountered at four reaches using both electrofishing and snorkeling.

In general, electrofishing detected rarer stream fishes when compared to snorkeling. For example, we detected Warmouth, which are typically found in low densities in our study area, at six additional reaches using electrofishing (Table 14). We also detected Orangespotted Sunfish *Lepomis humilis* (very rare in these streams) at two additional reaches with electrofishing and White Sucker *Catostomus commersonii* (relatively uncommon among suckers in these streams) at five additional reaches. Conversely, we detected the more abundant Bluegill and Rock Bass at only one additional reach using electrofishing. Smallmouth Bass were detected at every reach with both electrofishing and snorkeling, thus suggesting very high species detection with either sampling method. Longear Sunfish (the most common species in these streams) were, interestingly, not detected with either electrofishing or snorkeling at three reaches of Spring Creek. Longear Sunfish was likely absent from these Spring Creek reaches because the species was detected by both sampling methods at every other reach (i.e., high species detection). Redhorses and Northern Hogsucker (more common suckers) were also detected at all reaches by both electrofishing and snorkeling.

III. RECOMMENDATIONS

Our multispecies capture probability model indicated that multiple environmental variables were related to fish capture probability in Ozark Highlands streams. We identified wetted width, water depth, water clarity, stream discharge, emergent vegetation, proportion of riffle habitat, and lithology as variables that contribute to variation in tow-barge electrofishing capture probability for centrarchids. The relationship between capture probability and both water depth and discharge varied among species. Our results also suggest that standardizing electrofishing power may minimize variation in electrofishing capture probability if ambient water conductivity is not highly variable; however, stream-fish managers would benefit from adjusting centrarchid catch data using the multispecies capture probability model. We did not identify either instream large wood or water temperature as environmental variables that contributed to variation in electrofishing capture probability. Although both water temperature and instream large wood have been shown to be associated with electrofishing capture probability, neither likely varied enough among our study sites. In addition to environmental variables included in the multispecies electrofishing capture, we do recommend measuring water temperature at the time of sampling to ensure it is within the range we encountered during gear calibration. Model estimates may be unreliable if sampling occurs outside the range of water temperatures we encountered. We do not recommend measuring instream large wood at the time of sampling using the methods described here. It is often difficult to quantify the amount of instream large wood with a coarse surface estimation of patches. We recommend underwater observations or finer-scale measurements (e.g., measuring each piece individually) if a more intensive examination of the influence of instream large wood on electrofishing capture probability is desired. A primary advantage of the multispecies model is that additional stream fishes can be added; however, we recommend additional gear calibration to assess species-specific fit.

Our results indicate that single-pass snorkel counts are not reliable for estimating sunfish (e.g., Longear Sunfish) abundance in Ozark Highland streams. We recommend quantifying snorkeling capture probability as a function of covariates (i.e., modeling) either using repeat surveys (e.g., Royle 2004b) or calibrating against electrofishing abundance estimates derived from the multispecies capture-probability model if snorkeling to estimate sunfish abundance is desired. Data adjusted for variable snorkeling capture probability may result in reliable abundance estimates for Bluegill, Longear Sunfish, and Redear Sunfish; however, snorkeling efficiency is likely too low to be practical for Rock Bass, Green Sunfish, or Warmouth.

Our results suggest that, for Ozark Highland stream fishes considered here, single-pass tow-barge electrofishing will typically result in higher reach-scale stream-fish species detection than single-pass snorkeling in Ozark Highland streams. Single-pass snorkeling may be adequate for a reasonable assessment of presence-absence for common species, however; electrofishing was much more effective for less common stream fishes. The use of both tow-barge electrofishing and snorkeling may result in slightly higher species detection, although the additional effort may not be an acceptable trade-off. Both sampling methods were shown to have imperfect detection; thus, we recommend repeat surveys to quantify species detection probability and increase confidence in differentiating between true and false absences if either single-pass electrofishing or single-pass snorkeling is desirable for long-term stream fish monitoring. Our results do support that four-pass removal electrofishing can provide high, constant species detection (i.e., imperfect and variable detection probability can be ignored from a practical perspective) and may be a less labor-intensive option for species occurrence studies in the long run (i.e., repeat surveys and measurements of covariates to establish sources of heterogeneous detection probability is not required). Species-detection relationships may differ for other stream fishes (e.g., small-bodied minnow and darter) or in sampling conditions we did not encounter.

We assessed multiple sampling options for Smallmouth Bass. We do not recommend two-pass electrofishing removal to monitor Smallmouth Bass populations in Ozark Highland streams because our results suggest it consistently underestimates abundance even when accounting for variable capture probability. Additional electrofishing removal passes may result in more reliable Smallmouth Bass abundance estimates; however, we do not recommend that approach given the additional effort may not be practical and we have identified multiple viable approaches. We identified three plausible options to monitor Smallmouth Bass abundance, where the most desirable approach is associated with management objectives. (1) The multinomial capture-recapture model can be used to estimate Smallmouth Bass abundance using either one- or two-pass tow-barge electrofishing (although higher uncertainty in estimates would be expected using only one pass). The multinomial model provides reliable Smallmouth Bass-specific estimates of capture probability in higher density streams under typical summer conditions (i.e., lower flows). (2) The multispecies capture probability model can be used to estimate Smallmouth Bass abundance for two-pass tow-barge electrofishing.

Although estimates reflect both a trade-off of common (i.e., group) and species-specific relationships, the multispecies model encompasses a wider range of sampling conditions. Similarly, the multispecies model was developed with a much larger set of observations, which allowed us to include a larger number of variables associated with variable capture probability. Further, fit for species with relatively lower sample size, which included Smallmouth Bass, was improved due to the borrowing strength among stream fishes in the multispecies model. The multispecies model is a desirable option for Smallmouth Bass for long-term regional monitoring, particularly if Smallmouth Bass are incorporated into assemblage-level sampling efforts. (3) Single-pass snorkel counts with no consideration of variable capture probability are a viable option for Smallmouth Bass in "typical" Ozark streams during periods of good visibility for coarse abundance estimates (e.g., identifying major changes in reach-scale populations). We do not recommend using unadjusted snorkel counts in Boston Mountain border streams, even under ideal water clarity. We also suggest caution using unadjusted Smallmouth Bass snorkel counts as a primary sampling approach for sound long-term monitoring regardless of stream type or conditions without additional research. This approach is likely acceptable for determining coarse-scale differences, but for long-term monitoring, adjusted data would be desirable. We recommend repeat surveys to quantify Smallmouth Bass snorkeling capture probability if relying primarily on snorkeling for long-term population monitoring. Our results indicate that either single-pass snorkeling or single-pass tow-barge electrofishing is effective for establishing Smallmouth Bass presence-absence.

We identified both reach and channel unit sequence environmental characteristics that were related to stream fish abundance and occurrence. Reach-scale bankfull width was associated with sunfish abundance, where Rock Bass were the only species with a negative association. Bankfull width is an instream characteristic that can be manipulated by managers to achieve objectives. Reach-scale water temperature variation, substrate size, and residual pool depth were associated with stream-fish occurrence. The only channel unit sequence-scale characteristic associated with stream-fish occurrence was area. Differentiating at what scale stream fishes are "selecting" is important for specifying and quantifying management objectives. Additionally, both residual pool depth and stream area can be manipulated. Our findings also provide insight into stream-fish temperature relationships, which managers can use to identify species of concern if thermal regimes change over time. We did not identify any relationships between sunfish abundance and species traits.

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IV. SIGNIFICANT DEVIATIONS

There have been no significant deviations.

V. EQUIPMENT

No equipment was purchased in 2017. A tow barge electrofishing unit was purchased in 2013, but the equipment has been transferred to ODWC (since 2016).

VI. PREPARED BY:

Shannon Brewer and Robert Mollenhauer
Oklahoma Cooperative Fish and Wildlife Research Unit,
Oklahoma State University Stillwater, Oklahoma

DATE: August 10, 2017

APPROVED BY: ______________________________

Fisheries Division Administration
Oklahoma Department of Wildlife Conservation

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Andrea Crews, Federal Aid Coordinator
Oklahoma Department of Wildlife Conservation

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VII. TABLES

Table 1. Year and season of sampling and for 40 stream reaches (34 included in tow-barge calibration) nested wihin 20 stream segments surveyed using tow-barge electrofishing in the Ozark Highlands ecoregion of northeast Oklahoma and southwest Missouri in 2014-2015, where season one is May-June, season two is July-August, season three is September-October.

Reach	Year	Season	Segment	Tow-barge calibration
Baron Fork1	2014	2	Baron Fork1	Yes
Baron Fork2	2014	3	Baron Fork1	No
Baron Fork3	2015	2	Baron Fork1	Yes
Big Sugar Creek1	2015	2	Big Sugar Creek1	Yes
Buffalo Creek1	2014	2	Buffalo Creek1	Yes
Buffalo Creek2	2014	3	Buffalo Creek1	No
Buffalo Creek3	2015	2	Buffalo Creek1	Yes
Buffalo Creek4	2015	2	Buffalo Creek1	Yes
Butler Creek1	2015	2	Butler Creek1	Yes
Caney Creek1	2015	2	Caney Creek1	Yes
Evansville Creek1	2014	2	Evansville Creek1	Yes
Evansville Creek2	2014	3	Evansville Creek1	No
Five-mile Creek1	2015	2	Five-mile Creek1	Yes
Flint Creek1	2014	1	Flint Creek1	Yes
Flint Creek2	2014	2	Flint Creek1	Yes
14-mile Creek1	2014	2	14-mile Creek1	Yes
14-mile Creek2	2014	2	14-mile Creek1	Yes
14-mile Creek3	2015	2	14-mile Creek1	Yes
Greenleaf Creek1	2015	1	Greenleaf Creek1	Yes
Greenleaf Creek2	2015	1	Greenleaf Creek1	Yes
Honey Creek1	2015	1	Honey Creek1	Yes

Honey Creek2	2015	2	Honey Creek1	Yes
Indian Creek1	2015	2	Indian Creek1	Yes
Little Sac River	2015	1	Little Sac River1	Yes
Lost Creek1	2015	2	Lost Creek1	Yes
Saline Creek1	2014	2	Saline Creek1	Yes
Saline Creek2	2015	2	Saline Creek1	Yes
Sallisaw Creek	2015	2	Sallisaw Creek1	Yes
Spavinaw Creek1	2014	1	Spavinaw Creek1	Yes
Spavinaw Creek2	2014	2	Spavinaw Creek2	Yes
Spavinaw Creek3	2014	2	Spavinaw Creek1	Yes
Spavinaw Creek4	2014	3	Spavinaw Creek2	No
Spavinaw Creek5	2014	3	Spavinaw Creek1	No
Spavinaw Creek6	2015	1	Spavinaw Creek1	Yes
Spavinaw Creek7	2015	3	Spavinaw Creek1	Yes
Spring Creek1	2014	2	Spring Creek1	Yes
Spring Creek2	2014	3	Spring Creek1	No
Spring Creek3	2015	3	Spring Creek1	Yes
Spring Creek4	2015	3	Spring Creek1	Yes
Turnback Creek	2015	1	Turnback Creek1	Yes

Table 2. Results of ordinary-least-squares regression modeling the relationship between drainage area and two-year flood interval for hydrologically-similar Ozark Highland stream segments ($R^2 = 0.69$). Two-year flood interval was log-10 transformed.

Coefficients	Estimate	SE	*p*-value
Intercept	7.93	0.15	< 0.01
Drainage area (km^2)	0.01	0.01	< 0.01

Table 3. Mean density and traits for six stream-fish species sampled using tow-barge electrofishing at 40 stream reaches in the Ozark Highlands ecoregion of northeast Oklahoma and southwest Missouri in 2014-2015.

Common name	Scientific name	Mean density (fish/m) ± SD	Ecomorphological group	Longevity (yr)	Age at maturation (yr)
Bluegill	*Lepomis macrochirus*	0.48 ± 0.80	A	5	1
Green Sunfish	*Lepomis cyanellus*	0.48 ± 0.70	B	5	1
Longear Sunfish	*Lepomis megalotis*	1.77 ± 2.18	B	6	3
Redear Sunfish	*Lepomis microlophus*	0.07 ± 0.27	A	5	2
Rock Bass	*Ambloplites rupestris*	0.35 ± 0.31	B	7	3
Warmouth	*Lepomis gulosus*	0.06 ± 0.12	B	3	1

Table 4. Ecomorphological traits used for developing groups of Ozark stream fishes (*n* = 92) using hierarchical clustering with multiscale bootstrap resampling (HCMB). Asterisks indicate traits that were removed prior to the cluster analyses due to high correlations with other traits.

Trait	Definition
*Head depth	Vertical distance from dorsum to ventrum passing through the left pupil divided by standard length
*Head length	Horizontal distance from the anterior tip of the jaw to the posterior margin of opercular membrane divided by standard length
Flatness	Maximum body depth divided by body width
*Body depth below midline	Vertical distance along line of maximum body depth from the point of an imaginary perpendicular line coming from the pupil divided by standard length
Trunk length	Horizontal distance from the anterior tip of the jaw to point of maximum body depth divided by standard length
Caudal peduncle length	Horizontal distance from the anterior margin of the posterior base of the caudal fin to an imaginary vertical line from the posterior base of the anal fin divided by standard length
Caudal peduncle flatness	Caudal peduncle depth divided by caudal peduncle width
Eye position	Vertical distance from the center of the pupil to the ventrum divided by standard length

Eye diameter	Distance from between fleshy orbits of the eye along an anterior-posterior axis divided by standard length
Snout length	Distance from the pupil to the tip of the upper jaw with mouth shut divided by standard length
Mouth width	Horizontal distance across the mouth when fully open divided by standard length
*Mouth height	Vertical distance across the mouth when fully open divided by standard length
*Jaw length	Distance of the lower jaw mandible divided by standard length
*Dorsal fin area	Surface area of the dorsal fin divided by body area
Dorsal fin aspect ratio	Dorsal fin height divided by dorsal fin length
Caudal fin span	Maximum distance from the top edge of the caudal fin to the bottom edge of the caudal fin divided by maximum body depth
Caudal fin aspect ratio	Maximum distance from the top edge of the caudal fin to the bottom edge of the caudal fin squared divided by the surface area of the caudal fin
Anal fin area	Surface area of the anal fin divided by body area
Anal fin aspect ratio	Anal fin height divided by anal fin length
*Pelvic fin area	Surface area of the pelvic fin divided by body area
Pelvic fin aspect ratio	Pelvic fin height divided by pelvic fin

	length
Pectoral fin area	Surface area of the pectoral fin divided by body area
Pectoral fin aspect ratio	Pectoral fin height divided by pectoral fin length

Table 5. Flow and channel formation characteristics for 40 stream reaches nested within 20 stream segments in the Ozark Highlands ecoregion of northeast Oklahoma and southwest Missouri surveyed in 2014-2015 using tow-barge electrofishing.

Variable	Scale	Range	Mean ± SD
Bankfull width-depth ratio	Reach	17.00-72.00	45.90 ± 15.40
D50 (mm)	Reach	20.00-131.00	34.63 ± 23.25
Proportion riffle	Reach	0.07-0.42	0.23 ± 0.09
Residual pool depth (m)	Reach	0.50-2.00	1.23 ± 0.36
Sinuosity	Segment	1.09-1.73	1.35 ± 0.19
Two-year flood magnitude (m^3/sec)	Segment	87.00-355.00	160.93 ± 79.02

Table 6. Recapture rates and size variation for nine stream fishes included in tow-barge electrofishing gear calibration at 34 stream reaches in the Ozark Highlands in summer 2014-2015.

Species	Scientific name	# marked	Cumulative recapture	Mean TL ± SD (mm)	TL range (mm)
Bluegill	*Lepomis macrochirus*	1,922	25.0%	94 ± 25	50 - 215
Green Sunfish	*Lepomis cyanellus*	2,534	30.5%	106 ± 29	50 - 207
Largemouth Bass	*Micropterus salmoides*	517	23.6%	175 ± 73	50 - 196
Longear Sunfish	*Lepomis megalotis*	8,678	25.5%	94 ± 21	50 - 186
Redear Sunfish	*Lepomis microlophus*	158	26.6%	85 ± 24	50 - 286
Rock Bass	*Ambloplites rupestris*	2,052	19.0%	128 ± 37	52 - 182
Smallmouth Bass	*Micropterus dolomieu*	1,170	26.7%	200 ± 61	80 - 460
Spotted Bass	*Micropterus punctulatus*	30	20.0%	165 ± 67	80 - 404
Warmouth	*Lepomis gulosus*	240	39.6%	113 ± 23	90 - 360

Table 7. Instream variables to characterize 34 stream reaches in the Ozark Highlands ecoregion of northeast Oklahoma and southwest Missouri during summer 2014-2015 to develop a tow-barge electrofishing capture probability model Asterisks indicate variables included in the final model.

Variable	Definition	Mean ± SD	Range
Cross-sectional area (m^2)	Mean of wetted width times thalweg depth measured at 50 m transects	13.45 ± 6.52	5 - 39
*Discharge (m^3/s)	Mean of three replicates in a homogenous area of a run	1.86 ± 2.11	0.09 - 8.52
Proportion cover	Percent emergent vegetation plus percent instream large wood	0.06 ± 0.06	0.00 - 0.28
*Proportion emergent vegetation	Total area divided by sampling area	0.03 ± 0.06	0.00 - 0.25
Proportion instream large wood	Total area divided by sampling area	0.03 ± 0.02	0.00 - 0.10
*Proportion riffle	Total length divided by reach length	0.22 ± 0.09	0.07 - 0.42
Pool depth (m)	Mean maximum pool depth	1.35 ± 0.33	0.7 - 2.5
*Water clarity (m)	Horizontal distance an underwater observer could see fish silhouette	3.25 ± 1.53	1.0 - 8.5
Water conductivity (µS/cm)	Ambient water conductivity measured at the downstream end of reach	284.23 ± 80.35	160 - 510
*Water depth (m)	Mean thalweg depth measured at 50 m transects	0.82 ± 0.17	0.5 - 1.3
Water temperature (°C)	Measured at downstream end of the reach	21.60 ± 2.64	16.1 - 25.7
Wetted channel width (m)	Mean wetted width measured at 50 m transects	15.11 ± 4.73	9 -32
*Width-depth ratio	Mean wetted width of reach divided by mean thalweg depth of reach	18.69 ± 5.12	10 - 34

Table 8. Summary of snorkel surveys and capture-recapture estimates for 25 Smallmouth Bass populations in stream reaches of the Ozarks Highlands, where *M* is the total number of fish captured during the capture event, *C* is the total number of fish captured during the recapture event (both marked and unmarked), recap is the proportion of marked fish recaptured, and cumulative capture probability is the estimated proportion of Smallmouth Bass captured across both capture and recapture events and was calculated as the sum of the multinomial cell probabilities. Cumulative capture probability, multinomial abundance estimates, and multinomial 95% confidence intervals (CI) were derived from a multinomial negative-binomial-mixture model. Petersen abundance estimates were calculated using Peterson capture-recapture with the Chapman (1954) bias correction. Petersen 95% CI were calculated with a bias-corrected SE (Seber 1970) as $\hat{N} \pm z_{\alpha/2}$(SE). NA's for snorkel count indicate sites that were not surveyed due to insufficient water clarity. NA's for Petersen abundance estimates and Petersen 95% CI indicate sites where assumptions for unbiased estimates were not met.

Reach	*M*	*C*	Recap	Cumulative capture probability	Snorkel count	Multinomial abundance estimate	Petersen abundance estimate	Multinomial 95% CI	Petersen 95% CI
Baron Fork4	71	37	0.13	0.46	NA	214	299	185 - 246	147 - 451
Baron Fork5	16	11	0.38	0.54	NA	41	NA	30 - 53	NA
Big Sugar Creek2	24	11	0.08	0.23	138	144	NA	105 - 188	NA
Buffalo Creek5	79	96	0.47	0.75	153	183	204	169 - 199	167 - 241
Buffalo Creek6	11	17	0.09	0.32	85	86	NA	62 - 114	NA
Buffalo Creek7	15	17	0.07	0.35	59	91	NA	68 - 118	NA
Butler Creek2	64	60	0.36	0.43	244	232	164	200 - 267	123 - 204
Caney Creek2	8	6	0.13	0.45	18	31	NA	21 - 45	NA

Evansville Creek3	70	44	0.36	0.52	NA	170	128	147 - 195	101 - 155
Five-mile Creek2	15	13	0.20	0.50	61	52	NA	39 - 67	NA
Flint Creek3	14	21	0.00	0.33	NA	107	NA	81 - 137	NA
Flint Creek4	28	45	0.18	0.30	208	224	NA	183 - 270	NA
14-mile Creek4	32	40	0.25	0.40	NA	159	150	131 - 190	75 - 224
14-mile Creek5	36	25	0.14	0.51	NA	109	162	91 - 130	57 - 268
14-mile Creek6	120	97	0.57	0.84	NA	176	173	166 - 188	157 - 188
Honey Creek3	59	50	0.42	0.62	NA	135	117	118 - 153	93 - 141
Honey Creek4	12	12	0.17	0.30	NA	75	NA	52 - 103	NA
Indian Creek2	14	19	0.21	0.31	NA	97	NA	7 2 - 128	NA
Lost Creek2	9	4	0.22	0.62	NA	19	NA	13 - 27	NA
Saline Creek3	48	29	0.04	0.40	156	186	NA	156 - 220	NA
Spavinaw Creek8	63	59	0.13	0.40	167	282	NA	244 - 323	NA
Spavinaw Creek9	41	32	0.34	0.44	NA	133	92	110 - 160	64 - 120
Spring Creek5	29	39	0.31	0.33	247	176	120	142 - 214	66 - 174
Spring Creek6	85	84	0.40	0.62	225	215	207	194 - 238	167 - 248
Spring Creek7	17	18	0.47	0.40	84	69	NA	51 - 90	NA

Table 9. Results from 12 candidate multinomial capture-recapture models fitted with a negative-binomial error distribution to estimate site-specific abundance and capture probability for Smallmouth Bass using tow-barge electrofishing instream reaches of the Ozarks Highlands in northeast Oklahoma and southwest Missouri from July to October 2014-2015. In the model set, λ is latent abundance, *p* is the estimated detection probability, effort is electrofishing effort, width is average wetted width., Q is discharge, depth is average thalweg depth, clarity is horizontal water clarity and *θ* is the overdispersion parameter. K is the number of the parameters in each model, AICc is the Akaike information criterion score for the model corrected for small sample size, and w_i is the relative support for the model.

Model	K	Log-likelihood	AICc	Δ AICc	w_i
λ, *p*(effort + width * depth + clarity), *θ*	8	3604.54	-7184.09	0.00	0.95
λ, *p*(effort + width + depth + clarity), *θ*	7	3598.67	-7176.75	7.34	0.02
λ, *p*(effort + width * depth), *θ*	7	3598.34	-7176.09	8.00	0.02
λ, *p*(effort + width + depth), *θ*	6	3595.29	-7173.91	10.18	0.01
λ, *p*(effort + depth), *θ*	5	3591.39	-7169.61	14.48	0.00
λ, *p*(effort + width * Q + clarity), *θ*	8	3594.18	-7163.36	20.73	0.00
λ, *p*(effort + width + Q + clarity), *θ*	7	3591.58	-7162.58	21.51	0.00
λ, *p*(effort + width + Q), *θ*	6	3587.67	-7158.68	25.41	0.00
λ, *p*(effort + width + Q), *θ*	7	3588.97	-7157.35	25.41	0.00
λ, *p*(effort + Q), *θ*	5	3883.79	-7154.42	29.67	0.00
λ, *p*(effort + width), *θ*	5	3572.88	-7132.61	51.48	0.00
λ, *p*(effort), *θ*	4	3562.54	-7115.08	69.01	0.00

Table 10. Results from 12 candidate multinomial removal models fitted with a negative-binomial error distribution to estimate site-specific abundance and capture probability for Smallmouth Bass using tow-barge electrofishing in 25 stream reaches of the Ozarks Highlands in northeast Oklahoma and southwest Missouri from July to October 2014-2015. In the model set, λ is latent abundance, p is the estimated capture probability, effort is electrofishing effort, width is average wetted width., Q is discharge, depth is average thalweg depth, clarity is horizontal water clarity and θ is the overdispersion parameter. K is the number of the parameters in each model, AICc is the Akaike information criterion score for the model corrected for small sample size, and w_i is the relative support for the model.

Model	K	Log-likelihood	AICc	ΔAICc	w_i
λ, p(width + depth), θ	5	2044.57	-4075.97	0.00	0.51
λ, p(width * depth), θ	6	2044.97	-4073.28	2.69	0.13
λ, p(width * depth + clarity), θ	6	2044.93	-4073.20	2.77	0.13
λ, p(width * Q), θ	6	2044.76	-4072.84	3.13	0.11
λ, p(width + Q), θ	5	2042.28	-4071.41	4.56	0.05
λ, p(depth), θ	4	2039.89	-4069.78	6.19	0.02
λ, p(width + depth + clarity), θ	7	2045.11	-4069.64	6.33	0.02
λ, p(width * Q + clarity), θ	7	2044.81	-4069.02	6.95	0.02
λ, p(width + Q + clarity), θ	6	2042.56	-4068.46	7.51	0.01
λ, p(width), θ	4	2038.38	-4066.76	9.22	0.01
λ, p(Q), θ	4	2036.71	-4063.41	12.56	0.00
λ, p, θ	3	2028.66	-4050.17	25.80	0.00

Table 11. Comparison of snorkel counts and tow-barge electrofishing abundance estimates for 6 stream fishes at 20 stream reaches of the Ozarks Highlands in northeast Oklahoma and southwest Missouri. Reaches were sampled from July to October 2014-2015 (see also Table 1 and Figure 2).

Site	Bluegill E/S	Green Sunfish E/S	Longear Sunfish E/S	Redear Sunfish E/S	Rock Bass E/S	Warmouth E/S
Baron Fork2	0/0	16/1	386/182	0/0	82/3	0/0
Big Sugar Creek1	23/130	87/44	4,208/819	15/17	102/14	0/0
Buffalo Creek1	145/25	228/3	381/121	2/0	59/1	69/6
Buffalo Creek2	200/149	186/5	568/374	0/1	88/11	87/0
Buffalo Creek3	61/30	286/7	63/26	0/0	56/1	4/0
Buffalo Creek4	279/125	151/7	456/231	14/6	37/7	223/8
Butler Creek1	263/207	380/15	591/597	8/1	162/6	5/0
Caney Creek1	285/140	234/11	801/424	5/0	6/1	51/8
Evansville Creek1	23/27	94/2	720/344	0/0	62/7	0/0
Evansville Creek2	14/81	95/2	333/500	0/0	19/18	0/0
Fivemile Creek1	268/192	168/3	198/264	3/0	75/4	4/0
Flint Creek2	8/4	35/1	1,500/686	2/2	808/184	0/0
14-mile Creek1	162/83	344/22	1,779/536	0/0	73/1	7/0
14-mile Creek2	36/4	276/5	708/142	2/0	10/17	0/0
Saline Creek1	33/16	43/0	306/216	0/0	339/23	0/0
Spavinaw Creek3	55/16	61/0	270/278	0/0	643/114	0/0
Spavinaw Creek4	1,472/603	118/0	310/153	641/143	78/0	62/0
Spavinaw Creek5	131/54	46/0	166/127	6/0	154/9	0/0
Spring Creek1	0/0	13/1	0/0	0/0	156/53	0/0
Spring Creek2	6/0	21/4	0/0	0/0	370/11	0/0

Table 12. Ozark Highlands stream fishes included in an examination of both factors related to stream-fish occurrence at multiple spatial scales and species detection.

Common name	Latin name	Proportion of channel unit sequences encountered
Banded Sculpin	*Cottus carolinae*	0.74
Bluegill	*Lepomis macrochirus*	0.79
Creek Chub	*Semotilus atromaculatus*	0.45
Green Sunfish	*Lepomis cyanellus*	0.88
Largemouth Bass (subadult)	*Micropterus salmoides*	0.70
Largemouth Bass (adult)	*Micropterus salmoides*	0.28
Longear Sunfish	*Lepomis megalotis*	0.88
Northern Hogsucker	*Hypentelium nigricans*	0.78
Northern Studfish	*Fundulus catenatus*	0.23
Orangespotted Sunfish	*Lepomis humilis*	0.00
Redhorses	*Moxostoma* spp.	0.66
Redear Sunfish	*Lepomis microlophus*	0.31
Redspot Chub	*Nocomis asper*	0.87
Rock Bass	*Ambloplites rupestris*	0.74
Smallmouth Bass (subadult)	*Micropterus dolomieu*	0.79
Smallmouth Bass (adult)	*Micropterus dolomieu*	0.58
Spotted Bass	*Micropterus punctulatus*	0.18
Spotted Sucker	*Minytrema melanops*	0.09
Warmouth	*Lepomis gulosus*	0.38
White Crappie	*Pomoxis annularis*	0.10
White Sucker	*Catostomus commersoni*	0.18

Table 13. Summary of instream characteristics and spatial location for 80 channel unit sequences (CUS) nested within 22 stream reaches in the Ozark Highlands ecoregion of northeast Oklahoma and southwest Missouri surveyed from late spring to early autumn 2015. Water temperature variation is reported as a *z*-score.

Variable	Scale	Range	Mean ± SD
Mean water depth (m)	CUS	0.23-1.55	0.81 ± 0.28
Mean wetted channel width (m)	CUS	4.00-34.50	13.42 ± 6.25
Wetted width-depth ratio	CUS	4.55- 49.17	17.75 ± 8.48
Proportion emergent vegetation	CUS	0.00-0.31	0.03 ± 0.05
Proportion instream large wood	CUS	0.00-0.40	0.04 ± 0.06
Proportion total cover	CUS	0.00-0.40	0.06 ± 0.08
Proportion riffle	CUS	0.02-0.68	0.26 ± 0.16
Area (m^2)	CUS	75.00-12938.00	2422.96 ± 2433.16
Length (m)	CUS	15.00-456.00	154.53 ± 99.71
Bankfull width-depth ratio	Reach	17.00-67.00	38.59 ± 14.37
D50 (mm)	Reach	20.00-131.00	38.11 ± 27.41
Water temperature variation	Reach	-1.37-1.87	-0.10 ± 0.92
Proportion riffle	Reach	0.12-0.42	0.24 ± 0.09
Residual pool depth (m)	Reach	0.50-1.70	1.09 ± 0.35
Proportion emergent vegetation	Reach	0.00-0.21	0.03 ± 0.05
Proportion instream large wood	Reach	0.00-0.10	0.04 ± 0.02
Proportion total cover	Reach	0.00-0.23	0.06 ± 0.05
Distance to impoundment (km)	Reach	2.00-71.00	18.35 ± 15.94

Table 14. Comparison of the number of species detected using single-pass electrofishing and single-pass snorkeling for 19 stream fishes at 23 stream reaches in the Ozark Highlands ecoregion (see Appendix 2 to interpret reach codes). Columns 2-20 provide records of species detection at each site (see Appendix 1 to interpret two-letter species codes), where E indicates electrofishing, S indicates snorkeling, zero indicates detection, and one indicates non-detection. Species records with an NA correspond to sites sampled in 2014, where less species were considered. The last column provides both the number of species detected with each sampling method at each site and the total number of species detected at each site, where the number in the parentheses is the total number of species detected considering both sampling methods.

	BS	BG	CC	GS	LB	LS	MX	NH	NS	OS	RS	RB	RC	SB	SS	WA	WC	WS	Total detected
Site	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S	E/S (Both)
BAFO2	NA	0/0	NA	1/1	1/1	1/1	NA	NA	NA	0/0	0/0	1/1	NA	1/1	NA	0/0	0/0	NA	5/5 (6)
BISU1	1/0	1/1	0/0	1/1	1/1	1/1	1/1	1/1	1/1	0/0	1/1	1/1	1/1	1/1	0/0	0/0	0/0	1/1	13/12 (13)
BUFF1	NA	1/1	NA	1/1	1/1	1/1	NA	NA	NA	0/0	1/0	1/1	NA	1/1	NA	1/1	0/0	NA	8/7 (8)
BUFF2	NA	1/1	NA	1/1	1/1	1/1	NA	NA	NA	1/0	0/1	1/1	NA	1/1	NA	1/0	0/0	NA	8/7 9)
BUFF3	1/1	1/1	1/1	1/1	1/1	1/1	1/1	1/1	1/0	0/0	0/0	1/1	1/1	1/1	0/0	1/0	1/0	1/0	15/11 (15)
BUFF4	1/0	1/1	0/0	1/1	1/1	1/1	1/1	1/1	0/0	0/0	1/1	1/1	1/1	1/1	0/0	1/1	0/0	1/0	13/11 (13)
BUTL1	1/0	1/1	1/1	1/1	1/1	1/1	1/1	1/1	1/1	0/0	1/1	1/1	1/1	1/1	0/0	1/0	0/0	1/0	15/12 (15)
CANE1	1/1	1/1	0/0	1/1	1/1	1/1	1/1	1/1	0/0	0/0	1/0	1/1	1/1	1/1	1/0	1/1	0/0	1/0	14/11 (14)
EVAN1	NA	1/1	NA	1/1	1/1	1/1	NA	NA	NA	0/0	0/0	1/1	NA	1/1	NA	0/0	0/0	NA	6/6 (6)

EVAN2	NA	1/1	NA	1/1	0/1	1/1	NA	NA	NA	0/0	0/0	1/1	NA	1/1	NA	0/0	0/1	NA	5/7 (7)
FIVE1	1/1	1/1	0/1	1/1	1/1	1/1	1/1	1/1	1/1	0/0	1/0	1/1	1/0	1/1	0/0	1/0	0/0	0/0	13/11 (14)
FLIN2	NA	1/1	NA	1/1	1/1	1/1	NA	NA	NA	0/0	1/1	1/1	NA	1/1	NA	0/0	0/0	NA	7/7 (7)
FOUR1	NA	1/1	NA	1/1	1/1	1/1	NA	NA	NA	1/0	0/0	1/1	NA	1/1	NA	1/0	0/0	NA	8/6 (8)
FOUR2	NA	1/1	NA	1/1	1/1	1/1	NA	NA	NA	0/0	1/0	1/1	NA	1/1	NA	0/0	0/0	NA	7/6 (7)
SALI1	NA	1/1	NA	1/0	1/0	1/1	NA	NA	NA	0/0	0/0	1/1	NA	1/1	NA	0/0	0/0	NA	6/4 (6)
SPAV1	NA	1/1	NA	1/1	1/1	1/1	NA	NA	NA	0/0	0/0	1/1	NA	1/1	NA	0/0	0/0	NA	6/6 (6)
SPAV3	NA	1/1	NA	1/0	1/1	1/1	NA	NA	NA	0/0	0/0	1/1	NA	1/1	NA	0/0	0/0	NA	6/5 (6)
SPAV4	NA	1/1	NA	1/0	1/1	1/1	NA	NA	NA	0/0	1/1	1/0	NA	1/1	NA	1/0	0/0	NA	8/6 (9)
SPAV5	NA	1/1	NA	1/0	1/1	1/1	NA	NA	NA	0/0	1/0	1/1	NA	1/1	NA	0/0	0/0	NA	7/6 (8)
SPRI1	NA	0/0	NA	1/1	0/0	0/0	NA	NA	NA	0/0	0/0	1/1	NA	1/1	NA	0/0	0/0	NA	3/3 (3)
SPRI2	NA	1/0	NA	1/1	1/0	0/0	NA	NA	NA	0/0	0/0	1/1	NA	1/1	NA	0/0	0/0	NA	5/3 (5)
SPRI3	1/1	1/1	1/1	1/1	1/1	0/0	1/1	1/1	0/0	0/0	0/0	1/1	1/1	1/1	0/0	1/1	0/0	1/0	12/11 (12)
SPRI4	1/1	1/1	1/1	1/1	1/1	1/1	1/1	1/1	0/0	0/0	0/0	1/1	1/1	1/1	0/0	1/0	0/0	1/0	13/11 (13)

Table 15. Percentage of lotic stream-fish studies (n = 204) that evaluated sampling gears or methods in freshwater ecosystems that were given a response of "yes" for each of five criteria.

Criteria	Total	Percent of total
Did the study quantitatively evaluate a sampling gear or method?	193	94.4
Did the study quantitatively compare sampling gear types?	107	52.5
Did the study identify fish-level or habitat-level sources of variability between gear types or methods?	145	70.9
Did the study quantify capture probability	76	37.4
Did the study quantify species detection probability?	17	8.4

Table 16. Fixed effects and species-dependent coefficients (random effects) from a linear mixed model to identify environmental factors related to variation in sunfish density, where sunfish density was natural-log transformed. Standard error is reported for fixed effects and standard deviation is reported for random effects. All continuous variables were standardized such that each had a mean of zero and a standard deviation of one, where the model intercept estimates sunfish density at mean conditions and coefficients for continuous variables represent a unit change of one standard deviation.

Effect	Intercept	Bankfull width
Fixed	-4.62 ± 0.56	-0.30 ± 0.21
Bluegill	-4.35 ± 0.21	-0.73 ± 0.17
Green Sunfish	-3.98 ± 0.21	-0.57 ± 0.17
Longear Sunfish	-2.86 ± 0.21	-0.44 ± 0.17
Redear Sunfish	-6.21 ± 0.21	-0.33 ± 0.17
Rock Bass	-4.12 ± 0.21	0.46 ± 0.17
Warmouth	-6.17 ± 0.21	-0.19 ± 0.17

Table 17. Coefficients and bootstrapped 95% confidence intervals (CI) for main effects from a generalized linear mixed model developed to examine heterogeneity in electrofishing capture probability among stream fishes of the Ozark Highlands ecoregion. Coefficients are reported on a logit scale. All continuous variables were standardized such that each had a mean of zero and a standard deviation of one, where the model intercept estimates capture probability at mean conditions and coefficients for continuous variables represent a unit change of one standard deviation. Geosoil was defined as a categorical variable with shale as the reference category. Asterisks indicate reach-level variables that were also modeled using random species slopes (Table 17).

Parameter	Coefficent ± SE	95% CI
Intercept	-1.71 ± 0.19	(-2.09, -1.35)
TL	0.31 ± 0.02	(0.25, 0.36)
*Discharge	-0.25 ± 0.09	(-0.42, -0.07)
Percent emergent vegetation	-0.24 ± 0.06	(-0.37, -0.12)
Percent riffle	0.08 ± 0.06	(-0.04, 0.20)
Water conductivity	0.18 ± 0.07	(0.05, 0.31)
Water clarity	-0.19 ± 0.06	(-0.31, -0.09)
*Water depth	-0.10 ± 0.09	(-0.27, 0.06)
Width-depth ratio	-0.13 ± 0.06	(-0.25, -0.01)
Discharge x percent riffle	-0.16 ± 0.05	(-0.26, -0.05)
Geosoil (Cherty alluvium)	0.54 ± 0.19	(0.16, 0.95)
Geosoil (Stony alluvium)	1.22 ± 0.24	(0.75, 1.71)
Geosoil (Cherty limestone)	0.38 ± 0.19	(0.03, 0.76)

Table 18. Species-dependent coefficients (i.e., random intercepts and slopes) from a generalized linear mixed model developed to examine heterogeneity in electrofishing capture probability among stream fishes of the Ozark Highlands ecoregion (Table 17).

Species	Intercept ± SD	Discharge ± SD	Water depth ± SD
Bluegill	-1.53 ± 0.06	-0.25 ± 0.07	-0.06 ± 0.06
Green sunfish	-1.48 ± 0.06	-0.07 ± 0.06	-0.10 ± 0.05
Largemouth bass	-1.91 ± 0.10	-0.32 ± 0.09	-0.13 ± 0.08
Longear sunfish	-1.61 ± 0.05	-0.30 ± 0.04	0.02 ± 0.04
Redear sunfish	-1.54 ± 0.15	-0.31 ± 0.11	0.01 ± 0.11
Rock bass	-1.91 ± 0.07	-0.14 ± 0.07	-0.15 ± 0.06
Smallmouth bass	-2.20 ± 0.08	-0.32 ± 0.07	-0.42 ± 0.06
Spotted bass	-1.80 ± 0.21	-0.28 ± 0.11	-0.12 ± 0.13
Warmouth	-1.37 ± 0.12	-0.25 ± 0.10	0.06 ± 0.09

Table 19. Results of a leave-one-out cross validation that evaluated the performance of a generalized linear mixed model for estimating tow-barge electrofishing capture probability among fishes in 34 steam reaches of the Ozark Highlands: root mean square deviation (RMSD), the number of estimates contained in the 95% binomial probability confidence intervals (CI; contained), and the number of times predicted 95% CI overlapped with the binomial probability 95% CI (overlapped).

Species	Bias	RMSD	Contained 95% CI	Overlapped 95% CI
All	0.00	0.10	90% (112 of 124)	99% (123 of 124)
Bluegill	-0.01	0.10	95% (18 of 19)	100% (19 of 19)
Green sunfish	0.01	0.09	91% (20 of 22)	100% (22 of 22)
Largemouth bass	0.00	0.07	100% (10 of 10)	100% (10 of 10)
Longear sunfish	0.02	0.08	87% (26 of 30)	97% (29 of 30)
Redear sunfish	-0.04	0.07	100% (2 of 2)	100% (2 of 2)
Rock bass	0.01	0.11	90% (18 of 20)	100% (20 of 20)
Smallmouth bass	0.00	0.13	88% (15 of 17)	100% (17 of 17)
Warmouth	-0.05	0.11	75% (3 of 4)	100% (4of 4)

Table 20. Coefficients from a multinomial negative-binomial-mixture capture-recapture model to estimate the abundance of 25 Smallmouth Bass subpopulations in streams of the Ozarks Highlands surveyed using tow-barge electrofishing. The model had a high level of support (AICc weight = 0.95; Table 9). Capture probability covariates were reported on a logit scale and were standardized such that the intercept estimates detection at mean values and the coefficients represent a unit change of one SD.

Parameter	Estimate ± SE	Lower 95% CI	Upper 95% CI
Latent abundance	136 ± 17.6	4.66	5.18
Capture probability - intercept	-1.08 ± 0.08	-1.24	-0.92
Capture probability - water clarity	0.24 ± 0.07	0.11	0.36
Capture probability - effort	0.28 ± 0.06	0.17	0.39
Capture probability - wetted width	-0.13 ± 0.09	-0.30	0.04
Capture probability - depth	-0.39 ± 0.07	-0.52	-0.25
Capture probability – width x depth	0.21 ± 0.06	0.09	0.33
Overdispersion	3.01 ± 0.88	0.49	1.65

Table 21. Coefficients from a multinomial negative-binomial-mixture removal model to estimate the abundance of 25 Smallmouth Bass subpopulations in streams of the Ozarks Highlands surveyed using tow-barge electrofishing. The model was chosen using AICc from a set of 12 candidate models that incorporated the capture probability covariates discharge, mean wetted width, mean depth, water clarity, and electrofishing effort. The model had a high level of support (AICc weight = 0.51; Appendix 1). Capture probability covariates were reported on a logit scale and were standardized such that the intercept estimates detection at mean values and the coefficients represent a unit change of one SD. The estimate of c-hat for the model from the Chi-squared test (c-hat =1.02) did not indicate overdispersion in the model. A plot of predicted versus fitted residuals (n = 50) also suggested adequate model fit (i.e., no evidence of heteroscedasticity).

Parameter	Estimate ± SE	Lower 95% CI	Upper 95% CI
Latent abundance	59 ± 9.2	3.79	4.41
Capture probability - intercept	-0.36 ± 0.17	-0.70	-0.05
Capture probability - wetted width	-0.48 ± 0.14	-0.79	-0.17
Capture probability - mean depth	-0.47 ± 0.16	-0.75	-0.21
Overdispersion	2.61 ± 0.76	0.35	1.51

Table 22. Summary and comparison of Smallmouth Bass tow-barge electrofishing abundance estimates derived from a multinomial negative-binomial-mixture removal model for 25 subpopulations in streams of the Ozarks Highlands, where pass 1 is the number of fish captured on removal pass 1 and pass 2 is the number of fish captured on removal pass 2 ,and cumulative detection is the estimated proportion of Smallmouth Bass captured across both capture and recapture events and was calculated as the sum of the multinomial cell probabilities. Petersen abundance estimates were calculated using Peterson capture-recapture with the Chapman (1954) bias correction. Petersen 95% CI were calculated with a bias-corrected SE (Seber 1970) as $\hat{N} \pm z\alpha/2(\mathrm{SE})$. NA's for Petersen abundance estimates and Petersen 95% CI indicate sites where assumptions for unbiased estimates were not met.

Site	Pass 1	Pass 2	Removal cumulative capture probability	Capture-recapture cumulative capture probability	Snorkel count	Removal 95% CI	Petersen 95% CI
Baron Fork4	28	9	0.77	0.46	NA	41 - 56	147 - 451
Baron Fork5	13	5	0.85	0.54	NA	18 - 26	NA
Big Sugar Creek2	17	11	0.25	0.23	138	76 - 141	NA
Buffalo Creek5	79	18	0.90	0.75	153	102 - 115	167 - 241
Buffalo Creek6	9	1	0.50	0.32	85	14 - 32	NA
Buffalo Creek7	7	8	0.28	0.35	59	34 - 77	NA
Butler Creek2	43	32	0.65	0.43	244	101 - 131	123 - 204
Caney Creek2	4	4	0.69	0.45	18	9 - 16	NA
Evansville Creek3	28	16	0.69	0.52	NA	54 - 75	101 - 155
Five-mile Creek2	6	6	0.78	0.50	61	12 - 21	NA

Flint Creek3	12	9	0.55	0.33	NA	29 - 51	NA
Flint Creek4	23	22	0.55	0.30	208	67 - 98	NA
14-mile Creek4	26	14	0.69	0.40	NA	49 - 69	75 - 224
14-mile Creek5	29	8	0.78	0.51	NA	41 - 56	57 - 268
14-mile Creek6	78	19	0.95	0.84	NA	98 - 106	157 - 188
Honey Creek3	30	16	0.81	0.62	NA	50 - 64	93 - 141
Honey Creek4	8	5	0.36	0.30	NA	24 - 55	NA
Indian Creek2	10	4	0.45	0.31	NA	22 - 46	NA
Lost Creek2	7	2	0.81	0.62	NA	9 - 16	NA
Saline Creek3	19	10	0.73	0.40	156	33 - 48	NA
Spavinaw Creek8	30	29	0.55	0.40	167	89 - 125	NA
Spavinaw Creek9	32	10	0.55	0.44	NA	62 - 92	64 - 120
Spring Creek5	23	16	0.50	0.33	247	62 - 95	66 - 174
Spring Creek6	47	38	0.69	0.62	225	109 - 137	167 - 248
Spring Creek7	11	6	0.69	0.40	84	19 - 33	NA

Table 23. Fixed effects and species-dependent coefficients with standard error from a multiscale GLMM to identify environmental and spatial factors related to variation in occurrence for 20 stream fishes of the Ozark Highlands ecoregion. Standard error is reported for fixed effects and standard deviation is reported for species-dependent terms. Stream-fish occurrence probability is reported on a logit scale. All variables were standardized such that each had a mean of zero and a standard deviation of one, where the model intercept estimated the probability of stream-fish occurrence at mean conditions and coefficients for represent a unit change of one standard deviation.

Effect	Intercept	Channel unit sequence area (m^2)	Reach D50 (mm)	Reach water temperature variation	Reach residual pool depth (m)
Fixed	-0.01 ± 0.56	0.92 ± 0.13	0.03 ± 0.18	0.04 ± 0.24	-0.04 ± 0.15
Banded Sculpin	1.13 ± 0.38	0.69 ± 0.14	-0.09 ± 0.26	-0.67 ± 0.26	-0.14 ± 0.15
Bluegill	1.36 ± 0.38	0.95 ± 0.16	0.16 ± 0.27	0.64 ± 0.30	0.17 ± 0.15
Creek Chub	-0.39 ± 0.36	1.09 ± 0.14	-0.24 ± 0.22	0.27 ± 0.25	-0.15 ± 0.14
Green Sunfish	2.12 ± 0.44	0.88 ± 0.17	0.36 ± 0.32	0.91 ± 0.39	0.38 ± 0.19
Largemouth Bass (subadult)	0.86 ± 0.36	0.69 ± 0.15	0.68 ± 0.27	0.36 ± 0.27	0.38 ± 0.15
Largemouth Bass (adult)	-1.27 ± 0.38	0.77 ± 0.15	0.70 ± 0.25	0.29 ± 0.26	0.29 ± 0.14
Longear Sunfish	2.15 ± 0.45	0.86 ± 0.17	0.41 ± 0.32	0.93 ± 0.40	0.41 ± 0.19
Northern Hogsucker	1.37 ± 0.39	0.81 ± 0.14	-0.27 ± 0.26	-0.51 ± 0.26	-0.21 ± 0.15
Northern Studfish	-1.83 ± 0.42	1.38 ± 0.17	-0.69 ± 0.30	0.10 ± 0.30	-0.53 ± 0.18
Redear Sunfish	-1.09 ± 0.37	1.03 ± 0.14	0.25 ± 0.23	0.66 ± 0.26	0.14 ± 0.14
Redhorses	0.62 ± 0.36	0.92 ± 0.14	0.11 ± 0.24	0.29 ± 0.25	0.07 ± 0.14

IX. FIGURES

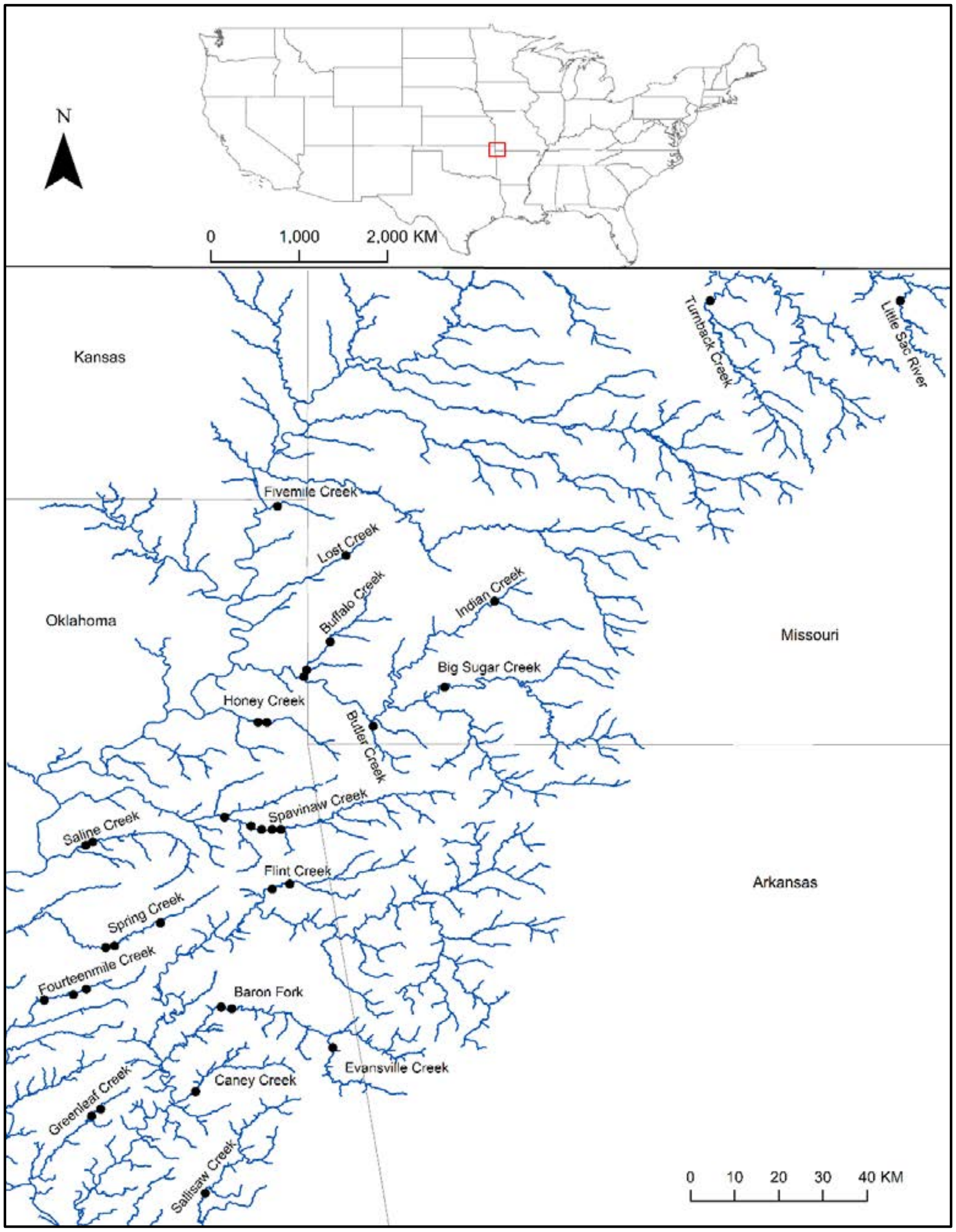

Figure 1. Study sites in the Ozark Highland ecoregion of northeast Oklahoma and southwest Missouri.

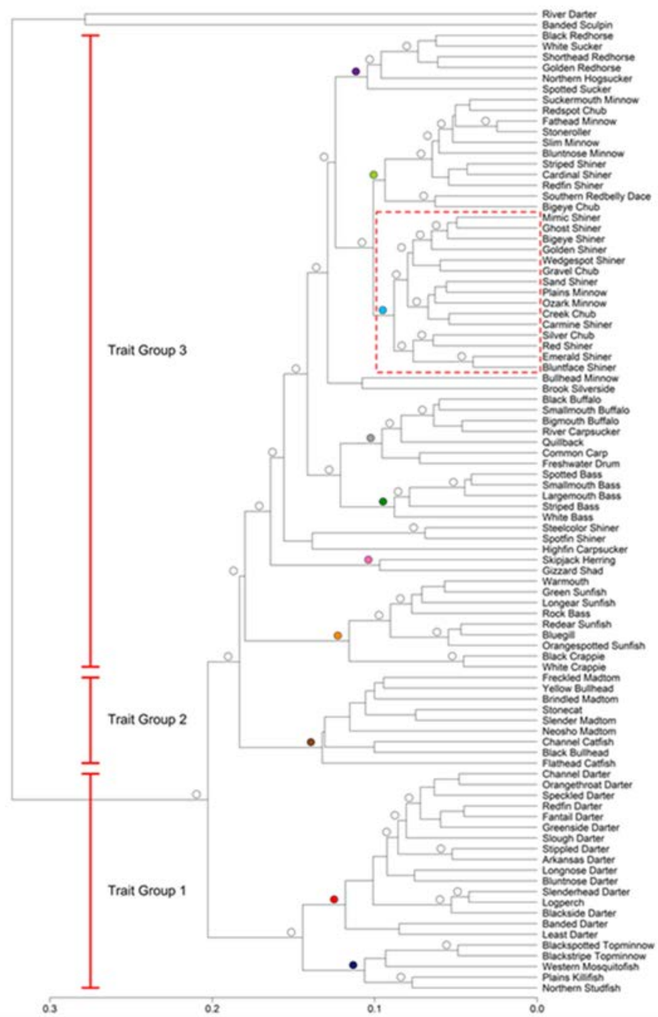

Figure 2. Hierarchical clustering with multiscale bootstrapping (HCMB) for 92 Ozark Highland stream fishes using external morphological traits. Values at nodes of the dendrogram are approximately unbiased (AU) p-values. Standard error was ≤ 0.01 for all AU p-values. Significant clusters (n = 54) were identified as (AU p-value minus SE) ≥ 0.95 and are indicated by circles at nodes of the dendrogram. Vertical red lines with bars indicate species members of trait groups 1-3 comprising 90 species. Color-coded circles represent 10 significant clusters comprising 85 species (trait groups A through J), where purple is trait group A, light green is trait group B, light blue is trait group C, light gray is trait group D, dark green is trait group E, pink is trait group F, orange is trait group G, brown is trait group H, red is trait group I, and dark blue is trait group J. The red box around trait group C highlights one example of significant

nested clusters.

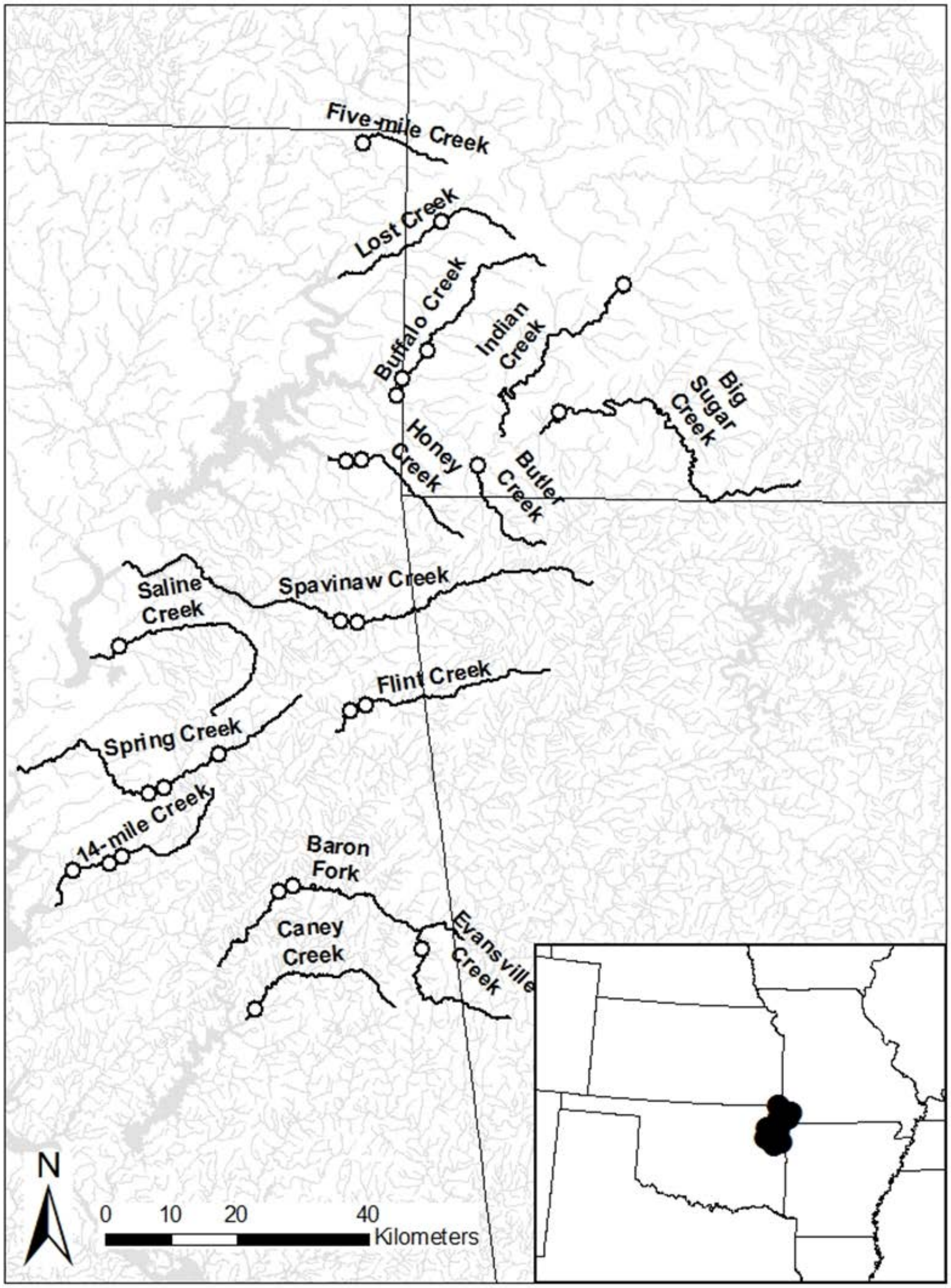

Figure 3. Location of 25 stream reaches of the Ozarks Highlands in northeast Oklahoma and southwest Missouri where Smallmouth Bass populations were surveyed using tow-barge electrofishing and snorkeling from July to October 2014-2015.

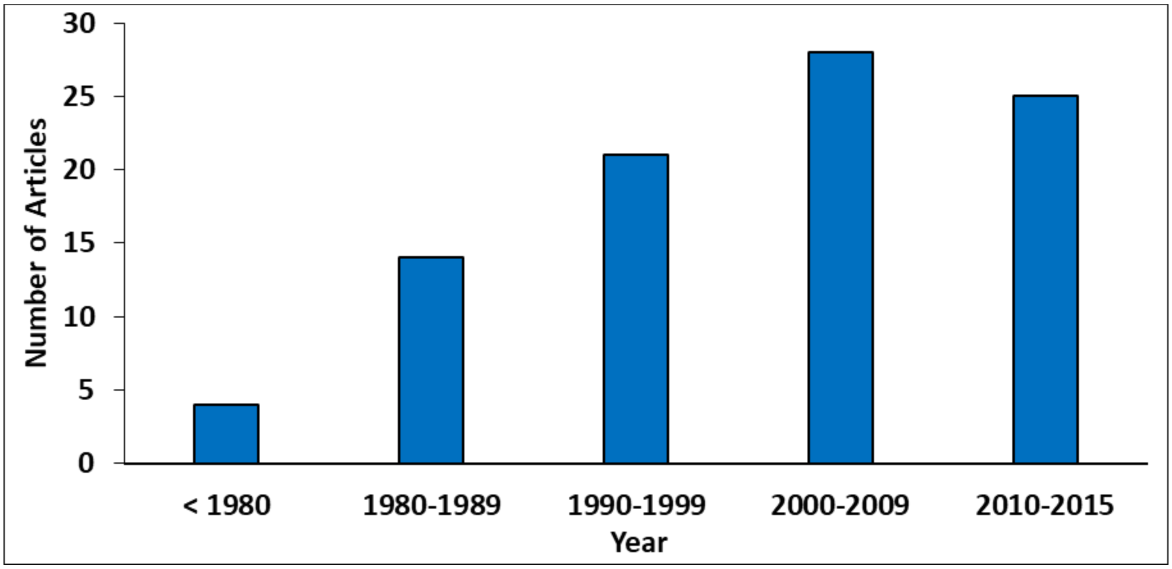

Figure 4. The number of articles where authors considered capture efficiency of their sampling gear from 1945-2015.

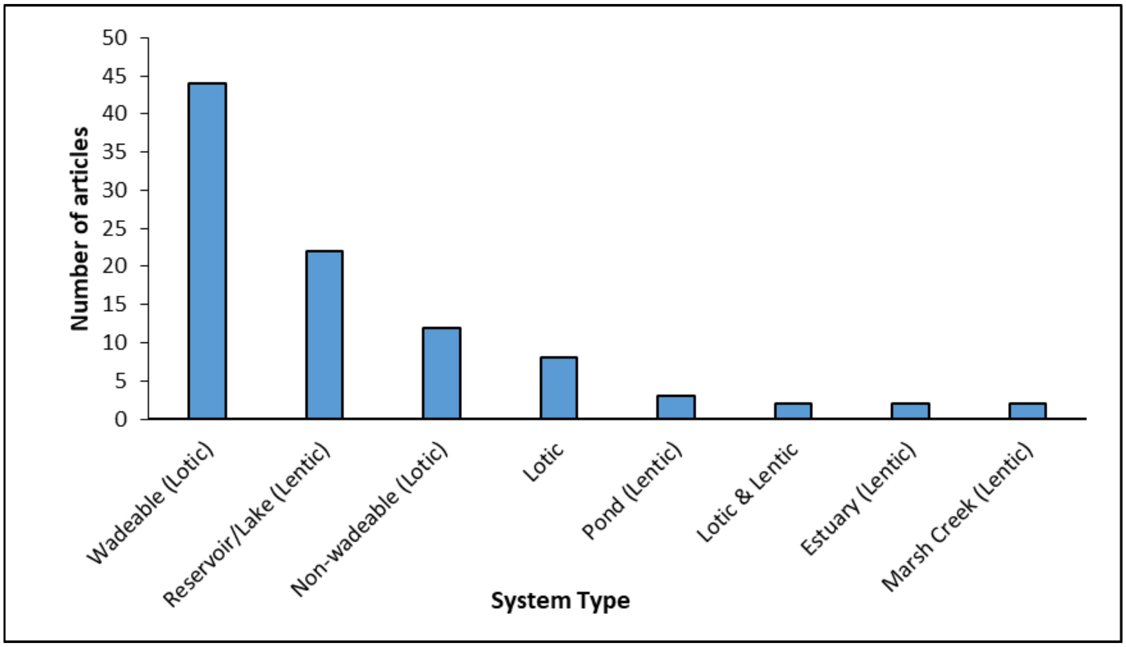

Figure 5. The system types where capture efficiency was the focus of a published study. Reviwed articles were obtained using key words that targed manuscripts published from 1945-2015.

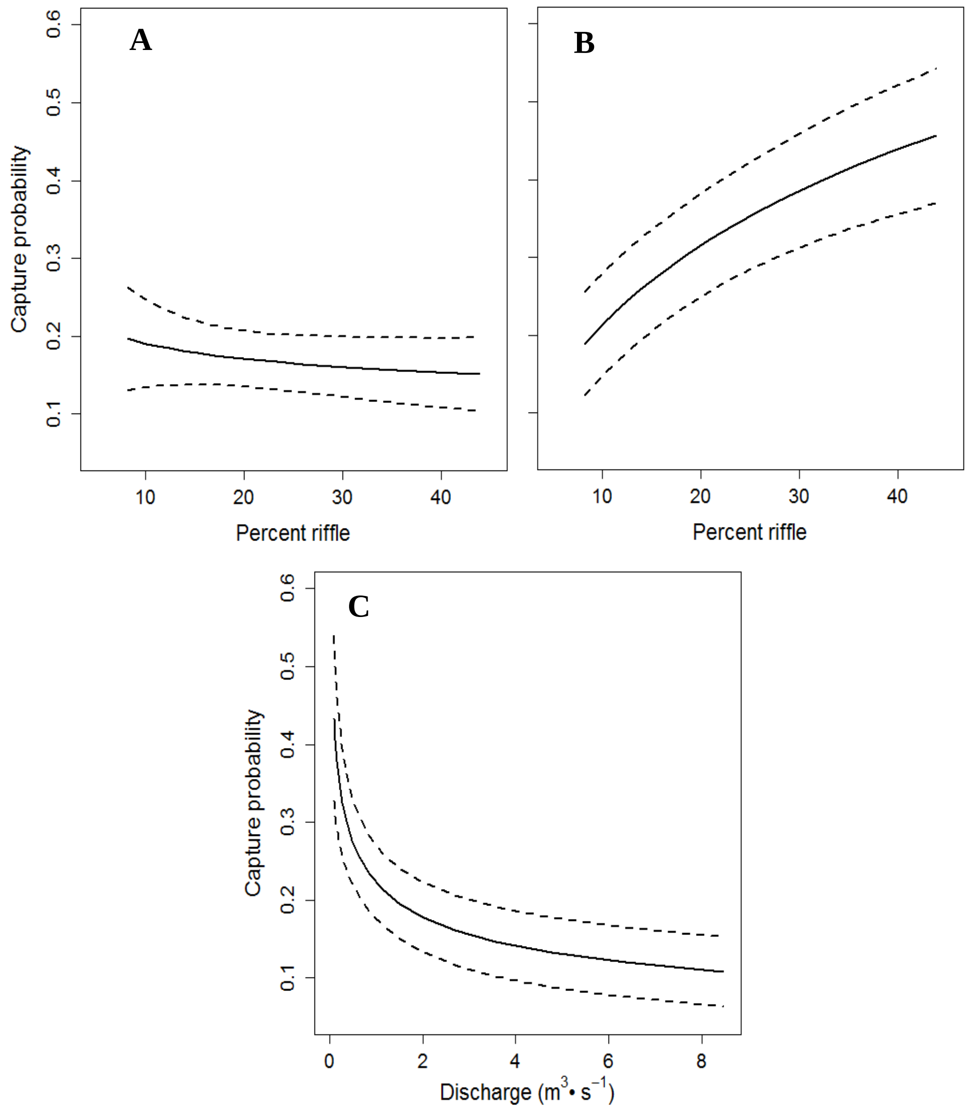

Figure 6. Electrofishing capture probability across levels of discharge and percent riffle habitat using estimates from a generalized linear mixed model (Table 16). Panel A shows the relationship between capture probability and riffle habitat at higher flows (discharge held at 1 SD). Panel B shows the relationship between capture probability and riffle habitat at low flows (discharge held at -1.5 SD). Panel C shows the relationship between capture probability and discharge with percent riffle held at 1 SD, where capture probability increases sharply at lower flow due to an interaction. The x-axes represent levels of either percent riffle or discharge from -2 to 2 SD. Capture probability was calculated for the average-sized Longear Sunfish, although the general relationships were similar among species and across fish size (see also Figure 5).

Other reach-level predictor variables included in the model were held at mean values and the geosoil category was cherty limestone (see Table 3). Dashed lines are 95% confidence intervals.

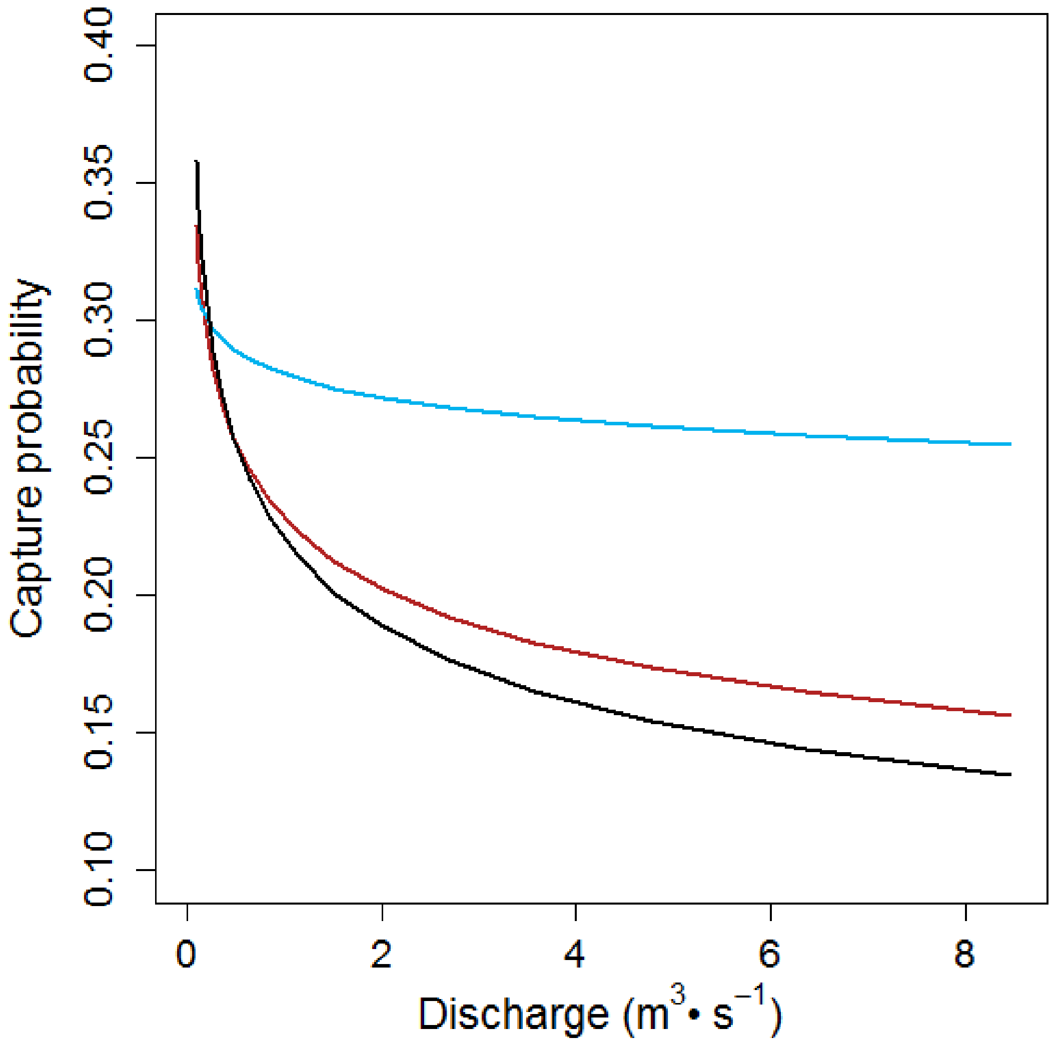

Figure 7. Relationship between electrofishing capture probability and discharge among stream fishes of the Ozark Highlands using species-dependent random terms derived from a generalized linear mixed model (Table 17). Red represents the average-size Bluegill, blue represents the average-size Green Sunfish, and black represents the average-size Smallmouth Bass. The x-axis represents levels of discharge from -2 to 2 SD. Other reach-level variables included in the model were held at mean values and the geosoil category was cherty limestone.

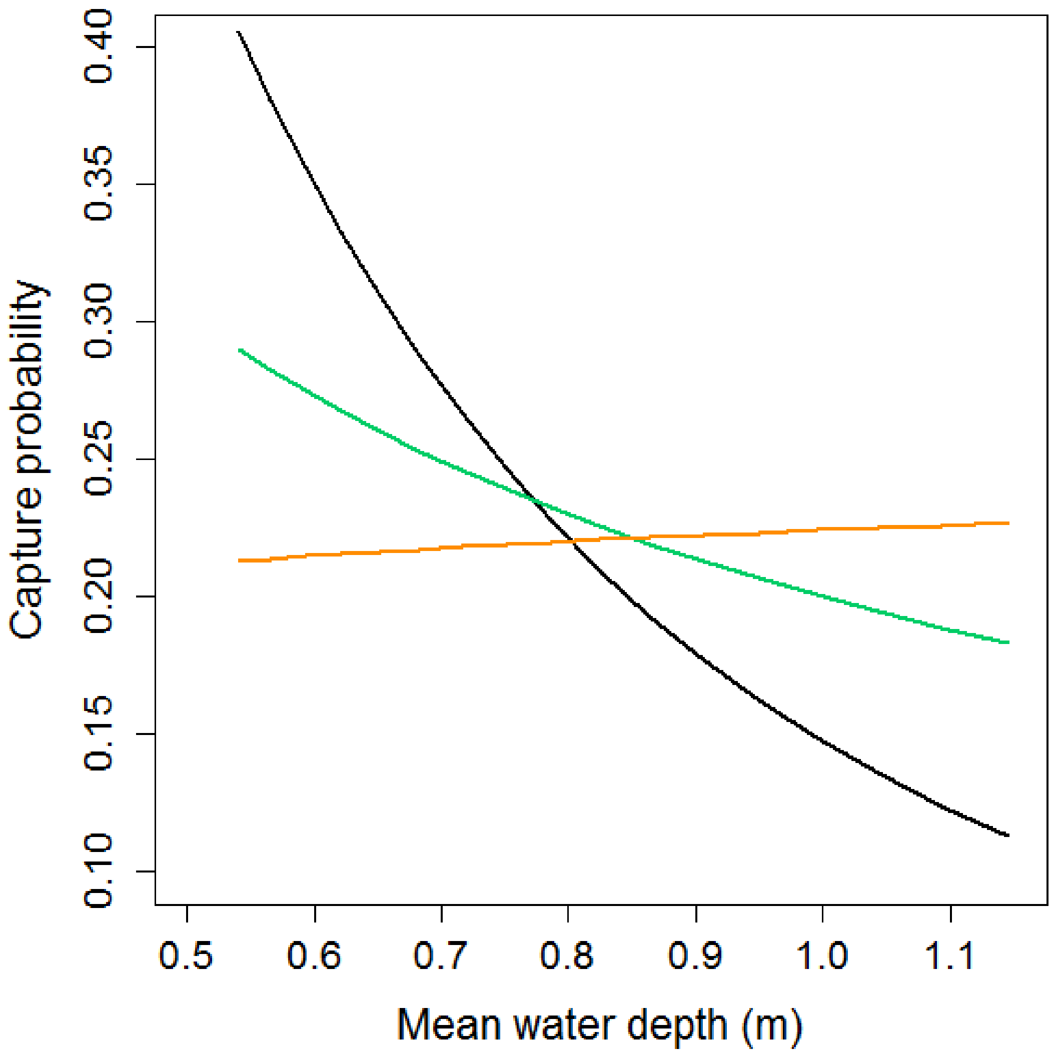

Figure 8. Relationship between electrofishing capture probability and water depth among stream fishes of the Ozark Highlands using species-dependent random terms derived from a generalized linear mixed model (Table 17). Orange represents the average-size Longear Sunfish, green represents the average-size Rock Bass, and black represents the average-size Smallmouth Bass. The x-axis represents levels of mean water depth from -2 to 2 SD. Other reach-level variables included in the model were held at mean values and the geosoil category was cherty limestone.

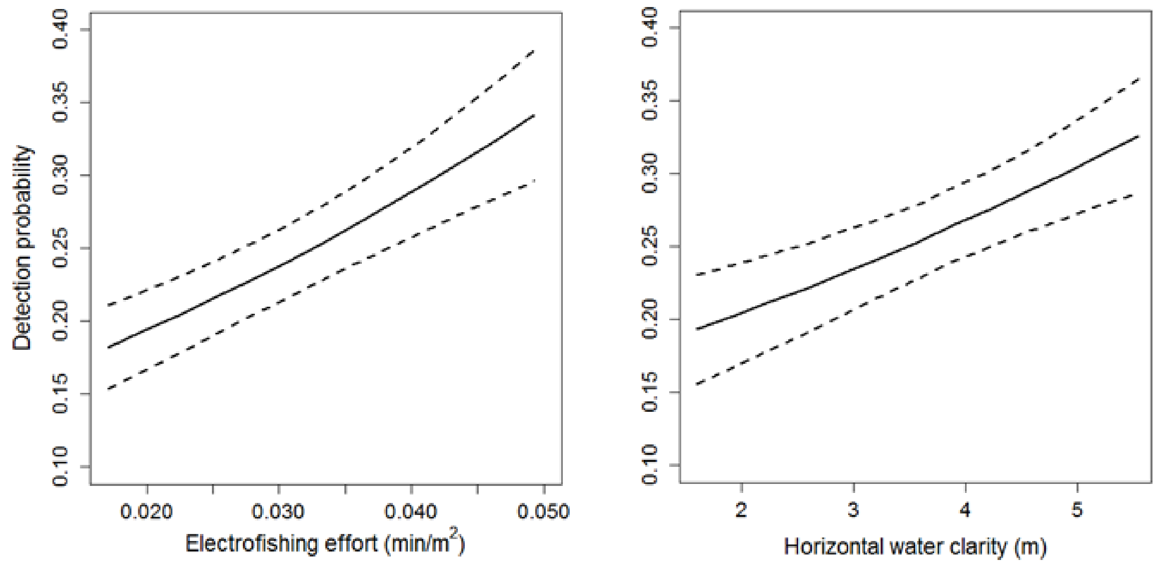

Figure 9. The relationship between Smallmouth Bass electrofishing capture probability and increasing effort (a) and water clarity (b) in Ozarks Highland streams. Capture probability estimates were derived from a multinomial negative-binomial capture-recapture model with mean wetted channel width and mean depth held at mean sampling levels. Dashed lines are 95% confidence intervals.

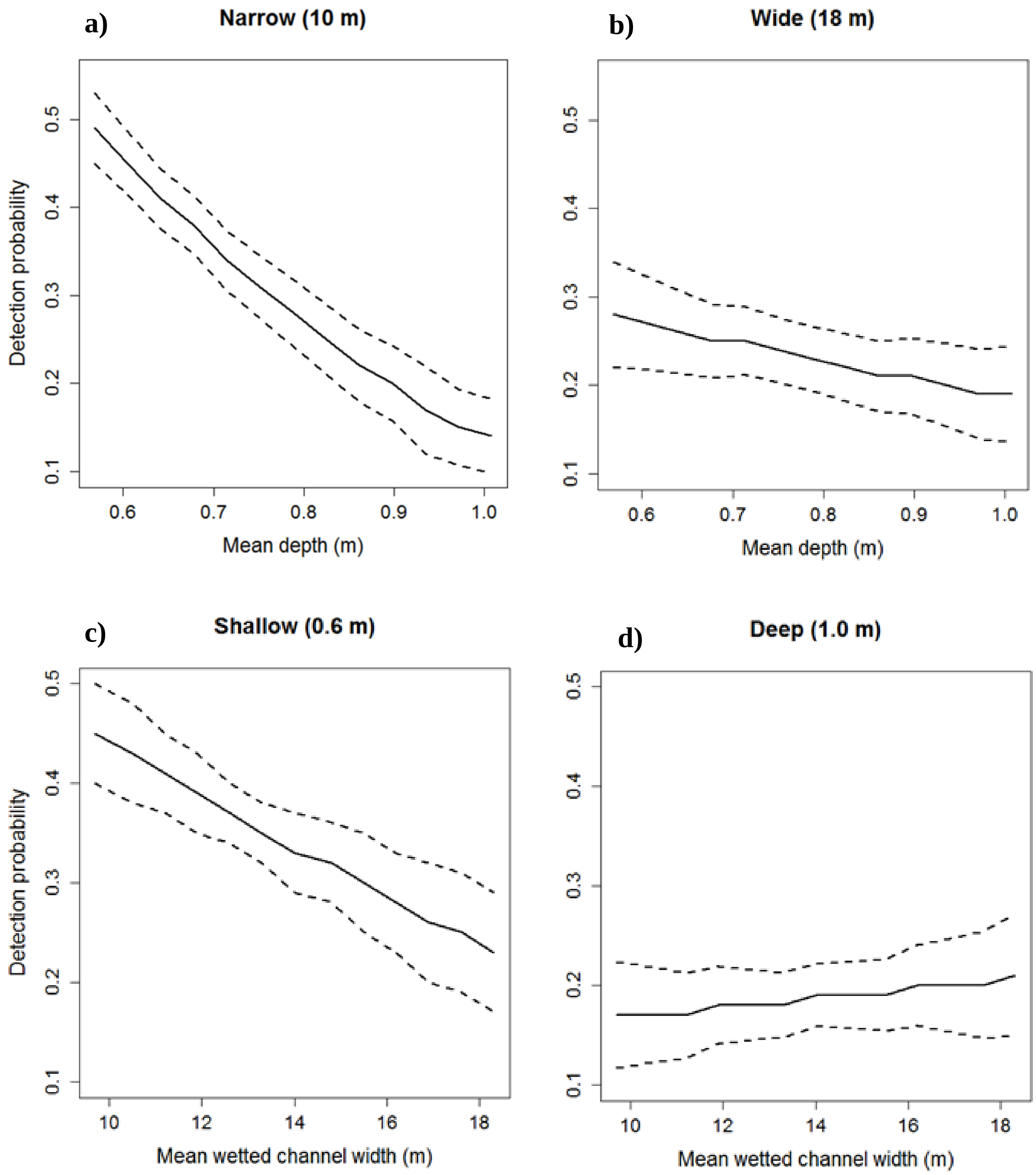

Figure 10. Interaction of mean wetted width and mean depth on Smallmouth Bass electrofishing capture probability in Ozark Highland streams. Narrow and wide represent values of mean wetted width of -1 and +1 SD, respectively. Shallow and deep represent values of mean depth of -1 and +1 SD, respectively. Dashed lines indicate 95% confidence intervals

X. APPENDICES

Appendix 1. Sites and site code for 23 stream reaches in 19 streams of the Ozark Highlands ecoregion sampled to evaluate tow-barge electrofishing and snorkeling for species detection and evaluate snorkeling to provide reliable stream fish abundance estimates (see also Table 1 and Figure 2)

Site code	Site
BAFO2	Baron Fork2
BISU1	Big Sugar Creek
BUFF1	Buffalo Creek1
BUFF2	Buffalo Creek2
BUFF3	Buffalo Creek3
BUFF4	Buffalo Creek4
BUTL1	Butler Creek
CANE1	Caney Creek
EVAN1	Evansville Creek1
EVAN2	Evansville Creek2
FIVE1	Five-mile Creek
FLIN2	Flint Creek2
FOUR1	14-mile Creek1
FOUR2	14-mile Creek2
SALI1	Saline Creek1
SPAV1	Spavinaw Creek1
SPAV3	Spavinaw Creek3
SPAV4	Spavinaw Creek4
SPAV5	Spavinaw Creek5
SPRI1	Spring Creek1
SPRI2	Spring Creek2
SPRI3	Spring Creek3
SPRI4	Spring Creek4

Appendix 2. Species code, common name, and scientific name of 19 Ozark Highland stream fishes included in a comparison species detection between single-pass tow-barge electrofishing and single-pass snorkeling for species detection and an evaluation of snorkeling to provide reliable stream fish abundance estimates. Species with a single asterisk were considered species in both 2014 and 2015. Species with a double asterisk were considered only in 2015.

Species code	Common name	Latin name
MX	**Redhorses	*Moxostoma* spp.
MH	**Northern Hogsucker	*Hypentelium nigricans*
SS	**Spotted Sucker	*Minytrema melanops*
WS	**White Sucker	*Catostomus commersoni*
BB	*Largemouth Bass	*Micropterus salmoides*
BB	*Spotted Bass	*Micropterus punctulatus*
SB	*Smallmouth Bass	*Micropterus dolomieu*
BG	*Bluegill	*Lepomis macrochirus*
GF	*Green Sunfish	*Lepomis cyanellus*
LS	*Longear Sunfish	*Lepomis megalotis*
OS	*Orangespotted Sunfish	*Lepomis humilis*
RS	*Redear Sunfish	*Lepomis microlophus*
RB	*Rock Bass	*Ambloplites rupestris*
WH	*Warmouth	*Lepomis gulosus*
WC	*White Crappie	*Pomoxis annularis*
RC	**Redspot Chub	*Nocomis asper*
CC	**Creek Chub	*Semotilus atromaculatus*
NS	**Northern Studfish	*Fundulus catenatus*
BS	**Banded Sculpin	*Cottus carolinae*