DEVELOPMENT AND EVALUATION OF SAMPLING PROTOCOLS FOR AT-RISK FISHES IN WADEABLE WARMWATER STREAMS

by

ALISON LOUISE PRICE

(Under the direction of James T. Peterson)

ABSTRACT

Natural resource managers make decisions based on the analyses of fish sample data; hence it is imperative to use unbiased data with low variance. The ability to capture fishes (capture efficiency) varies among methods and site- and species-characteristics. Failure to account for capture efficiency can bias sample data. High variance also influences data quality and can obfuscate important relationships and population trends. I evaluated two commonly-used fish sampling protocols in 31 streams the Upper Coosa Basin, Georgia, modeled fish capture efficiency, and evaluated the sources of sample variance. Capture efficiency was low and varied among species, between methods, and was related to stream habitat characteristics. I also estimated fish movement out of unblocked sample units during sampling and calculated unconditional capture efficiency. Failure to account for incomplete capture of fish introduces bias and variance in sampling data and may lead to poor inference, particularly for at-risk fishes.

INDEX WORDS: stream fish, sampling, protocol, capture efficiency, capture-recapture, backpack electrofishing, seining

DEVELOPMENT AND EVALUATION OF SAMPLING PROTOCOLS FOR AT-RISK FISHES IN WADEABLE WARMWATER STREAMS

by

ALISON LOUISE PRICE

B.S., North Carolina State University, 2004

A Thesis Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment
of the Requirements for the Degree

MASTER OF SCIENCE

ATHENS, GA

2008

DEVELOPMENT AND EVALUATION OF SAMPLING PROTOCOLS FOR AT-RISK FISHES IN WADEABLE WARMWATER STREAMS

by

ALISON LOUISE PRICE

Major Professor: James T. Peterson

Committee: Mary C. Freeman
Robert B. Bringolf

Electronic Version Approved:

Maureen Grasso
Dean of the Graduate School
The University of Georgia
December 2008

ACKNOWLEDGEMENTS

I would like to extend my gratitude to the many people who made this project possible. My major advisor, Jim Peterson, deserves many thanks for his continual guidance and helpful hints throughout the progression of my project. His analytical support and knowledge helped shape the project and bolstered my problem-solving skills. My committee members, Mary Freeman and Robert Bringolf provided field training, helpful suggestions and necessary moral support. I am grateful to the U.S. Fish and Wildlife Service for providing funding support, the Cohutta State Fish Hatchery and Shane Kornberg for providing lodging facilities, and many landowners in Georgia that allowed access to their property.

I would like to thank those that provided assistance during field sampling, including Justin Bezold, Will Bickerstaff, Scott Craven, Abby Farakesh, Becky Fauver, Megan Hagler Kevin McAbee, Jason Meador, Matt Mundy, Jill Olin and Kate Sparks. I received GIS assistance from Carrie Strait, and I am grateful to Dr. Bud Freeman and the Georgia Museum of Natural History for preserving collected specimens for perpetuity. In addition to those that assisted in the field, I would like to thank other lab-mates (Colin Shea), office-mates, book-lenders, and fellow members of the Georgia Cooperative Fish and Wildlife Unit for their humor, generosity, and words of wisdom.

I am especially grateful to my family (Karen & Hendal, Brandon, Justin) and my friends for their unwavering support, encouragement of my career, and jokes at my expense. This project would not have been as much fun without you!

TABLE OF CONTENTS

CHAPTER 1

INTRODUCTION

Natural resource managers are often faced with decisions that affect the status of fish and wildlife populations, and these decisions are typically based on analyses of fish sample data. Sample estimates of the abundance, distribution, and composition of fish communities also are used to assess the quality of aquatic resources (Karr et al. 1986). Because the quality of sample data can influence management decisions, it is imperative to use high quality data (Peterson and Rabeni 1995). The two main factors affecting the quality of fish sample data include variance and bias, with low bias and variance indicating higher quality data.

No fish sampling method can reliably collect all individuals or species in a sample unit. This incomplete capture results in biased estimates of fish abundance and distribution and community structure. The proportion or percentage of fishes captured, henceforth termed capture efficiency or catchability, is influenced by sample method, site-specific characteristics, and fish characteristics (Anderson 1995; Bayley and Dowling 1990; Peterson and Rabeni 2001; Peterson et al. 2004). Additionally, factors that influence fish populations, such as habitat suitability or complexity, also influence capture efficiency; this introduces systematic bias and can confound analyses of fish sample data. Fish sampling data are often used for species richness estimates or abundance estimates. By not accounting for differences in capture efficiency while sampling, systematic error is introduced into the dataset, which can bias abundance, relative abundance, or richness estimates (Bayley 1993; Peterson et al. 2004).

Sample bias can be minimized by using unbiased statistical methods for adjusting catch data, such as capture-recapture estimators.

High variation within a dataset can obfuscate population trends or estimates of the status of a population (Peterson and Rabeni 1995; Williams et al. 2002). Variance can be reduced by accounting for sources of variation through sampling design, increasing sample size, and by using the most efficient and effective methods (Williams et al. 2002). Systematic variance is generally predictable and consistent, such as variance introduced from sampling different habitat types (i.e. riffles or pools). Systematic variance can be eliminated by using certain designs, such as random stratified sampling and adaptive cluster sampling (Williams et al. 2002). Sample-to-sample variance, or variance due to differences between samples (i.e. due to habitat variables), can be accounted for using relevant covariates that explain a particular type of variation (Williams et al. 2002). Variation is also influenced by capture efficiency, with low efficiency leading to high variance. This source of variance can be reduced by using a highly efficient method (Peterson and Rabeni 1995).

As human population increases, the potential for increased negative anthropogenic influences to aquatic resources also will increase, further imperiling aquatic species. The U. S. Fish and Wildlife Service (USFWS) Section 7 consultations mandate that threatened and endangered species must be conserved; the consultations are meant to "ensure that [other agencies'] actions do not jeopardize listed species or destroy or adversely modify critical habitat" (U. S. Fish and Wildlife Service 2007). Required consultations, habitat conservation planning, pre-development planning, work with USFWS Partners, and other species' recovery efforts can only be effective if managers know the location and status of at-risk fish populations.

Because sampling protocols often differ among managing and consulting agencies, it is difficult to compare information on fish populations. Additionally, many contemporary sampling protocols do not take into account incomplete capture of fish; rare or difficult-to-capture fishes are assumed to be absent when the species could be present but undetected. There remains a need for developing a sampling protocol for at-risk stream fish that produces minimally biased, precise estimates of fish population status with a minimum of sampling effort by natural resource managers. While minimizing bias and variance is important, management agencies have finite financial resources, and collecting field data can be time-consuming and expensive. Increasing effort or sample size to reduce variance may eventually be cost prohibitive, therefore methods to collect fish sample data should be cost-effective and allocate sampling effort only where necessary. The purpose of this study is to develop a sampling protocol that incorporates sample design and models of fish detection rates to provide information on the status and distribution of at risk fishes for meeting USFWS Section 7 objectives.

Objectives

The goal of this research is to develop protocols for estimating the status of populations of at risk, stream-dwelling fishes. To do so, I will meet the following objectives:

1) Estimate and model fish capture efficiency for particular sampling conditions based on gear used, fish species collected, and site-specific habitat.
2) Estimate and partition the different sources of variance in fish abundance, distribution, and fish community metrics (e.g., species richness).
3) Develop an optimal sampling protocol that minimizes variance and sampling effort.

CHAPTER 2

LITERATURE REVIEW

Common stream sampling methods and capture efficiencies

The ideal method for sampling at-risk stream fish would be highly efficient to minimize variance and bias, require minimal effort to reduce costs, and have no adverse physiological effects on sampled fishes (Peterson and Rabeni 1995). Currently, no single sampling method meets all of these criteria, so biologists and natural resource managers generally choose among commonly accepted methods to sample streams and rivers for fishes based on the availability of equipment, the objectives, and other practical considerations. Some common methods for actively sampling wadeable streams include electrofishing and seining. Practical considerations include site-specific characteristics, such as stream size, amount of woody debris, or chemical water properties, as well as the characteristics of the target fish species, such as morphology, size, or behavior.

Electrofishing is a common fish sampling gear that uses an electrical field to attract and stun fish for retrieval (Reynolds 1996). Electrofishing can be performed using a backpack electrofisher, an electric seine, or a barge or boat electrofisher, and the type of electrofisher typically depends on the size of the waterway. Modern backpack electrofishers generally use a 12-V deep-charge battery to generate an electrical current and are strapped to the back of a sampler (Reynolds 1996). Stream size should determine how many backpack units to deploy to efficiently sample a stream; each backpack unit deployed requires a minimum of two people, as one person must operate the electrofisher and one person must net stunned fish (Georgia

Department of Natural Resources 2005; Peterson and Rabeni 2001). Generally, electrofishing requires purchasing an electrofishing unit and establishing a knowledgeable field crew to operate the unit and appropriate nets. While electrofishing is a very common sampling method, the risk of fish injury and death should be considered, particularly in at-risk populations (Reynolds 1996).

Seines are also a common gear for fish sampling. In wadeable streams, a seine is a mesh fabric stretched between two poles, termed as brailles (Hayes et al. 1996). Seines can be used independently, such as through a series of "hauls" in wadeable waters (Bayley and Dowling 1990), or to supplement electrofishing (Kennard et al. 2006). Swift water generally requires placing the seine downstream of the swift water and collecting fish as a sampler disturbs substrate upstream of the net, termed kick seining (Rabeni et al. In press). A seine is typically manned by at least one crew member on either braille, and may require an additional crew member to disturb substrate and herd fish into the seine during a haul. Using a seine without supplemental electrofishing places little physiological stress on fish.

Capture efficiency of sampling methods is affected by chemical properties of water, such as conductivity, turbidity, or temperature. In electrofishing, water conductivity affects the size and strength of an electrical field, and extremely low or high conductivity reduces capture efficiency by exceeding the capacity of the power source (Bayley and Dowling 1990; Pusey et al. 1998; Reynolds 1996). Very low water temperatures influence capture efficiency by altering fish distribution, decreasing fish floatation, and increasing fish recovery time. Alternately, high water temperatures increase fish metabolism rates (i.e., fish are more active) and water conductivity (Bayley and Dowling 1990; Holland-Bartels and Dewey 1997; Reynolds 1996). High turbidity, in combination with water depth, can affect the ability to see stunned fish and can negatively affect capture efficiency while electrofishing. Conversely, high turbidity may increase seining efficiency, as the fish are unable to see the seine and do not try to escape (Bayley and Dowling 1990; Hayes et al. 1996; Holland-Bartels and Dewey 1997).

Site-specific characteristics, such as stream substrate, structure, and size also affect the ability to sample. Habitat complexity such as woody debris, coarse substrate or undercut banks can negatively influence capture efficiency; fish may remain concealed in cover or under banks and certain habitat features may reduce the ability of a sampler to net fish or haul a seine (Bayley and Dowling 1990; Bayley and Peterson 2001; Hayes et al. 1996; Holland-Bartels and Dewey 1997; Wildman and Neumann 2003). Capture efficiencies are lower when stream width or depth exceeds the effective catch area (e.g., size of an electrical field or width of a seine; Peterson et al. 2004). In deep or wide streams, fish may avoid the electrical field or swim over, under or around the seine; conversely, fish may be startled by an approaching seine or sampler in very shallow water (Bayley and Dowling 1990).

Fish body size, morphology and behavior also influence capture efficiency. Small fish are generally not as susceptible as large fish to electrofishing, due to the amount of tissue susceptible to electroshock and the visibility of the fish to the sampler (Anderson 1995; Reynolds 1996). Small fish may slip through the mesh of a seine (>6mm mesh size; Peterson and Rabeni 2001), whereas large fish may be able to swim out of a seine prior to capture (Bayley and Dowling 1990). Position in the water column influences capture efficiency; benthic fishes are typically in contact with the streambed and may not be available or visible to a sampler when stunned during electrofishing or during a seine haul. Fishes that swim in the water column are generally more visible when stunned, but also may be able to escape a dip-netter (Peterson and Rabeni 2001; Reynolds 1996). Fish behavior also influences capture efficiency while sampling, as some fish may remain concealed under banks, vegetation, or larger substrates during sampling (i.e. exhibit "cryptic behavior") or may be nocturnal and unavailable to a sampler (Hayes et al. 1996; Holland-Bartels and Dewey 1997; Kennard et al. 2006; Peterson et al. 2004). Cryptic behavior may be an attribute of certain species or may be in response to disturbance by the sampler.

Minimizing bias

Incomplete capture of fish due to environmental variables and size, species, and shape of fish introduces bias into estimates of fish population status and can influence a manager's ability to make sound management decisions. Although it can be assumed that no fish sampling method is 100% effective, methods have been adapted to account for differing capture efficiencies. In general, managers have attempted to minimize bias through the use of statistical estimation techniques that model population size, species presence, and species richness, or through sample design, such as increasing the size of sample units (Bayley and Peterson 2001).

Estimation techniques or model-based estimators include removal methods and capture-recapture methods that can be useful for estimating species richness, population size, or presence of a species. Multiple-pass electrofishing or depletion sampling, termed the removal method, is frequently used by biologists to estimate fish abundance (Edwards et al. 2003; Rodgers et al. 1992; Simonson and Lyons 1995). There are two types of removal estimators: the constant capture probability estimator (i.e. Zippin removal estimator; Zippin 1956) and an estimator that allows heterogeneity in capture probabilities from removal pass to pass (Riley and Fausch 1992; Williams et al. 2002). Capture probability refers to the ability to capture an individual, whereas capture efficiency refers to the proportion of individuals or species captured. Both operate on the assumption that the population is closed to emigration and immigration and require substantial sampling effort. In addition, capture efficiencies need to be relatively high and fishes relatively abundant to obtain unbiased estimates (Peterson et al. 2004). At-risk fishes typically have very low capture efficiencies and are relatively rare; therefore, obtaining a valid sample size for a removal model is likely problematic. Capture-recapture estimates are also widely used for obtaining unbiased or minimally biased abundance estimates and are often used in place of removal estimates (Anderson 1995; Otis et al. 1978; Peterson and Cederholm 1984; Rosenberger and Dunham 2005). Similar to removal estimators, capture-recapture estimators typically assume the population is closed and require considerable effort. For example, at least two samples, a marking sample and recapture sample, are required to estimate population size. Capture-recapture methods often have a minimum sample-size requirement and, in fish sampling, captured fish must be allowed to rest a minimum of 24 hours before a recapture-event; labor- and time-intensive methods may be cost-prohibitive (Anderson 1995; Peterson et al. 2004). However, estimators have been developed that allow capture efficiencies to vary as a function of site and species-specific characteristics (Williams et al. 2002).

To obtain an unbiased estimate of capture efficiency, managers also can develop models for estimating capture efficiencies and apply them to sampling data, which is termed gear calibration. Gear calibration compares the number of fish collected or counted to the known number (or an unbiased estimate of the true number) of fish or species present. A common way of calibrating gear is through a capture-recapture event, which includes sampling a population of fish using a standardized method, marking the fish, returning them and allowing for acclimation, and then resampling with a standardized method (Peterson et al. 2004). Population estimators can be used to estimate capture efficiencies for both closed and open populations, and the estimated capture efficiencies can be applied to the number of collected or counted fish

(Thompson and Seber 1994; Williams et al. 2002). Fish data can then be adjusted using modeled capture efficiencies for various methods and conditions (Peterson and Rabeni 1995). Potential drawbacks to gear calibration via capture-recapture methods include introduced bias due to fish handling during capture (i.e., marked fish may have different capture probabilities than unmarked fish). Additionally, capture-recapture events can be time consuming and expensive, particularly if capture efficiency models do not yet exist for the particular sampling conditions. Knowledge of nonlinear statistical modeling and population estimation is required to calibrate sampling data using capture efficiency models (Bayley 1993).

In contrast to estimation methods, managers have attempted to minimize bias through sample design, and these have been defined as maximum effort approaches (Bayley and Peterson 2001). In stream sampling, this is typically a single upstream pass with a sampling reach-length being a multiple of mean wetted width (typically 35 to 40 times stream width; Lyons 1992; Simonson and Lyons 1995). Single-pass upstream sampling assumes that the cumulative catch of fish species will increase with stream length, eventually reaching a maximum or asymptote, thus providing a reasonable estimate of species richness (Lyons 1992). Maximum effort designs attempt to account for differences in species-specific detection probabilities (or the probability of capturing a certain species) by increasing the amount of sample effort, hence increasing species collected (Jones and Stockwell 1995; Lyons 1992). A single-pass approach is sometimes preferred to a more time- and labor-intensive mark-recapture or removal method approach (Simonson and Lyons 1995). Species accumulation curves do not account for differences in capture efficiency, therefore sampling effectiveness may be misrepresented (i.e., overestimated; Cam et al. 2002). Some species accumulation curves may never reach an asymptote or maximum as new species accumulate due to increased habitat heterogeneity. Additionally, as mean stream width increases, not only does the recommended length of stream increase, the ability to capture fish decreases, which introduces further bias into analyses (Meador et al. 2003). Due to the stringent budget that many agencies operate under, some published protocols require that sampling cease after 500 meters regardless of mean wetted width, even though sampling shorter distances may yield poor estimates (Angermeier and Smogor 1995; Georgia Department of Natural Resources 2005).

Minimizing sampling variance

Managers need tools to minimize temporal and spatial sampling variance to provide meaningful, precise estimates of species richness or abundance in fish populations. Using appropriate sampling design and collecting adequate sample sizes are the best ways to minimize variance (Peterson and Rabeni 1995). Temporal variance is the variability in estimates (e.g., abundance, species richness) through time and often indicates large-scale changes in fish distribution or population size (Matthews 1990). Spatial variance is the variability in estimates from location to location, and it typically exceeds temporal variance for most fish species and habitat types (Matthews 1990; Peterson and Rabeni 1995). Managers should identify the source of greatest variation (i.e. temporal or spatial) and modify the sampling strategy to account for such variation. For example, if temporal variance exceeds spatial variance, a manager would try to maximize sampling over time rather than space (Peterson and Rabeni 1995). Physically dissimilar habitat types (i.e. riffles, pools, runs) may control the structure of the fish community; therefore adopting a stratified design can reduce this source of variance in analyses (Peterson and Rabeni 1995; Williams et al. 2002).

Managers should also consider variance due to low sampling efficiency or error in capture efficiency estimates. Low capture efficiency increases sample variance (Peterson and

Rabeni 1995). When sampling data are adjusted using estimated capture efficiencies, additional model error may be introduced into analyses. Increased variance can decrease a manager's ability to understand population trends or estimates; variance from sampling efficiency can be minimized by using the most efficient method for the circumstances (Peterson and Rabeni 1995). Although using modeled capture efficiency may increase variance (i.e., due to error in model predictions), using capture efficiency models to adjust data can minimize sampling bias and effort compared to designs that either (1) ignore differences in capture efficiency or (2) that use capture-recapture techniques at all sample sites.

CHAPTER 3

METHODS

Study Area

The Upper Coosa River Basin is located in northwest Georgia and southeast Tennessee. The Coosa River is formed by the junction of the Etowah and Oostanaula Rivers, and the Oostanaula River is formed by the junction of the Conasauga and Coosawattee Rivers (Figure 1). The Upper Coosa River system drains an area of approximately 14,500 km^2, including 3 major physiographic provinces (Blue Ridge, Piedmont, Ridge and Valley). The Coosa River system historically supported more than 147 species of fish and had the largest diversity worldwide of freshwater snails and mussels. However, aquatic biodiversity has declined significantly since the mid-1900s due to riverine habitat loss, modification, and fragmentation by impoundments. Watersheds in the Upper Coosa face impending development from Atlanta, GA, and Chattanooga, TN, due to large numbers of subdivisions, highways, and metropolitan areas being constructed. Changes to stream habitats, water quality, and hydrologic regimes associated with increased anthropogenic development are the dominant problems threatening fishes and other aquatic biota in the Upper Coosa Basin.

Study Site Selection

To evaluate and model the efficiency of fish sampling methods under common conditions encountered in wadeable streams in the Upper Coosa Basin, sampling sites must represent the sampling conditions and fish communities found throughout the basin. Sites were selected using the random stratified sampling method, similar to that which is outlined in Peterson and Banish

(2002). Strata were developed for each province (Blue Ridge, Piedmont, Ridge and Valley) within the 4 respective sub-basins (Etowah, Oostanaula, Coosawattee, and Conasauga rivers), with each province serving as a sample stratum (9 total strata; Figure 1). By using sub-basin and geographic province as strata, sampling sites varied spatially and included habitat differences due to province (such as gradient and water conductivity).

The Georgia Museum of Natural History maintains collection records of many years of research in Georgia and elsewhere and has established an extensive database of site specific habitat and organism data (Freeman and Freeman, unpublished). Using the existing database (comprising over 1500 Upper Coosa sites sampled since 1996), sites within the Upper Coosa Basin were sorted into each designated stratum and randomly assigned a priority number. Sites were then ranked via priority number and assessed for wadeability and accessibility. Wadeability was assessed using Strahler stream order (Strahler 1952), and sites that were greater than 3rd order were assumed to be too large to wade (M.C. Freeman, University of Georgia, personal communication). Accessibility was evaluated by examining proximity of each potential site to a vehicular access point, and sites that were not wadeable or accessible by road were rejected from the potential site list.

Because at-risk fishes are a focal point of this project (Table 1), taxonomic representatives of at-risk fish in the Upper Coosa Basin needed to be present at each site to properly evaluate capture efficiencies of those particular species groups. Historical collection data were used to determine the presence of taxonomic representatives from genera *Etheostoma* or *Percina* and from *Cyprinella* or *Notropis* or *Luxilus* (genera with similar characteristics as *Cyprinella*) at each potential site. Additionally, due to the fragile nature of some endangered fish populations, any area known to contain substantial populations of federally listed fish species were avoided (M. C. Freeman, University of Georgia, personal communication). Sites that met the above criteria were considered for sampling and crews attempted to sample no less than 3 sites per stratum, with at least 30 sites sampled overall.

Site design

At each site, crews established 3 sampling units (henceforth termed unit 1, 2, or 3) ranging in length from 50-100 meters (Figure 2). Each unit consisted of a representative mixture of habitats available at the study site, such as a complete riffle-run-pool sequence, because both fish community structure and capture efficiency vary by habitat. Hydraulic controls were selected for upper and lower boundaries of all sampling units. Block nets (mesh size ≤ 7 mm) were used to block off units 1 and 2 to prevent fish movement while sampling. Unit 3 was 50 to 100 m upstream of unit 2 and was not blocked off. However, the upstream boundary used hydraulic controls to prevent movement; hydraulic controls are areas where water velocity naturally slows due to abrupt changes in channel gradient (Peterson et al. 2004) or a physical obstruction. In sampling sites with high leaf litter (e.g., sites sampled from October – November), a second set of block nets (henceforth termed double block nets) were installed approximately 2 meters upstream and downstream of the outermost block nets. Double block nets provided an additional barrier to fish movement and would also allow for estimates of escape rate. Block nets were visually and tactilely (i.e. feeling the bottom of the net) inspected before, during, and after sampling to minimize fish escape.

Sampling methods

Primary fish sampling gear (backpack electrofishing or seining with electrofishing) was randomly selected without replacement for unit 1; unit 2 used the alternate fish sampling gear. For backpack electrofishing, 3 passes were conducted with Smith-Root LR 24 pulsed DC backpack electrofishers. Voltage and duty cycle of each backpack electrofisher were adjusted according to the conductivity of the stream and amperage output was approximately 0.10 continuous amperes and 1.0 peak amperes. In narrow streams (≤ 4 meters mean wetted width), 1 backpack electrofisher was employed by 1 crewmember and 2 crewmembers retrieved stunned fish with dipnets. Streams wider than 4 meters mean wetted width required 2 backpack electrofishers and 1 crewmember per electrofisher to retrieve stunned fish. The first pass was completed in an upstream direction, followed by a second downstream pass, which was followed by a third upstream pass. Generally, each deployed backpack electrofisher and crewmember(s) zigzagged upstream or downstream and thoroughly sampled all available habitats, such as woody debris, undercut banks, pools, or riffles (Rabeni et al. In press). Fishes were stored separately by electrofisher pass in aerated buckets. Following all 3 passes, crewmembers collected fish that drifted into the downstream block net and stored them separately.

For seining with electrofishing, crews deployed 1 backpack electrofisher and a minnow seine (width = 2.5 m, height = 1.5 m, mesh size = 5 mm). Crews sampled all available mesohabitats, such as pools or riffles, in the unit and moved through the unit in an upstream direction. In slow flowing areas, such as glides or pools, downstream seining in the main channel was used. Fast flowing water required kick-seining (placing the seine at the downstream end of swift current and disturbing the substrate upstream of the seine); generally, crewmembers would disturb substrate no further than 3 meters upstream of the seine.

A backpack electrofisher was used in conjunction with the seine in the following manner:

a) During downstream seining, two crewmembers moved the seine slowly downstream while 1-2 additional crewmembers flanked the sides of the seine (to prevent fish escape around the seine). As the seine moved toward an area of shallow water or the bank, a crewmember facing the seine deployed the electrofisher to stun fish near the edge of the seine.

b) During kick-seining, the seine was placed at the downstream end of swift current, 1 to 2 crewmembers (with 1 carrying a backpack electrofisher) disturbed the substrate and applied electricity while moving downstream toward the seine.

c) In areas with woody debris or undercut banks, the seine was placed around the area to be sampled (to block fish escape while electrofishing). A crewmember applied electricity with the backpack electrofisher while a second crewmember dip-netted stunned fish. Often, the water or substrate surrounding the sampling area (inside the blocked seine) was disturbed to wash stunned fish into the seine.

At the end of each seine haul or set, the seine was lifted and fish were collected and placed in aerated buckets. One upstream sampling pass was completed with the seining with electrofishing method.

Following fish sampling in each unit, crews identified each fish to species or genus for *Gambusia* or age 0 *Moxostoma* species, measured total length (TL) to the nearest millimeter, and marked each fish with a unit-specific fin clip (upper or lower caudal fin). Fish were redistributed throughout the unit they were collected in and allowed to recover for 14 to 22 hours. Prior to redistribution, fish were examined to determine if they had recovered (i.e. swimming and respiring normally), and only fish that fully recovered were placed back into the unit.

Crewmembers returned to the site after a recovery period of 14 to 20 hours. Crewmembers checked the sampling area and lower block nets for mortalities. Any fish collected in the bottom block nets were identified to species, examined for fin clips, measured to the nearest millimeter, and removed from the sampling area. Each unit then was resampled using the alternative method. For example, if unit 1 was sampled via backpack electrofishing during the marking period, it was resampled via seining with electrofishing. Sampling and data collection procedures during the second sampling period followed the same protocol as the initial capture period. All fish captured were identified to species, measured to nearest millimeter, and recorded as previously caught (if marked with a fin clip) or unmarked (if no fin clip was present). If double block nets were installed at the site, crewmembers sampled the area between the outermost block nets using a backpack electrofisher and recorded the species and TL of fin-clipped individuals. Additionally, captures of marked fishes that had escaped from the adjacent unit (i.e., a fish marked in unit 1 and recaptured in unit 2) were recorded as such.

Prior to the removal of block nets, unit 3 was sampled using a backpack electrofisher with single upstream pass, and the number of electrofishers deployed duplicated that used in units 1 and 2. At the end of the single upstream pass, all fish were identified to species, measured to the nearest millimeter and released into the stream or preserved for later identification. Unit 3 data were collected to serve as a non-block netted comparison to the first upstream pass collected in units 1 and 2. All preserved fish were fixed in a 10% formalin solution for up to one month then transferred to a 70% ethanol solution. Following sampling, all block nets were removed.

Physicochemical data collection

Chemical water quality information and physical habitat characteristics were measured to provide site-specific parameters for modeling capture efficiencies of fish. Water quality data were recorded using calibrated meters; water temperature (ºC), specific conductivity (µs), and turbidity (NTU) were measured and recorded each day prior to sampling. A threshold value to distinguish high turbidity from low turbidity was established at 30 NTU and each site was binary coded as high turbidity or low turbidity. At most sites, 30 NTU represented a threshold where visibility was significantly reduced, which presumably reduced a sampler's ability to see a stunned fish in the water column or on the stream bottom. Following block net removal, a line-transect method was used to measure physical habitat characteristics within each unit by establishing transects perpendicular to the flow (Figure 3). Transects began at the previous location of the lowermost block net and continued at 10-m intervals until crewmembers reached the previous location of the uppermost block net. At each transect, crewmembers measured wetted width and depth of undercut bank (if applicable). Current velocity (m/s) and water depth (m) measurements were taken at 8 evenly-spaced points along each transect. Velocity was measured at 0.6 depth with a Marsh McBirney model 2000 water current meter attached to a standard, top-set wading rod. Mean cross-sectional area for each unit was estimated by averaging the mean wetted width and multiplying it by the mean depth at each transect. Banks that were undercut at least 10 cm were counted as an undercut bank (i.e., an area that may provide a hiding place for fish). Undercut banks were summed and divided by the total number of bank measurements to estimate the proportion of undercut banks at a site. Mean velocity was estimated by averaging velocity measurements for all transects in each unit. The area of each sample unit was estimated by multiplying the length of the sampling unit by the mean wetted width.

Substrate composition was visually estimated in a 1-meter-wide band centered across each transect and was categorized as percentages of clay (<0.03mm), silt (0.03-0.5 mm), sand (0.5-2 mm), gravel (2-20 mm), cobble (20-200 mm), boulder ($\geq$ 200 mm), and bedrock (no particles), similar to Peterson et al. (2005). Substrate percentages were averaged for each unit. Between each transect, crews counted the pieces of woody debris, which were defined as woody objects $\geq$ 1 meter long and 10 cm in diameter or as an aggregation of smaller pieces lying within the wetted channel. I estimated wood by area, or wood density, by dividing the number of total wood pieces in a unit by the total sample area (length of unit times mean wetted width).

Capture efficiency modeling

The first objective was to estimate and model fish capture efficiency using different gear types under site-specific conditions. However, traditional two-sample capture-recapture estimators, such as the Lincoln-Petersen estimator, require the assumption that capture and recapture probabilities are equal (Williams et al. 2002). Thus, we created a two sample capture-recapture model using the capture histories of fishes collected during sampling as:

$$p(1,0) = p_1*(1-p_2),$$

$$p(0,1) = (1-p_1)*p_2,$$

$$p(1,1) = p_1 * p_2,$$

$$p(\geq 1) = 1- (1-p1)*(1-p_2)$$

where $p(1,0)$ is the probability that an individual is captured on the first occasion, but not the second; $p(0,1)$ is the probability that an individual is missed on the first occasion, but captured on the second; $p(1,1)$ is the probability that an individual is captured on both occasions; $p(\geq 1)$ is the probability that the individual was captured on at least 1 occasion; p_1 is the capture probability for the primary sampling method and p_2 is the capture probability for the secondary sampling method. The number of fish collected with each capture history was modeled assuming a binomial distribution with site (i) and species size group-specific(j) abundance (N_{ij}). To accommodate the complex model structure, I used Markov Chain Monte Carlo (MCMC) as implemented in BUGS software, version 1.4 (Spiegelhalter et al. 2006) to fit models of the primary (p_1) and secondary (p_2) method capture probabilities as a function of method, site specific characteristics, and species characteristics, described below. All models were fit using 100,000 iterations with 25,000 burn in (i.e., the first 25,000 MCMC iterations were dropped) and diffuse, or non-informative, priors. I was primarily interested in evaluating capture efficiencies for a single upstream pass with either a backpack electrofisher or via seining with electrofishing. Hence, only fish collected in the first pass were included in the analysis. To format the data for analysis, each individual recaptured fish was matched with a marked fish of the same species and fish length (to the nearest millimeter) from the first upstream pass. Fish without a representative match from the first pass (i.e., a fish collected in the second or third pass) were considered unmarked.

The ability to capture fishes is reportedly related to fish size, shape, and behavior (Peterson and Rabeni 2001; Peterson et al. 2004; Reynolds 1996). To evaluate the effect of fish size on capture efficiency, I divided each taxon into 30 millimeter size class groups, and mean length for each taxon-size group at a site was used as an explanatory variable in further models. I chose 30-millimeter size classes as a compromise between providing an adequate sample size for each size class and incorporating variability in capture efficiency due to fish size. To evaluate the effect of species (i.e., shape and behavior) on capture efficiency, I separated species into family or genera groups; each fish species was binary coded as 1 or 0 for each species group. Differences among groups were then examined by treating each group as a covariate (following Peterson and Rabeni 2001). I used the minnow group as the baseline, which means that parameter estimates associated with a species groups should be interpreted relative to the minnow group. The minnow group was chosen as the baseline because it contained the greatest number of marked and captured individuals. The number of backpack shockers deployed at a site varied with stream width, so I included a binary coded covariate to indicate whether 1 or 2 backpack shockers were used in electrofishing sampling. Additionally, the secondary sampling gear was binary coded with a 1 (otherwise 0) to evaluate the relative support for a marking effect (i.e., recaptured fish may exhibit altered behavior due to previous capture; Manly 1971; Mesa and Schreck 1989). Previous studies suggest that stream characteristics, such as depth, substrate, woody debris, and specific conductance, influence fish capture efficiencies (Bayley and Dowling 1990; Bayley and Peterson 2001; Hayes et al. 1996; Holland-Bartels and Dewey 1997; Wildman and Neumann 2003). Therefore, habitat parameters also were included as potential explanatory variables in capture efficiency models. Prior to fitting models, I first evaluated the relationship among predictor variables by examining pairwise Pearson correlations for all pairs of variables. Strongly correlated variables ($r^2 > 0.49$) were not included in the same models to avoid multicollinearity (Moore and McCabe 1993).

My primary goal was to obtain the most accurate model for predicting fish capture probabilities rather than to identify the factors influencing capture efficiency. However, fitting models with all combinations of predictor variables was not feasible using computationally intensive MCMC methods. Therefore, I used a two-step procedure similar to Howell et al. (2008), where the relative support of predictor variables was initially evaluated using traditional maximum likelihood methods, and the most plausible variables then were included in models fit using MCMC methods. For each gear-type, I identified parameters that most strongly related to fish capture efficiency by fitting logistic regression models using only the number of marked fish recaptured by the secondary gear. All possible combinations of predictor variables and two-way interactions (i.e., all subsets) were included and models were fit using the GENMOD procedure in SAS (SAS Institute, Cary, NC). A preliminary analysis of the models indicated overdispersion, which occurs when the variance exceeds the presumed binomial distribution. To account for overdispersion, I used quasi-likelihood logistic regression, which is similar to logistic regression with an additional element, the extra-binomial variance (Agresti 1990). To assess the relative importance of each predictor, I calculated quasi-likelihood Akaike's Information Criterion (AIC; Akaike 1973) and Akaike parameter importance weights (following Burnham and Anderson 2002). Akaike importance weights range from 0-1 and represent the weight of evidence for a given predictor variable with larger weights indicating greater support (Burnham and Anderson 2002). Parameters with Akaike importance weights ≥ 0.05, henceforth termed the most-plausible predictors, were included in the hierarchical capture efficiency models.

During the second step of the modeling procedure, I fit the capture-recapture models described above using the most-plausible predictors. The primary (p_1) and secondary (p_2) method capture probabilities were modeled as a linear function of these predictors using a logit link. However, there was a possibility that the fishes collected at unit or a site were dependent (i.e., autocorrelated). Additionally, I expected the relationship between capture efficiency and study site characteristics to vary among species. To account for dependence among fishes at a site and allow the model parameters to vary among species, I fit hierarchical logistic regression models of fish capture efficiency. Hierarchical models can account for within site dependence by including random effects corresponding to each sample site (or unit). These random effects represented differences in site-to-site (or unit-to-unit) capture efficiencies that could not be explained by the covariates. Similarly, the variability in species relationships was incorporated through random effects that allowed model intercepts and slopes to vary among species (Snijders and Bosker 1999). Here, the random effects represented differences in species-specific capture efficiencies that could not be accounted for by the species group covariates. Random effects were assumed to be normally distributed with mean of zero and random effect specific variance (Snijders and Bosker 1999) and a covariance when two or more random effects were included.

To determine the best approximating random effects error structure, I initially fit a global model, or the model containing all of the most-plausible predictors, and evaluated the relative support of models with site, unit, and species random effects with and without a covariance using deviance information criterion (DIC; Spiegelhalter et al. 2002). DIC is a Bayesian measure of model fit or adequacy that is calculated using the expected model fit and the number of effective parameters, p_D. Models are penalized for complexity using p_D, therefore a smaller DIC indicates a better approximating model (Spiegelhalter et al. 2002). Using the best approximating random effects error structure, candidate capture efficiency models were fit using all subsets of the most-plausible parameters. Candidate models were ranked using DIC, and parameter estimates, standard deviations (SD), and 90% Bayesian credibility intervals (CI; Congdon 2001), which are analogous to 90% confidence intervals, were generated for each model. The best-approximating capture efficiency model was determined using DIC, and only the best-approximating model was used for inference. I also examined the precision of parameter estimates of the best-fitting model, and I considered parameter estimates to be precise if the upper and lower credibility intervals did not contain 0. I calculated scaled odds ratios for each predictor in the best-approximating model to ease interpretation of parameter estimates (Hosmer and Lemeshow 1989) using scalars that I believed represented biologically-relevant changes.

To estimate the predictive ability of the best-approximating model, I estimated out-of-sample prediction performance using 10-fold cross validation (Fukunaga and Kessell 1971). During this procedure, I randomly partitioned the data into tenths and used 90% of the data to fit the models and estimate capture efficiency. I then used the fitted models to predict the remaining 10% of the data and calculated prediction error. Prediction accuracy was estimated by the square root of the mean of the squared difference between known and predicted capture efficiencies, termed the root mean square error (RMSE). Known efficiency could only be calculated for secondary sampling method and was the ratio of recaptured individuals divided by marked individuals.

Evaluation of fish movement

Block nets can be labor-intensive and difficult to maintain when sampling; therefore it is important to understand the utility of block nets in a sampling protocol. If a substantial number of fish leave non-block netted sample units, abundance estimates could be biased low. I hypothesized that abundance estimates would be consistently lower in non-block netted units compared to block-netted units, which I interpreted as an indication that fish moved out of the sampling area. This analysis, however, required the assumption that assignment of block netted or non-block netted status was random (i.e., there was no consistent difference between block netted units versus non-block netted units) and that all units at a site were reasonably similar. I believe these assumptions were met because non-block netted units were established a reasonable distance away from block netted units (~50-100 meters upstream or downstream of the nearest block netted unit) and block netted and non-block netted units at a site had comparable habitat. I hypothesized that: (*i*) movement out of non-block netted sample units was lower for benthic fishes (e.g., darters, catfish, and sculpin); (*ii*) movement was greater for more mobile fishes (e.g., suckers, minnows and topminnows); (*iii*) movement was lower in areas with greater amounts of woody debris (i.e., habitat cover); (*iv*) movement was lower in areas with higher velocity; and/or (*v*) movement was greater in deeper water (i.e., more room for fish to escape).

For each sample unit, I estimated species-specific capture efficiencies using the best-approximating capture efficiency model for 1-pass backpack electrofishing. I averaged the total fish length for each species at each site and used the mean fish length in the capture efficiency model, rather than estimating capture efficiency for all fish-length groups. Using the estimated capture efficiencies, I estimated abundance ($\hat{N}$) for species collected in each sample unit as:

$$\hat{N} = C/\hat{p}$$

where C is the actual number of fish of a given species collected and $\hat{p}$ is the estimated capture efficiency (Williams et al. 2002).

I was primarily interested in estimating the difference in fish abundance between block netted and non-block netted units, which I interpreted as the number of fish moving out of non-block netted sampling units, and determining the factors that influence fish movement. However, stream fish abundance and assemblage structure are known to be related to stream habitat characteristics (Angermeier and Smogor 1995; Bayley and Dowling 1993; Gorman and Karr 1978; Peterson et al. 2004). All candidate fish abundance models included wood density, current velocity, sample unit area, and depth as "nuisance" variables to account for the effects of differences in habitat structure among sample units (similar to Rieman et al. 2006). I also created predictor variables to evaluate the relative support for my hypotheses regarding fish movement. I created a block net variable that was coded as 1 for block netted units and 0 otherwise. Similarly, I created a benthic fishes variable that was binary coded as 1 for benthic species (darters, catfish, and sculpin; Table 4) and 0 otherwise, and mobile fishes that was binary coded with a 1 for mobile species (suckers, minnows, and topminnows; Table 4) and 0 otherwise. Prior to analyses, I ran Pearson correlations on all pairs of predictor variables and excluded variables with $r^2 > 0.49$ to avoid multicollinearity (Moore and McCabe 1993).

I was concerned that samples collected from individual sample units at a site would be autocorrelated, which would preclude the use of traditional regression techniques (Sokal and Rohlf 1995). I also believed that the relationship between fish abundance and habitat characteristics was likely to vary among species. Thus, I initially fit a linear regression model of estimated abundance using all predictors (the global model; Table 8). An analysis of variance from the residuals from the global model indicated significant dependence among species ($F = 7.2$; $df = 50, 970$; $P < 0.0001$) and streams ($F = 3.5$; $df = 26, 970$; $P < 0.0001$), determined using the GLM procedure in SAS (SAS Institute; Cary, NC). To account for the spatial autocorrelation and the varying response of fishes, I examined relationships between habitat variables, block netted sites, species traits, and fish abundance using hierarchical linear models. Random effects represented variability in the relationship between abundance and the presence of a block net and habitat characteristics (wood density, velocity, depth, and block net) among sites or species. Random effects were assumed to be normally distributed with mean of zero and random effect-specific variance (Sniders and Bosker 1999). All models were fit using the MIXED procedure in SAS (Littell et al. 1996).

I used the information-theoretic approach (Burnham and Anderson 2002) to evaluate the relative plausibility of models relating the difference in abundance between non-block netted units to habitat and species-specific characteristics (Table 8). I constructed a global model with all predictors, a null model with only nuisance variables, and 18 additional candidate models representing the hypotheses regarding fish movement relative to behavioral traits, depth, velocity, or wood density. To assess the relative fit of each candidate model, I calculated AIC_c and Akaike weights (Burnham and Anderson 2002). The number of parameters used to estimate AIC_c included the fixed effects and random effects, when included in the error structure

(Burnham and Anderson 2002). Akaike weights can be used to compare one model to another to assess the degree of evidence for that particular model. The confidence set of models included candidate models with Akaike weights that were within 10% of the largest weight (similar to 1/8 rule suggested by Royall 1997). I reported parameter estimates, standard errors, and 95% confidence limits for models in the confidence set. I calculated relative importance weights of individual predictor variables by summing the Akaike weights for candidate models in which each predictor occurred (Burnham and Anderson 2002). Goodness-of-fit was assessed for each candidate model by examining normal probability plots of the lower level residuals.

Prior to evaluating the fit of candidate models, I evaluated the relative fit of several different variance structures for the hierarchical model random effects using the global model. The first set modeled the random effects due to site-specific habitat characteristics (wood density, velocity, depth, block net) as varying among species; the second, among sites; the third, varying among species with an additional random effect for site; and the fourth, varying among sites with an additional random effect for species. To assess the relative fit of each error structure, I calculated AIC_c values for each error structure model (Akaike 1973). The best approximating variance structure then was used during the evaluation of the relative plausibility of the candidate models of estimated abundance.

The capture efficiencies estimated using the capture-recapture data are defined as conditional capture efficiencies. That is, they are conditional on the fish existing in the (block netted) unit. Because some fish may leave the unit if block nets are not in place, I was interested in estimating the number of fish escaping the non-block netted units. For each site, I estimated species-specific abundance for identical (with respect to habitat characteristics) block netted and non-block netted units using the best-approximating model. I used these to estimate the proportion of fish remaining (not moving) in non-block netted units as the non-block netted abundance divided by the block netted abundance. For example, for predicted abundances of 20 and 25 fish in identical non-block netted and block netted units, respectively, the approximate proportion of fish staying in a non-block netted unit at that particular site would be 20/25, or 80%. I then estimated mean proportion of fish remaining for each species group and 95% confidence intervals. I summarized the results by species group rather than by individual species to coincide with the best-approximating capture efficiency model (see Results).

Using the proportion of fish remaining in the (non-block netted) unit, I estimated unconditional capture efficiency. Unconditional capture efficiency incorporates capture efficiency with the proportion of fish remaining in a sample unit during sampling. Using the mean proportion of fish remaining in non-block netted units, I estimated unconditional capture efficiencies ($\hat{p}^{\circ}$) as:

$$\hat{p}^{\circ} = \hat{p}_i * (1 - \gamma_i),$$

where $(1 - \gamma_i)$ is the proportion of fish remaining in unit (i) and $\hat{p}$ is the estimated capture efficiency for unit (i) (following notation by Bailey et al. 2004) . These unconditional capture efficiencies are used to adjust fish sample data collected in non-block netted units to minimize biases associated with incomplete capture *and* fish movement.

The variance of the fish abundance estimates is influenced by variance in the true abundance from sample to sample, capture efficiency, and error in the capture efficiency estimate (Thompson and Seber 1994). Capture efficiency and precision of the capture efficiency estimate likely varied among sampling methods and with the use of block nets, which suggests that under identical sampling situations, the use of a certain fish sampling method and/or the use of block nets can affect sample variance. To evaluate the expected differences in variance among methods, I calculated the variance of $\hat{N}$ using the approximate formula for variance of $\hat{N}$ adapted from Thompson and Seber (1994) as:

$$\operatorname{var}(\hat{N}) \approx \operatorname{var}(\hat{N}) + \frac{\hat{N}^2}{\hat{p}^2} * \operatorname{var}(\hat{p}),$$

following the notation as described above. For non-block netted units, $\hat{p}^{\circ}$ was substituted for $\hat{p}$, and the variability in the estimate of the proportion of fish remaining in the sampling unit was added to the estimate of variance of $\hat{p}$. I then calculated the coefficient of variation for each species group for 1-pass backpack electrofishing samples collected within block netted units and non-block netted units and for seining with backpack electrofishing within block netted units.

CHAPTER 4

RESULTS

Thirty-one sites were sampled from November 2006 to November 2007, with capture-recapture samples being collected at a total of 61 units; the majority of samples were collected between June and September 2007 (Figure 1; Table 2). At 26 sites, fish were sampled in all three units, but due to logistical constraints (e.g., equipment failure or severe weather), only capture-recapture samples were collected at four sites, and one site consisted of a single capture-recapture sample and two non-block netted sample units. Samples were collected during base flow conditions, and on average, sampling sites were in small (5.7-m-wide), warm (20.3 °C), wadeable (0.17-m-deep) streams and covered a relatively wide range of habitat characteristics (Table 3). Fifty fish species were used in the analysis and these fishes were divided into eight species groups for model fitting (Table 4). On average, 159 fish were marked in each unit, with a total of 9,691 total individuals marked. Trout and lamprey species were not included in the analysis because insufficient numbers of trout were collected and lamprey species were not targeted during sampling events.

Fish escape

Using the double block netted sites (n=2; Table 2), I was able to evaluate fish escape under normal circumstances. I assumed that samplers collected all fish within the area between the double block nets because the sampling area was small and the sampling effort was intensive. Because only two sites were double block netted and relatively few marked fish escaped the block netted units, I evaluated escape rate by simply dividing the number of marked fish collected in the double block netted units by the total number of fish marked per unit. The total proportion of fish escaping was 8/1021 (proportion of escape by unit was 1/234, 5/516, 2/136, and 0/135), with an overall average escape rate of 0.72%.

Capture efficiency modeling

I had difficulty maintaining closure in 12 sample units. At three sites, animals chewed one or more holes in the block nets (compromising eight units). A block net also failed at two additional sites due to soft substrate or water level change from a rain event (compromising four units). I was concerned about the violation of the closure assumption at sites with holes or net failures. Eighteen of 2,116 (0.85%) marked fish had escaped (e.g., fish marked in unit 1 and recaptured in unit 2) and were collected in the 12 units with net failures, whereas 30 of 7,575 (0.39%) marked fish had escaped and were collected in the 49 units without net failures. Marked fish that escaped and were collected were not included in the capture efficiency models, since those individuals were collected in the wrong sample unit during recapture sampling. To evaluate the bias of closure violations on capture efficiency modeling, I developed a binary coded predictor variable termed 'net failure' to indicate an instance where a hole appeared in a block net overnight or where a block net shifted and included this variable in capture efficiency models. There was no support for the net failure variable in the 1-pass backpack electrofishing model during the first step of the capture efficiency modeling procedure (Akaike importance weight < 0.05), therefore net failure variable was not included as one of the most plausible predictors in the hierarchical models. There was minimal support for a net failure effect (Akaike importance weight > 0.05) in the seining with backpack electrofishing efficiency models, so it was included in the hierarchical models. However, there was no support for models containing the net failure variable as it was not included in the best approximating model. In addition, the parameter estimate for the net failure variable in the best-approximating hierarchical model containing the net failure variable was positive (estimate = 0.15), indicating that net failure and fish movement *increased* capture efficiency due to escape, and imprecise (SD = 0.13; lower CI= -0.09; upper CI= 0.40).

The best approximating error structure in the hierarchical capture efficiency models included a unit-level random effect and no species random effects; hence all candidate capture efficiency models included a unit level random effect. The best-approximating model for 1-pass backpack electrofishing included 25 predictors: 6 species group effects, fish length, mean cross-sectional area, a marking effect, and 14 interactions between species group, fish length, and site-specific habitat effects (Table 5). The greatest differences in capture efficiencies were among species groups. Under average sampling conditions and average fish length by species group, capture efficiency was greatest for suckers (23%) and lowest for catfish (2%); the average capture efficiencies for the remaining species groups were 20% (bass), 8% (darters), 21% (minnows), 12% (sculpin), 19% (sunfish), and 7% (topminnows). Capture efficiency generally increased with mean fish length under average habitat conditions (Figure 4). However, capture efficiency for sunfish species group was unrelated to fish length. With the exception of the bass species group, all parameter estimates for species group effects and fish length effects were relatively precise (Table 5).

Habitat predictors in the best approximating 1-pass backpack electrofishing model consisted of turbidity, velocity, cross-sectional area, cobble, and wood density. On average, fish were estimated to be less catchable (i.e., had lower capture efficiencies) under high turbidity conditions for most species groups, although the parameter estimate was relatively imprecise (Table 5; Figure 6). In contrast to most species groups, the capture efficiency was positively related to high turbidity for the sculpin group (Figure 6). As velocity increased, the catchability of sunfish decreased, while the catchability of darters increased or remained relatively static with increasing velocity (Figure 8). Using unit scalars, for each 0.1 m/s increase in velocity, darters were 1.15 times more likely to be captured and sunfish were 2.11 times less likely to be captured (Table 5). Capture efficiency also was negatively related to cross-sectional area (Figure 9). I estimate that for each 1 m^2 increase in cross-sectional area, fish, on average, were 2.31 times less likely to be captured. Capture efficiency also was strongly related to wood density for bass, suckers and sunfish. Bass catchability increased with wood density, whereas catchability for suckers and sunfish decreased with wood density (Figure 10a). It should be noted, however, that the parameter estimate for wood density and bass and sucker species groups were relatively imprecise (Table 5). A marking effect was incorporated for each sampling method to account for differences in capture efficiency due to handling during the marking event. The marking effect for 1-pass backpack electrofishing was negatively related to capture efficiency, although the parameter estimate was relatively small and very imprecise. I estimate that marked fishes were as were much as 1.66 times less likely to be collected to as much as 1.15 times more likely to be collected than unmarked fishes.

Cross-validation of the 1-pass backpack electrofishing model indicated that it was relatively unbiased for most species groups (Table 6). The greatest mean differences between predicted and known capture efficiencies were 1.2% and 2.5% for minnows and sculpin, respectively. Cross-validated root mean square error indicated predicted efficiency for the 1-pass electrofishing model was most precise for bass (8.4%), catfish (0%), sunfish (14.6%) and topminnows (8%), and least precise for darters (16.9%), minnows (19.6%), sculpin (20.6%), and suckers (18.1%).

The best-approximating model for seining with backpack electrofishing included 24 predictors: 6 species group effects, mean cross-sectional area, mean velocity, percent cobble, wood density, a marking effect, and 12 interactions between species groups, fish length, and site-specific habitat effects (Table 6). The greatest differences in capture efficiencies were among species groups. Under average sampling conditions and average fish length by species group, capture efficiency was greatest for suckers (27%) and least for catfish (1%); the average capture efficiencies for the remaining species groups were 14% (bass), 16% (darters), 18% (minnows), 14% (sculpin), 19% (sunfish), and 12% (topminnows). Mean fish length was a relatively weak predictor of capture efficiency for this sampling method, although capture efficiency increased with fish length for bass and suckers and remained relatively static for topminnows (Figure 5). Parameter estimates for species effects for darters, suckers, sunfish, as well as the sucker fish length effect, were imprecise for this sampling method (Table 6).

Habitat effects used in the seining with electrofishing models consisted of velocity, cross-sectional area, percent cobble, and wood density. As velocity increased, the catchability of fish increased (Figure 8). Using unit scalars, for each 0.1 m/s increase in velocity, darters were 1.55 times more likely to be captured and all other fishes were 1.36 times more likely to be captured (Table 6). Capture efficiency was negatively related to cross-sectional area (Figure 9). I estimate that for each 1 m^2 increase in cross-sectional area, fish were 1.23 times less likely to be captured (Table 6). In contrast to most species groups, the capture efficiency was negatively related to cobble substrate percentage for the darter group (Figure 7). Capture efficiency was positively related to wood density for the bass, minnow, sculpin, and sucker species groups, but was negatively related to wood density for the darter, sunfish and topminnow species groups (Figure 10b). It should be noted, however, that the parameter estimate for wood density and the bass species group was relatively imprecise (Table 6). The marking effect for seining with electrofishing was negatively related to capture efficiency, although the marking effect was relatively small and very imprecise. I estimate that marked fishes were as much as much as 1.61 times less likely to be collected to as much 1.18 times more likely to be collected than unmarked fishes.

Cross-validation of the seining with backpack electrofishing model indicated that it was relatively unbiased for most species groups (Table 6). The greatest mean differences between predicted and known efficiencies were 2.3% and 3.4% for sunfish and topminnows, respectively. Cross-validated root mean square error indicated predicted efficiency for the seining with electrofishing model was most precise for catfish (0%), darters (14.1%), sculpin (17.8%), sunfish (17.5%) and topminnows (11.1%). The root mean square error indicated that this model was least precise for bass (23.1%), minnows (24.2%) and sunfish (20.9%).

Evaluation of fish movement

One-pass backpack electrofishing was completed at 27 sites, with a total of 81 units being sampled overall. Fifty-three block netted units and 28 non-block netted units were sampled, and abundance was estimated for 51 species in 8 species groups (Table 9). An examination of the normal probability plot of the lower-level residuals from all candidate models relating site- and species-specific characteristics to estimated fish abundance indicated that the residuals were slightly heteroskedastic. I attempted to normalize the data via square root and log transformation, yet these transformations did not normalize the data. I proceeded with model fit and selection using untransformed data to increase the interpretability of parameter estimates.

The error structure that best explained variance in the hierarchical fish abundance models included an intercept that randomly varied with species, habitat variables (wood density, velocity, depth) with slopes that randomly varied with species, a block net effect that randomly varied with species and an additional random effect corresponding to sample site, and these random effects were included in all candidate models. The best-approximating model relating fish abundance to site- and species-specific characteristics included the nuisance variables, block net presence, and the interaction between block net presence and water depth but was only 1.1 times more likely than the next best approximating model (Table 10). Although I intended to use the 10% rule referenced in the methods section to determine the confidence set of models, modeling results indicated that 19 of the 20 models had Akaike weights within 10% of the best-fitting model (Table 10). I chose to include the 2 models with the highest Akaike weights in the confidence set of models, because these models had similar Akaike weights ($\leq$ 0.12) and the remainder of the models in the candidate set had less support ($\geq$ 0.08), relatively. The confidence set of models consisted of 2 models that included the nuisance habitat variables with block net by depth interaction and with two interactions, block net by depth and block net by benthic species (Table 11).

I was most interested in evaluating my predictions of fish movement relative to the presence of block nets. It is notable, though, that the effects of wood density, velocity, and depth varied widely among species (Table 11). The block net parameter estimates were consistently positive and precise across the confidence set of models, which would indicate a consistent difference in estimated abundance of fish in block netted and non-block netted units (i.e., movement out of non-block netted sampling areas). Both models in the confidence set included a block net by depth effect and the parameter estimate for this effect was negative, indicating smaller differences in abundance in deeper units (Table 11). The block net by depth interaction accounted for 1.5% of the variation in estimated abundance of non-block netted units (Table 11) and had the greatest importance weight of the predictor variables (Table 12). The inclusion of the block net by benthic species interaction accounted for 29% of the variation in block net effect (Table 11) and this interaction had the second highest importance weight of the predictor variables (Table 12). Although some variation in the block net effect was accounted with covariates, the remaining variation in the block net effect was relatively high (Table 11). There was little or no evidence that current velocity and wood density were related to fish movement in this model set, and there was no evidence that movement differed for mobile species (Table 12).

Using the best-approximating fish abundance model, I estimate that, on average, 69% of fish remained in non-block netted units, or, alternately, 31% of fish left the sampling area when block nets were absent. I examined the proportion of fish remaining in non-block netted units using species groups for the best-approximating model. Fairly precise estimates of the proportion of fish remaining in a non-block netted unit were obtained by averaging across species group for darters, minnows, and sculpin (Figure 11). Darter species had the greatest estimated proportion of fish remaining in a non-block netted unit (mean = 0.89, SE=0.18) and catfish species had the least estimated proportion of fish remaining in a non-block netted unit, although the catfish estimate was very imprecise (mean=0.32, SE=0.73). For all species groups, the estimated unconditional capture efficiencies (no block nets) were lower than the estimated conditional capture efficiencies (block nets) and the difference between conditional and unconditional capture efficiencies was greatest for bass species and the least for sculpin (Figure 12). The estimates of coefficient of variation (CV) of abundance estimates were greatest for all species groups for non-block netted samples. Topminnow and catfish species had particularly large CVs, with values of 17.13 and 115.28, respectively (Table 13; Figure 13). The CVs for abundance estimates for block netted units also varied with species group. The CVs of samples collected using 1-pass backpack electrofishing ranged from 1.22 for bass species to 2.37 for darter species (Table 13). The CVs for samples collected with seining and electrofishing ranged from 1.53 for sculpin to 2.26 for sunfish (Table 13).

CHAPTER 5

DISCUSSION

Capture efficiency modeling

Many studies have shown that no fish sampling method is 100% effective (Bayley and Dowling 1990; Bayley and Dowling 1993; Peterson et al. 2004), and the results of my study indicated that capture efficiencies for both methods were much less than 100% and varied with species and habitat characteristics. The estimated capture efficiencies in my study were relatively low and average values for species groups and methods ranged from 1% -27%. Studies that have evaluated capture efficiency of backpack electrofishing or seining using an unbiased estimator (such as capture-recapture) have found similarly low capture efficiencies (Bayley and Dowling 1990; Bayley and Peterson 2001; Rosenberger and Dunham 2005). More specifically, Bayley and Peterson (2001) observed capture efficiencies of 4%-25% for backpack electrofishing and 5%-50% for 2-pass seining (for similar species). Other studies have found greater capture efficiencies using estimates obtained via the removal model (Heimbuch et al. 1997; Kennard et al. 2006; Simonson and Lyons 1995), although this method likely overestimates capture efficiency (Peterson et al. 2004; Peterson and Cederholm 1984; Rosenberger and Dunham 2005). Removal methods are relatively unbiased if first pass capture efficiency is >35% (Peterson et al. 2004), but this study lends more evidence that such capture efficiencies are rarely encountered in field sampling situations. Because I used an unbiased capture-recapture estimator and my results were similar to other studies, I assume that my study produced reliable estimates of capture efficiency of stream fish.

Capture efficiency of both 1-pass backpack electrofishing and seining with electrofishing varied with species group and fish length, which is consistent with similar studies (Bayley and Dowling 1990; Bayley and Dowling 1993; Bayley and Peterson 2001; Heimbuch et al. 1997; Kennard et al. 2006). For example, I estimated that capture efficiency was much greater for sucker species (~25%) than for catfish species (~2%), regardless of sampling method. Capture efficiency has also been shown to increase with fish length for backpack electrofishing (Anderson 1995; Bayley and Dowling 1990; Riley and Fausch 1992) and to be non-linearly related to fish length for seining methods (i.e., fish capture peaked at some fish length and then declined, indicating that small fish and large fish were less susceptible to seine capture; Bayley and Dowling 1990; Bayley and Dowling 1993; Bayley and Herendeen 2000). I found that capture efficiencies of most species groups increased with fish length for 1-pass backpack electrofishing (Figure 4), but was relatively weak and non-parabolic for seining with electrofishing (Figure 5). Community level assessments that use metrics related to species groups, such as the index of biotic integrity (IBI; Karr et al. 1986), may be biased due to differences in capture efficiency due to species effects. Consider for example, a simplified stream that contains 100 individuals each of sensitive species (e.g., darters) and members of the Centrarchidae (e.g., sunfish) is sampled under average habitat conditions with 1-pass backpack electrofishing. The expected catch would include 8 of 100 darters and 19 of 100 sunfish and falsely indicate that there was a higher proportion of sunfish in the stream, which would bias water quality inferences or calculation of IBI. Failure to adjust data with species-specific capture efficiencies may misrepresent the fish community and bias measures of fish population status and biotic integrity.

Capture efficiency of both 1-pass backpack electrofishing and seining with electrofishing varied with habitat characteristics, such as cross sectional area or current velocity, and with increasing habitat complexity, such as wood density or cobble substrate composition. I estimated that fish were more difficult to catch in larger streams (Figure 9), which was likely due to fish swimming under, over, or around the gear to avoid capture. However, the relationship between capture efficiency and habitat characteristics also differed among species. Catchability of most species decreased with increasing wood density, but increased for bass with 1-pass backpack electrofishing and for bass and suckers for seining with electrofishing (Figure 10). I believe that these differences were due to the behavioral response of fishes to samplers, because fish often seek cover when frightened, such as during sampling or during interspecific interactions (Rahel and Stein 1988). During sampling, these species groups likely sought cover in areas of greater wood density and remained in those locations where they could then be targeted during sampling. Wood density was negatively related to other species, such as sunfish or darters. These species presumably were more difficult to capture as wood density increased because they may use woody debris in a different manner than bass or because the sampler's netting ability was hindered by debris. Regardless of the mechanism, failure to account for differences in species' capture efficiency due to habitat complexity may obscure or suggest false population trends. For example, suppose woody debris was added to a stream as a means of habitat improvement, and fish were sampled prior to and following the addition. My capture efficiency models indicate that even if there was no change in the fish community, more bass and fewer other species would be collected after the addition because capture efficiency is positively related to wood density for basses and lower for other species. Reliance on these biased data would likely result in a manager falsely concluding that the addition was successful at improving the bass population. Because fish populations are influenced by the same factors that influence capture efficiency (e.g., habitat), failure to account for differences in capture efficiency can confound population trends, leading to poor inference.

Capture efficiency varied by sampling method across species groups and habitats. One sampling method may be more effective than another at collecting a particular species or sampling a specific habitat type, and I believe it is due to how each method collects fish in a stream. Backpack electrofishing stuns fish with an electrical field, allowing fish to be netted by a sampler. Fish can be targeted in specific habitats, such as in woody debris or along banks. Backpack electrofishing may be less effective in deep or wide streams, as the effective range of the backpack electrofisher may diminish in larger streams (Peterson et al. 2004). Swift water, such as riffles, may be difficult to sample with an electrofisher, as fish presumably are swept downstream prior to being netted (Rabeni et al. In press). Seining with electrofishing, conversely, can be used in open water through "hauling" to collect schools of minnows or in swift water through kick seining. Complex habitats, such as areas of woody debris or large cobble, cannot be effectively seined through hauling because the seine becomes snagged (Holland-Bartels and Dewey 1997). Each gear utilizes a different collection strategy, thus species composition of a sample likely reflects the specific properties of the method. In my study, 1-pass backpack electrofishing was slightly more effective at collecting bass species (20% compared to 14%), and seining with electrofishing was twice as effective at collecting darter and topminnow species (8% compared to 16% and 7% compared to 12%, respectively). Bass species may be more susceptible to a 'hunting' gear such as backpack electrofishing by being targeted in likely habitats (per Bayley and Dowling 1990), and bass' strong swimming ability allows them to avoid a seine. Kick-seining in swift water may collect more darters than backpack electrofishing, due to fish being dislodged and swept into the seine. Seine hauling may be more effective than backpack electrofishing at collecting topminnows, because the fish can be "herded" into shallow water and collected in a seine. Additionally, minnows and topminnows exhibit schooling behavior, thus a school can be collected efficiently with a seine. Conversely, the school may avoid capture by collectively moving away from the seine if the entire stream cannot be covered by a seine haul. My study clearly demonstrates that site and species characteristics influence the capture efficiency of each gear. Because efficiency bias of each sampling method is not consistent across habitats or species, failure to account for differences in capture efficiency due to gear properties can confound inferences based on collected samples.

The predictive ability of capture efficiency models for each sampling method varied by species. The mean error from the cross-validation procedure was a measure of the expected bias of the models, whereas the root mean squared error (RMSE) measured the expected accuracy (i.e., bias and precision) of the capture efficiency models applied under real world circumstances. Ideally, all the models would have a very low mean error and RMSE, indicating that the models perfectly predicted capture efficiency (i.e., all variability in capture efficiency can be accounted for through species or site covariates). Variability in the accuracy of each model may be due to unknown site- or species-specific covariates not included in the model or due to some erratic or unpredictable response of fishes to sampling. Factors that contribute to sample variance include the accuracy of an estimator as well as the efficiency of the sampling method, and high sample variance can hinder a manager's ability to make sound judgment (Peterson and Rabeni 1995). Thus, a manager should consider the accuracy or predictive ability of a particular gear. While my capture efficiency models were fairly accurate for most species groups (Table 7), accuracy varied among species groups and with sampling method. For instance, the 1-pass backpack electrofishing model was more accurate than the seining with electrofishing model for all species groups but darters and sculpin. Therefore, if the objective of a sampling study was to obtain community-wide estimates (e.g., species richness), the 1-pass backpack electrofishing model would provide the most reliable estimates of efficiency among all species groups. Conversely, if the objective of a sampling study was to obtain abundance estimates of darters or sculpin, the seining with electrofishing model would be most appropriate. Because accuracy of a model contributes to sample variance, choosing the most predictive model can reduce variance and provide better estimates of population parameters.

Evaluation of Fish Movement

Fish abundance was lower in non-block netted units versus block netted units, suggesting that fish likely moved out of the non-block netted units sampling unit during sampling. Several studies reported that up to 29% of fish move upstream during sampling (Edwards et al. 2003; Peterson et al. 2005) and that fish movement is related to species, sampling method, and habitat characteristics (Edwards et al. 2003; Peterson et al. 2005; Young and Schmetterling 2004). Similarly, my study suggested that fish moved out of non-block netted sampling units (Table 11) and that as depth increased, the difference in abundance between block netted and non-block netted units diminished. Several studies have indicated that fish movement due to sampling pressure is lower in areas with available cover (e.g., undercut banks or complex substrate), because fish seek refuge for concealment when frightened (Heggenes et al. 1990; Peterson et al. 2005; Young and Schmetterling 2004). Although I did not find support that movement was influenced by available cover, my data indicates that fish likely sought refuge in deeper water within the sampling unit if deep water was available. If deep water was not available within the unit, fish fled upstream or downstream. Additionally, I predicted that benthic species would be less likely to move due to behavioral differences, yet my results indicate that benthic species actually had greater movement rates than other species. However, the block net effect varied substantially by species, thus the movement rate of benthic fishes may not be consistent for all benthic species, but might be more representative of the large sample size of certain benthic species (e.g., sculpin; Table 9). That movement differed among species and with habitat characteristics indicates that movement of fish out of the sampling area is not constant or random. Rather, differential movement alters the fish community available to samplers imparting a bias to fish sample data and should be considered when developing protocols and analyzing existing data.

Block nets can be labor intensive and difficult to maintain in certain situations (e.g., high gradient streams or areas with heavy leaf litter), but if block nets are not installed, data should be adjusted with unconditional capture efficiencies. Unconditional capture efficiencies account for the proportion of fish leaving a sampling unit and for the capture efficiency of those fish that remain. Adjusting data using estimated unconditional capture efficiency minimizes bias, but it introduces additional variance into sample data due to the greater variability in the estimated unconditional capture efficiency. I found that movement rates were highly variable among species and habitats (Table 11) and considerable variation remained unaccounted for in fish movement. This resulted in higher variance in estimated abundance for non-block netted units compared to block netted units across species groups (Table 13). The only means to overcome high variance for a fixed sample design is to increase sample size (Sokal and Rohlf 1995). Thus, greater numbers of sample units need to be collected without block nets to obtain similar precision for abundance estimates. For example, using the estimator in Peterson and Rabeni (1995) and the CV in Table 13, I estimate that 36 times more samples need to be collected using

1-pass backpack electrofishing in non-block netted units versus in block netted units to achieve 30% precision with 90% confidence for bass species. While block nets add labor and time to field sampling, they more than likely result in less effort over a sampling season. For example, assume that non-block netted sites can be sampled with half the effort compared to block netted sites (e.g., two hours versus four hours, respectively). For every one hour of sampling with block nets, 18 hours of sampling without block nets would be needed to achieve a comparable level of precision without block nets (i.e., half the effort times 36 samples). In this instance, the effort required to precisely estimate abundance without block nets would likely be cost-prohibitive.

Sampling protocol recommendations

An ideal method for sampling at-risk stream fish would be highly efficient to minimize bias and variance, require minimal effort to reduce costs, and have no adverse physiological effects on sampled fishes. Based on the results of this study, I am able to make suggestions regarding protocol development for sampling at-risk fishes. Bias from incomplete capture can be minimized by adjusting data for site- and species-specific capture efficiency. The models I created in this study can be used to adjust data collected under a similar range of conditions, or these models can serve as a guideline for other efficiency estimates. Additionally, sampling method or gear should also be chosen to provide the most efficient and accurate results. I found that 1-pass backpack electrofishing was more effective and reliable for sampling the entire fish community, but seining with electrofishing may be more reliable for collecting particular benthic species, such as darters or sculpin. Variance due to fish movement can be minimized through block net installation, when feasible. My results indicate that it would be cost-prohibitive to sample enough non-block netted units to reach a level of precision similar to block netted units.

It should be noted, however, that estimates of movement in my study were highly variable, and more research on movement rates is needed. While the initial investment of a capture efficiency calibration project may seem costly, a calibration project can be created with a single investment. The resulting models can be used to adjust both historical data and future (less-intensive) samples. Following the above guidelines can improve the quality of data and inferences by providing unbiased, precise, and cost-efficient estimates.

Capture efficiency adjustment can improve the quality of sampling data by reducing bias, but estimating capture efficiency can be logistically difficult. Prior capture-recapture studies in streams have indicated that previously handled fish may have a lower recapture probability than unhandled fish, therefore a recovery period of at least 24 hours is necessary to eliminate a handling effect in resampled individuals (Peterson et al. 2004). Average recovery time in my study ranged from 14-20 hours, thus previously handled fishes were likely to have lower recapture probabilities. To account for bias in recapture sampling, I incorporated a marking effect into the capture efficiency models. Although the marking effect was small and imprecise, the effect was supported (via DIC) during model selection, thus the variability in estimated capture efficiency due to a handling effect should be considered. While a recovery period of at least 24 hours is recommended, the escape rate of fishes from blocked off sites increases with recovery period (Peterson et al. 2004) and fish escape can bias capture efficiency estimates. Rosenberger and Dunham (2005) observed an escape rate of 3%, similar to the one estimated for my study, and they found no significant effect on mark-recapture estimates. Because escape rate observed in my study was relatively low, consistent with other studies, and the net failure variable was not supported in the models, I do not believe that the violations of closure biased my estimates. Efficiency estimates may be biased if precautions are not taken to reduce handling effects on fishes (i.e., using appropriate recovery times) and sampling areas do not remain closed throughout the sampling and recovery periods.

REFERENCES

Agresti, A. 1990. Categorical data analysis. Wiley, New York.

Akaike, H. 1973. Information theory and an extension of the maximum likelihood principle. Pages 267-281 *in* B. N. Petrov, and F. Csaki, editors. Second International Symposium on Information Theory, Budapest, Hungary.

Anderson, C. S. 1995. Measuring and correcting for size selection in electrofishing mark-recapture experiments. Transactions of the American Fisheries Society 124(5):663-676.

Angermeier, P. L., and R. A. Smogor. 1995. Estimating number of species and relative abundances in stream-fish communities: effects of sampling effort and discontinuous spatial distributions. Canadian Journal of Fisheries & Aquatic Sciences 52:936-949.

Bailey, L. L., T. R. Simons, and K. H. Pollock. 2004. Estimating detection probability parameters for Plethodon salamanders using the robust capture-recapture design. Journal of Wildlife Management 68(1):1-13.

Bayley, P. B. 1993. Quasi-likelihood estimation of marked fish recapture. Canadian Journal of Fisheries and Aquatic Sciences 50(10):2077-2085.

Bayley, P. B., and D. C. Dowling. 1990. Gear efficiency calibrations for stream and river sampling. Illinois Natural History Survey Aquatic Ecology Technical Report 90/08:Champaign, Illinois.

Bayley, P. B., and D. C. Dowling. 1993. The effect of habitat in biasing fish abundance and species richness estimates when using various sampling methods in streams. Polskie Archiwum Hydrobiologii 40(1):5-14.

Bayley, P. B., and R. A. Herendeen. 2000. The efficiency of a seine net. Transactions of the American Fisheries Society 129(4):901-923.

Bayley, P. B., and J. T. Peterson. 2001. An approach to estimate probability of presence and richness of fish species. Transactions of the American Fisheries Society 130(4):620-633.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach, Second edition. Springer, New York, New York, USA.

Cam, E., J. D. Nichols, J. R. Sauer, and J. E. Hines. 2002. On the estimation of species richness based on the accumulation of previously unrecorded species. Ecography 25(1):102-108.

Congdon, P. 2001. Bayesian Statistical Modelling. John Wiley & Sons, Ltd, Chichester, UK.

Edwards, M. R., D. L. Combs, S. B. Cook, and M. Allen. 2003. Comparison of single-pass electrofishing to depletion sampling for surveying fish assemblages in small warmwater streams. Journal of Freshwater Ecology 18(4):625-634.

Freeman, B. J., and M. C. Freeman. Unpublished museum collection records. Georgia Museum of Natural History, University of Georgia, Athens, GA.

Freeman, M. C., Athens, Georgia, University of Georgia.

Fukunaga, K., and D. L. Kessell. 1971. Estimation of Classification Error. IEEE Trans. Comput. 20(12):1521-1527.

Georgia Department of Natural Resources. 2005. Part I: Standard operating procedures for conducting biomonitoring on fish communities in wadeable streams in Georgia. Pages 1-61 *in*.

Gorman, O. T., and J. R. Karr. 1978. Habitat Structure and Stream Fish Communities. Ecology 59(3):507-515.

Hayes, D. B., C. P. Ferreri, and W. W. Taylor. 1996. Active fish capture methods. Pages 193-220 *in* B. R. Murphy, and D. W. Willis, editors. Fisheries Techniques, 2nd edition. American Fisheries Society, Bethseda, Maryland.

Heggenes, J., A. Brabrand, and S. J. Slatveit. 1990. Comparison of three methods for studies of stream habitat use by young brown trout and Atlantic salmon. Transactions of the American Fisheries Society 119:101-110.

Heimbuch, D. G., H. T. Wilson, S. B. Weisberg, J. H. Volstad, and P. F. Kazyak. 1997. Estimating fish abundance in stream surveys by using double-pass removal sampling. Transactions of the American Fisheries Society 126:795-803.

Holland-Bartels, L. E., and M. R. Dewey. 1997. The influence of seine capture efficiency on fish abundance estimates in the upper Mississippi River. Journal of Freshwater Ecology 12(1):101-111.

Hosmer, D. W., and S. Lemeshow. 1989. Applied logistic regression Wiley, New York.

Howell, J. E., J. T. Peterson, and M. J. Conroy. 2008. Building hierarchical models of avian distributions for the state of Georgia. Journal of Wildlife Management 72(1):168-178.

Jones, M. L., and J. D. Stockwell. 1995. A rapid assessment procedure for the enumeration of salmonine populations in streams. North American Journal of Fisheries Management 15(3):551-562.

Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale. Illinois Natural History Survey Special Publication 5:Champaign, Illinois.

Kennard, M. J., B. J. Pusey, B. D. Harch, E. Dore, and A. H. Arthington. 2006. Estimating local stream fish assemblage attributes: sampling effort and efficiency at two spatial scales. Marine and Freshwater Research 57(6):635-653.

Lyons, J. 1992. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. North American Journal of Fisheries Management 12(1):198-203.

Manly, B. F. J. 1971. Estimates of a marking effect with capture-recapture sampling. The Journal of Applied Ecology 8(1):181-189.

Matthews, W. J. 1990. Spatial and temporal variation in fishes of riffle habitats - a comparison of analytical approaches for the Roanoke River. American Midland Naturalist 124(1):31-45.

Meador, M. R., J. P. McIntyre, and K. H. Pollock. 2003. Assessing the efficacy of single-pass backpack electrofishing to characterize fish community structure. Transactions of the American Fisheries Society 132(1):39-46.

Mesa, M. G., and C. B. Schreck. 1989. Electrofishing mark recapture and depletion methodologies evoke behavioral and physiological-changes in cutthroat trout. Transactions of the American Fisheries Society 118(6):644-658.

Moore, D. S., and G. P. McCabe. 1993. Introduction to the Practice of Statistics. W. H. Freeman and Company, New York.

Otis, D. L., K. P. Burnham, G. C. White, and D. R. Anderson. 1978. Statistical-inference from capture data on closed animal populations. Wildlife Monographs (62):7-135.

Peterson, J. I., N. P. Banish, and R. F. Thurow. 2005. Are block nets necessary?: Movement of stream-dwelling salmonids in response to three common survey methods. North American Journal of Fisheries Management 25(2):732-743.

Peterson, J. T., and N. P. Banish. 2002. The evaluation of sampling conditions across the bull trout range in Washington State, Lacey, WA.

Peterson, J. T., and C. F. Rabeni. 1995. Optimizing sampling effort for sampling warmwater stream fish communities. North American Journal of Fisheries Management 15(3):528-541.

Peterson, J. T., and C. F. Rabeni. 2001. Evaluating the efficiency of a one-square-meter quadrat sampler for riffle-dwelling fish. North American Journal of Fisheries Management 21(1):76-85.

Peterson, J. T., R. F. Thurow, and J. W. Guzevich. 2004. An evaluation of multipass electrofishing for estimating the abundance of stream-dwelling salmonids. Transactions of the American Fisheries Society 133(2):462-475.

Peterson, N. P., and C. J. Cederholm. 1984. A comparison of the removal and mark-recapture methods of population estimation for juvenile coho salmon in a small stream. North American Journal of Fisheries Management 4(1):99-102.

Pusey, B. J., M. J. Kennard, J. M. Arthur, and A. H. Arthington. 1998. Quantitative sampling of stream fish assemblages: Single- vs multiple-pass electrofishing. Australian Journal of Ecology 23(4):365-374.

Rabeni, C. F., J. T. Peterson, J. Lyons, and N. Mercado-Silva. In press. Warmwater fish in wadeable streams. S. Bonar, D. W. Willis, and W. A. Hubert, editors. Standard sampling methods for North American freshwater fishes. American Fisheries Society, Bethesda, Maryland.

Rahel, F. J., and R. A. Stein. 1988. Complex Predator-Prey Interactions and Predator Intimidation among Crayfish, Piscivorous Fish, and Small Benthic Fish. Oecologia 75(1):94-98.

Reynolds, J. B. 1996. Electrofishing. Pages 221-254 *in* B. R. Murphy, and D. W. Willis, editors. Fisheries Techniques, 2nd edition. American Fisheries Society, Bethseda, Maryland.

Rieman, B. E., J. T. Peterson, and D. L. Myers. 2006. Have brook trout (Salvelinus fontinalis) displaced bull trout (Salvelinus confluentus) along longitudinal gradients in central Idaho streams? Canadian Journal of Fisheries & Aquatic Sciences 63(1):63-78.

Riley, S. C., and K. D. Fausch. 1992. Underestimation of trout population size by maximum-likelihood removal estimates in small streams. North American Journal of Fisheries Management 12(4):768-776.

Rodgers, J. D., M. F. Solazzi, S. L. Johnson, and M. A. Buckman. 1992. Comparison of three techniques to estimate juvenile coho salmon populations in small streams. North American Journal of Fisheries Management 12(1):79-86.

Rosenberger, A. E., and J. B. Dunham. 2005. Validation of abundance estimates from mark-recapture and removal techniques for rainbow trout captured by electrofishing in small streams. North American Journal of Fisheries Management 25(4):1395-1410.

Royall, R. M. 1997. Statistical evidence: a likelihood paradigm. Chapman & Hall, New York.

SAS Institute. 2002-2003. SAS 9.1.3 Help and Documentation. SAS Institute, Inc., Cary, North Carolina.

Simonson, T. D., and J. Lyons. 1995. Comparison of catch per effort and removal procedures for sampling stream fish assemblages. North American Journal of Fisheries Management 15(2):419-427.

Snijders, T. A. B., and R. J. Bosker. 1999. Multilevel analysis: an introduction to basic and advanced multilevel modeling. Sage Publications, Thousand Oaks, CA.

Sokal, R. R., and F. J. Rohlf. 1995. Biometry: the principles and practice of statistics in biological research, 3rd edition. W. H. Freeman and Co., New York.

Spiegelhalter, D. J., N. G. Best, B. R. Carlin, and A. van der Linde. 2002. Bayesian measures of model complexity and fit. Journal of the Royal Statistical Society Series B-Statistical Methodology 64:583-616.

Strahler, A. N. 1952. Hypsometric (area-altitude) analysis of erosional topography. Geological Society of America Bulletin 63(11):1117-1142.

Thompson, S. K., and G. A. F. Seber. 1994. Detectability in conventional and adaptive sampling. Biometrics 50(3):712-724.

U. S. Fish and Wildlife Service. 2007. Endangered Species Program: Consultations with Federal Agencies. Arlington, VA.

Wildman, T. L., and R. M. Neumann. 2003. Comparison of snorkeling and electrofishing for estimating abundance and size structure of brook trout and brown trout in two southern New England streams. Fisheries Research 60(1):131-139.

Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and management of animal populations : modeling, estimation, and decision making. Academic Press, San Diego.

Young, M. K., and D. A. Schmetterling. 2004. Electrofishing and salmonid movement: reciprocal effects in two montane streams. Journal of Fish Biology 64:750-761.

Zippin, C. 1956. An evaluation of the removal method of estimating animal populations. Biometrics 12(2):163-189.

Table 1. Federal and state listed fishes in Upper Coosa River sub-basins, Georgia.

Basin	Federally Listed	State Listed
Conasauga	Blue shiner (*Cyprinella caerulea*)	Coldwater darter (*Etheostoma ditrema*)
	Amber darter (*Percina antesella*)	Trispot darter (*Etheostoma trisella*)
	Conasauga logperch (*Percina jenkinsi*)	River redhorse (*Moxostoma carinatum*)
		Frecklebelly madtom (*Noturus munitus*)
		Holiday darter (*Etheostoma brevirostrum)*
		Freckled darter (*Percina lenticula*)
		River darter (*Percina shumardi*)
		Bridled darter (*Percina kusha*)
Coosawattee	Goldline darter (*Percina aurolineata)*	Holiday darter (*Etheostoma brevirostrum*)
		Trispot darter (*Etheostoma trisella*)
		River redhorse (*Moxostoma carinatum)*
Etowah	Etowah darter (*Etheostoma etowahae*)	Holiday darter (*Etheostoma brevirostrum*)
	Cherokee darter (*Etheostoma scotti*)	Frecklebelly madtom (*Noturus munitus*)
	Amber darter (*Percina antesella)*	Freckled madtom (*Noturus nocturnus*)
		Freckled darter (*Percina lenticula*)
		Bridled darter (*Percina kusha*)
Oostanaula		Coldwater darter (*Etheostoma ditrema*)
		Trispot darter (*Etheostoma trisella*)
		River redhorse (*Moxostoma carinatum*)

Table 2. Site identification, name, stream link magnitude, sub-basin, physiographic province, county, and location of sample sites (n=31) in the Upper Coosa River Basin in Georgia.

Site ID	Stream Name	Link	Sub-Basin	Physiographic Province	County	Latitude	Longitude
S1-con4	Dills Creek	2	Conasauga	Blue Ridge	Murphy	N 34.81567	W 84.66691
S2-con2	N. Prong Sumac Creek	8	Conasauga	Ridge and Valley	Murray	N 34.92835	W 84.74210
S2-con4	Sumac Creek	8	Conasauga	Ridge and Valley	Murray	N 34.88974	W 84.73922
S2-con7	Mill Creek	3	Conasauga	Ridge and Valley	Murray	N 34.80351	W 84.69343
S2-con8	Kenyon Creek	2	Conasauga	Ridge and Valley	Whitfield	N 34.89569	W 84.96321
S3-coo3	Boardtown Creek	10	Coosawattee	Blue Ridge	Gilmer	N 34.77238	W 84.41951
S3-coo4	Mountaintown Creek	5	Coosawattee	Blue Ridge	Gilmer	N 34.80172	W 84.56461
S3-coo5[a]	Rolston Creek	5	Coosawattee	Blue Ridge	Gilmer	N 34.67633	W 84.33771
S3-coo7	Rock Creek	6	Coosawattee	Blue Ridge	Gilmer	N 34.77875	W 84.38984
S3-coo9	Kells Creek	4	Coosawattee	Blue Ridge	Gilmer	N 34.73069	W 84.47404
S4-coo1	Scarecorn Creek	4	Coosawattee	Piedmont	Pickens	N 34.45304	W 84.55756
S4-coo3	Tails Creek	4	Coosawattee	Piedmont	Gilmer	N 34.69505	W 84.60833
S4-coo5	Owltown Creek	4	Coosawattee	Piedmont	Gilmer	N 34.66396	W 84.44096
S4-coo8	Owltown Creek	4	Coosawattee	Piedmont	Gilmer	N 34.66712	W 84.44794
S5-coo5	Sugar Creek	8	Coosawattee	Ridge and Valley	Murray	N 34.63742	W 84.74260
S5-coo7	Rock Creek	4	Coosawattee	Ridge and Valley	Bartow	N 34.37428	W 84.77479
S5-coo8	Sugar Creek	6	Coosawattee	Ridge and Valley	Murray	N 34.68309	W 84.70526
S6-eto1[a]	Poverty Creek	4	Etowah	Blue Ridge	Dawson	N 34.51869	W 84.18246
S6-eto3	Nimblewilll Creek	7	Etowah	Blue Ridge	Lumpkin	N 34.55755	W 84.12822
S6-eto4	Cochrans Creek	3	Etowah	Blue Ridge	Dawson	N 34.49485	W 84.19645
S6-eto6	Ward Creek	3	Etowah	Blue Ridge	Lumpkin	N 34.61168	W 84.09688
S7-eto1	Boston Creek	6	Etowah	Piedmont	Bartow	N 34.21672	W 84.67629
S7-eto2	Champion Creek	7	Etowah	Piedmont	Pickens	N 34.47309	W 84.40404
S7-eto5	Long Swamp Creek	13	Etowah	Piedmont	Pickens	N 34.46718	W 84.39997

Table 2. Continued.

Site ID	Stream Name	Link	Sub-Basin	Physiographic Province	County	Latitude	Longitude
S8-eto1	Toms Creek	9	Etowah	Ridge and Valley	Bartow	N 34.24703	W 85.01454
S8-eto3	Nancy Creek	7	Etowah	Ridge and Valley	Bartow	N 34.18367	W 84.82729
S8-eto5	Silver Creek	10	Etowah	Ridge and Valley	Floyd	N 34.17635	W 85.16421
S9-oos12	Gunn Creek	3	Oostanaula	Ridge and Valley	Bartow	N 34.35814	W 84.92497
S9-oos2	Johns Creek	17	Oostanaula	Ridge and Valley	Floyd	N 34.51509	W 85.11104
S9-oos4	Dry Creek	14	Oostanaula	Ridge and Valley	Gordon	N 34.57578	W 84.95558
S9-oos7	Johns Creek	4	Oostanaula	Ridge and Valley	Gordon	N 34.57968	W 85.09173

[a]Indicates a double block netted site

Table 3. Mean, standard deviation (SD), and range of habitat characteristics of the 31 sites included in the capture efficiency models.

Habitat Characteristic	Mean	SD	Range Of Means
Unit Length (m)	69.34	12.27	50 – 100
Wetted Width (m)	5.70	2.26	2.5 – 9.3
Sampling Area (m^2)	413.67	175.66	171.6 – 782.9
Depth (m)	0.17	0.14	0.07 – 0.29
Wood Density[a,b,c,d] (no./m^2)	0.03	0.03	0.00 – 0.19
Temperature (°C)	20.00	3.91	6.1 – 24.6
Conductivity (µs)	91.60	87.60	14 – 298
Turbidity (NTU)[a,b,c]	16.70	18.82	2 – 137
Mean Cross-Sectional-Area[a,b,c,d] (m^2)	1.00	0.66	0.16 – 3.11
Mean Velocity (m/s)[a,b,c,d]	0.08	0.07	0.00 – 0.31
Percent Gravel[a,b]	21.65	13.13	0 – 62
Percent Cobble[a,b,c,d]	27.48	16.20	0 – 60
Percent Boulder	12.32	11.19	0 – 36
Percent Bedrock	7.86	15.57	0 – 82
Proportion Of Undercut Bank	0.09	0.10	0 – 0.50

[a]Included in the most plausible set of parameters for the backpack electrofishing model

[b]Included in the most plausible set of parameters for the seining with electrofishing model

[c]Included in the best-fitting 1-pass backpack electrofishing model

[d]Included in the best-fitting seining with electrofishing model

Table 4. Species name, common name, family and species group for all species used in capture efficiency model development.

Species name	Common name	Family	Species group	Number marked	Mean TL	TL range
Ambloplites ariommus	Shadow bass	Centrarchidae	Sunfish	3	143.3	88 - 185
Ameiurus natalis	Yellow bullhead	Ictaluridae	Catfish	3	95.5	65 - 150
Campostoma oligolepis	Largescale stoneroller	Cyprinidae	Minnow	1490	74.1	27 - 224
Catostomus commersoni	White sucker	Catostomidae	Sucker	1	187.0	187 - 187
Cottus carolinae	Banded sculpin	Cottidae	Sculpin	2211	62.1	23 - 110
Cyprinella callistia	Alabama shiner	Cyprinidae	Minnow	293	69.3	20 - 130
Cyprinella trichroistia	Tricolor shiner	Cyprinidae	Minnow	882	60.0	27 - 113
Cyprinella venusta	Blacktail shiner	Cyprinidae	Minnow	45	70.7	35 - 130
Etheostoma brevirostrum	Holiday darter	Percidae	Darter	24	53.1	43 - 70
Etheostoma coosae	Coosa darter	Percidae	Darter	364	50.8	30 - 91
Etheostoma etowahae	Etowah darter	Percidae	Darter	12	53.6	39 - 65
Etheostoma jordani	Greenbreast darter	Percidae	Darter	14	56.3	41 - 71
Etheostoma scotti	Cherokee darter	Percidae	Darter	66	51.4	32 - 70
Etheostoma stigmaeum	Speckled darter	Percidae	Darter	24	50.4	41 - 60
Fundulus stellifer	Southern studfish	Fundulidae	Topminnow	93	68.4	42 - 111
Gambusia species	Mosquitofish	Poeciliidae	Topminnow	13	34.7	20 - 55
Hybopsis lineapunctata	Lined chub	Cyprinidae	Minnow	8	62.2	55 - 80
Hybopsis sp. H. winchelli	Coastal chub	Cyprinidae	Minnow	9	70.5	60 - 80
Hypentelium etowanum	Alabama hogsucker	Catostomidae	Sucker	646	97.1	30 - 232
Lepomis auritus	Redbreast sunfish	Centrarchidae	Sunfish	388	79.9	30 - 205
Lepomis cyanellus	Green sunfish	Centrarchidae	Sunfish	99	89.1	40 - 176
Lepomis gulosus	Warmouth	Centrarchidae	Sunfish	7	153.9	106 - 210
Lepomis hybrid	Sunfish hybrid	Centrarchidae	Sunfish	4	111.0	66 - 150
Lepomis macrochirus	Bluegill	Centrarchidae	Sunfish	178	86.0	36 - 202

Table 4. Continued.

Species name	Common name	Family	Species group	Number marked	Mean TL	TL range
Lepomis megalotis	Longear sunfish	Centrarchidae	Sunfish	69	78.4	42 - 140
Lepomis microlophus	Redear sunfish	Centrarchidae	Sunfish	22	125.4	92 - 251
Lepomis punctatus	Spotted sunfish	Centrarchidae	Sunfish	44	92.4	39 - 145
Luxilus chrysocephalus	Striped shiner	Cyprinidae	Minnow	127	76.2	28 - 180
Luxilus zonistius	Bandfin shiner	Cyprinidae	Minnow	583	66.8	36 - 122
Lythrurus lirus	Mountain shiner	Cyprinidae	Minnow	3	71.6	55 - 84
Micropterus coosae	Redeye bass	Centrarchidae	Bass	172	112.0	32 - 275
Micropterus salmoides	Largemouth bass	Centrarchidae	Bass	8	74.1	34 - 206
Minytrema melanops	Spotted sucker	Catostomidae	Sucker	13	211.1	180 - 271
Moxostoma duquesnei	Black redhorse	Catostomidae	Sucker	21	132.4	31 - 327
Moxostoma erythrurum	Golden redhorse	Catostomidae	Sucker	2	130.0	88 - 161
Moxostoma species	Redhorse species	Catostomidae	Sucker	17	90.5	40 - 380
Nocomis leptocephalus	Bluehead chub	Cyprinidae	Minnow	212	72.9	35 - 214
Notropis chrosomus	Rainbow shiner	Cyprinidae	Minnow	204	58.8	29 - 92
Notropis lutipinnis	Yellowfin shiner	Cyprinidae	Minnow	249	56.0	25 - 87
Notropis stilbius	Silverstripe shiner	Cyprinidae	Minnow	26	73.1	35 - 102
Notropis xaenocephalus	Coosa shiner	Cyprinidae	Minnow	347	57.8	30 - 94
Noturus leptacanthus	Speckled madtom	Ictaluridae	Catfish	15	66.8	30 - 90
Percina aurolineata	Goldline darter	Percidae	Darter	2	67.0	59 - 74
Percina kathae	Mobile logperch	Percidae	Darter	18	98.8	70 - 135
Percina nigrofasciata	Blackbanded darter	Percidae	Darter	99	72.7	40 - 113
Percina palmaris	Bronze darter	Percidae	Darter	104	67.6	39 - 100
Phenacobius catostomus	Riffle minnow	Cyprinidae	Minnow	9	73.0	44 - 113
Pomoxis nigromaculatus	Black crappie	Centrarchidae	Sunfish	3	140.2	60 - 256
Rhynicthys atratulus	Blacknose dace	Cyprinidae	Minnow	1	69.8	61 - 85
Semotilus atromaculatus	Creek chub	Cyprinidae	Minnow	444	68.3	24 - 186

Table 5. Parameter estimates, standard deviation, lower and upper Bayesian credibility intervals, unit scalars, and scaled odds ratios (OR) for 1-pass backpack electrofishing model, with the minnow species group serving as the baseline.

Parameter	Mean	SD	Lower	Upper	Unit Scalar	Scaled OR
Fixed Effects						
Intercept	-2.161	0.12	-2.303	-1.934	-	-
Bass	-0.584	0.37	-1.425	0.041	1	0.558
Catfish	-2.591	0.189	-2.976	-2.236	1	0.075
Darter	-1.175	0.24	-1.645	-0.603	1	0.309
Sculpin	-0.312	0.182	-0.739	-0.036	1	0.732
Sucker	0.774	0.264	0.11	1.091	1	2.168
Sunfish	2.425	0.296	1.704	2.849	1	11.302
Mean Cross-Sectional Area	-0.839	0.051	-0.911	-0.725	1	0.432
Mean Total Length	0.022	0.002	0.018	0.023	30	1.935
High Turbidity	-0.175	0.108	-0.393	0.02	1	0.839
High Turbidity*Sculpin	0.443	0.203	0.054	0.841	1	1.557
Mean Cross-Sectional Area*Bass	0.559	0.249	0.072	1.044	1	1.749
Mean Cross-Sectional Area*Sculpin	0.784	0.099	0.594	0.975	1	2.190
Mean Total Length*Bass	-0.010	0.003	-0.015	-0.003	30	0.741
Mean Total Length*Sculpin	-0.014	0.004	-0.019	-0.005	30	0.657
Mean Total Length*Sucker	-0.010	0.003	-0.013	-0.003	30	0.741
Mean Total Length*Sunfish	-0.022	0.003	-0.025	-0.015	30	0.517
Mean Total Length*Topminnow	-0.015	0.004	-0.023	-0.006	30	0.638
Mean Velocity* Darter	1.354	0.544	0.340	2.453	0.1	1.145
Mean Velocity* Sunfish	-7.497	3.157	-13.607	-1.226	0.1	0.473
Percent Cobble*Darter	0.010	0.005	-0.001	0.021	10	1.105
Wood Density*Bass	10.443	6.916	-3.217	24.139	0.05	1.686
Wood Density*Sucker	-11.015	6.258	-23.421	0.947	0.05	0.577
Wood Density*Sunfish	-8.254	3.214	-14.952	-2.266	0.05	0.662
Marking Effect	-0.186	0.165	-0.509	0.135	1	0.830
Random Effect						
Random Effect For Unit	1.592	0.649	1.001	2.849	-	-

Table 6. Parameter estimates, standard deviation, upper and lower Bayesian credibility intervals, odds ratio unit scalars and scaled odds ratios for seining with electrofishing model, with the minnow species group serving as the baseline.

Parameter	Mean	SD	Lower	Upper	Unit Scalar	Scaled OR
Fixed Effects						
Intercept	-1.936	0.108	-2.155	-1.671	-	-
Bass	-1.945	0.409	-2.746	-1.139	1	0.143
Catfish	-3.259	0.213	-3.689	-2.867	1	0.038
Darter	0.577	0.321	-0.054	1.258	1	1.781
Sculpin	-0.144	0.054	-0.247	-0.040	1	0.866
Sucker	0.417	0.383	-0.334	1.195	1	1.517
Sunfish	0.355	0.197	-0.014	0.761	1	1.426
Mean Cross-Sectional Area	-0.214	0.040	-0.28	-0.123	1	0.807
Mean Velocity	3.109	0.366	2.401	3.823	0.1	1.365
Percent Cobble	0.006	0.002	0.002	0.009	10	1.062
Wood Density	7.885	1.070	5.791	9.913	0.05	1.483
Mean Cross-Sectional Area*Darter	-0.428	0.151	-0.723	-0.127	1	0.652
Mean Cross-Sectional Area*Sucker	-0.203	0.112	-0.420	0.016	1	0.816
Mean Total Length*Bass	0.012	0.003	0.006	0.019	30	1.433
Mean Total Length*Sucker	0.003	0.002	-0.002	0.007	30	1.094
Mean Total Length*Topminnow	-0.001	0.001	-0.003	0.000	30	0.970
Mean Velocity* Darter	4.401	1.676	1.139	7.689	0.1	1.553
Percent Cobble*Darter	-0.020	0.006	-0.032	-0.010	10	0.819
Percent Cobble*Sculpin	-0.006	0.003	-0.012	0.000	10	0.942
Wood Density*Bass	3.806	2.053	-0.181	7.843	0.05	1.210
Wood Density*Darter	-11.595	2.671	-16.911	-6.350	0.05	0.560
Wood Density*Sunfish	-11.440	2.119	-15.656	-7.307	0.05	0.564
Wood Density*Topminnow	-13.584	3.073	-19.596	-7.607	0.05	0.507
Marking Effect	-0.158	0.164	-0.478	0.164	1	0.854
Random Effect						
Random Effect For Unit	0.262	0.056	0.184	0.397	-	-

Table 7. Cross-validation results, including mean error and root mean squared error (RMSE) for best-fitting capture efficiency models for 1-pass backpack electrofishing and seining with electrofishing.

1-Pass backpack electrofishing model

Species group	Mean Error	RMSE
Bass	-0.006	0.084
Catfish	0.000	0.000
Darter	0.005	0.169
Minnow	0.012	0.196
Sculpin	0.025	0.206
Sucker	0.006	0.181
Sunfish	-0.007	0.146
Topminnow	0.006	0.080

Seining with electrofishing model

Species group	Mean Error	RMSE
Bass	-0.007	0.231
Catfish	0.000	0.000
Darter	-0.010	0.141
Minnow	-0.006	0.242
Sculpin	-0.017	0.178
Sucker	-0.015	0.209
Sunfish	-0.023	0.175
Topminnow	-0.034	0.111

Table 8. Interpretation of predictor variables on estimated abundance used in candidate models relating abundance to the presence of block nets.

Predictor variable	Biological interpretation
sampling area	area of sampling unit may influence the number of fish available to sampler
depth, velocity, wood density	channel characteristics influencing cover and the amount and capacity of local habitats
block net	indicates whether a unit was block netted to prevent fish movement
block net x depth	direct interaction influencing fish movement due to differences in depth
block net x velocity	direct interaction influencing fish movement differences due to velocity by altering the ability to swim upstream or sweeping fish out of electrical field
block net x wood density	direct interaction influencing fish movement difference due to available cover
block net x benthic fishes, block net x mobile fishes	direct interaction influencing fish movement differences due to fish behavior, shape, or size

Table 9. Species, mean estimated abundance ($\hat{\bar{N}}$), standard error (SE) and number of block netted (BN) and non-block netted units where each species was collected for abundance estimation models using 1-pass backpack electrofishing.

Species name	Common name	Family	Species group	$\hat{\bar{N}}$	SE	No. BN units	No. non-BN units
Ambloplites ariommus	Shadow bass	Centrarchidae	Sunfish	3	1.59	2	0
Ameiurus natalis	Yellow bullhead	Ictaluridae	Catfish	21	7.89	4	1
Cyprinella callistia	Alabama shiner	Cyprinidae	Minnow	68	16.04	22	13
Cottus carolinae	Banded sculpin	Cottidae	Sculpin	196	20.60	47	22
Campostoma oligolepis	Largescale stoneroller	Cyprinidae	Minnow	75	10.55	47	26
Cyprinella trichroistia	Tricolor shiner	Cyprinidae	Minnow	62	9.61	31	18
Cyprinella venusta	Blacktail shiner	Cyprinidae	Minnow	5	2.15	4	1
Etheostoma brevirostrum	Holiday darter	Percidae	Darter	18	6.31	4	2
Etheostoma coosae	Coosa darter	Percidae	Darter	98	13.99	24	12
Etheostoma etowahae	Etowah darter	Percidae	Darter	55	21.44	2	2
Etheostoma jordani	Greenbreast darter	Percidae	Darter	48	19.95	4	1
Etheostoma scotti	Cherokee darter	Percidae	Darter	101	15.53	6	2
Etheostoma stigmaeum	Speckled darter	Percidae	Darter	18	5.82	5	3
Fundulus stellifer	Southern studfish	Fundulidae	Topminnow	14	5.05	6	1
Gambusia species	Mosquitofish	Poeciliidae	Topminnow	17	8.13	2	2
Hypentelium etowanum	Alabama hogsucker	Catostomidae	Sucker	31	4.62	49	22
Hybopsis sp. H. winchelli	Coastal chub	Cyprinidae	Minnow	26	18.28	2	1
Lepomis hybrid	Sunfish hybrid	Centrarchidae	Sunfish	5	2.91	2	0
Lepomis auritus	Redbreast sunfish	Centrarchidae	Sunfish	32	8.06	20	13
Luxilus chrysocephalus	Striped shiner	Cyprinidae	Minnow	22	14.02	7	4
Lepomis cyanellus	Green sunfish	Centrarchidae	Sunfish	27	7.71	13	7
Lepomis gulosus	Warmouth	Centrarchidae	Sunfish	2	0.93	3	0
Lythrurus lirus	Mountain shiner	Cyprinidae	Minnow	8	7.67	0	1
Lepomis macrochirus	Bluegill	Centrarchidae	Sunfish	14	3.11	18	12
Lepomis megalotis	Longear sunfish	Centrarchidae	Sunfish	12	3.74	11	5

Table 9. Continued.

Species name	Common name	Family	Species group	$\hat{N}$	SE	No. BN units	No. non-BN units
Lepomis microlophus	Redear sunfish	Centrarchidae	Sunfish	6	1.71	4	2
Lepomis punctatus	Spotted sunfish	Centrarchidae	Sunfish	10	2.96	11	4
Luxilus zonistius	Bandfin shiner	Cyprinidae	Minnow	135	37.82	8	4
Micropterus coosae	Redeye bass	Centrarchidae	Bass	13	1.63	39	17
Moxostoma duquesnei	Black redhorse	Catostomidae	Sucker	4	1.59	5	1
Moxostoma erythrurum	Golden redhorse	Catostomidae	Sucker	1	1.00	1	0
Minytrema melanops	Spotted sucker	Catostomidae	Sucker	11	10.35	2	0
Micropterus punctulatus	Spotted bass	Centrarchidae	bass	2	2.33	1	0
Micropterus salmoides	Largemouth bass	Centrarchidae	Bass	1	1.00	1	0
Moxostoma species	Redhorse species	Catostomidae	Sucker	6	5.23	1	1
Notropis chrosomus	Rainbow shiner	Cyprinidae	Minnow	38	13.24	5	5
Notemigonus crysoleucas	Golden shiner	Cyprinidae	minnow	1	1.00	1	0
Noturus leptacanthus	Speckled madtom	Ictaluridae	Catfish	58	26.64	4	2
Nocomis leptocephalus	Bluehead chub	Cyprinidae	Minnow	55	20.07	12	6
Notropis lutipinnis	Yellowfin shiner	Cyprinidae	Minnow	141	25.63	4	2
Notropis stilbius	Silverstripe shiner	Cyprinidae	Minnow	12	7.76	3	1
Notropis xaenocephalus	Coosa shiner	Cyprinidae	Minnow	36	7.13	27	13
Percina aurolineata	Goldline darter	Percidae	Darter	4	4.33	1	0
Phenacobius catostomus	Riffle minnow	Cyprinidae	Minnow	21	4.73	2	1
Perca flavescens	Yellow perch	Percidae	darter	1	1.00	0	1
Percina kathae	Mobile logperch	Percidae	Darter	12	4.10	3	4
Percina nigrofasciata	Blackbanded darter	Percidae	Darter	32	6.46	17	8
Percina palmaris	Bronze darter	Percidae	Darter	41	11.14	19	9
Pimephales vigilax	Bullhead minnow	Cyprinidae	Minnow	1	1.33	0	1
Rhynicthys atratulus	Blacknose dace	Cyprinidae	Minnow	1	1.33	0	1
Semotilus atromaculatus	Creek chub	Cyprinidae	Minnow	22	3.57	30	13

Table 10. Predictor variables, with habitat variables referring to the nuisance variables of wood density, mean velocity, mean depth, sampling area, and block net presence, number of parameters (K), Akaike's Information Criterion with the small sample bias adjustment, ΔAICc, and Akaike weights (*w*) for the set of candidate models (*i*) for predicting fish abundance in sampling units (n=81) in the Upper Coosa River Basin, Georgia.

Candidate model	K	AIC_c	ΔAIC_c	w_i
habitat variables, block net x depth	14	11950.6	0	0.132
habitat variables, block net x benthic species, block net x depth	15	11950.8	0.178	0.121
habitat variables	13	11951.5	0.907	0.084
habitat variables, block net x wood density, block net x depth	15	11951.6	1.019	0.08
habitat variables, block net x benthic species	14	11951.7	1.127	0.075
habitat variables, block net x wood density, block net x benthic species, block net x depth	16	11951.7	1.165	0.074
habitat variables, block net x depth, block net x mobile species	15	11952.5	1.923	0.051
habitat variables, block net x depth, block net x benthic species, block net x mobile species	16	11952.6	2.002	0.049
habitat variables, block net x velocity, block net x depth	15	11952.6	2.008	0.049
habitat variables, block net x velocity, block net x benthic species, block net x depth	16	11952.7	2.17	0.045
habitat variables, block net x wood density	14	11953.1	2.558	0.037
habitat variables, block net x mobile species	14	11953.5	2.896	0.031
habitat variables, block net x wood density, block net x depth, block net x mobile species	16	11953.5	2.921	0.031
habitat variables, block net x velocity	14	11953.5	2.935	0.031
habitat variables, block net x depth, block net x wood density, block net x velocity	16	11953.7	3.082	0.028
habitat variables, block net x velocity, block net x benthic species	15	11953.7	3.145	0.027
habitat variables, block net x velocity, block net x depth, block net x mobile species	16	11954.5	3.92	0.019
habitat variables, block net x wood density, block net x mobile species	15	11955.1	4.545	0.014
habitat variables, block net x wood density, block net x velocity	15	11955.2	4.617	0.013
habitat variables, block net x benthic species, block net x depth, block net x mobile species, block net x wood density, block net x velocity	18	11955.7	5.097	0.01

Note: Akaike weights are interpreted as relative plausibility of candidate models.

Table 11. Estimates, standard errors (in parentheses), and lower and upper 95% confidence limits for confidence set of hierarchical linear models of fish abundance in streams (n=27) in the Upper Coosa Basin.

Parameter Estimate	Estimate (SE)	Lower	Upper
Wood density, velocity, depth, sampling area, block net, block net x depth			
Fixed Effect			
Intercept	-7.55 (18.86)	-46.33	31.22
Wood density	-3.06 (186.39)	-377.63	371.51
Velocity	-158.19 (82.55)	-325.46	9.079
Mean depth	135.54 (103.95)	-73.47	344.54
Sampling area	0.067 (0.027)	0.014	0.120
Block net	28.53 (12.38)	3.53	53.53
Mean depth*Block net	-146.51 (84.98)	-313.32	20.31
Random Effect			
Intercept[a]	1245.14 (472.87)	734.44	2664.68
Wood[a]	71617 (67861)	24829	1082769
Velocity[a]	49660 (23283)	26490	133488
Mean depth[a]	31348 (15058)	16515	86936
Block Net[a]	205.48 (141.01)	88.27	1074.84
Site	416.22 (161.00)	243.49	906.21
Residual	4331.61 (210.41)	4006.23	4700.79
Wood density, velocity, mean depth, sampling area, block net, block net x mean depth, block net x benthic species			
Fixed Effect			
Intercept	-8.08 (18.84)	-46.80	30.65
Wood density	-3.11 (186.01)	-376.92	370.70
Velocity	-159.17 (82.51)	-326.35	8.01
Mean depth	136.07 (103.87)	-72.78	344.92
Sampling area	0.07 (0.03)	0.01	0.12
Block net	25.27 (12.56)	-0.11	50.65
Mean depth*Block net	-147.57 (84.93)	-314.28	19.14
Benthic species*Block net	15.07 (10.29)	-5.12	35.27
Random Effect			
Intercept[a]	1238.42 (472.64)	728.87	2662.48
Wood[a]	69514 (67232)	23757	1153863
Velocity[a]	49688 (23256)	26528	133273
Mean depth[a]	31226 (15078)	16406	87192
Block Net[a]	142.02 (134.48)	49.26	2140.73
Site	414.14 (160.61)	241.99	903.91
Residual	4338.68 (211.07)	4012.31	4709.06

[a]Indicates a random effect that varies with species

Table 12. Akaike importance weights for parameters from candidate models of fish abundance.

Model parameter	No. of candidate models	Importance weights
Block net x depth	12	0.6876
Block net x benthic species	7	0.4037
Block net x wood density	8	0.2865
Block net x velocity	8	0.2217
Block net x mobile species	7	0.2037

Table 13. Variance of estimated abundance by species group for 1-pass backpack electrofishing and seining with electrofishing in block netted units (Y) and non-block netted units (N).

Sampling method	Species group	Block Net	$\hat{p}$	var($\hat{p}$)	$\hat{N}$	Var($\hat{N}$)	Variance of $\tilde{N}$	CV
1-pass BPEF	bass	Y	0.22	0.01	12	186.12	207.13	1.22
Seine + EF	bass	Y	0.13	0.05	12	186.12	613.87	2.10
1-pass BPEF	bass	N	0.14	0.98	12	186.12	7572.88	7.36
1-pass BPEF	catfish	Y	0.02	0.00	40	4798.68	4798.68	1.75
Seine + EF	catfish	Y	0.01	0.00	40	4798.68	4798.68	1.75
1-pass BPEF	catfish	N	0.01	0.73	40	4798.68	20866628.09	115.28
1-pass BPEF	darter	Y	0.09	0.03	50	4228.37	13997.46	2.37
Seine + EF	darter	Y	0.14	0.02	50	4228.37	6598.89	1.63
1-pass BPEF	darter	N	0.08	0.34	50	4228.37	153064.12	7.83
1-pass BPEF	minnow	Y	0.21	0.04	52	5971.88	8288.79	1.76
Seine + EF	minnow	Y	0.18	0.06	52	5971.88	10596.35	1.99
1-pass BPEF	minnow	N	0.18	0.39	52	5971.88	37167.11	3.72
1-pass BPEF	sculpin	Y	0.12	0.04	196	31820.79	152663.78	1.99
Seine + EF	sculpin	Y	0.14	0.03	196	31820.79	90024.69	1.53
1-pass BPEF	sculpin	N	0.09	0.28	196	31820.79	1380512.81	5.99
1-pass BPEF	sucker	Y	0.24	0.03	25	1472.13	1818.85	1.70
Seine + EF	sucker	Y	0.26	0.04	25	1472.13	1880.26	1.73
1-pass BPEF	sucker	N	0.19	0.62	25	1472.13	12056.14	4.39
1-pass BPEF	sunfish	Y	0.21	0.02	18	1200.72	1356.95	2.08
Seine + EF	sunfish	Y	0.15	0.03	18	1200.72	1606.59	2.26
1-pass BPEF	sunfish	N	0.16	0.53	18	1200.72	7533.75	4.90
1-pass BPEF	topminnow	Y	0.09	0.01	15	315.70	488.85	1.47
Seine + EF	topminnow	Y	0.11	0.01	15	315.70	546.69	1.55
1-pass BPEF	topminnow	N	0.05	0.82	15	315.70	66555.23	17.13

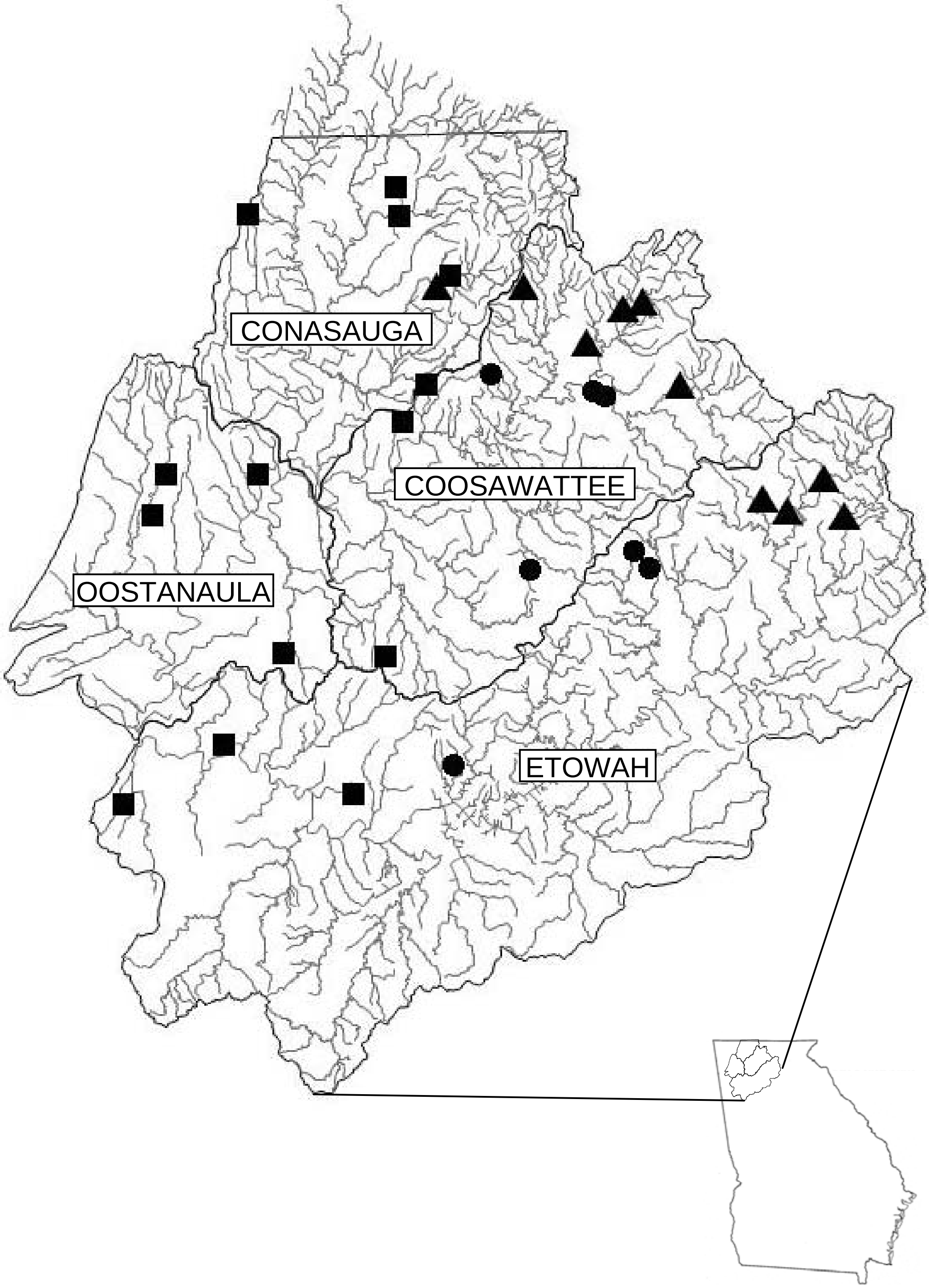


Figure 1. Locations of sample sites in the Upper Coosa River Basin of Georgia that were sampled during 2006 and 2007; sub-basins are labeled in text boxes. The physiographic province that each site lies in is designated by a separate shape: ▲ = Blue Ridge, ● = Piedmont, and ■ = Ridge and Valley.

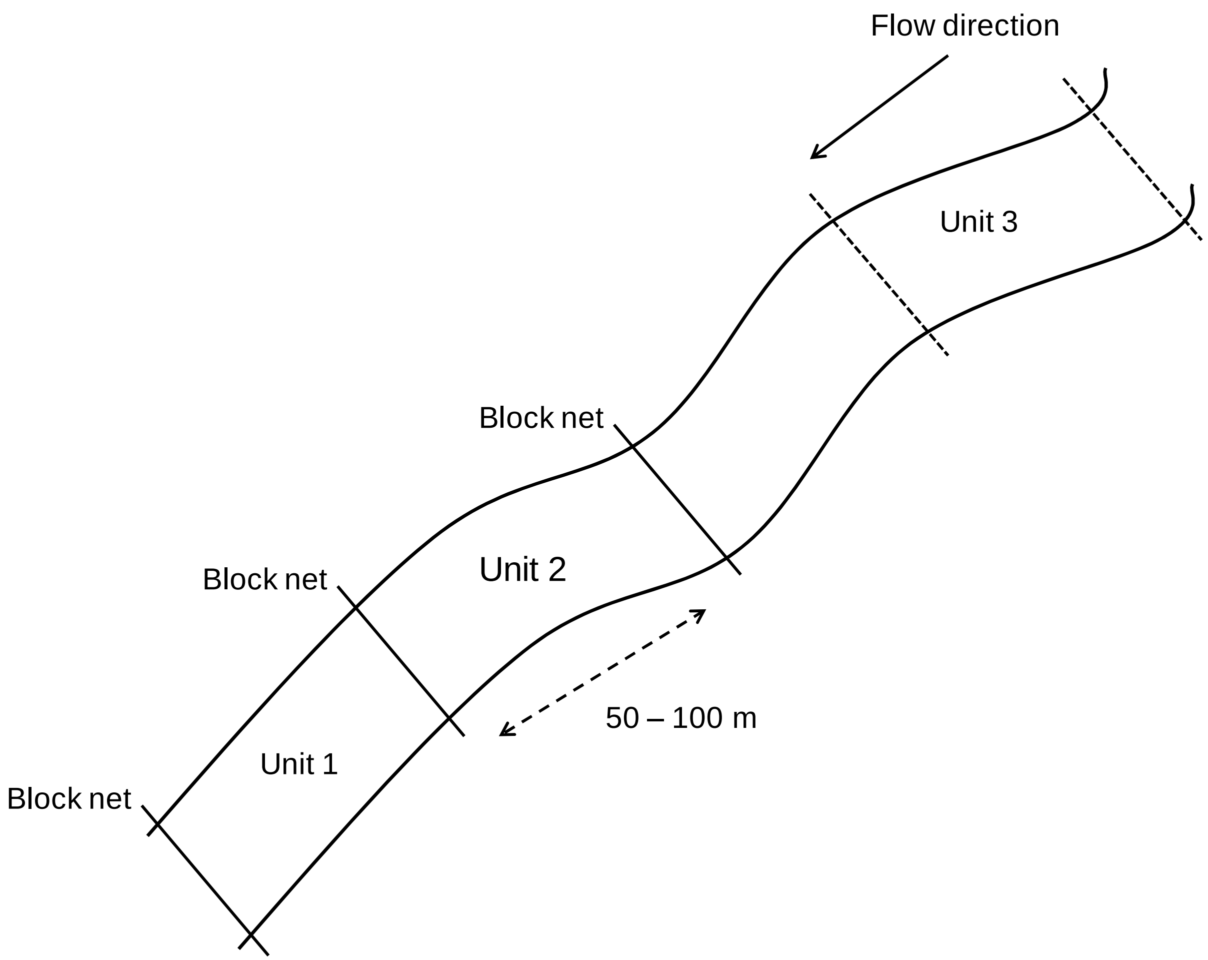


Figure 2. Representation of a sampling site designating units 1, 2 and 3, placement of block nets, and length of sampling units.

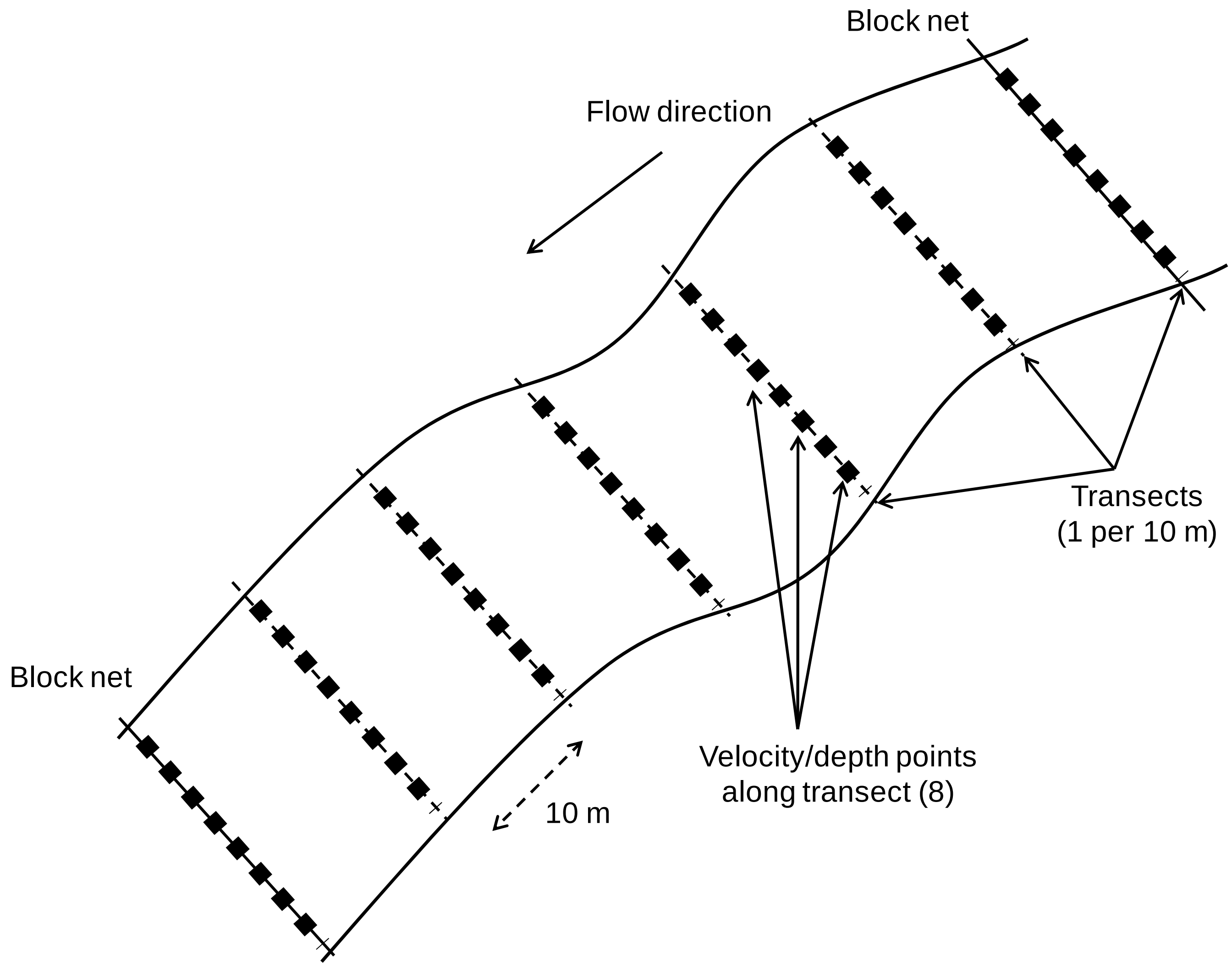
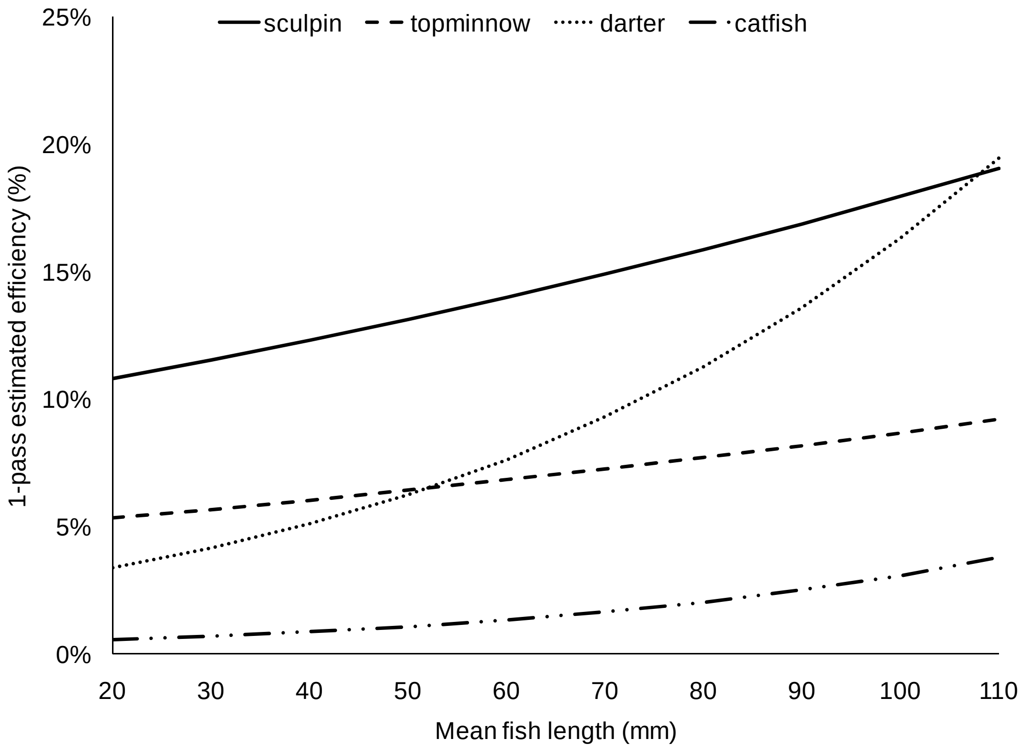


Figure 3. Representation of a unit with transects and points along transects for recording velocity and depth data.

a.



b.

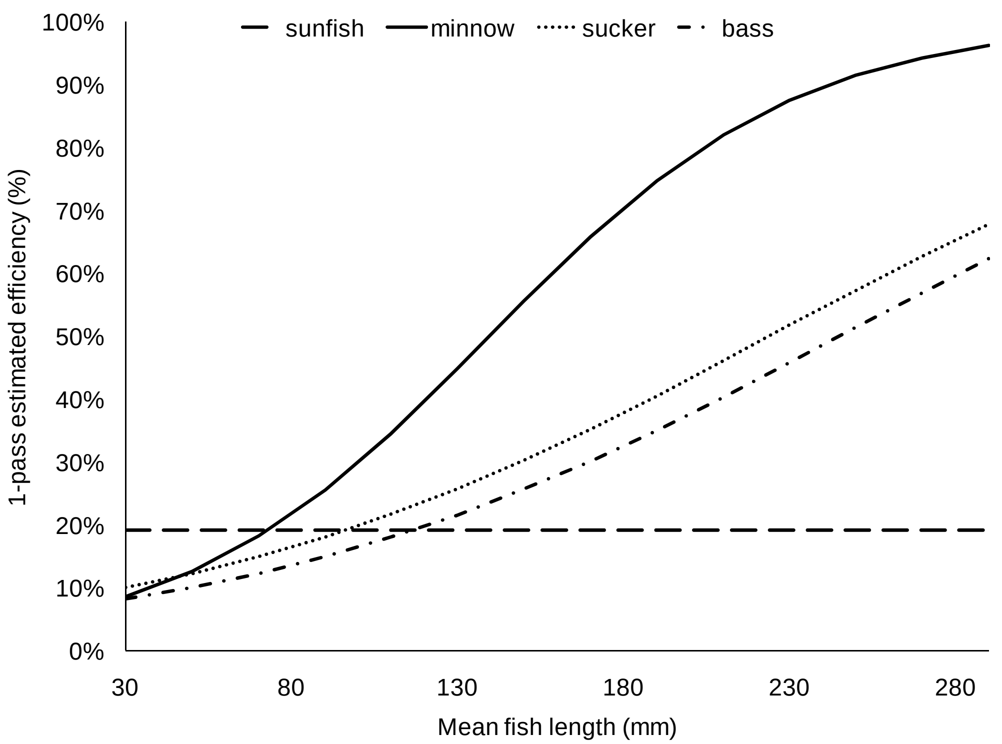


Figure 4. Estimated efficiency for 1-pass backpack electrofishing for a) catfish, darters, sculpin, and topminnows and b) bass, minnows, suckers, and sunfish versus mean fish length (mm) under average habitat conditions.

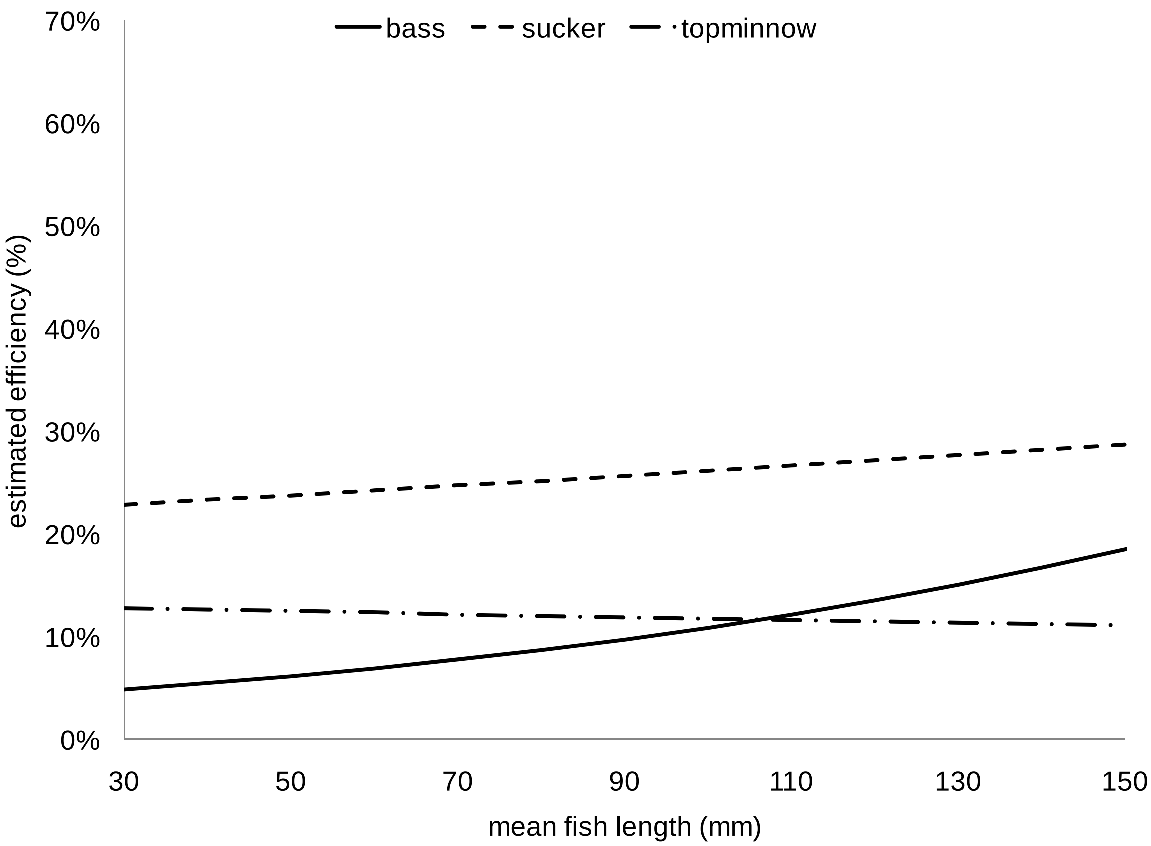


Figure 5. Estimated efficiency for seining with electrofishing for bass, suckers, and topminnows versus mean fish length (mm) under average habitat conditions.

a.

b.

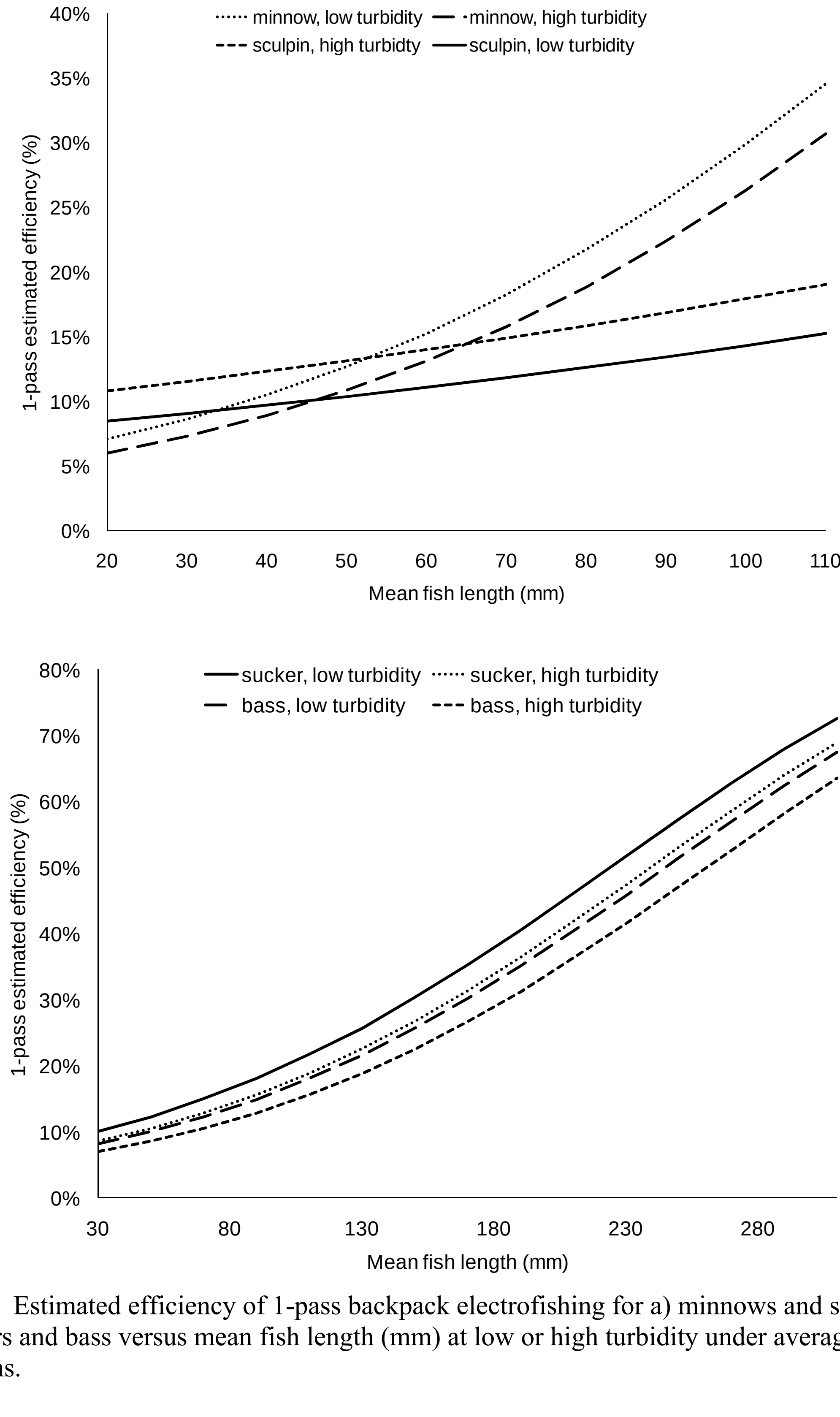


Figure 6. Estimated efficiency of 1-pass backpack electrofishing for a) minnows and sculpin and b) suckers and bass versus mean fish length (mm) at low or high turbidity under average habitat conditions.

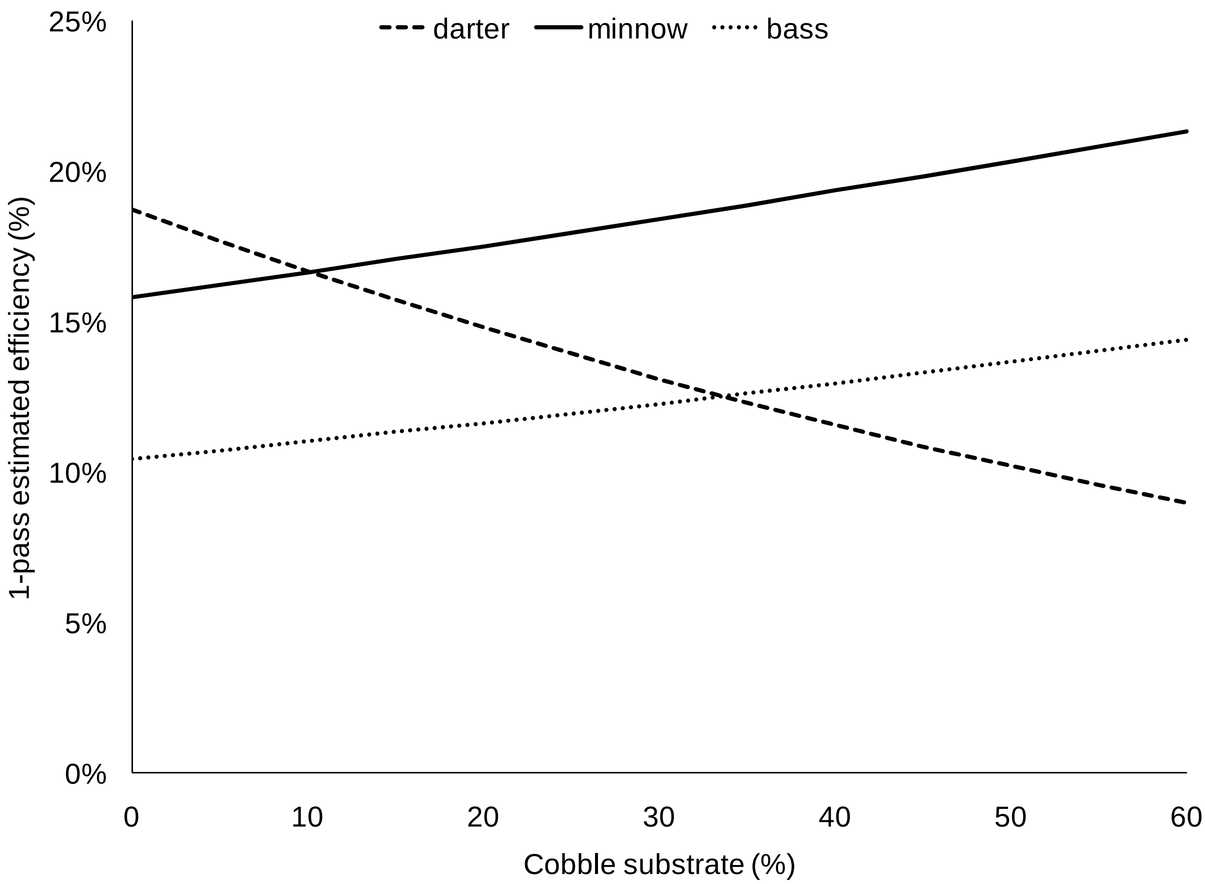


Figure 7. Estimated efficiency under average habitat conditions for seining with backpack electrofishing with respect to fish species group and percent cobble substrate.

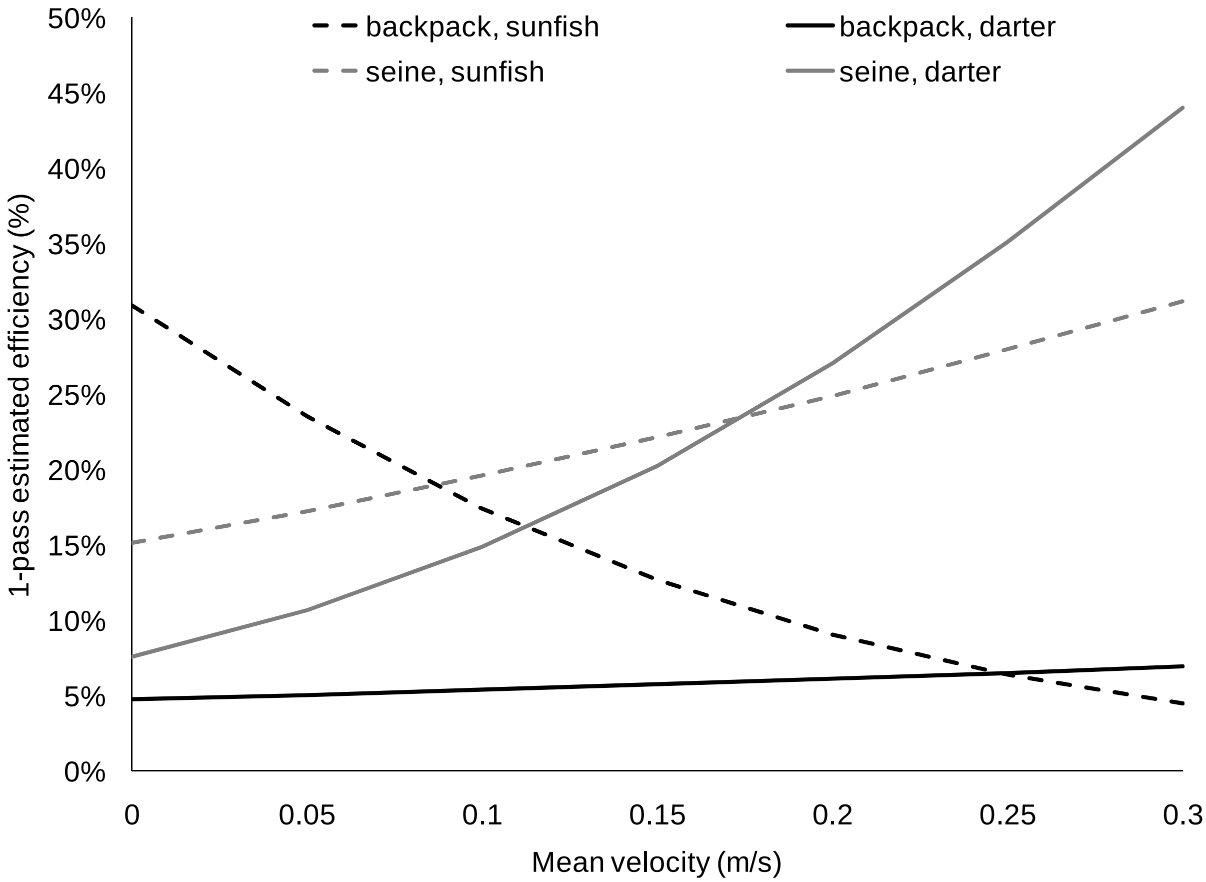


Figure 8. Comparison of estimated efficiency by sampling method under average habitat conditions for darter and sunfish species groups and mean velocity (m/s).

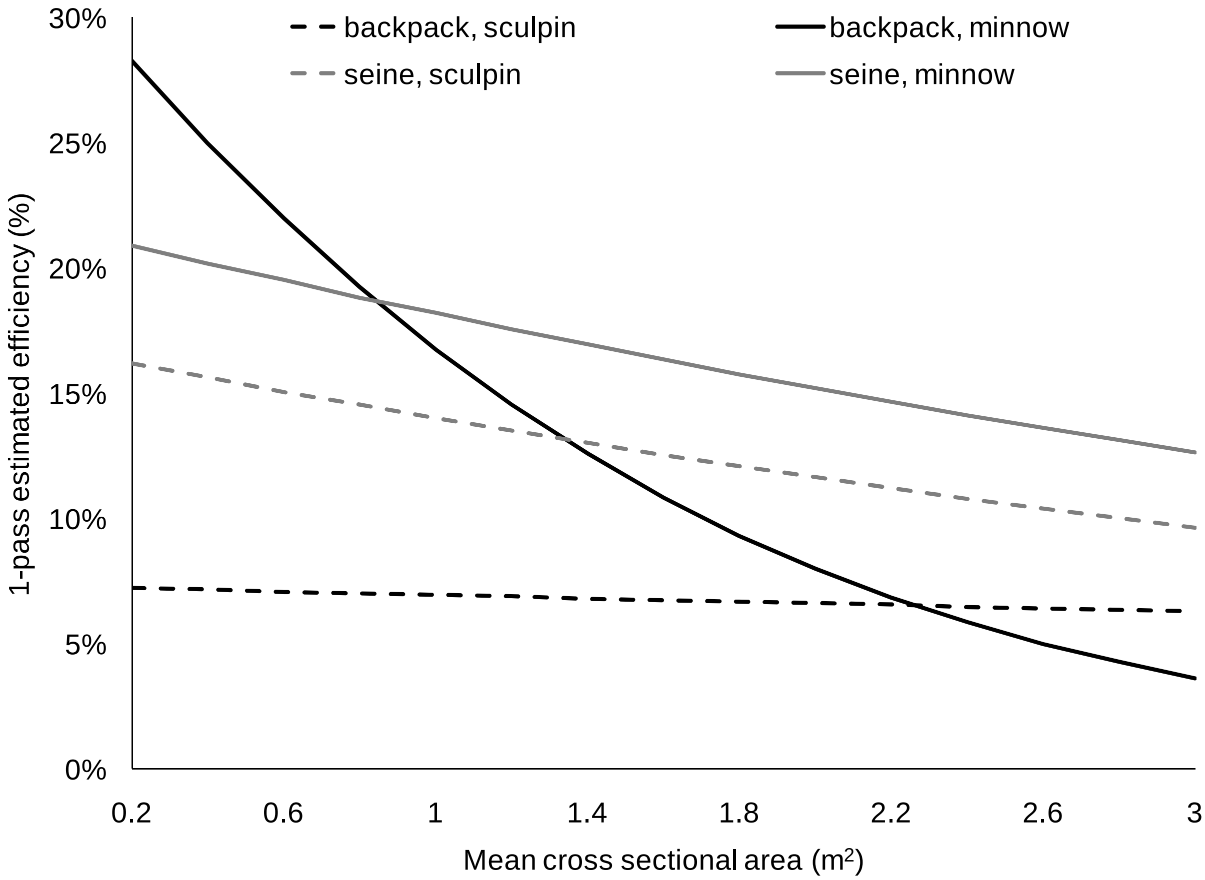
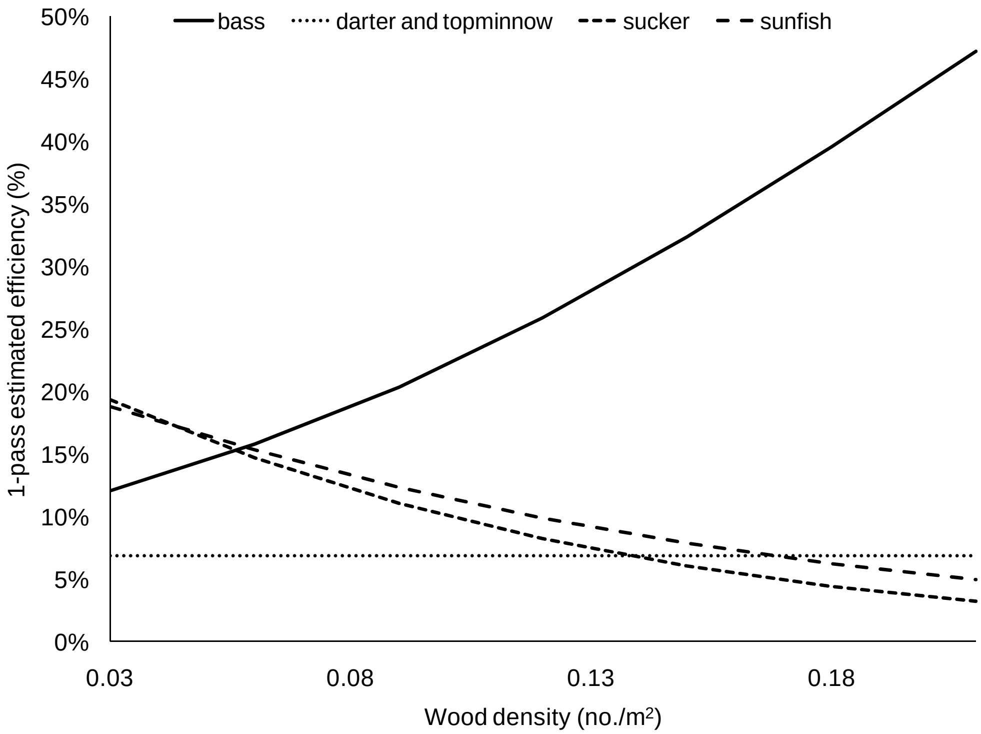


Figure 9. Comparison of estimated efficiency by sampling method for minnow and sculpin species groups and mean cross-sectional area (m^2).

a.



b.

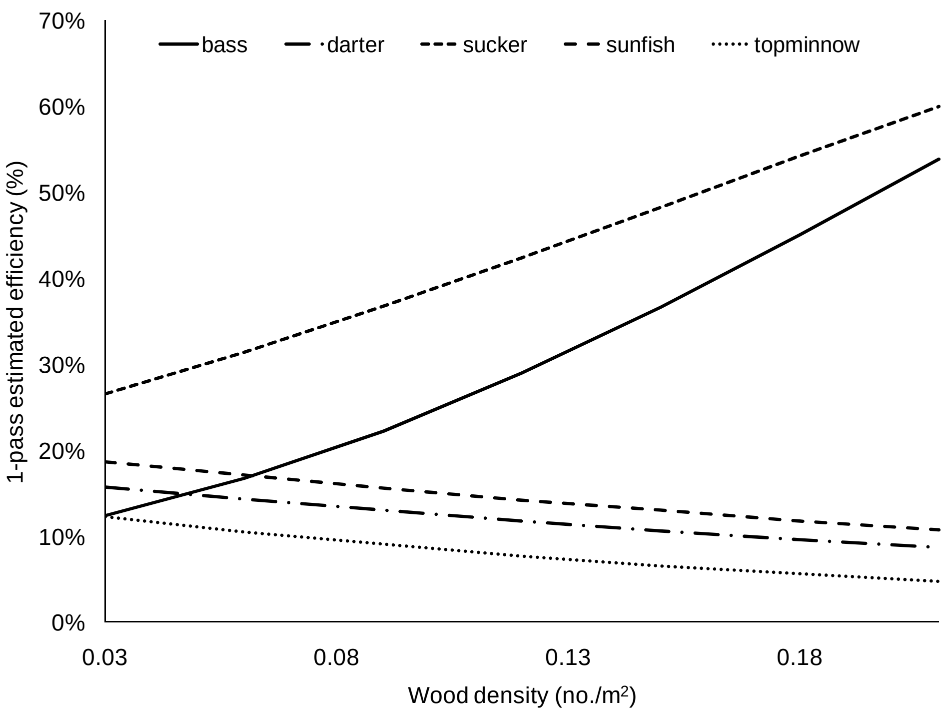


Figure 10. Estimated efficiency for fish species groups versus wood density (no./m^2) for a) 1-pass backpack electrofishing and b) seining with electrofishing.

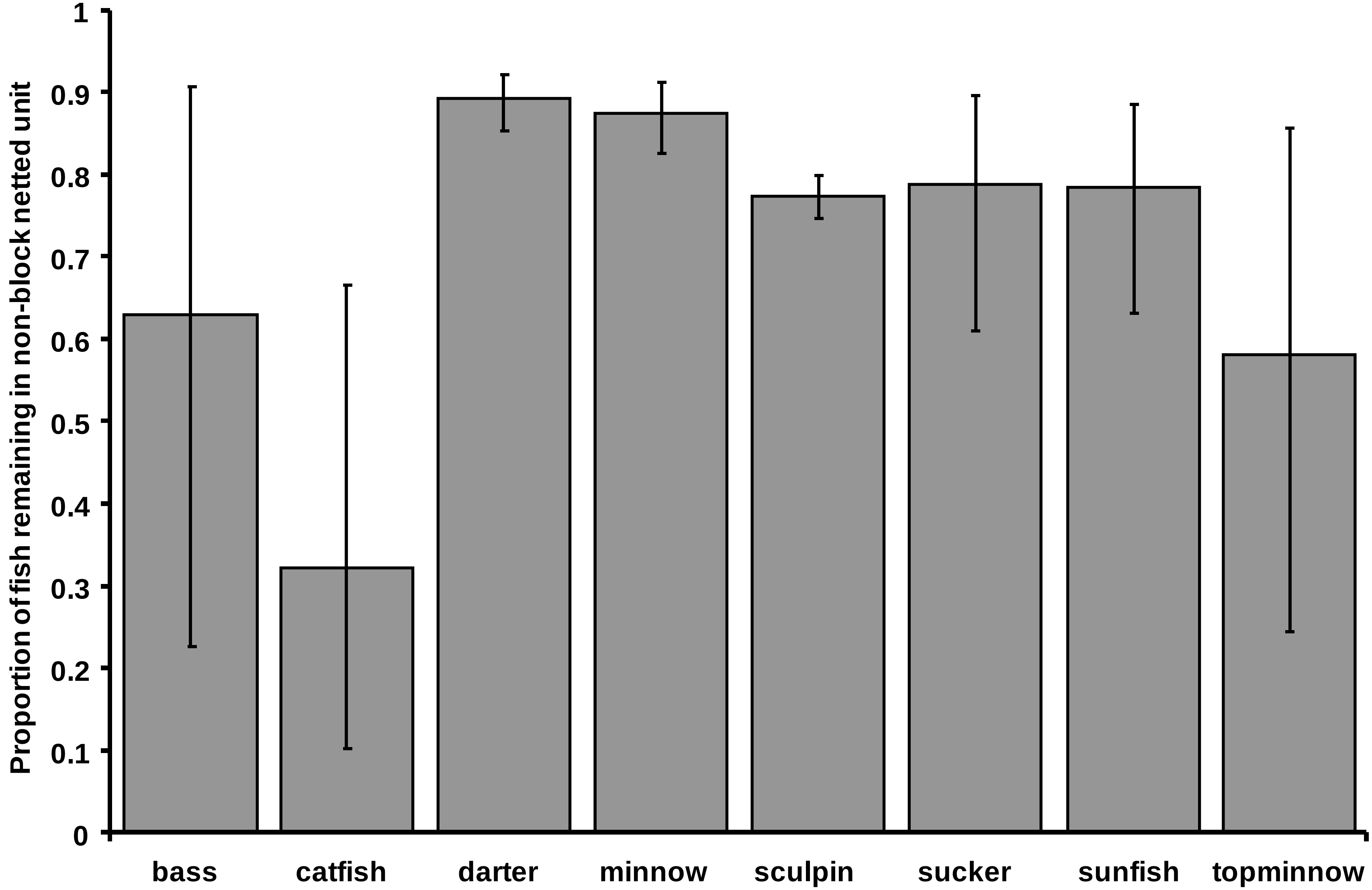
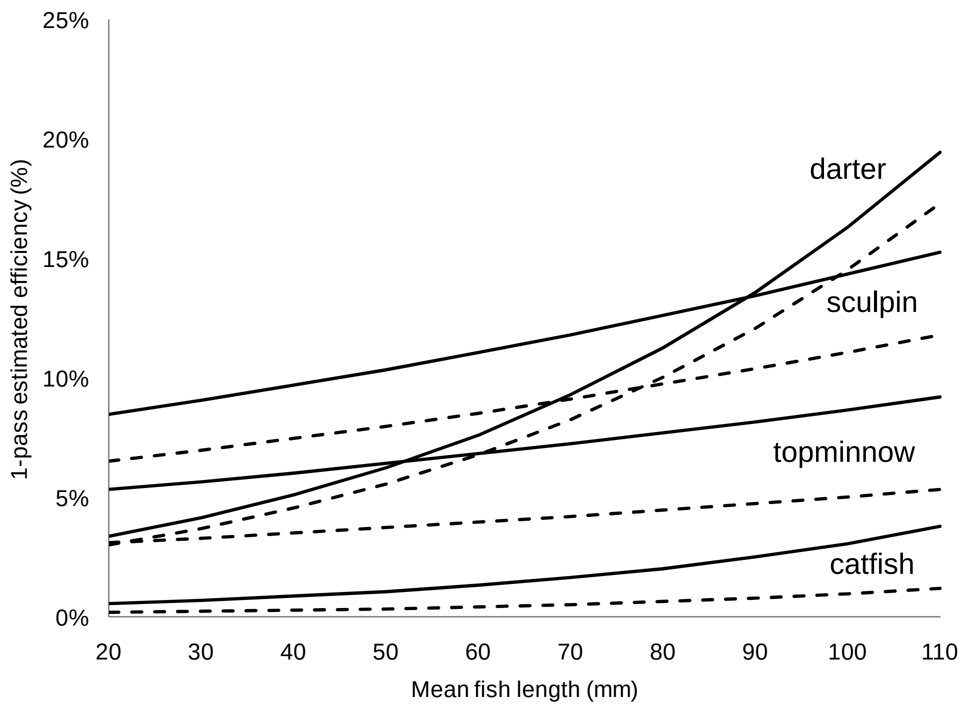


Figure 11. Estimates and standard errors of proportion of fish remaining in non-block netted sampling unit by species group for the best-approximating abundance model.

a.



b.

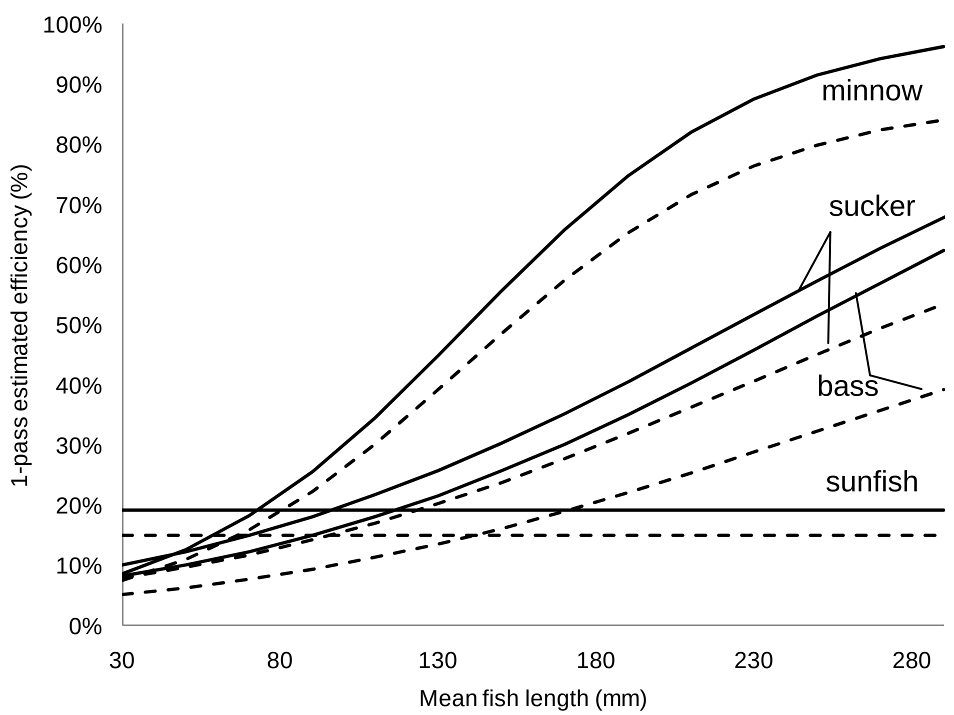


Figure 12. Estimated capture efficiencies conditional on block net installation (solid lines) and unconditional capture efficiencies (dashed line) for 1-pass backpack electrofishing for a) darters, sculpin, and catfish and b) bass, minnows, suckers, and sunfish versus mean fish length (mm) under average habitat conditions.

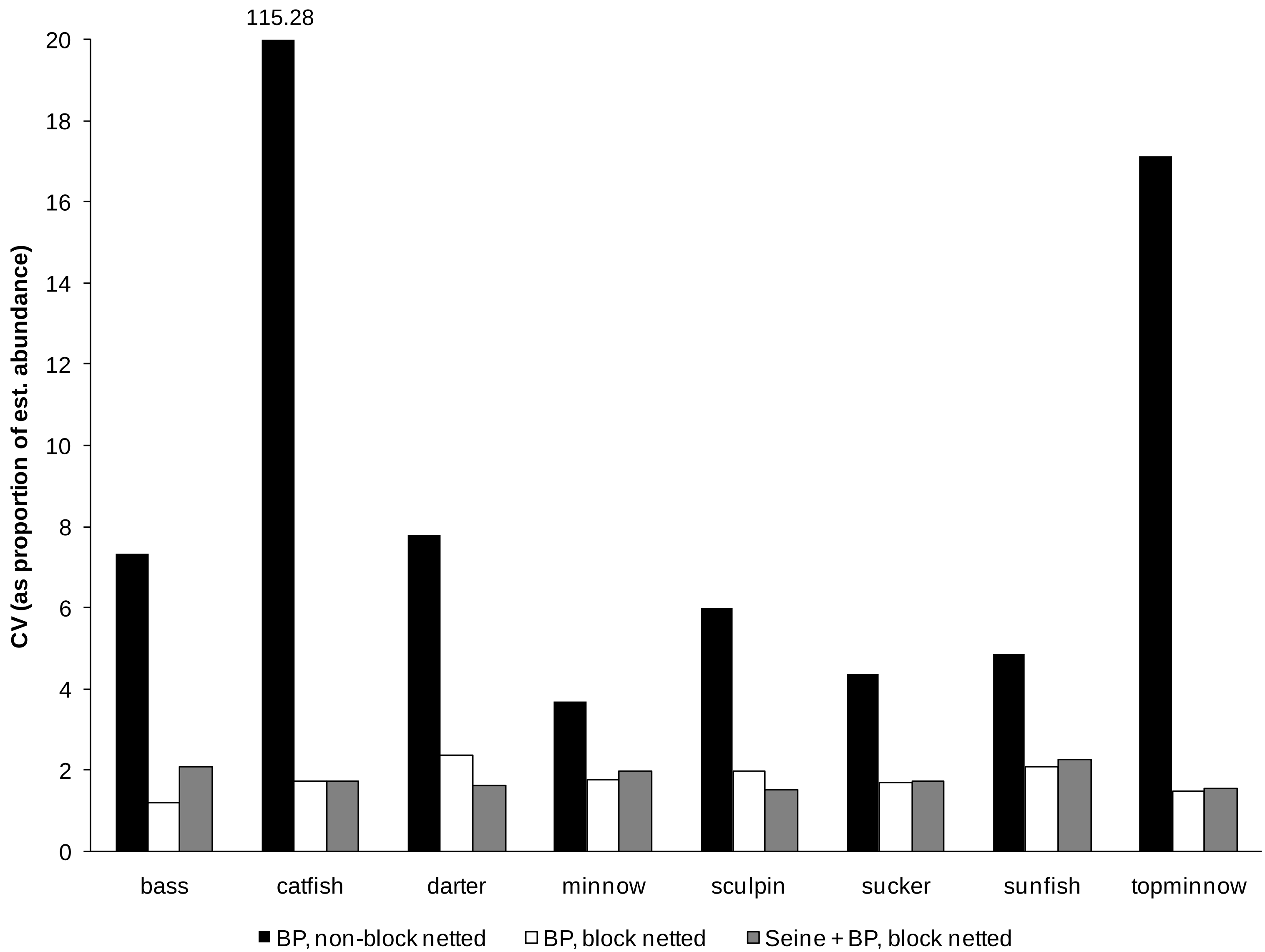


Figure 13. Coefficient of variation for 1-pass backpack electrofishing (BP) and seining with backpack electrofishing for block netted and non-block netted units by species group