3-22-2012

Heterogeneous detection probabilities for imperiled Missouri River fishes: implications for large-river monitoring programs

Joshua T. Schloesser
Kansas Cooperative Fish and Wildlife Research Unit, Division of Biology, Kansas State University & Present address: U.S. Fish and Wildlife Service, Ashland Fish and Wildlife Conservation Office, joshua_schloesser@fws.gov

Craig P. Paukert
U.S. Geological Survey, Kansas Cooperative Fish and Wildlife Research Unit, Division of Biology, Kansas State University & Present address: U.S. Geological Survey, Missouri Cooperative Fish and Wildlife Research Unit, Department of Fisheries and Wildlife Sciences, University of Missouri

Wyatt J. Doyle
U.S. Fish and Wildlife Service, Columbia Fish and Wildlife Conservation Office

Tracy D. Hill
U.S. Fish and Wildlife Service, Columbia Fish and Wildlife Conservation Office

Kirk D. Steffensen
Nebraska Game and Parks Commission

next page for additional authors

Schloesser, Joshua T.; Paukert, Craig P.; Doyle, Wyatt J.; Hill, Tracy D.; Steffensen, Kirk D.; and Travnichek, Vince H., "Heterogeneous detection probabilities for imperiled Missouri River fishes: implications for large-river monitoring programs" (2012). *USGS Staff -- Published Research*. 1061.
http://digitalcommons.unl.edu/usgsstaffpub/1061

Authors
Joshua T. Schloesser, Craig P. Paukert, Wyatt J. Doyle, Tracy D. Hill, Kirk D. Steffensen, and Vince H. Travnichek

Vol. 16: 211–224, 2012
doi: 10.3354/esr00399

ENDANGERED SPECIES RESEARCH
Endang Species Res

Contribution to the Theme Section 'Endangered river fish: threats and conservation options'

Heterogeneous detection probabilities for imperiled Missouri River fishes: implications for large-river monitoring programs

Joshua T. Schloesser[1,6,*], Craig P. Paukert[2,7], Wyatt J. Doyle[3], Tracy D. Hill[3], Kirk D. Steffensen[4], Vince H. Travnichek[5]

[1]Kansas Cooperative Fish and Wildlife Research Unit, Division of Biology, Kansas State University, Manhattan, Kansas 66506, USA
[2]U.S. Geological Survey, Kansas Cooperative Fish and Wildlife Research Unit, Division of Biology, Kansas State University, Manhattan, Kansas 66506, USA
[3]U.S. Fish and Wildlife Service, Columbia Fish and Wildlife Conservation Office, Columbia, Missouri 65203, USA
[4]Nebraska Game and Parks Commission, Lincoln, Nebraska 68503, USA
[5]Missouri Department of Conservation, St. Joseph, Missouri 64507, USA

[6]*Present address:* U.S. Fish and Wildlife Service, Ashland Fish and Wildlife Conservation Office, Ashland, Wisconsin 54806, USA

[7]*Present address*: U.S. Geological Survey, Missouri Cooperative Fish and Wildlife Research Unit, Department of Fisheries and Wildlife Sciences, University of Missouri, Columbia, Missouri 65211, USA

ABSTRACT: Occupancy modeling was used to determine (1) if detection probabilities (p) for 7 regionally imperiled Missouri River fishes (*Scaphirhynchus albus*, *Scaphirhynchus platorynchus*, *Cycleptus elongatus*, *Sander canadensis*, *Macrhybopsis aestivalis*, *Macrhybopsis gelida*, and *Macrhybopsis meeki*) differed among gear types (i.e. stationary gill nets, drifted trammel nets, and otter trawls), and (2) how detection probabilities were affected by habitat (i.e. pool, bar, and open water), longitudinal position (five 189 to 367 rkm long segments), sampling year (2003 to 2006), and season (July 1 to October 30 and October 31 to June 30). Adult, large-bodied fishes were best detected with gill nets (p: 0.02–0.74), but most juvenile large-bodied and all small-bodied species were best detected with otter trawls (p: 0.02–0.58). Trammel nets may be a redundant sampling gear for imperiled fishes in the lower Missouri River because most species had greater detection probabilities with gill nets or otter trawls. Detection probabilities varied with river segment for *S. platorynchus*, *C. elongatus*, and all small-bodied fishes, suggesting that changes in habitat influenced gear efficiency or abundance changes among river segments. Detection probabilities varied by habitat for adult *S. albus* and *S. canadensis*, year for juvenile *S. albus*, *C. elongatus*, and *S. canadensis*, and season for adult *S. albus*. Concentrating sampling effort on gears with the greatest detection probabilities may increase species detections to better monitor a population's response to environmental change and the effects of management actions on large-river fishes.

KEY WORDS: Detection probabilities · Large rivers · Missouri River · Gear evaluation

INTRODUCTION

Identifying and accounting for variation in detection probabilities among sampling gears and sampling occasions in a large-scale monitoring program is critical to effectively determine the population status or trends of rare animals (Yoccoz et al. 2001). Detection probabilities are defined as the probability of at least 1 individual of a species being detected during a survey at any particular site, given the spe-

*Email: joshua_schloesser@fws.gov

cies is present at that site (MacKenzie et al. 2002). In ecological studies, species are detected, present but not detected, or absent. The failure to account for imperfect detection probabilities when the species is actually present, but not detected, may confound any inference about changes in a species status and inhibit the ability of a fishery manager to make informed management decisions (MacKenzie et al. 2002, 2006, MacKenzie 2005). This is especially true for long-term assessments that may cover large areas with multiple sampling methods, where detectability varies by gear type, sampling location, and other spatial or temporal variables.

Gear selection is 1 aspect of a fishery monitoring program influencing the ability to efficiently collect the information needed to make informed management decisions for imperiled river fishes. Large-river fish assemblages, including those in the Missouri River (USA), are typically sampled with multiple gear types (Casselman et al. 1990, Lapointe et al. 2006, Doyle et al. 2008, Guy et al. 2009b) because detection probabilities may vary for any species among gear types due to bias for different sizes of fish and different gear efficiencies among habitats (Quist et al. 2006). However, multi-gear assessments make monitoring more difficult because of gear bias and a loss of power in statistical analyses as effort is divided among multiple gears (Beamesderfer & Rieman 1988, Berry et al. 2005). Sampling with gears that have lower species detection probabilities may result in misallocation of effort and inadequate statistical inference for monitoring programs (Paukert 2004, Noble et al. 2007).

Species detection probabilities may also be influenced by spatial and temporal factors such as longitudinal position in the river, habitat, season, or year. For example, some native large-river fishes have highly specialized ecological needs inherently leading to a patchy distribution, as specific habitat and environmental conditions are limited (Ridenour et al. 2009). Additionally, the variability in environmental conditions of large rivers, such as high water, complex river morphologies, variable substrates, or temperature changes, can alter the ability to detect a species seasonally or annually. Understanding these types of spatial and temporal factors that create differences in detection probabilities is essential in a large-scale monitoring program for rare fishes in order to draw proper inferences about the study population.

Monitoring programs on the Missouri River, use multiple gears to sample the entire fish community, including several species in need of conservation, to determine long-term trends in abundance and how these trends relate to predictable and stochastic environmental changes and management actions (Quist et al. 2004). We chose to focus the present paper on 7 species of conservation concern: *Scaphirhynchus albus*, *S. platorynchus*, *Cycleptus elongatus*, *Sander canadensis*, *Macrhybopsis aestivalis*, *M. gelida*, and *M. meeki* (see Table 1 for threat status). These fishes were primarily collected in gill nets, trammel nets, and otter trawls. At present, variable conditions in the Missouri River and low catches of rare and endangered species result in low statistical power to detect trends using abundance indices or responses to management actions (SEI 2004, Bryan et al. 2009).

The objective of this research was to (1) determine the effect of gear type on detection probabilities for gill nets, trammel nets, and otter trawls used to sample 7 imperiled Lower Missouri River fishes and (2) determine the effects of river segment, habitat, year, and season on probability of detection among sample gears. There is a dire need to estimate detection probabilities in order to properly allocate sampling effort to achieve the specific objectives of a monitoring program for rare fishes in large rivers.

MATERIALS AND METHODS

Study area

Fish sampling was conducted as part of the federally (USA) endangered pallid sturgeon *Scaphirhynchus albus* population monitoring and assessment program on the Missouri River (Drobish 2008, Wanner et al. 2010). The study area includes the lower 1212 rkm of the Missouri River from the Lower Ponca Bend at Ponca, Nebraska (river kilometer [rkm] 1212), to the confluence with the Mississippi River (rkm 0) at St. Louis, Missouri (Fig. 1). This area is completely channelized with rock dike structures that maintain a navigational channel depth of at least 2.7 m (National Research Council 2002, Galat et al. 2005). Dikes are static engineered rock structures that direct current towards the thalweg, but also provide low-velocity habitats immediately behind them (Ridenour et al. 2009). The physical features of the river change substantially throughout the study area. The Missouri River near Ponca, is approximately 200 m wide and dikes are spaced every 150 to 300 m, but the river near St. Louis, (rkm 0), is 400 to 500 m wide, with dikes spaced every 250 to 400 m. The length of the dike extending from the bank also increases from 30–50 m near Ponca, to 75–150 m

Table 1. Conservation status of 7 Missouri River species

Species	Status	Source
Pallid sturgeon *Scaphirhynchus albus*	Endangered	IUCN (2011)
Shovelnose sturgeon *Scaphiryhnchus platorynchus*	Vulnerable	IUCN (2011)
Blue sucker *Cycleptus elongatus*	Lower Risk/Near Threatened	IUCN (2011)
Sauger *Sander canadensis*	Secure/Decreasing	Galat et al. (2005)
Speckled chub *Macrhybopsis aestivalis*	Vulnerable/Apparently secure	Galat et al. (2005)
Sturgeon chub *Macrhybopsis gelida*	Vulnerable	IUCN (2011)
Sicklefin chub *Macrhybopsis meeki*	Lower Risk/Near Threatened	IUCN (2011)

near St. Louis. Lower Missouri River flows are partially controlled through Gavin's Point Dam at Yankton, South Dakota (rkm 1305), which has altered discharge to be more constant for navigation (Hesse & Mestl 1993). Fluctuations in gauge height may still change by 4.3 m near Omaha, Nebraska (rkm 991), and up to 8.2 m near St. Charles, Missouri (rkm 44.4), with significant rain events (USGS 2009; period of record January 2000 to December 2006).

Data collection

The channelized lower Missouri River was divided into 5 segments based on changes in physical attributes (e.g. water temperature, turbidity, tributary influences, natural hydrograph, and flow fluctuations) and river modifications (Drobish 2008). Segments 8, 9, 10, 13, and 14 encompassed rkm 1212–958, 958–591, 591–402, 402–209, and 209–0, respectively, and followed segments defined by Drobish (2008). Segment borders occurred at the confluence of major tributaries because these substantially increased total discharge and size of the adjacent downstream segment (Fig. 1). Sampling occurred during 2 seasons each year: the cold-water season started when river water temperatures declined below 13°C (generally starting around October 31) and lasted until June 30 (Doyle et al. 2008, Wildhaber et al. 2011). The warm-water season extended from July 1 until water temperatures dropped below 13°C (generally ending October 30; Wildhaber et al. 2011). A water temperature of 13°C defines sampling seasons, to minimize mortality of the Endangered *Scaphirhynchus albus* by restricting gill nets to the cold-water season only (USFWS 2005). For clarity, we will reference the cold-water season as starting on October 31 and the warm-water season as ending on October 30. Sampling occurred from January 2003 to October 2006. Sample years were defined starting with the cold-water season and ending with the warm-water season (e.g. start of the cold-water season on October 31, 2004 to the end of the warm-water season on October 30, 2005 was sample year 2005).

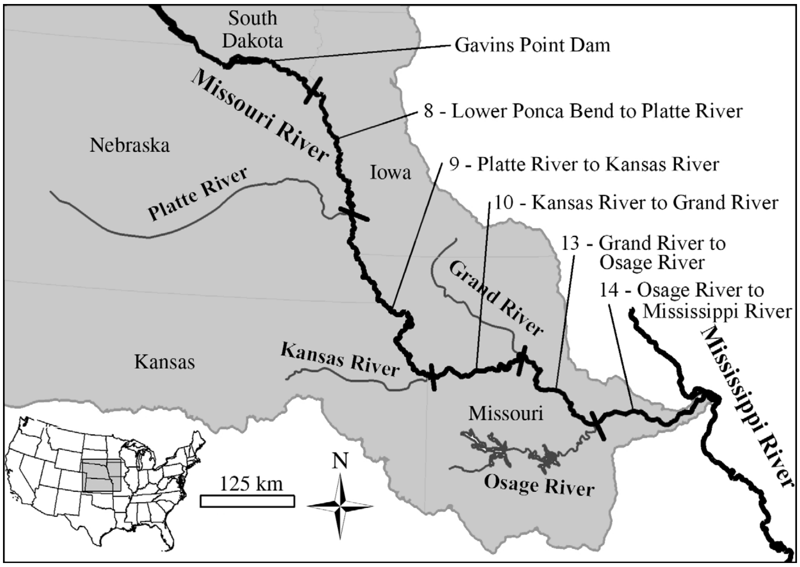

Fig. 1. Lower Missouri River (USA) with Sampling Segments 8 (river kilometer [rkm] 1212 to 958), 9 (rkm 958 to 591), 10 (rkm 591 to 402), 13 (rkm 402 to 209), and 14 (rkm 209 to 0) labeled from the Lower Ponca Bend at Ponca, Nebraska (rkm 1212) to the confluence of the Mississippi River (rkm 0) at St. Louis, Missouri

The entire study area was divided into river bends (n = 346), and at least 21% of the bends were randomly selected and sampled each year with a suite of gears during both seasons (Table 2). Over 4 yr, a total of 227 different river bends (hereafter referred to as sites) were sampled (range: 73 to 157 yr^{-1}). A site began

Table 2. Number of sites sampled in the Missouri River in 5 river segments during 2 seasons (cold-water season from October 31 to June 30; warm-water season from July 1 to October 30), and in 4 years (2003 to 2006). The values for habitat represent the mean percentage of samples taken in a habitat type at any given site. No gill nets were used in the warm-water season. n: number of sites sampled; rkm: river kilometer

Variable	Gill net (n = 209)	Trammel net (n = 356)	Otter trawl (n = 417)
Segment			
8 (rkm 1212.0–958.4)	32	76	78
9 (rkm 958.4–591.4)	61	104	127
10 (rkm 591.4–402.3)	11	20	20
13 (rkm 402.3–209.2)	64	67	92
14 (rkm 209.2–0.0)	41	89	100
Habitat (% of samples)			
Pool	51.9	0.5	5.5
Bar	9.5	16.3	16.2
Open water	38.5	83.1	78.2
Season			
Cold-water	209	170	214
Warm-water		186	203
Year			
2003	31	23	70
2004	35	57	66
2005	54	137	137
2006	89	139	144

at the upstream origin of the channel crossover, included the adjacent downstream outside/inside bend complex, and ended at the start of the next channel crossover (Drobish 2008, Doyle et al. 2008). At all sites, the sampling gear was deployed at a minimum of 8 locations to provide a consistent level of effort among sites during each season (Wanner et al. 2007, Drobish 2008). Sample locations were allocated throughout the site according to a macro- and meso-habitat classification system (Drobish 2008). A minimum of 2 sub-samples were required for specific macro- and meso-habitat combinations, but in some instances an exception had to be made if river conditions jeopardized crew safety. Within meso-habitats, the specific gear deployment location was identified as pool, bar, or open-water habitat as defined by Ridenour et al. (2009; collectively referred to as habitat hereafter). Pools were relatively small areas (50 to 125 m wide and long) defined as the area immediately downstream from a dike or other obstruction that formed a scour hole >1.2 m deep. Bars were the terrestrial/aquatic interface areas associated with a sand bar or shallow bankline where sediments deposit and water was <1.2 m deep. Open water was considered the area >1.2 m and not associated with a dike or its scour hole. A depth criterion of 1.2 m was used to define habitats because water <1.2 m deep met the depth criterion for shallow-water habitat as defined in the 2000 Biological Opinion on Missouri River management.

Stationary gill nets, drifted trammel nets, and towed otter trawls were deployed at each site during the cold-water season. Cold-water season gill nets were generally deployed during late October to April, and trammel nets and otter trawls during April to June. Warm-water season sampling used only trammel nets and otter trawls. Gill nets were not used in the warm-water season to minimize fish mortality during this period of higher water temperatures (Doyle et al. 2008). Within a sample site, only 1 gear type was deployed at a time to avoid depleting or affecting the catch in a different gear.

Each gear type was deployed in all habitat types (pool, bar, and open water), but primarily in 1 or 2 habitat types (Table 2). Gill nets were primarily set in pool and open-water habitats parallel to the flow overnight for a minimum of 12 h and maximum of 24 h. A gill net (61 m long and 2.4 m high) consisted of eight 7.6 m horizontal panels comprised of 3.8, 5.1, 7.6, and 10.2 cm bar multi-filament mesh, with panels organized in ascending order on each half of the net. Trammel nets were oriented perpendicular to the current and drifted downstream primarily in bar and open-water habitats (Doyle et al. 2008). Nets were 38.1 m long with a 2.4 m center wall of 2.5 cm multi-filament nylon mesh and a 1.8 m outer wall of 20.3 cm multi-filament nylon mesh on both sides. Otter trawls were 4.9 m wide, 0.9 m high, 7.6 m long, with 0.64 cm inner bar mesh and 3.8 cm outer chafing mesh (Doyle et al. 2008). Trawls were towed downstream just faster than the current, primarily in bar and open-water habitats. Trammel net and otter trawl samples had a minimum 75 m drift with a target of 300 m, which was usually limited by snags or dike spacing. All fishes from each gear were counted and measured (mm) for total length, and in the case of *Scaphirhynchus* spp., also for fork length.

Data analyses

Four of the study species were considered large-bodied fishes and divided into adults and juveniles based on length at maturity (Becker 1983, Robison & Buchanan 1988, Keenlyne & Jenkins 1993, Jenkins & Burkhead 1994, Pflieger 1997). Large-bodied fishes included *Scaphirhynchus albus* (adults were ≥550 mm fork length), *S. platorynchus* (adults were ≥550 mm fork length), *Cycleptus elongatus* (adults were

≥500 mm total length), and *Sander canadensis* (adults were ≥250 mm total length). Analyses were conducted separately by maturity class because susceptibility to capture may be influenced by size and behavioral traits (Hamley 1975, Kjelson & Johnson 1978, Argent & Kimmel 2005). Small-bodied species (which were not divided into maturity classes) were *Macrhybopsis aestivalis*, *M. gelida*, and *M. meeki*.

The program PRESENCE (Hines 2006) was used to estimate probabilities of detection (p) for each species by maturity class. Detection probabilities represent the probability of detecting a species, given that it is present at the sampling location. Species presence at a site (i.e. river bend) is known only if that species was detected during sampling. If the species was not detected, it may be present, but not detected, or truly absent from the site. Program PRESENCE calculates detection probabilities from sites where the species was known to be present (i.e. detected at least once). Sites where the species was not detected cannot be used to estimate detection probabilities because the species may not even be available for capture within the site (which would result in an infinite number of samples with zero chance of detection). Multiple samples at each site are required where the species was either detected or not detected in each sample. Sites are assumed to be closed to immigration or emigration at the species level. This means that individuals of the species may enter or leave the study area, but the state of species presence (either being present or absent) at the site must remain unchanged. The program PRESENCE also estimates an occupancy (ψ) parameter, which is the probability of a site containing at least 1 individual of a species, referred to as occupancy (MacKenzie et al. 2002). The occupancy parameter is used to assign a probability that a site is actually occupied, even though the species may not have been detected there due to imperfect detection probabilities. Since probability of detection was our only interest, the occupancy parameter was modeled as a constant (i.e. not allowed to vary as a function of a site level variable) in all candidate models to minimize model variation associated with parameters that were not of interest (MacKenzie et al. 2002, Longoria & Weckerly 2007).

A 2-step approach was used to fit candidate models, similar to that used by Washburn et al. (2004). First, detection probability was modeled as a function of gear type and as a constant (i.e. a model with no gear type effect) to assess if gear type had an effect on detection probabilities. If Akaike weights indicated gear type had some weight of evidence it was the best model, the second step was to test for an interaction between gear type and the spatial and temporal variables we selected (segment, habitat, year, and season). If gear type was not a strong factor affecting detection probabilities for a given species, a suite of 5 models was run; a constant model for no effect on detection probability and 1 model each for segment, habitat, year, and season. We selected this approach because gear type is known to be highly influential in fish captures (Sheehan & Rasmussen 1999), and our primary objective was to determine the effect of gear type on detection probabilities. We selected segment and habitat as spatial variables to account for the change in size of the river over the 1212 rkm study site and because each habitat type has its own unique functions and set of conditions in the river. Year and sample season were chosen to represent broad-scale discharge regimes and seasonal changes in fish behavior.

Since small-bodied fishes were generally not captured in the larger mesh of gill nets and trammel nets, only catches from otter trawls were used to model detection probabilities. This resulted in a slightly different model structure for small-bodied fishes than was used for large-bodied fishes. Five candidate models were selected for small-bodied fishes; the first model was a constant model to represent the only gear type, otter trawl. The remaining 4 candidate models were the interaction models between otter trawls and segment, habitat, season, and year. Interaction models among the spatial and temporal variables were not run due to concerns of over-parameterization with rare species that may have few detections during sampling.

Models were run under a single-season framework as described by MacKenzie et al. (2002). We chose this model structure because sites were not re-sampled consistently each year and season (i.e. sites were randomly selected each year and season) and a multi-season model was not a suitable fit to the sampling structure. Single-season models assume that sites are closed to immigration or emigration at the species level. Individuals may move in or out of the study area but the status of the species (either present or absent at the site) must remain the same throughout the survey period. With our sampling protocol, each gear type may have been fished at the same site (river bend) during the same season and year, but may have been fished months apart. We felt that analyzing gear type as a sample level variable for each site, and assuming presence status did not change over several months, would violate the closure assumption, especially for large and mobile fishes. To account for this, gear type was modeled as a site-level variable where each site (river bend) was broken into 3 independent periods, 1 for each gear type and the associated fish catch, and assigned a unique number so that species presence was considered unknown and independent among gear types deployed at the same river bend. All the repeat samples from 1 gear type at any particular site (river bend) were conducted within 5 d of each other, and the short time period was assumed as site closure to changes in species presence (MacKenzie et al. 2002, 2006). The repeat samples from each gear type within a site were considered independent of each other because field crews deployed sampling gears according to the macro- and meso-habitat classification system described previously (Drobish 2008), which typically sampled different dike structures or sand bars with every deployment. Furthermore, gear types deployed during both seasons were treated independently between seasons to account for the lack of warm-water gill netting.

We ranked models using Akaike's information criterion (AIC) because there were a large number of sites relative to the number of explanatory variables being tested (Burnham & Anderson 2002, MacKenzie et al. 2006). The gear-type model in the first step of the analysis was tested for overdispersion with the Pearson statistic and a parametric bootstrap procedure as described in MacKenzie & Bailey (2004). The variance inflation factor (ĉ) was used to adjust standard errors due to lack of fit, as overdispersed data indicate the expected model structure is adequate, but the variance structure is inadequate (Burnham & Anderson 2002, MacKenzie et al. 2006). Because small-bodied fishes did not have a specific gear-type model, the model with the most parameters (i.e. segment) was tested for overdispersion. Candidate models were considered to be equally well supported by the data if the difference (Δ_i) between AIC or the quasi-Akaike information criterion (QAIC; AIC adjusted for lack of fit) values was ≤2 (Burnham & Anderson 2002). The number of modeled parameters (K) included 1 parameter for each detection probability variable plus 1 for the occupancy parameter (estimates were not reported). One additional parameter was added to K when QAIC was used for model selection (Burnham & Anderson 2002). The relative importance of each explanatory variable on detection probabilities was assessed using Akaike weights (w_i) and log-likelihood (log(L)).

RESULTS

A total of 69 342 fish of the 7 study species were collected from 982 sites (7675 gear deployments) at 227 river bends over the 4 yr study period (Table 2). The first-stage evaluation for the effect of gear type on detection probabilities indicated the gear-type model was better supported than a constant model (Table 3). The gear-type model was considered the better of the 2 models for all adult and juvenile large-bodied fishes, except juvenile *Scaphirhynchus albus* and *Cycleptus elongatus*. The juvenile *S. albus* gear-type model was considered to be equally well sup-

Table 3. Model rankings for the effect of gear type on detection probabilities (p) of large-bodied fishes in the Missouri River, from 2003 to 2006. Either Akaike's information criterion (AIC) or quasi-Akaike's information criterion (QAIC) was used to rank models based on the variance inflation factor (ĉ). The 2 competing models, gear type and constant (.), were considered parsimonious if the ΔAIC was <2. Number of modeled parameters (K), Akaike weights (w_i), and log-likelihood (log(L)) were used to determine the relative effect of gear type

Adults						Juveniles					
Model	ΔAIC/QAIC	K	w_i	log(L)	ĉ	Model	ΔAIC/QAIC	K	w_i	log(L)	ĉ
Scaphirhynchus albus											
p(Gear)	0.00	4	1.00	−281.1	<1	p(.)	0.00	2	0.69	−429.0	
p(.)	56.96	2	0.00	−311.5		p(Gear)	1.60	4	0.31	−427.8	<1
Scaphirhynchus platorynchus											
p(Gear)	0.00	4	1.00	−4478.8	<1	p(Gear)	0.00	4	1.00	−4882.0	<1
p(.)	922.04	2	0.00	−4941.8		p(.)	473.83	2	0.00	−5120.9	
Cycleptus elongatus											
p(Gear)	0.00	5	1.00	−3291.9	1.98	p(.)	0.00	2	0.85	−767.1	
p(.)	49.00	2	0.00	−3344.4		p(Gear)	3.48	5	0.15	−766.7	1.23
Sander canadensis											
p(Gear)	0.00	4	1.00	−1107.3	<1	p(Gear)	0.00	4	1.00	−275.1	<1
p(.)	302.08	2	0.00	−1260.3		p(.)	42.25	2	0.00	298.2	

ported as the constant model (ΔAIC of 1.60 with an Akaike weight of 0.31). Additionally, gear type for juvenile *C. elongatus* was considered because that model had a ΔQAIC of 3.48 with an Akaike weight of 0.15. For clarity and consistency, we chose to model detection probabilities as a function of gear type for juvenile *S. albus* and *C. elongatus* along with all the other species. The gear-type models for all large-bodied fishes generally fit the data well, with $\hat{c} < 1$ and only minor overdispersion in the case of adult and juvenile *C. elongatus* ($\hat{c}$ = 1.23 to 1.98) which was accounted for during model selection.

Probability of detection estimates from the gear-type model in the first-stage evaluation varied greatly among large-bodied species and even between adults and juveniles of the same species (estimates of p ranged from 0.0 to 0.74; Fig. 2). The probability of detecting the federally endangered *Scaphirhynchus albus* was <0.03 for all gear types. Adult *S. albus* were best detected with gill nets (p = 0.02), but all gear types had similar p for juvenile *S. albus*. *S. platorynchus* were approximately twice as likely to be detected with gill nets (p = 0.74 and 0.66 for adults and juveniles, respectively) than with trammel nets or otter trawls. Adult *Cycleptus elongatus* had similar detection probabilities with gill nets and trammel nets, as indicated by overlapping 95% confidence intervals. Probability of detection estimates for all gears were similar for juvenile *C. elongatus*, as would be expected when a constant model ranks higher than a gear-type model. Adult *Sander canadensis* had a probability of detection estimate 10-fold greater in gill nets (p = 0.18) than with the other 2 gear types. The probability of detection for juvenile *S. canadensis* was <0.01 with gill nets and trammel nets, and only 0.06 with otter trawls.

The second stage of the analysis assessed how spatial and temporal variables interacted with gear type. Gear type was analyzed as an interaction variable for all adult and juvenile large-bodied fishes because it contributed a minimum of 15% of the weight during the first-stage model selection analysis. Of the candidate set of gear-interaction

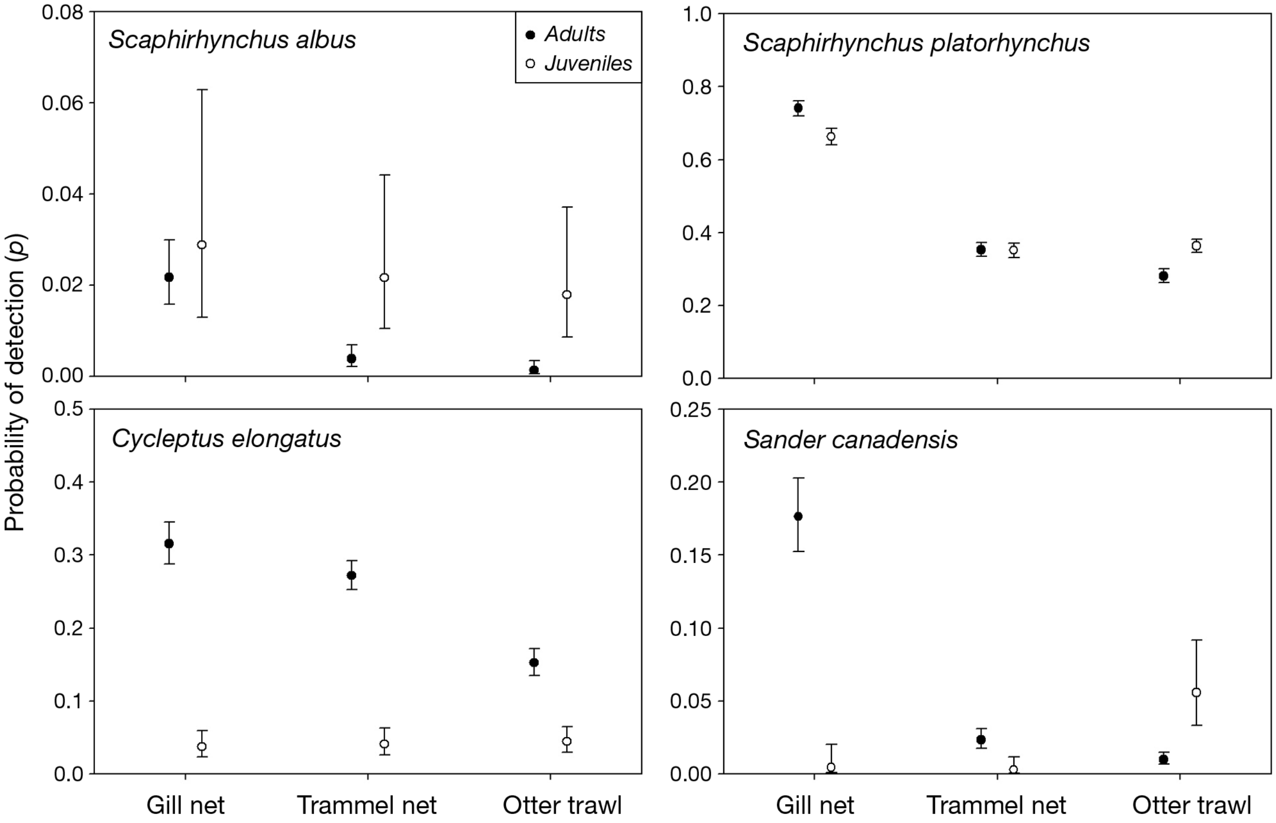

Fig. 2. Probability of detection (p) model estimates and 95% confidence intervals for the gear-type model during the first-stage evaluation to test for a gear-type effect on the probability of detecting large-bodied species. Adult lengths for *Scaphirhynchus albus* (≥550 mm), *S. platorynchus* (≥550 mm), *Cycleptus elongatus* (≥500 mm), and *Sander canadensis* (≥250 mm) were based on length at maturity (fork length for *Scaphirhynchus* spp., total length for other species)

models for adult large-bodied fishes, Gear × Segment was ranked best for *Scaphirhynchus platorynchus* and *Cycleptus elongatus*, Gear × Habitat for *Sander canadensis*, and adult *Scaphirhynchus albus* had 2 competing models (Gear × Season and Gear × Habitat) which were both considered the best approximating models (Table 4). Juvenile large-bodied fishes had *S. albus* and *S. canadensis* with Gear × Year as the best model, *S. platorynchus* with Gear × Segment as the best model, and *C. elongatus* had both Gear × Year and Gear × Segment as the best approximating models. Only *S. platorynchus* and *C. elongatus* had the same best models for both maturity classes.

Probability of detection estimates for the Gear × Segment interaction model resulted in unique patterns for those species where it was the best model (Fig. 3). For example, *Scaphirhynchus platorynchus* detection probabilities increased in downstream river segments, but at the segment nearest the confluence of the Mississippi River (14) they decreased slightly. Conversely, *Cycleptus elongatus* were most likely to be detected in upstream segments, while the middle segment (10) had the lowest probability of detection. While the magnitude of the response varied by gear type, the patterns generally remained the same among gear types and maturity classes (for which Gear × Segment was the best model).

The Gear × Habitat interaction model was the best model for adult *Sander canadensis* and was equally as well supported as Gear × Season for adult *Scaphirhynchus albus* (Fig. 4). Pool-type habitats had higher detection probabilities than the other habitat types for adult *S. canadensis*, but the magnitude of the difference varied by gear type. There were no clear differences in detection probabilities (all were <0.05) among habitat types for adult *S. albus*. The other parsimonious model for adult *S. albus*, Gear × Season, also had low detection probability estimates with cold-water gill nets being the greatest at $p = 0.02$. The Gear × Year interaction model was the best ranking model for juvenile *S. albus*, *S. canadensis*, and was equally as well supported as Gear × Segment for *Cycleptus elongatus*. Probability of detection estimates for juvenile *S. albus*, *C. elongatus*, and *S. canadensis* were low (<0.10) for all years and all gear types. All the estimates within a species had overlapping 95% confidence intervals, indicating no statistical difference in detection probabilities among years or gear types for these 3 juvenile species.

Table 4. Ranking of probability of detection (p) models for a gear-type interaction with river segment, habitat, season, and year. The change in Akaike's information criterion (ΔAIC) or quasi-Akaike's information criterion (QAIC) was used to rank models relative to the best approximating model. Parsimonious models were those with an AIC/QAIC value <2. Number of modeled parameters (K), Akaike weights (w_i), and log-likelihood (log(L)) were used to determine model support for each variable

Adults					Juveniles				
Model	ΔAIC/QAIC	K	w_i	log(L)	Model	ΔAIC/QAIC	K	w_i	log(L)
Scaphirhynchus albus									
p(Gear × Season)	0.00	5	0.56	−280.6	p(Gear × Year)	0.00	7	0.91	−421.0
p(Gear × Habitat)	1.20	6	0.31	−280.2	p(Gear × Segment)	4.94	8	0.08	−422.5
p(Gear × Year)	3.76	7	0.08	−280.4	p(Gear × Season)	9.26	5	0.01	−427.6
p(Gear × Segment)	4.64	8	0.05	−279.9	p(Gear × Habitat)	10.23	6	0.01	−427.1
Scaphirhynchus platorynchus									
p(Gear × Segment)	0.00	8	1.00	−4302.3	p(Gear × Segment)	0.00	8	1.00	−4781.8
p(Gear × Year)	132.79	7	0.00	−4369.7	p(Gear × Year)	67.71	7	0.00	−4816.7
p(Gear × Habitat)	150.99	6	0.00	−4379.8	p(Gear × Habitat)	79.15	6	0.00	−4823.4
p(Gear × Season)	345.44	5	0.00	−4478.1	p(Gear × Season)	193.39	5	0.00	−4881.5
Cycleptus elongatus									
p(Gear × Segment)	0.00	8	1.00	−3167.5	p(Gear × Year)	0.00	7	0.52	−759.9
p(Gear × Year)	192.57	7	0.00	−3264.8	p(Gear × Segment)	0.40	8	0.43	−759.1
p(Gear × Season)	222.24	5	0.00	−3281.7	p(Gear × Habitat)	4.91	6	0.04	−763.4
p(Gear × Habitat)	236.21	6	0.00	−3287.6	p(Gear × Season)	9.47	5	0.00	−766.7
Sander canadensis									
p(Gear × Habitat)	0.00	6	1.00	−1091.4	p(Gear × Year)	0.00	7	0.99	−265.2
p(Gear × Season)	25.39	5	0.00	−1105.1	p(Gear × Segment)	9.59	8	0.01	−269.0
p(Gear × Segment)	25.85	8	0.00	−1102.3	p(Gear × Season)	14.48	5	0.00	−274.4
p(Gear × Year)	31.68	7	0.00	−1106.2	p(Gear × Habitat)	15.69	6	0.00	−274.0

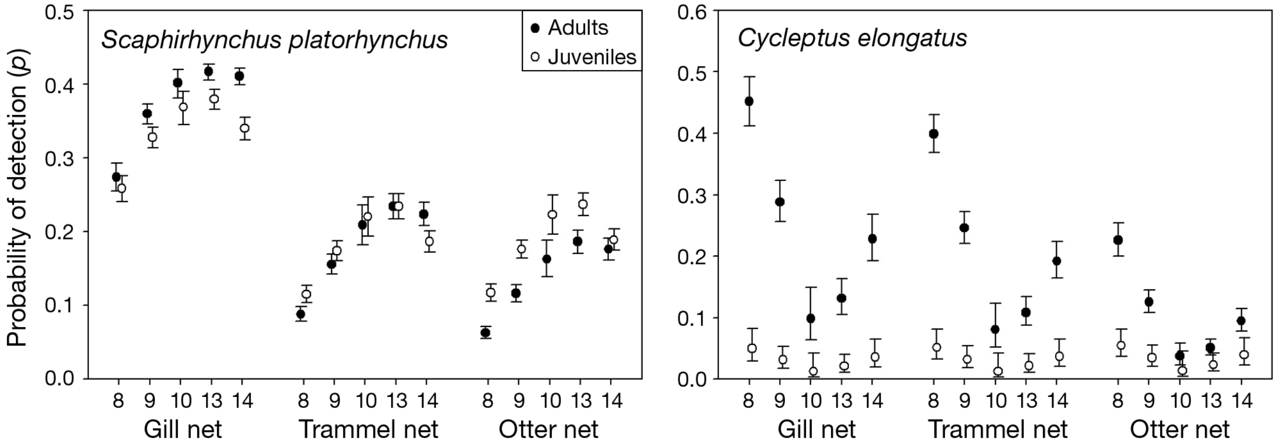

Fig. 3. Probability of detection (*p*) model estimates for 2 large-bodied species with Gear × Segment as the best approximating model. Estimates and 95% confidence intervals are given for gill nets, trammel nets, and otter trawls deployed in Segments 8 (rkm 1212 to 958), 9 (rkm 958 to 591), 10 (rkm 591 to 402), 13 (rkm 402 to 209), and 14 (rkm 209 to 0) in the Lower Missouri River. Adult lengths for *Scaphirhynchus platorynchus* (≥550 mm fork length) and *Cycleptus elongatus* (≥500 mm total length) were based on length at maturity

Small-bodied fish models strongly supported river segment as the best descriptor of detection probabilities (otter trawl was the only gear modeled for these fishes), where all model weights were 1.0 (Table 5). Habitat was the second ranked candidate model for *Macrhybopsis aestivalis* and *M. meeki*, and year was second for *M. gelida*. However, the differences in QAIC and AIC between the segment model and the next best candidate model was >20 for all small-bodied fishes, indicating little support for other candidate models. *M. gelida* and *M. meeki* showed no overdispersion in the data ($\hat{c} < 1$), but *M. aestivalis* did show minor overdispersion ($\hat{c}$ = 1.6), which was accounted for during model selection.

Small-bodied fishes had species-specific spatial trends in detection probabilities (Fig. 5). Detection probabilities of *Macrhybopsis aestivalis* and *M. meeki* increased in downstream segments to nearly 0.50 in Segments 13 and 14. *M. gelida* detection probabilities were similar among Segments 9, 10, 13, and 14 (mean $p = 0.14$), but lower in Segment 8 ($p = 0.02$).

DISCUSSION

The use of detection probabilities would be a valuable component to multi-gear evaluations for imperiled large-river fishes when spatial or temporal heterogeneity exists in fish captures. The use of simpler indices, such as the percent of samples in which a species was present, should not be used to evaluate sampling gears because it underestimates the true ability of the sampling gear to detect the presence of a species. This occurs because percent presence includes sites where a species was truly absent in calculations (i.e. gear efficiency should not be calculated from sites where the species was not even available for capture). The use of detection probabilities accounts for species absence and non-detection at a site, thus leading to a more accurate estimate of the gear's ability to detect species presence. Large-river monitoring programs for rare fishes typically sample over several seasons, and in many habitat types that cover a large spatial area where species detectability is likely to differ. Using detection probabilities to understand differences in detectability can help monitoring programs structure sampling effort to achieve specific objectives.

Incorporating detection probabilities into an adaptive management monitoring program can help refine sampling designs in a variety of ways. For example, fishery managers can incorporate environmental variables when modeling detection probabilities to determine thresholds for effective sampling conditions (e.g. detection probabilities decline below an acceptable level when rivers reach flood stage). Detection probabilities are also useful for multi-gear programs because estimates can be compared among active and passive gears (i.e. the comparison is based on a probability and not a measurement that is unit specific to a single gear type) as we demonstrated. Detection probabilities may be especially useful for biologists examining gear efficiencies prior to implementation in monitoring programs. Lastly,

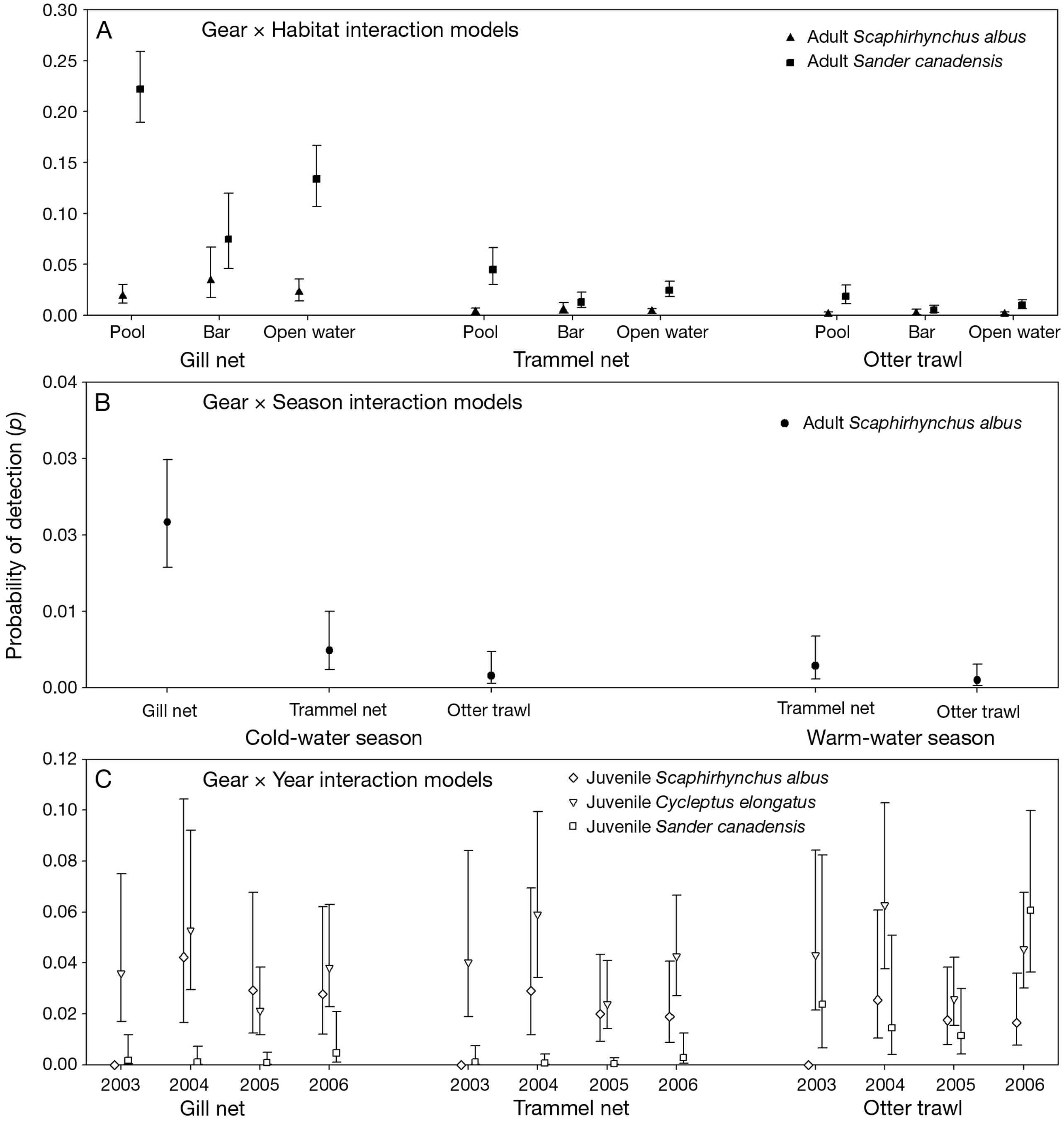

Fig. 4. Probability of detection (p) estimates and 95 % confidence intervals from the best approximating models of 3 large-bodied species. The best models were (A) Gear × Habitat (gill net, trammel net, otter trawl; pool, bar, open water) for adult *Scaphirhynchus albus* (≥550 mm fork length) and *Sander canadensis* (≥250 mm total length), (B) Gear × Season (cold-water season from October 31 to June 30; warm-water season from July 1 to October 30) for adult *S. albus*, and (C) Gear × Year (2003 to 2006) for juvenile *S. albus* (<550 mm fork length), *Cycleptus elongatus* (<500 mm total length), and *Sander canadensis* (<250 mm total length). Two candidate models were parsimonious for adult *S. albus* (Gear × Habitat and Gear × Season) and juvenile *C. elongatus* (Gear × Year and Gear × Segment)

evaluating gear efficiency in an open large-river system is difficult due to the river's dynamic nature (Casselman et al. 1990). While indices such as catchability (i.e. the portion of a population removed with a single unit of effort) have been used to evaluate gear effectiveness in large rivers (Guy et al. 2009a),

Table 5. Ranking of probability of detection (p) models for small-bodied fishes according to Akaike's information criterion (AIC) or quasi-Akaike's information criterion (QAIC). We modeled the probability of detection from fish catches only in otter trawls as constant (.), a function of segment, habitat, season, or year. The variance inflation factor ($\hat{c}$) was estimated from the model with the greatest number of parameters (i.e. segment). Number of modeled parameters (K), Akaike weights (w_i), and log-likelihood (log(L)) were used to determine model support for each variable

Model	ΔAIC/QAIC	K	w_i	log(L)	$\hat{c}$
Macrhybopsis aestivalis					
p(Segment)	0.00	7	1.00	−1740.0	1.6
p(Habitat)	72.49	4	0.00	−1800.5	
p(Season)	94.27	3	0.00	−1819.4	
p(Year)	108.84	5	0.00	−1827.7	
p(.)	119.96	2	0.00	−1841.3	
Macrhybopsis gelida					
p(Segment)	0.00	6	1.00	−839.3	<1
p(Year)	44.69	5	0.00	−862.7	
p(Season)	46.52	3	0.00	−865.6	
p(.)	51.27	2	0.00	−869.0	
p(Habitat)	52.88	4	0.00	−867.8	
Macrhybopsis meeki					
p(Segment)	0.00	6	1.00	−1388.1	<1
p(Habitat)	123.00	4	0.00	−1451.6	
p(Season)	153.36	3	0.00	−1467.8	
p(Year)	158.61	5	0.00	−1468.4	
p(.)	162.13	2	0.00	−1473.2	

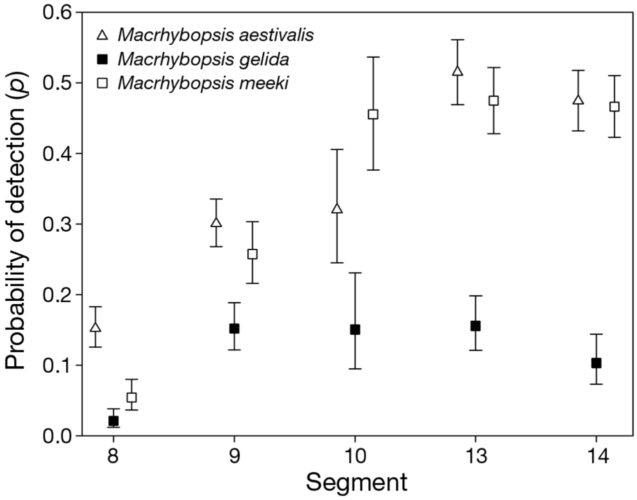

Fig. 5. Probability of detection (p) estimates by river segment for *Macrhybopsis aestivalis*, *M. gelida*, and *M. meeki* caught in otter trawls. Estimates and 95% confidence intervals are shown for Segments 8 (rkm 1212 to 958), 9 (rkm 958 to 591), 10 (rkm 591 to 402), 13 (rkm 402 to 209), and 14 (rkm 209 to 0) in the Lower Missouri River

they may not be feasible for many species (e.g. small-bodied or young-of-year fishes) and generally require an experimental design, which may not provide suitable results applicable to an extensive fish sampling program.

Our results supported our hypothesis that detection probabilities would be influenced by gear type, which was based on known bias in fish collections due to mesh size, net material, and restricted deployment locations (Hubert 1996, Guy et al. 2009b). In the channelized Missouri River, gill nets had a greater ability to detect most adult large-bodied fishes than trammel nets, indicating that gill nets may be a more appropriate gear for sampling an array of adult large-bodied fishes. Drifting trammel nets for *Scaphirhynchus* spp. can produce highly variable catches, many of which are zero catches (Hubert & Schmitt 1982, Wanner et al. 2007, Doyle et al. 2008), which can be problematic for monitoring programs. The small mesh (6 mm inner bar) of the otter trawl allowed it to effectively capture most juvenile and all small-bodied fishes. Herzog et al. (2005) fished an otter trawl similar to that used in our study in the Mississippi River and found higher species detection and catch rates of small-bodied and larval fishes with a small mesh (<5 mm) trawl when compared to a 19 mm mesh trawl.

Sampling with gears that have low detection probabilities for juvenile long-lived fishes (e.g. *Scaphirhynchus albus*, *Cycleptus elongatus*, and *Sander canadensis*) may be problematic, because the inability to detect changes in recruitment patterns slows the understanding of ecological responses to management actions and hinders changes in management policy (Eitzmann et al. 2007, Doyle et al. 2008). The otter trawl may have the greatest potential for collection of these and other small, imperiled benthic fishes in large rivers (Herzog et al. 2005, Braaten & Fuller 2007, Doyle et al. 2008, Guy et al. 2009b). However, juvenile fishes may not have been fully vulnerable with our current trawl design, as indicated by low detection probabilities for some species. Identifying other small mesh gears that can capture juvenile fishes is warranted and important for understanding recruitment dynamics (Maceina & Pereira 2007), particularly with rare fishes such as sturgeons (Paragamian & Hansen 2008).

Fishery biologists should not assume that gears fish similarly over a large stretch of river. Our study encompassed over 1200 rkm of a large river and provided an opportunity to evaluate detection probabilities longitudinally. Physical features (i.e. turbidity, habitat complexity, size) change throughout a river and likely affect detection probabilities (e.g. Speas et al. 2004). In addition, longitudinal changes in abundance for some species will likely affect detection probabilities because the more fish available in an area to be captured, the greater the likelihood of capturing at least one of the species (Royle & Nichols 2003, MacKenzie 2005, MacKenzie et al. 2006). Species exhibiting a gear-by-segment interaction (i.e. *Scaphirhynchus platorynchus* and *Cycleptus elongatus*) had a probability of detection estimates that generally corresponded to the relative abundances reported by Berry et al. (2005; i.e. high relative abundance equates to a high detection probability). MacKenzie et al. (2005) discussed that rare animals are likely to be associated with low detection probabilities, even when the sampling units are occupied, which is similar to our finding for *S. albus* which had $p \leq 0.03$. Since detection probabilities varied greatly among segments for some species (e.g. *C. elongatus*), monitoring programs distributed over a large area might consider stratifying sampling efforts relative to the abundance of targeted species or prioritize study reaches.

Differences in detection probabilities as a function of habitat were evident in adult large-bodied fishes. Madejczyk et al. (1998) found that *Sander canadensis* might select for areas near wing dikes on the Mississippi River for the rocky cover near areas of higher current velocities, the type of habitats where gill nets were deployed in our study. The higher detection probabilities observed for adult fishes in pool-type habitats may be a function of preferred habitat use, greater sampling efficiency, or a combination of both. Programs focused on species that exhibit a patchy habitat distribution should account for differences in detection probabilities among habitats in order to appropriately allocate sampling effort according to program objectives. For example, a rapid assessment program may choose to sample habitats with high detection probabilities, whereas a habitat assessment program may choose to allocate more effort to low detection habitats to ensure that a species which is present is not missed.

The spatial and temporal variables chosen did not predict detection probabilities well for juvenile *Scaphirhynchus albus*, *Cycleptus elongatus*, or *Sander canadensis*. Similar detection probabilities among all estimates, along with multiple top-ranked models, indicate that other variables may better model detection probabilities. It is unlikely that rare or endangered juvenile fishes would be ubiquitous throughout a study site, because they generally require specialized habitats (Niles & Hartman 2009). Fishery assessments focused on juvenile fishes or recruitment surveys should select variables which take into consideration the biological reasons for the variation in detection probabilities among gear types, sample locations, or timing of surveys.

Continued refinements to the sampling protocol of a large-scale monitoring program working under an adaptive management framework are important. Program evaluations ensure that the best sampling protocols are used to meet objectives, such as stocking evaluations, effects of habitat modifications, and flow regulation. Allocating sampling effort towards gears with the highest detection probabilities can minimize the number of zero catches, increase catches of target species, reduce gear-related bias associated with multi-gear sampling, and lead to a more efficient sampling protocol to detect long-term trends and fish responses to management actions. Some monitoring programs should even consider using occupancy models to first estimate detection probabilities and then estimate occupancy to index the status of rare species instead of using traditional indices such as catch per unit effort (MacKenzie et al. 2006).

Gill nets and otter trawls had greater detection probabilities for most species in the lower Missouri River, so the use of trammel nets may be redundant for species in need of conservation in this area. Trammel nets would still have utility in target applications. Our results clearly demonstrate that differences in detection probabilities exist among species and spatially throughout the study area and that sampling strategies would differ for species-specific sampling programs. The use of detection probabilities to evaluate gear types should be applicable in most other large-river systems, but the results will likely differ, as the species assemblages, habitats, and river conditions will vary from those on the Missouri River.

Acknowledgements. We thank the U.S. Geological Survey Science Support Partnership and U.S. Army Corps of Engineers for funding this project. Mark Drobish of the U.S. Army Corps of Engineers coordinated field work activities through the Pallid Sturgeon Population Monitoring and Assessment Program. Fish sampling was carried out by the U.S. Fish and Wildlife Service, Nebraska Game and Parks Commission, and Missouri Department of Conservation. Clayton Ridenour with U.S. Fish and Wildlife Service compiled the study area map. This manuscript greatly benefitted from the reviews of Keith Gido, Brett Sandercock, Phil Bettoli, and Steve Chipps. The Kansas Cooperative Fish and Wildlife Research Unit is jointly sponsored by the Kansas Department of Wildlife and Parks, Kansas State University, the U.S. Geological Survey, and the Wildlife Management Institute. The use of trade, product, industry, or firm names, or products is for informative purposes only and does not constitute an endorsement by the U.S. Government, the U.S. Geological Survey, or other sponsoring or participating agencies.

LITERATURE CITED

Argent DG, Kimmel WG (2005) Efficiency and selectivity of gill nets for assessing fish community composition of large rivers. N Am J Fish Manag 25:1315–1320

Beamesderfer RC, Rieman BE (1988) Size selectivity and bias in estimates of population statistics of smallmouth bass, walleye, and northern squawfish in a Columbia River reservoir. N Am J Fish Manag 8:505–510

Becker GC (1983) Fishes of Wisconsin. University of Wisconsin Press, Madison, WI

Berry CR Jr, Wildhaber ML, Galat DL (2005) Fish distribution and abundance. Population structure and habitat use of benthic fishes along the Missouri and lower Yellowstone Rivers, Vol 3. US Geological Survey, Cooperative Research Units, South Dakota State University, Brookings, SD

Braaten PJ, Fuller DB (2007) Growth rates of young-of-year shovelnose sturgeon in the upper Missouri River. J Appl Ichthyol 23:506–515

Bryan JL, Wildhaber ML, Gladish D, Holan S, Ellerseick M (2009) The power to detect trends in Missouri River fish populations within the Pallid Sturgeon Population Assessment Program. Final report to the Army Corps of Engineers. US Geological Survey, Reston, VA

Burnham KP, Anderson DR (2002) Model selection and multimodal inference, 2nd edn. Springer-Verlag, New York, NY

Casselman JM, Penczak T, Carl L, Mann RHK, Holcik J, Woitowich WA (1990) An evaluation of fish sampling methodologies for large river systems. Pol Arch Hydrobiol 37:521–552

Doyle W, Paukert CP, Starostka A, Hill T (2008) A comparison of four sampling gears used to collect shovelnose sturgeon in the Lower Missouri River. J Appl Ichthyol 24: 637–642

Drobish MR (2008) Pallid sturgeon population assessment project, Vol 1.3. US Army Corps of Engineers, Omaha District, Yankton, SD

Eitzmann JL, Makinster AS, Paukert CP (2007) Distribution and growth of blue sucker in a Great Plains river, USA. Fish Manag Ecol 14:255–262

Galat DL, Berry CR, Gardner WM, Hendrickson JC and others (2005) Spatiotemporal patterns and changes in Missouri River fishes. Am Fish Soc Symp 45:249–291

Guy CS, Oldenburg EW, Gerrity PC (2009a) Conditional capture probability of *Scaphirhynchus* spp. in drifting trammel nets. N Am J Fish Manag 29:817–822

Guy CS, Braaten PJ, Herzog DP, Pitlo J, Rogers RS (2009b) Warmwater fish in rivers. In: Bonar S, Willis DW, Hubert WA (eds) Standard methods for sampling North American freshwater fishes. American Fisheries Society, Bethesda, MD

Hamley JM (1975) Review of gillnet selectivity. J Fish Res Board Can 32:1943–1969

Herzog DP, Barko VA, Scheibe JS, Hrabik RA, Ostendorf DE (2005) Efficacy of a benthic trawl for sampling small-bodied fishes in large river systems. N Am J Fish Manag 25: 594–603

Hesse LW, Mestl GE (1993) An alternative hydrograph for the Missouri River based on the precontrol condition. N Am J Fish Manag 13:360–366

Hines JE (2006) PRESENCE 2.0 Software to estimate patch occupancy and related parameters. USGS-PWRC. Available at: www.mbr-pwrc.usgs.gov/software/presence.html (accessed on 1 July 2007)

Hubert WA (1996) Passive capture techniques In: Murphy BR, Willis DW (eds) Fisheries techniques, 2nd edn. American Fisheries Society, Bethesda, MD

Hubert WA, Schmitt DN (1982) Factors influencing catches of drifted trammel nets in a pool of the upper Mississippi River. Proc Iowa Acad Sci 89:153–154

IUCN (International Union for Conservation of Nature) (2011) IUCN Red List of Threatened Species, Version 2010.4. Available at: www.iucnredlist.org (accessed on 6 April 2011)

Jenkins RE, Burkhead NM (1994) Freshwater fishes of Virginia. American Fisheries Society, Bethesda, MD

Keenlyne KD, Jenkins LG (1993) Age at sexual maturity of the pallid sturgeon. Trans Am Fish Soc 122:393–396

Kjelson MA, Johnson GN (1978) Catch efficiencies of a 6.1-meter otter trawl for estuarine fish populations. Trans Am Fish Soc 107:246–254

Lapointe NW, Corkum LD, Mandrak NE (2006) A comparison of methods for sampling fish diversity in shallow offshore waters of large rivers. N Am J Fish Manag 26: 503–513

Longoria MP, Weckerly FW (2007) Estimating detection probabilities from sign of collared peccary. J Wildl Manag 71:652–655

Maceina MJ, Pereira DL (2007) Recruitment. In: Guy CS, Brown ML (eds) Analysis and interpretations of freshwater fisheries data. American Fisheries Society, Bethesda, MD

MacKenzie DI (2005) What are the issues with presence–absence data for wildlife managers? J Wildl Manag 69: 849–860

MacKenzie DI, Bailey LL (2004) Assessing the fit of site-occupancy models. J Agric Biol Environ Stat 9:300–318

MacKenzie DI, Nichols JD, Lachman GB, Droege S, Royle JA, Langtimm CA (2002) Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248–2255

MacKenzie DI, Nichols JD, Sutton N, Kawanishi K, Bailey LL (2005) Improving inferences in population studies of rare species that are detected imperfectly. Ecology 86: 1101–1113

MacKenzie DI, Nichols JD, Royle JA, Pollock KH, Bailey LL, Hines JE (2006) Occupancy estimation and modeling. Academic Press, Burlington, MA

Madejczyk JC, Mundahl ND, Lehtinen RM (1998) Fish assemblages of natural and artificial habitats within the channel border of the upper Mississippi River. Am Midl Nat 139:296–310

Niles JM, Hartman KJ (2009) Larval fish use of dike structures on a navigable river. N Am J Fish Manag 29: 1035–1045

Noble RL, Austen DJ, Pegg MA (2007) Fisheries management study design considerations. In: Guy CS, Brown ML (eds) Analysis and interpretations of freshwater fisheries data. American Fisheries Society, Bethesda, MD

NRC (National Research Council) (2002) The Missouri River ecosystem: exploring the prospects for recovery. National Academy Press, Washington, DC

Paragamian VL, Hansen MJ (2008) Evaluation of recovery goals for endangered white sturgeon in the Kootenai River, Idaho. N Am J Fish Manag 28:463–470

Paukert CP (2004) Comparison of electrofishing and trammel netting variability for sampling native fishes. J Fish Biol 65:1643–1652

Pflieger WL (1997) The fishes of Missouri, 2nd edn. Missouri Department of Conservation, Jefferson City, MI

Quist MC, Boelter AM, Lovato JM, Korfanta NM and others (2004) Research and assessment needs for pallid sturgeon recovery in the Missouri River. Final Report to the US Geological Survey, US Army Corps of Engineers, US Fish and Wildlife Service, and US Environmental Protection Agency. William D. Ruckelshaus Institute of Environment and Natural Resources, University of Wyoming, Laramie, WY

➤ Quist MC, Gerow KG, Bower MR, Hubert WA (2006) Random versus fixed-site sampling when monitoring relative abundance of fishes in headwater streams of the Upper Colorado River Basin. N Am J Fish Manag 26:1011–1019

➤ Ridenour CJ, Starostka AB, Doyle WJ, Hill TD (2009) Habitat used by *Macrhybopsis* chubs associated with channel modifying structures in a large regulated river: implications for river modification. River Res Appl 25:472–485

Robison HW, Buchanan TM (1988) Fishes of Arkansas. University of Arkansas Press, Fayetteville, AR

➤ Royle JA, Nichols JD (2003) Estimating abundance from repeated presence–absence data or point counts. Ecology 84:777–790

SEI (Sustainable Ecosystems Institute) (2004) Power to detect trends in pallid and shovelnose sturgeon populations in the Missouri River. SEI, Portland, OR

Sheehan RJ, Rasmussen JL (1999) Large rivers. In: Kohler CC, Hubert WA (eds) Inland fisheries management in North America, 2nd edn. American Fisheries Society, Bethesda, MD

➤ Speas DW, Walters CJ, Ward DL, Rogers RS (2004) Effects of intraspecific density and environmental variables on electrofishing catchability of brown and rainbow trout in the Colorado River. N Am J Fish Manag 24:586–596

USFWS (US Fish and Wildlife Service) (2005) Biological procedures and protocol for collecting, tagging, sampling, holding, culture, transporting, and data recording for researchers and managers handling pallid sturgeon. USFWS, Missouri River Fish and Wildlife Conservation Office, Bismark, ND

USGS (US Geological Survey) (2009) USGS surface-water daily data for the nation. Available at: http://waterdata.usgs.gov/nwis/dv? (accessed on 2 May 2011)

➤ Wanner GA, Shuman DA, Brown ML, Willis DW (2007) An initial assessment of sampling procedures for juvenile pallid sturgeon in the Missouri River downstream of Fort Randall Dam, South Dakota and Nebraska. J Appl Ichthyology 23:529–538

➤ Wanner GA, Klumb RA, Shuman DA, Steffensen K, Stukel S, Utrup NJ (2010) Comparison of green and white mesh trammel nets and gill nets to assess the fish community in a large river. N Am J Fish Manag 30:12–25

➤ Washburn BE, Tempel DJ, Millspaugh JJ, Gutiérrez RJ, Seamans ME (2004) Factors related to fecal estrogens and fecal testosterone in California spotted owls. Condor 106: 567–579

➤ Wildhaber ML, Holen SH, Bryan JL, Gladish DW, Ellersieck M (2011) Assessing power of large river fish monitoring programs to detect population changes: the Missouri River example. J Appl Ichthyol 27:282–290

➤ Yoccoz NG, Nichols JD, Boulinier T (2001) Monitoring of biological diversity in space and time. Trends Ecol Evol 16:446–453

Editorial responsibility: Steven Cooke,
Ottawa, Ontario, Canada

Submitted: May 18, 2011; Accepted: October 31, 2011
Proofs received from author(s): February 18, 2012