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Using angling and electric fishing to estimate smallmouth bass abundance in a river

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Abstract

Estimating abundance is fundamental to effective fishery management but can be challenging in a river where spatial and temporal heterogeneity may preclude the consistent use of a single sampling gear and different gears have differing size selectivity and capture probabilities of fish. In this study, the number of smallmouth bass, *Micropterus dolomieu* Lacepède, was estimated based on mark–recapture data from angling and boat electric fishing in a 4.2-km regulated section (mean width = 115 m) of the Broad River, South Carolina, USA. Closed-population capture–mark–recapture models were fit in the Bayesian hierarchical modelling framework with an estimated number of 2,380 fish (95% credible interval: 1,578–3,693) over 200 mm TL, although simulations indicated that abundance would be slightly overestimated (<20%) when two gears selected for different individuals. Integrating two gear types into a mark–recapture study can provide a method for assessing abundance in spatially or temporally heterogeneous habitats.

KEYWORDS
gear bias, hierarchical models, mark–recapture, *Micropterus dolomieu*, Piedmont, regulated river

1 | INTRODUCTION

Abundance estimation is a fundamental aspect of fisheries management. Sufficient knowledge of fish population size is critical to informing management decisions such as length limits and stocking rates. Abundance can be estimated by various methods. Catch-per-unit-effort (CPUE) from standardised sampling is commonly used (Copeland, Orth & Palmer, 2006; Balcombe & Arthington, 2009). Depletion and removal techniques are employed by blocking off a habitat section in streams (Rosenberger & Dunham, 2005; Habera, Kulp, Moore & Henry, 2010) and rivers (Odenkirk & Smith, 2005). Mark–recapture methods exist for abundance estimation in closed populations (i.e. no births, deaths, immigration or emigration; Otis, Burnham, White & Anderson, 1978). The two-sample Lincoln–Petersen and multiple-sample Schnabel methods require batch mark data over a short period to satisfy population closure assumptions (Modde, Burnham & Wick, 1996). Mark–recapture data of unique individuals provide the richest data for estimating abundance of a closed population because they record individual capture histories over sampling occasions (Pine, Pollock, Hightower, Kwak & Rice, 2003).

Despite the availability of various methods, abundance estimation is complicated in rivers due to their large size and fluctuating discharge (i.e. spatial and temporal heterogeneity). Multiple gears are often used to sample populations and assemblages in large lentic (Weaver, Magnuson & Clayton, 1993; Jackson & Harvey, 1997; Rogers, Hansen & Beard, 2005; Ruetz, Uzarski, Krueger & Rutherford, 2007) and lotic (Arab, Wildhaber, Wikle & Gentry, 2008; Pregler, Vokoun, Jensen & Hagstrom, 2015) waterbodies. River discharge changes seasonally following precipitation patterns and also changes greatly and rapidly due to anthropogenic causes such as water releases from dams (Dynesius & Nilsson, 1994). Changes in flow condition could prevent the use of a single gear even within the same habitat types on different days (Casselman et al., 1990).

Spatial and temporal heterogeneity of rivers necessitates a creative approach of combining sampling methods to estimate abundance. However, information is lacking on how data from multiple gears could be combined to inform abundance estimation. Gears have inherent sampling biases (Beamesderfer & Rieman, 1991). For example, electric fishing selects for larger individuals than fyke netting (Ruetz et al., 2007). Thus, different gears target different groups of individuals of the same species, and count data collected by different gears may not be directly comparable. Even if a single gear is consistently used, abundance estimation is further complicated by variable sampling efficiency resulting from spatial and temporal heterogeneity in rivers. Thus, CPUE, a common method for assessing abundance, cannot be applied reliably because it measures the product of true abundance and capture probability. Changes in CPUE can result solely from varying capture probability due to sampling conditions. Mark–recapture surveys typically require several days of sampling, and capture probability of individuals may vary under different discharge conditions during the sample period. Incorporating this temporal variation in capture efficiency should result in more accurate abundance estimates, which is particularly important when different gears are selective of different individuals.

In this study, abundance of smallmouth bass, *Micropterus dolomieu* Lacepède, was estimated in a 4.2-km section (mean river width = 115 m) of the Broad River, a flow-regulated river located in South Carolina, USA, using a mark–recapture method. Although not native to the Broad River (Brewer & Orth, 2015), smallmouth bass have been stocked since 1984 to enhance recreational fishing opportunities, and natural reproduction occurs in the river (Bettinger, 2013). Population monitoring of smallmouth bass is important in the study river to infer predation impacts on native fish populations (Fritts & Pearsons, 2004) and assess stocking need (Buynak, Kornman, Surmont & Mitchell, 1991; Weidel, Josephson & Kraft, 2007; Zipkin et al., 2008). To estimate smallmouth bass abundance, angling and boat electric fishing were used on different sampling days to accommodate fluctuating river discharge conditions due to water release from dams. Imperfect and variable capture probabilities of individuals were addressed in a Bayesian state-space model. Simulations were used to explore the behaviour of these abundance models integrating two different gears that likely had differing sampling biases.

2 | MATERIALS AND METHODS

2.1 | Study area

This study was conducted in the Broad River (South Carolina, USA) between the Ninety-Nine Islands and Lockhart dams (Figure 1). The river has a total drainage area of 9,819 km^2 and flows approximately 240 km through the lower Blue Ridge and northern Piedmont regions of North and South Carolina. River discharge is regulated via several run-of-the-river dams. Surrounding land use is dominated by pasture lands with mixed forest.

Abundance sampling was performed in the 4.2-km section immediately below the Ninety-Nine Islands Hydroelectric Station. During the last 15 years, mean annual discharge below the station ranged between 22 and 119 m^3/s (USGS Gage 02153551). Habitats consist primarily of shoals underlain by bedrock, cobble/gravel-dominated riffles and scoured sandy pools.

2.2 | Mark-recapture sampling

The study goal was to estimate smallmouth bass abundance (>200 mm TL) in the 4.2-km section below the Ninety-Nine Islands dam. In the Broad River, 200 mm is the body length that is likely to be the minimum size of smallmouth bass targeted by anglers, and fish of this size are typically age 1 based on otolith reading (J.M. Bettinger, unpublished data). To achieve the study goal, a mark–recapture survey was conducted on 5 days (hereafter "occasions") during autumn 2015. Two gear types were used due to fluctuations in river discharge during the

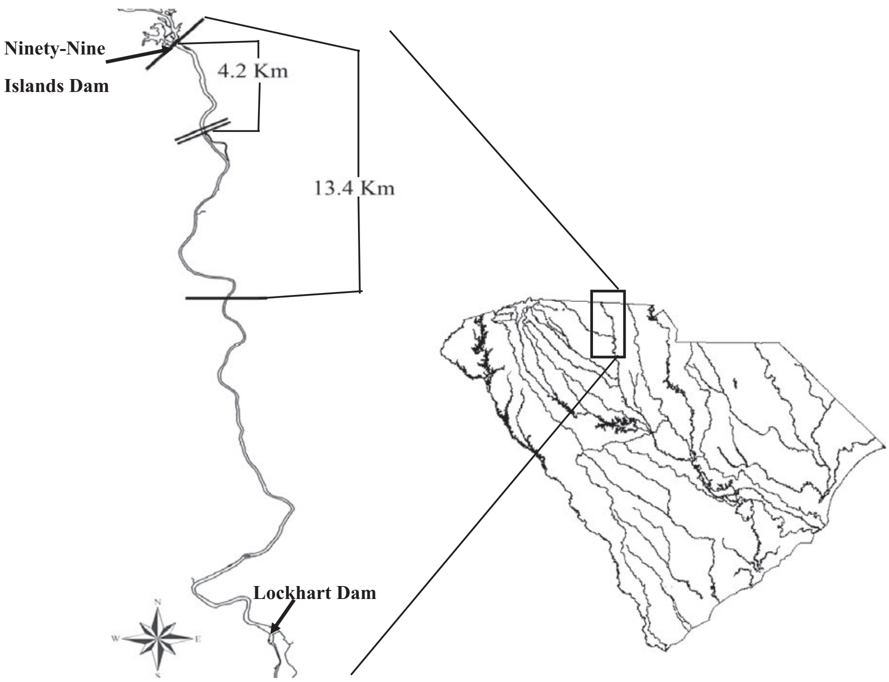

FIGURE 1 Major rivers of South Carolina with expanded study area of the Broad River between Ninety-Nine Islands and Lockhart Dams depicting the mark–recapture section (4.2 km) within the radio tracking segment (13.4 km). The river flows from the north to the south

study. At low discharge (<42 m^3/s), boat electric fishing is difficult as bedrock shoals are inaccessible with motor boats, and a combination of few access points, river width (115 m) and target fish size (>200 mm TL) made angling the only logistically feasible method of sampling the entire 4.2-km study section. At high discharge (>42 m^3/s), angling is dangerous because of swift currents, but an electric fishing boat can navigate through the whole study section. Angling was performed on 20 and 22 October (first and second occasions), and electric fishing was performed on 29 October and 4 and 11 November (third, fourth and fifth occasions). Sampling was completed over the 22-day period to conform to the population closure assumption.

Angling was used on the first two occasions when 11 anglers floated downstream from the Ninety-Nine Islands dam for 4.2 km in kayaks and canoes (Figure 1). Anglers were supplied with and instructed to use at least one of three types of light-weight (<4 g) lures (in-line spinners, jigged grubs and soft plastic minnows). Anglers got out of their kayaks or canoes at shallow shoals to wade and sampled all accessible river habitats. Non-wadeable habitats (i.e. deep pools) were fished from the edge of shoals and riffles as well as from canoes and kayaks as anglers floated downstream. Once smallmouth bass were captured with angling, they were quickly transported to the closer of two tagging teams for processing.

Electric fishing was conducted at high discharge (>42 m^3/s) when anglers could not wade safely (occasions three through five). A single boat equipped with a Smith-Root GPP 2.5 electric fisher (Smith-Root Inc., Vancouver, WA, USA) was used sampling downstream from the upper extent of the study section. The 4.2-km study section was divided into four sub-sections using three large shoal areas as natural breaks where sampling was stopped and captured smallmouth bass were transferred to another boat for processing. After handling, the fish were released at the mid-point of each sub-section.

Fish handling procedures were identical regardless of the capture method. Smallmouth bass were held in an aerated live well of river water for <10 min until an 8-mm passive integrated transponder (PIT) tag (Oregon RFID Inc., Portland, OR, USA) was injected intracoelomically using a Biomark (Boise, ID, USA) MK165 implanter and N165 needle. Tags were implanted ventrally approximately 20 mm posterior to the pelvic girdle (Roussel, Haro & Cunjak, 2000). To aid in the identification of recaptured smallmouth bass, the left pectoral fin was also clipped after their initial capture. All recaptured smallmouth bass were measured, scanned for a tag number using either an Avid Identification Systems Power Tracker (Norco, CA, USA) or Oregon RFID Easy Tracer II scanner (Portland, OR, USA). After handling, fish were promptly released back to the river. The mean daily river temperature was 12.6–17.0°C during the five sampling occasions (HOBO U22-001 data logger; Onset Computer Corp., Bourne, MA, USA), which is well below the upper thermal threshold of smallmouth bass (Pease & Paukert, 2014). Therefore, thermal stress was considered low.

2.3 | Abundance modelling

Abundance of smallmouth bass >200 mm TL in the 4.2-km section was estimated using a mark–recapture model assuming population closure. That is, no recruitment to 200 mm TL, death, immigration or emigration was assumed to occur. Given the short period of time (22 days) over which mark–recapture data were collected, it was plausible that recruitment to the minimum body size and death were negligible. It was assumed that smallmouth bass implanted with PIT tags suffered no post-release mortalities, except 8 individuals (5 captured by angling and 3 by electric fishing) for which recovery to normal behaviour was not noted at the time of release due presumably to injuries during capture and that were omitted from all analyses. Care was taken to minimise handling time of fish, and the field protocol was similar to others that assumed no post-release mortalities (Beamesderfer & Rieman, 1991). In addition, the mark–recapture study was conducted in autumn when river temperatures were cool (≤17.0°C). Immigration and emigration were also assumed negligible based on movement of 5 transmitter-implanted individuals (mean = 379 mm; range = 300–476 mm TL) released downstream of the Ninety-Nine Islands dam and tracked in a 13.4-km river segment which included the 4.2-km study section (Figure 1). All 5 fish were relocated within the 4.2-km mark–recapture section when they were tracked once daily over 15 days (16 June–1 July 2015), and the median riverine distance between each successive location was 0 m (range: 0–476 m; Mycko, 2017). Four of the same five fish were again relocated within the 4.2-km section when they were located just before (13 October) and after (14 November) the mark–recapture study. Limited movement of smallmouth bass has been reported in other riverine systems during summer (Langhurst & Schoenike, 1990; Lyons & Kanehl, 2002) and autumn (Todd & Rabeni, 1989), reinforcing the assumption of limited emigration and immigration.

Capture–recapture data were analysed using Bayesian state-space models (Kéry & Schaub, 2012). Capture histories of all individuals (i) across sampling occasions (j) were created as a two-dimensional array, $y_{i,j}$, where 1s represent captures and 0s, non-captures. Capture–recapture models for closed populations infer how many more unique individuals (unobserved) should have been observed based on capture probabilities of observed individuals. In this regard, 7,000 rows of all 0 entries were added to the data, $y_{i,j}$, to represent individuals that were potentially part of the population but never observed (Royle & Dorazio, 2012). The objective of capture–recapture models for abundance is then to estimate the proportion (Ω) of individuals within this augmented data set, $y'_{i,j}$, that should belong to the population. The following general form of capture–recapture models was fit on the augmented dataset $y'_{i,j}$:

$$z_i \sim \text{Bernoulli}(\Omega)$$

$$y'_{i,j} \sim \text{Bernoulli}\left(z_i * P\right)$$

where z_i is the latent state of the membership in the population ($z_i = 1$ if a true member of the population; $z_i = 0$ otherwise) and P is the capture probability of individuals. Abundance (N) is then estimated by dividing observed count (C) by capture probability ($N = C/P$). Three different hypotheses of capture probability were tested: capture probability was constant over five occasions (M_0), varied by occasion (M_t) and varied by sampling gear (M_g). Models were compared using deviance information criterion (DIC) values; the model with the lowest DIC value was selected as the top-ranked model (Spiegelhalter, Best, Carlin & Van Der Linde, 2002). Capture–recapture models can accommodate more complex structures such as behaviour or individual variation (Otis et al., 1978), but convergence of these models was not achieved with the data presumably due to low recapture probabilities.

Models were analysed through Markov-Chain Monte Carlo (MCMC) sampling methods in JAGS (Plummer, 2012) called from Program R (R Development Core Team 2015). Minimally informative priors were used in all models (i.e. $\Omega \sim$ Uniform(0,1), P~Uniform(0,1)). Posterior distributions of parameters were estimated by keeping the hundredth sample from 30,000 iterations of three chains after a 20,000 iteration burn-in period. Model convergence was assessed by examining plots of MCMC chains and visually ensuring mixture of all three chains. Gelman and Rubin diagnostics provided potential scale reduction factors for model parameters. Convergence of MCMC chains was assumed when Gelman-Rubin values of Ω and P were <1.1 (Brooks & Gelman, 1998).

2.4 | Model validation with simulations

The models above assume that sampling is from a single homogenous population (Otis et al., 1978). However, there was a statistically significant difference in body length between individuals captured by angling and electric fishing. Thus, it was likely that the two gears at least partially selected for different groups of individuals. To assess model performance when the assumption of equal catchability of individuals by both gears is violated, simulations were conducted in which two independent groups of individuals were targeted by different sampling gears, but data from both gears were analysed simultaneously as a single data set.

The top-ranked model, M_t (time varying), was used in simulations to investigate if known abundance can be estimated accurately. Five possible sampling scenarios were simulated, with three sampling occasions each, by varying abundance and capture probability between two groups of individuals (Table 1). Abundance was set to be equal between two groups in Scenarios A and B, but capture probability was equal (Scenario A) or unequal (Scenario B) between two groups. Abundance of one group was twice that of the other in three scenarios, but capture probability was equal between two groups (Scenario C), higher in the larger group (Scenario D) or higher in the smaller group (Scenario E). Abundance of each group was set at 1,000 or 2,000 individuals so that the sum of these values (2,000 or 3,000) was similar to the empirical abundance estimate obtained from the study area. Capture probability between two groups was similarly set equal in some simulations or varying in others to address uncertainties with gear bias and efficiency. Ranges of capture probability (10–30%) were higher than the empirical estimates reported in this study, but these settings were chosen to speed computational time. A sixth scenario was also simulated, as a control, in which a single homogenous population was assumed. In all scenarios, a range of capture probabilities was specified (Table 1) and a capture probability was randomly drawn from the range for each sampling occasion. Each scenario was simulated 1,000 times. Distributions of posterior mean values of estimated abundance across 1,000 replicates were compared to the true known abundance, which is the sum of abundances of the two groups. Simulations were completed using Clemson University's Palmetto Cluster supercomputer.

3 | RESULTS

3.1 | Abundance estimate

A total of 321 unique individuals >200 mm TL were captured and marked (Table 2). Of these fish, angling captured 175 unique individuals on the first two occasions and electric fishing collected 146 unique individuals on the subsequent three occasions. Across all five sampling occasions, 1 individual was captured three times, 16 individuals twice and 304 individuals only once. No recapture was recorded on the second occasion (angling), and all recaptures were recorded from subsequent electric fishing occasions. Eleven of 16 individuals recaptured with electric fishing were originally captured by angling, indicating that individuals were susceptible to both gear types. However, electric fishing captured larger bass than angling (Figure 2; Kolmogorov–Smirnov test: D = 0.221, df = 363, P < .001). The largest smallmouth

TABLE 1 Settings of simulation scenarios used to assess performance of the abundance estimation model (M_t model) when two groups of individuals were targeted by different sampling gears, but data from both gears were analysed as a single data set

Scenario	Abundance of each group	Range of capture probability for each group	Posterior mean estimate of total abundance	Per cent upward bias
A	1,000/1,000	0.10-0.30/0.10-0.30	2,364	18.2
B	1,000/1,000	0.10–0.20/0.20–0.30	2,164	8.2
C	1,000/2,000	0.10–0.30/0.10–0.30	3,609	20.3
D	1,000/2,000	0.10–0.20/0.20–0.30	3,268	8.9
E	1,000/2,000	0.20–0.30/0.10–0.20	3,422	14.1
F	3,000	0.20–0.30	3,068	2.3

Control scenario (F) assumed a single group of individuals. For each of 1,000 iterations per scenario, capture probabilities were randomly drawn from a uniform distribution for each sampling occasion. Per cent upward bias was calculated as posterior mean abundance estimated divided by known abundance of two population groups (2,000 or 3,000).

TABLE 2 Number of smallmouth bass (>200 mm TL) captured, marked and recaptured on each sampling occasion in 2015

Occasion	Date	Gear	Captured	Marked	Recaptured
1	20 October	Angling	90	90	0
2	22 October	Angling	85	85	0
3	29 October	Electric fishing	57	52	5
4	4 November	Electric fishing	76	69	7
5	11 November	Electric fishing	29	25	4
		Total	337	321	16

bass captured with angling was 416 mm TL, and the largest one captured with electric fishing was 520 mm TL.

The time-varying model (M_t) was the top ranked of the three abundance models, and no competing models were identified based on ΔDIC values; the next supported model (M_0) had a ΔDIC = 736.77 relative to the top-ranked model (Table 3). Capture probabilities differed by sampling occasion and generally were low. The M_t model indicated that mean capture probabilities were 0.040 during both angling occasions and 0.025, 0.034 and 0.013 for three electric fishing occasions. Based on model M_t, 2,380 individuals (95% CI: 1,578–3,693) >200 mm TL were estimated to be present in the 4.2-km section (Table 3).

3.2 | Simulations

Abundance was consistently overestimated in all five scenarios in which two different groups of individuals targeted by two different gears were simulated but analysed simultaneously as if they were a single group (Figure 3). Posterior mean abundance was overestimated by 8–20% among scenarios, with the most biased estimates observed in the scenario with unequal abundance (1,000 and 2,000 individuals) and equal capture probability of both groups (0.1–0.3). The control scenario without two groups only slightly (2%) overestimated abundance (Table 1).

4 | DISCUSSION

The dual-gear sampling technique was employed in this study to accommodate varying flow conditions in a regulated river. The top-ranked abundance model (M_t) indicated that capture probability varied by sampling occasion and ranged between 1.3 and 4.0%. Varying capture probability highlighted the importance of quantifying it for unbiased abundance estimates and lends support for field-intensive mark–recapture surveys. In the Broad River, the range of capture probabilities observed (1.3–4.0%) suggested that catch could vary threefold depending on sampling conditions even if abundance remains unchanged. In addition, heterogeneity in capture probabilities, when not accounted for in the closed-population abundance estimates, results in lower estimates of capture probabilities (*P*) and thus higher estimates of abundance (*N*; Otis et al., 1978; Kéry & Schaub, 2012). Abundance estimate of smallmouth bass with the M_0 model (posterior mean = 2,933) was approximately 23% higher than that with the M_t model (posterior mean = 2,380). Thus, modelling temporal heterogeneity in capture probabilities was important to reduce bias in abundance estimates.

Simulations suggested that the abundance estimate of smallmouth bass in the Broad River was likely an overestimate to an unknown but modest degree. This result is not surprising because the presence of groups of individuals with different catchability, when not explicitly accounted for, lowers mean capture probabilities and biases abundance estimates upward, similar to the failure to account for temporal heterogeneity in capture probabilities described above. Electric fishing selected for larger individuals than angling, and this result was probably due to a combination of factors. First, electric fishing inherently selects for larger individuals (Dolan & Miranda, 2003; Ruetz et al., 2007). Second, smallmouth bass select different habitat types through ontogeny with larger fish occupying deeper habitats (Probst, Rabeni, Covington & Marteney, 1984; Lawrence, Olden &

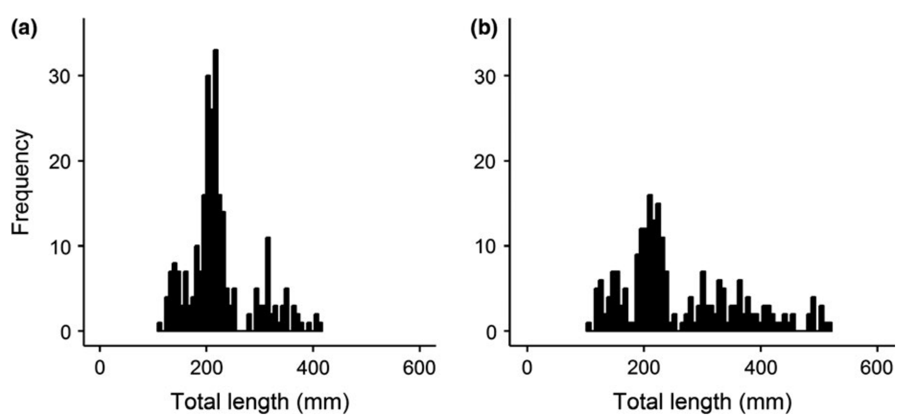

FIGURE 2 Length frequency histograms of all smallmouth bass captured using angling (a) and electric fishing (b)

TABLE 3 Estimated abundance of smallmouth bass >200 mm total length, in a 4.2-km section of the Broad River, and deviance information criterion (DIC) rankings of three abundance models

Model	Estimate (95% CI)	DIC
M_t	2,380 (1,578-3,693)	14,495.86
M_0	2,933 (1,868-3,264)	15,232.63
M_g	2,792 (1,755-4,520)	17,849.65

M_0, constant capture probability; M_t, occasion-varying capture probability; M_g, gear-varying capture probability.

Torgersen, 2012). It is perhaps possible that different gears sample different habitat types more effectively (Anderson, 1995; Buckmeier & Schlechte, 2009). Third, angling occurred earlier in sampling season when river discharge was lower and anglers used light-weight lures. It is still noteworthy that 11 of 16 individuals recaptured with electric fishing were originally captured by angling. Regardless of the mechanism, it is reasonable to suspect that there were two partially overlapping groups of individuals with different catchability during the mark–recapture study period. Scenarios simulated two completely independent groups of individuals targeted by different gears to assess the potential extent of bias in abundance estimates. Thus, the upward bias in abundance estimate in the Broad River should be smaller than those considered in the simulation scenarios (i.e. 8–20% overestimation). Violations of the population closure assumption are another potential source of upward bias in abundance estimation. Tag loss, whether due to emigration or mortality, results in lower capture probabilities and thus upward bias in abundance (Otis et al., 1978). This possibility cannot be excluded as a factor contributing to upward bias of the smallmouth bass abundance estimate in this study, although telemetry data suggested that emigration would most likely have a minor effect on the abundance estimate (Mycko, 2017).

Potential upward bias with selectivity of two gears or tag loss should not immediately discredit its application in population monitoring. Assessment of spatial and temporal trends in abundance is of great interest in fisheries management and conservation. In the case of the Broad River, assessing a temporal trend of smallmouth bass abundance was needed to inform their potential impact on native fishes, future stocking efforts and angler success. Trend assessment ideally is conducted with unbiased estimates, but biased estimates could identify such trends accurately as long as the magnitude and direction of bias remain consistent (Rosenberger & Dunham, 2005). Standardised sampling protocols should be designed to minimise variation in bias when unbiased estimates are not obtainable.

No PIT-tagged fish were recaptured by angling on the second sampling occasion (the only occasion on which fish could be recaptured by angling) and all recaptures were by electric fishing on subsequent occasions. The second occasion took place just 2 days after the first occasion. This short interval might have led to "trap shyness" of individuals on the second occasion that were collected by angling

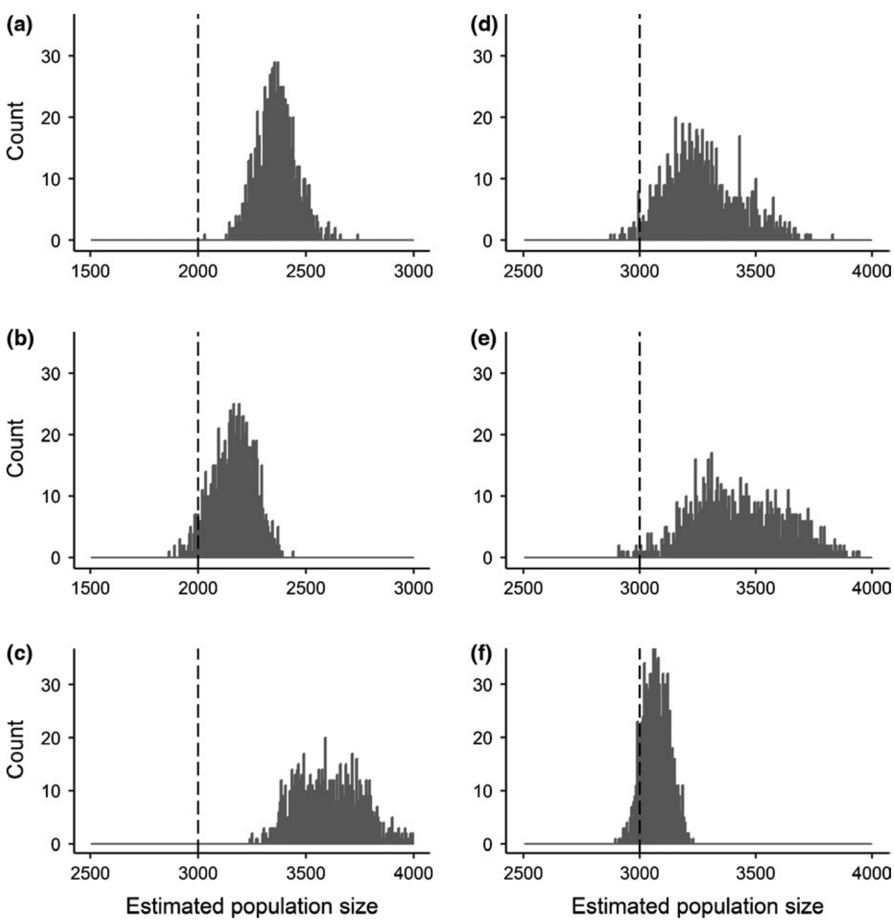

FIGURE 3 Distributions of posterior means of 1,000 replicate simulations when capture probability varied among sampling occasions (M_t models) for six different scenarios: (a) Equal capture probability in two equal-size groups of 1,000 individuals each. (b) Unequal capture probability in two equal-size group. (c) Equal capture probability in unequal-size groups. (d) Higher capture probability in a larger group. (e) Higher capture probability in a smaller group. (f) A control simulation estimating a single population size of 3000 individuals. See Table 1 for a description of each scenario. Dotted vertical lines indicate the true total abundance (i.e. sum of known abundance of two groups in Scenarios a-e and known abundance of the single group in Scenario f)

on the first occasion. Post-angling behavioural effects and tendencies to evade areas of prior capture have been reported for largemouth bass, *Micropterus salmoides* Lacepède (Thompson et al., 2008). If such effects of angling exist for smallmouth bass, timing of angling would need to be carefully assessed, for example, by limiting angling to the first occasion or ensuring a longer interval between angling occasions.

Estimating abundance in large water bodies such as rivers and lakes remains challenging. Standardised protocols are less common in such habitats, particularly in temporally heterogeneous rivers (Bonar, Hubert & Willis, 2009). Efficient sampling is particularly important for mark–recapture methods. Kéry and Royle (2016) state that the first law of capture–recapture methods is that "things become more difficult when *P* (capture probability) gets small (p. 246)" because data represent a smaller portion of the population (i.e. the truth). Low capture probabilities can be partly addressed by increasing sampling effort; in this study, more anglers or electric fishing boats could be used to capture more smallmouth bass. Improvements in analyses of abundance are an important area for further studies, and innovative analytical approaches can lead to less biased estimates of fish abundance (Korman, Schick & Mossop, 2016; Mollenhauer & Brewer, 2017). Simultaneous analysis of multiple-gear data is becoming more common (Arab et al., 2008; Carrier, Rosenfeld & Johnson, 2009) but warrants further investigations. Integrating multiple gears can be an effective and probably a needed method for assessing abundance in spatially and temporally heterogeneous habitats.

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