Transactions of the American Fisheries Society 147:729–739, 2018

ISSN: 0002-8487 print / 1548-8659 online
DOI: 10.1002/tafs.10061

ARTICLE

Can Single-Pass Electrofishing Replace Three-Pass Depletion for Population Trend Detection?

R. Daniel Hanks
Department of Forestry and Environmental Conservation, Clemson University, 261 Lehotsky Hall, Clemson, South Carolina 29634-0317, USA

Yoichiro Kanno*
Department of Fish, Wildlife, and Conservation Biology, Graduate Degree Program in Ecology, Colorado State University, 1474 Campus Drive, Fort Collins, Colorado 80523-1474, USA

Jacob M. Rash
North Carolina Wildlife Resources Commission, 645 Fish Hatchery Road, Marion, North Carolina 28752, USA

Abstract
Since changes in climate and land use operate at broad spatial scales, efficient monitoring of temporal trends in fisheries resources over large geographic areas is vital to appropriate management. We compared the statistical power of single- versus three-pass electrofishing surveys in detecting temporal trends of age-1 and older Brook Trout *Salvelinus fontinalis* populations in western North Carolina. Empirical estimates of abundance and capture probabilities were obtained from annual three-pass depletion surveys at 14 headwater stream sites between 2012 and 2017. The CVs in abundance averaged 26% (SD = 14.2%) across study sites, and the mean capture probability per pass was 0.72 (range = 0.57–0.84, SD = 0.09). Captures from single-pass sampling and abundance estimates from three-pass removal sampling were highly correlated ($r^2 = 0.98$). In simulations, under the range of years sampled (5–25 years) and annual declines (2.5–7.5%) considered, the power to detect temporal trends was similar (Δ power < 0.1) between the two methods when five or more sites were monitored. An additional set of simulations with varying capture probabilities demonstrated that differences in power between the two methods increased as mean capture probabilities decreased (0.8, 0.5, and 0.2) accompanied by larger variation in capture probabilities among the samples, indicating results obtained with Brook Trout populations in North Carolina might not be applicable to other habitat types or species. Variation in fish abundance did not affect the difference in power between the two methods. Single-pass electrofishing surveys can be an efficient survey method to monitor temporal population trends for habitat types and species characterized with high capture probabilities and low variation among samples. However, single-pass data would not typically allow for inferences of capture probabilities and thus abundance. This can be problematic when environmental factors vary and when data sets collected using different protocols are compared. This trade-off should be carefully considered when designing monitoring programs.

Fisheries managers are charged with understanding temporal trends of regional resources to inform management actions (Larsen et al. 2001; Gerow 2007; Wagner et al. 2007). A major objective of state monitoring programs is to detect declining or increasing population abundance over time, and monitoring programs need to be designed to confidently detect temporal trends in the most efficient manner given the limited resources

---

*Corresponding author: kanno@colostate.edu
Received October 31, 2017; accepted March 19, 2018

available. Managers typically monitor a set of sites when making inferences on temporal trends of regional resources (Ham and Pearsons 2000; Dauwalter et al. 2010), and such a task is becoming more important as large-scale drivers such as changes in climate and land use can affect fisheries resources at a broad spatial scale (Kanno et al. 2015; Midway et al. 2016). A critical issue when designing state-wide monitoring programs is to determine adequate sampling effort needed at local sites, which will then dictate the number of sites to be surveyed (Smith and Jones 2008; Fischer and Paukert 2009). Specifically, managers are challenged with making decisions on allocating effort within versus among local sites. In such cases single-pass electrofishing would increase the number of sites at the cost of collecting less intensive data, and multipass electrofishing would allow collection of more intensive data at fewer sites.

Multipass depletion electrofishing is commonly used when estimating population abundance, particularly in smaller waters (e.g., headwater streams) where a section of habitat can be physically isolated with relative ease. Fewer individuals captured through successive passes inform capture probabilities, and thus abundance, based on assumptions of population closure, equal catchability of individuals, and constant effort and catchability among passes (Riley and Fausch 1992; Jones and Stockwell 1995; Peoples and Frimpong 2011). Several analytical methods for depletion data have been developed for abundance estimates (e.g., DeLurey 1947; Zippin 1956; Carle and Strub 1978), and the importance of inferring capture probabilities is widely recognized in wildlife and fisheries surveys (Royle 2004; Dail and Madsen 2011). However, multipass depletion surveys require intensive effort and may limit the number of sites that can be surveyed (i.e., spatial coverage). As an alternative to multipass electrofishing, single-pass electrofishing has been used as a reliable index of population abundance in some cases (Jones and Stockwell 1995; Kruse et al. 1998) but not in others (Thompson and Rahel 1996). Less site-level effort with single-pass electrofishing is suitable for sampling a greater number of sites. However, the single-pass method cannot typically infer capture probabilities of individuals, which can vary over time and space (Letcher et al. 2015; Mollenhauer and Brewer 2017), making temporal and spatial comparisons of single-pass data challenging.

Statistical power analysis assesses the likelihood that an effect will be detected, given it exists, with different levels of sampling effort. Power is equal to one minus the type II error, where the type II error occurs when failing to reject the null hypothesis when it is false (Gotelli and Ellison 2004:100–102; Brown and Guy 2007). Managers should be wary of making decisions based on the failure to reject null hypotheses unless power is high (e.g., 0.80) (Peterman 1989). Power analysis to detect temporal trends in abundance is common in fisheries, and power to detect such trends is influenced by the magnitude of a trend, numbers of sites and years sampled, distribution of effort across time and space (i.e., sampling designs), temporal variation in abundance (i.e., population stability over time), and the level of the type I statistical error (Urquhart and Kincaid 1999; Larsen et al. 2001; Wagner et al. 2007; Dauwalter et al. 2009, 2010). Despite an array of studies comparing the power of fisheries monitoring efforts and designs, comparisons of power between single- and multipass methods in detecting a temporal trend of fish population abundance have rarely been examined (Wagner et al. 2014).

In this study, we compared the power of single- versus three-pass electrofishing methods to detect temporal declines of Brook Trout *Salvelinus fontinalis* populations in western North Carolina, based on empirical estimates of abundance and capture probabilities derived from a state monitoring program. Power of the two methods was assessed with various levels of population declines and numbers of sites and years sampled. Additionally, to assess whether the results from Brook Trout in North Carolina streams are generally applicable to other situations (e.g., different species or habitat characteristics), we evaluated the power to detect declining population trends with a range of detection probability and abundance.

METHODS

Study Area

This study was based on a subset of the state-wide monitoring data on Brook Trout collected by the North Carolina Wildlife Resources Commission (NCWRC). Since 2012, fisheries biologists at NCWRC have been using the current sampling protocol aimed at monitoring spatial and temporal patterns of trout populations in the state. Sites are sampled with either a single- or three-pass electrofishing method.

We selected 14 fixed sites sampled annually with three-pass depletion surveys between 2012 and 2017 to infer spatial and temporal variation in abundance and capture probabilities among Brook Trout populations. These stream sites were dispersed geographically across the major river basins within North Carolina's mountainous geographic region. State fisheries biologists identified streams with characteristics (i.e., stream type, available habitat, species composition) representative of coldwater resources within each river basin. Additionally, streams containing Brook Trout populations, including those that occur in sympatry with other trout species, were selected to increase our understanding of these native trout resources. Brook Trout data were used for this study because it was the only native salmonid to the region, the most common species in the data set, and its mean capture probability (0.72, SD = 0.09) was similar to that of Brown Trout *Salmo trutta* (0.69, SD = 0.11) and Rainbow Trout *Oncorhynchus mykiss* (0.72, SD = 0.10). The mean elevation of sample sites was 901 m (range = 566–1,376 m). Stream widths and lengths averaged 3.9 m (range = 2.1–5.7 m) and 167 m (range = 93–209), respectively (Table 1).

Field Sampling

Study sites were sampled between July and September using a depletion method following protocols detailed by the Southern Division of the American Fisheries Society Trout Committee (1992). Briefly, block nets (1.2 × 7.6–15.2 m, with 6.4-mm mesh) or natural barriers were used at upstream and downstream boundaries of each study site to help meet the assumption of population closure during depletion sampling. Three successive electrofishing passes were conducted in an upstream direction using one backpack shocker (model 12-B, Smith-Root, Vancouver, Washington) equipped with 1.8-m anode and cathode poles and 290-mm diameter rings (setting controls: 70 Hz, 8 ms, and 800 V) and a netter per every 3 m of average wetted stream width. Fish captured were temporarily housed in live cages located upstream from the study site. Fish were weighed (g), measured (mm, TL), and returned alive to the study site following completion of all passes. Individual Brook Trout were segregated into two size-groups based on clear gaps on length-frequency histograms: young of the year ($\leq$90 mm TL), and age 1 and older (>90 mm TL) (Besler 2006; Habera et al. 2009, 2010).

Statistical Analysis

Our goal was to compare the statistical power of single- versus three-pass electrofishing methods to detect a temporal decline of Brook Trout populations at various levels of population declines, numbers of sites, and numbers of years. While any number of passes (two, three, four, or more) could be simulated, we focused on the comparison of single- versus three-pass electrofishing in this study because they are the most commonly employed protocols in Brook Trout monitoring programs in the southern range of this species. To this end, empirical estimates of spatial and temporal variation in capture probabilities and abundance were derived based on capture data collected at the 14 study sites between 2012 and 2017. The empirical estimates were then used to simulate data sets, on which statistical power to detect a linear declining trend over time was assessed. Finally, to assess the applicability of Brook Trout power analysis, we evaluated the power to detect declining population trends with a range of capture probabilities and abundance. The following analyses were based on captures of age-1 and older Brook Trout. We examined the statistical power to detect temporal trends for both age-1 and older fish and young of the year but found results to be nearly identical between the two size-classes as far as the differences in power between single- and three-pass methods were concerned.

TABLE 1. Site characteristics for the 14 North Carolina stream sites used to obtain empirical estimates of abundance and capture probabilities for Brook Trout.

Site	Mean day of the year sampled (range)	Number of times sampled	Elevation (m)	Mean stream width (m)	Mean stream section length (m)	Longitude (°)	Latitude (°)
Boone Fork, lower	210 (210–210)	5	1,147	5.1	195	−81.786163	36.119033
Boone Fork, upper	214 (208–226)	2	1,231	4.4	189	−84.01477	35.33079
Cove Creek, lower	215 (205–223)	5	868	4.3	196	−83.983015	35.363319
Cove Creek, upper	215 (205–223)	5	958	3.7	209	−83.982697	35.359748
Elk Hollow Branch, lower	216 (215–216)	5	1,152	3.8	203	−83.99466	35.36251
Elk Hollow Branch, upper	219 (215–226)	4	1,317	3.4	197	−83.995468	35.358826
Garden Creek, lower	209 (209–209)	1	476	4.8	196	−81.777383	36.121005
Garden Creek, upper	216 (209–228)	1	566	4.9	195	−82.8277	35.30292
Gragg Prong, upper	220 (216–227)	4	836	3.9	199	−82.81772	35.29448
Sand Branch, lower	249 (246–256)	4	1,331	2.5	100	−82.057974	36.105121
Sand Branch, upper	249 (246–256)	5	1,331	2.7	100	−82.045108	36.099412
Whigg Branch	251 (245–256)	5	1,205	4.2	95	−81.103255	36.408302
Wolf Laurel, lower	250 (245–255)	4	1,350	3.1	97	−81.08684	36.39299
Wolf Laurel, upper	248 (245–255)	5	1,376	4.1	93	−81.815197	36.058241

Empirical estimates of capture probabilities and abundance.— Three-pass capture data were analyzed using the Carle–Strub method (Carle and Strub 1978) to infer capture probabilities and abundance of annual samples at study sites. The Carle–Strub depletion method is a modification of the Zippin (1956) depletion method, which fails to provide abundance estimates when captures from passes two or three are higher than the previous pass. The Carle–Strub depletion method overcomes this deficiency by substituting total catch for the estimated abundance when captures in passes two or three is higher than the previous pass. The Carle–Strub method was chosen in this study because the Zippin method had an estimation problem with some samples in our simulations, although, we noted that abundance estimates were identical for simulated data when both the Zippin and Carle–Strub methods worked. The Carle–Strub method was implemented using the FSA package (Ogle 2017) in R (R Core Team 2016).

Simple linear regression was used to assess whether there was a significant relationship between first-pass captures and abundance estimates obtained from the three-pass depletion method using the Carle–Strub method. Variation of Brook Trout population abundance through time at each site was characterized using CV (100 × SD/mean), which is commonly used as a measure of animal population stability (Grossman et al. 1990; Gaston and McArdle 1994).

Power analysis required empirical variation in capture probabilities and abundance among sites and years. To this end, capture probabilities and abundance estimated using the Carle–Strub method were used as response variables in linear mixed-effects models. Estimated Brook Trout abundance (N) at site i in year t was modeled with a Poisson distribution:

$$N_{i,t} \sim \text{Poisson}(\lambda_{i,t}), \tag{1}$$

$$\log(\lambda_{i,t}) = \mu + \alpha_i + \beta_t + \log(\text{site length}/200), \tag{2}$$

$$\alpha_i \sim \text{Normal}(0, \sigma_\alpha^2), \tag{3}$$

$$\beta_t \sim \text{Normal}(0, \sigma_\beta^2), \tag{4}$$

where μ was the overall mean abundance, α_i was a random site effect with variance of σ_α^2, and β_t was a random year effect with variance of σ_β^2. An offset term was included as the logarithm of study site length (m) standardized by 200 m to account for varying lengths among study sites (Table 1).

Capture probability (p) at site i in year t, obtained from the Carle–Strub method, was modeled on a logit scale with a mixed-effects approach and included one covariate, which we hypothesized to affect capture efficiency:

$$\text{logit}(p_{i,t}) = \gamma + \delta \times (\text{stream width}_{i,t}) + \varepsilon_{i,t}, \tag{5}$$

$$\varepsilon_{i,t} \sim \text{Normal}(0, \sigma_\varepsilon^2), \tag{6}$$

where γ is the overall mean capture probability and δ is the coefficient of stream width (m) at site i and year t, and $\varepsilon_{i,t}$ is the residual error with variance equal to σ_ε^2. Stream width was standardized by mean divided by SD. We hypothesized that capture efficiency could be negatively affected with increasing width as fish have more space to escape from the electrical field (Wagner et al. 2014).

Power analysis for Brook Trout populations.— Three-pass capture data were simulated based on empirical estimates of capture probability and abundance, with the addition of percent annual population declines, in order to assess the statistical power to detect these declines under varying sampling effort levels (i.e., number of years, sites, and passes). Following Dauwalter et al. (2009), an exponential growth model was used in which the population was assumed to decline annually with a constant rate over time for each site (r_i), and equation (1) above was modified:

$$\log(N_{i,t}) = \mu + \alpha_i + \beta_t + \log(1 + r_i) \times t, \tag{7}$$

where t continues to represent year of survey. Annual rate of decline was assumed to vary among sites (r_i) because it is unlikely that all sites in a study region would undergo a constant rate of annual decline. With varying rates of decline, an interaction between sites and years was implicitly incorporated in equation (7).

Capture probability ($p_{i,t}$) at site i in year t was simulated based on equations (5) and (6), where values of stream width were randomly drawn from a normal distribution with mean = 0 and SD = 1 due to mean standardization. Finally, three-pass capture data ($y_{i,t,j}$) at site i, year t, and pass j were simulated following binomial distributions based on successive depletions of individuals through passes:

$$y_{i,t,1} \sim \text{Binomial}(N_{i,j}, p_{i,t}), \tag{8}$$

$$y_{i,t,2} \sim \text{Binomial}(N_{i,j} - y_{i,t,1}, p_{i,t}), \tag{9}$$

$$y_{i,t,3} \sim \text{Binomial}(N_{i,j} - y_{i,t,1} - y_{i,t,2}, p_{i,t}). \tag{10}$$

Data were simulated with various levels of mean percent annual population declines (2.5, 5.0, and 7.5%), numbers of sites (1, 5, 10, and 15 sites), and numbers of years (5, 10, 15, 20, and 25 years). Annual rates of population declines were set to vary among study sites with a mean value ±2.5%. For example, a site was assigned an annual rate of decline drawn randomly between 0 and 5% when the mean rate of decline was 2.5% annually ($r_i \approx$ Uniform [0, −0.05] in equation 7). All unique combinations of levels of percent annual declines, numbers of sites, and years were simulated, and data were generated 1,000 times for each unique combination. The primary objective was to assess whether statistical power to detect annual declines differed between single- versus three-pass electrofishing methods. To this end, first-pass captures and abundance estimates based on the Carle–Strub method were modeled as a function of year (a numeric vector ranging consecutively from 1 to the number of years simulated) with random year and site effects for multiple-site simulations and with only random year effects for single-site simulations. Power was defined as the proportion of 1,000 simulations in which the coefficient of numeric year was significantly negative at $P = 0.05$. The R code for simulations can be found in the Supplement available in the online version of this article.

Influence of capture probability and abundance on power.— Statistical power may depend on capture probability and abundance (Zipkin et al. 2014; Kéry and Royle 2016). A final set of simulations was conducted to assess how the power to detect declining population trends with single- versus three-pass methods is influenced by a range of capture probabilities and abundance for two reasons. First, the depletion method is a commonly used sampling technique, and its application is not limited to Brook Trout populations in southern Appalachian (North Carolina) streams. However, capture probability and abundance vary by environmental conditions and species (Rosenberger and Dunham 2005; Hense et al. 2010), and the generality of our empirical data needs to be assessed in broader conditions. Second, the depletion method is known to overestimate capture probability, which leads to an underestimation of abundance (Rosenberger and Dunham 2005; Habera et al. 2010). Meyer and High (2011) found that the abundance of Brook Trout and Rainbow Trout (>10 cm TL) was underestimated by 17–25% depending on the number of passes in Idaho streams, and this bias was caused due to violations of equal capture probabilities among passes. The modest degree of potential bias was addressed by simulating a range of capture probability and abundance and assessing sensitivities of power to these different settings.

For this additional set of simulations, we evaluated all combinations of a 2.5, 5.0, and 7.5% annual decline at 5 and 15 sites for 5, 10, 15, and 20 years. Three levels of mean and two levels of variation (SD) of capture probabilities were simulated. Mean capture probabilities per pass were specified as high (0.8), medium (0.5), and low (0.2) (Gerow 2007); for example, $\gamma = \text{logit}\ (0.8)$ for high capture probability in equation (5). Empirical estimates of capture probabilities for Brook Trout were between the high and medium values but closer to the former (see Results) and thus acknowledges that the depletion method has a tendency to overestimate capture probabilities (Rosenberger and Dunham 2005; Meyer and High 2011). With these three mean capture probabilities, 99% of individuals would, on average, be captured after three passes under the high capture probability (i.e., $1 - [1 - 0.8]^3$), 88% under the medium capture probability, and 49% under the low capture probability. Variation of capture probabilities was set at two levels by supplying the SD of 0.419 from empirically derived capture probabilities based on the Carle–Strub method to σ_ε of equation (6) and supplying 1.5 times the SD value of 0.628. Finally, local abundance at sites was halved or doubled to simulate scenarios where half (100 m) or twice (400 m) as long a stream length was sampled on average where capture probability and SD were set to 0.73 and 0.472, respectively. Power was similarly defined as the proportion of 1,000 simulations in which the coefficient of numeric year was significantly negative at $P = 0.05$.

To facilitate interpretation of power analysis throughout, we report results of single- versus three-pass methods by focusing on the following two standards. First, we considered that a difference in power of >0.1 (10%) is large enough to provide support for three-pass methods over single-pass methods. Thus, we highlight simulation settings in which power differed by >0.1 between the two methods. Second, the number of years required to achieve power of 0.8 was compared between the two methods for selected simulation settings. A value of 0.8 is routinely used as a criterion of strong statistical power (Peterman and Bradford 1987; Thomas 1997; Brown and Guy 2007).

RESULTS

Empirical Estimates of Capture Probabilities and Abundance

Abundance estimates from three-pass removal sampling were highly correlated with first-pass captures ($r^2 = 0.98$; Figure 1). Estimated abundance of age-1 and older Brook Trout standardized at 200 m averaged 48 individuals (SD = 35) and ranged from 13 to 113. The CV for age-1 and older Brook Trout abundance averaged 26% (SD = 14.2%) across study sites and ranged from 10% to 54%. Spatial variation in abundance (random site effect: $\sigma_\alpha^2 = 0.33$ in equation 3) was larger than temporal variation (random year effect: $\sigma_\beta^2 = 0.09$ in equation 4). Mean capture probability per pass was 0.72 (SD = 0.09) and ranged from 0.57 to 0.84 (Table 2). Capture probability was negatively correlated with stream width ($\delta = -0.14$ in equation 5; $P = 0.03$) and the model residual error (σ_ε^2) was 0.46 in equation (6).

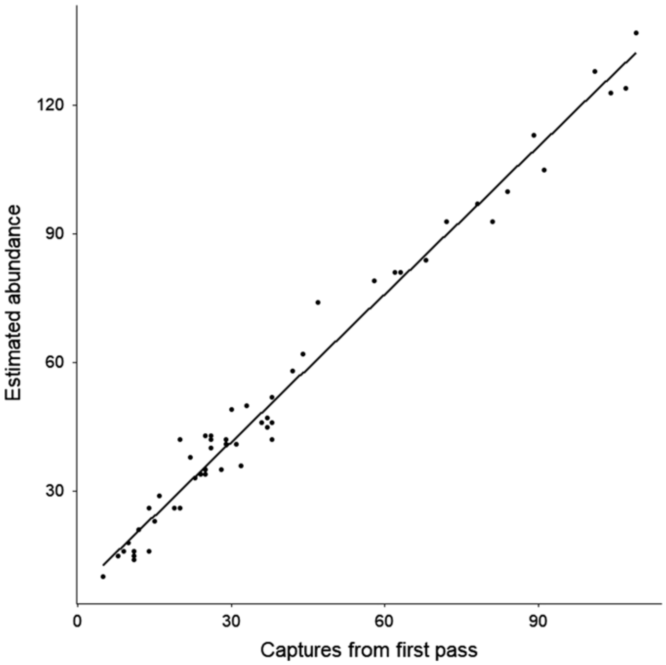

FIGURE 1. Abundance estimates plotted against capture data from the first pass for Brook Trout ($P < 0.001$, $r^2 = 0.98$). All sites and years are shown.

Power Analysis for Brook Trout Populations

Power to detect a declining population trend was similar between three-pass depletion and single-pass sampling (Δ power < 0.1) in all but three of the 120 simulation settings (Figure 2). These three settings occurred when only a single site was monitored for 10 years with an annual decline of 2.5% (Δ power = 0.12), 5.0% (Δ power = 0.16), and 7.5% (Δ power = 0.11). The difference in the power of single- versus three-pass methods diminished as the number of sites increased and became indistinguishable when 15 sites were monitored (Δ power < 0.05), indicating that the differences in the power of single- versus three-pass method depended on the number of sites monitored (Figure 2).

As the number of sites increased, the number of years required to achieve a power of 0.8 became more similar between single- and three-pass methods. When a single site was monitored with an annual decline of 5.0%, the number of years required to reach a power of 0.8 was 11 years with three-pass and 13 years with single-pass methods. When 15 sites were monitored for 10 years with an annual decline of 5.0%, 8 years were required to achieve a power of 0.8 for both three-pass and single-pass methods. Not surprisingly, power typically increased with the number of years and the magnitude of annual declines. Additionally, when at least five sites were monitored, 15 years of monitoring were adequate to achieve a power of 0.8, regardless of sampling methods and the magnitude of annual decline (Figure 2).

Influence of Capture Probability and Abundance on Power

The difference in power to detect a trend between single- and three-pass sampling methods was most pronounced when mean capture probability was low or variance was large. Of the 180 simulation settings with varying annual trends and number of sites and years considered, 28 had Δ power > 0.1 between three- and single-pass methods. Half of these scenarios were when mean capture probability was low (0.2), and the other half had

TABLE 2. Mean capture data (number of fish) of the first pass (pass 1), mean population estimate, %CV for the estimated abundance standardized at 200 m, and mean capture probability (SD) for Brook Trout in each of the 14 North Carolina stream sites. NE = not able to estimate.

Site	Mean pass 1	Mean abundance	Estimated abundance CV (%)	Mean capture probability (SD)
Boone Fork, lower	7	13	28	0.62 (0.05)
Boone Fork, upper	67	89	45	0.72 (0.09)
Cove Creek, lower	22	32	58	0.69 (0.08)
Cove Creek, upper	74	90	6	0.84 (0.04)
Elk Hollow Branch, lower	12	18	21	0.74 (0.07)
Elk Hollow Branch, upper	93	114	20	0.79 (0.05)
Garden Creek, lower	11	14	NE	NE
Garden Creek, upper	30	43	NE	NE
Gragg Prong, upper	77	94	13	0.79 (0.04)
Sand Branch, lower	32	45	9	0.73 (0.05)
Sand Branch, upper	33	42	16	0.8 (0.04)
Whigg Branch	15	24	51	0.57 (0.01)
Wolf Laurel, lower	27	41	13	0.7 (0.12)
Wolf Laurel, upper	31	46	13	0.68 (0.06)

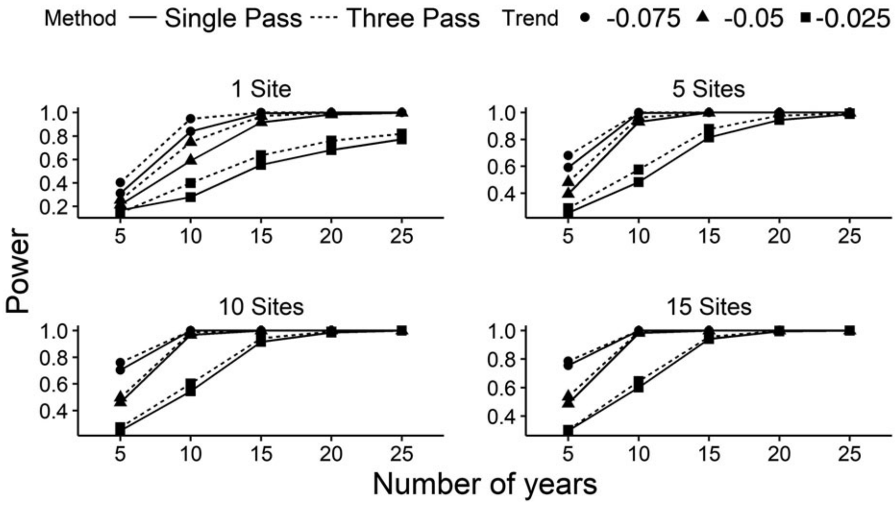

FIGURE 2. Power of single- and three-pass methods to detect annual declines of 2.5, 5.0, and 7.5% in Brook Trout abundance given combinations of sampling for 5–25 years and 1–15 sites (α = 0.05).

medium capture probability (0.5). More scenarios were under large variance in capture probability (0.628) than small variance (0.419). Power between single- and three-pass methods differed as much as 0.23, when populations were monitored at five sites for 15 years with an assumed annual decline of 2.5% with medium mean capture probability and large variance. Under this setting, the number of years required to reach a power of 0.8 was 15 years with the three-pass method versus 20 years with the single-pass method. Using the same number of sites (five) and annual decline (2.5%) but with high mean capture probability and small variance as settings, 13 years were required to reach a power of 0.8 for both single- and three-pass methods (Figure 3).

Changes in abundance had little influence on Δ power to detect a trend between single- and three-pass methods. Of the 180 simulation settings with varying annual trends and number of sites and years considered, 12 had Δ power > 0.1 between single- and three-pass methods. Four, two, and six of these scenarios were from half the original abundance, original abundance, and double the original abundance, respectively. Power between single- and three-pass methods differed as much as 0.17, when populations were monitored at one site for 10 years with an assumed annual decline of 5.0%, and abundance was doubled. Under this setting, the number of years required to reach 0.8 power was nine for three-pass and 13 for single-pass methods. Yet, using the same number of sites (one) and annual decline (5.0%) with half the original abundance. The number of years required to reach 0.8 power was 14 for three-pass and 15 for single-pass methods. Additionally, as the number of sites sampled increased, the difference in power between single- and three-pass sampling was reduced for each level of abundance (Figure 4).

DISCUSSION

The statistical power required to detect annual population declines of age-1 and older Brook Trout abundance between single- and three-pass electrofishing methods differed as a function of the number of sites. Specifically, the difference in power between the two methods diminished as a greater number of sites were surveyed and power was nearly identical between them (Δ power < 0.1) when five or more sites were surveyed. We surmise that several factors were responsible for the observed simulation outcomes. First and foremost, first-pass capture data was a robust predictor of population abundance in Brook Trout at study sites ($P < 0.001$, $r^2 = 0.98$). Second, the use of stream sites as a random effect facilitated detection of declining population trends over time as more sites were analyzed simultaneously. Third, we let annual rates of decline, with a mean value ±2.5%, vary among study sites. Thus, when only a single site was monitored, the site could have more likely been assigned, by chance, a small percent decline or 0% decline when an annual decline of 2.5% was simulated. As the number of sites increased, the sample populations more likely conformed to the true mean annual rate of decline. Dauwalter et al. (2009) similarly reported the importance of monitoring multiple sites versus a single site in temporal trend detection, although their study did not include comparison of single- versus multipass methods. Our results were also congruent with those of previous studies in that power to detect temporal trends typically increased with the magnitude of an effect

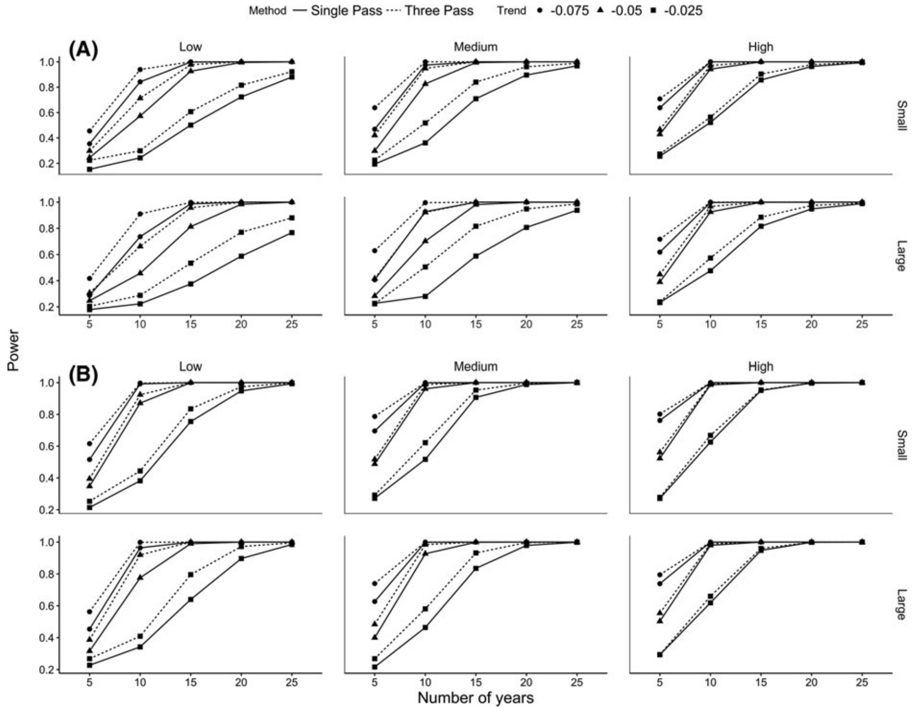

FIGURE 3. Power of single- and three-pass methods to detect annual declines of 2.5, 5.0, and 7.5% in Brook Trout abundance given combinations of three levels of mean capture probability (low = 0.2, medium = 0.5, high = 0.8) and its SD (logit scale; small = 0.419, large = 0.628), sampled for 5–25 years, and for **(A)** 5 and **(B)** 15 sites ($\alpha = 0.05$).

(annual percent declines) and numbers of sites and years, but there were also threshold levels above which increasing sampling effort results in diminishing returns in power (Larsen et al. 2001; Wagner et al. 2007; Dauwalter et al. 2009, 2010; Wagner et al. 2013, 2014).

Abundance estimates based on the depletion method can be biased compared with the mark–recapture method (Peterson et al. 2004; Meyer and High 2011), but we consider that the depletion method is a suitable approach for power analysis of state-level population trend detection. First, bias is typically modest (Meyer and High 2011), and this is particularly true in our study sites characterized with low physical habitat complexity (e.g., lack of log jams). Habera et al. (2010) reported a small negative bias (−12%) in an estimate of Rainbow Trout abundance using the depletion method in a southern Appalachian stream. Second, bias is consistently towards underestimation. The assessment of population trends should not be greatly affected as long the direction and magnitude of bias remains unchanged (Rosenberger and Dunham 2005). Finally, a regional population assessment is based on sampling multiple sites, often across a broad geographical range with limited resources, making the mark–recapture method logistically difficult, if not prohibitive.

The range of sites (~15 sites annually) considered in our simulations is routinely surveyed in state monitoring programs. In fact, the NCWRC samples Brook Trout populations at 45 sites annually with a single-pass electrofishing on a 3- to 5-year rotational period, in addition to the 14 fixed sites sampled every year with the three-pass depletion method that were used in this study. Given our simulation results, single- and three-pass methods are likely to achieve similar power to detect temporal trends when this range of sites (45–60 sites) is surveyed annually. Thus, the single-pass method is likely a preferred choice as it obtains a similar level of power with less effort, and it could allow managers to survey an even greater number of sites with single-pass electrofishing. Maintaining an adequate number of sites will result in more accurate characterization of spatial variability in abundance, and it can be further advantageous because spatial variation in abundance was greater than temporal variation in our Brook Trout data.

Sampling strategies inferred from North Carolina populations may not be universally applicable to other situations or species that are characterized with different capture probabilities. In the additional set of simulations conducted with varying capture probabilities, there were more pronounced differences in power between single- and three-pass methods as the mean capture probability per pass decreased from 0.8 to 0.2, especially when variation in capture probability was larger among samples. Capture

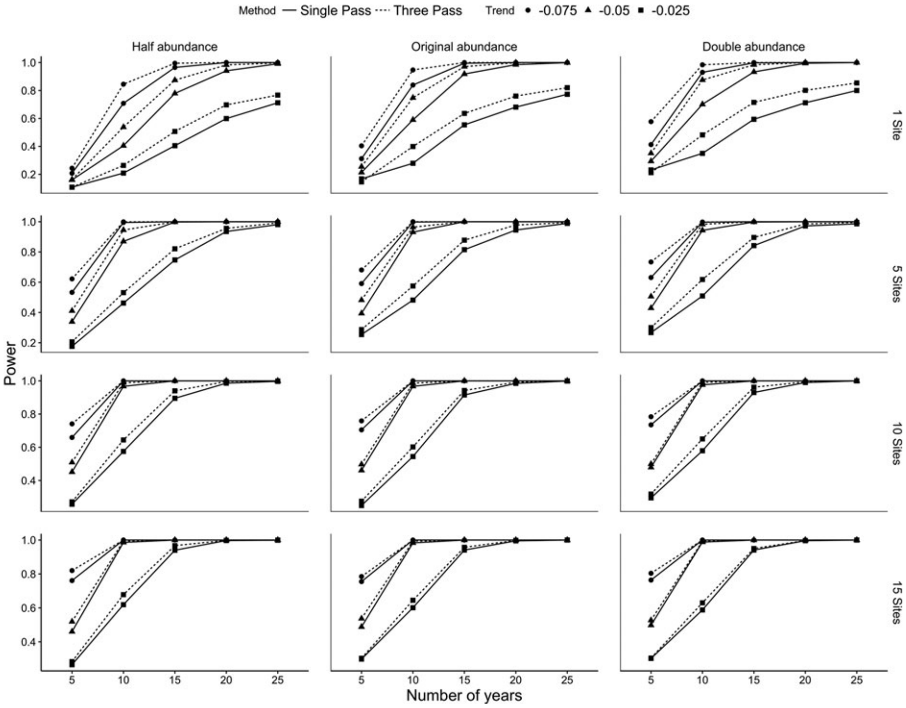

FIGURE 4. Power of single- and three-pass methods to detect annual declines of 2.5, 5.0, and 7.5 in Brook Trout abundance given combinations of three levels of abundance (one-half abundance, original abundance, and double abundance), sampled for 5–25 years and for 1–15 sites (α = 0.05).

probability of electrofishing is influenced by physical (e.g., undercut banks, log jams) and chemical (e.g., conductivity) characteristics (Riley and Fausch 1992; Kruse et al. 1998; Habera et al. 2010). For example, capture probability was negatively influenced by stream width in this study. Riley and Fausch (1992) found that trout abundance was more accurately estimated with increasing capture probability in Colorado streams. Capture probability also differs by species. Lower capture probabilities have been reported when depletion methods were used in Smallmouth Bass *Micropterus dolomieu* (0.35–0.65: Dauwalter and Fisher 2007), Central Stoneroller *Campostoma anomalum* (0.54–0.64: Wiley and Tsai 1983; Hense et al. 2010), and Rock Bass *Ambloplites rupestris* (0.47: Hense et al. 2010) in larger streams. As capture probabilities decrease and become more variable among samples, multipass depletion would be a more effective approach than single-pass electrofishing for increasing power to detect temporal trends. This finding can also provide an insight on population surveys of larger waterbodies in which depletion methods are not logistically feasible but capture probabilities are presumably low. Although depletion methods may not be an option, managers should be aware that the power to detect temporal trends would likely be compromised with a standardized sampling protocol of capture data collection, and other field techniques (e.g., mark–recapture) and innovative analytical approaches might be needed (Mollenhauer and Brewer 2017; Mycko et al. 2018).

Statistical power for trend detection was not dependent on local abundance at sites. Contrary to our results, Wagner et al. (2007) reported moderate gains in power to detect trends in length at age of Walleyes *Sander vitreus* in Michigan and Wisconsin lakes when the number of fish sampled was increased. We intended this simulation analysis to reflect stream lengths (100, 200, and 400 m) sampled for Brook Trout populations in the southeastern USA (Habera et al. 2010; Kanno et al. 2016). Stream lengths required to characterize fish assemblages and species accumulation curves have received more scrutiny (Fischer and Paukert 2009; Kanno et al. 2009) than stream length requirements for abundance assessment. Nevertheless, it should be remembered that mean abundance (μ in equation 2) was the primary factor of varying statistical power, and sampling stream lengths should be determined by considering local abundance among other factors.

Single-pass methods performed as well as three-pass methods in detecting temporal trends when as few as five sites were sampled in North Carolina trout populations, but managers should also acknowledge that single-pass methods have their shortcomings, and trend detection may be one of several other monitoring objectives. Single-pass capture data are an index of abundance, but do not quantify capture probability and thus abundance. Stakeholders and anglers are frequently interested in absolute abundance or density, for which multipass methods provide more robust answers. Single-pass data may also hamper comparisons of trout populations with other regions that use different sampling protocols and thus different capture probabilities. Broad-scale analysis and identification of spatial variation in environmental drivers are becoming increasingly more important and it can be done only by combining data sets. Similarly, single-pass data may make temporal comparisons more challenging. For example, NCWRC implemented a focused effort to collect Brook Trout abundance data between 1989 and 1996, and temporal comparisons with the current data set (2012–2017) were greatly facilitated, because both data sets contained at least a subset of sites sampled with three-pass methods. Choices between single- and multi-pass methods in state monitoring programs should probably not be dichotomous. Hybrid approaches in monitoring designs are frequently employed (Urquhart and Kincaid 1999), and managers need to make judicious decisions on the allocation of effort in statewide sampling protocols. Nevertheless, this study indicates that the single-pass method can be an alternative method to the three-pass depletion method when capture probability is high (e.g., habitats with limited physical characteristics) and is constant temporally and spatially (e.g., sites with stable flow regimes fed by groundwater), and the goal of monitoring programs is primarily one of temporal trend detection of regional resources.

ACKNOWLEDGMENTS

This research was financially supported by the North Carolina Wildlife Resources Commission. Field and technical assistance was provided by D. Arfmann, C. Barden, D. Besler, N. Bolick, A. Bushon, S. Byars, J. Coleman, W. Duncan, J. England, J. Farmer, A. Franks, D. Goodfred, C. Goudreau, S. Gouge, C. Harrill, J. Harrill, K. Hining, T. Hobbs, R. Hoch, K. Hodges, W. Humphries, B. Hyde, A. Isenhower, J. Keepfer, J. Miles, D. Owensby, T. Pate, C. Patton, T. Penland, A. Paoletta, M. Rimmer, L. Stroup, T. Johnson, K. Lingerfelt, S. Loftis, B. Rau, J. Sanders, N. Shaver, T. Wait, P. Wheeler, C. Wood, and J. Yale. We thank N. Hitt and two anonymous reviewers for their constructive comments that improved an earlier version of this manuscript. There is no conflict of interest declared in this article.

REFERENCES

Besler, D. A. 2006. A comparison of trout abundance estimates from mark–recapture and multiple-pass electrofishing in six North Carolina streams. North Carolina Wildlife Resources Commission, Federal Aid in Sport Fish Restoration, Final Report, Raleigh.

Brown, M. L., and C. S. Guy. 2007. Science and statistics in fisheries research. Pages 1–30 *in* C. S. Guy and M. L. Brown, editors. Analysis and interpretation of freshwater fisheries data. American Fisheries Society, Bethesda, Maryland.

Carle, F. L., and M. R. Strub. 1978. A new method for estimating population size from removal data. Biometics 34:621–630.

Dail, D., and L. Madsen. 2011. Models for estimating abundance from repeated counts of an open metapopulation. Biometrics 67:577–587.

Dauwalter, D. C., and W. L. Fisher. 2007. Electrofishing capture probability of Smallmouth Bass in streams. North American Journal of Fisheries Management 27:162–171.

Dauwalter, D. C., F. J. Rahel, and K. G. Gerow. 2009. Temporal variation in trout populations: implications for monitoring and trend detection. Transactions of the American Fisheries Society 138:38–51.

Dauwalter, D. C., F. J. Rahel, and K. G. Gerow. 2010. Power of revisit monitoring to detect forestwide declines in trout populations. North American Journal of Fisheries Management 30:1462–1468.

DeLurey, D. B. 1947. On the estimation of biological populations. Biometrics 3:145–147.

Fischer, J. R., and C. P. Paukert. 2009. Effects of sampling effort, assemblage similarity, and habitat heterogeneity on estimates of species richness and relative abundance of stream fishes. Canadian Journal of Fisheries and Aquatic Sciences 66:277–290.

Gaston, K. J., and B. H. McArdle. 1994. The temporal variability of animal abundances: measures, methods, and patterns. Philosophical Transactions of the Royal Society of London B 345:335–358.

Gerow, K. G. 2007. Power and sample size estimation techniques for fisheries management: assessment and a new computational tool. North American Journal of Fisheries Management 27:397–404.

Gotelli, N. J., and A. M. Ellison. 2004. Testing statistical hypotheses. Pages 100–102 *in* A primer of ecological statistics. Sinauer and Associates, Sunderland, Massachusetts.

Grossman, G. D., J. F. Dowd, and M. Crawford. 1990. Assemblage stability in stream fishes: a review. Environmental Management 14:661–671.

Habera, J. W., R. D. Bivens, B. D. Carter, and C. E. Williams. 2009. Region IV trout fisheries report: 2009. Tennessee Wildlife Resources Agency, Federal Aid in Sport Fish Restoration, Final Report, Nashville.

Habera, J. W., M. A. Kulp, S. E. Moore, and T. B. Henry. 2010. Three-pass depletion sampling accuracy of two electric fields for estimating trout abundance in a low-conductivity stream with limited habitat complexity. North American Journal of Fisheries Management 30:757–766.

Ham, K. D., and T. N. Pearsons. 2000. Can reduced salmonid population abundance be detected in time to limit management impacts? Canadian Journal of Fisheries and Aquatic Sciences 57:17–24.

Hense, Z., R. W. Martin, and J. T. Petty. 2010. Electrofishing capture efficiencies for common stream fish species to support watershed-scale studies in the central Appalachians. North American Journal of Fisheries Management 30:1041–1050.

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 15:551–562.

Kanno, Y., B. H. Letcher, N. P. Hitt, D. A. Boughton, J. E. B. Wofford, and E. F. Zipkin. 2015. Seasonal weather patterns drive population vital rates and persistence in a stream fish. Global Change Biology 21:1856–1870.

Kanno, Y., K. C. Pregler, N. P. Hitt, B. H. Letcher, D. J. Hocking, and J. E. B. Wofford. 2016. Seasonal temperature and precipitation regulate Brook Trout young-of-the-year abundance and population dynamics. Freshwater Biology 61:88–99.

Kanno, Y., J. C. Vokoun, D. C. Dauwalter, R. M. Hughes, A. T. Herlihy, T. R. Maret, and T. M. Patton. 2009. Influence of rare species on electrofishing distance when estimating species richness of stream and river reaches. Transactions of the American Fisheries Society 138:1240–1251.

Kéry, M., and J. A. Royle. 2016. Applied hierarchical modeling in ecology: analysis of distribution, abundance and species richness in R and BUGS. Elsevier, Amsterdam.

Kruse, C. G., W. A. Hubert, and F. J. Rahel. 1998. Single-pass electrofishing predicts trout abundance in mountain streams with sparse habitat. North American Journal of Fisheries Management 18:940–946.

Larsen, D. P., T. M. Kincaid, S. E. Jacobs, and N. S. Urquhart. 2001. Designs for evaluating local and regional scale trends. BioScience 61:1069–1078.

Letcher, B. H., P. Schueller, R. D. Bassar, K. H. Nislow, J. A. Coombs, K. Sakrejda, M. Morrissey, D. B. Sigourney, A. R. Whiteley, M. J. O'Donnell, and T. L. Dubreuil. 2015. Robust estimates of environmental effects on population vital rates: an integrated capture–recapture model of seasonal Brook Trout growth, survival and movement in a stream network. Journal of Animal Ecology 84:337–352.

Meyer, K. A., and B. High. 2011. Accuracy of removal electrofishing estimates of trout abundance in Rocky Mountain streams. North American Journal of Fisheries Management 31:923–933.

Midway, S. R., T. Wagner, J. D. Zydlewski, B. J. Irwin, and C. P. Paukert. 2016. Transboundary fisheries science: meeting the challenges of inland fisheries management in the 21st century. Fisheries 41:536–546.

Mollenhauer, R., and S. K. Brewer. 2017. Multinomial *N*-mixture models improve the applicability of electrofishing for developing population estimates of stream-dwelling Smallmouth Bass. North American Journal of Fisheries Management 37:211–224.

Mycko, S. A., Y. Kanno, and J. M. Bettinger. 2018. Using angling and electrofishing to estimate Smallmouth Bass abundance in a regulated river. Fisheries Management and Ecology 25:77–84.

Ogle, D. H. 2017. FSA: fisheries stock analysis. R package version 0.8.16. R Foundation for Statistical Computing, Vienna.

Peoples, B. K., and E. A. Frimpong. 2011. Among-pass, interregional, and single- versus multiple-season comparisons of detection probabilities of stream fishes. Transactions of the American Fisheries Society 140:67–83.

Peterman, R. M. 1989. Application of statistical power analysis to the Oregon Coho Salmon (*Oncorhynchus kisutch*) problem. Canadian Journal of Fisheries and Aquatic Sciences 46:1183–1187.

Peterman, R. M., and M. J. Bradford. 1987. Statistical power of trends in fish abundance. Canadian Journal of Fisheries and Aquatic Sciences 44:1879–1889.

Peterson, N. P., R. F. Thurow, and J. Guzevich. 2004. An evaluation of mutipass electrofishing for estimating the abundance of stream-dwelling salmonids. Transactions of the American Fisheries Society 133:462–475.

R Core Team. 2016. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

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 12:768–776.

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 25:1395–1410.

Royle, J. A. 2004. *N*-mixture models for estimating population size from spatially replicated counts. Biometrics 60:108–115.

Smith, K. L., and M. L. Jones. 2008. Allocation of sampling effort to optimize efficiency of watershed-level ichthyofaunal inventories. Transactions of the American Fisheries Society 137:1500–1506.

Southern Division of the American Fisheries Society Trout Committee. 1992. Standardized sampling guidelines for wadeable trout streams. Trout Committee, Southern Division, American Fisheries Society, Bethesda, Maryland.

Thomas, L. 1997. Retrospective power analysis. Conservation Biology 11:276–280.

Thompson, P. D., and F. J. Rahel. 1996. Evaluation of depletion-removal electrofishing of Brook Trout in small Rocky Mountain streams. North American Journal of Fisheries Management 16:332–339.

Urquhart, N. S., and T. M. Kincaid. 1999. Designs for detecting trend from repeated surveys of ecological resources. Journal of Agricultural, Biological, and Environmental Statistics 4:404–414.

Wagner, T., J. R. Bence, M. T. Bremigan, D. B. Hayes, and M. J. Wilberg. 2007. Regional trends in fish mean length at age: components of variance and the power to detect trends. Canadian Journal of Fisheries and Aquatic Sciences 64:968–978.

Wagner, T., B. J. Irwin, J. R. Bence, and D. B. Hayes. 2013. Detecting temporal trends in freshwater fisheries surveys: statistical power and the important linkages between management questions and monitoring objectives. Fisheries 38:309–319.

Wagner, T. J. T., J. Deweber, D. Kristine Detar, and J. A. Sweka. 2014. Spatial and temporal dynamics in Brook Trout density: implications for population monitoring. North American Journal of Fisheries Management 34:258–269.

Wiley, M. L., and C. Tsai. 1983. The relative efficiency of electrofishing vs. seines in Piedmont streams of Maryland. North American Journal of Fisheries Management 3:243–253.

Zipkin, E. F., J. T. Thorson, K. See, H. J. Lynch, E. H. C. Grant, Y. Kanno, R. B. Chandler, B. H. Letcher, and J. A. Royle. 2014. Modeling structured population dynamics using data from unmarked individuals. Ecology 95:22–29.

Zippin, C. 1956. An evaluation of the removal method of estimating animal populations. Biometrics 12:163–169.

SUPPORTING INFORMATION

Additional supplemental material may be found online in the Supporting Information section at the end of the article.