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DEVELOPMENT OF INTERIM TOOLS FOR PREDICTING BULL TROUT DETECTION PROBABILITIES

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Development of Interim Tools for Predicting Bull Trout Detection Probabilities

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Abstract

The development of efficient and effective bull trout management depends, in part, upon the ability to accurately depict the combinations of factors (e.g., habitat, water temperature) that affect bull trout distribution. These in turn, require adequate sampling effort to ensure high quality data for analyses and model building. In contrast to previous efforts, probability of detection estimates and sample size requirements in this report were based on the distribution of empirical bull trout densities and sampling efficiency estimates rather than single low arbitrary threshold values. The effect of 16 combinations of 4 habitat characteristics on sampling efficiency and detection probabilities were also estimated to improve the cost-effectiveness and accuracy of sampling protocols. Three-pass electrofishing and single-pass night snorkeling were the most effective techniques for detecting bull trout and required, on average, 24 and 25 samples, respectively to detect bull trout with 80% power. Sample size requirements and detection probabilities for various combinations of stream habitat characteristics and sampling method are also provided in tabled format. Large and small sampling frames were defined according to their aerial extent. Large sampling frames require the random selection of sampling units within the frame, whereas small sampling frames require either repeated sampling or estimation of detection probabilities for the entire sampling frame. Recommendations for future research efforts include the incorporation of new bull trout sampling data from numerous sites within Washington State, the development of optimal sampling protocols, and the development of models for estimating the risk of management actions to bull trout.

Introduction

Presence and absence data are increasingly being used to assess the current status and changes in the distribution of bull trout (*Salvelinus confluentus*) at scales ranging from stream reaches to entire basins (e.g., Rieman et al. 1997; Dunham and Rieman 1999). These data, however, are significantly influenced by detectability (i.e., the ability to capture at least one individual), which is generally low for rare and difficult to sample species, such as bull trout. Sampling protocols that fail account for low detectabilities may fail to detect bull trout when they are present. Previous bull trout sampling protocols have attempted to account for low detectability or differences in detectability among streams by varying the sample size (e.g., number of sites sampled) to ensure a consistent or minimum detection probability (Hillman and Platts 1993; Bonar et al. 1997). These protocols employed the Poisson estimator to calculate the required sample sizes, which requires the assumption that bull trout are distributed randomly among locations (sites). Organisms, however, are rarely distributed randomly and hence, detection probabilities can rarely be estimated via the Poisson estimator (Pielou 1969; Poole 1975). Rather, fish distribution patterns tend to be highly variable and as a result, bull trout detection probabilities are often overestimated by as much as 100% (Peterson *in review*).

Bull trout detection probabilities are also affected by sampling efficiency because the probability of detecting a species is a function of the probability of capture (sampling efficiency) and the species' distribution and density (Bayley and Peterson *in review*). Previous studies have attempted to incorporate the influence of sampling efficiency on detection by relying on professional judgment to estimate sampling efficiency and required sample sizes (Bonar et al. 1997). Recent bull trout sampling studies, however, suggest that these professional judgments can overestimate sampling efficiency by an order of magnitude (Russ Thurow, USFS Rocky Mountain Research Station, unpublished data). In addition, professional judgments cannot account the influence of sampling variance (i.e., variance in sampling efficiency) that can also cause the overestimation of bull trout detection probabilities.

There remains a critical need to develop tools that allow fishery biologists to accurately estimate bull trout detection probabilities and develop sampling protocols that ensure high quality data. These tools should be based on high quality empirical data that is currently unavailable for streams across the range of bull trout distribution in Idaho, Montana, Nevada, Oregon, and Washington. Thus, the goal of this study was to develop interim guidelines for the estimating detection probabilities and sample size requirements for stream-dwelling bull trout using existing empirical data on bull trout density, distribution, and sampling efficiency.

When estimating detection probabilities (detectability) for sampling bull trout, it is important to first consider the process of sampling. For instance, capturing or counting bull trout in a sampling unit (detection) requires 2 conditions– bull trout have to be in the site *and* at least 1 individual has to be captured or seen (during snorkeling). Similarly, a probability of detection estimate requires 2 components– an estimate of the number of fish in a sampling site (i.e., the number of chances you get) *and* an estimate of sampling efficiency (i.e., the ability to capture or count fish). In what follows, each component (fish abundance and sampling efficiency) is discussed individually and preliminary estimates of each are made using existing empirical data. The 2 components are then combined to produce interim guidelines for estimating detection probabilities and sample size requirements. Finally, recommendations are provided for the uses and limitations of the interim guidelines. Note that this report outlines the theoretical development and sample size requirements for the interim guidelines for sampling bull trout and should be considered a first approximation that will ensure uniform sampling effort until better data are collected and the guidelines revised.

Fish Abundance and Distribution

To estimate bull trout detection probabilities, abundance estimates (i.e., the number of fish) should be modeled using a discrete statistical distribution. This is because fish are discrete units (e.g., a site cannot contain a fraction of a fish). Previous studies have modeled fish abundance using the Poisson and negative binomial distributions. Both consist of integer values that can range from zero to infinity. The Poisson distribution has one parameter (m) that is both the mean and the variance. Hence, the Poisson distribution assumes that the mean and variance are equal. In contrast, the negative binomial has two parameters, the mean (m) and dispersion (k), which is a measure of variability. Somewhat counter intuitively, *low* values of k indicate high dispersion (variance) and high values indicate low dispersion and as k gets very large, the negative binomial is equivalent to the Poisson. Thus, variance can exceed the mean for the negative binomial distribution.

The negative binomial dispersion parameter (k) can be viewed as an index of overdispersion relative to the Poisson. Overdispersion refers to the variance in excess of the presumed variance, which for the Poisson is equal to the mean. Thus, the negative binomial estimator can incorporate additional variability in fish density, whereas the Poisson cannot. Sources on the additional variability are varied and can depend such things as the characteristics of sampling units. For example, bull trout abundance can vary with stream width, which can affect detection (e.g., Rieman and McIntyre 1995). Underestimates of the total variation associated with fish sampling will cause estimates of the probability of detection to be biased high and required sample sizes to be underestimated (Peterson *in review*). The negative binomial is more flexible than the Poisson because it can incorporate additional variability. Thus, it is an ideal candidate for modeling bull trout abundance in highly variable sampling conditions likely to be encountered across the bull trout range.

The variability of bull trout abundance discussed above can be attributed to two basic sources: among sampling frames (e.g., watersheds, patches) and among sampling units within sampling frames. Temporal variability is also likely important but cannot be estimated given the current available data. One of the difficulties with using the negative binomial distribution is the relatively large amount of high quality data needed to obtain a reliable estimate of the dispersion parameter, k. Further, dispersion is likely to be influenced by factors that vary across the bull trout range, such as habitat structure and patchiness, and possibly through time. Thus, to obtain estimates of both sources would require sampling data from a large number of sampling frames each containing a large number of sample units. These data are not currently available for such a large geographical area as the bull trout range. Until they are, the Poisson will be used to approximate within sampling frame variability in bull trout abundance.

Previous studies have largely overlooked the among sampling-frame variability or alternatively, uncertainty in the value of a threshold density. For instance, Hillman and Platts (1993) and Bonar et al. (1997) used an arbitrary single low threshold value, based on literature reviews, which they assumed to be the minimum density for (presumably viable) bull trout populations. However, a recent study indicated that mean bull trout densities could be as low as 0.02 individuals per sampling unit (Peterson *in review*). Thus, reliance on an arbitrary single low threshold value (e.g., 0.02) is cost inefficient and wastes management resources. Choosing a single threshold based some estimate of viability or effective population size is also somewhat problematic given the well-known difficulties with obtaining reliable, defensible estimates of viable population size (see Beissinger and Westphal 1998; White 2000). For example, Vucetich and Waite (1998) found that even 10 years of demographic data were inadequate for estimating effective population size. A more efficient and defensible approach would be to define a *class* of sampling frames known as – occupied by bull trout. Using existing sampling data for areas that are known to be occupied, the variation of bull trout densities among class members then can be incorporated into the estimates of detection probabilities. The simplest means of modeling this variation is by fitting the empirical density data to a continuous distribution known as the gamma, which is a natural conjugate of the Poisson (Berger 1985). This empirical gamma *distribution* of densities then replaces the single threshold density value for the Poisson mean (m). The resulting mixture (Poisson with a gamma mean) is a negative binomial distribution (Pielou 1969). Thus, a single low arbitrary threshold value for the Poisson is replaced by an empirical distribution of bull trout densities eliminating the need to address contentious issues, such as "minimum viable" or "minimum effective population" density. The interpretation of zero catches (no detection) is also greatly simplified by basing the sample size requirements on a *class* of sampling frames known as occupied by bull trout (see *Interpretation*).

Preliminary estimates.- To include the greatest geographical range possible, preliminary estimates of bull trout densities were obtained from fish survey data collected by USDA Forest Service (FS) biologists from1991- 96 (Peterson and Wollrab 1999) and fish monitoring data collected by Idaho Department of Fish and Game, USDA Forest Service, Nez Perce Tribe and Shoshone-Bannock Tribes between 1984-97 (Rieman et al. 1999). All of the sampling locations were within the known range of bull trout and most were in the Salmon, Clearwater, and Boise River Basins in Idaho. These data were collected in a variety of streams over a broad geographical area and represent a wide variety of stream types and habitats encountered in the bull trout range. In all applications, trained personnel collected fish with standardized protocols (day-snorkeling) from various stream types. Assuming that most bull trout survey or monitoring is conducted under similar circumstances, these data were considered fairly typical. Note that these data represent the best currently available data on bull trout densities over a relatively large geographical area. However, they were collected on low-order streams (2^{nd} -4^{th} order) in relatively good habitat areas in Idaho. Consequently, estimates generated from this data should be considered first approximations that will be adjusted in future guidelines as data are collected.

To maintain some flexibility for field crews, data collected using 2 different sized sampling units were selected, approximately 50-m and 100-m long. Only density data from subwatersheds (USGS 6th code hydrologic units) known to contain bull trout populations (Lee et al. 1997) were used. The influence of sampling error (within subwatershed variation) was minimized by averaging density estimates for all sample units within each subwatershed (mean number of units = 10.7, range 8-36). The resulting subwatershed means were then fit to a gamma distribution and goodness-of-fit was assessed via chi-square tests (Sokal and Rohlf 1995).

Fifty-meter units were sampled in 40 subwatersheds, and 100-m units were sampled in101 subwatersheds. Bull trout density (fish per sampling unit) varied across subwatersheds and averaged 0.4456 and 0.6119 in the 50-m and 100-m long sites, respectively. For both unit sizes, variances greatly exceeded the means, which indicated overdispersion relative to the Poisson. Goodness-of fit tests indicated that the distribution of bull trout densities could be adequately represented by a gamma for both the 50 m (χ^2 = 8.232, 7 df, P= 0.3125) and 100 m (χ^2 = 8.934, 6 df, P= 0.1773) sampling units. The gamma shape (a) and scale parameters (b) were estimated for the 50-m units as: a = 0.1325, b = 3.4311, and the 100-m units as: a = 0.1211, b = 5.1268. Using these estimates, the probability (p) of a sampling unit containing i individuals can be estimated according to Pielou (1969) as:

$$p = \frac{b^i}{(1+b)^{a+i}} \frac{\Gamma(a+i)}{\Gamma(i+1)\Gamma(a)}. \qquad (1)$$

Sampling Efficiency

Both the negative binomial and Poisson estimators require that all individuals can be counted in a sampling unit. In practice, this is impossible for most fish collection efforts because fish sampling efficiency is rarely, if ever, 100%. For clarity, sampling efficiency is defined as the proportion of individuals, in a given area, that are captured or observed during sampling. Previous efforts have attempted to account for the influence of sampling efficiency by assuming that fish capture is a binomial process (Bonar et al.1997; Rieman and McIntyre 1993). That is, fish are either captured or missed (e.g., a fraction of a fish cannot be caught). Detection probabilities were then adjusted assuming a binomial distribution and using a point estimate of sampling efficiency (q). The binomial adjustment assumes that all individuals within a sampling unit have the same probability of capture and all respond independently. However, sampling efficiency is influenced by body size (Buttiker 1992; Dolloff et el. 1996) and thus, can vary among individuals within a site. In addition, the prevalence of overdispersion (variance in excess of the binomial distribution) in many fish sampling efficiency and mark and recapture models suggests that fish do not respond independently (Bayley 1993). Independence refers to how fish react to the sampling process and its influence on efficiency. For example during sampling, a fish school can swim away from the sampler, so none are captured (0% efficiency) or they swim toward the sampler, so all are captured (100% efficiency). In other words, the school is acting like a single individual. These violations of the binomial assumptions can result in the systematic underestimation of bull trout probability of detection (Peterson *in review*).

Similar to the Poisson-gamma model discussed above, the extra variability in sampling efficiency (q) can be modeled with a continuous distribution, in this instance the beta. Thus, the beta distribution of sampling efficiencies replaces a single mean estimate of sampling efficiency (q) in the binomial estimator, resulting in a beta-binomial distribution (Prentice 1986). The beta-binomial can explicitly account for extra variability (overdispersion) in sampling efficiency due to factors such as independence among individuals, which can improve the accuracy of the probability of detection estimates (Peterson *in review*). Using the beta-binomial, the probability (p_c) of capturing at least 1 individual (detection) is estimated as:

$$\mathrm{p_c} = 1 - \left[\frac{\Gamma(i+1)\Gamma(a+x)\Gamma(a+b)\Gamma(i+b-x)}{\Gamma(x+1)\Gamma(i-x+1)\Gamma(a+b+i)\Gamma(a)\Gamma(b)} \right], \qquad (2)$$

where a and b are the beta shape parameters, i is the number of individuals in the sampling unit as defined above, and x is the number of individuals captured (0 in this case). The beta-binomial shape parameters are estimated using the mean sampling efficiency estimate (q) and dispersion parameter (γ) from a beta-binomial regression as $a = q/\gamma$ and $b = (1\text{-q})/\gamma$. The dispersion parameter is a measure of the variability of sampling efficiency due a variety of sources including difference in catchability due to body size and non-independence of fish responses (i.e., the effect of the number of bull trout in a site on efficiency).

Sampling efficiency is also influenced by sampling method and the habitat characteristics of the sampling unit (Buttiker 1992; Bayley and Dowling 1993; Riley et al. 1993; Anderson 1995). Failure to account for differences in sampling efficiency introduces a systematic error or bias into the data. "Systematic error either in the imposition of treatments or in sampling or measurement procedures renders an experiment [or observational study] invalid or inconclusive" (Hurlbert 1984). For instance, reliance on a single average estimate of sampling efficiency for all sampling methods would systematically overestimate detection probabilities (and underestimate sample size requirements) in situations where actual sampling efficiency was lower. In contrast, the use of the lowest estimated sampling efficiency (e.g., among methods) would cause detection probabilities to be underestimated and sample sizes overestimated for some methods, resulting in the collection of too many samples, wasting valuable research and monitoring funds. Therefore, gear and habitat-specific sampling efficiency estimates should be used for estimating detection probabilities and sample size requirements.

Preliminary estimates.- To develop a more efficient and effective approach to sampling bull trout, preliminary sampling efficiency models were developed using sampling efficiency data collected from 24 streams, known to contain bull trout, by Rocky Mountain Research Station (Boise, ID) personnel in 1999. The fish sampling and habitat measurement procedures used are thoroughly discussed in Thurow and Schill (1996) and outlined in Thurow (1994), but are reviewed below. These estimates were collected on a relatively narrow range of conditions encountered in streams throughout the bull trout range, but represent the best available data for bull trout sampling efficiency data. Therefore, the estimates should be considered preliminary and subject to change. Additionally, these efficiency data differ significantly from previous attempts to estimate sampling efficiency because they relied on the recapture or resighting (for snorkeling estimates) of a known number of individual bull trout. This method was chosen because removal estimates used to estimate sampling efficiency for the snorkeling and backpack electrofishing are generally biased low (Buttiker 1992; Riley et al. 1993; R. Thurow, Rocky Mountain Research Station, unpublished data; Peter Bayley, Oregon State University, personal communication) and, as such, previous estimates of relative sampling efficiency were most likely biased high.

The sampling gear evaluation procedure consisted of blocking off approximately 100-m long stream sections with 6-mm opening mesh nets that were secured to the streambed. Bull trout were collected within the blocked-off area via a backpack electrofisher with unpulsed Direct Current (DC) and 2 passes, one up- and one downstream. Captured fish were held in live wells, total length measured, and the dorsal or caudal fin received either a hole from a paper punch or was notched in a manner that would be visible to snorkelers. All fish were released back into the unit and allowed to recover and disperse for 24 hours. Following the recovery period, fishes in the blocked-off section were sampled between the hours of 1000 and 1700, via snorkeling (hereafter, day snorkeling) that began at the downstream end of each blocked off section and completed on a single pass. Bull trout were then sampled the following night, between 2230 and 0230 hours via snorkeling (hereafter, night snorkeling) and identical procedure as used during the day except that night counts were completed with the aid of an underwater halogen light. Snorkeling counts consisted all bull trout >70 mm TL (Age 1+) and divers took great care to distinguish marked individuals. Bull trout were then sampled the following day (between 1000 and 1700 hours) with a backpack electrofisher using unpulsed Direct Current (DC) and 3 passes, each in an upstream direction. Following sampling, fish were measured for total length, the presence or absence of a mark recorded, and released back into the site.

The physical features of each unit believed to affect the efficiency of bull trout sampling, were measured or estimated. Stream gradient was measured from a USGS 1:24,000 topographic map. Temperature and visibility were measured prior to sampling. Underwater visibility was estimated by suspending a plastic silhouette of a salmonid in the water column. A diver would begin in front of the silhouette and move downstream until the object could not be distinguished. Mean wetted width and depth and substrate composition were estimated by averaging measurements taken at transects established along the thalweg at 20-m intervals. At each transect, wetted channel width and mean depth were measured and substrate composition was visually estimated for 2 size classes: cobble (75-150 mm), and rubble (> 150 mm). All pieces of woody debris in the wetted channel at least 3-m in length and 10-cm in diameter were counted. Current velocity was estimated by placing a rubber ball into a section of the site with relatively uniform flow across the width of the channel and estimating the time to travel 10 m. The average of 3 timings was used to estimate mean current velocity.

Preliminary sampling efficiencies and dispersion parameters were estimated for bull trout 70- 200 mm total length because it was assumed that the presence of these individuals is indicative of the presence of a local population (*sensu* Rieman and McIntyre 1995). The effects of habitat variables (Table 1) on sampling efficiency were estimated for day snorkeling, night snorkeling, and 3-pass backpack electrofishing were estimated using the number of marked and recaptured (resighted during snorkeling) individuals as dichotomous dependent variables (i.e., the number of success and trials, respectively) in beta-binomial regression (Prentice 1986). Goodness-of-fit for each regression model was assessed by examining residuals (Agresti 1990). To allow for ease of use in the field, the 4 significant factors (e.g., habitat characteristics) that had the greatest influence on bull trout sampling efficiency as indicated by standardized beta-binomial regression coefficients were selected (Table 2). Each factor was then separated into 2 categories (high, low) based, in part, on the range of values during the calibration resulting in 16 habitat combinations per sampling gear. Average sampling efficiencies for each combination were then estimated for the midpoint values of each category.

Among the variables considered (Table 1), day snorkeling efficiency was positively related to water temperature and visibility and negatively related to stream gradient and density of large wood (Table 2). Night snorkeling efficiencies were also positively related to water temperature and visibility and negatively related to wood density. However, gradient was positively related to night snorkeling sampling efficiency (Table 2). In contrast, mean stream width (wetted), depth, gradient, and large wood negatively influenced the efficiency of 3-pass backpack electrofishing (Table 2). Sampling efficiency was, on average, greatest for 3-pass electrofishing (26.4%) and lowest for day snorkeling (10.2%). Night snorkeling efficiencies averaged 24.8% across habitat combinations (Table 5).

Efficiency cross-comparison.- There remains the possibility that the capture and handling of bull trout during the marking process could have affected their vulnerability during subsequent sampling. This would have resulted in biased estimates of bull trout sampling efficiencies. To examine the possible effects of marking on the sampling efficiency estimates, I cross-compared sampling efficiency estimates for each method as follows.

Step 1: Estimate sampling efficiency (π_m) for each method (m), site combination using the beta-binomial efficiency models.

Step 2: Estimate the number of unmarked fish in a site using the method- specific estimates of the number of unmarked fish and the sampling efficiency as:

$$T_m = N_m / \pi_m \tag{3}$$

where: T_m = efficiency adjusted estimate of the number of unmarked bull trout, π_m = predicted sampling efficiency as a fraction, and N_m = the number of unmarked bull trout collected or counted with method m.

Step 3: Cross-calculate estimates of the sampling efficiency of each method by dividing the raw (actual) count /catch of unmarked fish by the efficiency adjusted estimates of unmarked fish of the other sampling methods. This results in 3 sampling efficiency estimates per method: (1) the efficiency estimate which is the ratio of recaptured (resighted) to marked individuals and (2-3) two cross-calculated efficiency estimates which are the ratio of the number of unmarked fish counted with that method to the efficiency adjusted estimates of the number of unmarked fish for the other 2 methods. For example, the 2 cross-calculated estimates for day snorkeling would be calculated by dividing N_d by T_n and T_e, where N_d = the number of unmarked bull trout counted during day snorkeling, T_n = efficiency adjusted estimate of the number of unmarked bull trout for night snorkeling, and T_e = the efficiency adjusted estimate of the number of unmarked bull trout for 3-pass electrofishing.

Each of the 3 estimates then was averaged across sites and 95% confidence intervals (CI) calculated to assess the relative accuracy of the methods. Large differences among the 3 efficiency estimates for each method would indicate that marking/ handling affected the vulnerability of marked fishes differently among methods or through time. The latter of these two is due to the fact that the time between marking and day snorkeling averaged approximately 24 hours, marking and night snorkeling 36 hours, and marking and electrofishing 48 hours.

The cross-comparison of the 3 sampling efficiency estimates for all methods indicated that they were similar to the mark-recapture (resight) estimates (Figure 1). In fact, all were well within the 95% CI of the all of the mark-recapture based estimates. This suggested that either there was no detectable effect of marking on bull trout vulnerability or that this effect did not differ among methods or through time. However, the time between sampling and sampling method were confounded; hence a specific time effect could not be separated from the method used. Reducing or eliminating the effect of the marking procedure on fish vulnerability is crucial for developing useful models. Therefore, the RMRS personnel are currently randomly vary times between fish marking and sampling and randomly determining the method that is employed first (i.e., day vs. night snorkeling). In addition, the possibility remains that marked fish might have left the sites, biasing the estimates. Consequently, RMRS personnel are also placing a second set of block nets a short distance up and downstream of the blocked-off site and sampling these areas following the calibrations to check for marked individuals.

Probability of Detection and Sample Size Estimates

Using the empirical estimates of fish density and sampling efficiency, the probability of detection is estimated as 1- the probability of capturing no (zero) fish. The probability of capturing no fish is estimated (in the general form) as:

Probability (No. of fish = 0)* Probability (catching 0 fish given 0 fish there) +
Probability (No. of fish = 1)* Probability (catching 0 fish given 1 fish there) +
Probability (No. of fish = 2)* Probability (catching 0 fish given 2 fish there) +

.

.

.

Probability (No. of fish = x)* Probability(catching 0 fish given X fish there),

where x is summed to infinity (in theory). In practice, x is summed until the Probability (No. of fish = x) gets very, very small (i.e., *Taylor expansion*). Again, note this approach is much different than the "minimum threshold" used in previous protocols. Here, the probability of a particular density, given the range of observed densities, is used in place of a minimum density. The latter approach can result in a considerable unnecessary sampling effort, whereas the current approach is more cost efficient and is directly related to the goals of bull trout presence absence surveys. That is, the determination of bull trout presence in *occupied* sampling frames (see *Considerations for Using the Guidelines*, below) rather than presence below a single threshold density.

Probability estimation.- The probability of there being x fish in a sampling unit is estimated using the gamma shape and scale parameters and equation 1. For 50 *m* long sampling units, the probability that a unit contains 0–5 fish is:

Number of fish, x	0	1	2	3	4	5
Probability (Number of fish in unit $= x$)	0.8210	0.0842	0.0369	0.0203	0.0123	0.0079

The probability of catching no fish when there are x fish in the sampling unit is estimated using the beta-binomial mean sampling efficiency and dispersion parameter to estimate a and b

and equation 2. For day snorkeling, the sampling efficiency is 0.095 and the beta-binomial dispersion parameter is 0.481. Thus, $a = 0.095/0.4811 = 0.1975$, $b = (1\text{-}0.095)/0.4811 = 1.8811$, and the probability of collecting no fish when there are 0–5 fish in the sampling unit is:

Number of fish in unit, x	0	1	2	3	4	5
Probability (catching 0 fish given = x in unit)	1.000	0.905	0.8470	0.8059	0.7746	0.7494

Using the values above their products are taken:

Probability (No. of fish in unit = x)* Probability(catching 0 fish given x fish there),

Number Of fish, x	Probability (No. of fish in unit = x)	Probability (catching 0 fish given x fish there)	Product
0	0.8210	1.0000	0.8210
1	0.0842	0.9050	0.0762
2	0.0369	0.8470	0.0313
3	0.0203	0.8059	0.0164
4	0.0123	0.7746	0.0095
5	0.0079	0.7494	0.0059
.	.	.	.
.	.	.	.
.	.	.	.
18	0.0001	0.6031	0.0001
19	0.0001	0.5972	0.0000
20	0.0001	0.5916	0.0000
		Probability of not detecting bull trout (the sum) =	0.9723
		Probability detecting bull trout (1 minus the sum)=	0.0277

The products are then summed to obtain the probability of catching no fish, and this value is subtracted from 1 to get the probability of collecting at least 1 fish (i.e., detection). Note that this value is identical to the single sample probability for 50-m sampling units in Table 3.

Sample size estimation.- Sample sizes requirements are easily estimated using the probability of catching no fish, from above, or subtracting 1 from the single sample probabilities of detection (Tables 3-5). In the notation of Green and Young (1993), the number of sampling units (samples) needed to detect with 'power' 1- β is simply:

$$n = \log(\beta) / \log[\text{Probability(catching no fish)}], \qquad (4)$$

where log is the natural log. The resulting values for 16 combinations of habitat characteristics are calculated for day and night snorkeling and electrofishing in Tables 3–5.

Application.- In practice, it won't be necessary for biologists to calculate the single sample probabilities of detection because they're provided in Tables 3–5 for the 16 combinations of habitat characteristics. All a biologist has to do is measure or estimate their stream habitat characteristics and look up the tabled values. For example, a sampling crew decides to use single-pass night snorkeling. They measure the stream temperature visibility, gradient and wood density and find that the stream is, on average, cold (<9 °C), and has high visibility, low gradient, and high wood density. Looking in Table 4 they see that they need to collect 22 samples in the stream to be 80% confident of detecting bull trout. If a stream is highly heterogeneous (temperature, gradient, etc), the crew could use the single site probabilities to estimate their probability of detection. For example, assume that 3 sites were sampled and that they had single sample detection estimates of 0.05, 0.046 and 0.057 (the first 3 lines from Table 4). The probability of detection would be 1-[(1-0.05)*(1-0.046)*(1-0.057)] = 0.145. These estimates could be made sequentially as a team proceeds with sampling until a desired level of detection (e.g., 80% power) is reached.

When using these tables, it is important to keep in mind that the sampling efficiency estimates were based on preliminary models developed under a limited range of habitat conditions (Table 1). Thus, these estimates may not be applicable when physical habitat characteristics differ from those observed during the sampling gear calibrations. In some instances, sampling under conditions that are out of the range of those encountered during sampling efficiency calibration cannot be avoided. Consequently, probability of detection estimates and required sample sizes for these sampling designs should be interpreted with caution.

Considerations for Using the Guidelines

Limitations.- As indicated throughout this report, these are interim guidelines and should be considered first approximations to estimating sampling efficiency and detection probabilities. In addition, the current guidelines are limited to resident and juvenile bull trout and to smaller-sized streams because the data used to estimate sampling efficiencies were collected from areas and habitats used by these life stages and life history forms. Because habitat for juvenile and resident bull trout is crucial for the bull trout persistence, it is a reasonable place to begin development of the sampling guidelines.

The probability of detection estimates and sample size requirements in this document are also based on density data collected in the Salmon, Clearwater, and Boise river basins in Idaho. Those basins contain relatively high quality habitat and hence, bull trout densities in other basins could differ. Consequently, the interim guidelines might not accurately depict bull trout densities/ distribution or all possible samplings conditions encountered in Washington streams. Thus, a zero catch with an 80% power of detection should be interpreted as an 80% chance that bull trout would have been detected in a sampling frame if their densities were similar to those in the known occupied subwatersheds that were used to develop the sampling guidelines. It would have been preferable to use existing data collected in Washington State to create more robust interim guidelines. Unfortunately, these data were not available (Scott Bonar, Washington Department of Natural Resources, *personal communication*). This highlights the importance and need for natural resource agencies and industry to adopt consistent sampling protocols and devise plans to compile and store sampling data for future, unanticipated uses.

Sampling design.- The most important aspect to consider while designing a survey is its goal(s). This is used to define the frame of inference (sampling frame) and hence, determine appropriate survey designs. For instance, if the goal of a study was to estimate bull trout occurrence in a specific stream, the frame of inference would be the stream and a potential survey design could include sampling fish in randomly selected sampling units within that stream. However, this design (sampling in a single stream) would be inadequate if the goal was to estimate bull trout occurrence in an entire watershed. For this goal, the sampling frame would be the *watershed* and a more appropriate design could include the random selection of streams within the watershed and sampling fish in randomly selected sampling units within each of these. Because the goals for many of the surveys using these guidelines will likely vary, specific sampling designs are not provided here. However, the applicability of these detection probability estimates and sample size requirements depend on certain aspects of sampling design. Below, two typical sampling situations based on the aerial extent of the sampling frame are considered: one for a large-scale (e.g., watershed) and the other for smaller-scale (e.g., reach) surveys.

Large-scale surveys.- Large-scale surveys are defined as those for which the area of the sampling frame exceeds the total area sampled by the required number of samples. For example, assuming 50-m sampling units and 10 required samples, a large-scale study would be one for which the sampling frame (total stream length) exceeded 500-m (i.e., 50 * 10 = 500). This could include one or several 6^{th} code HUCs (USGS hydrologic units). The interim detection probabilities and sample size requirements were estimated using existing data collected in known occupied subwatersheds (6^{th} code HUC) and represent the (spatial) variability from subwatershed to subwatershed and sampling unit to unit. Thus, the basic requirement for the large-scale surveys is the random selection of sampling units within the sampling frame. To reduce some of the effort, biologists could randomly select sample units and focus their initial sampling efforts on what they perceive to be more suitable bull trout habitat. However, all sample units must be sampled if bull trout are not detected to make valid inferences regarding detection probabilities. Potential designs for large-scale studies include completely randomized, stratified random, and random start designs.

Small-scale surveys.- Small-scale surveys are defined as those for which the total area sampled by the required number of samples exceeds the total area of the sampling frame. In these instances, the entire reach could be sampled and the power of detection for the reach would be limited to the sum of the probabilities of detection for the individual sampling units. A more efficient sampling method (e.g., night snorkeling) would help maximize the power possible. Alternatively, a sampling design could include repeated measurements of sampling units through time for a large enough sample size to reach a desired power of 80% or 95%. However, this introduces a temporal variance component into the sampling design, whereas the estimates of detection probabilities and required sample sizes in this document more accurately represent the variation in spatial distribution. Consequently, probability of detection estimates and required sample sizes for these sampling designs should be interpreted with caution.

Interpretation.- Unlike previous efforts (Bonar et al. 1997; Watson and Hillman 1997), interpretation of zero catches using the interim guidelines cannot be made relative to a single arbitrary threshold value. They are made relative to the densities measured in known occupied sampling frames (i.e., those in Idaho). If the sampling frames represented a true random sample of occupied sampling frames, then a zero catch would be interpreted relative to bull trout presence, as opposed to presence above some arbitrary threshold density. This interpretation is more directly related to the goals of bull trout presence-absence surveys.

Previous approaches also were unable to estimate the probability that an area contained bull trout, given that they weren't collected. Because the detection probabilities are conditional on the sampling frame being occupied (or containing a viable population size), this estimate requires the consideration of total probability. Total probability simply means that all of the events that might have caused a zero catch are considered. Using these, the probability that bull trout occur in a sampling frame, given that they weren't collected can be easily estimated via Bayes' formula as:

$$P(P \mid C_0) = \frac{P(C_0 \mid P)P(P)}{P(C_0 \mid P)P(P) + P(C_0 \mid A)P(A)} \qquad (5)$$

where $P(P|C_0)$ = probability that bull trout occur, given that they weren't collected
$P(C_0|P)$ = probability of not detecting bull trout, given that they are present (i.e., 1 minus the probability of detection, β)
$P(P)$ = prior probability that bull trout are present
$P(C_0|A)$ = probability of not detecting bull trout, given that they are absent (equals 1)
$P(A)$ = prior probability that bull trout are absent (1 minus the prior probability of presence, above)

Because the detection probabilities in the interim guidelines are based on detecting the presence of bull trout in occupied sampling frames, $P(P|C_0)$ is interpreted (in theory) as the probability that bull trout occur, given that they weren't collected. This is known as the *posterior* probability of presence. The prior probability of presence, $P(P)$, could be assumed to be unknown (i.e., an *uninformative prior*; Gelman et al. 1996). However, a more sound approach would be to develop priors based on empirical models of bull trout distribution (e.g., Dunham and Rieman 1999; Rieman et al. 1997). The *posterior* probability of presence can then be used in management decision-making, planning, and risk assessment processes (*see* Peterson 1999 for an example).

Future Research Needs

The interim guidelines also use the Poisson to approximate within sampling frame variability in bull trout abundance. However, Peterson (*in review*) found that within sampling frame variability in bull trout abundance is often greater than that assumed for the Poisson (overdispersed). Indeed, bull trout distribution (and dispersion) is likely to be influenced by factors that vary across the bull trout range, such as habitat structure and patchiness, and possibly through time. To obtain robust estimates for Washington streams, bull trout sampling data will need to be collected from a large number of sampling frames (e.g., watersheds, patches) and sample units within those frames. Probability of detection and sample size estimates then can be modeled using these data. Therefore, the most useful research efforts designed to improve the estimates in this document should focus on the collection of high quality data on bull trout distribution and abundance.

Although sampling and monitoring for rare and difficult to capture species, such as bull trout, can be expensive and time consuming, designs can be optimized by properly allocating samples in space and time (Peterson and Rabeni 1995). It is commonly known that stream fish distribution and density vary spatially and temporally (Angermeier 1987; Matthews 1990). These two sources can contribute significant amounts of variance to estimates of fish distribution and abundance. Previous studies indicate that that spatial distribution usually exceeds temporal variation for warmwater stream fishes (Matthews 1990; Peterson and Rabeni 1995). However, there are currently no estimates of temporal (within and among season) variability in bull trout abundance and distribution. The best means of evaluating the efficiency of various sampling and monitoring designs is through the analysis of empirical data (Cochran and Cox 1957). Thus, another important future research objective should be to examine the variability of bull trout abundance and distribution over space and time in order to develop sampling and monitoring protocols that maximize biological insight and minimize costs.

Sample size requirements are not the only factor that should be considered when developing bull trout monitoring protocols. The current guidelines for standardizing effort (e.g. sample size estimates) provide little information on which to make informed management decisions when bull trout are not detected. An alternative approach is to use empirical models to estimate the posterior probability of presence, given that bull trout weren't detected (Bayley and

Peterson *in review*, available from the second author). These estimates could then be used to formally assess the risk of management actions to potential bull trout populations, but they require high quality empirical data to develop prior estimates of the probability of presence, fish abundance, and sampling efficiency. Thus, future research should focus on the collection and integration of new high quality sampling and monitoring data into accurate models that then can be used to make informed management decisions.

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Table 1. Means, standard deviations (SD), minimum and maximum values of variables tested in method-specific sampling efficiency models.

Variable	Mean	SD	Minimum	Maximum
Mean water temperature (C)	9.46	2.160	5.50	14.50
Sampling unit length (m)	90.30	14.05	45.60	112.00
Gradient	3.90	1.327	2.00	9.90
Mean current velocity (m/s)	0.66	0.146	0.44	1.15
Mean depth (m)	0.17	0.025	0.12	0.21
Percent cobble substrate	24.43	8.966	5.00	44.20
Percent rubble substrate	40.95	17.251	6.70	78.30
Mean wetted width (m)	3.34	0.720	2.33	5.12
Large wood density (no./m2)	0.07	0.040	0.01	0.14
Visibility (proportion of mean width)	0.69	0.157	0.43	0.99

Table 2. Standardized coefficients, standard errors, and associated upper and lower confidence limits from beta-binomial regression of day snorkeling, night snorkeling and 3-pass electrofishing efficiencies for juvenile bull trout. Confidence limits were based on the t-statistic. Note that these are initial estimates collected under a limited range habitat conditions (Table 1) and should be used only for the interim guidelines.

Parameter	Standardized coefficient	Standard error	Upper 95% CL	Lower 95% CL	Upper 90% CL	Lower 90% CL
Day snorkeling						
Water temperature	0.398	0.116	0.625	0.172	0.588	0.209
Visibility	0.092	0.050	0.190	-0.006	0.174	0.010
Gradient	-0.144	0.069	-0.009	-0.280	-0.031	-0.258
Wood density	-0.206	0.101	-0.008	-0.403	-0.040	-0.371
Night snorkeling						
Water temperature	0.023	0.014	0.051	-0.004	0.046	0.001
Visibility	0.108	0.060	0.227	-0.011	0.207	0.009
Gradient	0.042	0.022	0.085	0.000	0.078	0.007
Wood density	-0.019	0.011	0.003	-0.041	-0.001	-0.037
Three pass electrofishing						
Gradient	-0.188	0.084	-0.022	-0.353	-0.049	-0.326
Mean depth	-0.063	0.037	0.009	-0.135	-0.003	-0.123
Mean wetted width	-0.072	0.038	0.001	-0.146	-0.011	-0.134
Wood density	-0.122	0.052	-0.019	-0.225	-0.036	-0.208

Table 3. Habitat-specific estimated mean sampling efficiencies for single-pass day snorkeling, single sample probabilities of detection, and number of samples required to detect bull trout (<200mm TL) with 80- 95% probabilities of detection in 50 and 100-m long sampling units.

				50-m sampling units				100-m sampling units			
				Single sample prob. of detection	Number of samples for desired power			Single sample prob. of detection	Number of samples for desired power		
Visibility	Gradient	Wood Density	Estimated sampling efficiency		80%	90%	95%		80%	90%	95%
Water temperature less than or equal to 9^0C											
Low	Low	Low	9.50%	0.028	56	81	105	0.034	46	67	86
		High	4.00%	0.012	130	191	243	0.015	105	152	196
	High	Low	5.00%	0.015	104	152	194	0.019	84	120	157
		High	2.00%	0.006	251	383	467	0.008	202	287	376
High	Low	Low	10.30%	0.031	52	73	97	0.037	42	61	79
		High	4.40%	0.013	119	176	221	0.017	96	134	179
	High	Low	5.50%	0.017	95	134	178	0.021	77	108	144
		High	2.30%	0.007	228	328	425	0.009	184	255	342
Water temperature greater than 9^0C											
Low	Low	Low	25.20%	0.067	23	33	43	0.080	19	28	36
		High	11.80%	0.034	46	67	86	0.042	38	54	70
	High	Low	14.50%	0.042	38	54	71	0.050	31	45	58
		High	6.30%	0.019	84	120	155	0.023	68	99	126
High	Low	Low	27.00%	0.072	22	31	40	0.085	18	26	34
		High	12.80%	0.037	42	61	79	0.045	35	50	65
	High	Low	15.70%	0.045	35	50	65	0.054	29	41	54
		High	6.90%	0.021	76	108	142	0.026	62	87	116

Visibility classes:

low- visibility less than or equal to 50% of the mean channel width

high- visibility greater than 50% of the mean channel width

Gradient classes:

low- less than or equal to 3.5%

high- greater than 3.5%

Wood density classes

low – less than or equal to 0.065 pieces per square meter

high – greater than 0.065 pieces per square meter

Table 4. Habitat-specific estimated mean sampling efficiencies for single-pass night snorkeling, single sample probabilities of detection, and number of samples required to detect bull trout (<200mm TL) with 80- 95% probabilities of detection in 50 and 100-m long sampling units.

				50-m sampling units				100-m sampling units			
					Number of samples for desired power				Number of samples for desired power		
Visibility	Gradient	Wood Density	Estimated sampling efficiency	Single sample prob. of detection	80%	90%	95%	Single sample prob. of detection	80%	90%	95%
Water temperature less than or equal to 9^{0}C											
Low	Low	Low	16.80%	0.050	32	45	59	0.058	27	39	50
		High	15.60%	0.046	34	49	63	0.055	29	41	53
	High	Low	19.80%	0.057	27	39	51	0.067	23	33	43
		High	18.50%	0.054	29	41	54	0.063	25	35	46
High	Low	Low	27.70%	0.076	20	29	38	0.088	18	25	33
		High	26.00%	0.072	22	31	40	0.083	18	27	34
	High	Low	31.90%	0.085	18	26	34	0.097	16	23	29
		High	30.10%	0.081	19	27	36	0.093	16	24	31
Water temperature greater than 9^{0}C											
Low	Low	Low	19.40%	0.056	28	40	52	0.066	24	34	44
		High	18.10%	0.053	30	42	55	0.062	25	36	47
	High	Low	22.70%	0.064	24	35	45	0.075	21	30	38
		High	21.30%	0.061	26	37	48	0.071	22	31	41
High	Low	Low	31.40%	0.084	18	26	34	0.096	16	23	30
		High	29.50%	0.080	19	28	36	0.092	17	24	31
	High	Low	35.90%	0.093	17	24	31	0.106	14	21	27
		High	33.90%	0.089	17	25	32	0.102	15	21	28

Visibility classes:

low- visibility less than or equal to 50% of the mean channel width
high- visibility greater than 50% of the mean channel width

Gradient classes:

low- less than or equal to 3.5%
high- greater than 3.5%

Wood density classes

low – less than or equal to 0.065 pieces per square meter
high – greater than 0.065 pieces per square meter

Table 5. Habitat-specific estimated mean sampling efficiencies for backpack electrofishing (3 passes), single sample probabilities of detection, and number of samples required to detect bull trout (<200mm TL) with 80- 95% probabilities of detection in 50 and 100-m long sampling units.

				50-m sampling units				100-m sampling units			
				Single sample prob. of detection	Number of samples for desired power			Single sample prob. of detection	Number of samples for desired power		
Mean depth	Gradient	Wood Density	Estimated sampling efficiency		80%	90%	95%		80%	90%	95%
Mean channel width less than or equal to 3 m											
Low	Low	Low	47.80%	0.117	13	19	24	0.132	11	16	21
		High	34.60%	0.092	17	24	31	0.106	14	21	27
	High	Low	27.30%	0.077	20	29	37	0.090	17	24	32
		High	17.80%	0.054	29	41	54	0.064	24	35	45
High	Low	Low	42.60%	0.107	14	20	26	0.122	12	18	23
		High	30.00%	0.083	19	27	35	0.096	16	23	30
	High	Low	23.30%	0.068	23	33	43	0.079	19	28	36
		High	14.90%	0.047	34	48	63	0.055	28	41	53
Mean channel width greater than 3 m											
Low	Low	Low	38.90%	0.101	15	22	28	0.115	13	19	24
		High	26.90%	0.076	20	29	38	0.089	17	25	32
	High	Low	20.70%	0.062	25	36	47	0.072	21	31	40
		High	13.10%	0.042	38	54	71	0.050	32	45	59
High	Low	Low	34.00%	0.091	17	24	31	0.105	14	21	27
		High	22.90%	0.067	23	33	43	0.078	20	28	37
	High	Low	17.40%	0.053	29	42	55	0.063	25	35	46
		High	10.90%	0.035	45	65	84	0.042	37	54	70

Depth classes:

low- mean depth less than 1m
high- mean depth greater than 1m

Gradient classes:

low- less than or equal to 3.5%
high- greater than 3.5%

Wood density classes

low – less than or equal to 0.065 pieces per square meter
high – greater than 0.065 pieces per square meter

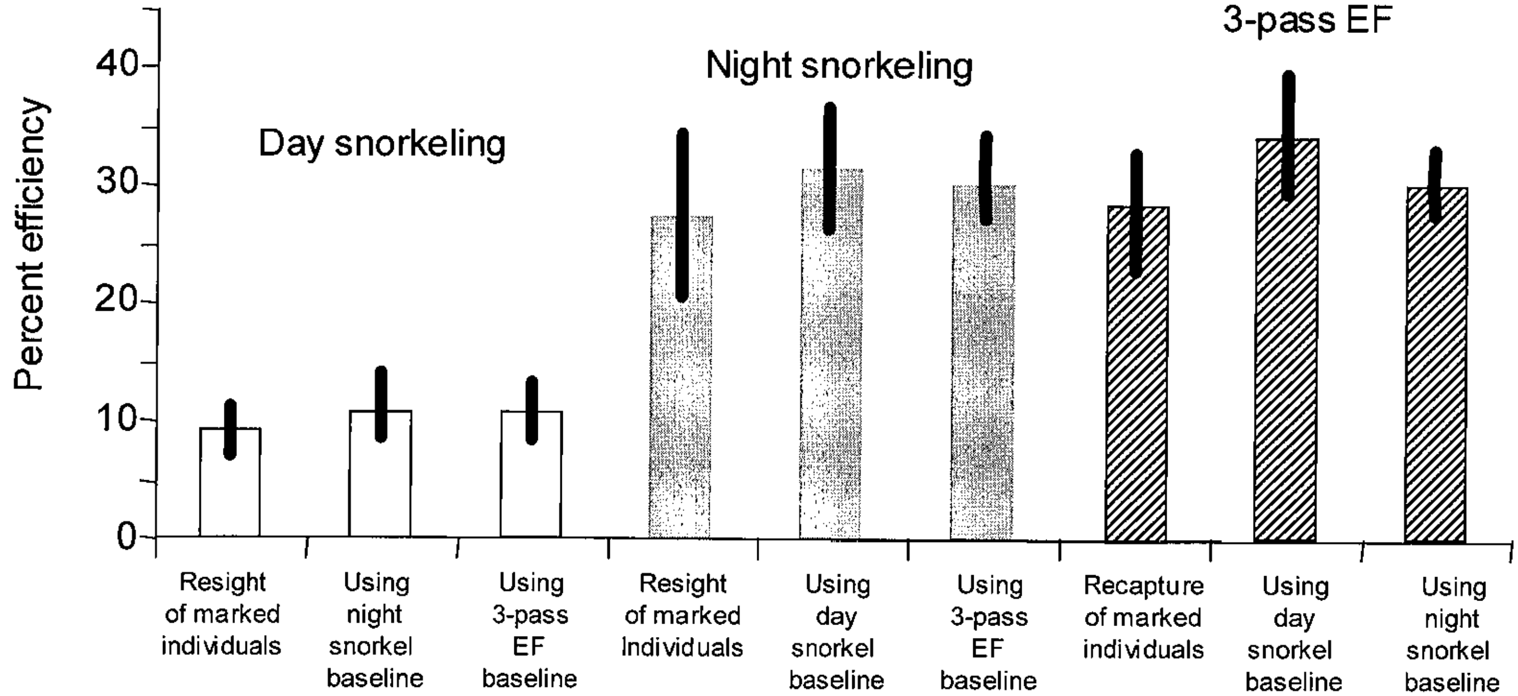

Figure 1. Cross-comparison of estimated average sampling efficiency, with log-based 95% confidence limits (black bars), of day and night snorkeling and 3-pass backpack electrofishing (EF) for bull trout 70-200 mm total length. Method-specific resight (snorkeling) or recapture efficiencies were estimated as the ratio of the number of recaptured or resighted individuals to number of marked individuals. Method-specific baselines were based on efficiency-adjusted estimates of the number of unmarked fish. See text for a complete explanation.