Review of Bull Trout Presence/Absence Protocol Development Including the Washington Validation Study

Annette Hoffmann

Kurt Reidinger

and

Molly Hallock

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Washington Department Of Fish And Wildlife
600 Capitol Way North
Olympia, Washington 98501-1091

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Abstract

Over the past decade, the U.S. Fish and Wildlife Service has worked toward defining sampling protocols for detection of threatened juvenile bull trout (*Salvelinus confluentus*) populations in the Coastal-Puget Sound and Columbia River regions. Previous detection protocols (Bonar et al. 1997) have set sample size guidelines using an assumed fixed capture efficiency and an arbitrary threshold density of bull trout. Sample sizes were selected so that if the capture efficiency was correct and no bull trout were detected, one would conclude it was unlikely that bull trout existed at or above the threshold density. Because of the fixed nature of the capture efficiency and threshold density, the sample size recommendations were also fixed. Recent protocols by Peterson et al. (2002) have incorporated two fundamental changes: (1) capture efficiencies that are modeled as a function of habitat characteristics; and (2) use of observed bull trout densities rather than arbitrary threshold densities. With the recent protocols, the resulting sample size recommendations vary with both the habitat characteristics and with density of juvenile bull trout used. Our review of the recent protocols resulted in substantial concern with both fundamental changes.

To address the utility of the habitat modeled capture efficiencies, validation studies were conducted in Washington streams during the summer of 2003. The natural variation in observed capture efficiencies within habitat categories was large making prediction difficult. The observed variation ranged as much as 60 percentage points (e.g., ranging from 10% to 70%) within a habitat category with much overlap among habitat categories. With so much variation and overlap, we felt that the stratification by habitat type suggested by the model acted more to dilute than improve the quality of the capture efficiency estimators. In fact, we found there to be less prediction error for both night snorkeling and one-pass electrofishing when we used the standard capture efficiency of 0.25 (Rieman and McIntyre 1995) across all habitat types than with the predicted capture efficiencies that we calculated from the habitat model. Until the practical value of the habitat-based capture efficiency model is more fully addressed, we recommend the resources used to measure habitat characteristics would be better devoted to enhance the direct sampling for bull trout.

Regarding the use of observed bull trout densities, we find it illogical to substitute actual densities for hypothetical thresholds. If one had actual densities, an effort to detect presence would be unnecessary. On the other hand, if actual densities measured elsewhere were to be used in sample size determination, then the density used will essentially function as the threshold. For example, using Idaho density patterns to set sample sizes in Washington protocols essentially declares that locations with densities less than those observed in Idaho are not important to detect in Washington. Because the choice of threshold has policy implications, we feel the choice of threshold needs input from a broader audience that includes managers and policy makers.

Based on our review, we have five general recommendations for Washington:

- Revisit the threshold density concept by creating a process that includes policy input to determine minimum threshold density and power (acceptable risk) criteria.

- Reanalyze the habitat data for practical value. This might be achieved by quantifying the actual cost vs. benefit (in terms of power and sample size) of using habitat modeled capture efficiencies and should include the uncertainties including the prediction error, the sampling error and the natural variability in observed efficiencies.

- Continue research to improve actual capture efficiencies as methods with better capture efficiencies will reduce the necessary sample sizes.

- Consider habitat-based models to help predict bull trout presence (as opposed to capture efficiency) so that the initial choice of sampling sites based on judgment will yield power greater than a strictly random selection would.

- Until the above steps are taken, continue to use the procedures outlined Bonar et al. (1997) for presence/absence sampling using a global mean value for capture efficiency updated with the data from all the studies conducted in Washington.

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Introduction

Washington bull trout (*Salvelinus confluentus*) populations in the Coastal-Puget Sound and Columbia River regions are currently listed as threatened (64FR58910, 63FR31647) under terms of the Endangered Species Act (ESA). To better understand the distribution of juvenile bull trout and regulate the activities that may affect them, field-sampling studies are conducted to determine the presence or absence of bull trout within streams and watersheds. By many accounts, bull trout are difficult to detect (see Thurow et al. 2004 for a review) so that not finding a bull trout in an area is no guarantee that a population does not exist there.

Presence/absence sampling protocols were developed in the past to control the probability of not detecting any bull trout if in fact they existed in the area sampled. The protocols assumed bull trout occurrence among sampling units was randomly distributed around some mean density (Green and Young 1993, Hillman and Platts 1993, Rieman and McIntyre 1995, and Bonar et al. 1997). Sample sizes were calculated as a function of a mean density (μ), a fixed capture efficiency (q), and some level of risk (β) where risk was defined as the conditional probability of not detecting bull trout given μ and q. Capture efficiency in this context is the probability of a sampling method detecting a bull trout given that it is in the sampling unit. The probability limit or risk was set at some arbitrary low value consistent with available resources. Mean densities or abundance thresholds were assumed or hypothesized as minimum values required for population viability (Rieman and McIntyre 1995). Table 1 summarizes the differences in the historic development of protocols.

With fixed capture efficiencies, sample sizes were calculated so that it would be unlikely to miss all bull trout if the hypothesized threshold densities existed. Higher threshold densities and/or higher capture efficiencies resulted in lower required sample sizes; lower threshold densities and/or capture efficiencies resulted in higher required sample sizes. With threshold densities and capture rates fixed, sample size guidelines were the same for all habitats.

Previous work (e.g., Hillman and Platts 1993, Rieman and McIntyre 1995) suggested there might be associations between stream habitat characteristics, capture efficiency, and bull trout abundance. Since then, Peterson et al. (2002) attempted to model capture efficiencies as a function of habitat characteristics and incorporated those into the sample size calculations. In considering the proposed new protocols, key issues for Washington managers are:

1) Logic and utility of replacing threshold densities with measured densities, and

2) Practical gains of using a habitat model to predict capture efficiency.

We have critically reviewed the rationale and feasibility of substituting actual density measures for hypothesized threshold values. We also addressed the practical gain of using a habitat model for capture efficiency in Washington by conducting a field validation study in 2003 and comparing the results with habitat model parameter values available at that time.

Table 1. Design differences among historical detection sampling protocols.

Study	Density Model	False Negative Risk Level	Capture Efficiency	Sample Size (Number of Units)	Site Selection Protocol
Green and Young (1993)	Threshold based on Poisson $\mu < 0.1$ sample unit size	$\beta = 0.05$	$q = 100\%$	$-\frac{\ln(\beta)}{\mu}$	None
Hillman and Platts (1993)	Threshold based on Poisson $\mu = 0.0025$/m	β based on study objective $= 0.05$, 0.20	$q = 100\%$	$-\frac{\ln(\beta)}{\mu}$	Random sample in each reach
Rieman and McIntyre (1995)	Threshold based on viable population size $\mu = 0.015$/m (or 0.45/30 m unit)	$\beta = 0.18$	q=25%	$-\frac{\ln(\beta)}{q \cdot \mu}$	Sequential based on judgment
Bonar et al. (1997)	Hypothesis test about a threshold mean density $\mu = 0.006$/m	$\beta = 0.2$	q=25%	$-\frac{\ln(\beta)}{q \cdot \mu}$	Judgment/Random
Current Proposal Peterson et al. (2002)	Actual densities modeled by gamma distribution	$\beta = 0.05$, 0.20	Habitat based	$-\frac{\ln(\beta)}{Compound(\Gamma, \beta - b)}$	Simple and Stratified Random

Incorporating Measured Densities in Place of Hypothesized Threshold Densities

Replacing threshold densities with field-measured densities poses conceptual problems since there is an obvious logic issue stemming from the fact that if actual densities were known, there would be no need to conduct a presence/absence survey. Even if actual densities could be known with certainty, replacing hypothesized thresholds with actual densities changes the meaning of finding no bull trout to a category Type I random error, i.e., the bull trout were there and should have been detected with the sample size selected, but unfortunately were not.

Because actual bull trout densities are extremely difficult to obtain, one might consider borrowing them from elsewhere. However, if the presumption is that the borrowed density mimics the study population, the interpretation concern noted above still exists. In addition, if the density from elsewhere is used as a threshold, then there needs to be discussion of the appropriateness of that threshold. For example, if sample sizes in Washington are set to detect Idaho densities then there may be great risk of missing Washington populations that are less dense. If the policy in Washington is to protect minimum viable populations that are less dense, then a protocol based on actual densities will not suffice.

To better focus on the density issue, Table 2 shows a range of hypothetical minimum densities illustrating the relationships between sample size, capture efficiency, and minimum threshold density for a given level of risk, β. These are obtained by rearranging the following sample size formula under the Poisson model and calculating underlying minimum threshold density, μ, for a given level of sample size, capture efficiency, and risk.

$$n = \frac{-\ln(\beta)}{\mu q}$$

For example, if a sample size of 10 was recommended and the search method had a capture efficiency of 10%, then one would have an 80% chance of detecting presence only if the population had an average density of 1.61 individuals or more per sampling unit. This table may be used to assess the minimum detectable density regardless of protocol used.

Table 2. Minimum threshold densities per sampling unit that are detectable (80% chance for detecting presence).

Number of Samples	Risk β = 0.20		
	Capture Efficiency		
	0.10	0.25	0.40
5	3.22	1.29	0.81
10	1.61	0.64	0.40
50	0.32	0.13	0.08
100	0.16	0.06	0.04
200	0.08	0.03	0.02

Table 3 explicitly shows the similarities and differences between the approaches outlined in Peterson et al. (2002) and the protocol of Bonar et al. (1997). The basic conceptual approach is the same in both cases, but the capture efficiency and density models are more complex in Peterson et al. (2002). The latter protocol attempts to model density from actual observations with a gamma distribution instead of using a minimum fixed threshold Poisson density. The key issue is not the choice of density model, but rather the use of actual densities instead of minimum threshold densities.

Habitat Based Capture Efficiency

Because bull trout capture efficiency is a key component in sample size calculations, Peterson et al. (2002) devoted a substantial effort into improving our understanding of this parameter by attempting to relate it to habitat and environmental characteristics. They used logistic regression to model the observed proportion of known marked fish recovered in different stream reaches against habitat and environmental parameters such as conductivity, percent undercut banks and cross sectional area. They also included an additional 'dispersion' parameter in the model fitting to scale the standard errors of the coefficient estimates upward because they felt the observed data exhibited more variability than the binomial variance of a logistic regression would allow.

Because the Peterson et al. (2002) protocol was partly based on measurements from Idaho streams, Thurow et al. (2003 and 2004) undertook field studies in Washington to empirically measure and model capture efficiencies for the three most common approaches to detecting bull trout:

1) day snorkeling,
2) night snorkeling, and
3) electrofishing.

They selected streams known to contain bull trout populations on both sides of the Washington Cascades. Observed capture efficiencies (Thurow et al. 2003) based on mark and recapture data were typically less than 50%. For bull trout, three-pass electrofishing appeared to be the most efficient method under average conditions (36.6%) with a 95% confidence range of 31.7% to 41.7%, followed by night snorkeling (24.5%) with a 95% confidence range of 20.5% to 29.0%, and day snorkeling (10.5%) with a 95% confidence range of 7.3% to 14.9%. They estimated the root mean squared error from a cross validation study with values ranging from 15.6% to 32.9%, implying the difference between the true and predicted capture efficiencies might be as much as 2*0.156 = 0.312 to 2*0.329 = 0.658, creating rather large discrepancies with significant impacts on sample size recommendations. With such large potential errors, we felt it was important to conduct an independent validation of the model.

Table 3. Comparison of statistical models used for bull trout detection.

Model component	General statistical interpretation	Bonar et al. 1997	Peterson et al. 2002
Threshold density	Density $f(x \mid \theta)$	Poisson $\rho(x \mid \mu) = \dfrac{e^{-\mu}\mu^{x}}{x!}$	Gamma estimated as $\mathcal{G}(x \mid a,b) = \dfrac{b^{x}}{(1+b)^{a+x}} \dfrac{\Gamma(a+x)}{\Gamma(x+1)\Gamma(a)}$
Probability of not detecting any bull trout given x are present	$P(0 \mid x)$	$\prod_{i=1}^{x}(1-q) = (1-q)^{x}$	$\dfrac{\Gamma\left(\frac{1}{\gamma}\right)\Gamma\left(x+\frac{1-q}{\gamma}\right)}{\Gamma\left(\frac{1-q}{\gamma}\right)\Gamma\left(x+\frac{1}{\gamma}\right)}$
Probability of no bull trout in *n* samples	$\left[\sum_{x=0}^{\infty} P(0 \mid x) f(x,\theta)\right]^{n}$	$\left[\sum_{x=0}^{\infty} (1-q)^{x} \dfrac{e^{-\mu}\mu^{x}}{x!}\right]^{n} = e^{-n\mu q}$	$\left[\sum_{x=0}^{\infty}\left(\dfrac{\Gamma\left(\frac{1}{\gamma}\right)\Gamma\left(x+\frac{1-q}{\gamma}\right)}{\Gamma\left(\frac{1-q}{\gamma}\right)\Gamma\left(x+\frac{1}{\gamma}\right)}\right)\dfrac{b^{x}}{(1+b)^{a+x}} \dfrac{\Gamma(a+x)}{\Gamma(x+1)\Gamma(a)}\right]^{n}$
Sample size guideline is solution to:	$\left[\sum_{x=0}^{\infty} P(0 \mid x) f(x,\theta)\right]^{n} = \beta$	$n = \dfrac{-\ln(\beta)}{\mu q}$	$n = \dfrac{\ln(\beta)}{\ln\left[\sum_{x=0}^{\infty}\left(\dfrac{\Gamma\left(\frac{1}{\gamma}\right)\Gamma\left(x+\frac{1-q}{\gamma}\right)}{\Gamma\left(\frac{1-q}{\gamma}\right)\Gamma\left(x+\frac{1}{\gamma}\right)}\right)\dfrac{b^{x}}{(1+b)^{a+x}} \dfrac{\Gamma(a+x)}{\Gamma(x+1)\Gamma(a)}\right]}$
Interpretation of finding no bull trout in *n* samples	It was unlikely that densities were higher than those described by $f(x \mid \theta)$.	It was unlikely that densities were higher than those described by a Poisson(μ).	It was unlikely that densities were higher than those described by the Gamma(a, b). With a and b estimated from Idaho data, it was unlikely that densities were higher than those in Idaho.

WDFW 2003 Validation Study

Washington Department of Fish and Wildlife (WDFW) sampled nine Washington streams from July through the first week in September in 2003 to measure bull trout capture efficiencies with electrofishing and night snorkel gear. The intent of the study was to see how well the habitat model predicted capture efficiency for the two sampling methods. The approach augmented the formal prediction error analysis of Thurow et al. (2004) and the streams were a subset of those studied by them in 2002. The habitat criteria measured in the study, however, were based on those reported by Peterson et al. (2003) in their study of effects of survey techniques on fish movement patterns. These criteria were categorical values, comparable to the categorical criteria specified in the Peterson et al. (2002) protocol.

Methods

Fish Surveys

Block nets were installed on two to three 100 m reaches per stream where bull trout were expected to be found. Sampling units in some streams were adjacent, but separated by non-sampled reaches in others. Block nets remained in place for the duration of the sampling and were regularly checked for tightness.

After block net installation, two passes with unpulsed DC electrofishing gear were made (one upstream, one downstream) to capture and mark bull trout. Only juvenile bull trout were collected, although the presence of other species was noted. Captured fish were anesthetized (MS-222), marked, and then released back into the netted stream after the anesthetic wore off. A combination adipose fin clip with either a dorsal or caudal punch was used to distinguish fish from any marked survivors of the 2002 field studies. Twenty-four stream reaches were studied. Most were east of the Cascades, but two streams in the Nooksack drainage were also included.

After a 24-hour waiting period, a night snorkel survey was made in each sample unit. Workers surveyed in an upstream direction using a diving light. Marked and unmarked bull trout were counted and recorded. A single-pass electrofishing survey was conducted the following day, also in the upstream direction. Both marked and unmarked fish captured by the gear were placed in live wells. After counting and recording, the fish were released back into the stream. Figure 1 shows the time sequence of activities within each sampling unit.

Habitat Surveys

Several key habitat measures were recorded at each sample site. Peterson et al. (2003) reported percent undercut banks (as a fraction of total bank length) to be a factor affecting night snorkel capture efficiency. Undercuts are areas defined by overhanging banks, boulders, bedrock or wood that are within a 0.5 m of the water surface. Submerged undercut banks, boulders, bedrock or wood within 0.5 m of the water surface are included as well. Length and average width of each undercut were measured within a sampling unit. Percent undercut was defined as the fraction of the total sample unit area occupied by undercut areas. Likewise, they considered conductivity and mean wetted cross-sectional area in addition to percent undercut banks as key factors in electrofishing capture efficiency. Conductivity was measured with a calibrated meter and approximately ten transects were established for measuring stream width and depth. Table 4 shows the habitat measurements of the target variables at each sample site.

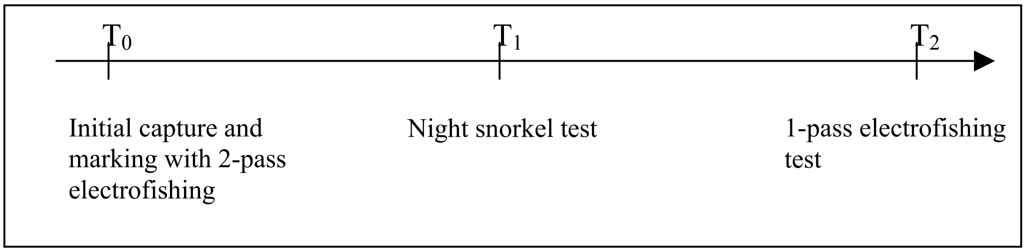

Figure 1. Schematic illustrating the sequence of fishing events in the WDFW validation study.

Table 4. Bull trout sample sites and measurements of habitat variables, WDFW 2003 sampling.

Basin	Stream	Unit	Number Transects	Conduct-ivity (μ ohms)	Temp. (°C)	Length (m)	Mean Wetted Width (m)	Mean Depth (m)	Mean Cross-sectional Area (m^2)	% Undercut Banks
Methow	Pine Creek	1	10	16	10.4	99.2	5.9	0.12	0.57	1.566
Methow	Pine Creek	2	10	16	10.4	95.0	3.6	0.35	0.65	1.881
Methow	Pine Creek	3	8	17	8.0	79.2	4.4	0.14	0.68	1.399
NF Nooksack	Whistler Creek	1	10	74	10.0	111.0	2.1	0.08	0.36	0.914
NF Nooksack	Whistler Creek	2	10	74	10.0	121.0	2.9	0.12	0.42	0.033
SF Nooksack	Bell Creek	1	10	87	9.0	107.0	7.4	0.12	0.77	0.257
SF Nooksack	Bell Creek	2	10	87	9.0	100.0	7.1	0.30	0.84	0.227
SF Nooksack	Bell Creek	3	10	87	9.0	105.0	3.9	0.18	0.65	1.316
Tucannon	Meadow	1	10	55	11.0	109.5	4.0	0.24	0.71	2.284
Tucannon	Meadow	2	10	55	11.0	103.5	3.8	0.15	0.39	2.173
Tucannon	Meadow	3	10	55	11.0	85.0	6.0	0.12	0.34	2.641
Twisp	EF Buttermilk	1	10	90	12.0	111.0	4.1	0.18	0.55	0.894
Twisp	EF Buttermilk	2	10	90	12.0	117.0	3.0	0.18	0.81	1.158
Twisp	EF Buttermilk	3	10	90	12.0	124.0	4.4	0.31	0.77	1.086
Twisp	Reynolds	1	11	71	9.5	106.0	4.1	0.20	0.70	1.291
Twisp	Reynolds	2	9	68	11.5	90.0	4.0	0.28	0.74	0.606
Twisp	Reynolds	3	10	68	11.5	88.0	4.2	0.09	0.81	0.558
Yakima	Deep Creek	1	13	36	9.0	139.6	5.3	0.18	0.75	1.344
Yakima	Deep Creek	2	11	36	9.0	113.5	5.6	0.20	1.45	1.012
Yakima	MF Ahtanum	1	10	45	7.5	94.0	3.1	0.16	0.43	4.064
Yakima	MF Ahtanum	2	10	42	8.0	105.0	3.7	0.10	0.42	4.314
Yakima	MF Ahtanum	3	11	42	8.0	123.0	2.9	0.06	0.33	5.662
Yakima	Shellneck	1	10	30	6.0	92.0	4.5	0.20	0.47	4.920
Yakima	Shellneck	2	9	30	6.0	76.5	3.8	0.13	0.45	1.255

Results for Night Snorkeling

Table 5 summarizes the results from the WDFW night snorkeling trials. The predicted capture efficiencies (from Table 9 in Peterson et al. 2003) are defined by percent undercut banks, i.e., greater or lesser than 1.6% of the bank length. Figure 2 shows a fair degree of scatter in the observed capture efficiencies. Arrayed separately by the number of bull trout marked, Figure 3 and Figure 4 show that 95% confidence intervals[1] cover the predictions for the most part, and as expected, confidence intervals generally narrow as the number of marked bull trout increases. Interestingly, the data do not suggest that capture efficiency increases with percent of undercut banks as in Peterson et al. (2003).

Table 5. Bull trout night snorkel results, WDFW 2003 sampling.

Basin	Stream	Unit	Mark Date	Marked Bull Trout	Snorkel Date	Marks Recap-tured	Observed[1] Capture Efficiency	Predicted[2] Capture Efficiency	% Undercut Banks
Methow	Pine Creek	1	14-Jul-03	18	15-Jul-03	3	0.17	0.222	1.57
Methow	Pine Creek	2	14-Jul-03	12	15-Jul-03	4	0.33	0.258	1.88
Methow	Pine Creek	3	15-Jul-03	18	16-Jul-03	4	0.22	0.222	1.40
NF Nooksack	Whistler Crk	1	02-Sep-03	57	03-Sep-03	19	0.33	0.222	0.91
NF Nooksack	Whistler Crk	2	02-Sep-03	75	03-Sep-03	38	0.51	0.222	0.03
SF Nooksack	Bell Creek	1	25-Aug-03	42	26-Aug-03	3	0.07	0.222	0.26
SF Nooksack	Bell Creek	2	25-Aug-03	46	26-Aug-03	6	0.13	0.222	0.23
SF Nooksack	Bell Creek	3	25-Aug-03	55	26-Aug-03	15	0.27	0.222	1.32
Tucannon	Meadow	1	29-Jul-03	3	30-Jul-03	0	0.00	0.258	2.28
Tucannon	Meadow	2	29-Jul-03	5	30-Jul-03	1	0.20	0.258	2.17
Tucannon	Meadow	3	29-Jul-03	3	30-Jul-03	1	0.33	0.258	2.64
Twisp	EF Buttermilk	1	19-Aug-03	14	20-Aug-03	3	0.21	0.222	0.89
Twisp	EF Buttermilk	2	19-Aug-03	15	20-Aug-03	2	0.13	0.222	1.16
Twisp	EF Buttermilk	3	19-Aug-03	28	20-Aug-03	13	0.46	0.222	1.09
Twisp	Reynolds	1	05-Aug-03	16	06-Aug-03	7	0.44	0.222	1.29
Twisp	Reynolds	2	04-Aug-03	9	05-Aug-03	2	0.22	0.222	0.61
Twisp	Reynolds	3	04-Aug-03	12	05-Aug-03	2	0.17	0.222	0.56
Yakima	Deep Creek	1	22-Jul-03	7	23-Jul-03	3	0.43	0.222	1.34
Yakima	Deep Creek	2	22-Jul-03	11	23-Jul-03	5	0.45	0.222	1.01
Yakima	MF Ahtanum	1	12-Aug-03	6	13-Aug-03	0	0.00	0.258	4.06
Yakima	MF Ahtanum	2	11-Aug-03	12	12-Aug-03	2	0.17	0.258	4.31
Yakima	MF Ahtanum	3	11-Aug-03	11	12-Aug-03	3	0.27	0.258	5.66
Yakima	Shellneck	1	07-Jul-03	16	08-Jul-03	3	0.19	0.258	4.92
Yakima	Shellneck	2	07-Jul-03	3	08-Jul-03	1	0.33	0.222	1.25

[1] The number of marks recaptured divided by the number of marked bull trout.

[2] From table 9 in Peterson et al. 2003.

[1] From the R package 'Hmisc' – uses preferred method of: Agresti, A. and B.A. Coull, Approximate is better than "exact" for interval estimation of binomial proportions, *American Statistician,* **52**:119–126, 1998

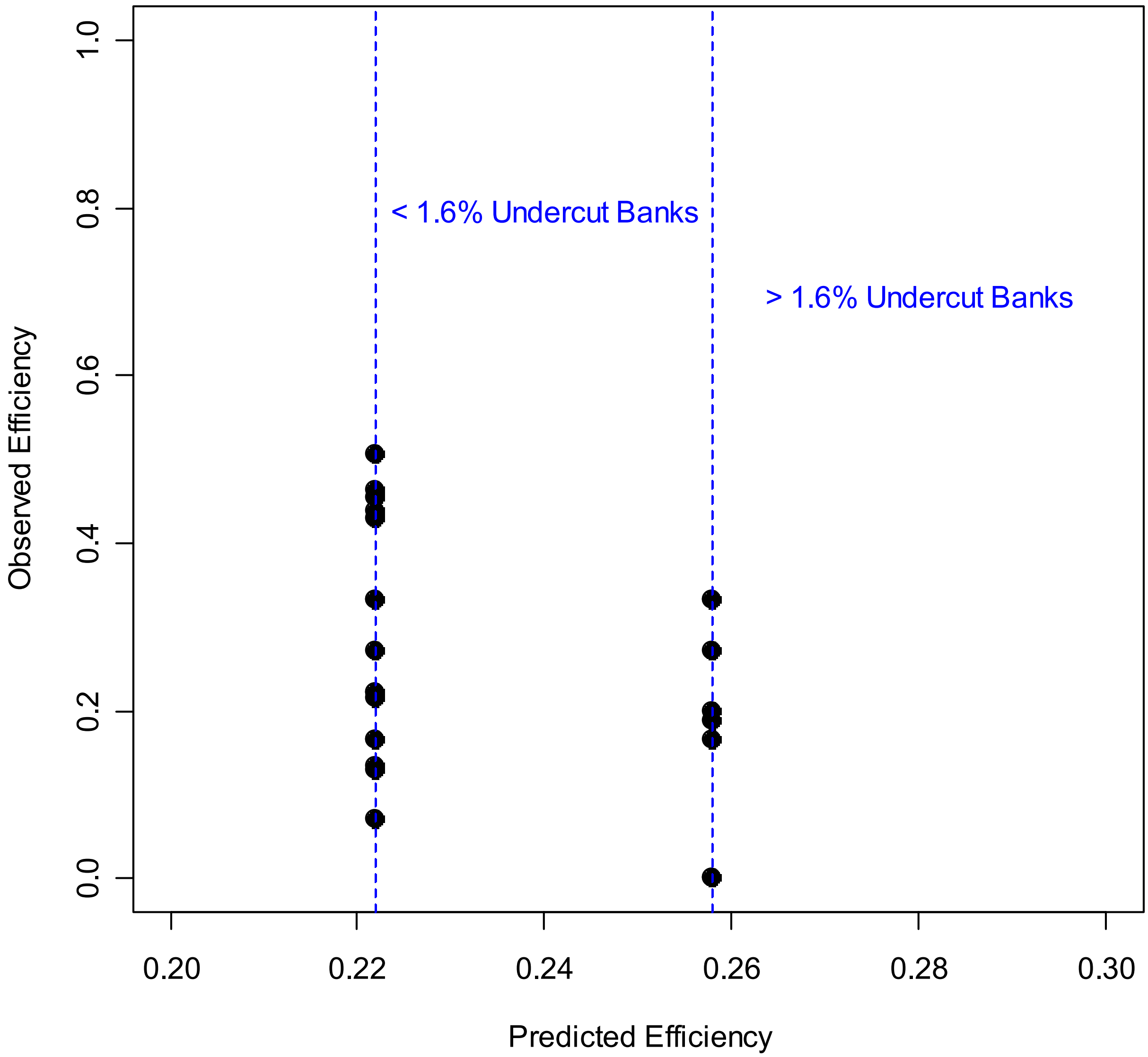

Figure 2. WDFW 2003 bull trout night snorkeling observed vs. predicted capture efficiency as a function of % undercut banks.

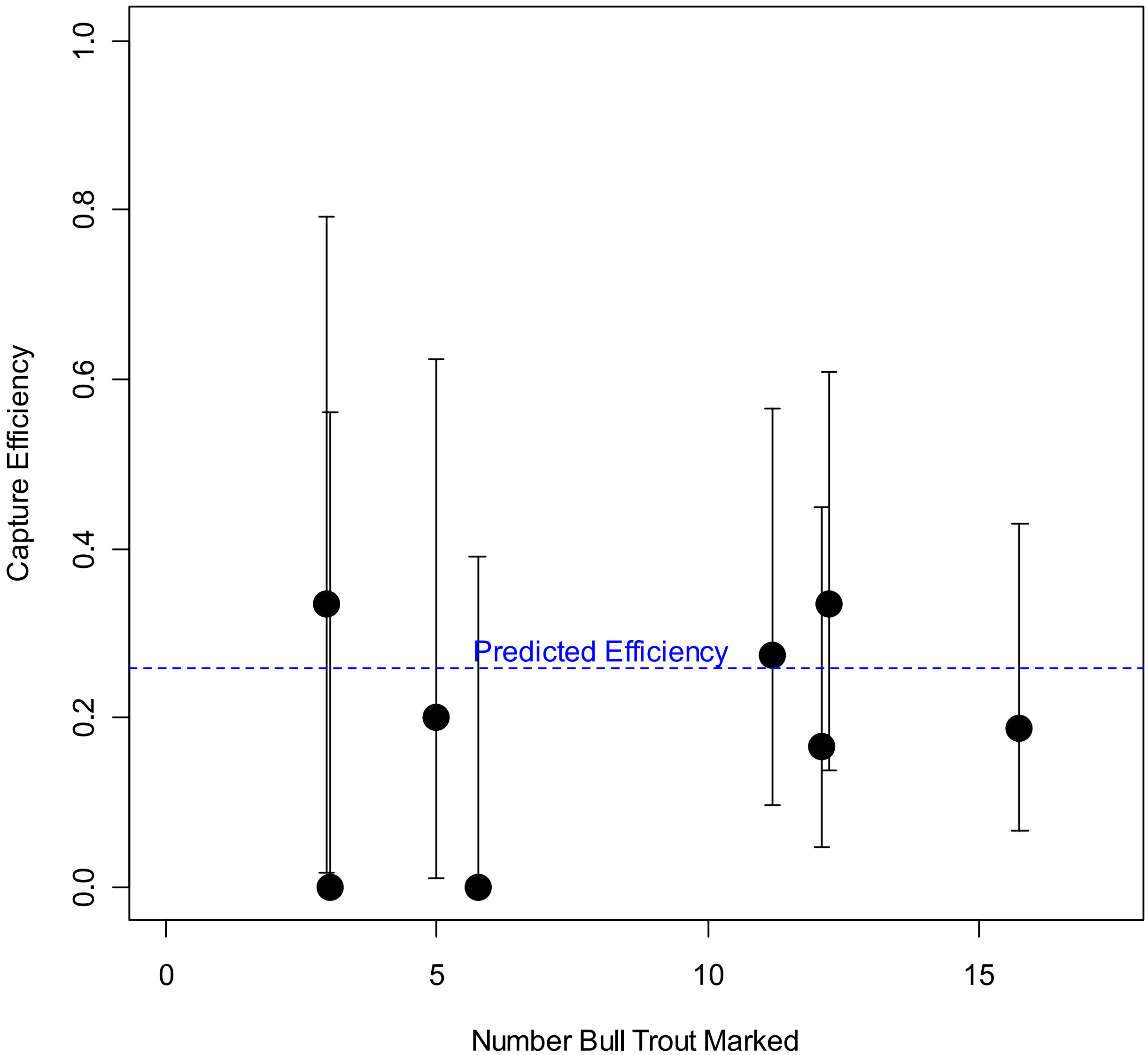

Figure 3. WDFW 2003 bull trout night snorkeling observed capture efficiency (95% CI) by number of marked bull trout for percent undercut banks > 1.6%.

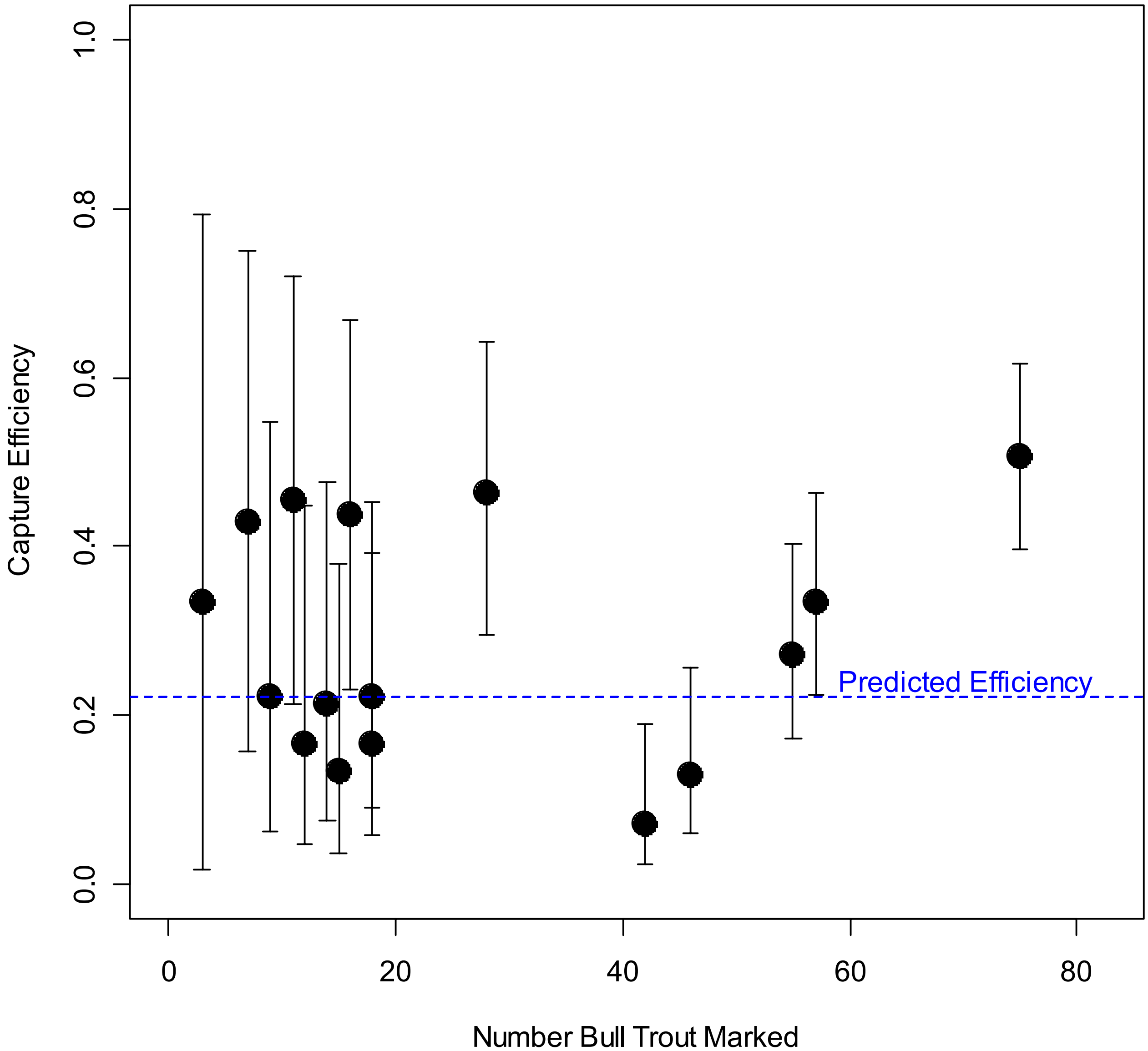

Figure 4. WDFW 2003 bull trout night snorkeling observed capture efficiency (95% CI) by number of marked bull trout for percent undercut banks < 1.6%.

Results for Electrofishing

The predicted efficiencies in Table 6 were again taken from Peterson et al. (2003, Table 10). Two additional habitat measures, conductivity (2 levels) and mean cross-sectional area (2 levels), were used in addition to percent undercut banks to categorize the predictions. This yielded a total of eight categories, reproduced in Table 7. However, not all the categories were encountered in the 2003 validation study.

Table 6. Bull trout electrofishing results, WDFW 2003 sampling.

Basin	Stream	Unit	Mark Date	Marked Bull Trout	Electro-fishing Date	Marks Recaptured	Observed[1] Capture Efficiency	Predicted[2] Capture Efficiency	% Undercut Banks	Conduct-ivity	Mean Cross-Sectional Area
Methow	Pine Creek	1	14-Jul-03	18	16-Jul-03	0	0.000	0.186	1.57	16	0.57
Methow	Pine Creek	2	14-Jul-03	12	16-Jul-03	1	0.083	0.118	1.88	16	0.65
Methow	Pine Creek	3	15-Jul-03	18	17-Jul-03	7	0.389	0.186	1.40	17	0.68
NF Nooksack	Whistler Creek	1	02-Sep-03	57	04-Sep-03	24	0.421	0.264	0.91	74	0.36
NF Nooksack	Whistler Creek	2	02-Sep-03	75	04-Sep-03	46	0.613	0.264	0.03	74	0.42
SF Nooksack	Bell Creek	1	25-Aug-03	42	27-Aug-03	5	0.119	0.264	0.26	87	0.77
SF Nooksack	Bell Creek	2	25-Aug-03	46	27-Aug-03	5	0.109	0.264	0.23	87	0.84
SF Nooksack	Bell Creek	3	25-Aug-03	55	27-Aug-03	12	0.218	0.264	1.32	87	0.65
Tucannon	Meadow	1	29-Jul-03	3	31-Jul-03	0	0.000	0.170	2.28	55	0.39
Tucannon	Meadow	2	29-Jul-03	5	31-Jul-03	0	0.000	0.170	2.17	55	0.39
Tucannon	Meadow	3	29-Jul-03	3	31-Jul-03	1	0.333	0.170	2.64	55	0.34
Twisp	EF Buttermilk	1	19-Aug-03	14	21-Aug-03	8	0.571	0.264	0.89	90	0.55
Twisp	EF Buttermilk	2	19-Aug-03	15	21-Aug-03	8	0.533	0.264	1.16	90	0.81
Twisp	EF Buttermilk	3	19-Aug-03	28	21-Aug-03	4	0.143	0.264	1.09	90	0.77
Twisp	Reynolds	1	05-Aug-03	16	07-Aug-03	3	0.188	0.264	1.29	71	0.70
Twisp	Reynolds	2	04-Aug-03	9	06-Aug-03	1	0.111	0.264	0.61	68	0.74
Twisp	Reynolds	3	04-Aug-03	12	06-Aug-03	4	0.333	0.264	0.56	68	0.81
Yakima	Deep Creek	1	22-Jul-03	7	24-Jul-03	1	0.143	0.186	1.34	36	0.75
Yakima	Deep Creek	2	22-Jul-03	11	24-Jul-03	2	0.182	0.140	1.01	36	1.45
Yakima	MF Ahtanum	1	12-Aug-03	6	14-Aug-03	4	0.667	0.118	4.06	45	0.43
Yakima	MF Ahtanum	2	11-Aug-03	12	13-Aug-03	4	0.333	0.118	4.31	42	0.42
Yakima	MF Ahtanum	3	11-Aug-03	11	13-Aug-03	4	0.364	0.118	5.66	42	0.33
Yakima	Shellneck	1	07-Jul-03	16	09-Jul-03	4	0.250	0.118	4.92	30	0.47
Yakima	Shellneck	2	07-Jul-03	3	09-Jul-03	0	0.000	0.186	1.25	30	0.45

1 The number of marks recaptured divided by the number of marked bull trout.

2 From table 10 in Peterson et al. 2003.

Table 7. Predicted single-pass electrofishing capture efficiency in 100 m stream sections by habitat variables (from Peterson et al. 2003, Table 10).

Percent Undercut Banks	Mean Cross-Sectional Area (m^2)	Conductivity (μ ohms)	Predicted Capture Efficiency
≤ 1.6%	≤ 1.00	> 53	0.264
≤ 1.6%	> 1.00	> 53	0.203
≤ 1.6%	≤ 1.00	≤ 53	0.186
> 1.6%	≤ 1.00	> 53	0.170
≤ 1.6%	> 1.00	≤ 53	0.140
> 1.6%	> 1.00	> 53	0.127
> 1.6%	≤ 1.00	≤ 53	0.118
> 1.6%	> 1.00	≤ 53	0.087

Figure 5 shows the degree of variability in the observed capture efficiencies. Capture efficiencies at the lowest predictive level (> 1.6% undercut banks, ≤ 1 m^2 cross-sectional area, and ≤ 53 μohms conductivity) were generally higher than expected as shown more clearly in Figure 6. Overall, slightly more than half (13 of 24) of the electrofishing capture efficiency estimates yielded 95% confidence limits that encompass the predicted or mean efficiency (Figure 6 through Figure 10). Of the eleven remaining estimates, the majority exhibited higher than expected efficiencies.

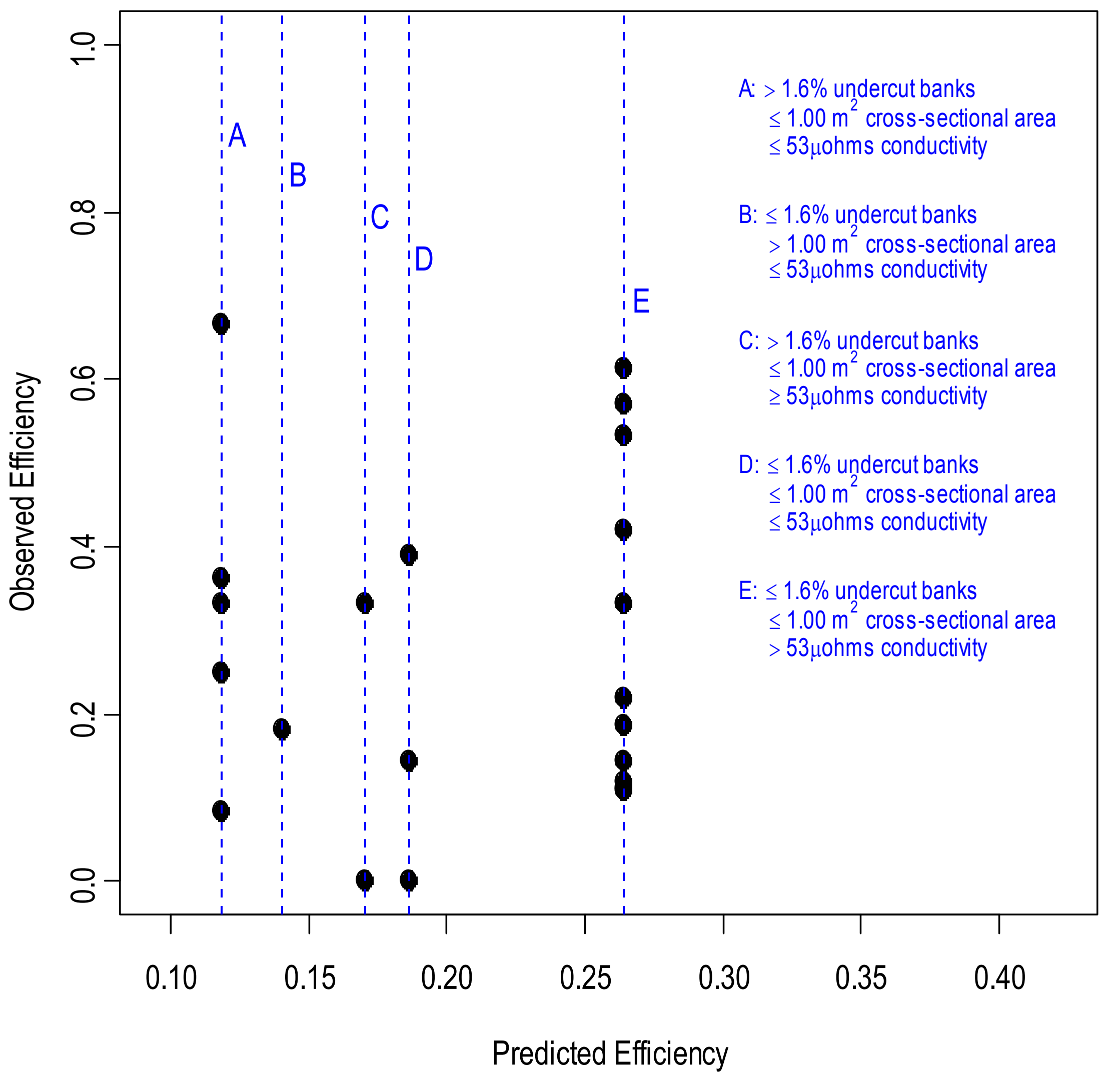

Figure 5. WDFW 2003 bull trout single-pass electro-fishing observed versus predicted capture efficiency by habitat measures.

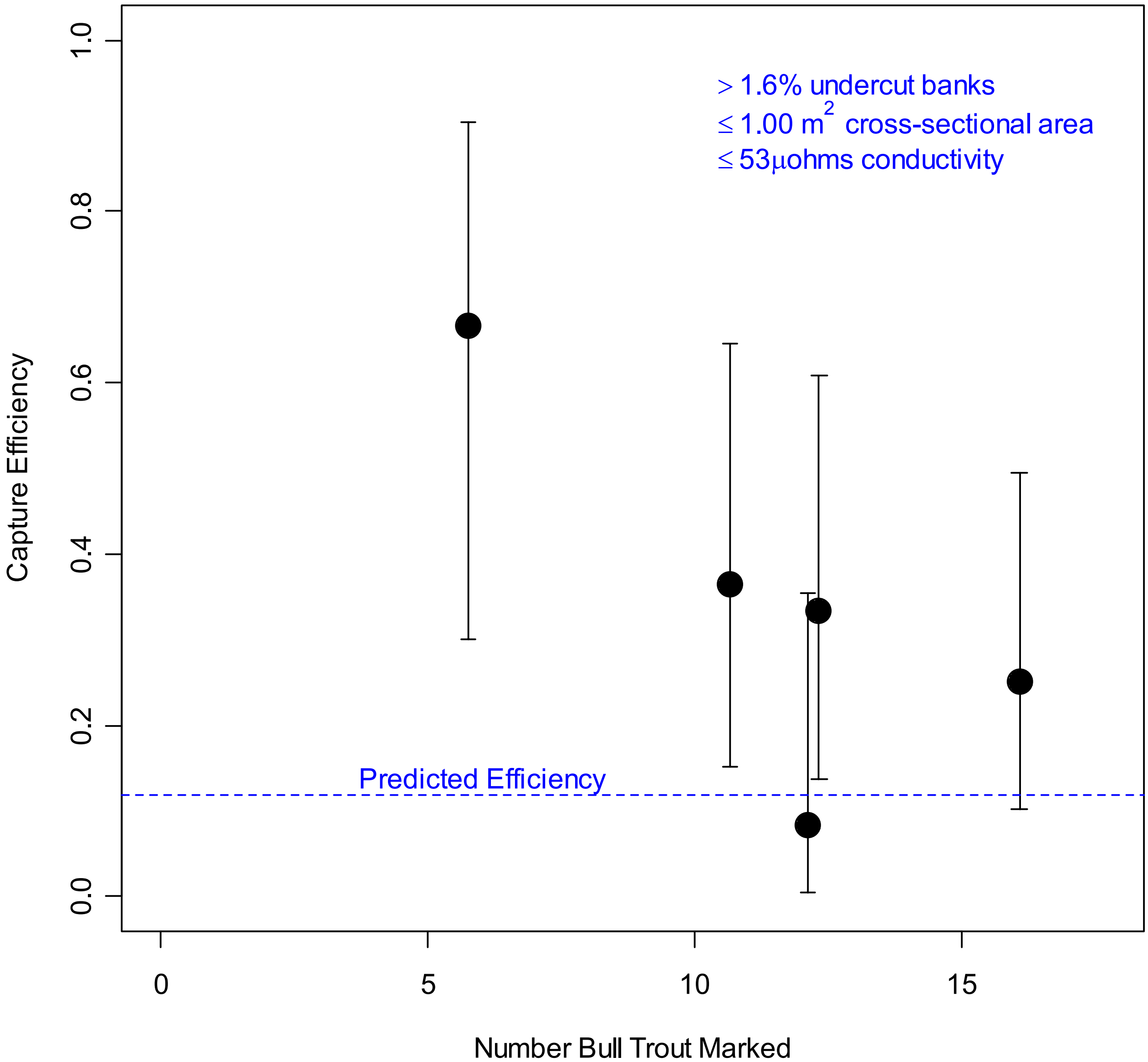

Figure 6. WDFW 2003 bull trout electro-fishing observed capture efficiency (95% CI) for habitat category 'A' by number of marked bull trout.

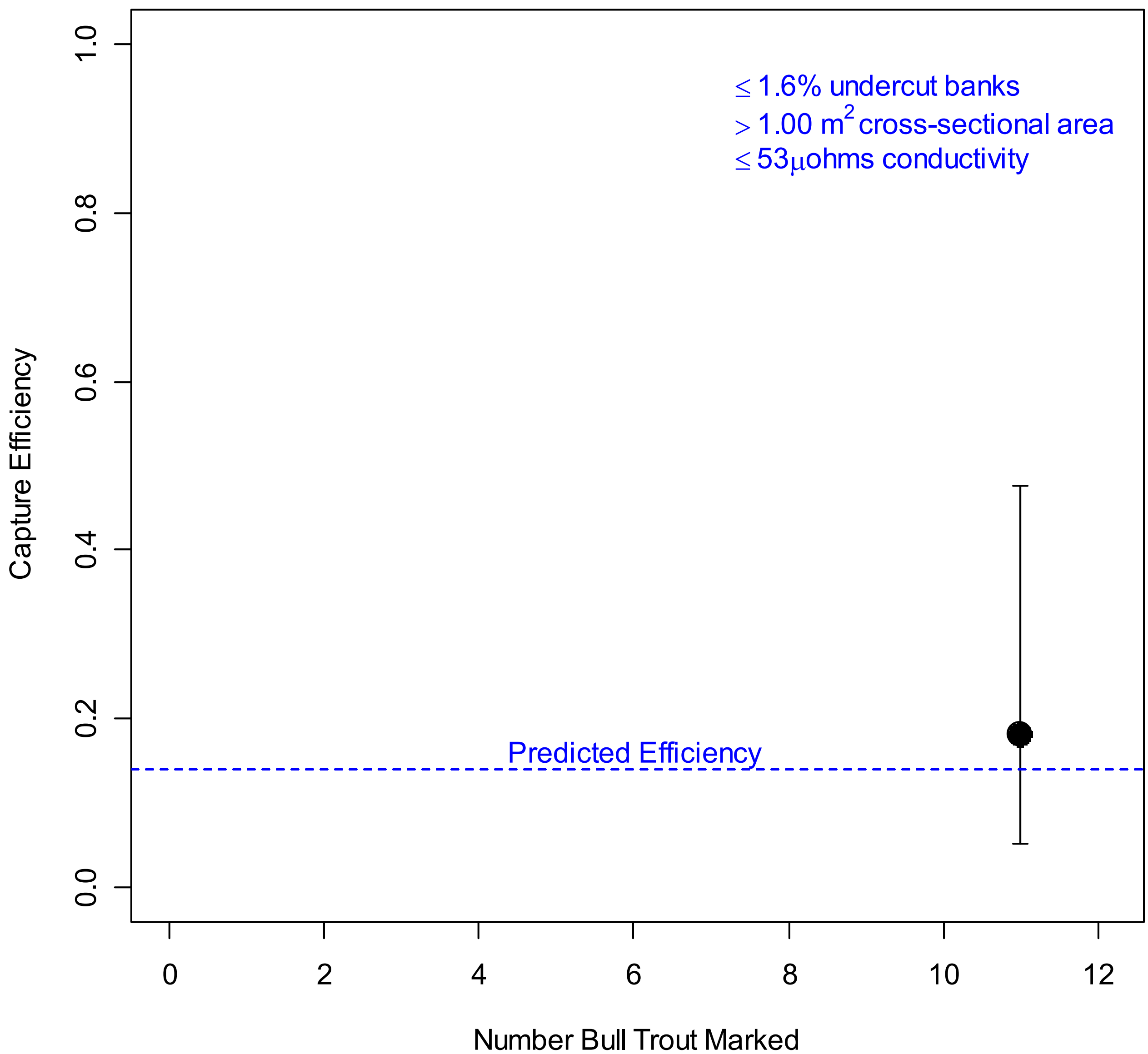

Figure 7. WDFW 2003 bull trout electro-fishing observed capture efficiency (95% CI) for habitat category 'B' by number of marked bull trout.

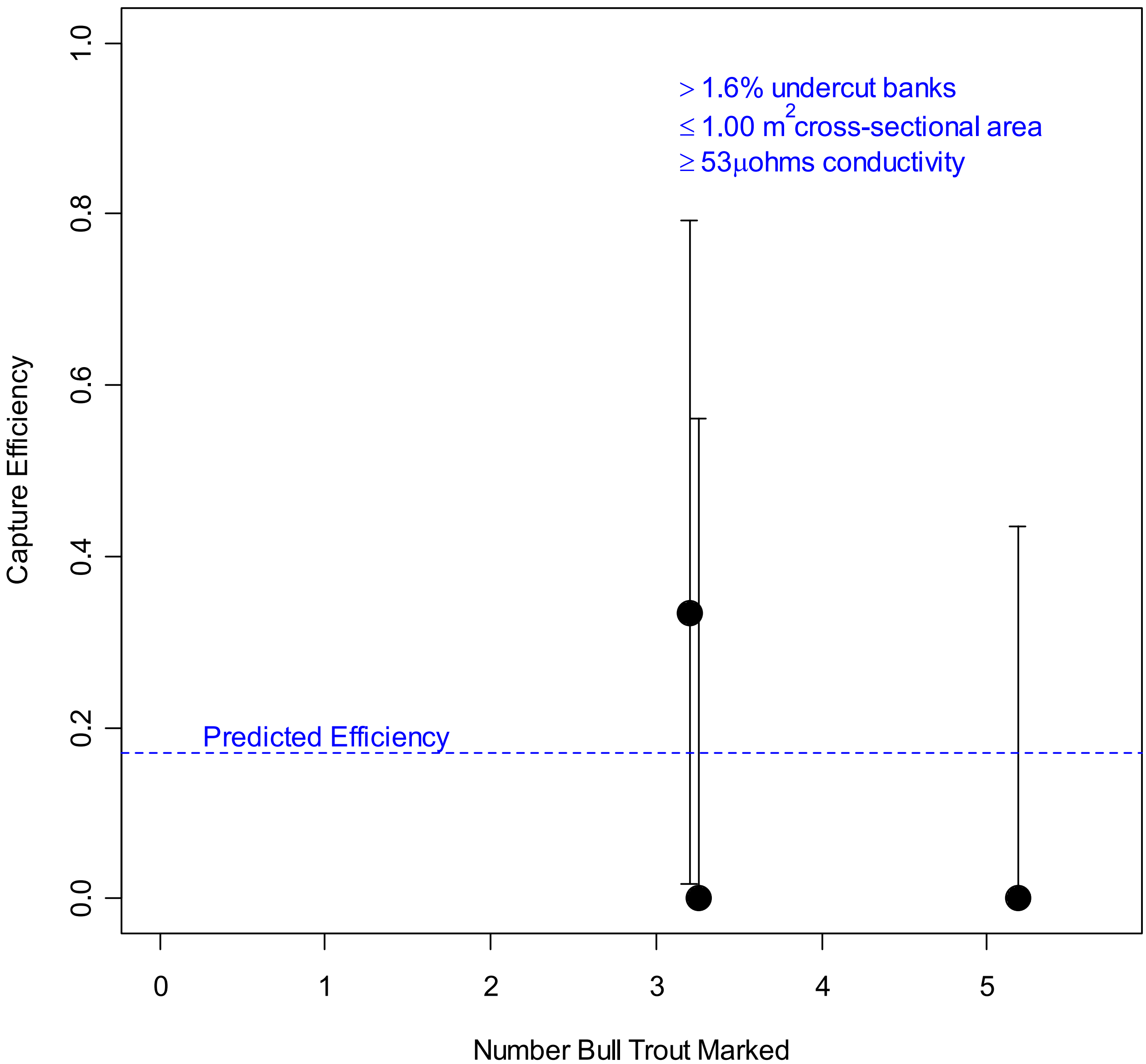

Figure 8. WDFW 2003 bull trout electrofishing observed capture efficiency (95% CI) for habitat category 'C' by number of marked bull trout.

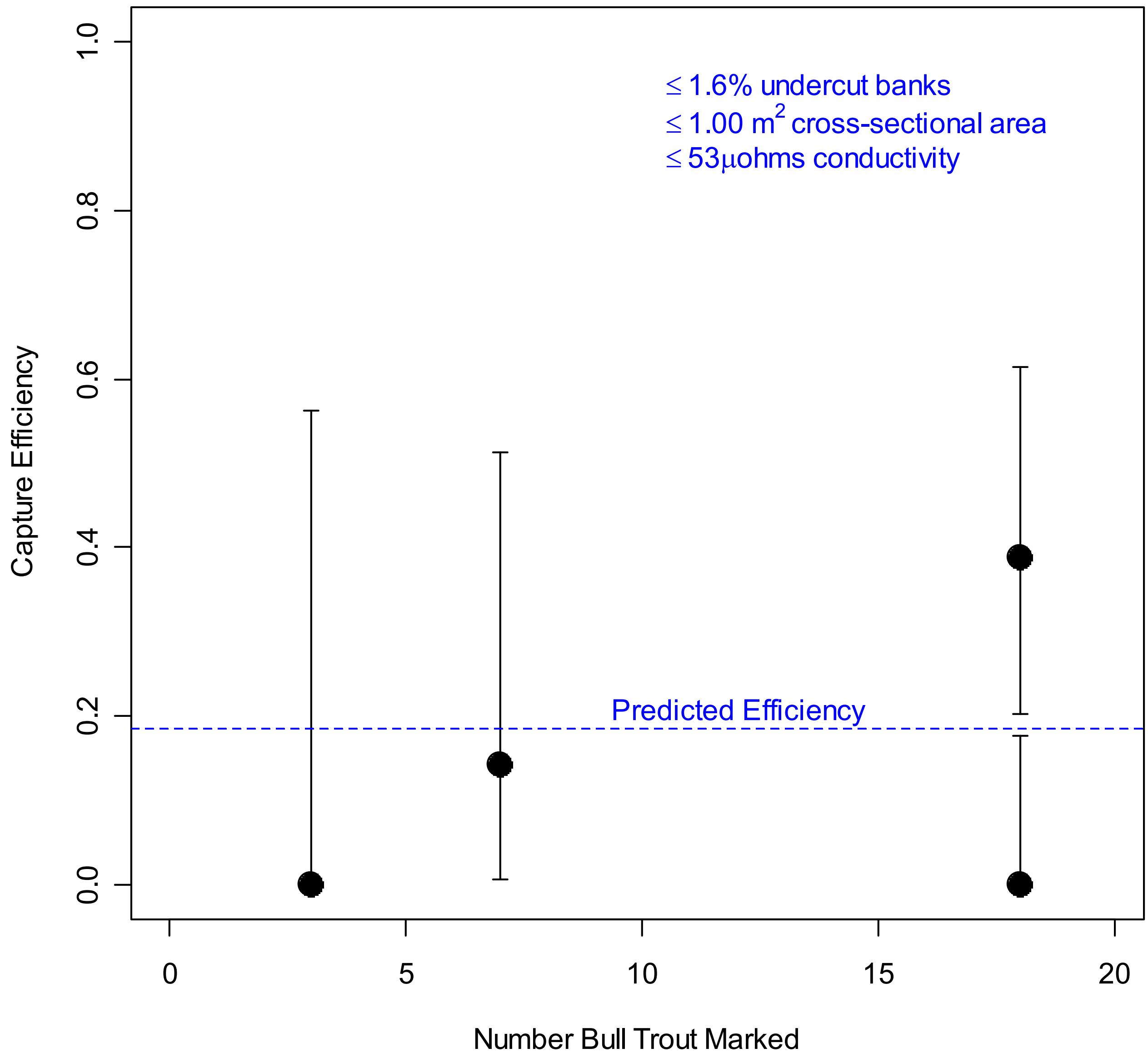

Figure 9. WDFW 2003 bull trout electro-fishing observed capture Efficiency (95% CI) for habitat category 'D' by number of marked bull trout.

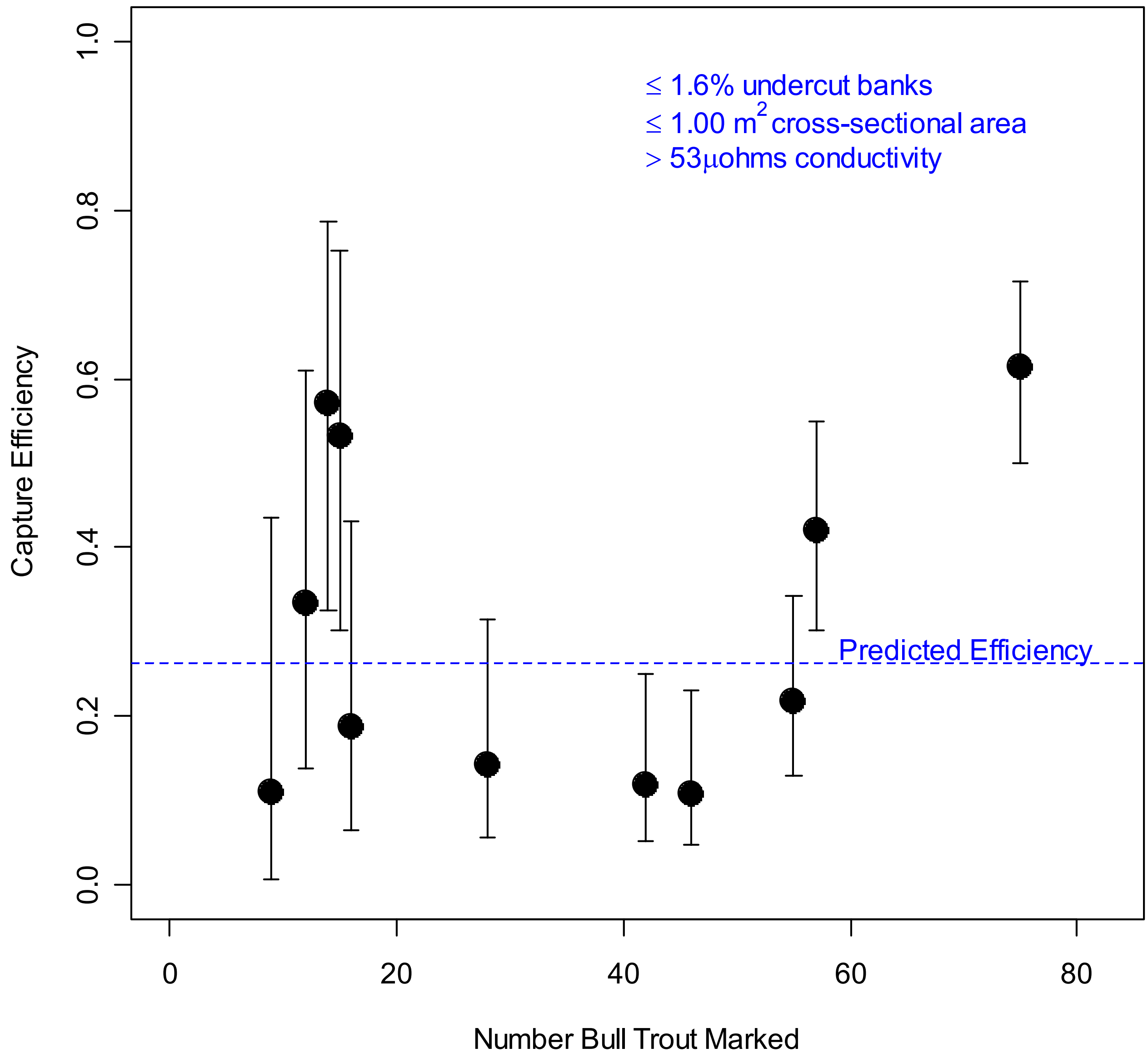

Figure 10. WDFW 2003 bull trout electro-fishing observed capture Efficiency (95% CI) for habitat category 'E' by number of marked bull trout.

Prediction Error

Prediction models inevitably contain prediction error or bias, i.e., the difference between the true parameter and the predicted parameter, since it is virtually impossible to incorporate every key variable into a model. Large prediction errors can render a model ineffective; therefore, evaluation of the prediction error can be informative on the utility of a particular model. As there is evidence of prediction error in the habitat model (Figures 2 and 5), we attempted to assess the magnitude of it using the data from the WDFW validation study.

In the WDFW validation study, observed capture efficiencies (OE) were calculated as the percent of known marked fish recaptured. For any sample unit (a stream section) there were *M* test fish initially marked at T_0 (Figure 1). For both the night snorkeling test conducted at T_1 and the electrofishing test conducted at T_2 capture efficiency was simply the proportion of marked fish observed. Since the number detected is a random variable, we expect some stochastic error in the observed efficiency so that differences in the OE and predicted efficiency (PE) are not wholly attributable to the prediction error. For some habitat category *H*, the habitat model will produce a PE. Any difference between the true capture efficiency (CE) and PE is the prediction error or bias (Figure 11).

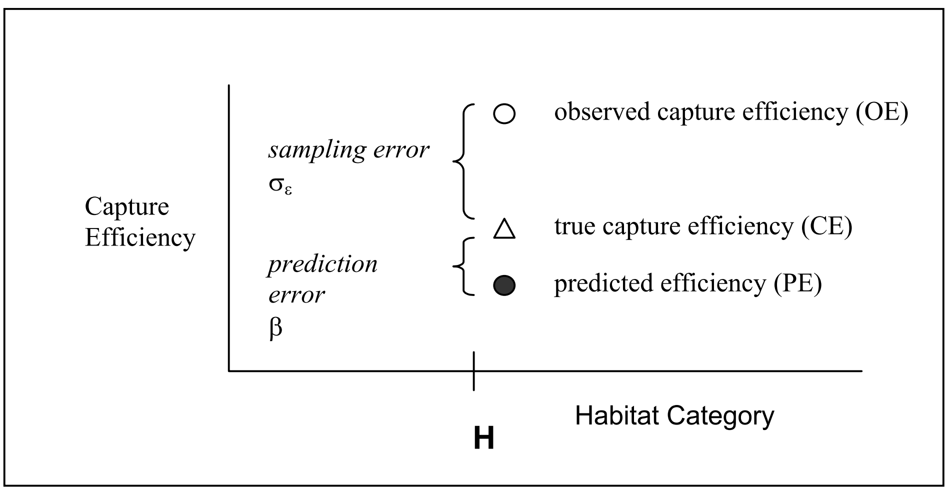

Figure 11. Schematic illustrating the differences among CE, PE and OE.

Although the CE is not observable, the bias of the prediction error can be estimated given a measure of the sampling variance and the mean squared error (MSE) of the observed efficiencies around the predicted efficiencies:

$$\begin{aligned} MSE(OE - PE) &= E(OE - PE)^2 \\ &= E(OE - CE + CE - PE)^2 \\ &= E(OE - CE)^2 + E(CE - PE)^2 \end{aligned}$$

$$= Variance(OE) + Bias^2(PE) = \sigma_\varepsilon^2 + \beta^2 \Rightarrow$$

$$\beta^2 = MSE - \sigma_\varepsilon^2.$$

where the sampling error (σ_ε^2) is estimated by the empirical variance among the observed capture efficiencies in a habitat category

In the validation study, the estimate of the MSE is the average squared difference between the OEs and the PEs.

One-pass Electrofishing

Table 8 provides the within and between error components for one-pass electrofishing for the habitat modeled PEs as well as for the historical PE of 0.25[2]. The two prediction error biases were estimated to be:

$$\beta^2_{\text{habitat model}} = 0.0431 - 0.0388 = 0.0043 \text{ and}$$

$$\beta^2_{\text{historical } 0.25} = 0.0391 - 0.0388 = 0.0003.$$

These biases translate into a prediction error of $2\sqrt{0.0043} = 0.131$ or 13.1% for the habitat based model and $2\sqrt{0.0003} = 0.034$ or 3.4% for the historical global value of 0.25.

Night Snorkeling

Table 9 provides the within and between error components for night snorkeling for both the habitat modeled PEs and for the historical PE of 0.25. The two prediction error biases were estimated to be:

$$\beta^2_{\text{habitat model}} = 0.0215 - 0.0184 = 0.0031 \text{ and}$$

$$\beta^2_{\text{historical } 0.25} = 0.0193 - 0.0184 = 0.0009.$$

These biases translate into a prediction error of $2\sqrt{0.0031} = 0.113$ or 11.3% for the habitat based model and $2\sqrt{0.0009} = 0.063$ or 6.3% for the historical global value of 0.25.

[2] See Bonar et al. 1997 pg. 10 and Rieman and McIntyre 1995 pg. 290.

Table 8. Variance statistics for the WDFW validation study 2003 for one-pass electroshocking.

Basin	Stream	Observed Capture Efficiency OE	Predicted[1] Capture Efficiency PE	OE-PE	OE-0.25	Observed Variance Within Each Habitat Grouping
Methow	Pine Creek	0.083	0.118	-0.035	-0.167	
Yakima	MF Ahtanum	0.667	0.118	0.549	0.417	
Yakima	MF Ahtanum	0.333	0.118	0.215	0.083	
Yakima	MF Ahtanum	0.364	0.118	0.246	0.114	
Yakima	Shellneck	0.250	0.118	0.132	0.000	$\sigma^2_{0.118}$= 0.0453
Yakima	Deep Creek	0.182	0.140	0.042	-0.068	$\sigma^2_{0.140}$= NA
Tucannon	Meadow	0.000	0.170	-0.170	-0.250	
Tucannon	Meadow	0.000	0.170	-0.170	-0.250	
Tucannon	Meadow	0.333	0.170	0.163	0.083	$\sigma^2_{0.170}$= 0.0370
Methow	Pine Creek	0.000	0.186	-0.186	-0.250	
Methow	Pine Creek	0.389	0.186	0.203	0.139	
Yakima	Deep Creek	0.143	0.186	-0.043	-0.107	
Yakima	Shellneck	0.000	0.186	-0.186	-0.250	$\sigma^2_{0.186}$= 0.0337
NF Nooksack	Whistler Crk	0.421	0.264	0.157	0.171	
NF Nooksack	Whistler Crk	0.613	0.264	0.349	0.363	
SF Nooksack	Bell Creek	0.119	0.264	-0.145	-0.131	
SF Nooksack	Bell Creek	0.109	0.264	-0.155	-0.141	
SF Nooksack	Bell Creek	0.218	0.264	-0.046	-0.032	
Twisp	EF Buttermilk	0.571	0.264	0.307	0.321	
Twisp	EF Buttermilk	0.533	0.264	0.269	0.283	
Twisp	EF Buttermilk	0.143	0.264	-0.121	-0.107	
Twisp	Reynolds	0.188	0.264	-0.077	-0.063	
Twisp	Reynolds	0.111	0.264	-0.153	-0.139	
Twisp	Reynolds	0.333	0.264	0.069	0.083	$\sigma^2_{0.264}$= 0.0391
Average						$\bar{\sigma}^2_{\varepsilon}$ = 0.0388
MSE				0.0431	0.0391	

[1] From Table 10 in Peterson et al. 2003.

Implications of Prediction Error

The implications of prediction error can be demonstrated by looking at the effects on sample size calculations and risk of concluding bull trout are not present at or above the threshold densities when infact they are for, say, one-pass electrofishing. In the case where the historical capture efficiency of 0.25 was used the prediction error was estimated to be ± 0.034, which indicates a likely interval capturing the true capture efficiency is (0.216, 0.284). For this illustration, say the true capture efficiency was 0.216. Then with a desired risk of β=0.2 and a threshold density defined by μ=0.06, the correct sample size would be 124 (1

From Table 9 in Peterson et al. 2003.

Table 10). However if the historical value of q=0.25 were used, the recommended sample size would be 107 that would result in a true risk of 0.25. On the other hand if the habitat model were used with a prediction error of 0.131 then the value of q used would be 0.347

resulting in a recommended sample size of 77 with an associated true risk of 0.37, almost twice that of the originally desired value.

Table 9. Variance statistics for the WDFW validation study 2003 for night snorkeling.

Basin	Stream	Observed Capture Efficiency OE	Predicted[1] Capture Efficiency PE	OE-PE	OE-0.25	Observed Variance within each Habitat Grouping
NF Nooksack	Whistler Ck	0.507	0.222	0.285	0.257	
SF Nooksack	Bell Creek	0.130	0.222	-0.092	-0.120	
SF Nooksack	Bell Creek	0.071	0.222	-0.151	-0.179	
Twisp	Reynolds	0.167	0.222	-0.055	-0.083	
Twisp	Reynolds	0.222	0.222	0.000	-0.028	
Twisp	EF Buttermilk	0.214	0.222	-0.008	-0.036	
NF Nooksack	Whistler Ck	0.333	0.222	0.111	0.083	
Yakima	Deep Creek	0.455	0.222	0.233	0.205	
Twisp	EF Buttermilk	0.464	0.222	0.242	0.214	
Twisp	EF Buttermilk	0.133	0.222	-0.089	-0.117	
Yakima	Shellneck	0.333	0.222	0.111	0.083	
Twisp	Reynolds	0.438	0.222	0.215	0.188	
SF Nooksack	Bell Creek	0.273	0.222	0.051	0.023	
Yakima	Deep Creek	0.429	0.222	0.207	0.179	
Methow	Pine Creek	0.222	0.222	0.000	-0.028	
Methow	Pine Creek	0.167	0.222	-0.055	-0.083	$\sigma^2_{0.222}$= 0.0195
Methow	Pine Creek	0.333	0.258	0.075	0.083	
Tucannon	Meadow	0.200	0.258	-0.058	-0.050	
Tucannon	Meadow	0.000	0.258	-0.258	-0.250	
Tucannon	Meadow	0.333	0.258	0.075	0.083	
Yakima	MF Ahtanum	0.000	0.258	-0.258	-0.250	
Yakima	MF Ahtanum	0.167	0.258	-0.091	-0.083	
Yakima	Shellneck	0.188	0.258	-0.070	-0.063	
Yakima	MF Ahtanum	0.273	0.258	0.015	0.023	$\sigma^2_{0.258}$= 0.0172
Average						$\bar{\sigma}^2_{\varepsilon}$ = 0.0184
MSE				0.0215	0.0193	

1 From Table 9 in Peterson et al. 2003.

Table 10. Comparison of errors in risk assessment as a function of prediction error in *q*. The sample size recommendation *n* was based on a desired risk of β=0.20 and a threshold density of μ=0.06.

Absolute Prediction Error	**Value used for *q***	**Recommended Sample Size (*n*)**	**True Risk (□)**
PE	*q* + PE	$n = \frac{-\ln(\beta_0)}{\mu_0 q} = \frac{-\ln(0.20)}{0.06\,q}$	$\beta = e^{-n\mu_0 q_0} = e^{-n(0.06)(0.216)}$
0 (true value)	0.216	124	0.20
0.034	0.250	107	0.25
0.131	0.347	77	0.37

Recommendations

Revisit the threshold density concept.

Sampling for bull trout is an inherently difficult problem, and the considerable efforts made by Peterson et al. (2003), Thurow et al. (2004), and others have improved the theoretical discussion of presence/absence sampling. However, there is substantial work to be done on refining the threshold density concept. Threshold densities are more likely to be relevant to management if they are based on biological criteria and incorporate policy and management considerations. For example, the choice of threshold might be based on a minimum viable population (biologically defined) over some time frame (defined by policy and management). The concept of using empirical bull trout densities from Idaho to set protocol sample sizes for Washington essentially declares that lesser densities are not important to detect. It is not clear that Washington policy makers and managers have agreed to that criterion or that such densities are relevant to assessing presence/absence in Washington.

Reanalyze habitat modeled capture efficiencies for practical value.

The fact that statistical models relating bull trout capture efficiency to various environmental variables have been developed in the past several years is an indicator that this is a potentially fruitful research area. However, concerns about excessive prediction error suggest the habitat models may not have improved on the past practice of using a constant average value for capture efficiency. If collection of habitat/environmental data is an expensive proposition, use of a habitat model approach could turn out to be an inefficient use of resources. Therefore, an assessment of the practical value of habitat modeled capture efficiencies ought to be conducted to facilitate the discussion of their utility.

Continue to research methods for improving actual gear sampling efficiency.

Gear capture efficiency is one component that is controllable to some degree and has a large influence on sample size requirements; more efficient gear requires fewer samples. Unless technological advances can be made in sampling gear (in the same manner, for example, that night snorkeling with artificial lights represents an improvement over day snorkeling) or other means can be found to improve the ability to detect bull trout, presence/absence protocols can at most provide blunt but expensive tools for detecting bull trout. Barring such improvements, if the protocols performance cannot meet requirements, then other methods will need to be devised.

Continue work on habitat-based models of bull trout presence.

Even though the practical value of habitat based capture efficiencies may not be evident, there may still be value in predicting bull trout presence from habitat parameters. Such relationships would be useful in using judgment (as in Rieman and McIntyre, 1995) to select sample sites to improve the power over using strictly randomly chosen sites or even to inform choices about protecting likely bull trout habitat. Studies by Watson and Hillman (1997), Dunham and Chandler (2001), Peterson and Banish (2002) and Rich et al. (2003) have illustrated the benefits likely to be encountered in this line of investigation as well as the difficulties. However, even with knowledge of habitat and presence, the sample sizes for presence/absence protocols should still be tied to threshold values that define key densities.

Continue to use Bonar et al. (1997) protocols until further suggested research is conducted.

Development of capture efficiency models as a function of habitat appears to be in an exploratory phase, as evidenced by the wide array of variables considered in recent modeling exercises (e.g., Peterson et al. 2002, Thurow et al. 2004). The habitat models do indicate some promising associations, but the practical value of the habitat-based capture efficiency approach is not yet verified. For these reasons, and because there is so much inherent variability in capture efficiency and so little of it appears to be explained by the habitat models developed so far, we recommend that Bonar et al.'s (1997) protocols be followed with the caveat that the threshold density value and the allowable risk be revisited and include policy and management considerations.

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