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[Article]

Utility of a Probabilistic Sampling Design to Determine Bull Trout Population Status Using Redd Counts in Basins of the Columbia River Plateau

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Abstract.—Redd counts are commonly used to monitor the current population status, trends in abundance, and distribution of bull trout *Salvelinus confluentus*. In many cases redd counts are conducted at subjectively selected sites, and there has been limited evaluation of statistical sampling designs. We evaluated the utility of the generalized random tessellation stratified (GRTS) sampling design to determine bull trout population status through redd counts. We tested a sampling effort that would be economically practical to implement on a continuous basis in multiple drainages within the southeastern Washington and Oregon portions of the Columbia River plateau. We evaluated the logistics of a pilot application of the GRTS design, compared GRTS-based estimates of redd abundance with those from census surveys, determined the precision of the GRTS estimates and the associated power for abundance comparisons, and compared the performance of the GRTS design with that of other probability sampling designs through simulation. A target of 50 sites per basin can be sampled by a two-person survey crew multiple times over the spawning season. At that level of effort, the precision of redd abundance estimates ranges from 15% to 35%, depending on the patchiness of the redd distribution and the extent of the target population. These levels of precision are suitable for detecting a 30–70% change in redd abundance. Direct comparisons of GRTS-based estimates with those obtained from a census showed mixed results. However, in a simulation study with three other probability sampling designs, GRTS consistently outperformed all but systematic sampling, which provided slightly better precision at intermediate sample sizes. Depending on the scale of inference, GRTS is useful in monitoring bull trout conservation units through redd counts, though a census may provide a more practical design for monitoring core areas as defined by the U.S. Fish and Wildlife Service.

The ability to accurately assess the population status, trends in abundance, and distribution of bull trout *Salvelinus confluentus* is central to conservation efforts for this species (USFWS 2002). Trends in population abundance are the central focus of most monitoring efforts, and the reproductive portion of a population is frequently used to estimate those trends (Al-Chokhachy et al. 2005). Redd counts are often the easiest and least costly way to monitor adult bull trout abundance. All riverine salmonids excavate nests (redds) in the stream bottom to deposit eggs. Although water velocity, redd size, and substrate preferences vary among species (DeVries 1997; Crisp and Carling 1989), in most cases newly formed redds are lighter in color than the undisturbed substrate around them. As a result, counting redds is easier and less expensive than more intrusive enumeration methods such as tagging, trapping, and underwater observation. Although there can be substantial error in redd counts (Bonneau and LaBar 1997; Dunham et al. 2001; Holecek and Walters 2007), research has shown that this metric is strongly correlated with estimates of adult escapement (Rieman and Myers 1997; Maxell 1999; Dunham et al. 2001), especially when experienced surveyors are used (Muhlfeld et al. 2006).

Common objectives of redd count surveys of salmonids are to measure the trends in the abundance of spawners, estimate the total abundance of spawning females, and determine both spatial and temporal spawning distributions (Gallagher et al. 2007). Two approaches to conducting basinwide redd counts have traditionally been taken. Census surveys require the

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enumeration of all redds throughout the entire range of spawning habitat, whereas index surveys are conducted in certain streams or reference reaches and the counts used to extrapolate estimates of population abundance. Both approaches have their advantages and disadvantages. Census surveys are often time-consuming because of the large spatial extent of the streams surveyed, some of which may not even contain redds. For this reason, multiple surveys of the basin are often unfeasible such that the temporal variability in redd distributions may not be detected. The advantage to this approach is that the true redd number (except for the observation error) is known and no extrapolation is needed as long as the census is completed after all spawning has occurred.

Index surveys have the advantage of being less time-consuming and therefore allowing multiple passes to be made. The disadvantage to this approach, besides the statistical uncertainty associated with such counts (Maxell 1999), is that the representativeness of the index reaches is open to question, especially when there is spatial and temporal variation in the distribution of redds (Rieman and McIntyre 1996; Isaak and Thurow 2006). Further, in the absence of any explicit randomization in site selection there is no theoretical basis for estimates of precision (Courbois et al. 2008). A sampling method that is statistically rigorous, minimizes field and budgetary constraints, and accounts for both the spatial and temporal variability in redd distributions is needed if redd counts are to be a reliable way to assess the abundance of adult bull trout.

The U.S. Environmental Protection Agency (EPA) developed the Environmental Monitoring and Assessment Program (EMAP) to evaluate the status of natural resources at regional and national scales (Stevens 1994). The goal of EMAP is to provide a scientific basis for monitoring programs that measure current and changing resource status. The program employs a probability survey design that allows resource assessment over large areas based on data from representative sample locations. It developed a sample design called a generalized random tessellation stratified (GRTS) design (Stevens and Olsen 2004) to achieve a spatially balanced point distribution that is nonetheless random. Briefly, the GRTS design achieves a random, nearly regular sample point pattern via a random function that maps two-dimensional space onto a one-dimensional line. Details of the construction of the random function and sample selection are provided in Stevens and Olsen (2004). Trends in status are best assessed by visiting randomly selected sampling sites on annual and multiyear cycles. The GRTS design allows the evaluation of status, trend, and distribution at multiple scales with statistical rigor (Firman and Jacobs 1999). The GRTS design takes into account the spatial patterns of resource distribution when calculating estimates of variance to provide higher precision for a given level of sampling effort (Stevens and Olsen 2003).

In the current study, we evaluate the utility of the GRTS design in determining population status and trends for bull trout through redd counts. We test a sampling effort that would be economically practical to implement on a continuous basis in multiple drainages within the southeastern Washington and Oregon portion of the Columbia River plateau. We evaluate the logistics of a pilot application of the GRTS design. We compare GRTS estimates of redd abundance with those from census surveys and determine the degree of accuracy and the level of precision associated with the protocol. Further, we investigate the performance of the GRTS design with respect to three other probability sampling designs using simulation.

Methods

Study area and data collection.—Our study was conducted in the Deschutes, John Day, Umatilla, and Walla Walla River basins in the portion of the Columbia River plateau located in central and northeastern Oregon and southeastern Washington (Figure 1). Surveys were conducted for three consecutive years, 2002–2004; however, no sampling took place in 2003 in the Deschutes watershed owing to a catastrophic wildfire that limited access. The target population among the four basins consisted of all wadeable stream reaches that contained known and potential spawning habitat for bull trout. The identification of suitable stream reaches was based on maps of the species' current distribution (derived from the EPA's 1:100,000 river reach data set) and input from local biologists and other fishery managers. Altogether, the target population constituted between 388 and 507 km during the 3 years of the study (Table 1). In 2004, owing to landowner restrictions, the sample frame length in the Deschutes basin was substantially reduced from its size in 2002 (Table 1). No sampling occurred in Shitike Creek or the Whitewater River (Figure 1).

Sample site selection was made following the methods described by Stevens and Olsen (2004). As bull trout spawning is less widely distributed in the Umatilla and Walla Walla River basins than in the other drainages, these two watersheds were combined (henceforth referred to as UWW). The site selection process produced 50 spatially balanced sites in each of the Deschutes, John Day, and UWW basins. We judged 50 sites to be the maximum number that a two-person field crew could effectively survey multiple times over the spawning period and required that a

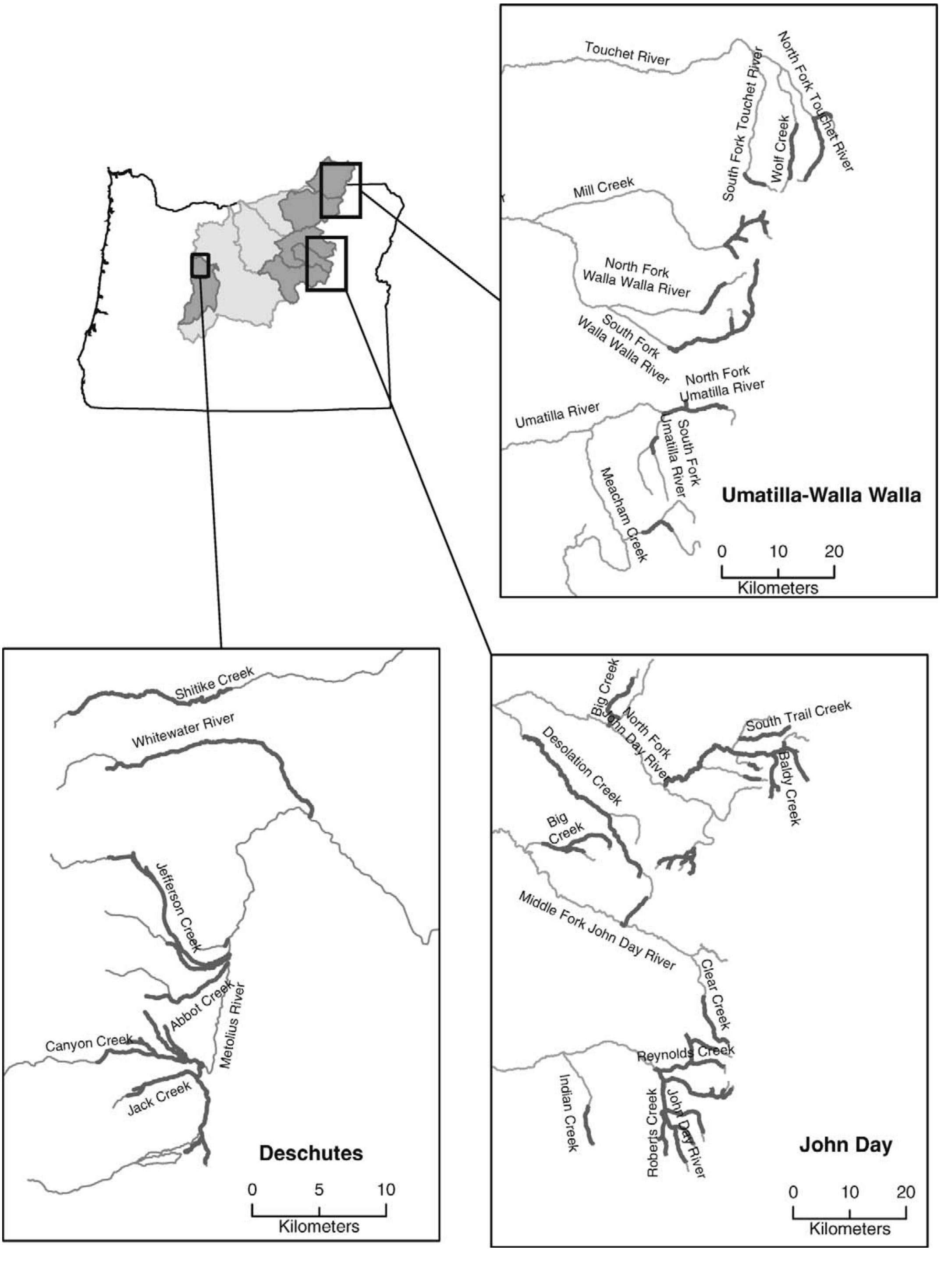

FIGURE 1.—Maps of the four sampled basins (Deschutes, John Day, Umatilla, and Walla Walla) within the Columbia River plateau.

TABLE 1.—Lengths of stream surveyed using the generalized random tessellation stratified (GRTS) design and total length (km) of target populations in four basins of the Columbia River plateau, 2002–2004.

	John Day		Umatilla–Walla Walla		Deschutes		Plateau	
Year	Surveyed	Total	Surveyed	Total	Surveyed	Total	Surveyed	Total
2002	71	274	59	117	46	115	176	507
2003	73	274	62	114	*Not sampled*		135	388
2004	73	274	68	114	56	67	197	454

minimum of 30 sites be surveyed in each basin. Fifty additional sites were selected in each basin for use as replacements in the event that some sites were unsuitable (e.g., located in a dry stream channel or on private property to which we could not get access). Each sample point served as the midpoint of a 1.6-km spawning survey section. During August, field crews located the midpoint of each survey section using Universal Transverse Mercator coordinates, maps, and a GPS receiver and measured the site to locate the endpoints. The suitability of each site was judged by the presence of potential spawning habitat and the absence of barriers to bull trout migration (unless bull trout were known to exist upstream from a barrier).

From early September through early November, sample sites were surveyed three to five times. Five two-person survey crews conducted the surveys. Single crews were assigned to the Deschutes, North Fork, and Upper–Middle Fork John Day River watersheds, and two crews sampled the UWW. The crews were trained in the identification of bull trout redds, and spawning surveys were conducted according to Oregon Department of Fish and Wildlife (ODFW) protocols (Bellerud et al. 1997). During the surveys, each newly observed redd was recorded and flagged. In streams where the presence of sympatric fall-spawning species made bull trout redd identification difficult, redds were counted only if they were observed to be occupied by bull trout.

To assess the accuracy of the GRTS design, all reaches of the target population in the UWW and Deschutes basins were also censused. The census was conducted three times throughout the spawning period in the UWW drainages and twice in the Deschutes drainage. The census surveys in the Deschutes basin were done by groups of trained biologists and untrained volunteers, whereas the UWW surveys were done by the same crews who conducted the GRTS surveys. The censused redd count within each basin was then compared with the estimated number of redds obtained using the GRTS design.

Additionally, in 2005, we conducted a census of known spawning streams in the upper main-stem and Middle Fork John Day basin and georeferenced the locations of all bull trout redds using handheld GPS units. These data were used to develop geographical information systems coverage of redd distribution. This coverage was then used to simulate the application of various sampling strategies. In addition to GRTS, we implemented simple random sampling, systematic sampling, and adaptive cluster sampling.

Our survey methods followed those used for the GRTS protocol except that survey reaches were contiguous across the entire target population and that the location of each redd was pinpointed with a handheld GPS unit. Redd coordinates were then displayed in Arc GIS software and snapped to the 1:100,000 digital line coverage representing the stream network.

Statistical techniques.—Bull trout population status was assessed based on the cumulative redd counts at the survey sites. These counts were extrapolated to the target population to provide estimates of total redd abundance and their associated precision by means of analytical algorithms developed for GRTS (Stevens and Olsen 2003, 2004). We assessed the performance of the GRTS design within each test basin and across all three collectively (hereafter referred to as the plateau). Statistical power analysis was conducted to estimate the sample sizes needed to obtain the target levels of precision and detectable effect size. Power simulations were done for each of the three basins and the plateau. Estimates of the variance of each of these simulation units were calculated as the variance pooled over each available sample year. Because the target levels of precision or detectable effect size were expressed as relative values (i.e., proportions of the point estimate), we used the coefficient of variation (100 × SD/mean) as our measure of variance.

The sample sizes needed to obtain the target levels of precision (95% confidence intervals) were calculated according to the formulae given in Cochran (1977) and Zar (1999), namely,

$$n_0 = [t_{0.05(2),\infty}]^2 (V^2/d^2),$$

where n_0 is the approximate sample size, $t_{0.05(2),\infty}$ is the two-tailed t-value with infinite degrees of freedom, V is the coefficient of variation, and d is the relative half-width of the 95% confidence interval.

TABLE 2.—Number of survey sites, estimated number of bull trout redds, and the associated level of precision (95% confidence interval expressed as a proportion of the estimated number of redds) in three basins of the Columbia River plateau, 2002–2004. the Deschutes basin was not sampled in 2003 because of a forest fire.

Sampling unit	2002			2003			2004		
	Sites	Redds	Precision	Sites	Redds	Precision	Sites	Redds	Precision
John Day	42	541	39	48	193	31	49	235	36
Umatilla–Walla Walla	40	716	23	48	684	15	50	511	19
Deschutes	34	1,704	29		*Not sampled*		50	709	12
Plateau	116	2,930	19	96	877	14	149	1,455	9

Estimates of the sample sizes needed to detect differences in population abundance were based on the formula of Snedecor and Cochran (1967), namely,

$$n_0 = 2(z_\alpha + z_\beta)^2\left(\frac{V^2}{\delta^2}\right),$$

where z_α is the standard normal deviation with α probability of committing a type I error, z_β is the standard normal deviation with β probability of committing a type II error, and δ is the detectable effect size expressed as a proportion of the initial abundance.

Because our sampling comprised an appreciable portion of the target population, we applied the finite population correction (Cochran 1977) to our estimates of samples size using the following formula:

$$n = \frac{n_0}{1 + (n_0/N)},$$

where n is the estimated sample size when sampling a finite population and N is the total number of units (sample points) in the target population.

Simulation study.—We evaluated the relative performance of GRTS and three other sampling designs for estimating redd abundance by applying each design to a data set representing the spatial distribution of redds in the John Day basin. This data set was obtained by a census conducted in 2005 in which redd locations were georeferenced (see preceding section). For the purposes of this investigation, we decomposed the 125 km of the surveyed portion of the river network into a finite population of 83 disjoint segments corresponding to potential survey reaches with an average length of 1.50 km. The segment redd counts ranged from 0 (48 segments) to 18 (1 segment), for a total of 154 redds. Each segment was assigned a unique pair of spatial coordinates by picking a single point along its length.

In addition to GRTS, we considered standard simple random sampling without replacement (SRS) and a version of systematic sampling (SYS) in which we first randomly ordered the groups of adjacent stream segments corresponding to different unique portions of the target population (individual watersheds) located in different tributaries and then drew a circular systematic sample (Cochran 1977:206) of the desired size. This procedure generally insures that the sampled segments are well distributed over the stream network for most sample sizes, producing a spatially balanced sample. In all cases, we estimated the total number of redds as

$$\hat{\tau} = N\sum_{k=1}^{n}\frac{y_k}{n},$$

where $N = 83$ is the total number of segments, y_k is the number of redds in segment k, and n is the sample size. Note that this is just the sample average redd count multiplied by the total number of segments. We also considered adaptive cluster sampling (ACS) with the modified Horvitz–Thompson estimator (Thompson 1990). For this purpose, we defined two segments to be neighbors if they were immediately adjacent to each other in the stream network either along a single tributary or at the junction of two or more tributaries. The condition triggering adaptive sampling was taken to be the presence of at least one redd within a surveyed unit.

As all four of these design (estimator) strategies are known to be theoretically unbiased, to assess their performance we used the variances of the estimators, which we estimated as the usual sample variance based on 10,000 independent samples for sample sizes from 10 to 50 for the three conventional (nonadaptive) estimators (SRS, SYS and GRTS) and an expected survey effort of (approximately) 10–50 units for the ACS. Whereas one specifies an initial sample size in adaptive cluster sampling, the final sample size is unknown in advance and varies from sample to sample, depending on the outcome of the sampling response. The three conventional designs considered here are by contrast fixed-size designs. We thus used the average number of units actually surveyed as the analog of sample size in comparing ACS with the other designs.

Results

Sampling Success

We sampled an average of 38% (range, 35–43%) of the entire plateau target population over the 3 years of the study (Table 1) and 34–50 sites per basin per year (Table 2). This comprised anywhere from 26% to 81% of the basin target populations. Only in 2004 did we meet our goal of surveying 50 sites for a single basin. The overwhelming majority of the target areas constituted potential spawning habitat; only 2% of the 368 sites we sampled were found to be dry or lacking spawning gravel.

Redd Distribution

Redds were not evenly distributed across the three basins. Despite this patchiness, the degree of fragmentation varied. In the UWW basin, consistently high redd densities occurred in three tributaries: Wolf Creek, Mill Creek, and the South Fork Walla Walla River (Figure 2). Similarly, the lower section of Jefferson Creek in the Deschutes basin had consistently high redd densities. No portions of the John Day basin showed consistently high redd densities during the three sampling seasons; however, the sampling sites on Desolation Creek and the North Fork John Day River either were consistently void of redds or had low redd densities. Locations that also consistently had little or no spawning activity included the North Fork Walla Walla River and Meacham Creek in the UWW and the upper Metolius River in the Deschutes basin.

Differences in the spatial patterns of redd distribution among the three basins are apparent in the cumulative distribution frequencies (Figure 3). Redds were most rare in the John Day basin, in which nearly 60% of the sites sampled in the three study seasons were devoid of them. Furthermore, less than 10% of the sites had redd densities exceeding 10 redds/km. This pattern contrasts sharply with the more uniform pattern observed in the Deschutes basin, in which less than 40% of the sites were devoid of redds and nearly 40% had redd densities exceeding 10 redds/km.

Estimates of Redd Abundance and Associated Precision

The GRTS-based estimates ranged from 193 to 1,704 redds per basin, with relative precisions (95% confidence interval half-widths) of 12% to 39% (Table 2). The redd abundance estimates for the John Day basin consistently had lower precision than those for the other two basins. This difference was partly due to our surveying only 26% of the relatively large target population in the John Day basin each year, compared with average sampling fractions of 55% for the UWW drainages and 62% for the Deschutes basin (Table 1). It was also due to the redds' having a more fragmented distribution in the John Day basin. Other studies have shown that precision is lower when one is sampling patchily distributed populations (Mier and Picquelle 2008; Courbois et al. 2008). The high precision of the Deschutes basin estimate in 2004 (Table 2) was largely due to the high sampling fraction and the use of the finite population correction in calculating estimates of variance. After the Warm Springs Indian Tribe denied us permission to conduct surveys on their reservation in 2004, the Deschutes target population was reduced 42% from 2002. This resulted in a sampling fraction exceeding 80%.

At the plateau scale, the abundance estimates ranged from about 900 to 2,900 redds and had precisions ranging from 9% to 19%. In each of the three study years, the precision at the plateau scale exceeded that at any of the constituent basins.

Power Analysis

The results of our power analysis show that less than half as many sample sites are needed in the Deschutes and UWW basins as in the John Day basin to achieve the same level of precision (Figure 4). For example, to achieve a precision of 20% in the UWW approximately 40 sites need to be surveyed. The same level of precision in the John Day basin requires that nearly 100 sites be surveyed. Further, surveying an additional 50 sites in the John Day basin only improves precision by approximately 13%. At the plateau scale, surveying 50 sites in each basin (for a total of 150 sites) yields a precision of 16%. If only 100 sites are surveyed, the precision is still well within 20%.

Our sample size targets of 50 sites at the basin scale and 150 sites at the plateau scale provide lower sensitivity when viewed in terms of the minimum detectable differences between two abundance estimates. Using the recommendations of Gryska et al. (1997) for rare species and those listed as threatened or endangered, we set the level of a type I error at 20% and that of a type II error at 10%. At these levels, a sample size of 50 would allow us to detect a minimum of about a 30% difference in redd numbers in either the Deschutes or UWW basins (Figure 5). Conversely, this sample size would only allow detection of a 70% abundance change in the John Day basin. With a sample size of 150 at the plateau scale we can detect about a 30% change in redd numbers, and with a sample size of 215 the minimum detectable difference decreases to a change of 20%. Reducing the probability of committing a type II error (missing a change in redd numbers) would increase the sensitivity of detecting differences.

Census versus GRTS

Comparison of the GRTS estimates and the census counts in the UWW and Deschutes basins showed variable results. In the UWW, the relationship was not

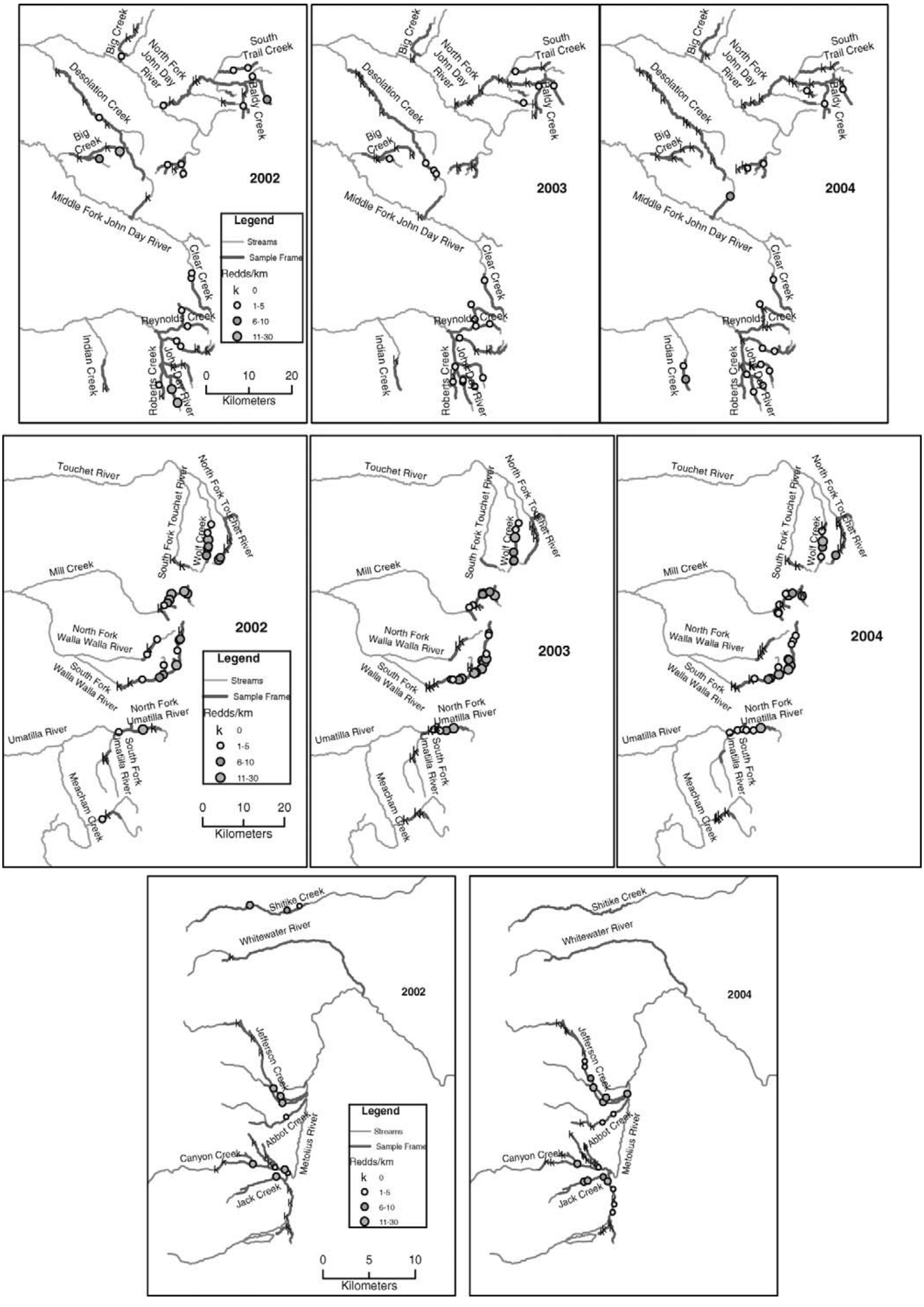

FIGURE 2.—Densities of bull trout redds at randomly selected survey sites in the John Day basin (top panel), Umatilla–Walla Walla basin (middle panel), and Deschutes basin (bottom panel) in 2002, 2003, and 2004. No sampling was conducted in the Deschutes basin in 2003 and none in Shitike Creek and the Whitewater River.

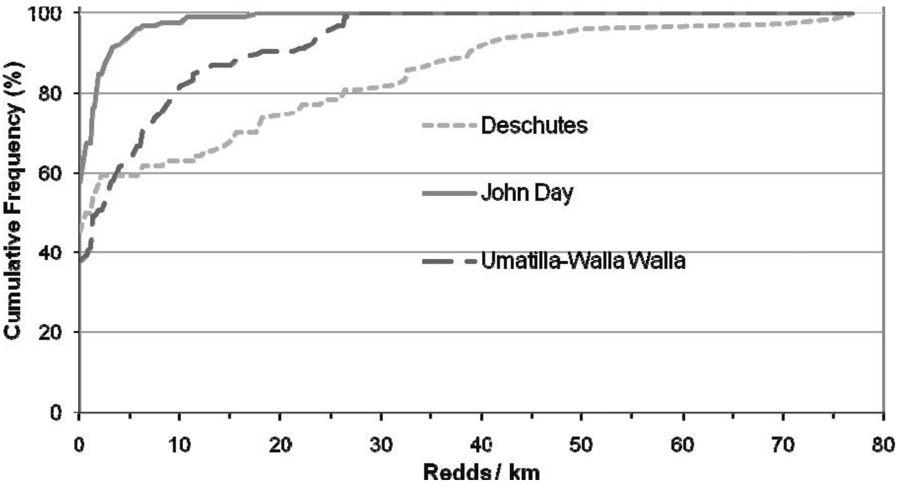

FIGURE 3.—Cumulative frequency distributions of redd density at generalized random tessellation stratified (GRTS) sample sites in the three study basins during 2002–2004. Density estimates were pooled for the three study years except for the Deschutes basin, for which data are only available for 2002 and 2004. Sample sizes are as follows: 84 (Deschutes), 139 (John Day), and 138 (Umatilla–Walla Walla).

consistent over the three study years. In 2002 and 2003, the census count and GRTS estimate differed by less than 2%; in 2004, however, the number of redds estimated through GRTS was significantly lower than the census redd count (Figure 6). We also compared our GRTS estimate in the Deschutes basin to the result of a census conducted by local ODFW district managers that included surveyors from the GRTS crew, U.S. Forest Service, and volunteers. There were some differences between the census surveys and the GRTS estimates: the target population for the census was 5% smaller than the GRTS target population, the census was only done twice during the season, and in some cases, different survey crews were used for each survey visit. Comparing the GRTS estimate with the district census in the Deschutes basin, we found a

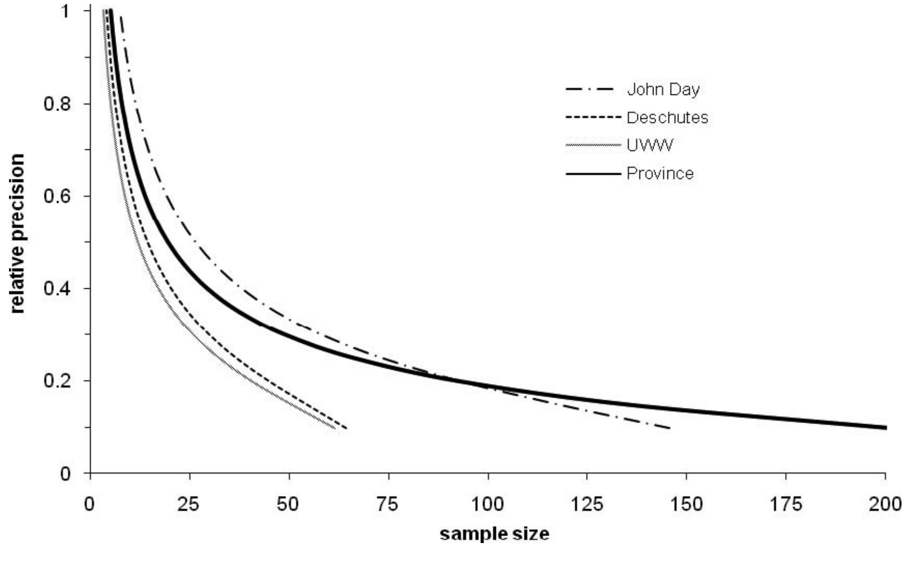

FIGURE 4.—Results of a power analysis showing the relative levels of precision (95% confidence intervals) of different GRTS sample sizes for three target populations in the Columbia plateau province with 3 years of pooled data. The pooled coefficients of variations are as follows: 1.42 (John Day), 1.08 (Deschutes), 0.96 (Umatilla–Walla Walla [UWW]), and 1.16 (plateau).

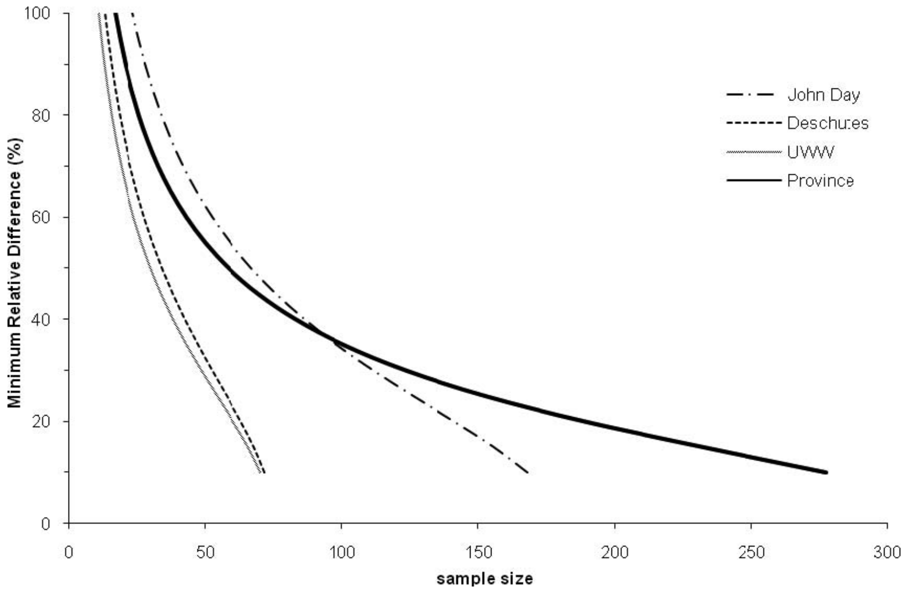

FIGURE 5.—Relationships between sample size and the minimum relative detectable difference between pairs of estimates of bull trout redd abundance for three target populations in the Columbia plateau province with 3 years of pooled data. The curves assume a probability of a type I error of 0.20 and a probability of a type II error of 0.10.

significant difference and greater apparent inaccuracy than in the UWW basin in both 2002 and 2004 (Figure 6). The 2002 GRTS estimate was 41% higher than the district census and the 2004 GRTS estimate was 32% lower.

The conflicting GRTS and census results raised concerns about the accuracy of the GRTS design for estimating bull trout redd abundance. To further address this question, we simulated the performance of the GRTS design using a known population of bull

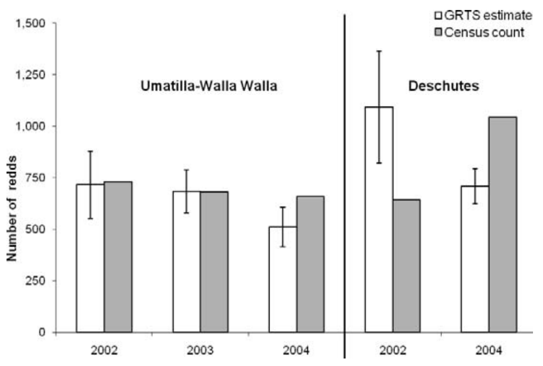

FIGURE 6.—Comparisons of GRTS estimates and census counts for bull trout redds in two basins of the Columbia plateau province, 2002–2004. The error bars show the 95% confidence intervals. The comparisons in the Deschutes basin are restricted to the Metolius watershed.

trout redds in the John Day basin. In our 2005 census we counted a total of 154 redds over 125 km of stream. These redds were notably unevenly distributed within the target population (Figure 7). In most areas there were no redds, and areas of high redd density were largely restricted to a limited portion of the stream network. With the exception of Clear Creek, only two redds were observed in Middle Fork John Day tributaries. Further, most of the observed redds in Clear Creek occurred within approximately 1 km of each other. In the main-stem John Day tributaries, the highest densities of redds were found in upper Deardorff and Rail creeks and the upper John Day River.

Simulation Study

As noted above, all four design (estimator strategies) considered here are theoretically unbiased, which our simulation results confirm. In all cases, the average value of the estimator over the 10,000 samples was very close to the true redd count of 154 (Figure 8). Among the conventional estimators, as measured in terms of estimator variance, both GRTS and SYS outperformed SRS, markedly so for sample sizes between about 15 and 30 (Figure 9). Over this same range, SYS did better than GRTS for this particular population, though GRTS was slightly more precise at both smaller and larger sample sizes. Adaptive cluster sampling with the modified Horvitz–Thompson esti-

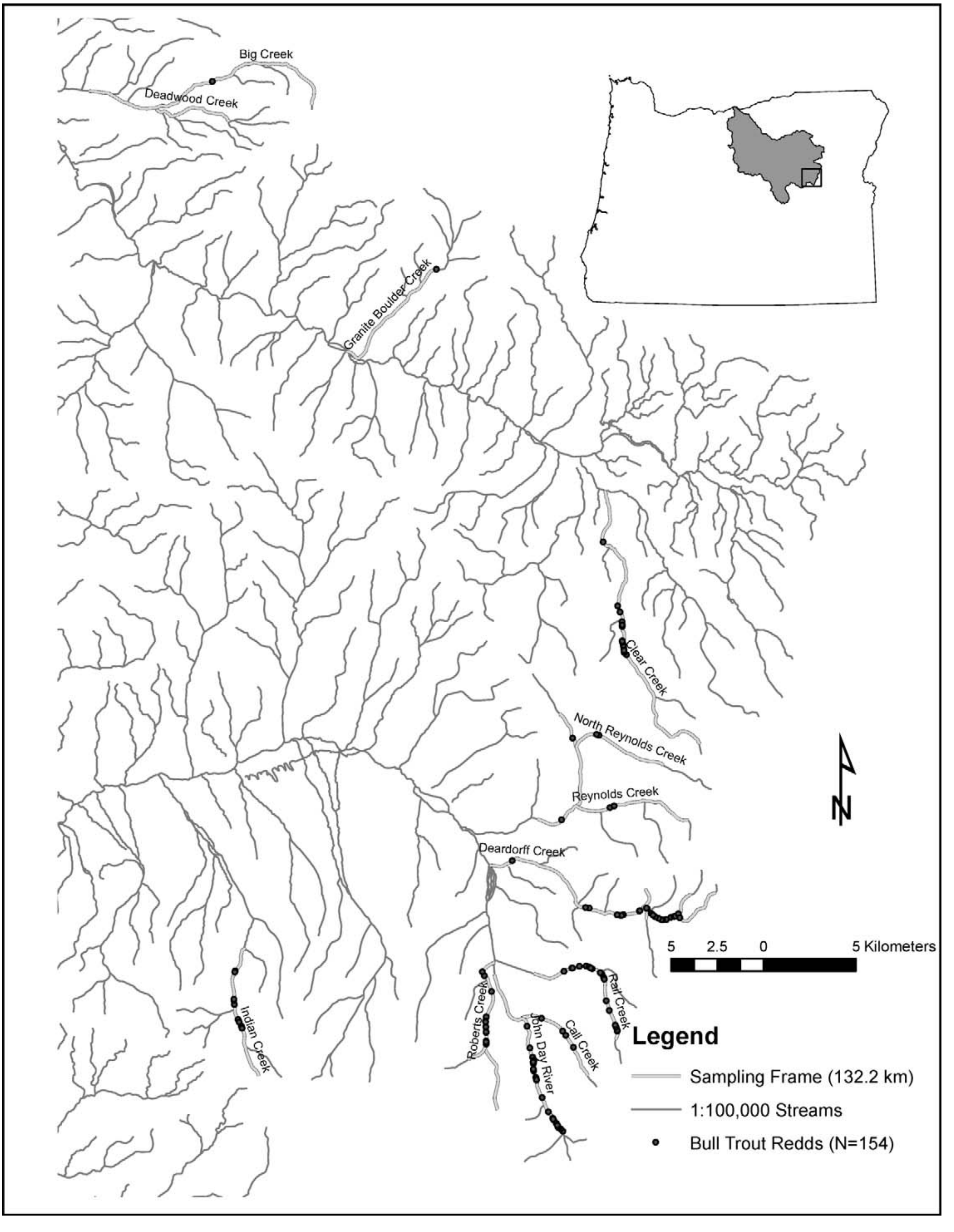

FIGURE 7.—Locations of bull trout redds observed during the 2005 census in the upper John Day watershed.

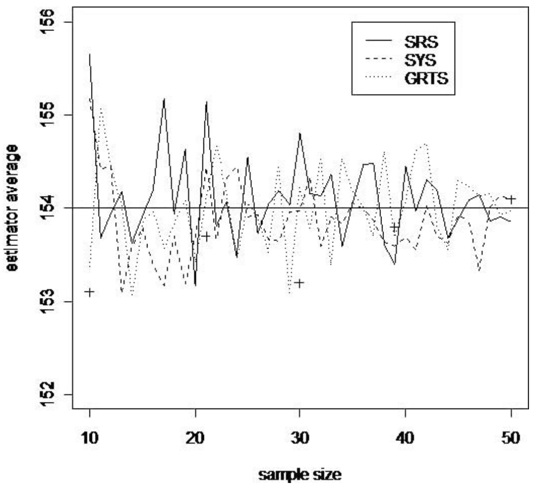

FIGURE 8.—Average estimated abundance over 10,000 independent samples as a function of sample size for three conventional sampling designs (GRTS, simple random sampling with replacement [SRS], and systematic sampling [SYS]) and adaptive cluster sampling (ACS). The plus signs represent ACS averages for expected final sample sizes of approximately 10, 21, 30, 39, and 50. All estimators are theoretically unbiased with respect to the true abundance of 154.

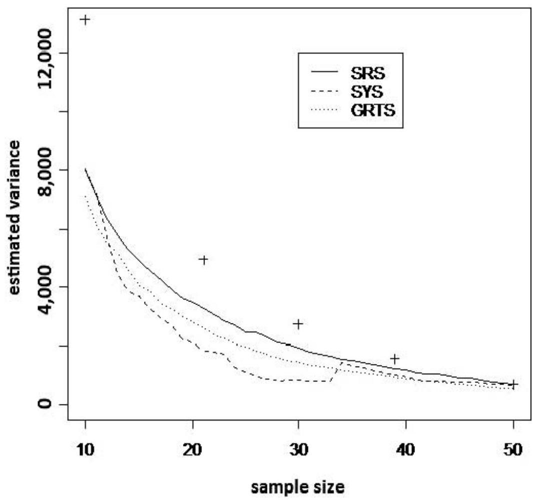

FIGURE 9.—Estimated variances in abundance based on 5,000 independent samples as a function of sample size for the four sampling designs noted in Figure 8. The plus signs represent ACS averages for expected final sample sizes of approximately 10, 21, 30, 39, and 50.

mator, on the other hand, generally performed very poorly even compared with SRS, though the differences decreased with increasing sample size. For a sample size of 50, adaptive cluster sampling compared well with both SRS and SYS, though GRTS provided the most precise estimate for this large sample size. Table 3 gives the estimated variances of SYS, GRTS, and ACS relative to SRS, termed the "design effect," for conventional sample sizes of 10, 21, 30, 39, and 50. The performance of SYS for a sample size of 30, with just 43% of the variance of SRS (compared with 75% for GRTS and 1.41% for ACS), is especially notable. Figure 10 shows the corresponding histograms. Whereas these distributions clearly reflect the relative precision of the estimators as documented in Table 3, all appear reasonably well behaved and symmetric about the target value (154) for this sample size and this population.

Discussion

Despite the disparity between some of the GRTS estimates and those obtained from censuses, our results suggest that the GRTS design is a viable tool for monitoring the status of bull trout. The GRTS design was (1) generally practical to implement logistically, (2) provided unbiased estimates of true redd numbers despite the patchy spatial distribution of redds common for bull trout populations, and (3) depending on the scale of inference, can provide levels of precision with relatively high sensitivity for detecting changes in population status and elucidating trends.

The simulation results showed that the GRTS survey design provided accurate estimates of redd abundance when applied to the actual georeferenced population of redds in the John Day basin. The estimator associated with the GRTS design was unbiased despite the highly fragmented spatial distribution of the redds. Further, at sample sizes ranging from 10 to 50 survey sites, there did not appear to be any reduction in accuracy associated with smaller sample sizes. These findings differ somewhat from the direct comparisons of GRTS estimates and censuses in the UWW and Deschutes basins, in which the census counts deviated substan-

TABLE 3.—Comparison of the estimated variances of three estimators relative to that of simple random sampling (SRS) for selected sample sizes. Abbreviations are as follows: SYS = systematic sampling, GRTS = generalized random tessellation stratified sampling, and ACS = adaptive cluster sampling. See text for details.

Sample size (initial ACS sample size)	Estimated variance relative to SRS sampling		
	SYS	GRTS	ACS
10 (4)	1.01	0.89	1.64
21 (9)	0.56	0.79	1.51
30 (14)	0.43	0.75	1.41
39 (20)	0.82	0.75	1.26
50 (29)	0.96	0.77	1.00

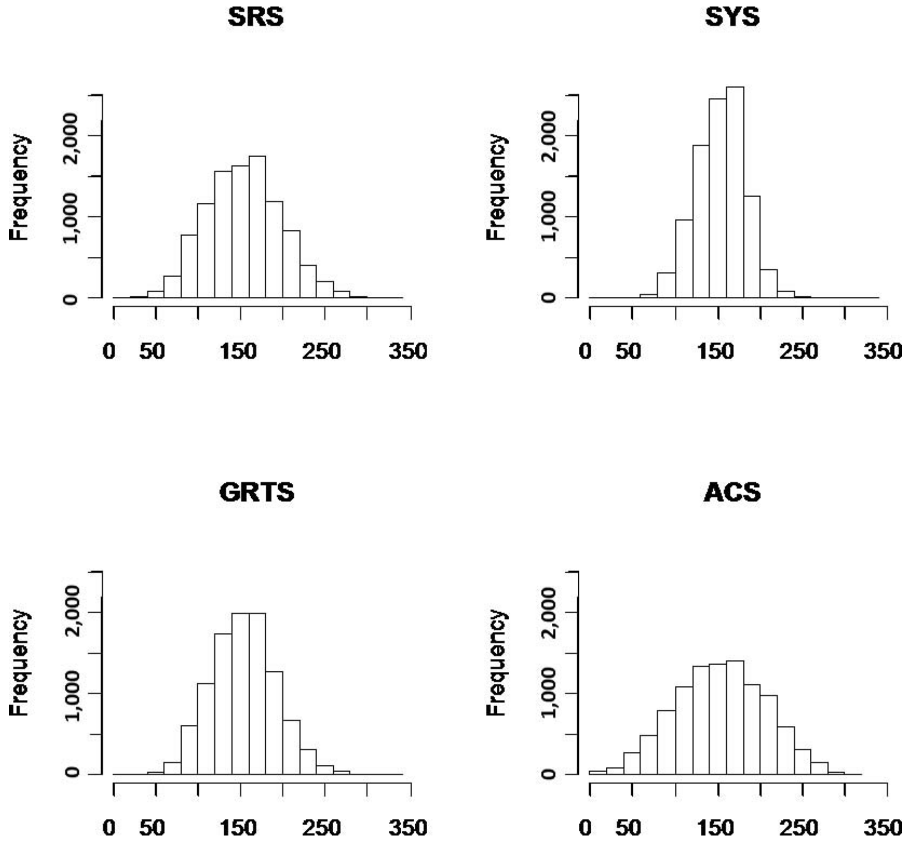

FIGURE 10.—Results of simulation modeling with different sampling designs and the 2005 census of bull trout redds in the upper John Day watershed. Shown are the empirical sampling distributions for the four estimators noted in Figure 8 based on 10,000 independent samples. Each sample simulation entailed drawing 30 sample units; on average, each sample unit comprised 1.5 km. The expected total adaptive cluster sample size is approximately 30. The actual abundance is 154.

tially from the GRTS estimates in three out of five cases. In the Deschutes basin, interobserver variability in redd recognition probably contributed to the lack of correspondence. In some cases, the survey crews that conducted the census were different from those that surveyed the GRTS sites. Prior studies have shown that there can be substantial variation in bull trout redd counts among different surveyors (Dunham et al. 2001; Muhlfeld et al. 2006). In the UWW basin, the same crews conducted the census and the GRTS surveys. In two of three cases, the GRTS estimates and census counts were remarkably consistent. In the case in which the two values differed markedly, we attribute the disparity to the statistical uncertainty associated with the GRTS estimates, under the assumption that the census counts provide a reliable assessment of abundance. Theoretically accurate sampling designs may not provide consistently accurate results in cases in which the target population has a spatially patchy distribution and a large portion of the sample points have a value of zero (Irvine et al. 1992; Courbois et al. 2008; Mier and Picquelle 2008).

Compared with the other three sampling designs evaluated, GRTS consistently outperformed all but SYS. Although none of the designs showed a consistently large bias, ACS had a slight but consistent negative bias in the simulations at all but the largest sample size. This, coupled with the substantially lower precision and logistical complexity of the ACS design, make it the least appropriate for this application. The poor performance of ACS may have resulted from the factor identified by Courbois et al. (2008), namely, that a high proportion of the initial sample did not containing redds, which would result in the failure to cluster sample sites and thus the lower precision of the ACS compared with other estimators at the smaller sample sizes (Figure 9). Poor statistical performance, the inability to control sample size, and difficulty in accommodating unpredictable sample locations make

ACS a poor sampling strategy for bull trout redd counts.

The GRTS design consistently provided higher precision than the SRS design when applied to the patchily distributed population of John Day bull trout redds at sample sizes ranging from 10 to 50 surveys. On average, the precision of the GRTS design was about 25% higher than that of the SRS design. Given that SRS and GRTS are virtually comparable in their application, the greater precision of GRTS makes it the obvious best choice. The SYS design provided higher precision than the GRTS design at intermediate sample sizes. The improvements in precision over GRTS can be as high as 25% for a sample size of 30. Thus, the SYS design deserves further evaluation. In a similar study involving redd counts for Chinook salmon, Courbois et al. (2008) also found that GRTS and SYS commonly provided greater accuracy and precision than SRS. The logistical factors that need to be evaluated before implementing an SYS design include the feasibility of predetermining all potential survey reaches within the target population and the effect of the inability to sample all selected sites. The GRTS design provides for replacing inaccessible or impractical sample sites while maintaining the design properties of the sample. By design, SYS requires sampling all selected sites.

Our study evaluated sampling efforts that were judged to be practicable to implement over the long term. A target of 50 sites per basin can be sampled by a two-person survey crew multiple times through the spawning season. At this level of effort, in two of the three basins we would achieve redd abundance estimates with a precision as high as 15%. This level of precision is suitable for detecting changes in abundance as low as 30%. In the John Day basin, however, the higher spatial variation in redd distribution and larger target population result in substantially lower sensitivity for this sampling effort. The same sample size only provides a precision of 35% and a minimum delectability of 70%. At the plateau scale, the effect of the higher variability in the John Day basin is dampened by the larger overall sample size. In this case, our target of 150 sites yields a precision of 15% and a minimal detectible difference of 30%. This study evaluated the performance of the GRTS sampling design in terms of the spatial variability of redd occurrence. When one is considering the changes in abundance over multiple spawning cohorts, the natural interannual variation in redd abundance must also be considered. Interannual variation can limit the sensitivity for detecting trends in bull trout (Maxell 1999) and thus the magnitude of the changes that can be detected from a baseline time series of various abundance indices of riverine salmonids (Ham and Pearsons (2000).

Application to Monitoring Progress toward Recovery

The USFWS draft bull trout recovery plan (USFWS 2002) established quantitative recovery criteria for bull trout. These criteria include elements addressing the species' distribution, abundance, habitat conditions, and genetic factors. This plan identifies discrete recovery units and defines specific goals for each of the four elements at each unit. Because the three basins addressed in this study also comprise individual recovery units, we are able to evaluate the application of the GRTS design to measuring progress toward achieving recovery goals.

Application of the GRTS design directly addresses the measurement of changes in distribution and abundance. As an example, the UWW recovery unit has the following recovery criteria: (1) bull trout are distributed among six or more local populations, (2) estimated abundance is in the range 3,500–10,000, and (3) adult bull trout exhibit a stable or increasing trend in abundance for at least two generations. With respect to the first criterion, the GRTS design provides an unbiased estimate of the distribution of redds throughout the recovery area. This distribution, coupled with geographic specification of the locations of local populations, could be used assess the criterion. In the UWW, redds were consistently observed in tributaries occupied by only four of the six local populations, indicating that the distribution criterion has not been attained. However, given that redd counts might not detect extremely low population abundances, it would be prudent to verify the absence of populations by additional sampling using more sensitive methods, such as electrofishing or night snorkeling. In addition to addressing the criteria contained in the recovery plan, the GRTS design allows for the development and monitoring of more subtle distribution metrics. Changes in the shape of the cumulative distribution frequency (as in Figure 3) could be used to track changes in distribution patterns within a recovery unit.

Redd counts directly reflect adult abundance; however, applying redd counts to adult abundance targets requires an estimate of the number of redds per spawning adult. Dunham et al. (2001) report results from a number of studies showing that the number of adults per redd averages 2.16 and ranges from 1.03 to 3.33. Such a range introduces additional uncertainty in using redd counts to estimate adult abundance. Without direct knowledge of the relationship between the number of redds and adult abundance for a given recovery unit, it would be prudent to include this uncertainty in the estimates of abundance derived from redd counts. Applying the published ratios to the average estimate of the number of redds (about 640 over the three study years) yields an estimated abundance of about 1,400 adult bull trout, with a low of 660 and a high of about 2,100. With our target of 50 sample sites, the precision would conservatively be within 18%. This level of precision provides strong certainty that the abundance of bull trout in the UWW recovery unit is no more than 60% of the lower end of the range of its abundance target.

With respect to status assessment, depending on the redd distribution in a given recovery unit, a target of 50 sample sites provides a 90% probability of detecting between a 30% and 70% difference in abundance. Given the low power of assessing trends with time series of redd counts (Maxell 1999), a criterion that compares annual abundance estimates with an established a priori abundance level might prove more sensitive. In the case of the UWW recovery unit, we could detect when the population abundance changes by 30%.

Recent assessment efforts for bull trout focus on core areas as the fundamental level of assessment (USFWS 2005). Core areas are generally viewed to function as metapopulations (Dunham and Rieman 1999) and usually are aggregated to constitute a recovery unit. For example, the UWW recovery unit is composed of two core areas. At the core area level of inference, using a probability sampling design such as GRTS to monitor bull trout redds may not be beneficial. Relatively high intersite variability, coupled with the small size of the target population, limits the utility of a survey design. Given the patchiness of bull trout redd distribution, it is likely that the sample sizes needed to achieve the necessary levels of precision for individual core areas will not be substantially lower than those needed for the associated recovery units. Further, the isolated and remote locations of bull trout spawning streams create logistical hurdles in accessing sampling sites. In many cases, surveyors need to walk through much of the stream network to access spatially balanced sample reaches. Because of their low sampling intensity, redd surveys do not require appreciable time beyond what is required to hike the stream channel. The time spent traveling to discrete sample reaches could be used to survey additional portions of the target population. Given these factors, a census may prove to be more practical design for monitoring bull trout redds at the core area scale of resolution.

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References

Al-Chokhachy, R., P. Budy, and H. Schaller. 2005. Understanding the significance of redd counts: a comparison between two methods for estimating the abundance and monitoring of bull trout populations. North American Journal of Fisheries Management 25:1505–1512.

Bellerud, B. L., S. Gunckel, A. R. Hemmingsen, D. V. Buchanan, and P. J. Howell. 1997. Bull trout life history, genetics, habitat needs, and limiting factors in central and northeast Oregon, 1996 annual report. Bonneville Power Administration, Project 199405400, Portland, Oregon.

Bonneau, J. L., and G. LaBar. 1997. Interobserver and temporal bull trout redd count variability in tributaries of Lake Pend Oreille, Idaho: completion report. Department of Fisheries and Wildlife, University of Idaho, Moscow.

Cochran, W. G. 1977. Sampling techniques, 3rd edition. Wiley, New York.

Courbois, J.-Y., S. L. Katz, D. J. Isaak, E. A. Steel, R. F. Thurow, A. M. Wargo Rub, T. Olsen, and C. E. Jordon. 2008. Evaluating probability sampling strategies for estimating redd counts: an example with Chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 65:1814–1830.

Crisp, D. T., and P. A. Carling. 1989. Observations on citing, dimensions, and structure of salmonid redds. Journal of Fish Biology 34:119–134.

DeVries, P. 1997. Riverine salmonid egg burial depths: review of published data and implications for scour studies. Canadian Journal of Fisheries and Aquatic Sciences 54:1685–1698.

Dunham, J. B., and B. E. Rieman. 1999. Metapopulation structure of bull trout: influences of physical, biotic, and geometrical landscape characteristics. Ecological Applications 9:642–655.

Dunham, J., B. Rieman, and K. Davis. 2001. Sources and magnitude of sampling error in redd counts for bull trout.

North American Journal of Fisheries Management 21:343–352.

Firman, J. C., and S. T. Jacobs. 1999. A survey design for integrated monitoring of salmonids. Oregon Department of Fish and Wildlife, Corvallis.

Gallagher, S. P., P. K. J. Hahn, and D. H. Johnson. 2007. Redd counts. Pages 197–234 *in* D. H. Johnson, B. M. Shrier, J. S. O'Neal, J. A. Knutzen, X. Augerot, T. A. O'Neil, and T. N. Pearsons. Salmonid field protocols handbook: techniques for assessing status and trends in salmon and trout populations. American Fisheries Society, Bethesda, Maryland.

Gryska, A. D., W. A. Hubert, and K. G. Gerow. 1997. Use of power analysis in developing monitoring protocols for the endangered Kendal Warm Springs dace. North American Journal of Fisheries Management 17:1005–1009.

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

Holecek, D. E., and J. P. Walters. 2007. Spawning characteristics of adfluvial rainbow trout in a north Idaho stream: implications for error in redd counts. North American Journal of Fisheries Management 27:1010–1017.

Irvine, J. R., R. C. Bocking, K. K. English, and M. Labelle. 1992. Estimating coho salmon (*Oncorhynchus kisutch*) spawning escapement by conducting visual surveys in areas selected using stratified random and stratified index sampling designs. Canadian Journal of Fisheries and Aquatic Sciences 49:1972–1981.

Isaak, D. J., and R. F. Thurow. 2006. Network-scale spatial and temporal variation in Chinook salmon (*Oncorhynchus tshawytscha*) redd distributions: patterns inferred from spatially continuous replicate surveys. Canadian Journal of Fisheries and Aquatic Sciences 63:285–296.

Maxell, B. A. 1999. A power analysis on the monitoring of bull trout stocks using redd counts. North American Journal of Fisheries Management 19:860–866.

Mier, K. L., and S. J. Picquelle. 2008. Estimating abundance of spatially aggregated populations: comparing adaptive sampling with other survey designs. Canadian Journal of Fisheries and Aquatic Sciences 65:176–197.

Muhlfeld, C. C., M. L. Taper, D. F. Staples, and B. B. Shepard. 2006. Observer error structure in bull trout redd counts in Montana streams: implications for inference on true redd numbers. Transactions of the American Fisheries Society 135:643–654.

Rieman, B. E., and J. D. McIntyre. 1996. Spatial and temporal variability in bull trout redd counts. North American Journal of Fisheries Management 16:132–141.

Rieman, B. E., and D. L. Myers. 1997. Use of redd counts to detect trends in bull trout (*Salvelinus confluentus*) populations. Conservation Biology 11:1015–1018.

Snedecor, G. W., and W. G. Cochran. 1967. Statistical methods. Iowa State College Press, Ames.

Stevens, D. L., Jr. 1994. Implementation of a national environmental monitoring program. Journal of Environmental Management 42:1–29.

Stevens, D. L., Jr., and A. R. Olsen. 2003. Variance estimation for spatially balanced samples of environmental resources. Environmetrics 14:593–610.

Stevens, D. L., Jr., and A. R. Olsen. 2004. Spatially balanced sampling of natural resources. Journal of the American Statistical Association 99(465):262–278.

Thompson, S. K. 1990. Adaptive cluster sampling. Journal of the American Statistical Association 85:1050–1059.

USFWS (U.S. Fish and Wildlife Service). 2002. Bull trout (*Salvelinus confluentus*) draft recovery plan. USFWS, Portland, Oregon.

USFWS (U.S. Fish and Wildlife Service). 2005. Bull trout core area conservation status assessment. USFWS, Portland, Oregon.

Zar, J. H. 1999. Biostatistical analysis. Prentice-Hall, Upper Saddle River, New Jersey.