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Evaluation of a Sampling Approach to Monitor the Status of Great Basin Redband Trout in Southeastern Oregon (2007 – 2009)

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ABSTRACT

The summer 2009 field season marked the completion of the third of a six year sampling effort to assess the distribution and abundance of redband trout (*Oncorhynchus mykiss newberrii*) in the six interior basins of Oregon's high desert: Catlow Valley, Chewaucan, Fort Rock, Goose Lake, Malheur Lakes, and Warner Valley Species Management Units (SMUs). Across all sampling years, sites were randomly selected using Generalized Random Tessellation Stratified (GRTS) design, which provides a random spatially balanced sample allowing for statistically rigorous evaluation of status, trend and distribution at multiple spatial scales. A total of 700 site surveys were conducted over the course of the study covering nearly 2% of the entire 2,420 km sampling frame. Abundance of age-1+ redband trout at the SMU level has remained relatively stable throughout the course of this study but has decreased since intensively sampled in 1999. Estimates of landscape-wide abundance of age-1+ redband trout were of similar magnitude and had comparable precision across all three study years, averaging 878,000 ± 16%. However, abundance at the SMU and population (stratum) level showed substantial variation, both spatially and inter-annually. Fish densities (fish/m) sampled at sites visited annually (2007-2009) showed significant differences between years, specifically between 2007 and 2009 in the Chewaucan and Fort Rock SMUs. Target levels of relative precision were achieved twice at the SMU level, and in approximately half of the strata. Increasing the number of sites sampled to increase precision is not likely, given limited funding. Yet, the current study design falls short of providing precise information to develop conservation management plans. Alternative sampling designs that would maximize data acquisition at the population level, while allowing for estimates of yearly variation were explored and suggested.

INTRODUCTION

Redband trout (*Oncorhynchus mykiss newberrii*) inhabit arid environments ranging from montane forests to desert shrub and grasslands (Benhke 1992) in streams where extreme fluctuation in flow and temperature are common. Hydrological cycles of flood and drought, paired with increased anthropogenic disturbance of already naturally fragmented habitat, have prompted concern over the status of desert trout populations (Zoellick and Cade 2006, Currens et al. 2009, Dambacher et al. 2009). Great Basin redband trout populations persist in fragmented habitats and are isolated from core riverine groups in the large river systems of the Columbia, Sacramento, and Klamath rivers (Currens et al. 2009, Dambacher et al. 2009). Redband trout populations in all of these pluvial lake basins have evolved adfluvial life histories, such that many populations may have further adapted to these unique environments, allowing populations to persist during extreme climatic fluctuations. However, flow diversions, migration barriers, degraded riparian habitat, competition with exotic salmonids, and changing climate regimes are likely affecting abundance in trout populations (Williams et al. 2007).

In response to a petition to list southeastern Oregon Great Basin redband trout as threatened or endangered under the Endangered Species Act, a landscape-level abundance survey was conducted in 1999 to assess population status (Dambacher et al. 2009). Although the overall abundance estimate of age-1+ redband trout in1999 was considered robust and viable (Dambacher et al. 2009), little information was provided on fine-scale distribution and abundance of individual populations of redband trout within each interior basin. In addition, without continued monitoring, no information was available to assess if these levels have persisted over time. In 2005, the Oregon Department of Fish and Wildlife reviewed the 1999 study and identified numerous data gaps that prevented a thorough review of status and trend (ODFW 2005). This review also defined conservation units for Great Basin redband trout consisting of local populations organized into Species Management Units (SMUs) (Figure 1). Prompted by the need for a consistent and continuous dataset, the Native Fish Investigations Project initiated a six-year pilot study to fill in this information, while testing the feasibility of using the Generalized Random Tessellation Stratified (GRTS) sampling design for long-term monitoring. The summer 2009 field season marked the completion of the third year of this six year sampling effort.

Using a GRTS design (Steven and Olsen, 2004), redband trout abundance in all six interior basins (SMUs) was assessed annually, starting the summer of 2007. In addition, all SMUs were assessed at the population (stratum) level once during the course of this study (Figure 1). This report evaluates the first three years of this study. Included are reports of sampling success, estimates of distribution and abundance, an evaluation of the precision of abundance relative to study goals and recommendations for future monitoring.

METHODS

Study Area and Design

Our study was conducted in the major pluvial lake basins (SMUs): Fort Rock, Malheur Lakes, Chewaucan, Goose Lake, Warner Lakes, and Catlow Valley comprising the southeastern Oregon portion of the Great Basin (Figure 1). Surveys were conducted for three consecutive years, 2007–2009. The sampling frames in each SMU consisted of all wadeable stream reaches that contained documented and/or potential summer distribution for redband trout. The identification of suitable stream reaches was based on documented distribution and the professional judgment of local biologists and other fishery managers. The resulting sample frame, mapped at a 1:24,000 scale, was the pool of possible locations from which sample sites were selected and represented our scope of inference.

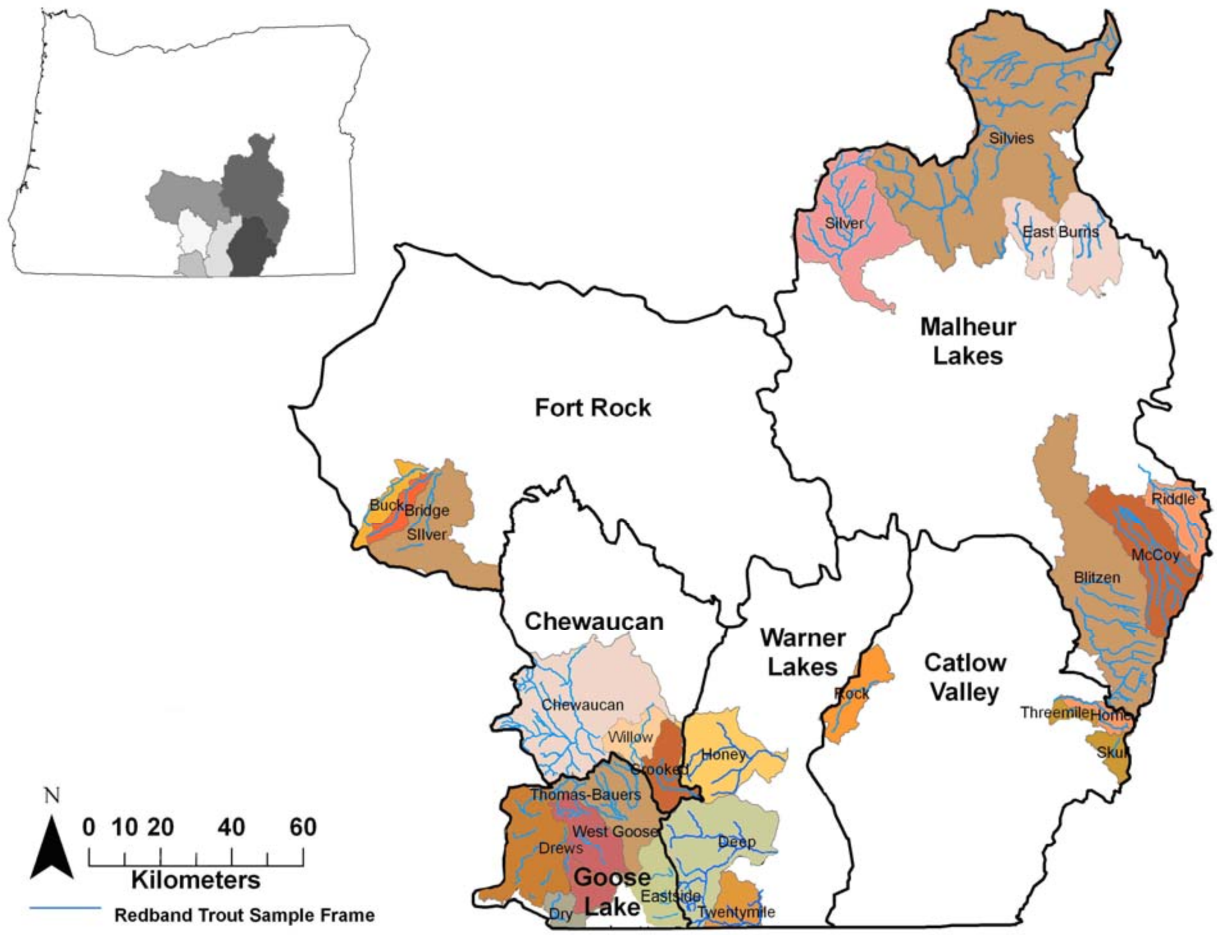


Figure1. Great Basin redband trout sample frame detailing species management units (SMUs) and constituent populations (strata) included in this study.

Sites were randomly selected using a Generalized Random Tessellation Stratified (GRTS) design developed by the U. S. Environmental Protection Agency (Stevens and Olsen 2003, 2004). The GRTS sampling design provides a random spatially balanced sample, allowing for statistically rigorous evaluation of status, trend and distribution at multiple scales. GRTS design provides higher precision for a given level of sampling effort by accounting for spatial patterns of resource distribution when calculating estimates of variance. In this study, sample allocation followed a rotating panel design where between 30 and 40 sites were targeted per SMU each year; of which, roughly half were defined as annual and targeted to be repeated yearly. The remaining sites were sampled only once, with replacement, each year. Each SMU contained several populations (strata) as described in the 2005 status review (ODFW 2005). Briefly, strata were typically defined as populations that were geographically isolated from each other with no or limited connectivity with other populations. At the stratum level, some small geographically proximate but disjunct populations were aggregated into one stratum, despite limited connectivity (**APPENDIX A**). This was done to allow for a logistically manageable sampling effort. These strata were used to provide better spatial coverage within a

SMU and allowed us to allocate samples proportional to the frame size of each stratum (**APPENDIX A**). Each stratum within every SMU was sampled once in three years, where 20-40 sites were targeted during stratum level sampling depending on sample frame size. To provide the sample sizes needed for stratum level (population level) inference on a three year interval, sampling intensity varied among SMUs each year. In total, this design sampled four to five SMUs at the SMU scale and one to two SMUs at the stratum (population) scale each year.

The site selection process produced 30-180 spatially balanced sites within the redband trout sampling frame in each SMU, each year. A minimum of 30 sites per year was targeted for evaluation, with additional sites selected as replacements in the event some sites were unsuitable (e.g., located in a dry stream channel or where access was denied to private property). The GRTS design selection process assigns sample sites in a numerical order that maintains the random and spatial structure of the sample. Sites were selected consistent with this order, such that replacement sites had the same design properties as the initial sample.

Survey Methods

Each sample point, defined by geographic coordinates, served as the downstream boundary of a survey reach. Field crews located each sample point using Universal Transverse Mercator (UTM) coordinates, maps, and a personal digital assistant (PDA), equipped with a Global Positioning System (GPS) receiver. Channel width was measured and a stream survey reach was calculated as 30 wetted channel widths with a minimum of 30 m and a maximum of 100 m. Survey reaches included a variety of stream habitat types and typically contained a minimum of three pool-riffle sequences. If channels were intermittent, 30 active channel widths were sampled; total and wetted lengths were measured. All side channels within a survey reach were included in the sample. Block nets were installed at the upper and lower site boundaries to prevent fish movement into or out of the site during a survey.

Surveys were conducted from mid-June through mid-September by four two-person survey crews. Crews were trained in the identification of redband trout and other fishes of southeastern Oregon. Multiple pass depletion backpack electrofishing was used to obtain abundance estimates at each site. A minimum of two passes were made, beginning at the downstream block net and employing two systematic upstream passes at full effort and two ½ - ¼ effort downstream sweeps to herd fish toward the lower block net. A single upstream and downstream sweep constituted one pass. Our field criterion targeted redband trout >60 mm, however an effort was made to collect smaller trout and all other species. If the number of redband trout in the second pass was greater than 50% of that in the first pass, two additional passes were completed. For sites where four passes were required, fish were summed for passes 1 and 2, and again for passes 3 and 4. A site failed (was not included in our estimates) if the total number of redband trout captured during passes 3 and 4 exceeded 50% of the total from passes 1 and 2. Sampling was limited to streams with water temperatures <21° C, measured prior to initiation of sampling.

Data Collection

Depletion Estimates

Captured fish were placed in an aerated bucket for processing after each pass. Fish were anesthetized prior to processing using a buffered Tricaine methanesulfonate (MS-222) solution, then identified to species, counted and measured. The fork length (FL) was measured to the nearest millimeter. If large numbers of other species were present, a subsample of 20 of each species, representing the range of size classes collected, was measured and the remaining individuals were tallied by tens. In 2009, a caudal fin clip was taken from 1 to 20 individual redband trout of different size classes at a subset of sites for genetic analysis. Fish were processed after each pass and released downstream of the lower block-net before starting the next pass. Amphibian and freshwater mussel presence were also recorded.

Mark-Recapture Calibration

Multiple pass removal procedures can result in biased estimates of abundance (Peterson et al. 2004). To evaluate the sampling bias, 10% of the site selections in each SMU were selected as mark-recapture calibration sites. Mark-recapture estimates provide less biased estimates of abundance, but take two site visits to complete, rather than a single visit. At calibration sites, after block nets were installed, a single electrofishing pass was conducted to capture a minimum of ten trout. These fish were anesthetized, measured for fork length, and if healthy and larger than 60 mm, were marked with a caudal fin clip and distributed back into the sample site, after recovery. The following day, the site was re-sampled using multiple pass depletion backpack electroshocking, as described above, recording the number of marked fish that were recaptured.

Habitat

Physical data describing habitat complexity were collected at each survey site. Overall maximum depth and thalweg length were measured at each site. Riparian vegetation was classified into one of four categories (conifer, deciduous, grass and sedge, limited vegetation). Obstructions that made electrofishing more difficult were counted and classified into categories describing the percent of the sample site obstructed: no obstructions, 0-33%, 33-66%, and 66-100% obstructed. Large wood (at least 10 cm) within the wetted channel was counted individually and the number of aggregates was tallied. Each site was then divided into 10 m transects, beginning 5 m upstream from the lower block-net for the survey site. For sites less than 50 m, five equally spaced transects were used. Maximum depth and depth at ¼, ½ and ¾ channel width, and wetted channel width were measured. Dominant substrate, proportion of aquatic vegetation, undercut bank volume, number of pools or channels, and the backwater area were also recorded.

Statistical Techniques

Length Frequency and Calibration Factor

Length-frequency analysis was used to separate age-1+ from age-0 trout for each SMU. These designations were not corroborated with scale or otolith analysis and therefore putative. Since age-0 fish are not completely vulnerable to our gear, they were not included in our estimates. A calibration factor was applied to estimates for all SMUs to adjust for biases associated with depletion estimates. The calibration factor was calculated as the mean ratio between depletion and mark-recapture density estimates for all years in all SMUs (including preliminary 2006 data) at a subsample of sites where both methods were employed. A generalized linear model 1-way ANOVA was used to test for differences in the calibration factor over the four years of sampling. Linear regression was used to determine the correlation between depletion and mark-recapture estimates. Data met all assumptions of normality and homogeneity of variance and did not require any transformation.

Fish Abundance Estimates

Abundance estimates for age-1+ redband trout in each SMU were extrapolated from depletion estimates (Zippen 1958) of fish density (fish/m) obtained at the sample sites. These densities were extrapolated to the extent of the sample frame by multiplying the surveyed reach lengths by the total sample frame length to provide estimates of total abundance and associated precision using analytical algorithms developed for GRTS design (Stevens and Olsen 2003, 2004), available as a part of the R package spsurvey (www.epa.gov.nheer/arm). Variance was estimated using an unbiased local neighborhood estimator that takes advantage of spatial patterns in the distribution, yielding lower variance (Steven and Olsen 2003, Dambacher et al. 2009). The mark-recapture calibration factor was applied to all abundance estimates to provide an unbiased estimate. Cumulative distribution functions (CDFs) were calculated for each SMU for each year to describe the proportion of the sampling frame that contained specific redband trout densities (or less).

Fish Abundance at Annual Sites

To determine whether fish densities changed at annual sites between years, and investigate the variance estimators at multiple spatial scales, redband trout abundance was analyzed separately at the SMU (landscape) and stratum (population) levels. Log-transformed densities of redband trout (fish/m) were compared among years, within each SMU at the stratum level, using a repeated-measures mixed effects model with a first order autoregressive covariance structure in Statistical Analytical Systems (SAS Institute 2001). Sites nested within strata and SMUs was used as both the random error term and between subject effects (Littell et al. 2006). Year was analyzed separately as a fixed factor at the landscape level and separately for each SMU; the covariance associated with time was also estimated. *Post-hoc* comparisons, using adjusted Tukey tests, were used to test for differences in means among years and strata. Assumptions of normality and homogeneity of variance were met.

The goal of using randomly selected annual sites was to increase our ability to assess trends at the SMU level. To do this, annual sites must consistently be representative of non-annual randomly selected sites within the SMU, and for population representation, at the stratum level. To determine whether annual sites densities reflect results from more the intensive non-annual site sampling, each SMU was analyzed separately using 2-way ANOVA with year and panel (annual site or non-annual site) as fixed factors. Log transformed density was the response variable. Assumptions of normality and homogeneity of variance were met.

Power Analysis

Statistical power analysis was conducted to estimate sample sizes needed to obtain target levels of precision and detectable effect size at both the SMU and stratum levels. Because target levels of precision or detectable effect size were expressed as relative values (proportion of the point estimate), we used the coefficient of variation averaged across years as our measure of variance.

Estimated sample size needed to obtain target levels of precision (95% confidence intervals) followed formula described Cochran (1978) and Zar (1996) and were calculated as:

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

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

Estimates of sample sizes needed to detect differences in population abundance were based on the formula of Snedecor and Cochran (1967) that was modified to substitute t for z as approximations for the probability distribution:

$$n_0 = \left(t_{\alpha(2),\infty} + t_{\beta(1),\infty}\right)^2 \left(V^2 / \delta^2\right)$$

where $t_{\alpha(2),\infty}$ = two-tailed t value with ∞ degrees of freedom associated with α probability of committing a type I error; $t_{\beta(1),\infty}$ = one-tailed t value with ∞ degrees of freedom associated with β probability of committing a type II error; and δ = the magnitude of the detectable effect size expressed as a proportion of the initial abundance.

Statistical Power to Detect Trends

Several scenarios were simulated to detect declining and increasing trends in fish abundance for individual SMUs. Two levels of statistical variation that spanned the range of the precision of our adult abundance estimates across five SMUs were modeled (note: Catlow was not included because of low sample size). To describe statistical variation, coefficients of variation (CV) of 0.16 and 0.21 were used. These CVs were the averages from years of SMU-level sampling and represented the range of precision across the five SMUs. To determine the power of our monitoring protocol to detect a declining trend, we used one-tailed tests and set α=0.2 and β=0.1 to reduce the chance of a Type II error (i.e., not detecting an actual decline in the population), which has been recommended for monitoring small populations of endangered species (Gryska et al. 1997). Statistical power of our scenarios were evaluated with the software program Trends (Gerrodette 1987).

Comparing 1999 Study Results to 2007-2009

In the 1999 study, abundance estimates were obtained at the landscape level across basins (SMU) using a GRTS design similar to that used in our current study, with deviations from the current study in stratification of the sampling frame, oversampling replacement site selection, and land access issues (Dambacher et al. 2009). Data collection of fish at the site level was similar to our study. Relative differences in abundance estimates from the 1999 sampling effort compared to this current study were assessed graphically. Comparisons of the numbers and densities of sites located on private and public lands were made to assess the potential bias within each study.

RESULTS

Sampling Success

Site Status

From 2007-2009, an average of 0.65% of the entire sample frame was sampled. The range per stratum among all years was 0.12% to 5.32%. Stratum-level sampling occurred in the Goose Lake and Warner Lakes SMUs in 2007, Malheur Lakes SMU in 2008, and Chewaucan and Fort Rock SMUs in 2009. For SMU level sampling, the number of sites sampled per SMU ranged between 7 and 39. For stratum level sampling, between 53 and 138 sites were sampled per SMU each year. Denied access to private land and time constraints prevented stratum level sampling in the Catlow Valley SMU, and any level of sampling in this SMU in 2008.

Thirty sites were targeted for each of the Fort Rock, Chewaucan, Goose Lake, and Warner Lakes SMUs and 40 sites for the Malheur Lakes SMU. However, the actual number of sites sampled in most SMUs approached, but did not always meet, these targets. Sixty-eight percent of the originally selected sites were sampled, which allowed us to maintain a representative sample across the range of each SMU, except for the Catlow Valley. The number of target sites varied by year, depending on the SMUs targeted for stratum level sampling. The proportion of targeted sites that had denied access or were dry at the time of sampling varied by year, but was highest in 2007 when the Goose Lake and Warner Lakes SMUs were sampled intensively (Table 1). **APPENDIX B** shows the locations and outcomes of sampling in each SMU. Lack of access across broad continuous blocks of the sample frame may affect our estimates of abundance, both at the stratum and SMU levels. For example, a large proportion of the Willow Creek stratum in the Chewaucan SMU was not sampled due to large blocks of denied access (**APPENDIX B**). Conversely, the Silver Creek stratum in Fort Rock SMU was represented by a more spatially balanced distribution of sites with minimal areas of denied access (**APPENDIX B**).

Table 1. Sampling outcome of sites selected for Great Basin redband trout surveys, 2007-09. The total number of sites represents the number of sites targeted for the entire sampling frame at the beginning of each season. Sites not included in the abundance estimate were categorized into six discrete categories.

Year	Total sites	Did not survey	Denied permission	Dry channel	Failed estimate	In-accessible	Not surveyable	Completed
2007	553	12	158	71	12	0	5	247
2008	323	7	47	31	1	0	2	235
2009	365	30	49	36	14	8	11	218

Of the total number of sites randomly selected across the entire sample frame over all years, 631 sites were targeted on private land and 602 on public land. Access on private land was limited, where access to approximately 40% of sites was denied. In some situations where it was necessary to cross private land (~3%), lack of access also limited our ability to visit sites on public land. Access in the Chewaucan, Malheur Lakes, and Catlow Valley SMUs was particularly difficult: access was denied to 50%, 47%, and 100% of the private sites, respectively. In other SMUs, denied access ranged from 26-35% of the sites on private land. Overall, the proportion of the number of completed sites on public land was 40% higher than on private land, based on the total number of originally allocated sites (Figure 2). Repeated contact and ODFW's fostering of working relationships with landowners was a likely cause for the relative increase in access between 2007 and in subsequent years (Table 1).

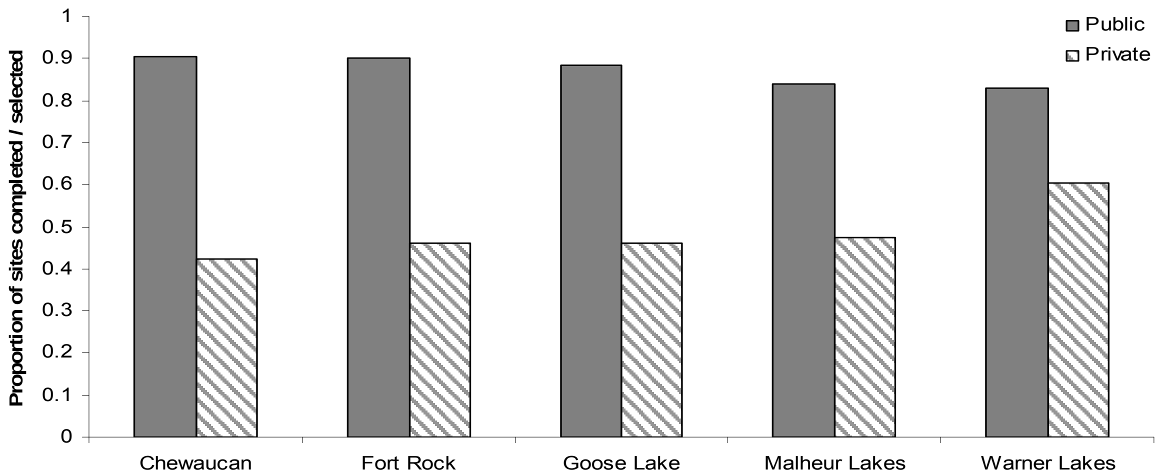


Figure 2. The proportion of sites within each SMU where sampling was completed based on the total number of sites originally selected for public and private lands. The number of sites completed includes dry sites, since permission was required to access all private land.

Sampling Intensity

The length of stream sampled at each sample site varied depending upon channel width and morphology. Sampled reaches averaged 70 m, which was approximately 20 times the wetted channel width, suggesting that crews underestimated site lengths. Sites were not accurately measured until the habitat survey was performed, which occurred after the fish survey was completed. The total number of sites completed was 247 (2007), 235 (2008), and 218 (2009); sampled at a rate of 5.1, 6.2 and 5.0 sites per day, respectively. The highest overall sampling intensity occurred in the Fort Rock SMU, where about 4% of the frame was sampled in 2009, which was a stratum level intensive sampling year in this SMU. Combining all three years of sampling, over 46 km of the total 2,420 km frame was sampled, or approximately 2% of the total sample frame (Table 2).

Table 2. The stream length (km) within the sample frame of each SMU stratum, calculated from 1:24,000 scale digital maps. The number of sites visited each year, and the length (m) of all sites sampled within that stratum were calculated. Stratum lengths were based on the 2007 version of the sample frame (2,420 km). The sample frame was adjusted in 2008 and 2009 to remove stream reaches judged not to support redband trout reducing the frame to 2340 km and 2273 km, respectively.

			2007		2008		2009	
SMU	**Stratum**	**Stratum length (km)**	**Number of sites**	**Sample length (m)**	**Number of sites**	**Sample length (m)**	**Number of sites**	**Sample length (m)**
Catlow	Home	39	-	-	-	-	1	74
	Rock	26	7	243	-	-	7	323
	Skull-3mile	10	-	-	-	-	2	142
Chewaucan	Chewaucan	283	24	1,731	19	1,262	24	1,955
	Crooked	38	4	225	3	140	22	1,439
	Willow	32	3	90	3	136	19	712
Fort Rock	Bridge	31	6	349	7	519	21	1,635
	Buck	38	9	814	7	545	19	2,018
	Silver	57	12	870	10	893	13	1,356
Goose Lake	Drews	87	20	1,274	6	312	8	381
	Dry	22	12	522	2	67	2	63
	Eastside	35	17	960	2	129	3	156
	Thomas-Bauers	129	21	1,057	10	624	12	582
	West Goose	56	20	1,139	4	179	4	224
Malheur Lakes	Blitzen	214	7	660	23	1,754	6	603
	East Burns	78	3	113	25	1,155	2	208
	McCoy	193	6	408	24	1,684	5	344
	Riddle	84	3	113	18	884	2	100
	Silver	186	6	282	25	1,494	3	273
	Silvies	468	14	744	23	1357	13	932
Warner Lakes	Deep	174	18	1509	13	900	16	1245
	Honey	94	17	1117	7	496	10	555
	Twentymile	46	18	1546	4	344	4	396

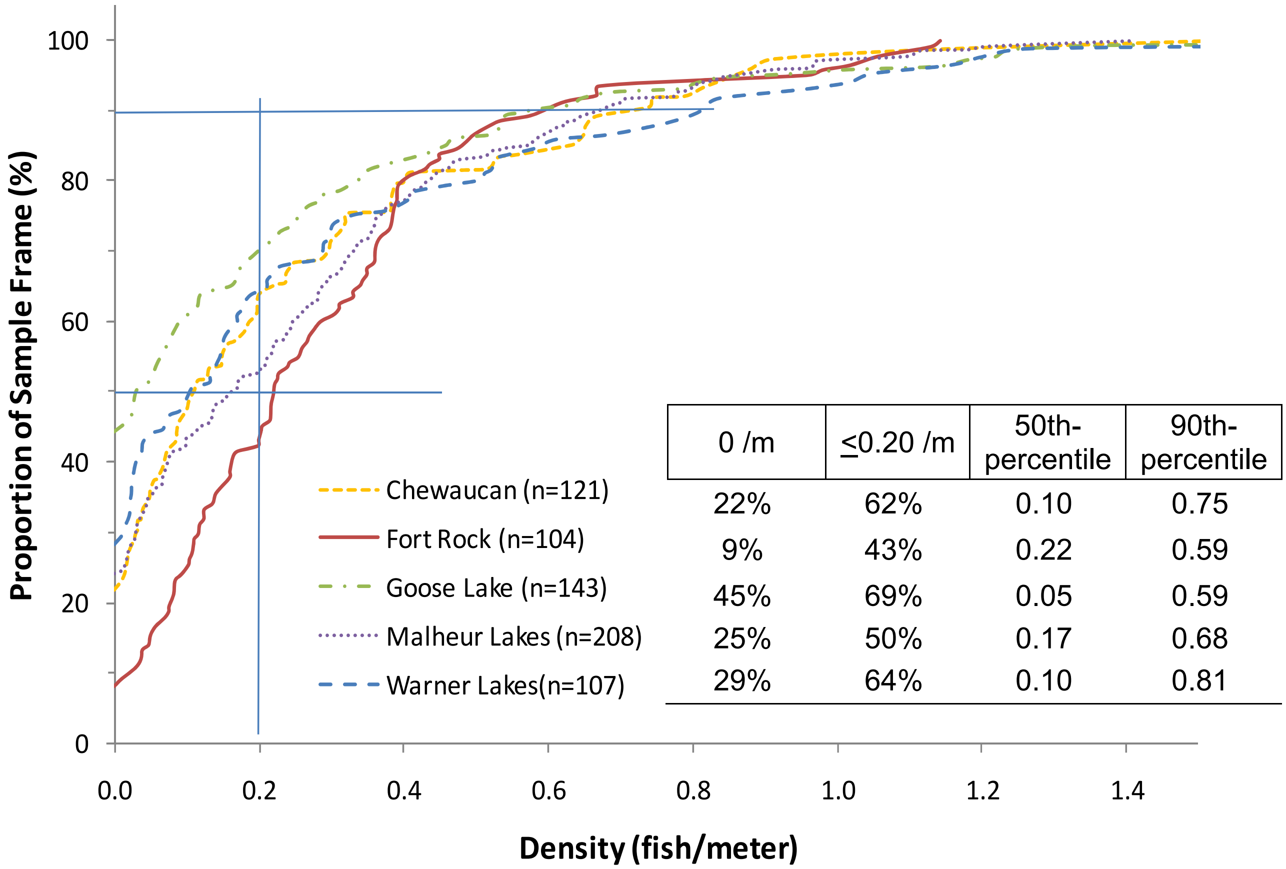
Depletion and Mark-Recapture Sampling Success

Depletion estimates were completed at 700 sites over the three years of this study (Table 1). To correct for the bias associated with depletion estimates, mark-recapture estimates were also obtained at 80 of these sites (37 in 2007, 14 in 2008, and 31 in 2009) and at an additional 19 sites sampled in the Chewaucan stratum in 2006, a preliminary study. There was a highly significant relationship between the two sets of estimates (R^2 = 0.85, $p<0.01$) and no significant difference was found between the calibration factor for any year (df = [3, 98], F = 1.43, p = 0.24). As such, a mean calibration factor of 0.60 was applied to all estimates.

Redband Trout Distribution

Redband trout were not evenly distributed at similar densities across the sampling frames of the six SMUs (**APPENDIX C**). Densities and distribution occasionally changed in strata within a SMU over the three years of this study (*see* **APPENDIX C**- Catlow Valley SMU, Rock Creek as an example). However, given our low sampling rate, changes in distribution were difficult to ascertain. With additional years of sampling, changes in distribution and density may be clearer. Although all basins showed patchy distributions of age-1+ redband trout, the degrees of fragmentation varied and were often dependent upon access to sampling sites, and whether the channel was dry (**APPENDIX B**). For example, in the Catlow Valley SMU, the Rock Creek stratum demonstrates how both density of fish and channel dryness can vary with year. In 2007, densities of fish at sites were sometimes five times those of 2009, but nearly half of the sampling frame was dry. In contrast, only two sites within the sampling frame were dry in 2009 yet density of fish per site was markedly lower.

Differences in spatial patterns of redband trout distribution among the SMUs were apparent by examining cumulative distribution frequencies (Figure 3). Occurrences were not rare in any SMU. However, the Goose Lake SMU consistently had the highest proportion of the sample frame void of redband (45%), which may be due to poorer habitat quality or a less accurate sample frame. This pattern contrasts sharply with the more uniform pattern observed in the Fort Rock SMU, where the portion of sites that lacked redband trout was only 9% over the three years of sampling. The Goose Lake SMU also showed the lowest densities of any SMU, as indicated by nearly 70% of the frame with densities of less than 0.2 fish/m (Figure 3). In nearly all SMUs, over 90% of the sample frame had redband trout densities less than 1.0 fish/m. The Warner SMU had the highest density in the upper 90% of the sample frame (0.81 fish/m) and 50% of the sample frame had densities less than 0.05 fish/m (Figure 3). All SMUs showed some evidence of higher fish densities upstream and lower densities downstream (**APPENDIX C**). CDFs and frequency distributions of site densities for individual sample years for each SMU, including the Catlow, are shown in **APPENDIX D**.



0 /m	≤0.20 /m	50th-percentile	90th-percentile
22%	62%	0.10	0.75
9%	43%	0.22	0.59
45%	69%	0.05	0.59
25%	50%	0.17	0.68
29%	64%	0.10	0.81

Figure 3. Cumulative distribution frequencies of the densities of age 1+ redband trout across the sample frames of five Species Management Units. CDFs were estimated from all density estimates obtained in each SMU during 2007-09 and are weighted to adjust for varying sample rates among different strata and years. Vertical and horizontal lines correspond to benchmark values used to compare distributions among SMUs, as tallied in the table within the figure. The Catlow SMU is not included in this analysis due to low sample size.

Abundance Estimates and Associated Error

Using the GRTS sampling design, the age-1+ redband trout abundance was estimated to be between 15,800 and 546,800 fish per SMU with relative precision (± 95% confidence interval) ranging from 15% to 59% (Table 3). Precision at the SMU level was within the 20% relative target twice during this study. Abundance estimates for the Catlow SMU consistently had lower precision than other basins. This difference was primarily due to surveying only a small number of sites and sampling primarily in only one stratum. Reduced precision at the SMU level was also due to redband trout showing fragmented distributions, particularly in the Goose Lake SMU. The high precision of the 2009 Fort Rock SMU abundance estimate (Table 3) may be largely attributed to the low number of sites with zero fish and relatively consistent densities among sample sites (**APPENDIX C**). At the stratum scale, abundance estimates varied greatly within a SMU. Precision ranged from ±15% to ±136%, and was within our 40% target precision level in 11 of the 20 strata sampled.

Table 3. Estimated abundance of age-1+ redband trout (adjusted with calibration factor), 95% confidence intervals expressed as percent of the estimate, coefficients of variation for density estimates (CV), and the number of sites sampled per SMU and stratum. Overall SMU values are in bold.

Year	SMU	Stratum	n	Estimate	Lower 95% confidence limit	Upper 95% confidence limit	Relative 95% confidence interval	CV
2007	**Catlow Valley**	(Rock Only)	**7**	**29,952**	**12,409**	**47,495**	**59%**	**0.81**
	Chewaucan		**31**	**147,174**	**86,678**	**207,670**	**41%**	**1.33**
	Fort Rock		**27**	**85,825**	**66,028**	**105,622**	**23%**	**0.68**
	Goose Lake		**90**	**125,543**	**84,796**	**166,290**	**33%**	**1.44**
		Drews	20	17,196	8,853	25,540	49%	1.54
		Dry	12	2,869	-1,026	6,765	136%	2.76
		Eastside	17	22,556	17,489	27,623	23%	0.57
		Thomas-Bauers	21	39,981	18,871	61,090	53%	1.33
		West Goose	20	42,941	9,711	76,171	77%	1.87
	Malheur Lakes		**39**	**546,783**	**368,497**	**725,070**	**33%**	**1.00**
	Warner Lakes		**53**	**82,586**	**47,683**	**117,488**	**42%**	**1.44**
		Deep	18	52,030	18,780	85,281	40%	1.22
		Honey	17	5,168	2,377	7,959	64%	1.57
		Twentymile	18	25,387	15,152	35,622	54%	1.21
2008	**Chewaucan**		**25**	**135,622**	**104,185**	**167,058**	**23%**	**0.87**
	Fort Rock		**24**	**48,116**	**30,875**	**65,358**	**36%**	**0.88**
	Goose Lake		**24**	**43,807**	**25,949**	**61,666**	**41%**	**0.91**
	Malheur Lakes		**138**	**478,170**	**406,391**	**549,949**	**15%**	**1.02**
		Blitzen	23	111,468	83,947	138,990	25%	0.85
		East Burns	25	29,159	18,512	39,806	37%	1.13
		McCoy	24	129,678	92,502	166,853	29%	0.85
		Riddle	18	47,943	30,650	65,235	36%	1.07
		Silver	25	42,957	28,336	57,578	24%	1.21
		Silvies	23	116,965	68,112	165,818	42%	1.25
	Warner Lakes		**24**	**146,013**	**96,341**	**195,685**	**34%**	**0.95**
2009	**Catlow Valley**		**9**	**15,869**	**7,767**	**23,971**	**51%**	**0.98**
		Rock	7	13,538	6,848	20,228	49%	0.98
	Chewaucan		**65**	**130,180**	**98,488**	**161,872**	**24%**	**1.05**
		Chewaucan	24	118,458	86,956	149,959	27%	1.03
		Crooked	22	9,776	6,408	13,144	35%	1.26
		Willow	19	1,946	1,111	2,782	43%	1.24
	Fort Rock		**53**	**44,888**	**36,676**	**53,099**	**18%**	**0.67**
		Bridge	21	15,662	13,002	18,322	17%	0.56
		Buck	19	12,629	10,800	14,457	15%	0.53
		Silver	13	16,598	9,047	24,148	46%	0.88
	Goose		**29**	**45,294**	**24,907**	**65,680**	**45%**	**1.02**
	Malheur Lakes		**31**	**398,478**	**273,487**	**523,468**	**31%**	**0.80**
	Warner Lakes		**30**	**129,569**	**82,004**	**177,134**	**37%**	**1.16**

Power Analysis

The target levels for relative precision at the SMU and stratum level were 20% and 40%, respectively. With the level of sampling achieved during 2007-09, desired targets were met in a few SMUs and about half the strata (Table 3). The results of the power analysis showed that to achieve target levels of relative precision, the number of sites in every SMU would need to be increased to a minimum to 50 sites (Figure 4). Further, for all SMUs, the rate of improvement in precision by adding additional samples decreases dramatically once the sample size exceeds about 50 sites. The between-site variability (CV) ranges from 0.67-1.44 and can dramatically influence the number of sites needed to meet precision targets. For example, to achieve precision within ± 20% in the Goose Lake SMU, approximately 156 sites would need to be surveyed; however, only 51 sites would be needed to be surveyed within the same level of precision in the Fort Rock SMU. If our target level of precision was reduced to 40%, only 39 and 13 sites in the Goose Lake and Fort Rock SMUs, respectively, would need to be surveyed.

At the stratum scale within each SMU, large between-site variability (CV) within some strata resulted in large differences in levels of precision (Table 3). For example, in the Dry stratum of the Goose Lake SMU, nearly 140 sites would need to be surveyed to achieve a ±40% relative precision; this is not realistic and essentially would constitute a census. In contrast, within the same SMU, only 5 sites are needed to obtain the same level of precision in the Eastside stratum, because of much lower between-site variability. Only 55% of the sampled strata were within the desired ± 40% relative precision target.

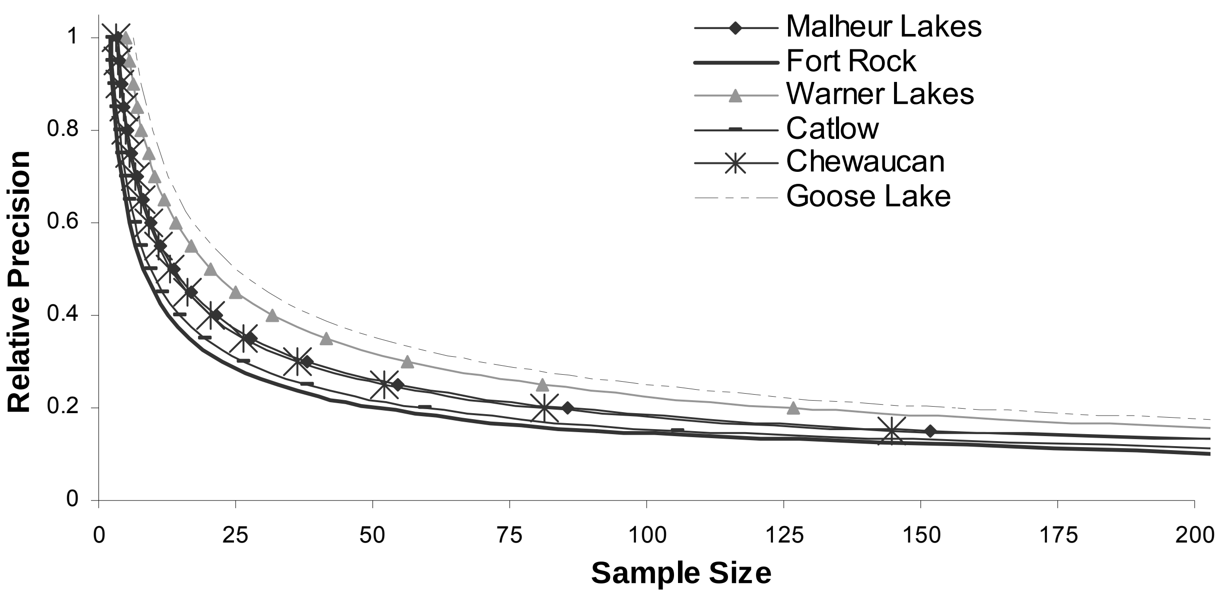


Figure 4. Results of the power analysis showing the relative level of precision of GRTS design for differing sample sizes (x-axis) for the six SMUs. Coefficients of variation were averaged across the three sample years.

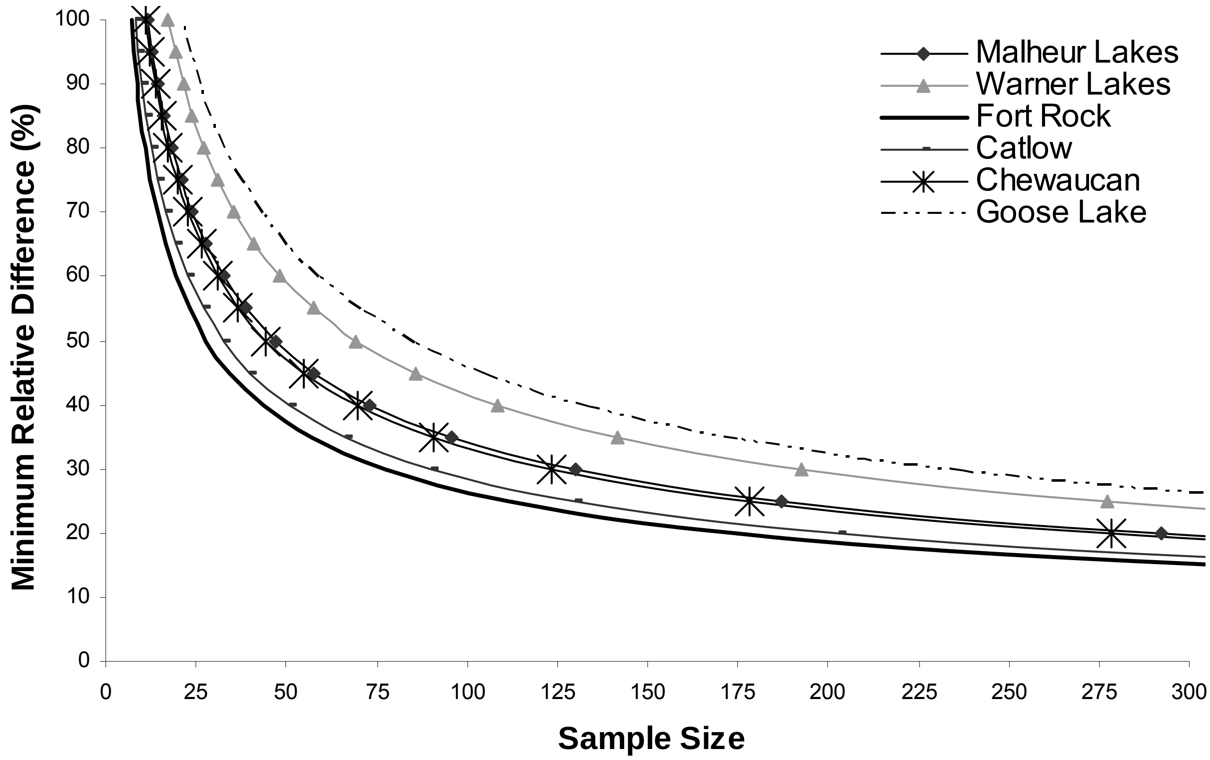


Figure 5. Results of the power analysis showing the minimum percent detectable difference (y-axis), based on differing sample sizes (x-axis), for the six SMUs. Coefficients of variation were averaged across the three sample years.

Sample size targets of 30-40 sites at the SMU (basin) scale and 30 sites in each stratum within a SMU (population-scale) provide lower sensitivity when viewed in terms of levels of minimum detectable differences. Using the recommendations of Gryska et al. (1997) for rare or listed species, the level of a type I error was set at 20% and the level of a type II error at 10%. Using these standards, a sample size of 47 sites would allow the detection of a 50% difference in redband trout numbers in the Malheur Lakes SMU (Figure 5). Conversely, this sample size would only be sensitive enough to detect a 60% abundance change in the Warner SMU. Current target levels of sampling at the SMU level (Malheur Lakes - 40 sites, all other SMUs 30 sites) allows detection of between 50-90% levels of change (Malheur Lakes – 55%, Fort Rock – 50%, Chewaucan – 60%, Warner Lakes – 75%, Catlow Valley – 50%, and Goose Lake – 85%). At the current levels of sampling at the stratum scale (30 sites per stratum), minimum levels of sampling needed to detect changes in abundance range from 20-100%. In roughly half of the strata, we can only detect abundance changes greater than 60%. Only the Bridge and Buck strata within the Fort Rock SMU and the Eastside stratum within the Goose Lake SMU have the power to detect smaller changes in abundance (less than the 40% target) estimates (Buck, Bridge, and Eastside, 21, 26 and 31% respectively). However, it should also be noted that our target of sampling 30 sites per stratum was not typically achieved and sampling fewer sites further reduces the ability to detect small changes in abundance. The actual number of sites sampled at the stratum level allows us to detect 60% or greater changes in abundance estimates in most (70%) of the strata. Reducing the probability of committing a type II error (not detecting a change in redband trout numbers) would increase the sensitivity of detecting differences.

Statistical Power to Detect Trends

Our ability to detect trends in abundance for individual SMUs depends on the duration of sampling (number of years of the study) and the precision associated with the abundance estimates. Additionally, our ability to detect declining trends is higher than our ability to detect increasing trends. At the range of precision we have calculated for SMU-level sampling, after five years of sampling we would have a 90% probability to detect a 20% annual decline and a 30-40% annual increase in abundance (Figure 6, upper panel). After 10 years of sampling, minimum annual rates of change that we can detect with 90% probability improve to a 3.5-4.5% decline and a 5-7% increase. When viewed in terms of minimum detectible overall changes in abundance, the annual rates of change listed above correspond to the ability of detecting 40-50% decreases and 60-90% increases after five years of sampling and 30-40% decreases and 40-60% increases after ten years of sampling (Figure 6, lower panel). These simulations assume sampling is continuous on an annual interval and that natural variation in SMU abundance is small relative to variation associated with the precision of abundance estimates (Gerrodette 1987).

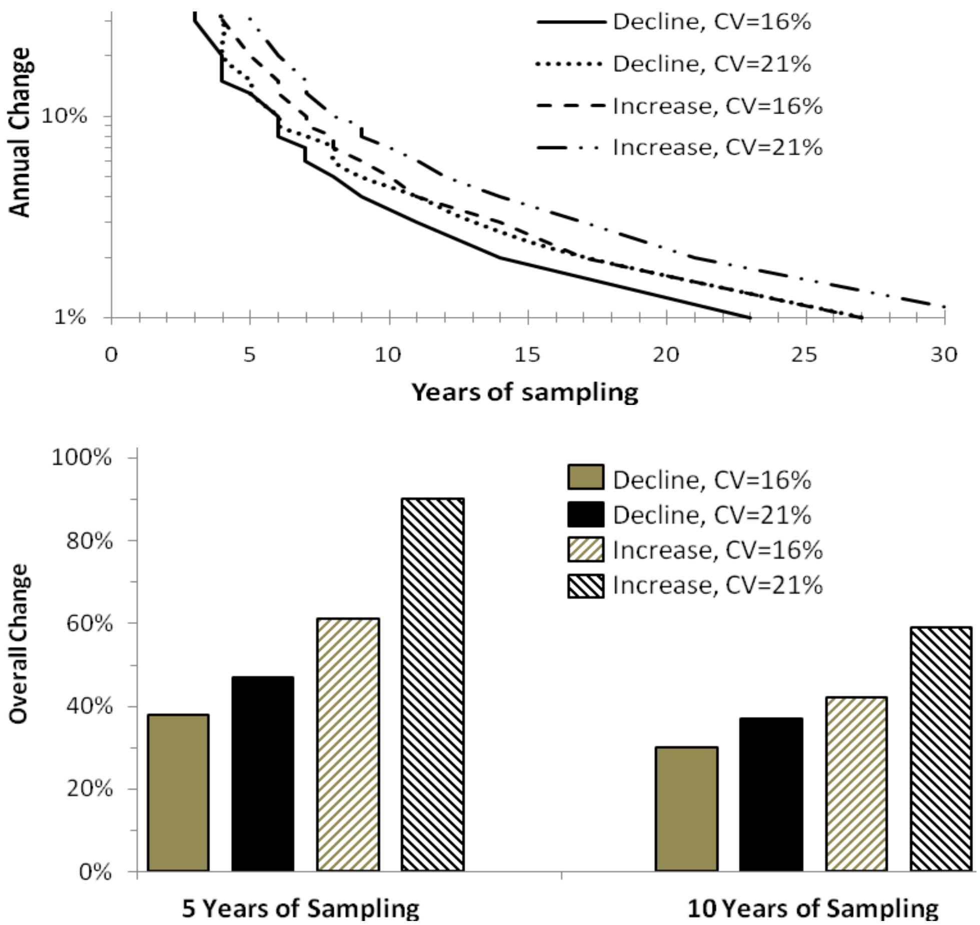


Figure 6. Power of trend detection for changes in redband trout abundance. Upper panel: relationships between sampling duration and ability to detect annual decreases or increases in abundance over the range of precision associated with abundance estimates for Species Management Units (SMUs). Note Y-axis is a log scale. Lower panel: minimum detectable decreases or increases in abundance within a SMU after five and ten years of continuous annual sampling.

Fish Abundance at Annual Sites

Across all SMUs, a total target of 95 sites was selected for annual sampling. Some sites proved to be dry and were replaced following the random selection order. Ultimately, a total of 85 annual sites were visited over the course of this study. Fifty-seven sites, distributed over five SMUs and all strata (Table 4), were visited in all years and were included in the analysis.

Table 4. The number of annual sites distributed among SMUs and within each stratum. The number of sites within each stratum was proportional to the overall size of the sample frame for that stratum, such that larger strata received more annual sites. The number of sites included in this table represents only sites that were visited in all three years of this study. The Catlow Valley SMU is not included in this analysis since no sites were visited in 2008.

SMU	Stratum	Number of annual sites
Chewaucan	Chewaucan	8
	Crooked	2
	Willow	1
	Total	**11**
Fort Rock	Bridge	4
	Buck	2
	Silver	5
	Total	**11**
Goose Lake	Drews	2
	Dry	1
	Eastside	1
	Thomas-Bauers	5
	West Goose	1
	Total	**10**
Malheur Lakes	Blitzen	3
	East Burns	1
	McCoy	3
	Riddle	1
	Silver	1
	Silvies	5
	Total	**14**
Warner Lakes	Deep	5
	Honey	4
	Twentymile	2
	Total	**11**

Redband trout densities at annual sites varied significantly by year (Table 5). The variance among the SMUs (0.002) was small compared to the between-site variance (0.02) and the within-site variance residual error (0.02), implying that temporal and smaller scale variability was more pronounced than large scale variability. At the landscape level (including all SMUs), *post hoc* Tukey tests revealed no difference in fish densities between 2007 and 2008 or between 2008 and 2009 (p = 0.67, p = 0.10, respectively). However, densities at repeated sites in 2007 and 2009 were significantly different (p = 0.03) (Table 5).

Table 5. Results of a repeated measures analysis of variance comparing all years and SMUs with covariance parameters. The site covariance parameter was the between site variance, basin was the variance among the basins, and the residual variance encompasses the within site variance and unexplained error. The AR(1) estimate reflects the correlation between two densities obtained one year apart; this estimate reduces the within-site variance between observations.

Type III fixed effect	num df	den df	F	p-value
Year	2	74.5	3.69	0.03
	Covariance parameters	**Estimate**		
	Site	0.02		
	Basin	0.002		
	AR(1)	0.22		
	Residual	0.02		

To determine if densities at annual sites, within each SMU, differed between years, each SMU was analyzed separately. Although a parameter of interest is whether strata within a SMU differed in densities each year, sample sizes in all strata are not adequate to make this comparison and instead can only be assessed every three years. Analyzing each SMU separately revealed significant yearly differences in fish densities within only the Fort Rock, and marginally within the Chewaucan SMUs (p = 0.03, p = 0.06, respectively). *Post hoc* Tukey tests revealed that the difference in densities at these annual sites occurred between sites in 2007 and 2009. In most SMUs (except Catlow Valley, which was not included in the analysis), the residual error explained slightly more variability than the between-site variance (results not shown). Only in the Chewaucan SMU was the between site variance greater than the residual error. Generally, the within and between-site variance was roughly the same, again suggesting the importance of temporal and small scale site-level variability. Correlations of density measures one year apart (AR(1)) ranged from -0.52 to 0.80, suggesting that densities at some sites changed little between years, depending on the SMU ((AR(1): Chewaucan = -0.26, Fort Rock = -0.52, Goose Lake = 0.02, Malheur Lakes = 0.73, Warner Lake = 0.68) . Using a repeated measures analysis allows for a measure of variability between and within annual sites, between years at the individual SMU level, and correlations between densities over time. Additional years of sampling should allow for a more clear determination of trends at these annual sites over time.

To determine whether annual sites within each SMU reflect results from the more intensive non-annual site sampling, each SMU was analyzed separately. Annual sites differed from non-annual sites in only the Chewaucan (2007 and 2009) and Goose Lake (2007) SMUs (Figure 7). If annual sites track yearly SMU level variation in abundance, then these sites may be useful in inferring trends at larger landscape levels; however, these limited data do not demonstrate that annual sites confidently reflect yearly average fish densities at the SMU level in every SMU, for every year.

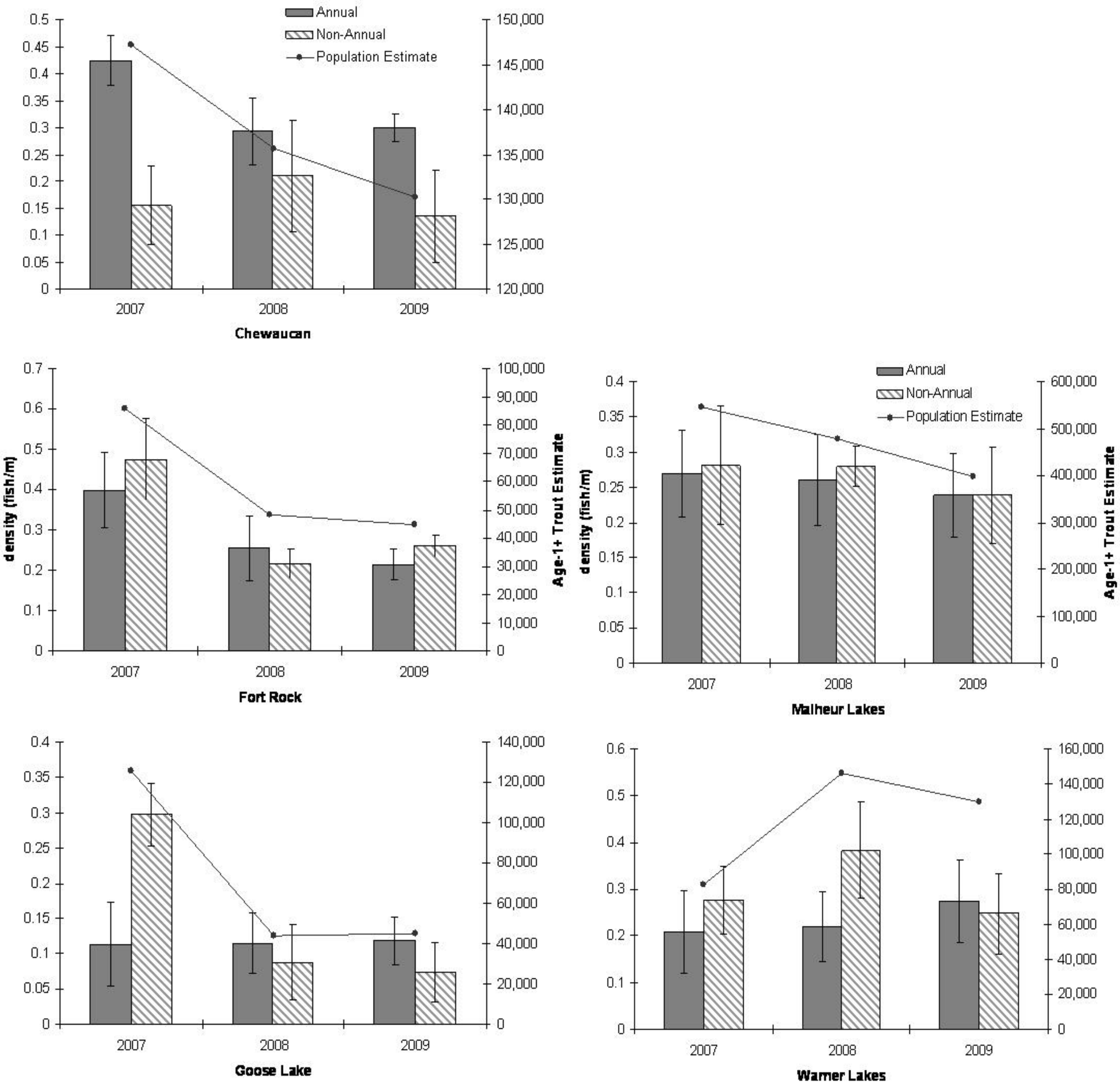


Figure 7. Redband trout densities at annual and non-annual sites + 1 S.E. (vertical bars) for each SMU. The calibrated population estimates were overlaid to show the estimated SMU population for each year relative to the calculated mean fish densities at annual and non-annual sites.

Comparing Studies: 1999 to 2007-2009

In 1999, the total population of age-1+ southeastern Oregon Great Basin Redband trout was calculated as roughly 970,000 with a 95% CI that was ± 15% of the estimate (Dambacher et al. 2009). After adjusting this estimate with the calibration factor used in our analysis, the population estimate increased to 1,609,000 age-1+ fish over the 2,167 km sampling frame used in that study. The 2007-2009 study estimates that overall abundance, at the landscape level, varied from 1,018,000 ± 19% in 2007, to 852,000 ± 11% in 2008, and 764,000 ± 18% in 2009. Across the entire sample frame (2,240 km), population estimates for 2007-2009 were less than 1999, but do not appear to differ at the landscape level from 2007 to 2009. However, abundance estimates in all SMUs were generally greater in 1999 than in 2007-2009 (Figure 8).

The 1999 study differed slightly in design, particularly with how replacement sample sites were chosen (Dambacher et al 2009). Yet, both studies suffered from the denial of access to roughly half of all sites on private land. The 1999 study's highest sampling denial rate was 60% in the Warner Lakes SMU and lowest in the Catlow Valley SMU (6%). In the largest SMU sampled, the Malheur Lakes had a denial rate of 36%. This is in contrast to the rate of private land access in our study, where no access was granted to private land in the Catlow Valley SMU. In fact, in the Catlow Valley, landowner permission was needed to gain access to a few public land sites. Permission was not obtained to cross private land to sample public land in Home Creek in the Catlow Valley until 2009, and due to time constraints, only one site was sampled in this particular stratum. The Goose Lake SMU had the fewest denials in the current study (26%) while access to roughly half of private sites in the Chewaucan and Malheur Lakes SMUs was denied. In both studies, densities of fish were often lower on private sites than public sites (Figure 9); this was always the pattern in the 2007-2009 study where the total number of sites completed was cumulatively greater than in 1999. Further, uncertainty was higher on private land, in both studies, due to lower sample size.

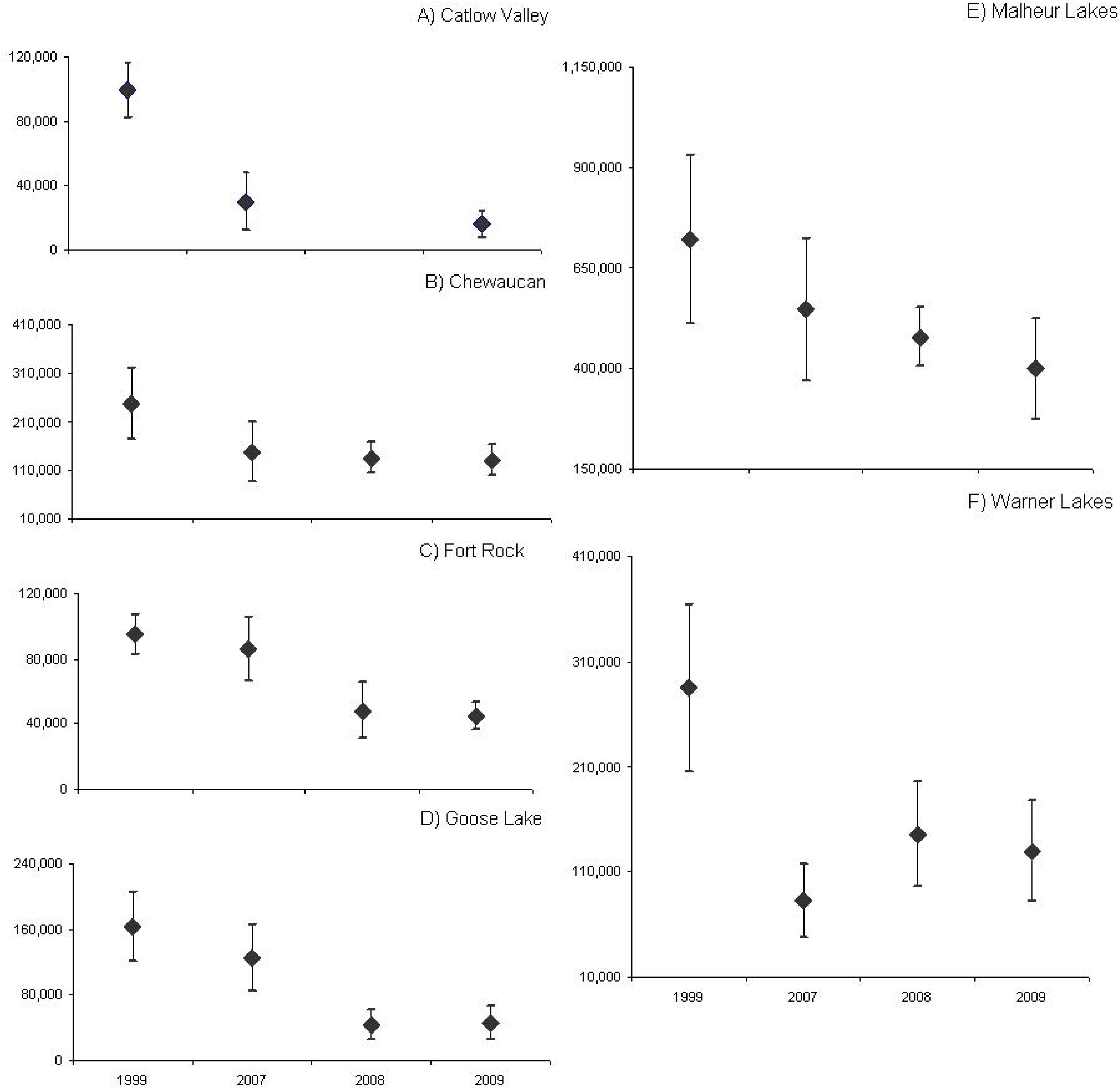


Figure 8. Abundance estimates for age-1+ redband trout in each SMU with 95% confidence intervals (vertical bars). Estimates include calibrated adjustments for each SMU using 1999 data (adapted from Dambacher et al 2009) paired with estimates obtained in the 2007-2009 study. A) Catlow Valley, B) Chewaucan, C) Goose Lake, D) Fort Rock, E) Malheur Lakes, F) Warner Lakes.

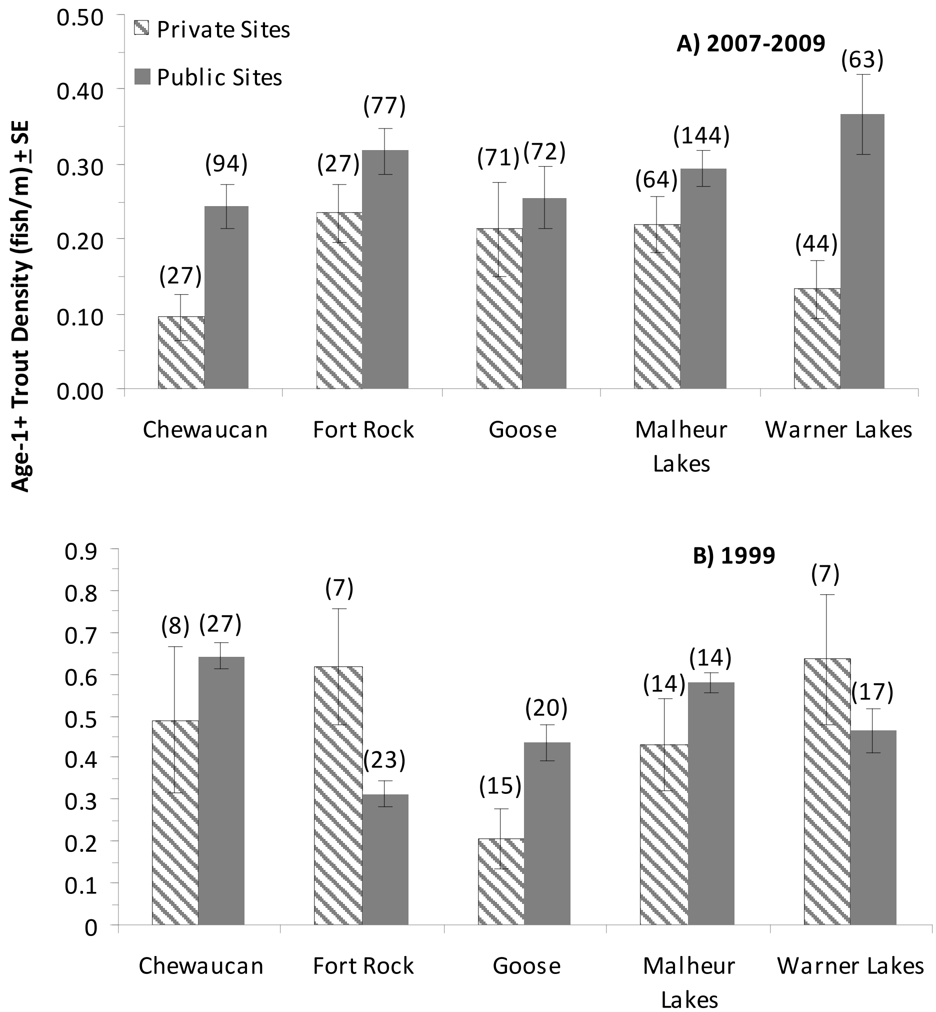


Figure 9. The average densities of redband trout (fish/m) at public and private sites within each SMU calculated for the (A) 2007-2009 and the (B) 1999 studies. The numbers in parentheses above each column represent the numbers of surveyed sites used to calculate that average.

DISCUSSION

Abundance estimates of age-1+ Great Basin redband trout in southeastern Oregon suggest abundant populations exist within most of the SMUs assessed in this study. Our current level of sampling in the Catlow Valley SMU precludes us from having the same degree of inference as for other SMUs. Employing a robust GRTS design and using the NHB variance estimator, the average abundance estimate at the landscape level across the three years was 878,000, with a relative 95% confidence interval of ± 16%. Abundance at the SMU level remained largely constant from 2007 to 2009 in most SMUs.

Target levels of relative precision at the SMU level were achieved in two SMUs, Malheur Lakes in 2008 and Fort Rock in 2009, both intensive sample years (Table 3). With additional resources, adding a hypothetically achievable number of additional sites (~10) at the SMU level would typically increase relative precision 5-10%, sometimes to within the original 40% target level. However, increasing the number of sites sampled to realize targets is impracticable, based on available funding and logistical constraints. Our power to detect change remains low and increasing the number of sites sampled to adequately detect small changes (< 40%) is not feasible. At the current level of effort, our ability to detect changes in abundance ranges between 40-90%, depending on the SMU. Our ability to detect trends in abundance improves with additional continuous sampling years, provided that natural variation in abundance is small relative to the precision of the estimate. Levels of change necessary to trigger specific management actions have not been established.

The low levels of precision and power to detect changes in these SMUs and strata was not necessarily a result of insufficient sample sizes, but rather of high levels of site variability. The question remains whether yearly differences, particularly between the 1999 (Dambacher et al. 2009) and current study, are an effect of sampling (residual error) or a product of natural variation (year to year or site level)?

Differences in sampling methods between 1999 and this current study may reflect part of the difference in these abundance estimates. In 1999, although the GRTS design was used, the authors used a different protocol when stratifying their samples and designating replacement sites. Although sample sites were spatially balanced and randomly allocated in both studies, the 1999 study sites were not stratified at the population level. Samples were instead allocated throughout a SMU, but were not spatially balanced proportionally within each population (stratum). As such, many of the sites were clustered in regions of the SMU, particularly towards the headwaters, and sites located far downstream were less represented in the sample. Downstream portions of the sample frame have a greater tendency to be privately owned and denial of access played a part in the distribution of sites completed. In the current study, sample sites were stratified within a SMU by proportionally allocating a number of sites within a stratum (population) based on the size of the frame. When extrapolated to SMU level abundance estimates, the 1999 study assumes that a site located in one population was representative of all sites within the entire SMU. Our study assumes that sites within a stratum (population) were more representative of other sites within that stratum. Abundance was estimated within each stratum separately and then all strata totals were aggregated to estimate the entire SMU. With both SMU and stratum level sampling, a more spatially balanced distribution of sites was achieved that better represents an entire stratum. This is particularly evident in the Warner Lakes SMU, where sites in the 1999 study largely fell in public lands in the headwaters, where densities of fish tend to be higher, biasing the estimates toward over estimation. This may partly account for the large decrease in abundance seen in the Warner Lakes SMU from 1999 to the present.

Designating replacement sites introduced the potential for bias into both studies. In 1999, the authors chose to non-randomly select sites that matched characteristics of the original denied site, rather than maintaining the ordered block that retains the random spatial balance of the design (Dambacher et al. 2009). This allowed them the logistical efficiency to fully complete their sampling plan at the cost of violating an assumption of the statistical model used to estimate the population. In our study, we instead used a randomly structured ordered block allowing us to maintain the spatial balance; however, sites used in the oversample may not reflect the characteristics of sites that were denied. Unlike the 1999 study, we were unable to reach our targeted sampling schedule reducing the overall number of sites visited. In both studies, inability to access the exact proportion of similar sites to the base sample introduces a level of bias to the estimates which was greatest in SMUs with higher levels of denied access to private sites. Lack of access across broad continuous blocks of private land may affect estimates of abundance both at the stratum and SMU levels.

Denied access to portions of the sample frame on private land occurred at approximately the same rates in each study and differed only in the SMU in which access denial was the greatest (e.g. in 1999 Warner Lakes had the highest rate of private access denials while in 2007-2009 the Catlow SMU had the highest denial rate). Based on the level of precision of the abundance estimate at the landscape scale, the bias associated with lack of access was considered to be relatively small in the 1999 study, but greatest for the Warner Lake SMU. Our study suggests that densities of fish on private lands were always lower than on public lands (Figure 9). This bias was particularly pronounced in the Chewaucan and Warner Lakes SMUs, where densities on public land were greater than on private lands by 58% and 65%, respectively. Comparing private versus public sites separately for each basin revealed that these differences were significant. This suggests that if sites used in the analysis come predominately from public sites, the resulting population estimate may over estimate the population despite precision levels within our target range. To account for the bias associated with access issues, the 1999 study post-stratified individual SMUs by landownership (private versus public) and weighted sites according to the prevalence within each SMU. This differed from our study where weights for each stratum were proportional to the number of sites targeted and the size (meters of stream in the sample frame) of the stratum. Weights were adjusted to account for the number of sites completed, failed estimates, and denied sites. Since sites randomly selected for our study were stratified using a probabilistic survey design at the stratum level, estimates of abundance followed that design. Further, post-stratifying by ownership type would not allow adequate sample sizes in each stratum to obtain precise estimates.

Depletion estimates are known to underestimate true population size. While mark-capture methods are generally unbiased, it was not practical to employ this technique at all sites because of the extra time needed to obtain the estimates. Instead, the mark-recapture method was used in addition the multiple-pass depletion method at a subset of locations in each SMU. The ratio of the depletion estimate to mark-recapture estimate was employed as calibration factor for estimating bias and all depletion estimates were adjusted using this calibration factor. This attempt to minimize bias is a common approach generally used to correct for the bias associated with visual observation versus mark-recapture estimates (Guy and Brown 2007); however, it does not incorporate all aspects of instability of capture efficiency. Capture efficiency declines with successive electroshocking passes and is also influenced by variation in habitat characteristics (Rosenburger and Dunham 2005). Habitat characteristics measured in our study will be paired with density estimates at mark-recapture sites to achieve a presumably more accurate calibration factor in future analyses, based on individual site level habitat characteristics (Cooperative project with USGS). The relative error associated with using the calibration adjustment was unknown.

The between-site variation (CV) varies among SMUs and changes year-to-year within each SMU (Figure 10). The CV does not appear to be dramatically lower in years of stratum level sampling (increased overall sampling rates), suggesting that large differences may be due to annual changes in fish densities, habitat, variation in the time of sampling (inter-annual), or site-to-site level differences. Only in 2008 does the CV remain relatively equal between SMUs, which may possibly be a result of a slight deviation in sampling logistics used that year. To be most efficient logistically, localized areas were chosen for sampling. Crews centralized to complete sample sites, sometimes all sites within a single stratum, before moving on to another area. During the two other sampling seasons crews were stationed geographically and were assigned to specific SMUs. The Malheur Lakes SMU, the largest SMU with the greatest number of strata, was sampled intensively at the stratum level in 2008. As such, the time it took to sample a localized area was, in some cases, under two weeks. The different sample schedule in 2008 may have reduced some of the between-site variation caused by more protracted sampling. In addition, if sites were visited earlier in the season (when more water was present) then that stratum may also have a different fish density then at a later point in the season (inter-annual variation). Unless every stratum was visited at the same time of the season, the yearly differences that appear to be attributed to differences in water year may actually be, in part, an artifact of changes in water levels within a single season (i.e. sampling at only the beginning of a season may reflect higher water levels than later in the season).

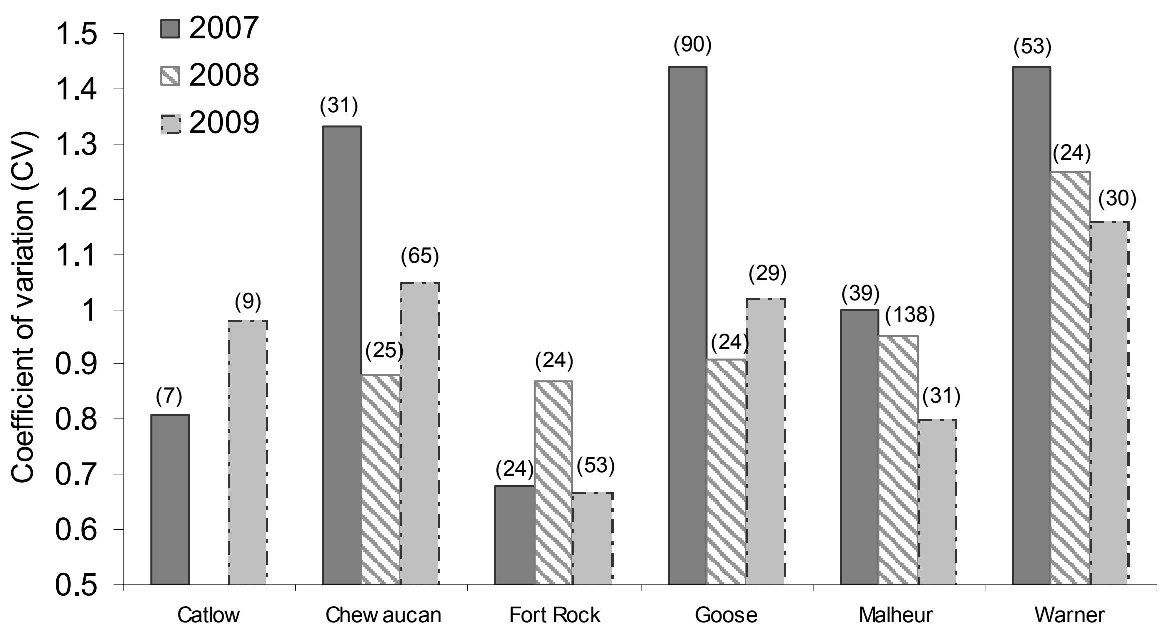


Figure 10. Coefficient of variation of redband trout density estimates for each SMU in each year. For each year, the total numbers of sites surveyed in the basins are shown in parentheses.

Given the hydrological history of the Great Basin and the diversity in water flows each year, it is difficult to compare the abundance of redband trout in the years of relatively high stream flow with lower flow years (Figure 11). Several years of above average water flows prior to the 1999 surveys may have provided more available habitat, potentially allowing for a higher abundance and broader distribution of fish in addition to higher levels of recruitment. Higher recruitment during 1999 is suggested when comparing abundance of age-0 redband trout between the two studies. Although not specially targeted by the sampling protocols in either study, age-0 trout were generally more abundant in 1999 than during 2007-09. While slight differences in capture effort for age 0 fish employed by each study may influence the total fish captured, generally among all six SMUs, the number of age-0 trout captured per site was highest in 1999 in 21 out of 23 cases.

Although flows differ in each stream in every basin, years of high or low flows generally followed the same trends in each SMU (Figure 11). With the limited time series we have available, little can be concluded regarding the relationship of redband trout abundance and distribution to climatic cycles in the Great Basin. Continued yearly sampling, in particular at the stratum level and at annual sites, will continue to elucidate potential relationships and assist in revealing yearly effects of natural variation.

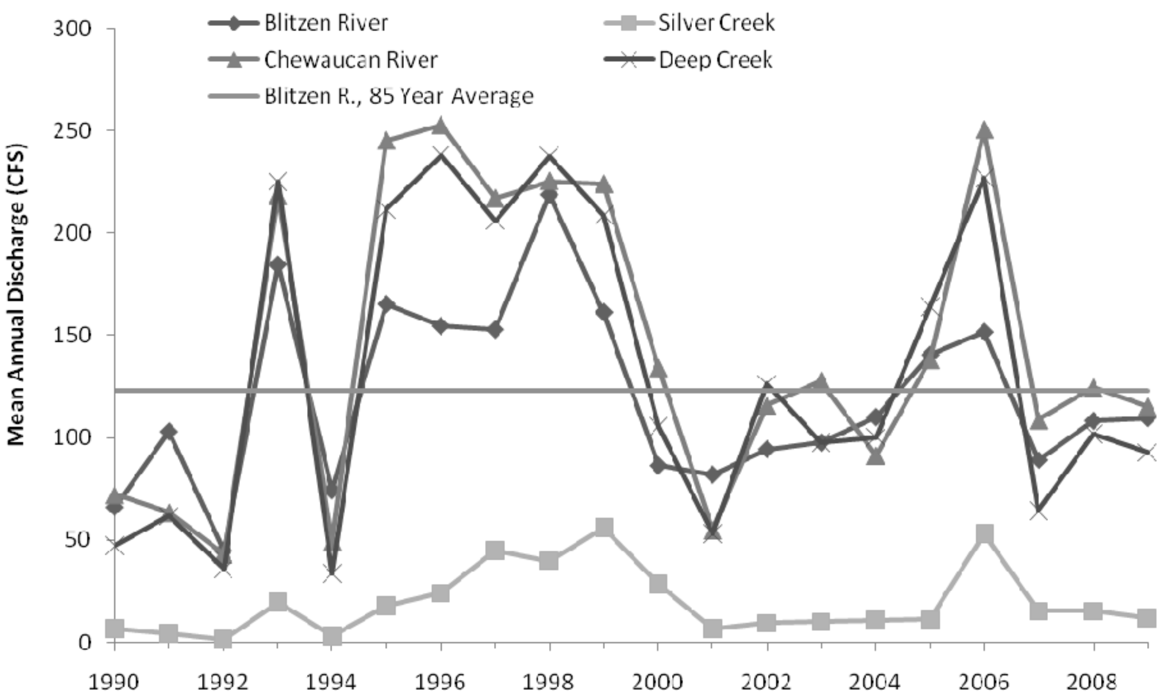


Figure 11. Mean annual discharge (cubic feet per second) based on the yearly average of mean daily flows for sampled streams in four SMUs. Fort Rock -Silver Creek, Warner Lake - Deep Creek, Chewaucan - Chewaucan River, and the Malheur Lakes - Donner und Blitzen River from January 1990 through December 2009. Data were downloaded from http://apps2.wrd.state.or.us/apps/sw/hydro_near_real_time on May 2010 for stations: 10390000, 10371500, 10384000, and 10396000. The 85 year average CFS, represented by the solid line, was calculated for the Donner und Blitzen to allow for a relative measure of higher and lower than average water years.

Repeated measures analysis suggests differences in densities at annual sites occurred during this study, particularly in the Fort Rock and Chewaucan SMUs between 2007 and 2009. Both of these SMUs had negative correlations between fish densities measured at sites one year apart, suggesting that use of these sites to detect trends may be difficult. In most SMUs, the within-site and the between-site variance were relatively equal, suggesting that sampling error and yearly variation was roughly the same as the variance between-sites. Temporal and small scale variability in fish densities at annual sites appears to be important in all SMUs. The large within-site variability may, in part, be an indication that the exact locations of these repeated sites may have varied. Setting up semi-permanent sampling boundaries at the beginning of the reach may reduce some of the within-site variability.

If annual sites reliably track yearly SMU level variation in abundance, then they could potentially be used to monitor trends at the landscape level. Further, annual sites are assumed to be as representative of all sites in the SMU as are the new sites that are randomly selected each year. In this study, annual sites differed from non-annual sites only occasionally, and these differences were within the Goose Lake and Chewaucan SMUs. No differences in densities between sites were found in any other SMU in any year, suggesting that if natural variation in population densities remains constant in future years, annual sites may be useful to detect trends in Fort Rock, Malheur Lakes, and Warner Lakes SMUs.

Our study over the last three years has provided the most current and accurate abundance estimates since the original baseline monitoring event occurred in 1999. Continued long term sampling is needed for effective management of redband trout. The Native Fish Conservation Policy (NFCP) of the Oregon Department of Fish and Wildlife (http://www.dfw.state.or.us/fish/nfcp/) calls for the development of conservation plans for each SMU. Further, the plan must address several key elements which include measureable criteria at the stratum level. The current sampling protocol for Great Basin redband trout did not consistently meet the precision targets. Further, stratum level sampling is necessary to provide consistency with the approach provided in the NFCP. Yet, increases in effort required to achieve the precision needed to assess individual populations at a relatively frequent intervals are not practical given available resources. To achieve this objective with the current level of funding necessitates restructuring the fundamental questions to be addressed by sampling, and investigation into alternative sampling designs.

Reducing the size of the sample frame could improve sampling efficiency and improve precision. Reduction in the sampling frame would allow for continued surveying of areas with greater potential for trout presence but forces a less conservative design (essentially where non-presence equals absence). Areas that were without trout or dry during this study may be potential habitat, depending on annual water availability. These areas are likely the most responsive to climate change. In each SMU, dry sites and areas without trout appear to be distributed furthest downstream from the headwaters (**APPENDICES B and C**). In higher water years, these downstream sites may have trout present. Since this study was largely during lower water years (Figure 11), we suggest leaving much of the sampling frame intact, removing only small sections of the frame that, through fairly intense sampling, have been shown to be consistently void of water or fish (and with no trout in close proximity). Small sections of the sample frames in Goose Lake, Malheur Lakes, Fort Rock, and Chewaucan SMUs have been removed using these criteria. Further reductions may be possible in the Goose Lake SMU. The numbers of continually dry sites and sites with no fish in Willow Creek in the Chewaucan SMU may warrant removal, if access to private sites is permitted in future years. For all other streams, no further reductions were apparent. However, it should be noted that reductions in the sampling frame have not appeared to reduce variation (based on CV) at the SMU level over the course of this study. Further sampling at the stratum level, particularly within the Goose Lake SMU (Dry Creek stratum), may elucidate if the reduction in sample frame helped to achieve a lower level of site-to-site variability. The additional information gathered with continued yearly sampling will determine whether any substantial reductions in the sample frame can be made.

Since stream reaches in the downstream portions of the watersheds appear to have a higher potential of being void of redband trout or being dry, it may be possible to re-stratify sample frames based on distance from headwaters. Blocking sites into upper, middle, or lower distribution, number of kilometers from mouth, or elevation may help reduce the variance in abundance estimates. This would be most helpful in basins where the trend of dry sites in lower reaches was most pronounced. To have utility, blocking would have to capture spatial patterns that are consistent across different flow regimes. The small size of some strata may also make it difficult to delineate appropriate blocks either by distance or elevation. Further, distribution of private sites in lower stream reaches (both in distance from headwaters and lower elevation) limits our ability to adequately sample these areas due to restricted access.

While both reduction of the sample frame and blocking may reduce variance, increasing sampling size would improve precision. Is it possible to increase sample size without additional funding or labor resources? The largest deficiency in the current sampling design was the lack of adequate power at the stratum level in Goose Lake and Warner Lakes SMUs. Since both stratum and SMU level information can be gathered during intensive population level sampling years, continued sampling at the stratum level in each SMU requires sampling to minimally continue at three-year intervals. A minimum of two stratum level sampling events (years) are needed to assess variance, requiring the full six years of sampling. Instead of using all of our resources to sample at the SMU level each year, adding more sites (to at least reach our target number) could be sampled within a single SMU in a single intensive year, rather than sampling in all six SMUs and one-two SMUs at the stratum level every year (Figure 12). This design would reduce the number of sites sampled each year by at least 86 sites and potentially allow for full sampling in the Catlow Valley SMU. These sites could be reallocated to increase the number of sites in each stratum during a "population intensive" sampling year, increasing precision at the stratum level, while also giving a more precise abundance estimate in that particular SMU.

The weakness of the alternative design is that differences that occur on a yearly basis at the SMU level are not captured every year (except through responses at annual sites). In this way, we fail to capture the variance associated with yearly fluctuation and may miss fluctuations in populations over intermediate time spans (Urquhart and Kincaid 1999). Our current design fails to address yearly fluctuations at the stratum level, since sampling is performed on a three year interval. To partly address this issue, continuation of sampling in each SMU at annual sites would allow for a measure of the yearly variability at the site level. Since annual sites were spatially balanced and randomly allocated within each stratum, differences found between years could potentially be used as a relative measure of year to year variation at the SMU level. However, differences in sites densities between annual and non-annual sites in the Chewaucan and Goose Lake SMUs make relying on annual sites for large-scale inference problematic. Using annual sites to extrapolate estimates to an entire stratum or SMU, while being logistically more feasible, assumes that these sites are consistently representative of the area of inference. Future years of sampling should confirm the adequacy of selected annual sites as reliable representations of larger scale patterns.

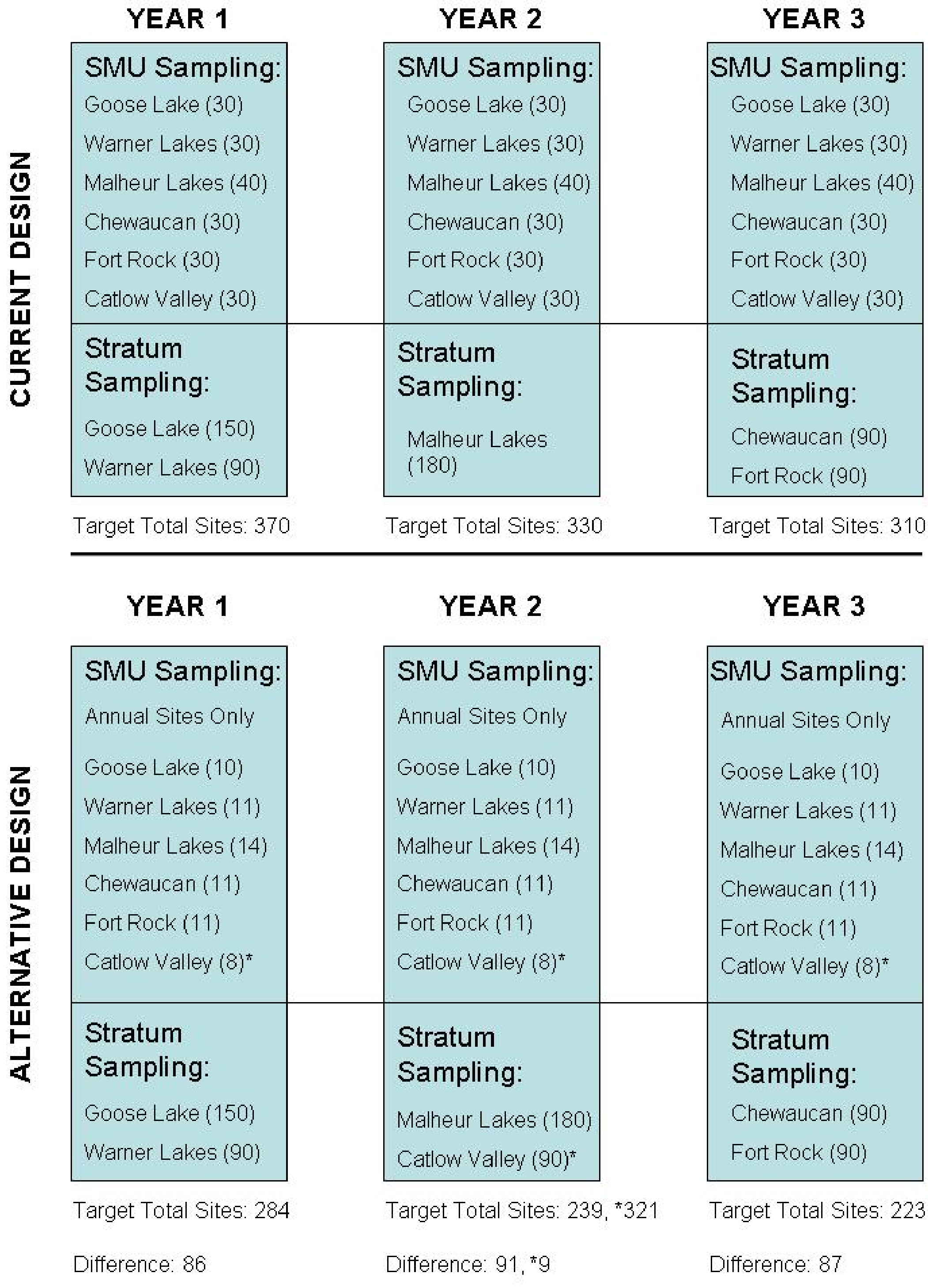


Figure 12. Comparisons between the current and an alternative sampling design. The numbers in parentheses following the specific SMU represent the target numbers of sample sites. Total target sites are the numbers proposed when totaling all individual SMUs, sampled at both the SMU and stratum level for that proposed year. The difference was calculated by subtracting the total number of target sites in the alternative design from the number in the current design. The asterisk represents the addition of the Catlow Valley SMU to the overall design. Although this SMU was originally a SMU of focus, full sampling has yet to be realized.

Although SMU level abundance of age-1+ redband trout appears robust, this does not mean that they are secure. Populations that are endangered by low abundance, spatial isolation, altered life histories or that are vulnerable to habitat alterations may not be detectable, despite having overall abundances at the SMU level that remain relatively constant. The accessibility to a diversity of high quality habitat allows populations to express multiple life history strategies including migratory ability. Lack of stable habitat tends to prevent persistence, increase population isolation, and increase extinction rates (Smith 1981, Smith et al. 2002, Currens et al. 2009). However, the persistence of robust landscape-scale populations of Great Basin redband trout suggests some level of stable habitat exists in part of their range. The ability of populations to express both resident and migratory life history strategies creates a higher potential to persist through low water years, colonize new habitats, and mix genetically with other populations. Genetic studies of redband trout show that large river systems in the upper Sacramento, Klamath and Columbia rivers provided long-term sources of stable and diverse habitat allowing trout to persist and evolve across the landscape (Currens et al. 2009). These sources, along with migratory movement in pluvial lake basins, were likely sources of ecological and evolutionary diversity, rather than isolated independent habitats (Currens et al. 2009). With reduced water flows, increasing barriers to movement, and presence of non-native fish species, the isolation of populations potentially continues to reduce genetic dispersion and may alter the fitness of distinct populations. Distinct genetic races appear across our study region with individuals in the Goose Lake, Warner Lakes, and Chewaucan SMUs being part of the Sacramento race *O. mykiss stonei.* Malheur Lakes and Fort Rock redband trout were most closely related to the Columbia River race (with Malheur Lakes associated with *O. mykiss gairdneri)* while the Catlow Valley basin redband trout remains unresolved (Currens et al 2009). Genetic samples collected from individuals within these SMUs during the course of our study may elucidate if any distinct populations exist to better inform the definition of conservation units and may offer information regarding the overall genetic diversity and health within these populations.

While differences in habitat are likely to contribute to variability in fish sampling at a site-by-site scale, the overall yearly differences in the abundance and distribution of redband trout may be largely attributed to annual fluctuations of in-stream flows. Streams become uninhabitable during drought years, when they dry up or do not offer thermal refugia (Ebersole et al 2001). These same streams are recolonized during wet cycles. Suitable habitat and sufficient water flows are likely indicators of redband trout presence in these basins (Dambacher 2001, Zoellick et al. 2005). Correlation of overall stream temperatures and distance from headwaters (Fausch et al 2002), riparian canopy (Li et al. 1994), and local stream temperatures (Ebersole et al 2001), as well as quality of physical habitat at a site, all appear to affect fish abundance and distribution (Zoellick and Cade 2006). Although other factors surely influence year-to-year and site-to-site variability, few studies have found clear relationships with measured habitat variables that can accurately predict trout abundance in stream environments (Dambacher et al 2009, Fausch et al 1988). In the 1999 study, these relationships were even more elusive, where habitat models showed little correspondence with measured variables and often these variables had opposite effects in different models (Dambacher et al. 2009). Variables such as body size (Dunham and Vinyard 1997), elevation (Dunham 1999), temperature (Ebersole et al. 2001), and the presence of nonnative brook trout (Dunham et al. 2003), have been found to explain large amounts of the variability in abundance of native desert trout species, yet these relationships remain unclear in redband trout systems in the Oregon portion of the Great Basin. Data collected during 2007-2009 will be used in future analyses to examine the relationships between fish abundance and distribution and measured habitat variables (Cooperative project with USGS).

Establishing acceptable benchmarks or criteria for evaluating habitats and status of populations is critical to the development of a useable conservation plan. To do this, the first step is to provide unbiased assessment of the abundance and distribution of age-1+ redband trout throughout the sampling frame. Given the variable nature of these populations, it is difficult to identify measurable criteria necessary to evaluate the status of these populations. Since it is unlikely that we will have all information needed to complete our assessment, it is important to distinguish between the data we would like to have and what is practical to obtain. With the data collected during this study, we have provided information on primary biological attributes including distribution of populations within SMUs, age-1+ fish abundance for strata, inferred population connectivity, and considered the likelihood of SMU level species persistence, at least in the short term. For wild trout to persist in these basins, they face many challenges. Redband trout productivity is likely limited by flow diversions, migration barriers, degraded riparian habitat, competition with exotic salmonids, and climate regime. Protection of current populations requires increasing the size and extent of populations, maintaining genetic and life history diversity, increasing connectivity, minimizing anthropomorphic stressors (i.e. irrigation withdrawals, exotic salmonids), and improving adaptive management (Williams et al. 2007). Maintaining robust populations at the SMU level is the first step in protecting this valuable resource and ensuring future fishing opportunities.

ACKNOWLEDGEMENTS

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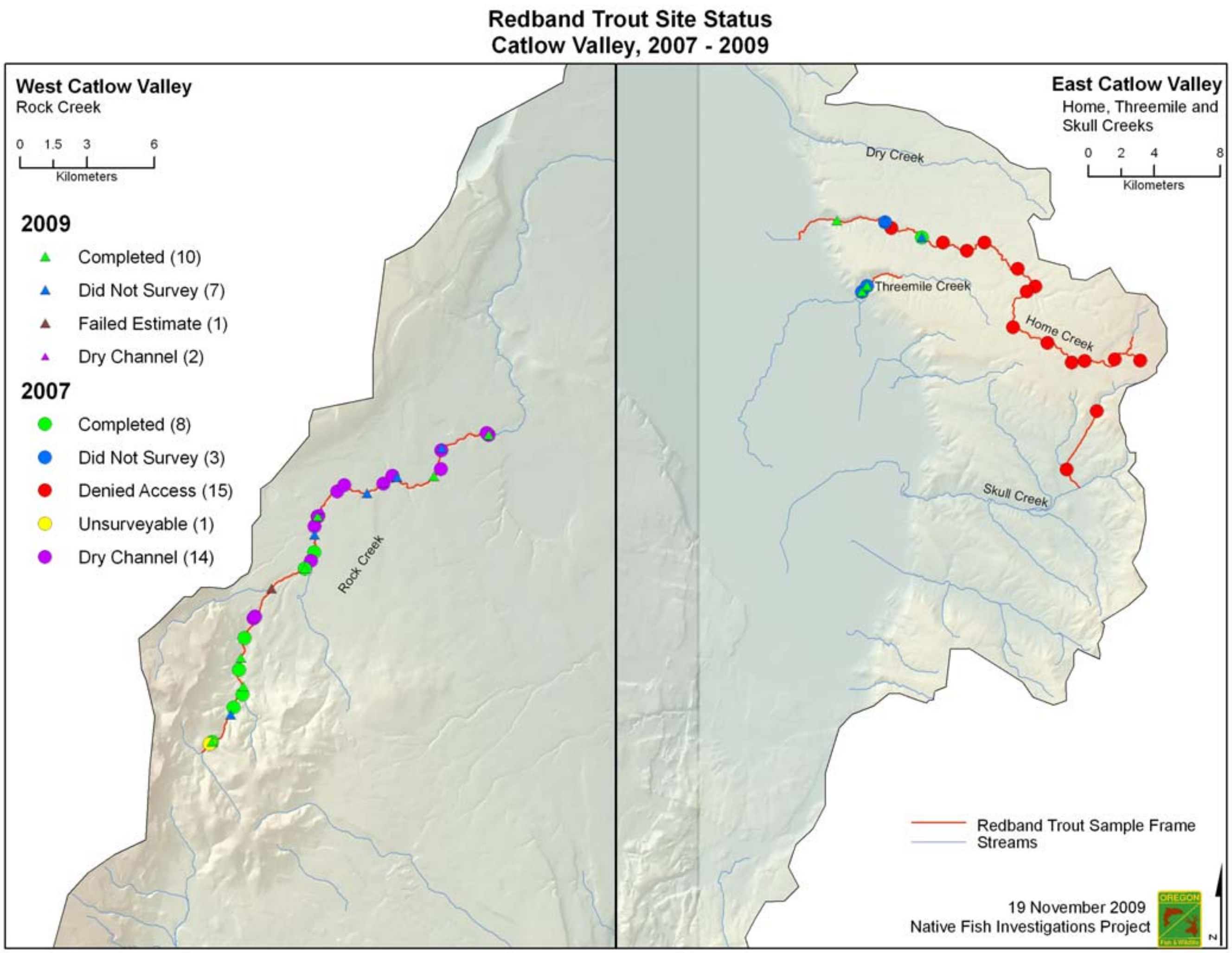
APPENDIX A. Constituent local populations and allocation of sample sites among Species Management Units (SMU) and strata used in this study.

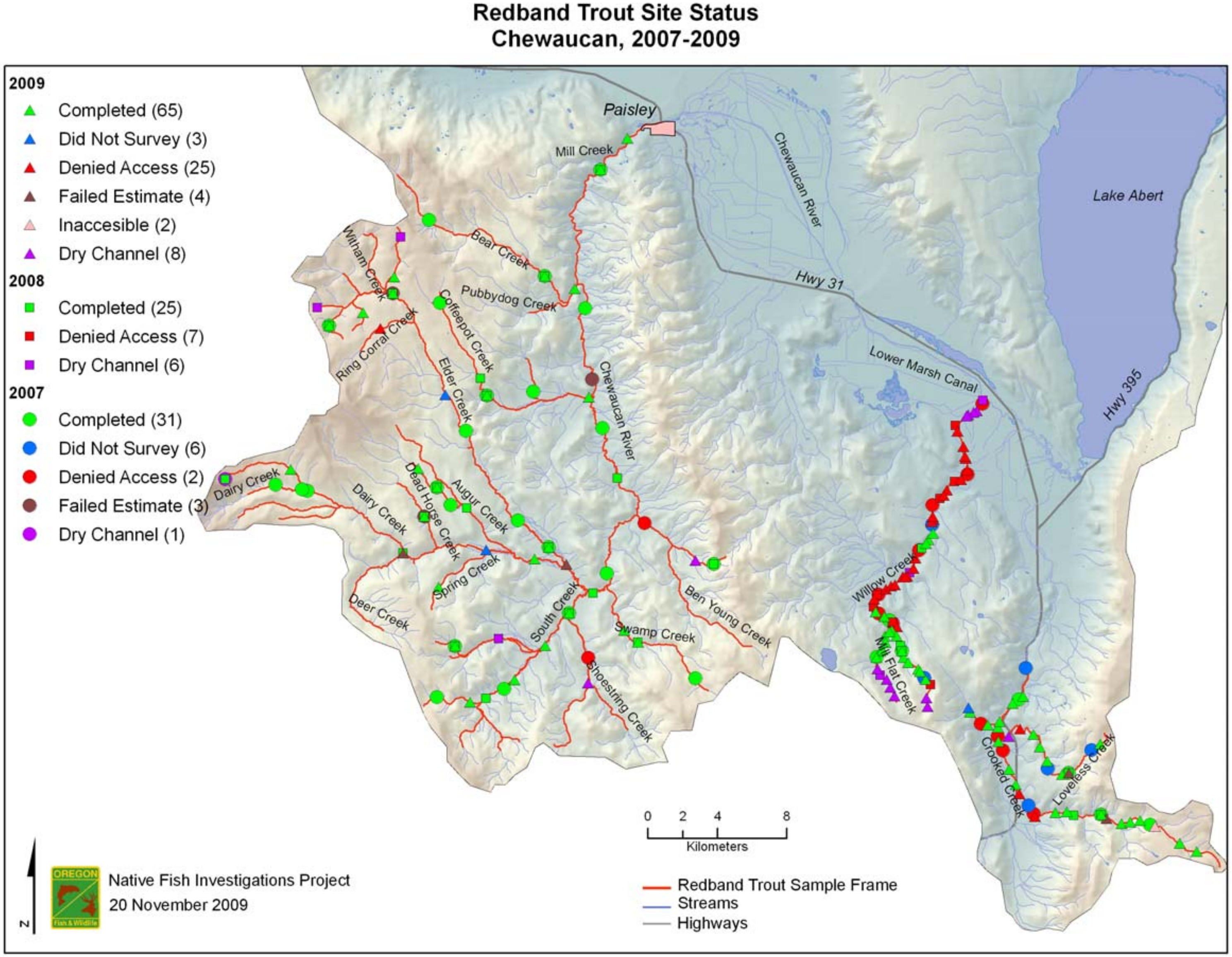
SMU	Stratum	Constituent Populations	Sample Frame Length (Km)	SMU level sampling		Stratum level sampling	
				Number of sites	Km/site	Number of sites	Km/site
Catlow Valley	Home	Home	39	2	19.4	2	19.4
	Rock	Rock	26	15	1.8	15	1.8
	Skull/3mile	Skull/3mile	10	2	5	2	5
Chewaucan	Chewaucan	Chewaucan	283	24	11.8	30	9.4
	Crooked	Crooked	38	3	12.5	30	1.3
	Willow	Willow	32	3	10.8	30	1.1
Fort Rock	Bridge	Bridge	31	7	4.4	30	1
	Buck	Buck	38	9	4.2	30	1.3
	Silver	Silver	57	14	4.1	30	1.9
Goose Lake	Drews	Upper Drews Lower Drews	87	8	10.9	30	2.9
	Dry	Dry, Fall	22	2	11.2	20	1.1
	Eastside	Deadman, Crane, Cogswell, Tandy, Kelly	35	3	11.8	30	1.2
	Thomas-Bauers	Thomas-Bauers	129	12	10.7	40	3.2
	West Goose	Muddy, Cottonwood, Antelope	56	5	11.1	30	1.9
Malheur Lakes	Blitzen	Blitzen	214	7	30.6	30	7.1
	East Burns	Poison, Prater, Cow Coffeepot, Rattlesnake	78	3	26.1	30	2.6
	McCoy	McCoy	193	6	32.1	30	6.4
	Riddle	Riddle	84	3	28	30	2.8
	Silver	Silver	186	6	31	30	6.2
	Silvies	Silvies	468	15	31.2	30	15.6
Warner Lakes	Deep	Lower Deep, Upper Deep	174	17	10.2	30	5.8
	Honey	Honey	94	9	10.4	30	3.1
	Twentymile	Twentymile	46	4	11.4	30	1.5
Total			2,420	179	13.5	619	3.9

Appendix B. Site status maps for each SMU. Each year is represented by a distinct shape. The sample frame used in these maps was derived from the original frame used in 2007.

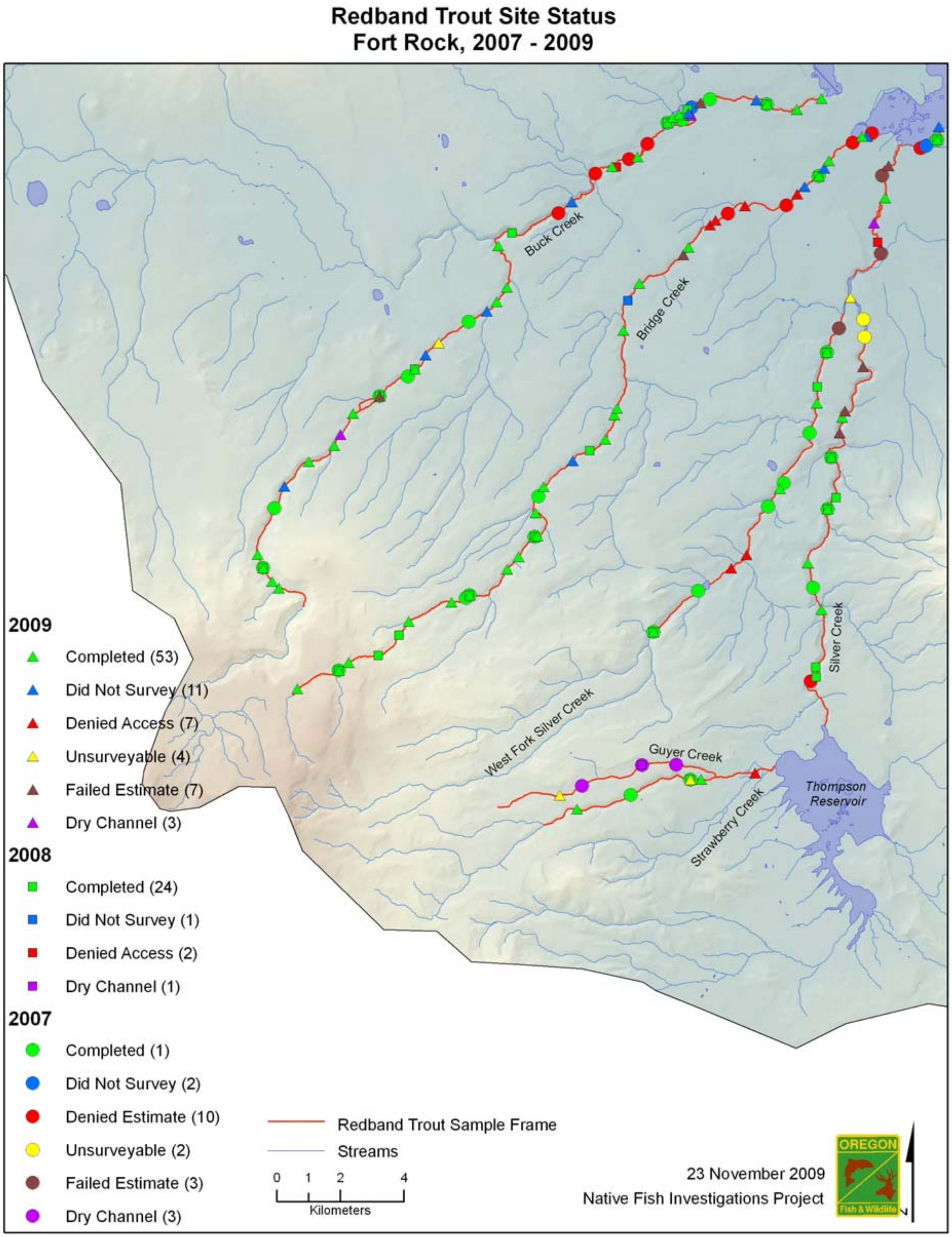
A) Catlow Valley



B) Chewaucan


Redband Trout Site Status
Chewaucan, 2007-2009
2009
Completed (65)
Did Not Survey (3)
Denied Access (25)
Failed Estimate (4)
Inaccesible (2)
Dry Channel (8)
2008
Completed (25)
Denied Access (7)
Dry Channel (6)
2007
Completed (31)
Did Not Survey (6)
Denied Access (2)
Failed Estimate (3)
Dry Channel (1)
Paisley
Mill Creek
Chewaucan River
Lake Abert
Bear Creek
Witham Creek
Hwy 31
Pubbydog Creek
Coffeepot Creek
Ring Corral Creek
Elder Creek
Lower Marsh Canal
Hwy 395
Dairy Creek
Dead Horse Creek
Augur Creek
Spring Creek
Deer Creek
South Creek
Swamp Creek
Ben Young Creek
Willow Creek
Mill Flat Creek
Shoestring Creek
Crooked Creek
Loveless Creek
0 2 4 8
Kilometers
N
Native Fish Investigations Project
20 November 2009
Redband Trout Sample Frame
Streams
Highways

C) Fort Rock



Redband Trout Site Status
Fort Rock, 2007 - 2009
Buck Creek
Bridge Creek
Silver Creek
West Fork Silver Creek
Guyer Creek
Strawberry Creek
Thompson Reservoir
2009
Completed (53)
Did Not Survey (11)
Denied Access (7)
Unsurveyable (4)
Failed Estimate (7)
Dry Channel (3)
2008
Completed (24)
Did Not Survey (1)
Denied Access (2)
Dry Channel (1)
2007
Completed (1)
Did Not Survey (2)
Denied Estimate (10)
Unsurveyable (2)
Failed Estimate (3)
Dry Channel (3)
Redband Trout Sample Frame
Streams
0 1 2 4
Kilometers
23 November 2009
Native Fish Investigations Project
OREGON
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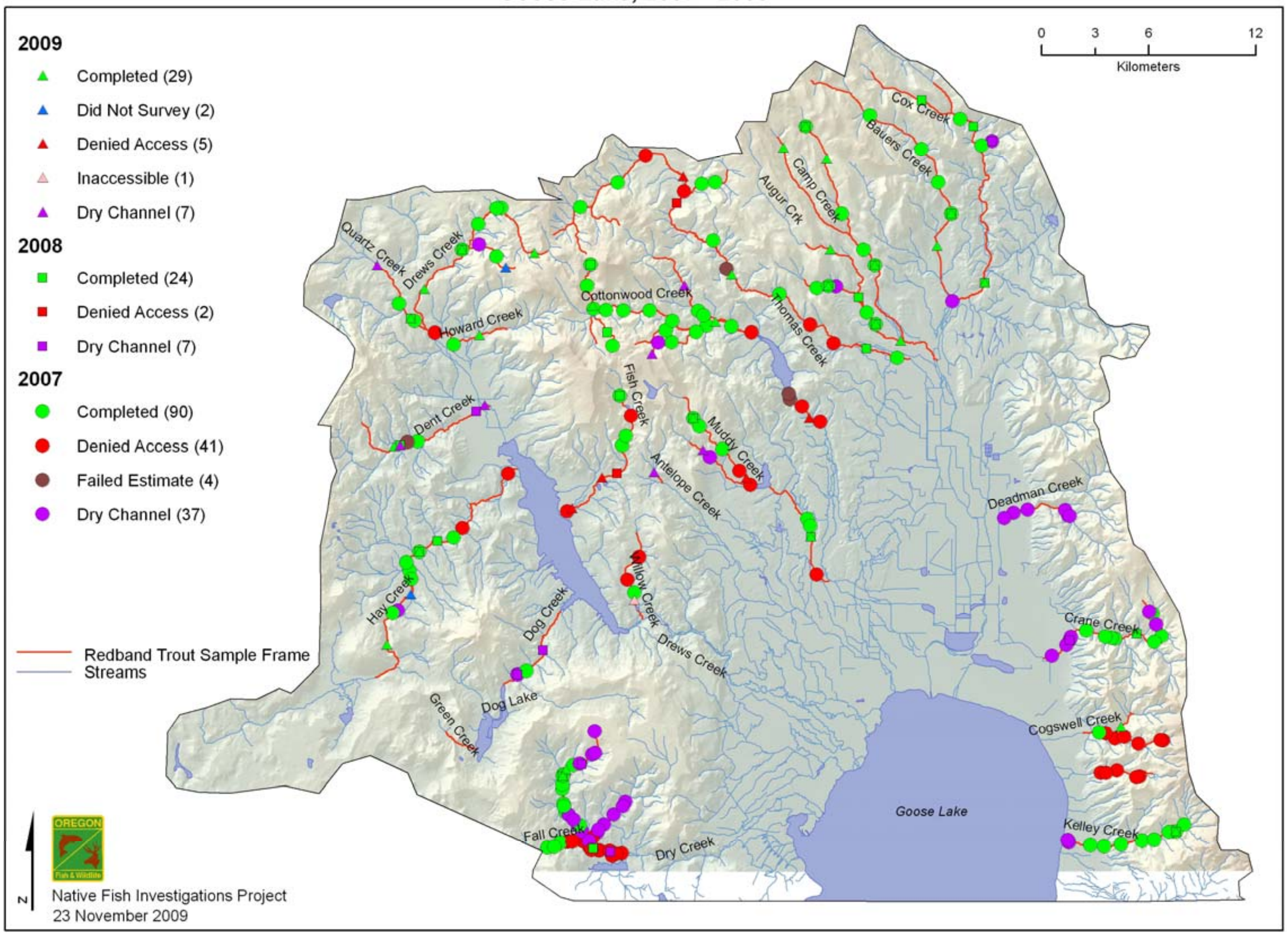
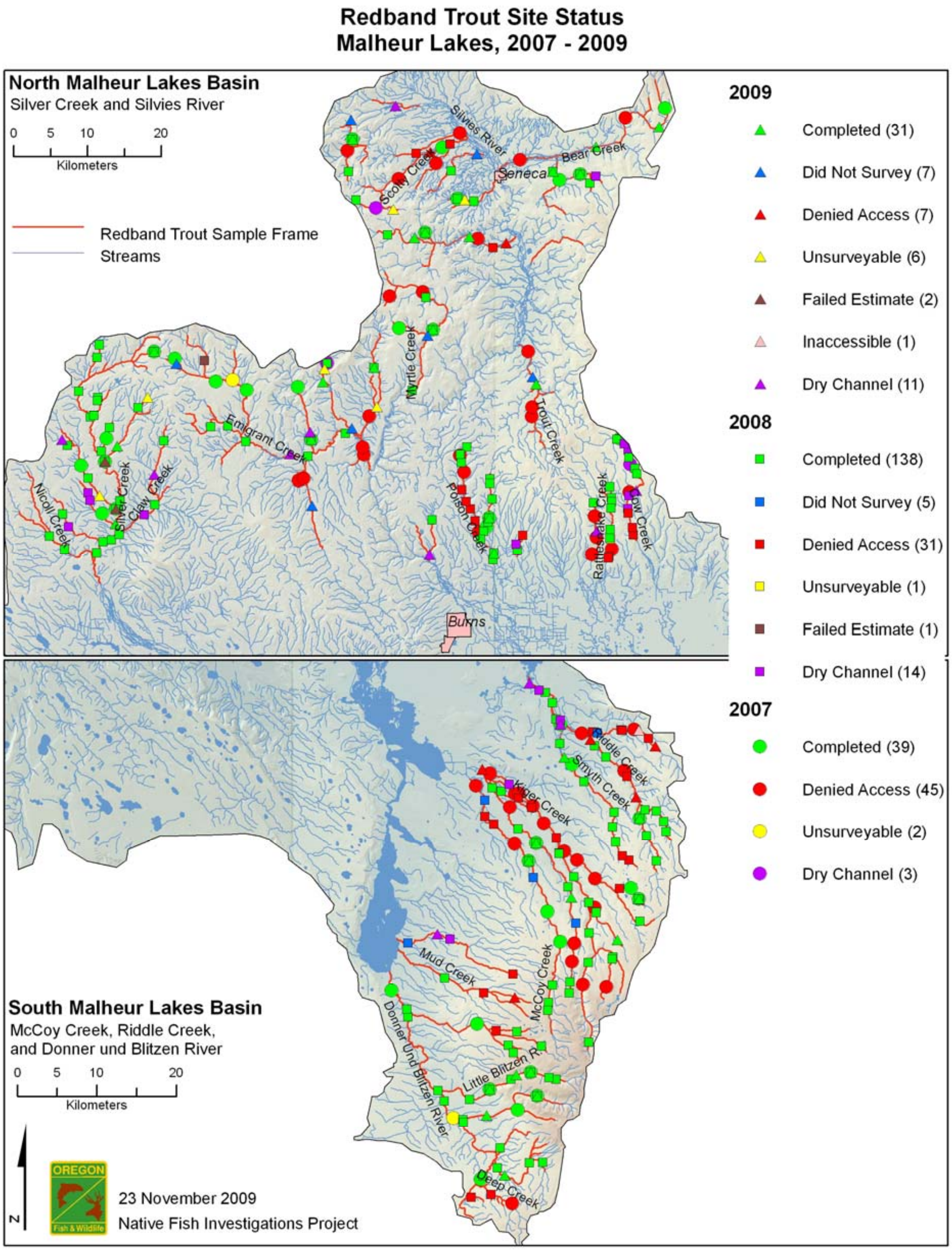
D) Goose Lake

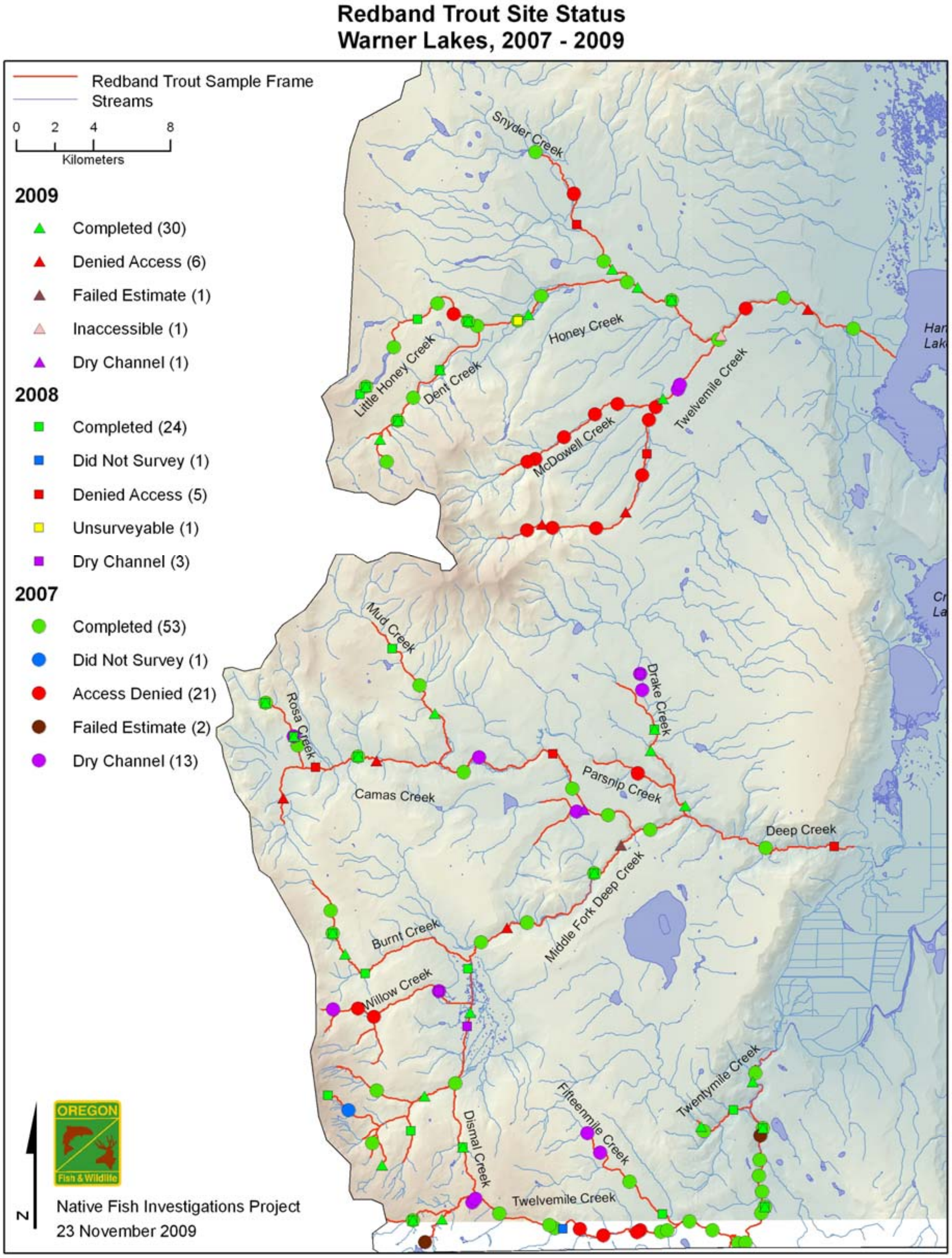
Redband Trout Site Status
Goose Lake, 2007 - 2009



E) Malheur Lakes

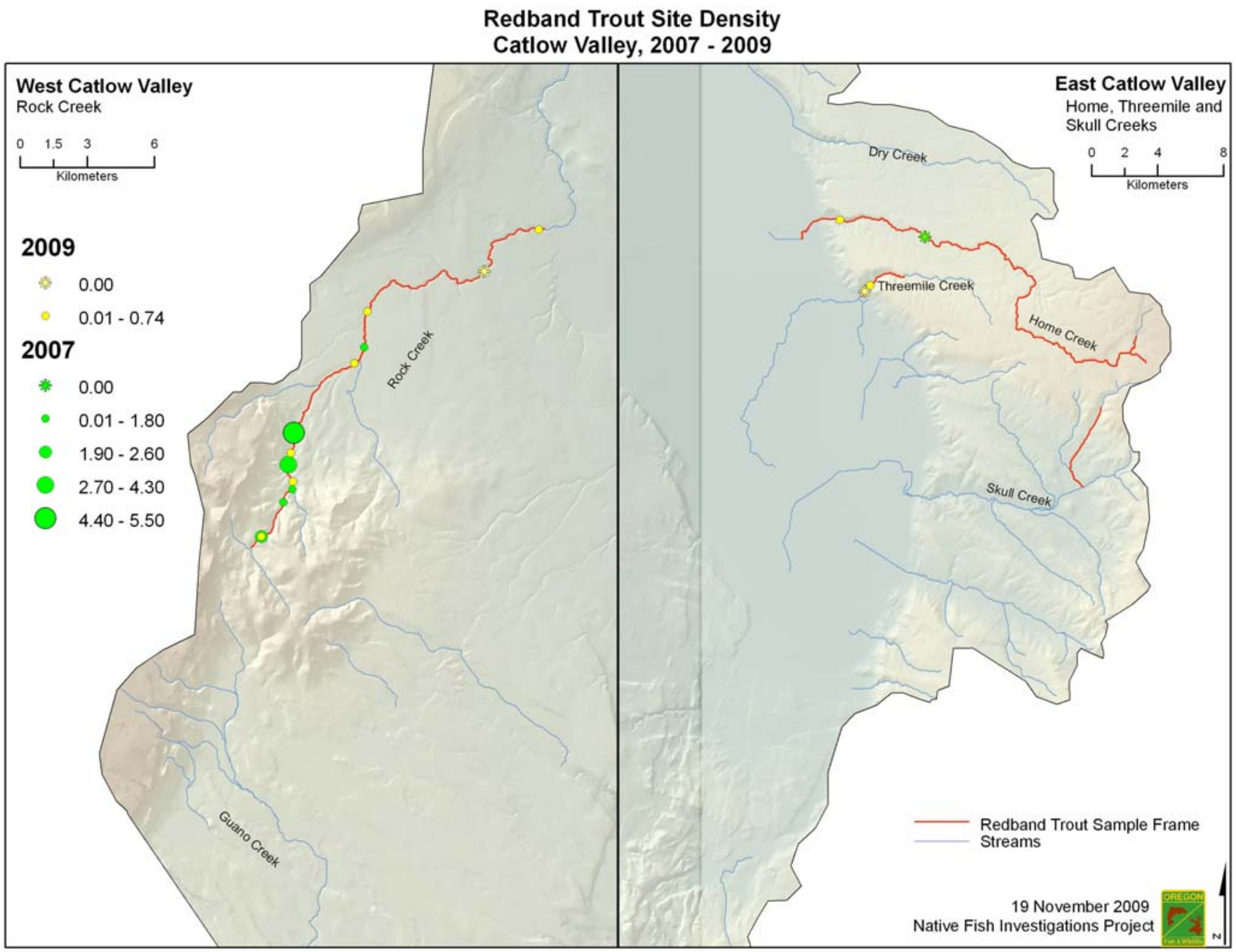


F) Warner Lakes

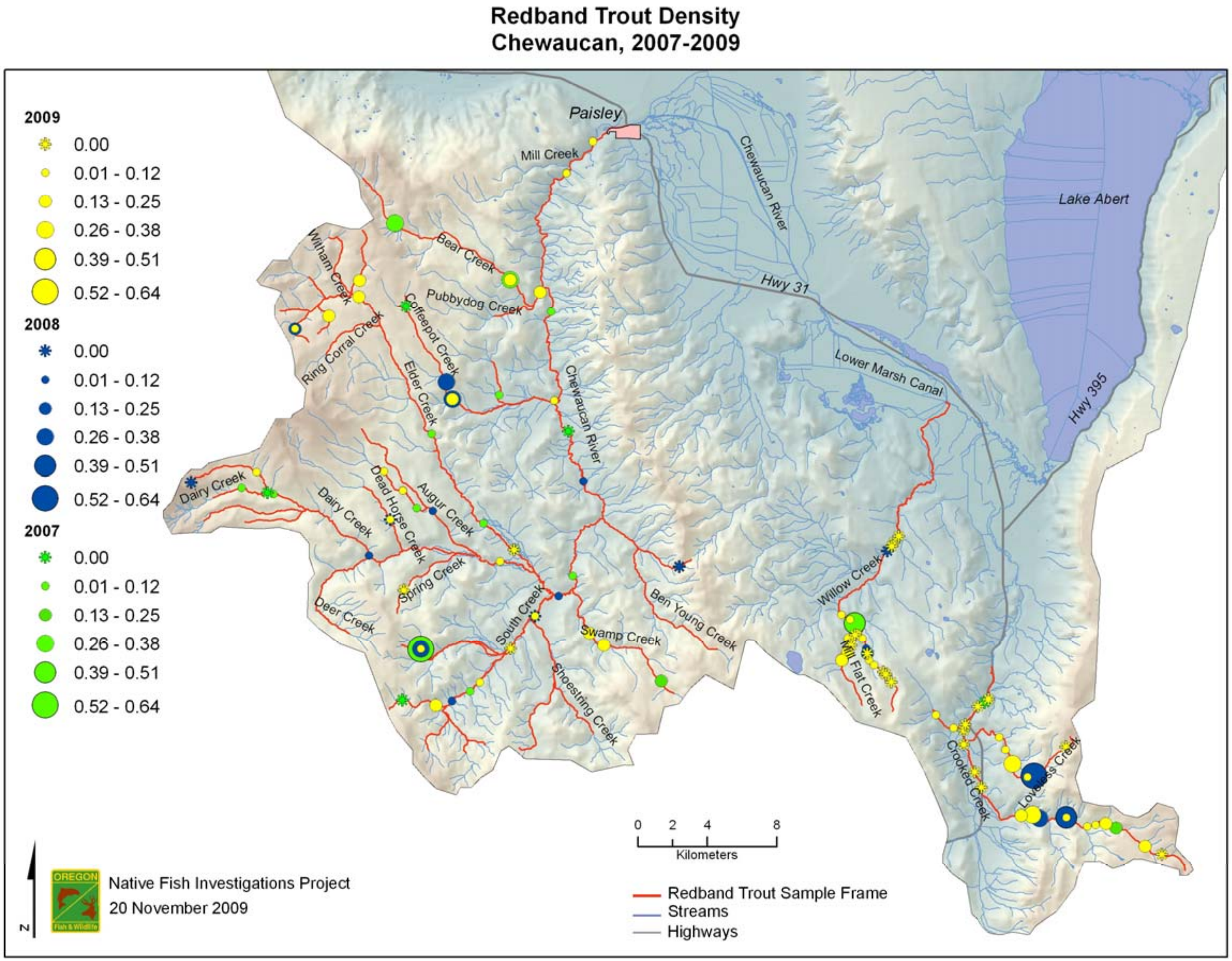


Appendix C. Site density maps (fish/m^2) for each SMU. The sample frame used in these maps was derived from the original frame used in 2007.

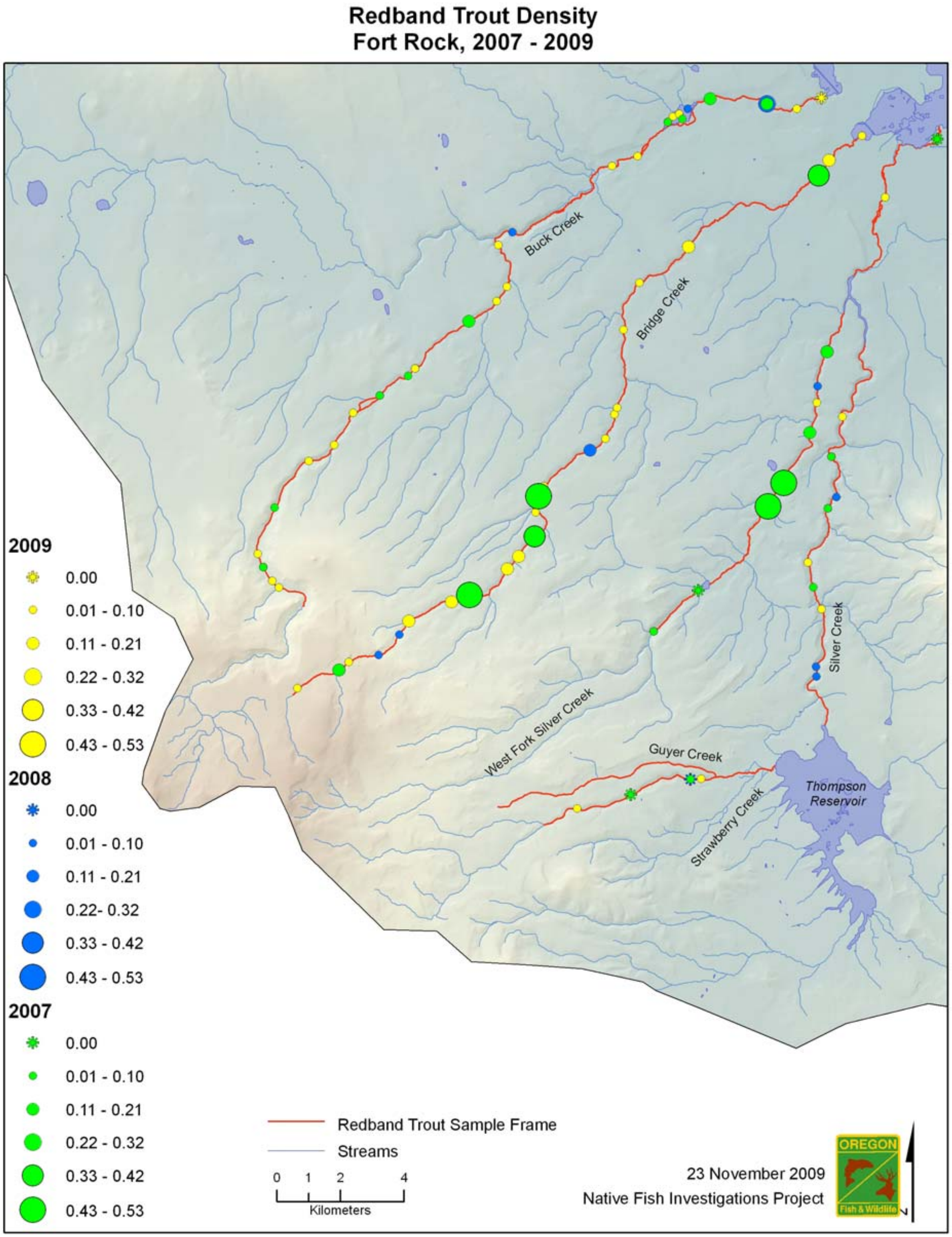
A) Catlow



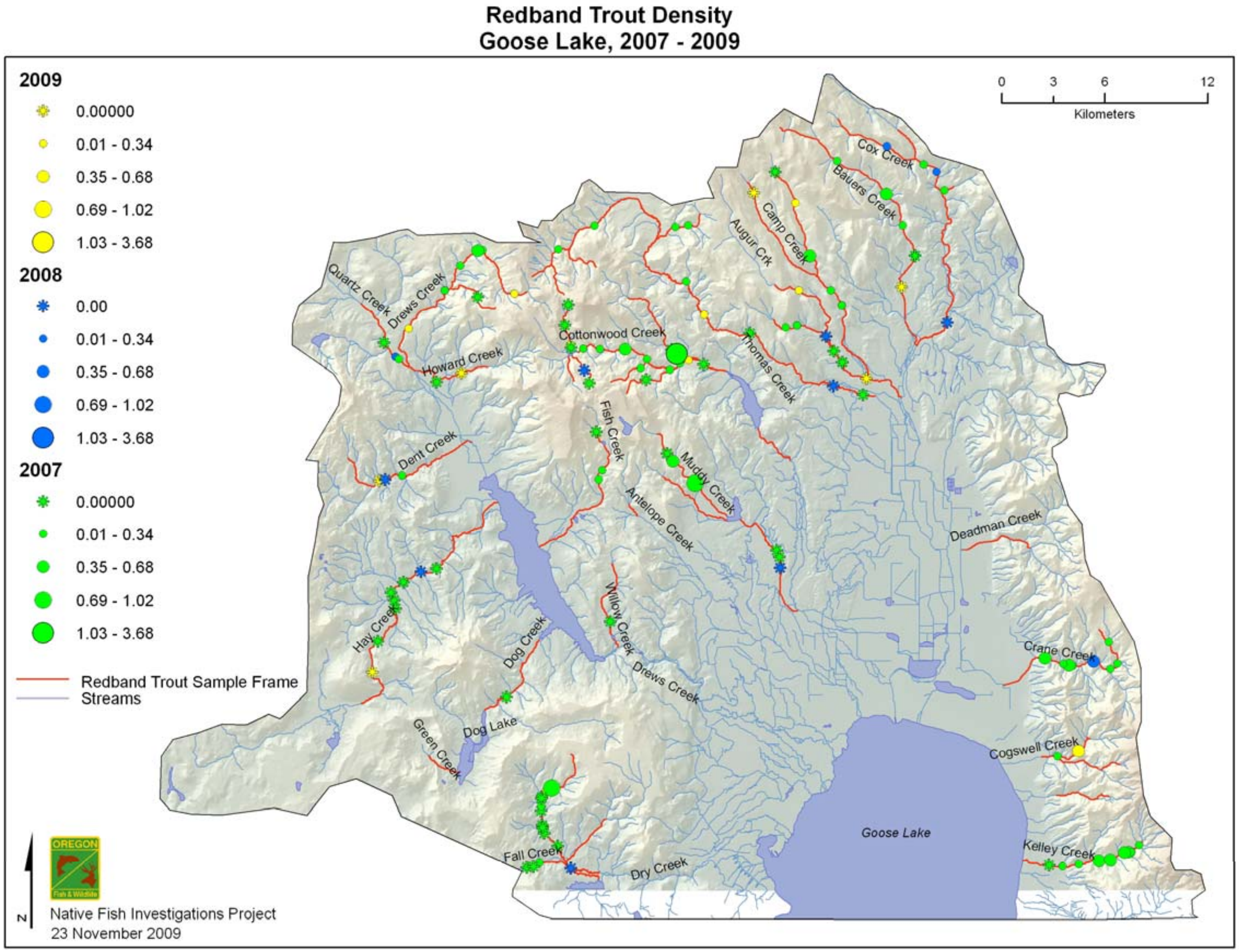
B) Chewaucan


Redband Trout Density
Chewaucan, 2007-2009
2009
0.00
0.01 - 0.12
0.13 - 0.25
0.26 - 0.38
0.39 - 0.51
0.52 - 0.64
2008
0.00
0.01 - 0.12
0.13 - 0.25
0.26 - 0.38
0.39 - 0.51
0.52 - 0.64
2007
0.00
0.01 - 0.12
0.13 - 0.25
0.26 - 0.38
0.39 - 0.51
0.52 - 0.64
Paisley
Mill Creek
Chewaucan River
Lake Abert
Hwy 31
Hwy 395
Lower Marsh Canal
Bear Creek
Wilham Creek
Pubbydog Creek
Coffeepot Creek
Ring Corral Creek
Elder Creek
Dairy Creek
Dead Horse Creek
Augur Creek
Spring Creek
Deer Creek
South Creek
Swamp Creek
Shoestring Creek
Ben Young Creek
Willow Creek
Mill Flat Creek
Crooked Creek
0 2 4 8
Kilometers
N
OREGON
Native Fish Investigations Project
20 November 2009
Redband Trout Sample Frame
Streams
Highways

C) Fort Rock


Redband Trout Density
Fort Rock, 2007 - 2009
Buck Creek
Bridge Creek
Silver Creek
West Fork Silver Creek
Guyer Creek
Strawberry Creek
Thompson Reservoir
2009
0.00
0.01 - 0.10
0.11 - 0.21
0.22 - 0.32
0.33 - 0.42
0.43 - 0.53
2008
0.00
0.01 - 0.10
0.11 - 0.21
0.22- 0.32
0.33 - 0.42
0.43 - 0.53
2007
0.00
0.01 - 0.10
0.11 - 0.21
0.22 - 0.32
0.33 - 0.42
0.43 - 0.53
Redband Trout Sample Frame
Streams
0 1 2 4
Kilometers
23 November 2009
Native Fish Investigations Project
OREGON
Fish & Wildlife

D) Goose Lake



Redband Trout Density
Goose Lake, 2007 - 2009
2009
0.00000
0.01 - 0.34
0.35 - 0.68
0.69 - 1.02
1.03 - 3.68
2008
0.00
0.01 - 0.34
0.35 - 0.68
0.69 - 1.02
1.03 - 3.68
2007
0.00000
0.01 - 0.34
0.35 - 0.68
0.69 - 1.02
1.03 - 3.68
Redband Trout Sample Frame
Streams
0
3
6
12
Kilometers
Cox Creek
Bauers Creek
Camp Creek
Augur Crk
Quartz Creek
Drews Creek
Howard Creek
Cottonwood Creek
Thomas Creek
Fish Creek
Dent Creek
Muddy Creek
Antelope Creek
Deadman Creek
Willow Creek
Hay Creek
Dog Creek
Drews Creek
Crane Creek
Green Creek
Dog Lake
Cogswell Creek
Goose Lake
Kelley Creek
Fall Creek
Dry Creek
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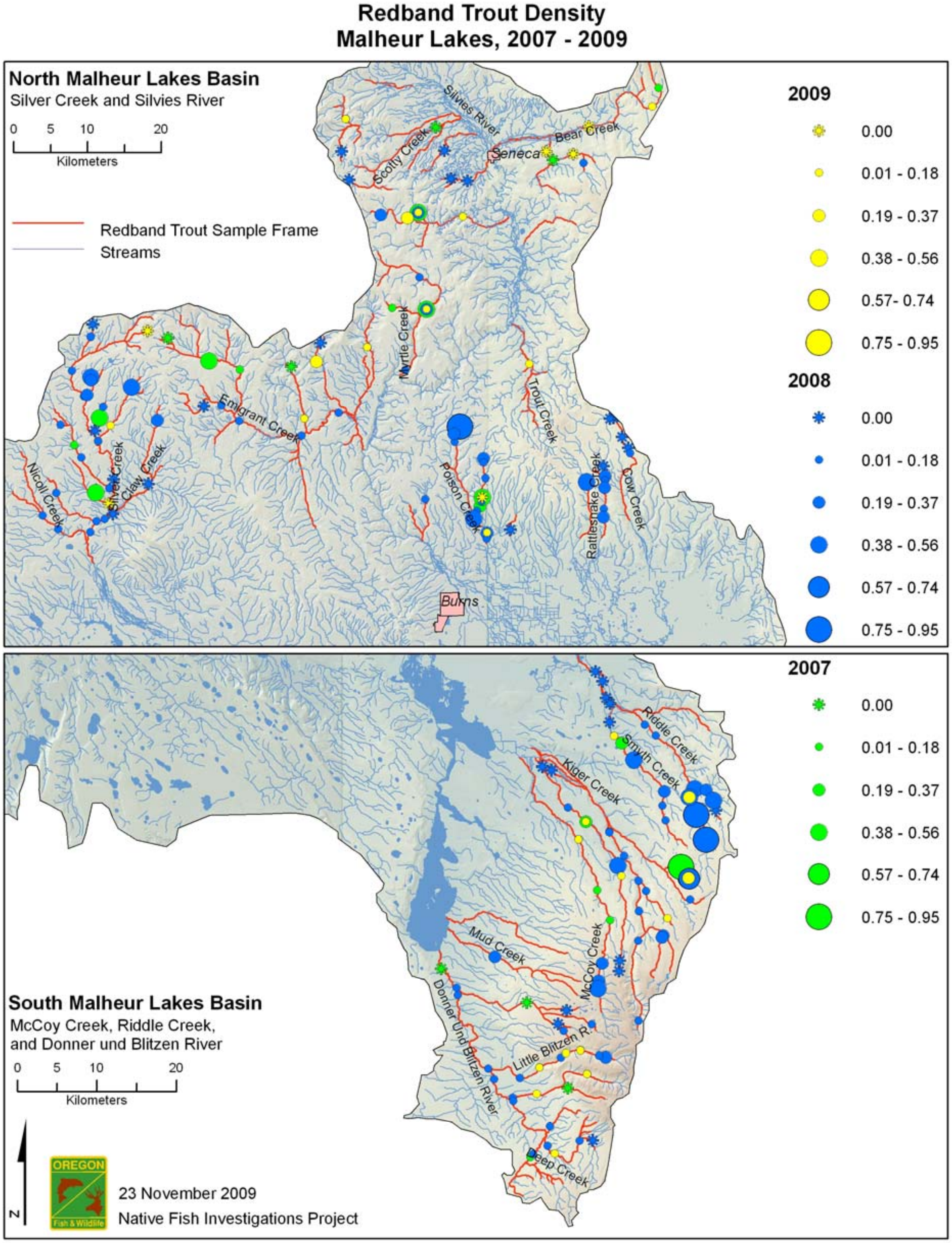
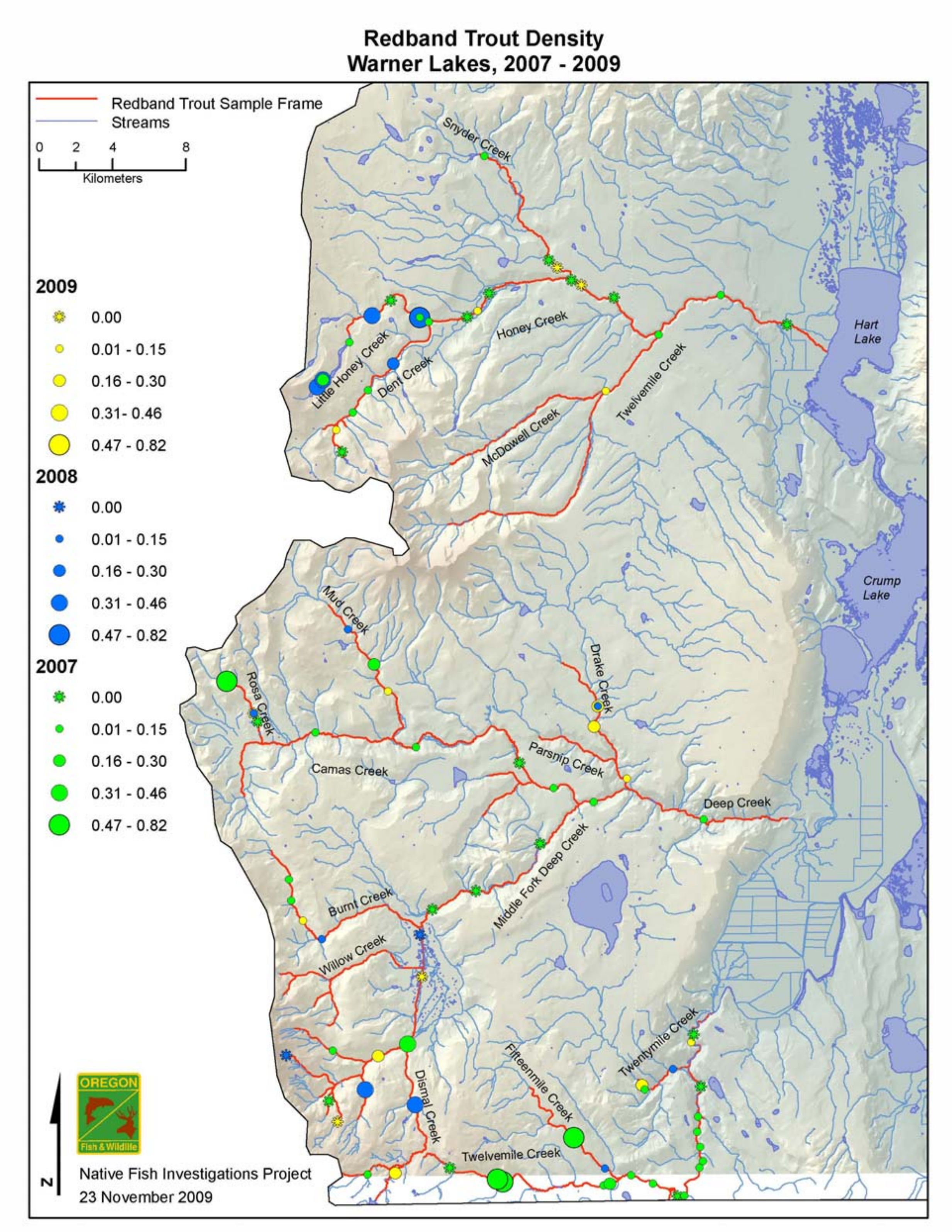
OREGON
Native Fish Investigations Project
23 November 2009

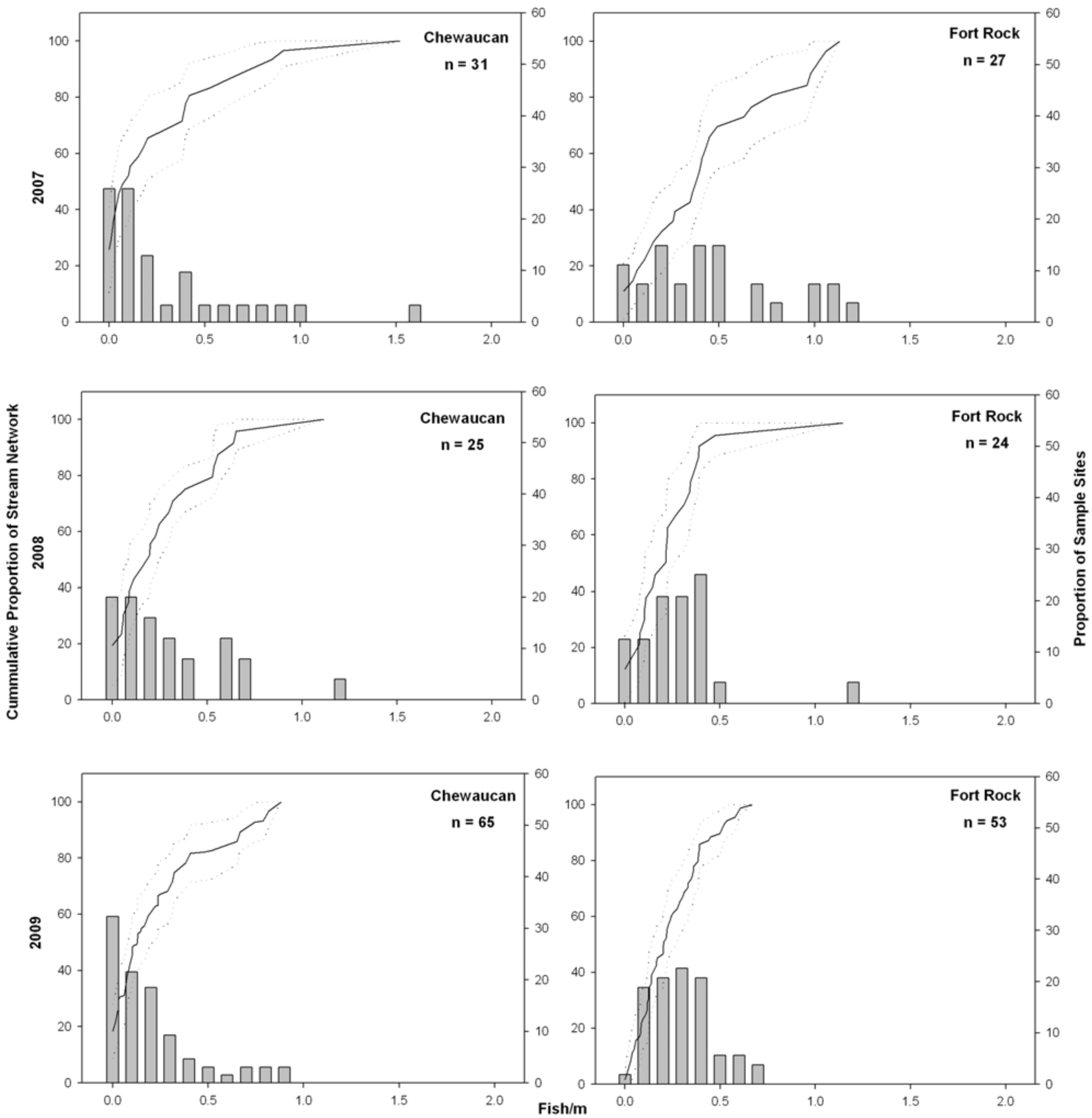
E) Malheur Lakes

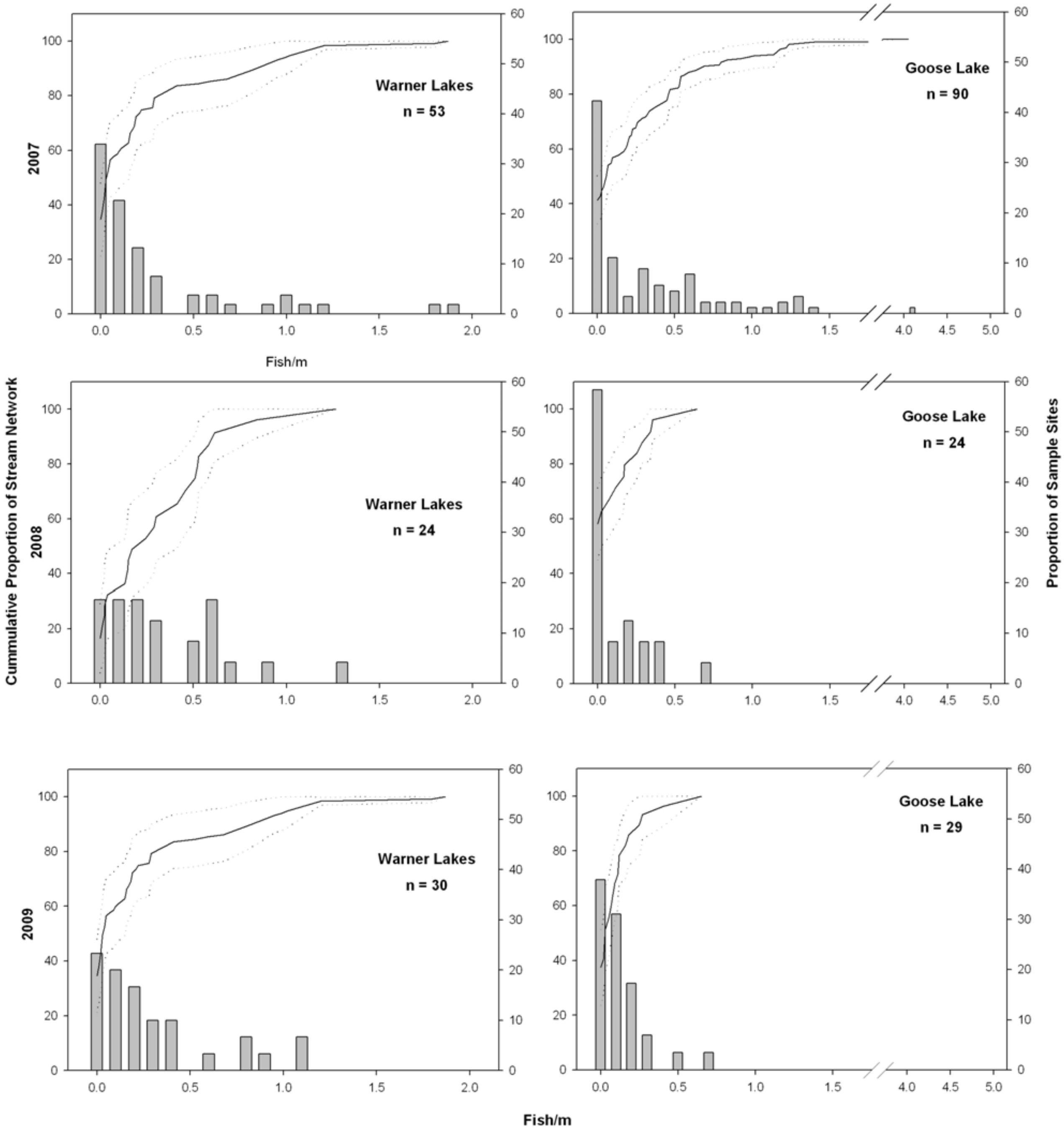


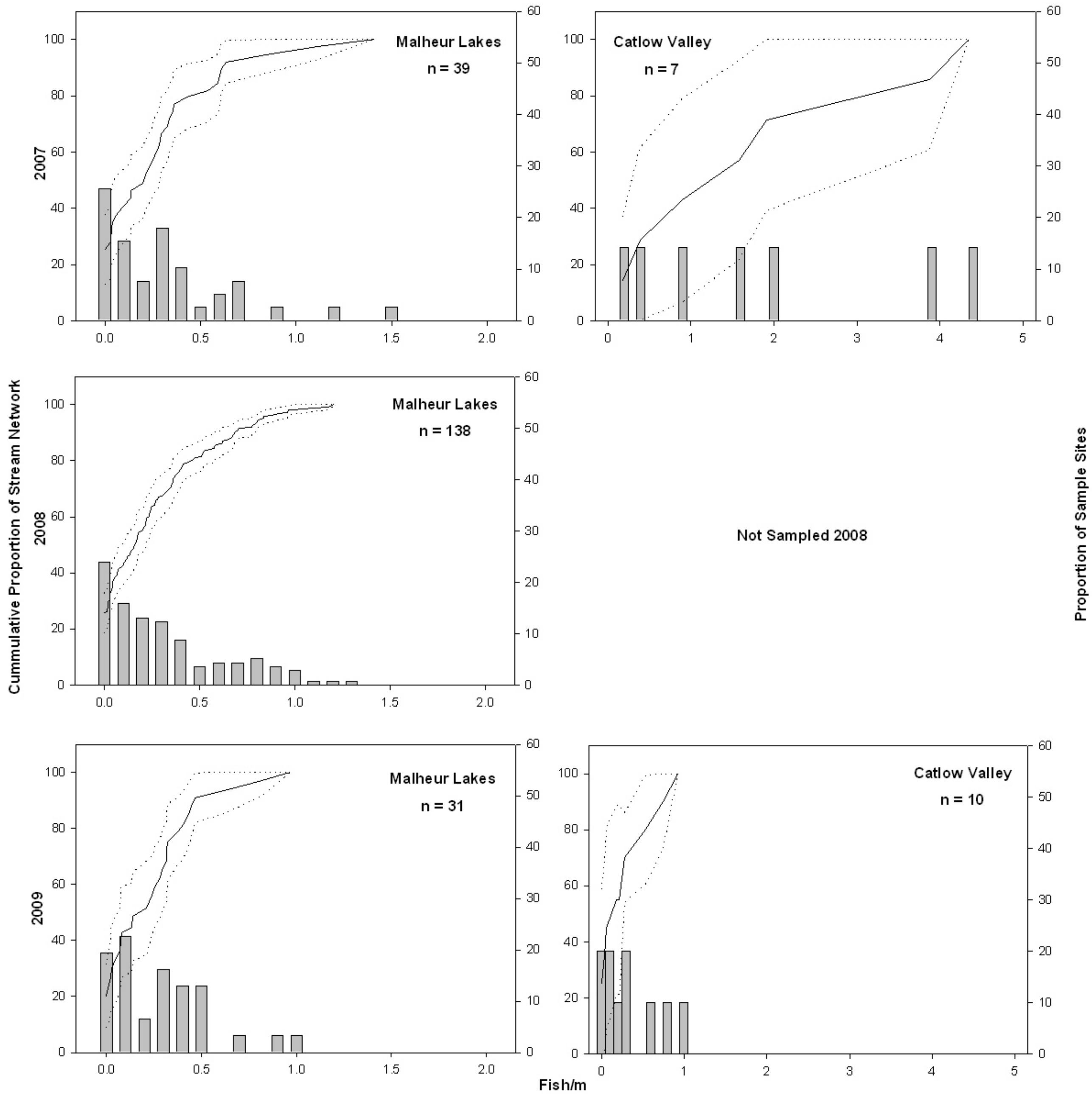
F) Warner Lakes



Appendix D. Frequencies of redband trout densities (fish/m) estimated at sample sites within each SMU during each year of the study. Each panel depicts a cumulative distribution function with an associated 95% confidence interval across the sample frame (left axis) and a frequency distribution of densities for sample sites (vertical bars on the right axis).




Warner Lakes
n = 53
Goose Lake
n = 90
Warner Lakes
n = 24
Goose Lake
n = 24
Warner Lakes
n = 30
Goose Lake
n = 29
2007
2008
2009
Cummulative Proportion of Stream Network
Proportion of Sample Sites
Fish/m
Fish/m



Cummulative Proportion of Stream Network
Proportion of Sample Sites
2007
Malheur Lakes
n = 39
Catlow Valley
n = 7
2008
Malheur Lakes
n = 138
Not Sampled 2008
2009
Malheur Lakes
n = 31
Catlow Valley
n = 10
Fish/m

OREGON

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