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USE OF DEPLETION ELECTROFISHING AND A GENERALIZED RANDOM-TESSELLATION STRATIFIED DESIGN TO ESTIMATE DENSITY AND ABUNDANCE OF REDBAND TROUT IN THE NORTHERN GREAT BASIN

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USE OF DEPLETION ELECTROFISHING AND A GENERALIZED RANDOM-TESSELLATION STRATIFIED DESIGN TO ESTIMATE DENSITY AND ABUNDANCE OF REDBAND TROUT IN THE NORTHERN GREAT BASIN

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Abstract – Redband trout (*Oncorhynchus mykiss newberrii*) in the Great Basin occupy a landscape characterized by spatial and temporal variability in environmental conditions that likely influences their abundance. Developing long-term monitoring protocols and effective conservation plans will require an understanding of spatial and temporal variability in abundance of redband trout as well as an understanding of the balance between sampling intensity and precision of empirical data. The objectives of this study were to 1) quantify the abundance and distribution of redband trout at two putative demographic scales over a six-year period in the northern Great Basin and 2) quantify variability in the sampling data to help guide development of conservation and monitoring plans for redband trout in the northern Great Basin. We used a generalized random-tessellation stratified design to select spatially well-distributed sample sites at the population and species management unit (SMU) scales. We conducted depletion electrofishing at about 30 to 40 sample sites within each of six SMUs annually from 2007 through 2012 and at about 30 sample sites per population within each SMU twice during the six year study. Electrofishing data were used to calculate site level density and abundance estimates that were extrapolated to the population and SMU levels. The abundance of redband trout varied from 1,716 to 191,690 among populations and from 17,709 to 566,514 among SMUs during the six years. Abundance was generally stable within populations and SMUs over the six-year study; however, inter-annual variation was observed in some populations and SMUs. We predicted that about 16 and 71 sample sites would need to be sampled at the population and SMU levels, respectively, to achieve desired levels of precision of abundance estimates (i.e., ≤ 80% and ≤ 40% relative confidence index at the population and SMU levels, respectively). Sampling intensities of > 60 and 100 sample sites at the population and SMU levels, respectively, result in little benefit in our ability to detect change in redband trout density using the current approach. Additionally, variability in point estimates of redband trout density did not decrease substantially at sampling intensities greater than about 20 sample sites at the population and SMU levels.

Redband trout (*Oncorhynchus mykiss newberrii*) colonized the Great Basin beginning about 70,000 years ago (Behnke 2002) and currently occupy six endorheic sub-basin in the northern Great Basin (Figure 1.1): Catlow Valley, Chewaucan, Fort Rock, Goose Lake, Malheur Lakes, and Warner Lakes (Goodson et al. 2005*a*; *b*). Streams within the Great Basin experience a relatively large degree of inter-annual variation in discharge (e.g., Figure 1.2) and temperature, and surface air temperatures in the Great Basin suggest a general warming trend during the 20th century (Tang and Arnone 2013). There is some evidence that redband trout have a higher metabolic scope at a given temperature than other salmonids, suggesting that they have specific adaptations to cope with warmer temperatures associated with desert environments (Rodnick et al. 2004).

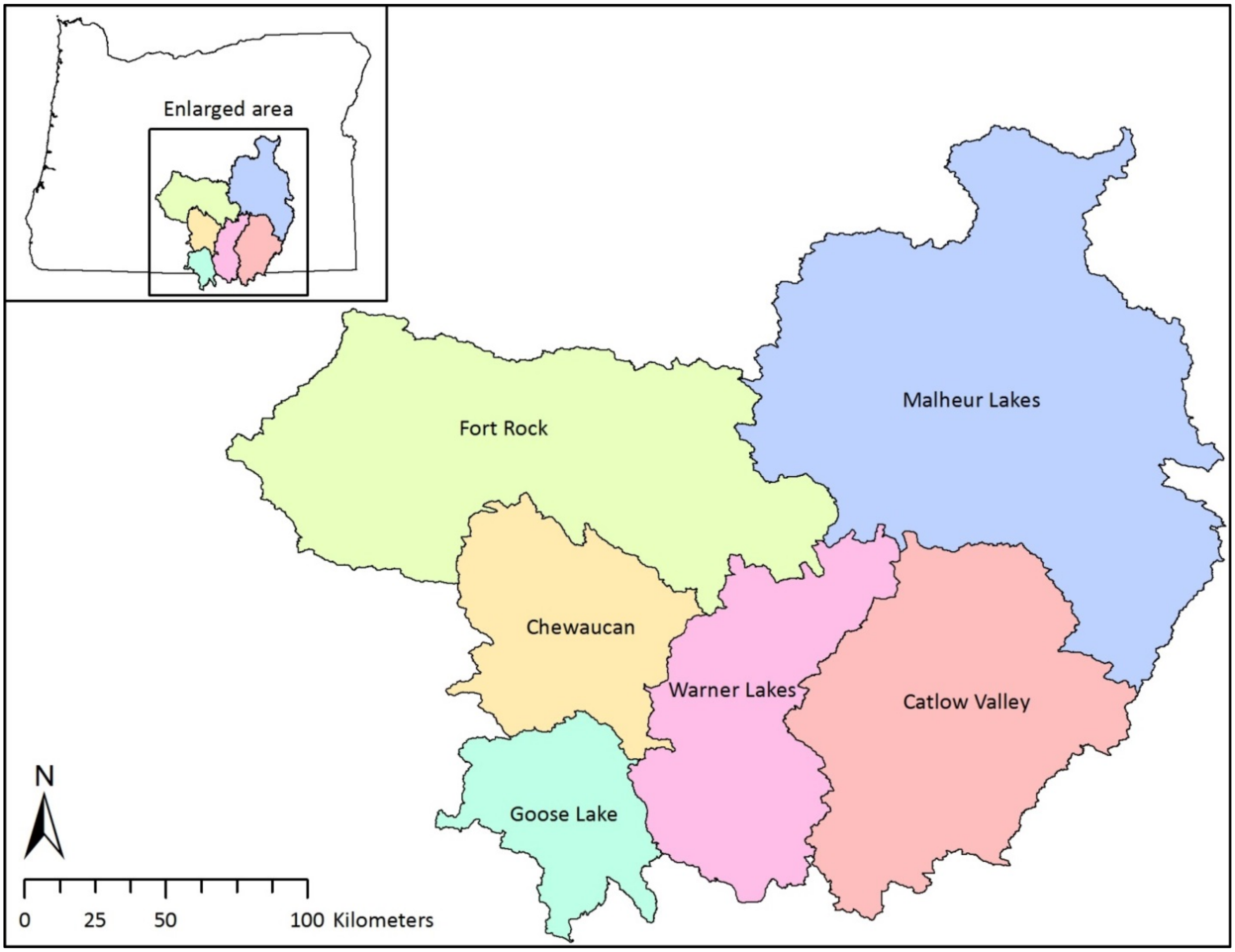


Figure 1.1 – Six endorheic sub-basins in the northern Great Basin. These basins correspond to the six redband trout species management units in the northern Great Basin described by Goodson et al. (2005*a*; *b*).

Survey data collected for redband trout in 1992 and 1994 suggested that some populations of redband trout in the northern Great Basin exhibited depressed abundance (Dambacher et al. 2009). Therefore, in 1997 the US Fish and Wildlife Service (USFWS) was petitioned to list redband trout in the northern Great Basin as threatened or endangered under the US Endangered Species Act (ESA). Following this petition, and to support a population status review, Oregon Department of Fish and Wildlife (ODFW) conducted a survey of redband trout abundance in each of the six-endorheic sub-basins of the Great Basin occupied by redband trout. Dambacher et al. (2009) estimated that there were 971,313 age-1+ redband trout within the northern Great Basin in 1999 based on this survey; abundance estimates varied from 57,270 to 435,045 among sub-basins (Dambacher et al. 2009). In 2000 the USFWS determined that listing redband trout as threatened or endangered under the ESA was not warranted based on findings of their status review and results of Dambacher et al. (2009).

In 2005 ODFW conducted a status review of native fishes in Oregon (Goodson et al. 2005*a*; *b*), which classified redband trout in the northern Great Basin as "at risk" or "potentially at risk". However, lack of data at the scale of individual populations and lack of data describing trends in abundance over time prevented a thorough review of status of redband trout in the northern Great Basin (Goodson et al. 2005a; b; Miller et al.

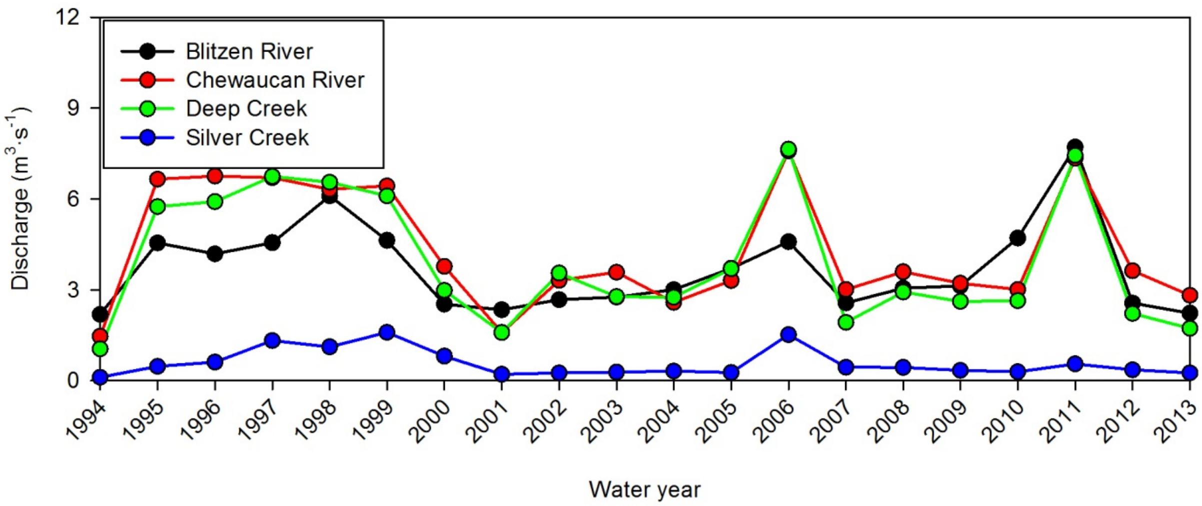


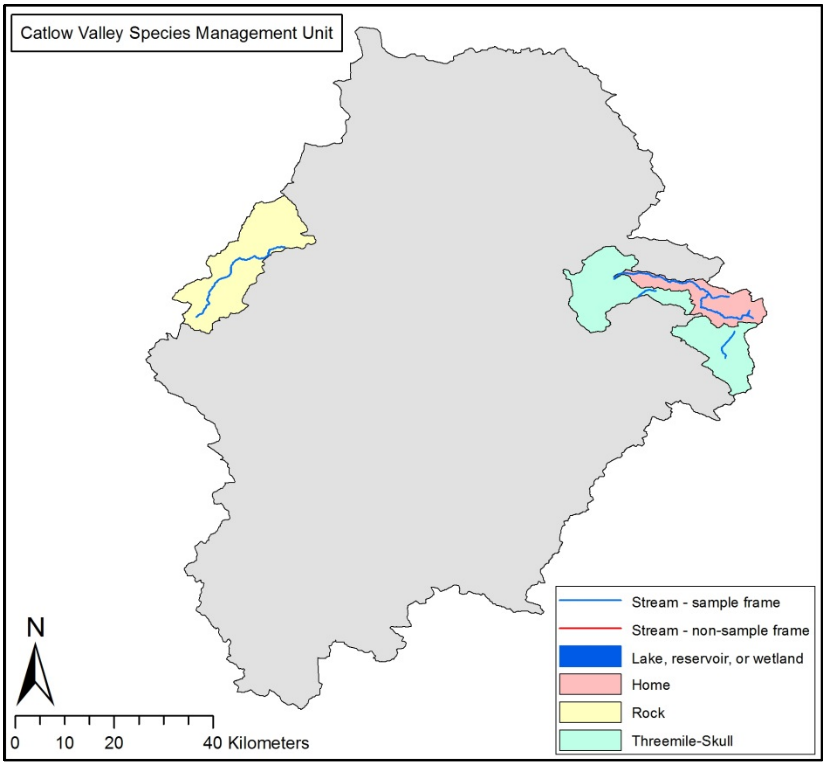
Figure 1.2 – Mean annual discharge ($m^3 \cdot s^{-1}$) for four streams within the northern Great Basin. Mean annual discharge for the last 20 water years was calculated from daily mean discharge data available from the Oregon Water Resources Department (available: http://apps.wrd.state.or.us/apps/sw/hydro_near_real_time/) for stations 10396000 (Blitzen River), 10384000 (Chewaucan River), 10371500 (Deep Creek), and 10390000 (Silver Creek).

2010). Therefore, ODFW began a six-year study of redband trout in the northern Great Basin to provide abundance and distribution data for redband trout at the population level and to evaluate trends in abundance over time. The specific objectives of this study were 1) to quantify the abundance and distribution of redband trout in the northern Great Basin at two spatial scales during a six-year time period and 2) quantify variability in the sampling data to help guide development of conservation and monitoring plans for redband trout in the northern Great Basin.

Methods

Study Area – This study was conducted in six endorheic sub-basins within the northern Great Basin: Catlow Valley, Chewaucan, Fort Rock, Goose Lake, Malheur Lakes, and Warner Lakes (Figure 1.1). These sub-basins correspond to the six redband trout species management units within the Great Basin identified by ODFW (Goodson et al. 2005*a*; *b*). Goodson et al. (2005*a*; *b*) define species management units (SMUs) as, '*groups of populations from a common geographic area that share similar life history, genetic, and ecological characteristics*.'

From three to six populations were identified within each sub-basin (Figure 1.3). The degree of demographic and genetic connectivity among and within these population groupings is unknown; therefore, the term population is used here to identify sample populations as opposed to biological populations. Population designations generally followed Goodson et al. (2005*a*; *b*), with the exception that some populations identified by Goodson et al. (2005*a*; *b*) were grouped into a single population due to small spatial extent or logistical constraints associated with achieving high enough sampling intensities to meet project objectives. Specifically, the Skull and Threemile populations were combined into a single population (Skull-Threemile), the Lower Drews and Upper Drews populations were combined into a single population (Drews), the Crane, Cogswell, Kelley, and Tandy populations were combined into a single population (Eastside), the Antelope, Cottonwood, and



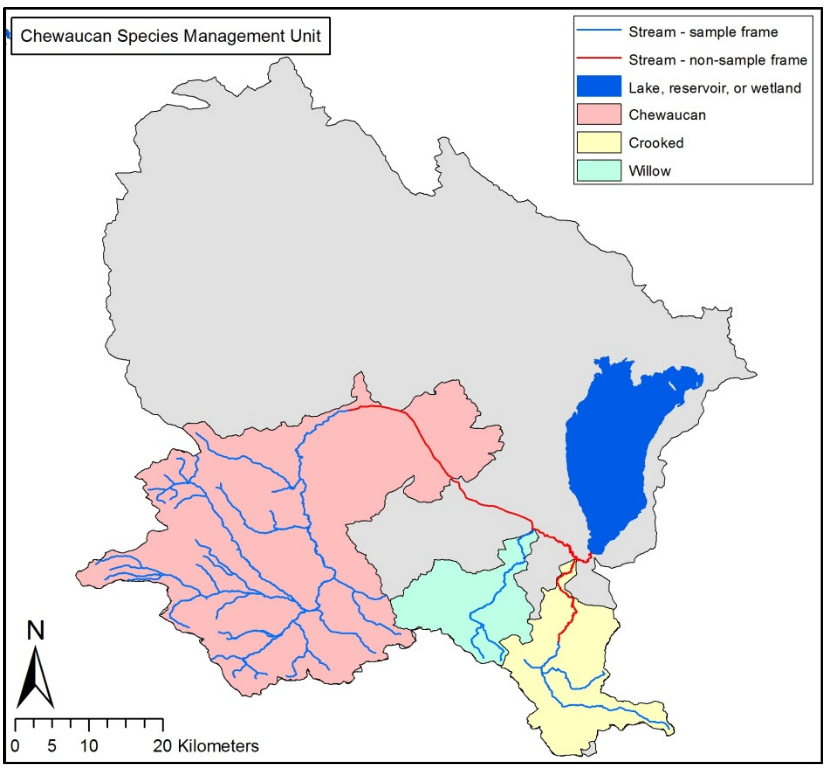


Figure 1.3 – Populations within species management units and sample frame designation; non-sample frame stream segments are provided to illustrate hydrologic connectivity.

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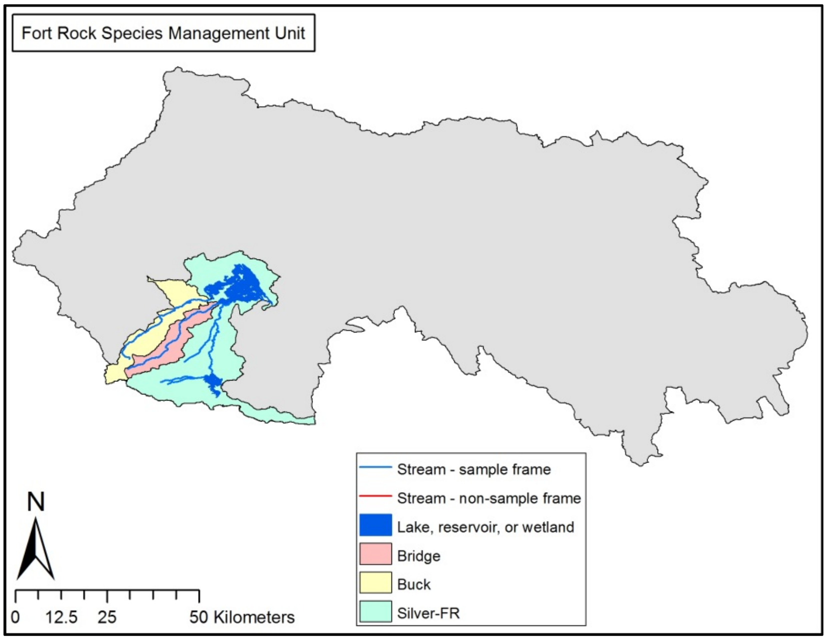

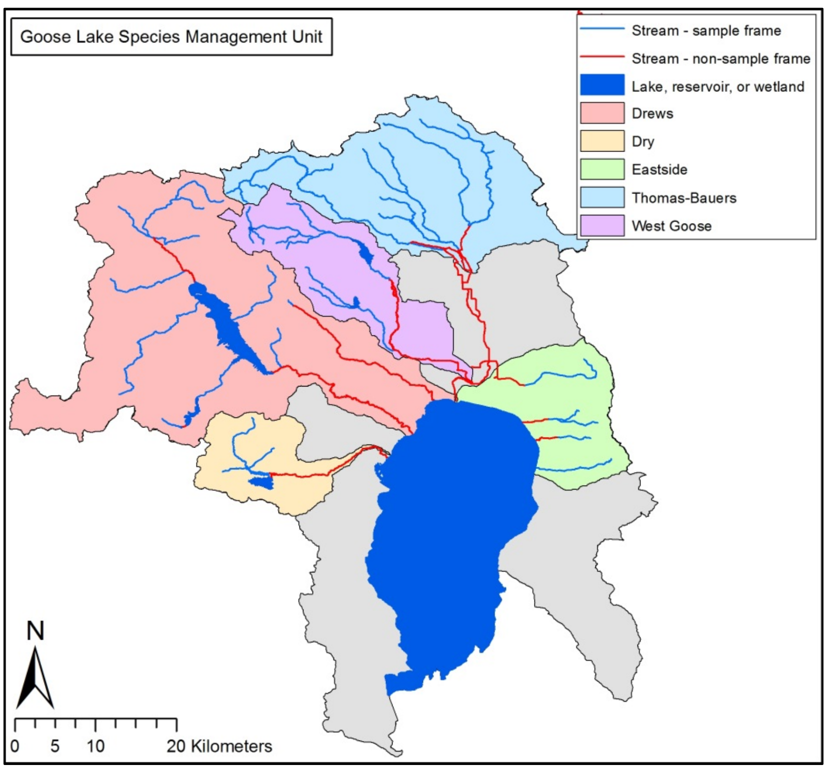

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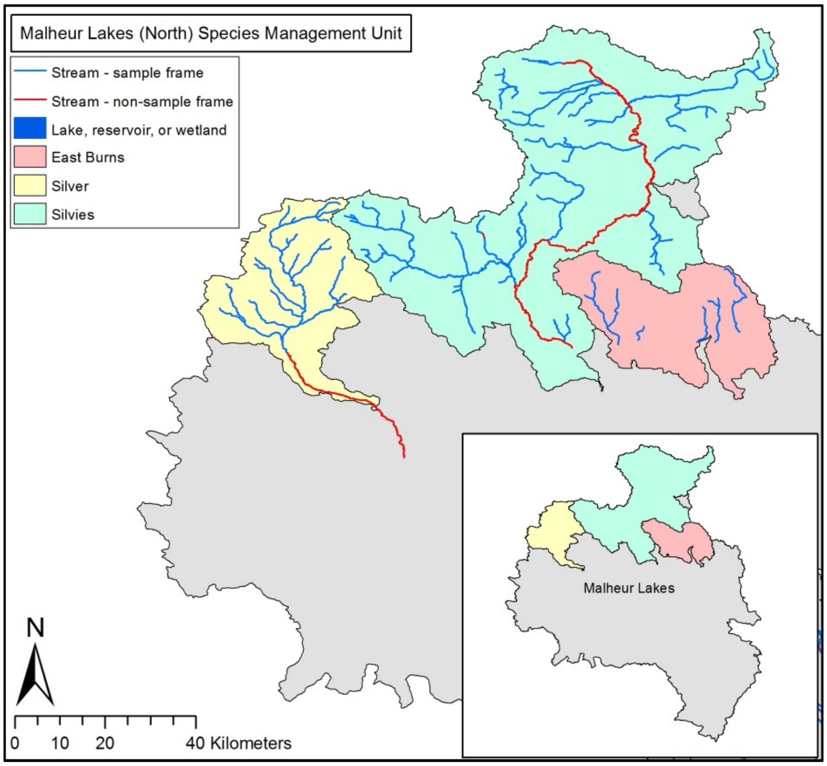

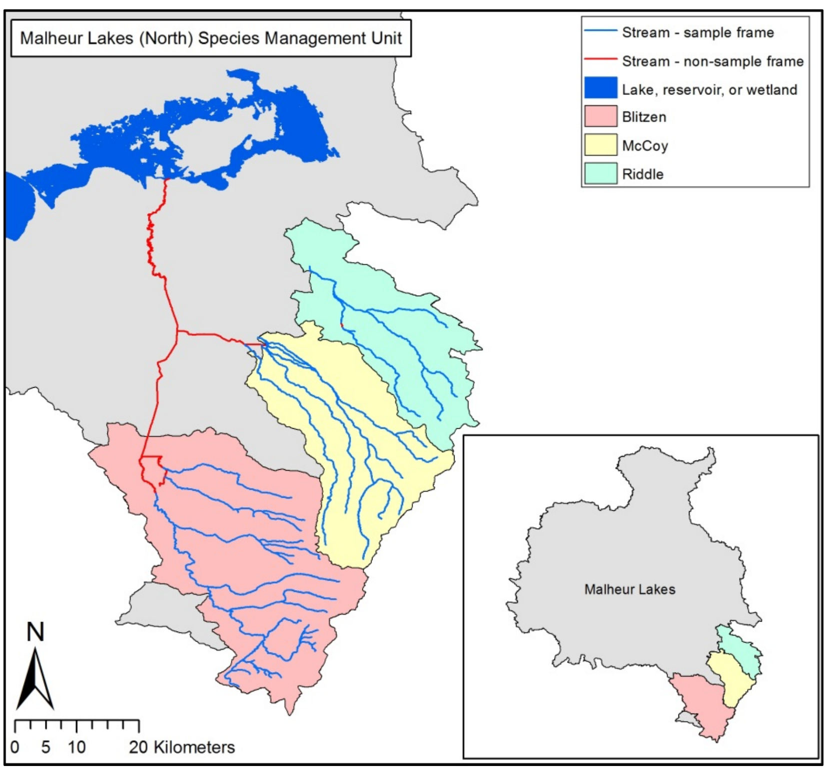

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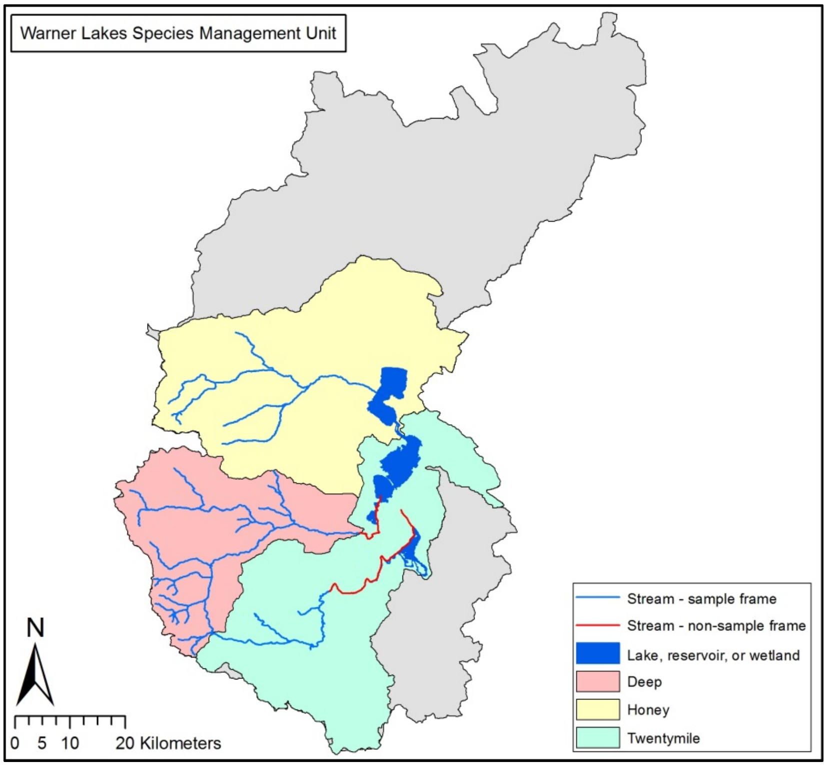

Muddy populations were combined into a single population (West Goose), the Coffeepot, Cow, Poison, Prater, and Rattlesnake populations were combined into a single population (East Burns), and the Lower Deep and Upper Deep populations were combined into a single population (Deep).

The study was conducted during the summers of 2007 through 2012 during base-flow conditions. This time period generally corresponded to below average water years (Figure 1.4); however, both 2011 (during the study) and 2006 (one year before the study) were above average water years based on a 20 year average.

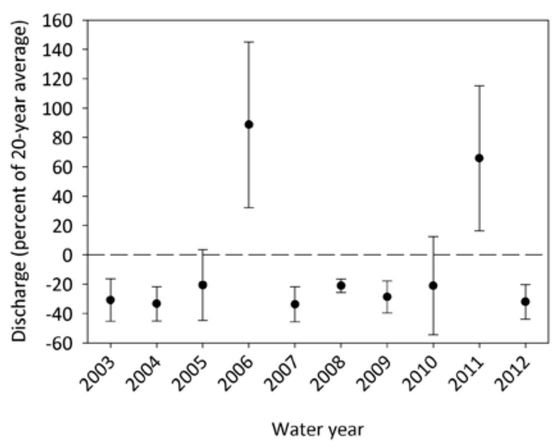

Figure 1.4 – Mean annual discharge for four years prior to the study and six years during the study expressed as a percent of the 20-year average discharge ($m^3 \cdot s^{-1}$). Mean (± SD) values were calculated from four streams within the study area (Figure 1.2).

Sample Frame and Sample Site Selection – The sample frame was defined as wadeable streams within the study area known or assumed to be occupied by redband trout based on previous sampling and expert opinion (Goodson et al. 2005*a*; *b*). The sample frame was mapped at the scale of 1:24,000 and divided into 100-m sample sites. A generalized random-tessellation stratified (GRTS) design was used to select sample sites. Briefly, the GRTS process uses restricted randomization (Bailey 1987; Stevens and Olsen 2000) to identify the order in which sites should be sampled to provide a spatially well-distributed probability sample (Stevens and Olsen 2004). The GRTS design was developed to sample finite linear and areal resources and has the benefits that 1) inclusion probabilities can be arbitrary and 2) sample points can be added dynamically while maintaining a spatially well-distributed probability sample (Stevens and Olsen 2004).

The GRTS design was used to select sample sites within the framework of an augmented serially alternating panel design (Table 1.1) (see Urquhart and Kincaid 1999). Seven panels were selected for each population, one panel that was repeated annually, and one panel for each of the six years of the study. Sites for each panel were selected sequentially from the GRTS list such that sites for the annually-repeated panel (hereafter, annual sites) were selected first and sites for each of the sample years (hereafter, non-annual sites) were subsequently selected; non-annual sites were selected without replacement over the course of the study. During four of the six years of the study a total of about 30 sites per SMU (annual + non-annual) were initially selected (hereafter SMU-level sampling) and during the two remaining years of the study a total of about 30 sites per population (annual + non-annual) were initially selected (hereafter population-level sampling) (Table 1.2). In general, sample sites were proportionally allocated among populations during SMU-level sampling years. We used this site selection scheme to provide SMU-level information during all six years of the study and population-level information during two years of the study.

Table 1.1. Conceptual framework for an augmented serially alternating panel design. An 'X' denotes the year in which each panel was sampled; e.g., in 2009 panels 0 and 3 were sampled.

	Sample year					
Panel	2007	2008	2009	2010	2011	2012
0	X	X	X	X	X	X
1	X					
2		X				
3			X			
4				X		
5					X	
6						X

Table 1.2. Sampling intensity [species management unit-level (SMU) or population-level (***POP***)] by sample year for six species management units in the northern Great Basin. The Catlow Valley SMU was not sampled in 2008 due to logistical and access constraints.

	Sample year					
SMU	2007	2008	2009	2010	2011	2012
Catlow Valley	SMU		SMU	SMU	SMU	SMU
Chewaucan	SMU	SMU	***POP***	SMU	SMU	***POP***
Fort Rock	SMU	SMU	***POP***	SMU	SMU	***POP***
Goose Lake	***POP***	SMU	SMU	***POP***	SMU	SMU
Malheur Lakes	SMU	***POP***	SMU	SMU	***POP***	SMU
Warner Lakes	***POP***	SMU	SMU	***POP***	SMU	SMU

Sample site accessibility was assessed prior to the field season each year of the study. Land ownership of sample sites was assessed based on public records supplied by county assessor offices and federal land-management agencies. Sample sites that were located on public land and that had public access were considered accessible. Sample sites that were located on private land that we were granted permission to access were also considered accessible. Samples sites that were on private land for which we were denied permission to access were classified as 'denied access'. Sample sites that were on public land, but that required access via private land for which we were denied permission were classified as 'not surveyed'. Sample sites that were too remote to access due to logistical constraints (e.g., lack of roads, extremely rugged terrain) were also classified as 'not surveyed'.

Sample Site Setup– Field crews were provided coordinates for the downstream boundary of each accessible sample site based on mapping of the sample frame at the scale of 1:24,000 (see above). Field crews located sample sites using handheld GPS, map, and compass. Once a sample site was located a preliminary assessment was made to determine if the site was sampleable. If a site was dry it was classified as 'dry channel'. If stream discharge at a site was too great to allow block nets to be used the site was classified as 'not surveyed'. If a site was too deep or wide to be effectively sampled it was classified as 'not surveyed'. All other sites were considered sampleable.

Sample site length was generally 30 times the average wetted width of the stream (estimated *in situ*) with the exception that sample sites had to be at least 30 m long and no longer than 100 m. Block nets were placed at the downstream and upstream boundary of each sample site to ensure that fish could not enter or leave a sample site during sampling (Dunham et al. 2009). The downstream block net was placed as close to the provided coordinates as feasible; however, the block net position was moved upstream or downstream such that it would be placed at an appropriate habitat unit break. The upstream block net was placed 30 to 100 m upstream from the lower boundary depending on sample site length (see above) at an appropriate habitat unit break.

Depletion and Mark-Recapture Electrofishing – Depletion and mark-recapture electrofishing were conducted at sample sites using a backpack electrofisher (Smith-Root model LR-12 or model LR-24). Electrofishing was generally conducted by a two-person crew (one electrofisher and one netter), but four-person crews (two electrofishers and two netters) were sometimes used at sites that were too wide to be effectively sampled by a two-person crew (Dunham et al. 2009). Once block nets were in place, field crews identified appropriate electrofisher settings (i.e., voltage, frequency, and duty cycle) based on expert opinion, prior knowledge of the sample site, or the electrofisher quick set-up feature (LR-24 only).

For depletion and mark-recapture electrofishing, field crews started at the downstream site boundary and electrofished in an upstream direction taking care to sample all available habitat. Once field crews reached the upstream site boundary they turned around and electrofished in a downstream direction rapidly moving through the site to direct fish towards the lower block net; fish were netted and placed in a bucket filled with aerated stream water as they were encountered. The combined upstream and downstream movement through the sample site constituted a single electrofishing pass.

Depletion electrofishing was conducted at all sampleable sites. Two or four passes were made through each site. If the number of redband trout sampled on the second pass was ≤ 50% of the number of redband trout sampled on the first pass then only two passes were completed. If the number of redband trout sampled on the second pass was > 50% of the number of redband trout sampled on the first pass then four passes were completed. Only redband trout ≥ 60 mm (fork length) were considered for the above criteria. After each electrofishing pass, redband trout were anesthetized in buffered MS-222 and measured for fork length; redband trout > 200 mm were not anesthetized. Redband trout were either returned to the stream outside of the sample site if sampling was not complete or back to the sample site if sampling was complete.

If only two electrofishing passes were conducted, but the number of redband trout sampled on the second pass was > 50% of the number of redband trout sampled on the first pass then the site was classified as 'estimate failed'. If four electrofishing passes were conducted and the total number of redband trout sampled on the third and fourth pass was > 50% of the total number of redband trout sampled on the first and second pass then the site was classified as 'estimate failed'. If block nets failed (i.e., were insufficient to limit movement of fish into or out of the sample site) then the site was classified as 'estimate failed'. Otherwise, sample sites were classified as 'estimate passed'.

Mark-recapture electrofishing was conducted at 11% of the sampleable sites visited in 2007 through 2009 (from 4 to 38% among SMUs and years). Mark-recapture sampling occurred over a two-day period. A single electrofishing pass was made through the sample site on day one (marking event). Fish were sampled as above with the exception that redband trout ≥ 60 mm were marked by removing a small portion of the upper lobe of their caudal fin. Fish were returned to the sample site after the marking event and block nets were left in place overnight. The site was revisited the next day and a single electrofishing pass was made through the site (recapture event); the numbers of marked and unmarked redband trout (≥ 60 mm) were recorded. Subsequent electrofishing passes were made through the site on day two to perform depletion sampling (i.e., the recapture event functioned as the first pass of the depletion electrofishing). If fewer than 10 redband trout ≥ 60 mm were captured during the marking event the site was not used for mark-recapture electrofishing, but was still used as a depletion electrofishing site.

Density and Abundance Estimates – Linear density of redband trout was calculated for each sample site where electrofishing was conducted. Linear density was used as the primary measurement for analyses because site lengths differed. Linear density (hereafter density), as opposed to areal or volumetric density, was calculated primarily to allow extrapolation of density estimates to population and SMU level abundance estimates based on sample frame length and secondarily to facilitate comparisons with previous reports (Miller et al. 2010) and publications (Dambacher et al. 2009). Redband trout < 60 mm were not included in any analyses related to redband trout density or abundance. For sample sites where two electrofishing passes were conducted, depletion density estimates were calculated as:

$$Density_{Dep} = \left(\frac{y_1^2}{y_1 - y_2}\right) \times \left(\frac{1}{Site\ Length}\right)$$

where $Density_{Dep}$ is the depletion density estimate, y_1 is the number of redband trout sampled on the first electrofishing pass, y_2 is the number of redband trout sampled on the second electrofishing pass, and *Site Length* is the length of the sample site (m) (Zippin 1958). The same equation was used for depletion density estimates for sample sites where four electrofishing passes were conducted with the exception that y_1 equaled the total number of redband trout sampled on electrofishing passes one and two and y_2 equaled the total number of redband trout sampled on electrofishing passes three and four.

Mark-recapture density estimates were calculated as:

$$Density_{MR} = \left(\frac{(M+1)(C+1)}{R+1} - 1\right) \times \left(\frac{1}{Site\ Length}\right)$$

where $Density_{MR}$ is the mark-recapture density estimate, *M* is the number of redband trout marked during the marking event, *C* is the total number of redband trout captured during the recapture event, *R* is the number of redband trout captured during the recapture event that were marked, and *Site Length* is the length of the sample site (m) (Chapman 1951).

Because depletion methods are biased (Peterson et al. 2004), we used robust regression (PROC ROBUSTREG, s-estimation; SAS software) to quantify the relationship between mark-recapture and depletion density estimates. This analysis was performed to develop a predictive model and is not intended to infer causality. Robust regression was used instead of simple linear regression because preliminary analyses indicated the presence of influential outliers (Neter et al. 1996). We used the results of this analysis to calculate corrected density estimates for all sample sites (hereafter, density estimate).

Density estimates were extrapolated to the population level (for population-level sampling years) and the SMU level (all years) and reported as redband trout abundance (total.est function in package 'spsurvey'; R software). Because these abundance estimates were calculated from corrected density estimates they represent redband trout abundance corrected for depletion sampling bias. Extrapolations were based on an adjusted site weight. First, an initial site weight was calculated for each population and year as:

$$W_I = \frac{Sample\ Frame\ Length}{EP + EF + NS + DA + DC}$$

where W_I is the initial site weight, $Sample\ Frame\ Length$ is the total length (m) of the sample frame (i.e., stream) for the population, EP is the number of sites classified as ‘estimate passed’, EF is the number of sites classified as ‘estimate failed’, NS is the number of sites classified as ‘not surveyed’, DA is the number of sites classified as ‘denied access’, and DC is the number of sites classified as ‘dry channel’. Second, an adjusted site weight was calculated for each population and year as:

$$W_A = W_I \frac{EP + EF + NS + DA\left(1 - \frac{DC}{EP + EF + NS + DC}\right)}{EP}$$

where W_A is the adjusted site weight and W_I, EP, EF, NS, DA, and DC are as above.

SMUs were stratified by population when estimating SMU level abundance. Variance was calculated using the local mean variance estimator when data were available for > 3 sample sites within strata for SMU level abundance estimates or total for population level abundance estimates. Variance was calculated using the simple random sampling estimator when ≤ 3 sample sites were available. Confidence intervals (95%) were calculated for all population-level and SMU-level abundance estimates. Additionally, we calculated a relative confidence index to determine whether current sampling intensity was sufficient to meet pre-determined abundance estimate precision goals. The relative confidence index was calculated as:

$$RCI = \frac{95\%\ CI}{Abundance} \times 100$$

where *RCI* is the relative confidence index, *Abundance* is the estimated abundance of redband trout at the population level or SMU level, and 95% *CI* is the 95% confidence interval for the abundance estimate. The *RCI* was calculated for population by year combinations for population-level sampling years and for SMU by year combinations for all SMUs with the exception of the Catlow Valley SMU; the Catlow Valley SMU had low sample sizes (Table 1.3). This *RCI* is analogous to the ± 95% relative confidence limits presented in Miller et al. (2010); therefore, target precision *RCI* values were ≤ 80% at the population level and ≤ 40% at the SMU level.

Cumulative frequency distributions of redband trout density were plotted for each SMU by year and all sites combined. Additionally, site-specific densities (all years combined) were plotted on maps for each SMU.

Distribution of Redband Trout – Distribution of redband trout was plotted on maps by SMU for each population among years. Distribution data were plotted for all redband trout as ‘detected’ or ‘not detected’. Trends in distribution were qualitatively summarized for all years combined because the sample frame selection and the panel design generally precluded formal analyses of distribution and changes in distribution of redband trout as a function of time. Specifically, the sample frame did not include areas of unknown or unsuspected occupancy and the sample frame did not include non-wadeable streams. Therefore, inference is limited to the known or suspected distribution of redband trout within the study area.

Differences in Redband Trout Density Between Annual Sites and Non-Annual Sites – We compared mean redband trout density between annual and non-annual sites using *t*-tests (α = 0.05; PROC TTEST; SAS software). We were primarily interested in the magnitude of potential differences between redband trout density at annual and non-annual sites as opposed to the direction of the difference (i.e., we were not interested in whether density was generally higher at annual sites compared to non-annual sites, or vice-versa), which would likely vary among populations. Therefore, we performed a series of *t*-tests as opposed to an analysis of

Table 1.3 – Number of annual (A) and non-annual (N-A) sample sites by species management unit (SMU) and population where redband trout abundance was estimated in the northern Great Basin from 2007 through 2012. The number of annual sites where redband trout abundance was estimated varied among years for some populations because some annual sites were dry in some years, depletion criteria were not met in some years (i.e., estimate failed) for some sites, and access was denied to some annual sites in some years.

		2007		2008		2009		2010		2011		2012	
SMU	Population	A	N-A	A	N-A	A	N-A	A	N-A	A	N-A	A	N-A
Catlow Valley	Home	1	0			0	1			2	2	2	1
	Rock	2	5			4	3	5	6	5	6	1	3
	Skull-Threemile					2	0	2	0	2	2		
Chewaucan	Chewaucan	8	16	11	7	11	13	12	12	12	10	12	17
	Crooked	2	2	2	1	1	21	2	1	2	2	2	25
	Willow	1	2	1	2	1	18	1	2	1	1	1	8
Fort Rock	Bridge	4	2	4	3	4	17	4	4	4	3	4	26
	Buck	4	5	5	2	4	15	5	5	4	4	4	25
	Silver-FR	6	6	6	4	5	8	6	3	2	4	5	18
Goose Lake	Drews	2	18	4	2	3	4	4	16	4	4	4	3
	Dry	1	11	1	1	1	1	1	3	1	1	0	2
	Eastside	1	16	2	0	2	1	2	19	2	1	2	1
	Thomas-Bauers	5	15	6	4	6	6	5	17	6	6	5	7
	West Goose	1	18	2	2	2	2	2	18	2	2	2	3
Malheur Lakes	Blitzen	3	4	3	20	3	3	3	2	3	23	3	3
	East Burns	1	2	2	23	2	0	2	1	2	23	2	1
	McCoy	3	3	3	21	3	2	1	2	3	22	3	3
	Riddle	2	1	2	16	1	1	2	1	2	23	1	2
	Silver	3	3	3	22	1	2	3	3	3	27	3	3
	Silvies	5	9	7	16	7	6	7	8	7	19	7	8
Warner Lakes	Deep	5	13	8	5	8	8	8	22	8	8	6	8
	Honey	4	14	5	2	5	5	5	15	5	4	5	4
	Twentymile	2	16	2	2	2	2	2	27	2	2	2	1

variance (ANOVA). t-tests were performed by population and year for population-level sampling years and by SMU and year for all SMUs with the exception of the Catlow Valley SMU.

Predicted Detectable Change in Density – A bootstrap resampling procedure was used to predict the percent change in redband trout density that could be detected at different levels of sampling intensity for population-level and SMU-level sampling based on empirical data from this study. For this analysis we assumed that redband trout densities at sites that we sampled were representative of densities throughout the sample frame. At the population level, a set of sample sites was randomly drawn with replacement from all sample sites for each population by year combination for years with population-level sampling. This procedure was replicated 1000 times for sample sizes from 2 to 150 sample sites in increments of 1.

The detectable change in abundance for each sample size by replicate combination was calculated as:

$$d = 100 \times \sqrt{\frac{t^2 \times \mathrm{CV}^2}{n}}$$

where d is the detectable change in redband trout density (%), t is the t-distribution deviate for a sample size of n, CV is the coefficient of variation (SD/mean), and n is the sample size (rearranged from Snedecor and Cochran 1989 in Quist et al. 2009). Mean detectable change (± SD) was plotted as a function of sample size. Data were combined by population between years and among populations and years by averaging means and variances.

A similar analysis was performed at the SMU level for all SMUs with the exceptions that 1) the analysis was conducted for all years, 2) random draws were proportionally allocated among populations within SMUs based on sample frame length, 3) sample sizes varied from 2 to 300, and 4) the Catlow Valley SMU was not evaluated.

Predicted Relative Confidence Index of Density Estimates – A bootstrap resampling procedure was used to predict the RCI that would be attained at different levels of sampling intensity for population-level and SMU-level sampling. For this analysis we assumed that redband trout densities at sites that we sampled were representative of densities throughout the sample frame.

At the population level, a set of sample sites was randomly drawn with replacement from all sample sites for each population by year combination for years with population-level sampling. This procedure was replicated 1000 times for sample sizes from 2 to 150 sample sites in increments of 1. Density estimates for the randomly drawn sites were used to calculate a mean density and 95% confidence interval for each replicate by sample size combination. Mean densities and 95% confidence intervals were used to calculate a RCI (as above) and the mean RCI was calculated for each sample size. Mean RCIs (± SD) were plotted as a function of sample size. Data were combined by population between years and among populations and years by averaging means and variances.

A similar analysis was performed at the SMU level with the exception that 1) the analysis was conducted for all years, 2) random draws were proportionally allocated among populations within SMUs based on sample frame length, 3) sample sizes varied from 2 to 300, and 4) the Catlow Valley SMU was not evaluated.

Predicted Variability of Redband Trout Density Estimates – A bootstrap resampling procedure was used to estimate the mean (± SD) density of redband trout for different levels of sampling intensity at the population-level and SMU-level. For this analysis we assumed that redband trout densities at sites that we sampled were representative of densities throughout the sample frame.

At the population level, a set of sample sites was randomly drawn with replacement from all sample sites for each population by year combination for years with population-level sampling. This procedure was replicated 1000 times for sample sizes from 2 to 100 sample sites in increments of 1. Density estimates for the randomly drawn sites were used to calculate a mean density for each replicate by sample size combination. The mean densities were averaged by sample size and SD was calculated. Mean densities (± SD) were plotted as a function of sample size. Data were combined by population between years and among populations and years by averaging means and variances.

A similar analysis was performed at the SMU level with the exception that 1) the analysis was conducted for all years, 2) random draws were proportionally allocated among populations within SMUs based on sample frame length, and 3) the Catlow Valley SMU was not evaluated.

Results

The GRTS design was used to draw a total of 2,393 sites for this study; about 10% of the total number of sites in the sample frame. Depletion criteria were achieved (estimate passed) at 1,500 sites, depletion criteria were not achieved (estimate failed) at 40 sites, 130 sites were not surveyed, we were denied access to 496 sites, and 227 sites were dry when visited

Table 1.4 – Percent of sites classified as 'estimate passed', 'estimate failed', 'not surveyed', 'denied access', and 'dry channel' by species management unit for all sites selected from 2007 through 2012.

	Percent				
Species Management Unit	Estimate passed	Estimate failed	Not surveyed	Denied access	Dry channel
Catlow Valley	47	1	7	24	22
Chewaucan	62	2	3	19	13
Fort Rock	69	5	9	14	2
Goose Lake	62	1	3	19	15
Malheur Lakes	60	1	7	27	6
Warner Lakes	69	1	4	17	9

Table 1.5 – Percent of sites classified as 'estimate passed', 'estimate failed', 'not surveyed', 'denied access', and 'dry channel' by study year for all sites selected among species management units; Catlow Valley, Chewaucan, Fort Rock, Goose Lake, Malheur Lakes, and Warner Lakes.

	Percent				
Year	Estimate passed	Estimate failed	Not surveyed	Denied access	Dry channel
2007	51	2	3	29	15
2008	72	1	3	15	10
2009	61	4	11	14	10
2010	68	1	6	19	7
2011	67	1	9	21	2
2012	62	< 1	2	23	12

(Supplemental Table 2.1). The percent of passed estimates was generally consistent among SMUs with the highest percent passed occurring in the Fort Rock and Warner Lakes SMUs and the lowest percent passed occurring in the Catlow Valley SMU (Tables 1.4 and 1.5). The percent of sites that had failed estimates was ≤ 5% among SMUs and among years and the percent of sites that were not surveyed was ≤ 11% among SMUs and among years. The percent of sites that we were denied access to was lowest in the Fort Rock SMU and highest in the Catlow Valley and Malheur Lakes SMUs; access was denied most in 2007 and least in 2009 among years. The percent of dry sites varied from 2% in the Fort Rock SMU to 22% in the Catlow Valley SMU. The percent of dry sites was only 2% in 2011 compared to 15% in 2007 and 12% in 2012.

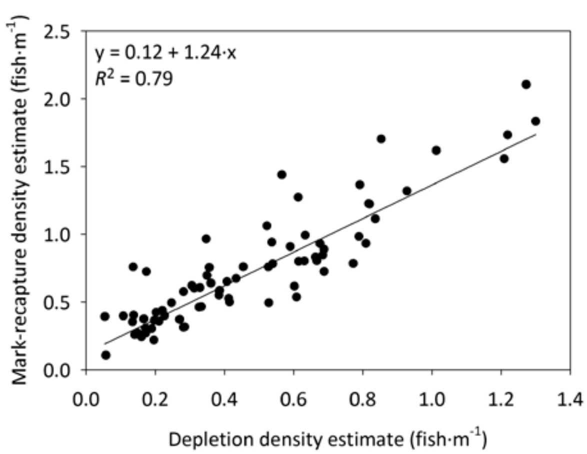

Figure 1.5 – Relationship between mark-recapture and depletion density estimates. Line fit using robust regression and shown with detected outliers removed.

Density and Abundance Estimates – Depletion electrofishing underestimated redband trout density relative to mark-recapture electrofishing. A linear relationship was observed between depletion density estimates and mark-recapture density estimates, and the intercept (B_0 = 0.12, p < 0.01) and slope (B_1 =1.24, p < 0.01) parameter estimates were both significantly different from zero (Figure 1.5). Therefore, depletion density estimates were corrected for bias using the equation:

$$Density = 0.12 + 1.24 \cdot Density_{Depletion}$$

where $Density$ is the density estimate (i.e., corrected density) and $Density_{Depletion}$ is the density estimate obtained from depletion electrofishing.

Abundance of redband trout varied from a low of 1,716 (population Dry sampled in 2010) to a high of 191,690 (population Silvies sampled in 2011) among populations during population-level sampling years (Figure 1.6). Population-level abundance estimates were generally similar between the two sample years within populations, with the exceptions of the Blitzen, Bridge, Buck, and West Goose populations (Figure 1.6). Abundance estimates met precision criteria (i.e., ≤ 80% RCI at the population level) for 92.5% of the population by year combinations examined. Abundance estimates did not meet precision criteria in the Deep population in 2007, the Dry population in 2007, and the West Goose population in 2007. Estimated abundance of redband trout varied from 17,996 to 560,201 among SMUs and years (Figure 1.7). Redband trout abundance was generally consistent among years in the Catlow Valley, Chewaucan, and Warner Lakes SMUs; although, some inter-annual variation was observed. A consistent negative trend in mean abundance was observed from 2007 to 2011 in the Fort Rock SMU; however, this trend was followed by an increase in abundance such that mean abundance in 2012 was similar to mean abundance in 2009 and 2010. Redband trout abundance decreased from 125,807 in 2007 to 76,368 in 2008 in the Goose Lake SMU, but remained consistent thereafter. Abundance estimates met precision criteria (i.e., ≤ 40% RCI at the SMU level) for 46.7% of the SMU by year combinations. Abundance estimates met precision criteria in the Chewaucan SMU in 2008 and 2009, the Fort Rock SMU in 2009, 2010, 2011, and 2012, the Goose Lake SMU in 2007, 2010, 2011, and 2012, the Malheur Lakes SMU in 2008, 2011, and 2012, and the Warner Lakes SMU in 2010.

No redband trout (≥ 60 mm) were detected in 26% of the sample sites. Redband trout density for these sites was corrected using the equation ($Density = 0.12 + 1.24 \cdot Density_{Depletion}$; where $Density_{Depletion} = 0$) resulting in densities of 0.12 redband trout·m^{-1} for these sites (Figure 1.8). Fifty percent of sample sites had densities less than 0.28 redband trout·m^{-1}, 80% of sample sites had densities less than 0.64 redband trout·m^{-1}, and 95% of sample sites had densities less than 1.22 redband trout·m^{-1}. The Goose Lake SMU had the highest percent of sample sites where no redband trout were detected; however, most SMUs had similar densities of redband trout (Supplemental Figure 2.1). The highest densities of redband trout were observed in the Catlow Valley SMU during 2007. No consistent trends were apparent with respect to the spatial distribution of high density sample sites within the study area (Figure 1.9).

Distribution of Redband Trout – Redband trout were generally present throughout the sample frame (Figure 1.9). Sites where redband trout were not detected (i.e., redband trout absent) were generally interspersed among sites where redband trout were detected. Consequently, delineation of upper or lower distribution limits (i.e., continuous portions of the sample frame where redband trout were absent) rarely occurred, with some possible exceptions. For example, an upper distribution may have been identified in the Silver-FR population and lower distributions may have been identified in the Thomas-Bauers and the Riddle populations (Figure 1.9). Additionally, relatively large areas of habitat where redband trout were not detected within the study system were observed (e.g., within the Drews, Deep, and Honey populations; Figure 1.9).

Differences in Redband Trout Density Between Annual Sites and Non-Annual Sites – Redband trout density at annual sites was similar to redband trout density at non-annual sites for most population by year combinations (Supplemental Figure 2.2). Redband trout density differed significantly between annual sites and non-annual sites for the Silver-FR population in 2009 (t = -2.33, df = 8.99, p = 0.04), the Silver population in 2011 (t = -2.36, df = 26.27, p = 0.03),

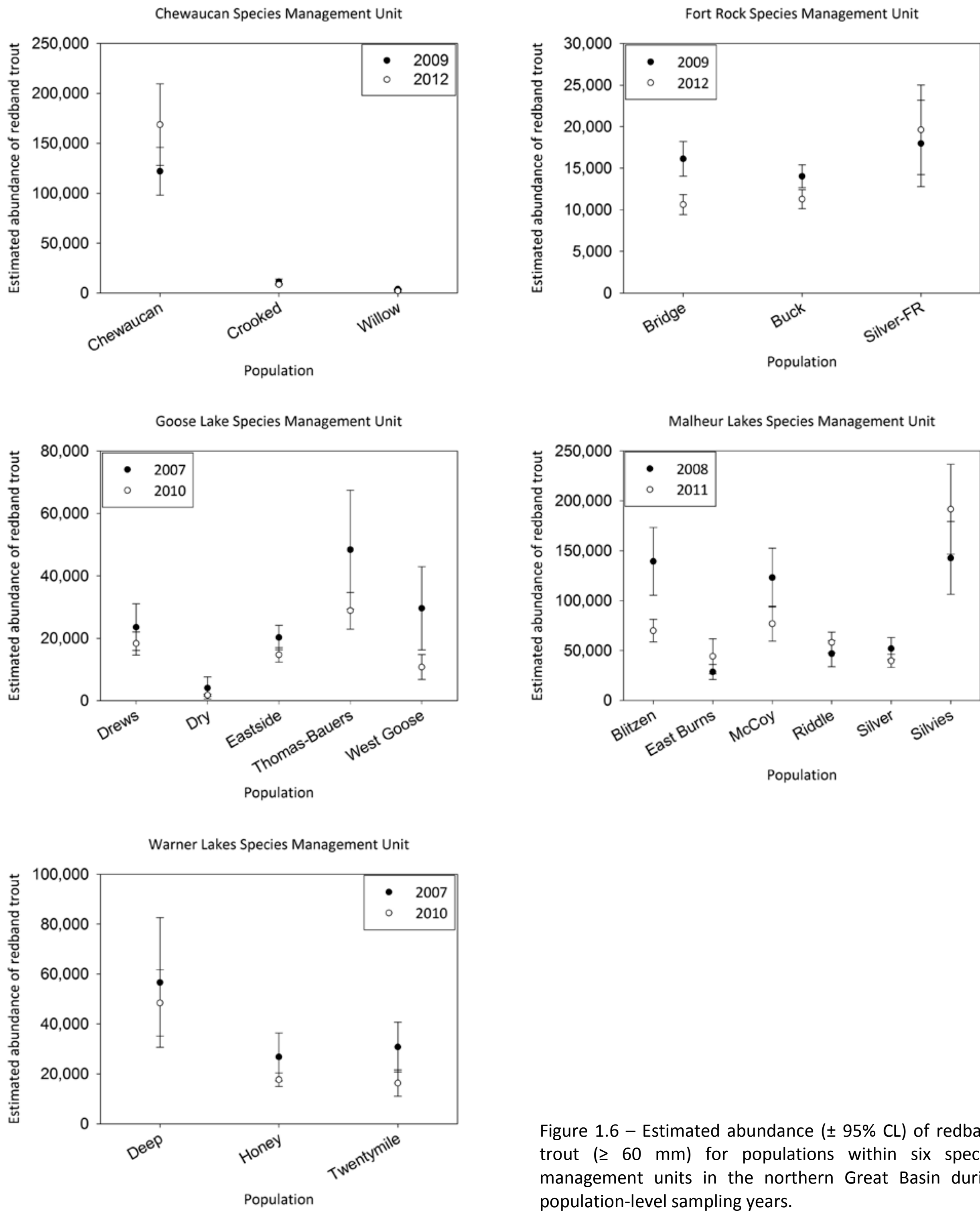

Figure 1.6 – Estimated abundance (± 95% CL) of redband trout (≥ 60 mm) for populations within six species management units in the northern Great Basin during population-level sampling years.

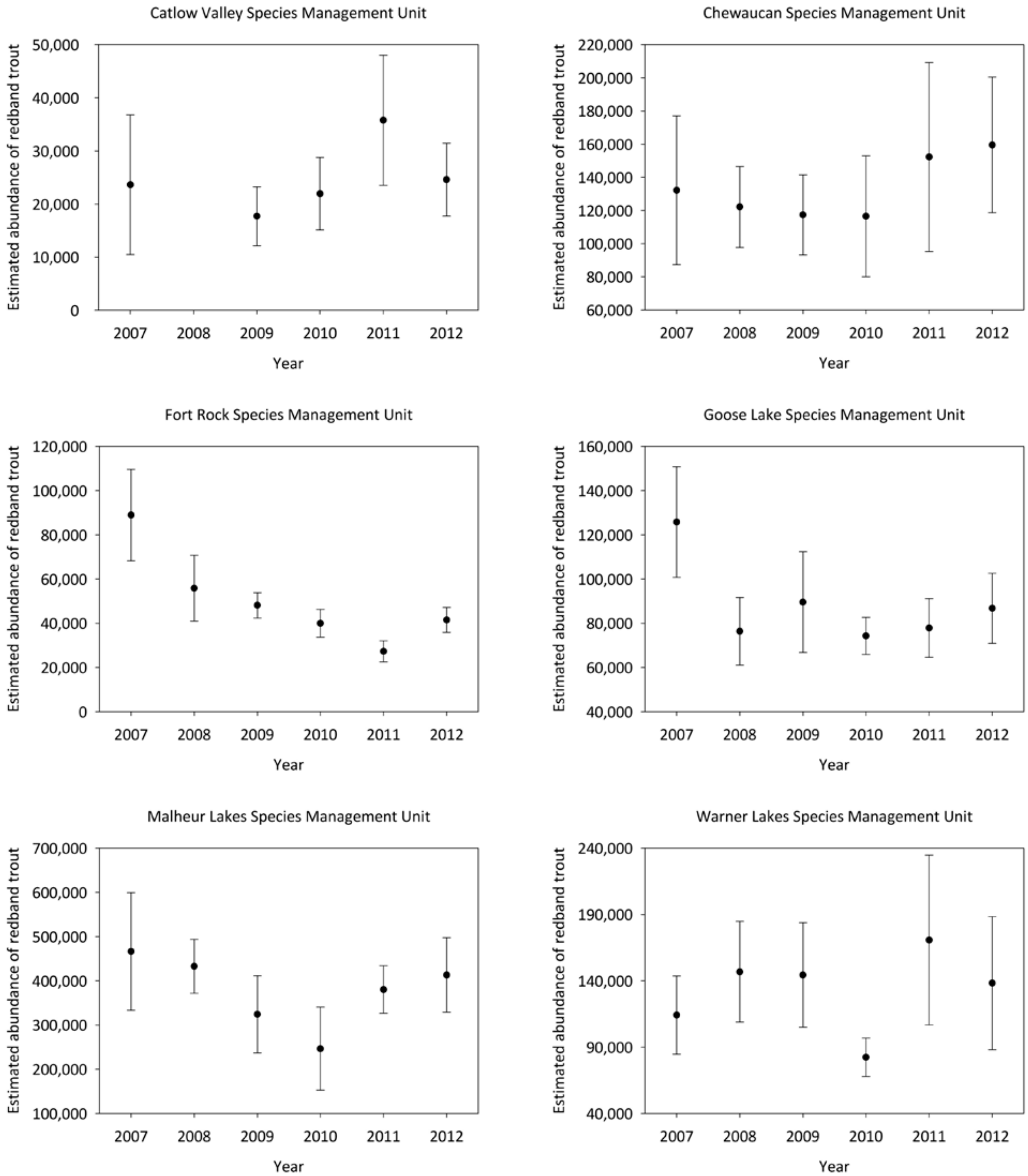

Figure 1.7 – Estimated abundance (± 95% CL) of redband trout (≥ 60 mm) in six species management units from 2007 through 2012 in the northern Great Basin.

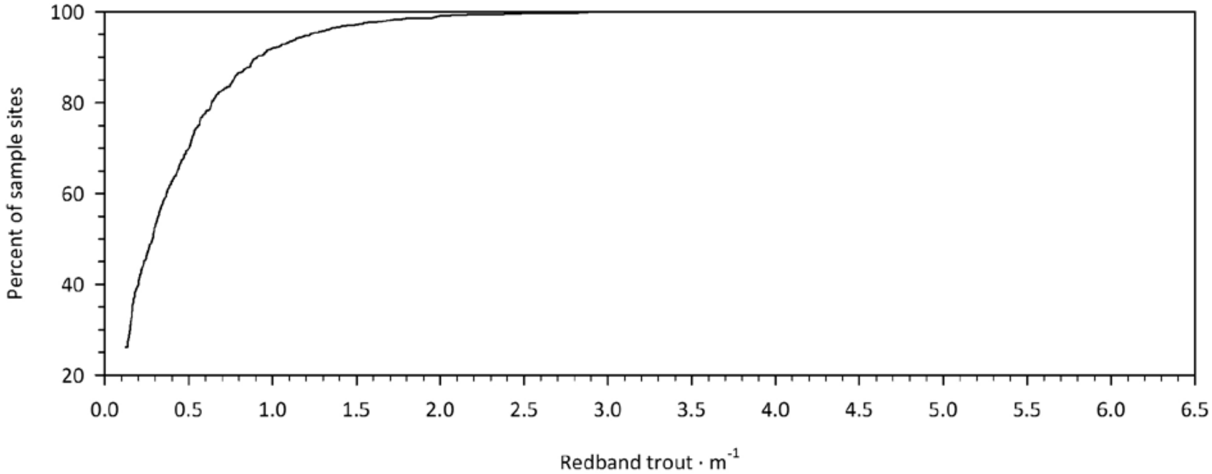

Figure 1.8. Cumulative frequency distribution of redband trout density for 1,500 sample sites examined in the northern Great Basin from 2007 through 2012.

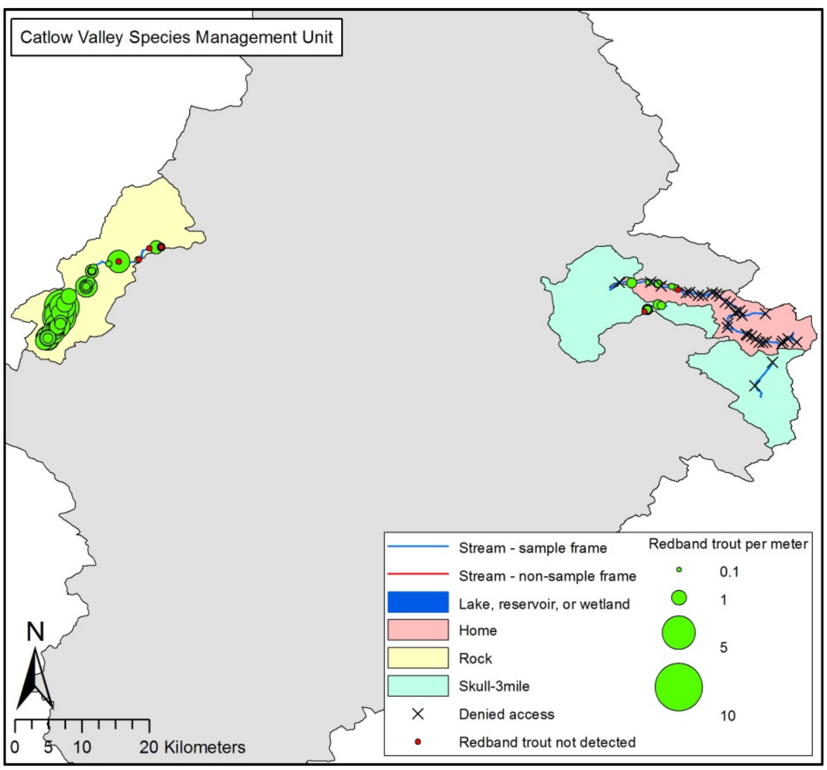

Figure 1.9 – Populations within species management units and sample frame designation; non-sample frame stream segments are provided to illustrate hydrologic connectivity. Sites where we were denied access are represented by an X, density of redband trout (fish·m^{-1}) is represented by green or red circles; the size of the circles is proportional to redband trout density. Sites where redband trout were detected are represented by a green circle. Sites where redband trout were not detected are represented by a red circle; we assumed that redband trout were present, but not captured at these sites and calculated density (0.12 fish·m^{-1}) based on the relationship between depletion density estimates and mark-recapture density estimates ($Density = 0.12 + 1.24 \cdot Density_{Depletion}$).

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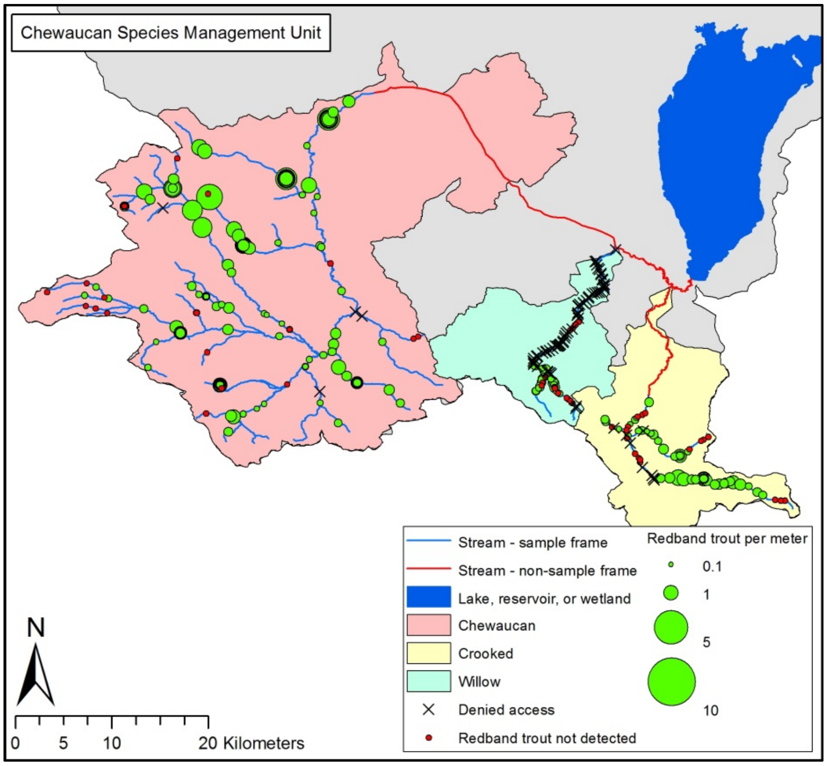

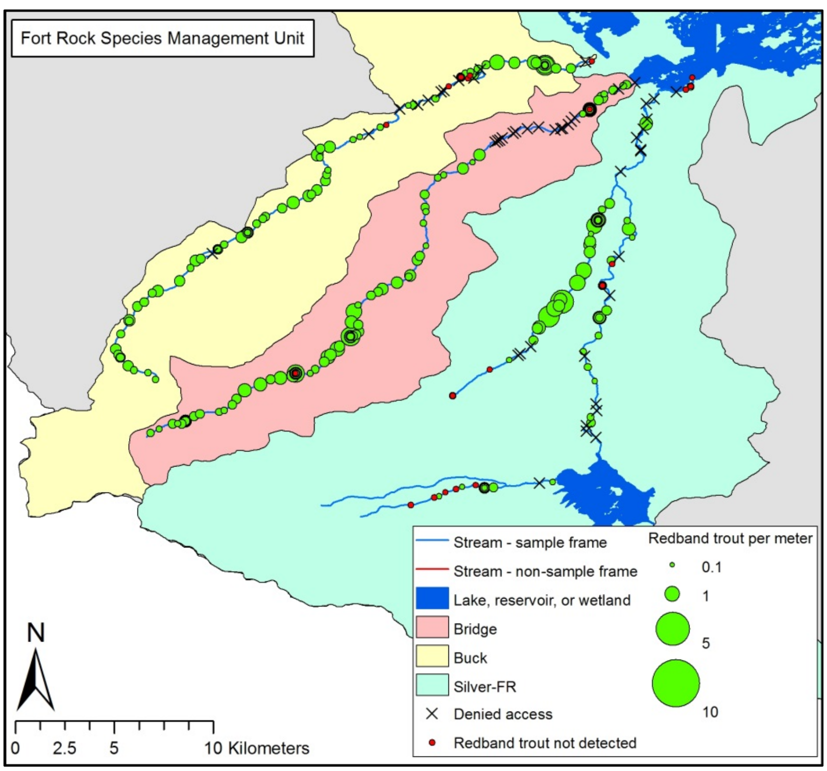

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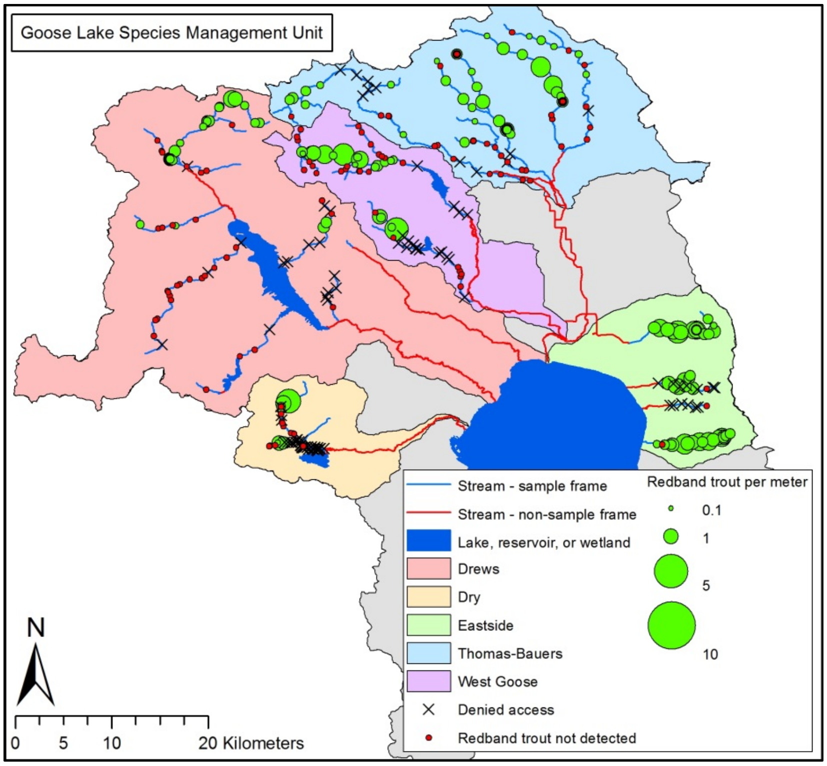

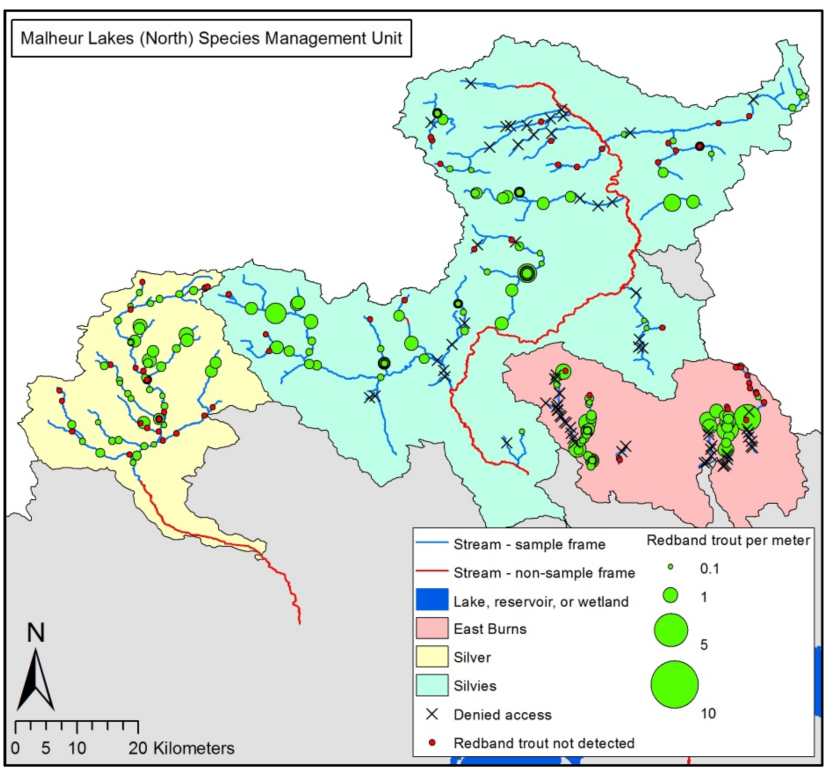

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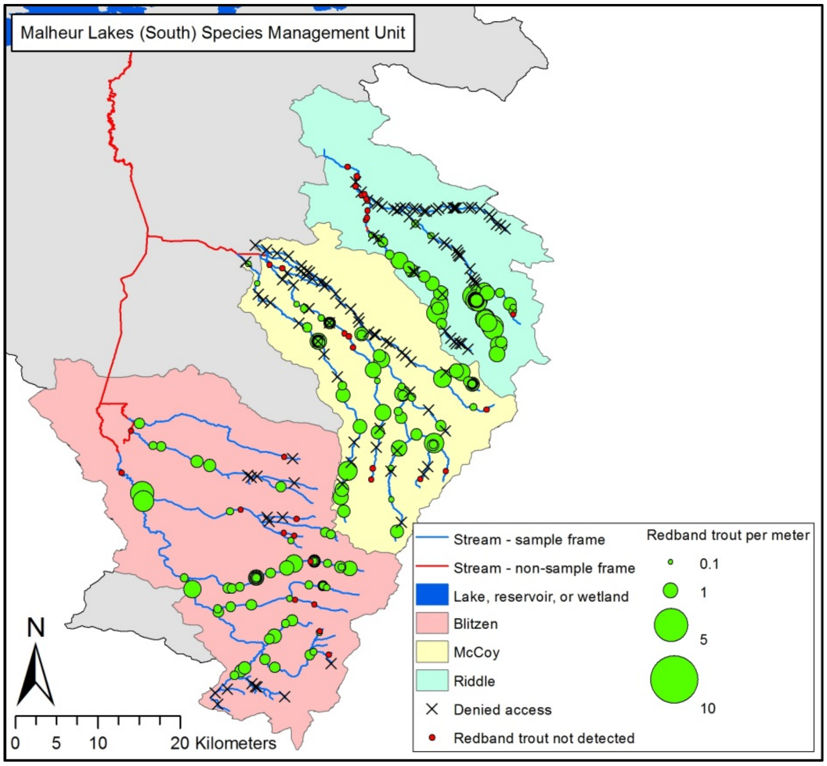

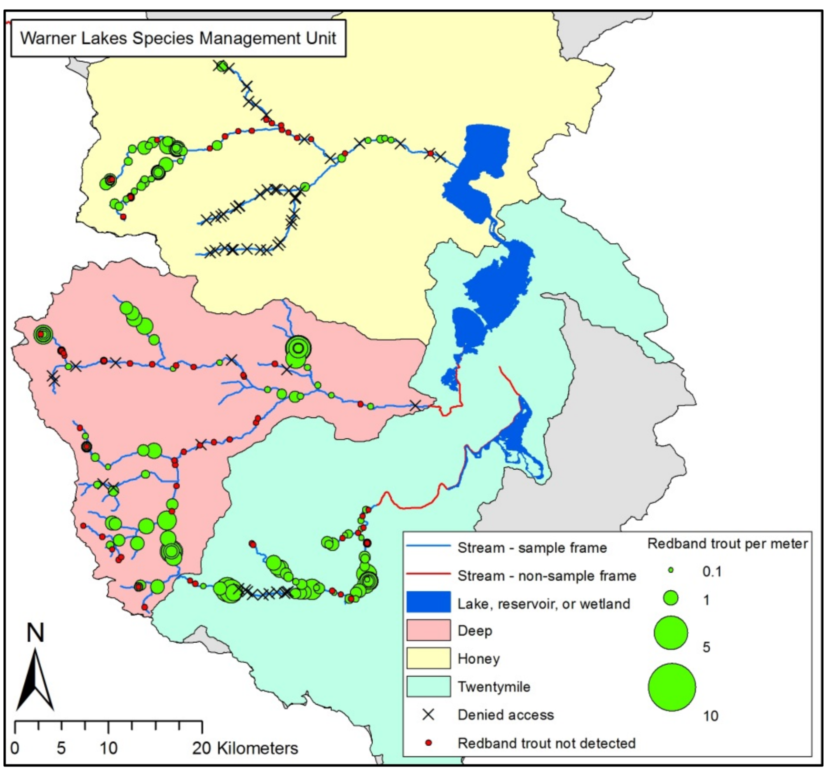

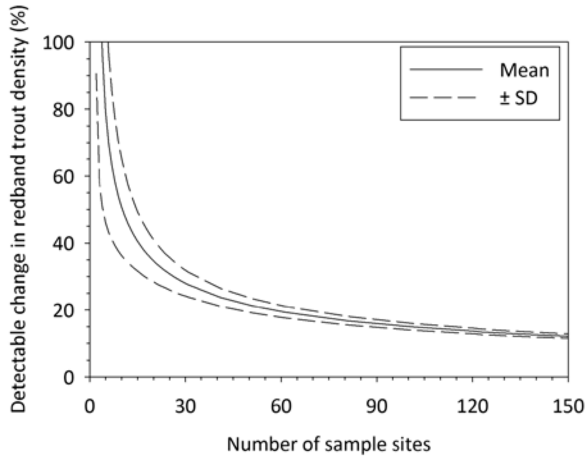

Figure 1.10 – Predicted detectable change in redband trout density (%) at the population level as a function of sampling intensity.

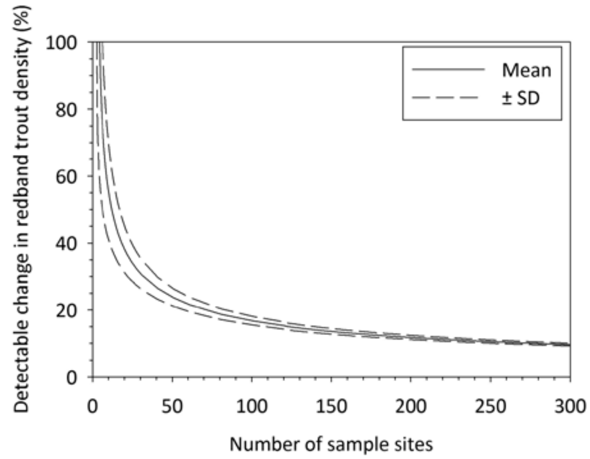

Figure 1.11 – Predicted detectable change in redband trout density (%) at the species management unit level as a function of sampling intensity.

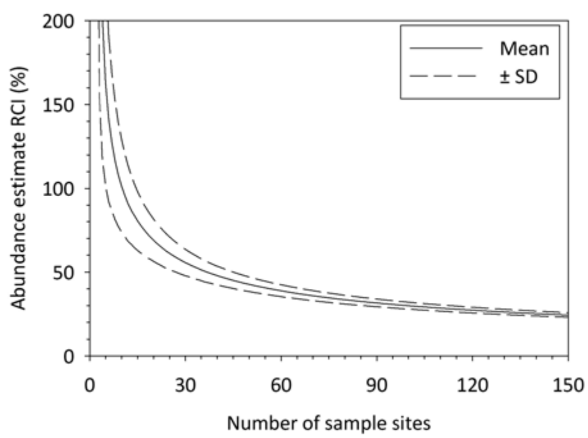

Figure 1.12 – Predicted relative confidence index (RCI) for population-level abundance estimates as a function of sampling intensity.

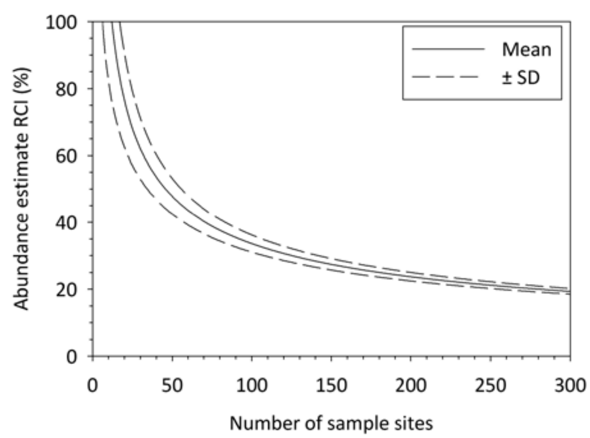

Figure 1.13 – Predicted relative confidence index (RCI) for species management unit level abundance estimates as a function of sampling intensity.

and the Silvies population in 2011 (t = -2.29, df = 23.99. p = 0.03). Insufficient sample sizes (i.e., N < 2) for annual sites or non-annual sites precluded the use of t-tests to compare mean differences in some instances (see Table 1.3 for sample sizes).

Redband trout density at annual sites was similar to redband trout density at non-annual sites for most SMU by year combinations (Supplemental Figure 2.3). Redband trout density differed significantly between annual sites and non-annual sites for the Goose Lake SMU in 2008 (t = 2.50, df = 16.28, p = 0.02).

Predicted Detectable Change in Density – The percent change in redband trout density that could be detected decreased rapidly from greater than 100% to about 30% as the number of sample sites increased from 2 to 30 at the population level (Figure 1.10). Sampling intensities of greater than about 60 sample sites provided little improvement in our ability to detect changes in abundance of redband trout at the population level. Similar patterns were observed for detecting changes in redband trout abundance at the SMU level (Figure 1.11). Large improvements in our ability to detect changes in abundance occurred when increasing the number of sample sites from 2 to about 50. Sampling intensities of greater than about 100 sample sites provide little improvement in our ability to detect changes in abundance of redband trout at the SMU level. Detectable changes in redband trout as a function of sampling intensity varied among populations and SMUs (Supplemental Figure 2.4, 2.5)

Predicted Relative Confidence Index of Density Estimates – Predicted RCIs for abundance estimates at the population level decreased rapidly as the number of sample sites increased from two to about 20 sites among all populations and years examined (Figure 1.12). Overall, 16 sample sites would be required to achieve a RCI of 80% or less at the population level; however, estimates for specific populations and SMUs varied (Supplemental Figure 2.6).

Predicted RCIs for abundance estimates decreased rapidly as the number of sample sites increases from two to about 100 among SMUs and years (Figure 1.13). On average a sample size of 71 sample sites would be required to achieve a RCI of 40% or less at the SMU level. More or less sampling intensity would be required to achieve the desired level of precision for some SMUs (Supplemental Figure 2.7).

Predicted Variability of Redband Trout Density Estimates – On average, predicted density of redband trout was consistent regardless of sample size (i.e., number of sample sites); however, variability in estimated density was relatively large for small sample sizes among all populations and years (Figure 1.14). Variability among density estimates decreased rapidly from a sample size of 2 to a sample size of about 12. Similar results were observed among all SMUs and years combined (Figure 1.15). These trends were similar among individual populations and SMUs with the exception that variability often decreased more rapidly with the addition of sample sites (Supplemental Figure 2.8, 2.9).

Discussion

Redband trout abundance was relatively consistent among years at the population and SMU levels and redband trout were generally present throughout the study area (Figures 1.3, 1.6, and 1.7). A decrease in abundance was observed in the Fort Rock SMU from 2007 through 2011; however, an increase in abundance was observed in 2012. The present analysis does not provide a causal mechanism for this observed trend; however, brook trout (*Salvelinus fontinalis*), which have been shown to negatively interact with *O. mykiss* (e.g., Miller et al. 2013), are present in the Fort Rock SMU. Further evaluation of the relationship between redband trout abundance and distribution and brook trout status (e.g., abundance, distribution, biomass) is warranted.

Data reported here are not directly comparable to results presented in previous studies; however, qualitative comparisons can be made after noting some caveats. Dambacher et al. (2009) did not provide correction for bias associated with depletion electrofishing, and a different bias correction method was used in the present study compared to Miller et al. (2010). Miller et al. (2010) used a constant calibration factor for correcting abundance estimates; however, we observed a linear relationship between depletion density estimates and mark-recapture density estimates with parameter estimates (slope and intercept) that were significantly different from zero. Therefore, the model we used predicted that bias between depletion density estimates and mark-recapture density estimates was dependent on the value of the depletion density estimate (i.e., not constant among all depletion density estimate values). Additionally, because we applied this model to calculate corrected density estimates, all sample

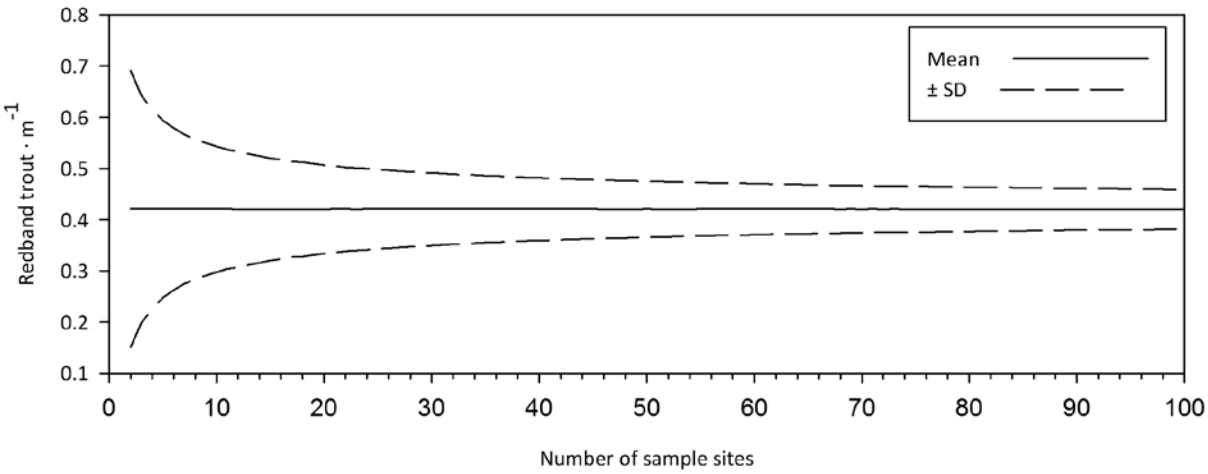

Figure 1.14 – Predicted density of redband trout (± SD) for different numbers of sample sites based on resampling abundance data at the population level and averaged among populations and years.

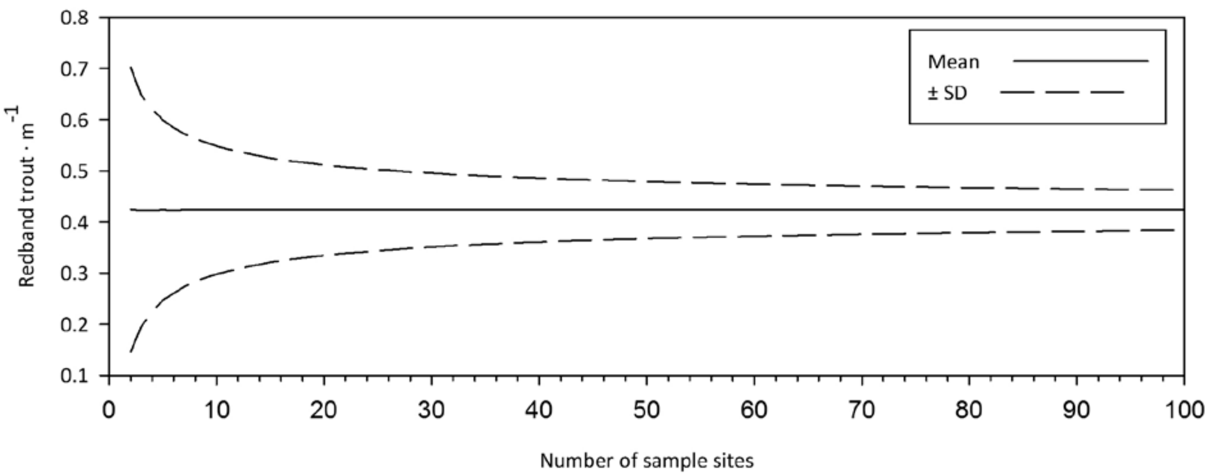

Figure 1.15 – Predicted density of redband trout (± SD) for different numbers of sample sites based on resampling abundance data at the species management unit level and averaged among populations and years.

sites where no redband trout were observed were estimated to have redband trout densities of 0.12 redband trout·m^{-1} (Figure 1.5); whereas calculations used by Miller et al. (2010) inherently assumed that sample sites where no redband trout were observed did not have redband trout.

The probability of capturing any redband trout individual by way of backpack electrofishing is less than 1.0 (as evidenced by depletion and mark-recapture data; this study); therefore, failing to detect redband trout may not be indicative of unoccupied habitat. However, it is also likely that some portion of the sample sites where redband trout were not detected truly were unoccupied. Consequently, alternative methods for identifying and correcting for bias associated with depletion electrofishing are necessary and critical improving our understanding of redband trout distribution and abundance. Detection probabilities for redband trout could be calculated from mark-recapture data and could be used to estimate bias associated with depletion density estimates. However, in the present study mark-recapture data were not collected for sample sites where less than 10 redband trout were detected during the marking event; therefore, if detection probability is influenced by abundance (e.g., McCarthy et al. 2013) then we cannot accurately estimate detection probabilities for a large proportion of the sites where redband trout abundance was estimated (e.g., less than 10 redband trout were sampled on the first pass for 61.3% of sites sampled). Additional studies and further analysis should be conducted to determine the bias associated with depletion density estimates over a broad range of environmental conditions, redband trout sizes, and redband trout abundances if depletion methods are to be used for management purposes.

In addition to differences associated with bias correction, previous studies (Dambacher et al. 2009; Miller et al. 2010) estimated abundance of age-1+ redband trout, where age estimates were based on visual interpretation of length-frequency histograms. Estimating the division between age-0 and age-1+ redband trout based on visual interpretation of length-frequency histograms may be relatively accurate in some instances, but is likely more subjective in other cases (e.g., Figure 1.16), and is subject to underlying assumptions (see Devries and Frie 1996). Additionally, mark-recapture abundance estimates were calculated for redband trout ≥ 60 mm; therefore, bias correction factors were also based on samples of redband trout ≥ 60 mm. Consequently, we used this size cutoff as opposed to a more subjective estimate of redband trout age.

Regardless of methodological differences, some trends in abundance among SMUs were similar between the present study and Dambacher et al. (2009); specifically, abundance was generally greatest in the Malheur Lakes SMU, lowest in the Catlow Valley and Fort Rock SMUs, and intermediate in the Chewaucan, Goose Lake, and Warner Lakes SMUs.

Precision criteria for abundance estimates were met 92.5% of the time at the population level and 46.7% of the time at the SMU level. Overall, we predicted that 16 sites at the population level and 71 sites at the SMU level would need to be sampled to meet precision criteria. These results differ greatly from Miller et al. (2010) who found that precision criteria were met 9.5% of the time at the population level from 2007 through 2009. However, much of this difference can be attributed to differences in the methods used to correct for bias associated with depletion electrofishing (see above). Specifically, the model used to correct for bias associated with depletion electrofishing in the present analysis estimated that 0.12 redband trout·m^{-1} were present at sample sites where redband trout were not detected. This correction can substantially reduce variability compared to a bias correction method that inherently assumes that sites where redband trout were not detected are unoccupied.

The benefit of achieving precision criteria in a management context is uncertain. Although precision of abundance estimates and the ability to detect changes in abundance (or density) are inextricably linked (i.e., both are dependent on sample size and variability in the data), the latter may be more useful for triggering conservation or management actions. For example, to meet abundance criteria for SMUs in Oregon, Goodson et al. (2005*a*; *b*) state that, '*the number of naturally-produced fish is greater than 25% of average levels in at least three of the last five years for at least 80% of existing populations*'. This criterion requires the ability to detect percent changes in populations as opposed to measuring how precise abundance estimates are; therefore, we suggest that results of this study should be used to identify sample sizes necessary to detect changes in redband trout density based on policy decisions and the management needs of natural resource managers.

The present study provides information on

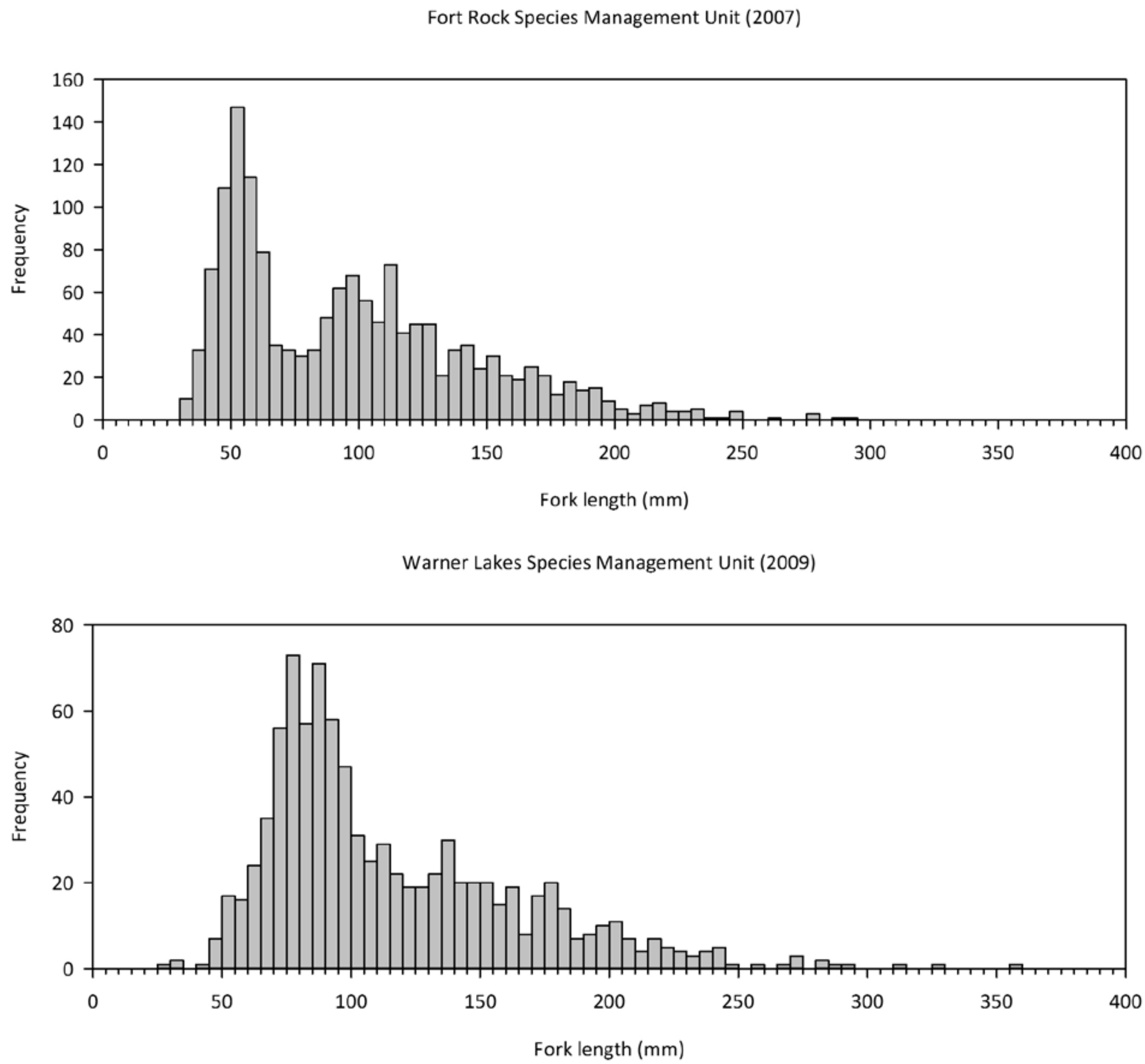

Figure 1.16 – Example length frequency histograms illustrating data with a relatively well-defined separation between age-0 and age-1+ redband trout (top panel; Fort Rock species management unit sampled in 2007) and data with a less well-defined separation between age-0 and age-1+ redband trout (bottom panel; Warner Lakes species management unit sampled in 2009).

The distribution of redband trout within the sample frame. However, the study design does not provide information on distribution limits or changes in distribution limits over time within the northern Great Basin. The original sample frame consisted of wadeable streams known or suspected to be occupied by redband trout within the northern Great Basin. Therefore, perennial or ephemeral use of non-wadeable streams by redband trout was not documented. Defining the sample frame based on known or suspected distribution was necessary given the large spatial extent of the northern Great Basin, but precluded the potential to sample areas of unknown occupancy. Additionally, the sample frame for this study was reduced in some circumstances when no redband trout were detected after one or two years of sampling; for example, the sample frame for the Eastside population was reduced after 2007 and the sample frames for the Deep, Dry, and Silver-FR populations were reduced after 2008. However, in other instances the sample frame was not reduced when redband trout were not detected;

for example, Hay Creek in the Drews population was sampled annually and redband trout were never detected. Reducing the sample frame after one or two years of sampling precludes the ability to detect colonization of previously unoccupied habitat. Furthermore, detection probability of redband trout by way of backpack electrofishing is less than 1.0 (as above). Consequently, distribution data presented in this study must be considered as the minimum distribution of redband trout. In addition to limiting our understanding of distribution limits, it is highly likely that sampling only wadeable streams results in an underestimate of redband trout abundance in the northern Great Basin, and may result in failure to detect migratory individuals. Consequently, alternative sample frame selection methods will be necessary if conservation or management strategies require detailed distribution data, information on changes in distribution over time, or comprehensive data on abundance of redband trout in the northern Great Basin.

The panel design used here included a combination of sample sites that were revisited annually (annual sites) and that were serially alternating (non-annual sites). In general, annual sites provided statistically similar density estimates for redband trout when compared to non-annual sites. However, care should be taken when interpreting the representativeness of annual sites for estimating abundance at the population-level due to low sample sizes within many populations (see Table 1.3). Resampling analyses based on random selection of sample sites from the empirical data indicated that, on average, point estimates for redband trout density were similar regardless of the number of sites sampled. However, sample size was important in reducing variability associated with density estimates. Variability among density estimates for a given sample size was relatively high for sample sizes ≤ 10, moderate for sample sizes from 10 to 20, and changed little for sample sizes > 20 sample sites at the population- and SMU-level. Therefore, adherence to sampling a set of reference sites may not be necessary if obtaining a point estimate of density is of primary interest; however, sufficient sample sizes should be used to reduce uncertainty in the representativeness of density estimates.

A GRTS design was used in this study to select a spatially well-distributed probability sample. However, in some instances we were denied access to large, spatially continuous portions of the sample frame (e.g., the East Burns, Home, McCoy, Riddle, and Willow populations), and in these instances the resulting sample of sites exhibits a clumped distribution (Figure 1.9). The method used to extrapolate abundance estimates to the population and SMU levels assumes that redband trout densities in areas where we were able to sample are representative of areas where we were denied access. The validity of this assumption is unknown. Although no consistent patterns were observed with respect to the spatial distribution of redband trout density among populations, SMUs, and years, gradients in density may be present within populations or population by year combinations. If spatial gradients in redband trout density do exist at some temporal or spatial scale then spatially explicit models (e.g., Peterson and Ver Hoef 2010) may be used to refine abundance estimates.

The role of environmental variability on patterns of distribution and abundance of redband trout in the northern Great Basin must also be considered. Redband trout in the northern Great Basin occupy a landscape characterized by substantial inter and intra-annual environmental variability. The present study occurred over a six-year time-span that included both below average and above average water years. Redband trout density estimates are likely influenced by this variability as well as time of sampling. For example, redband trout density was highest among all SMUs and years in the Catlow Valley SMU during 2007, which was also one of the driest years during the study. The Rock population was the only population sampled in the Catlow Valley SMU in 2007 and it had an average density of 2.46 redband trout·m^{-1};

densities varied from 0.34 to 5.51 redband trout·m^{-1} (the highest observed in the study). However, 67% of the selected sample sites in this population were dry during 2007. It is likely that during dry years, or during the driest period within a year, redband trout are concentrated into wetted and suitable habitat as streams begin to dry. The influence of drying habitat may have little effect on extrapolation of abundance estimates if sampling is conducted over a short time span. However, if sampling is conducted over a time span of sufficient length that habitat availability is variable then extrapolation of site-specific abundance estimates to the population or SMU level may bias results. This bias may be compounded if sites are visited in an order based on ease-of-sampling or known patterns of drying. For example, it may be tempting to sample sites early in the sample season that are known to dry up late in the sample season or to avoid high water sites early in the sample season in anticipation that they may be sampled as streams begin to dry later in the sample season. This type of strategy may maximize the number of sites for which data are gathered and reduce the chance of visiting dry sites or having failed estimates. However, if fish are moving in response to changing environmental conditions, this type of strategy has the potential to bias abundance estimates. Therefore, future studies should evaluate intra-annual distribution patterns of redband trout and how movement of individuals in response to changing environmental conditions may influence abundance estimates.

In conclusion, redband trout abundance was relatively consistent within populations and SMUs over the time period of 2007 through 2012, with few exceptions, and redband trout were distributed throughout much of the sample frame. Based on the prescribed level of sampling intensity, abundance estimates met precision criteria for 92.5% of the time at the population level and 46.7% of the time at the SMU level. We estimated that 16 and 71 sites would need to be sampled at the population and SMU levels, respectively, in order to achieve desired precision of abundance estimates. However, detectable changes in abundance of redband trout are more likely to trigger conservation and management actions than uncertainty in abundance estimates. Therefore, we suggest that resource managers make decisions on required sampling intensity based on the relationship between sampling intensity and ability to detect changes in abundance. Additionally, point estimates of redband trout density are relatively consistent among random sets of sample sites with sufficient sampling intensity. Therefore, strict adherence to sampling a set of reference sites may not be necessary to estimate redband trout density or abundance if obtaining a point estimate is the primary objective of sampling; however, if insufficient sampling intensity is applied then density or abundance estimates are likely to be imprecise, which may result in poor management decisions. We suggest that the sample frame for future surveys should be increased to include areas beyond the known distribution of redband trout in wadeable streams if distribution or changes in distribution of redband trout are of interest, and that spatial trends in density of redband trout should be evaluated with respect to assumptions associated with extrapolating density estimates to population and SMU levels. Depletion estimates are often easy to obtain, but are known to be systematically biased with respect to estimating density and abundance of salmonids (Peterson et al. 2004) and the method used to correct for bias associated with depletion electrofishing may have a substantial influence on the results of monitoring. Consequently, further research and analyses should be conducted to better understand factors influencing this bias if depletion electrofishing is to be used in the future for monitoring redband trout in the northern Great Basin.

Finally, this study relied on depletion and mark-recapture methodologies to estimate abundance of redband trout. These methods are often time-consuming, expensive, and often imprecise, but can be beneficial if information on abundance or density is needed for species management or conservation. However, abundance or density information may not always be of primary concern to resource managers. For example, resource managers may be primarily interested in distribution or occupancy across a species distribution. In such a case, conducting depletion or mark-recapture surveys may be unnecessary and a well-designed survey aimed at detecting presence or absence may be favored. Additionally, emerging technologies offer promise for reducing time, effort, and costs associated with species monitoring. For example, screening water bodies for target-species environmental DNA (e.g., Thomsen et al. 2012) may provide similar information to that provided by more traditional surveys aimed at detecting presence or absence (e.g., electrofishing or snorkel surveys), but with reduced field times and likely cost. Alternatively, genetic monitoring may provide detailed information on the number of breeders in a population, the genetic diversity of a population, and other genetic characteristics of a population (e.g., hybridization status, population genetic structure, etc.) at less cost than a tradition depletion of mark-recapture survey (Schwartz et al. 2007). Consequently, we suggest that resource managers identify their data requirements and select monitoring or survey designs and techniques that best meet their needs.

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PART 2:

SUPPLEMENTAL MATERIAL FOR: USE OF DEPLETION ELECTROFISHING AND A GENERALIZED RANDOM-TESSELLATION STRATIFIED DESIGN TO ESTIMATE DENSITY AND ABUNDANCE OF REDBAND TROUT IN THE NORTHERN GREAT BASIN

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Abstract – Supplemental tables and figures are available to provide detailed information related to sampling redband trout in the northern Great Basin. Briefly, Oregon Department of Fish and Wildlife performed depletion electrofishing following a generalized random-tessellation stratified design to estimate the density, abundance, and distribution of redband trout among 20 populations within six species management units (SMUs) in the northern Great Basin from 2007 through 2012.

Species management units were designated as Catlow Valley, Chewaucan, Fort Rock, Goose Lake, Malheur Lakes, and Warner Lakes following (Goodson et al. 2005*a*; *b*). Population designations generally followed (Goodson et al. 2005*a*; *b*), with the exception that some populations identified by Goodson et al. (2005*a*; *b*) were grouped into a single population due to small spatial extent or logistical constraints associated with achieving high enough sampling intensities to meet project objectives. Specifically, the Skull and Threemile populations were combined into a single population (Skull-Threemile), the Lower Drews and Upper Drews populations were combined into a single population (Drews), the Crane, Cogswell, Kelley, and Tandy populations were combined into a single population (Eastside), the Antelope, Cottonwood, and Muddy populations were combined into a single population (West Goose), the Coffeepot, Cow, Poison, Prater, and Rattlesnake populations were combined into a single population (East Burns), and the Lower Deep and Upper Deep populations were combined into a single population (Deep).

Supplemental tables and figures are presented by population, SMU-level, and, in some instances, sample year. This supplemental material will be useful for conservation planning and redband trout management purposes, which may be targeted towards specific populations. However, this supplemental material will be of less interest to individuals interested in overall patterns of redband trout density, abundance, and distribution in the northern Great Basin.

List of Supplemental Tables

Supplemental Table 2.1 – Number of sites where depletion criteria were achieved (estimate passed), where depletion criteria were not achieved (estimate failed), that were not surveyed, where we were denied access, and that were dry by species management unit (SMU), population, and year.

SMU	Population	Year	Estimate passed	Estimate failed	Not surveyed	Denied access	Dry channel
Catlow Valley	Home	2007	1	0	1	13	0
		2008					
		2009	1	0	2	0	0
		2010					
		2011	4	0	0	11	0
		2012	3	0	0	6	0
	Rock	2007	7	0	1	0	14
		2008					
		2009	7	1	5	0	2
		2010	11	0	0	0	0
		2011	11	0	0	0	0
		2012	4	0	0	0	12
	Skull-3mile	2007					
		2008					
		2009	2	0	0	0	0
		2010	2	0	0	0	0
		2011	4	0	0	0	0
		2012					
Chewaucan	Chewaucan	2007	24	3	0	2	1
		2008	18	1	0	0	3
		2009	24	2	3	1	2
		2010	24	0	0	1	3
		2011	22	1	2	0	1
		2012	29	0	0	0	1
	Crooked	2007	4	0	4	4	0
		2008	3	0	0	0	0
		2009	22	2	2	3	1
		2010	3	0	0	0	0
		2011	4	0	0	0	0
		2012	27	0	0	1	3
	Willow	2007	3	0	2	7	0
		2008	3	0	0	7	3
		2009	19	0	0	21	10
		2010	3	0	0	4	2
		2011	2	0	0	2	0
		2012	9	0	0	21	20
Fort Rock	Bridge	2007	6	0	0	4	0
		2008	7	0	1	0	0
		2009	21	1	3	4	0
		2010	8	0	0	0	0
		2011	7	0	0	1	0
		2012	30	0	0	7	0

Supplemental Table 2.1 – Continued on next page.

Supplemental Table 2.1 – Continued from previous page.

SMU	Population	Year	Estimate passed	Estimate failed	Not surveyed	Denied access	Dry channel
Fort Rock	Buck	2007	9	0	1	4	0
		2008	7	0	0	1	0
		2009	19	2	2	0	2
		2010	10	0	0	0	0
		2011	8	2	0	0	0
		2012	29	1	0	6	0
	Silver-FR	2007	12	3	3	2	3
		2008	10	0	0	1	1
		2009	13	4	3	3	1
		2010	9	1	4	0	0
		2011	6	2	9	0	0
		2012	23	1	5	15	0
Goose Lake	Drews	2007	20	1	0	7	3
		2008	6	0	0	1	3
		2009	7	1	3	2	3
		2010	20	0	0	5	1
		2011	8	0	0	1	0
		2012	7	0	0	0	1
	Dry	2007	12	0	0	13	15
		2008	2	0	0	0	2
		2009	2	0	0	0	0
		2010	4	0	8	18	6
		2011	2	0	0	0	0
		2012	2	0	0	1	0
	Eastside	2007	17	0	0	11	14
		2008	2	0	0	0	1
		2009	3	0	0	0	0
		2010	21	0	0	2	4
		2011	3	0	1	0	0
		2012	3	0	0	0	1
	Thomas-Bauers	2007	20	1	0	4	4
		2008	10	0	0	1	1
		2009	12	0	0	1	0
		2010	22	0	2	4	1
		2011	12	0	0	1	0
		2012	12	0	0	0	1
	West Goose	2007	19	2	0	6	3
		2008	4	0	0	0	0
		2009	4	0	0	2	4
		2010	20	0	1	7	1
		2011	4	0	0	1	0
		2012	5	0	0	1	0
Malheur Lakes	Blitzen	2007	7	0	1	1	0
		2008	23	0	1	5	1
		2009	6	0	0	1	1
		2010	5	0	3	1	1
		2011	26	0	13	7	1
		2012	6	0	1	1	1

Supplemental Table 2.1 – Continued on next page.

Supplemental Table 2.1 – Continued from previous page.

SMU	Population	Year	Estimate passed	Estimate failed	Not surveyed	Denied access	Dry channel
Malheur Lakes	East Burns	2007	3	0	0	8	1
		2008	25	0	1	10	5
		2009	2	0	0	0	2
		2010	3	0	0	3	0
		2011	25	0	1	16	1
		2012	3	0	0	3	1
	McCoy	2007	6	0	0	16	0
		2008	24	0	3	6	1
		2009	5	0	0	1	0
		2010	3	0	0	3	0
		2011	25	1	3	15	1
		2012	6	0	0	12	0
	Riddle	2007	3	0	0	3	0
		2008	18	0	1	7	2
		2009	2	0	1	3	2
		2010	3	0	0	0	0
		2011	25	0	3	26	3
		2012	3	0	0	15	3
	Silver	2007	6	0	0	0	0
		2008	25	0	0	0	4
		2009	3	2	2	0	2
		2010	6	0	0	0	0
		2011	30	0	0	0	0
		2012	6	0	0	0	1
	Silvies	2007	14	0	1	17	2
		2008	23	1	0	3	1
		2009	13	0	11	2	4
		2010	15	1	0	1	0
		2011	26	0	1	6	0
		2012	15	0	2	2	0
Warner Lakes	Deep	2007	18	1	1	3	7
		2008	13	0	0	3	3
		2009	16	1	0	3	1
		2010	30	0	1	1	1
		2011	16	0	2	0	0
		2012	14	0	0	0	4
	Honey	2007	18	0	0	13	2
		2008	7	0	1	2	0
		2009	10	0	1	3	0
		2010	20	0	1	18	3
		2011	9	0	0	3	1
		2012	9	0	0	2	0
	Twentymile	2007	18	1	0	4	4
		2008	4	0	1	0	0
		2009	4	0	0	0	0
		2010	29	0	2	6	5
		2011	4	0	3	0	0
		2012	3	0	0	0	1
		Sum	**1500**	**40**	**130**	**496**	**227**

Supplemental Table 2.2 – Sample size (*N*), estimated mean abundance, lower 95% confidence limit (L 95% CL), and upper 95% confidence limit (U 95% CL) by species management unit (SMU), population, and year. Confidence limits were calculated using a local mean variance estimator (when *N* > 3) and a simple random sample (SRS) variance estimator.

				Local mean variance estimator			SRS variance estimator		
SMU	Population	Year	*N*	Mean	L 95% CL	U 95% CL	Mean	L 95% CL	U 95% CL
Catlow Valley	Home	2007	1	.	.	.	.	.	.
		2008	0	.	.	.	.	.	.
		2009	1	.	.	.	.	.	.
		2010	0	.	.	.	.	.	.
		2011	4	7,603	5,972	9,233	7,603	5,657	9,548
		2012	3	.	.	.	7,913	4,088	11,737
	Rock	2007	7	23,638	10,491	36,786	23,638	9,118	38,159
		2008	0	.	.	.	.	.	.
		2009	7	14,725	10,375	19,074	14,725	7,568	21,882
		2010	11	19,632	13,154	26,109	19,632	8,452	30,811
		2011	11	25,391	13,284	37,499	25,391	10,879	39,903
		2012	4	16,676	10,975	22,378	16,676	9,609	23,744
	Skull-3mile	2007	0	.	.	.	.	.	.
		2008	0	.	.	.	.	.	.
		2009	2	.	.	.	2,985	-437	6,406
		2010	2	.	.	.	2,318	202	4,434
		2011	4	2,795	1,852	3,738	2,795	1,661	3,930
		2012	0	.	.	.	.	.	.
Chewaucan	Chewaucan	2007	24	121,451	78,467	164,435	121,451	68,257	174,644
		2008	18	106,662	82,553	130,771	106,662	66,753	146,571
		2009	24	122,040	98,015	146,064	122,040	84,443	159,636
		2010	24	119,901	83,527	156,276	119,901	69,648	170,154
		2011	22	155,568	99,009	212,127	155,568	85,336	225,800
		2012	29	168,776	127,897	209,655	168,776	113,770	223,783
	Crooked	2007	4	20,058	11,203	28,913	20,058	9,575	30,541
		2008	3	.	.	.	31,553	28,736	34,370
		2009	22	11,440	8,894	13,986	11,440	7,684	15,196
		2010	3	.	.	.	13,099	10,404	15,794
		2011	4	8,042	5,524	10,559	8,042	4,890	11,194
		2012	27	8,635	7,675	9,595	8,635	7,108	10,161
	Willow	2007	3	.	.	.	10,717	1,192	20,242
		2008	3	.	.	.	3,912	1,244	6,580
		2009	19	3,866	3,327	4,405	3,866	3,142	4,589
		2010	3	.	.	.	3,472	1,362	5,583
		2011	2	.	.	.	8,669	2,123	15,214
		2012	9	2,154	1,401	2,906	2,154	1,232	3,075
Fort Rock	Bridge	2007	6	31,267	22,664	39,870	31,267	21,527	41,007
		2008	7	14,283	10,074	18,492	14,283	8,890	19,677
		2009	21	16,130	14,058	18,201	16,130	13,231	19,028
		2010	8	12,912	9,619	16,205	12,912	8,886	16,938
		2011	7	8,370	6,931	9,810	8,370	6,022	10,719
		2012	30	10,613	9,404	11,822	10,613	9,128	12,097

Supplemental Table 2.2 – Continued on next page.

Supplemental Table 2.2 – Continued from previous page.

				Local mean variance estimator			SRS variance estimator		
SMU	Population	Year	*N*	Mean	L 95% CL	U 95% CL	Mean	L 95% CL	U 95% CL
Fort Rock	Buck	2007	9	23,543	16,998	30,088	23,543	15,309	31,777
		2008	7	22,138	9,363	34,912	22,138	7,401	36,874
		2009	19	14,031	12,660	15,402	14,031	11,588	16,474
		2010	10	12,290	9,281	15,298	12,290	8,825	15,754
		2011	8	9,591	7,882	11,300	9,591	7,249	11,932
		2012	29	11,275	10,130	12,420	11,275	9,051	13,498
	Silver-FR	2007	12	34,168	16,486	51,849	34,168	12,908	55,428
		2008	10	19,438	13,167	25,709	19,438	11,778	27,099
		2009	13	17,978	12,783	23,173	17,978	12,007	23,949
		2010	9	14,774	10,345	19,203	14,774	9,997	19,552
		2011	6	9,358	5,182	13,534	9,358	4,113	14,603
		2012	23	19,627	14,232	25,023	19,627	12,033	27,222
Goose Lake	Drews	2007	20	23,531	16,072	30,990	23,531	12,866	34,195
		2008	6	16,920	6,160	27,680	16,920	4,111	29,728
		2009	7	18,958	11,340	26,577	18,958	9,454	28,462
		2010	20	18,321	14,590	22,053	18,321	13,103	23,539
		2011	8	19,042	15,005	23,079	19,042	10,888	27,196
		2012	7	16,027	8,721	23,334	16,027	6,645	25,409
	Dry	2007	12	4,036	492	7,581	4,036	-153	8,225
		2008	2	.	.	.	1,379	1,379	1,379
		2009	2	.	.	.	13,141	-4	26,285
		2010	4	1,716	1,375	2,057	1,716	1,315	2,117
		2011	2	.	.	.	7,491	-2,748	17,730
		2012	2	.	.	.	2,520	2,024	3,015
	Eastside	2007	17	20,245	16,312	24,179	20,245	15,369	25,121
		2008	2	.	.	.	20,488	18,934	22,042
		2009	3	.	.	.	21,043	7,382	34,703
		2010	21	14,679	12,365	16,994	14,679	11,844	17,515
		2011	3	.	.	.	15,821	12,335	19,307
		2012	3	.	.	.	28,602	20,157	37,046
	Thomas-Bauers	2007	20	48,406	29,348	67,465	48,406	26,356	70,456
		2008	10	30,334	19,560	41,108	30,334	17,636	43,032
		2009	12	29,326	20,305	38,347	29,326	19,342	39,311
		2010	22	28,824	22,948	34,700	28,824	21,081	36,567
		2011	12	25,900	20,464	31,336	25,900	19,850	31,950
		2012	12	30,282	19,608	40,957	30,282	18,734	41,831
	West Goose	2007	19	29,589	16,227	42,950	29,589	14,014	45,163
		2008	4	7,247	6,635	7,858	7,247	6,527	7,966
		2009	4	7,078	2,648	11,509	7,078	1,742	12,414
		2010	20	10,740	6,720	14,760	10,740	6,188	15,293
		2011	4	9,598	5,855	13,341	9,598	5,066	14,130
		2012	5	9,314	5,757	12,871	9,314	5,009	13,618
Malheur Lakes	Blitzen	2007	7	67,457	38,305	96,608	67,457	32,137	102,777
		2008	23	139,422	105,442	173,402	139,422	90,717	188,128
		2009	6	132,840	87,624	178,057	132,840	80,683	184,997
		2010	5	68,628	37,471	99,784	68,628	30,926	106,329
		2011	26	69,890	58,561	81,219	69,890	57,366	82,413
		2012	6	97,055	55,213	138,896	97,055	46,815	147,294

Supplemental Table 2.2 – Continued on next page.

Supplemental Table 2.2 – Continued from previous page.

				Local mean variance estimator			SRS variance estimator		
SMU	Population	Year	*N*	Mean	L 95% CL	U 95% CL	Mean	L 95% CL	U 95% CL
Malheur Lakes	East Burns	2007	3	.	.	.	25,392	2,976	47,808
		2008	25	28,581	21,065	36,097	28,581	19,252	37,910
		2009	2	.	.	.	10,845	9,872	11,817
		2010	3	.	.	.	19,803	8,528	31,079
		2011	25	44,191	26,708	61,674	44,191	23,425	64,957
		2012	3	.	.	.	25,967	7,504	44,431
	McCoy	2007	6	145,389	102,896	187,881	145,389	95,868	194,909
		2008	24	123,174	93,707	152,642	123,174	87,673	158,675
		2009	5	67,679	38,842	96,517	67,679	35,158	100,201
		2010	3	.	.	.	54,636	1,288	107,984
		2011	25	76,859	59,269	94,448	76,859	54,586	99,131
		2012	6	139,657	88,608	190,706	139,657	74,189	205,125
	Riddle	2007	3	.	.	.	40,910	3,145	78,674
		2008	18	46,922	33,781	60,064	46,922	28,268	65,577
		2009	2	.	.	.	33,332	-14,179	80,843
		2010	3	.	.	.	62,996	1,799	124,194
		2011	25	58,112	47,675	68,549	58,112	37,219	79,004
		2012	3	.	.	.	56,614	29,385	83,843
	Silver	2007	6	86,248	48,504	123,993	86,248	44,035	128,462
		2008	25	51,987	41,087	62,888	51,987	36,367	67,607
		2009	3	.	.	.	33,914	2,491	65,337
		2010	6	23,503	22,609	24,397	23,503	22,498	24,508
		2011	30	39,703	33,067	46,338	39,703	30,036	49,369
		2012	6	28,074	16,399	39,749	28,074	14,836	41,312
	Silvies	2007	14	201,119	93,143	309,095	201,119	74,817	327,421
		2008	23	142,755	106,278	179,232	142,755	97,217	188,293
		2009	13	145,464	107,077	183,851	145,464	93,997	196,931
		2010	15	116,967	84,102	149,832	116,967	77,251	156,682
		2011	26	191,690	146,655	236,725	191,690	133,145	250,234
		2012	15	165,973	126,561	205,385	165,973	122,370	209,576
Warner Lakes	Deep	2007	18	56,632	30,627	82,637	56,632	26,279	86,985
		2008	13	63,089	35,336	90,842	63,089	28,474	97,705
		2009	16	90,367	54,429	126,305	90,367	46,911	133,823
		2010	30	48,444	35,169	61,719	48,444	32,363	64,525
		2011	16	98,516	39,992	157,039	98,516	31,191	165,840
		2012	14	93,055	45,831	140,279	93,055	40,881	145,229
	Honey	2007	18	26,775	17,204	36,347	26,775	14,353	39,198
		2008	7	64,702	40,970	88,435	64,702	38,055	91,350
		2009	10	37,260	23,340	51,179	37,260	17,708	56,811
		2010	20	17,633	14,936	20,329	17,633	13,981	21,284
		2011	9	35,787	23,434	48,140	35,787	21,416	50,157
		2012	9	30,342	17,764	42,920	30,342	14,390	46,294
	Twentymile	2007	18	30,754	20,796	40,713	30,754	16,855	44,654
		2008	4	19,051	8,763	29,339	19,051	6,735	31,367
		2009	4	16,804	8,783	24,824	16,804	7,288	26,319
		2010	29	16,281	10,991	21,570	16,281	10,010	22,551
		2011	4	36,465	13,872	59,057	36,465	10,017	62,912
		2012	3	.	.	.	14,894	3,383	26,404

Supplemental Table 2.3 – Sample size (N), estimated density (redband trout·m^{-1}), lower 95% confidence limit (L 95% CL), and upper 95% confidence limit (U 95% CL) by species management unit (SMU), population, and year. Confidence limits were calculated using a local mean variance estimator (when $N > 3$) and a simple random sample (SRS) variance estimator.

				Local mean variance estimator			SRS variance estimator		
SMU	Population	Year	*N*	Mean	L 95% CL	U 95% CL	Mean	L 95% CL	U 95% CL
Catlow Valley	Home	2007	1	.	.	.	.	.	.
		2008	0	.	.	.	.	.	.
		2009	1	.	.	.	.	.	.
		2010	0	.	.	.	.	.	.
		2011	4	0.1957	0.1537	0.2377	0.1957	0.1456	0.2458
		2012	3	.	.	.	0.2037	0.1052	0.3021
	Rock	2007	7	2.4579	1.0908	3.8249	2.4579	0.9480	3.9677
		2008	0	.	.	.	.	.	.
		2009	7	0.6424	0.4526	0.8322	0.6424	0.3302	0.9546
		2010	11	0.7423	0.4974	0.9872	0.7423	0.3196	1.1650
		2011	11	0.9600	0.5023	1.4178	0.9600	0.4113	1.5087
		2012	4	2.5221	1.6599	3.3844	2.5221	1.4533	3.5910
	Skull-3mile	2007	0	.	.	.	.	.	.
		2008	0	.	.	.	.	.	.
		2009	2	.	.	.	0.2978	-0.0437	0.6393
		2010	2	.	.	.	0.2313	0.0202	0.4424
		2011	4	0.2790	0.1849	0.3731	0.2790	0.1657	0.3922
		2012	0	.	.	.	.	.	.
Chewaucan	Chewaucan	2007	24	0.4447	0.2873	0.6021	0.4447	0.2499	0.6395
		2008	18	0.4361	0.3375	0.5347	0.4361	0.2729	0.5993
		2009	24	0.4606	0.3700	0.5513	0.4606	0.3187	0.6025
		2010	24	0.4763	0.3318	0.6208	0.4763	0.2767	0.6759
		2011	22	0.5713	0.3636	0.7790	0.5713	0.3134	0.8292
		2012	29	0.6165	0.4672	0.7658	0.6165	0.4156	0.8174
	Crooked	2007	4	0.5332	0.2978	0.7686	0.5332	0.2545	0.8119
		2008	3	.	.	.	0.8388	0.7639	0.9137
		2009	22	0.3158	0.2455	0.3861	0.3158	0.2121	0.4195
		2010	3	.	.	.	0.3482	0.2766	0.4199
		2011	4	0.2138	0.1469	0.2807	0.2138	0.1300	0.2976
		2012	27	0.2551	0.2267	0.2834	0.2551	0.2100	0.3001
	Willow	2007	3	.	.	.	0.3317	0.0369	0.6266
		2008	3	.	.	.	0.2422	0.0770	0.4074
		2009	19	0.1826	0.1572	0.2081	0.1826	0.1485	0.2168
		2010	3	.	.	.	0.1791	0.0703	0.2880
		2011	2	.	.	.	0.2683	0.0657	0.4709
		2012	9	0.2148	0.1397	0.2899	0.2148	0.1229	0.3067
Fort Rock	Bridge	2007	6	1.0050	0.7285	1.2815	1.0050	0.6919	1.3181
		2008	7	0.4591	0.3238	0.5944	0.4591	0.2858	0.6325
		2009	21	0.5185	0.4519	0.5850	0.5185	0.4253	0.6116
		2010	8	0.4150	0.3092	0.5209	0.4150	0.2856	0.5444
		2011	7	0.2690	0.2228	0.3153	0.2690	0.1936	0.3445
		2012	30	0.3411	0.3023	0.3800	0.3411	0.2934	0.3888

Supplemental Table 2.3 – Continued on next page.

Supplemental Table 2.3 – Continued from previous page.

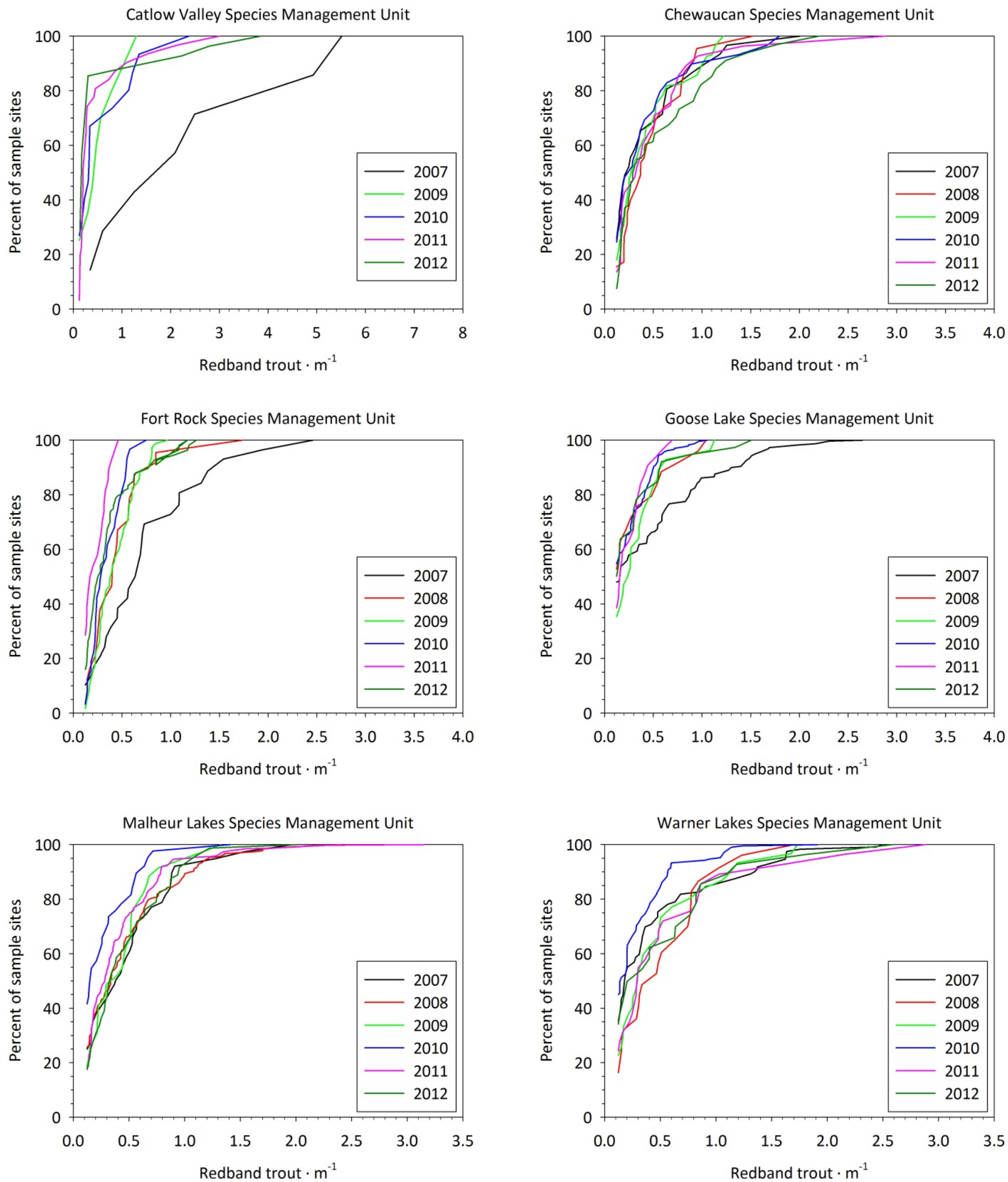
				Local mean variance estimator			SRS variance estimator		
SMU	Population	Year	*N*	Mean	L 95% CL	U 95% CL	Mean	L 95% CL	U 95% CL
Fort Rock	Buck	2007	9	0.6208	0.4482	0.7934	0.6208	0.4037	0.8380
		2008	7	0.5838	0.2469	0.9207	0.5838	0.1952	0.9724
		2009	19	0.4022	0.3629	0.4415	0.4022	0.3322	0.4722
		2010	10	0.3241	0.2447	0.4034	0.3241	0.2327	0.4154
		2011	8	0.2529	0.2078	0.2980	0.2529	0.1912	0.3146
		2012	29	0.2973	0.2671	0.3275	0.2973	0.2387	0.3560
	Silver-FR	2007	12	0.7020	0.3387	1.0652	0.7020	0.2652	1.1387
		2008	10	0.3765	0.2551	0.4980	0.3765	0.2281	0.5249
		2009	13	0.3842	0.2732	0.4952	0.3842	0.2566	0.5118
		2010	9	0.3007	0.2106	0.3909	0.3007	0.2035	0.3979
		2011	6	0.1905	0.1055	0.2755	0.1905	0.0837	0.2972
		2012	23	0.3995	0.2897	0.5093	0.3995	0.2449	0.5541
Goose Lake	Drews	2007	20	0.3089	0.2110	0.4068	0.3089	0.1689	0.4489
		2008	6	0.2915	0.1061	0.4769	0.2915	0.0708	0.5122
		2009	7	0.2771	0.1658	0.3885	0.2771	0.1382	0.4161
		2010	20	0.2209	0.1759	0.2659	0.2209	0.1580	0.2839
		2011	8	0.2187	0.1723	0.2651	0.2187	0.1251	0.3124
		2012	7	0.2104	0.1145	0.3063	0.2104	0.0872	0.3335
	Dry	2007	12	0.4069	0.0496	0.7642	0.4069	-0.0154	0.8292
		2008	2	.	.	.	0.1236	0.1236	0.1236
		2009	2	.	.	.	0.7164	-0.0002	1.4330
		2010	4	0.1403	0.1125	0.1682	0.1403	0.1075	0.1731
		2011	2	.	.	.	0.4084	-0.1498	0.9666
		2012	2	.	.	.	0.1374	0.1104	0.1644
	Eastside	2007	17	1.0420	0.8396	1.2445	1.0420	0.7911	1.2930
		2008	2	.	.	.	1.0009	0.9250	1.0769
		2009	3	.	.	.	0.6853	0.2404	1.1302
		2010	21	0.5691	0.4794	0.6589	0.5691	0.4592	0.6791
		2011	3	.	.	.	0.5153	0.4017	0.6288
		2012	3	.	.	.	1.2421	0.8753	1.6088
	Thomas-Bauers	2007	20	0.4475	0.2713	0.6237	0.4475	0.2437	0.6514
		2008	10	0.2591	0.1671	0.3512	0.2591	0.1507	0.3676
		2009	12	0.2277	0.1577	0.2978	0.2277	0.1502	0.3053
		2010	22	0.2332	0.1856	0.2807	0.2332	0.1705	0.2958
		2011	12	0.2011	0.1589	0.2433	0.2011	0.1542	0.2481
		2012	12	0.2548	0.1650	0.3446	0.2548	0.1576	0.3519
	West Goose	2007	19	0.6076	0.3332	0.8819	0.6076	0.2878	0.9274
		2008	4	0.1302	0.1192	0.1412	0.1302	0.1173	0.1431
		2009	4	0.2543	0.0951	0.4135	0.2543	0.0626	0.4461
		2010	20	0.2022	0.1265	0.2778	0.2022	0.1165	0.2878
		2011	4	0.1724	0.1052	0.2397	0.1724	0.0910	0.2539
		2012	5	0.1673	0.1034	0.2312	0.1673	0.0900	0.2447
Malheur Lakes	Blitzen	2007	7	0.3146	0.1786	0.4505	0.3146	0.1499	0.4792
		2008	23	0.6772	0.5122	0.8423	0.6772	0.4406	0.9138
		2009	6	0.7227	0.4767	0.9687	0.7227	0.4389	1.0064
		2010	5	0.3600	0.1966	0.5235	0.3600	0.1622	0.5578
		2011	26	0.3343	0.2801	0.3884	0.3343	0.2744	0.3941
		2012	6	0.5172	0.2942	0.7402	0.5172	0.2495	0.7850

Supplemental Table 2.3 – Continued on next page.

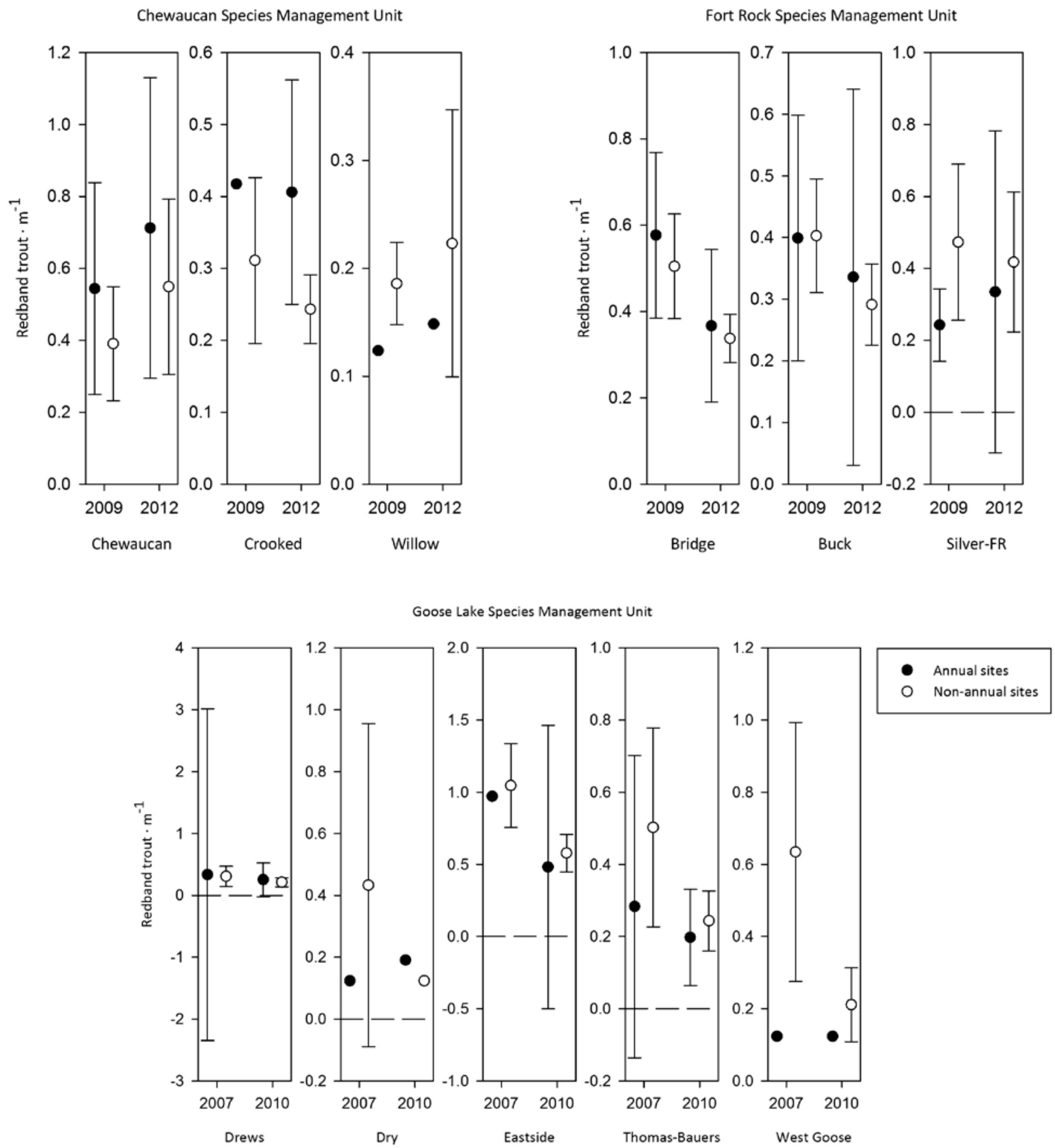
Supplemental Table 2.3 – Continued from previous page.

				Local mean variance estimator			SRS variance estimator		
SMU	Population	Year	*N*	Mean	L 95% CL	U 95% CL	Mean	L 95% CL	U 95% CL
Malheur Lakes	East Burns	2007	3	.	.	.	0.4330	0.0507	0.8153
		2008	25	0.4359	0.3212	0.5505	0.4359	0.2936	0.5781
		2009	2	.	.	.	0.2774	0.2525	0.3023
		2010	3	.	.	.	0.2533	0.1091	0.3975
		2011	25	0.5870	0.3547	0.8192	0.5870	0.3111	0.8628
		2012	3	.	.	.	0.4429	0.1280	0.7577
	McCoy	2007	6	0.7544	0.5339	0.9748	0.7544	0.4974	1.0113
		2008	24	0.6628	0.5042	0.8213	0.6628	0.4717	0.8538
		2009	5	0.3512	0.2015	0.5008	0.3512	0.1824	0.5199
		2010	3	.	.	.	0.2835	0.0067	0.5603
		2011	25	0.4125	0.3181	0.5069	0.4125	0.2930	0.5321
		2012	6	0.7246	0.4597	0.9895	0.7246	0.3849	1.0643
	Riddle	2007	3	.	.	.	0.4863	0.0374	0.9352
		2008	18	0.6165	0.4438	0.7891	0.6165	0.3714	0.8616
		2009	2	.	.	.	0.6604	-0.2809	1.6017
		2010	3	.	.	.	0.7488	0.0214	1.4763
		2011	25	0.7648	0.6274	0.9022	0.7648	0.4898	1.0398
		2012	3	.	.	.	1.3460	0.6986	1.9933
	Silver	2007	6	0.4637	0.2608	0.6666	0.4637	0.2367	0.6906
		2008	25	0.3242	0.2562	0.3922	0.3242	0.2268	0.4216
		2009	3	.	.	.	0.2344	0.0172	0.4516
		2010	6	0.1264	0.1215	0.1312	0.1264	0.1210	0.1318
		2011	30	0.2135	0.1778	0.2491	0.2135	0.1615	0.2654
		2012	6	0.1761	0.1029	0.2493	0.1761	0.0931	0.2591
	Silvies	2007	14	0.4869	0.2255	0.7483	0.4869	0.1811	0.7926
		2008	23	0.3176	0.2365	0.3988	0.3176	0.2163	0.4190
		2009	13	0.3625	0.2668	0.4582	0.3625	0.2342	0.4908
		2010	15	0.2498	0.1796	0.3200	0.2498	0.1650	0.3347
		2011	26	0.4095	0.3133	0.5057	0.4095	0.2844	0.5345
		2012	15	0.3545	0.2703	0.4387	0.3545	0.2614	0.4477
Warner Lakes	Deep	2007	18	0.4399	0.2379	0.6418	0.4399	0.2041	0.6756
		2008	13	0.4467	0.2502	0.6432	0.4467	0.2016	0.6918
		2009	16	0.5666	0.3413	0.7920	0.5666	0.2941	0.8391
		2010	30	0.2961	0.2150	0.3773	0.2961	0.1978	0.3944
		2011	16	0.5834	0.2368	0.9300	0.5834	0.1847	0.9821
		2012	14	0.7085	0.3490	1.0681	0.7085	0.3113	1.1058
	Honey	2007	18	0.3174	0.2040	0.4309	0.3174	0.1702	0.4647
		2008	7	0.6904	0.4371	0.9436	0.6904	0.4060	0.9747
		2009	10	0.3975	0.2490	0.5461	0.3975	0.1889	0.6062
		2010	20	0.2150	0.1821	0.2479	0.2150	0.1705	0.2595
		2011	9	0.4243	0.2778	0.5707	0.4243	0.2539	0.5946
		2012	9	0.3237	0.1895	0.4579	0.3237	0.1535	0.4939
	Twentymile	2007	18	0.8137	0.5502	1.0772	0.8137	0.4459	1.1815
		2008	4	0.4164	0.1915	0.6412	0.4164	0.1472	0.6856
		2009	4	0.3673	0.1920	0.5426	0.3673	0.1593	0.5752
		2010	29	0.4132	0.2790	0.5475	0.4132	0.2541	0.5724
		2011	4	0.7970	0.3032	1.2908	0.7970	0.2189	1.3750
		2012	3	.	.	.	0.4340	0.0986	0.7695

List of Supplemental Figures



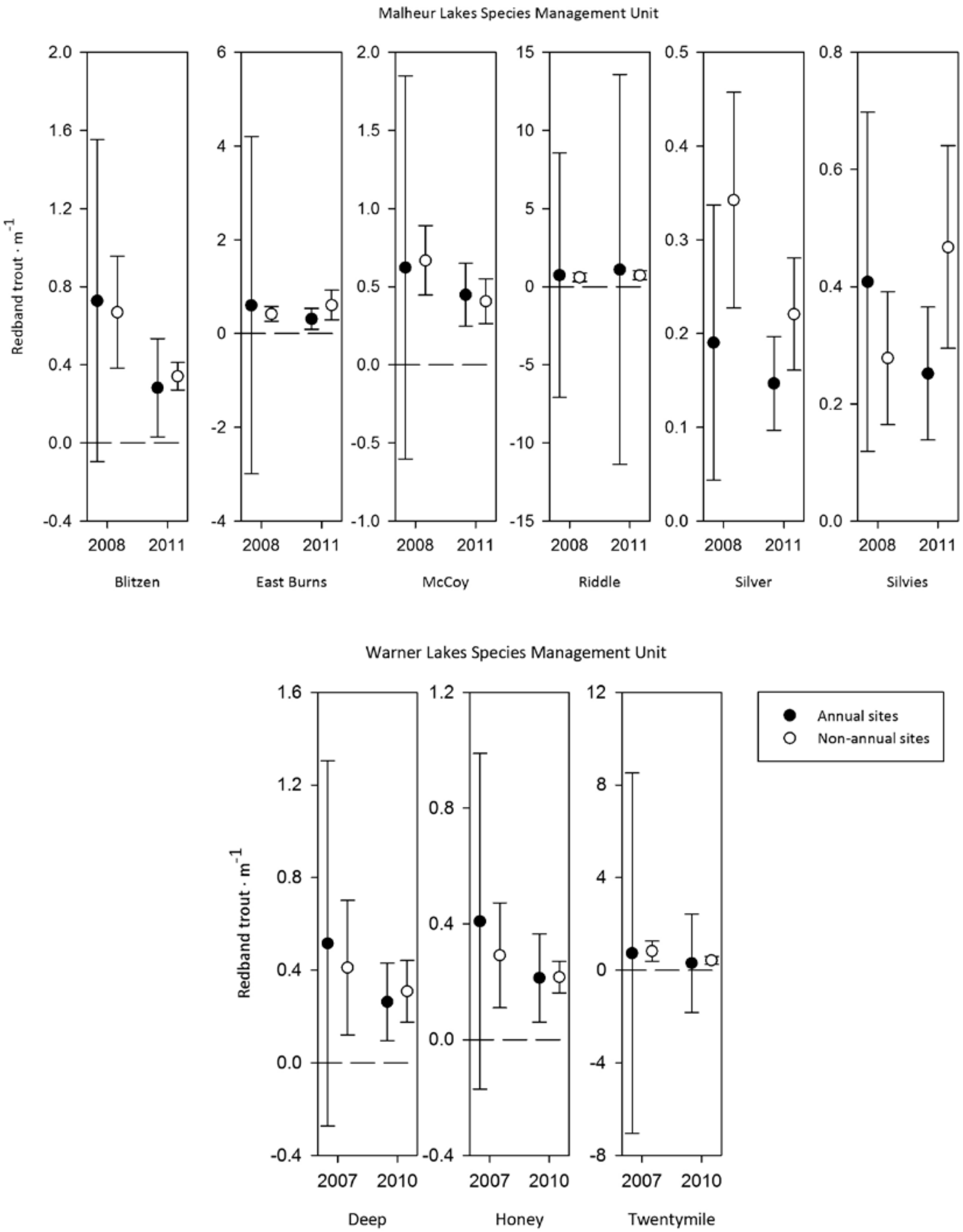
Supplemental Figure 2.1 – Cumulative frequency distribution of redband trout density for sample sites examined in six species management units in the northern Great Basin from 2007 through 2012. Redband trout density was calculated as *Density* = 0.12 + 1.24·$Density_{Depletion}$; therefore, redband trout densities of 0.12 represent sample sites where no redband trout (≥ 60 mm) were detected

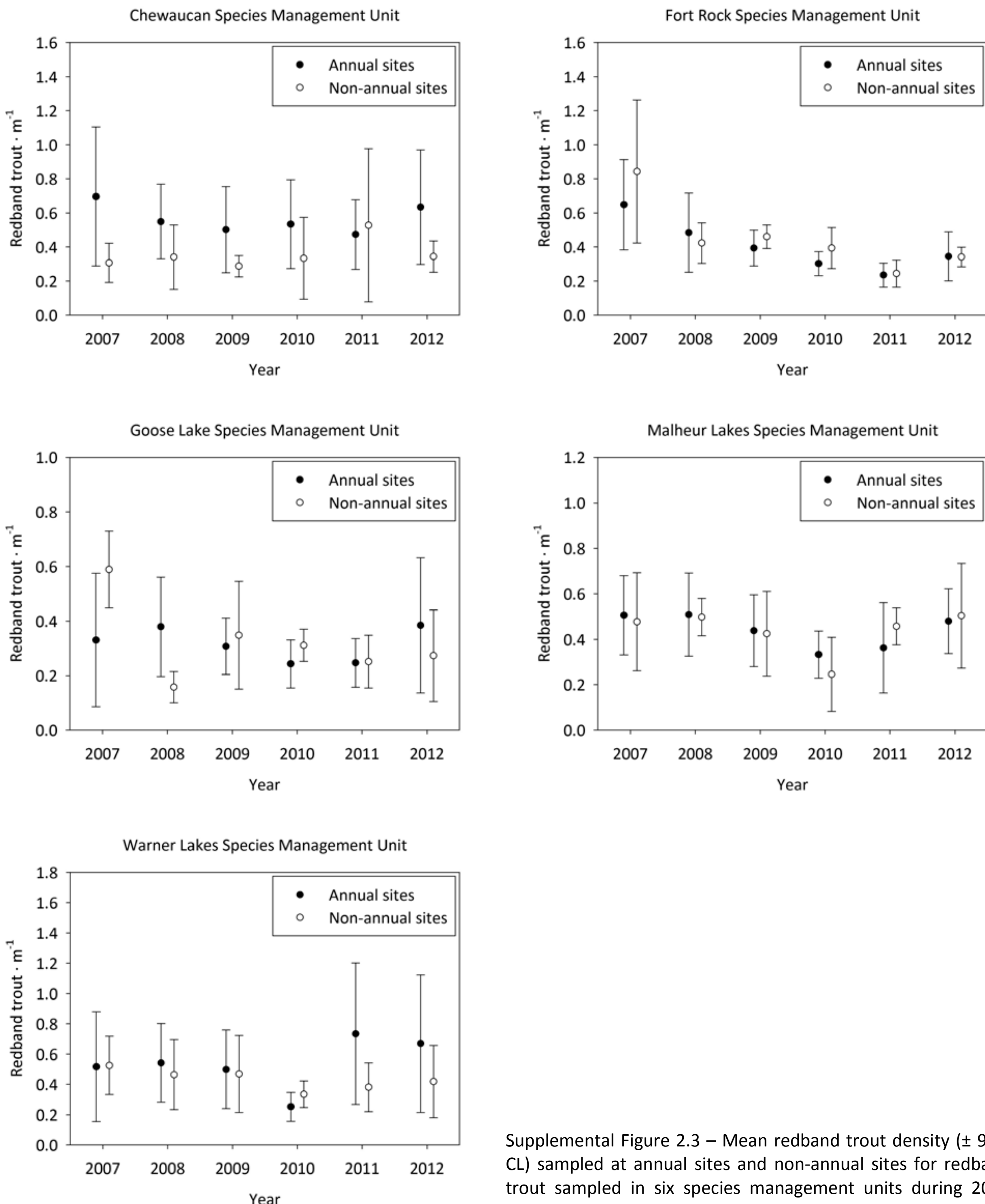


Supplemental Figure 2.2 – Mean redband trout density (± 95% CL) sampled at annual sites and non-annual sites for redband trout sampled in six species management units in the northern Great Basin. From three to six populations were present within each species management unit and each population was sampled twice at the population level during the study.

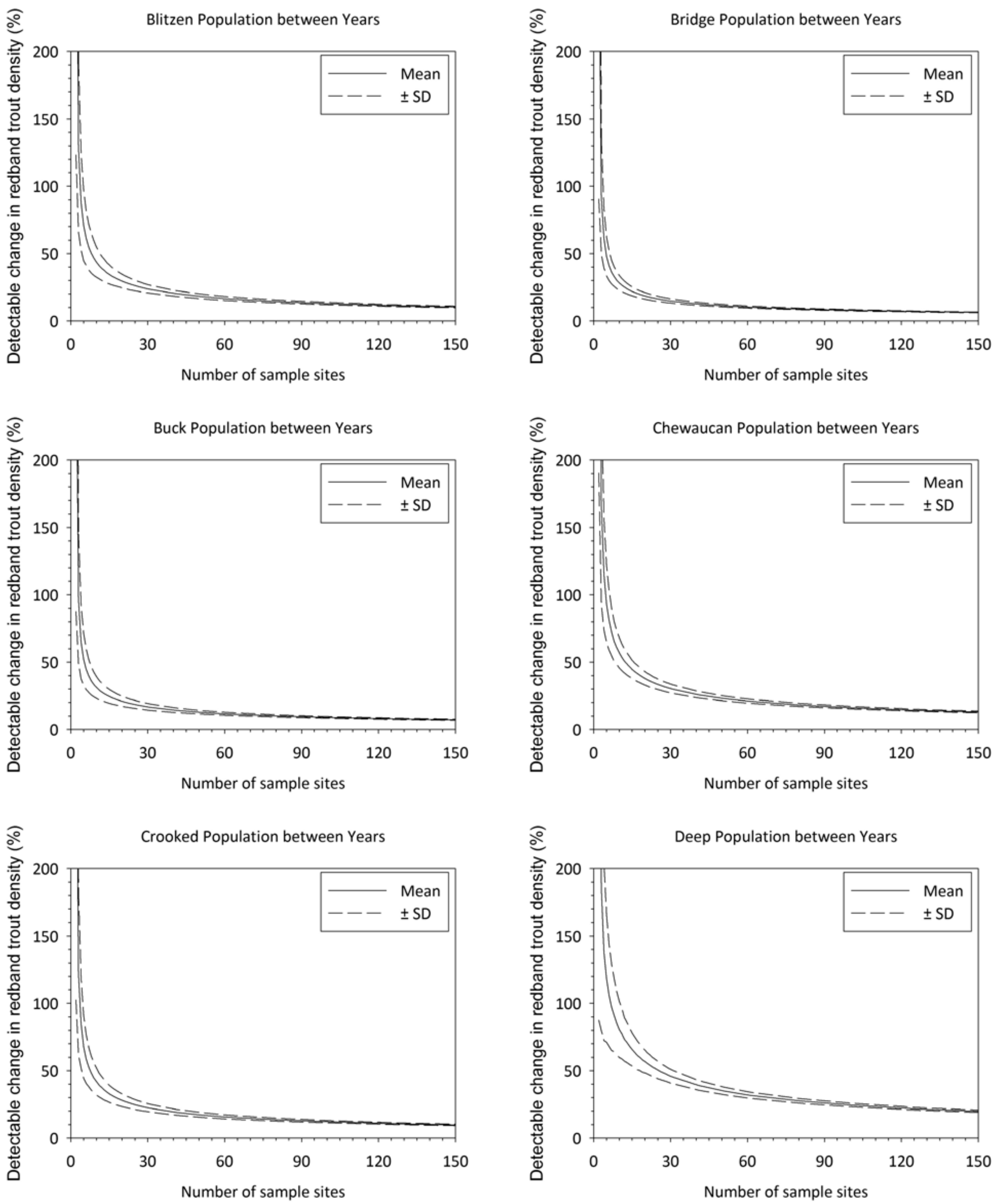
Supplemental Figure 2.2 – Continued on next page.

Supplemental Figure 2.2 – Continued from previous page.





Supplemental Figure 2.3 – Mean redband trout density (± 95% CL) sampled at annual sites and non-annual sites for redband trout sampled in six species management units during 2007 through 2012 in the northern Great Basin.



Supplemental Figure 2.4 – Predicted detectable change in redband trout density (%) for population-level density estimates as a function of sampling intensity. Analyses performed by population and year and averaged between population-level sampling years.

Supplemental Figure 2.4 – Continued on next page.

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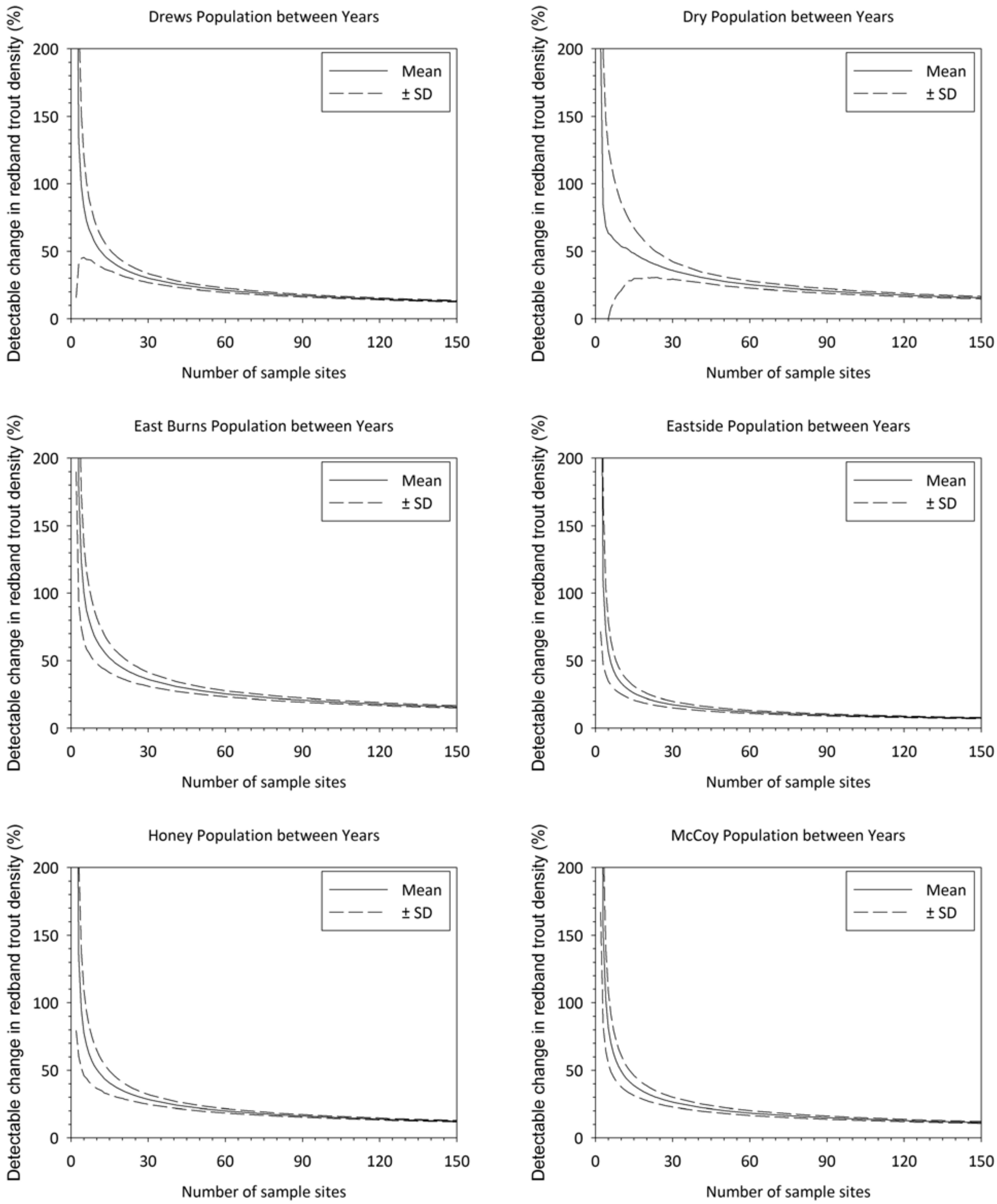

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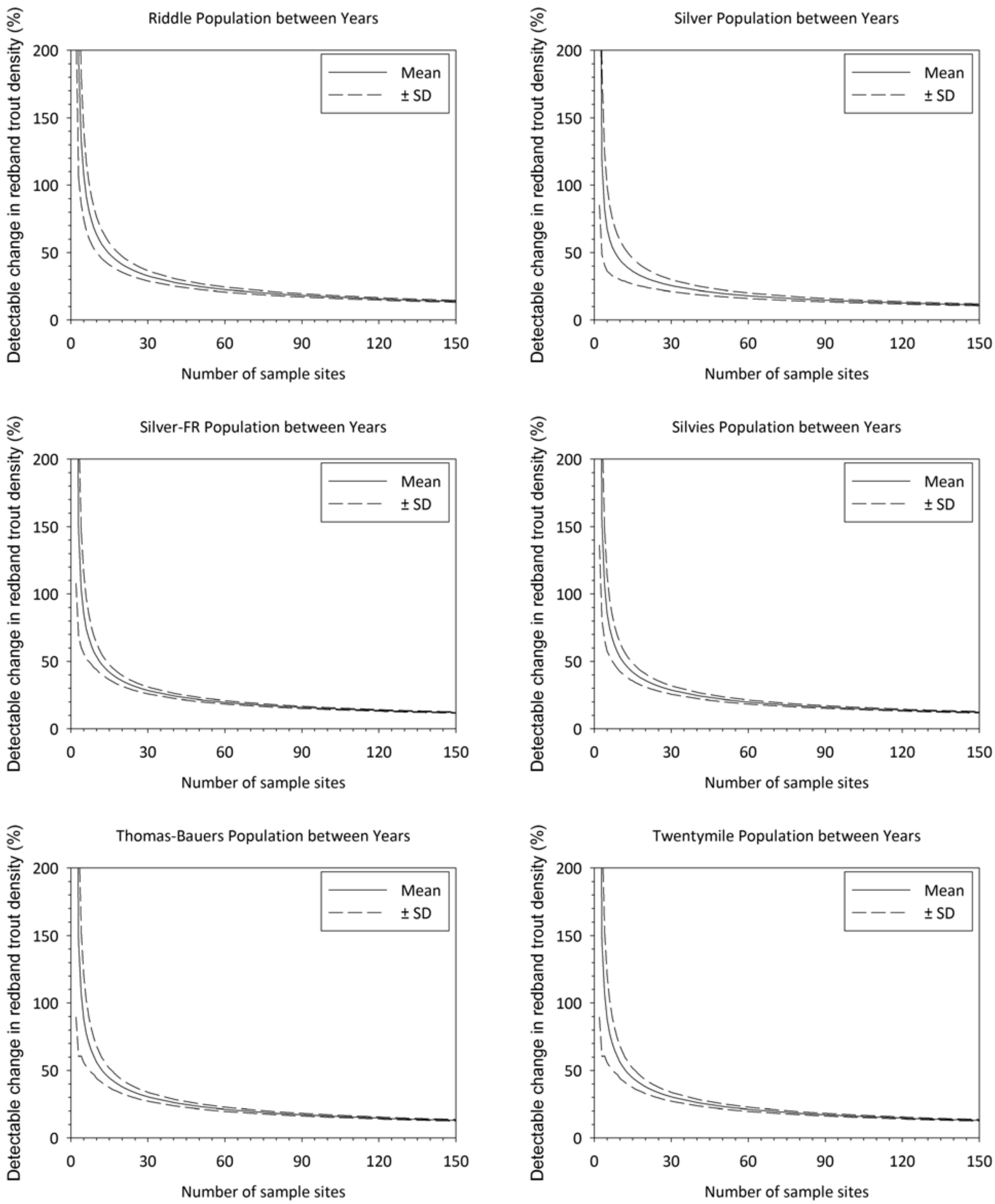

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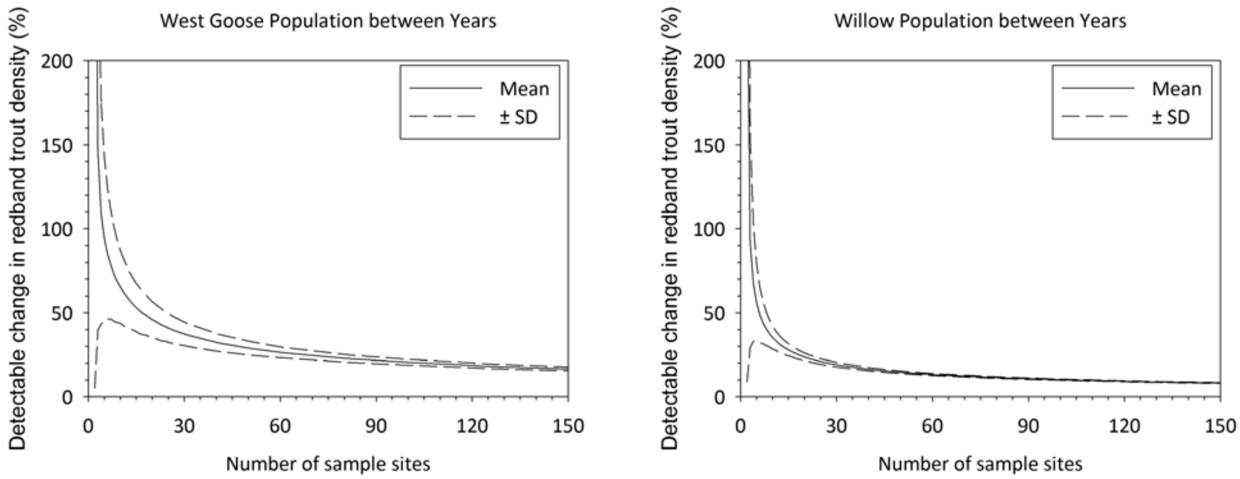
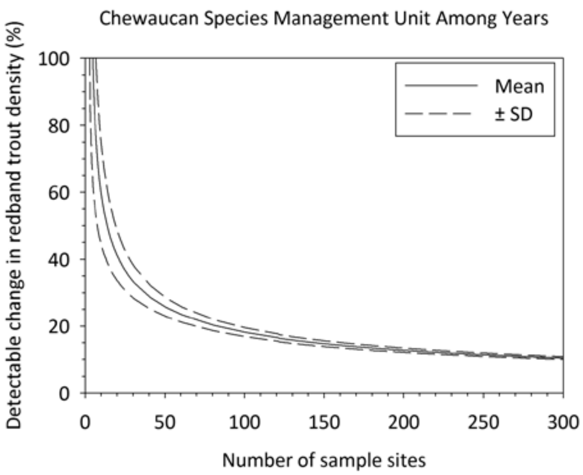
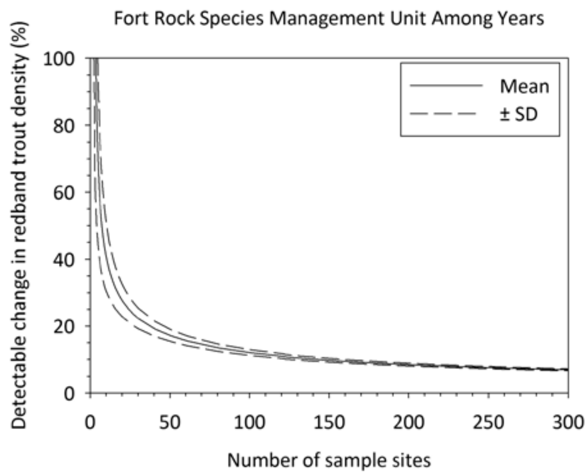
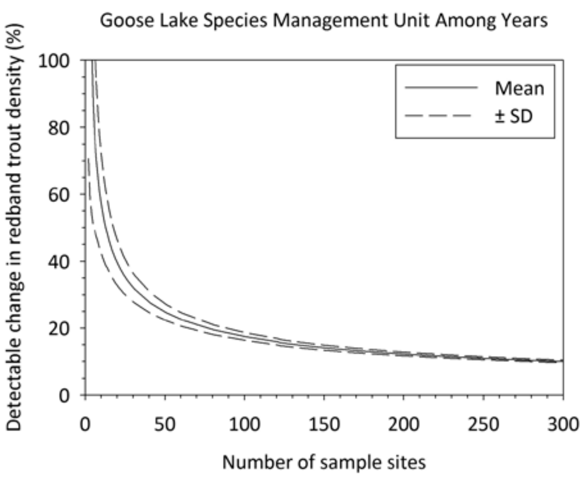
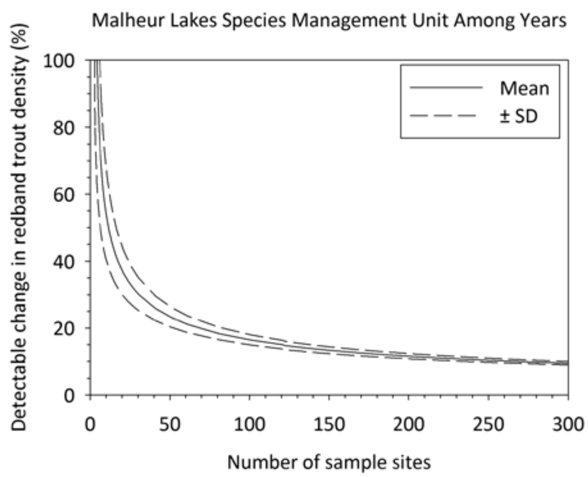
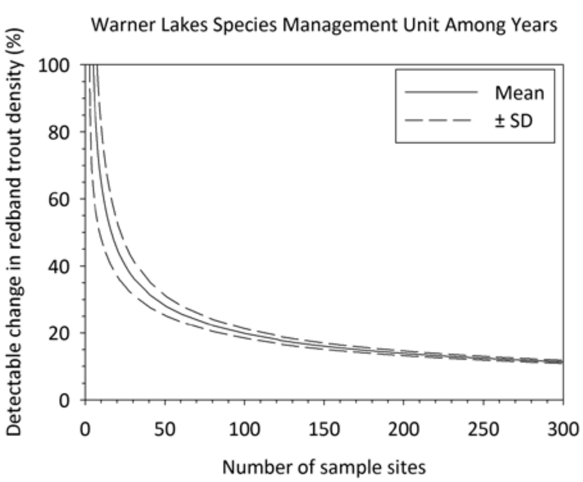




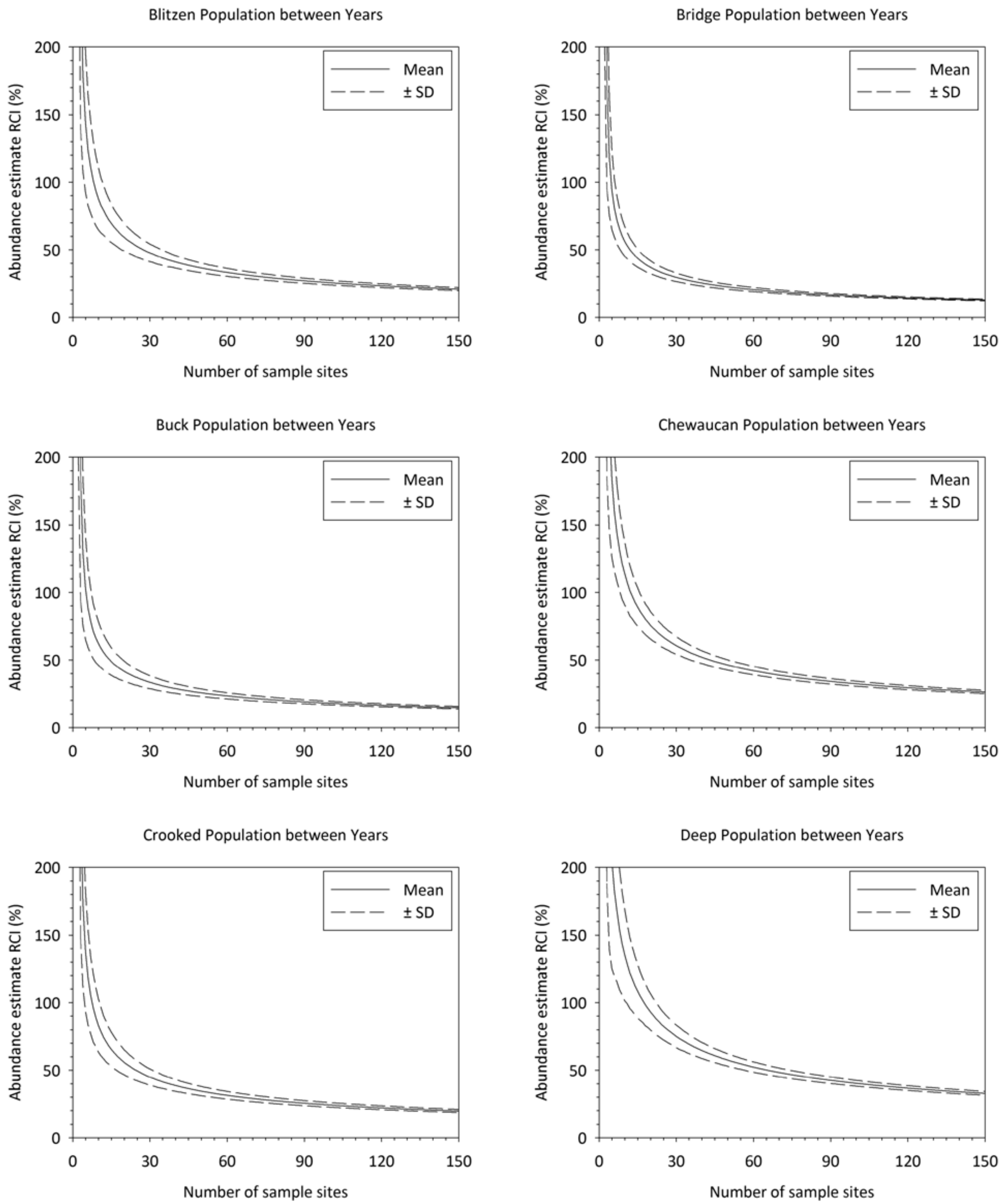








Supplemental Figure 2.5 – Predicted detectable change in redband trout density (%) for species management unit level density estimates as a function of sampling intensity. Analyses performed by species management unit and year and averaged among years.



Supplemental Figure 2.6 – Predicted relative confidence index (RCI) for population-level abundance estimates as a function of sampling intensity. Analyses performed by population and year and averaged between population-level sampling years.

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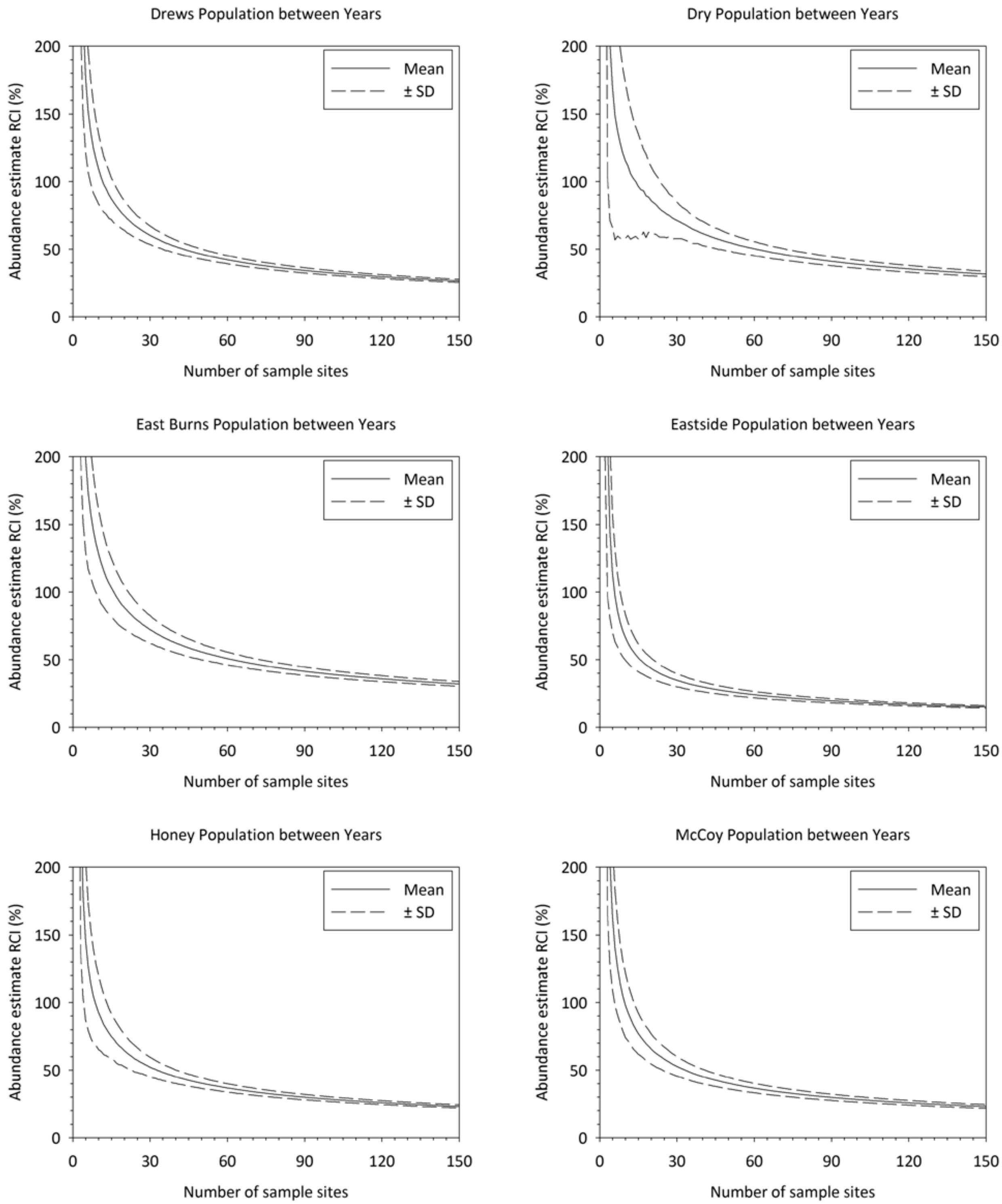

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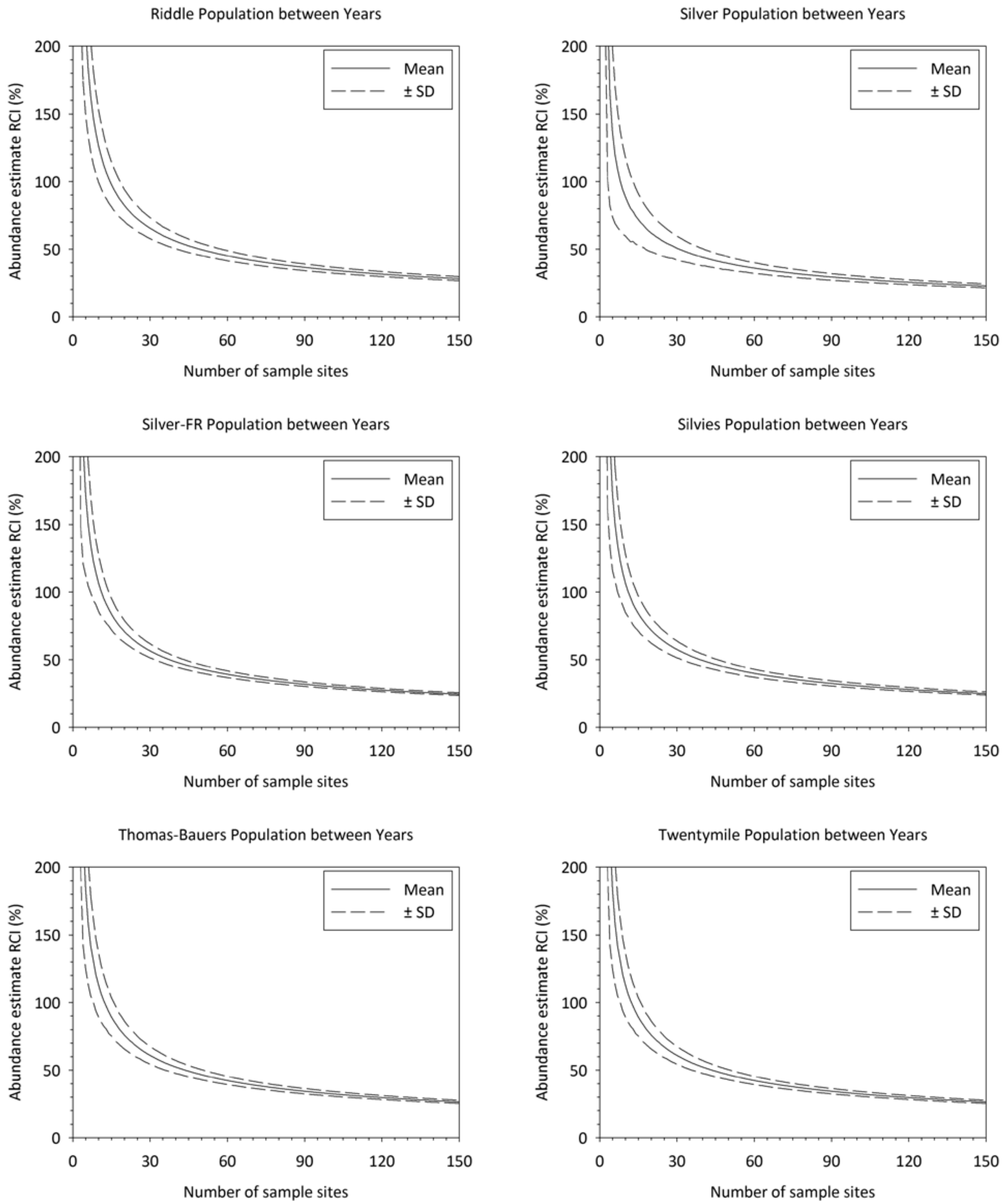

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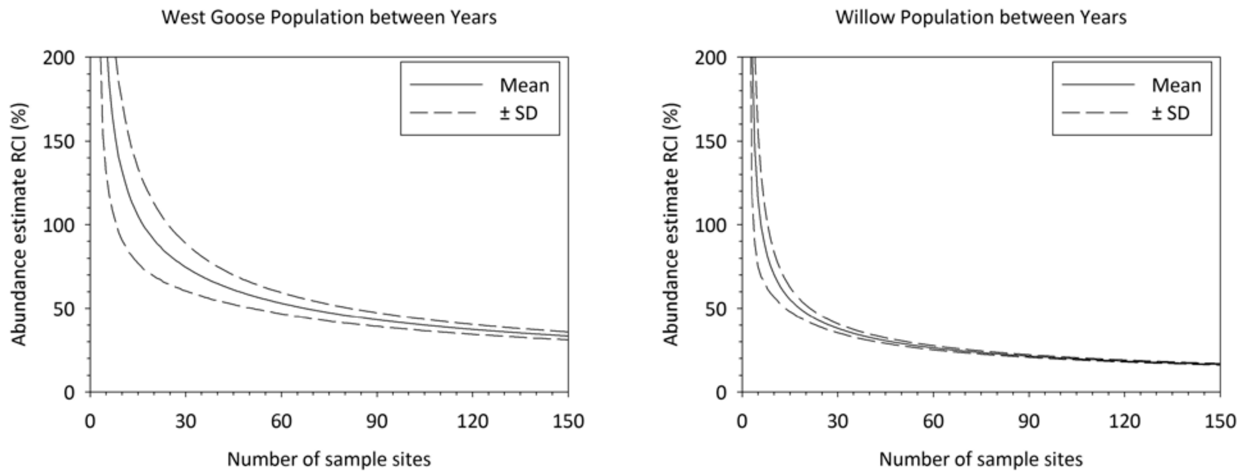
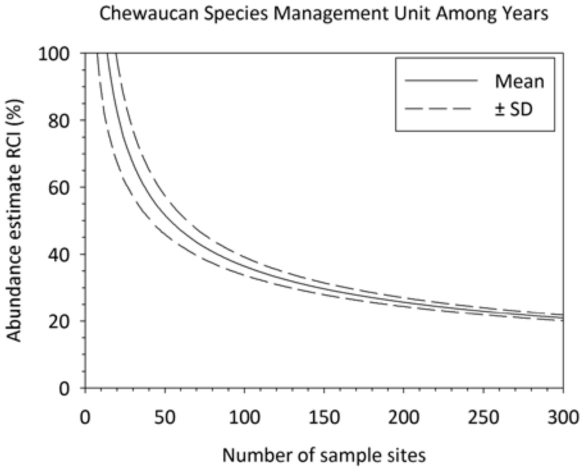
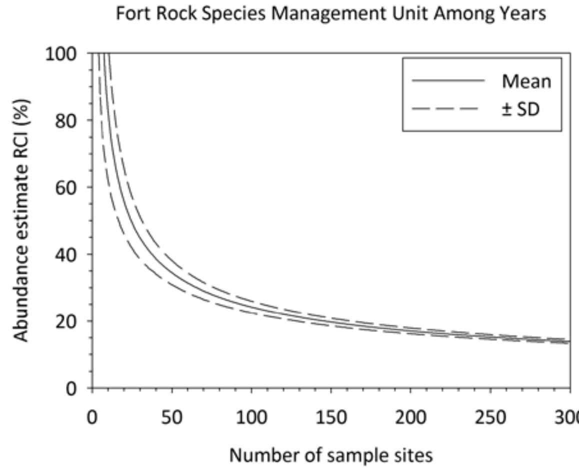
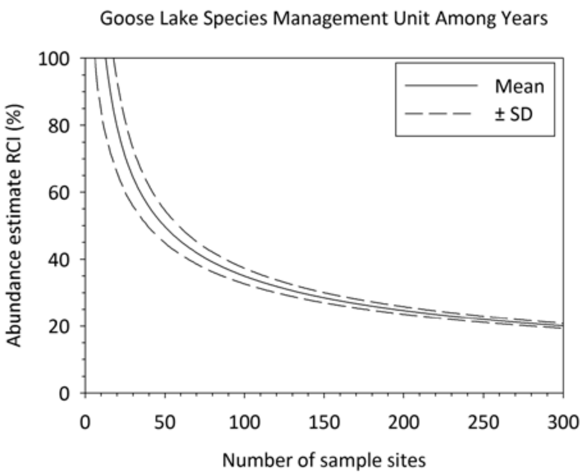
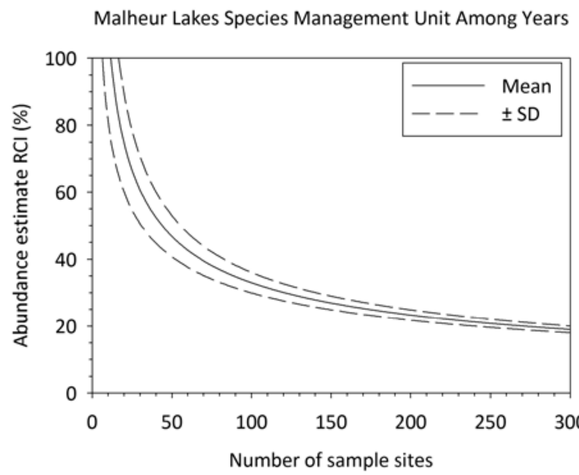
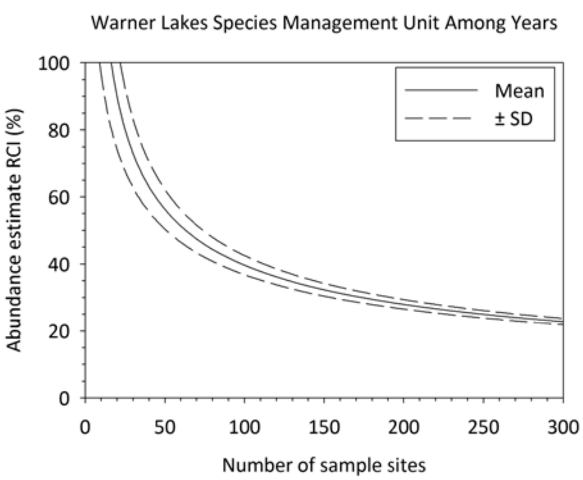




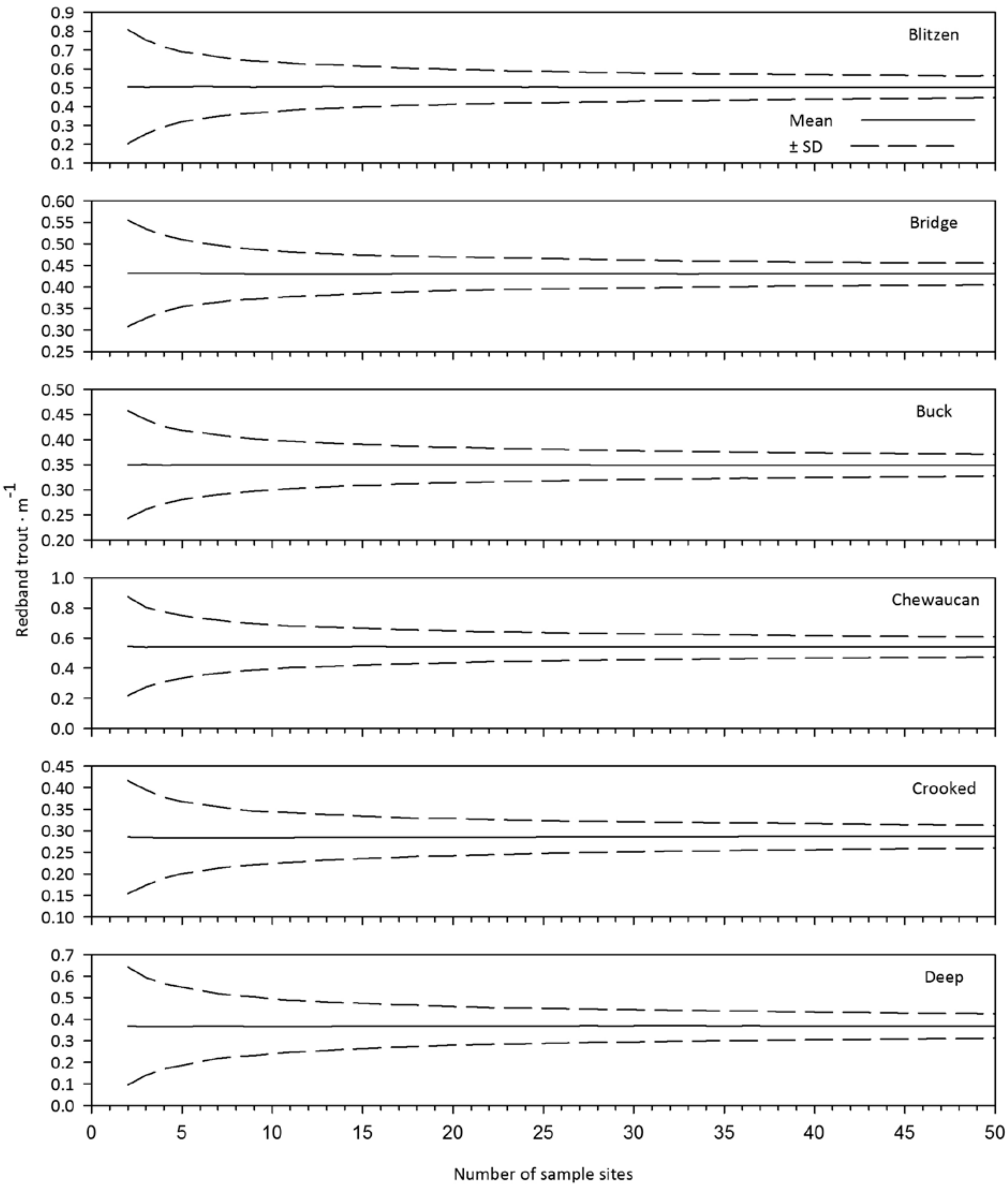








Supplemental Figure 2.7 – Predicted relative confidence index (RCI) for species management unit level abundance estimates as a function of sampling intensity. Analyses performed by species management unit and year and averaged among years.



Supplemental Figure 2.8 – Predicted variability of redband trout density estimates for population-level density estimates as a function of sampling intensity. Analyses performed by population and year and averaged between population-level sampling years.

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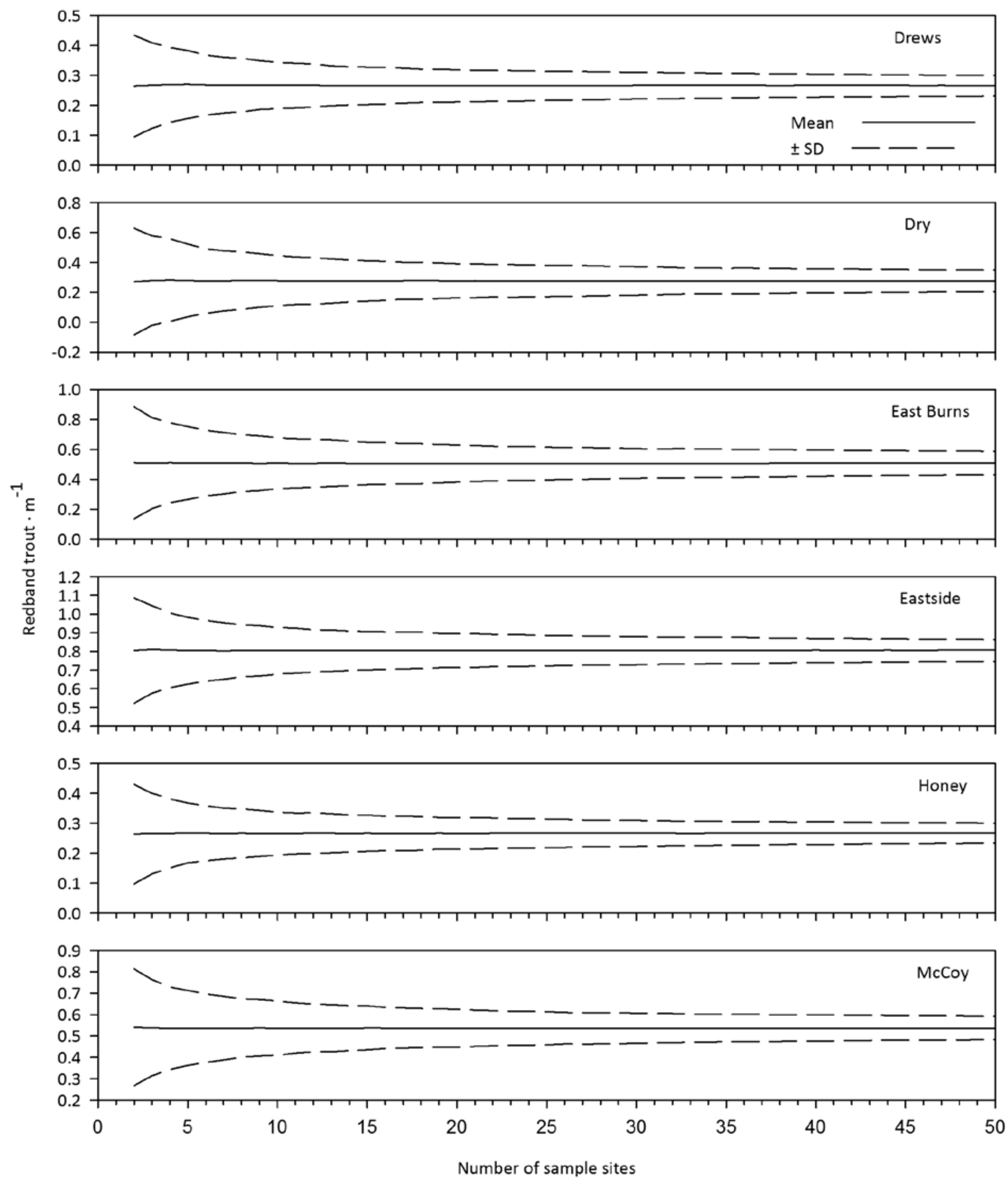

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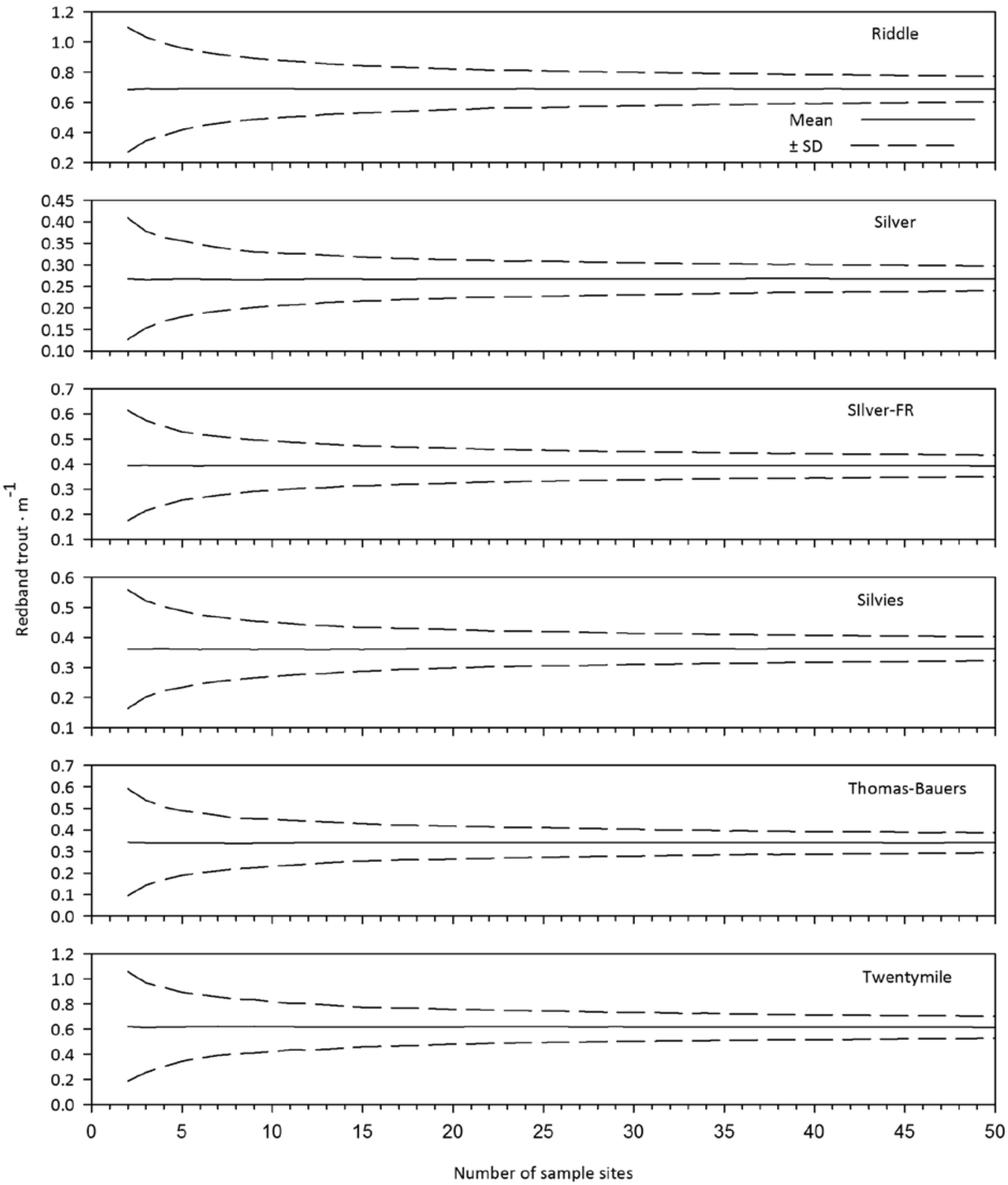

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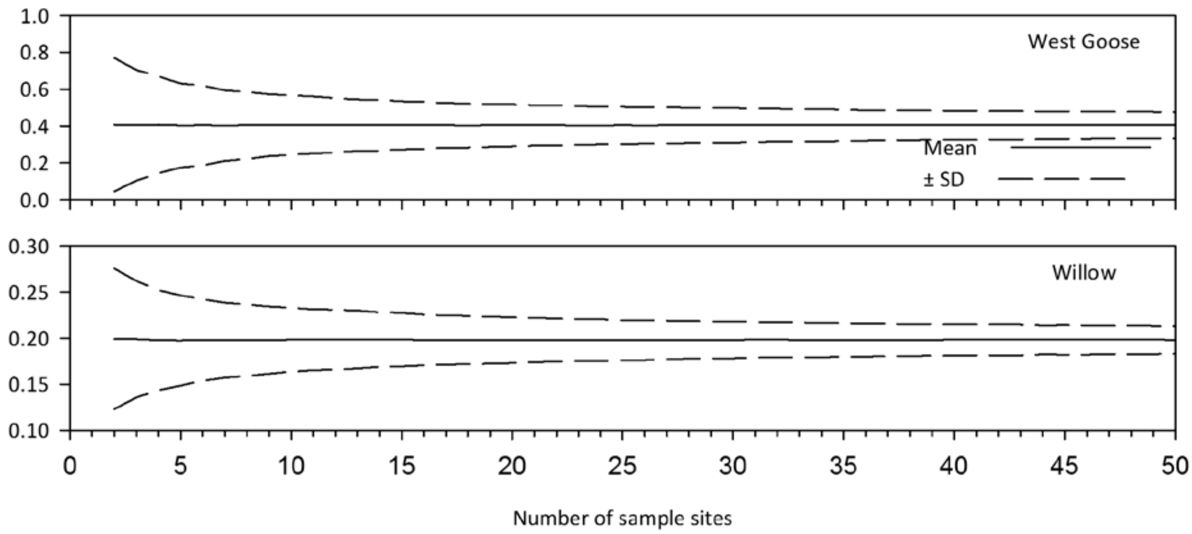

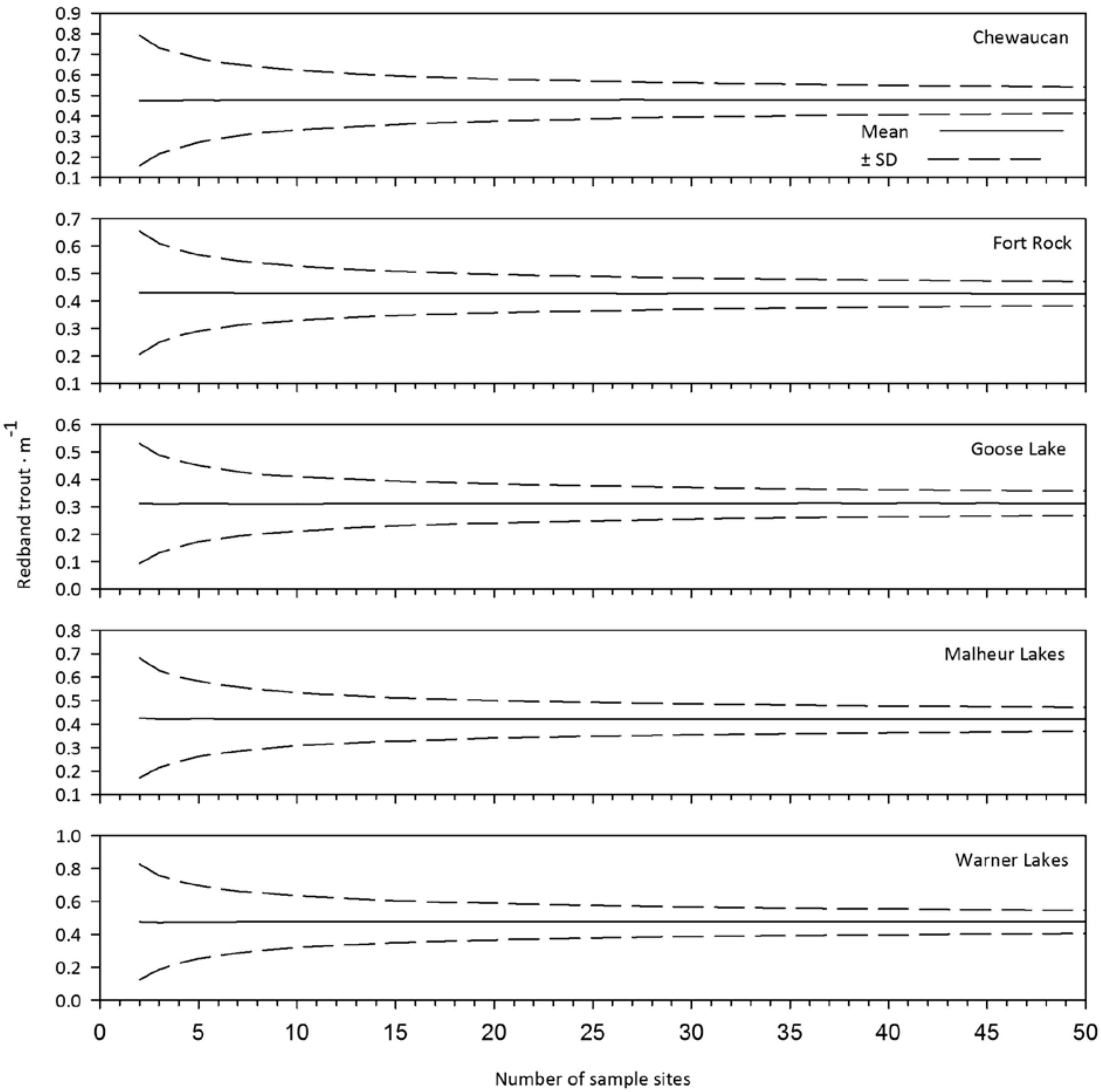

Supplemental Figure 2.9 – Predicted variability of redband trout density estimates for species management unit abundance estimates as a function of sampling intensity. Analyses performed by species management unit and year and averaged among years.

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