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Bull Trout Status Review and Assessment in the State of Idaho

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Abstract

Broad-scale declines in bull trout *Salvelinus confluentus* distribution and abundance over the past century or more led to state-wide no-harvest regulations in the state of Idaho in 1994, and ultimately to a threatened listing under the Endangered Species Act in 1998. Despite this listing, quantitative evaluations of trends in abundance and estimates of existing population sizes over most of its historical range have not been made. We evaluated long-term trends in bull trout abundance, estimated population sizes, and conducted population viability analysis (PVA) for bull trout in Idaho, based on stratified sampling extrapolations of fish surveys (snorkel and electrofishing) conducted at 2,287 study sites scattered across 77,447 km of stream within seven recovery units in Idaho. Bull trout were present in 871 (38%) of the study sites, and were most likely to be observed or captured in 2nd- and 3rd-order streams. Long-term trend estimates from bull trout redd counts, spawning weirs, and snorkel and electrofishing population surveys indicate that bull trout populations declined through the mid-1990s when state-wide no-harvest regulations were implemented, but in general have increased over the last 10 years. Moreover, abundance of all other species of salmonids appeared to increase at the same time, suggesting that bull trout recovery was not related to no-harvest regulation changes. Bull trout abundance was positively correlated with the abundance of all other salmonids (including non-native brook trout), but was most strongly correlated with westslope cutthroat trout abundance. We estimated there was approximately 1.24 million bull trout within the 269 designated local populations within the seven recovery units. Population viability analysis (PVA) indicates that if the general trend observed for the last 10 years is representative of the future, or if populations reach equilibrium at levels near the current estimated abundance, all 7 recovery units have > 0.95 probability of at least one local population persisting 100 years. If local populations in the

Kootenai River and Coeur d'Alene Lake basins experience negative growth with high variance, then there is a 0.30 and 0.20 probability of persistence, respectively, for at least one bull trout local population persisting 100 years containing 100 individuals. Our results suggest that bull trout remain relatively abundant in large stream networks throughout Idaho, that their abundance for most core areas in most recovery units has been increasing over the past decade, and that their risk to extinction in Idaho is minimal. Subsequently, we believe that bull trout fail to qualify as a threatened species since they are unlikely to be in danger of extinction in a substantial portion of their range within the foreseeable future.

Purpose

The purpose of this document is to provide comments to the U.S. Fish and Wildlife Service (USFWS) regarding their status review for bull trout *Salvelinus confluentus* under the Endangered Species Act (ESA).

Introduction

Like most other native salmonids in the western United States, the bull trout has over the past century or more experienced substantial declines in abundance and distribution in large portions of its historical range (Rieman and McIntyre 1993; Rieman et al. 1997). Declines have been ascribed to a number of factors, but most notably to 1) displacement by or hybridization with nonnative trout, 2) over-exploitation by anglers, and 3) habitat alterations and fragmentation due to water storage and diversion, grazing, mineral extraction, and timber harvest. Such declines led the Idaho Department of Fish and Game (IDFG) to implement state-wide no-harvest regulations in 1994, and ultimately led to a threatened listing under the ESA for bull trout in the Columbia River Basin (USFWS 1998).

Over the past decade several status assessments have been conducted for bull trout throughout most of their range in the coterminous United States (Ratliff and Howell 1992; Rieman and McIntyre 1993; Rieman et al. 1997). Most assessments have been qualitative in nature, focusing on the proportion of assumed historical range that is no longer occupied. None have included broad-scale evaluation of long-term and recent trends in abundance, and estimates of current population sizes at a variety of scales, such as individual local populations or entire, large river drainages. Such information is important because both the total number of bull trout and their trend in abundance are central to ESA listing. As part of the State of Idaho's comments to the USFWS for their 5-year review on bull trout status, our primary objective was to fill this information gap for Idaho.

Population viability analysis (PVA) has become a popular tool to use when managing populations at levels below carrying capacity. Essentially, PVA is a modeling exercise used to estimate future population sizes and risk of extinction based on population vulnerability to four categories of stochastic factors - genetic, demographic, environmental, and interactions between local populations (Shaffer 1981; Gilpin and Soulé 1986; Soulé 1987). Most methodologies for PVA are deterministic in nature, and are based on population simulations where a model is constructed using a number of population parameters. Consequently, PVA, in general, is data hungry (Beissinger and Westphal 1998). Although PVA approaches based on population simulation can be useful for prioritizing "at risk" populations or determining how different management strategies may affect populations, sufficient data are usually unavailable to adequately perform quantitative analyses, especially for rare or listed species (Dennis et al. 1991; Ralls et al. 2002). Furthermore, predicted extinction risk, particularly for longer time frames (i.e. 100 years), is extremely sensitive to the estimated growth rate (Dennis et al. 1991; Ludwig 1999; Fieberg and Ellner 2000) allowing small inaccuracies in parameter estimation to have a large effect on the model predictions.

An alternative to complicated PVA models requiring numerous population parameters is the stochastic exponential growth model of Dennis et al. (1991), which estimates growth rates and extinction probabilities based on time series data. This simple PVA model allowed us to model probabilities of population persistence over a representative range of likely population growth rates and variances, in order to assess the relative extinction risks faced by bull trout in Idaho. We are aware of only one formal evaluation of extinction risk for bull trout (Rieman and

McIntyre 1993) following the modeling approach of Dennis et al. (1991). Because their evaluation is more than a decade old, and because of the present USFWS status review for bull trout, our second objective was to re-evaluate bull trout extinction risk with updated information.

Study Area

The range-wide distribution of bull trout remains unclear because of its confusion in the past with Dolly Varden and Arctic char (Behnke 2002). In Idaho, bull trout were historically present in most of the Columbia River and Snake River basins up to Shoshone Falls, and apparently in the Little Lost River drainage above Shoshone Falls (Figure 1; Table 1). To facilitate summary of available information, and for consistency in terminology used by the USFWS in the draft bull trout recovery plan, we subdivided bull trout distribution in Idaho into 31 core areas within 7 recovery units (Figure 1). There are portions of three other recovery units in Idaho (Imnaha-Snake River basins, Snake River Basin, and Northeast Washington River basins), but we did not include these units in our analyses.

Because the historical range of bull trout is unknown and unknowable at small scales (i.e., individual streams), we have chosen to focus only on current distribution, abundance, trends, and population viability, and not on the amount of presumed historical habitat currently occupied. Within Idaho, bull trout maintain a number of life history patterns, including fluvial, adfluvial, and resident forms, but our surveys did not attempt to distinguish between these types of life histories.

Methods

We gathered as much geo-referenced, quantitative bull trout abundance data as possible from a number of sources, including IDFG, U.S. Forest Service, U.S. Bureau of Land Management, and U.S. Bureau of Reclamation. The data we were able to gather included annual redd counts, fixed weir counts, stream snorkel surveys, and one- and multiple-pass electrofishing surveys. We divided our analyses into long-term population trends, approximations of abundance, and population viability analysis.

Population trends

Long-term trends (at least from 1994 to 2003) in bull trout abundance were available from redd counts, weirs, electrofishing, and snorkel count data at five of the seven recovery units in Idaho. Redd count trends were available in five core areas within four recovery units (Table 2). Redd counts were summarized as total annual counts from one to six individual trend sites. Weir trends were available for three core areas in two recovery units, and were summarized simply as the total annual upstream spawning run. In the Lost River recovery unit, the only available trend data was from four electrofishing sites that were repeatedly (but sporadically) sampled from the 1980s to present; trend for this recovery unit was assessed using the yearly average density of bull trout from these sites.

Since 1985, daytime snorkel counts have been conducted by IDFG personnel via several Bonneville Power Administration-funded research projects, as part of what has been termed General Parr Monitoring (GPM). Although originally designed to track trends for anadromous species, observations on all resident fish have been recorded as well. The dataset contains density estimates for a few mainstem (fifth-order and larger; see Strahler 1964) river sites, but the bulk of the data are from smaller tributary streams typically snorkeled by crawling upstream.

Petrosky and Holubetz (1986) provide a more detailed description of snorkel techniques and sampling designs. All sampling occurred in the Salmon River and Clearwater River recovery units only, and sufficient long-term snorkel data were available for calculating trend in 10 core areas.

We examined trend before (when possible) and after bull trout no-harvest regulation changes were made in Idaho in 1994. Because of the wealth of snorkel data available for trend analysis, and in order to obtain adequate temporal data dispersion within consistently monitored sites, we included only those sites where multiple data points were available for each decade from the 1980s to 2000s to examine trend prior to and after the no-harvest regulation changes (n = 367). For the snorkel sites, it was also possible to compare the abundance of bull trout to that for other salmonids. For the weir and redd count data, records were more limited, and we used all available data.

We used linear regression with sample year as the independent variable and $\log_e$-transformations of the redd counts, weir, and snorkel abundance data as the dependent variables to analyze post-1994 trends in abundance. The slopes of the lines are equivalent to the intrinsic rates of change, r, for each population (Maxell 1999), which were then tested for positive slopes (i.e., one-tailed t-tests of the regression coefficients) to see if bull trout were stable or increasing across Idaho since 1994. Following the advice of Maxell (1999) and Peterman (1990), we used $\alpha = 0.10$ in order to increase the power of detecting true positive trends. Because zero values are incompatible with $\log_e$-transformations, and no bull trout were counted in the Lemhi River core area in 1995 (4 sites), and the Upper Salmon River in 1999 (12 sites), we inserted values of 0.01 bull trout per 100 meters for these years in order to calculate post-1994 r for these two core areas.

Approximation of abundance

To approximate bull trout abundance within each recovery unit, and within core areas where possible, we first coded (with the ArcView® geographic information system, ArcGIS) a standard 1:100,000 hydrography layer for bull trout presence using three categories (present, absent, or unknown). This was done with workshops held across the state, where numerous biologists used local knowledge to label all stream segments according to bull trout presence. This methodology followed the protocol that Shepard et al. (2003) and May et al. (2003) used for westslope and Yellowstone cutthroat trout.

We overlaid the stream hydrography layer with all geo-referenced bull trout abundance data we could gather (not at the workshops, but through later direct contact). We considered all snorkeling and electrofishing abundance data we could gather from 1997 to 2004 as useful in approximating current bull trout abundance, but the bulk (88%) of the data was collected from 1999 to 2003. For snorkel (n = 1,255) and one-pass electrofishing (n = 762) study sites, the total number of bull trout observed or captured was conservatively used as a minimal abundance. For multi-pass electrofishing study sites (n = 274), we estimated abundance using the maximum-likelihood method calculated with the MicroFish software package (Van Deventer and Platts 1989). Because no differentiation was made for daytime snorkel counts, one-pass electrofishing capture data, or multi-pass depletion estimates, our abundance estimates should be viewed, according to previous studies of fish estimation techniques, as gross underestimates. Indeed, Thurow and Schill (1996) and Mullner et al. (1998) estimated that daytime snorkeling accounted for only 77 and 65%, respectively, of depletion estimates. Similarly, Kruse et al. (1998) found that one-pass electrofishing accounted for 81% of depletion estimates. Moreover, Peterson et al.

(2004) estimated that even depletion electrofishing for bull trout underestimated true abundance, by an average of 116%. Thus, we realize that our bull trout abundance estimates are most likely extremely conservative, and we used them mainly as a starting point for extinction risk analysis.

We excluded fry from our analyses because of the inefficiencies in capturing them (Reynolds 1996). However, because data were gathered from several sources and collected in a variety of manners, we could not standardize fish size in our estimations of abundance. Subsequently, the percentage of study sites that included bull trout > 70, > 75, or > 100 mm (total length, TL) in abundance estimates were 41, 12, and 47%, respectively. Most (97%) of the data we gathered was collected during low to moderate flow conditions (i.e., between June and September, after spring runoff and before the onset of winter) which helped to standardize efficiencies in snorkel counts and electrofishing capture.

For each core area and recovery unit, we summed the total length of stream for each stream order using ArcGIS. We standardized our estimates of abundance to the number of trout per 100 meters of stream (study sites averaged 96 m in length), calculated a mean abundance within each stream order (with an associated variance), and multiplied that by the number of 100-meter reaches within each stream order, to calculate total abundance (and variance) of bull trout by stream order. We then summed these totals for an overall abundance (and variance) estimate. We used the stratified random sampling formulas from Scheaffer et al. (1996) to calculate population totals, variances, and subsequent 95% confidence intervals (CIs). We did this separately for each bull trout presence category, but because only 7% of the stream segments were coded for bull trout presence as unknown, and only 2% of the study sites were within these unknown stream segments, we lumped unknown and absent stream segments and study sites together for bull trout estimation purposes. We assumed that the data we were able to collect were somewhat randomly distributed, or at least behaved as if they were random, and that by making this assumption we did not inject any directional bias (negative or positive) in our results.

Core areas as defined by the USFWS are analogous to metapopulations since they contain one or more local populations, or groups of bull trout that spawn within a particular stream or portion of a stream system (Lohr et al. 2000). To approximate the number of bull trout in local populations, we divided the estimate of bull trout abundance in a particular core area by the number of local bull trout populations designated within that core area in the USFWS draft recovery plan. Because sample sizes were much smaller within individual local populations, confidence intervals around the abundance estimates were not calculated.

Our methodology included only stream abundance, and because data were unavailable or unreliable for lakes (such as Priest Lakes, Lake Pend O'reille, Arrowrock Reservoir, etc.), they were excluded from analysis.

Population viability analysis

Population viability analyses were performed at 3 spatial scales: state-wide, recovery unit, and core area. The stochastic exponential growth model of Dennis et al. (1991) was used to model population sizes and extinction risks using the computer software STOCHMVP (E.O. Garton, Department of Fish and Wildlife Resources, University of Idaho). Estimated total abundances obtained from the extrapolations explained above, were used as the initial population sizes when available. When specific abundance estimates for a particular core area were unavailable, we divided the recovery unit abundance estimate by the number of core areas within the recovery unit, and used this for the initial population size. The model was run based on 95%

probability of survival for 100 years. The lower thresholds for extinction were arbitrarily set at 10 and 100 individuals to represent elevated extinction risks due to demographic stochasticity and quasi-extinction, respectively (Quinn and Hastings 1987; Rieman and McIntyre 1993). The remaining parameters required by the model - instantaneous rate of population change (μ) and the variance in rate of change (σ^2) - were selected to fit three scenarios: 1) modest population growth and low variance; 2) equilibrium population (i.e., no growth) and modest variance; and 3) declining population and high variance. Values for μ and σ^2 appropriate for these 3 scenarios were selected based on the range of values observed while performing 11 PVAs on GPM and redd count data, where the time series contained a full compliment of data (1994 to 2003). The validity of selected values of μ and σ^2 was further substantiated by the fact that they were similar to those previously observed for bull trout in portions of Idaho and Montana (Rieman and McIntyre 1993). When observed values of μ and σ^2 were available for core areas, they were used in the analysis in addition to the three scenarios. When multiple core areas within a recovery unit had observed values of μ and σ^2 available, these were averaged and used in the PVA model for the recovery unit in addition to the three scenarios.

The modeling approach of Dennis et al. (1991) assumes local populations act independently, but most of the large, relatively pristine drainages in Idaho are known to harbor numerous local bull trout populations within a metapopulation structure (e.g., Bjornn and Mallet 1964; Dunham and Rieman 1999). Although the modeling approach of Dennis et al. (1991) does not incorporate metapopulation theory, we did attempt to assess the relative risk of bull trout extinction across a recovery unit. We did this by estimating the probability of a single large population (composed of multiple local populations) persisting 100 years, using the formula 1 – ($P_1 \cdot P_2 \cdot \ldots P_i$), where P_i is the probability of falling below the extinction threshold (i.e., 100 bull trout) in each of the i local populations (Rieman and McIntyre 1993). Again, values of μ and σ^2 representing the 3 scenarios described above were used to assess the relative extinction risks likely faced. This method still assumed no re-founding events would occur.

Results

The current assessment of bull trout, covering a total of 77,447 km of stream throughout the 7 recovery units in Idaho, was based on 2,287 surveys of bull trout abundance (Table 1). This included 1,565 surveys by IDFG, plus additional data obtained from the U. S. Forest Service (607 study sites), U.S. Bureau of Land Management (59 sites), and U.S. Bureau of Reclamation (56 sites) (Figure 1). Study sites occurred in stream reaches that were 18% first-order, 32% second-order, 23% third-order, 14% fourth-order, 10% fifth-order, and 2% sixth-order. The sample length of all study sites combined totaled 220 km of stream, or 0.3% of the entire stream network within the bull trout recovery units in Idaho. Bull trout were designated to occur in 14,551 km of stream (19%) within the recovery units.

Bull trout were captured in 871 (38%) of the study sites, including 823 (47%) of 1,762 study sites within the "bull trout present" stream segments (Table 1). Of the 485 study sites within the "bull trout absent" stream segments, bull trout were captured at 46 (9%) sites. Two of 40 sites within the unknown stream segments contained bull trout. Bull trout were most likely to occur at study sites in 2nd-order (present in 42% of sites) and 3rd-order (42%) streams, and least likely to occur at study sites in streams 5th-order and higher (20%).

Population trends

The trend data we were able to gather indicated that, in general, bull trout abundance appeared to decline through the mid-1990s (9 of 14 available trends), but has been at least stable and usually increasing across most of their range in Idaho over the last 10 years (Table 2, Figures 2 and 3). Intrinsic rate of growth (r) since 1994 was positive for 14 of 18 estimates, and 90% CIs did not overlap zero for 5 estimates (Table 2). Population trend since 1994 was consistently positive in the Salmon River (7 of 9 estimates) and Clearwater River (5 of 5 estimates) recovery units in particular, whereas in the remaining three recovery units for which trend data were available, population trend was positive in 2 of 4 estimates (Table 2).

Although increasing trends in bull trout abundance appeared to coincide with the implementation in 1994 of state-wide no-harvest regulations for bull trout, we found that for the long-term snorkel data in the Salmon River and Clearwater River recovery units (i.e., the GPM dataset), abundance of all other species of salmonid increased at the same time (Figure 4). Estimates of r for the GPM dataset was 0.11 for bull trout, and ranged from 0.07 to 0.20 for all other salmonids (Figure 5); no confidence intervals included zero, indicating that growth values for all species of salmonid were statistically positive. Bull trout abundance was positively correlated with abundance for all other salmonids, but was most strongly correlated with westslope cutthroat trout (Table 3). There were no negative correlations in abundance between any two species of salmonid.

Extrapolation of Abundance

We estimated there was approximately 1.24 million bull trout in the 7 recovery units in Idaho (Table 4). Sixty-two percent (0.76 million bull trout) of this abundance was estimated to occur within stream segments designated as containing bull trout, while the remaining 38% (0.47

million) was estimated to occur in stream segments designated to lack bull trout or where their status was unknown. Of the bull trout estimated to occur outside the stream reaches currently designated as containing bull trout, 78% was estimated to occur in first-order streams.

Over one-half (0.64 million) of the overall number of bull trout were estimated to occur in the Salmon River recovery unit, followed by the Southwest Idaho recovery unit (0.17 million). Estimates could not be made for the Coeur d'Alene River recovery unit, and estimates for two other recovery units (Kootenai River and Clark Fork River) were based on data from only a few study sites. Considering that the USFWS draft recovery plan designated 269 local populations within the areas we were able to approximate abundance, we estimated that an average local population contained about 4,600 bull trout, ranging from a low of 1,031 in the Clearwater River recovery unit to a high of 5,093 in the Salmon River recovery unit.

Nearly all (95%) of the overall abundance of bull trout occurred in first- through third-order streams (Figure 5). First-order streams comprised 46% of the total stream kilometers and 57% of the abundance, but only 18% of the study site surveys (Figure 5). Mean linear bull trout density (> 100 mm TL) at all study sites was highest in the Clark Fork River (22.1/100 m) and Little Lost River (18.4/100 m) recovery units, and lowest in the Clearwater River (1.2/100 m), Southwest Idaho (3.6/100 m), and Salmon River (4.4/100 m) recovery units (Figure 6). Average recovery unit density was 10.5 bull trout/100 m.

Population viability analysis

Probabilities of persistence for bull trout ranged from 0.37 to > 0.95, 0.17 to > 0.95, and 0.10 to > 0.95 for statewide, recovery unit, and core area spatial scales, respectively (Tables 5 and 6). Probabilities of persistence for bull trout populations, when observed values were used for μ and σ^2, ranged from 0.55 to > 0.95 (Tables 5 and 6). In general, probabilities of persistence were similar for all recovery units and core areas for each scenario of population growth and variance, indicating model results were more dependent on values of μ and σ^2 and less dependent on initial population size.

When persistence probabilities for populations of varying size were multiplied to assess relative risk of extinction for recovery units with multiple local populations, results indicate that the maximum number of local populations needed to ensure 95% probability survival of at least one local population ranged from 7 to 33 with initial population sizes ranging from 50,000 to 1,000, respectively (Figure 7). Given the average number of bull trout per local population in each recovery unit and the number of local populations each recovery unit contained, all recovery units would be expected have at least one local population persisting for the next 100 years given stable or increasing populations with moderate to low variance. Five of Idaho's seven recovery units fit in the conservative area of the graph (i.e., despite stable or decreasing population size and moderate to high variance, there is > 0.95 probability of persistence for at least one local population in the recovery unit). The Kootenai River Basin and Coeur d'Alene Lake Basin were two exceptions, requiring modest population growth and low variance conditions for probabilities of persistence to exceed 0.95 for at least one local population (Figure 7).

Discussion

Our results indicate that the bull trout are increasing or at least stable across most of their range in Idaho, that their distribution remains widespread, that they remain relatively abundant, and that their risk to extinction in Idaho was very low. We estimated that well over 1 million bull trout > 100 mm TL were scattered across 7 recovery units, and this was likely a gross underestimation of overall abundance (see below). Such findings suggest that, despite undoubtedly substantial declines from historical levels, bull trout in Idaho remain relatively secure, and appear to contain numerous strong populations, at least in terms of distribution and abundance. When considering that Idaho likely contains at least 1 million bull trout, and that Idaho constitutes only a small portion of the current range for the species, it is difficult to believe that they are at any threat to range-wide extinction in the near future.

There are some obvious limitations with the data used to approximate bull trout abundance, which may have resulted in biased estimates. First, there is much uncertainty regarding the upper range limit of bull trout in each stream, but our methodology usually assumed they were distributed to the uppermost end of perennial streamflow (i.e., the biologists did not usually attempt to pinpoint the exact upper range for each stream). If the distribution of our abundance data points and the actual distribution of bull trout were unequally balanced or biased in these upper reaches of first-order streams, we may have obtained biased bull trout abundance estimates (most likely, overestimations). For that matter, estimates for other stream orders may have been equally biased if the distribution of the data points and the actual distribution of bull trout distribution were incongruent, although that likelihood was lower for higher order stream segments. Another limitation (probably the biggest) was the use of snorkel and depletion (mostly one-pass) electrofishing to estimate bull trout abundance, methods which drastically underestimate true abundance (Thurow and Schill 1996; Mullner et al. 1998; Kruse et al. 1998; Peterson et al. 2004). Furthermore, we did not include bull trout < 100 mm TL, which ostensibly removes a large portion of overall bull trout abundance. Finally, the collection of fish abundance and distribution data at only one point in time may not accurately reflect true abundance when considering the amount of seasonal and annual variation known to exist in trout populations (e.g., Platts and Nelson 1988; Decker and Urman 1992; House 1995), although directional bias is unlikely from this limitation. Considering all potential sources of bias, we believe that bull trout abundance was most likely grossly underestimated for most core areas and recovery units. Nevertheless, we felt they served as good starting points for PVA modeling of extinction risk.

Bull trout trend was commonly negative prior to 1994, but since then it appears that bull trout have been recovering in abundance across much of their range in Idaho, especially in the Salmon River and Clearwater River recovery units. Unfortunately, available data was lacking in the Southwest Idaho and Kootenai River recovery units, and little was available for the Little Lost, Coeur d'Alene, and Clark Fork recovery units. The post-1994 increasing trend appeared to coincide with the implementation of state-wide no-harvest regulations for bull trout. However, the fact that all other fish increased at the same time suggests that the regulation changes were not responsible for the increased abundance. That evidence of increasing abundance was strongest in the two recovery units with anadromous salmonids, and that Chinook salmon and steelhead trout also increased during this time period, suggests a possible link between increased anadromous productivity in these recovery units and positive bull trout population growth. However, data are too minimal for us to draw strong conclusions, other than that it does appear that bull trout are stable or increasing in most recovery units and core areas in Idaho where data were available.

In general, PVA analysis on bull trout populations in Idaho suggests that future prospects of survival bodes well for the species. However, discussion of PVA results should be prefaced by stating that model predictions should be viewed cautiously and treated as testable hypotheses

(Reed et al. 2002; Shafer et al. 2002; Ralls et al. 2002). After all, there is no certain way to accurately predict events in the future. That being said, PVAs can provide strong input in management and regulatory decisions, and its benefits generally outweigh its limitations (Lindenmayer et al. 1993; Brook et al. 2000; Coulson et al. 2001). Brook et al. (2000) determined that PVA is a useful tool in making management decisions about threatened or endangered species because predictions are surprisingly accurate when quality data are used and when estimates of vital rates and their variances do not significantly change in the future. Coulson et al. (2001) defined quality PVA data as those datasets when 10 or more years of data are included; our data typically included 10 years of trend data (1994 to 2003).

Despite these favorable reviews of PVA, we opted to stay conservative where possible when estimating probabilities of persistence. This was accomplished by using conservative estimates of bull trout population sizes, relatively high thresholds of extinction, and by using the stochastic exponential growth model of Dennis et al. (1991), which has an inherently pessimistic nature when estimating probabilities of persistence (Belovsky et al. 2002), in part because the model does not account for density dependence or refounding (Rieman and McIntyre 1993). Conversely, persistence probabilities may be optimistic because the model lacks ability to account for deterministic factors and has limited ability to incorporate catastrophes. Overall, we believe those factors affecting the estimates in a conservative manner outweigh those that influence the model optimistically.

Probabilities of persistence were very similar among recovery units and core areas for each of the three scenarios. These results are similar to other PVA studies and reviews, which found estimates of persistence or extinction risks can be disproportionately influenced by μ (Dennis et al. 1991; Ludwig 1999; Ellner et al. 2002) and σ^2 (Rieman and McIntyre 1993).

While bull trout behavior and the juxtaposition of many local populations across the state of Idaho indicates this is likely not true in the light of metapopulation theory, we feel that assessing relative risk in this manner adds another level of conservatism to our estimates. Across the range of bull trout there is evidence of functioning metapopulations (Whiteley et al. 2004; Rieman and Dunham 2000), as well as data that fails to support metapopulation functionality (Spruell et al. 1999; Kanda and Allendorf 2001).

The relative extinction risk to bull trout in Idaho also greatly depends on the number of local populations. For example, despite the fact that we conservatively estimated there was over 1 million bull trout in Idaho, persistence probability at our pessimistic outlook for the future is estimated to be only 38 %. However, when the risk of extinction due to stochastic environmental conditions is spread over multiple local populations, only XXX local populations are needed to ensure persistence in 100 years at the 0.95 confidence level.

Theoretically, in the absence of nonnative trout, existing or additional habitat alterations may lead to continued or even further fragmentation of local populations of bull trout, but it is unlikely that current or further fragmentation would soon threaten the existence of the species in Idaho. However, that most recovery units can be divided into several (as many as 126) local populations that do not exchange gene flow on a regular basis, does suggest that many local populations of bull trout are facing a variety of risks inherent to their low abundance and fragmented existence, both directional (compensation and depensation) and random (catastrophes, and demographic, genetic, and environmental stochasticity) in effect. Small populations have been shown to lose adaptive genetic variation and gain maladaptive genetic variation at higher rates than larger populations (Lande 1995). However, most literature addressing small population sizes do not refer to species that contain at least 1 million individuals (even just within Idaho) scattered across a large geographic area and broken into numerous populations, some of which are extremely large.

At a smaller scale, it is difficult to resolve exactly how many bull trout are needed in local populations for long-term persistence and the maintenance of genetic diversity within that particular population, and there is no generally agreed upon standard. Franklin (1980) and Soule (1980) proposed that an N_e of at least 50 individuals are necessary for conservation of genetic diversity in the short term (i.e., several generations) to avoid inbreeding depression, while an N_e of 500 is needed to avoid serious genetic drift in the long term. Hilderbrand and Kershner (2000) suggested that census populations of at least 2,500 cutthroat trout were needed to avoid inbreeding depression, whereas Rieman and Allendorf (2001) recommended populations of at least 1,000 spawning adult bull trout to maintain genetic variation indefinitely. Regardless of how many individuals are needed to maintain genetic diversity, Lande (1988) argued that demography is likely to be more important in determining population viability.

We chose not to quantify the amount of bull trout historical range currently occupied, because of the difficulty in delineating actual historical distribution.

Conclusion

Despite the obvious substantial declines in distribution and abundance from historical levels, we found that bull trout were the widespread, numerous, and often increasing in abundance across their historical range in Idaho. We also found, however, that bull trout were often divided into numerous small and often unconnected local populations. Dual and seemingly conflicting conservation strategies of isolating local populations at risk of competition and introgression with nonnative salmonids, yet connecting local populations where feasible, must be balanced in their implementation for sound adaptive management of bull trout in Idaho. The distribution of nonnative salmonids must be controlled and reduced through selective removal or use of chemicals where feasible. Establishing and regularly monitoring bull trout presence and abundance, especially in recovery units and core areas where current data is lacking, would more completely address current trends and factors that influence that trend. Preserving meta-population function and multiple life-history strategies by connecting occupied habitats would help preserve more local populations. Where bull trout populations are depressed (< 500 adults) but seemingly stable, a temporary program of fish translocations may be needed to avoid inbreeding depression problems. The possibility of re-connecting isolated populations should be considered, keeping in mind the hybridization, competition, and disease risks such actions may pose. Such management objectives and activities will help insure that bull trout are a sustained part of the fish fauna of the state of Idaho.

References

Behnke, R. J. 2002. Trout and salmon of North America. The Free Press, New York.

Beissinger, S. R. and M. I. Westphal. 1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management 62:821-841.

Belovsky, G. E., C. Mellison, C. Larson, and P. A. Van Zandt. 2002. How good are PVA models? Testing their predicitions with experimental data on the brine shrimp. Pages 257-283 *in* S. R. Beissinger and D. R. McCullough, editors. Population viability analysis. The University of Chicago Press. Chicago, Illinois.

Bjornn, T. C., and J. Mallet. 1964. Movements of planted and wild trout in an Idaho River system. Transactions of the American Fisheries Society 93:70-76.

Brook, B. W., J. J. O'Grady, A. P. Chapman, M. A. Burgman, H. R. Akcakaya, and R. Frankham. 2000. Predictive accuracy of population viability analysis in conservation biology. Nature 404:385-387.

Coulson, T., G. M. Mace, E. Hudson, H. Possingham. 2001. The use and abuse of population viability analysis. Trends in Ecology and Evolution 16:219-221.

Decker, L. M., and D. C. Erman. 1992. Short-term seasonal changes in composition and abundance of fish in Sagehen Creek, California. Transactions of the American Fisheries Society 121:297-306.

Dennis, B. P. L. Munholland, and J. M. Scott. 1991. Estimation of growth and extinction parameters for endangered species. Ecological Monographs 61:115-143.

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

Ellner, S. P., J. Fieberg, D. Ludwig, and C. Wilcox. Precision of population viability analysis. Conservation Biology 16:258-261.

Fieberg, J., and S. P. Ellner. 2000. When is it meaningful to estimate an extinction probability? Ecology 81:2040-2047.

Gilpin, M. E., and M. E. Soulé. 1986. Minimum viable populations: processes of species extinction. Pages 13-34 *in* M. E. Soulé, editor. Conservation biology: the science of scarcity and diversity. Sinauer Associates, Sunderland, Massachutes.

House, R. 1995. Temporal variation in abundance of an isolated population of cutthroat trout in western Oregon, 1981-1991. North American Journal of Fisheries Management 15:33-41.

Kruse, C. G., W. A. Hubert, and F. J. Rahel. 1998. Single-pass electrofishing predicts trout abundance in mountain streams with sparse habitat. North American Journal of Fisheries Management 18:940-946.

Lindenmayer, D. B., T. W. Clark, R. C. Lacy, and V. C. Thomas. 1993. Population viability analysis as a tool in wildlife conservation policy: with reference to Australia. Environmental Management 17:745-758.

Lohr, S., T. Cummings, W. Fredenberg, and S. Duke. 2000. Listing and recovery planning for bull trout. Pages 80-87 in D. Schill, S. Moore, P. Byorth, and B. Hamre, editors. Wild Trout VII: Management in the new millennium, are we ready? Yellowstone National Park, Wyoming.

Ludwig, D. 1999. Is it meaningful to estimate a probability of extinction? Ecology 80:293-310.

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

May, B. E., W. Urie, and B. B. Shepard. 2003. Range-Wide Status of Yellowstone Cutthroat Trout (*Oncorhynchus clarki bouvieri*): 2001. Multi-state assessment document.

Mullner, S. A., W. A. Hubert, and T. A. Wesche. 1998. Snorkeling as an alternative to depletion electrofishing for estimating abundance and length-class frequencies of trout in small streams. North American Journal of Fisheries Management 18:947-953.

Peterman, R. M. 1990. Statistical power analysis can improve fisheries research and management. Canadian Journal of Fisheries and Aquatic Sciences 47:2.15.

Peterson, J. T., R. F. Thurow, and J. W. Guzevich. 2004. An evaluation of multipass electrofishing for estimating the abundance of stream-dwelling salmonids. Transactions of the American Fisheries Society 133:462-475.

Petrosky, C. E., and T. C. Holubetz. 1986. Idaho habitat evaluation of off-site mitigation record. Annual report to Bonneville Power Administration, Project 83-7. Idaho Department of Fish and Game, Boise.

REMOVE Platts, W. S. 1979. Relationships among stream order, fish populations, and aquatic geomorphology in an Idaho river drainage. Fisheries 4(2):5-9.

Platts, W. S., and R. L. Nelson. 1988. Fluctuations in trout populations and their implications for land-use evaluation. North American Journal of Fisheries Management 8:333-345.

Quinn, J. F., and A. Hastings. 1987. Extinction in subdivided habitats. Conservation Biology 1:198-207.

Ralls, K., S. R. Beissinger, and J. F. Cochrane. 2002. Guidelines for using population viability analysis in endangered-species management. Pages 521-550 *in* S. R. Beissinger and D. R. McCullough, editors. Population viability analysis. The University of Chicago Press. Chicago, Illinois.

Ratliff, D. E., and P. J. Howell. 1992. The status of bull trout populations in Oregon. Pages 10-17 *in* P. J. Howell and D. V. Buchanan, editors. Proceedings of the Gearhart Mountain bull trout workshop. American Fisheries Society, Oregon Chapter, Corvallis.

Reed, J. M., L. S. Mills, J. B. Dunning Jr., E. S. Menges, K. S. McKelvey, R. Frye, S. R. Beissinger, M. Anstett, and P. Miller. 2002. Emerging issues in population viability analysis. Conservation Biology 16:7-19.

Reynolds, J. B. 1996. Electrofishing. Pages 221-254 *in* B. Murphy and D. Willis, editors. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

REMOVE Rieman, B. E., and F. W. Allendorf. 2001. Effective population size and genetic conservation criteria for bull trout. North American Journal of Fisheries Management 21:756-764.

Schaffer, M. L. 1981. Minimum viable population sizes for conservation. Bioscience 31:131-134.

Schafer, M., L. H. Watchman, W. J. Snape III, and I. K. Latchis. 2002. Population viability analysis and conservation policy. Pages 123-142 *in* S. R. Beissinger and D. R. McCullough, editors. Population viability analysis. The University of Chicago Press. Chicago, Illinois.

Scheaffer, R. L., W. Mendenhall, and L. Ott. 1996. Elementary survey sampling, fifth edition. Duxbury Press. Belmont, CA.

Shepard, B. B., B. May, and W. Urie. 2003. Status of westslope cutthroat trout (*Oncorhynchus clarki lewisi*) in the United States. Multi-state assessment document.

Soulé, M. E. 1987. Viable populations for conservation. Cambridge University Press, Cambridge, England.

Strahler, A. N. 1964. Quantitative geomorphology of drainage basins and channel networks. Section 4-2 *in* V. T. Chow, editor. Handbook of Applied Hydrology. McGraw-Hill, New York, New York.

Thurow, R. F., and D. J. Schill. 1996. Comparison of day snorkeling, night snorkeling, and electrofishing to estimate bull trout abundance and size structure in a second-order Idaho stream. North American Journal of Fisheries Management 16:314-323.

U. S. Fish and Wildlife Service. 1998. Final rule to list Columbia River and Klamath River population segments of the bull trout as a threatened species. Federal Register 63:31647-31674.

Van Deventer, J., and W. S. Platts. 1989. Microcomputer software system for generating population statistics from electrofishing data - user's guide for MicroFish 3.0. U.S. Forest Service General Technical Report INT-254.

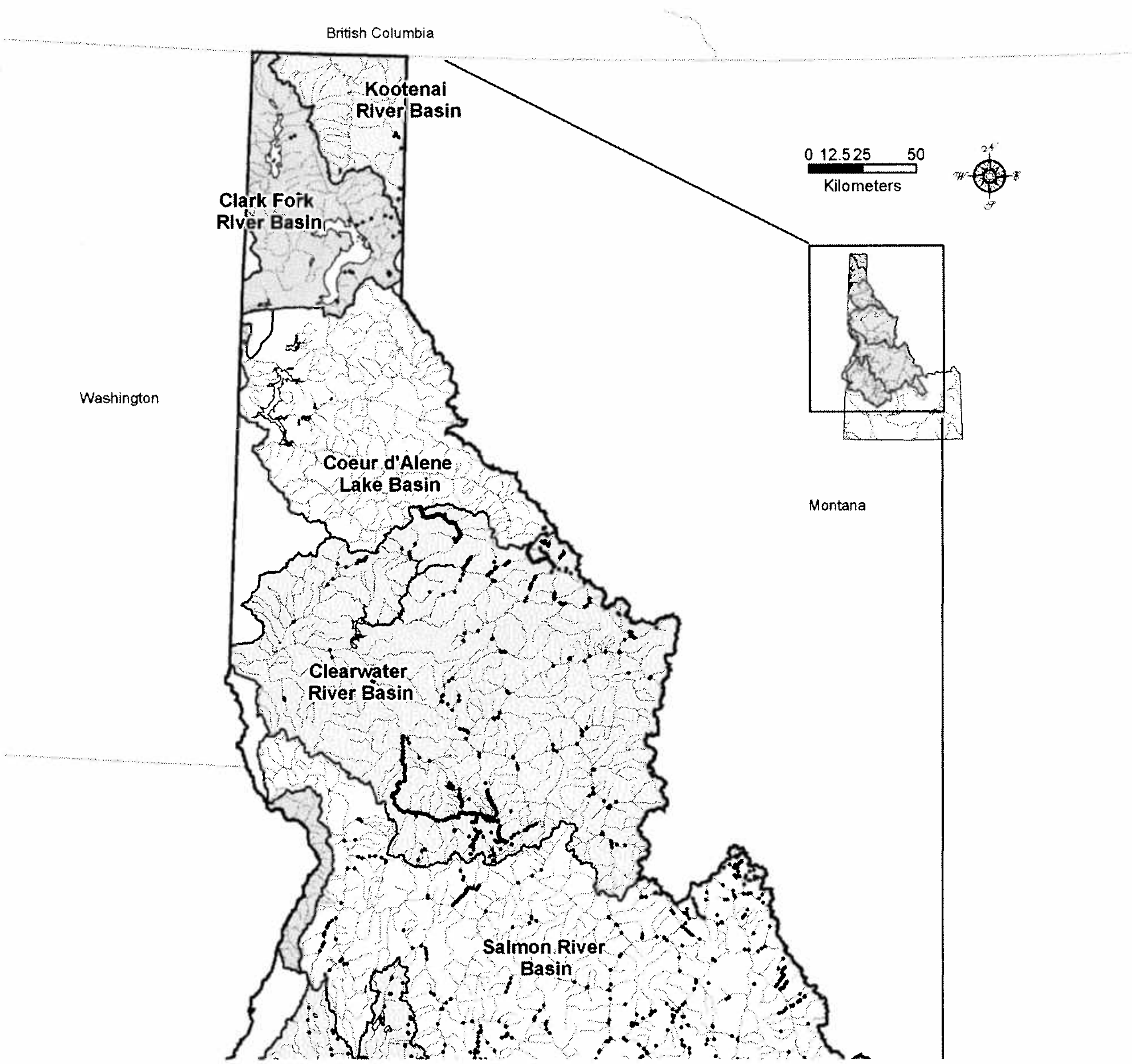

Figure 1. Distribution of study sites (dots) within the bull trout recovery areas in Idaho.