



**WILD TROUT EVALUATIONS: WESTSLOPE CUTTHROAT
TROUT TRENDS, YELLOWSTONE CUTTHROAT TROUT
STREAM PURIFICATION, PELICAN PREDATION, AND PIT-
TAG RETENTION**



Report Period July 1, 2013 to June 30, 2014

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**IDFG Report Number 14-05
February 2014**

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By

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**IDFG Report Number 14-05
April 2014**

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CHAPTER 1: WESTSLOPE CUTTHROAT TROUT TRENDS IN ABUNDANCE FOR IDAHO POPULATIONS

ABSTRACT

Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* are the most widely distributed subspecies of Cutthroat Trout in western North America and are an important resource for anglers. In Idaho, Westslope Cutthroat Trout populations exist in about 50% of their historic range. Numerous assessments have been conducted for Westslope Cutthroat Trout across the region in the past. To update trends in Westslope Cutthroat Trout abundance, we incorporated several years of recent data as well as additional data sets that were previously unpublished. We also examined several bioclimatic indices to help understand what environmental factors might be influencing Westslope Cutthroat Trout abundance. Lastly, we evaluated observation error between several survey methods. Seventeen data sets including ten populations and three survey methods were included. Abundances for Westslope Cutthroat Trout were increasing for 10 data sets, declining for 2 data sets, and stable for 5 data sets where the confidence intervals spanned zero. Covariates included in abundance models generally explained a low amount of the variation in Westslope Cutthroat Trout abundance. Chinook Salmon *O. tshawytscha* abundance was significantly related to Westslope Cutthroat Trout abundance most often; however, the influence of Chinook Salmon to Westslope Cutthroat Trout were equally likely to be positive or negative. The presence of high observation error was equally likely for snorkel data and screw trap data (50%). The results of our study suggest that most Westslope Cutthroat Trout populations in Idaho are currently stable or increasing in abundance.

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INTRODUCTION

Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* are the most widely distributed subspecies of Cutthroat Trout in western North America (Behnke 1992) and provide an important resource for anglers (Mallet 2013). In Idaho, Westslope Cutthroat Trout are native to the Salmon River and all major drainages north through the Idaho Panhandle (Wallace and Zaroban 2013).

In Idaho, populations exist in about 50% of their historic range (Wallace and Zaroban 2013). Concerns about the status of Westslope Cutthroat Trout have resulted in two petitions for listing under the U. S. Endangered Species Act (ESA) in 1997 and in 2001. Subsequent assessments found that this species did not warrant protection by the ESA because of their wide distribution in Idaho and Montana and the presence of isolated populations in Washington, Oregon, and Canada (U.S. Federal Register 1998, 2003). Still, the U.S. Forest Service and the Bureau of Land Management regard Westslope Cutthroat Trout as a sensitive species, and the Idaho Department of Fish and Game (IDFG) has designated it as a Species of Greatest Conservation Need (IDFG 2006). In 2013, IDFG published a management plan which outlined the status and conservation strategies for Westslope Cutthroat Trout in Idaho (IDFG 2013).

Many previous assessments of abundance or distribution have been conducted for Westslope Cutthroat Trout across their native range (e.g., Rieman and Apperson 1989; McIntyre and Rieman 1995; Thurow et al. 1997; Schill et al. 2004; Shepard et al. 2005; May 2009), but few have included analyses of trends in abundance. Schill et al. (2004) summarized a number of temporally and spatially extensive trend monitoring data sets in Idaho and concluded that Westslope Cutthroat Trout abundance in Idaho was generally stable or increasing. With a decade of additional data now available, our first objective was to re-evaluate trends in abundance for Westslope Cutthroat Trout in Idaho, including the addition of previously unpublished data sets.

Trend monitoring data for stream-dwelling salmonids are often subject to substantial observation error (Thurow and Schill 1996; Dunham et al. 2001; Thurow et al. 2006; Muhlfeld et al. 2006), which can diminish the ability to detect statistically significant changes in abundance (Dunham et al. 2001). To evaluate how observation error might be influencing our conclusions, a secondary objective of this study was to estimate how much observation error was present in the Westslope Cutthroat Trout abundance data.

Finally, a number of studies have shown relationships between environmental conditions in streams and the distribution and abundance of resident salmonids (e.g., Jackson et al. 2001; de la Hoz Franco and Budy 2005; Quist and Hubert 2005). Such relationships, if causative, could be useful in explaining any patterns we observed in Westslope Cutthroat Trout occupancy and abundance. Therefore, our final objective was to examine relationships between several common bioclimatic indices and Westslope Cutthroat Trout abundance.

METHODS

Areas of inference

Westslope Cutthroat Trout occurrence is well understood in Idaho, but metapopulation boundaries have not been well defined. Geographic management units (GMUs) have been delineated for Westslope Cutthroat Trout in Idaho (IDFG 2013; Figure 1). Geographic

management units were established to provide spatial reference for conservation efforts, but GMUs may include several river drainages and multiple Westslope Cutthroat Trout populations. Studies have shown that resident and adfluvial Westslope Cutthroat Trout can move between large drainages (Mallet 1963; Bjornn and Mallet 1964; Zurstadt and Stephan 2004; Schoby and Keeley 2011). We made inferences on Westslope Cutthroat Trout trends at the smallest scale possible, which were usually major river drainages. Areas of inference will hereafter be referred to as populations, though we acknowledge that several Westslope Cutthroat Trout populations may exist within these aggregates.

Available data

Snorkel data

Within nine populations, daytime summer snorkel surveys were used to index Westslope Cutthroat Trout abundance. Between one and five observers (depending on stream width) snorkeled slowly upstream or downstream counting and identifying all salmonids ≥ 75 mm total length (TL; see Copeland and Meyer 2011 for additional detail). Because Westslope Cutthroat Trout conceal themselves in the substrate at low temperatures (Griffith and Smith 1993; Jakober et al. 2000) and such behavior would have biased snorkel counts, we discarded all surveys conducted at water temperatures $< 6^{\circ}\text{C}$. Similarly, to reduce the potential for bias, we also discarded surveys when water visibility was < 2 m (Thurow 1994). During snorkel surveys, the identification of Westslope Cutthroat Trout can be confused with Rainbow Trout *Oncorhynchus mykiss* at TL ≤ 155 mm (T. Copeland, IDFG, personal communication); therefore, to also reduce the potential for biased estimates, all Cutthroat Trout ≤ 155 mm were removed from the snorkel survey data.

For many Cutthroat Trout populations, the same reaches were surveyed each year. However, in the Salmon and Clearwater basins, while there were typically dozens of reaches surveyed annually within each population, which reaches were surveyed in any given year was inconsistent. To help ensure that the snorkel data represented true Westslope Cutthroat Trout trends in abundance, rather than spatial variability in abundance, we only included reaches that were surveyed in consecutive years in order to obtain unbiased estimates of trends in abundance (following Connelly et al. 2004). For example, if snorkel surveys within a population were conducted at 20 reaches in 2002 and 30 reaches in 2003 but only 10 reaches were surveyed in both years, then trend from 2002 to 2003 was based only on data from the 10 shared reaches. After these criteria were applied to the data, densities (fish/m²) were calculated for each transect.

Screw trap data

Within four populations, 1.52 m rotary screw traps (Kennen et al. 1994) were used to capture Westslope Cutthroat Trout during routine monitoring of the anadromous salmonid outmigration (Copeland and Venditti 2009). Screw traps were deployed as early as possible in the spring, usually during the last week of February or the first week of March, and operated until ice-up (usually the first week of December). Screw trap data were included when a minimum of ten continuous years of data were available from a consistent sample location. Additional years of data may have been available at selected traps, though several years were discarded because of unequal effort between years, as determined by the total number of days sampled. This most often resulted in excluding a couple years early in the time series before sampling protocols were standardized. Westslope Cutthroat Trout (> 50 mm) were summed to

estimate total annual catch, which was used as an index of Westslope Cutthroat Trout abundance for each population.

Angling data

For many decades, survey trips have been conducted through the Middle Fork Salmon River and the Selway River to monitor resident and anadromous fish populations in the wilderness areas (Hand et al. 2012; Flinders et al. 2013). Hook and line surveys are one of the many types of data collected as crews descend the river. Angling surveys are used to monitor catch rates and average size of Westslope Cutthroat Trout, which describe the effectiveness of catch-and-release regulations implemented in the 1960s and 1970s (Mallet 2013). Westslope Cutthroat Trout of all size classes were summed to estimate total annual catch, which was used as an index of Westslope Cutthroat Trout abundance for these populations.

Estimating trends in abundance

We assessed trends in Westslope Cutthroat Trout abundance with least squares regression, using sample year as the independent variable and the index of abundance (\log_e -transformed) as the dependent variable (Meyer et al. 2014). A benefit of this approach is that the slope of the regression line fit to the \log_e -transformed abundance data is equivalent to the intrinsic rate of change (r_{intr}) for the population (Maxell 1999) and produces unbiased estimates of r_{intr} despite the potential presence of observation error within the data (Humbert et al. 2009). Values of $r_{intr} < 0$ indicate population declines whereas $r_{intr} > 0$ indicate population growth. We used a significance level of $\alpha = 0.10$ to increase the probability of detecting statistically significant trends (Peterman 1990; Maxell 1999).

Observation error

A Gompertz state-space model (Dennis et al. 2006) was used to estimate observation error for each sampling method in each population (Meyer et al. 2014). This model estimates the amount of observation or sampling error ($\hat{\tau}^2$) in abundance monitoring data that otherwise would be ascribed to process noise ($\hat{\sigma}^2$). The formula for the model is as follows:

$$\hat{r}_t = \hat{a} - \hat{b} \ln N_t + \hat{\tau}^2 + \hat{\sigma}^2,$$

where \hat{r}_t is the estimated instantaneous rate of change in year (t ($\ln N_t + 1 - \ln N_t$)), \hat{a} is the estimated intercept, \hat{b} is the estimated slope (a measure of the strength of density dependence), $\hat{\tau}^2$ is the estimated observation error, and $\hat{\sigma}^2$ is the estimated process noise (a measure of environmental and demographic variation).

We identified data sets with estimates of minimal observation error, which we arbitrarily set at $\hat{\tau}^2 < 0.10$; we assumed that minimal observation error may have only slightly inflated the error bounds on the estimates of trend (Meyer et al. 2014); We assumed that estimates of $\hat{\tau}^2 \geq 0.10$ would have produced error bounds around trend estimates that may have been substantially inflated and thus may have masked what otherwise would have been statistically significant trends.

Explanatory variables

Similar to Copeland and Meyer (2011), we assessed how abundance was influenced by several abiotic bioclimatic indicators including drought severity, mean winter stream flow, mean

annual air temperature, and the annual abundance of Snake River spring-summer Chinook Salmon *Oncorhynchus tshawytscha* (hereafter Chinook).

Drought has been implicated as a primary abiotic factor affecting Cutthroat Trout populations (Dunham et al. 1999; Haak et al. 2010; Gresswell 2011). Drought can result in direct (e.g., desiccation) and indirect (e.g., reduced forage) mortality fishes, or can cause shifts in distribution or species composition (Mathews and Marsh-Mathews 2003). To assess whether drought influenced Westslope Cutthroat Trout abundance, we compared the \log_e -transformed Westslope Cutthroat Trout abundance indices to the mean annual Palmer Drought Severity Index (PDSI) computed for each population by the National Climatic Data Center (Heddinghaus and Sabol 1991). A point local to each population was selected from an area central to the population and along each respective stream channel.

Stream flow is a common factor influencing fish survival for all salmonid life stages (Bjornn and Reiser 1991). Occupancy and carrying capacity can be altered with reduced stream flow (Mathews and Marsh-Mathews 2003; Copeland and Meyer 2011). We chose mean winter stream flow because the effect of anchor ice on egg-to-fry survival and habitat suitability has been identified as an important factor in other Cutthroat Trout populations (Jakober et al. 1998; Budy et al. 2012). Mean winter stream flow was calculated for December through February from the U.S. Geological Survey gauge station located most centrally within each population. For the Upper Salmon, Middle Fork Salmon, and the Middle Salmon rivers, several years of discharge were missing from the record when compared to the record of abundance. Reducing the time series and evaluating the relationship between abundance indices and available discharge truncated the data sets substantially; therefore we discarded discharge from the multiple regression analyses for these populations.

Water temperature can affect growth, survival, and migration timing for salmonids (Bjornn 1971; Scarnecchia and Bergersen 1986; Budy et al. 2012) and can also determine habitat suitability and carrying capacity for a given stream (Jakober et al. 1998). We used mean annual air temperature as a surrogate for water temperature because it was available for the entire record of our snorkel survey time series. Mean annual air temperature was calculated from the West Wide Drought Tracker. A point location was arbitrarily selected near the center of the dendritic stream network of each population.

As adult Chinook return to freshwater spawning grounds, they bring marine derived nutrients they have incorporated into their body tissues from years of feeding in the ocean. Increased Chinook abundance has been identified as a factor related to growth and fitness of resident fishes (Wipfli et al. 2003; Rinella et al. 2011). The number of Chinook redds was totaled annually for each Westslope Cutthroat Trout population. Exceptions were the Coeur d'Alene River and the St. Joe River because those rivers are outside of the natural range of anadromous salmon. Also, the Middle Fork Salmon River main stem and tributary abundance indices were combined for this comparison because the occurrence of Chinook spawning in the main stem is low relative to the tributaries.

Correlation coefficients (r) were calculated and a correlation matrix was used to assess multicollinearity between bioclimatic variables. We evaluated whether each bioclimatic variable was related to the \log_e -transformed Westslope Cutthroat Trout abundance metric using the correlation coefficients ($\alpha = 0.05$). However, because each variable could have potentially effected recruitment or had other delayed impacts that outweighed effects on within-year abundance, we related each bioclimatic variable to Westslope Cutthroat Trout abundance at one-year time lags as well (Copeland and Meyer 2011). Two-year time lags were assessed for

Chinook redds because Westslope Cutthroat Trout data were collected before Chinook returned to Idaho in a given year. Effects of Chinook abundance would not likely impact recruitment in the given year, but potentially would in subsequent years.

We fit multiple regression models to each data set using PROC REG in SAS (SAS Institute 2009). Akaike's information criterion corrected for small sample size (AIC_c) was used to compare the relative performance of each model. Akaike weights (w_i) were calculated to describe the likelihood that a particular model was the most correct model among all models (Burnham and Anderson 2002). Analysis of variance (ANOVA) tests were performed using PROC REG in SAS to test the significance of each variable in a given model. Additionally, r^2 or adjusted r^2 was used to describe the approximate amount of variation in annual rates of change explained by each model (Hamilton 1992). Residuals of each model were evaluated with Shapiro-Wilks and Kolmogorov-Smirnov tests ($\alpha = 0.05$) using PROC UNIVARIATE in SAS (SAS Institute 2009) to test for outliers, influential data points, unequal variance, and lack of normality.

RESULTS

Areas of inference

The spatial resolution of the Westslope Cutthroat Trout abundance data varied between major drainages. Many of the areas where we estimated abundance aligned with IDFG Westslope Cutthroat Trout GMUs (Figure 1; IDFG 2013), and in some areas we were able to estimate abundance at smaller scales (e.g., Coeur d'Alene and Middle Fork Salmon GMUs).

In northern Idaho, abundance data sets were available for two different populations within the Coeur d'Alene GMU: the Coeur d'Alene River, and the St. Joe River. In the Clearwater River basin, we estimated trends in abundance for the Lochsa GMU, the Selway GMU, and the South Fork Clearwater GMU.

In the Salmon River basin, we estimated trends in abundance in the South Fork Salmon GMU, the Middle Fork Salmon GMU, and several aggregated GMUs in the Middle Salmon. We split the Middle Fork Salmon River GMU and estimated trends for the tributaries and the main stem separately. The Middle Salmon, which included all tributary and main stem areas from the North Fork Salmon River downstream to Bargamin Creek including the Panther Creek GMU and the Chamberlain Creek GMU.

Snorkel surveys were available for all of the above populations except the Upper Salmon (Table 1). Trends were estimated at two screw traps in both the Lochsa and South Fork Clearwater GMUs and one screw trap each in the South Fork Salmon and the Upper Salmon GMUs (Table 2). Angling surveys were available for the Selway and the Middle Fork Salmon GMUs (Figure 2). Snorkel survey data generally had less variability than the angling and screw trap data sets (Figures 3, 4, 5). We did not have enough data for other important Westslope Cutthroat Trout areas such as the Lemhi River, North Fork Clearwater River, Moyie River, and several others; therefore we did not estimate trends for those locations (Figure 1).

Trends in abundance

Of the 17 data sets used to estimate trends, 10 showed statistically significant population growth, 2 showed statistically significant population decline, and 5 were considered stable with

90% error bounds that overlapped zero (Figure 5; Table 3). In the Coeur d'Alene GMU, the intrinsic rate of change, or trend, was estimated to be growing for both the Coeur d'Alene River and the St. Joe River. Trend in the Lochsa GMU was growing according to the snorkel densities and was substantiated with two screw traps. In the South Fork Clearwater GMU, trend from snorkel densities showed population growth and was also substantiated with two screw traps. Trend from snorkel densities in the Selway River GMU was growing. Trend in the Selway River population was also estimated using angling surveys (Figure 2). Angling surveys described the population as stable. In the South Fork Salmon GMU, trend from snorkeling data was declining which was substantiated by the screw trap in the South Fork Salmon River. Westslope Cutthroat Trout trends were estimated to be declining for the Middle Salmon population. In the Middle Fork Salmon River trends were increasing for both the main stem and the tributary populations.

Observation error

Of the 17 data sets available for WCT trend monitoring, 8 had no measurable observation error and 2 were estimated to have only minimal observation error (Table 6). High observation error was detected in four (44%) of the nine snorkel data sets and three (50%) of the six screw trap data sets (Table 7). There was no observation error detected in the angling data sets.

Explanatory variables

Correlation coefficients (r) for each bioclimatic indicator were generally low across all lags and variables (Table 4). However, correlation coefficients across all populations for PDSI, mean winter stream flow, and air temperature were generally higher for the +1 year lag, except for Chinook Salmon redds, for which correlation coefficients were generally higher for two-year lags (Table 4). For this reason, we chose to use the +1 year lag (+2 year lag for Chinook Salmon redds) in the multiple regression models. The direction of the relationships between environmental variables and cutthroat trout abundance were not consistent across populations (Table 4). For example, Chinook Salmon redds and air temperature were as likely to be positively correlated to Westslope Cutthroat Trout abundance as negatively correlated. The most consistent relationship was between winter stream flow and Westslope Cutthroat Trout abundance one year later, which were inversely related 13 of 17 instances.

Out of 82 combinations, the correlation matrix identified multicollinearity in two occasions. Mean winter discharge and mean annual air temperatures were significantly correlated in two populations: the Middle Salmon and the Upper Salmon.

The best combination of variables used to explain the variability in Westslope Cutthroat Trout trends varied between sampling methods and populations (Table 5). An average of only 21% of the variation in Westslope Cutthroat Trout abundance was explained using these variables. Chinook Salmon redd counts was the best predictor of Westslope Cutthroat Trout abundance for 56% of the data sets, followed by air temperature (28%), and PDSI (17%). Relatively weak but significant relationships between Chinook redds or mean winter discharge were identified with screw traps in the Lochsa River GMU, the Screw Trap 1 in the South Fork Clearwater GMU, at the screw trap in the South Fork Salmon River population. Air temperature and drought formed weak but significant relationships with trends in snorkeling and angling in the Selway River GMU. In the Middle Fork Salmon River populations, air temperature and Chinook redds were both strongly related to snorkel trends and the relationships were highly significant ($F = 19.35$; $r^2 = 0.68$; $P < 0.01$). All other data sets were weakly and non-significantly associated with trends. Evaluation of model residuals resulted in negligible issues regarding

normality, outliers, influential data points, or unequal variance between the terms of the regression model.

DISCUSSION

In our study, there were five times more statistically significant positive growth rates than significant negative growth rates, and several more stable growth rates, suggesting that Westslope Cutthroat Trout are generally stable or increasing in abundance across much of Idaho. Similar increases in population abundance have been observed for a number of salmonids in Idaho (Copeland and Meyer 2011). The only area in our study that appeared to have declining Westslope Cutthroat Trout populations was the South Fork Salmon River and nearby tributaries in other mid-Salmon tributaries.

We assumed that the trend data sets available for each Westslope Cutthroat Trout population were unbiased representations of the true trend within that population. For most populations, this assumption is tenuous because the trend data were obtained from only a portion of the Westslope Cutthroat Trout population. Nevertheless, for the populations where more than one data set was available, trends were generally in synchrony (Figure 6). In fact, there were no examples of trends being statistically positive and statistically negative for two different data sets within the same population. Furthermore, many of the trend data sets were initiated to monitor species other than Westslope Cutthroat Trout, such as the screw trap and snorkel data sets for the Salmon River and Clearwater River subbasins. Although these data sets contained data on all salmonids encountered, they were established to monitor trends in salmon and Steelhead, and it therefore seems unlikely that their use would have resulted in Westslope Cutthroat Trout data that were consistently more optimistic than the mean growth rate for the population would have been.

In the Selway River and Middle Fork Salmon River populations, angling surveys did not follow the trend of the snorkel surveys because the confidence intervals spanned zero, which suggests these populations were stable and not trending in either direction. These angling surveys were implemented to evaluate the effectiveness of catch-and-release regulations and the quality of angling (Hand et al. 2012; Flinders et al. 2013). Previous research has shown positive relationships between angler catch and fish abundance for salmonid populations in mountainous lotic environments (Tsuboi and Endou 2008). However, catch from angling surveys may not always provide consistent data for monitoring trends in abundance because angler experience, fishing techniques, terminal tackle, and environmental conditions can all vary widely through time, and such factors can influence angler catch rates (Figure 5; Bloom 2013; Sullivan et al. 2013; Heermann et al. 2013). This discrepancy may be the result of angling data tracking only a portion of the cutthroat trout populations (i.e., size large enough to be caught) whereas screw traps and snorkeling methods are geared to monitor (and do capture) fish of all sizes.

Observation error was low or not detected for over half of the Westslope Cutthroat Trout trend data sets we summarized herein. Furthermore, those data sets with high observation error in our study had little impact on our findings because for six of the seven data sets with high observation error, trends were still estimated to be statistically significant (despite the fact that CIs were likely inflated), and for the seventh data set, r_{intr} was very close to zero and likely would not have differed from zero even if the error bounds were not inflated. High observation error is often a problem in trend monitoring because it can obscure what otherwise might have been significant changes to a population's abundance (Dunham et al. 2001). The fact that screw

trap data sets were more likely to have high observation error than snorkeling data sets contrasts the findings of Meyer et al. (2014); these authors used many of the same data sets and found that for Bull Trout *Salvelinus confluentus*, snorkeling data sets were much more likely to have high observation error than data obtained from screw traps. These differences may stem from behavioral and life history differences between Bull Trout and Westslope Cutthroat Trout in Idaho. Bull Trout are cryptic, sporadically distributed, highly migratory salmonids (Pratt 1992). In contrast, Westslope Cutthroat Trout are usually more abundant (Copeland and Meyer 2011), less cryptic (and therefore more easily spotted by snorkelers), and - although more mobile than most salmonids - not as mobile as Bull Trout, at least during our sampling period (Schoby and Keeley 2011). It therefore should not be surprising that at least in Idaho, snorkeling data appears to index Westslope Cutthroat Trout abundance better than for Bull Trout, whereas screw traps appear to better index Bull Trout abundance.

Causative mechanisms for population trend are difficult to elucidate at such broad scales, but our results suggest that at least some of the positive growth in Westslope Cutthroat Trout populations in Idaho can be attributed to increases in wild Chinook returning from the Pacific Ocean. Chinook deliver marine-derived nutrients to the majority of Westslope Cutthroat Trout populations in Idaho (Cederholm et al. 1999) and marine-derived nutrients are particularly important for primary production in unproductive geologies which characterize much of our study area (Sanderson et al. 2008). Most bioclimatic variables were weakly correlated with Westslope Cutthroat Trout abundance, suggesting that environmental factors other than the ones we included in our study may have been influencing Westslope Cutthroat Trout abundance. Copeland and Meyer (2011) evaluated the relationships between bioclimatic conditions and fish density for six salmonids in central Idaho and their models generally explained a low amount (maximum adjusted $r^2 = 0.31$) of variation in Westslope Cutthroat Trout abundance. Our study differed by decreasing the spatial resolution to GMUs or smaller populations. Our results were similar because the majority of our models also poorly described the variance in Westslope Cutthroat Trout abundance (mean $r^2 = 0.21$). However, trends in abundance for both snorkel survey data sets in the Middle Fork Salmon River were notably higher with 68% and 42% of the variation in tributary and main stem trends explained respectively. Westslope Cutthroat Trout are often closely associated with headwater habitats (Shepard et al. 2005), which are typically more stochastic than downstream reaches (Richardson et al. 2005) and therefore may be less likely to be influenced by the large-scale bioclimatic indices we analyzed. Other factors we did not include in our study may be influencing Westslope Cutthroat Trout abundance in Idaho more strongly, such as improvements in land management practices (Valdal and Quinn 2011), and the increasing trend in catch-and-release regulations and mindsets among anglers (Quinn 1996; Mallet 2013).

The spatial resolution of this trend analysis could be expanded to provide a more complete understanding for Westslope Cutthroat Trout abundance across Idaho. Data were not available to estimate trends in several large GMUs such as the Moyie, North Fork Clearwater, Lemhi, Pahsimeroi, and several others (Figure 1). For the data that were available, the vast majority of it could not be included in this assessment because of missing transect dimensions (to calculate densities), low frequency of revisits (all electrofishing data, and the majority of snorkel data), and sometimes because of recognizably erroneous data which we therefore excluded. The snorkel data that were used in this assessment were obtained from multiple locations and required months of manipulation to compile datasets that were comparable with past Westslope Cutthroat Trout assessments. Future assessments could be completed more efficiently if all snorkel data were stored and maintained in a single database with more diligent QA/QC of the data. Nevertheless, the available data did cover most of the core Westslope Cutthroat Trout populations, which is impressive considering the expanse and terrain these

trends describe. The snorkel survey dataset in Idaho is extensive and provided the best spatial coverage among all available data for Westslope Cutthroat Trout.

As mentioned by Schill et al. in 2004, these data comprise one of the most extensive salmonid monitoring data sets in America. Our study can also be used to identify areas where monitoring can be improved. Several of the GMUs where we lacked adequate data to estimate trends could be considered to be on the fringes of Westslope Cutthroat Trout distribution, such as the Little Salmon and Lower Salmon GMUs. However, several other GMUs, such as the Lemhi River, Pahsimeroi River, and Upper Salmon River, might be regarded as core areas. Until Westslope Cutthroat Trout trend data are available for these drainages, assessment of Westslope Cutthroat Trout status in Idaho will be incomplete. Nevertheless, the results of our study suggest that most Westslope Cutthroat Trout populations in Idaho are stable or increasing in abundance.

RECOMMENDATIONS

1. Maintain the core General Parr Monitoring (GPM) snorkel trend surveys in the Clearwater River and Salmon River basins and the Westslope Cutthroat Trout snorkel trend monitoring surveys in the Coeur d'Alene River and St. Joe River.
2. Manage all relevant Westslope Cutthroat Trout data with increased emphasis on QA/QC. Store and maintain these data in the Standard Stream Survey database for rapid and consistent summary and to ensure continuity with future abundance and distribution assessments.
3. Several GMUs at the core of the Westslope Cutthroat Trout distribution in Idaho were not included in this analysis because there were not enough snorkel transects over a long enough time period. If IDFG fish managers want to know more about the Westslope Cutthroat Trout trend in abundance within these areas, more snorkel surveys or other population monitoring are needed to supplement the current data set.

ACKNOWLEDGEMENTS

We would like to thank many colleagues who shared abundance and trend information for this study, specifically Bruce Barnett, Mike Biggs, Tim Copeland, Tom Curet, Joe Dupont, Jon Flinders, Jim Fredericks, Robert Hand, Ryan Hardy, Mike Peterson, and Rob Ryan. Emanuel Ziolkowski provided summaries of Chinook Salmon data. Paul Bunn, Tony Lamansky, and Liz Mamer provided cartographic and database support. Matthew Corsi and Jordan Messner provided early reviews and Cheryl Zink helped format and edit this document. Funding for this work was provided by anglers and boaters through their purchase of Idaho fishing licenses, tags and permits and from federal excise taxes on fishing equipment and boat fuel through the Sport Fish Restoration Program.

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TABLES

Table 1. Average annual densities of Westslope Cutthroat Trout (>155 mm/m²) estimated from snorkel surveys in nine different populations in Idaho.

Survey year	Average annual density (>155 mm/m ²)								
	Middle Salmon tribs.	South Fork Salmon	Middle Fork Salmon main	Middle Fork Salmon tribs.	Lochsa	Selway	South Fork Clearwater	Coeur d'Alene	St. Joe
1969									0.0038
1973								0.0028	
1980								0.0025	
1981								0.0039	
1985			0.0038	0.0098	0.0066	0.0046	0.0022		
1986			0.0101	0.0103	0.0086		0.0026		
1987	0.0056			0.0064	0.0001		0.0002		
1988	0.0090		0.0080	0.0089	0.0003	0.0031	0.0026		
1989	0.0031		0.0207	0.0056	0.0003	0.0026	0.0032		0.0060
1990	0.0031		0.0052	0.0120	0.0010	0.0019	0.0014		0.0107
1991	0.0044	0.0045		0.0209	0.0010	0.0025	0.0015	0.0046	
1992	0.0213	0.0041	0.0078	0.0155	0.0008	0.0079	0.0014		
1993	0.0067	0.0034		0.0150	0.0015	0.0055	0.0003		
1994	0.0397	0.0031		0.0107	0.0005	0.0082	0.0006		
1995	0.0224	0.0001		0.0287	0.0006	0.0043	0.0005		
1996	0.0119	0.0016	0.0085	0.0252	0.0012	0.0040	0.0009		
1997	0.0035	0.0031		0.0352	0.0009	0.0037	0.0004		
1998	0.0053	0.0081		0.0649	0.0009	0.0079	0.0006	0.0046	0.0048
1999	0.0035	0.0007	0.0167	0.0412	0.0003	0.0083	0.0007		
2000	0.0042	0.0019	0.0271	0.1069	0.0014		0.0011		0.0129
2001	0.0302	0.0035	0.0286	0.0918	0.0020	0.0103	0.0032	0.0047	0.0087
2002	0.0165	0.0024		0.1759	0.0015	0.0064	0.0042	0.0054	0.0087
2003	0.0103	0.0081		0.1637	0.0020	0.0194	0.0033	0.0046	0.0117
2004	0.0047	0.0015	0.0194	0.3055	0.0021	0.0153	0.0026	0.0055	0.0116
2005	0.0053	0.0029	0.0447	0.0763	0.0032	0.0090	0.0016	0.0063	0.0122
2006	0.0062	0.0022		0.1281	0.0034	0.0056	0.0032	0.0071	
2007	0.0013		0.0321			0.0035	0.0003	0.0108	0.0115
2008			0.1506	0.0708		0.0139	0.0034	0.0084	0.0133
2009	0.0029		0.0614	0.1135	0.0057	0.0201	0.0092	0.0096	0.0116
2010	0.0004		0.0468	0.4678	0.0054	0.0291	0.0030	0.0093	0.0127
2011			0.0930	0.3917	0.0027	0.0095	0.0033	0.0075	
2012			0.0460	0.1940	0.0074	0.0079	0.0118	0.0033	0.0132

Table 2. Total catch of Westslope Cutthroat Trout (>50 mm) captured at selected screw traps in Idaho.

Screw trap location and total catch of Westslope Cutthroat Trout (>50 mm)						
Survey year	Colt Killed Creek	Crooked Fork Creek	Crooked River	Red River	Upper Salmon River	South Fork Salmon River
1996		162				
1997		100				24
1998		130	1	15	0	10
1999	13	44	1	12	2	5
2000	15	168	1	23	76	11
2001	37	314	0	0	23	8
2002	27	149	15	42	20	13
2003	9	124	0	6	32	7
2004	57	104	58	68	64	11
2005	18	176	11	23	9	3
2006	14	93	95	52	20	10
2007	31	241	25	33	5	4
2008	23	290	174	150	1	6
2009	40	308	26	59	20	20
2010	29	317	44	241	9	6

Table 3. Intrinsic rate of population change (r_{intr}) for Westslope Cutthroat Trout estimated using three sampling methods in 10 different populations in Idaho. Standard error ($\pm 1SE$) was estimated from least squares regression of the \log_e -transformed data.

Population	Sub-basin	Site name	Method	Time span (yr.)	Years of data	r_{intr}	SE
Coeur d'Alene	Spokane	CdA SN	Snorkeling	39	17	0.03	0.01
St. Joe	Spokane	St Joe SN	Snorkeling	43	18	0.03	0.01
Lochsa	Lochsa	Lochsa SN	Snorkeling	28	26	0.07	0.02
Colt Killed Creek	Lochsa	Lochsa ST1	Screw Trap	12	12	0.05	0.04
Crooked Fork Creek	Lochsa	Lochsa ST2	Screw Trap	15	15	0.07	0.03
SF Clearwater	SF Clearwater	SF CW SN	Snorkeling	28	28	0.06	0.02
Crooked River	SF Clearwater	SF CW ST1	Screw Trap	13	13	0.36	0.09
Red River	SF Clearwater	SF CW ST2	Screw Trap	13	13	0.24	0.08
Selway	Selway	Selway SN	Snorkeling	28	25	0.06	0.01
Selway	Selway	Selway AN	Angling	36	27	-0.01	0.01
Middle Salmon trib	Salmon	Mid Sal SN	Snorkeling	24	23	-0.06	0.03
South Fork Salmon	Salmon	SF Sal SN	Snorkeling	16	16	-0.11	0.06
South Fork Salmon	Salmon	Knox ST	Screw Trap	14	14	-0.04	0.03
MFSR Main stem	Salmon	MFSR Mn SN	Snorkeling	28	18	0.09	0.01
MFSR tributaries	Salmon	MFSR Tr SN	Snorkeling	28	27	0.15	0.01
Middle Fork Salmon	Salmon	MFSR AN	Angling	54	20	0.0002	0.01
Upper Salmon	Salmon	Sawtooth ST	Screw Trap	13	13	0.02	0.10

Table 4. Correlation coefficients (r) between four independent variables and Westslope Cutthroat Trout abundance indices in Idaho populations. Relationships are evaluated for two time periods to explain trends in abundance indices across Idaho. Statistically significant ($\alpha = 0.10$) estimates are in bold.

Population	Method	Mean annual air temperature		Palmer drought severity index		Mean winter (Dec.-Feb.) stream flow		Chinook Salmon redds	
		No lag	1 year lag	No lag	1 year lag	No lag	1 year lag	1 year lag	2 year lag
St. Joe	Snorkeling	0.21	0.13	0.23	-0.55	-0.01	-0.09	^a	^a
Coeur d'Alene	Snorkeling	-0.02	0.39	-0.10	0.12	-0.04	-0.14	^a	^a
Upper Salmon	Screw trap	-0.08	-0.26	-0.28	-0.59	-0.59	-0.52	0.40	0.37
Middle Salmon Main	Snorkeling	-0.26	-0.50	-0.22	-0.03	-0.63	-0.28	0.21	-0.07
South Fork Salmon	Snorkeling	0.15	-0.22	-0.40	-0.12	-0.12	0.26	0.27	0.01
South Fork Salmon	Screw trap	-0.26	-0.32	0.08	0.32	0.50	0.46	0.03	-0.37
MFSR Main stem	Snorkeling	0.61	0.72	-0.18	-0.41	0.09	-0.25	0.32	0.35
MFSR Tributaries	Snorkeling	0.75	0.73	-0.10	-0.36	-0.38	-0.59	^b	^b
Middle Fork Salmon	Angling	0.11	-0.07	-0.07	-0.15	-0.49	0.17	0.20	0.29
South Fork Clearwater	Snorkeling	0.09	-0.06	-0.18	-0.29	-0.19	-0.37	0.50	0.35
Crooked River	Screw trap	0.19	0.25	-0.29	-0.07	-0.25	-0.49	0.24	-0.16
Red River	Screw trap	-0.03	0.12	-0.37	-0.28	-0.05	-0.36	0.46	-0.05
Lochsa	Snorkeling	0.16	-0.03	0.14	0.26	0.08	-0.05	0.31	0.27
Colt Killed Creek	Screw trap	-0.15	-0.34	0.08	0.24	-0.50	-0.02	-0.05	-0.16
Crooked Fork Creek	Screw trap	0.04	-0.14	-0.21	0.35	-0.12	-0.26	0.09	-0.22
Selway	Snorkeling	0.47	0.41	-0.40	-0.28	-0.17	-0.07	0.22	0.46
Selway	Angling	-0.21	0.04	-0.08	0.09	-0.13	0.32	0.35	-0.41

^a These populations are outside of the range of anadromous salmon.

^b Densities in the Middle Fork Salmon River tributaries and main stem were combined for this comparison because main stem Chinook Salmon spawning occurrence is much lower than tributary spawning.

Table 5. Best least squares regression models relating bioclimatic variables to Westslope Cutthroat Trout abundance for individual populations in Idaho. Akaike information criterion weights (w_i) identified the likelihood that each model was the best model compared to all other models, and r^2 or adjusted r^2 estimated the amount of variation in trend described by the best model. PDSI is the Palmer Drought Severity Index, AirT is air temperature, Discharge is mean winter streamflow, and Redds is the annual count of Chinook Salmon redds.

Population and sampling method	w_i	r^2	Variables	F-value	P-value
Coeur d'Alene snorkeling	0.27	0.01	PDSI	0.14	0.71
St. Joe snorkeling	0.29	0.05	PDSI	0.75	0.40
Lochsa snorkeling	0.22	0.08	Redds	2.14	0.16
Colt Killed Cr screw trap (Lochsa)	0.25	0.25	Redds	3.29	0.10
Crooked Fork Cr screw trap (Lochsa)	0.41	0.27	Redds	4.72	0.05
SF Clearwater snorkel	0.19	0.07	Redds	1.96	0.17
Crooked R screw trap (SF Clearwater)	0.27	0.25	Redds	3.57	0.09
Red R screw trap (SF Clearwater)	0.22	0.18	Redds	2.36	0.15
Selway snorkeling	0.29	0.25	AirT	7.47	0.01
Selway angling	0.18	^a 0.14	AirT+PDSI	1.88	0.17
Middle Salmon main snorkel ^b	0.23	0.08	Redds	1.72	0.20
SFSR screw trap	0.30	^a 0.33	Redds+Discharge	4.14	0.05
SFSR snorkeling	0.27	0.16	PDSI	2.65	0.13
MFSR Main snorkeling ^b	0.42	^a 0.42	AirT+Redds	7.13	0.01
MFSR Trib snorkeling ^b	0.50	^a 0.68	AirT+PDSI+Redds	19.35	<0.01
MFSR Main angling ^b	0.26	0.01	AirT	0.21	0.65
Upper Salmon screw trap	0.35	0.34	Discharge	4.19	0.07
Upper Salmon screw trap ^b	0.52	0.29	Redds	4.47	0.06

^a Adjusted r^2 used because the best model utilized multiple variables.

^b Discharge was not included in the model comparison because the available record of stream flow was shorter than the record of abundance.

Table 6. Observation error ($\hat{\tau}^2$) estimated using the Gompertz state-space model for Westslope Cutthroat Trout populations in Idaho.

Population	Survey method	$\hat{\tau}^2$
Coeur d'Alene River	Snorkeling	0
St. Joe River	Snorkeling	0.07
Lochsa River	Snorkeling	0.19
Lochsa River	Screw trap	0
Lochsa River	Screw trap	0
Selway River	Angling	0
Selway River	Snorkeling	0
South Fork Clearwater River	Snorkeling	0.56
South Fork Clearwater River	Screw trap	0.61
South Fork Clearwater River	Screw trap	0.60
Middle Salmon Main River	Snorkeling	0.24
South Fork Salmon River	Snorkeling	0
South Fork Salmon River	Screw trap	0
Middle Fork Salmon River	Angling	0
Middle Fork Salmon River	Snorkeling	0.19
Middle Fork Salmon River	Snorkeling	0.09
Upper Salmon River	Screw trap	0.95

Table 7. Tally of Observation Error ($\hat{\tau}^2$) by method (Low= $0 < \hat{\tau}^2 < 0.1$; High= $\hat{\tau}^2 \geq 0.1$).

Method	%High	%Low	%Zero	High ($\hat{\tau}^2$)	Low ($\hat{\tau}^2$)	Zero ($\hat{\tau}^2$)	Total
Snorkel	44.4	22.2	33.3	4	2	3	9
Angling	0.0	0.0	100.0	0	0	2	2
Screw trap	50.0	0.0	50.0	3	0	3	6
Total =	41.2	11.8	47.1	7	2	8	17

FIGURES

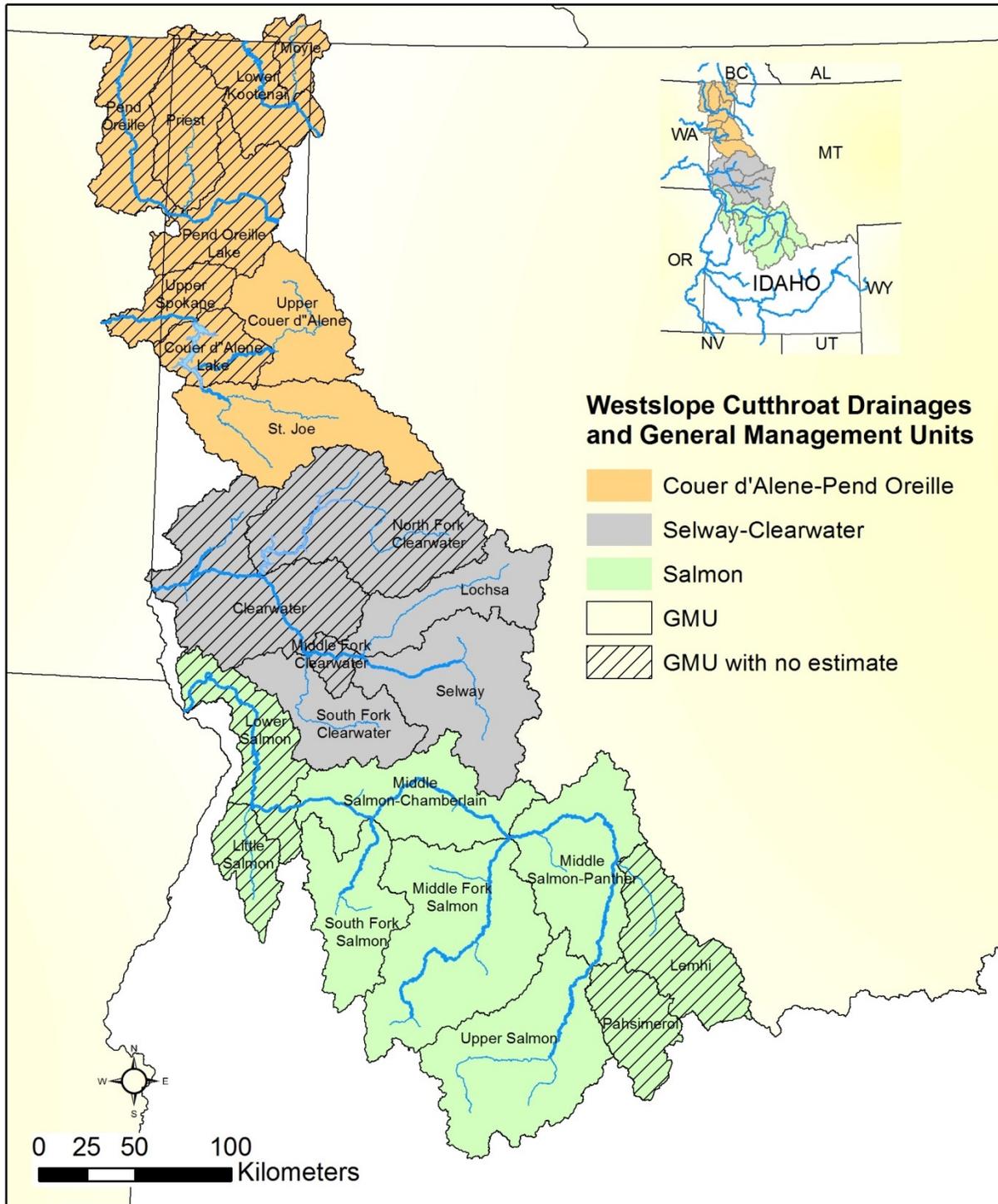


Figure 1. General Management Units (GMUs) and major drainages for Westslope Cutthroat Trout in Idaho. General Management Units with hash marks were not evaluated for trend because of data limitations.

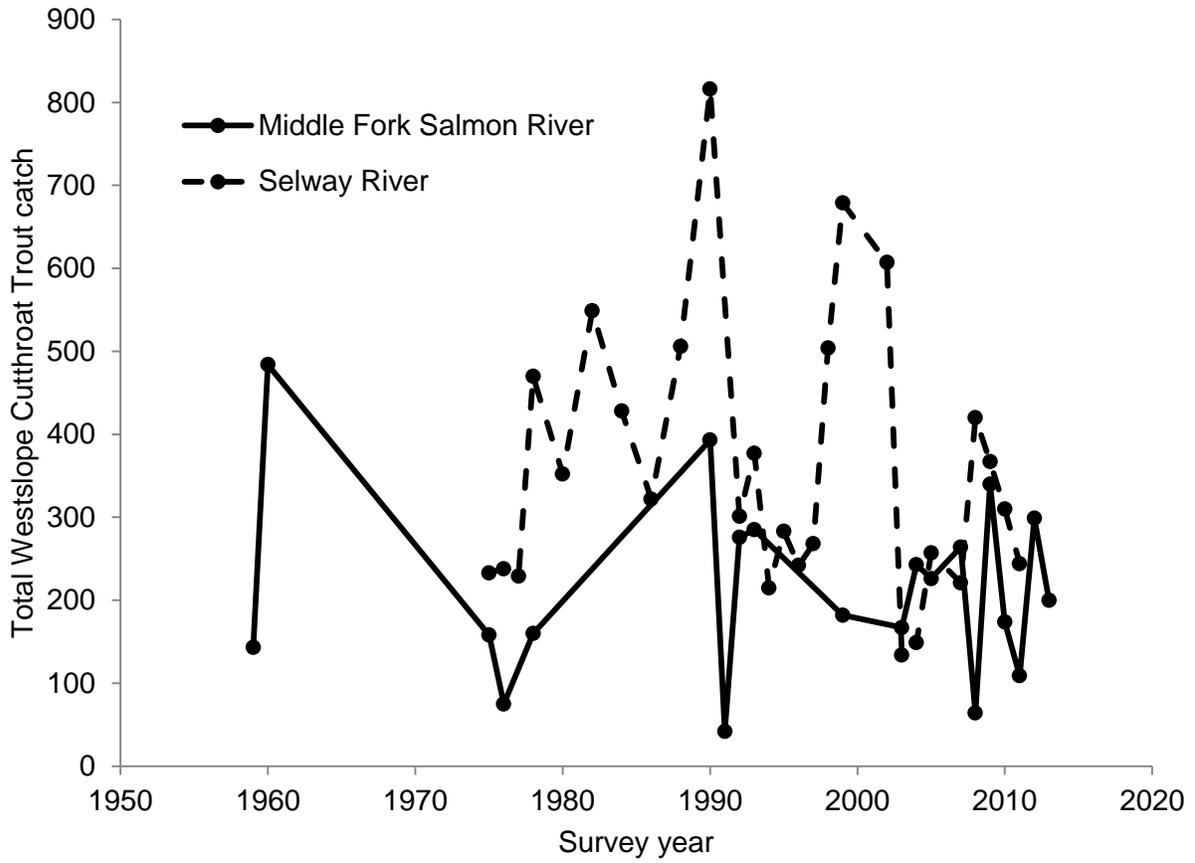


Figure 2. Total catch of Westslope Cutthroat Trout during angling surveys in the Middle Fork Salmon River and the Selway River.

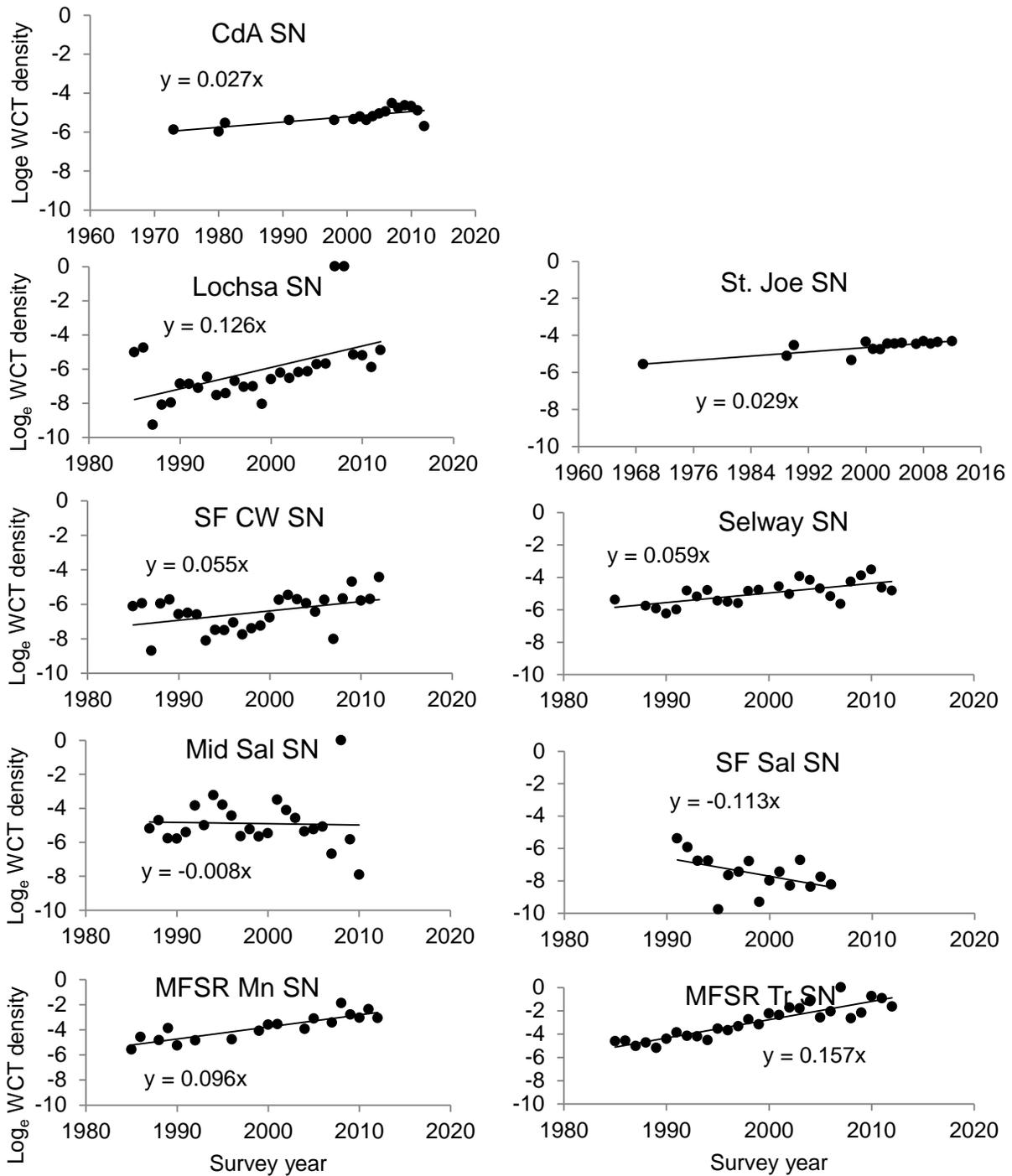


Figure 3. Intrinsic rate of population change (r_{intr} = slope) for Westslope Cutthroat Trout (WCT) estimated from annual average \log_e -transformed snorkel survey (SN) densities in eight different populations in Idaho. See Table 3 for location key.

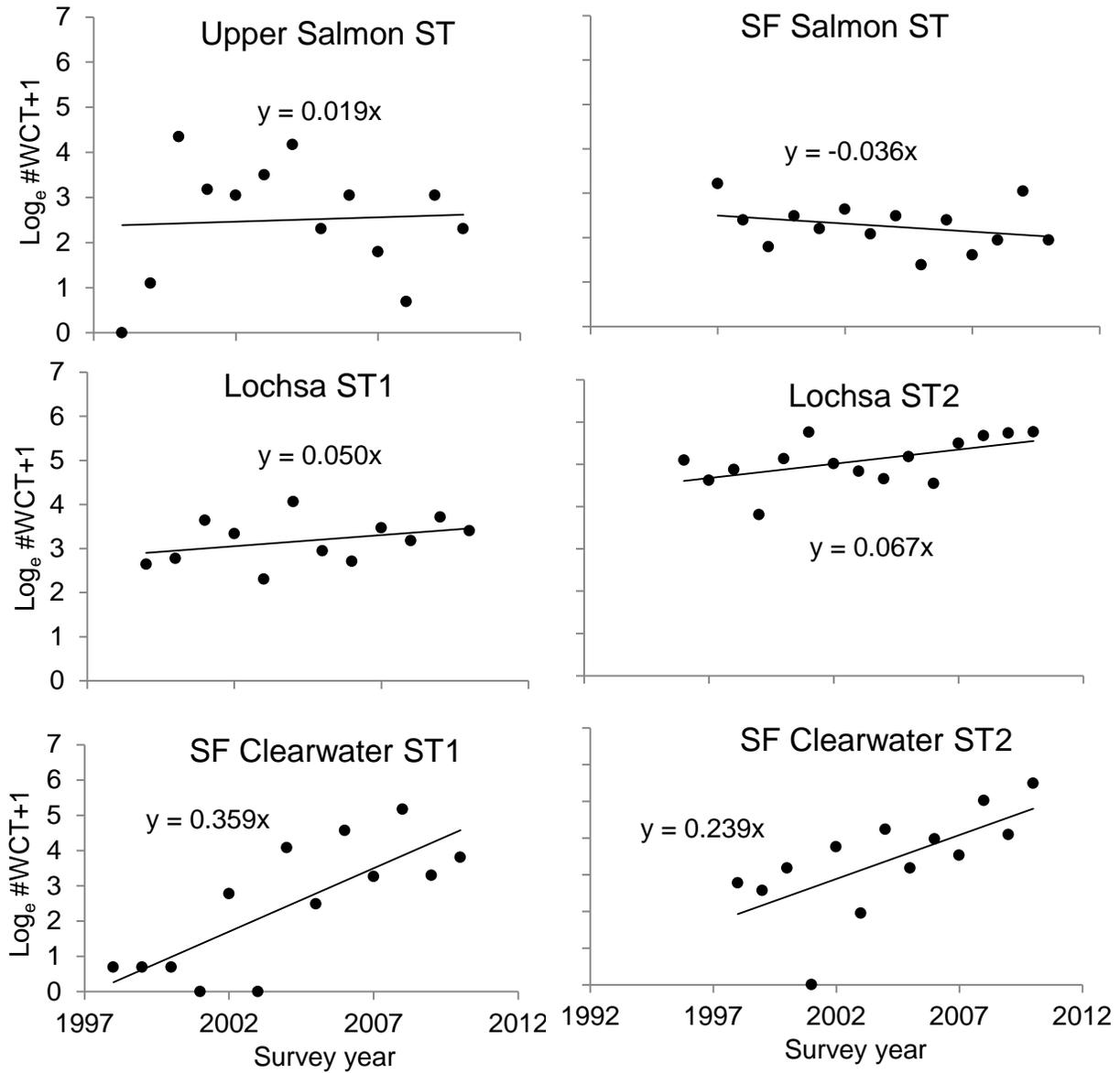


Figure 4. Intrinsic rate of population change (r_{intr} = slope) for Westslope Cutthroat Trout (WCT) estimated from \log_e -transformed total catch (+1) at screw traps (ST) in four different populations in Idaho. See Table 3 for location key.

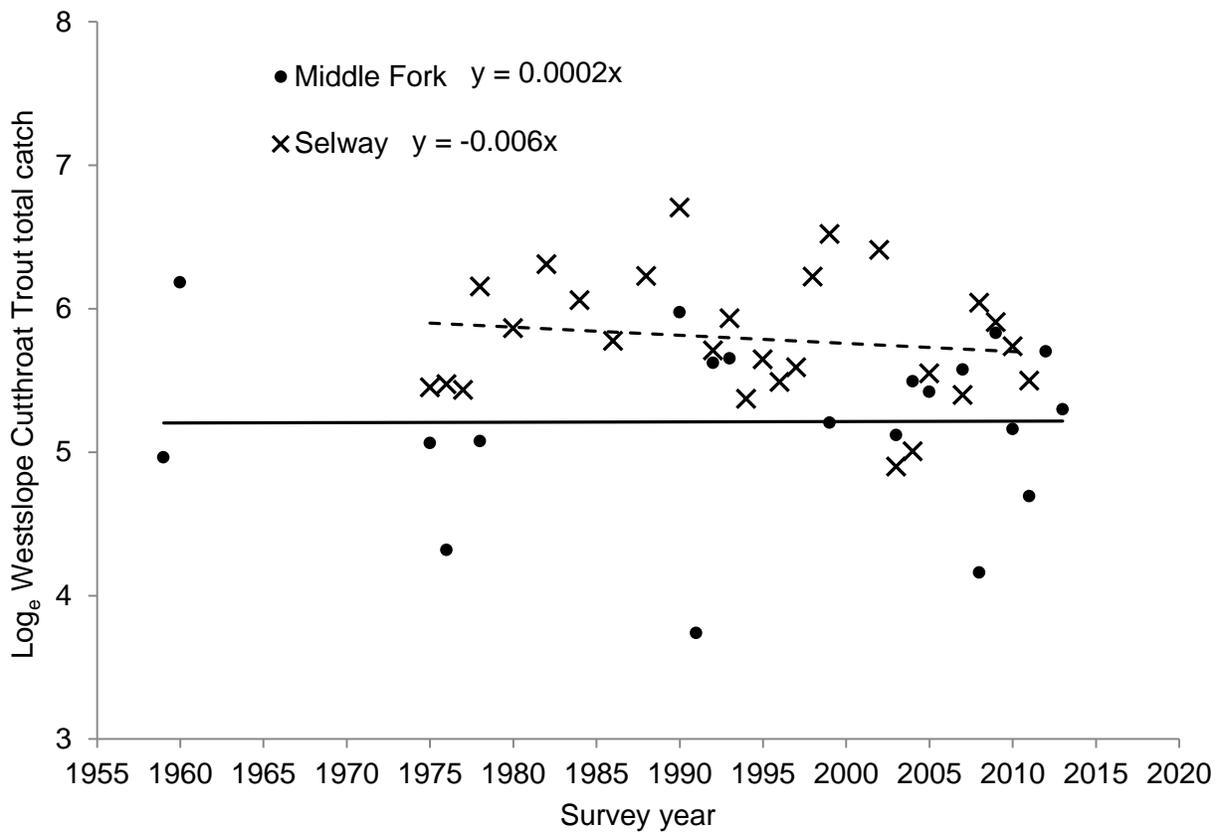


Figure 5. Intrinsic rate of population change (r_{intr} = slope) for Westslope Cutthroat Trout estimated from log_e-transformed angling surveys in the Middle Fork Salmon River and the Selway River.

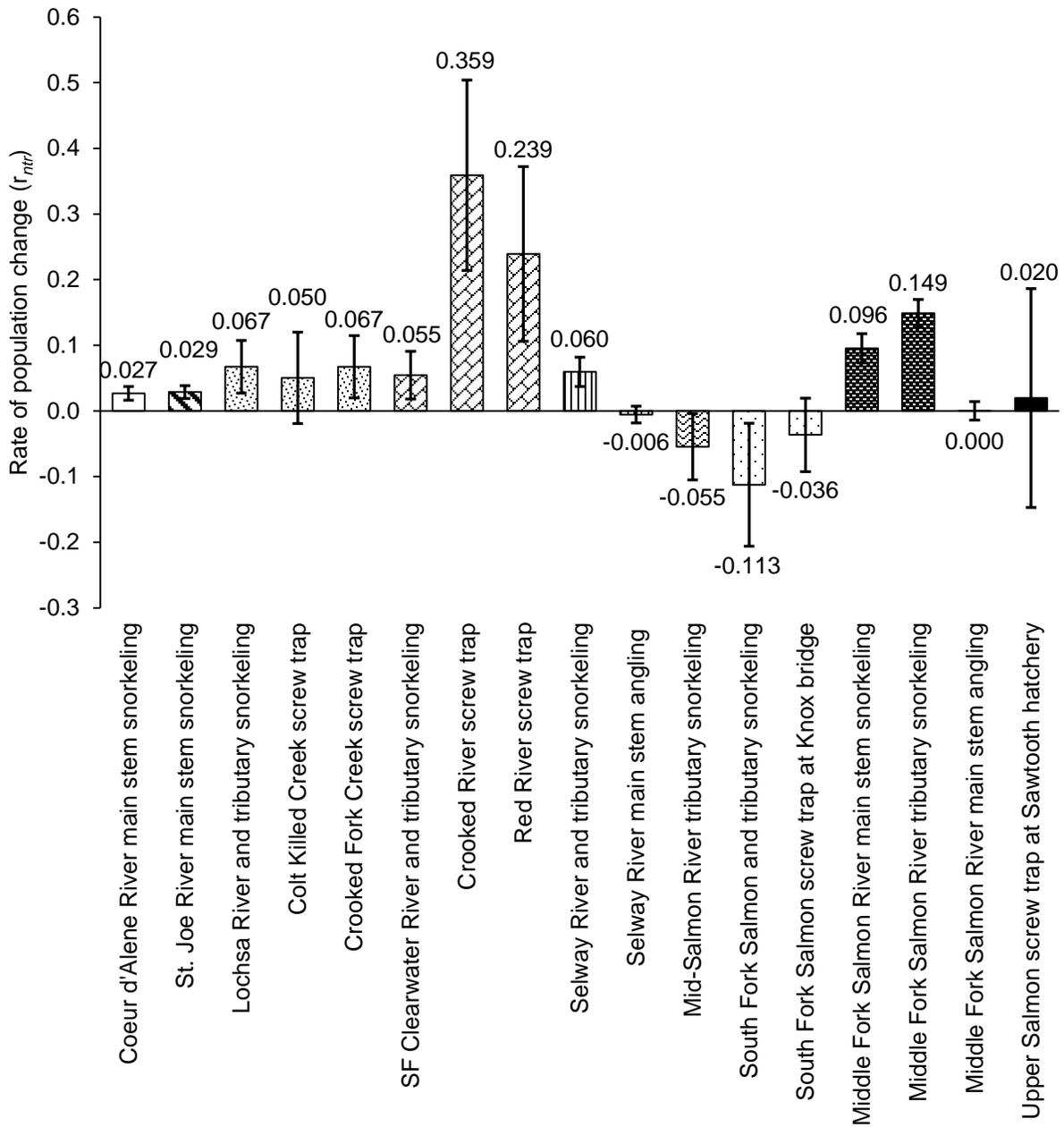


Figure 6. Intrinsic rate of population change (r_{intr}) for Westslope Cutthroat Trout in Idaho. Estimates generated from three different survey methods in ten populations. Error bars represent $\pm 90\%$ confidence intervals.

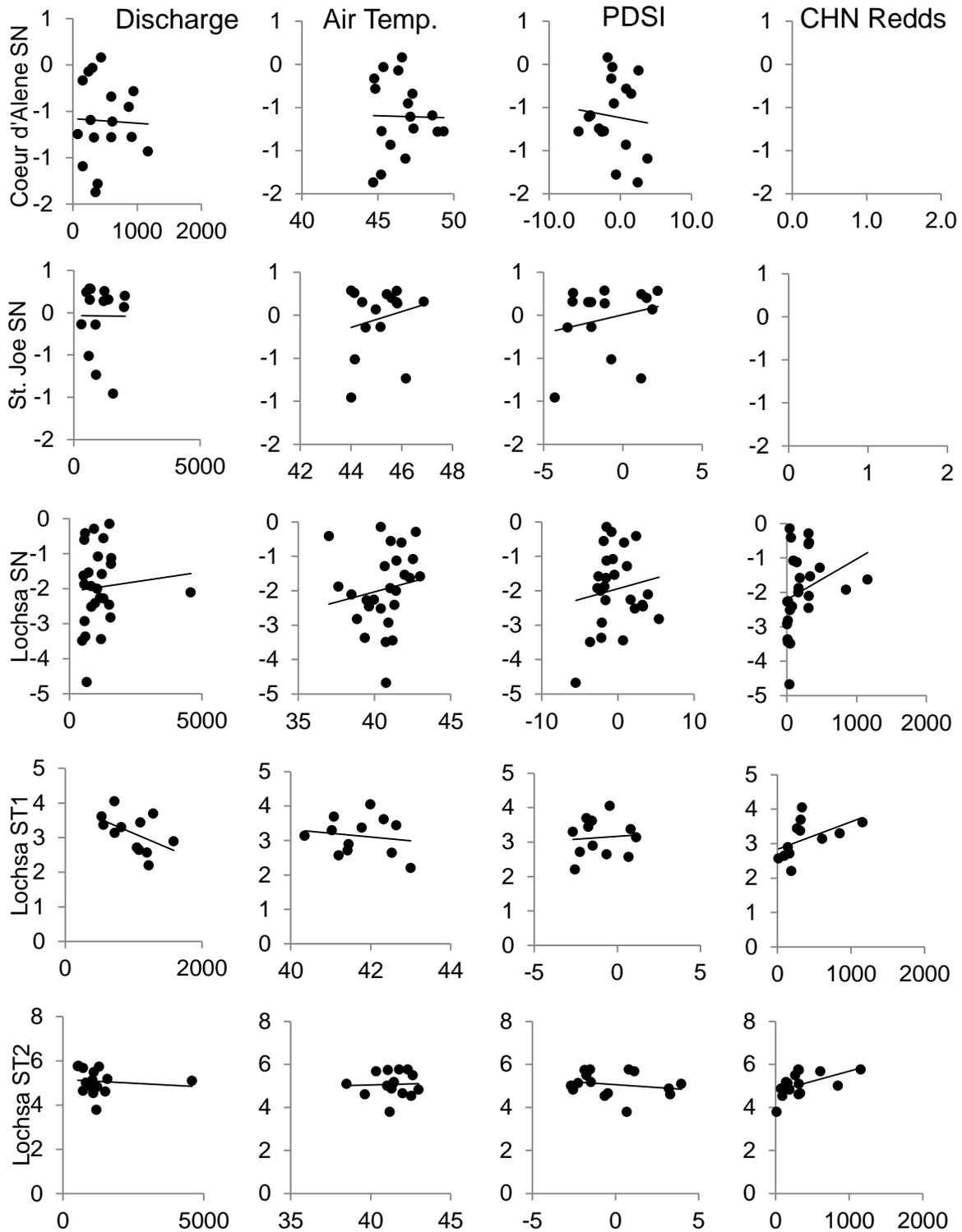


Figure 7. Scatter plots of four bioclimatic indicators (mean winter discharge, mean annual air temp., drought (PDSI) and Chinook Salmon abundance) vs. \log_e -transformed abundance indices (snorkeling [SN] and screw trap [ST]) for Westslope Cutthroat Trout in three different populations (Coeur d'Alene, St. Joe, Lochsa) in Idaho.

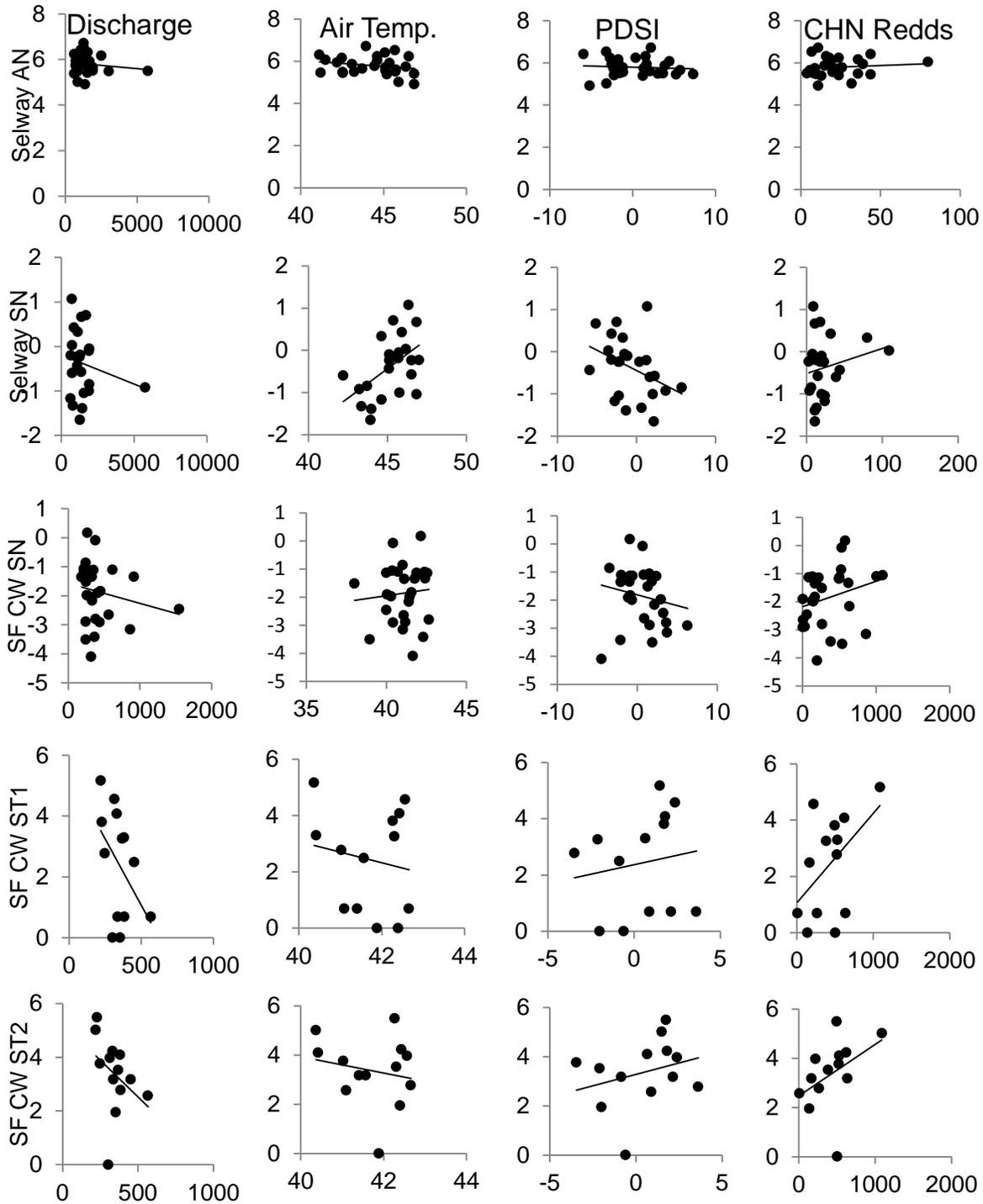


Figure 8. Scatter plots of four bioclimatic indicators (mean winter discharge, mean annual air temp., drought (PDSI) and Chinook Salmon abundance) vs. log_e-transformed abundance indices (snorkeling [SN] and screw trap [ST]) for Westslope Cutthroat Trout in two different populations (Selway, South Fork Clearwater) in Idaho.

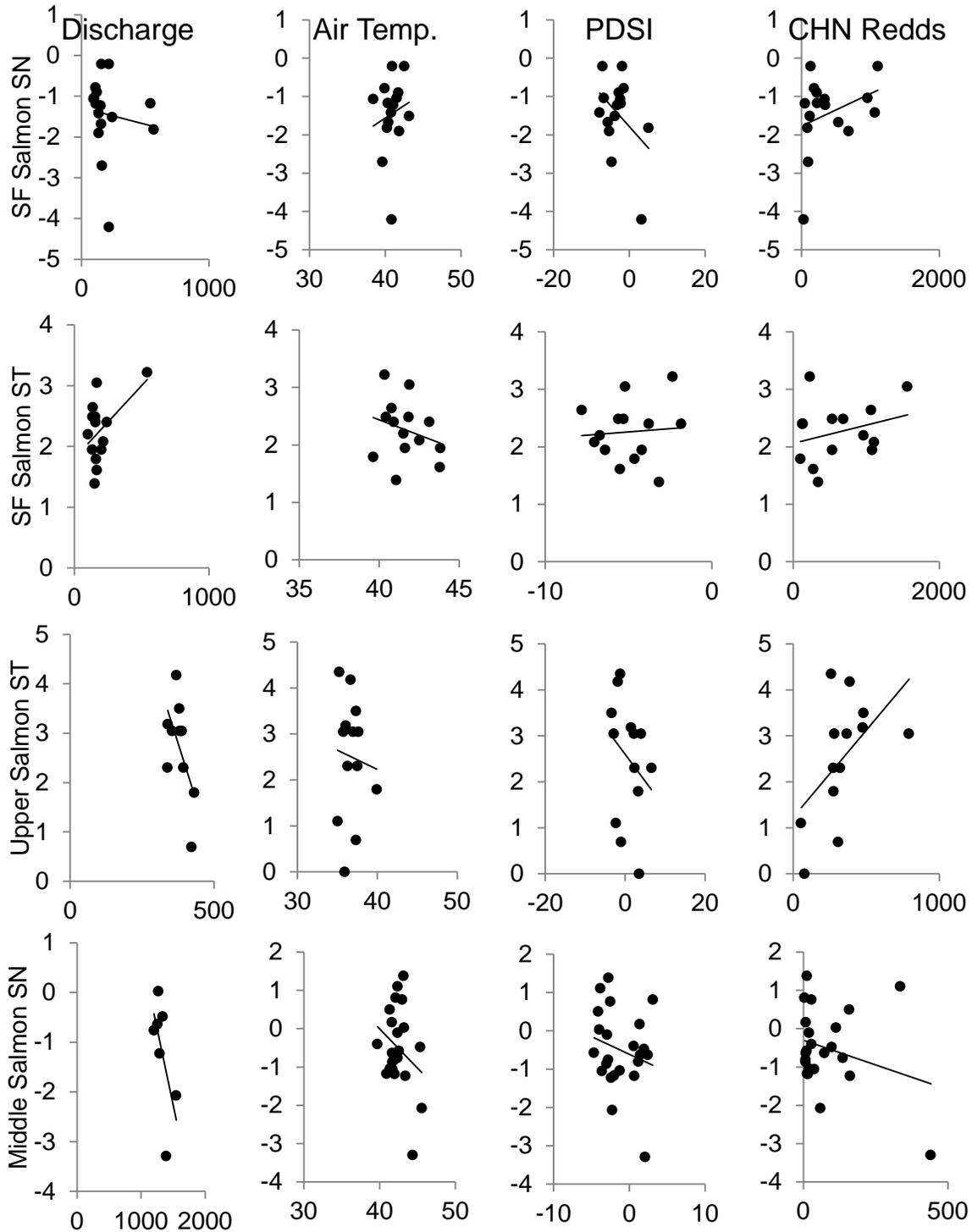


Figure 9. Scatter plots of four bioclimatic indicators (mean winter discharge, mean annual air temp., drought (PDSI) and Chinook salmon abundance) vs. \log_e -transformed abundance indices (snorkeling [SN] and screw trap [ST]) for Westslope Cutthroat Trout in three populations (South Fork Salmon, Upper Salmon River, Middle Salmon River) in Idaho.

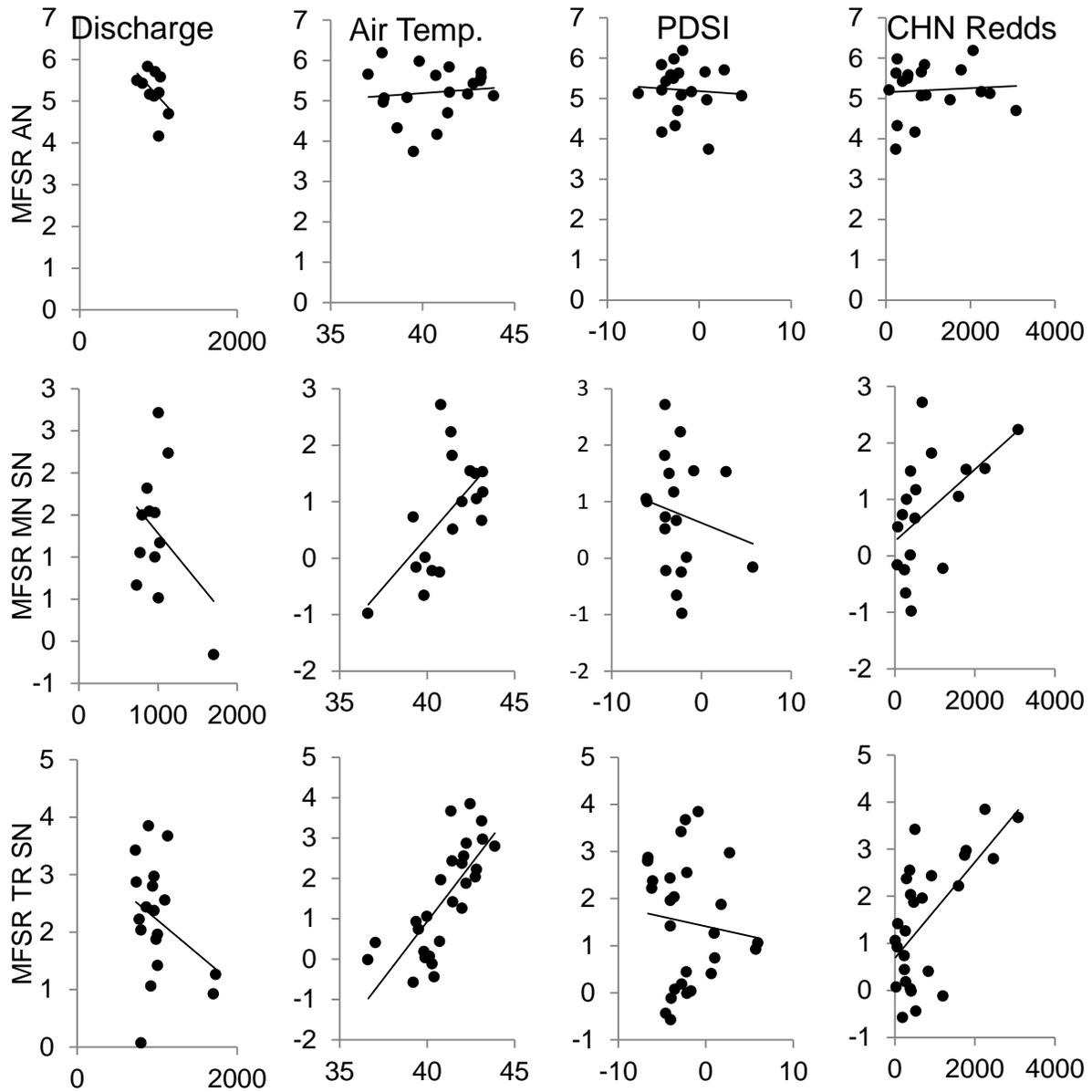


Figure 10. Scatter plots of four bioclimatic indicators (mean winter discharge, mean annual air temp., drought (PDSI) and Chinook Salmon abundance) vs. log_e-transformed abundance indices (snorkeling [SN] and screw trap [screw trap]) for Westslope Cutthroat Trout in the Middle Fork Salmon River main stem (MN) and tributaries (TR) in Idaho.

CHAPTER 2: AN ATTEMPT TO PURIFY A YELLOWSTONE CUTTHROAT TROUT STREAM BY REMOVING RAINBOW TROUT AND HYBRIDS VIA ELECTROFISHING

ABSTRACT

We used backpack electrofishing in 2010, 2012, and twice in 2013 to remove Rainbow Trout *Oncorhynchus mykiss* and Rainbow Trout X Cutthroat Trout *O. clarkii* hybrids from Palisades Creek (a tributary of the South Fork Snake River) and thereby evaluate whether an introgressed population of Yellowstone Cutthroat Trout *O. clarkii bouvieri* could be reverted to pure or nearly pure conditions. Removals were conducted from an electric weir (0.7 km from the mouth) – which prevents upstream migration of non-native salmonids - upstream approximately 10 km to a high velocity cascading section of stream that appears to be a complete fish barrier. A total of 2,626, 1,843, 1,279, and 2,228 fish were captured in Palisades Creek in 2010, 2012, and both 2013 removals, respectively; of these fish, 848, 484, 369, and 653 were Rainbow Trout or hybrids and were consequently removed. Removals were especially effective at targeting large fish; in fact, in 2010 we removed 289 Rainbow Trout and hybrids ≥ 25 cm in length, but by the time of the second 2013 removal, there were only an estimated 78 Rainbow Trout ≥ 25 cm left in the stream, of which 36 were removed during that survey. Unfortunately, the proportion of the entire trout population in the stream that Yellowstone Cutthroat Trout comprised increased only from 61% during the 2010 removal to about 70% during each removal thereafter. In general, there were fewer Rainbow Trout and hybrids in the upper reaches of the treatment section, especially in later years; by 2012, Yellowstone Cutthroat Trout comprised ~90% of the trout population in the highest 2 km of stream. Mean capture efficiency decreased from 0.51 in 2012 to 0.32 and 0.35 for the two 2013 removals. Despite positive signs that the removal is altering the abundance of Rainbow Trout and hybrids, unless capture efficiency is increased, or emigrating Rainbow Trout and hybrids are blocked more completely from entering the stream, it does not appear that the composition of trout in Palisades Creek can be increased to $\geq 90\%$ Yellowstone Cutthroat Trout throughout the entire study reach of the river.

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INTRODUCTION

In eastern Idaho, the South Fork Snake River supports one of the few remaining fluvial populations of Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (Thurow et al. 1988; Meyer et al. 2006a; Gresswell 2011). However, the long-term persistence of Cutthroat Trout in the South Fork Snake River drainage is threatened by an increasing abundance of Rainbow Trout *O. mykiss* (High 2010). Interactions between native Cutthroat Trout *O. clarkii* and non-native Rainbow Trout usually reduce or eliminate pure Cutthroat Trout populations via introgressive hybridization (Rainbow Trout X Cutthroat Trout; hereafter hybrids). If introgressive hybridization occurs for many generations throughout a population, the result is sometimes a hybrid swarm, where none of the remaining individuals in the population are pure (Allendorf et al. 2001). Once a hybrid swarm is established, without population replacement, it is essentially impossible to recover the population to a genetically pure state even with new introductions of Cutthroat Trout from a genetically pure population.

Rainbow Trout are not consistently distributed throughout the South Fork Snake River drainage (Meyer et al. 2006a), which is not surprising since hybridization is often not uniform within individual populations (Woodruff 1973). In areas where hybrid swarms have not formed, management actions that focus on removing Rainbow Trout and hybrids will almost certainly reduce both the rate and spread of hybridization and introgression in Cutthroat Trout populations and may actually be able to restore populations to a genetically pure or nearly pure condition (Leary et al. 1995). Recent debates on the value of hybridized populations of Cutthroat Trout have been fervent; opinions vary from discounting any population that is not entirely pure (Allendorf et al. 2004, 2005) to advocating preservation of populations that are up to 25% hybridized (USFWS 2003; Campton and Kaeding 2005). The Idaho Department of Fish and Game (IDFG) categorizes Cutthroat Trout into core, conservation, and sport fish populations as those with <1%, 1–10%, or >10% introgressive hybridization, respectively (Lentsch et al. 2000) and prioritizes fisheries management of each type of population differently. Converting conservation and sport fish populations to a higher category by eradicating Rainbow Trout and hybrids is one of the highest management priorities for Cutthroat Trout populations in Idaho by the IDFG (IDFG 2007).

While prior attempts to eradicate non-native fish species with electrofishing removals have typically not been successful (e.g., Thompson and Rahel 1996; Meyer et al. 2006b; Peterson et al. 2009; but see Kulp and Moore 2000 and Shepard et al. 2002), the life history characteristics of Rainbow Trout may result in higher success with removal efforts. In western North America, Brook Trout are the most commonly targeted non-native salmonid for removals (Dunham et al. 2002). Brook Trout mature at an earlier age and are more fecund than Rainbow Trout (Buss and McCreary 1960), providing Brook Trout populations better compensatory traits to offset removal efforts than Rainbow Trout populations might have (Meyer et al. 2006a). This should increase the likelihood of successfully removing Rainbow Trout by electrofishing, and in fact, the species has been eradicated from short reaches of a few small streams in eastern North America (Kulp and Moore 2000). Additionally, electric and velocity weirs were in place on several tributaries to the South Fork Snake River which was the source for non-native Rainbow Trout. These weirs prevent upstream passage of Rainbow Trout into Yellowstone Cutthroat Trout spawning habitat (High 2010). Removing the remaining Rainbow Trout and hybrids in these tributaries could provide a conservation measure for pure Yellowstone Cutthroat Trout.

OBJECTIVE

1. Evaluate whether a hybridized population of Yellowstone Cutthroat Trout in Palisades Creek (a tributary to the South Fork Snake River) could be purified of Rainbow Trout genetics (<10% introgression) with three consecutive years of electrofishing removals.

METHODS

Palisades Creek is a 4th order stream (1:24,000 hydrologic scale) with a mean width of approximately 10 m and an average gradient of approximately 2.0%. The fish community is predominantly Yellowstone Cutthroat Trout, Rainbow Trout, and hybrids. Although a few wild Brown Trout *Salmo trutta* and Mountain Whitefish *Prosopium williamsoni* are also present, Brown Trout and Mountain Whitefish comprise <1% of the salmonid population. Paiute Sculpin *Cottus beldingi* are the only known non-game fish present.

The main stem of Palisades Creek is about 30 kilometers in length. However, at approximately 11 km from the confluence with the South Fork Snake River there is a high-gradient, steep cascading section that fish surveys identify as a natural velocity barrier (Meyer et al. 2006a; K. Meyer, unpublished data). Rainbow Trout and hybrids have not been identified upstream from this barrier although Yellowstone Cutthroat Trout and Paiute Sculpin have. Removal of Rainbow Trout and hybrids occurred only in the lower 11 km of stream.

In an attempt to preclude upstream migration of Rainbow Trout and hybrids, several weirs have been operated on Palisades Creek since the year 2000. Most of these weirs were not efficient or could not be operated during high flows when Rainbow Trout are spawning. In 2009, a permanent electric weir was installed on Palisades Creek 0.7 km upstream from the confluence with the South Fork Snake River. The weir has six parallel electrodes made of metal railing embedded in an Insulcrete apron along the stream bottom, with the upper surfaces of the railings exposed to the water. The railings span the entire stream channel and continue up the Insulcrete walls that enclose the entire stream except for the fish trap. The fish trap is located on the east bank, outside the electric field. The lowermost and uppermost electrodes are parasitic, meaning that electrical current does not bleed upstream or downstream of these electrodes. Consequently, fish that approach the electric field from a downstream location can enter the fish trap without experiencing any electrical current. Electric settings are generally about 11.5 Hz, 2.5 ms pulse width, and 265 volts, which produce horizontal voltage gradients ranging from -11 to +12 V/cm but with most values in the -5 to +5 V/cm range.

Rainbow Trout and hybrid removals have occurred during base flow conditions in 2010, 2012, and 2013. In 2013, the removal consisted of two events separated by one month. Removal efforts were not possible during 2011 due to high flow conditions that rendered electrofishing inefficient and dangerous. Also, due to damaging high flows, the electric weir failed in 2011 early in the spawning run and had to be shut down for the remainder of the spawning period. During removals in other years, teams with backpack electrofishers and nets electrofished sequential sections of about 800 m. Removals started at the electric weir and proceeded approximately 10 km upstream to the natural barrier.

In 2010, three teams each consisted of two people with backpack electrofishers and two or more people with nets and buckets. In 2012, two teams each consisted of three people with backpack electrofishers and three or more people with nets and buckets. For the 2013 removal,

two teams consisted of three people with backpack electrofishers and one or more people with nets and buckets. Teams moved upstream in 800 m sections.

During sampling, persons with backpack electrofishers covered all available habitats. Where gradient was steepest, 1-2 electrofishers were used to block downstream movement of trout, and the remaining electrofishers were used to herd trout downstream through the steep gradient section into an area with slower water velocity where the fish were immobilized and netted more efficiently. All fish were anesthetized with MS-222, identified, and measured for total length (TL). Rainbow Trout and hybrids were euthanized and removed. Yellowstone Cutthroat Trout were released after recovering from anesthesia.

One to four weeks prior to the electrofishing removals in 2012 and 2013, Yellowstone Cutthroat Trout were collected, measured, and marked with a maxillary clip, caudal fin clip or anal fin clip, in order to estimate trout abundance, capture efficiency, and Rainbow Trout removal efficiency. Only Yellowstone Cutthroat Trout ≥ 100 mm total length were marked and released, whereas Rainbow Trout and hybrids that were encountered during the marking runs were removed from the stream.

The Fisheries Analysis + software package (Montana Fish, Wildlife and Parks 2004) was used to estimate trout abundance, using the modified Peterson approach. Separate estimates were made for the smallest size groups possible (usually 25-50 mm), having at least three marked fish per group in order to satisfy model assumptions. We assumed that there were: 1) no mortality of marked fish; 2) no movement of marked or unmarked fish out of Palisades Creek between the marking and recapture run; and 3) no difference in capture efficiency between Cutthroat Trout, Rainbow Trout, and hybrids. All trout were pooled for an overall estimate of abundance, and estimates for each species (hybrids were grouped with Rainbow Trout) were calculated based on the proportion of catch comprised by each species in each size class during the recapture run. Because Rainbow Trout first spawn at about 250 mm TL in the South Fork Snake River drainage (B. High, personal communication), population estimates were summarized by this size cutoff to determine if the potential spawning population of Rainbow Trout and hybrids, as well as the entire population, was decreasing in abundance.

RESULTS

A total of 2,626, 1,843, 1,279 and 2,228 fish were captured in Palisades Creek in 2010, 2012, and both 2013 removals, respectively; of these fish, 848, 484, 369, and 653 were Rainbow Trout or hybrids and were consequently removed. In the removal section of Palisades Creek, we estimated there were 4,141 trout (≥ 100 mm) of all species in 2012, compared to 5,967 in August 2013 and 6,422 in September 2013 (Table 8); no population estimate was conducted in 2010.

Population abundance of Rainbow Trout and hybrids increased from a total of 1,132 in 2012 to 2,243 for the first removal of 2013, and then declined slightly to 1,922 for the second removal of 2013. Most of these fish were subadults (< 250 mm); and at the time of the second removal in 2013, there were only an estimated 78 Rainbow Trout ≥ 25 cm remaining in the stream, of which 36 were removed during that survey.

The proportion of Yellowstone Cutthroat Trout increased from 61% to 73% from 2010 to 2012. Since then, that proportion has remained consistent at about 70% as identified by both removals in 2013. However, within each size class, the proportion of the total catch comprised

of Yellowstone Cutthroat Trout showed different trends (Figure 11). For the smallest fish (<150 mm), Cutthroat Trout comprised approximately 80% of the total catch during each removal except the first removal in 2013, where they comprised 62% of the total catch. For intermediate-sized fish (150-250 mm), Cutthroat Trout comprised about 60% of the total catch. For the largest fish (>250 mm), Cutthroat Trout comprised 61% of the total catch in 2010 and increased to over 90% by 2013.

In general, there were fewer Rainbow Trout and hybrids in the upper reaches of the treatment section (Figure 12). The lowest proportion of Yellowstone Cutthroat Trout during all four removals occurred in the lower reaches of the treatment section from rkm 3.2-4.8. The highest proportion of Yellowstone Cutthroat Trout in 2012 (0.92) and both removals in 2013 (0.91 and 0.89) occurred in the upper 2 km of the treatment section. Surprisingly, the highest proportion of Yellowstone Cutthroat Trout in 2010 (0.83) occurred in the bottom section of stream.

Mean capture efficiency decreased from 0.51 in 2012 to 0.32 and 0.35 for the two 2013 removals. However, capture efficiency generally increased as fish size increased for all three removal events for which capture efficiency was estimated (Table 9).

Mean length of Rainbow Trout and hybrids was 225 mm in 2010, 204 mm in 2012, 168 mm in the first 2013 removal, and 175 mm in the second 2013 removal. For Yellowstone Cutthroat Trout, mean length was 191 mm, 202 mm, 220 mm, and 200 mm, respectively, for each sequential removal. For Rainbow Trout and hybrids, by the second removal of 2013 we caught only five fish that were >300 mm in length. Size structure for Rainbow Trout and hybrids has shifted to smaller fish with subsequent removals. Alternatively, size structure for Yellowstone Cutthroat Trout has changed very little over time (Figure 12). There was less change in size structure for Rainbow Trout and hybrids from 2010 to 2012, likely because high water precluded electric weir operation and electrofishing removals in 2011. However by 2013, size structure shifted substantially toward smaller fish.

DISCUSSION

Palisades Creek is a wide, steep, swift, and deep stream that is difficult to sample with electrofishing equipment. Thus, it is not surprising that capture efficiency of marked fish was only between 32% and 51%. Even in small streams that are relatively less complex, achieving capture efficiency above 50% with backpack electrofishers is difficult (Meyer and High 2011). Capture efficiency was highest for spawning-sized fish, which is promising since removing the largest fish should reduce recruitment of Rainbow Trout and hybrids in the stream. Unfortunately, just the opposite seems to be occurring, with the abundance of subadult Rainbow Trout and hybrids doubling from 2012 to 2013 (Table 9). This disconnect suggests one or more of the following issues may have diminished the success of the removals. First, the weir may not be completely blocking Rainbow Trout and hybrids when in operation, which may be allowing adult Rainbow Trout to successfully enter and spawn in Palisades Creek. This explanation seems implausible because the abundance of spawning-sized Rainbow Trout and hybrids has declined since the removals began (Table 9). Second, perhaps new Rainbow Trout and hybrids (especially subadults) are entering the stream when the weir is not being operated (i.e., sometime other than the spawning migration). Third, the lack of operation of the weir in 2011, combined with our inability to remove Rainbow Trout and hybrids that year, may have allowed Rainbow Trout and hybrids to enter the stream, spawn, and produced a strong age-0 year class in 2011. This explanation seems unlikely because, based on length frequency analyses, a

strong age-0 year class of Rainbow Trout and hybrids in 2011 would have been about 150-200 mm in length in 2012, and no strong year class was evident during the 2012 removal (Figure 13). Finally, the remaining Rainbow Trout and hybrids that are missed during the removals may be undergoing a compensatory response to the removals via increased survival or reproductive success, thereby diminishing the success of the removal efforts (see Meyer et al. 2006b).

Regardless of the reason that the removals are not achieving the desired result, after four removals in three years, it now appears unlikely that continued removal efforts in Palisades Creek will result in a nearly pure Yellowstone Cutthroat Trout population in the foreseeable future. Nevertheless, although returning the Yellowstone Cutthroat Trout population in Palisades Creek to a nearly pure condition may not be feasible, removals of Rainbow Trout and hybrids at the electric weir, during the spawning run, may still be successful at maintaining a healthy population of Yellowstone Cutthroat Trout in this stream. Moreover, we did not analyze the genotypes of Rainbow Trout and hybrids in 2013, and although the total number of Rainbow Trout and hybrids may not be greatly diminishing, the number of Rainbow Trout alleles in the population may be diminishing at a faster rate, if the hybrids have fewer Rainbow Trout alleles.

The original goal of this project was to purify a Yellowstone Cutthroat Trout population with three consecutive years of removal effort. Because high water in 2011 precluded removals that year, 2014 will be the third consecutive year of removal effort. It will be important to do two removals in 2014, and to achieve high capture efficiencies. A final decision on whether to attempt removals in 2015 should be made once data from 2014 is analyzed.

RECOMMENDATIONS

1. Due to the high gradient of some sections of Palisades Creek, teams with three electrofishers should be used, with two shockers “blocking” the bottom ends of high-gradient reaches while a third shocker herds fish downstream out of the steepest habitat and into areas where netting fish is more efficient.
2. Because each removal effort achieves <50% reduction in Rainbow Trout and hybrids, two removals will be needed in 2014 during the base flow period to further reduce introgression.
3. Yellowstone Cutthroat Trout should continue to be marked and released before each removal so that capture efficiency (and therefore Rainbow Trout removal efficiency) can be estimated for each removal effort. Any Rainbow Trout or hybrids captured during the marking runs should be removed.
4. Perhaps the electric weir needs to be operated for a longer period of time, since the amount of Rainbow Trout and hybrid removal does not match the amount of reduction observed in our catch data, suggesting that Rainbow Trout and hybrids are perhaps still colonizing the stream.
5. In 2014, genetic samples should be collected from hybrids and pure YCT during marking runs and removals to characterize introgression in the population. The results of the genetic analyses should be compared to the genetic analyses of 2012.

ACKNOWLEDGEMENTS

We thank the numerous staff involved in the removal efforts including, Forrest Bohlen, Trevor Gipson, Jake Graham, Matt Hively, Kyle Jemmett, Lee Mabee, Conor McClure, Raleigh Scott, Joe Thiessen, and Chuck Traughber. We also appreciated the assistance from several local volunteers and we thank the local landowners for allowing access through their properties. Matthew Corsi and Jordan Messner provided early reviews and Cheryl Zink helped format and edit this document. Funding for this work was provided by anglers and boaters through their purchase of Idaho fishing licenses, tags and permits and from federal excise taxes on fishing equipment and boat fuel through the Sport Fish Restoration Program.

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TABLES

Table 8. Population abundance estimates and the number of Rainbow Trout and hybrids removed during each removal effort at Palisades Creek, Idaho.

Year	Rainbow trout and hybrids				Total trout estimate	
	<25 cm		≥25 cm		<25 cm	≥25 cm
	Estimate	Removed	Estimate	Removed		
2010	-	691	-	289	-	-
2012	996	357	136	137	3,228	913
2013a	2,121	227	122	29	4,504	1,463
2013b	1,844	613	78	36	5,333	1,089

Table 9. Number of marked fish (*m*) and capture efficiency for various size groups of Yellowstone Cutthroat Trout during one removal in 2012 and two removals in 2013 at Palisades Creek, Idaho.

size group (mm)	2012		2013a		2013b	
	<i>m</i>	Capture efficiency	<i>m</i>	Capture efficiency	<i>m</i>	Capture efficiency
100-149	13	0.00	13	0.00	24	0.13
150-199	32	0.38	16	0.25	20	0.50
200-249	33	0.45	15	0.20	25	0.24
250-299	47	0.60	13	0.08	30	0.57
300-349	20	0.85	13	0.31	22	0.32
350-499	8	0.75	11	1.27	8	0.25
total	153	0.51	81	0.32	129	0.35

FIGURES

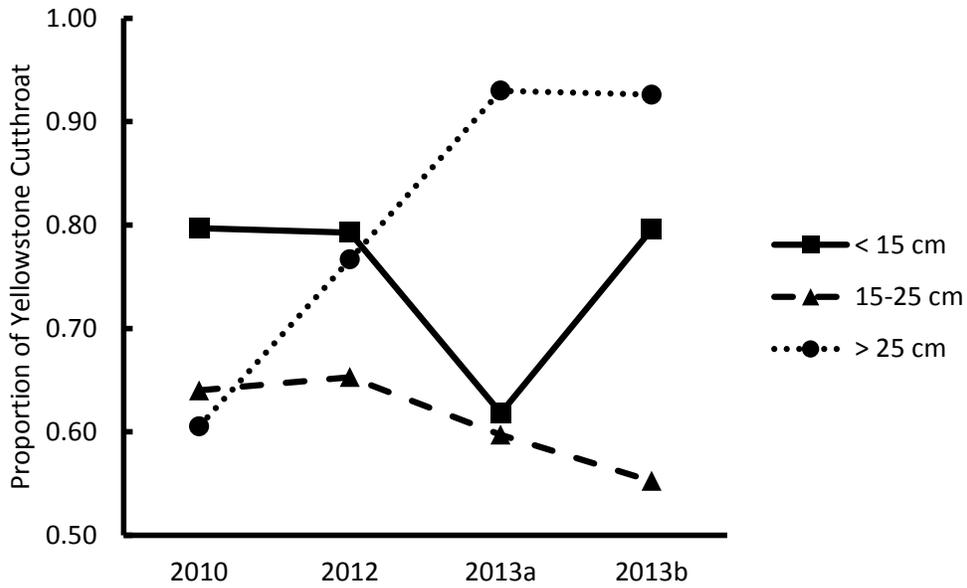


Figure 11. Proportion of three size classes of Yellowstone Cutthroat Trout during removals in 2010, 2012 and two removals in 2013 at Palisades Creek, Idaho.

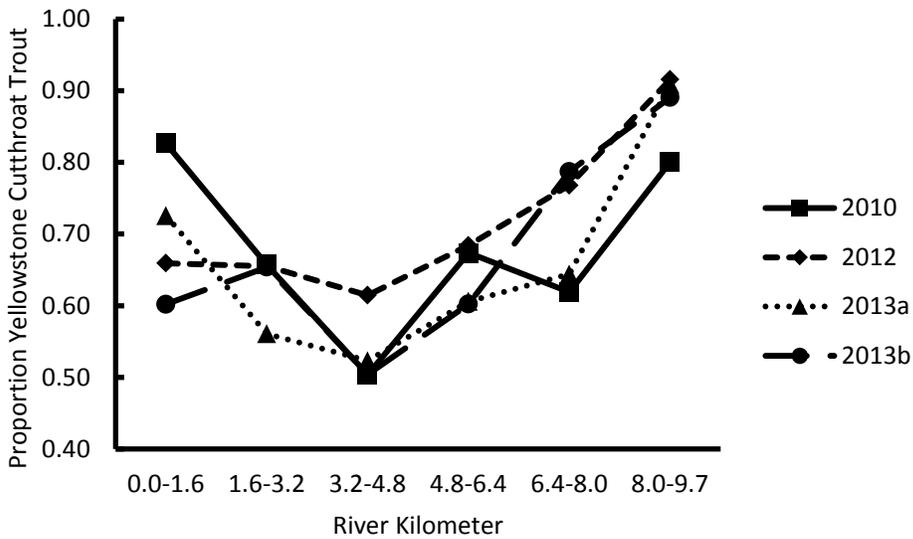


Figure 12. Proportion of Yellowstone Cutthroat Trout during removals in 2010, 2012, and two removals in 2013 in six sections of Palisades Creek, Idaho.

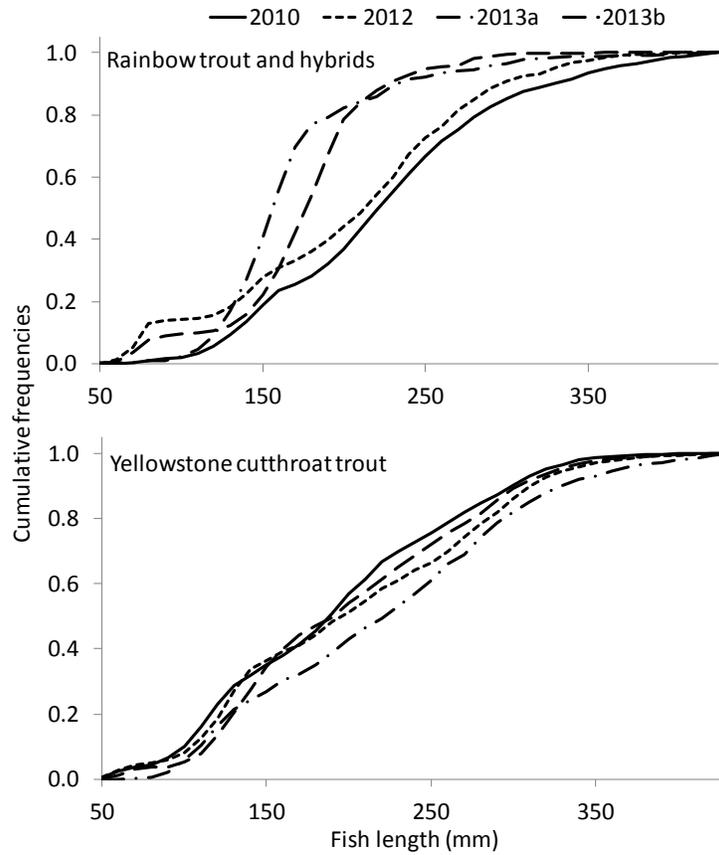


Figure 13. Cumulative length frequencies for fish caught during removals in 2010, 2012, and two removals in 2013 at Palisades Creek, Idaho.

CHAPTER 3: PREDATION RATE OF AMERICAN WHITE PELICANS ON STOCKED CATCHABLE RAINBOW TROUT IN IDAHO RESERVOIRS

ABSTRACT

In southern Idaho, growth of two American White Pelican *Pelicanus erythrorhynchos* nesting colonies since the early 1990s has generated fishery concerns about the effects of pelican predation on resident salmonid stocks, including wild and hatchery trout. To assess the level of impact that American White Pelican predation may be having on catchable Rainbow Trout *Oncorhynchus mykiss*, predation rates were estimated at several Idaho study waters, and comparisons were made to angler harvest at the same locations. In the second year of pelican predation research, we PIT tagged (to monitor bird predation) and anchor tagged (to monitor angler harvest) catchable Rainbow Trout and stocked them into 13 study waters, and tagged Yellowstone Cutthroat Trout at Henrys Lake, for a total of 4,318 PIT-tagged fish released at large. We also directly fed 827 PIT-tagged fish to pelicans across all study waters. Once the juvenile pelicans were fledged, we recovered 654 tags from three nearby pelican nesting colonies (at Minidoka, Blackfoot, and Yellowstone National Park), as well as from loafing areas for pelicans and Double-crested Cormorants *Phalacrocorax auritus*. Recoveries of PIT tags from fed fish allowed us to estimate PIT-tag recovery efficiency for fish that were known to have been consumed by pelicans. Tag recovery efficiency was then used to correct the unadjusted predation rate (based only on tag recoveries from fish at large), thereby enabling us to estimate the total pelican predation rate. Predation by American White Pelicans in 2013 study waters averaged 15% and ranged from 0-48%. In contrast, angler harvest averaged only 5%. In general there was an inverse relationship between distance from the nearest nesting colony and tag recovery efficiency, and between pelican predation and angler harvest. Further research in new and repeat waters in 2014 will finish this study.

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INTRODUCTION

American White Pelicans *Pelicanus erythrorhynchos* (hereafter pelicans) experienced long-term declines in abundance across North America until the 1960s (Knopf and Evans 2004). The cause of the decline is not clear but was likely related to a lack of federal and state protection and the heedless use of pesticides prior to the 1960s (Keith 2005). Regardless of what caused the decline, since the 1980s pelicans have experienced an almost exponential rebound in abundance (King and Anderson 2005), including at the two nesting colonies in Idaho. For example, from 1993 to 2013 the Blackfoot Reservoir pelican colony increased from approximately 200 to 1,980 breeding birds, while the Lake Walcott colony increased from approximately 1,000 to 4,100 breeding birds (Idaho Department of Fish and Game (IDFG), unpublished data). These two colonies generally constitute about 2/3 of the entire statewide pelican abundance, with non-breeding birds (and breeders from neighboring states) constituting the rest (IDFG, unpublished data). Although pelicans likely inhabited Idaho before European settlement, the Idaho nesting colonies are artificial by-products of reservoir construction, which created man-made islands largely free of terrestrial predators where pelicans have flourished (IDFG 2009).

While recent increases in abundance are positive signs for the conservation of American White Pelicans across North America, the increasing population size has also resulted in documented cases of pelican predation impacts on native fish populations and important recreational fisheries. For example, pelicans have been shown to capitalize on fish spawning migrations (Findholt and Anderson 1995; Murphy and Tracy 2005), and in Idaho, pelicans frequently consume substantial portions of the spawning migration of native Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (YCT) in the Blackfoot River system (Teuscher and Schill 2010). Substantial levels of pelican predation have also been documented on hatchery trout within days of individual stocking events (Derby and Lovvorn 1997). In Idaho, pelicans at Blackfoot Reservoir have been shown to target stocking areas shortly after hatchery trout are released (Teuscher et al. 2005), and anglers and biologists in southern Idaho have reported incidents of ponds being devoid of stocked trout within days after planting, with a noticeable increase in pelican activity during the same timeframe (D. Megargle, IDFG, personal communication). Such impacts are not surprising for a generalist predator such as the American White Pelican that exhibits plasticity in its opportunistic feeding habits (Hall 1925; Knopf and Kennedy 1980). With the noticeable increase in the presence of pelicans at local fisheries, anglers and fisheries management agencies are increasingly interested in understanding the impact that pelicans may be having on inland fisheries.

Recent innovative research investigating avian predation on salmonids in the Pacific Northwest has focused on recovery of Passive Integrated Transponder (PIT) tags at bird nesting colonies that were implanted in salmonids and subsequently consumed by nesting birds and deposited at the colonies (Evans et al. 2012; Sebring et al. 2013). Although PIT-tag recovery efficiency at the colonies has been estimated by intentionally sowing “control tags” onto the bird colony before PIT-tag recovery efforts are undertaken, a shortcoming to this approach is that off-colony deposition rate is unknown. Consequently, PIT-tag recoveries using this methodology only provide minimum predation estimates since not all tags that are consumed by birds are deposited at the colony. We used an updated modification to this approach that incorporates estimates of off-colony deposition (Osterback et al. 2013), thereby producing estimates of total predation by pelicans. The primary objective of this study was to estimate predation rates by American White Pelicans on hatchery catchable Rainbow Trout in several southern Idaho reservoirs and community ponds to gauge their general impact on hatchery trout fisheries in southern Idaho.

In instances where pelican predation of stocked hatchery fish is relatively high, it follows that angler catch of those same fish would likely be minimal since a large proportion of the stocked fish would likely be consumed by pelicans before anglers could successfully catch them. However, angler catch of hatchery trout stocked in lentic environments is affected by numerous factors other than avian predation, such as rearing conditions in the hatchery, season of stocking, the abundance of piscine predators, and water quality after release (e.g., Davison 1997; Yule et al. 2000; Barnes et al. 2009; Koenig and Meyer 2011). Thus a low rate of pelican predation would not necessarily translate directly to high rates of angler catch. Rather, a wedge-shaped pattern might be expected, where high pelican predation would nearly always lead to low rates of angler catch, but low pelican predation would not necessarily result in high rates of angler catch. Likewise, we expected that pelican predation, at least by breeding adults, would always be low at great distances from a nesting colony because breeding birds would choose to forage at waters closer to their nest. However, at waters in close proximity to nesting colonies, pelican predation would not necessarily be high because pelican predation is affected by more than just travel distance from the nest to the foraging water, such as water depth (Kaeding 2002; Ivey and Herziger 2006) and water clarity (Anderson 1991) where the birds are foraging. Thus we also expected a wedge-shaped pattern between the distance from the nearest colony to a particular study water and the rate of pelican predation at that water. Our second objective was to evaluate whether these two wedge-shaped patterns materialized in our data.

METHODS

American White Pelican nesting in Idaho consists of two islands at Blackfoot Reservoir and three islands at Lake Walcott (also a reservoir), the latter of which is part of the Minidoka National Wildlife Refuge (NWR). Pelicans in recent years have been attempting to nest at Island Park in eastern Idaho but success has been limited. Other nearby pelican nesting colonies include Molly Island at Yellowstone National Park and Gunnison Island at the Great Salt Lake. Study waters where hatchery trout were stocked were scattered across southern Idaho (Table 11). Study waters were not selected at random but rather to investigate predation losses in several southern Idaho fisheries known or suspected to be receiving substantial pelican use, and to gain perspective on possible geographical gradients in pelican predation rates across southern Idaho.

Distances between the two Idaho pelican colonies and the study waters ranged from 0 to 412 km (Table 11). The soaring ability of pelicans enables them to forage at distances of up to 300 km from their nests (Johnson and Sloan 1978; Trottier et al. 1980; O'Malley and Evans 1982). Double-crested Cormorants *Phalacrocorax auritus* (hereafter cormorants) also nest at the Blackfoot Reservoir and Lake Walcott islands, and they also have been shown to be effective predators on hatchery trout populations in some circumstances (e.g., Modde et al. 1996). However, the maximum foraging distance for cormorants is considered to be only about 50 km from their nesting colonies (Custer and Bunck 1992; Bugajski et al. 2012). Thus, for study waters within 50 km of the nesting colonies, PIT-tagged fish consumed by avian predators and deposited at the Blackfoot or Walcott islands could be the result of pelican or cormorant predation. Our study design attempted to account for PIT-tag deposition by cormorants in order to substantiate our pelican predation estimates (see below).

Estimating pelican predation

Estimating the rate of pelican predation involved four steps. The first step was to stock PIT-tagged catchable Rainbow Trout into the study waters that were then vulnerable to pelican (and cormorant) predation. A second step was to PIT tag hatchery Rainbow Trout and feed them directly to pelicans. The third step occurred after pelicans (and other birds) on the nesting colonies had fledged their young; at that time we searched the two colonies for regurgitated and defecated PIT tags from both of these PIT-tagged groups of fish (stocked and fed). The final step was to apportion the recovered tags into those known or assumed to have been consumed by either pelicans or cormorants.

By recovering tags from stocked fish at the nesting colony, and assigning them to either cormorant or pelican predation events, we were able to estimate an unadjusted rate of pelican predation and cormorant predation (i.e., the ratio of the number recovered to the number stocked). This unadjusted rate of predation did not account for stocked fish that were consumed by either pelicans or cormorants but were either not deposited on or not recovered at the nesting colony. By recovering tags from fish fed directly to pelicans, we were able to estimate tag recovery efficiency from fish that were known to be consumed by pelicans (i.e., the ratio of the number of recovered fed tags to the total number fed, stratified by water). These two independent estimates - unadjusted pelican predation and tag recovery efficiency - enabled us to estimate total pelican predation on stocked hatchery Rainbow Trout. Because cormorants were not fed directly, no such tag recovery efficiency could be estimated for cormorants. Thus, all tag recoveries by cormorants led only to minimum estimates of cormorant predation.

Fish stocking

To accomplish the first step in the methodology outlined above, we stocked 100-400 PIT-tagged catchable Rainbow Trout into each study water in conjunction with regularly scheduled hatchery stocking events. Mean size of stocked fish averaged about 250 mm TL and generally ranged from 200-300 mm. In addition, IDFG Region 6 personnel PIT-tagged 400 Yellowstone Cutthroat Trout at Henrys Lake, and they ranged in size from 250-500 mm.

Pelican feeding

During this same timeframe, we also fed fish directly to pelicans, using live or dead fish that were abdominally tagged with PIT tags (23 mm half-duplex tags). Feeding occurred between late May and mid-July, which encompassed the time when breeding pelicans were foraging and traveling between the breeding colonies and foraging sites to feed their chicks. For each feeding event, Rainbow Trout ~250 mm TL were transported from a nearby hatchery and injected with a PIT tag in the abdominal cavity and a small amount of air under the skin before being launched in the direction of loafing or foraging pelicans. Although pelicans were initially difficult to approach and feed, after a few days they became more comfortable with our close proximity and reticently consumed fish thrown in their direction. The purpose of the injected air was to ensure the tagged fish stayed on the surface after being launched, increasing the likelihood that pelicans would consume the fish. Pelicans were observed after they captured a tagged fish until a swallowing motion – known as a head toss (Anderson 1991) - was noted. Attempts were made to minimize the occurrence of individual birds consuming more than one tagged fish in order to achieve independence in tag recoveries. In addition, no more than 40 tagged fish were deployed on a given day to maximize tag dispersion and subsequent tag recovery independence, although at times 100 or more pelicans were attempting to consume

fish being fed to them. The few fed fish that were consumed by other birds (either gulls or herons) were omitted as lost tags.

PIT-tag recoveries

For the third step outlined above, we searched for regurgitated and defecated PIT tags from fed and stocked fish at the breeding colonies after the adult and juvenile birds had left the islands in the fall. Areas targeted for scanning on the colonies were marked into grids using surveyor flagging, and searchers scanned the grid systematically to ensure that all of the ground was covered. We used a backpack PIT tag reader (Oregon RFID HDX Backpack Reader), and when a tag was detected, surveyors used shovels and sieves to recover and remove the tags if they were not visible on the surface, in order to avoid interference with other PIT tags in the same area or in following years. In the few instances where we were unable to recover and remove the tag, attempts were made to ensure no other PIT tags were in the area, and individual PIT tag numbers were recorded. Tag recovery efficiency at the nesting colonies was unknown but likely extremely high (i.e., >95%) because often we searched for tags on the islands more than one time, and rarely did we find more than 1 or 2 tags on a second pass even though dozens if not hundreds were usually recovered during the first pass.

Apportioning tags to pelicans or cormorants at nesting colonies

To assign PIT-tag recoveries to either pelican predation or cormorant predation at Lake Walcott, any tags recovered directly in a pelican nest or cormorant nest were assigned to the nest type from which the tag was recovered. For any tag recovered from a stocking event >50 km from the nesting colony, predation was assumed to have arisen from pelicans because this distance exceeds the foraging range for cormorants (Custer and Bunck 1992; Bugajski et al. 2012). For tags recovered from stocking events within 50 km of the nesting colony, and not recovered in a pelican or cormorant nest, we assumed that predation occurred at a rate proportional to bird abundance at the Lake Walcott colony. This was determined by mounting cameras on fence posts placed strategically around the Lake Walcott islands to best capture images of birds present. These cameras periodically captured images each day, resulting in over 6,000 images from May to September. We randomly subsampled from these images to estimate the pelican to cormorant ratio at the Lake Walcott nesting colony.

A similar methodology was used to assign PIT-tag recoveries to either pelican or cormorant predation at the Blackfoot nesting colony for tags recovered directly in nests, and for tags recovered from waters >50 km from the colony. For tags recovered from stocking events within 50 km of the nesting colony, and not recovered in a pelican or cormorant nest, pelican or cormorant predation was assigned based on tag recovery location on the island, because pelican nesting and cormorant nesting did not overlap on either island. For tag recoveries at Molly Island in Yellowstone National Park, all tag recoveries were assumed to be the result of pelican predation since no study water was within 50 km of Molly Island.

Apportioning tags to pelicans or cormorants at loafing areas

At some waters, we searched for tags at cormorant and pelican loafing areas (often these birds loafed together). To separate these recovered PIT tags into pelican or cormorant predation events, and as a relative index of bird abundance at each study water, several counts of pelicans and cormorants were made each month from May to September. Count surveys were stratified by month and were made on randomly assigned days and times (daylight hours only), with counting start time also randomized. Counts were also stratified into weekday or

weekend surveys, but because pelican and cormorant counts did not differ between these two categories (data not shown), this stratification was discarded for further analyses. For the larger study waters where surveys could not be completed from one stationary location, the direction of counting around the study water (i.e., clockwise or counter-clockwise) was also randomized for each sample. During each survey at the larger study waters, a motorized boat was driven around the reservoir from which counts were made. High-powered binoculars and digital cameras were used to aid in counting. Blackfoot Reservoir was too large to fully count birds during each survey, and instead we randomly chose dates to sample only a portion of the reservoir. On these days, we separated the count data into that portion of the reservoir that was always counted and that portion that was only sometimes counted. The relationship between the partial and full counts was linear for both pelicans (correlation coefficient, $r = 0.97$) and cormorants ($r = 0.92$), and we used these relationships to estimate total counts of birds for days when partial counts were made. Based on these count data, tags recovered at loafing areas were assigned to either pelican or cormorant predation.

Calculating pelican predation rates

Proportions of recovered tags were calculated independently for both the fed tags (p_x) and stocked tags (p_y), where:

p_x = number of fed tags found on the colony divided by the total number of tags directly fed to pelicans on the individual study waters.

p_y = number of stocked PIT tags found on the colony divided by the total number of stocked PIT tags released in the study waters.

Variance for these proportions was calculated according to the formula in Fleiss (1981) as:

$$Var(\text{proportion}) = \sqrt{\frac{PQ}{n}}$$

where P is the numerator divided by the denominator for either p_x or p_y , Q is $1-P$, and n is the denominator in either p_x or p_y . We calculated the pelican predation rate ($Pelican_p$) for each water when both fed and stocked tags were recovered at a nesting colony according to the following formula:

$$Pelican_p = p_y/p_x$$

To calculate the variance for $Pelican_p$, we used the formula for the variance of a ratio (McFadden 1961; Yates 1980)

$$s^2\left(\frac{p_y}{p_x}\right) = \left(\frac{p_y}{p_x}\right)^2 \times \left(\frac{S_{p_y^2}}{p_y^2} + \frac{S_{p_x^2}}{p_x^2}\right)$$

For each water-specific estimate of pelican predation rate, we then calculated 90% CIs.

Estimating angler exploitation

To estimate angler exploitation, we used T-bar anchor tags to tag the same fish that were released with PIT tags in the study waters. For more details on anchor tagging methods and estimating angler exploitation, see Meyer et al. (2010, 2012). In short, anchor tags were labeled with the agency and phone number (i.e., "IDFG 1-866-258-0338") where tags could be reported. A toll-free automated telephone hotline and website were established through which anglers could report tags, although some tags were mailed to or dropped off at IDFG offices. Tag reporting by anglers in this program was voluntary, not mandatory.

Unadjusted angler exploitation rate (u) for each stocking event was calculated as the number of tagged fish reported as harvested divided by the number of fish released with tags; variance for this proportion was again calculated according to Fleiss (1981). Typically, angler exploitation is calculated as an annual rate, but at this time, a full year has not elapsed since tagged fish were released in 2013; therefore, the estimates presented for 2013 will be adjusted once a full year has elapsed. Adjusted angler exploitation rate (u') incorporated estimates of angler tag reporting rate (λ), anchor tag loss (tag_l), and tag mortality (Tag_m) (estimated to be 49.4%, 8.2%, and 1%, respectively; Meyer et al. 2010), and used the following formula:

$$u' = \frac{u}{\lambda(1 - Tag_l)(1 - Tag_m)}$$

Variance estimates for λ , tag_l , and tag_m came from Meyer et al. (2010). Variance for the entire denominator in the above equation was estimated using the approximate formula for the variance of a product in Yates (1980):

$$s_{x_1 x_2}^2 = x_1^2 \cdot s_{x_2}^2 + x_2^2 \cdot s_{x_1}^2$$

where $s_{x_1 x_2}^2$ is the variance of the product, x_1 and x_2 are independent estimates being multiplied together, and $s_{x_1}^2$ and $s_{x_2}^2$ are their respective variances. Variance for u' was calculated using the approximate formula for the variance of a ratio as previously noted, from which 90% CIs were derived.

While counting pelicans and cormorants at individual waters, we also counted the number of anglers present as a comparative count relative to the abundance of pelicans and cormorants. At Blackfoot Reservoir, the partial counts and full counts were highly correlated for anglers ($r = 0.99$), and we used this relationship to estimate angler counts on days when partial counts were made.

RESULTS

Bird counts

At the 14 waters in our study, the abundance of pelicans and cormorants varied greatly between seasons and waters (Figure 14). In general, pelicans tended to outnumber cormorants, although this relationship also varied seasonally at some waters. Summing across all waters and months, there was a mean monthly abundance of 2,236 pelicans and 1,661 cormorants. There was a surprisingly strong correlation ($r = 0.87$) between the monthly mean abundance of pelicans and cormorants across all waters (Figure 15).

Pelican predation rate

We stocked or captured and tagged a total of 4,318 trout for the 14 study waters during 2013, and directly fed an additional 827 PIT-tagged trout to pelicans in 9 of the 14 study waters (Table 12). At the two Idaho pelican nesting colonies (i.e., Blackfoot and Walcott) we recovered a total of 448 PIT tags, while at the remaining nesting colonies and the loafing areas at each study water we recovered an additional 202 PIT-tags (Table 12). This constituted 10% of all the stocked tags and 27% of all the fed tags from 2013. We apportioned 31% of all PIT-tag recoveries from stocked hatchery trout or captured-and-tagged wild trout to pelican predation while the remaining 69% were apportioned to cormorant predation. For tags fed directly to pelicans, there was a negative exponential relationship between the distance from a particular study water to the nearest pelican nesting colony and fed tag recovery efficiency at that study water (Figure 16).

At nine study waters, pelicans were fed and thus pelican predation could be estimated directly. At two of these waters, no stocked tags were recovered and thus pelican predation was 0% (Table 13), although Cascade Reservoir, being 304 km from the nearest Idaho pelican nesting colony, was at the limit of pelican foraging range from a colony so we expected to recover few tags from this location despite the fact that on average there were >300 pelicans at that location. At the other seven waters where pelican predation could be directly estimated, predation rate averaged 16% and ranged from 1-34% (Table 13).

At the five study waters where pelicans were not directly fed fish with PIT tags, pelican predation estimates could not be directly estimated. However, using the relationship between the distance from a particular study water to the nearest pelican nesting colony and the recovery efficiency of fed tags at that study water (Figure 16), we predicted that pelican predation at three of the waters averaged 32% and ranged from 2-48% (Table 13); at two of these waters we estimated predation to be 0%.

There was a negative relationship between distance from the nearest pelican nesting colony and pelican predation rates (Figure 17). At short distances to the nearest pelican colony, pelican predation varied essentially from zero to nearly 50%, but at greater distances to the nearest colony, pelican predation declined precipitously. This relationship was statistically significant ($t = -1.82$; $P = 0.09$). The maximum distance between fish stocking or pelican feeding locations and tag recovery locations for pelicans was 274 km (tagged at Henrys Lake and recovered at Lake Walcott) and for cormorants was 32 km (tagged at Freedom Park and Rupert Gun Club ponds and recovered at Lake Walcott).

Angler exploitation

Angler exploitation at the 10 waters where anchor-tagged fish were released averaged 7.6% and ranged from 0% to 27.3% (Table 13). There was no relationship ($r = 0.00$) between the rates of pelican predation and angler exploitation at individual waters.

DISCUSSION

Our results from both 2012 (not presented) and 2013 suggest that predation by American White Pelicans on hatchery trout stocked at catchable size in some southern Idaho waters may be relatively high (i.e., >25%). In Wyoming, pelicans (and cormorants) quickly increased their focus on trout species as soon as hatchery trout were stocked, and consumed

an estimated 80% of the trout stocked in one year (Derby and Lovvorn 1997). Although our study includes results from only a handful of study waters and should therefore be considered preliminary, our findings support the supposition that in southern Idaho, pelican predation of trout stocked at catchable size will negatively affect angler harvest of these fish. Additional years of data may help clarify the strength and the geographical extent of this relationship.

Estimated predation rates by pelicans on stocked Rainbow Trout in the study waters were variable and apparently were related to the distance of the study water to a breeding colony in Idaho. Results from CJ Strike Reservoir suggest that pelican predation in Idaho is almost exclusively associated with breeding-aged birds (Figure 16). This concurs with findings from patagial tagged pelicans in Idaho which suggest that pelicans do not return to Idaho after fledging until about age-4 (C. Moulton, IDFG, unpublished data).

Pelican abundance varied greatly between waters. For many non-colony waters, pelican abundance was lower in early spring and late fall, with peak abundance during the summer. This trend coincides with the activities associated with the breeding season and associated foraging behavior of pelicans. During May and June, breeding pelicans in southern Idaho are selecting nesting sites, incubating eggs, and are occupied with early chick rearing, all of which require the presence of one adult on the nest at all times (Madden and Restani 2005). At these times, pelican abundance was highest in particular at the two Idaho nesting colonies. When pelican chicks reach four weeks of age, both parents begin leaving the breeding colony for foraging trips simultaneously (O'Malley and Evans 1982); this period coincides with peak abundances observed at our reservoirs, and therefore likely represents the time of greatest predation potential on fish stocks within those reservoirs. Abundance of cormorants was surprisingly strongly related to the abundance of pelicans, perhaps because their nesting behaviors and chick feeding strategies or requirements are similar or that food resources are shared and therefore affect the abundance of both birds similarly.

The maximum recorded distance of which we are aware that American White Pelicans have been shown to travel one way from nesting colonies to foraging areas is 305 km (Johnson and Sloan 1978), suggesting that nearly all of the reservoirs and ponds in southern Idaho are vulnerable to pelican predation (Table 11). PIT-tag recoveries from the breeding colony at Minidoka NWR showed that pelicans traveled a maximum distance of 274 km to forage at Henrys Lake. We found Henrys Lake tags at three separate recovery locations and two nesting colonies; this was the only water where tags were recovered at more than one nesting colony.

There are other American White Pelican breeding colonies outside of Idaho that are within the foraging range of many southern Idaho reservoirs, the largest being colonies at Gunnison Island in the Great Salt Lake in northern Utah, Malheur National Wildlife Refuge (NWR) in eastern Oregon, and Badger Island on the Columbia River in southeastern Washington. It is likely that breeding adults from these colonies forage in Idaho reservoirs, and their impact is currently unknown. For instance, we did not recover any stocked or efficiency tags from Cascade Reservoir at Idaho breeding colonies, but we regularly observed 400-1,000 pelicans at Cascade Reservoir during June-September. Most likely these birds are non-breeding adults because all the nearest colonies are still a tremendous distance from Cascade Reservoir and it is unlikely that more than a few birds would travel great distances to forage there. Scanning for tags in fall 2013 by staff at Oregon State University resulted in no recoveries of fed or stocked fish from Cascade Reservoir at Malheur NWR (A. Evans, Real Time Research, personal communication). The pelicans using the breeding colony on Gunnison Island in the Great Salt Lake, Utah, pose a potential threat to southern Idaho fisheries due to the large number of pelicans (8,000 nesting pairs in 2005) nesting there (King and Anderson 2005) and

its close proximity to numerous reservoirs in southeastern Idaho. Scanning these additional colonies in future years would help gain a better understanding of the full impact of pelican predation at a wider geographical scale.

RECOMMENDATIONS

1. Continue one more year of additional pelican research (in 2014) to evaluate pelican predation in Idaho waters.
2. Install more cameras at the Minidoka nesting colony to help parse future tag recoveries into cormorant- or pelican-consumed fish.

ACKNOWLEDGEMENTS

We thank the following individuals for their help in this project: Chris Sullivan, Erin Larsen, Liz Mamer, Dennis Daw, Tony Lamansky, John Cassinelli, Joe Thiessen, Shane Knipper, Jake Graham, Trevor Gibson, and Ryan Schiferl. We thank D. Tommy King for assisting in project development and pelican identification, and David Teuscher, Arnie Brimmer, Ryan Hilliard, and Matt Green for developing methodologies for pelican feeding and tag recovery. Finally we thank Dr. Mike Quist at the University of Idaho for use of a boat-mounted PIT-tag antenna. Matthew Corsi and Jordan Messner provided early reviews and Cheryl Zink helped format and edit this document. Funding for this work was provided by anglers and boaters through their purchase of Idaho fishing licenses, tags and permits and from federal excise taxes on fishing equipment and boat fuel through the Sport Fish Restoration Program.

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TABLES

Table 11. Distance (km) from 2013 study waters in southern and eastern Idaho to nearest American White Pelican nesting colonies. Based on existing literature on maximum foraging ranges for pelicans and Double-crested Cormorants, grayed numbers indicate distances that only pelicans can presumably traverse, whereas boxed gray numbers indicate distances that both pelicans and cormorants can presumably traverse.

Study waters	Nearest nesting colonies				
	Yellowstone National Park	Island Park Reservoir ^a	Blackfoot Reservoir	Great Salt Lake	Lake Walcott
Cascade Reservoir	459	363	412	448	304
CJ Strike Reservoir	483	385	354	313	201
Freedom Park Pond	346	272	181	154	32
Riley Creek Pond	415	323	274	231	118
Rupert Gun Club Pond	347	271	181	156	32
Lake Walcott	315	244	147	152	0
American Falls Reservoir	259	199	95	170	56
Jensen Grove Pond	215	158	70	218	109
Blackfoot Reservoir	192	163	0	202	148
Chesterfield Reservoir	213	174	27	187	119
Foster Reservoir	275	252	84	111	140
Glendale Reservoir	275	253	83	113	141
Daniels Reservoir	294	245	93	123	95
Henry's Lake	97	28	193	384	274

^aPelican nesting is annually attempted here but successful offspring are rarely produced.

Table 12. Initial numbers of fed (F) and stocked (S; with PIT tags and anchor tags) hatchery Rainbow Trout and PIT tags recovered from American White Pelican nesting colonies or loafing areas from study waters in 2013.

Water	IDFG region	Initial PIT tags at large		Recovered tags																				Total				
		F	S ^a	Cascade Reservoir		CJ Strike Reservoir		Lake Walcott		Blackfoot Reservoir		Chesterfield Reservoir		Daniels Reservoir		Foster Reservoir		Smithfield Canal		Henry's Lake		Island Park Reservoir		Yellowstone Nat'l Park		F	S	
				F	S	F	S	F	S	F	S	F	S	F	S	F	S	F	S	F	S	F	S	F	S			
Tags known or assumed to be pelicans																												
Cascade Reservoir	3M	125	450	NA ^b																							0	NA ^b
CJ Strike Reservoir	3N	100	400			3	1	2	2																		5	3
Freedom Park Pond	4	0	100						16																		NA	16
Lake Walcott	4	91	397					44	65																		44	65
Riley pond	4	39	100					24	4																		24	4
Rupert Gun Club Pond	4	0	99						16																		NA	16
American Falls Reservoir	5	101	396					9	11																		9	11
Blackfoot Reservoir	5	143	300					3		98	2																101	2
Chesterfield Reservoir	5	80	385						1	19	5	1	0														20	6
Daniels Reservoir	5	50	299										5														5	0
Foster Reservoir	5	0	293												1			0									NA	1
Glendale Reservoir	5	0	399												0												NA	0
Jensen Grove Pond	5	0	300						1		1															NA	2	
Henry's Lake	6	98	400						1									2	1	3	1	12	5			15	7	
Total		827	4318	0	0	3	1	82	117	117	8	1	0	5	0	0	1	0	0	2	1	3	1	12	5	223	133	
Tags known or assumed to be cormorants																												
Cascade Reservoir	3M	0	450	NA ^b																							0	NA ^b
CJ Strike Reservoir	3N	0	400			0	1	0	0																		0	1
Freedom Park Pond	4	0	100						3																		0	3
Lake Walcott	4	0	397						17																		0	17
Riley pond	4	0	100						0																		0	0
Rupert Gun Club Pond	4	0	99						2																		0	2
American Falls Reservoir	5	0	396						0																		0	0
Blackfoot Reservoir	5	0	300					0		0	6																0	6
Chesterfield Reservoir	5	0	385						0	0	96	0	52														0	148
Daniels Reservoir	5	0	299										0														0	0
Foster Reservoir	5	0	293												97		2										0	99
Glendale Reservoir	5	0	399												20												0	20
Jensen Grove Pond	5	0	300						0		0																0	0
Henry's Lake	6	0	400						0									0	2	0	2	0	0	0	0	0	0	2
Total		0	4318	0	0	0	1	0	22	0	102	0	52	0	0	0	117	0	2	0	2	0	2	0	0	0	298	

^aStocked PIT-tagged fish were available for pelicans or cormorants to consume.
^bTwo tags were recovered but cormorant counts were not made, thus tags cannot be apportioned.

Table 13. Estimates of American White Pelican predation and angler harvest in select Idaho waters in 2012. Predicted predation estimates are derived from distance to the nearest pelican colony and the distance vs. tag recovery efficiency relationship in Figure 16. NA means data not available.

Water	IDFG region	Estimated pelican predation (%)	Predicted pelican predation (%)	Estimated angler harvest (%)
Cascade Reservoir	3M	0.0		6.2
CJ Strike Reservoir	3N	25.0		6.8
Freedom Park Pond	4		47.2	27.3
Lake Walcott	4	33.9		0.0
Riley pond	4	6.5		5.6
Rupert Gun Club Pond	4		47.6	0.0
American Falls Reservoir	5	31.2		0.0
Blackfoot Reservoir	5	1.0		NA
Chesterfield Reservoir	5	5.5		1.4
Daniels Reservoir	5			NA
Foster Reservoir	5		0.0	20.0
Glendale Reservoir	5		0.0	9.1
Jensen Grove Pond	5		2.3	NA
Henrys Lake	6	10.2		NA

FIGURES

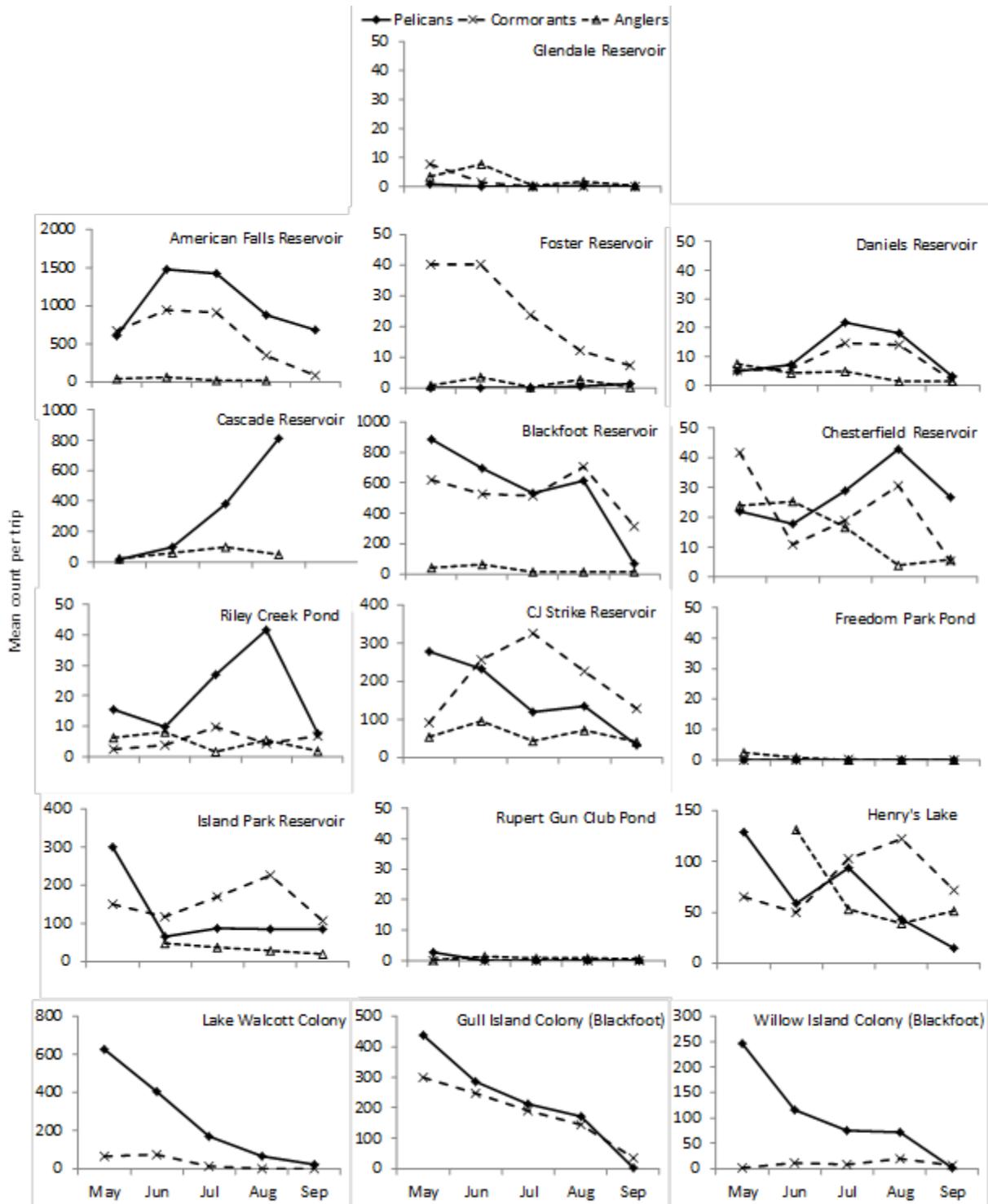


Figure 14. Mean monthly counts of American White Pelicans, Double-crested Cormorants, and anglers from May through September at study waters in 2013. Cormorants were not counted at Cascade Reservoir.

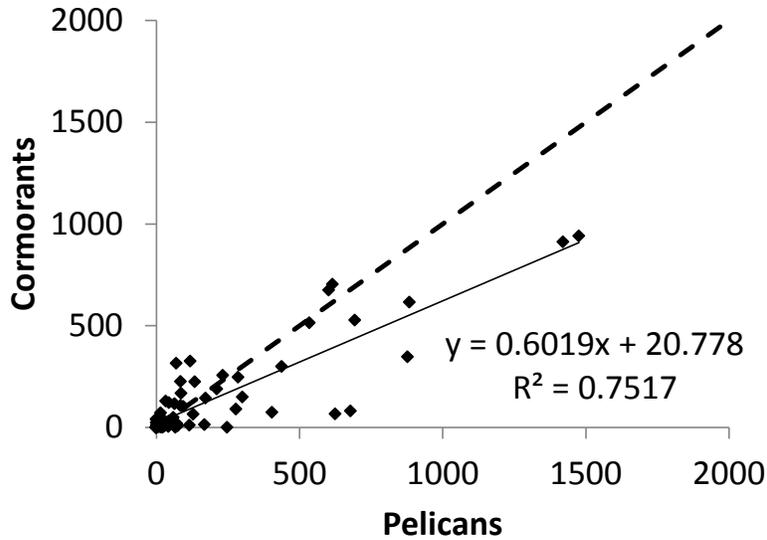


Figure 15. The relationship between each pair-wise comparison of the mean monthly abundance of American White Pelicans and Double-crested Cormorants at 14 study waters in southern Idaho in 2013. The solid line and equation were derived from fitting a least-squares regression to the data, whereas the dashed line depicts a 1:1 relationship.

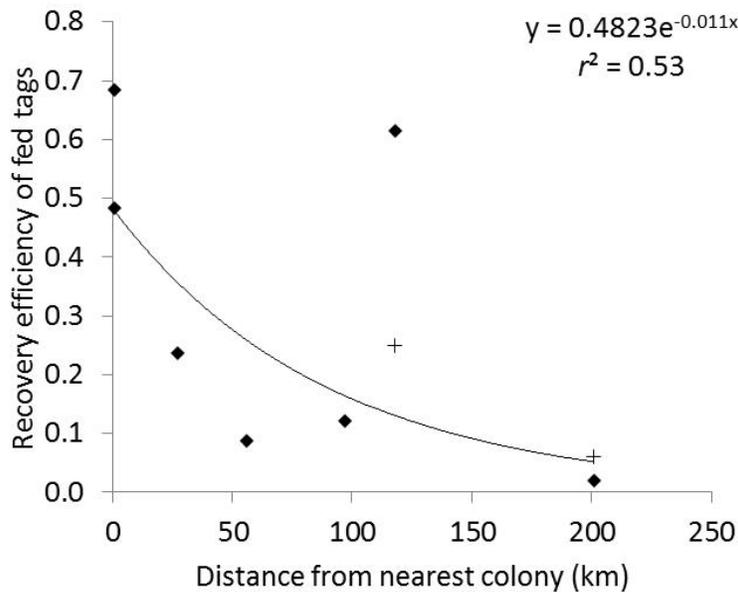


Figure 16. Relationship between a study waters' distance from the nearest American White Pelican colony and the recovery efficiency of PIT tags implanted in hatchery Rainbow Trout and fed directly to pelicans at that study water in 2012 and 2013; cross markers are for 2012 data.

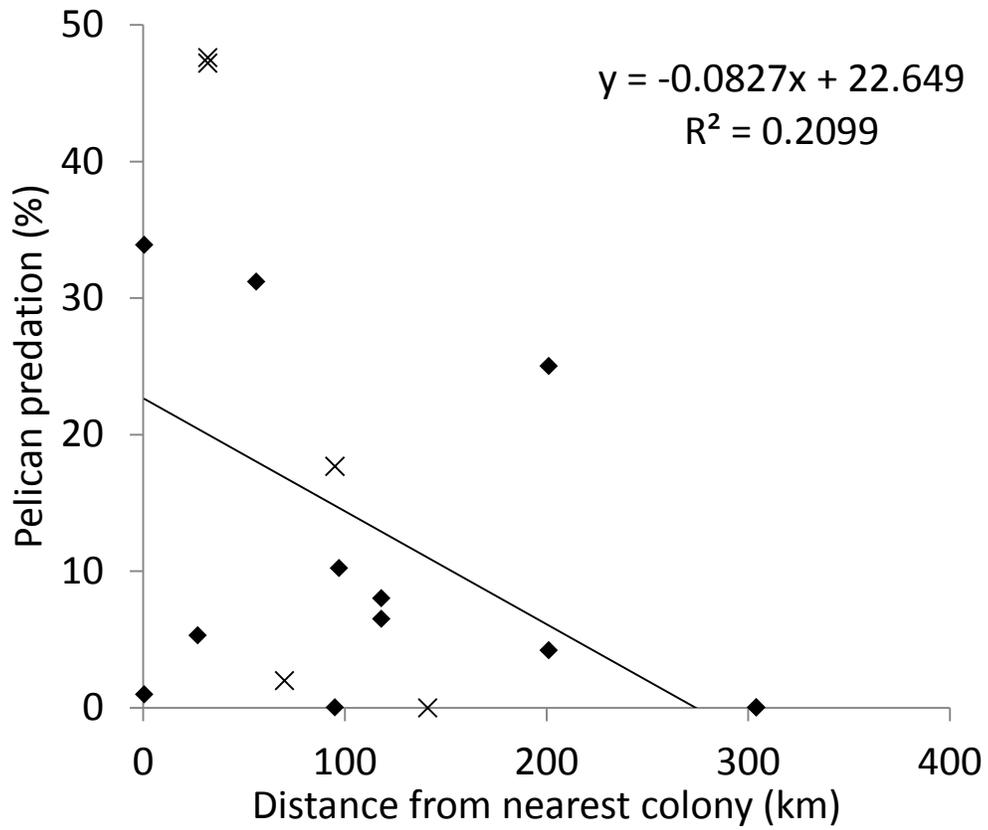


Figure 17. Relationship between a study waters' distance to the nearest American White Pelican nesting colony and the pelican predation rate on hatchery Rainbow Trout stocked at that water. Predation rates for the waters labeled with an "x" were generated from the relationship in Figure 16.

CHAPTER 4: COMPARISON OF TWO TAG TYPES AND FOUR TAGGING LOCATIONS ON LONG-TERM TAG RETENTION RATES IN WILD TROUT OF SPAWNING SIZE

ABSTRACT

Biologists often tag fish to identify them in the future, however, retention of tags is often compromised because tags can deteriorate or be shed over time. We evaluated retention rates for a visual implant elastomer (VIE) tag and for three injection sites of passive integrated transponder (PIT) tags in stream-dwelling Cutthroat Trout *Oncorhynchus clarkii* and Rainbow Trout *O. mykiss* of spawning size. In three streams in southeastern Idaho, 1,471 fish ≥ 150 mm (total length) were PIT tagged in the (1) body cavity, (2) musculature immediately posterior of the cleithrum bone, or (3) dorsal musculature. Maxillary clips provided a secondary physical mark. All streams were revisited approximately one year after tagging (range: 358-439 days). Retention of PIT tags was highest in the dorsal musculature (95%), followed by the musculature posterior of the cleithrum (90%), and the body cavity (69%). Retention of VIE tags was 94% and retention of maxillary clips was 100%. Streams will be revisited two years post marking to further evaluate spawning size on tag retention. Tagging locations other than the body cavity appear to offer greater retention in fish of spawning size, though concerns over angler interaction with tags during filleting and consumption should be considered.

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INTRODUCTION

Tags placed in fish have broad application as a tool for fisheries managers to attempt to quantify population dynamics of fish in the wild. A foremost challenge of any tag type is the ability to non-lethally and yet accurately identify individuals, at least to the group level, over time. Many options exist to meet this need; however, not all withstand the test of extended retention in the wild, requiring the quantification of these tag retention rates prior to analysis. Of particular interest are those tagging techniques that are retained throughout the life cycle of the fish, and are capable of withstanding biological changes exhibited by mature fish without tag shedding or degradation.

Tag retention varies with fish species, tag type, and tagging location. As described by Nielson (1992), some characteristics of the 'perfect mark' are: permanence throughout the fish's life cycle, ease of application, high likelihood of being observed by the untrained eye upon recapture, and low cost. Knowing that tag retention rates are affected by many factors is intrinsic to the ability to accurately describe the population under study when using tags as an identifier. How a tag type is effective as a long-term identification tool is greatly influenced by where the tag is applied and what effect location might have on retention throughout various growth stages of the fish.

Two tagging systems, passive integrated transponder (PIT) tags and visual implant elastomer (VIE) tags, are frequently used to identify individual fish (Bruyndoncx et al. 2002; Hopko et al. 2010; Younk et al. 2010; Soula et al. 2012). Multiple studies have evaluated retention rates for PIT and VIE tags in trout (Prentice et al. 1990; Bonneau et al. 1995; Hale and Gray 1998; Close 2000; Josephson and Robinson 2008; Knudsen et al. 2009); however, little information is available on long-term retention of tags in mature wild fish.

Passive integrated transponder tags inserted in the intraperitoneal cavity of trout have been shown to be shed at a higher rate as fish size increases, especially for females, presumably due to spawning activity. Of the few studies addressing the effect of stream-dwelling trout size and implied sexual maturity upon PIT-tag retention rates, Bateman et al. (2009) found that smaller (<140 mm) Westslope Cutthroat Trout residing in headwaters were 1.4 times as likely to retain PIT tags than larger (>174 mm) fish, though they were unable to identify to sex the population of fish marked. For stream-dwelling redband trout in southern Idaho, Meyer et al. (2011) demonstrated a difference in long-term PIT-tag retention rate (body cavity tagging location) between spawning size females (retention = 67%) and males (retention = 90%), suggesting female sexual maturation causes a higher expulsion rate due to spawning activity associated with egg deposition. When assessing 2-month retention rates of PIT tags in Brook Trout *Salvelinus fontinalis* and Brown Trout *Salmo trutta*, Dieterman and Hoxmeier (2009) found that retention in the body cavity was 70% and 56%, respectively, after the fall spawning period. This same study found dorsal musculature PIT-tag retention, over a two-month period not including a spawning event, was 100% for Brook Trout and 95% for Brown Trout.

The retention of VIE tags has been studied in many species and environments (Dewey and Zigler 1996; Haines et al. 1998; Goldsmith et al. 2003, Leblanc and Noakes 2012) yet the period of time involved has generally been too short to fully assess whether the tags were retained or degraded over time. Two studies found that VIE tags became fragmented in as little as 30-45 days after application (Astorga et al. 2005; Soula et al. 2012), but for long-term observations, Fitzgerald et al. (2004) found 92% retention in net pen Atlantic Salmon *Salmo salar* after 16 months while Willis et al. (2001) were able to identify adult Snapper *Pagrus auratus* three years post-tagging.

This study was initiated due to the concern over the lack of information regarding PIT and VIE tag retention in trout of spawning size. The objective of this study was to evaluate long-term (one and two year) PIT and VIE tag retention rates, differentiated by injection location, of mature-sized salmonids dwelling in a stream environment that would likely have gone through at least one spawning event. We sought to assess whether alternate PIT-tagging locations within the body can be used to maintain high retention rates throughout the entire life cycle of trout. We intend to revisit these study sites two years post-tagging in August 2014.

METHODS

In the fall of 2012, Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT) and Rainbow x Cutthroat hybrids (RbtHYB) were collected from three streams in southeastern Idaho (Table 14). The streams were selected because they had large numbers of mature fish, resident (i.e., non-migratory) populations, and essentially no angler exploitation to maximize recapture potential and avoid possible interaction with tags by anglers.

Two backpack electrofishing units were used to conduct a single upstream pass. Fish >150 mm total length (TL) were held in 19-liter buckets while electrofishing. At periodic intervals, fish were relaxed in an immersion bath of 15-20 ppm isoeugenol (AQUI-S[®], New Zealand). Once measured to the nearest mm (TL), fish were tagged, held in freshwater until recovered, and released near the area from which they were collected.

Three PIT-tag injection site locations were evaluated for likelihood of tag loss. A full-duplex PIT tag (12 mm long, 2 mm diameter) was injected using a 12-gauge stainless steel veterinary hypodermic needle and modified syringe (Prentice et al. 1990) in one of three areas. PIT-tag locations included (1) in the body cavity (injected into the abdominal cavity, anterior to the pelvic girdle, offset from the dorsoventral axis); (2) in the dorsal musculature (injected post-anteriorly, shallow subcutaneous depth, ventral to dorsal structure); and (3) behind the cleithrum (inserted subcutaneously, dorsoventrally, directly posterior and parallel to cleithrum) (Figure 18, Figure 19). PIT tags were read by a portable PIT-tag reader once injected into the fish. Tagging wounds were not sealed by any surgical glue or closure.

A single injection site location was evaluated for VIE. A biologically non-reactive elastomer, VIE (Northwest Marine Technologies, Shaw Island, Washington) was injected subdermally into the minimally pigmented tissue against and perpendicular to the bony structure of the lower mandible (Figure 20). Using a 28-gauge needle on a .33 cc syringe inserted into a manual VIE elastomer injector, the needle tip was shallowly inserted and a volume of elastomer injected as the needle was withdrawn, stopping just prior to the needle bezel exiting the tissue. Excess elastomer was removed by gently wiping the injection site with a fingertip to avoid leaving strands of pigment to harden outside of the dermis in an attempt to decrease the likelihood of shedding.

To differentiate between additional tagging events of future years, fish collected in this event (2012) had the adipose fin clipped. A corresponding year specific color of VIE was injected into the lower jaw tissue. A secondary physical mark in the form of maxillary clip combinations (either right, left, or both) was used to act as an indication of a PIT-tag location should the tag be shed (Siepker et al 2012).

In the fall of 2013, using the same backpack electrofishing method (single pass), fish were recaptured from 358 to 439 days post-tagging, and interrogated for all marking types and locations. Fish were scanned for PIT tags using a portable PIT-tag reader, and VIE marks were examined using incidental light. Any presence of elastomer material was considered a mark, but partial or fragmented conditions were not noted. Those fish identified as recaptures had marks recorded and were released. Those found to be new specimens were PIT tagged, maxillary clipped, and marked with the second year VIE color, though these fish were not ad clipped, so as to differentiate between year groups. Recaptured fish that had shed a PIT tag were not retagged with a new PIT tag or VIE color so that one-year and two-year retention rates could be estimated. Results of the second year recaptures will be available in 2014.

Long-term retention rates by injection location were calculated as the proportion of recaptured fish that had retained a tag relative to the number of all recaptured fish that presented an external physical mark (max clip or ad clip) indicating that it was part of the initial marking effort. Ninety-five percent confidence intervals (CIs) were calculated for the retention rate of each tag location, and differences in proportions were evaluated for statistical significance based on non-overlapping confidence intervals (Fleiss 1981). For the sake of analysis, all species were considered together when calculating tag retention rates.

RESULTS

A total of 1,471 fish were tagged in 2012 and the total length of fish tagged ranged from 150-415 mm (mean = 210 mm, SD = 45; Table 14). In 2013, of the 291 recaptured fish, 244 (84%) retained PIT tags, 276 (94%) retained a VIE tag, and 290 (99.6%) retained a maxillary clip mark for longer than one year (Figure 21). The rate of PIT-tag retention was highest for the dorsal musculature site ($95 \pm 4\%$; $n = 94$) compared to the cleithrum site ($90 \pm 6\%$; $n = 92$) and the body cavity site ($69 \pm 9\%$; $n = 105$). The VIE mark was observed in 274 of 290 ($94 \pm 2\%$; $n = 290$) recaptured fish (Table 15).

Retention rates for PIT tags were similar across all tagging locations for fish in the smallest size class (150-224 mm), ranging from 86-95% (Figure 22). However as fish size increased, PIT-tag retention rate for the body cavity location was markedly lower (63% and 65% for fish between 250-274 mm and fish >274 mm respectively) than for the cleithrum (91% and 91%) and dorsal musculature (96% and 100% for fish between 250-274 mm and fish >274 mm respectively) tagging locations. For fish greater than 249 mm, there was a significant difference between the proportion of tags retained in the body cavity location when compared to the remaining three tag types. There was no relationship between fish size and tag retention for the VIE tags.

DISCUSSION

Ease of application, tag retention, and duration of the study are some elements to be considered when choosing an appropriate tag type. In this study, over one year's duration, retention of PIT and VIE tags was similar in fish less than 225 mm; however, as body size increased, retention of PIT tags dropped significantly when injected into the body cavity when compared to cleithrum or dorsal musculature locations. Visual injectable elastomer tag retention when injected into the jaw appeared to be less affected by fish length.

Similar to the findings of Meyer et al. (2011), retention rates of PIT tags can be affected by fish size. We did not sacrifice fish in order to preserve the continuation of this study for one more year, and therefore were unable to attribute the noted difference in body cavity tag retention rates between size classes to a sexual component; however, the trend is similar. For PIT tags to remain effective throughout the life cycle of the fish, a location other than the body cavity should be used. Both the cleithrum and dorsal musculature locations appear to be unaffected by whatever physiological factors might be causing higher PIT-tag shedding, though these two locations come with an additional consideration of angler interaction upon harvest (e.g., Dieterman and Hoxmeier 2009). Sieper et al. (2012) investigated using Plastic Infusion Process (PIP) PIT tags in response to food safety hazards created by consumption of a glass PIT tag. Working with largemouth bass, they found that tag retention, when injected in either the intraperitoneal cavity or the dorsal musculature, was 100% over one year but that given the larger tag size, the utility of this type of tag might be limited to fish of larger size. McKenzie et al. (2006) noted that PIP tags, in snapper *Pagrus auratus*, could be considered reliable for as long as two years.

A VIE tag is easy to apply, is easily observed, and requires little special equipment; however, in some studies, size at tagging and length of time at large outlasted mark longevity. In muskellunge marked as fingerlings, Younk et al. (2010) found VIE to be a viable mark (100% retention) after 176 days, but rates dropped drastically 2-6 years later. Passive integrated transponder tags, while providing a unique identifier within a population, have demonstrated moderate retention rates in stream-dwelling trout, possibly due to natural spawning activity when the tag is forcefully ejected with the eggs (Bateman et al. 2009; Meyer et al. 2011).

Being able to easily see a mark is vital to its usefulness. The VIE colors used in this study (red in 2012, blue in 2013) are considered contrasting colors and of highest visibility (Astorga et al. 2005). However, this study did not quantify fragmentation of the elastomer in the tissue, which, should shedding continue to occur over time, could result in fewer VIE marks retained. Bangs et al. (2013) did not assume 'perfect detection' past 150 days due to fragmentation. Curtis (2006) noted that levels of tag visibility, color determination, and injection location with respect to pigmentation can all affect the utility of VIE tags.

Close and Jones (2002) described some possible limitations to VIE retention in fingerling Rainbow Trout as the development of pigmentation and tissue overgrowth affecting visibility and the occurrence of mark fragmentation. Josephson and Robinson (2008) studied brook trout in hatchery and lake environments, reporting a 50-72% VIE tag retention when observed under outdoor sunlight conditions approximately 400 d post-tagging, declining to 0% at 959 d. Upon dissection, however, they found the retention rate was 100%, raising the question of the difference between detection and retention. Fitzgerald et al. (2004) also noted the distinction between detection and retention, pointing out that in their study no effort was made to actually retrieve the tag from the fish (such as dissection) so defined 'detection' and 'retention' as "visual identification of a tag, if any, without the implicit assumption of tag presence." We rated the effectiveness of a VIE tag based upon its visible component only, though this tag type could possibly be used similarly to a coded wire tag requiring dissection to be quantified.

A weakness of this study is that there was not a consistent method of assessing PIT-tag reader error rate, which would potentially elevate the implied effect of fish size on PIT-tag retention; however, our study design should alleviate this issue with data from 2014 when dissections can be made to assess PIT-tag reader error rate.

RECOMMENDATIONS

PIT tags, when injected into the body cavity, are an effective method to identify individual fish, up to spawning size. After that stage in life history has been reached, however, retention rates are compromised. Secondary injection sites are equivalent to prespawning body cavity retention rates; however, in order to use PIT tags in the dorsal musculature or post-cleithrum injection locations, further study should be undertaken to assess the likelihood of anglers interacting with the tags while filleting their catch. In future studies, PIT-tag reader error rate should be recorded. If the size structure of the population under study supports the use of plastic encased PIT tags (fish >300 mm), it might be valuable to test the usefulness of this angler-friendly tag alternative. VIE tags are useful to identify, to the group level, fish populations of all range of sizes. Creating a scoring protocol to discern VIE fragmentation and condition, as suggested by Bangs et al. 2013, such as 0 = easily recognizable (1 obviously visible mark), 1 = visible but fragmented (multiple fragments), 2 = unrecognizable without assistance (required UV light, low light conditions, or dissection to observe) would help understand the level of degradation occurring over time.

ACKNOWLEDGEMENTS

For assistance with stream collection efforts, we would like to thank Chris Sullivan, Brad Wright, Erin Larson, Steve Elle, John Cassinelli, Jessica Buelow, Patrick Kennedy, and Region 6 seasonal staff. For data entry we thank Ryan Schiferl. Matthew Corsi and Jordan Messner provided early reviews and Cheryl Zink helped format and edit this document. Funding for this work was provided by anglers and boaters through their purchase of Idaho fishing licenses, tags and permits and from federal excise taxes on fishing equipment and boat fuel through the Sport Fish Restoration Program.

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TABLES

Table 14. Numbers and average length (mm, SD) at tagging and recapture of Rainbow Trout (RBT), Rainbow Trout-Yellowstone Cutthroat Trout hybrid (RbtHYB), and Yellowstone Cutthroat Trout (YCT) on three southeast Idaho streams.

Waterbody	Species	Tag			Recapture		
		Date	<i>n</i>	Ave. Length	Date	<i>n</i>	Ave. Length
Badger Cr	RBT	07/13/12	579	209 (± 40)	10/02/13	176	248 (± 34)
	RbtHYB & YCT		21	236 (± 54)			
Fall Cr	YCT	09/25/12	630	206 (± 41)	09/23/13	90	236 (± 31)
Rainey Cr	YCT	07/17/12	241	221 (± 58)	09/24/13	24	269 (± 44)

Table 15. Numbers of trout collected at least one year post-tagging with one of three different passive integrated transponder (PIT) tag insertion locations (body cavity; cleithrum; dorsal musculature) and visible injectable elastomer (VIE) with corresponding tag retention rates (percent retained) in three southwestern Idaho streams.

Waterbody	Days post tagging	Tagging Location											
		Body Cavity			Cleithrum			Dorsal musculature			VIE-tag		
		Number of tagged fish			Number of tagged fish			Number of tagged fish			Number of tagged fish		
		Recap	Wit h tag	Retenti on (%)	Recap	Wit h tag	Retenti on (%)	Recap	Wit h tag	Retenti on (%)	Recap	Wit h tag	Retenti on (%)
Badger Cr	439	65	44	68%	60	56	93%	52	52	100%	176	171	97%
Fall Cr	358	31	19	61%	25	21	84%	34	31	91%	90	82	91%
Rainey Cr	427	9	9	100%	7	6	86%	8	8	100%	24	23	96%
Total		105	72	69%	92	83	90%	94	89	95%	290	276	94%

FIGURES

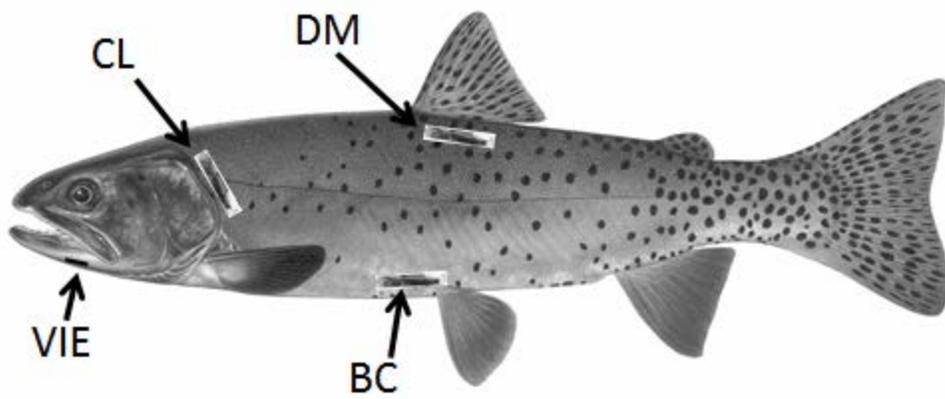


Figure 18. Locations of PIT-tag injection sites (BC, body cavity; CL, cleithrum; DM, dorsal musculature), and visible injectable elastomer (VIE) injection site.



Figure 19. Image of PIT-tagging location and appearance in the body cavity, cleithral, and dorsal musculature injection locations.



Figure 20. Location of VIE mark on lower jaw (indicated by arrow).

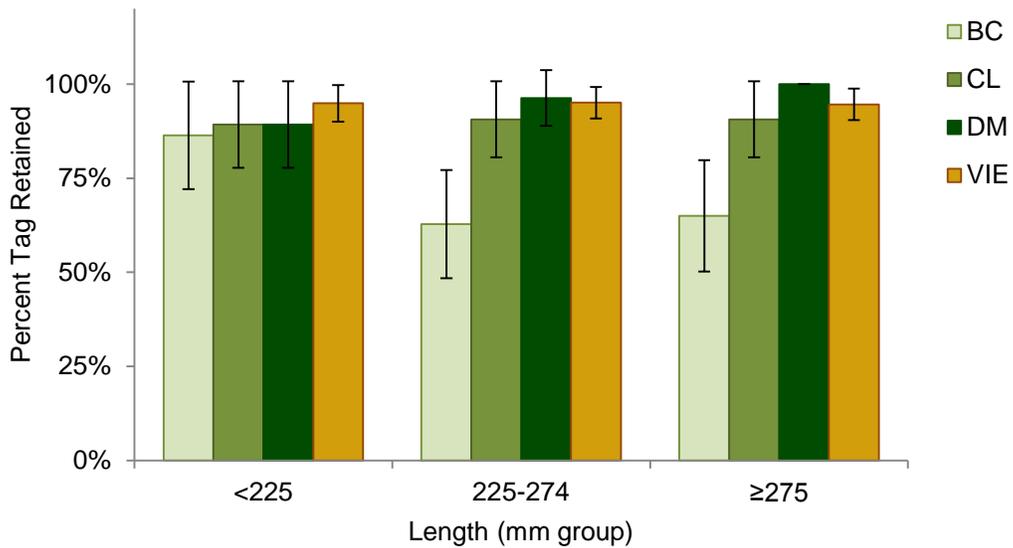


Figure 21. Effect of body size on percent tag retention, one year post tagging, evaluating three passive integrated transponder (PIT) tag locations (body cavity - BC; cleithrum - CL; dorsal musculature - DM), and a visible implant elastomer (VIE) tag .

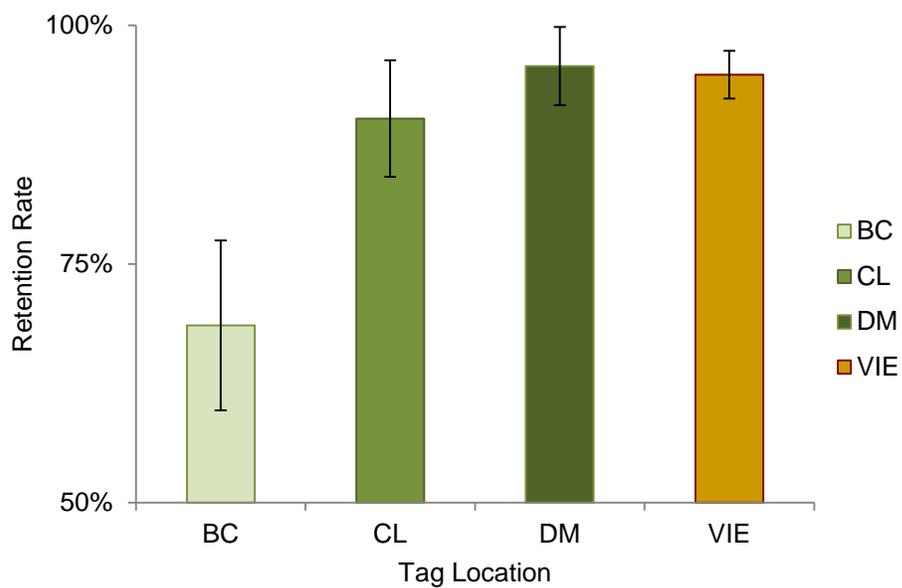


Figure 22. Retention rate, one year post tagging, evaluating three passive integrated transponder (PIT) tag locations (body cavity - BC; cleithrum - CL; dorsal musculature - DM), and a visible implant elastomer (VIE) tag.

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