Trends in the Abundance of Westslope Cutthroat Trout in Idaho

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Abstract—Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (WCT) are the most widely distributed subspecies of Cutthroat Trout in western North America. Despite known declines, data on trends in abundance are generally lacking range-wide for this subspecies. We evaluated WCT trends in abundance throughout much of Idaho using daytime snorkeling, screw trap, and angler catch data to index abundance. We also evaluated whether data sets contained observation error, and whether any easily-measured, broad-scale bioclimatic indices were correlated to WCT abundance through time. A total of 17 data sets were available within nine river drainages that contained WCT; on average, data sets covered a period of record of 25 years and averaged 19 years of data. Of these 17 data sets, 10 showed statistically significant population growth (at $\alpha = 0.10$), two showed statistically significant population declines, and five were stable (with 90% error bounds that overlapped zero). Seven of the 17 datasets were estimated to have high observation error, which likely inflated the error bounds around those trend estimates. The bioclimatic variables we included in our study (indices of streamflow, water temperature, marine-derived nutrient influx, and drought) explained little of the variation in WCT abundance.

Introduction

Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (WCT) are the most widely distributed subspecies of Cutthroat Trout in western North America (Behnke 2002), but despite their widespread distribution, declines in occupancy and abundance have occurred (Shepard et al. 2005). In Idaho, WCT now occupy approximately 50% of their historic range (Wallace and Zaroban 2013). Concerns about the status of WCT resulted in two petitions for listing under the Endangered Species Act, both of which were denied. Nevertheless, the U.S. Forest Service and the Bureau of Land Management regard WCT as a sensitive species, and the Idaho Department of Fish and Game (IDFG) has designated it as a Species of Greatest Conservation Need.

Concern over the status of WCT across their native range has resulted in numerous status assessments (e.g., Schill et al. 2004; Shepard et al. 2005). These assessments have highlighted a lack of information on trends in WCT abundance. One exception was a summary of several long-term trend monitoring data sets in Idaho (Schill et al. 2004); these authors concluded that WCT abundance in Idaho was generally stable or increasing. However, this study included only data from the St. Joe, Coeur d'Alene, Selway, and Middle Fork Salmon rivers, so that the area of inference from their study was small relative to the entire distribution of WCT in Idaho. Our primary objective was to more completely summarize trends in WCT abundance in Idaho, using all available data. Secondarily, because trend monitoring data is often subject to substantial observation error (Dennis et al. 2006), which can diminish the ability to detect statistically significant changes in abundance (Dunham et al. 2001), we estimated how much observation error was present in these WCT trend data sets. Finally, we examined whether trends in WCT abundance were correlated with several broad-scale bioclimatic variables, in an attempt to partially explain patterns in WCT trends in abundance that we observed in Idaho.

Methods

Trends in WCT Abundance

Westslope Cutthroat Trout occurrence is well documented in Idaho, but metapopulation boundaries have not been well defined. The IDFG delineated geographic management units (GMUs) to provide spatial reference for conservation efforts, and they included several river basins and multiple WCT populations. Studies have demonstrated that WCT can move substantially between large river drainages (Bjornn and Mallet 1964; Schoby and Keeley 2011). Herein, we make inferences on WCT trends at the smallest scale possible, which is generally at the scale of major river drainages. Areas of inference will hereafter be referred to as populations, though we acknowledge that several WCT populations may exist within these aggregates.

Several sources of data were used to index WCT abundance, including daytime summer snorkeling observations, screw trap catch, and angler catch. For snorkel surveys, from one to five observers (depending on stream width) counted all salmonids ≥150 mm total length (TL). Because Cutthroat Trout exhibit daytime concealment behavior at temperatures below 6-8°C (Griffith and Smith 1993), and such behavior would have negatively biased snorkel counts, we discarded all snorkel surveys conducted at water temperatures <6°C. We also discarded surveys when snorkeler visibility was <2 m (Thurow 1994).

For some populations, 1.5-m rotary screw-traps were used to capture WCT during routine monitoring of Chinook Salmon *O. tshawytscha* and Steelhead Trout *O. mykiss* outmigration. Screw traps were deployed as early as possible in the spring, usually in 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 10 continuous years of data were available from a consistent sample location. Total annual catch of WCT (>50 mm) at the screw trap was used as an index of abundance for the population.

In the Middle Fork Salmon River and Selway River, hook-and-line surveys have been collected annually by IDFG survey crews descending those rivers in raft trips to monitor resident and anadromous salmonid abundance. For the angling data, WCT of all size classes were summed as an index of abundance for the population.

We assessed trends in WCT abundance with least squares regression, using sample year as the independent variable and the index of abundance ($\log_e$-transformed) as the dependent variable. The regression line fit to these 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 α = 0.10 to increase the probability of detecting 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 (also see 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:

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

where $\hat{r}_t$ is the estimated instantaneous rate of change in year (t ($lnN_t + 1 - lnN_t$)), $\hat{a}$ 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). The Gompertz state-space model was therefore used to identify data sets that were estimated to have no observation error and thus (presumably) no bias in the error bounds around the trend estimates. We also 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 only slightly inflated the error bounds on estimates of trend. 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 were less reliable.

Bioclimatic Variables

We assessed whether abundance was related to several broad-scale bioclimatic variables, including drought, mean winter streamflow, mean annual air temperature (as a surrogate for water temperature), and the number of Chinook Salmon redds counted within the WCT population (as an index of marine-derived nutrient influx). The mean annual Palmer Drought Severity Index (PDSI) was computed for each population by the National Climatic Data Center (www.ncdc.noaa.gov). A point local to each population was selected from an area central to the population and along each respective stream channel. Mean winter streamflow was calculated for December through February from the U.S. Geological Survey gauge station (http://waterwatch.usgs.gov/?m=real&r=id&w=map) located most centrally within each WCT population. Mean annual air temperature was calculated from the West Wide Drought Tracker (http://www.wrcc.dri.edu/wwdt/time/) at a point near the center of the dendritic stream network of each WCT population. The number of Chinook Salmon redds was summed annually within each WCT population (IDFG, unpublished data), except for WCT populations outside the natural range of Chinook Salmon (i.e., the Coeur d'Alene and St. Joe rivers).

Because each bioclimatic variable could have potentially affected WCT recruitment or had other delayed impacts that outweighed effects on within-year abundance, we related each bioclimatic variable to WCT abundance within that same year as well as with a one-year time lag (Copeland and Meyer 2011). Because Chinook Salmon redds were often counted after WCT abundance data had been collected in a given year, evaluating whether Chinook Salmon redd abundance affected WCT abundance in the same year was illogical; instead, one-year and two-year time lags were used for this relationship. We used multiple linear regression models to relate bioclimatic data through time to WCT abundance through time. Akaike's information criterion was used to identify the best model for each data set.

Results

A total of 17 data sets were available within nine WCT populations that indexed WCT abundance through time (Table 1), including nine snorkeling data sets, six screw-trap data sets, and two angling data sets. Data sets on average covered a period of record of 25 years and averaged 19 years of data.

Of the 17 data sets used to estimate WCT trends, 10 showed statistically significant population growth, two showed statistically significant population decline, and five were considered stable with 90% error bounds that overlapped zero (Table 1; Figure 1). In the Coeur d'Alene River sub-basin, r_{intr} was statistically positive for both data sets. In the Clearwater River sub-basin, r_{intr} was statistically positive for six data sets, statistically negative for no data sets, and stable for two data sets. In contrast, r_{intr} in the Salmon River sub-basin was statistically positive for two data sets, statistically negative for two data sets, and stable for three data sets.

Of the 17 data sets available for WCT trend monitoring, eight had no measurable observation error and two were estimated to have only minimal observation error (Table 2). Screw-trap and snorkeling data sets were equally prone to high observation error,

Table 1. Description of available data sets, intrinsic rates of population change (r_{intr} with 90% confidence intervals), and estimated observation error ($\hat{\tau}^2$) for Westslope Cutthroat Trout populations in Idaho.

						r_{int}			
							90% CIs		
Sub-basin	WCT population	Data description	Collection method	Time Span (yrs)	Years of data	Estimate	Lower	Upper	$\hat{\tau}^2$
Coeur d'Alene	Coeur d'Alene River	Main stem data	Snorkeling	39	17	0.027	0.016	0.037	0.00
Coeur d'Alene	St. Joe River	Main stem data	Snorkeling	43	18	0.029	0.019	0.039	0.07
Clearwater	Lochsa River	Main stem and tributary data	Snorkeling	28	26	0.067	0.027	0.108	0.20
Clearwater	Lochsa River	Colt Killed Creek	Screw trap	12	12	0.050	-0.019	0.120	0.00
Clearwater	Lochsa River	Crooked Fork Creek	Screw trap	15	15	0.067	0.020	0.115	0.00
Clearwater	SF Clearwater River	Main stem and tributary data	Snorkeling	28	28	0.055	0.018	0.091	0.00
Clearwater	SF Clearwater River	Crooked River	Screw trap	13	13	0.359	0.214	0.504	0.00
Clearwater	SF Clearwater River	Red River	Screw trap	13	13	0.239	0.106	0.372	0.56
Clearwater	Selway River	Main stem and tributary data	Snorkeling	28	25	0.060	0.037	0.082	0.61
Clearwater	Selway River	Main stem data	Angling	36	27	-0.006	-0.018	0.007	0.60
Salmon	Mid-Salmon River	Tributary data	Snorkeling	24	23	-0.055	-0.105	-0.004	0.24
Salmon	South Fork Salmon River	Main stem and tributary data	Snorkeling	16	16	-0.113	-0.206	-0.019	0.00
Salmon	South Fork Salmon River	Knox bridge	Screw trap	14	14	-0.036	-0.092	0.020	0.00
Salmon	Middle Fork Salmon River	Main stem data	Snorkeling	28	18	0.096	0.073	0.118	0.00
Salmon	Middle Fork Salmon River	Tributary data	Snorkeling	28	27	0.149	0.128	0.170	0.20
Salmon	Middle Fork Salmon River	Main stem data	Angling	54	20	0.0002	-0.014	0.014	0.10
Salmon	Upper Salmon River	Sawtooth hatchery	Screw trap	13	13	0.020	-0.147	0.186	0.95

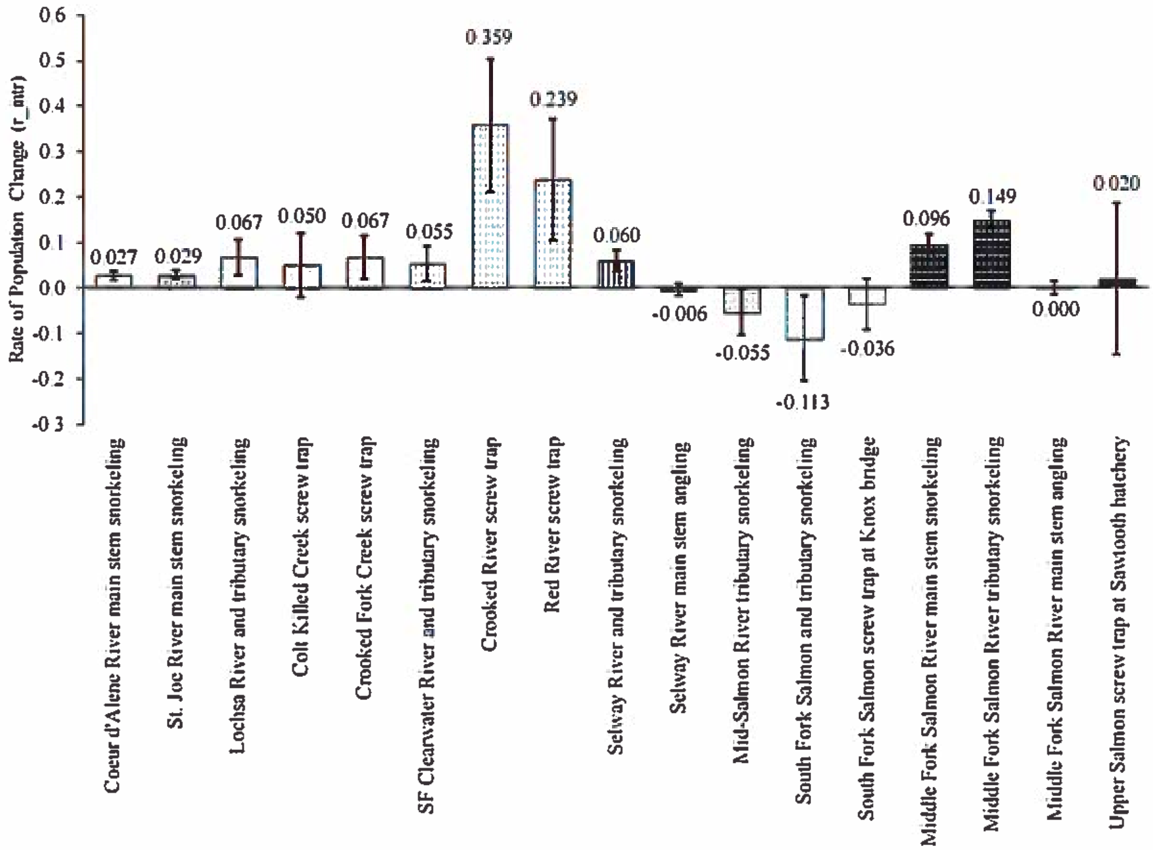

Figure 1. Intrinsic rates of population change (r_{intr}) for Westslope Cutthroat Trout populations in Idaho. Error bars represent ± 90% confidence intervals.

Table 2. Best models for each Westslope Cutthroat Trout trend monitoring data set relating trout abundance to bioclimatic variables. Akaike information criterion weights (w_i) indicate the probability that the given model is 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.

WCT population	Data description	Collection method	Variables	w_i	r^2	F-value	P-value
Coeur d'Alene River	Main stem data	Snorkeling	PDSI	0.27	0.01	0.14	0.71
St. Joe River	Main stem data	Snorkeling	PDSI	0.29	0.05	0.75	0.40
Lochsa River	Main stem and tributary data	Snorkeling	Redds	0.22	0.08	2.14	0.16
Lochsa River	Colt Killed Creek	Screw trap	Redds	0.25	0.25	3.29	0.10
Lochsa River	Crooked Fork Creek	Screw trap	Redds	0.41	0.27	4.72	0.05
SF Clearwater River	Main stem and tributary data	Snorkeling	Redds	0.19	0.07	1.96	0.17
SF Clearwater River	Crooked River	Screw trap	Redds	0.27	0.25	3.57	0.09
SF Clearwater River	Red River	Screw trap	Redds	0.22	0.18	2.36	0.15
Selway River	Main stem and tributary data	Snorkeling	AirT	0.29	0.25	7.47	0.01
Selway River	Main stem data	Angling	AirT + PDSI	0.18	0.14	1.88	0.07
Mid-Salmon River	Tributary data	Snorkeling	Redds	0.23	0.08	1.72	0.20
South Fork Salmon River	Main stem and tributary data	Snorkeling	PDSI	0.27	0.16	2.65	0.13
South Fork Salmon River	Knox bridge[a]	Screw trap	Redds + Discharge	0.30	0.33	4.14	0.09
Middle Fork Salmon River	Main stem data[a]	Snorkeling	AirT + Redds	0.42	0.42	7.13	0.03
Middle Fork Salmon River	Tributary data[a]	Snorkeling	AirT + PDSI + Redds	0.50	0.68	19.35	<0.001
Middle Fork Salmon River	Main stem data[a]	Angling	AirT	0.26	0.01	0.21	0.65
Upper Salmon River	Sawtooth hatchery	Screw trap	Discharge	0.35	0.34	4.19	0.07

[a] Streamflow data incomplete, therefore discharge was not tested in this model

with approximately 50% of those data sets estimated to have high observation error. Both angling data sets appeared to have no observation error.

For all bioclimatic variables except Chinook Salmon redds, correlation coefficients with WCT abundance were generally higher for one-year lags; for Chinook Salmon redds, correlation coefficients were generally higher for two-year time lags. Using these time lags in multiple regression models, the bioclimatic variables generally explained a statistically significant but low amount of variation in WCT abundance, with an average of 21% of the variation in WCT abundance being explained by the bioclimatic variables. Chinook Salmon redd counts was the most explanatory variable for 56% of the data sets, followed by air temperature (28%), and Palmer Drought Severity Index (17%). High redd counts had a positive effect and low air temperature and low PDSI had negative effects on WCT abundance.

Discussion

In our study, there were five times more statistically significant positive growth rate estimates than statistically significant negative estimates, and several more stable growth rates, suggesting that WCT 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 WCT populations is the South Fork Salmon River and nearby tributaries to the middle reaches of the main-stem Salmon River. Causative mechanisms are difficult to elucidate at such broad scales using mensurative (rather than manipulative) study designs, but our results suggest that at least some of the positive growth in WCT populations in Idaho can be attributed to increases in wild Chinook Salmon returning from the Pacific Ocean. Salmon deliver marine-derived nutrients to the majority of WCT populations in Idaho (Cederholm et al. 1999) and marine-derived nutrients are particularly important for primary production in unproductive geologies like Idaho (Sanderson et al. 2008). Nevertheless, most bioclimatic variables were weakly correlated to WCT abundance, suggesting that environmental factors other than the ones we included in our study may have been influencing WCT abundance. Copeland and Meyer (2011) evaluated the relationships between bioclimatic conditions and fish density for six salmonids in central Idaho and also found weak relationships for WCT. 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 that may be contributing to positive WCT population growth in Idaho include improvements in land management practices, and catch-and-release regulations (Quinn 1996; Mallet 2013).

We assumed that the trend data sets available for each WCT 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 WCT population. Nonetheless, for the WCT populations where more than one data set was available, trends were generally in synchrony within the population. In fact, there were no examples of trends being statistically positive and statistically negative for two different data sets within the same WCT population. Furthermore, many of the trend data sets were initiated to monitor species other than WCT, such as the screw trap and snorkel data sets for the Salmon River and Clearwater River basins. 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 WCT data that were consistently more optimistic than the mean growth rate for the population would have been.

Observation error was high for nearly one-half of the WCT trend data sets we summarized herein. Fortunately, in our study this had little impact on our findings because for six of the seven data sets with high observation error, trends were already 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). Observation error is also a concern because it can inflate estimates of a population's risk of extirpation

(Morris and Doak 2002). We did not estimate risk of extirpation in our study, because we had no estimates of adult population size, which is necessary for such modeling.

The fact that screw trap data sets were as likely to have high observation error as 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 WCT in Idaho. Bull Trout are cryptic, sporadically distributed, highly migratory salmonids (Pratt 1992). In contrast, WCT 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 appear to index WCT abundance better than for Bull Trout, whereas screw traps appear to better index Bull Trout abundance.

We suspected that any effect the bioclimatic variables would have on WCT abundance might be delayed by one year. Such a delayed response might indicate that the bioclimatic variables were influencing WCT recruitment (Copeland and Meyer 2011), and since for most data sets we either discarded small fish (snorkel data sets) or small fish were not vulnerable to the data collection method (angling data sets), a one-year delayed response would be expected. Although the bioclimatic variables were only weakly related to WCT abundance, in all instances these relationships were indeed strongest with a one-year lag (except the two-year lag for Chinook Salmon redds).

Our study highlights the lack of WCT trend monitoring data for many areas in Idaho, particularly the Moyie River, North Fork Clearwater River, Lemhi River, and Pahsimeroi River drainages. Until WCT trend data are available for these drainages, assessment of WCT status in Idaho will be incomplete. Nevertheless, the results of our study suggest that WCT in Idaho currently appear to be stable or increasing in abundance in most areas.

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