

# PROJECT 3: WILD TROUT EVALUATIONS 

## Annual Progress Report

Project 3 - Wild Trout investigations
Subproject \#1: Myy Brook Trout Field Evaluations 2022
Subproject \#2: Trends in the occupancy and abundance of Redband Trout and nonnative trout in the Wood River Basin

Subproject \#3: Factors affecting angler catch and harvest of trout in Idaho alpine lakes

Subproject \#4: Effects of elevated water temperatures on trout angler catch rates and catch-and-release mortality

Subproject \#5: Trends in the occupancy and abundance of Bonneville Cutthroat Trout and nonnative trout in the Bear River basin of Idaho

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## ANNUAL PROGRESS REPORT

# SUBPROJECT \#1: MyY BROOK TROUT FIELD EVALUATIONS 2022 

State of: Idaho
Project No.: $\underline{3} \quad$ Title: Wild Trout Evaluations

Subproject \#1: $\quad$ Myy Brook Trout Field Evaluations $\underline{2022}$


#### Abstract

Non-native Brook Trout Salvelinus fontinalis were introduced throughout western North America in the early 1900s, resulting in widespread self-sustaining non-native populations that are difficult to eradicate and often threaten native salmonid populations. A novel approach to eradicating undesirable Brook Trout populations is using YY male (Myy) Brook Trout. Myy Brook Trout are created in the hatchery by feminizing XY males and crossing them with normal XY males. When Myy Brook Trout reproduce successfully with wild females, all progeny are males. This can potentially be used to shift the sex ratio of the wild population toward males to reach a point where no females remain in the population to reproduce, thus eliminating the population. In 2022, we stocked fingerling (mean $=134 \mathrm{~mm}$; range $=82-176 \mathrm{~mm}$ ) My Brook Trout in four streams and four lakes, and catchable (mean = 240 mm ; range $=154-310 \mathrm{~mm}$ ) Myy Brook Trout in one stream and two lakes to attempt to eradicate wild Brook Trout in these study systems. These study waters are stocked annually beginning as early as 2015. Prior to stocking, we suppressed wild Brook Trout via mechanical removal in two streams and two lakes to potentially increase survival of stocked $M_{Y y}$ Brook Trout, and therefore decrease the time to eradication. Suppression via mechanical removal in 2022 was $64 \%$ in Dry Creek, $48 \%$ in Pikes Fork Creek, and $45 \%$ in Seafoam Lake \#4. Male sex ratio is as high as $86 \%$ in Dry Creek, where fingerling stocking occurs and suppression is annual. In other study streams, and in alpine lakes, there is little evidence of a shift in sex ratio. This long-term study is scheduled to be completed in 2026, but at this time, it appears that eradication of wild Brook Trout can only occur in a reasonable timeframe in streams (not alpine lakes) stocked with fingerlings (not catchables) and with annual suppression of wild fish.


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## INTRODUCTION

Brook Trout Salvelinus fontinalis were originally introduced outside their native range into waters of the western United States as early as 1872 by the California Fish Commission (MacCrimmon and Campbell 1969), and they continue to colonize new habitats in western North America (Benjamin et al. 2007). Brook Trout have contributed to declines in native fish abundance through hybridization, competition, and predation (Rahel 2000). Thus, fisheries managers have attempted to suppress or eliminate Brook Trout populations outside of their native range (reviewed in Dunham et al. 2004). There are several methods which fisheries managers use to eradicate non-native fish. Managers have used piscicides with some success (Gresswell 1991; Lee 2001; Lentsch et al. 2001; Hepworth et al. 2002), but piscicides may result in collateral damage to native fish populations (Britton et al. 2011), and other aquatic fauna (e.g., Hamilton et al. 2009; Billman et al. 2012). Multiple-pass electrofishing has been used to physically remove Brook Trout from streams (e.g., Thompson and Rahel 1996; Meyer et al. 2006; Shepard et al. 2014), but it has been questioned whether stream electrofishing removal alone can cause meaningful progress in Brook Trout eradication at the landscape scale (Meyer et al. 2006; Schill et al. 2017). Sterile predatory fish have been introduced in alpine lakes, but successful eradication of Brook Trout occurred in less than one-half of the lakes where the strategy was used (Koenig et al. 2015). The mixed success of these approaches suggests a need for additional methods for non-native fish eradication.

An alternative method, suggested decades ago for eradicating undesirable fish populations, is shifting the population sex ratio toward all males (Hamilton 1967). In this scenario, shifting the sex ratio over time could be accomplished by annual introductions of hatchery produced male fish with an YY genotype ( $\mathrm{M}_{\mathrm{YY}}$ ), eventually resulting in population eradication by eliminating females (Gutierrez and Teem 2006; Teem and Gutierrez 2010). To create a Myr brood stock, XY males can be feminized by exposing them to estrogen (Teem and Gutierrez 2010). After rearing to maturity, the resulting XY neo-females can be crossed with normal XY males and, on average, one-quarter of the progeny will be MYY. To develop a functional broodstock, half of the $M_{Y Y}$ can then be feminized by exposure to estrogen at an early age to create egg-bearing $\mathrm{Y} Y$ fish ( $F_{Y Y}$ ). Subsequent progeny of $F_{Y Y}$ and $M_{Y Y}$ crosses are all $M_{Y Y}$. These $M_{Y Y}$ progeny can then be stocked into wild fish populations in an effort to drive the sex ratio of the wild population to $100 \%$ males (Parshad 2011). Although YY fish culture is occasionally used in commercial hatcheries (e.g., Mair et al. 1997; Liu et al. 2013), a stocking program utilizing YY fish to eradicate a non-native fish species has not been tested in the wild (Wedekind 2012; Wedekind 2018).

In wild Brook Trout populations, sex ratios would only shift under such a stocking program if the M M Brook Trout survive and successfully reproduce after stocking. A pilot study estimated an average of $16 \%$ of $M_{Y y}$ Brook Trout survived for three months and successfully reproduced with wild females after they were stocked in four Idaho streams (Kennedy et al. 2018a). Hatchery trout encounter many challenges upon release into natural environments, and often exhibit low survival, especially in streams (e.g., Miller 1952; Bettinger and Bettoli 2002; High and Meyer 2009). Low survival of hatchery trout in streams is largely attributed to the stress associated with adjusting to natural stream flows and competition with resident fish (Schuck 1948; Miller 1954; Miller 1958; Hochachka and Sinclair 1962). Though rarely evaluated, past studies suggested that manual removal (hereafter suppression) of wild fish prior to stocking hatchery fish could markedly improve survival of the stocked hatchery trout (Miller 1958; Horner 1978). In addition, modelling by Schill et al. (2017) suggested that combining MYy stocking with suppression of wild fish may decrease the time-to-eradication in wild Brook Trout populations.

Size-at-release can also influence survival of hatchery-reared fish. Adult hatchery trout of catchable-size (avg. 220 mm ), hereafter referred to as catchables, generally return to creel at a much higher rate than juvenile hatchery trout, hereafter referred to as fingerlings (Wiley et al. 1993; Dillon and Jarcik 1994). The greater post-release performance of catchables in fisheries may result from larger energy reserves, reduced vulnerability to post-release predation, and reduced competition with wild fish. Catchables are also immediately vulnerable to anglers upon release, whereas fingerlings must survive and grow for months or perhaps more than a year before they grow to be vulnerable to anglers. Most work comparing survival between catchables and fingerlings has focused on overall return-to-creel, but the difference in short-term survival between fingerlings and catchables is unknown. The difference in survival between fingerlings and catchables is of particular interest in the case of $M_{Y Y}$ fish, because the objective is to maximize the abundance of mature $M_{Y Y}$ on the spawning grounds. However, it is unclear whether this is best achieved by stocking higher numbers of fingerling $\mathrm{M}_{\mathrm{YY}}$ or lower numbers of catchable $\mathrm{M}_{\mathrm{YY}}$ fish.

The Idaho Department of Fish and Game (IDFG) established a YY Brook Trout broodstock in 2012 that annually produces 20,000-30,000 Myy Brook Trout for eventual stocking into the wild (Schill et al. 2016). Prior to large-scale stocking, survival, and reproductive success of catchable Myy Brook Trout in the wild were evaluated in Kennedy et al. (2018a), and this study indicated that Myy Brook Trout could successfully survive and reproduce in the wild. Recent modelling suggests that annual stocking of Myy Brook Trout into streams and alpine lakes can result in eradication of the wild population within 10 years if Myy Brook Trout are stocked at a rate of 50\% of the wild Brook Trout abundance (Schill et al. 2017). In model simulations, eradication occurred faster as suppression of the wild population increased. However, these models are theoretical and need to be tested on wild Brook Trout populations to validate predictions.

## OBJECTIVE

1. Evaluate various $M_{Y Y}$ stocking and wild trout suppression strategies in both streams and lakes to identify where the MYy approach is most likely to result in complete eradication of wild Brook Trout populations in Idaho.

## METHODS

The IDFG experimentally feminized male Brook Trout fry with estrogen (in the form of $17 \beta$ estradiol) to create an adult broodstock of YY Brook Trout. For complete details of YY broodstock production, see Schill et al. (2016). Production and rearing of MyY Brook Trout occurred at the IDFG Mackay Fish Hatchery prior to 2019 and the IDFG Hayspur Fish Hatchery post 2019. Offspring were produced by crossing $\mathrm{F}_{Y Y}$ and $\mathrm{M}_{Y Y}$ broodstock, and fish were reared to fingerling and catchable sizes at the hatchery in outdoor concrete raceways in $10-12^{\circ} \mathrm{C}$ single-use spring water until the time of release. All study fish are adipose fin clipped so they can be differentiated from wild fish post-stocking. For this study, fingerlings average about 125 mm TL and are stocked 8 months post-hatching, while catchables average about 250 mm TL and are stocked 20 months post-hatch.

Study streams and lakes were selected with self-sustaining Brook Trout populations comprising greater than $80 \%$ of the wild fish species composition. Each study stream treatment reach exhibited a total stream length of less than 10 km from the upstream distribution of Brook Trout to a downstream passage barrier, which provided isolation from potential upstream
immigrating female Brook Trout from lower untreated reaches of the stream (Figure 1; Table 1). Lakes were also chosen based on the presence of passage barriers, which would prevent upstream immigration of Brook Trout (Figure 1; Table 2). Lakes varied in size from 2.5 to 15.8 hectares. During 2015-2017, streams and lakes were assigned to one of two treatment levels (Suppression and non-suppression) to evaluate fingerling and catchable Myr Brook Trout stocking. At two of the streams and two of the lakes, we manually suppressed the wild Brook Trout population on an annual basis to improve survival and spawning success of stocked fish. Suppression was achieved by the removal of wild Brook Trout using backpack electrofishing in streams, and gill nets in conjunction with boat/raft electrofishing in lakes. Non-suppression streams and lakes were stocked with Myy Brook Trout without the suppression of their wild counterparts. Two control streams and two control lakes were also selected to monitor the stochastic changes in wild Brook Trout populations in central Idaho. All treatment streams and lakes will be stocked annually, for a minimum of seven years, unless the population collapses and intensive sampling identifies that no female ( $F_{x x}$ ) Brook Trout remain. Sex ratios in each Brook Trout population will be assessed approximately every three years until the wild population is considered eradicated.

The first field evaluations of Myy Brook Trout in streams began in 2016 with additional streams included in 2017 (Table 1). Dry, East Fork Clear, and Tripod creeks have been under evaluation since 2016, with Pikes Fork and East Threemile creeks added to the evaluation in 2017. For a more complete discussion of previous study years for streams, see Kennedy et al. (2018c) and Roth et al. (2020). Field evaluations of Myy Brook Trout in Duck, Lloyds, Snowslide \#4, and Upper Hazard lakes began in 2015, with Black and Rainbow lakes added in 2016; and Martin Lake and Seafoam Lake \#4 added in 2017. For a more complete discussion of previous study years involving lake evaluations, see Kennedy et al. (2018b) and Roth et al. (2020). Due to the three-year cycle associated with sampling, surveys were conducted at all study lakes and the two study lakes that received annual suppression (i.e., Martin Lake and Seafoam Lake \#4) in 2022. Non-suppression lakes were not sampled in 2022. However, all lakes and streams were stocked with Myy Brook Trout in 2022. Full lake sampling is scheduled for 2024, and stream sampling for 2025.

## Stream surveys

Suppression of wild Brook Trout was conducted in Dry and Pikes Fork creeks after snowmelt subsided (to maximize electrofishing capture efficiency) but prior to annual Myy stocking. Before suppression, approximately 20 Brook Trout ( $\geq 100 \mathrm{~mm}$ ) were marked with an upper caudal clip at each $1 / 2 \mathrm{~km}$ of the stream, 1 day prior to suppression so recaptured fish could be used to estimate abundance and capture efficiency. Single-pass electrofishing was conducted to capture fish over the entire study reach (range 3.9-9.1 km). Electrofishing crews consisted of 2-3 people (depending on stream flow) with backpack electrofishers, and 1-3 people with nets and buckets ( 19 L ). We used a pulsed-DC waveform typically operated at $60 \mathrm{~Hz}, 300-900 \mathrm{~V}$, and a $25 \%$ duty cycle. During suppression, persons with backpack electrofishers covered all available habitats, moving methodically upstream in tandem. All wild Brook Trout captured were euthanized. Data collected from captured fish included: species, total length (TL; mm), and identification of marks (fin and jaw clips). Salmonids other than Brook Trout comprised less than $31 \%$ of the total catch among all study streams, were released unharmed, and were not included in further analyses.

At non-suppression streams (East Threemile, East Fork Clear, and Tripod creeks), wild Brook Trout abundance was estimated using multiple-pass depletion electrofishing during September at ten survey sites within each treatment reach. It should be noted that these surveys
occurred after annual stocking. Survey sites were selected systematically from each stream by dividing the total treatment reach into ten equal sections. The downstream end of each section was then identified as the downstream boundary of a survey site. Block nets were installed on the upper and lower boundary of each survey site. Survey site length (approximately 50 m ) was adjusted slightly as needed to utilize natural stream channel constraints. Electrofishing crew size and output settings were similar to those used during suppression efforts. If no salmonids were captured on the first pass, no more passes were made. If Brook Trout were captured on the first pass, a minimum of three electrofishing passes were conducted within the study reach, such that the last pass resulted in $\leq 50 \%$ of the wild Brook Trout captured as the prior pass for at least two consecutive passes. Captured fish were anesthetized, measured for length and inspected for marks as above, then placed in a bath of fresh water to recover from the anesthetic before being released back into the stream, but outside the survey site. Abundance will be estimated annually at suppression streams, and approximately every three years after the initial stocking in nonsuppression streams.

Prior to the initial $M_{y y}$ stocking in each study stream, we collected tissue samples from wild Brook Trout fry ( $<100 \mathrm{~mm}$ ) to estimate genetic sex ratios and parentage of the Brook Trout populations. Sex-biased survival was anticipated in mature Brook Trout due to the stresses associated with spawning and size-selective harvest by anglers (McFadden 1961). Fry were assumed exempt from these biases so equal sex ratios for males and females were anticipated (Fisher and Bennett 1999). Tissue samples were clipped from the caudal fin and preserved on Whatman ${ }^{\text {TM }} 3 \mathrm{MM}$ chromatography paper (Thermo Fisher Scientific, Inc., Pittsburgh, Pennsylvania). We sought a goal of 100 tissue samples from Brook Trout fry from each stream during suppression or abundance estimate surveys to characterize the sex ratio of each wild population. Tissue samples were then collected from Brook Trout fry in each stream with a goal of 100 samples every three years for non-suppression streams and every year from suppression streams to evaluate the change in sex ratio through time and quantify to amount of successful $M_{Y Y}$ offspring production. Fry collections occurred at multiple locations over the entire treatment reach to minimize family effects (Whiteley et al. 2012).

To evaluate presumed fish passage barriers, we collected Brook Trout via electrofishing downstream from the identified passage barrier. Passage barriers were either natural or manmade structures depending on the waterbody. All salmonids captured were anaesthetized and measured for TL as described above and were given a maxillary clip on both sides of the mouth, then released near their point of capture. Over time, any maxillary-clipped fish captured upstream from the assumed passage barrier will help us assess the effectiveness of the barrier and the degree of demographic isolation in study populations.

## Lake surveys

Only the two lakes that receive annual suppression (i.e., Martin Lake and Seafoam Lake \#4) were sampled in 2022. Sampling and suppression were conducted using gill nets and boat or raft electrofishing. Paired gill nets (floating and sinking) were set at three netting locations chosen to maximize catch based on professional experience. All nets were Swedish experimental gillnets ( $36-\mathrm{m}$ long and $1.8-\mathrm{m}$ deep) consisting of either floating or sinking types consisting of nylon mesh, with equal-length panels ordered from smallest to largest of 10-, 12.5-, 18.5-, 25-, 33-, and 38mm bar mesh. All gillnets were set at dusk and retrieved the following morning. Boat electrofishing was conducted in Martin Lake over two nights while raft electrofishing was conducted at Seafoam Lake \#4 over three nights. During electrofishing, Myy Brook Trout were identified based on adipose fin clips. In both systems, all Brook Trout captured during the first night were marked with an upper caudal clip (marking run 1) and then released. During the second night all trout were
marked with a lower caudal clip (marking run 2), noted for recaps from the first marking run and all wild Brook Trout were removed. The third night of electrofishing and gillnetting, all recapped fish were noted and all wild Brook Trout were removed. Data collected from captured fish included: species, TL (mm), and identification of fin and maxillary clips. In Martin Lake, Rainbow Trout Oncorhynchus mykiss comprised over $75 \%$ of captured fish while the only other salmonid present in Seafoam Lake \#4 was one Arctic Grayling Thymallus arcticus. These salmonids were released unharmed back into their respective systems and were not included in further analyses. Additionally, tissue samples from Brook Trout fry were collected in Seafoam Lake \#4 to estimate sex ratios and parentage. Collection of tissue samples was also attempted in Martin Lake, but we were unable to capture any fry. Tissue samples were clipped from the caudal fin and preserved on Whatman ${ }^{\text {TM }} 3 \mathrm{MM}$ chromatography paper (Thermo Fisher Scientific, Inc., Pittsburgh, Pennsylvania).


#### Abstract

Abundance For mark-recapture surveys at each suppression stream, survey data were pooled over the entire study area, then total Brook Trout abundance was estimated using the modified Peterson estimator from the FSA package (Ogle 2020) in statistical package R (R Core Team 2022). Ninety-five percent confidence intervals (Cls) were calculated by calculating the variance of a product and then converting that into a confidence interval (Goodman 1960). To account for differences in capture efficiency among size classes, abundance was estimated separately for the smallest size groups that still allowed for at least three recaptured fish per size group in order to satisfy model assumptions. We assumed there was 1) no mortality of marked fish between marking and recapture passes, and 2) no movement of marked or unmarked fish out of the study reach between marking and recapture passes. In all streams, estimates for both wild and Myy Brook Trout were calculated for all size classes $\geq 100 \mathrm{~mm}$ to describe abundance for the entire study area.


Wild Brook Trout abundance and Cls from depletion electrofishing surveys at each nonsuppression streams were estimated using the removal function from the FSA package (Ogle 2020) in statistical package R ( R Core Team 2022). When no Brook Trout were captured on the second and/or third pass, total catch from the first pass was assumed equivalent to abundance. Abundance estimates were only made for fish $\geq 100 \mathrm{~mm}$ TL to maintain consistency with the markrecapture surveys. Brook Trout abundance was then averaged across all 10 sites to determine the mean abundance per 50 m reach in the study stream. This average linear density was then multiplied by the length of the study reach to estimate total abundance of both wild and Myy Brook Trout in the study reach. Due to equipment error and the small sample size of Brook Trout encountered during the marking run, Brook Trout abundance could not be calculated in Martin Lake. In Seafoam Lake \#4, all marked fish (marking run 1 and 2) were pooled.

## Stocking

Stocking Myy Brook Trout occurred during the month of August for most streams and lakes. However, due to logistical constraints Martin Lake and Seafoam Lake \#4 were stocked in early September. All Myy Brook Trout at each waterbody were stocked in a single event, so stocking densities described here are annual total stocking densities. Fingerling-sized trout are rarely stocked in Idaho streams due to their low survival and return-to-creel (Schuck 1948). Catchables are commonly stocked in Idaho streams, though the selected study streams are considerably smaller than most rivers IDFG stocks with trout. Silver Creek (tributary to the Middle Fork Payette River) was the most comparable in size to study streams described here, that was regularly stocked with hatchery trout by IDFG. Stocking densities ranged from 96-128 trout/km at

Silver Creek. Therefore, we chose a priori stocking density of catchable Myy Brook Trout at 125 fish/km.

Fingerling stocking density was initially set at four times the stocking rate of catchables (i.e., 500 fingerlings $/ \mathrm{km}$ ) based on the ratio of juvenile fish to adult fish suggested in McFadden (1961; i.e., adult Brook Trout comprise $20 \%$ of the population). However, initial scouting trips to study streams identified major disparities in stream widths, to the extent that 500 fingerlings $/ \mathrm{km}$ may have been detrimental to survival of stocked fish at very narrow streams. Therefore, at narrow streams (i.e., East Fork Clear and Tripod creeks; Table 1; Figure 1), we reduced stocking densities to 250 fingerlings/km.

Estimates of wild Brook Trout abundance at each individual waterbody were used to adjust stocking densities after the first year of stocking and were re-evaluated in 2018. Because prior research has suggested that $50 \%$ fingerling stocking rates (relative to wild Brook Trout abundance) would result in eradication times of less than 10 years in streams (Schill et al. 2017), we adjusted fingerling stocking rates to $50 \%$ of the estimated total wild Brook Trout abundance for each stream. To maintain the $4: 1$ fingerling to catchable stocking ratio (which also approximately balanced the biomass of fish being stocked in each stream), the number of catchables stocked was adjusted to $50 \%$ of the total wild population estimate, divided by four. Resulting stocking densities for each stream were then held constant for the duration of the study to reduce bias when evaluating the rate of change in sex ratios.

Stocking rates in alpine lakes were initially set based on the typical stocking rate of fry in alpine lakes used in Idaho of 500 fry/ha. However, because fry are slightly smaller than stocked fingerling $M_{Y Y}$ Brook Trout we slightly reduced the stocking density to 438 fingerling/ha. To standardize the biomass being stocked, the stocking rate of catchables was adjusted to $1 / 5$ the stocking rate of fingerlings (i.e., $88 / \mathrm{ha}$ ) because preliminary testing indicated that fingerlings were approximately $1 / 5$ the weight of catchables. Additionally, this stocking rate is supported by the fact that fingerling Myy Brook Trout are typically immature at the time of stocking, catchables are typically mature, and wild Brook Trout populations typically exhibit a $4: 1$ ratio of mature to immature fish (McFadden 1961; Meyer et al. 2006). Therefore, a catchable stocking rate of $1 / 5$ the fingerling stocking rate makes sense from a biological standpoint as well as a biomass standpoint. Because population abundance estimates were never obtained for alpine lakes, these stocking rates will be used for the duration of the study.

Stocking fingerlings and catchables into streams near roads was usually completed using 19-L buckets from a 1 -ton or $3 / 4$-ton hatchery tanker truck. Fish were counted into buckets with hatchery water, then carried to the river and released into a pool or other low-velocity stream section. At suppression streams, mark-recapture abundance estimates of wild fish from every $1 / 2$ km were used to inform $\mathrm{M}_{\mathrm{Yy}}$ stocking distribution in the stream. Assuming stocked hatchery fish generally move downstream (High and Meyer 2009), Myy Brook Trout were distributed at a slightly higher density at the upstream extremities of each study reach and in reaches where electrofishing catch identified high abundances of wild fish. Hatchery trout generally exhibit minimal movement within streams (Heimer et al. 1985; High and Meyer 2009), so we dispersed $M_{Y y}$ fish longitudinally throughout the entire stream. To maximize the encounter rate of hatchery Myy males with spawning females, we backpacked fish into headwater reaches or other roadless areas. For stocking in roadless sections, a contractor-grade garbage bag was placed inside of 19-L buckets loaded into backpacks was filled with approximately 8-L of hatchery truck water $\left(\sim 12^{\circ} \mathrm{C}\right)$. Then, fish were loaded into the garbage bag. An air stone and hose (connected to a Quiet-Bubbles® air pump) were inserted into the opening of the garbage bag, and then the bag was sealed. Fish loading densities and water displacement were calculated following Piper et al.
(1982). To maintain fish health during transport, target fish loading densities were less than 3,392 g of fish/L. Depending on ambient temperatures, water temperature and dissolved oxygen were suitable for Brook Trout health for $\leq 45$ minutes. At some locations, fish were transported in coolers filled with hatchery truck water on ATVs oxygenated with a compressed oxygen cylinder with dual ceramic plate fine pore oxygen diffusers (Point Four®). Loading densities and water quality monitoring in coolers followed methods described above.

Fingerling and catchable Myy Brook Trout were stocked into alpine lakes primarily by helicopter and bucket (90-100-gal capacity SEI Industries Bambi Bucket®) because they were too large to be stocked with fixed-wing aircraft without significant mortality. Precise preliminary estimates of average fish weight (number of fish/lb) were helpful for the necessary helicopter load calculations. Fish loading densities and water displacement were calculated following Piper et al. (1982). To maximize fish health during transport, target fish loading densities were less than 1.0 lb fish/gal. Load calculations were estimated for the number of fish and water weight needed for each lake. The number of flights to each lake was determined by the helicopter's (Bell 206L-3) safe load capacity ( 600 lbs .) and to keep fish load densities under 1.0 lb of fish/gal, and total flight distances were planned to deliver the required number of fish and water weight. The number of fish for each flight (estimated via pound counts) was transferred from a hatchery tanker truck to a 100-gal Rubbermaid $®$ stock tank where dissolved oxygen was rigorously maintained at 10 ppm (Piper et al. 1982) using a YSI EcoSense® 200-4 dissolved oxygen probe. At each treatment lake, the pilot submerged the bucket and remotely removed the bottom seals of the bucket to allow the fish to swim free without dropping them. The pilot then filled the stocking bucket with lake water and returned to the helipad. Fish were quickly netted from the stock tanks into the helicopter bucket for transfer to the next lake. For safety purposes, coordinates were programmed into the helicopter GPS and the pilot navigated to each study lake so that no fisheries personnel were onboard. Because Martin Lake and Seafoam Lake \#4 have road access, fish were stocked in these lakes directly from the hatchery truck. Lengths and weights were measured from a subsample ( $n$ $=100$ ) of fingerling and catchable MYY Brook Trout immediately prior to loading the helicopter barrel or directly stocking from the truck.

## Genetic sex ratios and reproductive success

During scheduled sampling in each stream and both suppression lakes, approximately 100 tissue samples were collected from wild Brook Trout fry ( $<100 \mathrm{~mm}$ ) and wild Brook Trout adults ( $\geq 100 \mathrm{~mm}$ ) during July-September to estimate sex ratios and reproductive success. Tissue samples were clipped from the caudal fin and preserved on Whatman ${ }^{\text {TM }} 3 \mathrm{MM}$ chromatography paper (Thermo Fisher Scientific, Inc., Pittsburgh, Pennsylvania).

## Sex ratio monitoring

Samples were screened by the IDFG Eagle Genetics Lab using two genetic markers that differentiate sex in Brook Trout: SexY_Brook1 (Schill et al. 2016) and the master sex-determining gene sdY (Yano et al. 2013). These two markers were screened in a multiplex PCR reaction along with an autosomal microsatellite marker (Sco102) to act as an internal control. The forward primers of each marker were labeled with the carboxyfluorescein (FAM) fluorophore. Thermal cycling PCR reactions were performed in a $5 \mu \mathrm{~L}$ volume consisting of $0.50 \mu \mathrm{~L}$ of primer mix, 2.50 $\mu \mathrm{L}$ of Qiagen Master Mix (cat. 206143), $1.00 \mu \mathrm{~L} \mathrm{dH} 20$, and $1.00 \mu \mathrm{~L}$ template DNA (unknown concentration). Thermal cycling conditions were $95^{\circ} \mathrm{C}$ for 15 min followed by 25 cycles of $94^{\circ} \mathrm{C}$ for $30 \mathrm{~s}, 60^{\circ} \mathrm{C}$ for $1 \mathrm{~min} 30 \mathrm{~s}, 72^{\circ} \mathrm{C}$ for 60 s , and a final extension of $60^{\circ} \mathrm{C}$ for 30 min .

Amplification products were electrophoresed on a 3730 genetic fragment analyzer. Genetic sex was determined using the following rules: individuals that amplified at Sco102 (peak height $=\sim 131-135$ base pairs; b.p.) and both SexY_Brook1 (peak height $=\sim 161$ b.p.) and UsdYMod (peak height = ~222 b.p.) were scored as "males." Samples that amplified at Sco102 but not at SexY_Brook1 and UsdYMod were scored as "females." Individuals that failed to amplify at Sco102 were not scored.

The accuracy of this multiplex marker to differentiate sex in brook Trout was previously validated by screening them on samples of known genetic sex (Schill et al. 2016). Gonadal tissue from 25 individuals of each sex from each study stream, whose phenotypic sex was identified in the field by dissection, was tested to validate the sex marker described above. Sex assignments from tissue samples were compared with the phenotype determined from dissections. We calculated $95 \%$ Cls around the estimated male proportions, following Fleiss (1981).

## Genetic assignment evaluation

A second method to evaluate reproductive success of Myy Brook Trout involves the use of genetic assignment (GA) tests. Genetic assignment refers to a variety of genetic methods that ascertain population membership of individuals or groups of individuals (Manel et al. 2005). Under a GA approach, a sample is required from putative progeny and parents. This methodology is best used in scenarios where it is impossible (e.g., due to cost and time limitations) to genetically sample all $M_{Y Y}$ Brook Trout individually prior to release and when study designs require stocking thousands of Myy Brook Trout into large lakes or rivers.
$M_{Y y}$ Brook Trout offspring were identified with the program Structure (Pritchard et al. 2000; Kennedy et al. 2018a). Structure uses an admixture model that estimates a membership coefficient $(Q)$, which represents the portion of an individual's genotype that originated from a defined number of populations or genetic clusters (in the current study, two). This was accomplished prior to stocking Myy Brook Trout by genetically screening samples collected from both the $M_{Y y}$ population used for stocking and from the receiving wild population fish. The expectation was that progeny from $M_{Y Y}$ adults and wild adults had approximately equal probability of membership to each population $(Q=0.5)$.

Fry sampled during 2022 from all study streams, Martin Lake, and Seafoam Lake \#4 for sex ratio analysis were subjected to GA analysis to describe the origin of sampled fish as either progeny of wild or Myy Brook Trout. Determining the origin of the sampled fry will allow us to describe relative spawning success of $M_{Y y}$ Brook Trout and the proportion of the offspring in the system produced by Myy fish.

## RESULTS

## Stream surveys

At Dry Creek, 4,636 Brook Trout $\geq 100 \mathrm{~mm}$ were captured, of which 3,624 ( $78 \%$ ) were $\mathrm{M}_{\mathrm{YY}}$ Brook Trout and 1,012 (22\%) were wild Brook Trout, the latter being removed from the system. Myy abundance was estimated to be 5,676 fish ( $95 \% \mathrm{Cl}=4,724-6,628$ ), and wild Brook Trout abundance was estimated to be 1,570 fish ( $95 \% \mathrm{CI}=1,306-1,833$; Table 3). Suppression of wild Brook Trout was estimated to be $64 \%$. Wild Brook Trout TLs $\geq 100 \mathrm{~mm}$ averaged 166 mm (maximum $=283 \mathrm{~mm}$ ), while $\mathrm{M}_{\mathrm{Yy}}$ Brook Trout lengths $\geq 100 \mathrm{~mm}$ averaged 197 mm (maximum $=$

313 mm ; Figure 2). No fish with maxillary clips were observed, suggesting the passage barrier is effective at preventing recolonization at Dry Creek. An additional 77 Brook Trout which included 57 stocked $\mathrm{M}_{\mathrm{Yy}}$ and 20 wild Brook Trout were maxillary clipped (mean $=230 \mathrm{~mm}$; maximum = 295 mm ) below the downstream barrier of the study reach to continue barrier evaluations in future years. In addition to Brook Trout, 380 Yellowstone Cutthroat Trout Oncorhynchus clarkii bouvieri were also captured.

At Pikes Fork Creek, 2,183 Brook Trout $\geq 100 \mathrm{~mm}$ were captured, all of which were wild Brook Trout and were removed from the system. The abundance of wild Brook Trout $\geq 100 \mathrm{~mm}$ was estimated to be 4,221 ( $95 \% \mathrm{Cl}=3,605-4,837$; Table 3) and no Myy Brook Trout were captured. Suppression of wild Brook Trout in Pikes Fork Creek was estimated to be $48 \%$. Lengths of wild Brook Trout $\geq 100 \mathrm{~mm}$ averaged 153 mm (maximum = 286 mm ; Figure 2). No fish with maxillary clips were detected above the barrier, and 101 new wild Brook Trout (mean $=140 \mathrm{~mm}$; maximum $=198 \mathrm{~mm}$ ) were maxillary clipped below the barrier for future barrier evaluation. Additionally, 763 Rainbow Trout O. mykiss were captured in Pikes Fork Creek in 2022.

At East Fork Clear Creek, 144 Brook Trout $\geq 100 \mathrm{~mm}$ were captured from ten $50-\mathrm{m}$ reaches, 10 of which were stocked MYy Brook Trout. Total abundance of Brook Trout $\geq 100 \mathrm{~mm}$ was estimated to be $1,032(95 \% \mathrm{CI}=965-1,100$; Table 3) for wild and $79(95 \% \mathrm{Cl}=58-100)$ for Myy Brook Trout. Total length averaged 120 mm (maximum 205 mm ) for wild fish but was slightly higher for $M_{Y Y}$ fish at 130 mm (maximum 163 mm ; Figure 3). In addition, 4 Rainbow Trout were captured.

At East Threemile Creek, 362 Brook Trout $\geq 100 \mathrm{~mm}$ were captured from ten $50-\mathrm{m}$ reaches, all of which were wild Brook Trout. Total abundance of Brook Trout $\geq 100 \mathrm{~mm}$ was estimated to be $5,417(95 \% \mathrm{CI}=5,337-5,497$; Table 3) for wild. No stocked MyY Brook Trout were captured across all sites. Total length of wild Brook Trout averaged 128 mm (maximum 208 mm ; Figure 3). No additional trout species were captured in East Threemile Creek.

At Tripod Creek, 613 Brook Trout $\geq 100 \mathrm{~mm}$ were captured from ten $50-\mathrm{m}$ reaches, 348 of which were stocked Myy Brook Trout. Total abundance of Brook Trout $\geq 100 \mathrm{~mm}$ was estimated to be $4,892\left(95 \% \mathrm{Cl}=4,735-5,049\right.$; Table 3) for wild and $6,493(95 \% \mathrm{Cl}=6,245-6,740)$ for $\mathrm{M}_{\mathrm{YY}}$ Brook Trout. Length averaged 126 mm (maximum 221 mm ) for wild fish but was slightly higher for Myy fish at 136 mm (maximum 176 mm ; Figure 3). In addition, 77 Rainbow Trout were captured.

## Lake surveys

At Martin Lake, 25 Brook Trout $\geq 100 \mathrm{~mm}$ were captured, 12 ( $48 \%$ ) of which were $\mathrm{M}_{\mathrm{YY}}$ Brook Trout and 13 ( $52 \%$ ) were wild Brook Trout, the latter being removed from the system (Table 3). Due to boat issues in the field and a lack of fish marked ( $n=3$ ) and total Brook Trout captured ( $n=25$ ) an abundance estimate could not be calculated for Martin Lake. No Brook Trout <100 mm were captured during the survey. Lengths of wild Brook Trout $\geq 100 \mathrm{~mm}$ averaged 204 mm (maximum = 259; Figure 4) while $M_{Y Y}$ Brook Trout $\geq 100 \mathrm{~mm}$ averaged 197 mm (maximum $=222$ $\mathrm{mm})$. Additionally, 178 Rainbow Trout were captured.

At Seafoam Lake \#4, 311 Brook Trout $\geq 100 \mathrm{~mm}$ were captured, 182 (59\%) of which were Myy Brook Trout and 129 ( $41 \%$ ) were wild Brook Trout, with 89 wild Brook Trout being removed from the system (Table 3). The estimated abundance of wild Brook Trout $\geq 100 \mathrm{~mm}$ was 197 fish, and Myy Brook Trout abundance was estimated to be 331 fish. The suppression rate of wild Brook Trout in the lake was estimated to be $45 \%$. Length of wild Brook Trout $\geq 100 \mathrm{~mm}$ averaged 247
mm (maximum $=320 \mathrm{~mm}$; Figure 4), while $M_{y y}$ Brook Trout $\geq 100 \mathrm{~mm}$ averaged 230 mm (maximum = 324 mm ). No fish were observed with maxillary clips, suggesting the barrier between Seafoam Lakes \#3 and \#4 is effective. Additionally, one Arctic Grayling Thymallus arcticus was captured.

## Stocking

Fingerling Myy Brook Trout were stocked into Dry Creek, East Fork Clear Creek, Pikes Fork Creek, Tripod Creek, Duck Lake, Lloyds Lake, Martin Lake, and Seafoam Lake \#4 in 2022 (Table 4). Total lengths (mean $=134 \mathrm{~mm}$; range $=82-176 \mathrm{~mm}$ ) and weights (mean $=25 \mathrm{~g}$; range $=3.3-58 \mathrm{~g}$ ) of stocked fish were similar across waterbodies. Catchable Myy Brook Trout were stocked into East Threemile Creek, Black Lake, and Rainbow Lake in 2022 (Table 4). Total lengths (mean $=240 \mathrm{~mm}$; range $=154-310 \mathrm{~mm}$ ) and weights ( mean = 142 g ; range $=32-313 \mathrm{~g}$ ) of stocked fish were similar across waterbodies.

## Genetic sex ratios and reproductive success

Sex ratios of fry (<100 mm) varied from $51 \%$ to $84 \%$ male across study streams with both suppression streams exhibiting the highest ratios of $60 \%$ male in Pikes Fork Creek and $84 \%$ in Dry Creek. In study lakes the sex ratio of fry was $39 \%$ male in Seafoam Lake \#4 and could not be estimated in Martin Lake because no fry were captured (Table 5). Genetic assignment analyses indicated that the proportion of male offspring in the population that were produced by stocked $M_{y y}$ Brook Trout was highest in Dry Creek at 94\% followed by Tripod Creek (31\%), East Fork Clear Creek (18\%), and Pikes Fork Creek (8\%). No Myy offspring were detected in East Threemile Creek or Seafoam Lake \#4. Sex ratios of adult wild Brook Trout ( $\geq 100 \mathrm{~mm}$ ) was highest at $81 \%$ male in Dry Creek and lowest in Pikes Fork Creek at $46 \%$ male for suppression streams. In all three non-suppression streams, the male sex ratio ranged from 45\% to 59\% (Table 5).

## DISCUSSION

The most promising results are occurring at Dry Creek, the one suppression stream stocked with fingerling-sized $M_{y y}$ Brook Trout. Wild Brook Trout abundance declined by 39\% from 2021 to 2022, whereas Myy Brook Trout abundance increased by 21\%. Furthermore, of all study waters, Dry Creek exhibits the highest proportion of MYY Brook Trout captured (78\%) and most skewed sex ratio of $84 \%$ male in fry, of which $94 \%$ were attributed as $\mathrm{M}_{\mathrm{yy}}$ offspring. Considering suppression rates, abundance estimates, and sex ratios from 2022, we estimate that: (1) only 558 wild Brook Trout remained after suppression, 100 of which were likely females; (2) of all male Brook Trout (excluding fry) remaining in the stream, 93.5\% are Myr fish; and (3) of all Brook Trout (excluding fry) remaining in the stream, only $1.6 \%$ are likely females. This pattern of high male sex ratios and Myy offspring is also present in two other streams (Bear and Willow creeks) in nearby drainages which are also currently undergoing manual suppression and stocking with fingerling-sized MyY Brook Trout (D. Schill, Fisheries Management Solutions, unpublished data).

Conversely, these positive results are not echoed in Pikes Fork Creek, the second suppression stream. There has been a $53 \%$ decrease in wild Brook Trout abundance from 2021 to 2022, but that was not coupled with the capture of any MYY Brook Trout, and there was no shift in the male sex ratio or MYy offspring. The most notable difference between these two waters is the stocking regimen where Pikes Fork Creek is stocked with catchable-sized Myy Brook Trout compared to the fingerling stocking occurring in Dry Creek.

Survival of stocked catchable-sized $M_{y y}$ Brook Trout appears to be limited in study streams since inception of the study. East Threemile Creek has only been sampled twice, with 38 $M_{Y y}$ Brook Trout captured (adipose clipped) during the 2019 survey while no Myy Brook Trout were captured in the 2022 survey. One notable difference between years was the date of Myy stocking. In 2019, stocking occurred three weeks prior to the abundance survey while in 2022 stocking occurred after our abundance survey was completed. It is likely that most if not all of the $M_{Y y}$ Brook Trout encountered in 2019 were fish stocked in 2019 and not fish that had overwintered from the 2018 stocking. Additionally, we have only encountered a few Myr fish at Pikes Fork Creek, the second catchable study stream, over the last 5 years. It appears that the vast majority of the stocked catchable-sized $\mathrm{M}_{\mathrm{yy}}$ Brook Trout die during their first winter, as is common with catchable-sized stocked hatchery fish stocked into streams (Day et al. 2021; High and Meyer 2009; Kennedy et al. 2018a; Miller 1952). Although only limited numbers of stocked catchablesized fish have been regularly encountered, these stocked Myy Brook Trout have spawned successfully in these waters, as evidenced by the presence of $\mathrm{M}_{\mathrm{Yy}}$ offspring.

Throughout the study, the sex ratio of Brook Trout fry ( $<100 \mathrm{~mm}$ ) in Pikes Fork Creek has remained relatively stagnant ( $51 \%$ in 2017 to $60 \%$ in 2022). This coupled with the limited observed survival of catchable Myy Brook Trout in the creek, prompted us to switch our stocking regimen from stocking catchable-sized to fingerling-sized Myy Brook Trout effective in 2022. It appears that more pronounced shifts in sex ratios towards males occurs in waters which receive both manual suppression and fingerling MyY Brook Trout stocking (i.e., Dry Creek). The success of this combination will be even more evident if we start to encounter our stocked Myy Brook Trout during annual suppression surveys and more pronounced shifts in the sex ratio in years to come. Sex ratios in fry ( $<100 \mathrm{~mm}$ ) in 2022 at all non-suppression streams (East Fork Clear, East Threemile, and Tripod Creeks) appear similar to baseline sex ratios (Kennedy et al. 2018c) and remain close to $50 \%$ male (range $51-55 \%$ ). Additionally, these waters continue to have limited $M_{Y Y}$ offspring production compared to 2019 (Roth et al. 2020).

At Seafoam Lake \#4, we estimated wild abundance to be three times lower in 2022 compared to 2021. Although the abundance estimate calculation is different between the two sampling years, we were able to calculate an estimate directly from our mark -recapture survey. Despite the differences in estimate calculations, we captured less than half the number of wild Brook Trout in our 2022 survey compared to the 2021 survey indicating a likely reduction in the wild Brook Trout population. Due to boat issues in the field and a lack of fish marked ( $\mathrm{n}=3$ ) and total Brook Trout captured $(\mathrm{n}=25)$ coupled with the lack of sub 100 mm Brook Trout captured at Martin Lake we cannot draw any conclusions on the current Brook Trout abundance or sex ratio.

This year denotes the $6^{\text {th }}$ (East Threemile Creek, Pikes Fork Creek, Martin Lake, and Seafoam Lake \#4) and 7 ${ }^{\text {th }}$ (Dry, East Fork Clear, and Tripod Creeks) year of stocking of a planned 9 - to 10-year study. Preliminary results suggest fingerling Myy Brook Trout are surviving and reproducing in most waters while catchable-sized Myy Brook Trout are reproducing but not surviving their first winter. Additionally, it is evident that sex ratios are shifting towards males in some study waters, notably Dry Creek. However, whether complete eradication can be achieved at any study water remains to be seen.

## RECOMMENDATIONS

1. Continue suppression efforts and stocking in all four study waters within the study design that are designated for annual suppression for the duration of the study.
2. Continue annual stocking of fingerling- or catchable-sized Myy Brook Trout in remaining study waters until the effectiveness of the treatment has been determined using the current stocking numbers.
3. Continue to evaluate sex ratios and genetic assignment analyses approximately every three years to monitor reproductive success of Myy Brook Trout in all study waters.
4. Consider adding 1-2 manual (gill-netting) suppression lakes for field seasons in 2024 and 2025, followed by stocking in August for the next several years to determine if eradication occurs. Location would need to be near McCall for stocking to be possible.

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TABLES

Table 1. Study streams in central Idaho selected for $M_{Y Y}$ Brook Trout evaluations including treatment level, fish size stocked, location (WGS84), and physical stream characteristics.

| Stream Name | Start <br> Year | Treatment | $\begin{gathered} \text { Stocked fish } \\ \text { size } \\ \hline \end{gathered}$ | Reach length (km) | Avg. width (m) | Gradient (\%) | Max. elevation (m) | Latitude | Longitude |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Dry Creek | 2016 | Suppression | Fingerling | 6.5 | 5.2 | 1.5 | 2,377 | 44.1268 | -113.5681 |
| East Threemile Creek | 2017 | Non-suppression | Catchable | 6.5 | 2.7 | 5.3 | 2,320 | 44.3986 | -112.0898 |
| East Fork Clear Creek | 2016 | Non-suppression | Fingerling | 3.9 | 2.1 | 5.7 | 1,827 | 44.4757 | -115.8398 |
| Pikes Fork Creek | 2017 | Suppression | Fingerling ${ }^{1}$ | 7.5 | 3.7 | 3.3 | 1,871 | 43.9832 | -115.5484 |
| Tripod Creek | 2016 | Non-suppression | Fingerling | 9.1 | 1.4 | 1.0 | 1,625 | 44.3178 | -116.1200 |
| Alder Creek | 2016 | Control | n/a | 2.4 | 4.9 | 3.2 | 2,000 | 43.8234 | -113.6074 |
| Beaver Creek | 2016 | Control | n/a | 4.0 | 2.4 | 2.2 | 1,650 | 43.9889 | -115.6071 |

${ }^{1}$ Swtiched from stocking catchables to fingerlings in 2022

Table 2. Study lakes in central Idaho selected for Myy Brook Trout evaluations including treatment levels, fish size stocked, location (WGS84), and physical lake characteristics.

| Lake name | Start Year | Treatment | Stocked fish size | Surface area (ha) | Surface elevation (m) | Latitude | Longitude |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Black Lake | 2016 | Non-suppression | Catchable | 2.60 | 2,149 | 45.2454 | -116.1987 |
| Duck Lake | 2015 | Non-suppression | Fingerling | 4.96 | 2,177 | 45.1146 | -116.1573 |
| Llyods Lake | 2015 | Non-suppression | Fingerling | 2.91 | 2,094 | 45.1929 | -116.1637 |
| Martin Lake | 2017 | Suppression | Fingerling | 2.50 | 2,107 | 44.3033 | -115.2636 |
| Rainbow Lake | 2016 | Non-suppression | Catchable | 8.78 | 2,175 | 45.2541 | -116.1966 |
| Seafoam Lake \#4 | 2017 | Suppression | Fingerling | 2.72 | 2,423 | 44.5077 | -115.1258 |
| Snowslide Lake \#1 | 2015 | Control | n/a | 4.86 | 2,188 | 44.9834 | -115.9343 |
| Upper Hazard Lake | 2015 | Control | n/a | 15.84 | 2,265 | 45.1742 | -116.1350 |

Table 3. Abundance of wild Brook Trout Salvelinus fontinalis (BKT) and MYY Brook Trout $\geq 100 \mathrm{~mm}$ sampled in study waters in Idaho during 2022. Estimates of abundance were calculated using either a mark-recapture (MR) or depletion abundance (DE) survey with $95 \%$ confidence estimates (CI). Also included are the proportion of $\mathrm{M}_{\mathrm{YY}}$ Brook Trout composition in the population, the number of wild Brook Trout removed from the system during annual suppression, and suppression rate.

| Waterbody | Sample method | Wild BKT <br> Abundance | 95\% CI | $M_{Y Y} B K T$ <br> Abundance | 95\% CI | $\mathrm{M}_{\mathrm{YY}}$ BKT <br> Composition | \# of wild <br> BKT removed | $\begin{gathered} \text { Suppression } \\ \text { rate } \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Streams |  |  |  |  |  |  |  |  |
| Dry Creek | MR | 1,570 | 1,306-1,833 | 5,676 | 4,724-6,628 | 78\% | 997 | 64\% |
| East Threemile Creek | DE | 5,417 | 5,337-5,497 | 0 | - | - | - | - |
| East Fork Clear Creek | DE | 1,032 | 965-1,100 | 79 | 58-100 | 7\% | - | - |
| Pike's Fork Creek | MR | 4,221 | 3,605-4,837 | 0 | - | - | 2,030 | 48\% |
| Tripod Creek | DE | 4,892 | 4,735-5,049 | 6,493 | 6,245-6,740 | 57\% | - | - |
| Lakes |  |  |  |  |  |  |  |  |
| Martin Lake | MR | - | - | - | - | - | 13 | - |
| Seafoam Lake | MR | 197 | 154-239 | 331 | 259-403 | 63\% | 150 | 45\% |

Table 4. The number of $M_{Y Y}$ Brook Trout Salvelinus fontinalis stocked into study waters in Idaho during 2022, stocking date, and average size of stocked fish. Stocking rate of Myy Brook Trout compared to the wild Brook Trout population was calculated by dividing the number of fish stocked divided by the total number of wild Brook Trout in the population.

| Waterbody | Stocking size | Stocking date | \# of fish stocked | Mean length (mm) | SE | $\begin{gathered} \text { Mean } \\ \text { weight (g) } \end{gathered}$ | SE |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Streams |  |  |  |  |  |  |  |
| Dry Creek | Fingerling | 8/9/2022 | 3,978 | 133 | 1.8 | 26 | 1.2 |
| East Fork Clear Creek | Fingerling | 8/25/2022 | 105 | 134 | 1.6 | 25 | 0.9 |
| East Threemile Creek | Catchable | 8/18/2022 | 593 | 243 | 2.3 | 152 | 6.0 |
| Pike's Fork Creek | Fingerling | 8/11/2022 | 2,835 | 133 | 1.8 | 26 | 1.2 |
| Tripod Creek | Fingerling | 8/25/2022 | 6,966 | 134 | 1.6 | 25 | 0.9 |
| Lakes |  |  |  |  |  |  |  |
| Black Lake | Catchable | 8/2/2022 | 200 | 236 | 2.8 | 132 | 5.0 |
| Duck Lake | Fingerling | 8/2/2022 | 2,123 | 127 | 1.8 | 21 | 0.9 |
| Lloyds Lake | Fingerling | 8/2/2022 | 1,105 | 127 | 1.8 | 21 | 0.9 |
| Martin Lake | Fingerling | 9/1/2022 | 787 | 143 | 1.6 | 28 | 1.0 |
| Rainbow Lake | Catchable | 8/2/2022 | 768 | 236 | 2.8 | 132 | 5.0 |
| Seafoam Lake \#4 | Fingerling | 9/1/2022 | 1,176 | 143 | 1.6 | 28 | 1.0 |

Table 5. Results of genetic sex ratio of wild Brook Trout fry ( $<100 \mathrm{~mm}$ ) each study water at the inception of the study and from the 2022 surveys. Genetic sex and parental origin were then determined based on analysis of fin clips. Additionally, information on whether the system receives annual suppression to remove wild Brook Trout prior to stocking, and the size of the $\mathrm{M}_{\mathrm{Yy}}$ fish that are stocked into the system.

| Stream | Treatment | Stockingsize | Avg. annual \# stocked | \% Male offspring |  |  |  | $\begin{gathered} \% M_{r y} \\ \text { offspring } \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  |  | Start |  | Current |  |  |
|  |  |  |  | Year | \% (n) | Year | \% (n) |  |
| Streams |  |  |  |  |  |  |  |  |
| Dry Creek | Suppression | Fingerling | 3,886 | 2016 | 45\% (286) | 2022 | 84\% (57) | 94\% |
| East Fork Clear Creek | Non-suppression | Fingerling | 535 | 2016 | 57\% (98) | 2022 | 55\% (93) | 18\% |
| East Threemile Creek | Non-suppression | Catchable | 1,079 | 2017 | 51\% (97) | 2022 | 54\% (94) | 0\% |
| Pikes Fork Creek | Suppression | Fingerling* | 792 | 2017 | 48\% (145) | 2022 | 60\% (65) | 8\% |
| Tripod Creek | Non-suppression | Fingerling | 5,691 | 2016 | 27\% (100) | 2022 | 51\% (94) | 31\% |
| Lakes |  |  |  |  |  |  |  |  |
| Martin Lake | Suppression | Fingerling | 865 | 2017 | 50\% (120) | 2022 | -- | -- |
| Seafoam Lake \#4 | Suppression | Fingerling | 1,098 | 2017 | 51\% (99) | 2022 | 39\% (31) | 0\% |

*Switched from catchables to fingerlings stocking in 2022
${ }^{1}$ Estimates from the 2022 sampling.

FIGURES


Figure 1. Locations of study lakes (red) and streams (yellow) for My Brook Trout Salvelinus fontinalis field trials in Idaho.


Figure 2. Length distributions of wild Brook Trout Salvelinus fontinalis and Myy Brook Trout sampled in Dry Creek and Pikes Fork Creek, Idaho, in 2022. Note: no Myy Brook Trout were captured in Pikes Fork Creek in 2022.


Figure 3. Length distributions of wild Brook Trout Salvelinus fontinalis and Myr Brook Trout sampled in the three non-suppression study streams in 2022.


Figure 4. Length distributions of wild Brook Trout Salvelinus fontinalis and Myy Brook Trout sampled in Martin Lake and Seafoam Lake \#4, Idaho in 2022.

## ANNUAL PROGRESS REPORT

# SUBPROJECT \#2: TRENDS IN THE OCCUPANCY AND ABUNDANCE OF REDBAND TROUT AND NONNATIVE TROUT IN THE WOOD RIVER BASIN 

State of: Idaho

Project No.: $\underline{3}$
Title: $\quad$ Wild Trout Evaluations
Subproject \#2: Trends in the occupancy and abundance of Redband Trout and nonnative trout in the Wood River Basin


#### Abstract

The Wood River Basin in central Idaho has been isolated from the surrounding Snake River Basin by Malad Gorge Falls for at least 50,000 years, and recent genetic analyses suggest that Redband Trout Oncorhynchus mykiss in the basin represent a distinct previously undescribed lineage. To assess their contemporary status, we revisited 22 stream reaches in 2021-2022 that were originally surveyed in 2003 and that were occupied by Redband Trout. Our objective was to assess changes in the occupancy and abundance of Redband Trout as well as nonnative trout. In 2021-2022, Redband Trout were present in 17 of the 22 originally occupied reaches, with all 5 extirpated reaches now entirely comprised of Brook Trout Salvelinus fontinalis. Brook Trout were originally present at 20 of the 22 reaches and were extirpated from two reaches, both of which are now entirely comprised of Redband Trout. Brown Trout Salmo trutta colonized one new reach since 2003 and were present at all reaches $(\mathrm{n}=3)$ which exceeded 10 m average wetted stream width. Redband Trout were the dominant species ( $\geq 70 \%$ ) in 12 of the 22 sites in 2003 but only 8 reaches in 2021-2022. Because extensive historical hatchery stocking of Rainbow Trout throughout the basin has resulted in limited introgression, we deem the presence of Brook Trout in headwater habitats and Brown Trout in main stem habitats as the primary concern for the longterm conservation of Redband Trout in the Wood River Basin.


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## INTRODUCTION

The Wood River Basin in central Idaho has been hydrologically isolated from the surrounding Snake River Basin by Malad Gorge Falls for at least the last $\sim 50,000$ years (Lamb et al. 2014). This isolation has resulted in unique fishes occupying lotic habitat in the basin, including the endemic Wood River Sculpin Cottus leiopomus (Simpson and Wallace 1982), and genetically divergent populations of Bridgelip Sucker Catostomus columbianus (Smith 1966) and Mountain Whitefish Prosopium williamsoni (Miller 2006). A recent genetic investigation indicated that Redband Trout Oncorhynchus mykiss (the Interior form of O. mykiss, relative to the coastal O. mykiss designation of Rainbow Trout) in the Wood River Basin may also represent a unique lineage, one that has not been previously described (Campbell et al. 2022). While stocking of fertile hatchery Rainbow Trout (of various origins) in the basin ceased decades ago (Kozfkay et al. 2006), such stocking did occur for nearly a century. Surprisingly, introgression in the basin between native and nonnative O. mykiss appears to be limited (Campbell et al. 2022). Consequently, determining the status of this unique form of $O$. mykiss is an important conservation need.

Prior comprehensive research in the basin suggests that in 2003, Redband Trout occupied $1,246 \mathrm{~km}(19 \%)$ of a total of $6,480 \mathrm{~km}$ of stream in the basin (at a 1:100,000 hydrologic scale), with an estimated population of 197,000 Redband Trout present at that time (Meyer et al. 2014). Whether their distribution and abundance has changed since these surveys were conducted is unknown.

## OBJECTIVES

1. Compare contemporary Redband Trout occupancy and abundance in the Wood River Basin to the prior work conducted in 2003 by Meyer et al. (2014).
2. Assess whether nonnative Brook Trout and/or Brown Trout were experiencing changes in occupancy or abundance, and whether they were adversely impacting Redband Trout in the basin.

## STUDY AREA

The Wood River Basin located in central Idaho has a drainage area of $7,778 \mathrm{~km}^{2}$ and consists of three sub-basins: the Big Wood River, the Little Wood River, and Camas Creek (Figure 1). The Malad River forms at the confluence of the Big Wood and Little Wood rivers and flows downstream to the Snake River. Geologic processes of glaciation and episodic volcanic activity likely contributed to the isolation of fish populations in the basin through the formation of the Malad Gorge Falls, eliminating upstream fish passage from the Snake River (Lamb et al. 2014. Connectivity is further limited within the basin by multiple irrigation diversions on the main stem rivers and tributaries and two large irrigation storage reservoirs. Stream discharge is driven by alpine snowmelt, peaking between April and June, but is modified by irrigation reservoirs and diversions. Elevations are highest at the mountainous headwaters (over $3,000 \mathrm{~m}$ above sea level) and lowest ( 930 m ) at the confluence with the Snake River.

## METHODS

Native fish species in the Big Wood River Basin include the Wood River sculpin, Redband Trout, Mountain Whitefish, Bridgelip Sucker, Largescale Sucker Catostomus macrocheilus, Utah Chub Gila atraria, Redside Shiner Richardsonius balteatus, Longnose Dace Rhinichthys cataractae, and Speckled Dace Rhinichthys osculus. In addition, the introduced species Brook Trout Salvelinus fontinalis and Brown Trout Salmo trutta have established self-sustaining populations within the basin. Populations of the warmwater game fish Smallmouth Bass Micropterus dolomieu, Largemouth Bass Micropterus salmoides, Yellow Perch Perca flavescens, and Bluegill Lepomis macrochirus are also present, primarily in lentic waters.

In 2003, a list of spatially balanced randomly selected study reaches were generated with the help of the Environmental Protection Agency's Environmental Monitoring and Assessment Program. This technique maps two-dimensional space (in our study, a 1:100,000 scale hydrography layer) into one-dimensional space with defined, ordered spatial addresses and uses restricted randomization to randomly order the spaces. Systematic sampling of the randomly ordered spaces results in a spatially balanced sample (Stevens and Olsen 2004) of study reaches. For further details on sample reach selection, see Meyer et al. (2014).

Redband Trout were present at 24 of the 114 stream reaches that were sampled in the Wood River Basin in 2003. In 2021-2022, we resampled all reaches where Redband Trout were formerly present, except for two reaches on private property where access could not be obtained. Sampling occurred at baseflow conditions from July to October for both sampling periods (2003 and 2021-2022) to minimize differences in fish capture efficiency and shifts in habitat use. Fish were captured using electrofishing gear, generally using settings of $50-60 \mathrm{~Hz}, 10-25 \%$ duty cycle, and 200-500 volts. All trout collected by electrofishing were anesthetized, identified to species, enumerated, measured for total length to the nearest millimeter, and released. The few triploid hatchery Rainbow Trout encountered were readily identifiable based on fin condition, were released, and were not considered further.

At sample reaches less than $\sim 15 \mathrm{~m}$ wide, depending on stream size, crews of two to seven people performed multiple-pass depletions (two to four passes) using at least one but up to three backpack electrofishing units. Block nets and/or natural stream breaks were used to minimize fish movement into and out of the study reach during depletion surveys. The Zippin removal estimator was used to estimate trout abundance (Zippin 1958) for fish $\geq 100 \mathrm{~mm}$ total length. If all trout were captured on the first pass, we considered that catch to be the estimated abundance. At two reaches, only one electrofishing pass was conducted, for which abundance was estimated using the linear relationship between the first pass and the resulting abundance estimates from all other multi-pass study reaches (cf. Kruse et al. 1998). Reach length averaged 103 m in 2003 and 101 m in 2021-2022.

At wider, deeper reaches where fish depletion among passes was not feasible, markrecapture abundance estimates were conducted using a barge-mounted electrofishing unit with crews of seven to ten people. All trout were marked with a caudal fin clip during the single marking run, with marked and unmarked trout captured using a single recapture run a few days later. We assumed that there was no movement of trout into or out of the study reach between runs, and reaches were much longer to help minimize movement between the mark and recapture runs (mean reach lengths of 879 m in 2003 and 912 m in 2021-2022). Estimates of trout abundance for fish $\geq 100 \mathrm{~mm}$ were made with the Lincoln-Petersen mark-recapture model as modified by Chapman (1951). Estimates were made separately for the smallest size-classes possible (generally $25-50 \mathrm{~mm}$ ) while meeting the criteria that (1) the number of fish marked in the marking
run multiplied by the catch in the recapture run was at least four times the estimated population size, and (2) at least three recaptures occurred per size-class; meeting these criteria creates estimates that are biased by less than 2\% (Robson and Regier 1964).

For both depletion and mark-recapture electrofishing, all trout captured were pooled for an overall estimate of trout density in the study reach (e.g., Isaak and Hubert 2004; Carrier et al. 2009), and point estimates for each species were then calculated based on the proportion of catch comprised by each species (Meyer and High 2011).

## RESULTS

Of the 22 stream reaches in the Wood River Basin where Redband Trout were present in 2003, they were still present at 17 (77\%) reaches in 2021-2022 (Table 6). The five stream reaches where Redband Trout were apparently extirpated between 2003 and 2021-2022 were small (<4.1 m mean wetted stream width) and are now comprised entirely of Brook Trout (Table 6). Of the five extirpated reaches, Redband Trout were present within 400 m of the Federal Gulch reach and 6.13 km of the North Fork Big Wood River reach. At two study reaches, Redband Trout were not found within 441 m upstream (Iron Mine Creek) or 2 km upstream and downstream of the East Fork Fish Creek study reach, and one site was dry upstream for 1.6 km from the study reach (Cove Creek; Figure 6).

In comparison, Brook Trout were present at 20 of the 22 reaches ( $91 \%$ ) occupied by Redband Trout in 2003. By 2021-2022, Brook Trout were absent from two reaches that they occupied in 2003 and remained absent in the two reaches where they were absent in 2003. The two reaches where Brook Trout were apparently extirpated were also small ( 3.1 and 5.0 m wide in 2021-2022) and are now comprised entirely of Redband Trout. Brown Trout were present at two reaches in 2003 and three reaches in 2021-2022, having colonized one new reach (Figure 6). In 2021-2022, Brown Trout occupied all three stream reaches that exceeded 10 m average wetted stream width (Table 6).

Average abundance in 2003 was 3.38 fish $/ 100 \mathrm{~m}^{2}$ for Redband Trout, 2.01 fish $/ 100 \mathrm{~m}^{2}$ for Brook Trout, and 0.02 fish $/ 100 \mathrm{~m}^{2}$ for Brown Trout. Abundance in 2021-2022 increased for all species, with mean abundance of 6.15 fish $/ 100 \mathrm{~m}^{2}$ for Redband Trout, 4.67 fish $/ 100 \mathrm{~m}^{2}$ for Brook Trout, and 0.65 fish $/ 100 \mathrm{~m}^{2}$ for Brown Trout. There was little indication that an increase in nonnative trout abundance from 2003 to 2021-2022 resulted in a decrease in the abundance of Redband Trout (Figures 7 and 8).

## DISCUSSION

From a conservation perspective, Brook Trout likely pose the largest threat to the longterm persistence of Redband Trout in the Wood River Basin, as evidenced by the fact that during the last two decades, Brook Trout displaced Redband Trout at one-quarter of the reaches where they were formerly sympatric. However, while fewer in number, there were also some formerly sympatric reaches where Redband Trout persisted and Brook Trout surprisingly did not, indicating that displacement of Redband Trout by Brook Trout is not a foregone conclusion. In western North America, Brook Trout have repeatedly displaced or replaced both Cutthroat Trout and Bull Trout (reviewed in Dunham et al. 2002; Rieman et al. 2006), often resulting in little sympatric overlap between Brook Trout and either of these two native salmonids (e.g., Meyer et al. 2022; Voss et al. 2023). Unlike Cutthroat Trout and Bull Trout, our results suggest that Redband Trout can
exhibit some level of biotic resistance to Brook Trout invasion, and prior research has demonstrated similar resistance by Redband Trout to Brook Trout displacement (e.g., Benjamin et al. 2007; Miller et al. 2013).

Brown Trout may also pose a long-term conservation threat to Redband Trout in the basin (Budy and Gaeta 2017), although this threat is likely limited to the main stem, larger stream segments. Brown Trout can successfully occupy small, high-elevation Rocky Mountain streams (Young 1999; Al-Chokhachy and Sepulveda 2019), but recruitment in those waters is often limited by environmental conditions (Wood and Budy 2009), and Brown Trout are known to prefer warmer, lower elevation main stem rivers (de la Hoz Franco and Budy 2005). Considering the collective threat that Brown Trout pose in larger rivers and Brook Trout pose in headwater streams, an integrated pest management approach - in the form of rotenone treatments, manual electrofishing suppressions, and perhaps even Myy stocking - may be needed in some areas to control their spread in the Wood River Basin.

Despite being displaced from several study reaches in the Wood River Basin, Redband Trout density increased substantially from 2003 to 2021-2022, as did the densities of nonnative Brook Trout and Brown Trout. Salmonid populations are notoriously variable in nature (Platts and Nelson 1988; House 1995), with annual fluctuations that are driven by stochastic and demographic population responses (Cattanéo et al. 2003; Copeland and Meyer 2011), thus we cannot rule out the possibility that the 2003 surveys were by chance conducted in a period of lower overall trout abundance and the 2021-2022 surveys were conducted in a more favorable period. Alternatively, the increased abundance we observed may be a reflection of improved habitat or environmental conditions in the basin. Habitat improvement projects have indeed been implemented in the basin, including bank stabilization, removal of fish passage barriers, culvert replacements, irrigation ditch screening, fish entrainment recoveries, floodplain reconnection, and deployment of beaver dam analogs (M.P. Peterson, unpublished data). However, such activities have been widely implemented in countless river basins across the western United States and beyond, and although they are generally beneficial (reviewed in Roni et al. 2008), they probably do not fully explain the dramatic increases in stream fish abundance we observed. Though the few abiotic factors we measured in our analyses apparently had no influence on the temporal changes we observed in Redband Trout density, it is curious that of the study reaches with mean wetted width $<5 \mathrm{~m}$, one-half experienced extirpation of either Redband Trout or Brook Trout, whereas none of the study reaches with mean wetted width $>5 \mathrm{~m}$ experienced any species extirpations. Consequently, despite stream width having little influence on changes in Redband Trout abundance, we recommend that any Redband Trout restoration actions or Brook Trout eradication efforts be initially focused on the smallest streams in the basin.

It is admittedly difficult to draw firm conclusions on Redband Trout status and nonnative trout expansion in the Wood River Basin based on only two sets of surveys across time. As a result of only resurveying reaches where Redband Trout occurred in 2003, our study design only allowed us to detect range contraction (not expansion), although we deem it unlikely that Redband Trout have significantly expanded into previously unoccupied habitat in the basin; surveys to assess this notion are needed. Despite study limitations, there is reason to be concerned with the potential spread of nonnative salmonids - especially Brook Trout - in the basin. Continued periodic monitoring of these established reaches and other locations is needed to assess the status of this unique lineage of Redband Trout more thoroughly, and to better monitor the potential expansion of nonnative salmonids in the basin. While Redband Trout appear to be at least somewhat resistant to Brook Trout displacement, a concerted effort to implement an integrated pest management approach in the basin would seem prudent to control their spread.

## RECOMMENDATION

1. Continue to periodically survey these established monitoring reaches, at an increased frequency, to assess further changes in occupancy and abundance of salmonids in the Wood River Basin.
2. Evaluate the potential to manually eradicate Brook Trout from certain drainages to secure more allopatric water for Redband Trout in the basin. Eradications could use a combination of either rotenone or electrofishing, plus Myy Brook Trout stocking.
3. Consider evaluating whether Brook Trout have a competitive advantage over Redband Trout in Wood River Basin streams, perhaps via a DJ-funded graduate project.

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## TABLE

Table 6. Study reach descriptions, total fish density for Redband Trout (RBT), Brook Trout (BKT), and Brown Trout (BNT) expressed as fish >100 mm total length/100 $\mathrm{m}^{2}$ in the Wood River Basin surveyed in 2003 and 2021-2022.

| Stream name | Stream reach | Latitude | Longitude | Elevation (m) | Gradient | Average wetted width (m) ${ }^{1}$ | 2003 fish density |  |  | 2021/2022 fish density |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  |  |  |  |  | RBT | BKT | BNT | RBT | BKT | BNT |
| Baker Creek | 53 | 43.7446 | -114.5637 | 2,103 | 1.35 | 5.69 | 0.14 | 0.84 | 0.00 | 0.96 | 0.32 | 0.00 |
| Big Wood River | 33 | 43.8388 | -114.6310 | 2,164 | 1.29 | 4.97 | 1.28 | 0.28 | 0.00 | 4.81 | 0.00 | 0.00 |
| Big Wood River | Lower Hailey | 43.5135 | -114.3192 | 1,616 | 1.80 | 20.80 | 14.68 | 0.61 | 0.04 | 14.39 | 0.34 | 9.88 |
| Big Wood River | 181 (or "Boulder") | 43.7770 | -114.4945 | 2,009 | 0.57 | 8.38 | 0.85 | 0.02 | 0.00 | 7.16 | 0.07 | 0.00 |
| Big Wood River | 165 (or "Gimlet") | 43.5865 | -114.3489 | 1,678 | 0.66 | 19.86 | 21.37 | 0.03 | 0.00 | 11.56 | 0.12 | 2.80 |
| Cove Creek | 133 | 43.6077 | -114.1866 | 1,971 | 0.69 | 0.66 | 1.47 | 6.61 | 0.00 | 0.00 | 2.15 | 0.00 |
| East Fork Big Wood River | 57 | 43.6526 | -114.1967 | 1,960 | 1.92 | 4.67 | 4.96 | 6.20 | 0.00 | 11.28 | 10.58 | 0.00 |
| East Fork Big Wood River | 121 | 43.6183 | -114.2976 | 1,775 | 1.82 | 6.41 | 5.22 | 0.58 | 0.00 | 11.47 | 0.13 | 0.00 |
| East Fork Big Wood River | 185 | 43.6671 | -114.1524 | 2,065 | 3.10 | 3.53 | 2.46 | 2.73 | 0.00 | 2.38 | 5.09 | 0.00 |
| East Fork Fish Creek | 113 | 43.4514 | -113.7608 | 1,717 | 2.03 | 2.43 | 0.59 | 1.76 | 0.00 | 0.00 | 35.10 | 0.00 |
| Federal Gulch | NA | 43.6672 | -114.1493 | 2,083 | 6.91 | 1.22 | 0.82 | 1.64 | 0.00 | 0.00 | 1.24 | 0.00 |
| Fox Creek | 228 | 43.7444 | -114.4516 | 2,063 | 4.61 | 2.10 | 5.42 | 4.65 | 0.00 | 16.51 | 14.21 | 0.00 |
| Friedman Creek | 306 | 43.5331 | -113.8721 | 1,916 | 2.23 | 3.84 | 2.31 | 2.31 | 0.00 | 1.74 | 5.50 | 0.00 |
| Friedman Creek | 41 | 43.5133 | -113.8935 | 1,829 | 1.75 | 3.46 | 2.40 | 10.95 | 0.00 | 0.00 | 0.80 | 0.00 |
| Iron Mine Creek | 362 | 43.5370 | -113.7313 | 1,887 | 2.81 | 2.97 | 1.05 | 0.35 | 0.00 | 0.00 | 11.10 | 0.00 |
| Little Wood River | 137 | 43.4142 | -114.0073 | 1,549 | 0.47 | 12.40 | 0.77 | 0.00 | 0.00 | 2.96 | 0.00 | 0.00 |
| Little Wood River | 169 | 43.1773 | -114.0218 | 1,400 | 0.73 | 10.69 | 0.49 | 0.00 | 0.47 | 0.26 | 0.00 | 1.61 |
| North Fork Big Wood River | 222 | 43.8768 | -114.4478 | 2,195 | 6.03 | 4.05 | 0.18 | 1.95 | 0.00 | 0.00 | 7.97 | 0.00 |
| North Fork Thompson Creek | 360 | 43.6753 | -114.5446 | 2,097 | 3.92 | 3.07 | 2.77 | 2.08 | 0.00 | 21.19 | 0.00 | 0.00 |
| South Fork Warm Springs Creek | 21 | 43.6008 | -114.5991 | 2,103 | 1.56 | 2.19 | 1.41 | 0.35 | 0.00 | 22.33 | 2.73 | 0.00 |
| Trail Creek | 77 | 43.7234 | -114.3199 | 1,871 | 1.15 | 6.46 | 1.34 | 0.37 | 0.00 | 6.17 | 0.79 | 0.00 |
| Unnamed trib to Friedman Creek | 202 | 43.5339 | -113.8230 | 1,985 | 4.19 | 1.46 | 2.43 | 0.00 | 0.00 | 0.06 | 4.38 | 0.00 |

${ }^{1}$ Average of widths taken at 8-10 transects per site in 2021/2022

FIGURES


Figure 5. Location of study reaches resampled in 2021-2022 (black triangles), main stem rivers (dark grey), reservoirs, and streams (light grey) in the Wood River Basin, Idaho.


Figure 6. Fish species composition at each study reach sampled in 2003 and 2021-2022 in the Wood River Basin, Idaho where RBT = Redband Trout, BKT = Brook Trout, and BNT = Brown Trout.


Figure 7. Relationship between the densities of nonnative trout and Redband Trout in the Wood River Basin at reaches surveyed in 2003 and again in 2021-2022. Dotted lines represent fitted linear regressions.


Figure 8. Relationship between the change in density of nonnative trout and Redband Trout in the Wood River Basin at reaches surveyed in 2003 and again in 2021-2022. Dotted line represents fitted linear regression.

## ANNUAL PROGRESS REPORT

# SUBPROJECT \#3: FACTORS AFFECTING ANGLER CATCH AND HARVEST OF TROUT IN IDAHO ALPINE LAKES 

State of: Idaho
Project No.: $\underline{3} \quad$ Title: Wild Trout Evaluations

## Subproject \#3: Factors affecting angler catch and

 harvest of trout in Idaho alpine lakes
#### Abstract

Historically, most alpine lakes in western North America were devoid of fish, but over the last century many of these lakes have been stocked with trout (usually as fry) to provide angling opportunities. Basic information on rates of exploitation and catch-and-release in alpine lakes is lacking, as is information on factors affecting the catch of trout either stocked or naturally reproducing in such lakes. In the present study, a total of 1,265 tags were implanted in various species of salmonid in 113 lakes scattered across Idaho, of which 125 tags were eventually caught and reported by anglers. Anglers reported catching tagged fish from 1 to 1,465 days after they were originally tagged. Anglers caught and released more tagged fish in the first year (17.7\%) than they harvested (14.6\%). Logistic regression modeling indicated that a tagged salmonid in Idaho alpine lakes was more likely to be caught and reported by anglers when the fish was larger and when the lake was higher in elevation, had a more irregular shape, and contained less shallow habitat (Table 10). The likelihood of a tagged fish being caught and reported was also higher when hiking distance was shorter and more of the route was on a trail, and when the human population within 100 km of the lake was higher. For anglers who landed a tagged fish, they were apparently more likely to harvest the fish when the hike to the lake included less cumulative elevational gain to get there. Continued tagging of fish in alpine lakes during routine surveys is encouraged as an inexpensive means of monitoring angler use and catch.


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## INTRODUCTION

Historically, most alpine lakes in western North America were devoid of fish (Dunham et al. 2004), but over the last century many of these lakes have been stocked with hatchery trout to expand and diversify angling opportunities. Many alpine lakes now provide self-sustaining trout fisheries, although stocking programs continue in many U.S. states and Canadian provinces. Such fisheries provide a unique angling opportunity with solitude, dramatic scenery, and a backcountry experience seldom found in other settings. Not surprisingly, anglers visiting alpine lakes typically express high levels of satisfaction with their fishing experience (WGF 2002; IDFG 2007).

Although angling effort at remote alpine lakes is presumably diffuse (cf. McCormick 2015) compared to fisheries at lower elevations that are more easily accessed, fisheries managers must still make decisions regarding harvest regulations, which lakes and species to stock, and the rates and frequency of stocking. Due in part to this presumption of low angler effort at alpine lakes, detailed fisheries information on which to make management decisions is conspicuously lacking. Perhaps the most important information to garner in any fishery is the rate of angler exploitation, meaning the proportion of fish that are removed from the population annually via fishing harvest. A common technique for estimating exploitation is to release a known number of tagged fish, relying on anglers to report the tags as a means of estimating harvest (Pollock et al. 2001). Although trout anglers often choose to release their catch (Policansky 2002), rates of catch-andrelease (C-R) can also be estimated from reported tags by asking anglers whether they harvested or released the tagged fish. Combining estimates of harvest and C-R provides fisheries managers a measure of total angling utilization of the fishery. An additional benefit of such a fish tagging program is that total annual mortality rate can also be estimated for the population by comparing tag returns in year two to year one (Ricker 1975; McCammon and LaFaunce 1961). To our knowledge, the efficacy of a fish tagging program in alpine lakes to estimate rates of harvest, C$R$, and total annual mortality are lacking entirely, and our first objective was to fill this important information gap.

While basic information on rates of mortality and C-R in alpine lakes is long overdue, so too is information on factors affecting the catch of trout either stocked or naturally reproducing in such lakes. In Wyoming, angler accessibility was a primary factor affecting trout size structure in alpine lakes, suggesting that fisheries that were farther from roads and trailheads were less likely to receive appreciable angler harvest (Bailey and Hubert 2003). However, even for alpine lakes in closer proximity to roads or trailheads, an angler's interest in fishing a particular lake, and their ability to land fish there, can be influenced by numerous factors besides fish abundance, such as lake size (Ashe et al. 2014; Cassinelli and Meyer 2018) and morphology (Arlinghaus et al. 2017), the species (e.g., Brauhn and Kincaid 1982; Dwyer 1990) and size (Aas et al. 2000) of fish in the lake, and lake aesthetics (Hampton and Lackey 1976). Our second objective was to evaluate various factors associated with whether anglers caught tagged fish in alpine lakes. Because anglers informed us whether they chose to harvest or release the tagged fish they landed, we also evaluated factors associated with their decision to harvest or release the fish they caught.

## OBJECTIVES

1. Evaluate the efficacy of a tag release program in alpine lakes to evaluate estimates of exploitation and catch-and-release rates.
2. Evaluate what factors affect whether anglers caught tagged fish in alpine lakes, and whether they chose to harvest or release the fish they caught.

## METHODS

There are about 3,000 alpine lakes scattered across Idaho, with over 1,000 now containing some species of salmonid and over 600 on some sort of stocking rotation, the most common being a three-year rotation (Meyer and Schill 2007). Fish are generally stocked from fixed-wing aircraft as $40-60 \mathrm{~mm}$ fry, with stocking size varying annually depending on flight scheduling, and flight delays related to weather and fire conditions. For the present study, alpine lakes were generally sampled 2-3 years after the last stocking event so stocked fish could grow to a size desirable to anglers. However, it should be noted that self-sustaining wild trout comprise the majority of salmonids encountered in alpine lakes in Idaho (Koenig et al. 2011), and perhaps in other locations (Wiley 2003). Lakes included in this study (Figure 9) ranged from 1,610 to 3,158 m in elevation and 0.3 to 28.0 ha in surface area (Table 7). Hiking distance to access these lakes ranged from zero (i.e., accessible by vehicle) to 28 km . Fishing regulations at all lakes included a six-fish harvest limit, with no size or gear restrictions, except at one lake, which had a two-fish daily bag limit.

Shore-based angling was used to capture fish for tagging, using artificial flies or lures. Species landed by anglers included Rainbow Trout Oncorhynchus mykiss, Cutthroat Trout 0. clarkii, Rainbow Trout $\times$ Cutthroat Trout hybrids, Golden Trout O. aguabonita, Brook Trout Salvelinus fontinalis, and Arctic Grayling Thymallus arcticus. Landed fish were identified to species, measured to the nearest mm (total length), and implanted with a T-bar anchor tag at the base of the dorsal fin. In order to estimate tag loss, approximately $10 \%$ of all tagged fish were implanted with two tags adjacent to each other. Anchor tags were fluorescent orange, 70 mm in total length ( 51 mm of tubing) and treated with algaecide to prolong readability. Tags were labeled on two sides, with one side stating the agency and phone number and the other side listing a unique tag number.

Anglers were able to report tagged fish that they landed in a variety of ways. In addition to the phone number, a website has long been established in Idaho through which anglers can report fish tags (cf. Meyer et al. 2012). Some tags were also reported via mail or by returning tags to agency offices. Regardless of how the tags were reported, anglers were asked a series of questions, including the date the fish was captured, whether the fish was double tagged, and whether they harvested the fish or released it. Five tagged fish were reported twice by different anglers, and we included each reporting event as separate information for these tags.

Angler exploitation ( $u$ ) in Idaho alpine lake fisheries was calculated using the formula:

$$
u=\frac{r}{\lambda\left(1-\operatorname{tag}_{l}\right)\left(1-\operatorname{tag}_{m}\right)}
$$

where $r$ was the total number of tagged fish reported as harvested within 365 days of release divided by the total number of fish tagged in that same time period, $\lambda$ was the angler tag reporting rate, tag, was the tag loss rate, and $\operatorname{tag}_{m}$ was tagging mortality rate. We assumed that tagged fish caught by anglers were reported at a $54 \%$ reporting rate, and that tag loss was $9.7 \%$, as was observed for wild trout tagged with T-bar anchor tags in Idaho (Meyer et al. 2012; Meyer and Schill 2014). Tagging mortality was assumed to be $1 \%$ (Meyer and Schill 2014). The rate of C-R angling occurring in these fisheries was estimated in the same way as for $u$ except that the number
of tagged fish reported as harvested was replaced with the number reported as caught and released. Sample size was inadequate to estimate exploitation for individual lakes.

An adjustment to estimates of harvest and C-R was needed because tags were implanted over a four-month period, and they were therefore not vulnerable to anglers for the same amount of time in any given year. Indeed, fish tagged at the very end of the angling season were essentially not vulnerable to anglers until after they had survived harsh winter conditions at high elevation in relatively unproductive environments. Such lentic conditions can lead to considerable energy deficit and subsequent mortality (Biro et al. 2021), thus we assumed most natural mortality occurred over winter.

To account for this, we divided the number of tags reported the year after tagging occurred (but still within one full year of the tagging event) by the estimated rate of survival, which was calculated by estimating instantaneous total mortality rate ( $Z$ ) from tag return data in successive years (Ricker 1975; Miranda and Bettoli 2007). Specifically, tag returns were summed by 365-day intervals (from tagging date to date of capture) and plotted on a log scale in relation to the year the tagged fish was caught (i.e., year 1, year 2, etc.). The slope of the line was used to estimate Z, from which survival ( $S=e^{-z}$ ) was calculated. Estimating survival in this manner requires a number of assumptions, including that: (1) tagged fish are subject to the same survival rates as untagged fish; (2) survival is relatively constant during the study period and across all sizes of tagged fish; (3) tagged fish generally remain equally vulnerable each year they are at large; and (4) tag loss does not change through time. The final assumption was accounted for by adjusting tag returns each year by the rate of increase in tag loss previously observed for wild trout populations in Idaho from year one to year two (Meyer and Schill 2014), and we assumed the rate of tag loss continued at the same trajectory in subsequent years. This adjusted tag return total was added to the number of tags reported in the same calendar year as they were released; the sum was used to estimate harvest and C-R.

We measured a number of water-specific characteristics we felt might influence whether anglers caught a tagged fish in an alpine lake. Elevation (m) and surface area (km2) were estimated using TOPO® Version 2.7.3. Shoreline development index - the ratio of a lake's shore length to the circumference of a circle with the lake's area (Wetzel 2001) - was also estimated from measurements in Google Earth Pro. The proportion of the lake that was shallow (i.e., less than $2-3 \mathrm{~m}$ ) was indexed with satellite imagery because relative water depth is visible on satellite images when the water is clear (Stumpf et al. 2003). An index of shallow water was calculated by comparing the surface area of the lake that was darker in color (indicating deeper water) compared to lighter in color (indicating shallower water); such differentiation was visibly discernible in Google Earth Pro images (Figure 10).

Three measures of lake access difficulty were calculated using digital topographical maps, including (1) hiking distance (km) from the nearest trailhead or accessible road to the lake, (2) the proportion of the hike that was on a trail, and (3) the cumulative amount of gain in elevation (m) during the hike. Cumulative gain was used rather than net gain to account for instances where the hike required passing over a series of one or more elevational gains during the hike. In the few instances where the lake was lower in elevation than the starting point of the hike, we used the cumulative loss in elevation instead of the gain since the return hike from the lake to the vehicle would comprise the same amount of cumulative gain in elevation. A final measure of the likelihood of an angler visiting an alpine lake was the human population size living within 100 km of the lake; we assumed that more people living in the vicinity of the lake would increase the likelihood of a tagged fish being landed and reported by an angler (Post et al. 2008).

We used logistic regression to relate predictive variables to the reporting of tagged fish by anglers. Each tagged fish was considered the unit of observation, and the dependent variable in the model was a dummy variable of either 1 or 0 , which represented tags that were or were not reported as caught by anglers, respectively. All predictor variables were considered to be fixed effects, and candidate models included all combinations of predictor variables. Multicollinearity among continuous predictor variables was assessed by calculating variance inflation factors, all of which were <3.0, indicating that multicollinearity was low in our data set (Menard 1995). Moreover, correlation coefficients between independent variables were $<0.40$ for all but one comparison (total hiking distance vs. elevation gain, $r=0.73$ ).

We also used logistic regression to relate predictive variables to whether the angler harvested or released the tagged fish they caught. For this model, each reported fish was the unit of observation, and the dependent variable in the model was a dummy variable of either 1 or 0 , which represented whether the angler harvested or released the fish, respectively. Because we had little reason to suspect that lake morphology would affect angler harvest decisions, we limited the predictor variables to those related to the fish that was landed (i.e., species and length) and lake access difficulty.

For both modeling exercises, models were ranked using Akaike information criterion (AIC; Burnham and Anderson 2002), and we considered the most plausible models to be those with AIC scores within 2.0 of the best model (Burnham and Anderson 2004). We used AIC weights $\left(w_{i}\right)$ to assess the relative plausibility of each model within the set of most plausible models, and the adjusted coefficient of determination for discrete models (Nagelkerke 1991) was used to assess the amount of variation in tag returns that was explained by each model. The HosmerLemeshow goodness-of-fit statistic (Hosmer et al. 2013) was used to verify that the most plausible logistic regression models adequately fit the data. Coefficients were estimated and reported for all of the most plausible models, but coefficients were only considered influential if their 95\% confidence intervals (Cls) did not overlap zero. Statistical analyses were conducted using SAS (SAS Institute Inc, 2009).

## RESULTS

A total of 1,191 tags were implanted in various species of salmonid in 113 lakes scattered across Idaho (Figure 9), of which 125 tags were eventually reported by anglers from 52 different lakes. Anglers reported catching tagged fish from 1 to 1,465 days after they were originally tagged. We estimated that anglers caught and released more tagged fish in the first year (17.7\%) than they harvested (14.6\%; Table 8).

The most plausible logistic regression model explaining the variation in angler catch of tagged salmonids in alpine lakes indicated that the likelihood of a fish being caught by anglers was a function of shoreline development ratio, the proportion of the lake that was shallow, hiking distance, fish length, the proportion of the hike that was on a trail, nearby human density, lake elevation, and cumulative elevation gain on the hike (Table 9). Based on coefficient estimates and $95 \%$ Cls that did not overlap zero, tagged fish were more likely to be caught by anglers when the fish was larger and when the lake was higher in elevation, had a more irregular shape, and contained less shallow habitat (Table 10). The likelihood of a tagged fish being caught was also higher when hiking distance was shorter and more of the route was on a trail, and when the human population within 100 km of the lake was higher. There was also some support for two other models explaining variation in angler tag reporting (Table 9), but coefficient estimates and 95\% Cls around those estimates indicated no difference in the interpretation of those models (Table
10). The most plausible models explained only a small amount of the total variation in angler catch (Table 9).

The most plausible logistic regression model explaining variation in angler harvest of caught fish in alpine lakes indicated that anglers choosing to harvest a fish (rather than release it) was a function of the cumulative elevation gain on the hike and the species that was landed (Table 11). Interpretation of coefficient estimates and their $95 \% \mathrm{Cls}$ suggests that anglers were more likely to harvest a tagged fish (rather than release it) when the hike to the lake included less cumulative elevational gain to get there, but that harvest did not differ among species (Table 12). There was also support for several other models explaining angler harvest of tagged fish (Table 11), but $95 \%$ Cls around coefficient estimates included zero for every variable except cumulative elevational gain (Table 12), indicating no difference in model interpretation compared to the most plausible model. These models explained more variation in angler harvest than did models explaining angler catch (Tables 9 and 11).

## DISCUSSION

Because of the remote nature of many alpine lakes in Idaho and throughout the Rocky Mountains, and the rigorous elevation changes often associated with accessing these areas, it has been assumed that angling pressure is relatively low in such trout fisheries, compared to more easily accessed lowland waters. As expected, combined rates of harvest and C-R that we observed in Idaho alpine lakes were lower than results obtained in other types of wild trout and hatchery trout fisheries in Idaho (Table 13). While we cannot be certain that lower harvest and C$R$ rates were the result of lower angling pressure and not reduced catch rates, we presume the former explains our findings. These are the first estimates we are aware of for alpine lakes salmonid fisheries, and we encourage similar work in other regions of North America and beyond to allow for broader comparisons to our work. The fact that harvest was less than C-R is probably a recent phenomenon stemming from the proliferation of C-R practices among most contemporary trout anglers (Policansky 2002). Indeed, about 20 years ago Bailey and Hubert (2003) found evidence in northeastern Wyoming that cutthroat trout longevity was greater and larger fish were more plentiful in alpine lakes that were more difficult to access, which they attributed to angler harvest impacts.

Although we estimated that less than $20 \%$ of tagged salmonids in study lakes were either harvested or caught and released each year, we also observed higher tag return rates for larger fish, which suggests that our estimates of harvest and C-R may be biased low. Recall that we used estimates of annual survival to adjust the number of tags caught and reported within one year of tagging but not in the same calendar year as the tagging event occurred. This adjustment was needed because some fish were not tagged until the end of the angling season so they were virtually invulnerable to anglers until they had survived one winter in harsh high elevation conditions. Estimating survival based on tagged fish assumes that tagged fish remain equally vulnerable to capture and reporting each year they are at large. If smaller fish were less vulnerable to angling, and became more susceptible as they grew, their likelihood of being caught and reported by anglers would have increased in the years after tagging relative to the year in which they were tagged. This would have positively biased our survival estimates, resulting in an insufficient adjustment to the number of tags reported and thus an underestimate of harvest and C-R. Such an effect would have had minimal influence on our modeling results.

As expected, some of the characteristics of the lake and of the hike to get to the lake affected whether tagged fish were caught by anglers. Specifically, shallow water around the
perimeter of the lake and a more uniform perimeter to the lake hindered anglers from catching tagged fish. Trout are more wary in and generally avoid shallower water, likely due to predation concerns. This may reduce the likelihood of an angler encountering a fish in nearshore shallow habitat, and it may make those that are encountered more difficult to catch. More uniform shorelines provide less projections of land out into the lake, minimize effective casting distance, and provide less habitat complexity along the shoreline for trout to utilize. Shoreline depth and uniformity can both be scouted by anglers using satellite images prior to choosing a hiking destination (e.g., the Department's Fishing Planner webpage), thus making anglers aware of the benefits of irregular and deeper shorelines (perhaps via a blogpost) may improve angler catch rates and overall satisfaction.

Not surprisingly, fish that were tagged in more remote lakes with less trail access were less likely to be caught by anglers. However, the nuances of accessibility were difficult to fully characterize in this study. For example, at the lake in our study with the farthest hike ( 22 km ) and most cumulative elevation gain ( $2,634 \mathrm{~m}$ ), an angler reported the capture of one of the 18 fish that were tagged in that lake, but we should have expected no angler reports of tagged fish from such a remote lake. However, the angler who reported this fish likely accessed the lake via a nearby wilderness airstrip, not by hiking to the lake. Similarly, some lakes were open to all-terrain vehicles, but only a portion of the angling public owns such vehicles, thus access was not equivalent among all anglers. Such complexities in the overall study design may explain why our models, while useful in explaining angler tag returns from alpine lakes, explained little of the total variation in tag return data. Clearly, a number of other factors we did not include in our study likely affected an angler's likelihood of encountering (and reporting) a tagged fish. The most obvious of these is the esthetics and conveniences of a particular lake, such as the grandeur of the setting or the ability to locate flat, soft spaces large enough for overnight camping. Myriad other catchand non-catch-related determinants influence whether an angler can or chooses to exert angling pressure in remote, alpine fisheries, including fish size and catch rates, angling avidity, back country proficiency, angling regulations, and the length of time a lake and the surrounding area are free of snow and ice conditions.

While numerous factors appeared to affect angler catch in alpine lakes, of the conditions we included in our study, only the cumulative amount of elevation gain an angler experienced while hiking to a lake appeared to affect (in a negative manner) whether the angler chose to harvest or release the fish they caught. Considering that cumulative elevation gain and total hiking distance were correlated ( $r=0.73$ ), we speculate that the farther and more rigorous the hike was, the more avid the angler was and the less likely they were to rely on harvesting a fish for sustenance on the trip. Instead, lakes with less elevation gain (and less distance to travel) were perhaps more likely to experience day-use traffic and less avid anglers, and perhaps those anglers were more inclined to harvest a fish and carry it out with them. While such information could be used to adjust harvest regulations or stocking density, considering that harvest was $<10 \%$, such adjustments do not seem warranted at this time. Moreover, such low harvest in salmonid populations, combined with a lack of a fish size effect on angler harvest decisions, suggests that harvest was likely not substantially diminishing the size structure of fish populations in these alpine lakes. However, studies of the impacts of harvest on the size structure of salmonid populations are surprisingly absent in the literature.

We released well over 1,000 tagged fish in over 100 lakes, and tags were returned from nearly one-half of the lakes despite the fact that less than a dozen tagged fish were released in most lakes. Nevertheless, our conclusions regarding the factors that affected harvest or C-R of fish in Idaho alpine lakes were based on only 125 reported tags. Consequently, we encourage continued tagging of fish during routine monitoring of alpine lake fisheries to boost sample size.

This would improve our ability to determine species differences in harvest and C-R rates, and allow monitoring of changes through time in angler use. At present, our results indicate that anglers are clearly not overharvesting fisheries in Idaho alpine lakes.

## RECOMMENDATIONS

1. Department staff should continue to implant trout in alpine lakes with anchor tags, to broaden the geographical distribution and robust sample size from which to draw conclusions regarding alpine lake trout fisheries. As more fish are tagged, additional questions can be addressed, such as whether harvest or catch differs among species.
2. Some reward tags were released, but too few were released or reported by anglers to determine angler tag reporting rate. Rather than continue to assume reporting rate for trout in alpine lakes does not differ from reporting rates of wild trout in lowland fisheries, it is recommended that reward tags be released at 10\% of the total number of tagged fish until such time as sample sizes are adequate to empirically estimate tag reporting rate at Idaho alpine lakes.

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## TABLES

Table 7. Characteristics of alpine lakes in Idaho where salmonids were captured and implanted with T-bar anchor tags to assess angler exploitation and use.

| Lake characteristic | Mean | SD | Min | Max |
| :--- | :---: | :---: | :---: | :---: |
| Elevation $(\mathrm{m})$ | 2,485 | 372 | 1,610 | 3,158 |
| Surface area (ha) | 4.0 | 4.2 | 0.3 | 28.0 |
| Shoreline development ratio | 1.25 | 0.25 | 1.02 | 3.48 |
| Proportion shallow habitat | 0.31 | 0.17 | 0.02 | 0.98 |
| Hiking distance from nearest road or trailhead (km) | 6.1 | 5.5 | 0 | 27.9 |
| Proportion of hike on trail | 0.81 | 0.31 | 0.00 | 1.00 |
| Cumulative elevational gain on hike $(\mathrm{m})$ | 611 | 529 | 0 | 2,634 |
| Human population with 100 km of lake | 93,968 | 150,517 | 5,455 | 783,003 |
| Mean days-at-large for reported fish | 353 | 258 | 1 | 1,465 |

Table 8. The number of tags at-large for one full year in Idaho alpine lakes, the number reported by anglers as harvested or caught-and-released (C-R) within the same calendar year as tagging or in the next year (though still within one full year of the tagging event), and subsequent estimated rates of angler harvest and C-R. Rates for the next year were corrected for estimated mortality rates (see Methods).

|  |  | Harvested |  |  | Caught-and-released |  |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: |
| Metric | At-large | Same year | Next year |  | Same year | Next year |
| Tagged fish | 1,168 | 15 | 19 |  | 35 | 12 |
| Rate: |  | 0.027 | 0.047 |  | 0.062 | 0.030 |

Table 9. The most plausible logistic regression models relating angler reporting of tagged fish from Idaho alpine lakes to various lake morphology and angler accessibility conditions. Akaike information criterion (AIC), AIC difference ( $\triangle$ AIC), and AIC weights ( $w_{i}$ ) were used to select the most plausible models, whereas the adjusted coefficient of determination for discrete models $\left(\tilde{R}^{2}\right)$ was an indication of the amount of variation in tag reporting explained by the models.

| Model | AIC | $\boldsymbol{\Delta A I C}$ | $\boldsymbol{W}_{\boldsymbol{i}}$ | $\boldsymbol{R}^{2}$ |
| :--- | :---: | :---: | :---: | :---: |
| Shoreline development ratio + Percent littoral + hiking distance + fish <br> length + percent of hike on trail + nearby human density + elevation <br> + elevation gain | 732.18 | 0.00 | 0.39 | 0.10 |
| Shoreline development ratio + Percent littoral + hiking distance + fish <br> length + percent of hike on trail + nearby human density + elevation | 732.36 | 0.18 | 0.36 | 0.10 |
| Shoreline development ratio + Percent littoral + hiking distance + fish | 733.06 | 0.88 | 0.25 | 0.11 |
| length + percent of hike on trail + nearby human density + elevation <br> + elevation gain + surface area <br> Shoreline development ratio + Percent littoral + hiking distance + fish <br> length + percent of hike on trail + nearby human density + lake size |  |  |  |  |

Table 10. Coefficient estimates and $95 \%$ confidence intervals (Cls) for the most plausible models relating angler reporting of tagged fish from Idaho alpine lakes to various lake morphology and angler accessibility conditions.

| Coefficient | Estimate | 95\% CI |
| :--- | :---: | :---: |
|  | Best model |  |
| Intercept | -7.24 | $-9.30--5.17$ |
| Shoreline development ratio | 1.94 | $0.99-2.88$ |
| Shallow habitat | -0.03 | $-0.05--0.02$ |
| Hiking distance | -0.16 | $-0.25--0.07$ |
| Fish length | 0.005 | $0.002-0.008$ |
| Proportion of hike on trail | 0.91 | $0.21-1.61$ |
| Humans within 100 km (1000s) | 0.0015 | $0.0004-0.0026$ |
| Elevation | 0.0008 | $0.0002-0.0014$ |
| Elevational gain | 0.0006 | $-0.0002-0.0015$ |
|  | 2nd best | model |
| Intercept | -7.04 | $-9.08--4.99$ |
| Shoreline development ratio | 1.83 | $0.89-2.77$ |
| Shallow habitat | -0.03 | $-0.05--0.02$ |
| Hiking distance | -0.11 | $-0.17--0.05$ |
| Fish length | 0.005 | $0.002-0.008$ |
| Proportion of hike on trail | 0.83 | $0.15-1.52$ |
| Humans within 100 km (1000s) | 0.0014 | $0.0004-0.0025$ |
| Elevation | 0.0008 | $0.0002-0.0014$ |
|  |  |  |
| Intercept | 3rd best model |  |
| Shoreline development ratio | -6.81 | $-9.01--4.60$ |
| Shallow habitat | 2.01 | $1.05-2.97$ |
| Hiking distance | -0.04 | $-0.06--0.02$ |
| Fish length | -0.14 | $-0.24--0.05$ |
| Proportion of hike on trail | 0.005 | $0.002-0.008$ |
| Humans within 100 km (1000s) | 0.95 | $0.25-1.65$ |
| Elevation | 0.0014 | $0.0003-0.0025$ |
| Elevational gain | 0.0006 | $0.0000-0.0013$ |
| Surface area | 0.0006 | $-0.0003-0.0014$ |

Table 11. The most plausible logistic regression models relating whether anglers harvested the fish they landed in Idaho alpine lakes to various biological and angler accessibility conditions. Akaike information criterion (AIC), AIC difference ( $\triangle \mathrm{AIC}$ ), and AIC weights ( $w_{i}$ ) were used to select the most plausible models, whereas the adjusted coefficient of determination for discrete models ( $\tilde{R}^{2}$ ) was an indication of the amount of variation in harvest explained by the models.

| Model | AIC | $\boldsymbol{\Delta A I C}$ | $\boldsymbol{w}_{\boldsymbol{i}}$ | $\widetilde{\mathbf{R}}^{\mathbf{2}}$ |
| :--- | :---: | :---: | :---: | :---: |
| Elevational gain + species | 162.52 | 0.00 | 0.21 | 0.20 |
| Elevational gain | 162.82 | 0.30 | 0.18 | 0.12 |
| Elevational gain + species + percent of hike on trail | 162.83 | 0.31 | 0.18 | 0.22 |
| Elevational gain + fish length | 163.12 | 0.60 | 0.16 | 0.14 |
| Elevational gain + percent of hike on trail | 163.96 | 1.44 | 0.10 | 0.13 |
| Elevational gain + species + hiking distance | 164.5 | 1.98 | 0.08 | 0.20 |
| Elevational gain + species + fish length | 164.51 | 1.99 | 0.08 | 0.20 |
| Elevational gain + species + percent of hike on trail + hiking | 164.73 | 2.21 | - | 0.22 |
| distance |  |  |  |  |

Table 12. Coefficient estimates and 95\% confidence intervals (Cls) for the most plausible models relating whether anglers harvested the fish they landed in Idaho alpine lakes to various biological and angler accessibility conditions. Rainbow Trout was the reference species in the model.

| Coefficient | Estimate | 95\% Cl |
| :---: | :---: | :---: |
| Best model |  |  |
| Intercept | 1.10 | 0.16-2.03 |
| Elevational gain | -0.0025 | -0.0041-0.0009 |
| Species - Arctic Grayling | 0.48 | -0.76-1.72 |
| Species - Brook Trout | 0.63 | -0.49-1.76 |
| Species - Cutthroat Trout | -0.29 | -0.96-0.39 |
| Species - Rainbow/Cutthroat hybrids | 0.23 | -0.96-1.43 |
| 2nd best model |  |  |
| Intercept | 0.7616 | -0.08-1.60 |
| Elevational gain | -0.0025 | -0.0040-0.0010 |
| 3rd best model |  |  |
| Intercept | 0.26 | -1.33-1.86 |
| Elevational gain | -0.0025 | -0.0041--0.0009 |
| Species - Arctic Grayling | 0.43 | -0.82-1.68 |
| Species - Brook Trout | 0.62 | -0.51-1.74 |
| Species - Cutthroat Trout | -0.19 | -0.88-0.50 |
| Species - Rainbow/Cutthroat hybrids | 0.29 | -0.89-1.47 |
| Percent of hike on trail | 0.94 | -0.53-2.41 |
| 4th best model |  |  |
| Intercept | 1.90 | -0.05-3.86 |
| Elevational gain | -0.0023 | -0.0039--0.0008 |
| Fish length | -0.0043 | -0.0108-0.0023 |
| 5th best model |  |  |
| Intercept | 0.19 | -1.32-1.69 |
| Elevational gain | -0.0024 | -0.0039--0.0009 |
| Percent of hike on trail | 0.63 | -0.73-1.99 |
| 6th best model |  |  |
| Intercept | 1.12 | 0.13-2.11 |
| Elevational gain | -0.0024 | -0.0043-0.0006 |
| Species - Arctic Grayling | 0.49 | -0.76-1.74 |
| Species - Brook Trout | 0.62 | -0.51-1.76 |
| Species - Cutthroat Trout | -0.29 | -0.96-0.38 |
| Species - Rainbow/Cutthroat hybrids | 0.23 | -0.96-1.43 |
| Hiking distance | -0.01 | -0.18-0.15 |
| 7th best model |  |  |
| Intercept | 1.21 | -0.83-3.25 |
| Elevational gain | -0.0025 | -0.0041-0.0008 |
| Species - Arctic Grayling | 0.48 | -0.77-1.72 |
| Species - Brook Trout | 0.62 | -0.53-1.77 |
| Species - Cutthroat Trout | -0.28 | -0.96-0.40 |
| Species - Rainbow/Cutthroat hybrids | 0.23 | -0.98-1.43 |
| Fish length | -0.0004 | -0.0076-0.0068 |

Table 13. Rates (\%) of harvest and catch-and-release for wild and hatchery trout at various water types in Idaho with general harvest regulations (i.e., six fish bag limit with no size restriction).

| Water type | Origin | Harvest | Catch-and- <br> release | Source |
| :--- | :---: | :---: | :---: | :---: |
| Stream | Wild | 10.6 | 16.4 | Meyer and Schill 2014 |
| Stream | Hatchery | 18.7 | 16.9 | Meyer and Schill 2014 |
| Lake/reservoir | Wild | 14.6 | 5.2 | Meyer and Schill 2014 |
| Lake/reservoir | Hatchery | 17.3 | 5.0 | Meyer and Schill 2014 |
| Community pond | Hatchery | 46.1 | 7.0 | Chiaramonte and Meyer 2022 |
| Alpine lake | Variable | 7.3 | 9.2 | This study |

FIGURES


Figure 9. Locations of alpine lakes in Idaho where trout were tagged for this study.


Figure 10. Using color differences in satellite images to determine the proportion of Idaho alpine lakes that were shallow. Outer red line indicates the lake shore; inner red line differentiates relatively shallow from relatively deep water.

## ANNUAL PROGRESS REPORT

# SUBPROJECT \#4: EFFECTS OF ELEVATED WATER TEMPERATURES ON TROUT ANGLER CATCH RATES AND CATCH-AND-RELEASE MORTALITY 

State of: Idaho
Project No.: $\underline{3} \quad$ Title: Wild Trout Evaluations

| Subproject \#4: | Effect of elevated water <br> temperatures on trout angler catch |
| :--- | :--- |
|  | $\underline{\text { rates and catch-and-release }}$ |
|  | $\underline{\text { mortality }}$ |

Title: $\quad$ Wild Trout Evaluations


#### Abstract

During catch-and-release angling, some released fish do not survive, and there is growing concern that as climate change increases summer water temperatures in streams, occasional cessation of angling may be needed to protect fish populations. The objectives of the present study were to evaluate whether relative survival of fish caught by anglers was reduced when water temperature was elevated at the time of landing, and to evaluate the effect of temperature on angler catch rates. Anglers caught, marked, and released Cutthroat Trout Oncorhynchus clarkii ( $17-37 \mathrm{~cm}$ in length) in streams at temperatures from 13.5 to $25.7^{\circ} \mathrm{C}$. Recapture rate of marked fish (i.e., relative survival) declined from 0.58 for fish landed at water temperatures $<19^{\circ} \mathrm{C}$ to 0.30 for fish landed at temperatures $>21^{\circ} \mathrm{C}$, but angler catch rate declined similarly, with mean catch rates of 5.3 fish $/ \mathrm{h}$ at temperatures $<19^{\circ} \mathrm{C}$ and $3.4 \mathrm{fish} / \mathrm{h}$ at temperatures $>21^{\circ} \mathrm{C}$. Considering both declines, the number of fish mortalities/angler/h might be higher at cooler temperatures than at warmer temperatures, thus inhibiting fishing at elevated temperatures would likely have no more benefit to a trout population than it would at lower temperatures. Moreover, such temperatures are currently rare in Idaho's most popular trout fisheries. Consequently, we urge caution on implementing temperature-induced angling closures until population-level benefits are shown.


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## INTRODUCTION

Climate change models predict large reductions in salmonid occupancy of flowing waters during the $21^{\text {st }}$ century as some streams become too warm to support coldwater fish populations (Isaak et al. 2015). In addition to restricting the ability of salmonids to occupy warmer sections of streams, elevated stream temperature will also likely impact their ability to tolerate and recover from human-induced stressors they are subjected to in reaches where they can persist (reviewed in McCullough et al. 2009). One stressful event that salmonids are commonly exposed to is handling when they are caught and released by anglers.

Catch-and-release angling in recent decades has become very popular among anglers of all types, especially trout anglers (Policansky 2002). While catch-and-release angling-whether voluntary or mandatory-can be an effective tool to limit fishing-related mortality in recreational fisheries, not all fish that are released by anglers survive (e.g., High and Meyer 2014). In general, the level of fishing mortality induced by anglers during the catch-and-release process is directly related to the physical injury and level of stress a fish experiences while being hooked, landed, and handled prior to release (reviewed by Muoneke and Childress 1994). Some stress factors, such as fight duration and air exposure duration during the landing and releasing process (Lamansky and Meyer 2016), and the terminal tackle used (High and Meyer 2014), are within the control of anglers. Other factors, such as the water temperature that fish are experiencing while being hooked and landed, cannot be controlled by anglers unless they cease fishing when the temperature becomes elevated.

Temperature-related angling restrictions on trout and salmon in North America have been implemented in some Canadian provinces (Dempson et al. 2001) and some U.S. states, such as Montana as part of their drought fishing closure policy (Boyd et al. 2010). The Montana policy states that angling is closed (for all or part of the day) in waters containing salmonids when daily maximum water temperature is $\geq 23^{\circ} \mathrm{C}$ for three consecutive days. This policy was based on a study in Montana which found that catch-and-release angling on days in which maximum water temperature exceeded $23^{\circ} \mathrm{C}$ resulted in $13 \%$ mortality for Rainbow Trout Oncorhynchus mykiss and $3 \%$ mortality for Brown Trout Salmo trutta that were held in cages for three days, compared to zero mortality for both species for fish caught and held on days in which maximum water temperatures never exceeded $20^{\circ} \mathrm{C}$ (Boyd et al. 2010). However, differential mortality of freeranging trout in relation to water temperature at the time of capture has not been investigated. Moreover, angler catch rates for stream-dwelling salmonids may decline at higher water temperature (McMichael and Kaya 1991; Van Leeuwen et al. 2021), dampening the impact that elevated water temperature may have on lotic fish populations by reducing the number of fish landed by anglers when temperatures are warmer. We are unaware of any studies simultaneously investigating angler catch rates and the survival of stream-dwelling trout caught and released by anglers in relation to relatively warm summer stream temperatures. Consequently, our objective was to quantify the effect that elevated water temperature had on catch rates and catch-andrelease mortality in stream-dwelling trout populations.

## OBJECTIVE

1. Quantify effect of elevated temperatures on catch-and-release mortality and angler catch rates.

## METHODS

We conducted our study in four streams in eastern Idaho with summer water temperatures that were relatively high but that nonetheless maintained relatively abundant populations of stream-resident Cutthroat Trout O. clarkii (Table 14). Angling regulations prohibited the harvest of Cutthroat Trout in all study streams. Brook Trout Salvelinus fontinalis were also occasionally encountered; they were not included in the survival portion of the study because only a few were landed ( $n=5$ ), and it has been previously shown that survival of caught-and-released salmonids at elevated temperatures can differ among species (Boyd et al. 2010).

Angling occurred from July 27 to August 12, 2020, during some of the warmest days of the year. Anglers fished from about 0900 to 1800 hours each day as water temperatures increased from an overnight low and reached a peak for the day in late afternoon (Figure 11). One or two anglers fished each reach over part or all of any given day, but no reach was fished more than three times over the course of the study. Anglers recorded start and end times for each period of angling, and time recording was halted throughout the day for any nontrivial interruptions in angling effort (e.g., lunch break). Anglers used artificial flies exclusively to capture fish, and a landing net was used to minimize handling stress during data collection.

For each fish caught, species was recorded, and total length (to the nearest cm ) was measured in the landing net underwater using a tape measure. Time of capture was also recorded, and instantaneous water temperature at the time of capture was measured with a digital thermometer. Fight time was minimized to the extent possible but was not recorded explicitly. Landed fish were tagged with an individually-numbered anchor tag inserted just below the base of the dorsal fin. We assumed that tagging mortality was inconsequential. An adipose fin clip was used to evaluate whether any anchor tags were shed prior to recapture efforts. No fish were landed by anglers more than once. Fish were released at the point of capture, having received no air exposure during the catch-and-release process. Processing time from the point of landing the fish to releasing it was not measured but generally took 1-2 minutes.

Post-release relative survival was evaluated by recapturing tagged fish on August 25-27, 2020, using a single backpack electrofishing pass through each stream reach where angling occurred. Electrofishers were set at $60 \mathrm{~Hz}, 25 \%$ duty cycle, and enough volts to emit about 100 Watts of average power output. Captured fish were examined for anchor tags and adipose fin clips (none of the recaptured fish had lost their tag), measured for length (nearest cm), and released after recovering from being handled. Because recapture efficiency of fish landed by anglers was clearly not $100 \%$ with backpack electrofishing in our study streams, and some emigration of fish out of each study reach may have occurred, our analyses on relative survival in no way represent actual survival and is only meaningful in a comparative sense.

The effect of water temperature on catch-and-release relative survival was examined using logistic regression. Each landed fish was the experimental unit, with fish landed and tagged by anglers receiving dummy response variables for whether they were subsequently recaptured by electrofishing ( $0=$ not recaptured, $1=$ recaptured). Because sampling efficiency likely differed between streams, stream was included as a random effect in all models. Fish length was included as a fixed effect because relative mortality could be dependent on fish length, and because capture efficiency for stream-dwelling salmonids using backpack electrofishing is size dependent (Chiaramonte et al. 2020). Angler and instantaneous water temperature at the time of landing were also included as fixed effects; angler was included to account for potential differences in handling stress for landed fish among anglers. Finally, fish length $\times$ temperature and angler $\times$
temperature interaction terms were included to evaluate whether any effect of water temperature on relative survival of caught and released fish was mediated by fish length or the angler.

The effect of water temperature on catch rate was examined using general linear models. Each landed fish (including all salmonids caught) was the experimental unit. Catch rate (fish landed/h) for each landed fish (i.e., the response variable) was calculated by dividing 60 by the number of minutes since the last fish was landed; thus, for a fish that was landed 25 minutes after the previous fish, catch rate for that fish was calculated as $60 / 25=2.4$ fish $/ \mathrm{h}$. For each angler's last fish caught on each day, if fishing effort did not end at the time a fish was landed, then any extra fishing time that resulted in no fish landed was added to the time recorded for the last fish; this extra time averaged 18 minutes. Predictor variables included a random effect for stream and fixed effects for the angler and water temperature at the time of landing. An angler $\times$ temperature interaction term was included to evaluate whether any effect of water temperature on catch rate was mediated by the angler.

Candidate models included all combinations of predictive factors, and the random effect of stream was included in all models. Models were ranked using Akaike's information criterion corrected for small sample size ( $\mathrm{AIC}_{\mathrm{c}}$ ), and we considered the most plausible models to be those with $\mathrm{AlC}_{c}$ scores within 2.0 of the best model (Burnham and Anderson 2004). We used $\mathrm{AlC}_{c}$ weights ( $w_{i}$ ) to assess the relative plausibility of each model. Coefficients were only considered influential if their $90 \%$ confidence intervals (Cls) did not overlap zero. This more lenient interpretation of Cls was used to balance type I and type II errors, considering our relatively small sample size.

## RESULTS

In total, we landed 100 Cutthroat Trout and 5 Brook Trout Salvelinus fontinalis. Total length for Cutthroat Trout ranged from 17 to 37 cm , whereas Brook Trout ranged from 20 to 26 cm . The size of landed fish was similar for all streams (Table 14). Instantaneous water temperature at the time that fish were landed ranged from $13.5^{\circ} \mathrm{C}$ to $25.7^{\circ} \mathrm{C}$ (Figure 11). During electrofishing, 50 tagged Cutthroat Trout were recaptured (Brook Trout were not tagged).

Relative survival of angled trout declined as water temperature increased, with a recapture rate (all streams combined) of 0.58 for Cutthroat Trout caught at water temperatures $<19^{\circ} \mathrm{C}$ compared to 0.30 for those caught at temperatures $\geq 21^{\circ} \mathrm{C}$ and 0.17 for those caught at temperatures $\geq 23^{\circ} \mathrm{C}$ (Figure 12). The best model explaining the variation in catch-and-release relative survival included fish length, water temperature, and angler, as well as the random effect of stream (Table 15). There was also some support for models without some combination of water temperature, fish length, and angler, and for a model that included an interaction between length and water temperature. In the most parsimonious model, based on parameter estimates with $90 \%$ Cls that did not include zero, relative survival was reduced at higher water temperatures, for smaller fish, and for fish caught and released by angler 3 compared to angler 1 (Table 16). The effects of fish length and angler were also considered influential (i.e., 90\% Cls did not include zero) in the second-best model. Interaction terms for fish length $\times$ temperature and angler $\times$ temperature were not considered influential in any model based on $90 \%$ Cls.

Catch rate also declined as water temperature increased, with a mean catch rate of 5.3 fish $/ \mathrm{h}$ (SE $=0.7$ fish $/ \mathrm{h}$ ) at temperatures $<19^{\circ} \mathrm{C}$ compared to 3.4 fish $/ \mathrm{h}$ (SE $=0.7$ fish $/ \mathrm{h}$ ) at water temperatures $\geq 21^{\circ} \mathrm{C}$, or 1.2 fish $/ \mathrm{h}$ at $\geq 23^{\circ} \mathrm{C}$ (Figure 12). The best model explaining the variation observed in angler catch rate included water temperature and the random effect of stream (Table
17). There was also support for the null (random effect only) model, a model with water temperature, angler, and stream, and a model with only angler and stream. Based on parameter estimates with $90 \%$ Cls that did not include zero, water temperature was not considered influential in the best model but was considered influential in the $3^{\text {rd }}$ best model (Table 18) and indicated that catch rates declined at higher water temperatures. Catch rates also varied among anglers, but an angler $\times$ temperature interaction term was not included in any of the plausible models (Table 18).

## DISCUSSION

Myriad studies have been conducted on the effects of catch-and-release angling in recreational fisheries. Most of the work from the 1960s to the 1980s focused on the benefits of catch-and-release angling, and generally showed that in waters with high exploitation, both population abundance and angler catch rates increased when anglers switched to releasing most of their catch (see Barnhart 1989). In recent decades, the focus of most catch-and-release research has shifted to concerns that individual released fish may experience sub-lethal or lethal negative impacts after release due to stressful handling practices by anglers (reviewed in Cooke and Schramm 2007). Since water temperature can be a major stressor for coldwater species such as trout, as summer stream temperatures continue to rise due to climate change (Isaak et al. 2015), concern regarding the stress that catch-and-release angling may pose to stream-dwelling salmonid populations will also continue to rise in both the scientific literature (e.g., Isaak et al. 2015) and in popular articles and social media (Painter 2021; Peterson 2021).

In the present study, there was equivalent evidence that both relative survival and catch rate of stream-dwelling trout declined as water temperature increased. Both findings concur with previous research. Indeed, while elevated water temperatures have been shown to be negatively related to catch-and-release survival for Rainbow Trout, Brown Trout, Mountain Whitefish Prosopium williamsoni (Boyd et al. 2010), and Atlantic Salmon S. salar (Van Leeuwen et al. 2021), elevated temperature has also been shown to be negatively related to angler catch rates for Rainbow Trout (McMichael and Kaya 1991), Brown Trout (Taylor 1978), and Atlantic Salmon (L'Abée-Lund and Aspås 1999; Dempson et al. 2002; Van Leeuwen et al. 2021). The decline in angler catch rate at higher water temperatures is important because anglers presumably will either limit their fishing effort due to lack of success, or they will handle fewer fish at warmer water temperatures due to lower catch rates.

To scale this to actual fish mortality, let us assume that fly fishing catch-and-release mortality rate for stream-dwelling trout at non-elevated water temperatures averages about 0.05 (see High and Meyer 2014 and citations therein). Relative mortality (i.e., 1-relative survival) in the present study was 0.42 at cool temperatures ( $<19^{\circ} \mathrm{C}$ ) and 0.87 at high temperatures ( $\geq 23^{\circ} \mathrm{C}$ ). Let us therefore assume that mean fly-fishing mortality rate at high temperature is about $0.05 \times$ $(0.87 / 0.42)=0.10$, or about double the mortality rate at cool temperatures. Catch rate was 5.3 fish $/ \mathrm{h}$ at cool temperatures and 1.2 fish $/ \mathrm{h}$ at high temperatures. So when anglers fish at cool temperatures, they can be expected to incidentally cause mortality to 5.3 fish $/ \mathrm{h} \times 0.05=0.27$ fish/h. In contrast, when anglers fish at high temperatures, they can be expected to incidentally cause mortality to 1.2 fish $/ \mathrm{h} \times 0.10=0.12 \mathrm{fish} / \mathrm{h}$. This simple thought experiment suggests that, because angler catch rates are so much higher at cooler temperatures, inhibiting fishing at cooler water temperatures would actually be more beneficial to the trout populations than would fishing closures at high water temperatures. While we certainly do not recommend such closures, this highlights the need for caution in implementing temperature-related fishing closures until population-level benefits can be demonstrated in the trout populations they are purportedly
protecting. Indeed, seasonal angling closures restrict access to a public resource, so ideally they should be based on evidence of biological benefits for the fishery at the population scale.

How prevalent such high-water temperatures actually are in Idaho's trout streams can also be considered. Here we only considered the main stems of the most popular trout fisheries where water temperature data was available (Table 19), assuming that if the main stems do not experience high water temperatures, neither do the tributaries (regardless of whether they also are popular trout fisheries). From 2014 to 2021, we gathered $>300,000$ hours of summer (JunAug) hourly water temperature data at these waters and found that summer water temperatures reached or exceeded $23^{\circ} \mathrm{C}$ for only about 4,000 combined hours, or just over $1 \%$ of all the hours for which we had data. High water temperature occurrence was restricted to a few of the streams, usually at the lowest elevations in that stream.

The present study has a major limitation, that being a relatively small sample size. This resulted in wide confidence bounds on model parameter estimates, even when using more liberal $90 \% \mathrm{Cls}$, thus there is less certainty that the reductions we observed in relative survival and catch rate at higher water temperatures are reliable. Additional studies are clearly needed to confirm or refute our preliminary findings. In the meantime, as climate change increasingly leads to warmer water temperatures, concern is likely to accelerate on the part of fisheries managers as well as anglers with regard to potential impacts that warmer temperatures may have on the survival of caught-and-released fish. Research to date has largely focused on the impact of elevated water temperatures on the growth and survival of released fish, but the effect of increased temperature on angler catch rates should be given equal attention because if anglers' ability to land fish is diminished, so is their likelihood of causing incidental catch-and-release mortality. We expect that this topic will be hotly debated in the coming decades, but until there is evidence that trout populations are being negatively impacted by catch-and-release practices in areas where fishing is permitted at elevated water temperatures, we urge caution with the proliferation of angling restrictions at elevated water temperatures (often termed "hoot owl" regulations).

## RECOMMENDATION

1. Continue to periodically assess hourly water temperatures in Idaho's most prominent trout fisheries to evaluate increases in elevated water temperatures that trout are experiencing.

## ACKNOWLEDGEMENTS

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## TABLES

Table 14. Characteristics of streams in eastern Idaho with Cutthroat Trout and Brook Trout were landed to evaluate the effect of elevated summer water temperature on relative survival and angler catch rates.

| Stream | Latitude | Longitude | Reach length (km) | Elevation Gradient (m) (\%) |  | Mean width (m) | Fish landed |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  |  |  |  | n | Mean length (mm) |
| Willow Creek | $43.311^{\circ}$ | $111.777^{\circ}$ | 1.2 | 1,800 | 1.9 |  | 10.8 | 36 | 24.9 |
| Canyon Creek | $43.785^{\circ}$ | $111.445^{\circ}$ | 1.4 | 1,787 | 0.6 | 3.8 | 7 | 27.0 |
| McCoy Creek (lower) | $43.159^{\circ}$ | $111.206^{\circ}$ | 1.5 | 1,860 | 1 | 8.7 | 13 | 29.5 |
| McCoy Creek (upper) | $43.161^{\circ}$ | $111.275^{\circ}$ | 1.2 | 1,887 | 1 | 6.4 | 29 | 24.1 |
| Clear Creek | $43.162^{\circ}$ | $111.286^{\circ}$ | 1 | 1,896 | 0.9 | 3.2 | 15 | 26.2 |

Table 15. Comparison of models relative survival of Cutthroat Trout to water temperatures in eastern Idaho streams. Akaike's information criteria $\left(\mathrm{AIC}_{c}\right)$, change in $\mathrm{AIC}_{\mathrm{c}}\left(\triangle \mathrm{AIC}_{c}\right)$, and $\mathrm{AIC}_{\mathrm{c}}$ weights ( $w_{i}$ ) were used to assess models plausibility.

| Model | Log <br> likelihood | AIC $_{\mathbf{c}}$ | $\boldsymbol{\Delta A I C}_{c}$ | $\boldsymbol{w}_{\boldsymbol{i}}$ | AUC |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Length + temperature + angler + stream <br> Length + angler + stream | 121.70 | 136.94 | 0.00 | 0.28 | 0.77 |
| Length + temperature + length*temperature + <br> angler + stream | 125.06 | 137.98 | 1.04 | 0.17 | 0.75 |
| Temperature + stream | 121.22 | 138.82 | 1.88 | 0.11 | 0.78 |
| Temperature + angler + stream | 132.60 | 138.85 | 1.91 | 0.11 | 0.70 |
| Temperature + length + stream | 126.22 | 139.13 | 2.19 | 0.10 | 0.75 |
| Null (stream only) | 131.16 | 139.58 | 2.64 | 0.08 | 0.69 |
| Angler + stream | 136.15 | 140.28 | 3.34 | 0.05 | 0.67 |
| Length + stream <br> Length + temperature + angler + <br> temperature*angler + stream | 129.91 | 140.55 | 3.61 | 0.05 | 0.74 |

Table 16. Coefficient estimates and $90 \%$ confidence intervals (Cls) for the most plausible models constructed to evaluate the relative survival of Cutthroat Trout in relation to elevated summer water temperatures in eastern Idaho streams.

| Coefficient | Estimate | $\mathbf{9 0 \% ~ C l}$ |
| :--- | :---: | :---: |
|  | Model: length + temperature + angler + stream |  |
| Intercept | 1.23 | $-2.38-4.84$ |
| Fish length | 0.08 | $0.00-0.16$ |
| Water temperature | -0.16 | $-0.31--0.01$ |
| Angler 2 | -1.14 | $-2.85-0.57$ |
| Angler 3 | -1.33 | $-2.47--0.19$ |
| Angler 4 | 0.49 | $-0.56-1.53$ |
| Stream | 0.32 | $-0.53-1.16$ |
|  | Model: length + angler + stream |  |
| Intercept | -1.99 | $-4.12-0.13$ |
| Fish length | 0.08 | $0.01-0.16$ |
| Angler 2 | -1.13 | $-2.77-0.51$ |
| Angler 3 | -1.17 | $-2.25--0.09$ |
| Angler 4 | 0.64 | $-0.35-1.63$ |
| Stream | 0.32 | $-0.41-1.05$ |
| Model: length + temperature + length $\times$ temperature + angler + stream |  |  |
| Intercept | 6.78 | $-7.19-20.74$ |
| Fish length | -0.14 | $-0.67-0.40$ |
| Water temperature | -0.46 | $-1.20-0.28$ |
| fish length * water temperature | 0.01 | $-0.02-0.04$ |
| Angler 2 | -1.02 | $-2.73-0.70$ |
| Angler 3 | -1.30 | $-2.43--0.17$ |
| Angler 4 | 0.43 | $-0.62-1.49$ |
| Stream | 0.32 | $-0.51-1.15$ |
|  |  |  |
| Intercept |  |  |
| Water temperature |  |  |
| Stream |  |  |

Table 17. Comparison of linear regression models constructed to evaluate catch rates of trout in relation to elevated summer water temperatures in eastern Idaho streams. Estimates of log-likelihood, Akaike's information criteria (AICc), change in AICc ( $\triangle \mathrm{AICc}$ ), and AICc weights (wi) were used to assess models plausible models.

| Model | Log <br> likelihood | AIC $_{\mathbf{c}}$ | $\boldsymbol{\Delta A I C}_{\mathbf{c}}$ | $\boldsymbol{w}_{\boldsymbol{i}}$ |
| :--- | :---: | :---: | :---: | :---: |
| Temperature + stream | 640.10 | 648.49 | 0.00 | 0.34 |
| Null (stream only) | 642.73 | 648.97 | 0.48 | 0.27 |
| Temperature + angler + stream | 634.29 | 649.43 | 0.94 | 0.22 |
| Angler + stream | 637.18 | 650.03 | 1.54 | 0.16 |
| Temperature + angler + temperature*angler + stream | 633.53 | 655.84 | 7.35 | 0.01 |

Table 18. Coefficient estimates and $90 \%$ confidence intervals (CIs) for the most plausible models constructed to evaluate catch rates of trout in relation to elevated summer water temperatures in eastern Idaho streams. All parameters were fixed effects except stream, which was a random effect included in all models.

| Coefficient | Estimate <br> Model: temperature + stream | $\mathbf{9 0 \% ~ C I}$ |
| :--- | :---: | :---: |
| Intercept | 10.88 | $4.49-17.27$ |
| Water temperature | -0.32 | $-0.64-0.01$ |
| Stream | 4.12 | $-2.58-10.81$ |
|  | Model: null (stream only) |  |
| Intercept | 4.92 | $3.19-6.65$ |
| Stream | 3.19 | $-2.21-8.58$ |
|  | Model: temperature + angler + stream |  |
| Intercept | 12.07 | $5.72-18.41$ |
| Water temperature | -0.32 | $-0.64--0.01$ |
| Angler 2 | -3.30 | $-6.52--0.08$ |
| Angler 3 | -2.57 | $-4.81--0.34$ |
| Angler 4 | -1.05 | $-3.20-1.09$ |
| Stream | 3.61 | $-2.60-9.81$ |
|  | Model: angler + stream |  |
| Intercept | 5.90 | $4.00-7.79$ |
| Angler 2 | -3.45 | $-6.73--0.18$ |
| Angler 3 | -2.41 | $-4.66--0.15$ |
| Angler 4 | -0.88 | $-3.03-1.28$ |
| Stream | 2.85 | $-2.25-7.96$ |

Table 19. Summer (June-August) water temperature data from several of Idaho's most popular wild trout fisheries. Data sources include Idaho Department of Fish and Game (IDFG), United States Forest Service (USFS), United States Geological Survey (USGS), Silver Creek Alliance, Idaho Department of Environmental Quality (IDEQ), Henrys Fork Foundation (HFF), and Friends of the Teton River (FTR).

| Water temperature grand totals: |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Location |  |  | $>23^{\circ} \mathrm{C}$ |  | Missing data: |  |  |  |
| Waterbody | Lat. | Long. | Years of data | Days | Hours | Hourly records | Hours | $\begin{aligned} & \text { Likely } \\ & >23^{\circ} \mathrm{C} \end{aligned}$ | Data source |
| NF Coeur d'Alene River | 47.614 | -116.237 | 2016-20171 | 0 | 0 | 3,312 | 0 | - | IDFG |
| NF Coeur d'Alene River | 47.861 | -116.105 | 2016-2017 ${ }^{1}$ | 0 | 0 | 3,312 | 0 | - | IDFG |
| St Joe River | 47.323 | -116.292 | 2016-2017 ${ }^{1}$ | 4 | 16 | 3,294 | 0 | - | IDFG |
| NF Clearwater River | 46.841 | -115.621 | 2017-2021 | 0 | 0 | 11,035 | 5 | No | USGS |
| Lochsa River | 46.144 | -115.598 | 2020-2021 | 42 | 265 | 4,416 | 0 | - | IDFG |
| Selway River | 46.078 | -115.395 | 2016-2020 | 4 | 17 | 11,039 | 1 | No | IDFG |
| Selway River | 45.702 | -114.717 | 2017-2021 | 0 | 0 | 8,832 | 0 | - | IDFG |
| MF Salmon River | 45.296 | -114.595 | 2014 | 0 | 0 | 1,320 | 0 | - | USFS |
| MF Salmon River | 44.891 | -114.723 | 2014 | 0 | 0 | 1,488 | 0 | - | USFS |
| MF Salmon River | 44.766 | -115.095 | 2014 | 0 | 0 | 1,488 | 0 | - | USFS |
| MF Salmon River | 44.532 | -115.293 | 2014 | 0 | 0 | 1,488 | 0 | - | USFS |
| SF Boise River | 43.550 | -115.722 | 2016-2021 | 0 | 0 | 13,235 | 13 | No | USGS |
| Big Wood River | 43.329 | -114.319 | 2014 | 10 | 25 | 2,208 | 0 | - | USGS |
| Big Wood River | 43.517 | -114.322 | 2014 | 0 | 0 | 2,208 | 0 | - | USGS |
| Big Wood River | 43.786 | -114.425 | 2014 | 0 | 0 | 2,208 | 0 | - | USGS |
| Silver Creek | 43.236 | -113.986 | 2016-2020 | 142 | 888 | 8,832 | 0 | - | Silver Creek Alliance |
| Silver Creek | 43.284 | -114.008 | 2016-2020 | 102 | 602 | 11,040 | 0 | - | Silver Creek Alliance |
| Silver Creek | 43.324 | -114.108 | 2016-2021 | 20 | 96 | 13,203 | 45 | No | USGS |
| Silver Creek | 43.317 | -114.106 | 2016-2020 | 1 | 1 | 10,985 | 0 | - | Silver Creek Alliance |
| Silver Creek | 43.320 | -114.141 | 2016-2020 | 0 | 0 | 11,040 | 0 | - | Silver Creek Alliance |

Table 19. Continued


Table 19. Continued

## Water temperature grand totals

|  | Location |  | $>23^{\circ} \mathrm{C}$ |  |  | Missing data |  |  | Data source |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Waterbody | Lat. | Long. | Years of data | Days | Hours | Hourly records | Hours | $\begin{aligned} & \text { Likely } \\ & >23^{\circ} \mathrm{C} \end{aligned}$ |  |
| Teton River | 43.640 | -111.175 | $\begin{gathered} 2018,19, \& \\ 21^{1} \end{gathered}$ | 0 | 0 | 4,968 | 0 | - | FTR |
| Teton River | 43.696 | -111.165 | 2016-2021 ${ }^{1}$ | 0 | 0 | 9,936 | 0 | - | FTR |
|  |  |  |  | 777 | 4,226 | 325,999 | 7,978 |  |  |

[^0]FIGURES


Figure 11. Instantaneous water temperatures at the time that Cutthroat Trout were landed and tagged by fly anglers in eastern Idaho streams. Each symbol type depicts data at one stream, with any change in symbol color indicating different days at that same stream.


Figure 12. Recapture rate of Cutthroat Trout landed and marked by anglers, and angler catch rate of trout, in relation to summer instantaneous water temperature at the time that fish were landed. Sample size is provided inside the bars.

## ANNUAL PROGRESS REPORT

# SUBPROJECT \#5: TRENDS IN THE OCCUPANCY AND ABUNDANCE OF BONNEVILLE CUTTHROAT TROUT AND NONNATIVE TROUT IN THE BEAR RIVER BASIN IN IDAHO 

State of: Idaho<br>Project No.: $\underline{3}$<br>Title: $\quad$ Wild Trout Evaluations<br>Subproject \#5: Trends on the occupancy and abundance of Bonneville Cutthroat Trout and nonnative trout in the Bear River Basin in Idaho


#### Abstract

Bonneville Cutthroat Trout have experienced substantial declines in their historical distribution and abundance, and recent status assessments have noted a particular lack of information on trends in abundance for the species. From 1993 to 2020, a total of 184 backpack electrofishing surveys were conducted across 34 index reaches to monitor abundance of Bonneville Cutthroat Trout and nonnative salmonids in southeastern Idaho streams. Trout abundance (all species combined) averaged 7.6 fish $/ 100 \mathrm{~m}^{2}$ of stream. Bonneville Cutthroat Trout population growth rate ( $\lambda$ ) was generally stable through time (mean $\lambda=1.04$ across all reaches), whereas for nonnative trout considered collectively, estimates of $\lambda$ in general were declining over the entire study period. While the abundance of Bonneville Cutthroat Trout was negatively related to the abundance of nonnative trout for surveys in which both were captured, estimates of $\lambda$ for Cutthroat Trout were not related to the abundance of nonnative trout. Bonneville Cutthroat Trout $\lambda$ was also unrelated to all the reach-scale environmental conditions we measured except for conductivity, which was positively associated with $\lambda$. While conductivity is normally associated with the productivity of a water body, is also correlated to other important cations and anions (e.g., alkalinity and water hardness) that can influence fish populations in a number of ways, thus we cannot ascertain whether the relationship we observed was causative or correlative. We observed that Bonneville Cutthroat Trout abundance was higher in years when winter discharge and summer discharge was higher the prior year, which concurs with a large body of literature demonstrating that reduced baseflow during winter or summer can adversely impact salmonid recruitment, food resources, predatory avoidance, survival, or stream habitat conditions.


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## INTRODUCTION

Cutthroat Trout that occupy the Bear River drainage of southeastern Idaho and northern Utah are taxonomically designated as Bonneville Cutthroat Trout Oncorhynchus clarkii utah (Behnke 2002). However, recent investigations highlight the fact that they actually share a phylogenetic relationship with Yellowstone Cutthroat Trout O. clarkii bouvieri of the Snake River basin and Bonneville Cutthroat Trout of the Bonneville basin (e.g., Campbell et al. 2011, 2018; Loxterman and Keeley 2012). The shared phylogeny reflects the historical hydrologic connection between the Bear River and Snake River drainages (Martin et al. 1985; Smith et al. 2002) as well as periods of Bear River hydrologic isolation from the Bonneville Basin (Bouchard et al. 1998). Preserving the unique and diverse genetic, morphologic, and life history characteristics of Cutthroat Trout in the Bear River basin has been prioritized in several management plans (e.g., UDWR 2018; IDFG 2022).

As with nearly all salmonids, Bonneville Cutthroat Trout have experienced substantial declines in their historical distribution and abundance, due primarily to habitat loss and fragmentation as well as hybridization and competition with introduced nonnative salmonids (Duff 1988; Hepworth et al. 1997; McHugh and Budy 2006). Such declines were the basis of petitions made in 1998 for listing Bonneville Cutthroat Trout as threatened under the Endangered Species Act, though their listing was deemed not warranted in 2001 (USFWS 2001) and again (after a lawsuit) in 2008 (USFWS 2008). Nevertheless, it is estimated that Bonneville Cutthroat Trout currently occupy only $39 \%$ of their historical distributional range (UDWR 2018); in the Idaho portion of the Bear River basin, current occupancy is estimated to be $54 \%$ of their historical range (IDFG 2022).

Recent status assessments have noted a particular lack of information on trends in abundance for Bonneville Cutthroat Trout (Budy et al. 2007; IDFG 2022). To our knowledge, longterm trends have only been reported for southern Utah, where from the 1970s to 1990s, Bonneville Cutthroat Trout were estimated to occupy only 57 km , with abundance increasing in some streams and declining in others (Hepworth et al. 1997). Without a more thorough and contemporary understanding of trends in population abundance throughout their range, inferences regarding long-term population persistence cannot be made for the species. The primary objective of the present study was to estimate trends in population abundance for Bonneville Cutthroat Trout in the Idaho portion of their range. A secondary objective was to gain a better understanding of what factors might be influencing the status of Bonneville Cutthroat Trout in Idaho by relating several biotic and abiotic conditions to their abundance and trends in abundance.

## OBJECTIVES

1. Estimate trends in population abundance for Bonneville Cutthroat Trout in the Idaho portion of their range.
2. Evaluate what factors are associated with the status of Bonneville Cutthroat Trout in Idaho by relating several biotic and abiotic conditions to Bonneville Cutthroat Trout abundance and trends in abundance.

## METHODS

The upper Bear River basin originates in the Uinta Mountains in northeastern Utah, flows north into Wyoming before turning west into Idaho, and eventually turns back south, flowing back into Utah. The Bear River basin in Idaho can be characterized as a high desert region of the Intermountain West with streams that range from 1,300 to $2,500 \mathrm{~m}$ in elevation. Riparian vegetation at lower elevation generally consists of native grasses as well as dogwood Cornus spp., alder Alnus spp., willow Salix spp., and cottonwood Populus spp., whereas at higher elevation, riparian areas also include mixed conifers.

Besides Bonneville Cutthroat Trout, other salmonids occupying streams in the study area included nonnative Brown Trout Salmo trutta, Brook Trout Salvelinus fontinalis, Rainbow Trout $O$. mykiss, and cutthroat $\times$ rainbow hybrids (hereafter hybrids). Bonneville Cutthroat Trout can be readily (though not perfectly) differentiated from Rainbow Trout and hybrids using the phenotypic characteristics outlined in Meyer et al. (2022). In short, fish were considered to be Bonneville Cutthroat Trout when they had (1) few spots on top of the head, (2) no white leading edge on the pelvic, dorsal, or anal fins, (3) spots on the body that were large and concentrated posteriorly and dorsally, and (4) a strong or at least a faint throat slash. Rainbow Trout and hybrids were considered one taxa in the present study and were visually identified by some combination of the presence of white edges on the pelvic, dorsal, or anal fins, smaller spots evenly distributed throughout the body, many spots on the top of the head, or absence of a throat slash.

## Fish sampling

From 1993 to 2020, 34 trend-monitoring reaches in 16 different Bear River tributaries (Table 20; Figure 13) were repeatedly sampled to assess salmonid occurrence and abundance. These index reaches were established in streams known to contain Bonneville Cutthroat Trout. They were not drawn from a probability-based design, but rather, they were established near roads, bridges, culverts, or other access points. Latitude and longitude were determined at the lower end of each reach using a Global Positioning System (GPS). Once established, GPS units were used to relocate the lower ends of each index reach prior to each new survey. Reach length sampled by field crews was determined with a tape measure and varied from 34 to 815 m , but average reach length was 120 m and $>90 \%$ of the reaches were between 70 and 130 m in length.

Fish were sampled with backpack electrofishing units using pulsed DC, with output generally at 60 Hz and $25 \%$ duty cycle; voltage ranged from $200-800 \mathrm{~V}$ depending on fish response to the electric field. Captured fish were identified to species and measured for total length. Nongame species that were encountered were not enumerated.

Fish abundance was estimated based on either single-pass or multi-pass backpack electrofishing depletions. For multi-pass depletions, trout abundance was estimated using the maximum-likelihood model in the MicroFish software package (Van Deventer 1989). If no trout were captured on the second pass, we considered the catch on the first pass to be the estimated abundance. Using data from all 128 multi-pass depletion surveys that were conducted across all years, we developed a linear relationship (with the origin through zero) between the numbers of trout captured in first passes and the subsequent maximum-likelihood abundance estimates ( $F=$ 2877.3; $P<0.001 r^{2}=0.88$ ). From this relationship, we then predicted trout abundance for 58 additional surveys in which only a single removal pass was conducted (cf. Lobón-Cerviá et al. 1994; Kruse et al. 1998). Abundance was standardized to fish/100 m² of stream surveyed.

The length of age 0 fish was inconsistent across reaches and among species, and age 0 fish were difficult to sample effectively; therefore, we did not include fish $<100 \mathrm{~mm}$ in any of our estimates of trout abundance. Furthermore, separating abundance estimates for each species was often not possible because low abundance or limited catch precluded such partitioning at some index reaches. Therefore, in order to maintain consistency in methodology across reaches and time periods, all trout species were pooled for an overall estimate of trout abundance at the reach scale (e.g., Mullner et al. 1998; Isaak and Hubert 2004; Carrier et al. 2009), and point estimates for each species were calculated based on the proportion of the catch that each species comprised (cf. Meyer and High 2011).

## Estimating population growth rates

To estimate trends in fish abundance at individual reaches, we used linear regression with sample year as the independent variable and loge transformations of trout abundance as the dependent variable. Because the natural logarithm is undefined for zero, we added 0.1 fish/100 $\mathrm{m}^{2}$ to each estimate of abundance. The slope of the regression line is equivalent to the intrinsic rate of change (r) for the population (Gerrodette 1987; Morris and Doak 2002); this approach to monitoring trend assumes that the population changes in an exponential manner and that the rate of population change is constant over the sampling period. Confidence intervals (Cls) around the slope estimates were obtained from the linear regression models. Each estimate of $r$ was exponentiated to convert estimates to population growth rate ( $\lambda$ ).

Estimates of $\lambda$ were calculated for Bonneville Cutthroat Trout at each reach because they occupied every reach. Because all of the nonnative trout in the study area-Brook Trout, Brown Trout, and Rainbow Trout-have been previously demonstrated to have a negative effect on Cutthroat Trout (Dunham et al. 2002; McHugh and Budy 2006; Seiler and Keeley 2009), but which nonnative trout were present varied through time and among reaches, we grouped the abundance of all nonnative trout together to estimate $\lambda$ for nonnative trout where they occurred. Estimates of $\lambda$ with $90 \%$ Cls that overlapped unity (i.e., 1.00 ) were assumed to be stable populations, whereas those populations with $\lambda<1.00$ or $>1.00$ were assumed to be declining or increasing in abundance, respectively. We used a significance level of $\alpha=0.10$ for individual estimates of $\lambda$ and for the overall mean in order to increase the power of detecting trends in population abundance (Peterman 1990; Maxell 1999; Dauwalter et al. 2009).

## Relating reach-scale stream conditions to population growth

To assess whether population growth rate at each index reach was associated with various stream conditions at that reach, we treated each reach as the sample unit, and related several predictor variables to $\lambda$ using multiple linear regression. Elevation (often a surrogate for stream temperature: Isaak et al. 2010; Wenger et al. 2011; Eby et al. 2014), wetted width, and stream gradient can influence nonnative salmonid invasion success, mediate competitive interactions among salmonids, and explain partitioning of salmonids along stream networks (e.g., Fausch 1989; Bozek and Hubert 1992; Rahel and Nibbelink 1999; Peterson et al. 2004; Torgersen et al. 2006). Elevation (m) was determined from digital U.S. Geological Survey (USGS) 1:24,000scale topographic maps based on GPS-acquired latitude/longitude coordinates obtained in the field at the lower end of the reach. Mean wetted width ( m ) was calculated from the average of 10 transects spaced equally throughout each reach. Gradient (\%) was determined using the same digital topographic maps; the distance ( m ) between the two contour lines that bounded the study site latitude/longitude coordinates was traced, and gradient was calculated as the elevational increment between those contours divided by the traced distance (converted to a percentage).

Reaches averaged 1,918 m in elevation (range 1,478 to 2,438 m), 2.3\% in channel gradient ( 0.1 to $5.6 \%$ ), and 3.2 m in wetted width ( 0.9 to 8.1 m ; Table 1).

Using the GIS model constructed by Olson and Cormier (2019), conductivity was estimated for each index reach and was included in our analyses as a measure of stream productivity (McFadden and Cooper 1962; Scarnecchia and Bergersen 1987). Road density was included because western native trout are usually negatively impacted by roads that are near streams (Eaglin and Hubert 1993, Valdal and Quinn 2011). The 2019 Topologically Integrated Geographic Encoding and Referencing (TIGER) database (United States Census Bureau 2019) was used to map all the roads in Idaho, and road density was estimated by summing the total length of road within a $1.78-\mathrm{km}$ radius (i.e., a $10-\mathrm{km}^{2}$ area) of each index reach. We assumed that conductivity and road density at present was representative of those characteristics throughout the study period. A final predictor variable included the mean abundance of nonnative trout at the reach (across all surveys), which was $\log _{\mathrm{e}}$ transformed.

## Relating broad-scale factors to population abundance

In addition to the reach-scale evaluation just described, we also used multiple linear regression to assess whether annual Bonneville Cutthroat Trout abundance across the landscape was influenced by broad-scale bioclimatic predictor variables, including those representing stream flow, thermal regime, and drought. For this analysis, we treated each year as the sample unit.

Stream flow was included as a predictor variable because it is important for all life stages of stream-dwelling salmonids, including migration, spawning, and rearing (reviewed in Bjornn and Reiser 1991), and because both summer and winter stream flow can affect salmonid abundance (Bell et al. 2000; Mitro et al. 2003; Kanno et al. 2016). To characterize annual stream flow across the entire study area, we used mean daily discharge ( $\mathrm{m}^{3} / \mathrm{s}$ ) from three U.S. Geological Survey (USGS) stream gaging stations that (1) bounded the study area, (2) had similar magnitude of daily and mean annual flow, (3) were located in smaller streams not subject to intense upstream water management, and (4) were highly correlated with each other (mean correlation coefficient [ $r$ ] between these stations for average daily discharge $=0.82$ ). The stations included the Logan River (USGS station 10109000), Blacksmith Fork (USGS station 10113500), and the Portneuf River (USGS station 13073000). We averaged the mean daily discharge from these three stations, from which mean summer (Jun-Aug) and mean winter (Dec-Feb) discharge were calculated for each year.

Temperature was included as a predictor variable because the severity of both summer and winter water temperatures can affect the survival and abundance of stream-dwelling salmonids (e.g., Jowett 1992; Isaak and Hubert 2004; Meyer et al. 2010). Long-term stream temperature data were generally lacking across the study area. However, air temperature is often strongly correlated to stream water temperature (Crisp and Howson 1982), and summer air temperature is often correlated to the distribution and abundance of salmonids in Rocky Mountain streams (e.g., Dunham et al. 1999; Rahel and Nibbelink 1999) and elsewhere (Kanno et al. 2016). We therefore used annual air temperature variation to index annual water temperature variation. Accordingly, mean daily air temperature data were obtained from the National Oceanic Atmospheric Administration's (NOAA) Global Historical Climatology Network for three stations that bounded the Bear River basin in Idaho (Emigrant Summit, station USS0011G06S; Franklin Basin, station USS0011G32S; and Giveout, station USS0011G33S). We averaged the mean daily values from these three stations, from which mean summer (Jun-Aug) and mean winter (DecFeb) air temperatures were calculated for each year.

While stream flow and water temperature are experienced directly by salmonids in lotic habitats, drought can have a more nuanced impact on stream-dwelling salmonids. For instance, although drought may directly affect stream flow and water temperature, it may also indirectly influence stream-dwelling salmonids by impacting conditions such as riparian vegetation, fire, bank stability, and food resources (Zong et al. 1996; Dwire and Kauffman 2003; Boulton 2003; Garssen et al. 2014). Consequently, drought is often associated with fluctuations in the abundance of stream-dwelling salmonids (Elliott et al. 1997; Hakala and Hartman 2004; Meyer et al. 2014), including Bonneville Cutthroat Trout (White and Rahel 2008).

To characterize an annual drought index for the study area, estimates of the Palmer Drought Severity Index (PDSI) were obtained from NOAA's National Center for Environmental Information for the Southeast Division of Idaho. The PDSI is a monthly measure of dryness that is based on recent moisture supply, soil characteristics, and evapotranspiration (Palmer 1965). Negative PDSI values of 0 to -0.5 are normal, -0.5 to -1 indicate incipient drought, -1 to -2 indicate mild drought, -2 to -3 moderate drought, -3 to -4 indicate severe drought, and less than -4 indicate extreme drought. Positive PDSI values follow a similar qualitative categorization for wet weather. We averaged the 12 monthly values to compute a mean PDSI for each year.

To characterize annual variation in Bonneville Cutthroat Trout abundance, estimates for all sampling events at a reach were normalized to a z-score based on the mean abundance at the reach across all sampling periods, so that each reach had a mean abundance z-score of zero and a standard deviation of one. Normalizing the Cutthroat Trout abundance data had the effect of making all reaches contribute equally to the abundance-bioclimate relationships rather than hinging more heavily on the reaches with the highest abundance. For each year of fish sampling, we estimated a mean z-score for all reaches surveyed in that year. Since we surveyed fish abundance in 17 separate years, this gave us a sample size of 17 for this analysis. Because broad-scale bioclimatic conditions such as stream flow, temperature, and drought are likely to affect recruitment or have other delayed impacts that outweigh effects on within-year abundance (e.g., Bell et al. 2000; Copeland and Meyer 2011), we related bioclimatic conditions to Bonneville Cutthroat Trout abundance at a one-year time lag.

## Data analyses

Using simple linear regression, we assessed whether the abundance of Bonneville Cutthroat Trout was negatively associated with the abundance of nonnative trout. The sample unit for this analysis was each survey in which both taxa were captured.

For both the reach-scale and broad-scale modeling exercises described above, we considered all combinations of predictor variables during model construction, but interaction terms were not considered due to small sample size for both data sets. Models were ranked using Akaike's information criterion corrected for small sample size ( $\mathrm{AIC}_{\mathrm{c}}$; Burnham and Anderson 2002), and we considered the most plausible models to be those with AIC $_{c}$ scores within 2.0 of the best model (Burnham and Anderson 2004). AIC $_{c}$ weights ( $w_{i}$ ) were used to assess the relative plausibility of each of the most plausible models, and coefficients of determination ( $r^{2}$ ) or adjusted $r^{2}$ (for models with more than one predictor variable) were used to describe the amount of the variation in CPUE explained by the parameters in the models. Coefficient estimates are reported only for the most plausible models, and only those coefficients with $95 \%$ Cls that did not overlap zero were considered influential in a model, regardless of their inclusion. All analyses were conducted using the SAS statistical software package (SAS Institute 2009).

## RESULTS

Bonneville Cutthroat Trout >100 mm TL were captured during 171 of the 186 electrofishing surveys conducted. At three index reaches, Bonneville Cutthroat Trout were present during the initial survey but absent during the final survey, but there were also three reaches where they were absent during the initial survey (though they were known to be present in the stream) but present during the final survey (Table 21).

Nonnative trout were captured during 80 surveys and occurred at 20 of the 34 index reaches. Rainbow Trout were the most common nonnative salmonid encountered (captured in 43 surveys at 16 reaches), followed by Brook Trout ( 35 surveys at 8 reaches), and Brown Trout (26 surveys at 5 reaches). At 11 of the 34 reaches, at least one nonnative trout either appeared at or disappeared from the reach from the beginning to the end of the time period, and all three nonnative species experienced appearance and disappearance at one or more index reaches (Table 21).

Trout abundance (all species combined) averaged 7.6 fish $/ 100 \mathrm{~m}^{2}$ of stream and ranged from a low of zero on one occasion to a high of 29.2 fish $/ 100 \mathrm{~m}^{2}$. Bonneville Cutthroat Trout abundance averaged 5.6 fish $/ 100 \mathrm{~m}^{2}$ (or $160 / \mathrm{km}$ ) and ranged from 0 to 29.2 fish $/ 100 \mathrm{~m}^{2}$ ( 0 to $810 / \mathrm{km}$ ). Bonneville Cutthroat Trout abundance was negatively related to the abundance of nonnative trout for surveys where they were both captured (Figure 14).

Across all 34 index reaches combined, mean $\lambda$ was 1.04 for Bonneville Cutthroat Trout, and $90 \%$ Cls overlapped unity ( $0.98-1.10$; Table 21). Within individual reaches, Bonneville Cutthroat Trout population growth rate was generally stable, with statistically significant declines in $\lambda$ at three reaches, statistically significant increases in $\lambda$ at three other locations, and stable estimates of $\lambda$ (i.e., non-significant changes) at the remaining reaches. In comparison, estimates of mean $\lambda$ for all reaches combined averaged 0.93 for nonnative trout, and $90 \%$ Cls did not overlap unity (0.89-0.97), suggesting that nonnative trout in general were declining in the long-term monitoring reaches over the entire study period. However, few estimates of $\lambda$ were statistically significantly declining at individual reaches (Table 21).

All of the plausible models relating reach-scale stream conditions to estimates of $\lambda$ at each reach included conductivity (Table 22), and none of the coefficient estimate $95 \% \mathrm{Cls}$ included zero (Table 23); estimates indicated that Bonneville Cutthroat Trout population growth was higher at reaches with higher conductivity. All of the remaining stream conditions that we included in our analyses, including road density, nonnative trout density, wetted width, elevation, and stream gradient, appeared in some of the most plausible models (Table 22). However, in nearly all instances, the $95 \%$ Cls around these parameter estimates included zero (Table 23), indicating that none of the remaining variables were very influential in the models in which they appeared. These models explained 19-29\% of the variation we observed in Bonneville Cutthroat Trout estimates of $\lambda$ among index reaches (Table 22).

The mean annual z-scores of Bonneville Cutthroat Trout abundance at individual stream reaches were most strongly associated with annual variation in mean daily discharge at nearby USGS gaging stations the previous winter and the previous summer and were weakly associated with annual variation in nearby daily air temperatures the previous summer and previous winter and mean annual PDSI for southeast Idaho the previous year (Figure 15). The most parsimonious model relating bioclimatic factors to normalized Cutthroat Trout abundance included only winter discharge (Table 24), with the parameter estimate (and associated $95 \%$ CIs) indicating that Bonneville Cutthroat Trout abundance was generally higher in years when winter discharge was
higher the prior year (Table 25). There was also some support for two additional models, one including both winter and summer discharge, and the other including summer air temperature and discharge (Table 24). Based on parameter estimates and their 95\% Cls (Table 25), the secondbest model indicated that winter and summer flow the prior year did not influence annual variation in Cutthroat Trout abundance, whereas the third-best model indicated that Cutthroat Trout abundance was generally higher in years with higher summer discharge the prior year. The most plausible models (i.e., those with $\mathrm{AlC}_{\mathrm{c}}$ scores within 2.00 of the best model) explained from 24 to $31 \%$ of the annual variation we observed in normalized Bonneville Cutthroat Trout abundance across the landscape (Table 24).

## DISCUSSION

Bonneville Cutthroat Trout have unequivocally experienced a range-wide reduction in occupancy and abundance from historical levels, though much of this range contraction occurred decades ago due primarily to habitat alterations resulting from land use practices and the introduction of nonnative salmonids (Duff 1988). Our results suggest that in the last several decades, the distribution and abundance of Bonneville Cutthroat Trout at index reaches in southeastern Idaho are generally stable. Whether this is true in other portions of their range is unknown because additional published long-term trend data are lacking. Considering that Bonneville Cutthroat Trout occupy a higher proportion of their historical range in Idaho (54\%; IDFG 2022) than elsewhere, and that the Bear River basin is known to be a stronghold for Bonneville Cutthroat Trout (UDWR 2018), the index reaches in our study likely represent some of the best remaining lotic habitat for the species, and thus may not accurately represent trends in abundance across their range. Additional trend monitoring is clearly needed to better characterize the status of Bonneville Cutthroat Trout at a broader scale.

Bonneville Cutthroat Trout population growth rates were generally stable even at reaches where nonnative trout were present, and nonnative trout (taken collectively) showed declining population growth rates in the Bear River basin. This was unexpected, since all three nonnative trout generally have adverse impacts on Cutthroat Trout populations (e.g., Dunham et al. 2002; McHugh and Budy 2006; Seiler and Keeley 2009), although this effect is not ubiquitous in all Cutthroat Trout populations (e.g., Meyer et al. 2014). We found nearly twice as many index reaches experienced changes (either contractions or expansions) in the occupancy of at least one nonnative species $(n=13)$ as stability in their occupancy $(n=7)$. This concurs with a recent study in western Montana, which revealed that Brook Trout, Brown Trout, and Rainbow Trout were all undergoing long-term contractions and expansions in some watersheds (Bell et al. 2021). The temporal stability of stream fish assemblages varies dramatically among taxa and ecosystems but is generally thought to be driven by variation in density-dependent and density independent factors (Gido and Jackson 2010). Although Bonneville Cutthroat Trout trends in abundance were as stable at reaches where they were sympatric with nonnative trout as they were in allopatric reaches, nonnative trout abundance was nevertheless negatively associated with Bonneville Cutthroat Trout abundance in the surveys in which both were encountered. Despite the indication of some population resilience by Bonneville Cutthroat Trout to the presence of nonnative trout, the nearly ubiquitous negative relationship nonnative trout have on native trout (Krueger and May 1991; Buoro et al. 2016) suggests that management actions designed to curtail the spread or abundance of nonnative trout may eventually be needed for the long-term persistence of Bonneville Cutthroat Trout in Idaho.

Our results suggest that reduced baseflow in summer or winter months may have an adverse impact on Bonneville Cutthroat Trout abundance the following year. Considering that age

0 fish in one year were large enough the following year to be included in our abundance estimates, and they would likely have constituted the most abundant age class in most instances, the negative relationship between summer or winter baseflow levels and Bonneville Cutthroat Trout abundance is perhaps the result of poor survival or production of age 0 fish during low-flow years (Jespersen et al. 2021). Alternatively, reduced baseflow may have negative impacts on multiple age classes (Elliott et al. 1997; Hakala and Hartman 2004). Such an effect of reduced summer or winter baseflow on the abundance of age 0 fish or all age classes could be the result of: 1) reduced reproductive success (Elliott et al. 1997); 2) reduced habitat quality and availability (Hakala and Hartman 2004); 3) diminished food resources (Cowx et al. 1984); 4) intensified predation as subadults are forced into closer proximity to predators because of less available space (Larimore et al. 1959); and 5) lower winter flows, which may reduce overwinter habitat, and ultimately, survival of age 0 trout (Hakala and Hartman 2004) or older age classes (Meyer and Gregory 2000). Regardless of the mechanism(s), the negative effects of reduced streamflow on Cutthroat Trout abundance observed here portends that if climate change continues to diminish baseflow conditions in streams across the west (Luce and Holden 2009), the likelihood of long-term persistence for many Bonneville Cutthroat Trout populations in Idaho may be reduced.

Population growth rates for vertebrate species are clearly affected by density dependent processes (Morris and Doak 2002), but estimates of $\lambda$ for stream-dwelling trout populations have rarely if ever been directly linked to other biotic or abiotic stream conditions. Of the factors we investigated, only conductivity appeared to influence estimates of $\lambda$ for Bonneville Cutthroat Trout. Conductivity is normally associated with the productivity of a water body (Rawson 1951; Welch 1952) and has been previously shown to be positively associated with trout abundance in streams (e.g., McFadden and Cooper 1962; Scarnecchia and Bergersen 1987). In the present study, conductivity was $240-580 \mu \mathrm{~S} / \mathrm{cm}$, which is moderate to high for flowing waters in western North America (Griffith 2014). Conductivity is also correlated to other important cations and anions (e.g., alkalinity and water hardness) that can influence fish populations in a number of ways (Scarnecchia and Bergersen 1987), so we cannot ascertain whether the relationship we observed was causative or correlative. Although road density, elevation, gradient, stream size, and nonnative trout density were not important predictors of Bonneville Cutthroat Trout population growth, it should be noted that limiting factor analysis is notoriously challenging because such biotic and abiotic conditions can interact in complex ways to affect animal populations (Cade et al. 1999; Townsend et al. 2008). Nevertheless, continued monitoring of these and other Bonneville Cutthroat Trout populations should include limiting factor analysis whenever feasible to reveal environmental conditions that could be targeted by management or conservation activities.

We expected that drought conditions might adversely affect Bonneville Cutthroat Trout abundance in the study area, but we observed no such effect. In general, drought reduces the volume and complexity of stream habitat, resulting in diminished food resources (Cowx et al. 1984), reduced reproductive success (Elliott et al. 1997; White and Rahel 2008), shifts in species assemblages (Matthews and Marsh-Matthews 2003), and increased predation (Larimore et al. 1959). Not surprisingly, drought conditions have repeatedly been shown to negatively affect cutthroat trout populations (Dunham et al. 1999; White and Rahel 2008; Gresswell 2011; Meyer et al. 2014). However, Bonneville Cutthroat Trout are closely associated with headwater habitats (Kershner 1995), which are typically more stochastic in nature (Richardson et al. 2005) and less prone to climate-altered conditions (Isaak et al. 2016) than downstream reaches. As such, the headwater stream reaches we included in our study may have been less likely to be influenced by drought.

The primary limitation in our study was that sites selected for long-term monitoring were not originally drawn at random. Consequently, our results may not accurately depict trends in the distribution and abundance of Bonneville Cutthroat Trout and nonnative trout in streams within the Bear River and Bear Lake tributaries in Idaho that were not sampled. Despite the well-known importance of random sampling to ensure that ecological observations are drawn from the population of interest (Garton et al. 2012), it is common in long-trend monitoring programs tracking changes in stream-dwelling salmonid populations to utilize data from index reaches that were established in a nonprobabilistic manner (e.g., Gowan and Fausch 1996; Ham and Pearsons 2000; Cook et al. 2010). Courbois et al. (2008) highlight the importance of such index reaches because the temporal extent of the data allows examination of long-term population dynamics that would otherwise be unattainable. Nevertheless, we recommend that future efforts combine these index reaches with additional sites drawn probabilistically to augment the rigor of the current study design.

Nonprobabilistic sampling is not the only limitation of our study. A second shortcoming was our reliance on surrogate data for stream temperature (using elevation and air temperature) and stream flow (using nearby stream gages on larger nearby rivers). Using surrogates rather than direct field measurements for stream temperature and flow are common in fish-stream habitat studies (e.g., Dunham et al. 1999; Rahel and Nibbelink 1999; Kanno et al. 2016) because long-term water temperature and stream flow data are rarely available in headwater streams, but they are not always effective proxies (Isaak et al. 2016). Third, we used geospatial covariates to characterize reach conductivity and road density, and we assumed these conditions were relatively stable throughout the study. This assumption is supported for conductivity by Olson and Cormier (2019) who observed that conductivity, though not constant, was relatively stable through time. For road density, the correlation between point estimates from 2019 TIGER data at our study reaches compared to averaging point estimates from the beginning and end of the study (i.e., 2000 and 2019) was very high ( $r=0.96$ ). Fourth, sampling was conducted only at summer baseflows, but salmonid distribution and abundance inherently changes seasonally, thus sampling at other times of the year during baseflow conditions (e.g., late fall or winter) may have produced different results. Fifth, our estimates of cutthroat trout and rainbow trout and hybrid distribution and abundance may have been slightly biased because phenotype imperfectly differentiates these taxa; however, recent evidence suggests phenotype is quite accurate to differentiate these taxa (Meyer et al. 2022), so this source of bias is likely inconsequential to our general conclusions. Finally, none of the most plausible models we presented explained a large portion of the variation we observed in Bonneville Cutthroat Trout population growth rates or abundance, implying that other environmental or biological conditions not accounted for in our study (e.g., disease, land use activities, disturbance events) may have been important predictors.

Notwithstanding study limitations, our results suggest that Bonneville Cutthroat Trout are more stable in the Idaho portion of the Bear River basin than are nonnative salmonids. However, considering the inverse relationship we observed between summer and winter stream baseflow conditions in a given year and Bonneville Cutthroat Trout abundance the following year, the projection of further reductions in stream baseflow levels in western North America as the climate continues to warm (Luce and Holden 2009) is concerning. This is especially true in streams containing nonnative trout that may be better adapted to warmer streams (Shepard 2004; Peterson et al. 2004) or lower stream flows. Periodic revisitation of these long-term monitoring reaches would continue to provide valuable information on the status of Bonneville Cutthroat Trout in Idaho. Expansion of these monitoring reaches to include all areas occupied by Bonneville Cutthroat Trout (in Idaho and elsewhere) would help confirm or refute the more narrow conclusions that can be drawn from this trend-monitoring program.

## RECOMMENDATION

1. Revisit these long-term monitoring sites at least twice per decade to assess changes in the abundance of Bonneville Cutthroat Trout and nonnative trout in the Idaho portion of the Bear River basin.

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## TABLES

Table 20. Location and channel characteristics for 34 reaches sampled repeatedly with backpack electrofishing to determine trends in occupancy and abundance of salmonids in Bear River tributaries of southeast Idaho. Site numbers correspond to Figure 13.

|  |  |  |  | Wetted | Elevation | Reach |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: |
| Stream | Site | Latitude | Longitude | width $(\mathrm{m})$ | $(\mathrm{m})$ | slope $(\%)$ |
| Beaver Creek | 1 | 42.00668 | 111.5233 | 3.42 | 2342 | 1.6 |
| Beaver Creek | 2 | 42.04209 | 111.53921 | 3 | 2438 | 1.2 |
| Cottonwood Creek | 3 | 42.33583 | 111.78822 | 4.7 | 1593 | 2.8 |
| Cottonwood Creek | 4 | 42.36329 | 111.91115 | 4.7 | 1798 | 0.9 |
| Cottonwood Creek | 5 | 42.43579 | 111.91551 | 5.2 | 1950 | 2.3 |
| Dry Creek | 6 | 42.43843 | 111.08034 | 2 | 2016 | 2.2 |
| Dry Creek | 7 | 42.44483 | 111.09206 | 2 | 2058 | 3.6 |
| Eightmile Creek | 8 | 42.57513 | 111.55017 | 3.8 | 1822 | 0.7 |
| Eightmile Creek | 9 | 42.5321 | 111.57719 | 3.6 | 1900 | 1.8 |
| Eightmile Creek | 10 | 42.50363 | 111.57875 | 4.3 | 1976 | 2.2 |
| Giraffe Creek | 11 | 42.46874 | 111.05453 | 1.8 | 2183 | 2 |
| Giraffe Creek | 12 | 42.46919 | 111.06061 | 1.81 | 2190 | 2 |
| Hoopes Creek | 13 | 42.39604 | 111.76631 | 2.6 | 1585 | 5.1 |
| Kackley Spring | 14 | 42.53336 | 111.79376 | 3.2 | 1536 | 1.7 |
| Kackley Spring | 15 | 42.53363 | 111.79468 | 3.2 | 1535 | 1.7 |
| Logan River | 16 | 42.00854 | 111.59756 | 3.9 | 2349 | 2.5 |
| Logan River | 17 | 42.0014 | 111.59659 | 2.62 | 2319 | 2.8 |
| Maple Creek | 18 | 42.03643 | 111.75569 | 4 | 1478 | 1.8 |
| Maple Creek | 19 | 42.06861 | 111.69902 | 3.68 | 1791 | 5.6 |
| Montpelier Creek | 20 | 42.35642 | 111.21303 | 5.32 | 2055 | 4.3 |
| Montpelier Creek | 21 | 42.40182 | 111.17937 | 3.5 | 2024 | 1 |
| Preuss Creek | 22 | 42.4358 | 111.12568 | 1.79 | 2024 | 2.6 |
| Preuss Creek | 23 | 42.43858 | 111.12993 | 0.93 | 2031 | 1.3 |
| Preuss Creek | 24 | 42.45042 | 111.14856 | 1.37 | 2093 | 2.9 |
| Preuss Creek | 25 | 42.4563 | 111.1598 | 2.51 | 2130 | 2.2 |
| Preuss Creek | 26 | 42.46056 | 111.1657 | 2.32 | 2143 | 2.2 |
| Preuss Creek | 27 | 42.46647 | 111.17562 | 1.22 | 2185 | 3.2 |
| Stauffer Creek | 28 | 42.45095 | 111.41848 | 2.34 | 1800 | 0.1 |
| Stauffer Creek | 29 | 42.42092 | 111.44934 | 2.4 | 1866 | 2.3 |
| Stockton Creek | 30 | 42.31746 | 111.94935 | 2.51 | 1567 | 3.2 |
| Stockton Creek | 31 | 42.32958 | 111.91892 | 1.7 | 1664 | 3.1 |
| Trout Creek | 32 | 42.46549 | 111.66452 | 3.4 | 1645 | 4.7 |
| Whiskey Creek | 33 | 42.45533 | 111.7223 | 8.1 | 1565 | 0.5 |
| Whiskey Creek | 34 | 42.46587 | 111.70975 | 5.4 | 1575 | 1.1 |
|  |  |  |  |  |  |  |

Table 21. Mean abundance (with associated coefficient of variation [CV]) and population growth rates ( $\lambda$; with $90 \%$ lower and upper confidence intervals [CIs]) for Bonneville Cutthroat Trout and nonnative trout at 34 long-term monitoring reaches in Bear River tributaries of southeast Idaho. Nonnative trout species (spp.) included Brook Trout (BKT), Brown Trout (BNT), and Rainbow Trout and hybrids (RBT). Bold text highlights estimates in which Cls do not overlap zero. Arrows indicate where a species appeared at (up arrow) or disappeared from (down arrow) the reach over the study period. Site numbers correspond to Figure 13.

| Site Stream | Time period | Numbe of surveys | Bonneville Cutthroat Trout |  |  |  | Nonnative trout |  |  |  | Species present |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  | Fish/100m ${ }^{2}$ |  |  | $\lambda$ | Fish/10 | $00 \mathrm{~m}^{2}$ |  | $\lambda$ |  |
|  |  |  | Mean | CV | Est | 95\% Cl | Mean | CV | Est | 95\% Cl |  |
| 1 Beaver Creek | 2006-2017 | 6 | 3.62 | 0.80 | 0.95 | 0.77-1.16 | 1.19 | 1.47 | 0.72 | 0.53-0.96 | BCT, BKT $\uparrow$, RBT |
| 2 Beaver Creek | 2009-2017 | 5 | 1.81 | 1.30 | 1.22 | 0.66-2.28 | 5.24 | 0.49 | 0.95 | 0.81-1.11 | BCT, BKT, RBT $\downarrow$ |
| 3 Cottonwood Creek | 2006-2019 | 8 | 8.83 | 0.56 | 0.90 | 0.82-0.99 | 1.14 | 0.99 | 0.92 | 0.71-1.19 | BCT, RBT |
| 4 Cottonwood Creek | 2006-2019 | 6 | 1.79 | 0.48 | 0.96 | 0.88-1.04 | 0.05 | 2.45 | 0.97 | 0.87-1.09 | BCT, RBT |
| 5 Cottonwood Creek | 2011-2017 | 4 | 8.05 | 0.41 | 0.96 | 0.67-1.38 | 0.47 | 1.40 | 0.78 | 0.37-1.63 | BCT, RBT $\downarrow$ |
| 6 Dry Creek | 2008-2020 | 5 | 4.07 | 1.09 | 1.11 | 0.84-1.48 | - |  |  |  |  |
| 7 Dry Creek | 2012-2020 | 5 | 6.02 | 0.69 | 0.92 | 0.70-1.21 | - |  |  |  |  |
| 8 Eightmile Creek | 2010-2018 | 3 | 1.58 | 0.27 | 1.05 | 0.81-1.37 | 9.07 | 0.43 | 0.89 | 0.86-0.92 | BCT, BKT |
| 9 Eightmile Creek | 2006-2018 | 7 | 0.55 | 1.60 | 0.87 | 0.69-1.10 | 8.96 | 0.39 | 0.98 | 0.90-1.08 | BCT $\downarrow$, BKT |
| 10 Eightmile Creek | 2010-2020 | 6 | 1.03 | 0.60 | 1.18 | 0.92-1.52 | 17.5 | 0.47 | 0.98 | 0.80-1.19 | BCT, BKT, RBT $\uparrow$ |
| 11 Giraffe Creek | 2008-2020 | 6 | 14.5 | 0.29 | 0.99 | 0.92-1.06 | - |  |  |  |  |
| 12 Giraffe Creek | 2004-2020 | 7 | 14.3 | 0.47 | 1.08 | 1.02-1.15 | - |  |  |  |  |
| 13 Hoopes Creek | 2009-2019 | 5 | 2.73 | 0.87 | 1.13 | 0.92-1.38 | - |  |  |  |  |
| 14 Kackley Spring | 2009-2018 | 5 | 7.05 | 0.92 | 1.71 | 1.28-2.28 | 3.86 | 0.59 | 0.94 | 0.77-1.17 | BCT $\uparrow, \mathrm{BNT} \downarrow, \mathrm{RBT} \uparrow$ |
| 15 Kackley Spring | 2009-2018 | 6 | 5.46 | 0.53 | 1.58 | 1.20-2.08 | 4.53 | 0.40 | 0.95 | 0.86-1.07 | $\mathrm{BCT} \uparrow, \mathrm{BNT} \downarrow, \mathrm{RBT} \uparrow \uparrow$ |
| 16 Logan River | 2011-2019 | 3 | 9.66 | 0.25 | 1.04 | 0.80-1.34 | 0.11 | 1.73 | 0.82 | 0.60-1.12 | BCT, RBT $\downarrow$ |
| 17 Logan River | 2001-2019 | 7 | 4.2 | 0.56 | 0.98 | 0.89-1.09 | 0.28 | 1.89 | 0.99 | 0.83-1.17 | BCT, RBT |
| 18 Maple Creek | 2009-2019 | 6 | 7.47 | 0.49 | 0.88 | 0.80-0.97 | 1.77 | 1.02 | 1.14 | 0.71-1.82 | BCT, BNT $\uparrow$, RBT $\downarrow$ |
| 19 Maple Creek | 2006-2017 | 6 | 6.41 | 0.38 | 0.96 | 0.84-1.09 | 0.26 | 2.45 | 0.91 | 0.68-1.21 | BCT, RBT |
| 20 Montpelier Creek | 2006-2020 | 6 | 0.23 | 2.04 | 0.83 | 0.75-0.90 | 2.75 | 0.48 | 0.95 | 0.83-1.08 | BCT $\downarrow, \mathrm{BKT} \downarrow, \mathrm{BNT} \uparrow, \mathrm{RBT} \uparrow$ |
| 21 Montpelier Creek | 2006-2020 | 7 | 3.8 | 0.56 | 1.04 | 0.96-1.11 | 6.82 | 0.40 | 1.04 | 0.97-1.11 | $\mathrm{BCT}, \mathrm{BKT}, \mathrm{BNT} \uparrow, \mathrm{RBT}$ |
| 22 Preuss Creek | 1993-2010 | 4 | 5.68 | 0.69 | 1.06 | 0.94-1.20 | - |  |  |  |  |
| 23 Preuss Creek | 2004-2020 | 7 | 4.43 | 0.79 | 0.98 | 0.89-1.08 | - |  |  |  |  |
| 24 Preuss Creek | 1993-2020 | 6 | 7.92 | 0.76 | 0.93 | 0.77-1.11 | - |  |  |  |  |
| 25 Preuss Creek | 1993-2008 | 3 | 10.5 | 1.24 | 1.06 | 0.33-3.37 | - |  |  |  |  |
| 26 Preuss Creek | 1993-2008 | 3 | 9.91 | 0.76 | 1.16 | 0.81-1.67 | - |  |  |  |  |
| 27 Preuss Creek | 1993-2020 | 8 | 10.3 | 0.60 | 0.98 | 0.93-1.04 | - |  |  |  |  |
| 28 Stauffer Creek | 2012-2020 | 4 | 4.16 | 0.65 | 0.99 | 0.69-1.42 | - |  |  |  |  |
| 29 Stauffer Creek | 2012-2020 | 5 | 9.06 | 0.64 | 0.94 | 0.74-1.19 | - |  |  |  |  |
| 30 Stockton Creek | 2009-2019 | 7 | 2.9 | 0.71 | 1.06 | 0.94-1.19 | - |  |  |  |  |
| 31 Stockton Creek | 2010-2019 | 6 | 6.52 | 0.43 | 0.96 | 0.85-1.08 | 0.14 | 2.45 | 0.86 | 0.69-1.07 | BCT, RBT $\downarrow$ |
| 32 Trout Creek | 20112019 | 4 | 6.35 | 0.42 | 1.07 | 0.85-1.35 | 1.87 | 0.37 | 1.01 | 0.80-1.29 | BCT, BKT |
| 33 Whiskey Creek | 2011-2019 | 5 | 0.86 | 1.13 | 1.17 | 0.73-1.88 | 0.39 | 0.91 | 0.95 | 0.72-1.26 | BCT $\uparrow$, RBT |
| 34 Whiskey Creek | 2011-2019 | 5 | 0.36 | 1.28 | 0.81 | 0.56-1.19 | 0.12 | 0.74 | 0.89 | 0.66-1.20 | BCT $\downarrow$, RBT |

Table 22. Top models relating reach-scale conditions to Bonneville Cutthroat Trout population growth rate ( $\lambda$ ) at 34 long-term monitoring reaches in Bear River tributaries of southeast Idaho. Akaike's information criteria ( $\mathrm{AIC}_{\mathrm{c}}$ ), change in $\mathrm{AIC}_{\mathrm{c}}$ $\left(\triangle \mathrm{AIC}_{c}\right)$, and $\mathrm{AIC}_{c}$ weights $\left(w_{i}\right)$ were used to assess model plausibility, and coefficients of determination $\left(r^{2}\right)$ indicate the amount of variation explained in the models.

| Model | AICc | $\boldsymbol{\Delta A I C c}$ | $\boldsymbol{w}_{\boldsymbol{i}}$ | $\boldsymbol{r}^{\mathbf{2}}$ |
| :--- | :---: | :---: | :---: | :---: |
| Conductivity + road density | -121.54 | 0.00 | 0.10 | 0.24 |
| Conductivity + road density + Ln(nonnative trout density) + width | -121.20 | 0.35 | 0.08 | 0.29 |
| Conductivity + road density + Ln(nonnative trout density) | -120.51 | 1.04 | 0.06 | 0.24 |
| Conductivity + road density + elevation | -120.32 | 1.22 | 0.05 | 0.24 |
| Conductivity | -120.22 | 1.32 | 0.05 | 0.20 |
| Conductivity + road density + gradient | -120.19 | 1.36 | 0.05 | 0.23 |
| Conductivity + Ln(nonnative trout density) | -120.02 | 1.52 | 0.05 | 0.20 |
| Conductivity + road density + width | -119.83 | 1.71 | 0.04 | 0.23 |
| Conductivity + road density + Ln(nonnative trout density) + width | -119.77 | 1.78 | 0.04 | 0.29 |
| + gradient | -119.56 | 1.99 | 0.04 | 0.22 |
| Conductivity + Ln(nonnative trout density) + width | -119.50 | 2.04 | 0.03 | 0.19 |
| Conductivity + gradient |  |  |  |  |

Table 23. Parameter estimates and 95\% confidence intervals (Cls) for the top models relating reach-scale conditions to Bonneville Cutthroat Trout population growth rates $(\lambda)$ at 34 long-term monitoring reaches in Bear River tributaries of southeast Idaho.

| Parameter | Estimate | SE | 95\% Cls |
| :---: | :---: | :---: | :---: |
| Model 1 |  |  |  |
| Intercept | 0.53 | 0.16 | 0.22-0.85 |
| Conductivity | 0.0009 | 0.0004 | 0.0002-0.0017 |
| Road density | 0.011 | 0.006 | -0.001-0.022 |
| Model 2 |  |  |  |
| Intercept | 0.70 | 0.17 | 0.36-1.04 |
| Conductivity | 0.0009 | 0.0004 | 0.0002-0.0016 |
| Road density | 0.011 | 0.006 | -0.001-0.023 |
| Ln(nonnative trout density) | 0.023 | 0.012 | -0.001-0.046 |
| Wetted width | -0.038 | 0.022 | -0.081-0.006 |
| Model 3 |  |  |  |
| Intercept | 0.57 | 0.16 | 0.25-0.88 |
| Conductivity | 0.0009 | 0.0004 | 0.0002-0.0017 |
| Road density | 0.010 | 0.006 | -0.002-0.021 |
| Ln(nonnative trout density) | 0.012 | 0.010 | -0.001-0.032 |
| Model 4 |  |  |  |
| Intercept | 0.18 | 0.38 | -0.57-0.92 |
| Conductivity | 0.0011 | 0.0004 | 0.0003-0.0018 |
| Road density | 0.015 | 0.007 | 0.001-0.028 |
| Elevation | 0.000 | 0.000 | -0.0001-0.0004 |
| Model 5 |  |  |  |
| Intercept | 0.59 | 0.16 | 0.27-0.91 |
| Conductivity | $0.0011$ | 0.0004 | 0.0003-0.0018 |
| Model 6 |  |  |  |
| Intercept | 0.58 | 0.17 | 0.25-0.90 |
| Conductivity | 0.0010 | 0.0004 | 0.0002-0.0017 |
| Road density | 0.010 | 0.006 | -0.002-0.022 |
| Gradient | -0.022 | 0.022 | -0.007-0.022 |
| Model 7 |  |  |  |
| Intercept | 0.62 | 0.16 | 0.31-0.94 |
| Conductivity | 0.0011 | 0.0004 | 0.0003-0.0018 |
| Ln(nonnative trout density) | 0.015 | 0.011 | -0.006-0.035 |
| Model 8 |  |  |  |
| Intercept | 0.57 | 0.17 | 0.25-0.90 |
| Conductivity | 0.0009 | 0.0004 | 0.0002-0.0017 |
| Road density | 0.012 | 0.006 | 0.000-0.024 |
| Wetted width | -0.015 | 0.019 | -0.054-0.023 |
| Model 9 |  |  |  |
| Intercept | 0.75 | 0.18 | 0.40-1.10 |
| Conductivity | 0.0009 | 0.0004 | 0.0002-0.0017 |
| Road density | 0.011 | 0.006 | -0.001-0.022 |
| Ln(nonnative trout density) | 0.022 | 0.012 | -0.002-0.045 |
| Wetted width | -0.041 | 0.022 | -0.084-0.003 |
| Gradient | -0.023 | 0.022 | -0.065-0.020 |
|  | Model 10 |  |  |
| Intercept | 0.64 | 0.17 | 0.32-0.97 |
| Conductivity | 0.0011 | 0.0004 | 0.0004-0.0019 |
| Gradient | -0.027 | 0.023 | -0.071-0.017 |

Table 24. Top models relating broad-scale bioclimatic conditions to basin-wide Bonneville Cutthroat Trout abundance using 34 long-term monitoring reaches in Bear River tributaries of southeast Idaho. Akaike's information criteria ( $\mathrm{AIC}_{\mathrm{c}}$ ), change in $\mathrm{AIC}_{\mathrm{c}}$ $\left(\triangle \mathrm{AIC}_{c}\right)$, and $\mathrm{AIC}_{c}$ weights $\left(w_{i}\right)$ were used to assess model plausibility, and coefficients of determination $\left(r^{2}\right)$ indicate the amount of variation explained in the models.

| Bioclimatic predictor models | $\mathbf{A l C}_{\mathbf{c}}$ | $\boldsymbol{\Delta} \mathbf{A I C}_{\mathbf{c}}$ | $\boldsymbol{w}_{\boldsymbol{i}}$ | $\boldsymbol{r}^{\mathbf{2}}$ |
| :--- | :---: | :---: | :---: | :---: |
| Winter flow | -35.11 | 0.00 | 0.22 | 0.31 |
| Winter flow + summer flow | -33.65 | 1.46 | 0.11 | 0.26 |
| Summer air temperature + summer flow | -33.16 | 1.95 | 0.08 | 0.24 |

Table 25. Parameter estimates and $95 \%$ confidence intervals (Cls) for the top models relating broad-scale bioclimatic conditions to basin-wide Bonneville Cutthroat Trout abundance using 34 long-term monitoring reaches in Bear River tributaries of southeast Idaho.

| Parameter | Estimate | SE | 95\% CIs |
| :--- | :---: | :---: | :---: |
| Best model |  |  |  |
| Intercept | -1.04 | 0.43 | $-1.88--0.21$ |
| Winter flow | 0.38 | 0.15 | $0.09-0.68$ |
| Second-best model |  |  |  |
| Intercept | -1.02 | 0.43 | $-1.85--0.18$ |
| Winter flow | 0.32 | 0.16 | $0.00-0.64$ |
| Summer flow | 0.03 | 0.03 | $-0.03-0.09$ |
|  |  |  |  |
| Intercept | Third-best model |  |  |
| Summer air temperature | -2.58 | 1.30 | $-5.12--0.05$ |
| Summer flow | 0.15 | 0.08 | $-0.01-0.32$ |
|  | 0.07 | 0.03 | $0.01-0.13$ |

FIGURES


Figure 13. Location of reaches that were repeatedly electrofished to determine trends in the abundance of Bonneville Cutthroat Trout and nonnative trout in Bear River tributaries of southeast Idaho. Numbers correspond to Tables 20 and 21.


Figure 14. Relationship between the abundance of nonnative trout and Bonneville Cutthroat Trout for individual electrofishing surveys conducted at long-term monitoring reaches in Bear River tributaries of southeast Idaho where sympatry occurred.


Figure 15. Relationship between mean annual z-scores of Bonneville Cutthroat Trout abundance in a given year and nearby air temperature, stream discharge, and Palmer Drought Severity Index (PDSI) the previous year for Bear River tributaries of southeast Idaho from 1993 to 2020.

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[^0]:    ${ }^{1}$ Water temperature data recorded every 80 minutes rather than 60 minutes.
    ${ }^{2}$ Data starts on July 1 and ends on August 24.
    ${ }^{3}$ Data starts on July 1.
    ${ }^{4}$ Data starts on June 29.

