FISHERY RESEARCH

PROJECT 3: WILD TROUT EVALUATIONS

ANNUAL PROGRESS REPORT
JULY 1, 2021 — JUNE 30, 2022

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PROJECT 3: WILD TROUT EVALUATIONS

Annual Progress Report

July 1, 2021 to June 30, 2022

Project 3 – Wild Trout investigations

Subproject #1: M_{YY} Brook Trout Field Evaluations 2021

Subproject #2: Comparison of Growth and Body Condition of wild and hatchery M_{YY} Brook Trout in Streams and Alpine Lakes

Subproject #3: Evaluating Methods of Capturing Juvenile Trout in Alpine Lakes

Subproject #4: Factors Related to the Distribution and Abundance of Westslope Cutthroat Trout in Central Idaho

Subproject #5: Temporal Trends and Habitat Associations for Mountain Whitefish in Central Idaho

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ANNUAL PROGRESS REPORT

SUBPROJECT #1: M_{YY} BROOK TROUT FIELD EVALUATIONS 2021

State of: Idaho

Project No.: 3 Title: Wild Trout Evaluations

Subproject #1: M_{YY} Brook Trout Field Evaluations 2021

Time Period: July 1, 2021 to June 30, 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 (M_{YY}) Brook Trout. YY male Brook trout are created in the hatchery by feminizing XY males and crossing them with normal XY males. When M_{YY} Brook Trout reproduce successfully with wild females, all offspring are males. This can potentially be used to shift the sex ratio of the wild population toward males, potentially reaching a point where no females remain in the population to reproduce, thus eliminating the population. We stocked fingerling (mean = 131 mm; range = 72–192 mm) M_{YY} Brook Trout in three streams and four lakes in 2021, and catchable (mean = 255 mm; range = 168–320 mm) M_{YY} Brook Trout in two streams and two lakes in 2021 to attempt to eradicate wild Brook Trout in these study systems; these waters have now been stocked for several years, some 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_{YY} Brook Trout, and therefore decrease the time to eradication. Suppression via mechanical removal in 2021 was 86% in Dry Creek, 64% in Pike's Fork Creek, 24% in Martin Lake, and 41% in Seafoam Lake #4. This long-term study is scheduled to be completed in 2026.

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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 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-third 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 (M_{YY}), eventually resulting in population eradication by eliminating females (Gutierrez and Teem 2006; Teem and Gutierrez 2010). To create a M_{YY} brood stock, XY males are feminized by exposing them to estrogen (Teem and Gutierrez 2010). After rearing to maturity, the resulting XY neo-females are crossed with normal XY males and, on average, one-quarter of the progeny will be M_{YY}. To develop a functional broodstock, half of the M_{YY} are then feminized by exposure to estrogen at an early age to create egg-bearing YY fish (F_{YY}). Subsequent progeny of F_{YY} and M_{YY} crosses are all M_{YY}. These M_{YY} 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_{YY} Brook Trout survive and successfully reproduce after stocking. A pilot study estimated an average of 16% of M_{YY} 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 for food and space (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 M_{YY} 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. Hatchery trout of catchable-size, hereafter referred to as catchables, generally return to creel at a higher rate than smaller hatchery trout, hereafter referred to as fingerlings (Wiley et al. 1993; Dillon and Jarcik 1994). The greater survival of catchables 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 not well understood. The difference in survival between fingerlings and catchables is of particular interest in the case of M_{YY} fish, because the objective is to maximize abundance of mature M_{YY} at the time of spawning. However, it is unclear whether this is best achieved by stocking higher numbers of fingerling M_{YY} or lower numbers of catchable M_{YY}.

The Idaho Department of Fish and Game (IDFG) has produced a M_{YY} Brook Trout broodstock that annually produces 20,000-30,000 M_{YY} Brook Trout for eventual stocking into Idaho waters (Schill et al. 2016). Prior to large-scale stocking, survival and reproductive success of catchable M_{YY} Brook Trout in the wild were evaluated in Kennedy et al. (2018a), and this study indicated that M_{YY} Brook Trout could successfully survive and reproduce in the wild. Recent modelling suggests that annual stocking of M_{YY} Brook Trout into streams and alpine lakes can result in eradication of the wild population within 10 years if M_{YY} Brook Trout are stocked at a rate of 50% of the wild Brook Trout abundance and their fitness is equivalent to wild fish (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. For this study, the following objectives were developed to guide our work:

OBJECTIVE

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

METHODS

The IDFG experimentally feminized male Brook Trout fry with estrogen (in the form of 17β-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 M_{YY} Brook Trout prior to 2019 occurred at Mackay Hatchery and after 2019 at Hayspur Hatchery. Offspring are produced by crossing F_{YY} and M_{YY} broodstock, and fish are reared to fingerling and catchable sizes at the hatchery in outdoor concrete raceways in 10-12°C single-use spring water until the time of release. All study fish are adipose clipped so they can be differentiated from wild fish after stocking. For this study, fingerlings average about 125 mm and catchables average about 250 mm total length. They are stocked at approximately 8 and 20 months after hatching, respectively.

Study streams and lakes were selected with self-sustaining Brook Trout populations comprising a large majority of the fish species composition (>80%). Each study stream treatment reach had 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 2; 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 stocking. At two of the streams and two of the lakes, we manually suppressed the wild Brook Trout population to improve survival and spawning success of stocked fish. Suppression was achieved by the removal of wild Brook Trout with backpack electrofishing in streams, and gill nets in conjunction with boat/raft electrofishing in lakes. Non-suppression streams and lakes were stocked with M_{YY} Brook Trout without the suppression of their wild counterparts. Two control streams and two control lakes were 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_{XX}) 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 M_{YY} Brook Trout in streams began in 2016 with additional streams included in 2017 (Table 1). Dry, Tripod, East Fork Clear, Alder, and Beaver creeks have been under evaluation since 2016, whereas Pike's Fork and East Threemile creeks have been part of the evaluation since 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 M_{YY} 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). Because of the three-year cycle associated with sampling, surveys were conducted in all study lakes and the two study streams that receive annual suppression (i.e., Dry Creek and Pike's Fork Creek) in 2021. Non-suppression streams were not sampled in 2021. However, all lakes and streams were stocked with M_{YY} Brook Trout in 2021. Stream sampling is scheduled for 2022, and lake sampling for 2024.

Stream surveys

Suppression of wild Brook Trout was conducted in Dry and Pike's Fork creeks after snowmelt subsided (to maximize electrofishing capture efficiency) but prior to annual M_{YY} stocking. Before suppression, approximately 20 Brook Trout (≥100 mm) were marked with a lower caudal clip at each ½ km of the stream, 1-5 days 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) and wild Brook Trout were removed. 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 Hz, 300-990 V, and a 35% duty cycle. During suppression, persons with backpack electrofishers covered all available habitats, moving methodically upstream in tandem. All wild Brook Trout captured were euthanized with a lethal dose of anesthetic. 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 30% of the total catch among all study streams, were released unharmed, and were not included in further analyses.

Prior to M_{YY} stocking in each study stream, we collected tissue samples from wild Brook Trout fry (<100 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™ 3MM chromatography paper (Thermo Fisher Scientific, Inc., Pittsburgh, Pennsylvania). A sample size of 100 tissue samples from Brook Trout fry was targeted for each stream to characterize the sex ratio of each wild population. 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

All eight study lakes (i.e., Black Lake, Duck Lake, Lloyd Lake, Martin Lake, Rainbow Lake, Seafoam Lake #4, Snowslide Lake #1, and Upper Hazard Lake) were sampled in 2021. For non-suppression lakes, fish were captured using floating Swedish experimental gill nets (36 m long and 1.8 m deep) consisting of nylon mesh panels of 10, 12.5, 18.5, 25, 33, and 38 mm bar mesh set overnight. Net locations were selected to maximize catch, based on professional experience. One net per night constituted one unit of effort. Three nets were set in each lake. Angling using fly and spinning tackle was also used to capture Brook Trout at each lake. Data collected from captured fish included: species, TL (mm), weight (g), and identification of marks (fin and jaw clips). Gonads were exposed for observation by making a ventral mid-line incision along the entire body cavity. Males were classified as immature if testes were dorsally restricted, opaque, and thread-like, and mature if they were large and milky white. Females observed were classified as immature if their ovaries were small, translucent, granular, and dorsally restricted, and mature if they possessed eggs in advanced stages of development filling much of the abdominal cavity (Downs et al. 1997; Meyer et al. 2003; Schill et al. 2010). Additionally, tissue samples from adult and juvenile fish were collected to estimate sex ratios and parentage. Brook Trout fry were captured either by gill net or with backpack electrofishing when fry were observed at stream inlets or lake out-flows. Minnow traps (25.4 x 25.4 x 43.2 cm; 3.2 mm sq. mesh) baited with tuna in oil, tuna in oil and a submersible light emitting diode light (LED), or tuna in oil and a diphenyl oxalate chemical light stick (glow stick) were also used to target fry along shorelines, and lake inlets and outlets. In total, nine minnow traps (2 traps per bait type) were set in each lake.

For suppression lakes, sampling and suppression was conducted using gill nets and boat or raft electrofishing. Boat electrofishing was conducted in Martin Lake over two nights. Gill nets were also set each night to increase the number of wild Brook Trout that could be removed from the system. Suppression in Seafoam Lake #4 consisted of similar protocols, but electrofishing was conducted via raft rather than boat. During electrofishing, M_{YY} Brook Trout were identified based on adipose fin clips; these fish were marked with an additional lower caudal clip and then released so that a population estimate could be conducted via mark-recapture. In both systems all wild Brook Trout were removed. Data collected from captured fish included: species, TL (mm), and identification of fin and maxillary clips. Salmonids other than Brook Trout comprised less than 25% of the total catch among all study lakes, were released unharmed, 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™ 3MM chromatography paper (Thermo Fisher Scientific, Inc., Pittsburgh, Pennsylvania). A fish passage barrier assessment was conducted at Seafoam Lake #4 using the same methodology that was used in streams; Martin Lake has no inlet or outlet and is considered a closed system.

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 Development Core Team 2021). Ninety-five percent confidence intervals (CIs) 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. Due to small sample sizes (see below), Brook Trout abundance was calculated in Martin Lake by dividing the total catch of wild Brook Trout in the lake by the average capture efficiency (i.e., 0.24) from 2017 and 2018 sampling events. Similarly, M_{YY} Brook Trout abundance in Seafoam Lake #4 was calculated by dividing the total catch of wild Brook Trout in the lake by the average capture efficiency (i.e., 0.41) from 2018 and 2019 sampling events. After calculating total Brook Trout abundance, the abundance of wild and M_{YY} Brook trout was estimated based on their respective proportions of the total catch. In all waterbodies, estimates for both wild and M_{YY} Brook Trout were calculated for all size classes ≥100 mm to describe abundance for the entire study area.

Stocking

Stocking M_{YY} 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 M_{YY} 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 (a 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 M_{YY} Brook Trout at 125 fish/km.

Fingerling stocking rates were initially set at four times the stocking rate of catchables (i.e., 500 fingerlings/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/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. 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. Subsequent stocking densities should be consistent to reduce bias when evaluating the rate of change in sex ratios, because a higher stocking rate of M_{YY} Brook Trout could result in a faster rate of change in sex ratios (Schill et al. 2017) and obscure our ability to detect a difference between treatment groups.

Stocking rates in alpine lakes were 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_{YY} 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/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 M_{YY} 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. 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 ¾-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 ½ km were used to inform M_{YY} stocking distribution in the stream. Assuming stocked hatchery fish generally move downstream (High and Meyer 2009), M_{YY} Brook Trout were distributed at a 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_{YY} fish longitudinally throughout the entire stream. To maximize the encounter rate of hatchery M_{YY} males with spawning females, we backpacked fish into headwater reaches or other roadless areas. For stocking in roadless sections, a contractor-grade garbage bag inside of 19-L buckets inside backpacks was filled with approximately 8-L of hatchery truck water (~12°C). 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 ≤45 minutes. At some locations, fish were transported in coolers on ATVs. Loading densities and water quality monitoring in coolers followed methods described above.

Fingerling and catchable M_{YY} Brook Trout were stocked into alpine lakes primarily by helicopter. Fish were counted and placed into a 208-L bucket filled with water. The helicopter flew to the designated lake and dumped the fish into the lake by tipping the bucket over. For some larger lakes, the fish were delivered in more than one trip to ensure appropriate loading densities. 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 M_{YY} 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 of the suppression streams or lakes, tissue samples were collected from hatchery and wild fish to identify successful reproduction of M_{YY} Brook Trout in the wild and to monitor changes to the populations' sex ratio. Approximately 100 tissue samples were collected from wild Brook Trout fry (<100 mm) from each study system during July-September to estimate sex ratios and reproductive success. Tissue samples were clipped from the caudal fin and preserved on Whatman™ 3MM 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 µL volume consisting of 0.50 µL of primer mix, 2.50 µL of Qiagen Master Mix (cat. 206143), 1.00 µL dH20, and 1.00 µL template DNA (unknown concentration). Thermal cycling conditions were 95°C for 15 min followed by 25 cycles of 94°C for 30 s, 60°C for 1 min 30 s, 72°C for 60 s, and a final extension of 60°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 = ~131-135 base pairs; b.p.) and both SexY_Brook1 (peak height = ~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% CIs around the estimated male proportions, following Fleiss (1981).

Genetic assignment evaluation

A second method to evaluate reproductive success of M_{YY} 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_{YY} Brook Trout individually prior to release and when study designs require stocking thousands of M_{YY} Brook Trout into large lakes or rivers.

M_{YY} 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 represented 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 the introductions of M_{YY} Brook Trout by genetically screening samples

collected from both the M_{YY} population used for stocking and from the receiving wild population fish. The expectation was that progeny from M_{YY} adults and wild adults had approximately equal probability of membership to each population (Q = 0.5).

Fry sampled during 2021 from all study lakes, Dry Creek, and Pike's Fork Creek for sex ratio analysis were subjected to GA analysis to describe the origin of sampled fish as either progeny of wild or M_{YY} Brook Trout. Determining the origin of the sampled fry will allow us to describe relative spawning success of M_{YY} Brook Trout and the proportion of the offspring in the system produced by M_{YY} fish.

Water Chemistry

In 2021, measurements of water chemistry parameters (i.e., alkalinity and water hardness) were also taken at all study waters. These parameters were measured because differences in these parameters between the hatchery and the waterbody receiving fish can influence the post-release survival of stocked fish (Trushenski et al. 2019). Alkalinity was measured using a Hanna Instruments HI1775 Checker® Handheld Colorimeter (Hanna Instruments, Smithfield, Rhode Island). Total hardness was measured using a Hach Company Model HAC-DT Hardness Test Titration Kit (Hach Company, Loveland, Colorado). Measurements of both alkalinity and total hardness were also taken from Hayspur hatchery for comparison.

RESULTS

Stream surveys

At Dry Creek, 5,718 Brook Trout ≥100 mm were captured, of which 3,845 (67%) were M_{YY} Brook Trout and 1,873 (33%) were wild fish, the latter being removed from the system. M_{YY} abundance was estimated to be 4,472 fish (95% CI = 4,130–4,813), and wild Brook Trout abundance was estimated to be 2,178 fish (95% CI = 2,012–2,345; Table 3). Suppression of wild Brook Trout was estimated to be 86%. Wild Brook Trout TLs ≥100 mm averaged 173 mm (maximum = 290 mm), which was slightly smaller than M_{YY} Brook Trout lengths ≥100 mm, which averaged 200 mm (maximum = 358 mm; Figure 3). No fish with maxillary clips were observed, suggesting the barrier is effective at preventing recolonization at Dry Creek. An additional 143 Brook Trout were maxillary clipped (mean = 225 mm; maximum = 284 mm) below the downstream barrier on the study reach to continue barrier evaluations in future years. In addition to Brook Trout, 238 Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* were captured.

At Pike's Fork Creek, 2,954 Brook Trout ≥100 mm were captured, only two of which were M_{YY} Brook Trout. All 2,952 of the wild Brook Trout captured were removed from the system. The M_{YY} Brook Trout ≥100 mm abundance was estimated to be 3 fish (95% CI = 3–3) and the abundance of wild Brook Trout ≥100 mm in the stream was estimated to be 4,601 (95% CI = 4,136–5066; Table 3). Suppression of wild Brook Trout in Pike's Fork Creek was estimated to be 64%. Lengths of wild Brook Trout ≥100 mm averaged 144 mm (maximum = 257 mm), and M_{YY} Brook Trout ≥100 mm averaged 207 mm (maximum = 227 mm). Similar to Dry Creek, no fish with maxillary clips were detected above the barrier, and 101 new wild Brook Trout (mean = 134 mm; maximum = 231 mm) were maxillary clipped below the barrier for future barrier evaluation. Additionally, 1,179 Rainbow Trout *O. mykiss*, and two Bull Trout *S. confluentus* were captured in Pike's Fork Creek in 2021.

Lake surveys

At Black Lake, gill nets were set for an average of 12.7 (SE = 0.2) hours per night, over 3 net-nights (Table 4). In total, 22 Brook Trout ≥100 mm were sampled via gill net, all of which were wild fish. Gill net catch-per-unit effort (CPUE) was 7.3 Brook Trout/net-night (≥100 mm). Lengths of wild Brook Trout ≥100 mm averaged 227 mm (maximum = 299; Figure 4). No M_{YY} Brook Trout were captured in Black Lake.

At Duck Lake, we captured a total of 20 Brook Trout ≥100 mm via gill net. Gill nets were set for an average of 13.8 (SE = 0.3) hours per night, over 3 net-nights (Table 4). Gill net CPUE was 6.7 fish/net-night. Similar to Black Lake, all fish sampled via gill net were wild fish. Average lengths of wild Brook Trout ≥100 mm in Black Lake via gill net was 124 mm (maximum = 290 Figure 4). Additionally, one M_{YY} Brook Trout (300 mm) was sampled via angling.

At Lloyds Lake, we captured 30 Brook Trout ≥100 mm of which 19 (63%) were M_{YY} Brook Trout and 11 (37%) were wild fish. The majority of fish ≥100 mm were sampled via gill net (*n* = 22), but eight were sampled via angling. Gill nets were set for an average of 13.9 (SE = 0.2) hours per night, over 3 net-nights (Table 4). In gill nets, CPUE of Brook Trout ≥100 mm was 7.3 fish/net-night. Lengths of wild Brook Trout ≥100 mm averaged 207 mm (maximum = 264 mm; Figure 4), which were similar to M_{YY} Brook Trout ≥100 mm, which averaged 201 (maximum = 256 mm).

At Rainbow Lake, we captured 30 Brook Trout ≥100 mm, all of which were wild fish. Gill nets were set for an average of 12.9 (SE = 0.3) hours per night, over 3 net-nights (Table 4). The majority of Brook Trout ≥100 mm were sampled via gill net (*n* = 29), but one Brook Trout was sampled via minnow trap. Gill net CPUE was 6.7 fish/net-night. Lengths of wild Brook Trout ≥100 mm averaged 163 mm (maximum = 276; Figure 4). No M_{YY} Brook Trout were captured in Rainbow Lake.

At Snowslide Lake #1 Lake, we captured 74 Brook Trout ≥100 mm, all of which were wild fish captured in gill nets. Gill nets were set for an average of 13.4 (SE = 0.4) hours per night, over 3 net-nights (Table 4) with a CPUE = 24.7 fish/net-night. Lengths of wild Brook Trout ≥100 mm averaged 168 mm (maximum = 250; Figure 4). No M_{YY} Brook Trout were captured in Snowslide Lake #1 Lake.

At Upper Hazard Lake, we captured 15 Brook Trout ≥100 mm, all of which were wild fish. The majority of fish were sampled via gill net (*n* = 14; CPUE = 4.7 fish/net-night), but one fish was sampled via backpack electrofishing. Gill nets were set for an average of 13.4 (SE = 0.4) hours per night, over 3 net-nights (Table 4). Lengths of wild Brook Trout ≥100 mm averaged 140 mm (maximum = 251; Figure 4). One tiger muskellunge *Esox lucius* × *E. masquinongy* (855 mm) was also captured via gill net.

Across all six non-suppression lakes (Black, Duck, Lloyds, Rainbow, Snowslide #1, and Upper Hazard Lakes) 207 Brook Trout fry (i.e., <100 mm) were captured. The majority of fry were captured via backpack electrofishing (*n* = 150), but fry were also captured via gill net (*n* = 39) and minnow trap (*n* = 18). Average fry length was 55 mm, and varied from 23 to 99 mm.

At Martin Lake, 130 Brook Trout ≥100 mm were captured, four (3%) of which were M_{YY} Brook Trout and 126 (97%) were wild fish, the latter being removed from the system (Table 3). Based on the average capture efficiency (i.e., 0.24) from the 2017 and 2018 surveys, estimated abundance of Brook Trout ≥100 mm was 15 for M_{YY} Brook Trout and 532 for wild Brook Trout (Table 3). The suppression rate was estimated to be 24% in Martin Lake. Lengths of wild Brook

Trout ≥100 mm averaged 156 mm (maximum = 271; Figure 5) while average lengths of M_{YY} Brook Trout ≥100 mm were greater (mean = 206 mm; maximum = 216 mm).

At Seafoam Lake #4, 408 Brook Trout ≥100 mm were captured, 141 (35%) of which were M_{YY} Brook Trout and 248 (61%) were wild fish, the latter being removed from the system (Table 3). Based on the average capture efficiency (i.e., 0.41) from 2018 and 2019, estimated abundance of wild Brook Trout ≥100 mm was 594 fish, and M_{YY} Brook Trout abundance was estimated to be 387 fish. The suppression rate of wild Brook Trout in the lake was estimated to be 42%. Length of wild Brook Trout ≥100 mm averaged 222 mm (maximum = 359 mm; Figure 5), which was similar to M_{YY} Brook Trout ≥100 mm, which averaged 231 (maximum = 321 mm). No fish were observed with maxillary clips, suggesting the barrier is effective. Additionally, 5 wild Brook Trout (mean = 122 mm; maximum = 145 mm) were marked with a maxillary clip below the barrier into Seafoam Lake #4 to continue to assess the barrier.

Stocking

Fingerling M_{YY} Brook Trout were stocked into Dry Creek, East Fork Clear Creek, Tripod Creek, Duck Lake, Lloyds Lake, Martin Lake, and Seafoam Lake #4 during 2021 (Table 5). Lengths and weights of stocked fish were similar across waterbodies. However, stocking rates varied across the waterbodies in which stocking rates could be calculated (i.e., 148%-198% of wild Brook Trout ≥100 mm abundance).

Catchable M_{YY} Brook Trout were stocked into East Threemile Creek, Pike's Fork Creek, Black Lake, and Rainbow Lake during 2021 (Table 5). Again, lengths and weights of stocked fish were similar across waterbodies. Due to the three year sampling rotation, catchable stocking rates could only be calculated for Pike's Fork Creek (16%) during 2021. In suppression waters, stocking annually occurred after suppression was conducted.

Genetically determined sex ratios and reproductive success

Sex ratios varied from 46% to 64% male in study lakes, while in study creeks it was 82% for Dry Creek, and 49% for Pike's Fork Creek (Table 6). Genetic assignment analyses indicated that the proportion of offspring produced by stocked M_{YY} Brook Trout in study lakes varied from 0% to 9% (Table 6). No M_{YY} offspring were detected at Duck and Martin lakes. M_{YY} Brook Trout produced 64% of the offspring in Dry Creek and 6% in Pike's Fork Creek.

Water Chemistry

Water chemistry analysis revealed that both alkalinity and water hardness varied greatly among waterbodies and between Hayspur Hatchery and the waters stocked with M_{YY} Brook Trout (Table 7). Alkalinity and water hardness were higher at Hayspur Hatchery (alkalinity = 178 mg/L $CaCo_3$, water hardness = 224 mg/L $CaCo_3$) than at any of the stocked waters, and were most similar to the values observed in Dry Creek (alkalinity = 142 mg/L $CaCo_3$, water hardness = 163 mg/L $CaCo_3$). Lloyds Lake had the lowest alkalinity of 1 mg/L $CaCo_3$, while water hardness was lowest in Black Lake, Duck Lake, and Rainbow Lake (5 mg/L $CaCo_3$).

DISCUSSION

Due to the three-year sampling cycle of this study, the only two streams sampled in 2021 were Dry Creek and Pikes Fork Creek. At Dry Creek, abundance of wild fish declined by 27%

from 2020 to 2021, whereas M_{YY} Brook Trout was virtually unchanged. Dry Creek now has an M_{YY} composition of 67%, and among all the waterbodies stocked with M_{YY} Brook Trout, Dry Creek has the highest M_{YY} composition, the most skewed sex ratio (82% male), and the highest proportion of offspring attributed to M_{YY} fish (64%). Conversely, from 2020 to 2021 at Pike's Fork, wild Brook Trout abundance increased (20%) and M_{YY} Brook Trout abundance decreased (75%), M_{YY} Brook Trout composition remained chronically low (0% in 2021), and only 6% of Brook Trout offspring were attributed to M_{YY} fish. Considering that the habitat at Pike's Fork is adequate to support a robust wild Brook Trout population, we speculate that the discrepancy to date in program success between these two waters is attributable to stocking fingerlings in Dry Creek compared to stocking catchables in Pike's Fork.

Using CPUE as a measure of relative abundance, between surveys in 2018 and 2021, CPUE of wild Brook Trout declined in Black Lake (67%), Duck Lake (43%), Lloyds Lake (62%), Rainbow Lake (62%), and Upper Hazard Lake (65%), and only increased in Snowslide Lake #1 (12%). M_{YY} Brook Trout survival appears to be poor in the majority of waters, considering that none were captured at two lakes (i.e., Black Lake, Rainbow Lake), and only one was captured in Duck Lake. The one exception was Lloyds Lake, where M_{YY} comprised 63% of the catch. Survival of M_{YY} fish also remained low in Martin Lake, with M_{YY} composition declining by 2% and the M_{YY} abundance declining by 40% between 2020 and 2021. In Seafoam Lake #4, M_{YY} composition tripled and M_{YY} abundance increased almost four-fold from 2020 to 2021. Clearly, program success to date is much higher in Seafoam Lake #4 than in Martin Lake, although whether complete eradication can be achieved remains to be seen.

Sex ratios in all waterbodies except Dry Creek and Lloyds Lake have remained about 50% over the entire study. The successful shift to a male dominated sex ratio in these waters may in part be due to the use of fingerling M_{YY} Brook Trout. Indeed, all study waters stocked with fingerling M_{YY} Brook Trout have documented M_{YY} survival, while M_{YY} Brook Trout composition is currently 0% at all waters stocked with catchable M_{YY} Brook Trout.

Investigation into differences in water chemistry parameters between Hayspur Hatchery and the waters stocked with M_{YY} Brook Trout provided limited insight into the patterns of success observed in the current study systems. One pattern of note is that the waterbody that had alkalinity and water hardness measurements most similar to Hayspur Hatchery was Dry Creek, and M_{YY} appear to be having the most success in Dry Creek, whereas alkalinity and water hardness differed greatly between Hayspur Hatchery and the waters in which M_{YY} Brook Trout appear to be least successful (e.g., Martin Lake and Pikes Fork Creek). However, alkalinity and water hardness also vary greatly between Hayspur Hatchery and two waterbodies (i.e., Lloyds Lake and Seafoam Lake #4) that are showing some signs of success. These results suggest that while water chemistry may contribute to the success of M_{YY} Brook Trout, other environmental conditions may also play a role or act in conjunction with the effect of water chemistry.

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 M_{YY} 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 in future reports approximately every three years to monitor reproductive success of M_{YY} Brook Trout.

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TABLES

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

Stream name	Starting year	Treatment level	Stocked fish size	Reach length (km)	Avg. width (m)	Gradient (%)	Maximum elevation (m)	Latitude	Longitude
Dry Creek	2016	Suppression	Fingerling	6.5	5.2	1.5	2,377	44.1268	-113.5681
Pike's Fork Creek	2017	Suppression	Catchable	7.5	3.7	3.3	1,871	43.9832	-115.5484
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
Tripod Creek	2016	Non-suppression	Fingerling	9.1	1.4	1.0	1,625	44.3178	-116.1200
Alder Creek	2016	Control	Control	2.4	4.9	3.2	2,000	43.8234	-113.6074
Beaver Creek	2016	Control	Control	4.0	2.4	2.2	1,650	43.9889	-115.6071

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

Lake name	Starting year	Treatment	Stocked fish size	Surface area (ha)	Surface elevation	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
Lloyds Lake	2015	Non-suppression	Fingerling	2.91	2,092	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 #4	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* and M_{YY} Brook Trout ≥100 mm sampled in study waters in Idaho during 2021. Estimates of abundance were calculated using a mark-recapture survey (MR). Also included are the 95% confidence estimates (CI) on abundance, the proportion of the population that is comprised of M_{YY} Brook Trout, the number of fish removed from the system when annual suppression was conducted, the suppression rate, and capture efficiency. Capture efficiencies followed by * indicate that they were average capture efficiencies based on previous sampling years.

Waterbody	Sample method	Wild abundance	95% CI	M_{YY} abundance	95% CI	M_{YY} composition	Number of fish removed	Suppression rate	Capture efficiency
Dry Creek	MR	2,178	2,012 - 2,345	4,472	4,130 - 4,813	67%	1,873	86%	86%
Martin Lake	MR	532	-	15	-	3%	126	24%	24%*
Pike's Fork Creek	MR	4,601	4,136 - 5,066	3	3 - 3	<1%	2,952	64%	64%
Seafoam Lake #4	MR	594	-	387	-	35%	248	42%	41%*

Table 4. The number of Brook Trout *Salvelinus fontinalis* captured and catch-per-unit-effort (CPUE; fish/net-night) using experimental gill nets at six alpine lakes in central Idaho during 2021. Also shown is composition of M_{YY} Brook Trout in each lake.

Lake name	Treatment	Net nights	Brook Trout catch	CPUE	M_{YY} Composition
Black Lake	Catchable	3	22	7.3	0%
Duck Lake	Fingerling	3	20	6.7	0%
Lloyds Lake	Fingerling	3	22	7.3	59%
Rainbow Lake	Catchable	3	29	9.7	0%
Snowslide Lake #1	Control	3	74	24.7	0%
Upper Hazard Lake	Control	3	14	4.7	0%

Table 5. The number of M_{YY} Brook Trout *Salvelinus fontinalis* stocked into study waters in Idaho during 2021. Two sizes for M_{YY} Brook Trout were stocked into study waters. Additionally, the number of fish stocked divided by the total number of wild Brook Trout is included as an estimate of the stocking rate of M_{YY} Brook Trout compared to the wild Brook Trout population.

Waterbody	Size	Stocking date	Number of fish stocked	Mean length (mm)	SE	Mean weight (g)	SE	Stocking rate
			Streams					
Dry Creek	Fingerling	8/17/2021	4,255	129	2	22	1	195%
East Fork Clear Creek	Fingerling	8/23/2021	92	127	2	21	1	-
East Threemile Creek	Catchable	8/25/2021	606	256	2	178	4	-
Pike's Fork Creek	Catchable	8/19/2021	714	257	2	180	4	16%
Tripod Creek	Fingerling	8/23/2021	7,176	127	2	21	1	-
			Lakes					
Black Lake	Catchable	8/12/2021	227	253	2	189	5	-
Duck Lake	Fingerling	8/12/2021	2,137	132	2	25	1	-
Lloyds Lake	Fingerling	8/12/2021	1,172	132	2	25	1	-
Martin Lake	Fingerling	9/2/2021	788	136	2	26	1	148%
Rainbow Lake	Catchable	8/12/2021	735	253	2	189	5	-
Seafoam Lake #4	Fingerling	9/2/2021	1,176	136	2	26	1	198%

Table 6. Results of genetic sex ratio and genetic assignment analyses. Fry were sampled from study waters during 2021. 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 M_{YY} fish that are stocked into the system. Fingerlings averaged 131 mm (SE = 1) and catchable fish averaged 255 mm (SE = 1). One F_{YY} fry was identified in Black Lake and denoted as (1).

			Brook Trout fry sampled				Sex ratio (% male)		M_{YY} offspring (%)	
Waterbody	**Stocking size**	**Treatment**	**Total**	**Wild females**	**Wild males**	**M_{YY} offspring**	**Est.**	**SE**	**Est.**	**SE**
				Streams						
Dry Creek	Fingerling	Suppression	99	18	18	63	82	4	64	5
East Fork Clear Creek	Fingerling	Non-suppression	-	-	-	-	-	-	-	-
East Threemile Creek	Catchable	Non-suppression	-	-	-	-	-	-	-	-
Pike's Fork Creek	Catchable	Suppression	98	50	42	6	49	5	6	2
Tripod Creek	Fingerling	Non-suppression	-	-	-	-	-	-	-	-
				Lakes						
Black Lake	Catchable	Non-suppression	55	31	21	3	44	7	5	3
Duck Lake	Fingerling	Non-suppression	70	35	35	0	50	6	0	0
Lloyds Lake	Fingerling	Non-suppression	11	4	6	1	64	15	9	9
Martin Lake	Fingerling	Suppression	95	50	45	0	47	5	0	0
Rainbow Lake	Catchable	Non-suppression	92	47	38	7	49	5	8	3
Seafoam Lake #4	Fingerling	Suppression	104	52	49	3	50	5	3	2
Snowslide Lake #1	-	Control	75	38	37	0	49	6	0	0
Upper Hazard Lake	-	Control	60	29	31	0	52	6	0	0

Table 7. Water chemistry data collected from Hayspur Hatchery where the M_{YY} Brook Trout broodstock are raised and Idaho waterbodies stocked with M_{YY} Brook Trout during 2021.

Waterbody	Alkalinity (mg/L $CaCo_3$)	Hardness (mg/L $CaCo_3$)
Hayspur Hatchery	178	224
	Streams	
Dry Creek	142	163
East Fork Clear Creek	11	11
East Threemile Creek	56	54
Pike's Fork Creek	58	52
Tripod Creek	42	42
	Lakes	
Black Lake	8	5
Duck Lake	12	5
Lloyds Lake	1	18
Martin Lake	19	14
Rainbow Lake	10	5
Seafoam Lake #4	21	16

FIGURES

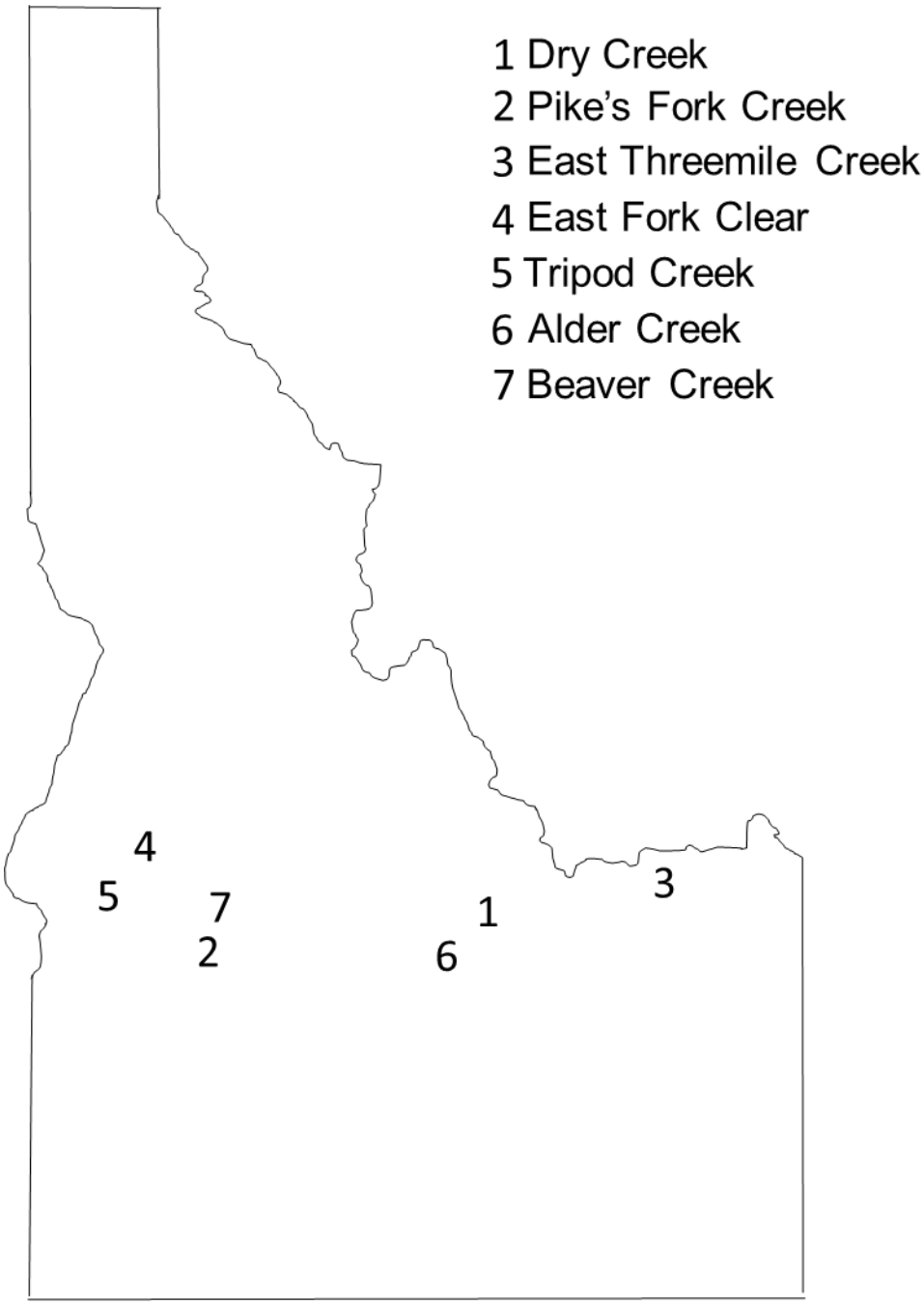

Figure 1. Locations of study streams for M_{YY} Brook Trout *Salvelinus fontinalis* field trials in Idaho.

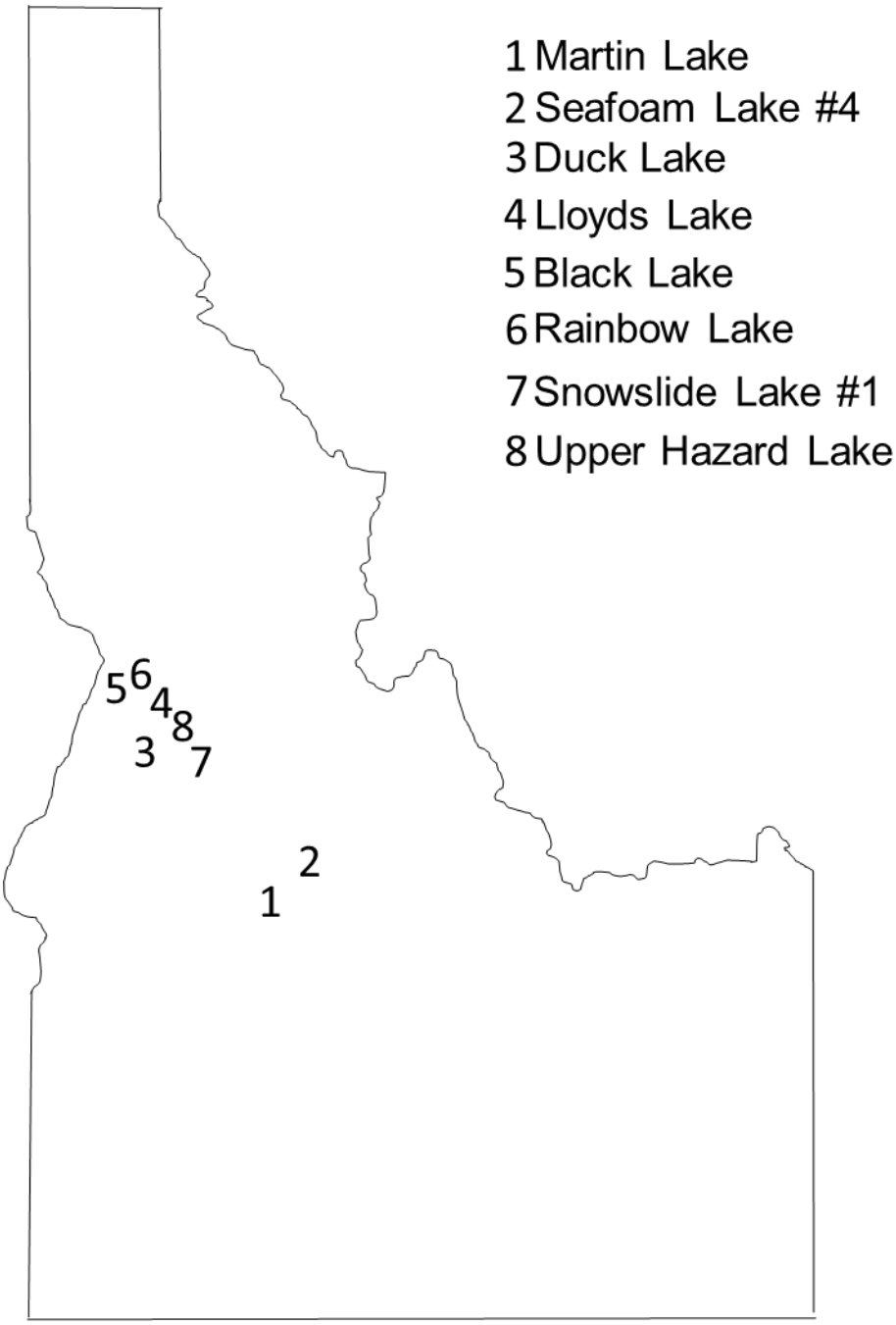

Figure 2. Locations of study lakes for M_{YY} Brook Trout *Salvelinus fontinalis* field trials in Idaho.

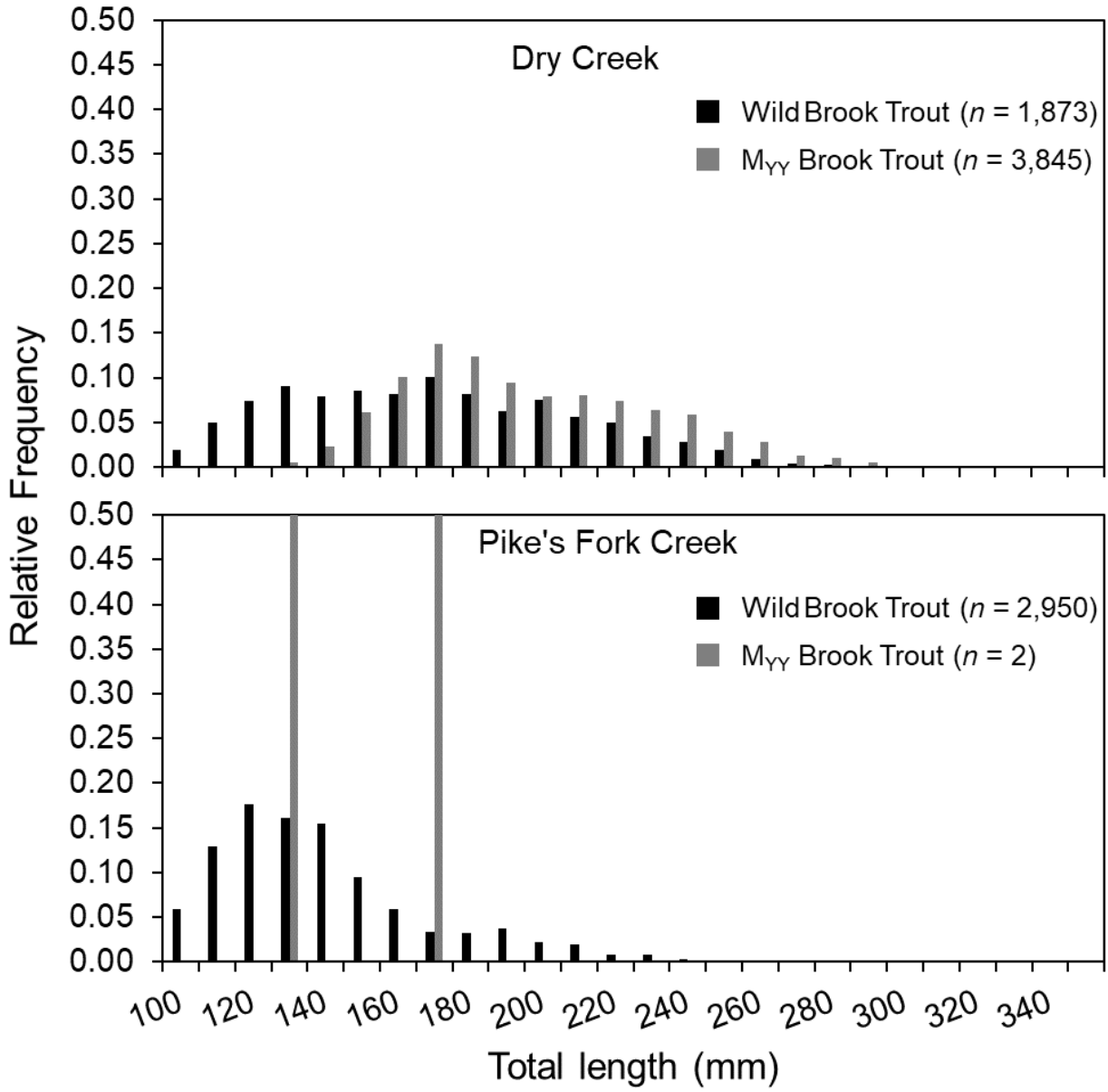

Figure 3. Length distributions of wild Brook Trout *Salvelinus fontinalis* and M_{YY} Brook Trout sampled in Dry Creek and Pike's Fork creeks, Idaho, during 2021.

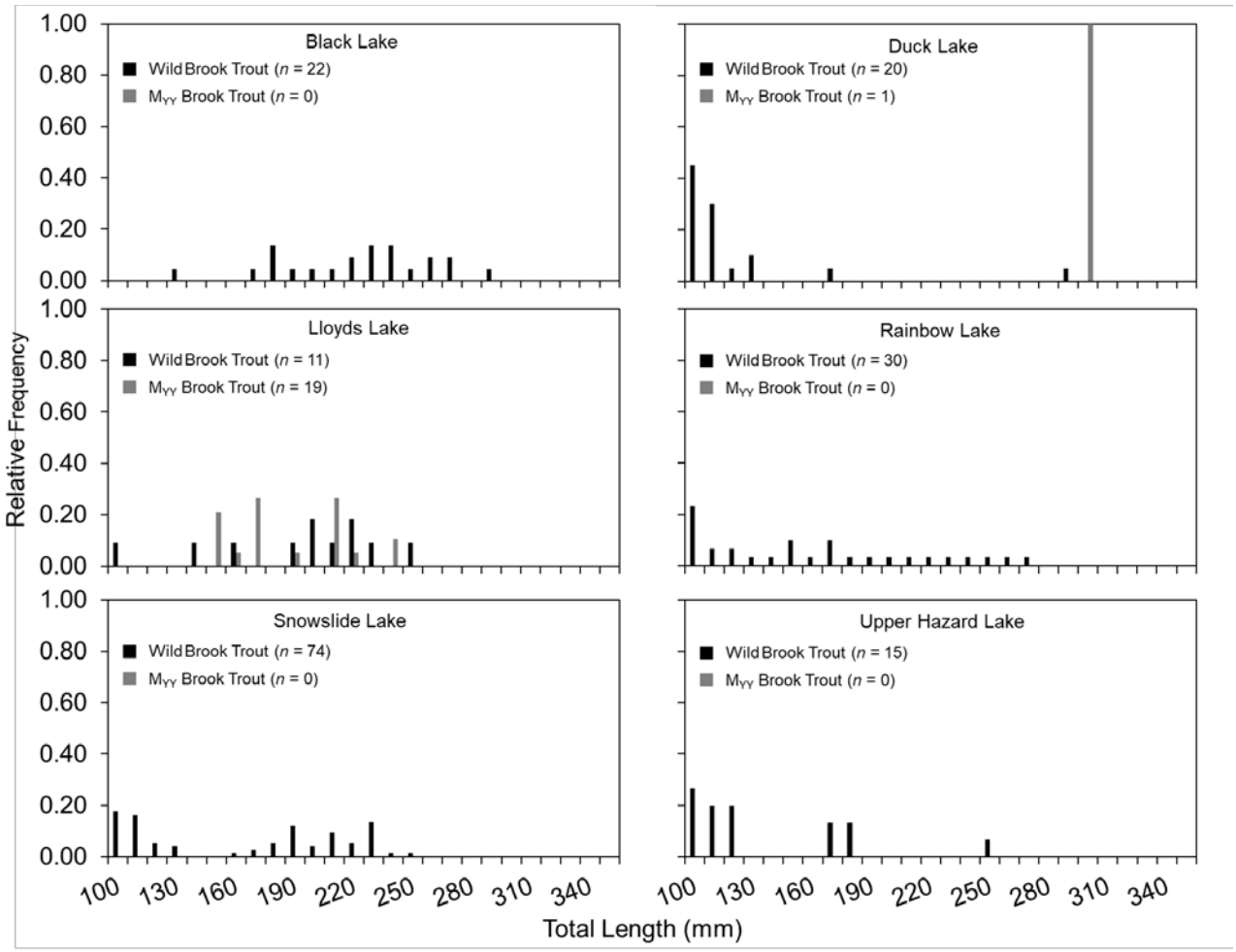

Figure 4. Length distributions of wild Brook Trout *Salvelinus fontinalis* and M_{YY} Brook Trout sampled in 6 Idaho alpine lakes, during 2021.

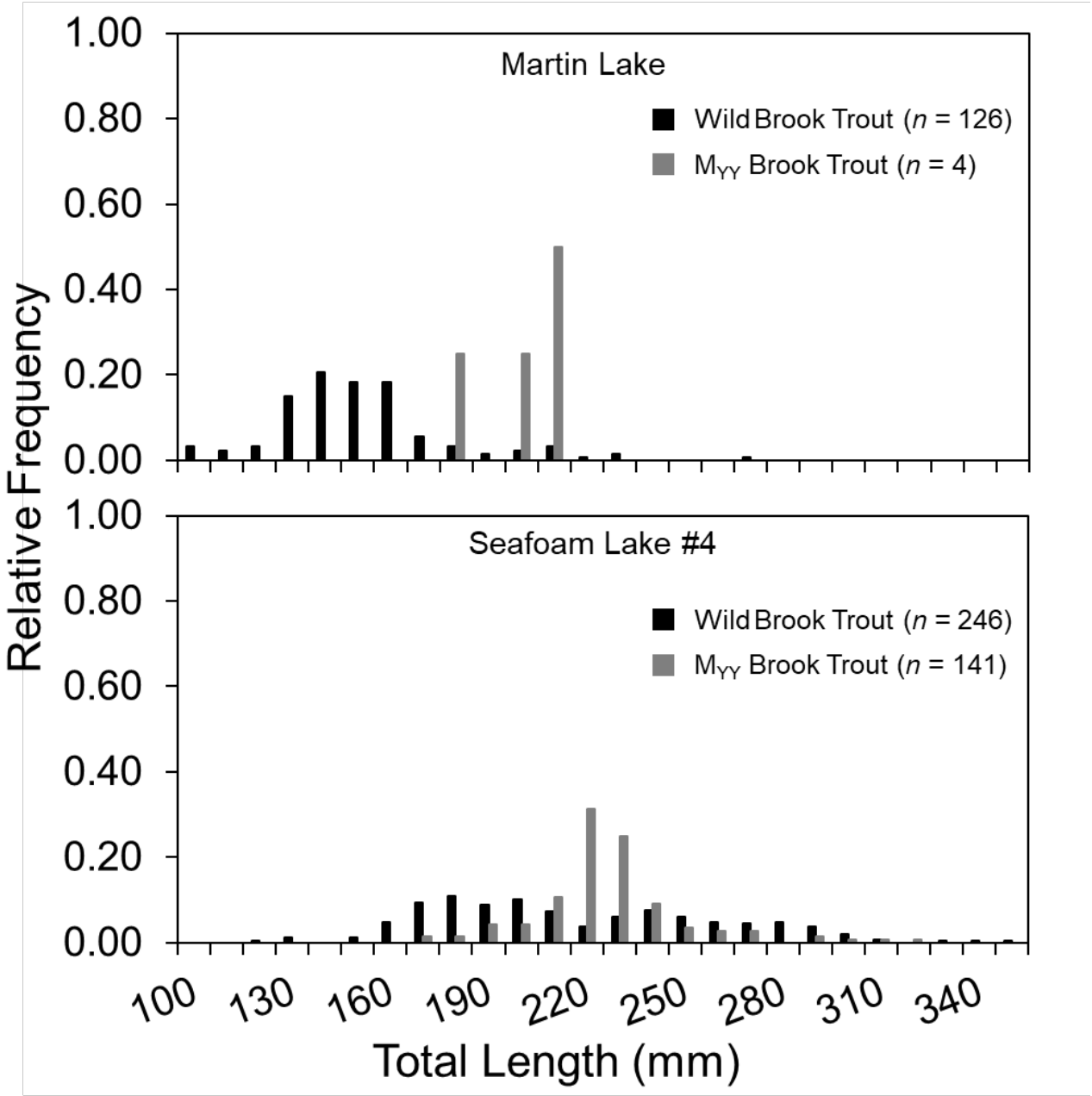

Figure 5. Length distributions of wild Brook Trout *Salvelinus fontinalis* and M_{YY} Brook Trout sampled in Martin and Seafoam lakes, Idaho, during 2021.

ANNUAL PROGRESS REPORT

SUBPROJECT #2: COMPARISON OF GROWTH AND BODY CONDITION OF WILD AND HATCHERY M_{YY} BROOK TROUT IN STREAMS AND ALPINE LAKES

State of: Idaho

Project No.: 3 Title: Wild Trout Evaluations

Subproject #2: Comparison of Growth and Body Condition of Wild and Hatchery M_{YY} Brook Trout in Streams and Alpine Lakes

Time Period: July 1, 2021 to June 30, 2022

ABSTRACT

Biologists have theorized that stocking YY male individuals (created via in-hatchery hormonal sex-reversal and selective breeding; hereafter M_{YY} fish) could be used to eradicate unwanted non-native vertebrate populations, but almost nothing is known about the fitness of M_{YY} individuals once released into the wild. After several years of stocking hatchery-reared M_{YY} Brook Trout (*Salvelinus fontinalis*) in two mountain streams and two alpine lakes, we sampled fish to compare their growth rates and body condition to wild conspecifics. Age estimates varied from 1-6 years for wild Brook Trout and 1-5 years for hatchery M_{YY} fish, whereas length varied from 103-359 mm for wild fish and 115-353 mm for hatchery M_{YY} fish. Results indicated that growth rates and body condition of stocked M_{YY} brook trout did not differ from wild brook trout in the same waters. Given that the success of M_{YY} eradication programs is primarily contingent upon M_{YY} individuals having fitness characteristics similar to wild conspecifics, these results provide further evidence that the stocking of hatchery-reared M_{YY} fish may be a viable tool for eradicating unwanted non-native fish populations.

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INTRODUCTION

Non-native fishes are considered one of the greatest threats to native freshwater fish populations worldwide (Gozlan et al. 2010). The negative effects of exotic fish on native fish populations can be attributed to a variety of mechanisms, including hybridization, predation, competition, habitat modification, and disease transmission (Zaret et al. 1973; Rhymer and Simberloff 1996; Vander Zanden et al. 1999; Fausch 2007; McMahon et al. 2007). While the introduction of non-native fish species outside their original distributional range includes myriad taxonomic groups, perhaps no taxa exemplifies this problem more than salmonids (Buoro et al. 2016). Using Brook Trout (*Salvelinus fontinalis*) as a paradoxical example, this species has been extirpated from much of its native range in eastern North America (Hudy et al. 2008) due in part to displacement by non-native Rainbow Trout (*Oncorhynchus mykiss* Walbaum; Habera and Moore 2005), yet in western North America (Dunham et al. 2002) and northern Europe (Spens et al. 2007; Korsu et al. 2010) Brook Trout are an invasive species that threatens the long-term viability of countless populations of various native salmonids.

In an attempt to diminish the negative effects of nonnative freshwater fishes on native taxa, fisheries managers and conservation biologists often implement eradication programs that are mechanically-based (e.g., Knapp and Mathews 1998; Meyer et al. 2006), chemically-based (e.g., Gresswell 1991; Treanor et al. 2017), and even biologically-based (e.g., Koenig et al. 2015), but such eradication projects are often unsuccessful (reviewed in Meronek et al. 1996, and Rytwinski et al. 2019). The fact that conventional eradication programs often fail to remove the undesirable species highlights the need for novel conservation methods for eradicating non-native vertebrate populations.

One eradication method that has long been considered is to shift the sex ratio of an unwanted population to all males, theoretically causing the population to collapse (Hamilton 1967; Gutierrez and Teem 2006). In fisheries, this could potentially be accomplished by rearing and stocking individuals with two Y chromosomes (i.e., males with a genotype of YY rather than XY - hereafter MYY; e.g., Schill et al. 2016). Population models considering possible eradication of non-native Brook Trout populations have indicated that eradication is theoretically feasible if the fitness of hatchery M_{YY} individuals approaches that of individuals in the wild population (Schill et al. 2017; Day et al. 2020). Due to the novelty of using M_{YY} vertebrates as an eradication method, almost nothing is known about the fitness of M_{YY} individuals once released into the wild. In the only such study ever conducted, hatchery M_{YY} Brook Trout were reared to about 225 mm total length and stocked in four North American mountain streams; these fish survived and spawned successfully with wild conspecifics, and produced all-male progeny, though reproductive success was lower for M_{YY} fish than for their wild counterparts (Kennedy et al. 2018a). While those preliminary results were insightful, additional evaluations of M_{YY} fitness are clearly needed.

Fitness in biological terms generally refers to reproductive success, but it can be quantified using a variety of metrics, many of which are correlated with growth and body condition. For instance, fish body length and body condition are positively correlated with milt potency (Wedekind et al. 2007), mate choice (de Gaudemar et al. 2000), post-spawning survival (Cunjak et al. 1987), and fry production (Blanchfield et al. 2003; Hargrove et al. 2021). In addition, dominance hierarchies, which regulate the use of optimal feeding and resting habitats, and thus overall energy budget, are largely related to body length and condition in salmonids (Fausch and White 1981; Nakano 1995). Because growth rates and body condition data are easily obtained, and are correlated with various metrics of fitness, we sought to compare growth and condition of hatchery M_{YY} and wild Brook Trout in the habitats where they commonly displace native salmonids in western North America.

OBJECTIVES

1. Compare growth and body condition of M_{YY} and wild Brook Trout in high-elevation streams and alpine lakes.

METHODS

Wild and hatchery M_{YY} Brook Trout were sampled from two streams (i.e., Dry Creek and Tripod Creek) and two alpine lakes (i.e., Seafoam Lake #4 and Lloyds Lake) in central Idaho (Table 8). Waters included in the present study are a subset of several streams and alpine lakes currently being stocked with M_{YY} Brook Trout to evaluate their ability to eradicate unwanted wild Brook Trout populations (Kennedy et al. 2017). Streams varied from 1.4 to 5.2 m in average width and 1,625 to 2,279 m in elevation. Lakes varied from 2.7 to 2.9 ha in surface area and 2,092 to 2,426 m in elevation.

M_{YY} Brook Trout were developed and reared either at Mackay or Hayspur fish hatcheries following the procedures in Schill et al. (2016). Fish were stocked annually as age-0 fingerlings (mean = 131 mm SD = 18.83; TL), and were adipose-clipped prior to release in order to differentiate them from wild fish. They were stocked at a rate (Table 8) equal to approximately 50% of the wild Brook Trout population abundance at the time the study began, which was determined from abundance estimates obtained for each population (Kennedy et al. 2017, 2018b). Stocking of M_{YY} fish occurred for several years prior to the field sampling presented herein, in order to assess fish growth for multiple age classes.

For the present study, fish were sampled in streams via backpack electrofishing in July of 2020 and 2021. Electrofisher settings were 60 Hz and 25% duty cycle, and voltage was adjusted until the electrofishing unit produced approximately 100 watts of average power output, which optimizes capture of salmonids in small streams (Meyer et al. 2021).

Seafoam Lake #4 was accessible by vehicle, so sampling was conducted using raft electrofishing and gill netting in September 2020 and September 2021. Raft electrofisher settings were 60 Hz, 25% duty cycle, and 300-400 volts, which produced 7-10 amps of peak current. Three pairs of gill nets were set in locations evenly dispersed around the lake. Nets were set each afternoon and pulled the following morning in the order in which they were set. Gill net pairs consisted of one floating and one sinking experimental gill net (46 m long and 2 m deep; consisting of nylon mesh panels of 19, 25, 32, 38, 51, and 64 mm bar mesh). All net sets were oriented perpendicular to the shoreline with the smaller mesh towards the shore.

Lloyds Lake could not be accessed by vehicle so raft electrofishing was not possible. Instead, sampling was conducted in June 2021 using gill nets and hook-and-line angling. Three floating gill nets (36 m long 1.8 m deep; consisting of nylon mesh panels of 10, 13, 19, 25, 32, and 38 mm bar mesh) were evenly distributed around the lake and oriented perpendicular to the shoreline, with the smaller mesh towards the shore. Nets were set in the afternoon and pulled the following morning in the order in which they were set.

Regardless of the water, a minimum of two hatchery M_{YY} Brook Trout and two wild Brook Trout were collected from every 10 mm length-bin, when present in the sample. The number of fish sacrificed was limited so that this study did not interfere with the ongoing M_{YY} hatchery Brook

Trout study (Kennedy et al. 2017, 2018b). As part of that overarching M_{YY} evaluation, wild Brook Trout were suppressed annually in two of the waters (i.e., Dry Creek and Seafoam Lake #4; Table 8), with an annual population suppression rate of about 50% (as determined from mark-recapture data); no suppression of wild Brook Trout occurred in the remaining two waters.

Each fish sampled was measured for total length (mm) and weight (g). Fish were euthanized with a lethal dose of anesthetic and sagittal otoliths were removed (Schneidervin and Hubert 1986). Fish were either processed on shore or placed in individually labeled plastic bags, preserved using ice, and later processed in a laboratory. One otolith from each fish was randomly selected and embedded in epoxy. Using a low-speed saw (Buehler Inc., Lake Bluff, Illinois), a 0.55-mm section of each otolith was cut through the transverse plane of the otolith to expose a cross-section of the nucleus. Sectioned otoliths were polished and then photographed in immersion oil using reflected light at 40x magnification with a Leica (model DFC450 C) digital camera and a Leica (model DM 4000 B) compound light microscope. Photographs were reviewed by two independent readers who were unaware of fish length, and age was estimated by enumerating presumptive annuli. In cases where the readers did not agree on the age of the fish, fish length was considered to determine a consensus age.

A combination of length, weight, and age data were used to compare growth rates and body condition (i.e., linear regression of log[length] on log[weight]) between hatchery M_{YY} and wild Brook Trout. Comparisons of growth rate and body condition between the two groups (hereafter we refer to them as wild and hatchery “strains”) were conducted using linear regression and von Bertalanffy growth models (von Bertalanffy 1938) in statistical software R (R Development Core Team 2021). Growth was modeled using either linear regression or a von Bertalanffy growth model because preliminary analysis indicated that growth was asymptotic in one water (i.e., Dry Creek) but linear in the remaining three waters. Asymptotic growth was estimated by fitting a von Bertalanffy growth function (von Bertalanffy 1938), and linear growth was estimated by fitting a linear regression model (Ogle et al. 2017). Within the asymptotic growth model, the effect of strain on growth was evaluated by estimating the theoretical maximum average length the population could achieve (L_∞), the Brody growth coefficient (K), and the theoretical age when length equals zero (t_0) for each strain. Ninety-five percent confidence intervals (CIs) were estimated for all parameters, and parameter estimates were considered statistically different between wild and hatchery strains if the CIs did not overlap (Ogle et al. 2017).

Linear growth models were developed with length at capture as the response variable; predictor variables were the estimated age of the fish at capture (age), a categorical variable that designated the fish as either M_{YY} or wild (strain), and an age × strain interaction term. The interaction term was used to evaluate if the length at a given age varied based on the strain of the fish. Age was chosen as a predictor variable due to its relationship with length in fish species that exhibit indeterminate growth (Neumann et al. 2012). Ninety-five percent CIs were constructed for each parameter estimate, and growth was considered statistically different between wild and hatchery M_{YY} Brook Trout if the interaction term in the model produced 95% CIs that did not overlap zero (Johnson 1999).

Body condition models were developed with loge transformed weight as the response variable. Predictor variables included loge transformed length at capture, strain, and a length × strain interaction term to evaluate if body condition varied based on the strain of the fish. The loge transformations linearized these data (Quinn and Deriso 1999). Statistical differences in body condition between wild and hatchery M_{YY} Brook Trout were considered to exist if the interaction term in the model produced 95% CIs that did not overlap zero (Johnson 1999).

RESULTS

In total, 381 Brook Trout were sampled across all four waters. Age estimates varied between 1-6 years for wild Brook Trout and 1-5 years for hatchery M_{YY} Brook Trout. Total length varied from 103-359 mm for wild Brook Trout and 115-353 mm for hatchery M_{YY} Brook Trout.

In Dry Creek, where growth was asymptotic, K and L_∞ were estimated at 0.37 (95% CI = 0.17–0.59) and 357 mm (311–500 mm), respectively, for hatchery M_{YY} Brook Trout, and 0.51 (0.28-0.81) and 306 mm (273–378 mm) for wild Brook Trout. In the remaining waters, where growth was linear, hatchery M_{YY} Brook Trout grew an estimated 24-43 mm per year, whereas wild Brook Trout grew an estimated 36-42 mm per year. However, analysis of growth models indicated that these differences were not significant (Figure 6; Table 9). Similarly, evaluation of body condition models indicated that body condition did not differ significantly between wild and hatchery M_{YY} Brook Trout (Figure 7; Table 10).

DISCUSSION

Results of this study indicate that hatchery M_{YY} Brook Trout stocked into mountain streams and alpine lakes as age-0 fingerlings grew at a similar rate and maintained a similar body condition as did wild Brook Trout. These results are contrary to much of the existing literature demonstrating poorer performance for hatchery salmonids relative to their wild counterparts (reviewed in Araki et al. 2008). For example, hatchery salmonids generally demonstrate poorer survival (Miller 1954; Jonsson et al. 2003), slower growth (Finstad and Heggberget 1993; Bohlin et al. 2002), and reduced reproductive fitness (reviewed in Christie et al. 2014) compared to wild salmonids in the same environments. Kennedy et al. (2018a) reported slightly reduced reproductive fitness for hatchery M_{YY} Brook Trout relative to wild conspecifics in several mountain streams, though their study was conducted on catchable-sized fish (as compared to fingerlings in the present study), and they did not compare growth or condition between wild and M_{YY} fish. Taken together, the results of Kennedy et al. (2018a) and the present study suggest that hatchery M_{YY} fish stocked in lentic and lotic waters may survive and grow similarly to wild fish, but once they reach maturity, they may have comparatively lower reproductive fitness. However, since these are the first studies ever to evaluate M_{YY} vertebrates liberated into the wild, more research is clearly needed on all aspects of their post-release performance.

An inherent limitation with our study design was that wild Brook Trout populations in two study waters - Dry Creek and Seafoam Lake #4 - were subjected to manual suppression for several years, which could have affected the growth and condition of fish in our study. However, there was no evidence that growth or condition differed in suppression and non-suppression waters for either wild or hatchery fish. The lack of a suppression effect on fish growth and condition in our study may be related to the well documented ability of Brook Trout to undergo compensatory responses to population changes (McFadden 1961, 1976, 1977; Meyer et al., 2006). The term ‘‘compensation’’ (taken from McFadden 1977) refers to the propensity of populations to exhibit reduced death rates or increased birth rates as a population declines. Previous studies have indicated that Brook Trout may compensate for increased exploitation (in our case, suppression) through a variety of methods, including decreased natural mortality (McFadden 1961; Meyer et al. 2006). A reduction in natural mortality may have sufficiently compensated for the suppression in our study waters such that growth and condition changes for either wild or hatchery fish did not materialize.

Our study had two other limitations. First, as noted above, we were limited by the constraints of the ongoing long-term M_{YY} study, meaning that we could not sacrifice many fish, and thus our sample size at each water was relatively low, as was our replication of study waters. Second, the wild components of the Brook Trout populations were composed of both male and female individuals, whereas the hatchery M_{YY} components of the populations were inherently composed of only males. In wild Brook Trout populations, male Brook Trout often grow faster than females (e.g., Hoover 1939; McFadden 1961; Toetz et al. 1991), so had we sexed the fish, we could have compared the growth of hatchery males to wild males. However, male Brook Trout do not always grow faster than females (e.g., Curry et al. 2003), and even when they do, the growth difference between sexes for Brook Trout is usually only a few millimeters at each age, so we consider this limitation minor.

Despite these study limitations, our results clearly indicate that hatchery M_{YY} Brook Trout can survive for several years, grow at an equivalent rate, and maintain an equivalent body condition relative to wild fish in both alpine lakes and mountain streams. Notwithstanding potentially reduced reproductive fitness (see Kennedy et al. 2018a), the results of the present study support the use of hatchery M_{YY} fish stocking as a prospective means of biological control for invasive fishes. Considering the general lack of information comparing M_{YY} vertebrates to wild counterparts, however, more research is clearly needed to confirm or refute this supposition.

RECOMMENDATIONS

1. Continue the larger M_{YY} Brook Trout study to evaluate the effectiveness of M_{YY} Brook Trout at eradicating unwanted Brook Trout populations.

2. Once the larger M_{YY} Brook Trout study has been completed, and more M_{YY} fish can be sacrificed, conduct this study again with larger sample sizes.

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TABLES

Table 8. Physical characteristics, treatment type, and initial stocking dates of four Idaho waters in which hatchery M_{YY} (i.e., males with two Y chromosomes) and wild Brook trout were sampled for growth and body condition comparisons.

Parameter	Seafoam Lake #4	Lloyd Lake	Dry Creek	Tripod Creek
Latitude	44.508	45.193	44.127	44.318
Longitude	-115.126	-116.164	-113.568	-112.076
Initial M_{YY} stocking year	2017	2015	2016	2016
Annual M_{YY} stocking number	1,176	1,194	4,326	6,938
Wild fish suppression	Yes	No	Yes	No
Surface area (ha)	2.7	2.9	-	-
Reach length (km)	-	-	6.5	9.1
Average wetted width (m)	-	-	5.2	1.4
Gradient (%)	-	-	1.5	1
Elevation (m)	2,423	2,092	2,377	2,146

Table 9. Parameter estimates from linear regression and von Bertalanffy growth models used to estimate growth for hatchery M_{YY} (i.e., individuals with two Y chromosomes) and wild Brook Trout in four Idaho waters. Lower (LCI) and upper (UCI) bounds for 95% confidence intervals are also included.

Parameter	Estimate	LCI	UCI
Dry Creek (VBGF)			
$L\infty$ (Wild)	306	273	378
$L\infty$ (YY)	357	311	500
K (Wild)	0.51	0.28	0.81
K (YY)	0.37	0.17	0.59
t_0 (Wild)	-0.31	-0.92	0.06
t_0 (YY)	-0.63	-1.40	-0.19
Lloyd Lake (Linear)			
Intercept	118.74	90.06	147.42
Age	38.63	27.01	50.25
Strain (YY)	20.93	-15.23	57.09
Age × Strain	-6.75	-22.29	8.78
Seafoam Lake #4 (Linear)			
Intercept	124.28	115.29	133.27
Age	41.71	38.46	44.96
Strain (YY)	10.98	2.31	34.53
Age × Strain	1.15	-5.31	7.60
Tripod Creek (Linear)			
Intercept	95.77	83.98	107.56
Age	35.66	29.95	41.37
Strain (YY)	30.21	9.59	50.83
Age × Strain	-11.55	-24.39	1.29

Table 10. Parameter estimates for linear regression body condition models used to estimate weight by length relationship for hatchery M_{YY} (i.e., individuals with two Y chromosomes) and wild Brook Trout at all four study waters. Lower (LCI) and upper (UCI) bounds for 95% confidence intervals are included.

Parameter	Estimate	LCI	UCI
Dry Creek			
Intercept	-11.49	-12.55	-10.42
Log (Length)	3.03	2.83	3.23
Strain (YY)	-0.39	-1.86	1.07
Log (Length) × Strain	0.05	-0.22	0.33
Lloyd Lake			
Intercept	-11.05	-12.03	-10.07
Log (Length)	2.91	2.73	3.10
Strain (YY)	-0.73	-2.23	0.76
Log (Length) × Strain	0.13	-0.15	0.42
Seafoam Lake #4			
Intercept	-10.23	-10.76	-9.70
Log (Length)	2.80	2.70	2.90
Strain (YY)	-1.08	-2.20	0.04
Log (Length) × Strain	0.17	-0.04	0.37
Tripod Creek			
Intercept	-12.93	-13.75	-12.11
Log (Length)	3.27	3.11	3.43
Strain (YY)	0.71	-1.29	2.70
Log (Length) × Strain	-0.16	-0.56	0.23

FIGURES

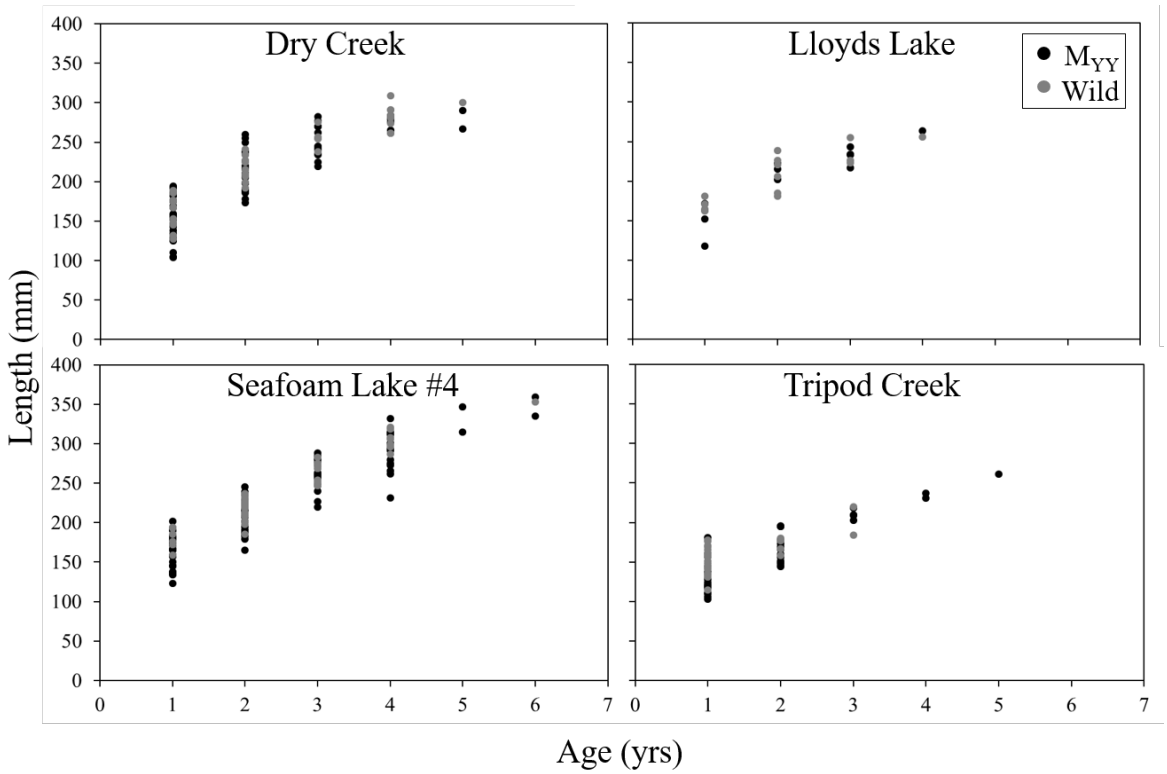

Figure 6. Length at age relationships for hatchery M_{YY} (i.e., individuals with two Y chromosomes; M_{YY}) and wild Brook Trout (Wild) sampled in four Idaho waters. The dots represent individual fish.

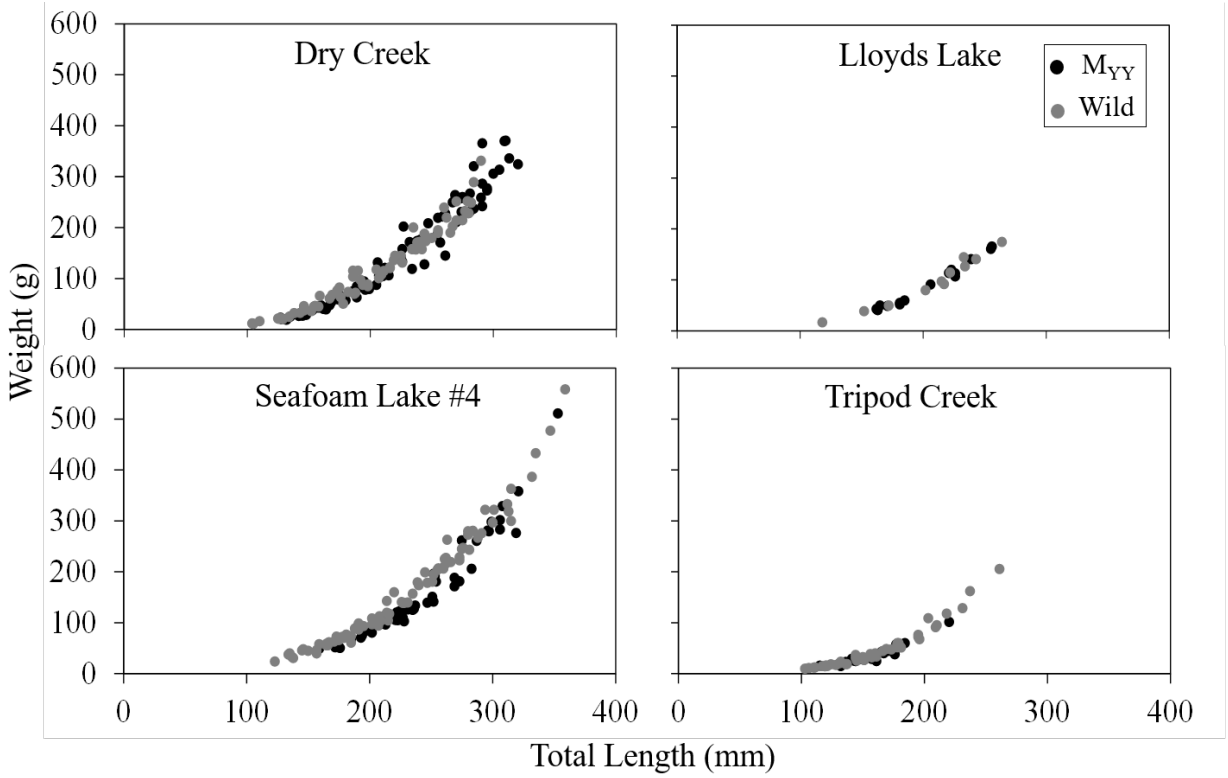

Figure 7. Weight to length relationships for hatchery M_{YY} (i.e., individuals with two Y chromosomes; M_{YY}) and wild Brook Trout (Wild) sampled in four Idaho waters. The dots represent individual fish.

ANNUAL PROGRESS REPORT

SUBPROJECT #3: EVALUATING METHODS OF CAPTURING JUVENILE TROUT IN ALPINE LAKES

State of: Idaho

Project No.: 3 Title: Wild Trout Evaluations

Subproject #3: Evaluating Methods of Capturing Juvenile Trout in Alpine Lakes

Time Period: July 1, 2021 to June 30, 2022

ABSTRACT

Fisheries managers sample alpine lakes for a variety of reasons, some of which necessitate the capture of juvenile fish, which are often difficult to catch with conventional sampling gears. We sampled six alpine lakes to compare the catch of juvenile Brook Trout *Salvelinus fontinalis* using backpack electrofishing, floating gill nets, and baited minnow traps to evaluate whether adding artificial light to baited minnow traps augmented their catch. Electrofishing was the most effective sampling technique, and gill nets and minnow traps were equally ineffective. Additionally, artificial light failed to augment the catch of minnow traps.

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INTRODUCTION

Fisheries managers sample fish in alpine lakes for a variety of reasons. Most commonly, alpine lake surveys are conducted to evaluate size structure (Milbrink et al. 2008), index abundance (James and Graynoth 2002), or tag fish in order to estimate angler use or harvest (Cassinelli et al. 2019). Fisheries surveys in alpine lakes may also be conducted to evaluate growth rates (Thaulow et al. 2017), to eradicate nonnative fish species (Knapp and Matthews 1998), or to determine if natural reproduction is occurring (Armstrong and Knapp 2004; Messner et al. 2013). In these latter instances, it is important to collect not only adult fish, but also subadults, especially fry.

Gill netting is typically the most appropriate gear to sample fish in alpine lakes (Lester et al. 2009), but this gear usually has a selectivity bias against smaller fish (Miranda and Colvin 2017), including when targeting salmonids (Hansen et al. 1997; Roth et al. *In Press*). Electrofishing is another gear commonly used to sample fish in lentic systems (Miranda and Boxrucker 2009), and while there is also a size selective bias to electrofishing (Meyer and High 2011), it is nevertheless quite effective at sampling even the smallest fish (Randall 1982; Mitro and Zale 2002). The primary restriction to using electrofishing in alpine lakes is that such areas are usually rather inaccessible, although backpack electrofishers can be used in shallow lentic areas (Vaux et al. 2000) and are light enough to transport to remote areas such as alpine lakes.

A third potential gear to target the capture of small fish in alpine lakes is baited minnow traps (Rowe et al. 2003). While minnow traps have shown promise in sampling juvenile salmonids in both lentic (Stauffer 1981) and lotic (Bloom 1976) environments, to our knowledge the use of minnow traps in alpine lakes to capture juvenile salmonids has not been evaluated. Moreover, a direct comparison of the relative sampling efficiency of all three gears to capture juvenile salmonids has also not be undertaken, though Vaux et al. (2000) found that in lowland lakes <20 ha, backpack electrofishing (from a small boat) was nearly as efficient as barge electrofishing and was more efficient than gill netting and minnow traps at assessing species richness. Given the potential limitations of all of these techniques, our study had two objectives: 1) determine the relative sampling efficiency of backpack electrofishing, gill netting, and baited minnow traps for sampling juvenile trout in alpine lakes; and 2) determine if the inclusion of artificial light sources would augment the catch of juvenile trout in baited minnow traps by taking advantage of the positive phototaxic behavior of salmonids (Gibson and Keenleyside 1966; Tabor et al. 2017, 2021).

METHODS

This study was conducted in six alpine lakes located in central Idaho during the summer of 2021 (Table 11). Alpine lakes varied from 2.6 to 15.8 ha in surface area, from 2,092 to 2,265 m in elevation, and from 4 to 13 µS/cm in specific conductivity. Based on earlier gill netting surveys (Kennedy et al. 2017), Brook Trout *Salvelinus fontinalis* were the predominant species captured in every lake. Rainbow Trout *Oncorhynchus mykiss* were reportedly present in one of the lakes according to previous unpublished surveys (P. Kennedy, Idaho Department of Fish and Game, personal communication), but they were not encountered during the present study.

Three floating experimental gill nets (36 m long and 1.8 m deep) were set at each lake, consisting of nylon mesh panels of 10.0, 12.5, 18.5, 25.0, 33.0, and 38.0 mm bar mesh. Gill nets were set perpendicular to the shoreline with the small-mesh end tied to the shoreline.

Nine minnow traps (25.4 × 25.4 × 43.2 cm; 3.2 mm square mesh) were deployed at each lake, using three different bait configurations: (1) a single 141 g punctured can of tuna in oil (control); (2) a can of tuna and a submersible light emitting diode light (LED; 2,541 lux); and, (3) a can of tuna and a diphenyl oxalate chemical light stick (i.e., a glow stick; 45 lux). Both artificial light sources emitted green wavelength light. Once configured, minnow traps were divided into three groups with three traps in each group (i.e., one of each configuration).

For both gill nets and minnow traps, the best locations for deployment were established based on the presence of tributaries, appropriate water depth, the dominate substrate at the location, the visual presence of juvenile fish, and the presence of cover such as large woody debris or aquatic vegetation (Wurtsbaugh et al. 1975; Tabor and Wurtsbaugh 1991; Graynoth 1999; Tiberti et al. 2017). However, for gill nets, locations were also selected to maximize catch of all sizes of fish, not just juveniles. For minnow traps, traps were set at what were considered to be the nine best locations based on professional experience. Minnow traps from group one were randomly assigned to sites ranked one to three, traps from group two were randomly assigned to sites four to six, and traps from group three were randomly assigned to sites seven to nine. In an attempt to ensure that the artificial light from one minnow trap did not interfere with other traps in the lake, traps were set so that the minimum distance between traps was at least two secchi depths. Both gill nets and minnow traps were set in the afternoon and retrieved the following morning in the order in which they were set.

Daytime backpack electrofishing was conducted by wading shallow water (<1 m deep) along the shoreline and sampling the nearest 2-3 m of littoral habitat. The electrofisher consisted of a 1.8-m electrical pole and 28-cm stainless-steel ring (1.0-cm thickness) anode and a trailing 3.3-m braided stainless steel rat tail (0.5-cm thickness) for the cathode. The unit was set at pulsed DC current using 60 Hz, 25% duty cycle, and enough voltage (500–1,000 V) to produce ~100 watts of average power output, which has been demonstrated to be effective for sampling salmonids in small streams (Meyer et al. 2021). Daytime electrofishing was always conducted after gill nets and minnow traps were pulled.

All captured Brook Trout were measured for total length (mm), and were considered to be juvenile fish when they were <100 mm (cf. Roth et al. *In press*). The total time each sampling technique was actively sampling fish was recorded to account for differences in effort between sampling gears.

Juvenile Brook Trout catch among gear types were compared using Analysis of Variance (ANOVA). Although preliminary analysis indicated that the data deviated somewhat from a normal distribution, ANOVA is robust to violations of normality (Zar 1999) and is more powerful than nonparametric methods (Sokal and Rohlf 1995). For gear comparisons, minnow trap catch was pooled across trap configurations because there was no difference between configurations (see results). Tukey's multiple comparisons test was used to determine group differences (Zar 1999). Prior to analysis, all catch data were converted to catch-per-unit-effort (CPUE) by dividing the total catch by the length of time the gear was actively sampling. All statistical analysis was conducted in statistical package R (R Development Core Team 2021) with alpha level set at 0.05. No formal test was used to compare catch between minnow trap configurations because the vast majority of traps (95%) failed to capture juvenile Brook Trout, and catch was very low for the few traps that did capture fish.

RESULTS

In total, 207 juvenile Brook Trout were sampled across all six lakes (Table 12). The majority of fish were captured via electrofishing (*n* = 150), but fish were also captured via gill nets (*n* = 39) and minnow traps (*n* = 18). Juvenile Brook Trout were captured at every lake and captured Brook Trout <100 mm TL averaged 47.8 mm (SE = 1.3) for electrofishing, 91.4 mm (SE = 0.9) in gill nets, and 35.2 mm in minnow traps (SE = 2.4; Figure 8).

Results of the ANOVA indicated that CPUE was significantly different between gear types (*df* = 2, *F* = 8.43; *P* = 0.004). Results of the Tukey's multiple comparisons test indicated that CPUE was significantly higher for backpack electrofishing ($\bar{X}$ = 50.0; SE = 17.19) than for floating gill nets ($\bar{X}$ = 0.17; SE = 0.04) or minnow traps ($\bar{X}$ = 0.02; SE = 0.02), but gill net and minnow trap catch did not differ.

The addition of artificial light did not influence minnow trap CPUE (Table 13). Of the fish captured via minnow traps, one fish was captured in a control trap in Black Lake, two fish were captured in a control trap in Duck Lake, and 14 fish were captured in a LED trap in Duck Lake (Table 13). All other minnow traps failed to capture juvenile Brook Trout.

DISCUSSION

Based on CPUE of the three gears, backpack electrofishing was substantially more effective at capturing juvenile Brook Trout in the littoral areas of alpine lakes than were floating gill nets or baited minnow traps. Backpack electrofishing is generally used to sample fish in flowing water, but it can also be effective for sampling small fish in littoral lake habitat (e.g., Schoenebeck et al. 2005; Poole and Bajer 2019). In the low-conductivity (<20 µS/cm) alpine lakes included in the present study, daytime electrofishing resulted in the capture of almost four times as many fish as overnight gill netting and over eight times as many fish as overnight baited minnow traps. Vaux et al. (2000) found that nighttime backpack electrofishing was more efficient at characterizing lake fish assemblage than gill nets, trap nets, minnow traps, and seines, even at conductivities <30 µS/cm. Most juvenile salmonids in the lakes we surveyed were likely small enough to pass through the smallest gill net mesh, as evidenced by the fact that no fish <77 mm were captured via gill net, whereas 93% of the fish captured via electrofishing were <77 mm. Nevertheless, while CPUE was very low for both gill nets and minnow traps, gill nets captured juvenile trout at all but one lake, including one lake where electrofishing failed to capture juvenile fish, whereas minnow traps captured juvenile trout in only two lakes.

Baited minnow traps were ineffective at capturing juvenile Brook Trout in alpine lakes, regardless of the addition of artificial light. Considering that only one light color was tested in the current study, it is possible that a different color light would have yielded different results. Indeed, light color has been demonstrated to influence the effect of light as an attractant for White Sturgeon *Acipenser transmontanus* (Ford et al. 2018) and Lumpfish *Cyclopterus lumpus* (Foss et al. 2020). Light intensity of the two artificial light sources used in the current study may also have been inappropriate to illicit a positive phototaxic response (Gibson and Keenleyside 1966; Tabor et al. 2017, 2021). Gibson and Keenleyside (1966) reported that light intensities outside of the optimal range (4–1,722 lux) could actually cause Brook Trout to exhibit negative phototaxic behavior, although only one of the two light sources in the present study (LED) were outside this optimal range. Alternatively, low catch rates observed in the current study could simply indicate that minnow traps are ineffective at sampling juvenile salmonids in alpine lakes, although they have been successful in other settings (e.g., Bloom 1976; Stauffer 1981; Bryant 2000).

Although electrofishing appeared to be the most effective gear in the current study in terms of CPUE, catch is not the only consideration when choosing sampling gear (Hayes et al. 2012). For instance, the remote nature of many alpine lakes coupled with the fact that electrofishing requires an electrofishing unit and an energy source may prohibit the use of this technique at many alpine lakes. In such cases, gill netting may be the best choice of gear, as they are easily transported and deployed.

RECOMMENDATIONS

1. When feasible, use backpack electrofishing to sample lakes for juvenile salmonids.

2. In cases where backpack electrofishing is not feasible for sampling alpine lakes use gill nets, but other gears (e.g., dip nets) may be viable options.

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TABLES

Table 11. Characteristics of several alpine lakes in central Idaho where juvenile Brook Trout (<100 mm) were sampled using backpacking electrofishing, floating gill nets, and minnow traps.

Water	Secchi depth (m)	Surface area (ha)	Elevation (m)	Conduct-ivity (µS/cm)	Latitude	Longitude
Black Lake	1.8	2.6	2,149	6	45.24539	-116.19867
Duck Lake	4.6	4.9	2,177	6	45.11459	-116.15726
Lloyds Lake	5.2	2.9	2,092	10	45.19290	-116.16370
Rainbow Lake	5.2	8.8	2,175	7	45.25406	-116.19663
Snowslide Lake #1	4.2	4.9	2,188	13	44.50766	-115.93431
Upper Hazard Lake	5.9	15.8	2,265	4	45.17423	-116.13500

Table 12. Juvenile Brook Trout (<100 mm) catch, effort, and catch-per-unit-effort (CPUE) in alpine lakes surveyed with various gears in central Idaho.

	Backpack electrofishing			Floating gill nets			Minnow traps		
Lake	**Catch**	**Effort (hrs)**	**CPUE**	**Catch**	**Effort (hrs)**	**CPUE**	**Catch**	**Effort (hrs)**	**CPUE**
Black Lake	24	0.5	48.00	11	38.2	0.29	1	112.5	0.01
Duck Lake	32	0.5	64.00	6	41.5	0.14	17	127.8	0.13
Lloyd Lake	1	0.5	2.00	0	41.7	0.00	0	123.6	0.00
Rainbow Lake	51	0.5	102.00	11	38.7	0.28	0	109.2	0.00
Snowslide Lake #1	0	0.5	0.00	5	40.1	0.12	0	113.1	0.00
Upper Hazard Lake	42	0.5	84.00	6	40.1	0.15	0	120.3	0.00
Average	25	0.5	50.00	7	40.1	0.17	3	117.8	0.02

Table 13. Minnow trap (<100 mm) catch-per-unit-effort (catch/hr) of Brook Trout in seven alpine lakes in central Idaho. Configurations included three replicates (groups) of (1) baiting the trap with a single 141 g punctured can of tuna in oil Cntrl), (2) baiting the trap with tuna in oil and a submersible light emitting diode light (LED), and (3) baiting the trap with tuna in oil and a diphenyl oxalate chemical light stick (Gstick).

	Group 1			**Group 2**			**Group 3**		
Lake	**Cntrl**	**LED**	**Gstick**	**Cntrl**	**LED**	**Gstick**	**Cntrl**	**LED**	**Gstick**
Black Lake	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	0.00
Duck Lake	0.14	0.97	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Lloyd Lake	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rainbow Lake	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Snowslide Lake #1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Upper Hazard Lake	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

FIGURES

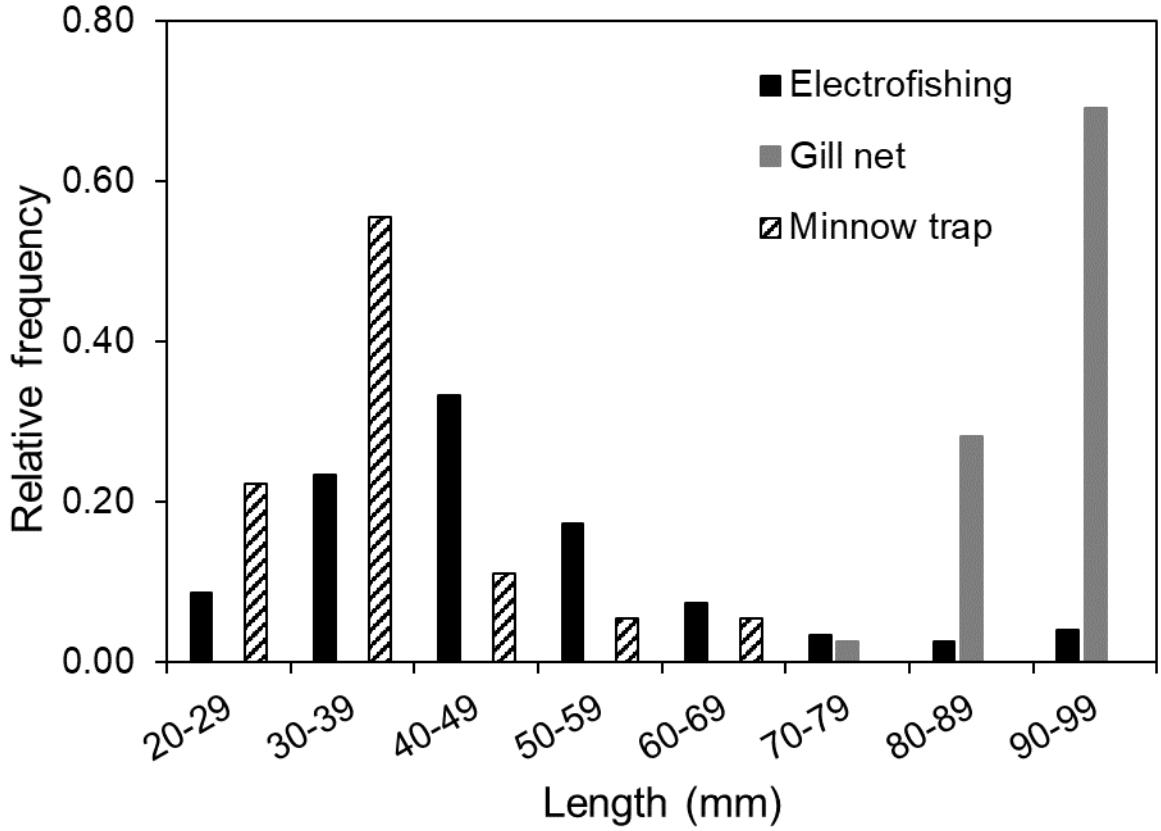

Figure 8. Length frequency distribution of juvenile Brook Trout (<100 mm) sampled in Idaho alpine lakes during the summer of 2021. Fish were captured via backpack electrofishing, gill net, and baited minnow trap.

ANNUAL PROGRESS REPORT

SUBPROJECT #4: FACTORS RELATED TO THE DISTRIBUTION AND ABUNDANCE OF WESTSLOPE CUTTHROAT TROUT IN CENTRAL IDAHO

State of: Idaho

Project No.: 3 Title: Wild Trout Evaluations

Subproject #4: Factors Related to the Distribution and Abundance of Westslope Cutthroat Trout in Central Idaho

Time Period: July 1, 2021 to June 30, 2022

ABSTRACT

Native resident salmonids throughout North America have experienced population declines, and understanding factors that influence their contemporary distribution and abundance may help conserve and manage such species. We examined the influence of several environmental factors on the current distribution and abundance of Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* in central Idaho, based on snorkel survey data collected from 2010–2019. In total, 2,758 snorkel surveys were conducted at 1,000 sites; Cutthroat Trout were present during 1,277 of the surveys, and their occupancy rate was higher if Brook Trout *Salvelinus fontinalis* were absent (0.31) than present (0.48). During surveys where Cutthroat Trout were present, mean density was 1.81 fish/100m^2. Underlying lithology was associated with Westslope Cutthroat Trout distribution but not their abundance, suggesting that lithology may influence broader habitat features that affect their ability to fulfill a component of their life history, such as spawning or overwinter survival, more so than characteristics that affect their abundance, such as microhabitat suitability. Not surprisingly, Westslope Cutthroat Trout occupancy was negatively influenced by the abundance of nonnative Brook Trout, but in central Idaho this affect is tempered by the limited distribution of Brook Trout. Both the occupancy and the abundance of Westslope Cutthroat Trout were related in a non-linear, dome-shaped manner to site elevation; considering that elevation was included as a surrogate for stream water temperature, which is also commonly related to trout occupancy and abundance in a dome-shaped manner, this suggests that intermediate stream elevations (in central Idaho, 800 to 1,600 m) currently provide an ideal thermal regime for Westslope Cutthroat Trout.

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INTRODUCTION

Native resident salmonids throughout North America have experienced declines in distribution and abundance, including species in the Pacific Northwest such as Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* (Shepard et al. 2005). In response to these declines, fisheries managers and various partners have developed recovery plans, multistate conservation agreements and strategies, and restoration projects (e.g., Lohr et al. 2000; Hirsch et al. 2006). However, for management actions to be effective, they must be developed with a good understanding of factors contributing to the status of the species (Milner et al. 1993).

Factors limiting the distribution and abundance of Westslope Cutthroat Trout have been repeatedly investigated across their range (e.g., Sloat et al. 2005; D'Angelo and Muhlfeld 2013; Peterson et al. 2014; Heckel et al. 2020; Heinle et al. 2021). However, the relationship between environmental conditions and the status of this species can vary among populations. For example, road density has been reported to be positively associated with Westslope Cutthroat Trout abundance in the St. Maries River basin, Idaho (Heckel et al. 2020) but was identified as a key limiting factor in British Columbia streams (Valdal and Quinn 2011). Such regional disparities in factors associated with the status of this species highlight the importance of determining limiting factors at geographic scales appropriate for regional management.

The Salmon River and Clearwater River basins of central Idaho comprise a large contiguous network of stream habitats in mountainous terrain that is dominated by coniferous forests at higher elevations (up to 3,800 m in elevation) and sagebrush–grass steppe at lower elevations. Over 80% of the study area is publicly owned, and nearly 25% is designated wilderness, with many large expanses functioning as de facto wilderness. Due to the high elevation, remoteness, and relatively pristine nature of this ecosystem, central Idaho serves as a stronghold for Westslope Cutthroat Trout (Kennedy and Meyer 2015). Nevertheless, the distribution and abundance of the species in this area is patchy (Shepard et al. 2005). To better understand what biotic and abiotic factors contribute to this patchiness, we investigated landscape-level factors that might be influencing the contemporary distribution and abundance of Westslope Cutthroat Trout in central Idaho.

METHODS

Study Area

The current study incorporated data from the Clearwater River and Salmon River basins of central Idaho (Figure 9). The Clearwater River originates in the Bitterroot Mountains and has a drainage area of approximately 25,000 km^2 and a mean basin elevation of 1,311 m. Originating in the Sawtooth Mountains, the Salmon River has a larger drainage area of approximately 37,000 km^2 and a higher mean basin elevation of 2,020 m. Salmonid species present in these river basins include Westslope Cutthroat Trout, Bull Trout *Salvelinus confluentus*, Brook Trout *S. fontinalis*, lake trout *S. namaycush*, mountain whitefish *Prosopium williamsoni*, Chinook salmon *O. tshawytscha*, coho salmon *O. kisutch*, and resident and anadromous forms of *O. nerka* and *O. mykiss.*

Fish Surveys

Westslope Cutthroat Trout distribution and abundance were assessed via daytime snorkel surveys conducted from 2010–2019 as part of the Idaho Department of Fish and Game's Natural

Production Monitoring and Evaluation Program. These surveys typically occurred from June to August each year. Sites were selected either subjectively to represent the general habitat of the waterbody of interest, or using a generalized random-tessellation stratified design (see Apperson et al. 2015 for details). Survey crews attempted to survey approximately 100 linear meters of stream, but upstream and downstream site boundaries were adjusted to fit within hydraulic controls (Apperson et al. 2015). Because these data were from a long-term monitoring program, some sites (38%) were surveyed more than once during the study period. The frequency with which each site was surveyed was subjective depending on crew size, annual streamflow variation, and regional fisheries management emphasis.

For each snorkeling survey, fish counting protocols followed those described in Thurow (1994). In short, one or more snorkelers moved upstream or downstream visually observing and recording fish in all available habitat. Maximum underwater visibility at each site was measured with a tape measure prior to the snorkel survey. The measurement of maximum underwater visibility was used to determine how many snorkelers were required to ensure that the distance between snorkelers did not exceed the visibility. Snorkel surveys were predominantly conducted in an upstream direction except on occasions (approximately 10% of the surveys) when water velocities were too high or when the water was too deep for the snorkelers to survey in that direction. Each snorkeler recorded all observed fish, identifying fish to species based on phenotypic characteristics and recording fish length to the nearest 25 mm (total length). Snorkelers did not record any observed fish <50 mm due to difficulty in identifying those fish to species. Fish density for each survey was standardized to fish/100m^2, but it should be recognized that densities of stream-dwelling salmonids as determined from snorkel surveys are inherently underestimated because detection probability is not 100% (Thurow and Schill 1996; Mullner et al. 1998; Korman et al. 2010). However, we assumed the bias in abundance estimates was equivalent across all surveys.

Environmental variables

Several site-level and landscape-level measurements were made, either in the field at the time of snorkeling or later using a geographic information system (GIS), to characterize stream or watershed environmental conditions. During each field survey, instantaneous water temperature (°C) was recorded and was included in our analyses because water temperature influences daytime concealment behavior in salmonids, which can directly alter their detection probability and thus their visual abundance (O'Neal 2007). Stream width (m) at the site was estimated by averaging wetted width measurements collected every 10 m throughout the reach, and was included because stream size influences habitat complexity and biotic integrity (Fausch et al. 1984). The density of Brook Trout (fish/100m^2) was included as an explanatory variable because they consistently have a negative effect on Westslope Cutthroat Trout occupancy and abundance (Dunham et al. 2002; Shepard 2004; Heckel et al. 2020).

Using a GIS, stream slope (%) at each site was estimated using the National Hydrography Plus Version 2 dataset (McKay et al. 2012); stream slope was included in our analyses to account for the influence it has on stream habitat characteristics (Bozek and Hubert 1992; Isaak and Hubert 2000; Wenger et al. 2011). However, sites where slope exceeded 15% were not included in the analysis (n = 1) because they rarely support salmonid populations (Isaak et al. 2018). The National Hydrography Plus Version 2 dataset was also used to estimate stream order. Similar to wetted width, stream order was included as a measure of stream size (Vannote et al. 1980) to account for its effect on fish assemblage (Fausch et al. 1984) and abundance (Eklöv et al. 1999).

Elevation (m) at each site was estimated using a digital elevation model in ArcMap 10.6 (Environmental Systems Research Institute, Redlands, California) and was included in our analyses to account for the influence it often has on stream-dwelling salmonids (Jowett et al. 1996; Dunham and Rieman 1999; Rieman et al. 2006). Conductivity was estimated at each site using the GIS-based model constructed by Olson and Cormier (2019), and was included due to its influence on stream productivity (McFadden and Cooper 1962; Scarnecchia and Bergersen 1987). Lithology was included because it influences stream morphology (Hack 1957; Minshall et al. 1985), substrate particle size (Connolly and Hall 1999), primary productivity (Minshall et al. 1985; Sanderson et al. 2009), and the availability of physical habitat (Baxter and Hauer 2000), all of which can influence salmonid communities (Lanka and Hubert 1987). Lithology at each of our snorkel sites was estimated using the Idaho State Geologic Map at a scale of 1:750,000 (Lewis et al. 2012) and was categorized as acid volcanic (rhyolite), basalt, sedimentary (including alluvium, sandstone and quartzite), shale, and shield (metamorphic and plutonic rock; Suchet et al. 2003).

Road density was included because western native trout are usually less likely to occur and less abundant where there are roads 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 km of road within 10 km^2 (i.e., within a 1.78 km radius) of each survey site. Slope, conductivity, elevation, lithology, and road density measurements were all taken from the downstream end of the snorkel survey site and were considered to be representative of the entire site, as it is unlikely that these factors varied greatly given the relatively short length of the sites (mean = 95.7 m, range = 40.0 – 300.0).

Statistical Analysis

Evaluation of factors affecting Westslope Cutthroat Trout occupancy and abundance was conducted using generalized linear models. Prior to any model construction, we excluded all data collected in the Potlatch River subbasin because although cutthroat trout are present in a few locations in the subbasin, they were never encountered in the snorkel surveys conducted in that subbasin. Multicollinearity among all continuous predictor variables was evaluated with pairwise Pearson correlation coefficients (*r*), but no comparisons exceeded $|r| > 0.70$ (Table 14) so we considered collinearity to be inconsequential (Dormann et al. 2013).

All variables were included in models as fixed effects. Fish density, instantaneous water temperature, and wetted width were averaged across all visits for survey sites with more than one visit during the study period. Averaging these variables across visits alleviated temporal autocorrelation (Sokal and Rohlf 1995) and pseudoreplication (Zar 1999) issues with the data. No such treatment was needed for conductivity, elevation, lithology, road density, stream slope, or stream order, as they were all derived from GIS spatial layers and thus were static values for each site. Instantaneous water temperature was assumed to potentially have a quadratic influence on Westslope Cutthroat Trout distribution and abundance because at low temperatures, salmonid activity is diminished as concealment behavior is triggered (O'Neal 2007), but at high temperatures, activity for salmonids may also be reduced as they seek thermal refuge or cover (Thurow 1994). Elevation is often used as a surrogate for the water temperatures that stream-dwelling fish experience (e.g., Isaak et al. 2010; Eby et al. 2014) and as such, it was also assumed to potentially have a quadratic effect because salmonids such as Westslope Cutthroat Trout have a thermal optima and upper thermal tolerance (Bear et al. 2007).

To relate Westslope Cutthroat Trout occupancy to predictor variables, logistic regression was used with a dummy response variable of one if they were present at a site and zero if they were absent. To relate cutthroat trout abundance (at sites where they were present) to predictor variables, a general linear model (GLM) was used. For both logistic and GLM models, we constructed the following models for comparison: a null (intercept-only) model; a full model with all nine predictor variables included; and nine reduced models, with each model systematically missing one of the predictor variables. Plausible models were considered to be those with Akaike's information criterion (AIC; Akaike 1973) scores within 2.00 of the best (i.e., most parsimonious) model. Akaike weights (w_i) were used to rank the relative plausibility of the candidate models (Burnham and Anderson 2004), whereas adjusted coefficient of determination (R^2; for GLMs) and adjusted pseudo-R^2 ($\tilde{R}^2$, Nagelkerke 1991); for logistic regression) were used to assess the amount of variation explained by the models. The Hosmer and Lemeshow goodness-of-fit statistic (Hosmer et al. 2013) was used to check that the most plausible logistic regression models adequately fit the data, whereas diagnostic analyses of residuals was used for checking the adequacy of model fit for GLMs. Natural log transformations of the fish density data were needed to normalize the residuals of the GLM models.

For both logistic and GLM models, model averaged coefficient estimates and 95% confidence intervals (CIs) were derived only from the plausible models (i.e., those with AIC scores within 2.00 of the best model) and were calculated according to the formulas in Burnham and Anderson (1998). We considered model averaged coefficient estimates with 95% CIs that did not overlap zero to be influential in the occupancy and abundance models. For all statistical analyses, SAS statistical software (SAS Institute 2009) was used.

RESULTS

During 2010–2019, a total of 2,758 snorkel surveys were conducted at 1,000 sites in the Clearwater River and Salmon River basins. Across both basins, conductivity at the survey sites averaged 69 µS/cm (range = 30–327), elevation averaged 1,345 m (range = 278–2,431 m), total km of road in a 10 km^2 radius around survey sites averaged 7.3 km (range = 0–53.2 km^2), stream slope averaged 1.9% (range = 0.0–11.2%), water temperature at the time of the survey averaged 13.0°C (range = 4.0–24.0°C), and mean wetted width averaged 10.8 m (range = 0.8–64.6 m; Table 15). The most common lithology across all sites was shield (46%) followed by shale (21%), sedimentary (17%), basalt (9%), acid volcanic (6%).

Westslope Cutthroat Trout were observed during 1,277 (46%) surveys. They were present during at least one survey at 560 sites, and among the 279 occupied sites that were surveyed more than once, Westslope Cutthroat Trout were more often intermittently present (198 sites) than always present (81 sites). However, even at sites where Westslope Cutthroat Trout were observed intermittently, they were present 65% of the time on average. Brook trout were observed during 288 (10%) surveys and at 176 (18%) sites, and Westslope Cutthroat Trout were more likely to be present during surveys in which Brook Trout were absent (48% of the time) than present (31%).

At the sites occupied by cutthroat trout, their mean density was 1.81 fish/100m^2, but this varied from a low of 0.02 fish/100m^2 to a high of 31.68 fish/100m^2. Mean cutthroat trout abundance was higher in the Clearwater River basin (2.32 fish/100m^2) than in the Salmon River basin (0.86 fish/100m^2; Figure 10). In comparison, Brook Trout mean density (at sites they occupied) was 1.71 fish/100m^2, and this varied from a low of 0.01 fish/100m^2 to a high of 41.43 fish/100m^2. Mean density was >1.0 fish/100m^2 for both species at only three sites (Figure 11).

The most plausible logistic regression models (of those we considered) explaining the variation observed in Westslope Cutthroat Trout occupancy included all predictor variables except either instantaneous water temperature, stream slope, stream order, road density, or stream width (Table 16). Based on model-averaged parameter estimates (from the most plausible models only) with 95% CIs that did not overlap zero, results indicated that Westslope Cutthroat Trout were more likely to occupy sites with lower conductivity, containing fewer Brook Trout, and at an intermediate elevation (Table 16), with occupancy peaking at sites that were 800–1,400 m in elevation and declining at lower or higher elevation (Figure 12). Westslope Cutthroat Trout were also more likely to occupy sites with underlying lithologies of shield (59% occupancy rate) and acid volcanic (54%) than basalt (32%; Table 16).

The most plausible GLM models (of those we considered) explaining the variation observed in Westslope Cutthroat Trout abundance included all predictor variables except either instantaneous water temperature, lithology, or stream slope (Table 16). Based on model-averaged parameter estimates (from the most plausible models only) with 95% CIs that did not overlap zero, results indicated that where Westslope Cutthroat Trout were present, their density was higher in smaller streams with fewer Brook Trout and lower conductivity. Cutthroat trout density peaked at sites that were 1,400–1,600 m in elevation and declined at lower and higher elevation (Figure 10).

DISCUSSION

Westslope Cutthroat Trout remain widely distributed and abundant in central Idaho, especially in comparison to much of the rest of their native range (Shepard et al. 2005; Kennedy and Meyer 2015). Nevertheless, they are certainly not as widespread or as abundant as they were historically. While many biotic and abiotic factors have contributed to the historical decline of the species (Shepard et al. 2005), our results focus on several broad-scale environmental factors that appear to be influencing their contemporary distribution and abundance.

Our results suggest that in central Idaho, Westslope Cutthroat Trout are more likely to be encountered in stream reaches underlain by shield and acid volcanic lithologies than basalt lithology. Lithology influences stream morphology (Minshall et al. 1985), substrate particle size (Connolly and Hall 1999), the productivity of the waterbody (Minshall et al. 1985; Sanderson et al. 2009), and the availability of physical habitat (e.g., overhead cover, aquatic vegetation, and instream cover; Baxter and Hauer 2000), all of which can influence salmonid communities (Lanka and Hubert 1987). For Westslope Cutthroat Trout, the lower probability of occurrence in stream reaches with basalt lithology could be due to the fact that basaltic landscapes tend to produce less complex drainage patterns and more riverine migration barriers than landscapes formed on softer underlying rock (Guy et al. 2008), and both ecosystem complexity and connectivity have been shown to be important for Westslope Cutthroat Trout populations (Pierce et al. 2014). Additionally, basalt lithology typically produces larger stream substrate particle sizes (Kaufmann and Hughes 2006; Kaufmann et al. 2009), and Westslope Cutthroat Trout tend to spawn in areas with relatively small substrates (Magee et al. 1996) compared to other sympatric native salmonids in central Idaho (e.g., Riebe et al. 2014; Guzevich and Thurow 2017), thus Westslope Cutthroat Trout may have more difficulty successfully spawning in stream reaches with basalt lithology than shield and acid volcanic lithologies. The fact that lithology was more influential for fish distribution than abundance suggests that lithology may influence broader habitat features that affect the ability of Westslope Cutthroat Trout to fulfill a component of their life history, such as spawning or overwinter survival, more so than characteristics that affect fish abundance, such as microhabitat suitability. However, surprisingly little research has been conducted on the direct effects of

lithology on fish distribution or abundance (but see Nelson et al. 1992), thus further research is needed to establish better causative links between lithology and fish ecology.

The negative influence that Brook Trout abundance had on Westslope Cutthroat Trout distribution and abundance in our study was not surprising, considering that such a pattern has been observed for many subspecies of cutthroat trout (reviewed in Dunham et al. 2002). A number of potential explanations exist for this relationship, but the most prevalent is competition (Dunham et al. 2002; Peterson et al. 2004). Competition from Brook Trout has often restricted cutthroat trout populations to small, steeper headwater streams where they are more protected from Brook Trout invasion (Shepard et al. 2005), but Brook Trout were present at only 83 (18%) of the 460 sites where Westslope Cutthroat Trout were absent, indicating that in central Idaho, Brook Trout distribution is too limited to be a primary factor influencing contemporary Westslope Cutthroat Trout occupancy. However, due to the relatively intact nature of riverscapes in the Salmon River and Clearwater River basins (e.g., Schoby and Keeley 2011; Feeken et al. 2019), continued expansion of Brook Trout in central Idaho is likely to occur (e.g., Adams et al. 2002; Benjamin et al. 2007) without active suppression in streams where they currently exist.

Given the thermal requirements of Westslope Cutthroat Trout (Bear et al. 2007; Macnaughton et al. 2021), water temperature clearly influences their distribution and abundance (e.g., Shepard 2004). While water temperature was included as a predictor variable in our model, it was based on instantaneous measurements at the time of each snorkel survey and therefore obviously did not represent the thermal regime that fish experienced over the course of each year. Rather, water temperature was included to account for its potential influence on fish behavior and thus their detectability (O'Neal 2007), but it was not useful in explaining the variation we observed in Westslope Cutthroat Trout occupancy or abundance. In contrast, both the occupancy and the abundance of Westslope Cutthroat Trout were related to elevation in a non-linear, dome-shaped manner. Elevation was included as a surrogate for stream water temperature (Isaak et al. 2010; Eby et al. 2014), which has previously been shown to be related to trout abundance in a dome-shaped manner (Isaak and Hubert 2004; Meyer et al. 2010). We speculate that in central Idaho, there is a range of stream elevations (perhaps from 800 to 1,600 m) that currently provides an ideal thermal regime for Westslope Cutthroat Trout, though their distribution in the future will likely shift to higher elevations as stream temperatures continue to warm due to climate change (Isaak et al. 2012).

Road density apparently had no effect on Westslope Cutthroat Trout distribution, but where they were present, higher road density reduced their abundance, which concurs with previous studies reporting similar relationships (e.g., Muhlfeld et al. 2009; Valdal and Quinn 2011). Roads can negatively affect salmonid populations through sedimentation and habitat alteration (Dunham and Rieman 1999), as well as by creating barriers to fish movement (Simpkins and Mistak 2010). The influence of roads on fish occupancy and abundance can be difficult to ascertain using GIS data because the database used to map roads in Idaho has a lag on the inclusion of recently closed or decommissioned roads, and because closed and decommissioned roads, though not actively in use, can negatively affect salmonid communities through legacy effects of increased stream sedimentation, at least until vegetative regrowth can stabilize the soil (McCaffery et al. 2007).

Conductivity was negatively associated with both the distribution and abundance of Westslope Cutthroat Trout, though conductivity is normally associated with the fertility 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 at nearly all of the sites was <100 μS cm^{-1}, which is

considered low for flowing waters (Griffith 2014), thus the negative effect we observed for the limited range of low conductivities in central Idaho streams may have been more correlative than causative in nature. For example, conductivity is generally lower in smaller headwater streams (Wilcox et al. 1956) where we observed Westslope Cutthroat Trout densities to be higher. Moreover, conductivity is 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).

In fact, we fully recognize that for many of the associations reported herein, correlation between predictor variables and the distribution and abundance of Westslope Cutthroat Trout does not necessarily imply a causative relationship; this is a well-recognized weakness of any non-manipulative ecological investigation (Hilborn 2016). In addition, our analyses included only some of the environmental characteristics that may influence the distribution and density of Westslope Cutthroat Trout in central Idaho or elsewhere, thus our models explained only a small amount of the overall variation we observed in Westslope Cutthroat Trout occupancy and abundance. Nevertheless, assuming the general patterns that we observed at least suggest causative links, some findings relevant to the management of Westslope Cutthroat Trout emerge. First, where Westslope Cutthroat Trout were present, road density negatively influenced their abundance, suggesting that restoration of stream habitat impaired by roads could improve the status of the species in central Idaho (Pierce et al. 2013). Second, while management actions to improve degraded stream habitat usually target areas that are most degraded or where funding can be secured, it should also be recognized that in central Idaho, improving habitat in stream reaches underlain with shield and acid volcanic lithology may provide the most benefit to Westslope Cutthroat Trout. Third, efforts to control or eradicate Brook Trout populations would likely improve the security of Westslope Cutthroat Trout populations in central Idaho. Lastly, while protecting central Idaho streams at intermediate elevation may currently provide the most benefit for Westslope Cutthroat Trout, recognition of the future importance of upstream habitat as the climate continues to warm (Isaak et al. 2015) is critical for the long-term persistence of Westslope Cutthroat Trout in central Idaho.

Management Recommendations

Insight from these findings can be extrapolated to individual subbasins in central Idaho. For example, the Middle Fork Salmon River and the Upper Salmon River subbasins are geographically proximate, similar in size and elevation, and have similar stream slopes, but Westslope Cutthroat occupancy and abundance in the Middle Fork Salmon River subbasin was nearly double that observed in the Upper Salmon River subbasin (Table 17). While the Middle Fork Salmon River remains a world-class Cutthroat Trout fishery, the upper Salmon River subbasin does not, although historically, large Cutthroat Trout were abundant throughout that subbasin (Mallet and Thurow 2022). Considering the negative influence that Brook Trout had on Westslope Cutthroat Trout, this discrepancy may be due in part to the fact that Brook Trout occupancy rates (2.5 times) and densities (2.8 times) are much higher in the Upper Salmon River subbasin than in the Middle Fork Salmon River subbasin. Additionally, road density, which was 4.8 times higher in the Upper Salmon River subbasin than the Middle Fork Salmon River subbasin, may be a contributing factor.

In the South Fork Clearwater River subbasin, Westslope Cutthroat Trout occupancy and abundance were surprisingly high considering how dense roads are in the subbasin, and considering the widespread occurrence of Brook Trout. Further spread of Brook Trout in this subbasin is likely. Taken collectively, these patterns highlight the fact that decommissioning roads and minimizing Brook Trout expansion may be useful management actions to conserve

Westslope Cutthroat Trout populations in central Idaho. Focusing such efforts in areas with shield and acid volcanic lithologies may reap the highest benefit for the species.

RECOMMENDATIONS

1. Additional research investigating the effects of other broad-scale environmental factors (e.g., land use, water temperature, and water chemistry) would be useful to better understand their influence on Westslope Cutthroat Trout populations in Idaho.

2. Snorkel sites that have been established in central Idaho should continue to be visited with enough frequency (perhaps at least three times per decade) that trends for Westslope Cutthroat Trout and other resident salmonids can continue to be calculated.

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TABLES

Table 14. Correlation matrix for continuous predictor variables used to explain Westslope Cutthroat Trout occupancy and abundance during snorkel surveys conducted in streams throughout the Clearwater and Salmon river basins of central Idaho.

	ELEV	SLOPE	COND	SO	TEMP	ROAD	WIDTH	BKT
Elevation (ELEV)	1.00							
Slope (SLOPE)	0.16	1.00						
Conductivity (COND)	0.26	0.04	1.00					
Stream order (SO)	-0.42	-0.62	-0.11	1.00				
Instantaneous water temperature (TEMP)	-0.46	-0.36	-0.16	0.55	1.00			
Road density (ROAD)	-0.03	-0.18	0.12	-0.01	0.02	1.00		
Mean wetted width (WIDTH)	-0.44	-0.38	-0.28	0.65	0.42	0.02	1.00	
Brook trout density (BKT)	0.13	-0.01	0.06	-0.15	0.05	0.11	-0.09	1.00

Table 15. Summary of fish survey data (i.e., Brook Trout density, length, survey area, water temperature, and average wetted width) and environmental data (i.e., conductivity, elevation, road density, stream slope, and stream order) collected via snorkel surveys (2010–2019) and geographic information system analysis. See methods for variable descriptions.

Variable	Mean	SD	Minimum	Maximum
Brook trout desnity (fish/100m^2)	0.35	2.01	0.00	41.40
Conductivity (µS/cm)	70	32	30	327
Elevation (m)	1,290	443	278	2,431
Length (m)	95.7	23.5	40.0	300.0
Road density (m)	7,715	8,113	0	53,172
Slope (%)	1.8	1.8	0.0	11.2
Stream order	3	1	1	6
Survey area (m^2)	999	976	59	12,874
Temperature (°C)	13.2	3.1	4.0	24.0
Wetted width (m)	10.3	8.4	0.6	64.6

Table 16. Parameter estimates generalized linear models predicting the distribution and density of Westslope Cutthroat Trout in the Clearwater and Salmon River basins, Idaho. Parameter estimates and 95% confidence (95% CI) intervals have been exponentiated to increase interpretability. See methods for complete variable and model descriptions.

Parameter	Estimate	LCI	UCI
Distribution			
Intercept	0.00	0.00	0.00
Brook Trout density	0.64	0.42	0.86
Conductivity	0.55	0.45	0.67
Elevation	0.92	0.77	1.09
Lithology (acid volcanic)	7.94	3.48	18.56
Lithology (sedimentary)	5.75	2.96	11.50
Lithology (shale)	2.87	1.50	5.62
Lithology (shield)	3.06	1.63	5.90
Road density	0.86	0.74	0.99
Slope	1.26	1.06	1.50
Stream order	0.73	0.60	0.89
Temperature	1.07	0.90	1.26
Density			
Intercept	0.02	0.01	0.04
Brook Trout density	0.94	0.85	1.05
Conductivity	0.85	0.76	0.96
Elevation	1.11	0.98	1.26
Lithology (acid volcanic rock)	0.62	0.32	1.16
Lithology (sedimentary)	0.69	0.40	1.17
Lithology (shale)	0.60	0.34	1.01
Lithology (shield)	0.58	0.33	0.97
Road density	0.72	0.66	0.80
Slope	1.17	1.02	1.36
Stream order	0.55	0.48	0.63
Temperature	1.02	0.90	1.15

Table 17. Summary of environmental data (i.e., conductivity, elevation, road density, and stream slope) and survey data (average wetted width) by major subbasin in central Idaho. Data were collected with either geographic information system analysis or during snorkel surveys from 2010–2019. Values included in parenthesis are equal to one standard deviation. For waterbodies, MF is Middle Fork, NF is North Fork, and SF is South Fork. See methods for complete variable descriptions.

		Westslope Cutthroat Trout		Brook Trout								
Basin	*n*	Occupancy	Density (fish/100m^2)	Occupancy	Density (fish/100m^2)	Conductivity (µS/cm)	Elevation (m)	Predominate lithology	Secondary lithology	Road density (m)	Slope (%)	Average wetted width (m)
Clearwater River												
Clearwater River	68	0.47	3.19 (3.70)	0.18	1.22 (2.14)	70 (28)	821 (316)	Shale	Basalt/Shield	9,529 (8,285)	2.3 (2.2)	7.7 (4.4)
Lochsa River	72	0.92	2.15 (2.13)	0.01	0.03 (-)	36 (4)	1,147 (329)	Shield	Sedimentary	10,397 (11,249)	1.8 (1.1)	11.3 (5.8)
NF Clearwater River	41	0.54	0.52 (0.94)	0.00	0.00 (-)	39 (3)	879 (154)	Shield	Shale	4,770 (4,006)	0.6 (0.5)	32.3 (11.9)
Potlatch River	112	0.00	0.00 (-)	0.46	1.56 (1.74)	80 (29)	801 (166)	Shale	Basalt	11,180 (5,474)	1.1 (0.9)	5.4 (3.5)
Selway River	82	0.89	2.78 (4.44)	0.04	1.55 (1.80)	49 (15)	1,045 (349)	Shield	Sedimentary	2,793 (3,705)	1.8 (1.8)	13.8 (8.8)
SF Clearwater River	173	0.73	2.31 (3.48)	0.31	0.61 (1.18)	59 (14)	1,257 (268)	Shale	Shield	11,132 (8,468)	1.7 (1.8)	9.8 (8.0)
Salmon River												
Lower Salmon River	50	0.22	0.82 (2.04)	0.12	2.09 (2.35)	98 (35)	1,002 (383)	Basalt	Shield	10,566 (9,277)	2.7 (1.8)	9.0 (4.6)
Middle Salmon River	134	0.29	0.79 (0.99)	0.08	0.56 (0.88)	69 (28)	1,457 (395)	Shield	Sedimentary	3,446 (4,963)	2.7 (2.0)	8.4 (5.2)
MF Salmon River	190	0.52	1.29 (1.82)	0.16	0.71 (2.00)	75 (17)	1,663 (331)	Shield	Acid volcano	2,431 (3,980)	1.7 (1.8)	8.7 (5.9)
SF Salmon River	102	0.38	0.41 (1.00)	0.23	5.18 (9.89)	66 (8)	1,515 (328)	Shield	Shale	10,723 (7,015)	1.8 (2.1)	14.3 (9.6)
Upper Salmon River	88	0.27	0.71 (0.82)	0.41	2.76 (5.19)	122 (55)	1,721 (284)	Sedimentary	Acid volcano	11,755 (10,567)	1.4 (1.2)	6.9 (5.3)

FIGURES

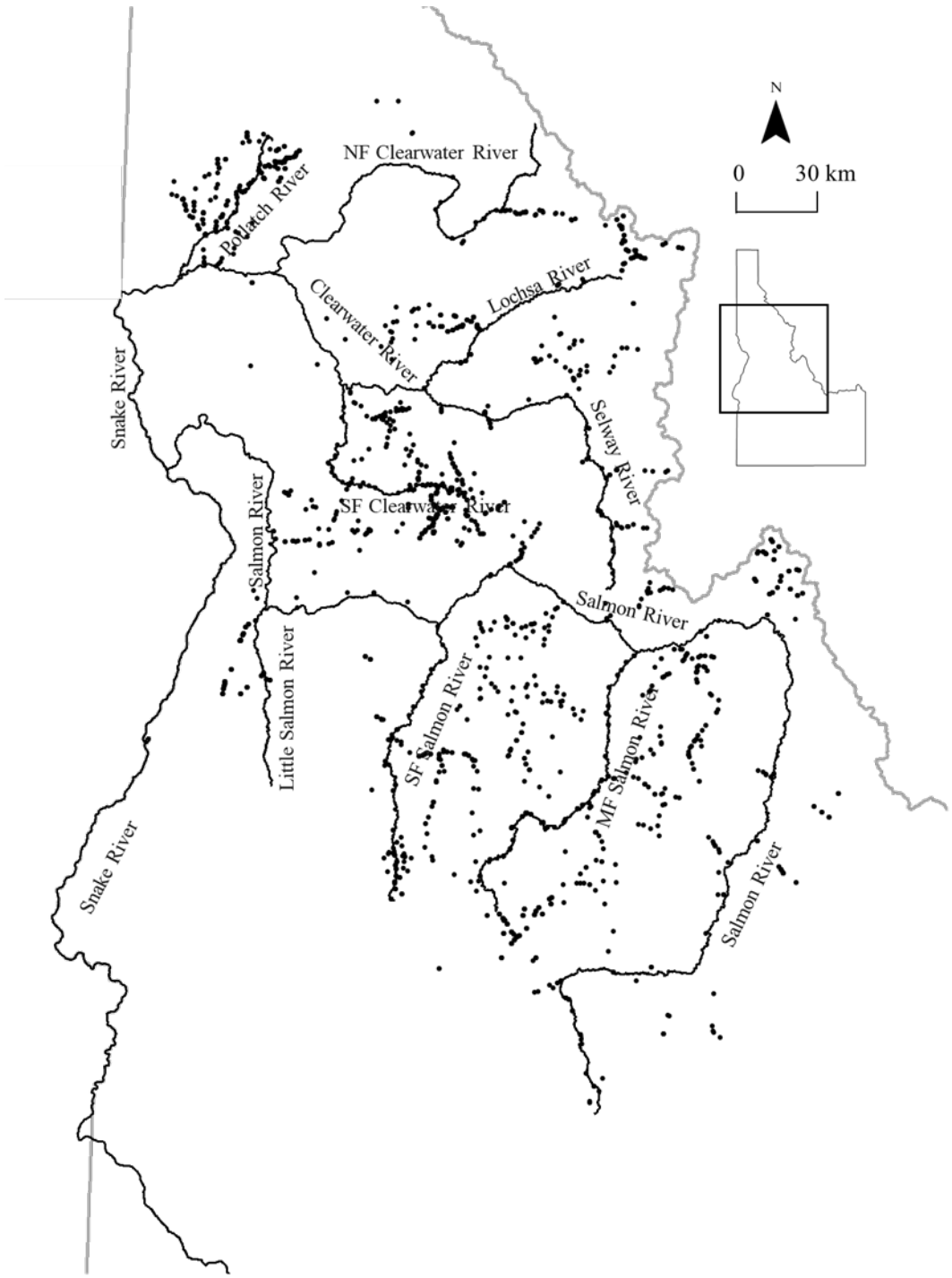

Figure 9. Map of the Clearwater and Salmon River basins, Idaho, including major subbasins. Dots represent locations where snorkel surveys were conducted during 2010–2019. For waterbodies, MF is Middle Fork, NF is North Fork, and SF is South Fork.

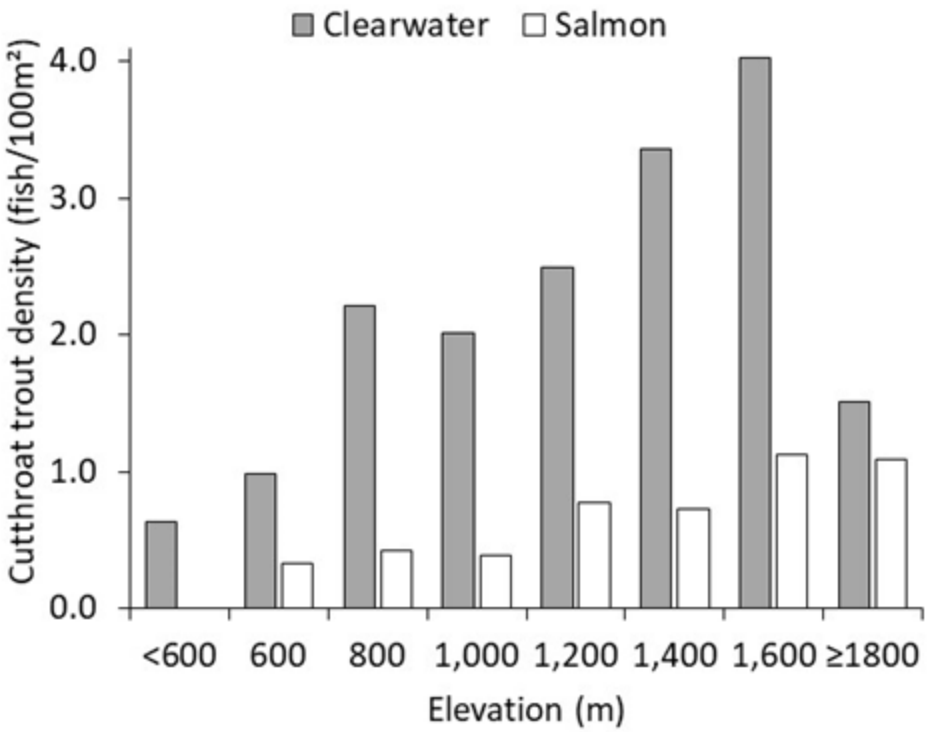

Figure 10. Mean Westslope Cutthroat Trout density (as determined from snorkel surveys) in relation to site elevation in streams throughout the Clearwater River and Salmon River basins of central Idaho.

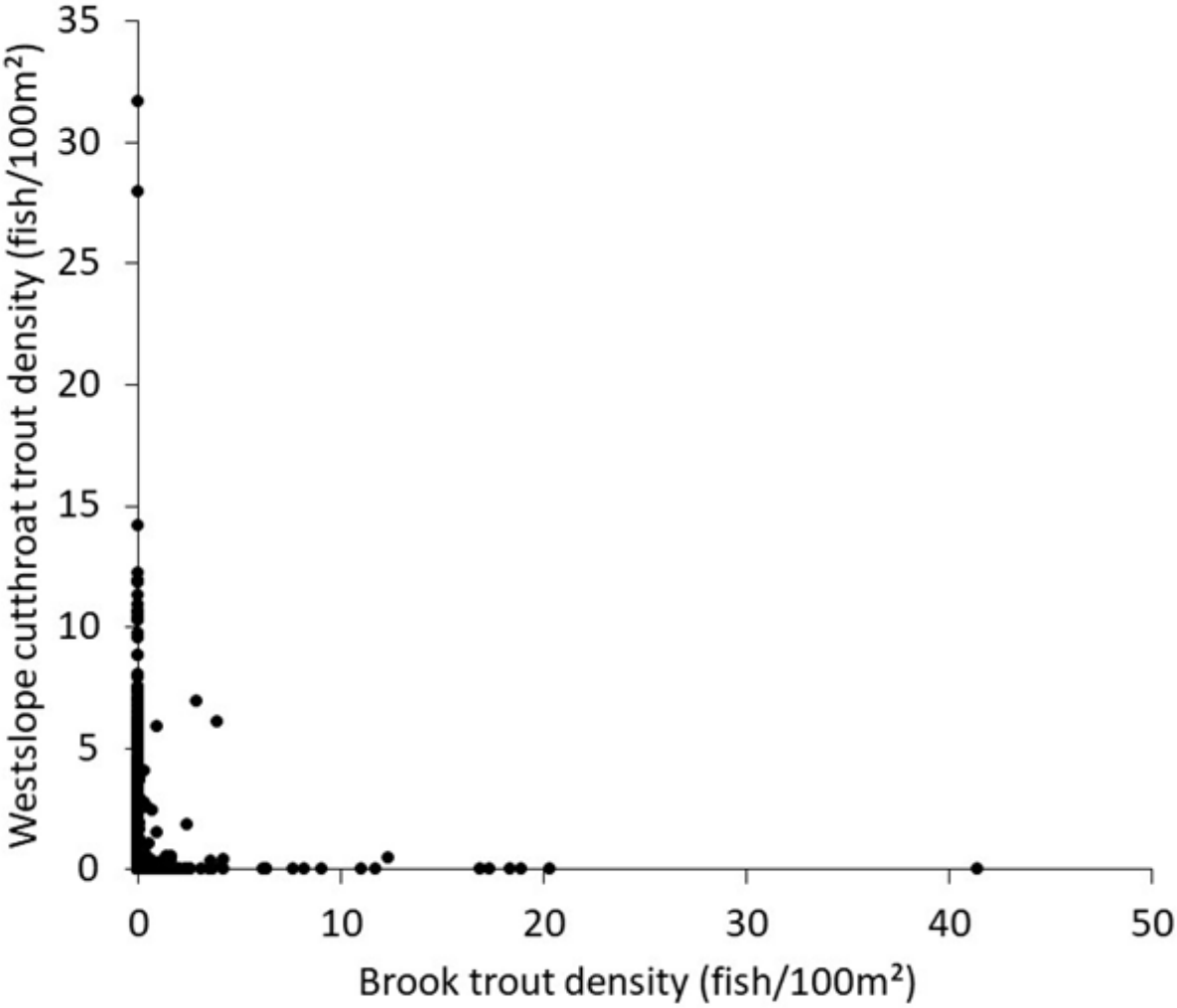

Figure 11. Paired estimates of westslope cutthroat trout density and brook trout density (as determined from snorkel surveys) at stream sites throughout the Clearwater River and Salmon River basins of central Idaho.

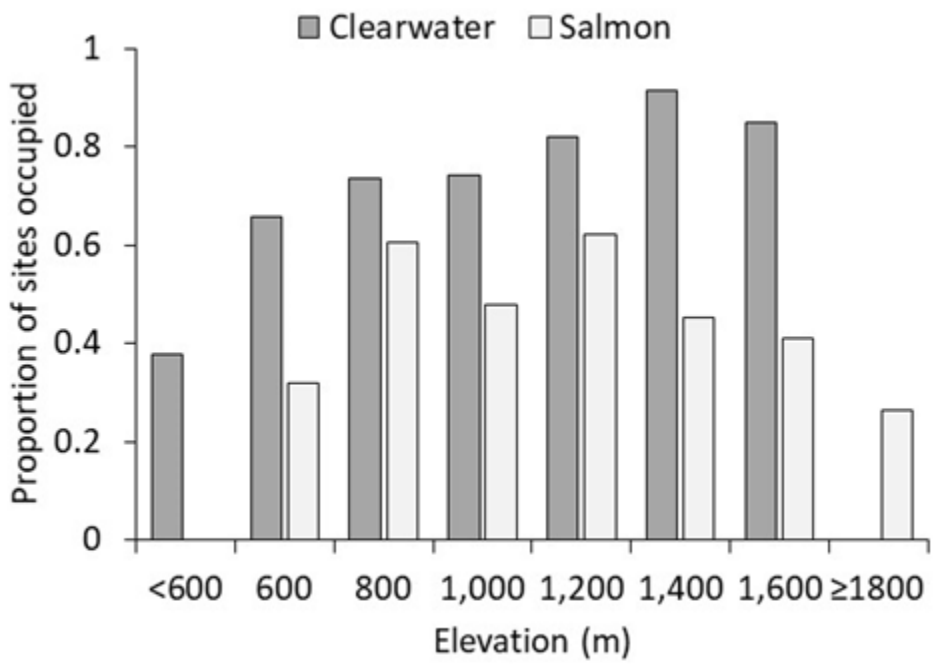

Figure 12. The proportion of stream sites occupied by westslope cutthroat trout (as determined from snorkel surveys) in relation to site elevation throughout the Clearwater and Salmon river basins of central Idaho.

ANNUAL PROGRESS REPORT

SUBPROJECT #5: TEMPORAL TRENDS AND HABITAT ASSOCIATIONS OF MOUNTAIN WHITEFISH IN CENTRAL IDAHO

State of: Idaho

Project No.: 3 Title: Wild Trout Evaluations

Subproject #5: Temporal Trends and Habitat Associations for Mountain Whitefish in Central Idaho

Time Period: July 1, 2021 to June 30, 2022

ABSTRACT

Mountain Whitefish *Prosopium williamsoni* have failed to garner the same level of attention as other members of the salmonid family in terms of scientific investigations, especially with regard to limiting factor analysis and population status. Consequently, we used snorkel survey data from 1985 to 2019 to relate a suite of environmental factors to Mountain Whitefish occupancy and abundance and to estimate population growth rates (λ) in central Idaho. Mountain Whitefish λ in the majority of subbasins in central Idaho appear to be stable or increasing over the last several decades, but more so in the Salmon River basin than the Clearwater River basin. Mountain Whitefish occupancy and abundance were higher in stream reaches that were lower in elevation and gradient and larger in size, with an occupancy rate of <0.10 in stream reaches that were <6 m average wetted width but >0.50 in stream reaches that were ≥9 m average wetted width. Road density was positively associated with the occupancy and abundance of Mountain Whitefish, contrasting previous studies that generally report negative associations between road density and salmonid population metrics; in the relatively sterile lotic environment of central Idaho, such anthropogenic disturbance may inadvertently result in nutrient enrichment, potentially benefitting the forage base of Mountain Whitefish. We also observed that conductivity positively influenced Mountain Whitefish abundance, likely stemming from its direct effect on stream productivity. Although the status of Mountain Whitefish in central Idaho appears generally stable, the paucity of studies reporting on the status of this species highlights the need for additional research devoted to a better understanding of trends in Mountain Whitefish abundance across their range.

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INTRODUCTION

Mountain Whitefish *Prosopium williamsoni* are one of the most widely distributed (Behnke 2002) and abundant (Meyer et al. 2009) species of salmonid in western North America. However, unlike other members of the salmonid family, they have failed to garner much attention from fisheries managers (Brown 2010), likely because anglers generally do not target them. Limited interest in Mountain Whitefish from these groups has resulted in a shortage of studies targeted at understanding the ecology of the species (Northcote and Ennis 1994).

Although studies focusing on Mountain Whitefish are relatively scarce compared to other salmonids, attempts have been made to address this knowledge gap. For example, it has been established that Mountain Whitefish primarily occupy riverine habitat (Northcote and Ennis 1994; Sigler and Zaroban 2018), are broadcast spawners in autumn (Sigler 1951; Northcote and Ennis 1994; Boyer et al. 2017), can achieve lengths exceeding 600 mm (Taylor et al. 2012), commonly live from 8-24 years (Thompson and Davies 1976; Meyer et al. 2009; Watkins et al. 2017), and feed almost exclusively on aquatic insects (Sigler 1951; DosSantos 1985). These and other studies have also investigated Mountain Whitefish behavior (DosSantos 1985; Taylor et al. 2012), fecundity (Sigler 1951; Brown 1952; Wydoski 2001; Meyer et al. 2009; Boyer et al. 2017), growth (Pettit and Wallace 1975; Benjamin et al. 2014), movement (Pettit and Wallace 1975; Davies and Thompson 1976; Benjamin et al. 2014), recruitment (Watkins et al. 2017) and survival (Thompson and Davies 1976; Meyer et al. 2009; Watkins et al. 2017). What has rarely been evaluated is population growth rates for Mountain Whitefish, though there is anecdotal information that some populations across western North America are in decline (Boyer et al. 2017). Also lacking are studies investigating factors that limit the distribution and abundance of Mountain Whitefish, although habitat alterations (Erman 1973; Northcote and Ennis 1994; Paragamian 2002; Brinkman et al. 2013) and water temperature thresholds (Quinn et al. 2010) have been shown to influence their distribution.

OBJECTIVES

1. Use data from a long-term, broad-scale snorkeling program to evaluate trends in Mountain Whitefish abundance across central Idaho.

2. Investigate environmental factors limiting the contemporary distribution and abundance of Mountain Whitefish in central Idaho.

METHODS

Study area

Data for the current study were collected from the Clearwater River and Salmon River basins in central Idaho (Figure 13). The Clearwater River originates in the Bitterroot Mountains and has a drainage area of approximately 25,000 km^2 and a mean basin elevation of 1,311 m. Originating in the Sawtooth Mountains, the Salmon River has a larger drainage area of approximately 37,000 km^2 and a higher mean basin elevation of 2,020 m. In addition to Mountain Whitefish, other salmonids in these basins include Bull Trout *Salvelinus confluentus*, Brook Trout *S. fontinalis,* Chinook Salmon *Oncorhynchus tshawytscha*, Coho Salmon *O. kisutch*, Westslope Cutthroat Trout *O. clarkii lewisi*, Lake Trout *S. namaycush*, and resident and anadromous forms of *O. mykiss* and *O. nerka*.

Data collection

From 1985 to 2019, 11,692 fish surveys were conducted at 2,935 sites across the Clearwater River and Salmon River basins, from which Mountain Whitefish occupancy, abundance, and population growth rates were determined. Fish surveys were conducted via daytime (i.e., daylight hours) snorkel surveys, typically from June to August of each year. The length of stream snorkeled at each location averaged 97 m (range = 40–300 m). Protocols for snorkel surveys (Apperson et al. 2015) were similar to those established by Thurow (1994). Prior to the start of each survey, underwater visibility (i.e., distance to distinguish patterns on an object [e.g., boot or tape measure], used as a surrogate for spotting patterns on fish) was measured before each survey and averaged 2.5 m. Visibility was measured to determine the number of snorkelers required to survey the site so that the distance between snorkelers did not exceed the visibility. The majority of surveys (~90%) were conducted in an upstream manner, but water conditions (e.g., high water velocities or deep water) occasionally required the snorkelers to move in a downstream direction. Salmonids >50 mm were identified to species and total lengths were estimated to the nearest 25 mm in total length. Salmonids <50 mm were not recorded due to the difficulty in identifying some of those fish to species using phenotypic characteristics.

To evaluate factors that influenced Mountain Whitefish occupancy and abundance, various environmental data for each site that was snorkeled (Table 17) were collected using field measurements or a geographic information system (GIS). Field measurements were collected at the time of snorkeling. Average stream width for each site was calculated by averaging wetted width measurements collected every 20 m throughout the survey site, and was included in our analyses because stream size can influence Mountain Whitefish occupancy (Meyer et al. 2009). Instantaneous water temperature (°C) at the time of the snorkel survey was collected because it can affect detection probability for salmonids during snorkel surveys (O'Neal 2007). The area of the snorkel survey (m^2) was quantified to account for differences in effort among surveys.

Broad-scale GIS layers were linked to Mountain Whitefish occupancy and abundance data using ArcMap 10.6 (Environmental Systems Research Institute, Redlands, California). Using the GIS model constructed by Olson and Cormier (2019), conductivity was estimated for each site, and was included in our analyses as a measure of stream productivity (McFadden and Cooper 1962; Scarnecchia and Bergersen 1987). Elevation at each site was calculated from a GIS digital elevation model, and was included because it can influence Mountain Whitefish occupancy and abundance (Rahel and Hubert 1991). Underlying lithology was determined at each site using the GIS Idaho State Geologic Map at a scale of 1:750,000 (Lewis et al. 2012) and was categorized as acid volcanic (rhyolite), basalt, sedimentary (including alluvium, sandstone, and quartzite), shale, and shield (metamorphic and plutonic rock; Suchet et al. 2003). Lithology was included in our analyses due to its effect on water chemistry (Olson and Hawkins 2012), habitat complexity (Guy et al. 2008) and substrate size (Kaufmann et al. 2009), all of which can influence Mountain Whitefish occupancy and abundance (DosSantos 1985; Kennedy 2009; Smith et al. 2016).

Road density was included to account for the effect that road construction and maintenance often have on salmonids in general (Dunham and Rieman 1999; Valdal and Quinn 2011) and Mountain Whitefish in particular (Northcote and Ennis 1994). Estimates of road density at each site were calculated by mapping the roads in Idaho using the 2019 Topologically Integrated Geographic Encoding and Referencing (TIGER) database (United States Census Bureau 2019), and summing the total meters of road within 10 km^2 (i.e., within a 1.78 km radius) of each survey site. Stream slope and stream order (Strahler 1957) were estimated using the National Hydrography Dataset Plus Version 2 dataset (https://nhdplus.com/NHDPlus/NHDPlusV2_home.php) and were included to account for their

influence on Mountain Whitefish distribution and abundance (Platts 1979; Meyer et al. 2009). Conductivity, elevation, lithology, road density, and slope were all point estimates at the downstream end of the site.

Data analyses

Temporal trends in Mountain Whitefish abundance in central Idaho were evaluated by estimating λ for subbasins in the Clearwater River and Salmon River basins. Population growth rates were estimated for the entire long-term dataset (1985–2019) and for a more contemporary time period (2010–2019). For both time frames, a site was only included in the trend analyses if Mountain Whitefish had been observed at the site at least once over the course of all surveys, and a minimum of three surveys had been completed at the site. For long-term surveys, at least one survey had to be conducted at a site during 2000–2009 and 2010–2019. For contemporary surveys, three surveys must have occurred at a site during 2010–2019. Of the 2,935 sites surveyed by snorkelers, 286 sites met the long-term survey criteria and 308 sites met the contemporary survey criteria (Table 18), but sample size was inadequate to estimate λ for the Clearwater and the North Fork Clearwater subbasins. Sites were divided into headwater streams (stream orders 1-3; Vannote et al. 1980) and large rivers (stream orders >3) because Mountain Whitefish tend to occupy larger rivers (Platts 1979; Maret et al. 1997; Meyer et al. 2009), and headwater streams likely represent suboptimal habitat for the species.

Population growth rates were estimated by fitting snorkel count data to a linear regression model in Program R (R Development Core Team 2021). The independent variable within the model was the sample year, and the dependent variable was the $\log_e$ transformation of the mean Mountain Whitefish density (fish/100m^2) from snorkel surveys from that year that met the criteria noted above. Density estimates were used rather than raw count data to account for differences in survey effort between surveys. When constructed in this manner, the slope of the model is equal to the intrinsic rate of change for the population (r; Morris and Doak 2002), which can be exponentiated to estimate λ. Ninety-five percent confidence intervals (CIs) were calculated using the error surrounding the estimate of λ from the linear regression (i.e., 95% CI = 1.96 × SE). Because values of zero cannot be $\log_e$ transformed, all surveys with a density of zero were replaced with 0.01 fish/100m^2. The insertions altered the mean and standard error for these data sets by less than 0.01% compared with the untransformed trend data. Populations were considered stable when the 95% CIs overlapped one, whereas they were considered to be increasing or decreasing when point estimates of λ were either >1 or <1, respectively, and 95% CIs did not overlap one.

Modeling factors that influenced Mountain Whitefish occupancy was conducted using logistic regression, with a dummy response variable of one if they were present and zero if they were absent. For sites surveyed more than once, Mountain Whitefish were considered present if they were observed at least once at the site. Because preliminary analyses indicated that the fish count data were overdispersed, modeling factors that influenced Mountain Whitefish abundance was conducted using negative binomial regression. Both models were fit in Program R (R Development Core Team 2021) using the MASS package (Venables and Ripley 2002). Because survey area (i.e., effort) was different between surveys, both models included survey area as an offset to control for this difference. Of the 11,692 snorkel surveys conducted over the entire study period, 3,516 surveys (at 1,293 sites) occurred during the contemporary time period (2010–2019) and were used in these analyses.

Prior to any model fitting, correlation was assessed between covariates, and highly correlated covariates (i.e., r >0.70; Dormann et al. 2013) were not included in the same model.

Both stream width and stream order were metrics of stream size; because stream width and survey area were highly correlated (r = 0.84) and not independent (i.e., surveyed area = width × reach length), and because survey area was needed in models as an offset, we only included stream order as a metric of stream size in our modeling efforts. To avoid pseudoreplication (Zar 1999) and temporal autocorrelation (Sokal and Rohlf 1995) in the fish count data, survey area, water temperature, and wetted width were averaged across all surveys at a site when sites were visited more than once during the entire study period. No changes were made to the remaining covariates (i.e., conductivity, elevation, lithology, road density, and stream order) as they did not differ between surveys.

All variables were included as fixed effects, and continuous variables were scaled (Schroeder et al. 1986) so that the mean was equal to zero and a one unit increase in the variable was equal to one standard deviation. Variables were included as linear terms except water temperature, which was included as a quadratic term due to the fact that salmonids, like nearly all poikilotherms, have an optimal temperature range, and outside this range, activity (and therefore detectability) usually decreases (Thurow 1994; O'Neal 2007). Because of inherent differences in Mountain Whitefish distribution and abundance between subbasins, subbasin was included as a fixed effect. Data from the Potlatch subbasin were discarded from our analyses because although Mountain Whitefish are present in the subbasin, they were entirely absent from the sites surveyed in this study. For each parameter estimate, 95% CIs were calculated via profiling (Venables and Ripley 2002). Parameter estimates and CIs were exponentiated to increase interpretability, and parameters were considered significant when the 95% CIs did not overlap one (for continuous variables) or each other (for discrete variables).

RESULTS

Trends in abundance

In the Clearwater River basin, Mountain Whitefish λ was stable in headwater streams for all subbasins except the South Fork Clearwater River, where λ declined over the entire study period, but was stable in the last 10 years (Table 18, Figure 14). In larger rivers, λ was negative in the South Fork Clearwater across the entire study period. Combining data across subbasins to look at overall trends in the Clearwater River basin indicated that Mountain Whitefish λ in the last 10 years was stable in headwater streams and in larger rivers, but long-term λ declined in headwater streams. In the Salmon River basin, Mountain Whitefish λ – both contemporary and long-term – was stable in headwater streams and in larger rivers for all subbasins. However, combining data across subbasins to assess overall trends in the Salmon River basin indicated that Mountain Whitefish λ was increasing in headwater streams and in larger rivers for long-term and contemporary timeframes.

Factors affecting distribution and abundance

Mountain Whitefish were present during at least one visit at 561 (43%) of the 1,293 sites used to relate environmental conditions to their contemporary distribution and abundance in central Idaho. Where Mountain Whitefish were present at these sites, they were observed during every survey at 60% (n = 337) of the sites and sporadically at 40% (n = 224) of the sites. At sites where they were sporadically present, Mountain Whitefish were observed during 52% of the surveys. Their abundance averaged 1.02 fish/100m2 (range = 0.01–19.01 fish/100m2) at the 561 sites they occupied.

Mountain Whitefish occupancy was influenced by a variety of factors, including elevation, road density, slope, and stream order (Table 19). Parameter estimates indicated that Mountain Whitefish were more likely to occupy larger, lower elevation stream reaches with a lower stream slope and a higher density of roads in the vicinity of the site. In comparison, parameter estimates from the Mountain Whitefish abundance model indicated that their density was higher in stream reaches with higher conductivity, lower elevation, a higher density of roads in the vicinity of the site, and lower water temperature at the time of the survey (Table 19). Additionally, Mountain Whitefish density was higher in stream reaches with sedimentary and shale lithologies than in reaches with an underlying lithology of basalt.

DISCUSSION

Mountain Whitefish have failed to garner the same level of attention as other members of the salmonid family in terms of scientific investigations, especially with regard to their status. However, there is some evidence of population declines in various portions of their range (Boyer et al. 2017), highlighting the need for a better understanding of trends in Mountain Whitefish abundance. Because Mountain Whitefish are sensitive to alterations in riverine habitat (e.g., Erman 1973; Northcote and Ennis 1994; Paragamian 2002; Brinkman et al. 2013), they are often used as an indicator species for local environmental assessments (e.g., Bergstedt and Bergersen 1997; McPhail and Troffe 1998; Cash et al. 2000; Quinn et al. 2010) despite the lack of trend data across their range. Based on results from the current study, the vast majority of Mountain Whitefish populations in central Idaho appear to be stable or increasing over the last several decades, although this is true more so in the Salmon River basin than in the Clearwater River basin. These results are not surprising given that more than one-third of central Idaho is designated or de facto wilderness and has been demonstrated to be a stronghold for other resident salmonids (e.g., Meyer et al. 2014; Kennedy and Meyer 2015). Nevertheless, not all of these areas are pristine, especially in the South Fork Clearwater River subbasin, where habitat alterations are more prevalent (Northwest Power and Conservation Council 2003) than in adjacent subbasins; this may explain the declining population growth rates for this subbasin. Considering that this is the first study we are aware of that has assessed Mountain Whitefish population growth rates, we encourage more research devoted to investigating trends in Mountain Whitefish abundance across their range so their broad-scale status is better understood.

Although elevation was inversely related to the occupancy and abundance of Mountain Whitefish, it is unlikely that this indicates a direct causative relationship. Rather, changes in elevation were likely correlated to changes in other environmental factors that more directly elicited behavioral or physiological responses in Mountain Whitefish. For example, lower elevation streams tend to be larger in size and lower in gradient, and we observed higher Mountain Whitefish occupancy in such conditions, as have others (Sigler 1951; Torgersen et al. 2006; Meyer et al. 2009). Stream width was not included in our models because (as mentioned above) stream width and survey area were highly correlated and not independent, and survey area was needed to control for differences in snorkeling “effort” between sites. However, the relationship between stream width and Mountain Whitefish occupancy was striking (Figure 15), with an occupancy rate of <0.10 in reaches with stream width <6 m but >0.50 in reaches with average stream width ≥9 m. Relatively large, low gradient stream reaches likely provide better spawning, rearing, and overwinter habitat for Mountain Whitefish than steeper, headwater reaches. Unfortunately, lower elevation lotic habitats in the intermountain west are more vulnerable to temperature warming due to climate change than are headwater streams (Isaak et al. 2016). Considering their aversion to smaller headwater streams, the ability of Mountain Whitefish to

colonize upstream habitat as temperatures warm in the lower elevation reaches they currently occupy may be diminished compared to other native salmonids (e.g., Isaak et al. 2015).

Not only are larger, lower-elevation river segments more vulnerable to climate change displacement of native salmonids, but they also tend to be more heavily altered by anthropogenic disturbances, including road construction and maintenance. Considering that Mountain Whitefish are positively associated with such larger, lower elevation streams, it should not be surprising that we found positive associations between road density and their occupancy and abundance. In general, roads negatively affect salmonid populations through sedimentation and habitat alteration (Dunham and Rieman 1999), as well as by creating barriers to fish movement (Diebel et al. 2015). However, Mountain Whitefish differ from most other stream-dwelling salmonids in that they have a stronger tendency to school in deeper pool and run habitat, and they broadcast spawn rather than building redds; these behaviors may render Mountain Whitefish less vulnerable to impacts from roads. Alternatively, Scrimgeour et al. (2008) argued that the positive association they observed between Mountain Whitefish occupancy and road density may have been caused by road construction and maintenance creating a trophic cascade of nutrient enrichment mediated by forest land-use practices (e.g., Carignan et al. 2000; Lamontagne et al. 2000), resulting in increased invertebrate abundance. However, this explanation is not supported by the population growth rates we observed, which were lowest in the South Fork Clearwater River basin, a relatively disturbed watershed.

As with stream size, conductivity in streams also changes longitudinally, although not always in a linear manner (e.g., McGuire et al. 2014). We observed that conductivity positively influenced Mountain Whitefish abundance, as has been demonstrated for Mountain Whitefish in other parts of Idaho (Meyer et al. 2009). Such a relationship likely stems from conductivity directly affecting stream productivity (Rawson 1951; Welch 1952). Stream conductivity in our study area is relatively low (cf. Merovich et al. 2007) compared to other areas where Mountain Whitefish occur (e.g., Meyer et al. 2009; Boyer et al. 2017). As such, our results emphasize that in the relatively sterile lotic environments of the central Idaho mountains (Sanderson et al. 2009), higher stream productivity increases Mountain Whitefish productivity.

Lithology can have a profound effect on channel morphology (Minshall et al. 1985), water chemistry (Clow and Sueker 2000; Wanty et al. 2009), substrate composition (Connolly and Hall 1999), and habitat availability (Lanka and Hubert 1987; Baxter and Hauer 2000) within a stream. In the present study, lithology had no apparent influence on Mountain Whitefish occupancy and only limited influence on their abundance, with densities being highest in stream reaches with sedimentary and shale lithologies and lowest in stream reaches with basalt lithology. The fact that lithology was more influential for Mountain Whitefish abundance than their distribution suggests that lithology influenced habitat suitability more than their ability to fulfill a particular component of their life history. However, surprisingly little research has been conducted regarding the direct effects of lithology on fish distribution or abundance, thus further research is needed to establish better causative links between lithology and fish ecology.

Our study had some important limitations that may have influenced our findings. First, snorkel data has been shown to be more prone to observation error than data generated from other traditional fish sampling techniques, such as electrofishing, screw traps, and weirs (Meyer et al. 2104), likely because detection probability is less consistent between surveys. However, estimates of λ using linear regression is robust to the potential presence of observation error within the data (Humbert et al. 2009), and there is no reason to suspect directional bias in the relationships we observed between environmental factors and Mountain Whitefish occupancy or abundance. Second, although snorkel survey locations were not established randomly, we

assume that the data they generated did not result in any directional bias in our results, based on the sheer volume of surveys (n >11,000) included in the study (cf. Kadmon et al. 2003). Third, we only included a few of the myriad factors that could have influenced Mountain Whitefish occupancy and abundance in the study area, although we would argue that this is an inherent weakness of any limiting factor analysis of fish populations. Notwithstanding these and other limitations, our results suggest that although Mountain Whitefish in central Idaho appear to be stable, conserving wider, lower elevation stream reaches, especially in sedimentary and shale lithologies, would likely positively benefit Mountain Whitefish populations.

MANAGEMENT RECOMMENDATIONS

In general, Mountain Whitefish trends appear to be stable for central Idaho (Table 18). Two exceptions to this statement are the rivers of the South Fork Clearwater River subbasin and the headwater streams of the Clearwater River basin as a whole. However, the status of Mountain Whitefish in the South Fork Clearwater River subbasin are likely controlling the overall Clearwater River basin trend as South Fork Clearwater River subbasin sites make up the vast majority of headwater stream sites in the Clearwater River basin. Therefore, to better characterize the status of Mountain Whitefish in the headwater streams of the Clearwater River basin, additional surveys should be conducted in the headwater streams of the Lochsa and Selway River subbasins at regular intervals. Similarly, only one site in the headwaters of the lower Salmon River subbasin met the criteria to be used to evaluate the trends of Mountain Whitefish; therefore, additional surveys at additional sites should be conducted throughout the lower Salmon River subbasin to more thoroughly assess the status of the species in the area. Lastly, given the declining trend of Mountain Whitefish in the rivers of the South Fork of the Clearwater River lower Salmon River subbasin basin over the past 10 years, management actions targeted at improving the status of Mountain Whitefish in this lower Salmon River subbasin basin should be implemented. Furthermore, given that majority of sites used to estimate trends in the headwaters of the Clearwater River basin are in the South Fork Clearwater River lower Salmon River subbasin basin, it is likely that management actions implemented to improve conditions in the South Fork of the Clearwater River lower Salmon River subbasin basin would improve the overall status of Mountain Whitefish in headwaters of the Clearwater River basin.

In addition to the broad scale insight provided by the habitat associations describe above, inference can be drawn specifically regarding Mountain Whitefish fisheries in central Idaho. For example, the importance of stream size for Idaho Mountain Whitefish populations is highlighted by the fact that Mountain Whitefish occupancy and density are highest in the North Fork of the Clearwater River basin where average wetted width is also the largest (Table 20). Another pattern of note for Idaho Mountain Whitefish populations is that basins underlain by lithologies with strong positive correlations to Mountain Whitefish distribution and abundance have higher rates of occupancy and higher densities (e.g., North Fork Clearwater River basin, Middle Salmon River basin, Upper Salmon River basin) than basins underlain with less favorable lithologies. In fact, the three subbasins (i.e., Clearwater River subbasin, Potlatch River basin, Lower Salmon River basin) that have basalt as either the predominate or secondary lithology have the lowest occupancy and density among all basins. Taken collectively, these patterns suggest that managers interested in improving the status of Idaho Mountain Whitefish populations should implement management actions targeted at improving large river habitat, particularly in areas underlain by lithologies other than basalt. However, it is important to note that these inferences are only relevant to the sites observed during the current survey and are not necessarily representative of a given river basin as a whole.

RECOMMENDATIONS

1. Given the declining trend of Mountain Whitefish in the South Fork of the Clearwater River over the past 10 years, management actions targeted at improving their status in this subbasin should be implemented.

2. Continue monitoring Mountain Whitefish populations throughout the Clearwater River and Salmon River basins, and conduct another formal trend analysis in 10 years to determine if their status has changed.

3. Identify and monitor areas where Mountain Whitefish may encounter Smallmouth Bass to evaluate the effects of Smallmouth Bass range expansion, due to climate change, on Mountain Whitefish populations.

ACKNOWLEDGEMENTS

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TABLES

Table 17. Summary of environmental data (i.e., conductivity, elevation, road density, stream slope, and stream order) and survey data (length, survey area, water temperature, and average wetted width) collected either with geographic information system analysis or during snorkel surveys from 2010–2019 in the Clearwater River and Salmon River basins of central Idaho. Data were divided into sites where Mountain Whitefish were either absent or present.

	Absent				Present			
Variable	Mean	SD	Minimum	Maximum	Mean	SD	Minimum	Maximum
Conductivity (µS/cm)	73	39	28	327	67	30	29	260
Elevation (m)	1,410	455	278	2,431	1,292	400	413	2,229
Road density (m)	8,515	9,389	0	57,885	10,135	9,098	0	48,385
Slope (%)	2.4	2.0	0.0	14.8	0.8	0.9	0.0	11.8
Temperature (°C)	12.4	3.1	4.0	24.0	14.3	2.7	8.0	22.0
Wetted width (m)	7.3	5.8	0.8	55.3	16.2	10.0	2.6	52.7

Table 18. Mountain Whitefish population growth rates (λ) for major subbasins of the Clearwater River and Salmon River basins of central Idaho. Ninety-five percent confidence intervals are also included in parentheses. Survey sites were divided into headwater sites (stream orders 1-3) and river sites (stream orders >3). SF is South Fork, MF is Middle Fork.

	Long term (1985 - 2019)				Contemporary (2010 - 2019)			
Sub-basin	Timeframe	Sites	Surveys	λ	Timeframe	Sites	Surveys	λ
Clearwater River (headwaters)								
Lochsa	1988 - 2019	6	110	1.03 (0.94 - 1.14)	2010 - 2019	9	37	1.18 (0.76 - 1.85)
Selway	1986 - 2019	7	119	0.93 (0.82 - 1.06)	2010 - 2019	7	35	0.59 (0.31 - 1.14)
SF Clearwater	1986 - 2019	42	795	0.90 (0.82 - 1.00)	2010 - 2019	39	294	0.79 (0.49 - 1.26)
Overall	1986 - 2019	55	1024	0.95 (0.93 - 0.98)	2010 - 2019	55	366	0.93 (0.81 - 1.08)
Clearwater River (rivers)								
Lochsa	1988 - 2019	15	297	1.04 (0.99 - 1.10)	2010 - 2019	23	149	1.10 (0.88 - 1.37)
Selway	1988 - 2019	19	329	1.01 (0.93 - 1.08)	2010 - 2019	14	88	1.00 (0.70 - 1.42)
South Fork Clearwater	1986 - 2019	17	296	0.83 (0.77 - 0.90)	2010 - 2019	12	63	0.68 (0.47 - 0.98)
Overall	1986 - 2019	51	922	0.99 (0.95 - 1.02)	2010 - 2019	49	300	0.99 (0.85 - 1.16)
Salmon River (headwaters)								
Lower Salmon River	1991 - 2017	1	19	0.93 (0.70 - 1.23)	2010 - 2017	1	3	0.85 (0.16 - 4.45)
Middle Salmon River	1987 - 2019	9	152	0.96 (0.85 - 1.09)	2010 - 2019	8	383	1.00 (0.53 - 1.92)
MF Salmon River	1985 - 2019	23	378	0.95 (0.86 - 1.06)	2010 - 2019	21	153	0.95 (0.58 - 1.57)
SF Salmon River	1986 - 2019	11	259	1.03 (0.93 - 1.16)	2010 - 2019	17	113	0.85 (0.51 - 1.44)
Upper Salmon River	1986 - 2019	15	266	1.01 (0.90 - 1.12)	2010 - 2019	29	147	1.15 (0.68 - 1.95)
Overall	1985 - 2019	59	1074	1.07 (1.03 - 1.11)	2010 - 2019	76	799	1.30 (1.06 - 1.58)
Salmon River (rivers)								
Lower Salmon River	1987 - 2019	6	106	0.94 (0.85 - 1.05)	2010 - 2019	6	29	0.79 (0.47 - 1.31)
Middle Salmon River	1987 - 2019	11	197	1.09 (1.00 - 1.18)	2011 - 2019	8	37	1.07 (0.62 - 1.87)
MF Salmon River	1985 - 2019	66	1,099	1.00 (0.94 - 1.06)	2010 - 2019	65	481	1.16 (0.90 - 1.49)
SF Salmon River	1988 - 2019	15	309	1.05 (0.98 - 1.13)	2010 - 2019	26	137	1.13 (0.81 - 1.58)
Upper Salmon River	1987 - 2019	23	405	1.03 (0.96 - 1.10)	2010 - 2019	23	137	1.24 (0.87 - 1.76)
Overall	1985 - 2019	121	2116	1.08 (1.04 - 1.12)	2010 - 2019	128	821	1.25 (1.04 - 1.50)

Table 19. Parameter estimates from generalized linear models evaluating the distribution and density of Mountain Whitefish in the Clearwater River and Salmon River basins of central Idaho. Also included are lower (LCI) and upper (UCI) 95% confidence limits. Parameter estimates and confidence limits were exponentiated to increase interpretability. See methods for complete variable and model descriptions.

Parameter	Estimate	LCI	UCI
Distribution			
Intercept	0.00	0.00	0.00
Conductivity	1.01	0.81	1.25
Elevation	0.66	0.48	0.89
Lithology (acid volcanic)	1.79	0.34	9.16
Lithology (sedimentary)	2.86	0.59	13.15
Lithology (shale)	4.73	0.99	21.77
Lithology (shield)	3.17	0.69	14.02
Road density	1.40	1.15	1.70
Slope	0.42	0.28	0.60
Stream order	2.48	1.89	3.30
Temperature	0.98	0.80	1.19
Subbasin (SF Clearwater)	44.47	10.43	319.82
Subbasin (NF Clearwater)	279.22	47.20	2621.69
Subbasin (Lochsa)	19.50	4.24	146.37
Subbasin (Selway)	32.35	6.95	246.51
Subbasin (Lower Salmon)	9.94	1.62	88.08
Subbasin (SF Salmon)	57.40	12.24	436.92
Subbasin (Mid Salmon)	69.23	14.42	538.37
Subbasin (MF Salmon)	95.55	19.02	765.83
Subbasin (Upper Salmon)	123.47	24.49	981.61
Abundance			
Intercept	0.00	0.00	0.00
Conductivity	1.52	1.31	1.79
Elevation	0.84	0.71	0.99
Lithology (acid volcanic)	2.53	0.61	8.63
Lithology (sedimentary)	3.98	1.03	12.25
Lithology (shale)	4.19	1.09	12.83
Lithology (shield)	3.48	0.91	10.58
Road density	1.14	1.02	1.27
Slope	0.90	0.80	1.02
Stream order	1.07	0.95	1.20
Temperature	0.90	0.83	0.99
Subbasin (SF Clearwater)	3.42	0.42	22.89
Subbasin (NF Clearwater)	10.70	1.30	72.31
Subbasin (Lochsa)	4.13	0.49	28.69
Subbasin (Selway)	4.88	0.59	33.23
Subbasin (Lower Salmon)	1.25	0.12	13.15
Subbasin (SF Salmon)	7.30	0.88	49.70
Subbasin (Mid Salmon)	6.92	0.84	47.08
Subbasin (MF Salmon)	11.99	1.43	83.29
Subbasin (Upper Salmon)	5.40	0.64	37.90

Table 20. Summary of environmental data (i.e., conductivity, elevation, road density, and stream slope) and survey data (average wetted width) by major subbasin in central Idaho. Data were collected with either geographic information system analysis or during snorkel surveys from 2010–2019. Values included in parenthesis are equal to one standard deviation. MF is Middle Fork, NF is North Fork, and SF is South Fork. See methods for complete variable descriptions.

		Mountain Whitefish								
Basin	*n*	Occupancy	Density (fish/100m^2)	Conductivity (µS/cm)	Elevation (m)	Predominate Lithology	Secondary lithology	Road density (m)	Slope (%)	Average wetted width (m)
Clearwater River										
Clearwater River	75	0.03	0.08 (0.02)	74 (32)	800 (317)	Shield	Basalt	12,850 (12,228)	2.1 (2.0)	7.6 (4.2)
Lochsa River	123	0.31	0.35 (0.53)	37 (6)	1,156 (327)	Shield	Shale	10,265 (11,109)	2.0 (1.5)	11.7 (9.4)
NF Clearwater River	74	0.95	1.40 (2.60)	45 (9)	909 (162)	Shield	Shale	6,765 (7,298)	0.5 (0.4)	29.9 (10.8)
Potlatch River	156	0.02	0.03 (0.02)	79 (28)	812 (155)	Shale	Basalt	13,117 (5,876)	1.0 (0.9)	5.1 (3.4)
Selway River	86	0.42	0.48 (0.59)	50 (15)	1,039 (347)	Shield	Sedimentary	3,723 (5,078)	1.8 (1.8)	14.0 (8.7)
SF Clearwater River	192	0.46	0.47 (0.83)	59 (14)	1,253 (270)	Shale	Shield	13,224 (7,581)	1.5 (1.5)	10.1 (8.4)
Salmon River										
Lower Salmon River	59	0.12	0.08 (0.04)	105 (41)	969 (408)	Basalt	Shield	13,320 (10,209)	2.7 (1.8)	8.6 (4.6)
Middle Salmon River	235	0.51	1.31 (1.49)	67 (24)	1,462 (396)	Shield	Sedimentary	4,897 (6,679)	2.6 (2.0)	8.5 (5.2)
MF Salmon River	139	0.33	0.87 (0.73)	75 (17)	1,668 (332)	Shield	Acid volcano	3,033 (4,805)	1.4 (1.8)	9.5 (6.4)
SF Salmon River	159	0.49	0.91 (0.87)	66 (8)	1,543 (324)	Shield	Shale	12,952 (8,687)	1.8 (2.1)	12.8 (9.1)
Upper Salmon River	151	0.50	1.40 (2.38)	124 (55)	1,726 (282)	Sedimentary	Shale	13,868 (9,528)	1.2 (1.4)	7.2 (5.6)

FIGURES

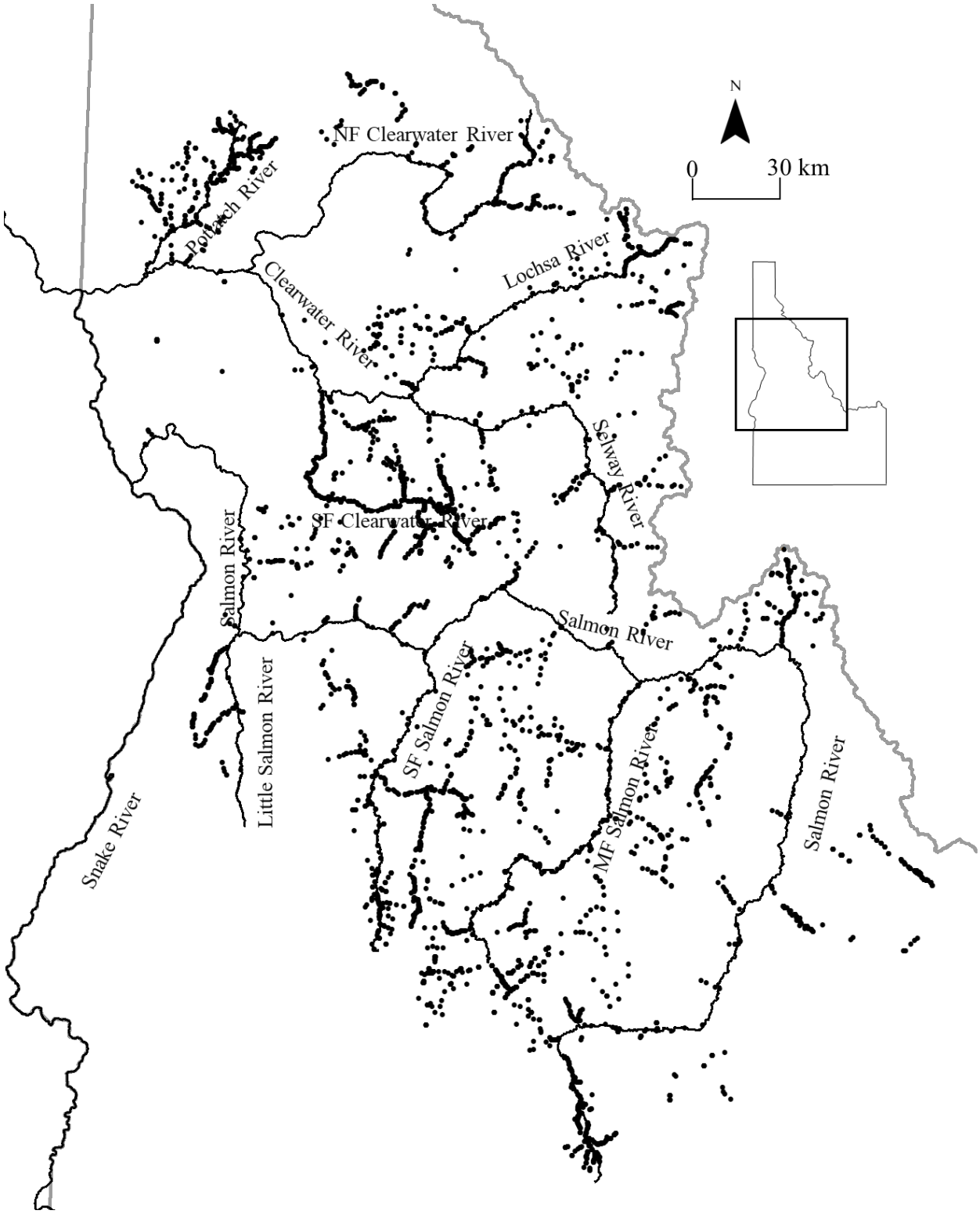

Figure 13. Map of the Clearwater River and Salmon River basins of central Idaho. Black dots represent locations where snorkel surveys were conducted during 1985–2019. MF is Middle Fork, NF is North Fork, and SF is South Fork.

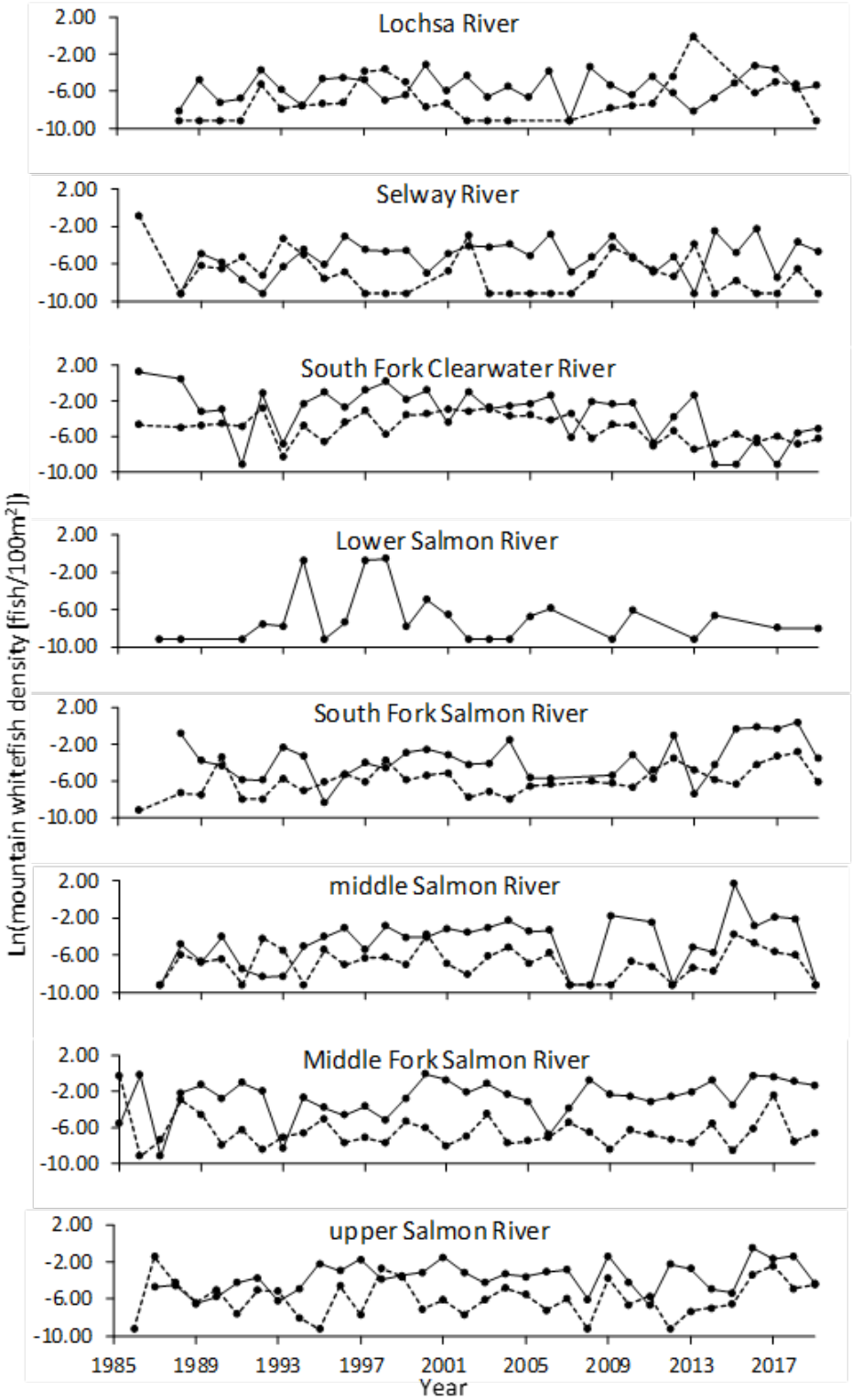

Figure 14. Trends in the density of Mountain Whitefish in various subbasins of the Clearwater River and Salmon River basins of central Idaho. Dashed lines are headwater streams (stream orders 1-3) whereas solid lines are large rivers (stream orders >3).

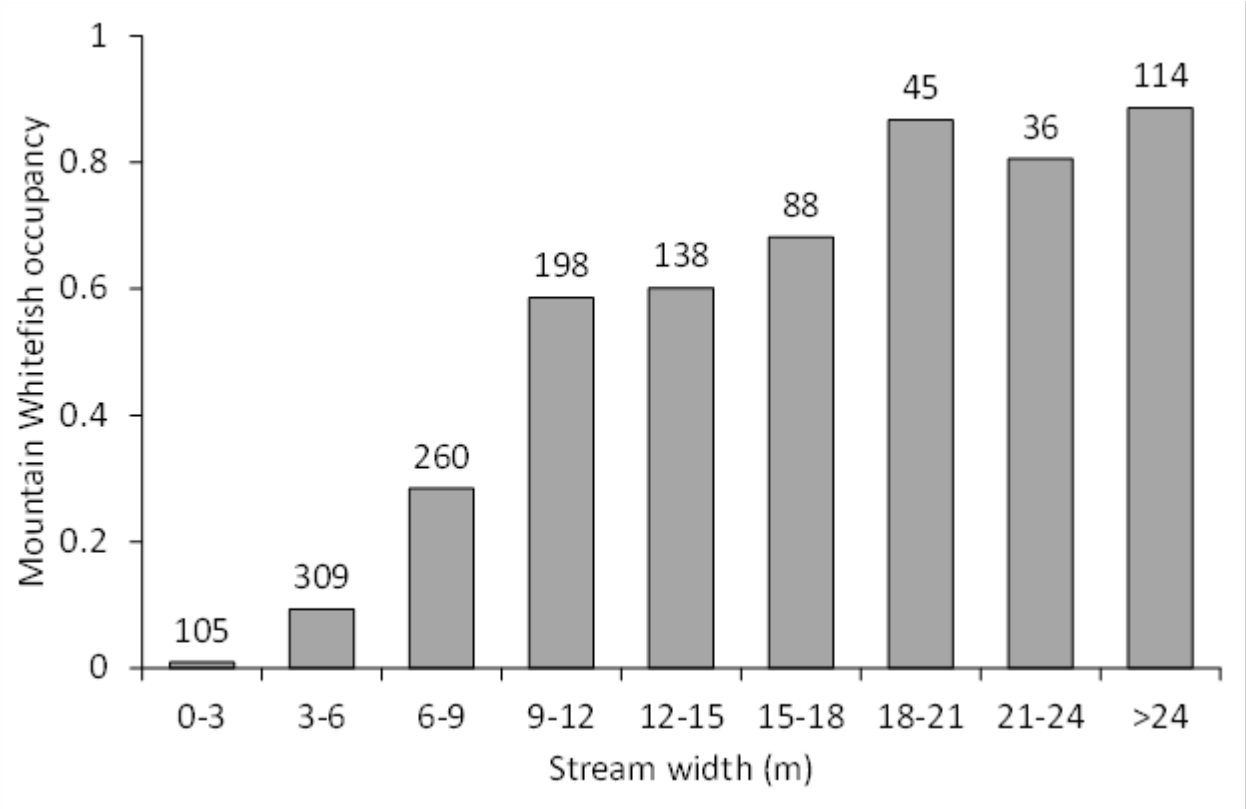

Figure 15. Observed occupancy rates of Mountain Whitefish in the Clearwater River and Salmon River basins, Idaho, in stream reaches with various average stream widths. Numbers on the top of bars indicated sample size for each bin.

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