FISHERY MANAGEMENT INVESTIGATIONS

IDAHO DEPARTMENT OF FISH AND GAME
FISHERY MANAGEMENT ANNUAL REPORT

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UPPER SNAKE REGION

2021

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February 2024
IDFG 22-103

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SOUTH FORK SNAKE RIVER

ABSTRACT

The South Fork Snake River (SFSR) supports the largest population of native Yellowstone Cutthroat Trout *Ocorhynchus clarkii bouvieri* (YCT) in Idaho and is one of the few populations with fluvial and resident life histories intact. Rainbow Trout *Oncorhynchus mykiss* (RBT) are their largest threat through hybridization and competition. Suppression of RBT using tributary weirs, incentivized angler harvest, and electrofishing may be effective tools for managing RBT abundance in the SFSR. Fisheries staff operated weirs at four tributaries and weir efficiencies averaged 87.0%, near the ten-year average (87.5%). Spawning run sizes for YCT indexed at four large tributaries exceeded the ten-year average, but we only trapped six YCT at Rainey Creek, the largest of the four tributaries. For the RBT harvest incentive program, staff marked 837 RBT with coded wire tags. Anglers turned in 2,279 RBT heads including 48 with tags (2.1%) worth $4,850. Since 2010, anglers have turned in 34,619 RBT heads including 764 with tags (2.2%) worth $63,400. Abundances of total trout in the upper SFSR at the Conant monitoring reach were above the ten-year average, and long-term intrinsic rates of population change identify that abundances are stable or increasing for all trout species at all monitoring reaches. Unfortunately, RBT abundances have significantly increased at the Lorenzo monitoring reach. Similarly, RBT abundance at the Lufkin Bottom monitoring reach increased, but the increase was not significant. The IDFG Fish Management Plan has an objective for the SFSR that RBT (including Yellowstone Cutthroat Trout *x* Rainbow Trout hybrids) represent less than 10% of the trout species composition. Though logistical improvements were made, little progress was noted toward IDFG's management goal of 10% RBT. However, significant effort needs to continue for the next several years to determine if RBT suppression is a viable tool to achieve our management goals.

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INTRODUCTION

The South Fork Snake River (SFSR) supports the strongest remaining fluvial population of native Yellowstone Cutthroat Trout *Ocorhynchus clarkii bouvieri* (YCT) within their historical range in Idaho. The SFSR is one of only a handful of large rivers in the species' range that supports a robust population of YCT (Thurow et al. 1988; Van Kirk and Benjamin 2001; Meyer et al. 2006). Across the majority of the species' range, YCT have experienced dramatic reductions in abundance and distribution (Behnke 1992). In August 1998, conservation groups petitioned the United States Fish and Wildlife Service (USFWS) to list YCT under the Endangered Species Act (ESA). In February 2001, the USFWS denied the listing petition, and conservation groups filed a lawsuit in January 2004, which led to a 12-month review of the status of YCT. The USFWS determined that YCT did not warrant ESA listing in February 2006 (USFWS 2006). However, in many areas across their historical range YCT have continued to sustain declines in their abundance and distribution (Endicott et al. 2016).

The primary goal for the SFSR, as directed by our constituency (Koenig 2020) and the Idaho Fish and Game Commission, is the preservation of the genetic integrity and population viability of YCT (IDFG 2019). In the late 1990s and early 2000s, Rainbow Trout *O. mykiss* (RBT) abundance increased in the main stem SFSR. During the same period, RBT increasingly pioneered tributaries for spawning. The propensity for RBT to hybridize with YCT and create Yellowstone Cutthroat Trout *Ocorhynchus clarkii bouvieri x* Rainbow Trout *O. mykiss* hybrids (collectively RBT hereafter) are the biggest threat to the continued persistence of YCT in the SFSR (Moller and Van Kirk 2003, IDFG 2007; Range-wide YCT Conservation Team 2009; Van Kirk et al. 2010) because of risks through competition (Seiler and Keely 2007a) and hybridization (Henderson et. al 2000). Interspecific competition can cause increased mortality as individual fish aggressively compete for food resources or niche space (Seiler and Keely 2007a; Seiler and Keely 2007b; Van Kirk et al. 2010). Additionally, hybridization may result in the loss of genetically distinct YCT as gene flow transfers from one species to another through backcrossing of interspecific hybrids (Young 1995; Huxel 1999; Kruse et al. 2000; Kozfkay et al. 2007; Gunnell et al. 2008). Abundance of RBT has increased significantly in recent years to the extent that RBT were twice as abundant as YCT in 2018, increasing the potential for hybridization and competition.

The primary management objective in the Idaho Department of Fish and Game (IDFG) Fisheries Management Plan (IDFG 2019) is to preserve the genetic integrity and population viability of native YCT. The secondary objective is to limit RBT prevalence to less than 10% of the trout species composition of the catch during annual fall electrofishing surveys in the Conant monitoring reach.. The 10% threshold would return species compositions similar to those documented during the early to mid-1980s. Since 2004, the Idaho Department of Fish and Game (IDFG) and collaborators have implemented several YCT conservation management strategies in the SFSR drainage to support the viability and genetic integrity of this population.

The first management strategy for YCT conservation utilizes fish weirs and traps on four, primary spawning tributaries. The IDFG has constructed weirs and traps as far downstream in tributaries as possible. During the spawning season, weirs and traps are checked frequently. Yellowstone Cutthroat Trout are passed upstream to spawn, while RBT are removed. The weir program on these tributaries has greatly reduced the occurrence of RBT accessing spawning areas upstream of weirs, improving the long-term viability of fluvial Snake River YCT populations (Van Kirk et al. 2010). Modeling by Van Kirk et al. (2010) highlighted the importance of reproductive segregation resulting from this weir program and noted that if RBT were to invade the major spawning tributaries, there may be little chance of securing long-term viability of YCT in the SFSR. Commonly, managers have installed in-stream passage barriers, without fish traps, to reduce hybridization, which unfortunately reduces life history variation and gene flow (Neville et al. 2006). Weir management strategies in the SFSR tributaries are unique because they maintain the YCT fluvial life history while reducing threats from RBT invasion and hybridization. However, weirs alone only provide areas of refuge for YCT to spawn in the absence of RBT, and do not affect the status of RBT in other portions of the watershed.

Though weirs are important for maintaining YCT population viability and integrity in tributaries, there is a larger component of the YCT population that spawn in the main stem SFSR; therefore, successfully managing threats from RBT require additional efforts in the main stem. In addition to weirs, fishing seasons and limits were altered as a second management strategy to encourage RBT harvest and limit YCT harvest. Prior to 2004, the fishing season was from Memorial Day to November 30, and the bag limit was six trout, except only two YCT or BNT, none under 16”. In 2004, the Idaho Fish and Game Commission removed the bag limit on RBT and hybrids, and prohibited the harvest of YCT, while also extending the season to year-round. IDFG did not change the regulations on Brown Trout at that time. This modification resulted in a brief increase in RBT harvest (Schrader and Fredericks 2006). However, now that the regulations have been in place for several years, RBT harvest has decreased. To increase harvest of RBT, we implemented a third management strategy intended to further reduce the RBT population through incentivizing angler harvest in the main stem SFSR. Anglers have the ability to play a key role in YCT conservation efforts on the SFSR should they choose to harvest RBT. Unfortunately, annual harvest rates of RBT have been low largely because the prevalent catch-and-release ethic embraced by many trout anglers. Population modeling suggested that RBT harvest and manual suppression combined must exceed 20% annually to result in a decreasing RBT population (Van Kirk et al. 2010; Devita 2014; Oldemeyer and Van Kirk 2018). Despite attempts to incentivize harvest, angler harvest rates of RBT have been less than 20% except for one year since 2004 (High et al. 2011).

Beginning in 2018, IDFG tested the effectiveness of and using boat electrofishing to suppress RBT in the main stem of the SFSR. We removed RBT from the SFSR and transported them to other RBT fisheries within the Upper Snake Region. In effect, this fourth management strategy could additionally suppress RBT abundances to exceed 20%. In 2018, IDFG sampled known RBT spawning areas to determine if localized suppression of RBT would result in reduced catch in these areas, or if other RBT would repopulate these areas. If RBT repopulated these locations, then suppression at a larger scale and yielding higher numbers of RBT would be feasible. This pilot study in 2018 identified that electrofishing catch did not decline after initial suppression, suggesting larger suppression efforts might be effective. In 2019, IDFG tested available equipment and staffing to estimate the number of RBT that a moderate level of suppression could achieve (Heckel et al. 2020). This effort resulted in the suppression of approximately 6,000 RBT suggesting suppression via boat electrofishing may help managers achieve or exceed 20% total suppression rate. Several more years of successive suppression will help estimate the efficacy of these efforts in significantly reducing RBT abundances.

The IDFG designed the above management strategies to achieve the primary goal to preserve the genetic integrity of YCT in the SFSR and maintain the YCT population’s long-term viability (IDFG 2007; IDFG 2019). To assess the effectiveness of the aforementioned management strategies, IDFG evaluated trends in abundance and recruitment. This report summarizes IDFG management actions in the SFSR during 2021.

STUDY AREA

The Snake River originates in Yellowstone National Park and flows south through Grand Teton National Park and the Jackson Hole valley before turning west and flowing into Palisades Reservoir at the Idaho – Wyoming state line. The SFSR is the 106 km (65 mi) portion of the Snake River that flows from Palisades Dam to the confluence with the Henrys Fork Snake River. This section of the SFSR is a world-class trout fishery and is an important factor to local economies. Anglers and biologists divide the SFSR into three segments. The first section, known as "the upper river," flows from Palisades Dam to Pine Creek through a relatively unconfined valley. A simple channel characterizes the first 13 km of the upper river downstream of the dam. From this point, the river braids around numerous islands. Three of the four main YCT spawning tributaries enter the SFSR in this upper river, including Palisades, Rainey, and Pine creeks. The second section of the SFSR, also known as "the canyon", flows from Pine Creek downstream to the town of Heise. Burns Creek, the fourth major YCT spawning tributary enters the SFSR in the canyon. The last section of the SFSR, known as "the lower river", flows from Heise to the confluence with the Henrys Fork Snake River. There are no major YCT spawning tributaries in the lower river. Stable water temperatures from Palisades Dam moderate winter conditions in the upper river and canyon sections. Winter conditions in the lower river are usually more severe than upstream with colder temperatures (Moller and Van Kirk 2003). The Conant and Lorenzo monitoring reaches of the SFSR are in the upper and lower river sections, respectively.

In addition to YCT and RBT, other salmonids present in the SFSR include Brown Trout *Salmo trutta* (BNT), Brook Trout *Salvelinus fontinalis,* Lake Trout *Salvelinus namaycush*, and Mountain Whitefish *Propsopium williamsoni*. Native catostomids include Utah Sucker *Catostomus ardens*, Bluehead Sucker *C. discobolus*, and Mountain Sucker *C. platyrhynchus*. The native cottids include Paiute Sculpin *Cottus beldingii* and members of the Mottled Sculpin complex *C. sp.*. Native cyprinids include Redside Shiner *Richardsonius balteatus*, Longnose Dace *Rhinichthys cataractae*, Speckled Dace *R. osculus*, and Utah Chub *Gila atraria*.

OBJECTIVES

1. Operate weirs on Burns, Pine, Rainey, and Palisades creeks to pass YCT and remove RBT, thereby limiting RBT access to YCT spawning and rearing areas. Monitor efficiency and make adjustments where needed and possible.

2. Increase recreational angler suppression of RBT in the main stem SFSR by maintaining liberal fishing regulations, promoting the RBT harvest incentive program, and by other creative solutions.

3. Manually suppress resident RBT in Palisades Creek, and in the main-stem SFSR to reduce hybridization and competition with YCT.

4. Continue monitoring trout abundances and species composition at Conant and Lorenzo monitoring reaches of the main-stem SFSR.

5. Monitor genetic integrity of the spawning YCT run at tributary weirs, and YCT abundance and distribution at Rainey Creek.

6. Rehabilitate stream habitat in Rainey Creek to restore fish passage and mitigate limiting environmental factors for fluvial YCT.

METHODS

Tributary Weirs

To limit RBT access to YCT spawning and rearing areas, and to monitor the genetic integrity of the fluvial YCT population, one combination vertical and velocity barrier (Burns Creek) and three electric weirs (Pine, Rainey, Palisades creeks) were maintained and operated at the four main spawning tributaries of the SFSR. In the past, we started the electrical weirs and installed the trap boxes at least one day prior to the earliest dates we captured RBT in previous years. In 2021, our goal was to begin trapping the first week in April. We checked each trap every three days until daily catch exceeded 20 fish, then we checked traps daily. We operated weirs through mid-July, until the number of trapped fish was less than one YCT per day.

We modified weirs by adding check boards before peak discharge to increase the head of the pool upstream of the weir, and to provide an obstacle for trout. The obstacle requires the fish to stop or slow down, causing the electricity to be more effective as a barrier, and forces fish into the trap. This resulted in increased weir efficiency in past years (Heckel et al. 2020). We operated the electric weir at Palisades Creek through August to prevent late-spawning RBT and hatchery YCT from accessing tributary habitat. To prevent BNT and RBT from re-colonizing habitat upstream of the weir, the fish trap and ladder boards were removed from Burns Creek after the weekly YCT capture reached zero and the spawning run was complete.

We identified all fish captured at weirs to species, determined sex according to expression of gametes or based on head morphology, and measured length to the nearest mm. Cutthroat Trout were marked with a passive integrated transponder (PIT;12 mm; FDX) tag and an adipose clip, or a caudal fin punch and released upstream of the weir. We removed the adipose fin from PIT-tagged YCT as a secondary mark to make future scanning for PIT tags more efficient, and to estimate tag retention. All YCT captured in the trap with adipose fin clips were scanned for PIT tags. We identified fish that moved below the weir after interrogation at the trap by caudal fin punches or fresh adipose clip scars. Similarly, we used these marks to evaluate weir efficiency. We identified YCT that fell back below the weir (hereafter "fallback") to calculate accurate numbers of adult spawner abundance and fallback rates. We estimated the peak of the spawning run for each tributary as the date when we passed 50% of the total run of spawning fluvial YCT. We removed RBT from the tributary.

All fish that were phenotypically determined (Meyer et al. 2017a) to be pure YCT were tissue sampled and passed upstream of the weir. Tissue samples were provided to the IDFG Eagle Fish Genetics Lab (EFGL) in the fall, where a subsample (*n* = 200) from each tributary was randomly selected from all samples for evaluation of hybridization and introgression. At Rainey Creek, where the spawning run of fluvial YCT is lower than 200, all tissue samples were processed. To identify hybrids and evaluate introgression, samples were analyzed using Cocut GTseq v 3.0 201 single nucleotide polymorphism (SNP) panel (Matthew Campbell, IDFG, personal communication). There were 38 SNP markers in this panel that were diagnostic between RBT and YCT. We estimated genotypes using Bayesian model-based program, NEWHYBRIDS (Anderson 2008) to calculate the posterior probability of individuals belonging to one of six categories: (1) YCT, (2) RBT, (3) first generation hybrids, (4) second generation hybrids, (5) YCT backcrosses, and (6) RBT backcrosses. We used results from this test to assess the accuracy of identifying YCT x RBT hybrids by staff at the weirs. We retained all tissue samples at the EFGL for archival purposes.

We used backpack electrofishers on Burns and Pine creeks during the spawning season to estimate weir efficiencies. We captured fluvial-sized (≥ 300 mm) YCT upstream of the fish weirs and assessed each for marks and evidence of prior interrogation at the weirs. There is a canal head gate associated with the Palisades Creek weir and a fish screen downstream from the head gate in the canal. A bypass channel upstream of the fish screen returns downstream migrants back to the creek. We used a downstream trap on the Palisades Canal screen bypass channel to estimate weir efficiencies at Palisades Creek. We could not evaluate weir efficiencies at Rainey Creek due to the limited number of YCT passed upstream. We calculated efficiencies for weirs on Burns, Pine, and Palisades creeks as the number of YCT ≥ 300 mm with PIT tags or caudal fin punches divided by the total number of YCT ≥ 300 mm captured. The length cutoffs, used to discriminate between fluvial and resident fish, were previously calculated for each year from 2009 to 2020 (Vincent et al. *in press*). The length cutoffs were similar from year to year through that period, so we averaged (means of the ratio) the yearly length cutoffs to form a standard cutoff length (300 mm) at all the SFSR tributary weirs.

Cutthroat were PIT-tagged during several sampling objectives throughout the course of each year in tributaries and in the main-stem SFSR. We evaluated tag retention at each weir by sex since YCT were sexually dimorphic during sampling at the weirs. We estimated PIT tag retention by subtracting from 100, the number of adipose clipped YCT without a PIT tag and divided by the total number of adipose clipped YCT. We estimated tag retention by sex at each weir and calculated the average tag retention (means of the ratio) for all weirs for 2013 – 2020, and separately for 2021.

Rainbow Trout Harvest Incentive Program

In March, RBT were individually marked with coded wire tags (CWT) in the snout. We used boat electrofishing to capture RBT. We tagged all RBT with total lengths between 150 and 400 mm to avoid tag loss associated with fish mortality of young and old RBT. We captured, tagged, and released RBT from Palisades Dam downstream to Heise. Coded-wire tags were marked with five different six-digit numbers corresponding to the following monetary values: $50, $100, $200, $500, and $1,000. Currently, we attempt to tag approximately 575 fish annually from Palisades Dam to Byington boat ramp including 300 $50 tags, 200 $100 tags, 50 $200 tags, 20 $500 tags, and 5 $1,000 tags. Anglers wishing to participate in the program were required to turn in the heads of RBT and provide their contact information to the IDFG regional office directly or via freezers placed at the Byington and Conant boat ramp areas. On the first Friday of every month, "Fishead Friday", we collected RBT heads from the freezers and scanned (Northwest Marine Technology blue or T-wand) each for CWTs. Anglers were welcome to observe the scanning process in the fish lab at the IDFG regional office in Idaho Falls, Idaho. When we identified CWTs, we notified the angler with a phone call to verify their address, inform them of the reward amount, and thank them for their participation. We mailed rewards as checks to winners.

Main-stem Rainbow Trout Suppression

Prior modeling results (Devita 2014) suggested that the RBT abundance needed to be reduced by 20% in a given year, to provide substantial benefits to the YCT population. Since this modelling was completed, model parameters have changed with angler harvest likely decreasing and the RBT abundances increasing. To account for these changes and provide substantive benefits to the YCT population in 2021, IDFG set a goal of removing 30% of the RBT abundance from Palisades Dam, downstream to Dry Canyon (30.6 km), where RBT abundances are highest. The RBT abundance estimate at the Conant monitoring reach from 2020 (1,383 RBT > 152 mm/km) was expanded throughout the river length, to conservatively estimate 42,320 RBT in the SFSR. This informed our goal to remove approximately 12,696 RBT (42,320 RBT *0.30) in the spring of 2021.

Suppression was planned for early-April through May, which corresponded with spawn timing of RBT in the SFSR. Four sections were identified to focus effort in areas of known high densities of RBT. The first river reach was from Palisades Dam to the mouth of Palisades Creek (Table 1). The second reach was from the mouth of Indian Creek downstream to the U.S. Highway 26 Bridge (Swan Valley Bridge). The third reach was from the Swan Valley Bridge downstream to Pine Creek. The fourth river reach was from Dry Canyon downstream to Lufkin Bottom. One section was chosen for each day of sampling. Sampling was planned for four days per week from Monday (section 1) through Thursday (section 4) progressing downstream each day. We did not suppress on Fridays to reduce congestion with other boaters, and we did not start until 10:00 a.m. on Monday morning to provide anglers a chance to fish downstream of our efforts in reach one. We used two electrofishing boats for the majority of the suppression efforts but also attempted to use three electrofishing boats during weeks when spawning or CPUE was highest in the prior year. For boat electrofishing throughout this chapter, we used pulsed direct current (DC) at 7 – 12 amps, 200 – 350 volts, 50% pulse width, and a frequency of 60 Hertz. Electrofishing effort was concentrated on known RBT spawning locations or was focused on areas that typified spawning habitat on the downstream tails of pools, upstream head of riffles, or shoreline areas lateral to the channel. Electrofishing effort (minutes) was recorded for each boat daily, to estimate catch-per-unit-effort.

Captured fish were scanned for CWT tags during transfer from boats to 0.38 m^3 in-stream live-wells. Tagged RBT were counted and released back into the stream. A subsample (n = 100) of fish were measured for length and weight each week, over multiple river sections. When RBT were ripe and the expression of gametes was available, sex was recorded as well. Captured RBT were held in 0.38 m^3, partially submerged, in-stream live-wells until sampling was complete or the live well had reached capacity. Up to four 0.38 m^3 live-wells were utilized. Removed RBT were transported by boat to the boat ramp where they were counted into a 1.14 m^3 transport tank, then they were relocated to other fisheries. All RBT that succumbed to mortality between the initial capture to the final release were enumerated daily. Dissolved oxygen levels were maintained at 100% during transport and temperature was mitigated as necessary before release.

Main-stem Abundance Monitoring

Since 2009, fisheries staff have estimated trout abundances annually at the Lorenzo and Conant monitoring reaches of the SFSR, using mark-recapture boat electrofishing. Staff attempted surveys regularly at Conant since 1982 and at Lorenzo since 1987; though the frequency of surveys at Lorenzo was more sporadic in the late 1990s and early 2000s. We conducted our survey during the fall when river flows decreased near the end of the irrigation season. The Conant reach is representative of the upper SFSR, and begins at the Swan Valley Bridge (43.450674°, -111.397284°) continuing downstream 4.9 km (43.478871°, -111.428777°). The Lorenzo reach is representative of the lower SFSR and is 4.8 km long, approximately equally distributed upstream (43.721271°, -111.859816°) and downstream (43.746023°, -111.89158°) of U.S. Highway 20. In 2021, an additional reach in between Conant and Lorenzo, near Lufkin Bottoms, was surveyed in the canyon of the SFSR to describe the downstream expansion and increased species composition of RBT. This reach begins approximately 1.5 km upstream of Lufkin Bottom (43.574612°, -111.442951°) and extends 3.3 km downstream (43.586008°, -111.466400°).

Crewmembers with nets attempted to capture all trout encountered. Fisheries staff identified captured fish to species and measured them [mm; total length (TL)]. We weighed (g) a subsample (n = 100) for each species. Prior to release, trout were marked with a hole-punch in the caudal fin during our marking pass so that previously captured fish could be identified during our recapture pass 5 to 7 days later.

In 2021, we sampled the Lorenzo monitoring reach September 14 – 15 (marking pass) and September 20 – 21 (recapture pass). We sampled Lufkin Bottom monitoring reach September 22 – 23 (marking pass) and September 28 – 29 (recapture pass). We sampled the Conant monitoring reach October 4 – 6 (marking pass) and October 13 – 15 (recapture pass). We estimated abundance (trout/km) separately for each species for age-1 trout (at Conant and Lorenzo only) and for age-1 and older trout, at each monitoring reach. The age-1 trout lengths for each species (YCT 102 mm – 254 mm, BNT 178 mm – 279 mm, and RBT 152 mm – 279 mm) was previously estimated from an ageing study using sagittal otoliths (Schrader and Fredericks 2006; see Vincent et al. *in press* for BNT). We also estimated abundance for all trout species combined which included all trout ≥102 mm at each monitoring reach. We used the Fisheries Analysis+ program developed by the Montana Department of Fish, Wildlife, and Parks to calculate abundance estimates and standard deviations using the Log-likelihood method for 25.4 mm size groups. Confidence intervals (CIs; 95%) were calculated by multiplying the standard deviation by 1.96. Modified Peterson methods were used to estimate abundance instead of Log-likelihood if the P-value of the Pooled Chi-squared test of the Log-likelihood estimate was less than 0.05 or if the Modified Peterson abundance estimate was greater than 10% different from the Log-likelihood estimate. Typically, these conditions were met and Modified Peterson methods were used when the number of recaptures were limiting. The species composition of the trout community (age-1 and older) at Conant and Lufkin Bottom was estimated using the estimated abundances of YCT, RBT, and BNT.

We calculated the trend of the estimated abundance for the past 10 years for each trout species at the Lorenzo and Conant monitoring reaches. We chose this duration to monitor changes with trout abundances that would include multiple cohorts from each species in the SFSR. For trends at Lufkin Bottom, we used the entire time series which included five survey years from 2014 to 2021. We used linear regression to estimate the intrinsic rate of change in abundance for each species, with sample year as the independent variable and the $\log_e$ transformed abundance estimate (trout/km) as the dependent variable. The slope of the regression line fit to the $\log_e$ transformed abundance data is the intrinsic rates of change (r) for the population (Maxell 1999; High et al. 2008; Kennedy and Meyer 2015). Positive intrinsic rates of change ($r > 0$) indicate that abundance is increasing, and negative estimates of r indicate decreasing abundance in the population. Confidence intervals were estimated (90%) around the slope of the regression line. If 90% CIs did not include zero, the trend was considered significant. We used 90% CIs for better ability to detect significant trends in abundance (Peterman 1990; Maxell 1999).

We compared length-weight relationships for each trout species caught at all monitoring reaches of the SFSR. We compared observed weights with standard weights published for each species. We used the standards published by Kruse and Hubert (1997) for YCT, Simpkins and Hubert (1996) for RBT, and Milewski and Brown (1994) for BNT. We calculated relative weights (W_r) for each of the trout that were weighed and compared these with W_r from 2012 to 2018, and 2020 for trout at the Lorenzo and Conant reaches. We made comparisons using 95% CIs and 100 mm length groups, where we determined that non-overlapping intervals were statistically significant.

Tributary YCT Abundance and Distribution

Since 2016, each year we used multiple-pass, depletion, backpack electrofishing techniques to estimate abundance and distribution of YCT in one of the four main tributaries to the South Fork. We originally selected sites using a stratified random design, where the length that the stream order composed of the entire tributary weighted the number of sites selected in each stream order. During random selection of sites, we first identified all of the potential 100-m reaches in the drainage at the 1:24,000 scale using Forest Service maps projected in the web-based program CalTopo (CalTopo LLC, Truckee, California). To maximize precision of the drainage-wide estimate, we limited the number of sites in first-order streams known to be intermittent, and proportionally increased the number in perennial first order, and third-order streams (Meyer et al. 2006). We estimated the proportion of stream length (m) in first-order intermittent and perennial first-order, second order, and third-order streams. We used a random number generator to select sites from all available 100-m reaches in each stream order. In 2021, we surveyed 9.5% of possible first-order sites, 3.1% of second-order sites, and 8.2% of third-order sites. We utilized natural transitions between pools and riffles instead of block nets, and we adjusted site lengths accordingly.

We completed the first drainage-wide survey in Rainey Creek during the late summer and fall of 2017; with 53 sites surveyed. Of these sites, 23 were first order, 5 were second order, and 21 were third order. In 2021, we re-surveyed all of the original sites in Rainey Creek, plus one additional third-order site (n = 54 sites). For the majority of the sites in Rainey Creek, multiple-pass backpack electrofishing surveys were used to estimate abundance. Electrofishing teams consisted of one person with a backpack electrofisher and one person with a net and bucket. When stream widths exceeded 3 m, we used two to three backpack electrofishers. For backpack electrofishing throughout this chapter, we used a pulsed DC waveform operated at 60 Hz, and a goal of 100 Watts (Meyer et al. 2021). During sampling, persons with backpack electrofishers moved upstream covering all available habitats. We planned three passes at each site, unless we encountered zero fish on the first pass. If we captured fish on the first pass, we completed subsequent passes until we captured < 50% of the prior pass, regarding trout > 100 mm. At two sites in lower Rainey Creek, we used mark/recapture electrofishing from rafts, because the stream channel was too wide and deep for backpack electrofishing. During mark/recapture surveys we used pulsed DC at 9 – 12 amps, 200 – 350 volts, 50% pulse width, and a frequency of 60 Hertz. Crewmembers with nets attempted to capture all trout encountered. For reaches where mark/recapture techniques were used, a caudal fin punch was given to all trout and we returned 6-8 days after the marking pass to complete the recapture pass. For all survey sites, we identified captured fish to species, measured length to the nearest mm, and checked for marks and tags.

For the two mark/recapture survey sites, we used the Log-likelihood method for YCT > 100 mm in the Fisheries Analysis+ program (developed by the Montana Department of Fish, Wildlife, and Parks) to calculate abundance estimates and 95% CIs. Modified Peterson methods were used to estimate abundance instead of Log-likelihood if the P-value of the Pooled Chi-squared test was less than 0.05 or if the Modified Peterson abundance estimate was greater than 10% different from the Log-likelihood estimate. The conditions were met and Modified Peterson methods were used when the number of recaptures were limiting. For multiple-pass surveys, we estimated Maximum-Likelihood abundance estimates and 95% CIs using the program MicroFish 5.0 (Van Deventer 1989). For each survey site, estimates were calculated for each trout species > 100 mm and where sufficient capture allowed, for each species < 100 mm. We calculated the drainage-wide estimate of abundance (trout > 100 mm) and associated variances, and abundance by stream order and associated variances using the stratified-random-sampling formulas in Schaeffer et al. (1996).

Palisades Creek Rainbow Trout Suppression

We used backpack electrofishing units to capture trout in Palisades Creek and manually removed phenotypically identified RBT (Meyer et al. 2017a). We released all YCT. We completed electrofishing surveys during midsummer during base flow conditions to maximize capture efficiencies. In 2021, stream flows were lower than average because of drought conditions and poor snowpack the preceding winter. Steep gradient and associated high water velocities in combination with complex habitat in the upper 6.4 km of the suppression reach make electrofishing less efficient. Because previous efforts had successfully reduced RBT in the upper 6.4 km of Palisades Creek (Meyer et al. 2017b), we did not attempt suppression efforts in this upper reaches in 2021. We performed one removal pass in the lower 3.2 km of Palisades Creek. Electrofishing suppression started at the weir and proceeded upstream. We identified captured fish to species, measured length to the nearest mm, and checked for marks and tags.

Rainey Creek Eyed-Eggs

At the beginning of June, we built a temporary picket weir in Rainey Creek approximately 1.0 km upstream from the U.S. Forest Service boundary and upstream of private property. Forest Service property currently provides better stream habitat with fewer passage impediments and little concern for loss of fish due to entrainment in irrigation canals. Adult wild YCT brood fish were collected from the SFSR near the mouth of Rainey Creek near the end of June using jet boat electrofishing. Electrofishing effort was focused on areas near the confluence of Rainey Creek and focused on habitats that typified spawning areas on the downstream tails of pools, upstream head of riffles, or shoreline areas lateral to the channel. Ripe brood YCT were transported by boat to a 1.14 m^3 water tank in a pickup for relocation 0.36 km upstream from the picket weir (43.46375°, -111.25772°), and were released to spawn naturally.

In the fall, when YCT fry were large enough to be recruited to backpack electrofishing equipment (> 50 mm), we sampled fry from Rainey Creek using backpack electrofishing (spot shocking) in shallow, low-velocity areas lateral to the thalweg commonly inhabited by fry with a goal of 100 tissue samples from YCT fry smaller than 80 mm. Sampling was conducted in the upper reaches of Rainey Creek on private property and in the lower reaches of Rainey Creek on the U.S. Forest Service property. Genetic samples from fry as well as the adults used for brood were analyzed at the Eagle Fish Genetics Lab using Parental Based Tagging (PBT) techniques to identify fry resulting from adult YCT brood transfers into Rainey Creek.

Rainey Creek Habitat Rehabilitation

Before and after habitat projects were completed, fish surveys were conducted to describe changes in fish populations that might be attributed to changes in habitat improvements. Following the stream rehabilitation completed at lower Third Creek in 2018, post-project monitoring was conducted through the entire treatment reach for the second year. Similarly, post-project fish surveys were completed for the first year at two sections of Rainey Creek near Swan Valley. Two pre-project fish surveys were completed as well, one at upper Third Creek and another section in lower Rainey Creek. We conducted single-pass fish survey using one backpack electrofishing unit with an additional technician for netting fish. Data collected from captured fish included: species, length, and any marks/tags. Stream temperature (°C) and conductivity (µS) were also documented at the time of survey. Rehabilitation of stream habitat in upper Third Creek was initiated in late-September 2021 and is progressing into 2022.

RESULTS

Tributary Weirs

From April 14 through July 9, we captured 2,060 migrating trout at the Burns Creek weir, including six male RBT, 14 female RBT, and 2,040 YCT (918 males, 1,109 females, and 13 YCT of unknown sex; Table 8). The YCT spawning run peaked in Burns Creek on June 14. Fallback rates were 1.0% for male YCT and 0.6% for female YCT. To estimate efficiency, we captured 68 fluvial-sized YCT upstream of the Burns Creek weir and found that 66 were marked. Thus, the 2021 weir efficiency estimate for the Burns Creek weir was 97%. There was one YCT that succumbed to short-term mortality associated with the weir operations.

We operated the Pine Creek weir from April 13 through July 5, capturing 2,065 trout, of which 14 were RBT (7 males and 7 females; Table 8). The 2,051 YCT included 827 males, 1,212 females, and 12 YCT of unknown sex. The YCT spawning run in Pine Creek peaked on June 5. The fallback rates were 11% for female and 13% for male YCT. Upstream of the weir, we caught 55 YCT, of which 49 had been marked. We installed the uprights and check boards on June 1 and the efficiency estimate for the Pine Creek weir was 89%. There were 30 YCT that succumbed to short-term mortality associated with the weir operations.

We operated the Rainey Creek weir from April 1 through June 21, and passed six YCT, including five male, and one female (Table 8). The YCT spawning run peaked in Rainey Creek on June 1. We observed no fallback through the Rainey Cr weir. There were eight YCT that succumbed to mortality associated with the weir operations, not including the six that were successfully passed upstream.

At the Palisades Creek weir, we caught 669 trout between April 1 and July 13. We caught 16 RBT including 6 males and 10 females (Table 8). The 653 YCT included 298 male, 347 female, and 8 of unknown sex. The YCT spawning run in Palisades Creek peaked on June 18. Fallback rates for male YCT were 1.0% and 0.6% for female YCT. We captured 97 YCT and 73 had been marked. We installed the uprights and check boards on June 1 and the efficiency estimate for the Palisades Creek weir was 75%. There were 20 YCT that succumbed to mortality associated with the weir operations and another 18 that died during the efficiency estimates.

In 2021, tag retention was 96.5% for males and 79.8% for females. Tag retention (PIT tags) averaged 89.8% and 62.7% in males and females, from 2013 through 2020 (Table 9). In 2020, one RBT and one first generation hybrid (YCT x RBT) were passed upstream of Palisades Creek weir, otherwise the YCT passed upstream of each weir were pure (Table 10). We will not complete hybridization and introgression analyses of YCT passed upstream of each tributary weir in 2021 until summer of 2022. Those results will be added to the time series for these metrics (Table 10).

Rainbow Trout Harvest Incentive Program

In 2021, we continued the RBT harvest incentive program for the twelfth year. We marked 201 RBT with CWT in March 2021. This was significantly fewer than the average tagged annually from 2010 to 2019 (average = 1,020). We supplemented the shortfall from winter tagging by tagging additional RBT during the marking passes during the fall abundance surveys at Lorenzo (*n* = 56), Lufkin Bottom (*n* = 221), and Conant (*n* = 382). Tags released in 2021 included 567 RBT with $50 tags, 238 with $100 tags, 34 with $200 tags, 15 with $500 tags, and 6 fish with $1,000 tags. Anglers (*n* = 128) turned in 2,279 RBT in 2021 (Table 4). There were 48-tagged fish for a reward rate of 2.1%. The tag values and number that were turned in were $50 (*n* = 29), $100 (*n* = 15), $200 (*n* = 2), $500 (*n* = 1), and $1,000 (*n* = 1) for a total of $4,850, which was below the twelve-year average (average = $5,323). Of the tagged fish turned in by anglers, seven (14.6%) were tagged in 2020 and 2021.

Since 2010, we tagged 11,258 RBT (average = 936; Table 4). From 2010 through 2015, 650 tags were released annually on average, but from 2016 through 2019 the number of tags released annually increased by 142.6% (average = 1,575). Anglers have turned in 34,619 RBT and 765 (2.2%) tags were returned by anglers for a total of $63,400 rewarded to anglers during the duration of the program. The average reward rate (2.2%) ranged from a low (0.6%) in 2010 to a high (3.2%) in 2018 but has remained relatively consistent since 2013.

Main-stem Rainbow Trout Suppression

We completed 24 days of RBT suppression from April 19 – May 27, and we removed 10,654 RBT from the main-stem SFSR (25.2% of the RBT abundance). An additional 426 RBT (4.2%) were captured and then released when a CWT was identified. A subsample (*n* = 578) of fish were measured for length and weight in multiple river sections (Table 5). By extrapolating average weights by total catch per section we estimate 7,644.8 kg (> eight tons) of RBT were removed from the SFSR. Of the fish we measured, we also determined sex on 290 RBT and 66.9% were female.

We operated three electrofishing boats for 10 days, and two boats for 14 days. Daily catch combined for all boats averaged 444 RBT/d (Table 6). Combining all boats, we averaged 377.0 min of electrofishing per day (range 201.7 – 584.1) and totaled 8,426 min during the duration of spring suppression. Combined average catch-per-unit-effort (CPUE) was 1.3 RBT/min (range 0.6

– 3.8). Among boats, average CPUE per boat was consistent and ranged from 1.3 to 1.4 RBT/min Average CPUE was 1.4 RBT/min on days using two boats and 1.0 on days using three boats. We removed 2,956 RBT from reach one (average CPUE = 2.1 RBT/min), 3,125 from reach two (average CPUE = 1.4), 2,057 from reach three (average CPUE = 0.9), and 2,516 from reach four (average CPUE = 1.1).

We relocated RBT removed from the SFSR to five other fisheries (Table 7). We released RBT in the lower Henry's Fork of the Snake River (*n* = 7,000) at four different locations below Ashton Dam, the Big Lost River (*n* = 923), Louis Pond in Swan Valley (*n* = 647), Trail Creek Pond near Victor (*n* = 581), and Jim Moore Pond near Roberts (*n* = 1,095). We recorded 211 RBT (2.0%) that succumbed to short-term mortality between removal and final release.

Additionally in October 2021, 455 RBT (202.3 kg) were removed from the Conant reach during the recapture pass of the fall abundance surveys. We relocated these fish to Louis Pond (*n* = 106) and the main-stem Snake River in Idaho Falls (*n* = 347). Average CPUE was 0.9 RBT/min and two RBT succumbed to mortality (1.3%).

Main-stem Abundance Monitoring

During 2021, we captured 1,132 trout at the Lorenzo monitoring reach, including 401 YCT, 90 RBT, and 641 BNT of all age classes older than age-0. The estimated abundance for all trout, age-1 and greater, was 1,465 YCT and BNT per km. There were an estimated (via Modified Peterson) 561 age-1 and older YCT/km (± 255) and 904 age-1 and older BNT (± 223; Figure 1). We captured too few marked RBT to calculate an estimate using mark-recapture techniques, but RBT did compose 8.0% of the catch. Extrapolating 8.0% with the total trout estimate (1,465 trout/km), RBT abundance was approximately 117 RBT/km (all age classes) at Lorenzo. Trends in abundance at Lorenzo from 2012 to 2021 identified significant increases in both RBT (*r* = 0.016; ± 0.003 90% CI) and YCT (*r* = 0.098; ± 0.096 90% CI). There were no significant changes in BNT abundance during this period (*r* = -0.006; ± 0.047 90% CI). At Lorenzo, we have surveyed the abundance of age-1 and older RBT since 2010 using the mark/recapture methods above, though we still have not been able to estimate the number of age-1 RBT due to low sample sizes. Sample sizes of age-1 sized YCT limited our ability to estimate abundance of age-1 YCT too. We estimated 511 age-1 BNT/km (± 283), which was similar to the 10-yr average (547 age-1 BNT). Average W_r, for all size classes, for YCT at Lorenzo in past years (2012 – 2018, 2020) ranged from 93 to 105. Relative weights for YCT were higher in 2021 than past years with an average of 108. Brown Trout relative weights were consistent with past estimates and averaged 96, whereas past estimates ranged from 92 to 97. Too few RBT weights were measured to estimate relative weights at Lorenzo.

At the Conant monitoring reach, we captured 2,935 trout during the 2021 survey. This included 1,261 YCT, 977 RBT, 696 BNT, and 1 Brook Trout of all age classes. The estimated abundance for all trout, age-1 and greater, was 3,106 trout/km. There were an estimated 1,160 age-1 and older YCT/km (± 115), 666 age-1 and older BNT (± 168), and 1,280 age-1 and older RBT (± 179; Figure 2). Higher samples sizes at Conant allowed for the estimation of age-1 fish for all three major trout species including 365 YCT/km (± 210), 674 RBT (± 331), and 298 BNT (± 91; Table 2). At Conant, the trend in abundance did not significantly change from 2012 to 2021 for any trout species [YCT (*r* = 0.022; ± 0.042 90% CI), RBT (*r* = 0.045; ± 0.060 90% CI), and BNT (*r* = -0.010; ± 0.049 90% CI)]. Relative weights, averaged for all size classes, for YCT at Conant in past years (2012 – 2018, 2020) ranged from 89 to 102. Relative weights for the same period for BNT ranged from 92 to 99 and from 92 to 102 for RBT. In 2021, relative weights for YCT, BNT, and RBT at Conant were consistent with past estimates and averaged 102, 98, and

101, respectively. Of age-1 and older trout, RBT currently compose 41.2% of the total trout composition, while YCT and BNT compose 37.3% and 21.5% of the trout composition respectively (Table 3).

At the Lufkin Bottom monitoring reach, we captured 1,232 trout during the 2021 survey. This included 419 YCT, 373 RBT, and 440 BNT of all age classes. The estimated abundance for all trout, age-1 and greater, was 2,901 trout/km. There were an estimated 947 age-1 and older YCT/km (± 260), 1,699 age-1 and older BNT (± 746), and 670 age-1 and older RBT (± 152; Figure 3). Currently, RBT compose 20.0% of the total trout composition, while YCT represent 28.5% of the trout composition. At Lufkin Bottom, the intrinsic rate of change was calculated for all years surveyed which included 2014 – 2016, 2018 and 2021. Trends were stable with no significant changes in abundance [YCT (r = -0.033; ± 0.149 90% CI), RBT (r = 0.080; ± 0.124 90% CI), and BNT (r = 0.139; ± 0.143 90% CI)]. Relative weights, averaged for all size classes of YCT in past years (2014 – 2016, 2018) ranged from 90 to 98, but were 104 in 2021. Relative weights for the same period for BNT and RBT ranged from 90 to 99. In 2021, relative weights for BNT, and RBT at Lufkin Bottom were consistent with past estimates and averaged 99, and 96, respectively.

Tributary YCT Abundance and Distribution

Mapping the Rainey Creek drainage included a total of 71.2 km of stream. Of the total stream length, 28.4 km were first-order, 16.1 were second-order, and 26.7 were third-order streams. We surveyed 54 sites in the Rainey Creek drainage in 2021 between July 7 and July 21. These included 27 first-order sites, of which no fish were present, five second-order sites, and 22 third-order sites. First-order sites were generally dry (n = 21), or too thick with underbrush to sample (n = 3). Fish were absent from the remaining first-order sites that were sampled. The majority of fish were captured in second-order and third-order sites (Figure 5). Two third-order sites where mark/recapture techniques were used were combined into one continuous 1,337 m site. We captured YCT, RBT and BNT during the survey. We captured 28 RBT at six third-order sites, three of which (n = 4 RBT) were upstream of the weir. The majority of RBT were captured near the mouth of Rainey Creek. Brown Trout were captured at 17 third-order sites. Yellowstone Cutthroat Trout were captured at five second-order sites, and 23 third-order sites (Figure 5). Fluvial-sized YCT (≥ 300 mm) were captured at 12 of the 55 sites, all of which were third-order sites, however there were no tagged or marked fish captured which would identify that a fluvial-sized fish had previously been interrogated at the weir. In 2021, the average density of YCT/100 m^2 was 4.1, and 12.3 for second- and third-order streams. Densities in second- and third--order streams were lower in 2021 compared to 2017 (8.8 and 18.9 YCT/100 m^2; IDFG unpublished data). In 2021, 3,932 (± 1,445) YCT > 100 mm were estimated throughout Rainey Creek drainage. This was lower than the previous estimate conducted in 2017 [6,469 YCT > 100 mm (± 1,877)] IDFG unpublished data].

Palisades Creek Rainbow Trout Suppression

We completed one electrofishing pass of suppression and removal of RBT in the lower 3.2 km of Palisades Creek during two days on August 11 – 12. We caught 392 trout, including 273 YCT and 119 RBT. Rainbow Trout made up 30.4% of the catch. The average total length of of RBT was 203 mm.

Rainey Creek Eyed-Eggs

We relocated a total of 100 YCT collected from the SFSR on June 25 and June 28 using boat electrofishing to capture adult fish. This included 46 females (average TL = 393 mm) and 54 males (average TL = 394 mm).
We captured 90 YCT fry (average TL = 52 mm) using backpack electrofishing from Rainey Creek on October 28 and obtained a tissue sample from each fish. Samples are pending analyses at IDFG's Eagle Fish Genetic Lab and results are anticipated in the summer of 2022.

Rainey Creek Habitat Rehabilitation

Monitoring the benefits of stream rehabilitation occurred at six survey sites in 2021. On July 8, we completed post-project monitoring at one site in lower Third Creek for the second year and pre-project survey occurred at two sites in upper Third Creek. No fish were observed in Third Creek during these surveys. Water temperatures ranged from 14.3 to 18.3°C at the time of surveys in Third Creek.

DISCUSSION

Fish management for the SFSR YCT population has maintained three primary goals; 1) preserve genetic integrity and population viability of native cutthroat trout; 2) reduce RBT and hybrid trout to less than 10% of the species composition at the Conant monitoring reach (IDFG 2019); and 3) maximize juvenile cutthroat trout production from the tributaries (IDFG 2007). Sometimes, we design management goals to dually satisfy biological and societal goals. We identify the preferences of IDFG's angling constituency through angler opinion surveys (Koenig 2020). The primary goals for the SFSR align with anglers' preferences both in the recent and previous surveys, because anglers identified managing for native trout fisheries (specifically cutthroat trout) as a primary goal. Respondents to the survey identified native trout management as the third most important management activity only behind protecting and improving fish habitat and maintaining and improving existing fishing access sites. Results from this study, which were similar to previous surveys, identify that the majority of anglers that fish in Idaho prefer management for native trout fisheries over managing for quality/trophy trout in rivers and streams.

The SFSR weir program is one of the most important strategies for managing threats from RBT to fluvial and resident YCT in tributaries (Van Kirk et al. 2010). The proportion of SFSR YCT that spawn in the tributaries is thought to be lower than the proportion that spawn in the main stem (IDFG unpublished data), though precise estimates of these proportions are not available. If the majority of YCT spawn in the main-stem SFSR, then tributary weirs alone will not be sufficient to protect the genetic integrity of the YCT population. This was the eighth consecutive year since 2010 that we were able to operate weirs on all four major spawning tributaries of the SFSR with high efficiency. The number of adult spawning fluvial YCT trapped at tributary weirs was higher than the 10-yr. average (average = 3,999) but lower than the past two years. Weir efficiency was significantly lower at Palisades Creek than 2013 – 2018, but higher than 2020. There is a trend to suggest the problem is persistent, so in 2021 we had the system inspected by the manufacturer (Smith-Root, Inc.). Few deficiencies were revealed through inspection, and with drought conditions in 2021 there was very little water remaining in the downstream creek channel after irrigation needs were met. This should have increased weir efficiency significantly as the majority of the remaining water was passing through the fish trap and should have attracted fish, while leaving little to no water in the creek channel for a fish to potentially jump a > 1.2 m diversion dam, within an electric field maintained at the highest electrical settings recommended (Larson et al. 2014). Data from a PIT tag array identify a unique life history strategy, where adult fluvial YCT ascend the tributary to spawn, and then reside in the tributary upstream of the weir for 12 – 14 months until the next spawning event. If fish remained upstream of the weir through the winter, they would not have been interrogated and marked at the weir and would therefore reduce the estimated efficiency. Data from PIT tag detections at the Palisades Creek array identify that this life history strategy exists, though current sample sizes of fish exhibiting this behavior are limiting (data not shown). PIT tag data also substantiate the low estimated weir efficiency with marked fish recorded by antennae upstream of the weir that were not interrogated at the Palisades Creek trap.

Angler harvest and manual suppression of RBT are currently the most promising tools for managing competitive threats to YCT in the main-stem SFSR. We removed 13,388 RBT from the SFSR through combined angler harvest and manual suppression in the spring and fall. These efforts combined, resulted in the removal of 31.6 % of the RBT abundance estimated in 2020. Suppression alone fell short of removing 12,695 RBT, however the angler contributions from the incentive program helped to achieve our goal. Anglers participating in the Harvest Incentive Program clearly contributed to fisheries management in 2021, by helping to achieve our suppression goal. During the twelve years of this program, anglers have harvested over 34,000 RBT. This program provides important interactions between biologists and the fishing public. Through conversations with anglers on “Fishhead Friday,” it is apparent the program does successfully motivate additional angler harvest of RBT on the SFSR. During 2021, we tagged the fewest RBT in twelve years and the reward rate is beginning to drop. Additional tagging during spring RBT suppression and fall abundance surveys needs to continue. Suppression of RBT should help increase this reward rate as we returned tagged fish to the river, to further incentivize RBT harvest. If angler harvest were increased, manual suppression effort could decrease commensurately.

Using the same methods to develop our spring RBT suppression goal as in 2021, we can estimate that in 2022 we need to remove 11,750 RBT > 152 mm. If average CPUE is similar to 2021 (1.4 RBT/min), and we can average the same amount of effort per day (377.0 min), then we need to suppress for approximately 22 days in the spring of 2022 to reduce abundances of RBT by 30% through suppression alone. If more effort were focused on reach one, where CPUE was highest, we could achieve our suppression goals sooner. This suppression goal of 11,750 RBT is similar to our goal of 12,695 RBT in 2021 and might be discouraging to some who might expect the suppression goal, and RBT abundances to decrease annually. However, model results presented by Devita (2014) suggest that under our suppression scenario it might take at least five years before significant declines are observed in the RBT abundances. We need several consecutive years of suppression to determine if manual suppression is a viable tool to achieve management goals.

The current species composition of RBT (33%) is substantially higher than our goal of RBT species composition of 10% at Conant (IDFG 2019), though several management efforts continue to pursue that goal. Annual abundance monitoring in the SFSR in 2021 identified that abundances were slightly above the 10-year average (3,027 trout/km). At Conant, YCT are now the most abundant species and currently compose 43% of the trout population. Cutthroat have not been the most abundant trout species since 2011. The high abundances of YCT observed in the past several years are likely due to several successive good winters that resulted in higher stream flows in the SFSR and tributaries, specifically in the winter months (Battle et al. 2010), though these good water years likely benefitted RBT too. However, with relatively even proportions of RBT and YCT, there are increased opportunities for density-dependent population effects from intra- and interspecific competition (McCormick and High 2020). McCormick and High

(2020) suggest age-0 – age-2 YCT are at a competitive disadvantage to RBT and hybrids (Seiler and Keely 2007a; Seiler and Keely 2009; Van Kirk et al. 2010). If true, then the effects of density dependence to the YCT population may lead to lower year-class strength for affected cohorts. During the past ten years, the intrinsic rate of change identified a significant increase in abundance for YCT and RBT at Lorenzo. At Lorenzo, the abundance of YCT continues to be high, though CIs were wide around the point estimate. In 2019 and 2020, the RBT species composition of RBT was 2.3% and 4.8%, respectively. In 2021, we observed the third consecutive year where the proportion of RBT in the catch (8.0%) has nearly doubled. We identified an increase in RBT abundance at Lorenzo the past two years (data not shown) but this was the first year the trend was statistically significant.

Main stem abundance monitoring identified an increased prevalence of RBT in the SFSR despite earlier management efforts. This suggests that additional management actions, such as manual suppression, need to continue. Suppression of nonnative species is not a novel management strategy and there are many examples of other fish management regulations in other states that require anglers to carry out the mandatory kill/harvest regulations on other nonnative species, such as Walleye *Sander vitreus* in Wyoming, BNT in Colorado, and RBT in Yellowstone National Park to name a few. Additional efforts to manage for native salmonids include netting and removal of Lake Trout *Salvelinus namaycush* for native trout conservation at Yellowstone Lake and Lake Pend Oreille (Syslo et al. 2011; Dux et al. 2019). Manual suppression of RBT likely represents the most efficient management tool to achieve IDFG's management goal of less than 10% RBT in the main-stem SFSR (IDFG 2019) and less than 10% introgression between RBT and YCT (IDFG 2007). At the Conant monitoring reach, the species composition of YCT and RBT estimated in 2021 do not show that significant progress was achieved toward IDFG's management goal of 10% RBT. Suppression at similar levels to 2021 needs to continue for several more years to determine if manual suppression is going to help achieve this goal.

Basin-wide surveys of SFSR tributaries document a diversity of life history strategies persisting in the SFSR YCT population. Both resident and fluvial YCT use all reaches including the headwater stream sections. Stochastic variability in weather patterns (e.g., snowpack, summer air temp., etc.) are likely notable factors influencing stream flow, which also influence fluvial YCT distributions and abundance. This was the second time we surveyed Rainey Creek in its entirety, and we need to complete additional surveys here to assess trends in abundance and distribution.

Gunnell et al. (2008) recommended evaluating the effectiveness of the tributary weirs at reducing hybridization and introgression between RBT and YCT at SFSR tributaries. Results herein suggest that introgression and hybridization is very low in fluvial fish passed upstream of the weirs. Estimates of introgression and hybridization herein (Table 10) reflect the successful identification and removal of RBT and hybrids at tributaries, but do not necessarily describe the spawning population upstream of the weir. For example, during the past several years during tributary YCT abundance and distribution surveys, we have estimated thousands of YCT in each of the major tributaries after the majority of the fluvial fish have returned to the main stem (Heckel et al. 2020; Vincent et al. *in press*). Thus, assuming resident fish are several times as abundant as fluvial fish, estimating introgression at the weir may be under representative or spatially biased by the spawning YCT population upstream. Gunnell et al. (2008) measured hybridization and introgression at multiple locations within each tributary and found significantly higher introgression (average = 4.5%) than current estimates (average = 0.2%). Upstream of the weirs there may be relict populations of hybrids and RBT that continue to provide threats to pure YCT. Manual suppression efforts upstream of the weirs in Burns and Palisades creeks have contributed positively to reducing hybridization and introgression in tributaries as well (Meyer et al. 2017b).

However, RBT in Palisades Creek persist upstream from the weir. The average sizes of RBT in Palisades Creek suggest this is a resident population and they are increasing in species composition. This is not likely the case in Burns Creek, where efficiency has been consistently higher. Replicating the sampling conducted by the Gunnell et al. (2008) and evaluating hybridization and introgression throughout a more thorough spatial scale may confirm the low levels of introgression measured at the weirs and describe the positive impact associated with the weir program.

Rainey Creek is the only major tributary that is not demonstrating the success relative to the other tributaries. Poor weir efficiency from 2001 to 2009 resulted in low catch and reduced effectiveness in removing RBT. In 2011, we constructed a new weir downstream, closer to the mouth of Rainey Creek to protect more of the system from invading RBT. We anticipated higher catches with a weir located upstream from the mouth only 5.1 km versus 14 km in an area with multiple irrigation diversions and potential passage problems. The higher catches have yet to materialize. Cutthroat Trout spawning runs at Rainey Creek continue to be stagnant despite increased adult YCT abundance in the SFSR, and catch has similarly increased at adjacent spawning tributaries in Pine and Palisades creeks during the same period. It is possible that fluvial YCT in Rainey have gone through a bottleneck, which Campbell (1990) defined as a severe reduction in the demographic size of a population. If bottlenecks are severe enough, inbreeding depression can occur which limits the ability of the population to recover because of reduced levels of reproductive fitness (Frankham 1995). The abundant resident YCT population in the upper portions of the system maintaining genetic diversity likely mitigates for adverse effects of genetic bottlenecks in Rainey Creek. However, the fluvial component of the Rainey Creek sub-population has not recovered and does not show evidence for a trend towards recovery. Transferring adult fluvial YCT from the main-stem SFSR into Rainey Creek might help mitigate inbreeding depression. However, if progeny drift downstream or try to migrate to the main stem in late summer it is likely they will be entrained by irrigation diversions, eliminating any benefits from the past egg box program or adult fish transfers in 2021.

Past agricultural practices have degraded trout habitat in several areas of Rainey Creek, and persistent issues with passage barriers and over allocation of stream water for irrigation (RivHab 2020) limit the biological productivity of Rainey Creek. The lower river is degraded to the extent that there are no reference conditions to target for stream restoration, thus we refer to this work as rehabilitation. Spawning habitat is limited due to sedimentation, limiting riparian cover, and high stream temperatures (Lehotsky 2019; RivHab 2020). In 2018, a first-order tributary (Third Creek) to Rainey Creek was rehabilitated to mitigate water temperatures in lower Rainey Creek. In 2020, land managers rehabilitated two additional reaches in lower Rainey Creek. In 2021, IDFG in collaboration with the private landowner, Henry's Fork Foundation – South Fork Initiative, Natural Resources Conservation Service, and the U.S. Fish and Wildlife Service completed rehabilitation of an additional reach. Comparisons of pre and post project surveys at Rainey Creek identify no benefit to Third Creek with fish still absent in rehabilitated habitat. Similar results have been observed at reaches rehabilitated by management partners, where little to no short-term benefits have been documented. In 2021, access was granted by a private landowner to a collaborative group of resource managers in July, and we identified a river reach (approximately 100 m) dewatered by irrigation. This and several other impediments to passage are detrimental to the fluvial life history of YCT in the SFSR as upstream migrating adults may not successfully access suitable spawning habitat. However, even if upstream migrants manage to successfully spawn, out-migrating juveniles are likely to be entrained by irrigation diversions. Rehabilitation of additional reaches to restore connectivity to the upper river are critical for the fluvial YCT in Rainey Creek (Cegelski et al. 2006).

MANAGEMENT RECOMMENDATIONS

1. Maintain weir program in the four primary YCT tributaries of the SFSR to provide spawning and rearing areas with reduced possibility of competition or hybridization with RBT.

2. Increase CWT tagging during spring suppression and fall abundance surveys with the goal of increasing angler harvest of RBT.

3. Complete 22 days of boat electrofishing in 2022 in known spawning areas of the main-stem SFSR to suppress RBT and reduce hybridization and competition between RBT and YCT.

4. Develop model simulations to estimate what levels of suppression and duration are needed to achieve the current management goal of <10% RBT in the main-stem SFSR with observed suppression rates and current angler harvest rates.

5. Complete a study to describe the age, growth, and survival of Rainbow Trout in the SFSR.

6. Replicate electrofishing and tissue sampling conducted by Gunnell (2008) in SFSR tributaries to estimate introgression between RBT and YCT and describe the effectiveness of the tributary weir program.

7. Evaluate introgression between RBT and YCT in the main-stem SFSR and compare introgression levels with YCT spawning in tributaries.

Table 1. Rainbow Trout suppression approximate boundary locations and days of the week where electrofishing boats were operating in 2021 on the South Fork Snake River.

Day	River reach	Launch site	Upstream boundary	Downstream boundary
Monday	1	Palisades Ramp	Palisades Dam	Palisades Creek
Tuesday	2	Spring Cr Ramp	Indian Creek	Swan Valley Bridge
Wednesday	3	Conant Ramp	Swan Valley Bridge	Pine Creek
Thursday	4	Conant Ramp	Dry Creek	Lufkin Bottom

Table 2. Age-1 abundance and 95% confidence intervals (CIs) of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) at the Conant monitoring reach of the South Fork of the Snake River in 2021.

Year	YCT/km	95% CI	RBT/km	95% CI	BNT/km	95% CI
2010	---	---	370	308	248	209
2011	560	391	582	431	448	165
2012	428	117	755	316	531	116
2013	251	127	298	88	244	54
2014	---	---	296	205	72	37
2015	126	77	365	129	270	107
2016	217	110	261	65	273	58
2017	339	133	790	170	325	52
2018	265	198	884	343	392	159
2019	180	83	466	143	198	62
2020	---	---	514	153	453	197
2021	365	210	674	331	298	91

Table 3. Composition of age-1 and older Yellowstone Cutthroat, Brown, and Rainbow trout captured at the Conant monitoring reach of the South Fork Snake River, from 2010 through 2021.

Survey year	Yellowstone Cutthroat Trout (%)	Rainbow Trout (%)	Brown Trout (%)
2010	42.3	41.0	16.7
2011	38.2	37.1	24.8
2012	33.6	38.0	28.4
2013	42.0	35.4	22.6
2014	40.5	38.6	20.9
2015	42.7	26.1	31.2
2016	38.6	34.5	26.9
2017	33.9	44.1	22.0
2018	29.3	49.2	21.5
2019	39.6	44.6	15.8
2020	43.3	35.4	21.3
2021	37.3	41.2	21.5

Table 4. Summary of the Rainbow Trout (RBT) Harvest Incentive Program on the South Fork Snake River from 2010 through 2021.

Year	RBT tagged	# Anglers	RBT turned in	Tags found	Reward rate (%)	Tag value ($) 50	100	200	500	1,000	Annual payout ($)
2010	575	640	2,861	18	0.6	12	3	2	0	1	2,300
2011	600	237	2,001	16	0.8	5	8	1	2	0	2,250
2012	860	207	1,854	37	2.0	23	9	3	2	0	3,650
2013	530	239	2,441	75	3.1	55	13	2	2	1	6,450
2014	705	175	3,587	75	2.1	40	29	7	0	0	6,300
2015	628	137	2,599	72	2.8	48	13	12	0	0	6,100
2016	1,338	117	2,681	45	1.7	35	7	2	1	0	3,350
2017	1,540	87	3,303	71	2.1	58	10	2	0	0	4,300
2018	2,068	104	3,205	102	3.2	75	18	8	2	1	9,150
2019	1,353	186	4,538	114	2.5	93	13	5	2	0	7,950
2020	201	143	3,270	92	2.8	67	20	2	2	0	6,750
2021	860	128	2,279	48	2.1	29	15	2	1	1	4,850
Total	11,258	2,400	34,619	765	2.2	540	158	48	14	4	63,400

Table 5. Summary of lengths (TL), weights and sex of Rainbow Trout (RBT) removed the South Fork Snake River by river reach in 2021.

River reach	Sample size	Avg. length (mm)	Avg. weight (g)	# Male	# Female	#Unknown	Female (%)
1	96	409.1	858.1	21	17	2	0.45
2	145	408.9	726.1	34	58	53	0.63
3	194	304.8	650.8	20	61	113	0.75
4	142	380.5	596.5	21	58	64	0.73
Total	577			96	194	232	
Average		375.8	707.9				0.67

Table 6. Spring Rainbow Trout (RBT) suppression electrofishing summary statistics including catch-per-unit-effort (CPUE), and fish with coded wire tags (CWT) detected on the South Fork Snake River in 2021.

Date	River reach	Total effort (min)	CPUE (#RBT/min)	RBT removed	Biomass removed (kg)	#CWT	Cumulative catch
19-Apr	1	249.3	1.4	347	297.8	7	354
20-Apr	2	249.1	2.5	613	445.1	23	990
21-Apr	3	292.1	0.8	227	147.7	10	1,227
22-Apr	4	345.2	1.0	341	203.4	10	1,578
26-Apr	1	264.6	1.3	345	296.1	7	1,930
27-Apr	2	494.4	0.9	459	333.3	23	2,412
28-Apr	3	483.1	0.9	440	286.4	21	2,873
29-Apr	4	488.6	0.8	410	244.6	18	3,301
3-May	1	---	---	442	379.3	18	3,761
4-May	2	---	---	605	439.3	18	4,384
5-May	3	395.0	0.8	328	213.5	25	4,737
6-May	4	361.2	1.5	548	326.9	13	5,298
10-May	1	230.1	2.5	584	501.1	20	5,902
11-May	2	404.7	1.5	595	432.0	16	6,513
12-May	3	584.1	0.8	478	311.1	25	7,016
13-May	4	376.2	1.0	379	226.1	8	7,403
17-May	1	240.1	3.3	790	677.9	27	8,220
18-May	2	508.4	0.9	453	328.9	24	8,697
19-May	3	364.7	0.9	311	202.4	26	9,034
20-May	4	538.9	0.9	482	287.5	9	9,525
24-May	1	345.5	1.3	448	384.4	29	10,002
25-May	2	374.5	1.1	400	290.4	17	10,419
26-May	3	440.5	0.6	273	177.7	25	10,717
27-May	4	395.9	0.9	356	212.3	7	11,080
Total		8,426		10,654	7,644.8	426	11,080
Average		377.0	1.3	444		18	

Table 7. Stocking summary statistics from Rainbow Trout (RBT) relocated during suppression on the South Fork Snake River in 2021.

Pond name	RBT	Biomass stocked (kg)	Deceased	Mortality (%)
Louis Pond	347	297.8	0	0.0
Trail Cr Pond	253	183.7	5	1.9
Louis Pond	300	217.8	0	0.0
Trail Cr Pond	227	147.7	0	0.0
Jim Moore Pond	320	190.9	10	3.0
Jim Moore Pond	343	294.3	2	0.6
Trail Cr Pond	101	73.3	3	2.9
Del Rio - Henry's Fork	337	244.7	8	2.3
Del Rio - Henry's Fork	441	287.0	0	0.0
Del Rio - Henry's Fork	400	238.6	10	2.4
Jim Moore Pond	432	370.7	10	2.3
Upper Big Lost	597	433.5	8	1.3
Upper Big Lost	326	212.1	2	0.6
Del Rio - Henry's Fork	444	264.9	60	11.9
Bubble Land - Henry's Fork	582	499.4	0	0.0
Bubble Land - Henry's Fork	593	430.6	0	0.0
Bubble Land - Henry's Fork	476	309.8	1	0.2
Bubble Land - Henry's Fork	318	189.7	57	15.2
Warm Slough - Henry's Fork	778	667.6	12	1.5
Warm Slough - Henry's Fork	423	307.2	4	0.9
Warm Slough - Henry's Fork	304	197.8	7	2.3
Warm Slough - Henry's Fork	481	286.9	1	0.2
Beaver Dick - Henry's Fork	447	383.6	1	0.2
Beaver Dick - Henry's Fork	392	284.6	8	2.0
Beaver Dick - Henry's Fork	263	171.1	0	0.0
Beaver Dick - Henry's Fork	321	191.5	2	0.6
Total	10,246	7,376.8	211	
Average	394	283.9	8	2.0

Table 8. South Fork Snake River tributary weir summary statistics from 2009 through 2021.

Location and year	Weir type	Operation dates	Estimated weir efficiency (%)	Catch: Cutthroat Trout	Catch: Rainbow Trout	Catch: Total
		Burns Creek				
2009	Fall/velocity	April 9 - July 22	98	1,491	2	1,493
2010	Fall/velocity	March 26 - July 14	100	1,550	2	1,552
2011	Fall/velocity	March 23 - July 12	90	891	5	896
2012	Fall/velocity	March 24 - July 11	90	496	0	496
2013	Fall/velocity	April 4 - July 2	98	888	6	894
2014	Fall/velocity	April 1 - July 3	90	833	12	845
2015	Fall/velocity	April 6 - July 3	94	1,357	1	1,358
2016	Fall/velocity	April 4 - July 3	98	1,528	7	1,535
2017	Fall/velocity	April 1 - June 27	87	759	4	763
2018	Fall/velocity	April 3 - July 6	100	1,303	9	1,312
2019	Fall/velocity	April 8 - July 8	94	1,322	6	1,328
2020	Fall/velocity	April 3 - July 9	95	1,800	16	1,816
2021	Fall/velocity	April 14 - July 9	97	2,040	20	2,060
		Pine Creek				
2010	Electric	April 13 - July 6	NE	2,972	3	2,975
2011	Electric	April 11 - July 9	49	1,509	1	1,510
2012	Electric	March 28 - July 1	NE	1,427	3	1,430
2013	Electric	April 5 - June 22	89	1,908	1	1,909
2014	Electric	April 7- June 30	70	899	7	906
2015	Electric	April 1 - June 25	78	1,864	3	1,867
2016	Electric	April 1- June 22	93	3,240	8	3,248
2017	Electric	April 3 - June 26	67	2,695	2	2,697
2018	Electric	April 2 - June 26	94	2,075	6	2,081
2019	Electric	April 8 - July 5	72	3,191	8	3,199
2020	Electric	April 6 - July 13	85	3,183	16	3,199
2021	Electric	April 13 - July 5	89	2,051	14	2,065
		Rainey Creek				
2011	Electric	March 28 - June 28	NE	0	0	0

Table 8 (continued)

Location and year	Weir type	Operation dates	Estimated weir efficiency (%)	Catch Cutthroat Trout	Location and year	Weir type
2012	Electric	April 18 - June 23	NE	7	0	7
2013	Electric	ND	ND	ND	ND	ND
2014	Electric	April 29 - June 25	NE	56	2	58
2015	Electric	April 2 - June 21	NE	73	2	75
2016	Electric	April 1 - June 23	NE	19	2	21
2017	Electric	April 3 - June 26	NE	37	2	39
2018	Electric	April 2 - June 26	NE	37	0	37
2019	Electric	April 8 - June 24	NE	70	0	70
2020	Electric	April 3 - July 3	NE	85	2	87
2021	Electric	April 1 - June 21	NE	6	0	6
		Palisades Creek				
2009	Electric	May 12 - July 20	26	202	4	206
2010	Electric	March 19 - July 18	86	545	50	595
2011	Electric	April 7 - June 15	NE	30	13	43
2012	Electric	March 24 - July 2	88	232	20	252
2013	Electric	April 5 - July 8	96	619	23	642
2014	Electric	April 2 - July 18	98	734	63	797
2015	Electric	April 2 - July 18	95	832	14	846
2016	Electric	April 1 - July 6	99	958	27	985
2017	Electric	April 3 - July 21	100	755	63	818
2018	Electric	April 2 - July 10	92	478	18	496
2019	Electric	April 8 - July 23	83	627	6	633
2020	Electric	April 3 - July 8	68	786	13	799
2021	Electric	April 1 - July 13	75	653	16	669

Table 9. Tag retention (PIT) by sex for fluvial Yellowstone Cutthroat Trout. Retention was measured at three tributary weirs on the South Fork Snake River from 2013 through 2021.

	Palisades		Pine		Burns		Annual average	
Year	Male (%)	Female (%)	Male (%)	Female (%)	Male (%)	Female (%)	Male (%)	Female (%)
2013	91.3	71.9	95.7	49.8	91.7	49.4	92.9	57.0
2014	96.4	47.1	71.4	63.9	91.3	60.0	86.4	57.0
2015	92.7	78.6	94.7	75.0	87.0	56.3	91.5	69.9
2016	100.0	84.2	88.2	68.4	93.9	68.6	94.0	73.7
2017	81.0	31.8	88.4	47.1	---	---	84.7	39.4
2018	95.5	64.7	81.7	82.4	91.7	35.4	89.6	60.8
2019	80.8	94.7	87.8	79.2	90.2	56.9	86.3	76.9
2020	95.2	75.0	88.2	71.4	95.0	52.6	92.8	66.4
2021	94.7	88.9	97.2	92.9	97.6	57.7	96.5	79.8
Average	92.0	70.8	88.2	70.0	92.3	54.6	89.8	62.7

---- Estimates were unavailable due to small sample sizes (<3)

Table 10. Estimated percent hybridization and percent introgression, and sample sizes (*n*), of Yellowstone Cutthroat Trout passed upstream of weirs at the four primary tributaries to the South Fork Snake River during 2015, 2018 – 2020.

	Burns			Pine			Rainey			Palisades		
Year	*n*	Hybrid. (%)	Intro. (%)	*n*	Hybrid. (%)	Intro. (%)	*n*	Hybrid. (%)	Intro. (%)	*n*	Hybrid. (%)	Intro. (%)
2015	30	0.00	---	30	0.00	---	30	0.00	---	30	6.67	---
2018	198	1.00	0.03	128	2.29	0.41	124	0.00	0.00	195	2.01	0.30
2019	199	0.50	0.17	200	0.00	0.01	81	2.47	0.23	200	3.00	0.55
2020	186	0.00	0.00	191	0.00	0.00	47	0.00	0.00	200	2.00	0.15

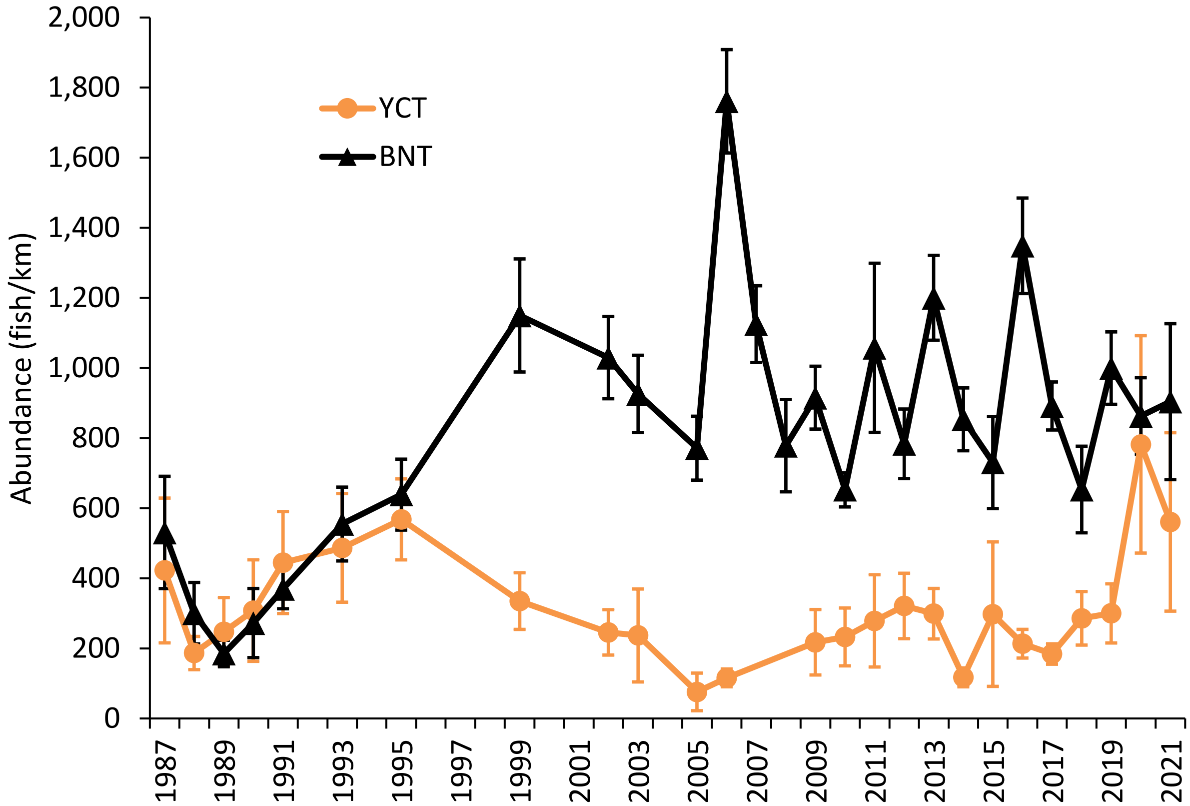

Figure 1. Abundance estimates (fish/km) and 95% confidence intervals for Yellowstone Cutthroat Trout (YCT) and Brown Trout (BNT) at the Lorenzo monitoring reach on the South Fork Snake River from 1987 through 2021.

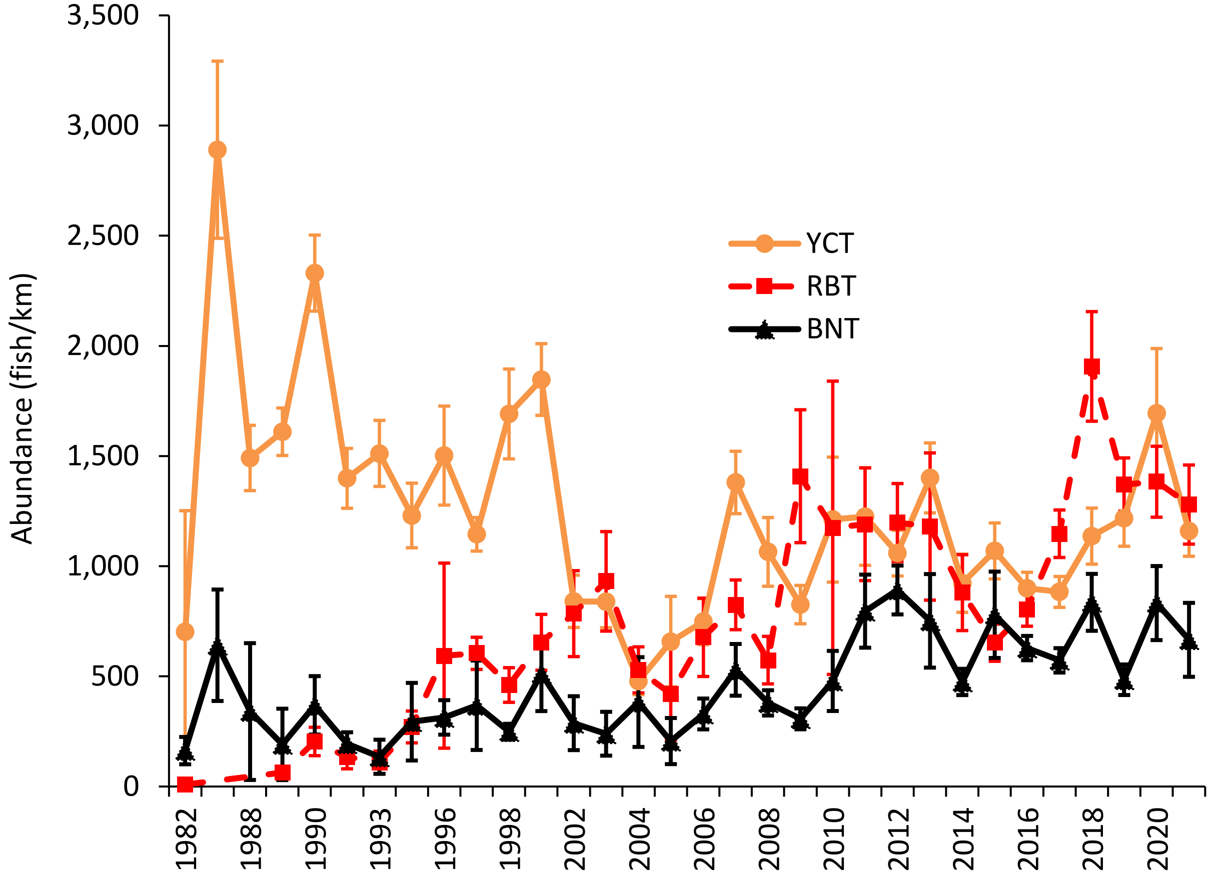

Figure 2. Abundance estimates (fish/km) and 95% confidence intervals for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) at the Conant monitoring reach on the South Fork Snake River from 1982 through 2021.

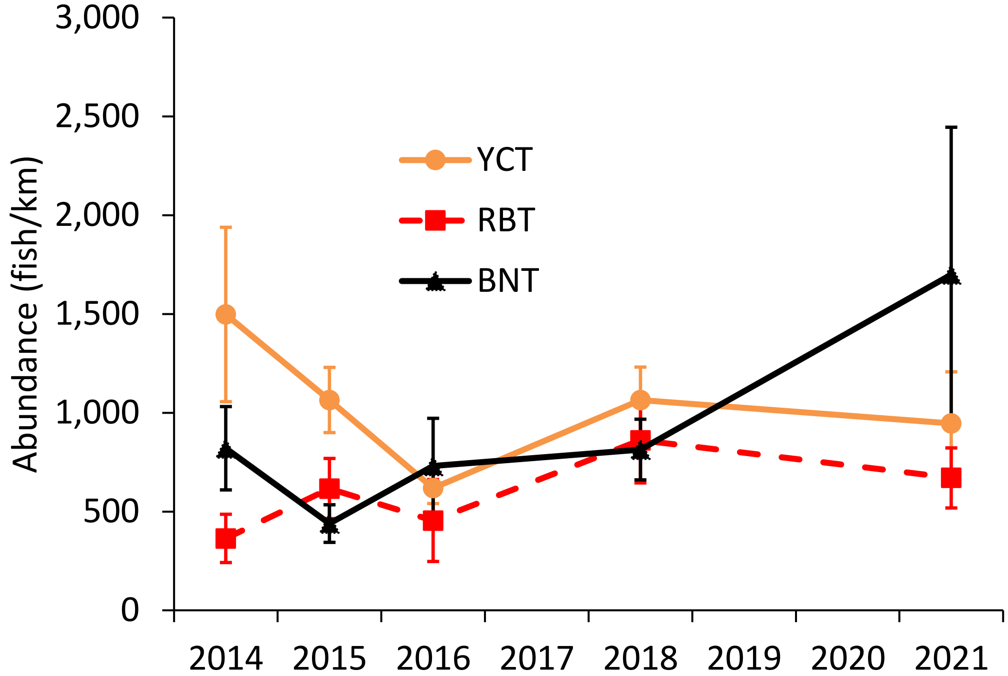

Figure 3. Abundance estimates (fish/km) and 95% confidence intervals for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) at the Lufkin Bottom monitoring reach on the South Fork Snake River from 2014 through 2021.

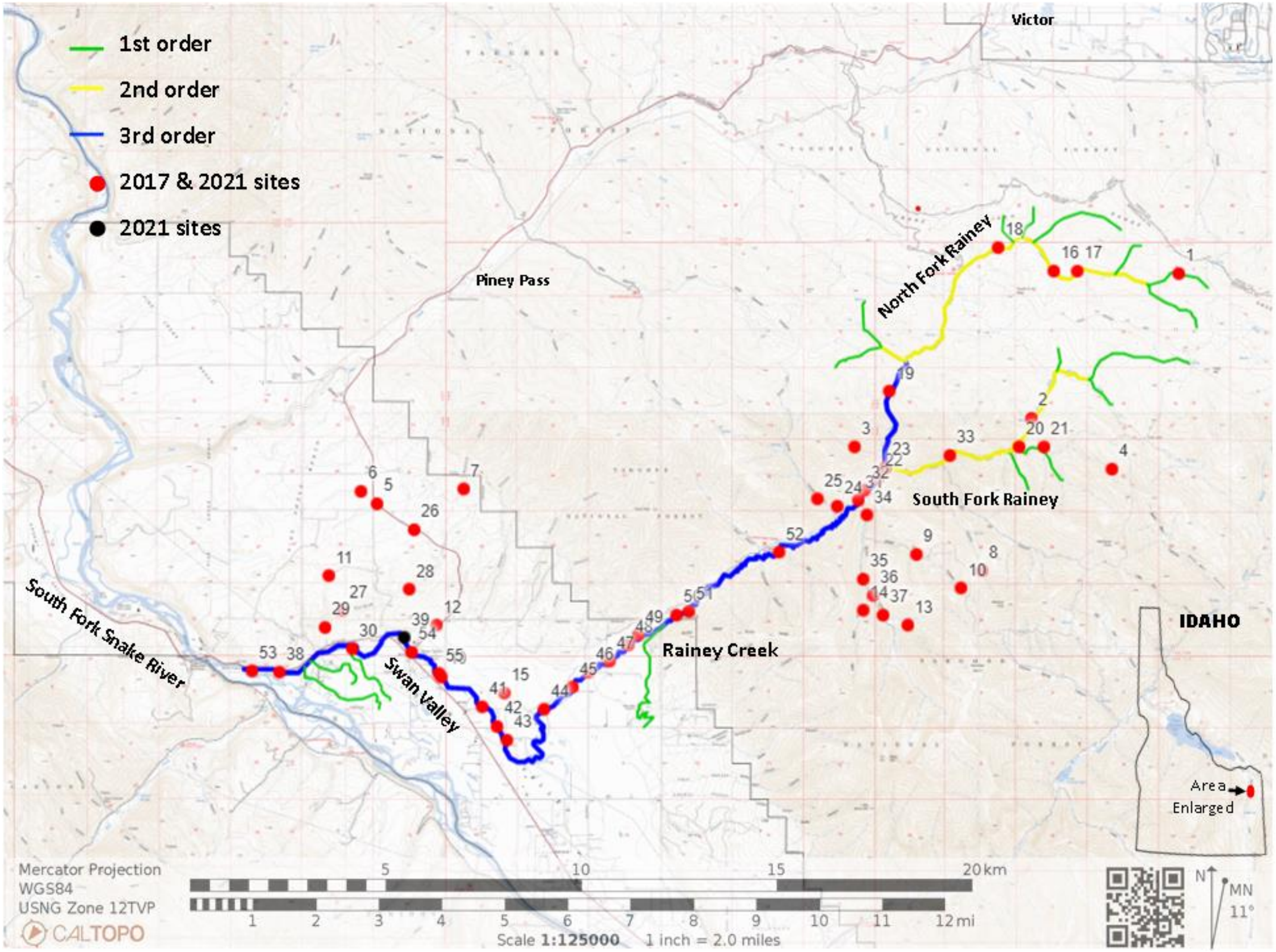

Figure 4. Stream-order designations and survey sites for the 2021 Pine Creek drainage-wide surveys of Yellowstone Cutthroat Trout abundance and distribution.

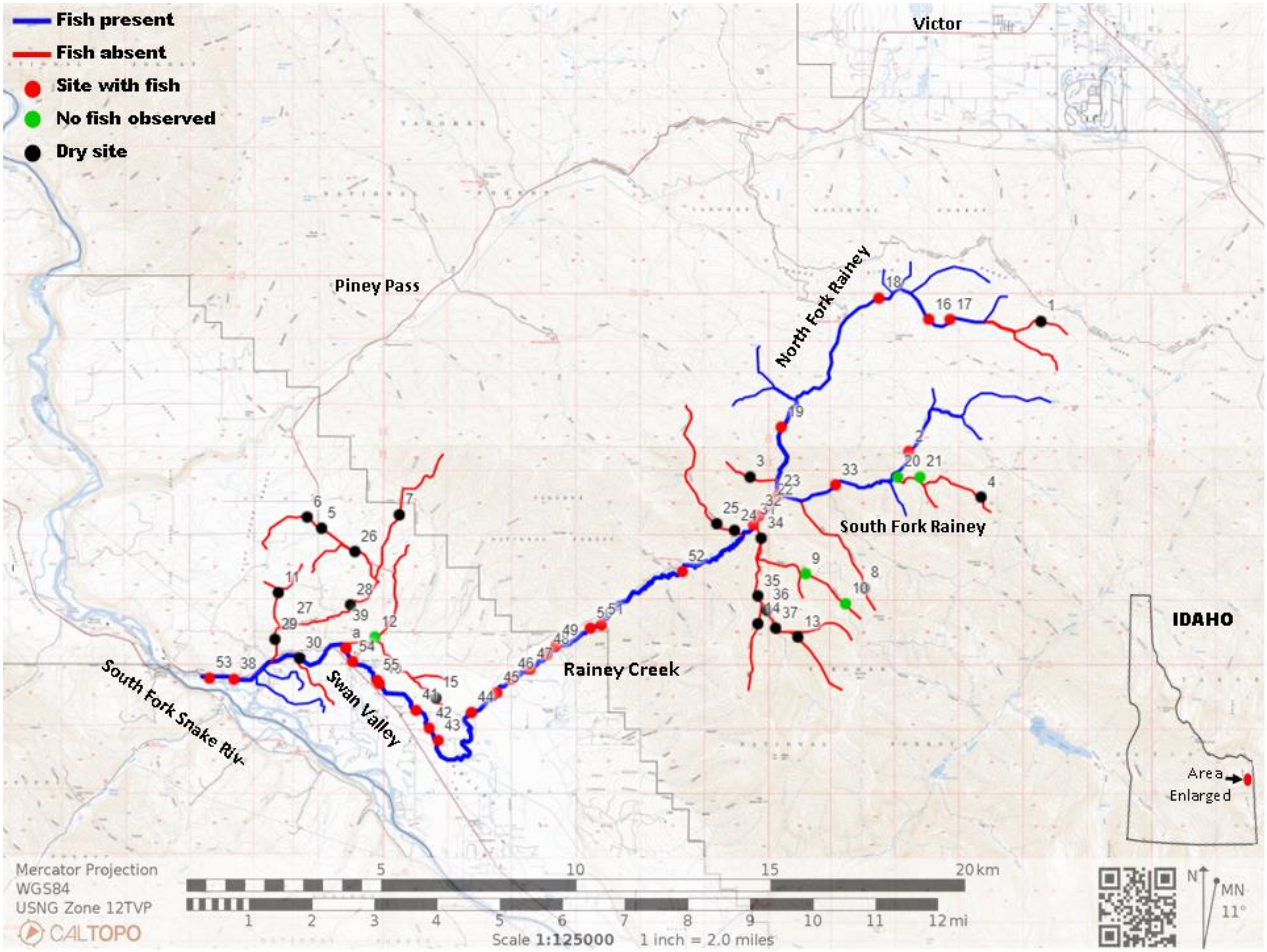

Figure 5. Yellowstone Cutthroat Trout distribution at sites surveyed for the 2021 Rainey Creek drainage-wide survey.

HENRYS FORK SNAKE RIVER

ABSTRACT

We used boat mounted electrofishing equipment to assess fish populations in the Macks Inn, Box Canyon, Vernon, and Chester reaches of the Henrys Fork Snake River during May 2021. In the Macks Inn reach, we estimated 1,246 trout/km ± 656 (95% CI), which included Rainbow Trout *Oncorhynchus mykiss* (752 fish/km ± 459; RBT) and Brook Trout *Salvelinus fontinalis* (571 fish/km ± 533, BKT), and the total trout estimate increased from the 2016 estimate (i.e., 338 fish/km ± 323). We estimated Mountain Whitefish *Prosopium williamsoni* (MWF) abundance in the Macks Inn reach at 1,085 fish/km ± 358, which decreased from our 2007 estimate (1,132 fish/km ± 306). In the Box Canyon reach, RBT abundance (2,616 fish/km ± 603) was less than the 2019 estimate (3,061 fish/km ± 237), but greater than the average of the previous 24 surveys (i.e., 1,929 fish/km). We estimated 845 fish/km ± 354 for MWF, which was greater than our 2019 estimate of 779 fish/km ± 466. In the Vernon reach, we estimated 885 trout/km ± 271, which included RBT (803 fish/km ± 438) and Brown Trout *Salmo trutta* (265 fish/km ± 87; BNT). Trout abundances for all species increased since our 2018 estimates; which were 678 trout/km ± 119 for all trout, 503 trout/km ± 94 for RBT, and 185 trout/km ± 42 for BNT. Mountain Whitefish abundance (44 fish/km ± 28) declined since our 2016 estimate of 116 fish/km ± 51. In the Chester reach, total trout abundance (628 trout/km ± 129) was less than our 2018 estimate of 699 trout/km ± 120. Similarly, RBT (427 fish/km ± 214) and BNT (275 fish/km ±68) abundances were less than in 2018; 529 ± 77 and 250 ± 33, respectively. Mountain Whitefish abundance (90 fish/km ± 81) was also less than our 2018 estimate (121 fish/km ± 82).

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INTRODUCTION

The Henrys Fork Snake River (HFSR) is a popular fishery that attracts anglers from throughout the nation and across the globe. The Henrys Fork Snake River forms at the confluence of Big Springs Creek and the Henrys Lake Outlet, and flows approximately 25 km before reaching Island Park Dam. Downstream of Island Park Dam, the HFSR flows approximately 147 km and through two smaller dams and four irrigation check dams before joining the South Fork Snake River to form the Snake River. The HFSR upstream of Island Park Reservoir is primarily supported by stocking, but it also includes wild trout. The fishery is also supported by trout that migrate out of Henrys Lake or Island Park Reservoir. Management of the HFSR downstream of Island Park Dam emphasizes wild, natural populations without hatchery supplementation. The HFSR downstream of Island Park Dam, particularly Box Canyon, Harriman Ranch, and Pinehaven reaches, support world famous wild Rainbow Trout *Oncorhynchus mykiss* (RBT) fisheries. Downstream of Harriman Ranch, the Henrys Fork flows over Mesa Falls and is joined by Warm River before it is impounded by Ashton Dam. Brown Trout *Salmo trutta* (BNT) are present in the Henrys Fork downstream of Mesa Falls, and densities increase in downstream reaches. Eventually, BNT dominate the species composition (>80%) near the town of St. Anthony and downstream. Mountain Whitefish *Prosopium williamsoni* (MWF) are present throughout all reaches of the HFSR,Yellowstone Cutthroat Trout *O. clarkii bouvieri* and Yellowstone Cutthroat Trout *Ocorhynchus clarkii bouvieri x* Rainbow Trout *O. mykiss* (hybrids) are present in some reaches predominately upstream of Mesa Falls and to a greater extent upstream of Island Park Reservoir. Brook Trout *Salvelinus fontinalis* (BKT) are present upstream of Island Park Reservoir, and they have also been sampled in the Box Canyon reach. Multiple species of nongame fishes have been sampled throughout the HFSR; including Mottled Sculpin *Cottus bairdii*, Paiute Sculpin *C. beldingii*, Utah Sucker *Catostomus ardens*, Bluehead Sucker *C. discobolus*, Longnose Dace *Rhinichthys cataractae*, Speckled Dace *R. osculus*, Redside Shiner *Richardsonius balteatus*, Utah Chub *Gila atraria*, and Least Chub *Lotichthys phlegethontis* have recently been identified in the watershed.

Previous research has emphasized the importance of winter river flows to the survival of age-0 RBT in the Box Canyon reach (Garren et al. 2006a; Mitro 1999). Higher winter flows (i.e., ≥ 500 cfs) in this reach results in significantly higher overwinter survival of juvenile trout and subsequent recruitment to the fishery downstream of Island Park Reservoir. Implementation of a congressionally-mandated Drought Management Plan has improved communications among interested parties and planning regarding winter discharges. We will continue to work cooperatively with stakeholders of this group to maximize wild trout survival based on timing and magnitude of winter releases from Island Park Dam.

STUDY SITE

In May 2021, we sampled four standardized reaches of the HFSR (Figure 6). The Macks Inn reach is sampled on an opportunistic schedule, which occurs roughly every five years. The reach begins at the confluence of the Henrys Lake outlet and Big Springs, then extends downstream about 2.5 km. The Box Canyon reach is sampled annually as part of our long-term monitoring program for the HFSR. The Box Canyon reach starts near Island Park Dam at the confluence with the Buffalo River and extends downstream 3.7 km to a large pool. Due to low flows in May 2021, our sampling reach was shortened to about 3.2 km because we could not float several sections of the reach because of low water. The Vernon and Chester reaches are sampled on a triennial basis. The Vernon reach begins at the Vernon boat ramp and extends downstream approximately 4.4 km to the beginning of the Chester backwaters. The Chester reach begins at the Chester boat ramp and extends downstream approximately 5.1 km to the island upstream of the Fun Farm boat ramp.

OBJECTIVES

Obtain current information on fish population characteristics by estimating abundance, species composition, and trends to inform fishery management decisions on the HFSR.

METHODS

Population Monitoring

We used boat-mounted electrofishing units to sample all reaches. We used two electrofishing rafts and one chase boat, we used a chase boat to expedite fish processing and decrease mortality, in the Macks Inn reach in which we marked fish on May 12 and recaptured fish on May 19. In the Box Canyon, Vernon, and Chester reaches, we used two electrofishing rafts and one electrofishing drift boat followed by a chase boat to process fish. We marked fish on May 18 and recaptured fish on May 20 in Box Canyon; marked fish on May 10 and recaptured on May 13 in the Vernon reach; and marked fish on May 11 and recaptured on May 13 in the Chester reach. One pass was conducted for each run (marking and recapture) in all reaches.

We recorded water temperature (°C) and conductivity (µS/cm) prior to active electrofishing using a handheld probe. We used pulsed direct current power provided by a 5000-W generator and standardized to 2,750-3,250-W based on water conductivity (Miranda 2009). We applied electricity to the water using an Infinity model electrofisher (Midwest Lake Management, Inc., Polo, Missouri). Electrofishing began at the uppermost point of the sampling reach and proceeded in a downstream direction. One netter positioned at the bow of each watercraft used a 2.4-m long dip net with 6-mm bar knotless mesh. The netter attempted to net all sport fish and place them into a live well located in the raft. For all sport fish captured, we identified species, measured total length (TL) to the nearest mm, and marked fish ≥ 150 mm TL with a hole punch in the caudal fin prior to release. This mark was used to identify previously captured fish during the recapture run.

We estimated abundance for all trout combined, and separately for RBT, BNT, and MWF ≥ 150 mm using a Petersen estimator with a Chapman modification using the mrClosed() function in the FSA package in program R (R Core Team 2021). We adjusted for size selectivity by distributing the catch into size classes to fit the recapture data, and then we calculated 95% CI for the abundance estimates. We evaluated species composition in each reach and catch rates (CPUE) for each species on the marking run as the number of fish caught per minute of electrofishing. We estimated capture efficiency as the number of marked fish that were recaptured divided by the total number of marked fish that were available for capture, multiplied by 100 (Chiaramonte et al. 2020). After calculating abundance estimates, we estimated trends in population growth in each reach using an exponential model and the intrinsic rate of population change (r) as described by Maxell (1999) using $\alpha = 0.10$. Proportional size distribution (PSD) was calculated as the number of fish ≥ 300 mm divided by the number of fish ≥ 200 mm multiplied by 100. Similarly, relative stock densities (RSD-400 and RSD-500) used the same formula, with the numerator replaced by the number of fish ≥ 400 and 500 mm (Anderson and Neumann 1996; Neumann et al. 2012).

In the Box Canyon reach, all RBT mortalities were kept and sagittal otoliths were extracted for assessing age, growth, natural mortality, and survival estimates. We recorded TL and weight (g) of all mortalities. We extracted otoliths from all RBT mortalities, then otoliths were sectioned otoliths across the transverse plain following standard techniques (Koch and Quist 2007; Long and Grabowski 2017), and we used a compound microscope in conjunction with imaging software (LAS 4.6; Leica; Buffalo Grove, IL) to take digital images of each otolith. We used ImageJ 1.53e (National Institutes of Health, USA) software to measure annuli growth increments, and then we estimated back-calculated length-at-age using the Dahl-Lea method (Shoup and Michaletz 2017). Growth was evaluated with a von Bertalanffy growth model (Ogle et al. 2017). We built an age-length key from our subsample of RBT and applied the key to our unaged sample. Then, we used a Chapman-Robson estimator (Chapman and Robson 1960) to estimate the survival rate (*S*), instantaneous mortality rate (*Z*), and total annual mortality (*A*; Ricker 1975). All analyses were conducted using the FSA package (Ogle et al. 2021) in program R (R Core Team 2021). Fish condition was evaluated by calculating relative weight (W_r) and plotting against TL. We used the standard weight (W_s) equation

$$log_{10}(W_s) = -5.023 + 3.024 * log_{10}(L),$$

for RBT (Simpkins and Hubert 1996) where *L* is the total length, then used the equation

$$W_r = \left(\frac{W}{W_s}\right) * 100,$$

where W_r is relative weight and *W* is measured weight. We used the same equations for MWF (i.e., intercept = -5.086 and slope = 3.036; Rogers et al. 1996) and BNT (i.e., intercept = -4.867 and slope = 2.960; Milewski and Brown 1994).

We also evaluated the effectiveness of using the mean (i.e., ratio of means) winter (Dec. 1–Feb. 28) discharge (cubic feet per second [cfs]) from Island Park Dam (USGS gage #13042500) during the first winter of age-0 RBT to predict their abundance at age 2 using linear regression as described by Garren et al (2006a). We log-transformed observed age-2 RBT abundance and mean winter flow data from the past 23 surveys to establish the following relationship:

log10 age-2 RBT abundance = 0.6191 × log10 winter streamflow at age 0 + 1.9086

Using this equation, we estimated the expected abundance of age-2 RBT in our 2021 population survey based on mean winter discharge measured during December 2019–February 2020. Winter streamflow data represented conditions experienced by age-0 trout during their first winter (i.e. the 2019 RBT year-class). To investigate this relationship, we estimated age-2 RBT abundance during the 2021 electrofishing surveys, which correlates to the TL (230–329 mm) of age-2 RBT in past surveys. Age-2 RBT were determined to be the first year-class fully recruited to the electrofishing gear (Garren 2006b). We then compared predicted and observed age-2 RBT abundance in Box Canyon to evaluate the ability of the equation to predict year-class strength based on winter streamflow. Data from 2019 was added to the streamflow vs. age-2 abundance regression model, which will continue to be used in recommending management of winter streamflow releases from Island Park Dam.

Fish Tagging

In addition to conducting population estimates in four reaches of the HFSR, we also conducted a “Tag You’re It” (Meyer et al. 2012; Meyer and Schill 2014) study in the Macks Inn reach on RBT that were stocked at Macks Inn from May through August 2021. American Falls Fish Hatchery reared all 36,336 catchable-sized, triploid RBT that were stocked in the Macks Inn reach of the HFSR in 2021. A proportion (i.e., 5%) of fish that were stocked were implanted with non-reward, T-bar anchor tags at the base of the dorsal fin according to standard methods (Dell 1968) prior to stocking. Prior to each stocking event, IDFG personnel tagged 5% of fish to be stocked at American Falls Fish Hatchery, then those fish were kept in a separate raceway, and added to the stocking tank on the day of stocking. Tagged and un-tagged fish were randomly distributed in the stocking tank. We tagged 1,816 catchable-sized RBT at American Falls Fish Hatchery with non-reward tags. In addition, we tagged a total of 75 of the hatchery RBT with $50 reward T-bar anchor tags. Anchor tags were printed with a unique identification number, phone number, and website address where anglers could report the tag using the “Tag You’re It” statewide tag reporting system (Meyer and Schill 2014). We used data obtained from reported tags to estimate exploitation, caught and released fish, and total angler use.

We used the angler reporting rate (*λ*) of 48.3% (Meyer et al 2012) for non-reward tagged fish and a reporting rate of 88.4% for $50 reward tagged fish to estimate exploitation. We estimated angler exploitation (*u'*) using the equation:

$$u' = \frac{u}{\lambda\,(1 - Tagl)(1 - Tagm)},$$

where *u was* the number of non-reward tagged fish that were reported as harvested divided by the total number of non-reward tagged fish stocked, $Tagl$ was the first year tag loss rate (i.e., 0.088), and $Tagm$ was the tagging mortality rate (i.e., 0.01). We used the tag loss and tagging mortality as reported by Meyer and Schill (2014). We also estimated angler use by modifying *u* to also include fish reported as caught and released. We queried the “Tag You’re It” database for tag return data in February 2022, so the tags were not at-large for an entire year.

RESULTS

Macks Inn

The total trout abundance (i.e., RBT and BKT) in the Macks Inn reach has increased since 2016. We observed large confidence bounds in our estimate due to recapturing a small number of marked fish. We sampled 235 RBT, 141 BKT, 1 YCT, and 572 MWF during 2 days of electrofishing in the Macks Inn reach. We estimated 1,246 trout/km (± 656; 95% CI), 755 RBT/km ± 459, 574 BKT/km ± 531, and 1,090 MWF/km ± 357 (Table 11). Species composition of trout was 62% RBT, 37% BKT, and 1% YCT. Of all RBT sampled, 3% (n = 8) were of hatchery-origin, which we determined based on the presence of fin erosion, and these fish were not included in our abundance calculations. Capture efficiency and CPUE were 6% and 0.80 fish/min for RBT, 5% and 0.26 fish/min for BKT, and 13% and 2.0 fish/min for MWF. The TL of RBT ranged from 66 mm to 561 mm (mean = 294 mm; SD = 129; Figure 7). The intrinsic rate of population growth for all trout in this reach has been stable over seven abundance estimates (r = 0.03, $F_{1,5}$ = 0.71, P = 0.44; Figure 8). The MWF population has also been stable (r = 0.027) over time (three years) in this reach ($F_{1,1}$ = 172.1, P = 0.048; Figure 9). We estimated the mean relative weights of RBT, BKT, and MWF as 81, 76, and 97 (Figure 10).

Anglers reported 8% of non-reward tags (i.e., 75 tags reported as fish released and 79 tags reported as fish harvested). For non-reward tagged fish, we estimated exploitation (u') to be 10% and total angler use at 18% (i.e., the number of tagged fish reported as released and the number of tagged fish reported as harvested combined). We received nine $50 reward tags, which was a reporting rate of 12% (i.e., six reported as harvested and three reported as released). We estimated exploitation of reward-tagged fish at 10% and total angler use of reward-tagged fish at 15%.

Box Canyon

We sampled 1,451 RBT, 14 BKT, and 443 MWF during two days of electrofishing in the Box Canyon reach. We estimated 2,616 RBT/km (± 603; 95% CI), 845 MWF/km ± 569, and we did not recapture a BKT to estimate their abundance in the reach. Species composition of trout was 99% RBT and 1% BKT. Capture efficiency and CPUE were 9% and 4.73 fish/min for RBT, and 10% and 1.14 fish/min for MWF. The TL of RBT ranged from 100 mm to 545 mm (mean = 325 mm; SD = 63; Table 12; Figure 11). Mountain Whitefish TL ranged from 126 to 502 mm (mean = 370 mm; SD = 61. We estimated the average W_r for RBT to be 93 with a declining W_r as TL increased. The average W_r of MWF was 99 with an increasing relative weight with an increasing TL. The intrinsic rate of population growth for RBT in the Box Canyon reach has been stable and not increasing or decreasing significantly over time (i.e., 25 years; r = 0.01, $F_{1,23}$ = 0.87, P = 0.36; Figure 12). In addition, the MWF population has also been stable over time (seven years; r = -0.02, $F_{1,5}$ = 1.76, P = 0.24; Figure 13).

Despite using a chase boat to expedite fish processing, we observed 47 RBT mortalities over 2 days of electrofishing. Because we used these mortalities opportunistically and we did not sample 10 RBT per 10-mm length group, as would usually be done for an age and growth study, we sampled an unequal number of RBT per length bin. Using the von Bertalanffy growth function, we estimated L_∞ = 466.7, K = 0.29, and t_0 = -0.56 (Figure 14). We estimated S at 48%, Z at 0.73, and A at 52% (Figure 15).

We estimated the expected abundance of age-2 RBT (i.e., 230–329 mm) in our 2021 population estimate based on mean winter streamflow measured during December 2019–February 2020 (i.e., 560 cfs) to be 4,074 age-2 RBT in the population (Figure 16). Using the size selective model in R to estimate abundance for age-2 RBT, we estimated age-2 RBT abundance to be 4,109 (± 1,635; 95% CI).

Vernon

We sampled 352 RBT, 307 BNT, and 52 MWF during 2 days of electrofishing in the Vernon reach of the HFSR. We estimated 885 trout/km (± 271; 95% CI), and separated by species that was 803 RBT/km ± 438, 265 BNT/km ± 87, and 44 MWF/km ± 28. Species composition of all trout in the catch including all length classes was 55% RBT and 45% BNT. Capture efficiency and CPUE were 12% and 1.53 fish/min for all trout combined, 7% and 0.70 fish/min for RBT, 17% and 0.83 fish/min for BNT, and 18% and 0.10 fish/min for MWF. The mean TL of trout in the Vernon reach were 386 mm (± 107) for RBT, 363 mm (± 137) for BNT, and 458 mm (± 61) for MWF (Figure 17). Although total trout abundance has been increasing over the last two surveys (Figure 18), there has not been a significant increase in the intrinsic rate of population growth (r = 0.02, $F_{1,6}$ = 0.93, P = 0.37) over time (eight years). However, BNT abundance has been significantly increasing over time (eight years; r = 0.13, $F_{1,6}$ = 7.80, P = 0.03; Figure 19), but RBT abundance has been stable over time (nine years; r = 0.00, $F_{1,7}$ = 0.01, P = 0.94). We estimated the intrinsic rate of population growth for MWF to be significantly declining (r = -0.13, $F_{1,4}$ = 4.50, P = 0.10; Figure 20). We estimated the mean W_r of RBT, BNT, and MWF as 101, 97, and 108.

Chester

We sampled 334 RBT, 450 BNT, and 69 MWF during two days of electrofishing in the Chester reach of the HFSR. We estimated 606 trout/km (± 153; 95% CI), 427 RBT/km ± 222, 275 BNT/km ± 71, and 87 MWF/km ± 81. Species composition of trout was 43% RBT and 57% BNT. Capture efficiency and CPUE were 9% and 2.60 fish/min for all trout combined, 5% and 1.09 fish/min for RBT, 12% and 1.50 fish/min for BNT, and 5% and 0.20 fish/min for MWF. The average TL of RBT in the Chester reach was 345 mm ± 101, 356 mm ± 112 for BNT, and 395 mm ± 105 for MWF (Figure 21). The intrinsic rate of population growth for all trout in the Chester reach has been stable (r = 0.02, $F_{1,5}$ = 2.76, P = 0.16; Figure 22). Rainbow Trout intrinsic rate of population growth has been stable over time (r = -0.02, $F_{1,5}$ = 0.50, P = 0.51; Figure 23), while BNT population growth has been significantly increasing over time (r = 0.09, $F_{1,5}$ = 15.84, P = 0.01). We estimated the intrinsic rate of population growth for MWF to be significantly declining at (r = -0.15, $F_{1,4}$ = 15.80, P = 0.02; Figure 24). We estimated the mean relative weights of RBT, BNT, and MWF as 97, 98, and 109.

DISCUSSION

Macks Inn

The timing of sampling for this reach may skew the number of RBT that are actually residing in the Macks Inn reach of the HFSR, but it does provide us with data about the migratory (e.g., adfluvial) component of the population. This reach of the HFSR is known as an important spawning reach for RBT in the upper HFSR and large numbers of RBT migrate through this reach to spawn in Big Springs. Therefore, we presume that the low number of recaptures was due to the migratory behavior of RBT in this reach during this timeframe. Shortening the number of days between the marking and recapturing runs is warranted to try and capture a better estimate with less variability. In addition, sampling this reach later in the summer may produce a better estimate for the number of trout in the reach, rather than trying to estimate the number of trout in the reach during their spawning migration. However, due to the increase in popularity for recreational floating in this reach (Van Kirk et al. 2019), May is likely the best month to sample the Macks Inn section of the HFSR and it provides us with information and continued monitoring of the migratory component of the population.

Mountain Whitefish abundance is a concern we've heard from some of our anglers of this reach. Mountain Whitefish body condition was good in this reach and their abundance was near the same level as trout, which indicates that environmental conditions can support thousands of fish per km. We observed multiple size classes of MWF, as well as RBT, BKT, suggesting that recruitment for all salmonid species was occurring in this reach and may be one of the reasons why our analysis indicates the abundance of the MWF population has been stable over time. Furthermore, the majority (97%) of RBT that we sampled in the reach were observed to be wild-origin fish.

The Macks Inn reach of the HFSR is open to general trout regulations (6 trout per day) from Memorial Day weekend to January 1, but only two Cutthroat Trout may be harvested and they cannot be less than 16 inches. Our results suggest angler harvest is not influencing trout abundance in the Macks Inn reach, but stocked, catchable-sized RBT are providing some opportunity for anglers. Using the "Tag You're It" angler reporting system, we estimated exploitation in this reach to be 10% and total angler use at 18% where we would anticipate the population could withstand an exploitation rate of 40-50% before a negative impact to population abundance would be expected, based on the annual mortality (*A*) of RBT we estimated for the Box Canyon reach at 52%. Our harvest estimate only includes hatchery-origin RBT, but if similar harvest levels exist for wild fish we do not expect that level of harvest to have a population-level effect. Exploitation in this reach was within the range of what was estimated for hatchery catchable RBT in the North Fork Boise River (i.e., 11.2% and 20.1%; Branigan 2018) and was similar to what we estimated in the Copper Basin area of the Big Lost River (10%; Heckel et al. 2020). At both of those fisheries, we do not consider exploitation as a significant factor influencing abundance because natural mortality was significantly higher with those populations. The increased interest in the Macks Inn reach by recreational floaters may lead to an increase in fishing pressure, which should be investigated by a creel survey to learn more about angler effort and harvest.

To mitigate for the potential increase in harvest and for continued harvest at the current level, we should continue supplementing this reach with hatchery-origin catchable RBT to continue providing anglers with opportunities to harvest trout without having a population-level effect on the wild RBT population. If we continue to hear concerns from anglers about harvest of wild trout, then we can replicate the "Tag You're It" study on wild fish and compare the estimate to the hatchery fish exploitation estimate. But, based off of the proportion of wild, presumably migratory RBT, in the abundance estimate wild fish are likely migrating out of the reach where harvest is focused. Furthermore, at a 10% rate of exploitation of hatchery-catchable RBT we would not expect a population-level effect through additive mortality if that same rate of exploitation were applied to wild RBT (Allen et al. 1998).

Box Canyon

We currently have a robust population of RBT in the Box Canyon reach with two strong year-classes, and we observed mortality rates that are lower than previous estimates. The current abundance estimate was 30% greater than the 24-year average. We also estimated that the age-4 year-class was the strongest in the population suggesting that the RBT population was robust following several good water years, which led to winter flows above 500 cfs during their first winter. The current estimate was the fourth highest RBT abundance in our 25th population survey in the Box Canyon reach since 1994. The mean TL of RBT in the current estimate was 325 mm, which was 64 mm longer than the mean TL in our last population estimate in 2019, and based off of W_r, their body condition was good. This suggests that trout density is not limiting their body condition and that environmental factors (e.g., streamflow, water temperatures, prey availability) can support a high density of RBT. Mortality does begin to increase after age-4, but the strong year-class of 2017 should provide anglers with more opportunities to catch larger fish in the Box Canyon over the next two years. Our mortality estimate (Z = 52%) from ages 4 to 7 was lower than the last estimate in 2014 (Z = 74%; Flinders et al. 2016) from ages 2 to 4. Although we did not take a subsample of fish from each length bin (e.g., 10 fish per cm length group), our mortality estimate does represent the current population dynamics, which was supported by the large number of RBT represented in our length frequency and von Bertalanffy growth model from 290-400 mm, i.e., RBT estimated to be age 4.

Winter streamflows continue to be an important predictor for RBT abundance in the Box Canyon reach of the HFSR (Garren et al. 2006a). Fausch et al. (2001) found Rainbow Trout recruitment was higher in tailwaters exhibiting high winter and/or low spring flows. Observed age-2 RBT (4,109 RBT ± 1,635) abundance was very similar to what we predicted (4,074 RBT) using the regression formula from the flow model. In previous surveys in Box Canyon (Heckel et al. 2020; Heckel et al. 2021), the age-2 RBT year-class generally comprised the largest proportion of the abundance estimate. However, the age-2 year-class was not the most abundant age class estimated in the population. This could be an artifact of overlap among length-at-ages in the population in applying our age length key to the unaged sample because age-4 was most represented in the ages we estimated from otoliths. But, this could also be evidence that there was a high abundance of older (age-4+) fish that make up more of the population than younger (age 2 and 3) fish. If this was the case, it may be a couple more years until we see this pattern change to more young fish in the population and fewer old fish. For example, Carline (2006) found that the interactive effects of streamflow and trout density accounted for most of the annual variation in mortality, recruitment, and growth in regulating an unexploited Brown Trout population. It appears that streamflow and trout abundance may also explain changes in the Box Canyon wild RBT population.

Mountain Whitefish abundance has been stable since 2019, and there has not been a significant change in abundance since a large decline from 1991 to 2002. Although MWF are often not the targeted species by anglers in the Box Canyon reach, they do offer additional opportunities for anglers to “save the day” when RBT catch rates are low, and based off of W_r their condition was good. Our MWF abundance estimates have been conducted opportunistically, but they should be conducted on a biennial basis to monitor MWF population health, since their abundance is in decline in other sections of the Henrys Fork and MWF abundance has been used as an indicator to describe the health of a fishery (*see* Vernon and Chester results section of this report; McPhail and Troffe 1998.; Meyer et al. 2009). There was a decline in the MWF population sometime between 1992 and 2001; however, we did not conduct a MWF population estimate during this time frame, so we cannot make inferences into why there was a decline. Since that decline, however, the MWF population abundance has remained stable since 2002.

Vernon

One of the management objectives for this reach of the HFSR is to provide a wild trout fishery with quality-length trout (IDFG 2019). Proportionally, there was a high abundance of trout greater than 400 mm in this reach of the HFSR, which provides anglers plenty of opportunities at quality-length fish. Additionally, trout in this reach have an average W_r near 100 indicating that they are in good condition as well. For both RBT and BNT, we estimated three distinct size classes in the length frequency distribution with strong length classes for fish between 130 and 180 mm and for fish ≥ 370 mm. The strong younger year-class suggests that juvenile recruitment was occurring and should continue to contribute to this wild trout population.

Brown Trout abundance continues to increase in relation to RBT in the Vernon reach, which is a similar pattern that we have been observing farther downstream in the St. Anthony reach of the HFSR (Vincent et al. *in press*). Species composition remains predominately RBT (i.e., 55%), but BNT abundance has been steadily increasing over time. As summer water temperatures are forecasted to increase (Isaak et al. 2016), BNT abundance will likely also increase because they have a higher thermal tolerance than RBT (Carline and Machung 2001; Wehrly et al. 2007). Both species were represented in nearly equal proportions of the trout community, which diversifies the fishery for anglers. However, some anglers may favor RBT over BNT, which could complicate managing this fishery. If BNT are becoming more abundant because of their higher thermal tolerance, then they will be more suitable to warming stream conditions and there may not be much that we can do for RBT, but altering water management could help mitigate warm stream temperature effects with changing flow regimes during critical RBT life history periods. During the next annual creel survey on the HFSR, it would be advisable to investigate angler opinion on the shifting trout composition.

Although trout abundance was high, MWF abundance was the lowest point estimate that we have on record since 2002. The 2007 estimate was an outlier with very wide confidence bounds and it is unclear how much effort was focused on collecting MWF during electrofishing. Wider rivers and streams generally have MWF present at higher abundances (Meyer et al. 2009), but our 2021 data suggests the opposite in that a wide HFSR reach has lower MWF abundance than narrower reaches upstream. There are other environmental factors that are limiting MWF abundance in the Vernon reach because we saw an increase in MWF abundance in the St. Anthony reach in 2020 (Vincent et al. *in press*), which is farther downstream than the Vernon reach. A lack of good overwintering habitat and water conditions when MWF spawn and a lower abundance of pools may be affecting their overall abundance in the Vernon reach. In addition, the St. Anthony reach has groundwater seeps and springs that contribute to its streamflow and stable water temperature, whereas the Vernon reach does not. Higher mean summer water temperatures could also be affecting the health of the MWF population in this reach. Each abundance estimate conducted in the Vernon reach should include trout and MWF so that we have consistent data collection to more closely monitor both populations and allow more insight into how environmental factors are affecting the population.

Chester

The length-frequency distribution of trout sampled in the Chester reach indicates that we should anticipate several strong year-classes and consistent recruitment to the fishery. In addition, there was not a statistically significant population increase for all trout combined or RBT, but BNT abundance has been significantly increasing over time. In each length frequency group, BNT average TL was longer than RBT, especially of trout measuring 100 to 180 mm. Therefore, BNT may have a size advantage over RBT. In this size class, BNT are on average longer than RBT, which could be advantageous going into winter if BNT carry this size advantage throughout the summer and fall. Differences in the timing of fry emergence has been shown to segregate populations such that larger individuals move into progressively faster and deeper water and become able to compete with larger individuals in the population (Hearn 1987). Fish in a population that can occupy habitat with more and larger prey will grow faster and be more competitive than smaller individuals. This factor, in addition to warmer average summer water temperatures, could contribute to increasing BNT composition in the Chester reach. Both species are in good condition with mean W_r near 100, suggesting that there are adequate prey resources for this density of trout in Chester.

Similar to the Vernon reach, MWF abundance has been significantly decreasing over time. Because these reaches are separated by Chester Dam and similar in geomorphology, the same factors that are affecting MWF abundance could be at play in both reaches. Farther downstream in the St. Anthony reach, MWF abundance was increasing (Vincent et al. *in press*), but there are more spring and groundwater seeps in that reach in addition to more pool habitat compared to the Chester reach. Temperature tolerances for MWF are much lower than those for RBT and BNT (Brinkman et al. 2013) and we are seeing an increasing abundance of BNT, which has a higher thermal tolerance than RBT. Warming water temperature could be a primary limiting factor for MWF abundance in both the Vernon and Chester reaches. It was apparent in the MWF length frequency distributions that there was low juvenile abundance in both reaches. Increasing water temperatures will have an effect on MWF egg survival, hatching success, and fry survival, so as mean water temperature increases MWF mortality will likely also increase leading to a population-level effect (Brinkman et al. 2013). Mountain Whitefish abundance should continue to be monitored every year that a trout abundance is conducted in this reach.

MANAGEMENT RECOMMENDATIONS

1. Continue, at a minimum, biennial population surveys in the Box Canyon reach to quantify population response to changes in the flow regime over time and to forecast expected abundance to anglers.

2. Work with the irrigation community and other agencies to continue recommending winter flows of 500 cfs out of Island Park Dam to benefit trout recruitment, stressing the importance of early winter streamflows (December, January and February) to benefit age-0 trout survival.

3. Conduct a formal age and growth study on RBT in Box Canyon that includes a full subsample of RBT from all length bins.

4. Continue triennial population estimates in the Vernon and Chester reaches to monitor trout and MWF population trends.

Table 11. Population demographics for Rainbow Trout (RBT), Brook Trout (BKT), Yellowstone Cutthroat Trout (YCT), Mountain Whitefish (MWF), and all trout combined at four index reaches in the Henry's Fork Snake River sampled in May, 2021.

Reach	Species	Abundance estimate (± 95% CI)	No./km (± 95% CI)	% Composition	Capture efficiency (%)	CPUE (fish/min)	Mean TL (± SD)
Macks Inn	RBT	1,812 (1,106)	755 (459)	62	6	0.80	294 (129)
	BKT	1,377 (1,279)	574 (531)	37	5	0.26	201 (78)
	YCT	--	--	<1	--	--	460
	All trout	2,991 (1,580)	1,246 (656)	--	6	1.07	--
	MWF	2,617 (860)	1,090 (357)	--	13	2.00	361 (91)
Box Canyon	RBT	8,417 (1,940)	2,616 (603)	99	9	4.73	325 (63)
	BKT	--	--	<1	0	0.03	197 (62)
	MWF	2,718 (1,138)	845 (569)	--	10	1.14	370 (61)
Vernon	RBT	3,058 (1,927)	803 (438)	55	7	0.70	386 (107)
	BNT	1,011 (382)	265 (87)	45	17	0.83	363 (137)
	All trout	3,373 (1,193)	885 (271)	--	12	1.53	--
	MWF	145 (122)	44 (28)	--	18	0.10	458 (61)
Chester	RBT	2,134 (1,071)	427 (222)	43	5	1.09	345 (101)
	BNT	1,377 (342)	275 (71)	57	12	1.50	356 (112)
	All trout	3,032 (737)	606 (153)	--	9	2.60	--
	MWF	436 (393)	87 (81)	--	5	0.20	395 (105)

Table 12. Mean total length (TL; mm)-at-age, estimated from sagittal otoliths, for Rainbow Trout sampled via raft electrofishing in the Box Canyon reach of the Henrys Fork Snake River in May 2021.

Age	*n*	Mean TL (mm)	SD
1	89	169	27.9
2	102	259	25.5
3	266	290	33.6
4	438	332	28.1
5	371	356	19.4
6	188	374	27.4
7	88	424	34.8

Figure 6. Map of the Henrys Fork Snake River watershed illustrating the approximate locations of Macks Inn, Box Canyon, Vernon, and Chester survey reaches.

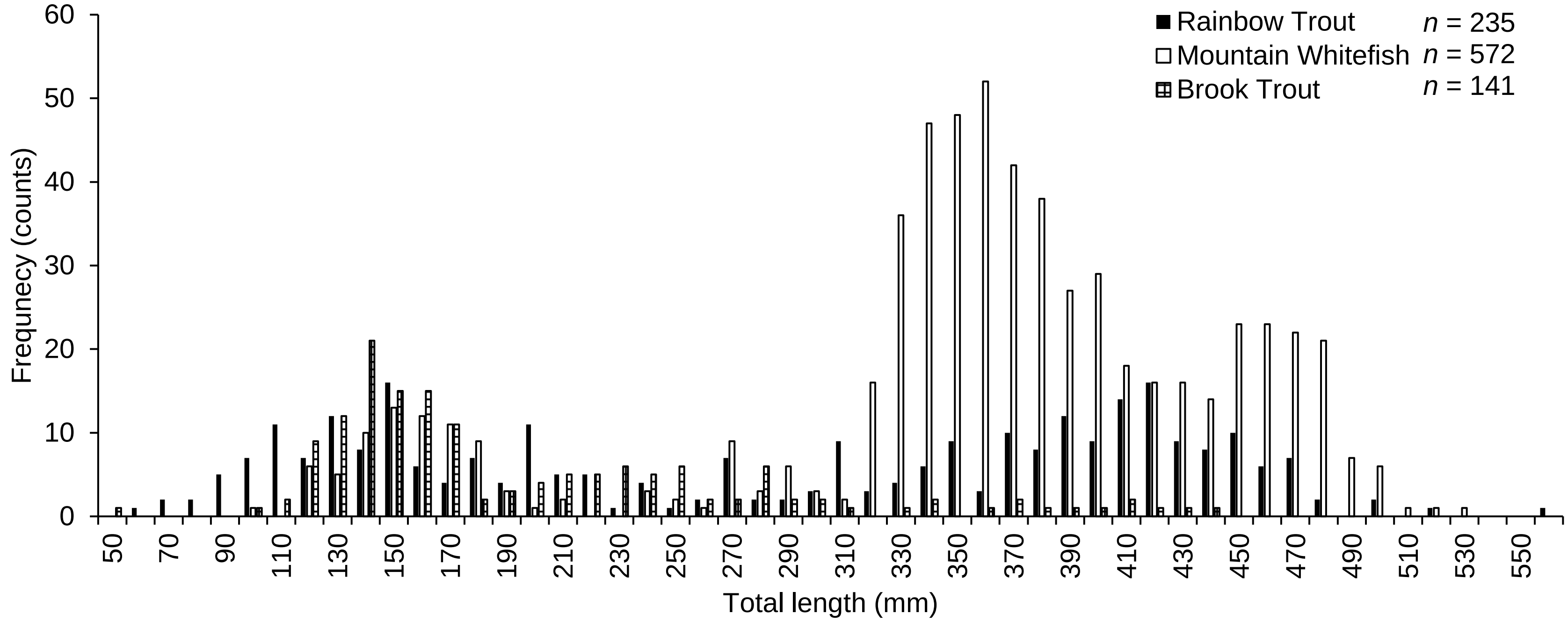

Figure 7. The length-frequency distributions of Rainbow Trout, Mountain Whitefish, and Brook Trout captured via raft electrofishing in the Macks Inn reach of the Henrys Fork Snake River 2021.

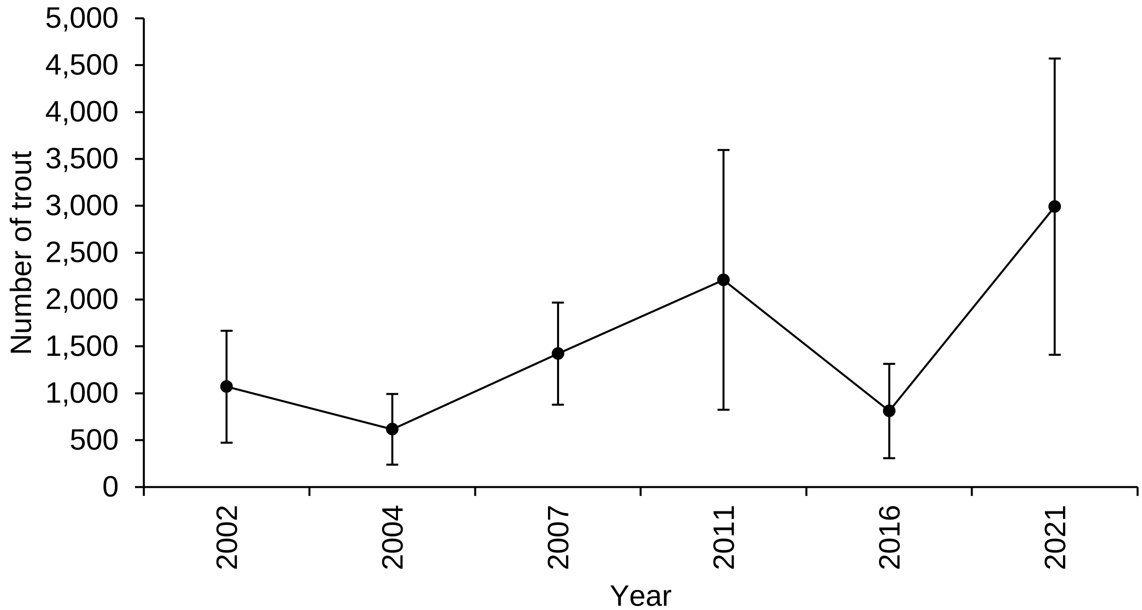

Figure 8. Modified Petersen abundance estimates for all trout combined in the Macks Inn reach of the Henrys Fork Snake River from 2002 to 2021. Error bars represent 95% confidence intervals around the abundance estimate.

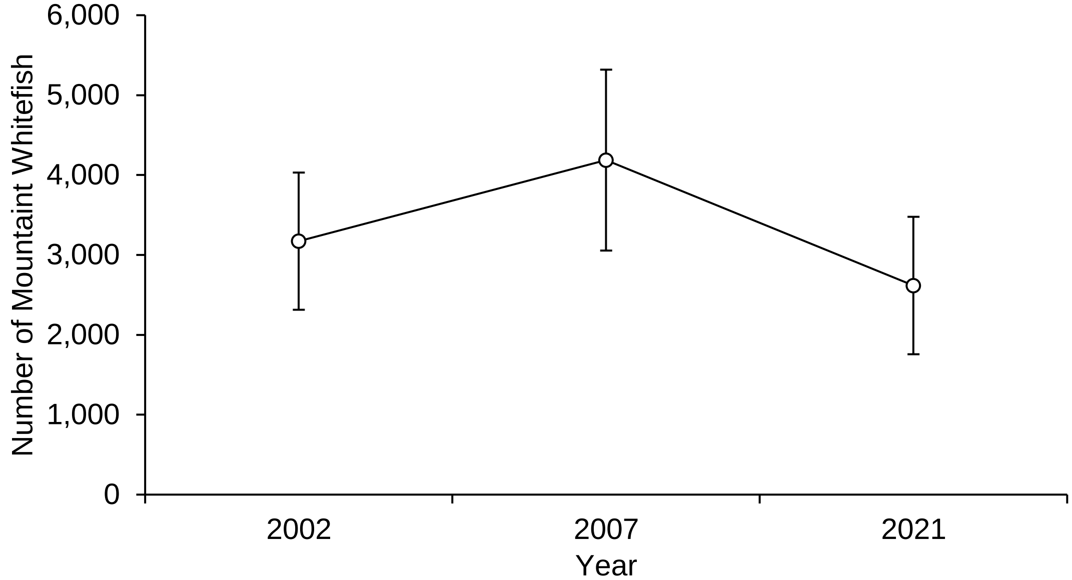

Figure 9. Modified Petersen abundance estimates for Mountain Whitefish in the Macks Inn reach of the Henrys Fork Snake River in 2002, 2007, and 2021. Error bars represent 95% confidence intervals around the abundance estimate.

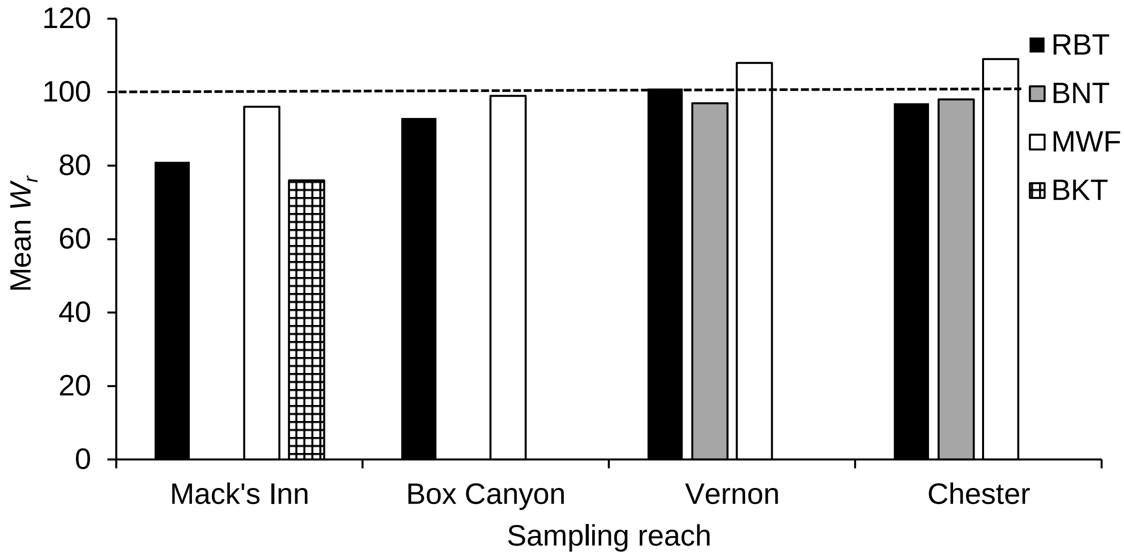

Figure 10. The mean relative weight (W_r) of Rainbow Trout (RBT), Brown Trout (BNT), Mountain Whitefish (MWF), and Brook Trout (BKT) in four reaches of the Henrys Fork Snake River sampled in May 2021. The dashed line designates a W_r of 100, which represents the 75th percentile of mean weights for a given length of the respective species across its range.

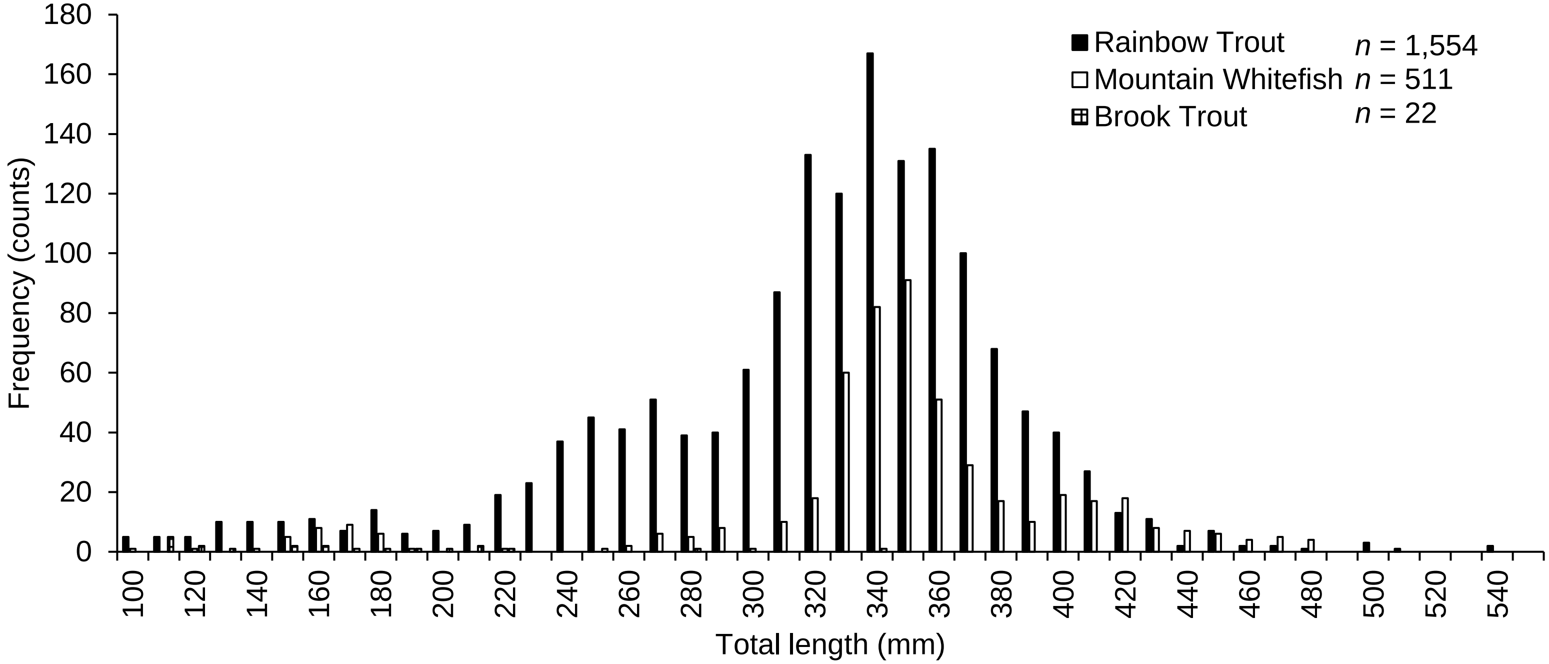

Figure 11. The length-frequency distributions of Rainbow Trout, Mountain Whitefish, and Brook Trout captured via raft electrofishing in the Box Canyon reach of the Henrys Fork Snake River in May 2021.

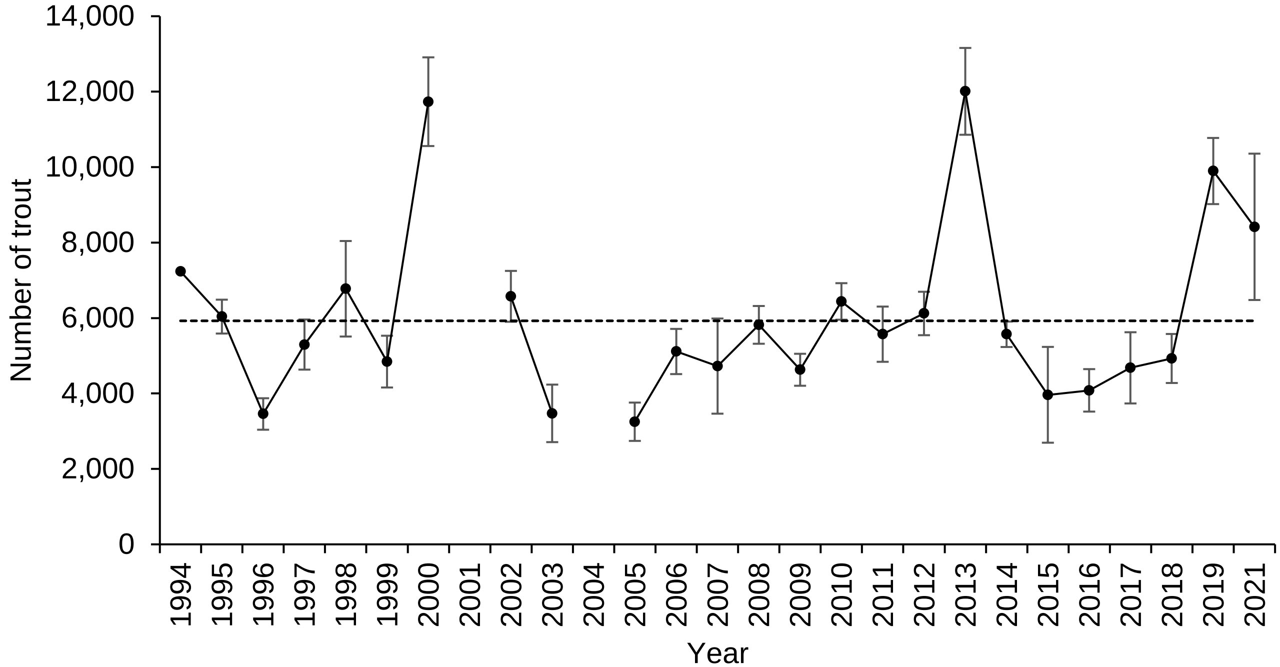

Figure 12. Modified Petersen abundance estimates for Rainbow Trout in the Box Canyon reach of the Henrys Fork Snake River in the spring from 1994 to 2021. Error bars represent 95% confidence intervals around the abundance estimate. The dashed line represents the mean abundance estimate (5,925 trout) from 1994-2019.

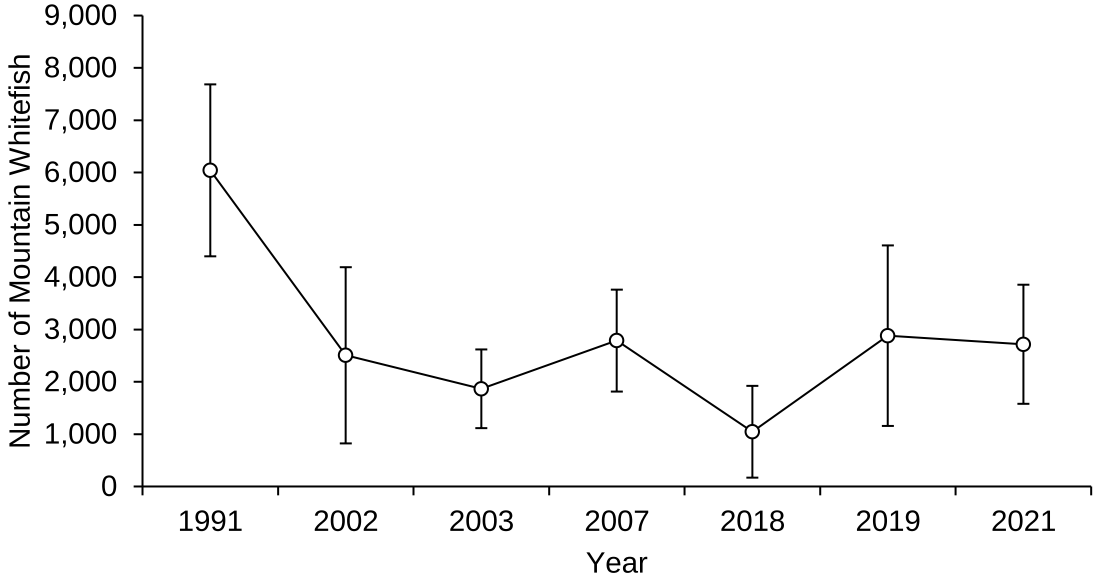

Figure 13. Modified Petersen abundance estimates for Mountain Whitefish in the Box Canyon reach of the Henrys Fork Snake River in the spring from 1991 to2021. Error bars represent 95% confidence intervals around the abundance estimate.

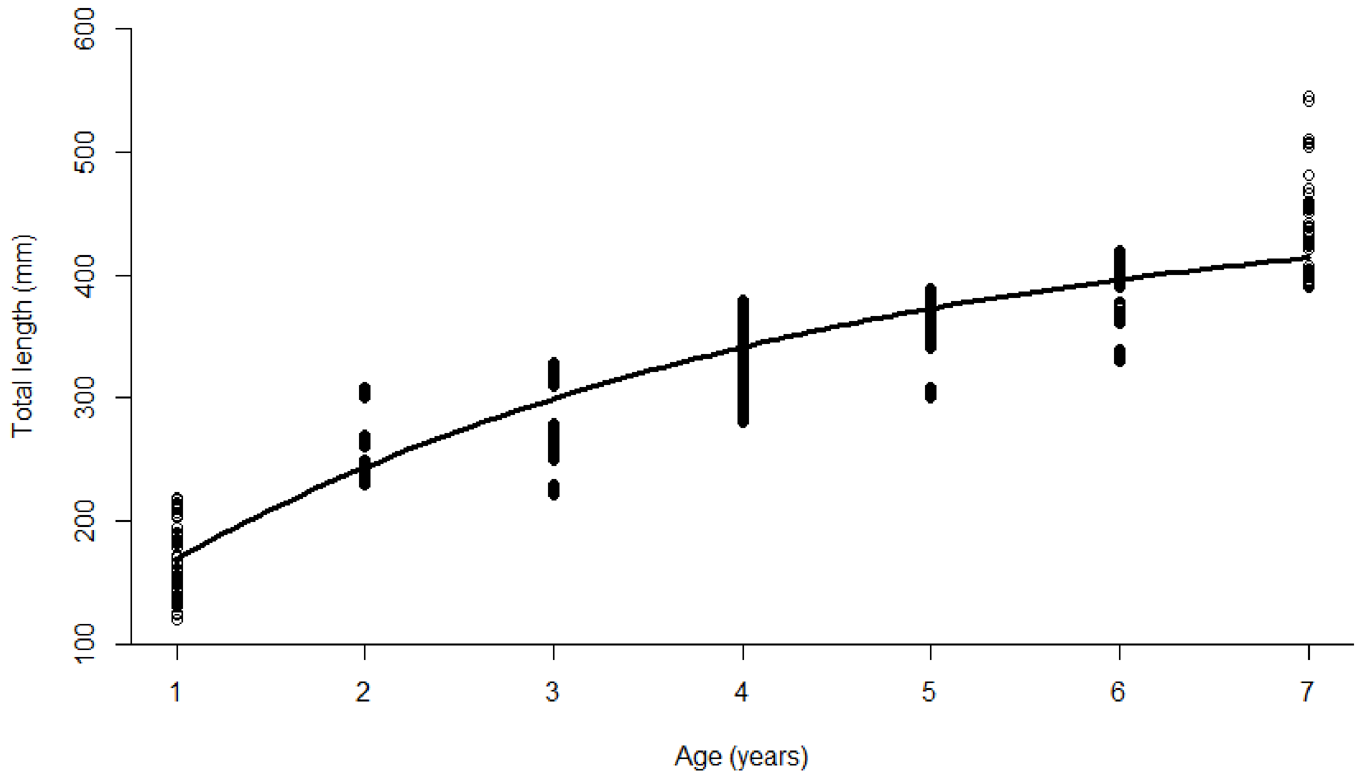

Figure 14. The Von Bertalanffy growth function (solid black curved line) fitted to Rainbow Trout total length-at-age captured via raft electrofishing in Box Canyon in May 2021.

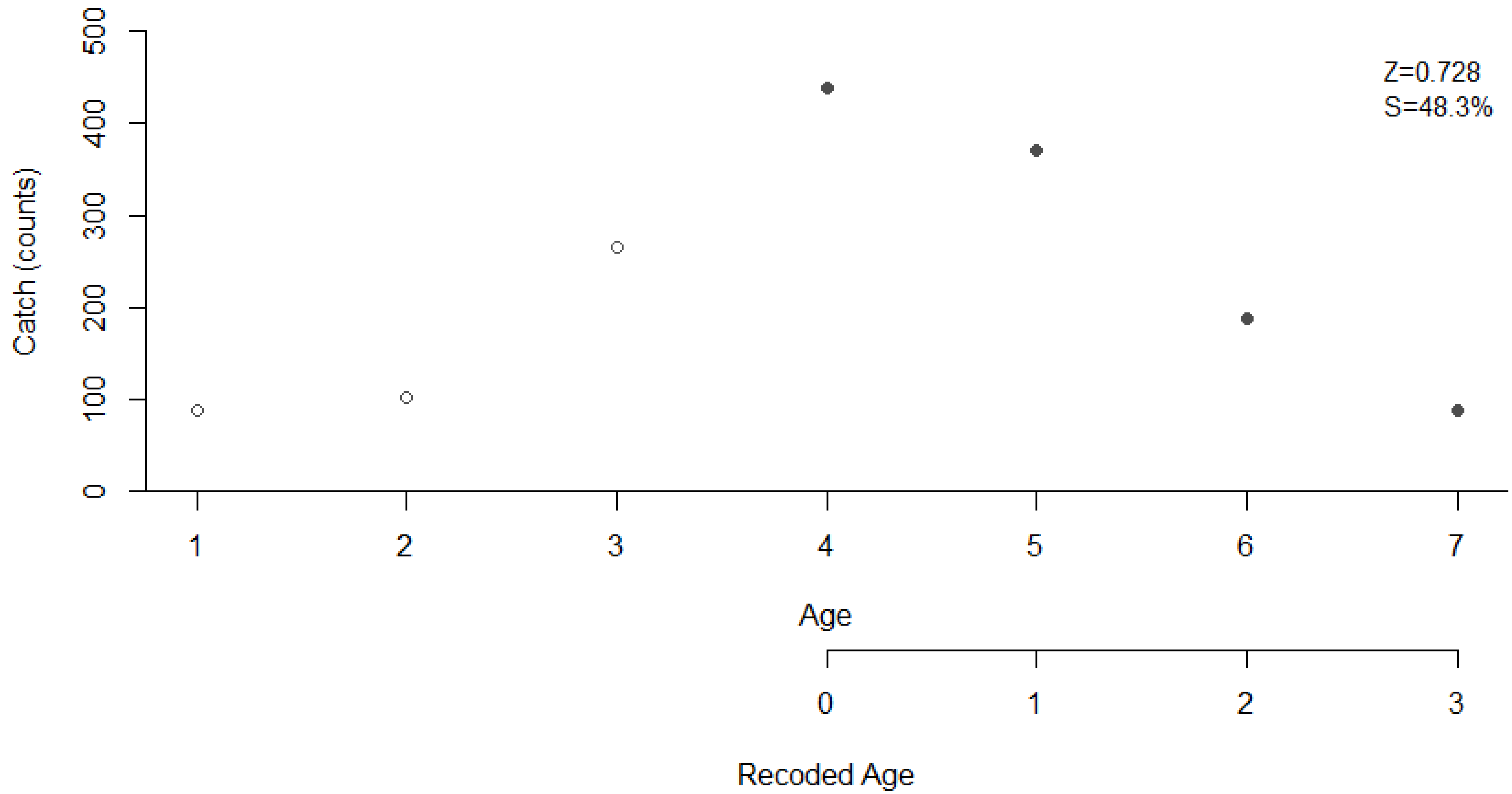

Figure 15. Catches at estimated ages for Rainbow Trout captured in the Box Canyon reach of the Henrys Fork Snake River in May 2021. The solid points were used to calculate the Chapman-Robson estimates of *S* and *Z*.

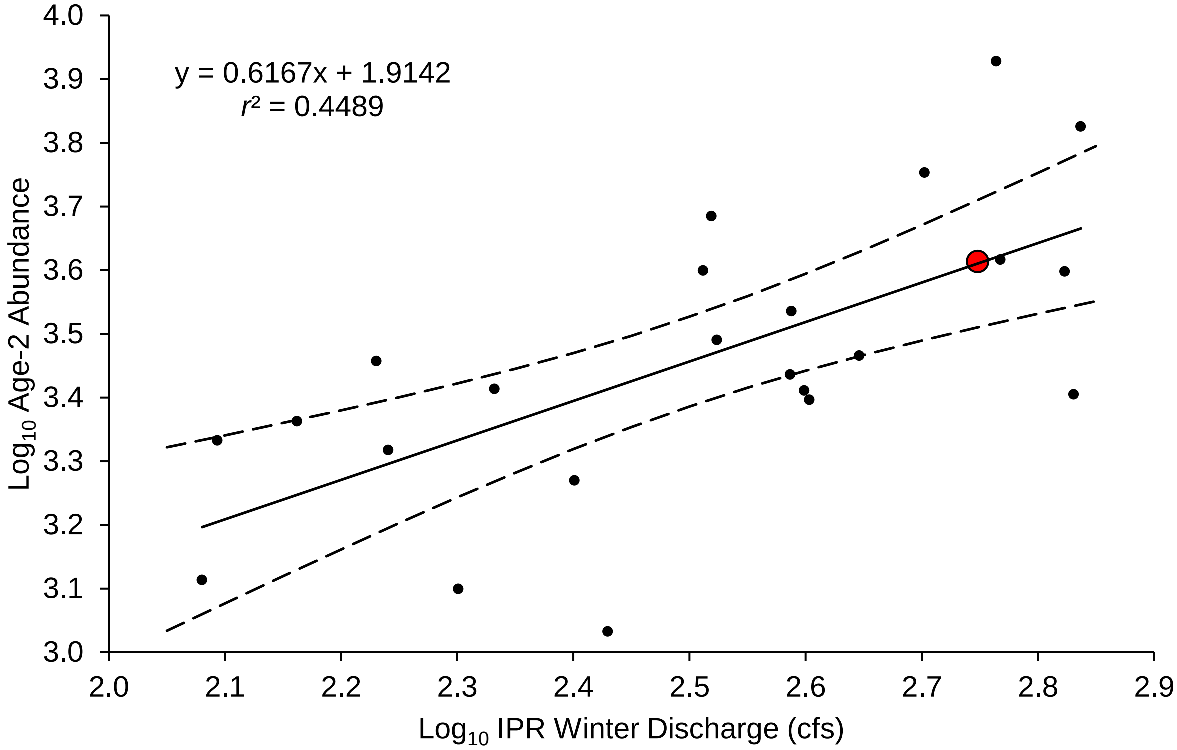

Figure 16. Linear regression (solid black line) and 95% confidence intervals (dashed lines) of the relationship between age-2 Rainbow Trout abundance and mean winter streamflow (cubic feet per second; cfs) during the first winter of a fish's life from 1995–2021. The black points are the observed age-2 abundances from 1995 to2019, and the red dot is the predicted age-2 abundance in 2021 using the regression formula of years 1995–2019. $\log_{10}$ age-2 Rainbow Trout abundance = 0.6167*$\log_{10}$ flow (cfs) + 1.9142, (r^2 = 0.45, $F_{1, 22}$ = 18.11, P < 0.01).

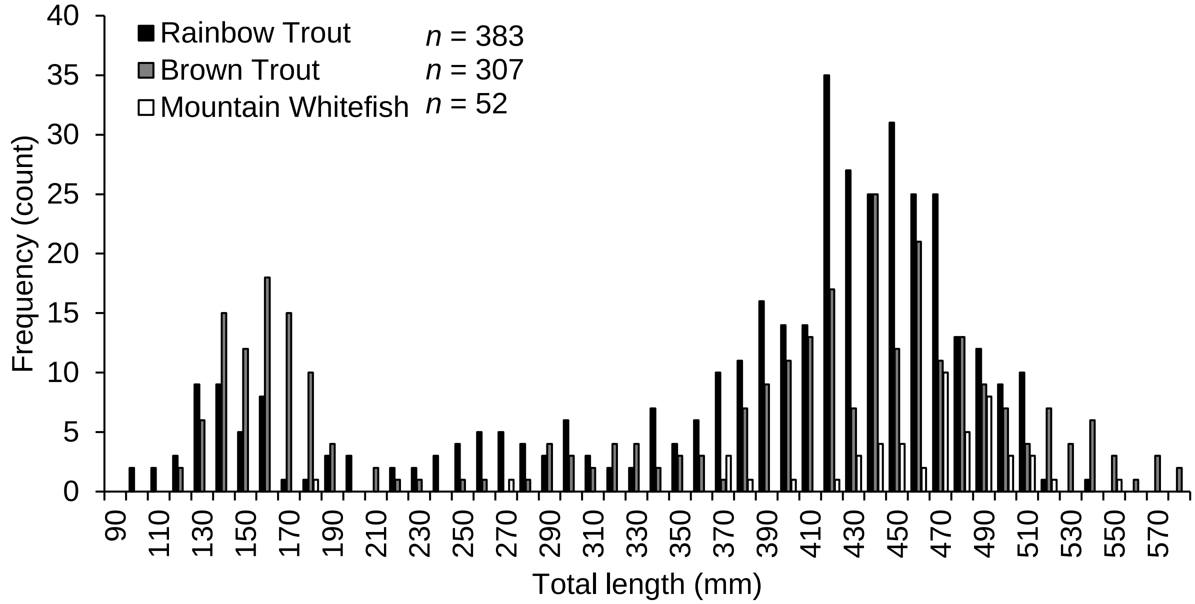

Figure 17. The length-frequency distributions of Rainbow Trout, Brown Trout, and Mountain Whitefish captured in the Vernon reach of the Henrys Fork Snake River in May 2021.

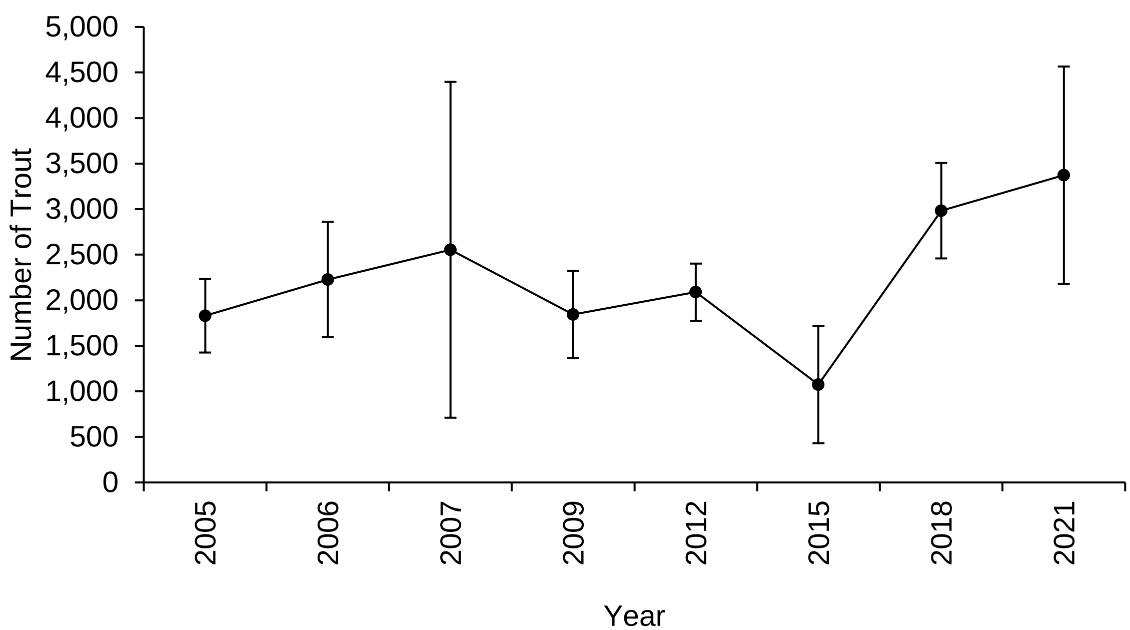

Figure 18. Modified Petersen abundances estimates for all trout combined in the Vernon reach of the Henrys Fork Snake River in spring from 2005–2021. Error bars represent 95% confidence intervals around the abundance estimate.

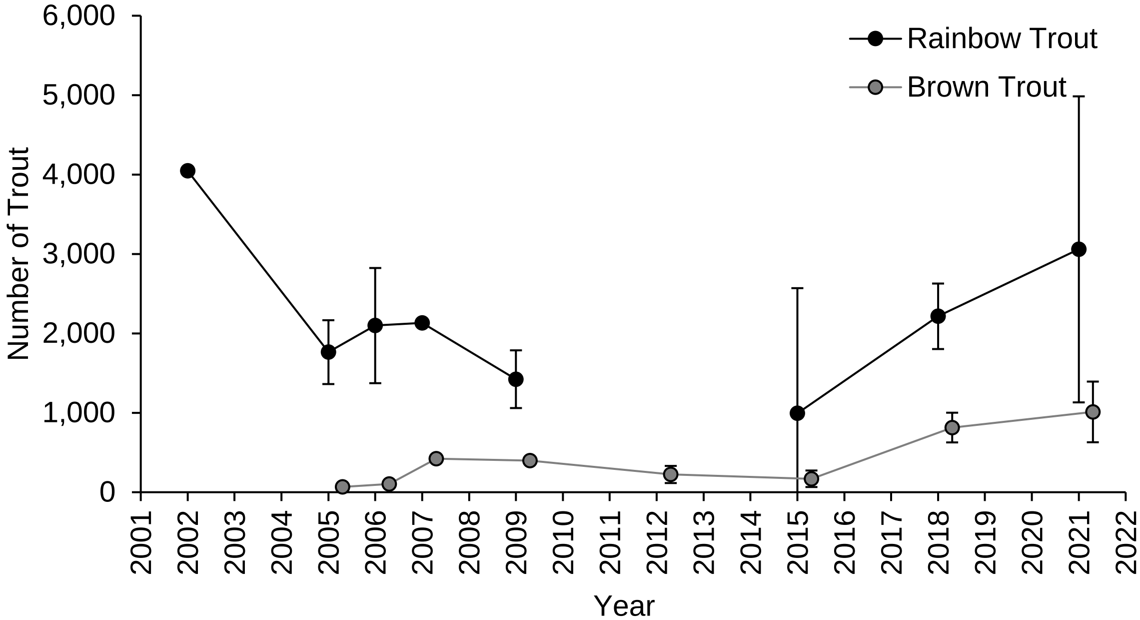

Figure 19. Modified Petersen abundance estimates for Rainbow Trout and Brown Trout in the Vernon reach of the Henrys Fork Snake River in spring from 2002–2021. Error bars represent 95% confidence intervals around the abundance estimate.

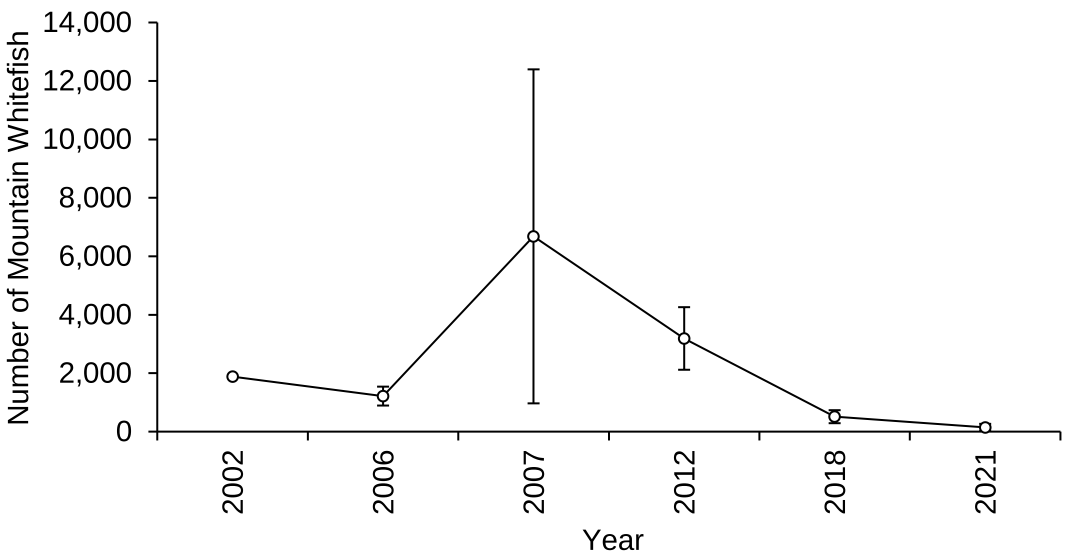

Figure 20. Modified Petersen abundance estimates for Mountain Whitefish in the Vernon reach of the Henrys Fork Snake River in spring from 2002–2021. Error bars represent 95% confidence intervals around the abundance estimate.

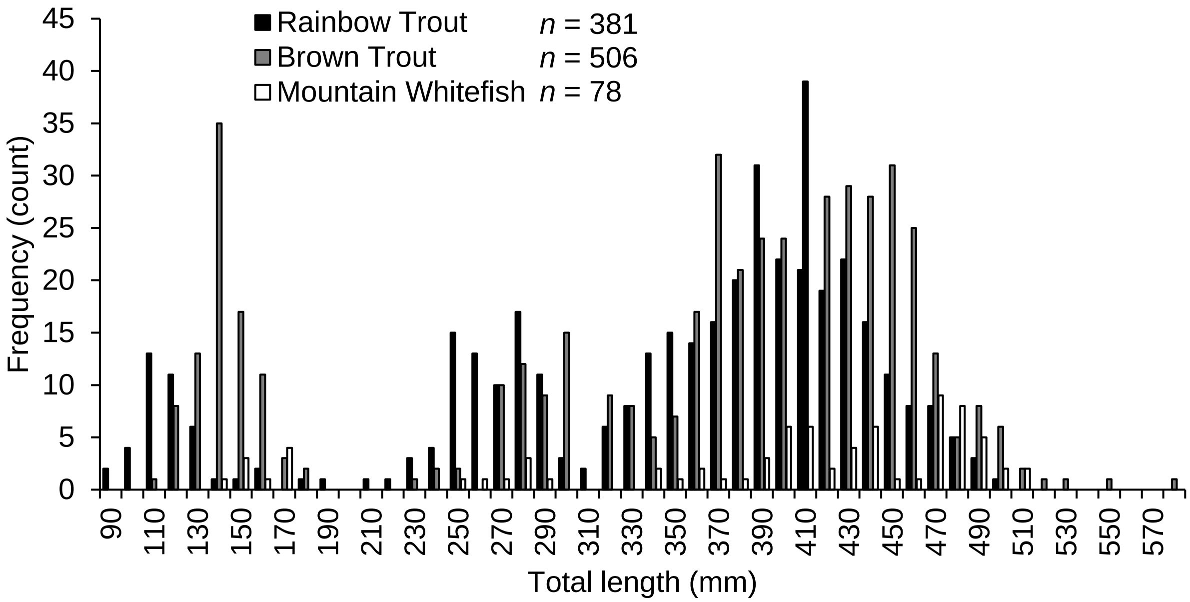

Figure 21. The length-frequency distributions of Rainbow Trout, Brown Trout, and Mountain Whitefish captured in the Chester reach of the Henrys Fork Snake River in May 2021.

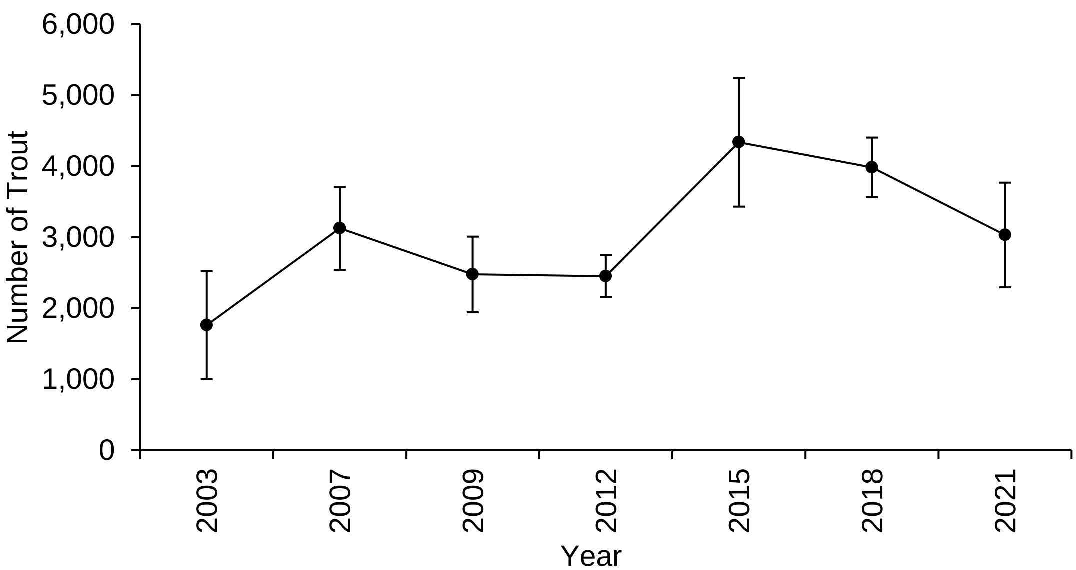

Figure 22. Modified Petersen abundance estimates for all trout combined in the Chester reach of the Henrys Fork Snake River in spring from 2003 to 2021. Error bars represent 95% confidence intervals around the abundance estimate.

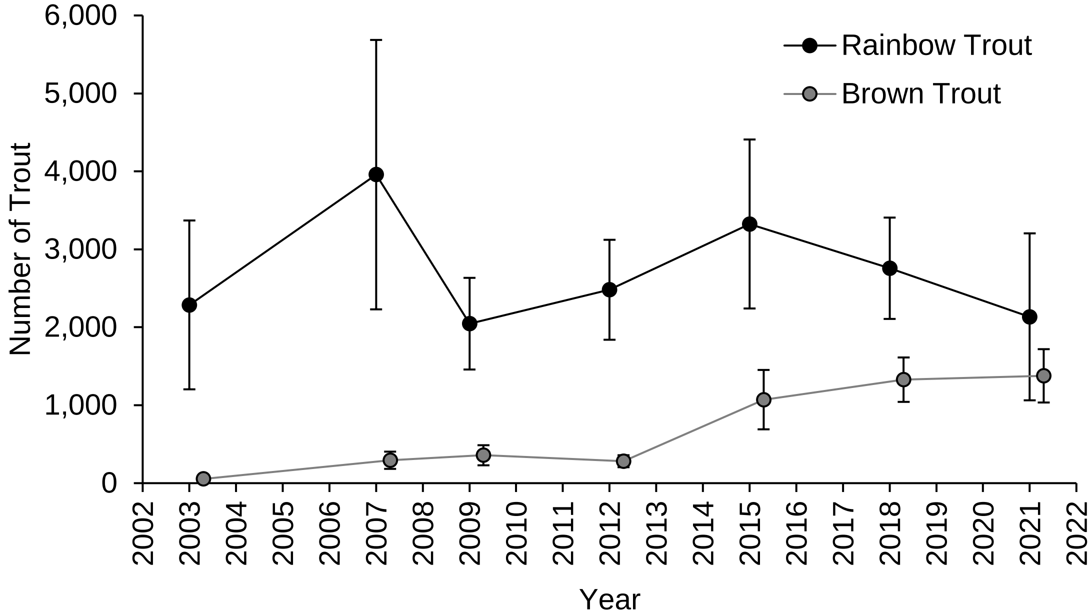

Figure 23. Modified Petersen abundance estimates for Rainbow Trout and Brown Trout in the Chester reach of the Henrys Fork Snake River in spring from 2003 to 2021. Error bars represent 95% confidence intervals around the abundance estimate.

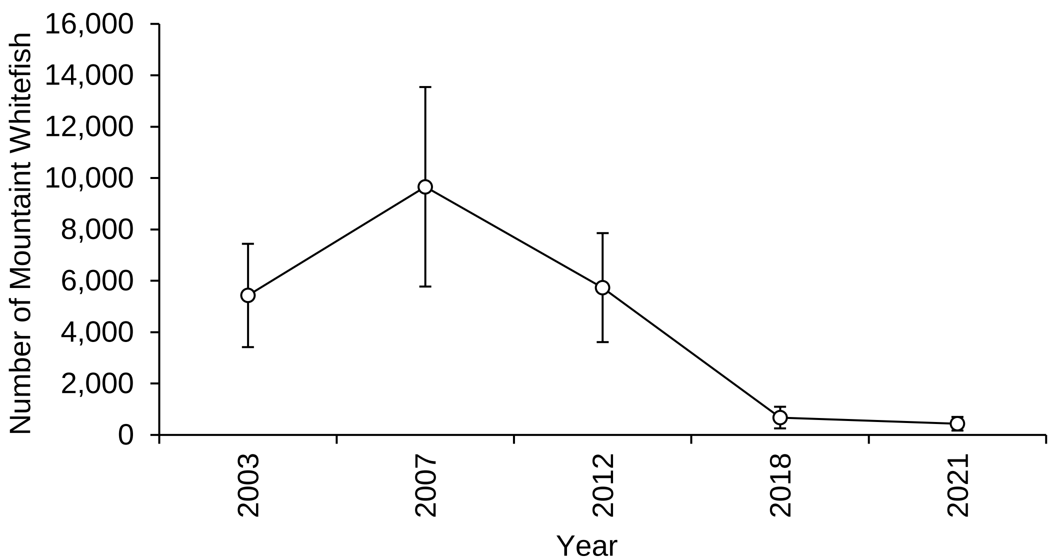

Figure 24. Modified Petersen abundance estimates for Mountain Whitefish in the Chester reach of the Henrys Fork Snake River in spring from 2003 to 2021. Error bars represent 95% confidence intervals around the abundance estimate

TETON AND SOUTH FORK TETON RIVERS

ABSTRACT

We estimated trout and Mountain Whitefish *Prosopium williamsoni* (MWF) abundance, species composition, average lengths, and relative weights in the Teton River and South Fork Teton River (SFTR). The Teton River and SFTR are home to a variety of trout species which include Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), Brook Trout (BKT), and Brown Trout (BNT). We surveyed all trout in two sites of the Teton Valley section (Nickerson and Breckenridge) and one site in the lower Teton River section (SFTR). We also surveyed MWF in one reach of the Teton Valley section (Buxton Site) and the FTR site. Total trout abundance in both the Teton Valley sites increased since historic lows in 2003. Trout abundance estimates (± 95% confident intervals) at Nickerson included 232 YCT/km (± 32), 340 RBT (± 94), and 518 BKT (± 92). Downstream at Breckenridge, trout estimates were 35 YCT/km (± 21), 386 RBT (± 87), 287 BKT (± 141), and 120 BNT (± 235). In the SFTR, we estimated 15 YCT/km (± 11), 12 RBT (± 5), and 52 BNT (± 18). Relative weights for all species were high. Non-native trout abundance (RBT, BKT, and BNT) has increased at all sites. Brown Trout are now the dominant species in the SFTR. We estimated there were 1,402 MWF/km (± 223) in the Buxton site and 24 (±13) MWF in the SFTR. Mountain Whitefish abundances decreased in both site since prior surveys. A fish ladder and trap on the SFTR was operated during the spring. Overall, results indicate a shift in survey sites to more abundant non-native species. Continued monitoring is warranted to better understand the current effects of non-native trout on YCT and factors affecting species trends.

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INTRODUCTION

The Teton River and South Fork Teton River (SFTR), tributaries of the Henrys Fork Snake River in eastern Idaho, support robust populations of wild trout including an important population of native Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (YCT). Other prevalent species include Rainbow Trout *O. mykiss* (RBT), Brook Trout *Salvelinus fontinalis* (BKT), Brown Trout *Salmo trutta* (BNT), Mountain Whitefish *Prosopium williamsoni* (MWF), Utah Sucker *Catostomus ardens* (UTS), and Bluehead Sucker *Catostomus discobolus* (BHS).

Yellowstone Cutthroat Trout in the Teton River are separated into three distinct independent meta-populations sub-sectioned into the lower Teton River which includes the SFTR and North Fork Teton River, the Teton Canyon, and the Teton Valley (Schrader and Jones 2004). Each of these sections exhibit a unique species composition, where YCT dominate the catch in the North Fork Teton River and Teton Canyon sections (76% in 2020) while non-native BKT, BNT, and RBT comprise the majority of the trout community in the Teton Valley section (96% Breckenridge site 2019; Heckel et al. 2020). The boundary between the Teton Valley and Teton Canyon section of the River is the Felt Hydroelectric Dam. In 2020, a fish trap was operated at Felt Dam over a 7-month period with 116 unique adult YCT and 589 RBT (including RBT/cutthroat hybrids) captured (Gregory 2020) further indicating low YCT movement between these populations.

Habitat and gradients are different between these three sections. The lower Teton River section extends downstream of Teton Dam through the town of Rexburg to the Henrys Fork Snake River and encompasses both the SFTR and North Fork Teton Rivers. Numerous diversion structures in the lower Teton River section have likely further fragmented fish populations as these diversions were originally rebuilt following the Teton Dam collapse without adequate fish passage for juveniles or adults (Moore and Andrews 1983). An established spawning run for YCT is present in the SFTR with YCT moving upstream through the SFTR fish ladder with the majority spawning in Moody Creek and the remainder of the spawn relatively unknown (Schrader and Jones 2004). The SFTR fish ladder was built by IDFG in 1994 as a fish bypass to the Rexburg City Ditch, a major irrigation diversion. In the Teton Canyon, the majority of YCT spawn in tributaries, mainly Bitch Creek which is the largest undiverted tributary draining the western slope of the Teton Range and discharging around 130,000 acre feet per year. The hydrology of the canyon is driven by snowmelt with highly variable flows. Spawning has also been documented in Canyon Creek, the second major tributary of the Teton Canyon section. Trout in the Teton Valley section upstream of the Teton Canyon section, also spawn in tributaries (Schrader and Jones 2004), but these tributaries exhibit moderate spring flows, higher, more consistent summer flows, and consistent temperature regimes, similar to spring-fed creek aquatic environments. These are seasonally connected streams with a large component of irrigation demand which return warmer water to the river (Van Kirk and Jenkins 2005). These spring creek conditions in the Teton Valley section favor non-native trout production, while the more natural snowmelt driven system in Teton Canyon section favor YCT production (Moller and Van Kirk 2003; Van Kirk and Jenkins 2005).

The Teton River has been managed by Idaho Department of Fish and Game (IDFG) as a wild trout fishery since the early 1990s with the following objectives:

- Preserve genetic integrity and population viability of YCT.
- Restore connectivity and natural hydrology as possible to improve spawning, rearing migration success of YCT.
- Minimize impacts of land use and development on fish habitat and water quality.

- Increase consumptive trout fishing opportunity for anglers near population centers.
- Minimize loss of juvenile fish to irrigation diversions and tributary dewatering where these losses are deemed to be having a population-level impact on the resource.
- Obtain adult fish passage around or through barriers (IDFG 2019).

Native YCT abundance has fluctuated in the Teton River since monitoring efforts began in 1987, and management objectives have evolved. The primary management objectives hen sampling efforts began were providing quality angling and harvest opportunities (IDFG 1991), and hatchery fish (Rainbow Trout and Yellowstone Cutthroat Trout) were stocked annually through 1992 and 1994 for each species, respectively. A shift in angler expectations and management objectives occurred during the early 90s. In 1990, a slot limit regulation was adopted for the Teton River, and wild trout became the management focus in 1995. Through the late 90s and early 2000s YCT abundance in the Teton River declined. Abundance of YCT was estimated at an all-time low in 2003 of 3 YCT/km (Garren et al. 2006a). This led to the elimination of legal YCT harvest from the river and its tributaries in beginning in 2006. Since 2003, YCT abundance has rebounded but non-native trout continue to threaten YCT in the drainage (IDFG 2019).

Non-native trout abundance has increased along with increasing YCT abundance since 2003. Non-native RBT are a risk to the genetic integrity and population viability of YCT in the Teton River drainage through competition and hybridization. Interspecific competition can lead to increased mortality through completion for food resources and niche space (Seiler and Keeley 2007a, 2007b). Hybridization, the more significant threat, may result in the loss of genetically distinct YCT through gene flow from one species to another through backcrossing of interspecific hybrids (Allendorf and Leary 1988; Kruse et al. 2000; Kozfkay et al. 2007). To mitigate these threats, IDFG removed the RBT daily bag limit in fishing regulations in 2015. Since 2003, YCT population trends have been positive in the Teton Valley section (Heckel et al. 2019), and Teton Canyon section (Vincent et al. *in press*) while remaining stable in the lower Teton River section (Flinders et al. 2016). The Teton River is currently managed as a native trout fishery, and Idaho anglers continue to identify the management for native trout fisheries as a preferred and important management objective (Koenig 2020).

Bluehead Sucker *Catostomus discobolus* are another species native to the Teton River that receives management attention. We have limited information available for this nongame native species (IDFG 2019). However, we do encounter this species during electrofishing surveys and fish trapping efforts in the Teton River drainage. We encounter Bluehead Suckers infrequently, which indicates they may be low in abundance. Because of this concern and the need for more biological information for the species, this nongame species is protected and harvest is not allowed. There is some discussion among taxonomists relative to the scientific name for Bluehead Sucker. Some researchers and taxonomists have suggested this species of sucker that we encounter in the Teton River are more accurately labeled as Green Sucker *Pantosteu virescens* (Sigler and Zaroban 2018).

Beginning in 1987, we have routinely sampled the Nickerson and Breckenridge site in the Teton Valley section to estimate trends in trout abundance and species composition. Periodically, other sites of the river have also been sampled in the Teton River drainage. This report summarizes findings for trout abundance from two sites in the Teton Valley section (Nickerson and Breckenridge), and one site in the lower Teton River section, the SFTR. We also report MWF abundance in one site each from the Teton Valley section (Buxton) and the lower Teton River section (SFTR).

OBJECTIVES

Our objective was to monitor trends in species composition, fish abundance, and fish metrics to gain insight into potential limiting factors affecting fish populations in the Teton River so that we could improve conservation efforts and fishing.

METHODS

We estimated trout abundance by species using mark/recapture techniques at the Nickerson and Breckenridge sites in Teton Valley section, and the SFTR trout site in the lower Teton River section (Figure 25). We also estimated MWF abundance using mark/recapture at the Buxton site in Teton Valley section and in the SFTR MWF site in the lower Teton River section. We conducted all electrofishing surveys using drift boat and raft-mounted gear in the fall when river flows reached base levels of approximately 300 cubic feet per second (cfs). We completed two electrofishing passes with a one week interval between passes at each site. Captured fish which were identified to species and measured to the nearest total length in mm. A subsample of fish for each species were weighed to the nearest gram (g). When less than 100 fish were sampled all encountered fish were weighed. During the marking pass, fish were marked using a hole punch in the caudal fin, and this mark was used to identify previously captured fish in the recapture pass. During the second pass, captured fish were again measured, identified to species, and inspected for caudal fin marks. After calculating density estimates for each species as described in Schoby et al. (2013), we used linear regression to estimate the intrinsic rate of change in abundance for each species, with sample year as the independent variable and the log transformed abundance estimate (trout/km) as the dependent variable. The slope of the regression line fit to the $\log_e$ transformed abundance data is the intrinsic rates of change (r) for the population (Maxell 1999; High et al. 2008; Kennedy and Meyer 2015). Positive intrinsic rates of change ($r > 0$) indicate that abundance is increasing, and negative estimates of r indicate decreasing abundance in the population. Two separate time frames were used to estimate abundance changes which included 1987 to present to encompass the entire dataset and from 2003 to present as the rate of change from historic lows in abundance observed in 2003.

Relative weights (W_r) were calculated by dividing the weight of each fish (by a standard weight (W_s) for the same length for that species multiplied by 100 (Anderson and Neumann 1996). Relative weights were then averaged for each length class (< 150 mm, 150-249 mm, 250-349 mm, 350-449 mm, and fish > 449 mm). We used the following formulae to calculate W_r:

$$\log W_s = -5.192 + 3.086 \log \text{TL for YCT (Kruse and Hubert 1997)}$$

$$\log W_s = -5.023 + 3.024 \log \text{TL for RBT (Simpkins and Hubert 1996)}$$

$$\log W_s = -5.186 + 3.103 \log \text{TL for BKT (Hyatt and Hubert 2001a)}$$

$$\log W_s = -4.867 + 2.96 \log \text{TL for BNT (Milewski and Brown 1994)}$$

$$\log W_s = -5.086 + 3.036 \log \text{TL for MWF (Rogers et al. 1996).}$$

We compared W_r among size groups using 95% confidence intervals (CI) where statistical difference was inferred with non-overlapping CIs.

South Fork Teton Fish Trap

A cement fish ladder was installed in the SFTR on a large irrigation diversion on the SFTR near the confluence of Moody Creek by IDFG in 1994 to facilitate upstream fish passage. A fish trap was also installed at this location to quantify fish passage through the ladder. This fish trap was retrofitted with the installation of a funnel entrance on the downstream end and a screen across the upstream end. The trap was installed on March 19, 2021, with the first sampling occurring on March 22, 2021. The trap was maintained and checked between the hours of 1200 and 1600 every 1-3 days until the trap was pulled on June 21, 2021. During each visit, the trap was checked for fish, cleaned; and, water temperature (°C), and water level (inch) was measured. Water level was measured from the top of the concrete fish ladder to the current water level at the downstream end of the ladder.

We identified captured fish to species, measured them to the nearest mm (TL), determined sex (if phenotypically possible), and moved fish upstream of the diversion. A subsample of YCT and BHS greater than 350 mm in length were marked with half-duplex PIT tags as part of an ongoing movement study in the drainage. A subsample of BNT and RBT were implanted with non-reward T-bar anchor tags at the base of the dorsal fin according to standard methods (Dell 1968). Anchor tags were printed with a unique identification number, phone number, and website address where anglers could report the tag using the “Tag You’re It” statewide tag reporting system (Meyer and Schill 2014). Genetic samples in the form of fin clips were collected from a sub-sample of phenotypically identified YCT and BHS which were stored on Whatman data sheets. All captured BKT were euthanized. All phenotypically-identified BHS were checked for tubercles on the anal and tail fins to determine sex. The presence of tubercles indicated the fish were male while the absence was indicative of females.

RESULTS

In the Teton Valley section, we sampled the Nickerson site on September 7 and 14, 2021 and captured 1,556 unique trout including 390 YCT, 316 RBT, 845 BKT, and 5 BNT (Figure 26). Of these captured trout, weights were taken from 187 YCT, 703 RBT, 165 BKT, and 5 BNT. We estimated there were 232 YCT/km (± 32), 340 RBT (± 94), and 518 BKT (± 92; Figure 27). The average length of YCT was 272 mm (Figure 26), and their average W_r was 98 (Figure 28). Rainbow Trout averaged 295 mm and their average W_r was 99 (Figure 28). Brook Trout averaged 210 mm with an average W_r of 104. Brown trout averaged 255 mm and their average W_r was 122 (Table 13). Abundance trends for YCT at Nickerson since 1987 have been stable (r = 0.02, F = 0.49, df = 15, P = 0.49). However, since 2003 YCT abundance increased significantly (r = 0.18, F = 15.75, df = 9, P = 0.004). Rainbow Trout abundance at Nickerson significantly increased since 1987 (r = 0.08, F = 10.95, df = 12, P = 0.007). Abundance of RBT at Nickerson from 2003 through 2021 also increased, although not significantly (r = 0.07, F = 3.17, df = 9, P = 0.11). Brook Trout abundance at Nickerson significantly increased for both the entire data set (r = 0.10, F = 30.29, df = 13, P < 0.001) and since 2003 (r = 0.10, F = 10.5, df = 9, P = 0.01; Figure 26). Prior to 2003, species composition was on average 56% YCT, and 45% non-native trout (RBT, BKT, and BNT combined). A significant change occurred post-2003 (2003-2021) to an average of 21% YCT and 79% non-native trout in the population (Figure 29). No notable differences in species composition occurred between the 2019 and 2021 surveys.

We sampled the Breckenridge site on September 8 and 15, 2021 and captured a total of 1,021 unique trout. This included 58 YCT, 354 RBT, 444 BKT, and 165 BNT (Figure 30). Weights were obtained from 31 YCT, 174 RBT, 206 BKT, and 83 BNT. We estimated there were 35

YCT/km (± 21), 386 RBT (± 87), 287 BKT (± 141), and 120 BNT (± 235; Figure 31). The average length of YCT at Breckenridge was 304 mm with an average W_r of 90. The average length of RBT was 293 mm with an average W_r of 102. Brook Trout length averaged 181 mm with an average W_r of 102 (Table 13; Figure 32). We observed multiple size classes of Brown Trout (Figure 29). Brown Trout averaged 253 mm in total length and had an average W_r of 102. Between 1987 and 2001, the intrinsic rate of population change for YCT was -0.01, but this estimate was not significantly different than zero, indicating a stable trend (F = 0.54, df = 13, P = 0.54). The intrinsic rate of population change since 2003 was significantly different than zero at Breckenridge, suggesting the YCT population is increasing (r = 0.12, F = 27.11, df = 8, P = 0.001). Rainbow Trout abundance at Breckenridge has significantly increased over the duration of the dataset (r = 0.05, F = 5.95, df = 14, P = 0.03). Since fish populations decreased in 2003, RBT abundances have remained relatively stable (r = 0.02, F = 0.57, df = 9, P = 0.47). Brook Trout trends have significantly increased at Breckenridge from 1987 through 2021 (r = 0.12, F = 6.55, df = 8, P = 0.04), and since 2003 (r = 0.21, F = 21.6, df = 7, P = 0.004). Prior to 2003, the trout population was on average 25% YCT, and 75% non-native trout (RBT, BKT, and BNT combined). Post-2003 (2003 - 2021) the fish composition has averaged 6% YCT and 94% non-native trout (Figure 33). No notable differences in species composition occurred since the previous survey at this site. Genetics were obtained from 37 BNT, 2 RBT, and 69 YCT in the Valley section. A total of 106 YCT, 3 HYB, 102 RBT, and 36 BNT were tagged with half-duplex Passive Integrated Transponder (PIT) tags in the Valley section.

We sampled the Buxton site on October 13 and 20, 2021 and captured 1,063 unique MWF (Figure 25). Average TL of MWF was 299 mm (Figure 34) and mean W_r was 96 (Table 13; Figure 35). We estimated there were 1,402 MWF/km (± 223) in our 2021 survey which was less than our 2005 estimate of 2,485 (± 356; Meyer et al. 2009).

In the lower Teton River section, we sampled the SFTR site on September 22 and 29, 2021 (Figure 25). We captured a total of 136 trout. This included 20 YCT, 29 RBT, and 87 BNT (Figure 36). We measured weights for 20 YCT, 24 RBT, and 85 BNT. We estimated there were 15 YCT/km (± 11), 12 RBT (± 5), and 52 BNT (± 18; Figure 37). The average TL of YCT was 327 mm with an average W_r of 98. The average TL of RBT was 344 mm with an average W_r of 101. The average TL of BNT was 334 mm with an average W_r of 96 (Table 13; Figure 38). This is the sixth year of sampling for this site. Over the entire dataset (1993 – 2021), the trend, for YCT has been stable (r = -0.02, F = 0.15, df = 5, P = 0.72). Rainbow Trout abundance was variable across years but trend was stable (r = -0.03, F = 0.032, df = 2, P = 0.16). Brown Trout abundance increased over the sampling period, but the trend was not significant (r = 0.95, df = 2, P = 0.10). Species composition has changed from the 2015 survey. Brown Trout currently are 64% of the fish community compared to 35% in 2015. Prior to this survey YCT comprised 52% of the fish community compared to 15% in this survey (Figure 39). Genetic samples were obtained from 65 BNT, 11 HYB, 15 RBT, and 12 YCT. We also captured 65 MWF in the SFSR site (Figure 40). We estimated there were 24 MWF/km (± 13; Table 13) which was significantly lower than our 2015 estimate of 210 MWF/km (± 102). The average TL for MWF was 241 mm with an average W_r of 89 (Table 13).

South Fork Teton Fish Trap

We captured 2,170 fish including 2,002 UTS, 88 BHS, 35 YCT, 13 RBT, 11 BNT, 11 MWF, and 10 BKT in the SFTR fish trap. We PIT tagged 17 YCT and 76 BHS, and T-bar anchor tagged 10 BNT and 13 RBT. We collected genetic samples from 32 YCT and 87 BHS. Overall species composition of catch at the trap was dominated by UTS (92%). Bluehead Sucker and YCT comprised 4% and 1.6% of the catch respectively with <1% of the catch represented each by

RBT, BNT, and BKT. Utah Sucker were first captured on 5 April, 2021. There were eight individual trapping dates where we captured >100 UTS, with the peak of trapped UTS (*n* = 417) on May 18. Over a six day period, we captured 55% of the total UTS trapped during the sampling period. This occurred from May 14 through May 19 (Figure 41). Yellowstone Cutthroat Trout were captured between April 26 and June 20, 2021 (Figure 42) with 50% of the run passed on May 18. Yellowstone Cutthroat Trout average TL at the trap was 324 mm (Figure 43).

DISCUSSION

Within the Teton Valley section of the Teton River, the Nickerson and Breckenridge sites have been sampled since 1987 and are the standard IDFG sites. To date, the time series for the surveys in this section is the longest running and most comprehensive information on fish abundance in the Teton River (Schrader and Brendon 2004; Garren et al. 2006c). Throughout the Teton Valley section, YCT abundances continue to be stable since or increasing since 2003.

Non-native trout (BKT and RBT) populations continue to persist in abundances greater than or similar to YCT. Over the duration of the data set, species compositions have been variable. Since 2003, RBT and BKT collectively comprise a greater percentage of the population compared to the native YCT in the Teton Valley section. Brown Trout, an additional non-native species in the Teton Valley section, have also experienced a positive trend in abundance for unknown reasons. Low numbers of BNT (< 7 fish) were captured in our surveys from 2007 to 2013. Since 2013, we have captured an increasing number of individuals and size classes. This population of non-native trout to the Teton River drainage poses additional threats to the persistence of YCT. Non-native trout have been found to outcompete YCT for niche space, thermal refuge, and prey resources (Al-Chokhachy and Sepulveda 2019; Budy and Gaeta 2018; Gregory and Griffith 2000; Meyer et al. 2006; Seiler and Keeley 2007a; Seiler and Keeley 2007b). The greatest threat to the population viability and genetic integrity of Teton River YCT is hybridization with RBT resulting in viable fertile offspring. Hybridization may result in the loss of genetic integrity through gene flow from one species to the other through backcrossing and interspecific hybrids, eventually resulting in the loss of genetically distinct YCT (Kruse et al. 2000; Kozfkay et al. 2007; Gunnell et al. 2008). Increases in hybridization coupled with interspecific competition for niche space and food resources points towards a disconcerting outlook for the persistence of YCT in the Teton drainage.

Although our estimates identify that BNT are expanding into the Teton Valley section, overall species composition of total non-native trout (BKT, RBT, and BNT) has not changed drastically since 2003.

In the SFTR, species composition has shifted from YCT as the most common species to BNT. Several studies suggest that Brown Trout and Brook Trout displace native trout through aggressive behavior, predation, and competition for food and space throughout western North America (Budy and Gaeta 2018; Dunham et al. 2002). In the Teton River basin, Al-Chokhachy et al. (2021) found at sites where YCT and BKT were sympatric, BKT abundance exceeded YCT abundance at 86% of the sites. Furthermore, a study on a native YCT stream in Montana showed reduced growth rates, juvenile recruitment, and survival of YCT with the invasion of BNT into the stream (Al-Chokhachy and Sepulveda 2019). Bonneville Cutthroat Trout have been showed to change their diet in the presence of Brown Tout, leading to suppressed growth and movement in two streams of northern Utah (McHugh and Budy 2006). To fully understand the effects of non-native trout on native YCT, further studies are warranted throughout the Teton River drainage.

Drought conditions in the Pacific Northwest in 2021 resulted in lower-than-average flows due to poor snowpack the preceding winter and higher water temperatures to the Teton drainage. A USGS stream flow gauge located at the start of the Breckenridge site (USGS Teton River gauge above South Leigh Creek) measured an average daily stream flow average of 169 cfs (0.08 cms) during the 2021 survey compared to 288 cfs (0.14 cms) in 2019. The average flow for the Teton River at this gauge on September 10th (when electofishing surveys are generally conducted) from 1961 to 2021 is 314 cfs (0.15 cms). Drought conditions usually have negative effects on fish populations in streams (Matthews and Marsh-Matthews 2003). These can include short-term changes due to habitat loss, physiological stress, or delayed changes such as reduced reproductive success (Elliott et al. 1997). Specific to Cutthroat Trout, drought is a primary abiotic factor negatively affecting resident populations (Dunham et al. 1999; Haak et al. 2010; Gresswell 2011). Meyer et al. (2014) found that drought exhibits negative effects on YCT abundance the following year indicating drought may contributed to lower catch rates in future surveys.

The spawning run timing of when 50% of the YCT spawning run had passed the SFTR fish trap was 17 days later than in 2019 (Heckel et al. 2020), and 28 days later than 2018 (IDFG unpublished data). We observed YCT at the trap during periods of generally declining stream flows. When flows were increasing, we generally did not capture YCT. In the upper Snake River basin, YCT generally migrate into spawning tributaries on the descending limb of the hydrograph, and they spawn just after peak discharge (Van Kirk and Jenkins 2005). A shift in later run timing in 2021 may be explained by inter-annual variability tied to stream flows. When comparing hydrographs between 2018, 2019, and 2021, this season exhibited lower, more consistent flows attributing to a drier spring than observed in 2019 and 2018. Run size appears to be declining since 2018 and 2019. However, the diversion structure associated with this fish trap is a partial fish barrier which some size classes of fish can pass depending on flows. This makes comparisons of catch at the SFSR trap among years difficult. We have not quantified the proportion of the run which uses the ladder or the environmental variables that affects this. We recommend quantifying ladder use rates in future studies to better quantify YCT spawning run size in the SFTR.

Bluehead Sucker are currently ranked as a rare vulnerable species (S3) in Idaho (IDFG 2015). Increasing numbers of BHS have been encountered at the SFTR trap indicating the SFTR may be an important location for spawning and rearing for this species that we know so little about. Multiple studies are currently ongoing in the upper Snake River watershed to understand the range, distribution, and genetic diversity of BHS and information gathered at the SFTR trap could add to these efforts. Basic life history studies should be completed fill in the existing information gap for BHS. Continued monitoring of this species throughout the Teton River drainage will be important to determine their current status in Idaho.

Mountain Whitefish abundance decreased in both the Buxton and SFTR sites from the previous surveys of 2005 (Meyer et al. 2009) and 2015 (Flinders et al. 2015) though the magnitude of the decrease was greater in the SFTR. The occurrence of juvenile MWF indicates some successful spawning and recruitment to the population. Information on the limiting factors specific to MWF in the scientific literature are largely un-defined. Potential factors relevant to the Teton River MWF may include water diversions, sport fisheries (Northcote and Ennis 1994), habitat degradation, water quality, water flows, and grazing (Gregory 2005). It is probable that the drought conditions outlined above may be a factor limiting MWF. We suggest managers continue to monitor MWF abundances at existing study sites throughout the Teton drainage to document trend and help identify factors responsible for the trend so appropriate management actions can be identified and enacted to conserve MWF.

MANAGEMENT RECOMMENDATIONS

1. Continue to monitor the abundance and trends of trout in the Teton River and identify factors affecting those trends.

2. Periodically monitor the abundance of MWF in existing sites of the Teton River to establish MWF abundance trends in the Teton River basin and identify factors that may be limiting recruitment.

3. Continue to operate the SFTR fish ladder to better understand factors affecting YCT in the lower Teton River section and Bluehead Sucker life history and relative abundance.

4. Monitor genetic purity of YCT to evaluate introgression and hybridization levels with RBT in order to gauge population status relative to objectives outlined in the Fisheries Management Plan.

5. Complete basic life history studies of Bluehead Sucker in the Teton River drainage and quantify distribution and abundance where possible.

Table 13. Summary statistics (± 95% CI) for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), Brook Trout (BKT), Brown Trout (BNT), and Mountain Whitefish (MWF) captured in each site of the Teton and South Fork Teton Rivers, 2021.

Site	Species	Mean TL (mm)	Mean weight (g)	Mean W_r	Density (# fish/km)
Teton River - Nickerson	YCT	272 (± 10.2)	181 (± 348)	98 (± 1.9)	232 (± 32)
	RBT	295 (± 10.2)	207 (± 38.3)	99 (± 2.7)	340 (± 94)
	BKT	210 (± 3.7)	145 (± 14.1)	104 (± 3.7)	518 (± 92)
	BNT	255 (± 115)	244 (± 395)	122 (± 72.4)	
Teton River - Breckenridge	YCT	304 (± 16.5)	319 (± 71.2)	90 (± 6.0)	35 (± 21)
	RBT	293 (± 8.8)	284 (± 30.3)	102 (± 5.3)	386 (± 87)
	BKT	181 (± 6.1)	104 (± 15.4)	102 (± 2.9)	287 (± 141)
	BNT	253 (± 15.6)	234 (± 64.5)	102 (± 4.6)	120 (± 235)
Teton River - Buxton	MWF	299 (± 4.8)	310 (± 17.9)	96 (± 1.0)	1,402 (± 223)
South Fork Teton River - Trout Reach	YCT	327 (± 51)	475 (± 189)	98 (± 5.1)	15 (± 11)
	RBT	344 (± 43)	705 (± 291)	101 (± 5.0)	12 (± 5)
	BNT	334 (± 25)	575 (± 132)	96 (± 2.3)	52 (± 18)
South Fork Teton River - MWF Reach	MWF	241 (± 27)	203 (± 58)	89 (± 3.4)	24 (± 13)

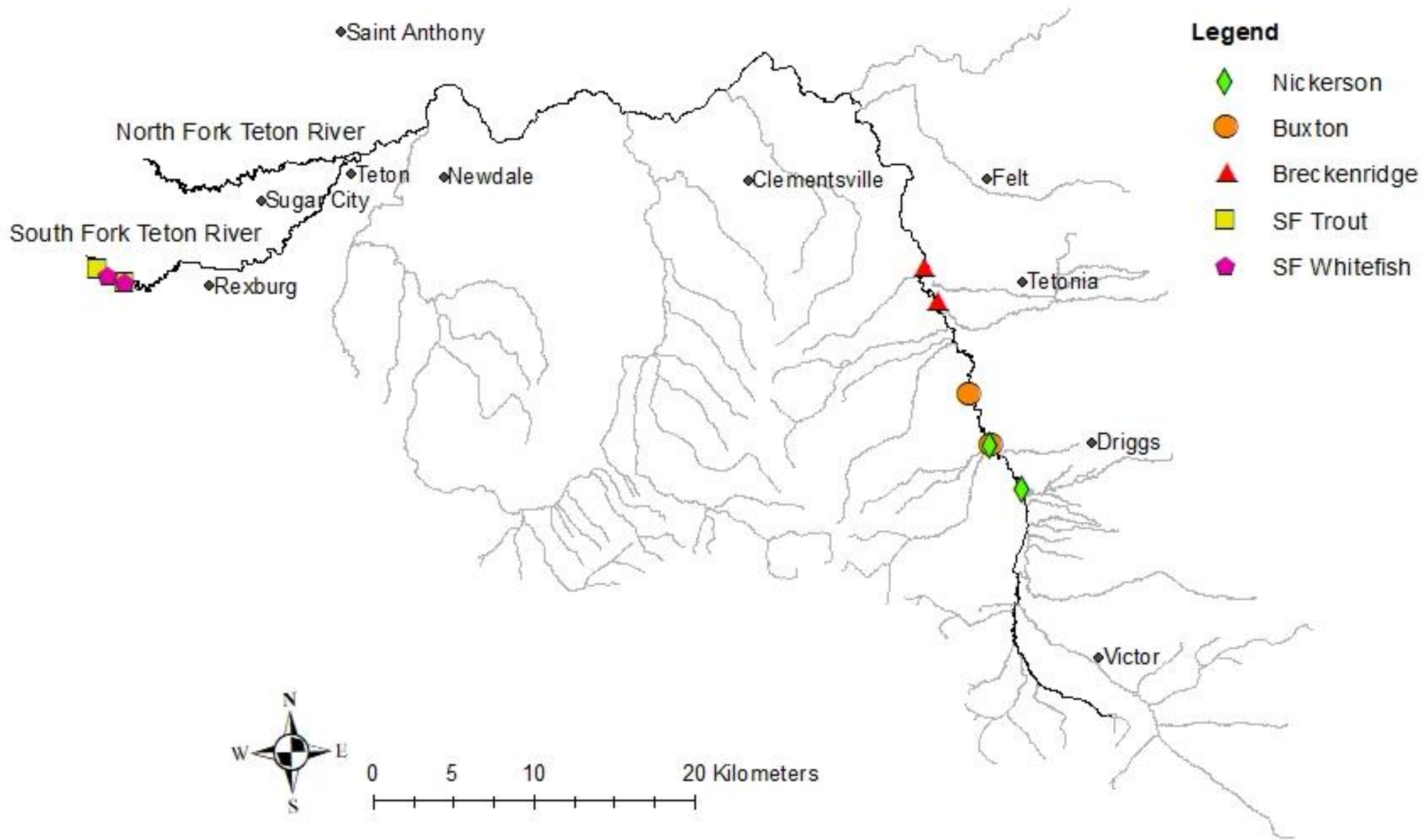

Figure 25. Electrofishing site boundaries in the Teton River and South Fork Teton River (SFTR) fall 2021.

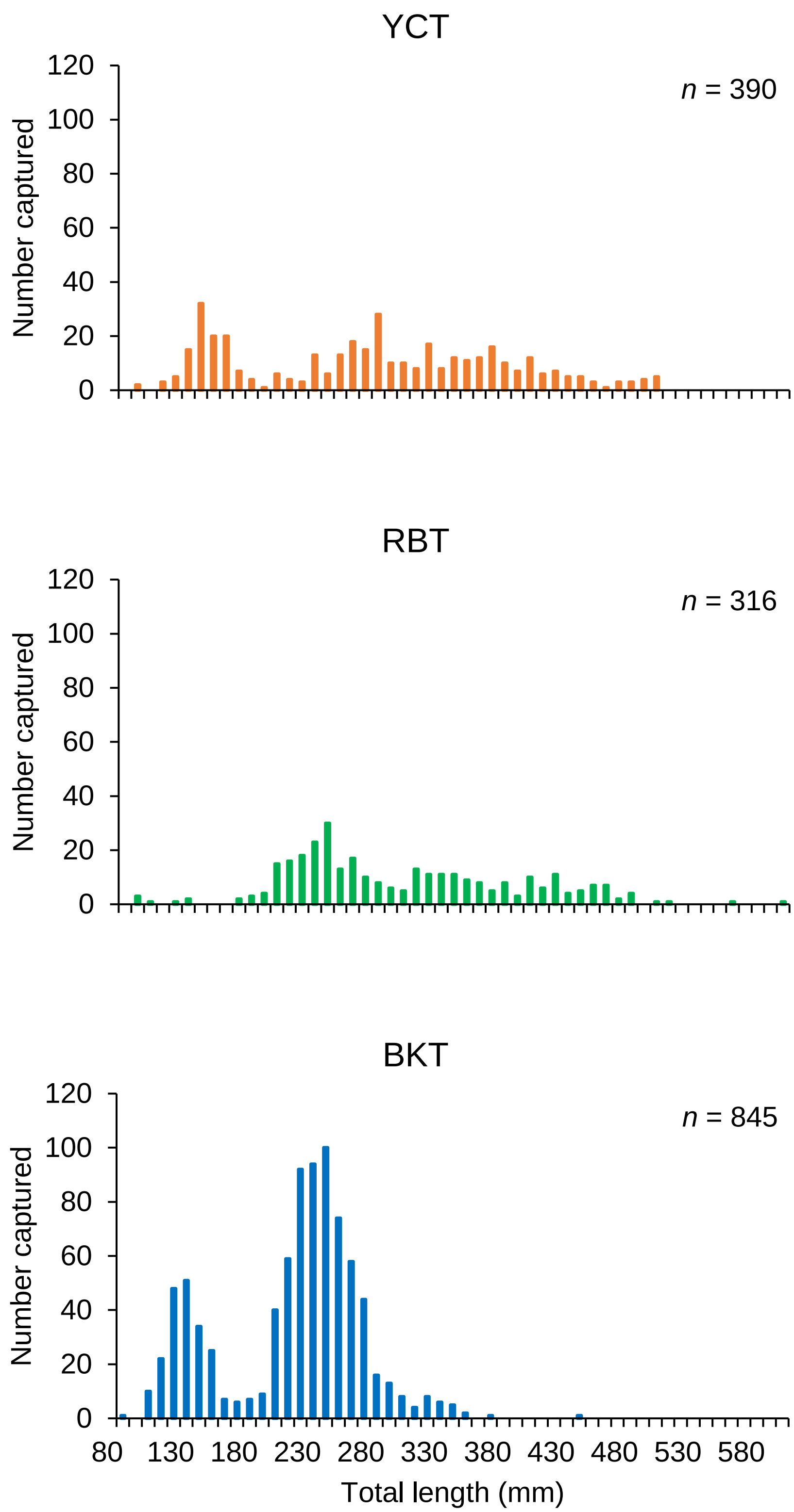

Figure 26. Length-frequency distributions for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brook Trout (BKT) at the Nickerson site of the Teton River, fall 2021.

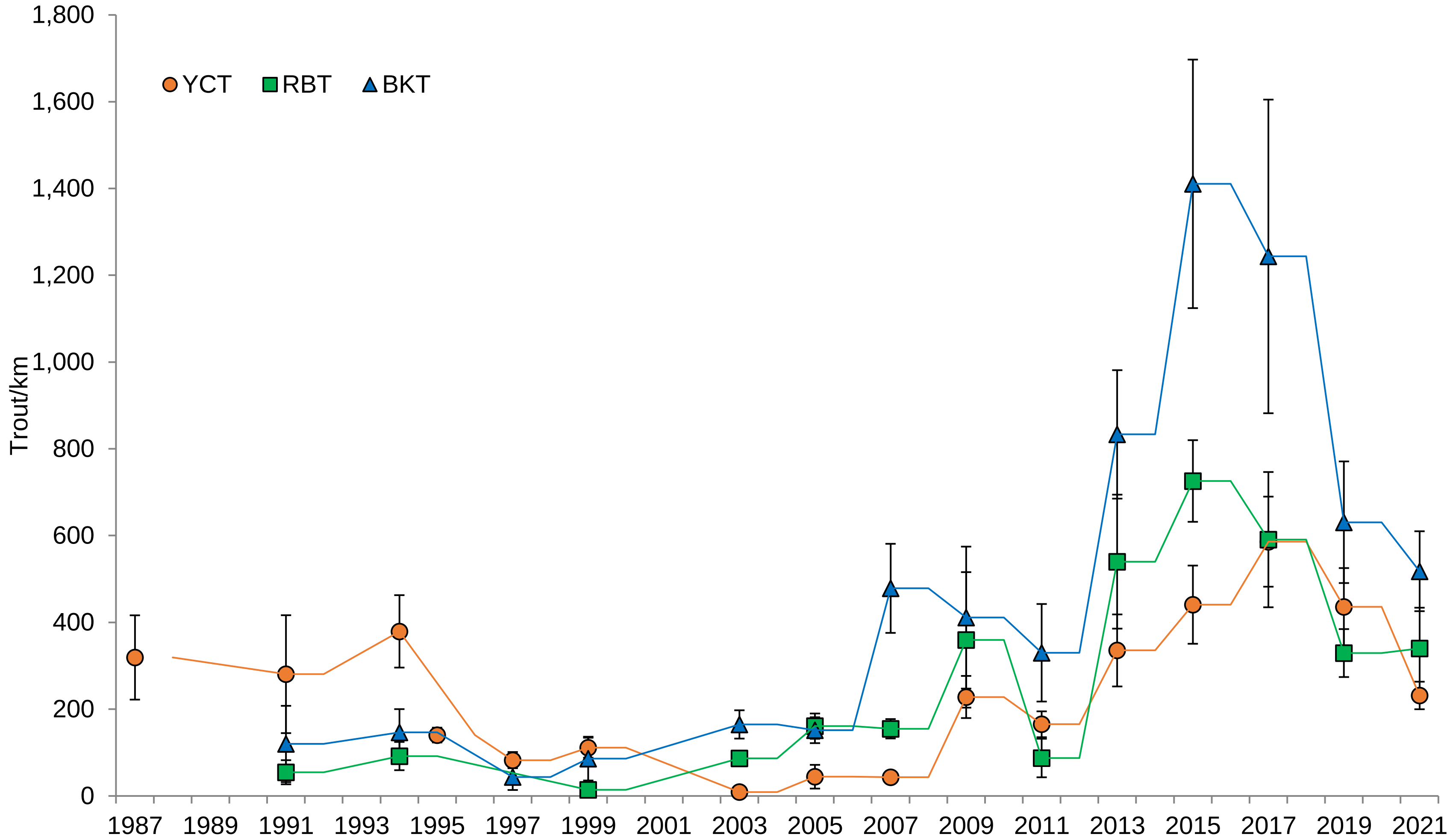

Figure 27. Estimated abundance (trout/km) with 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brook Trout (BKT) in the Teton River at the Nickerson site from 1987 to 2021.

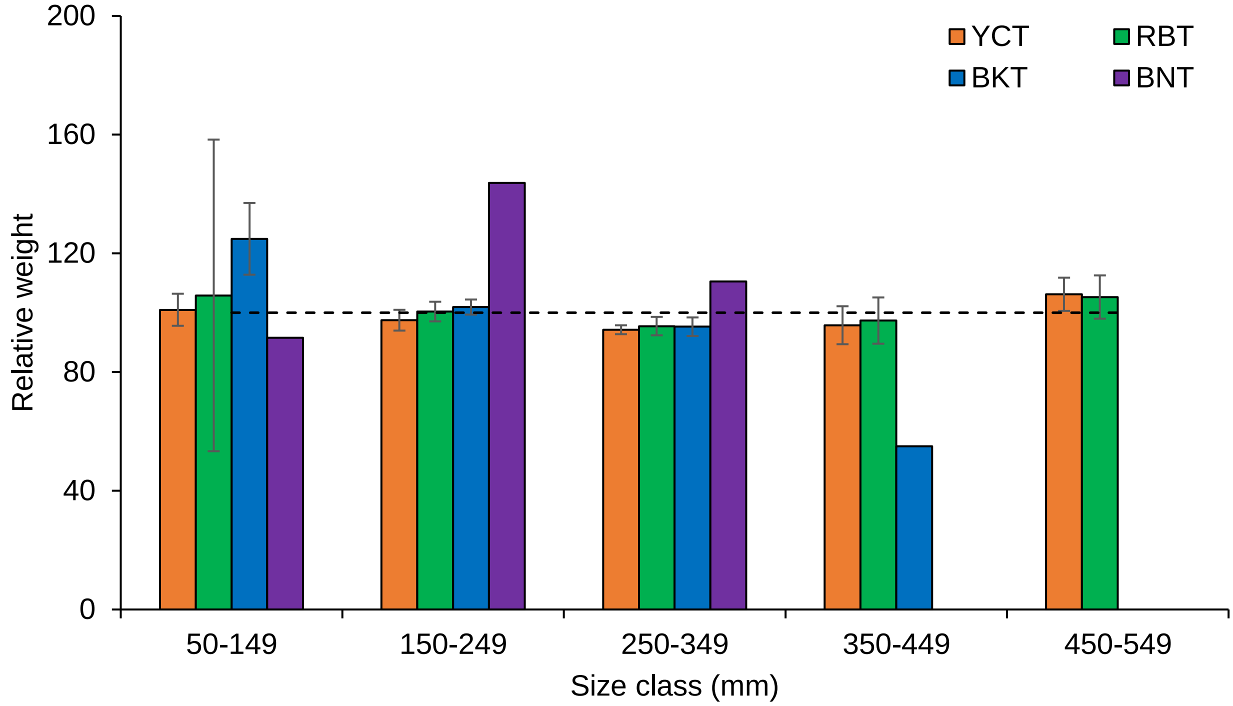

Figure 28. Mean relative weight and 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), Brook Trout (BKT), and Brown Trout (BNT) in the Teton River at the Nickerson site, 2021. Missing confidence intervals are attributed to low sample sizes. The dashed line is the standard for the species or W_r = 100.

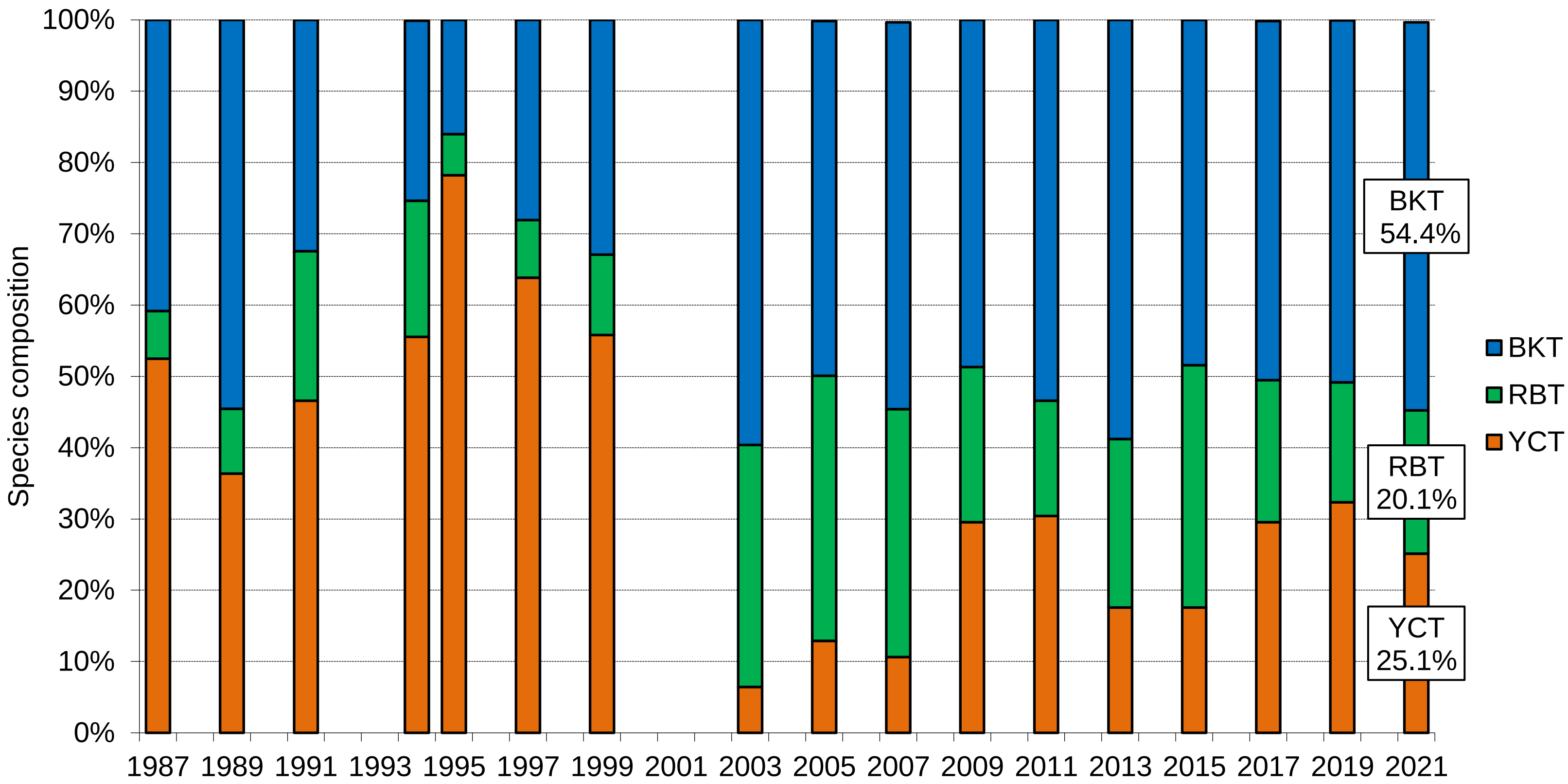

Figure 29. Species composition (%) of Brook Trout (BKT), Rainbow Trout (RBT), and Yellowstone Cutthroat Trout (YCT) from 1987 to 2021 in the Teton River at the Nickerson site. Brown Trout represented between 0 and 0.3% of the catch and were omitted from this figure.

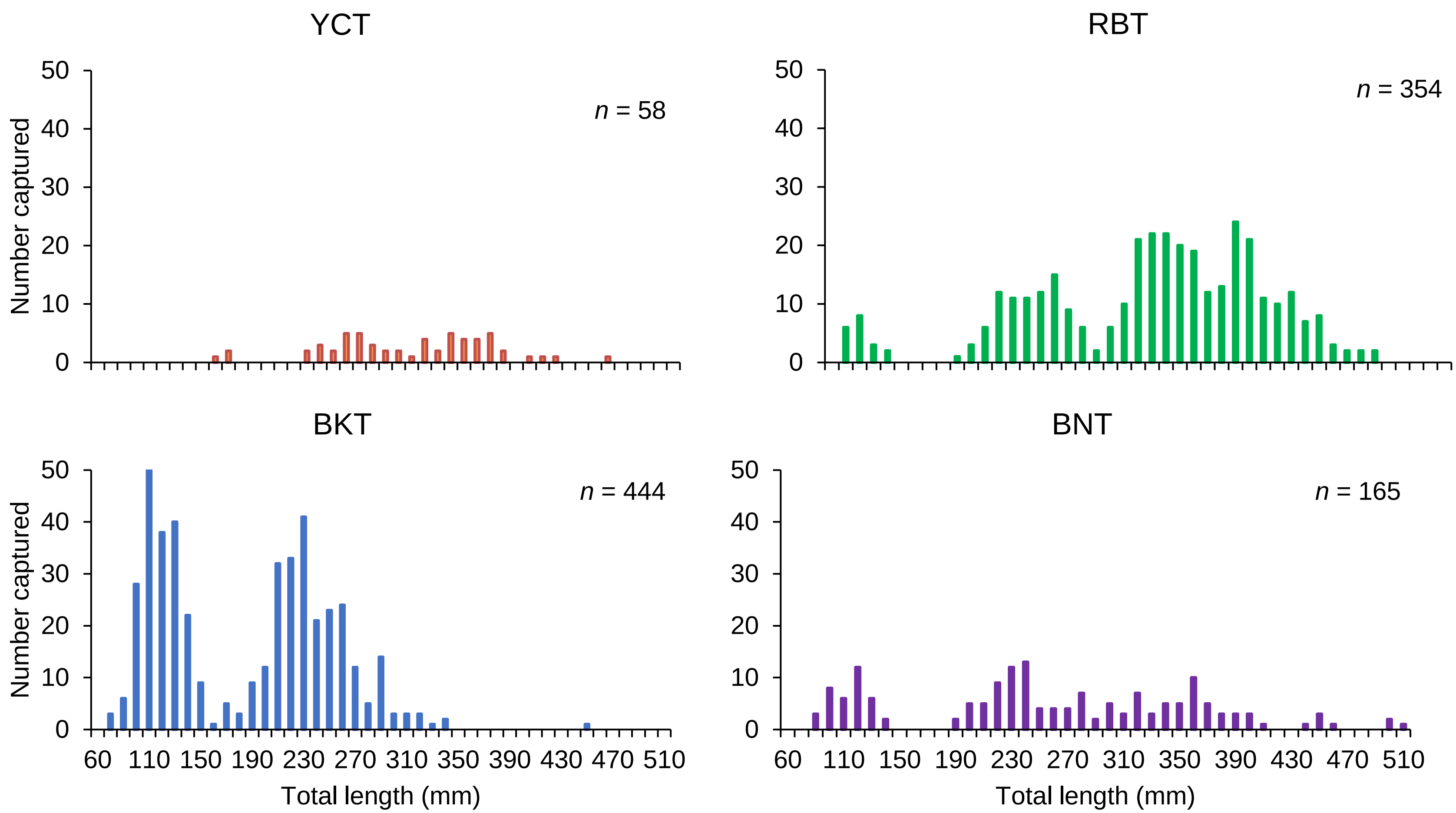

Figure 30. Length-frequency distributions for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), Brook Trout (BKT), and Brown Trout (BNT) at the Breckenridge site of the Teton River, fall 2021.

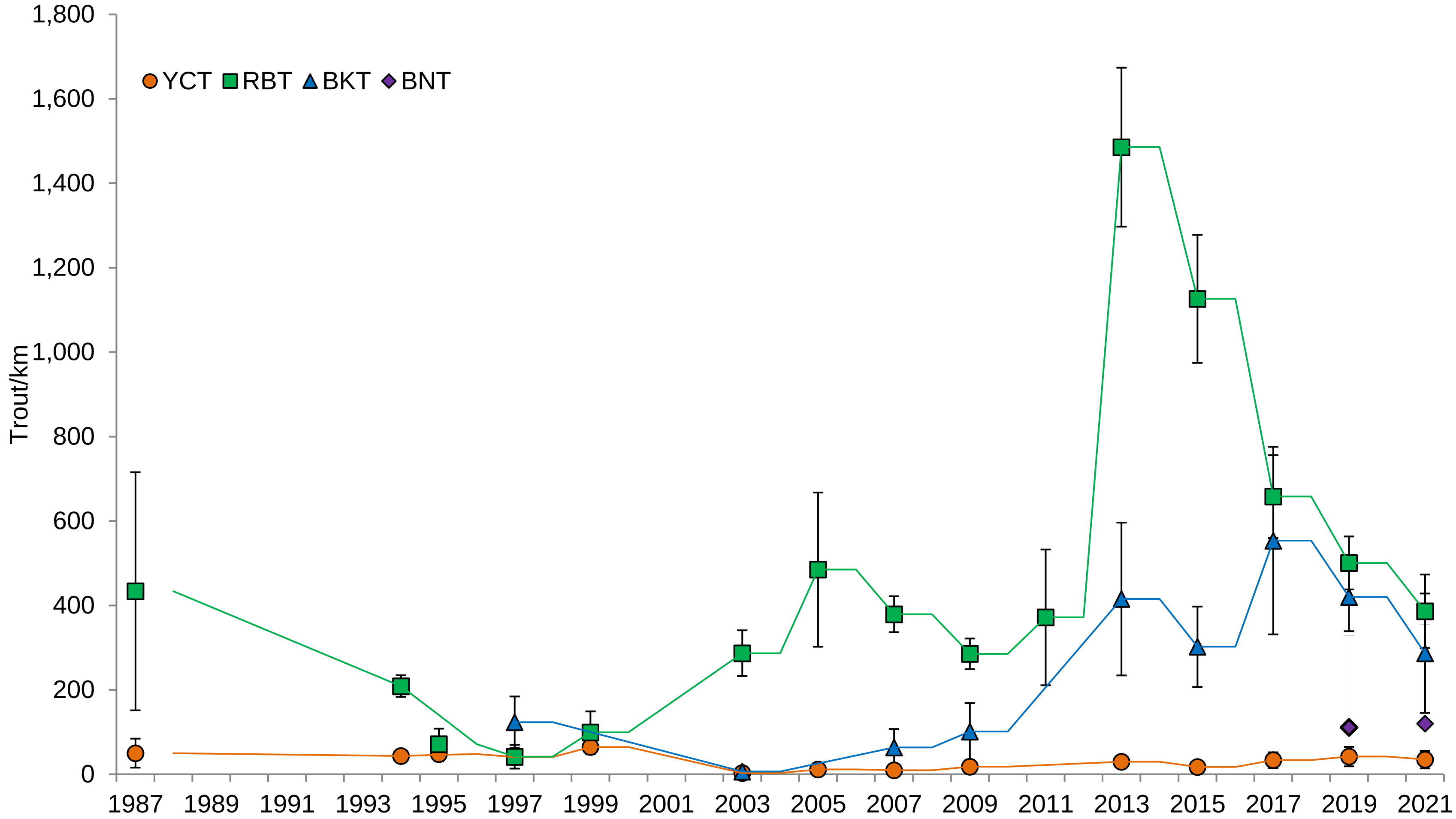

Figure 31. Estimated abundance (trout/km) with 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brook Trout (BKT) in the Teton River at the Breckenridge site from 1987 to 2021.

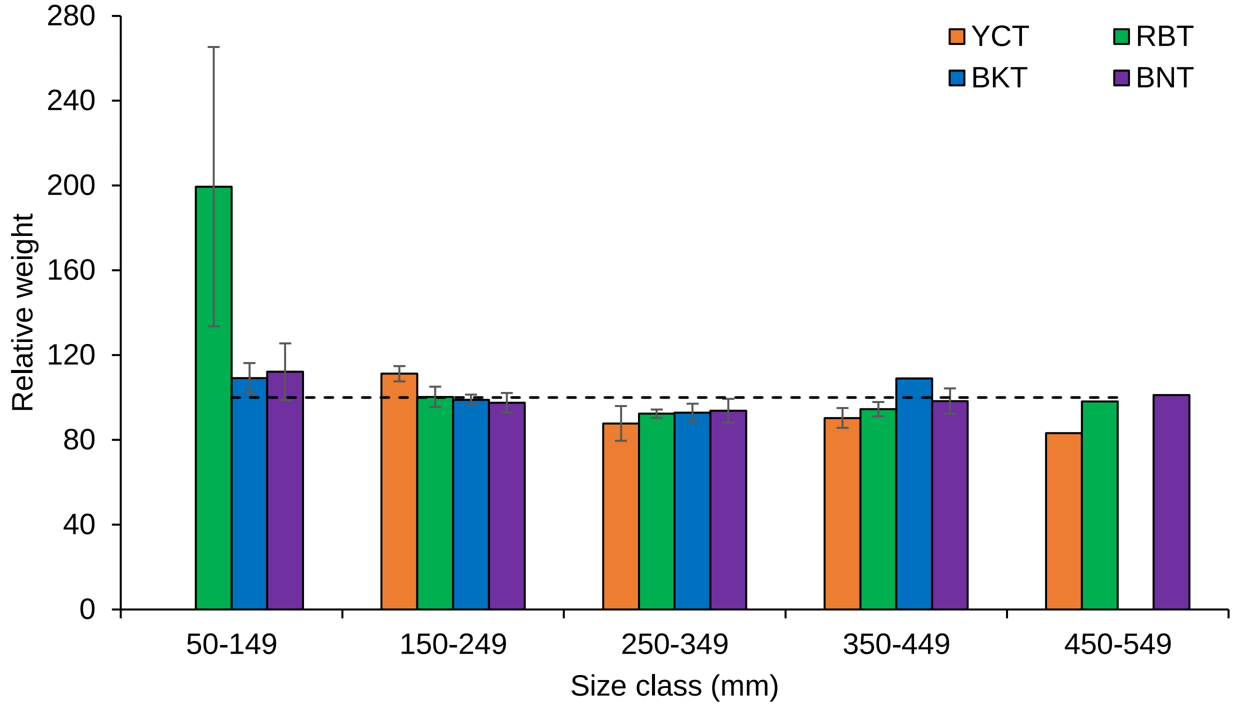

Figure 32. Mean relative weight and 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), Brook Trout (BKT), and Brown Trout (BNT) in the Teton River at the Breckenridge site, fall 2021. Missing confidence intervals are attributed to low sample sizes. The dashed line is the standard for the species or W_r = 100.

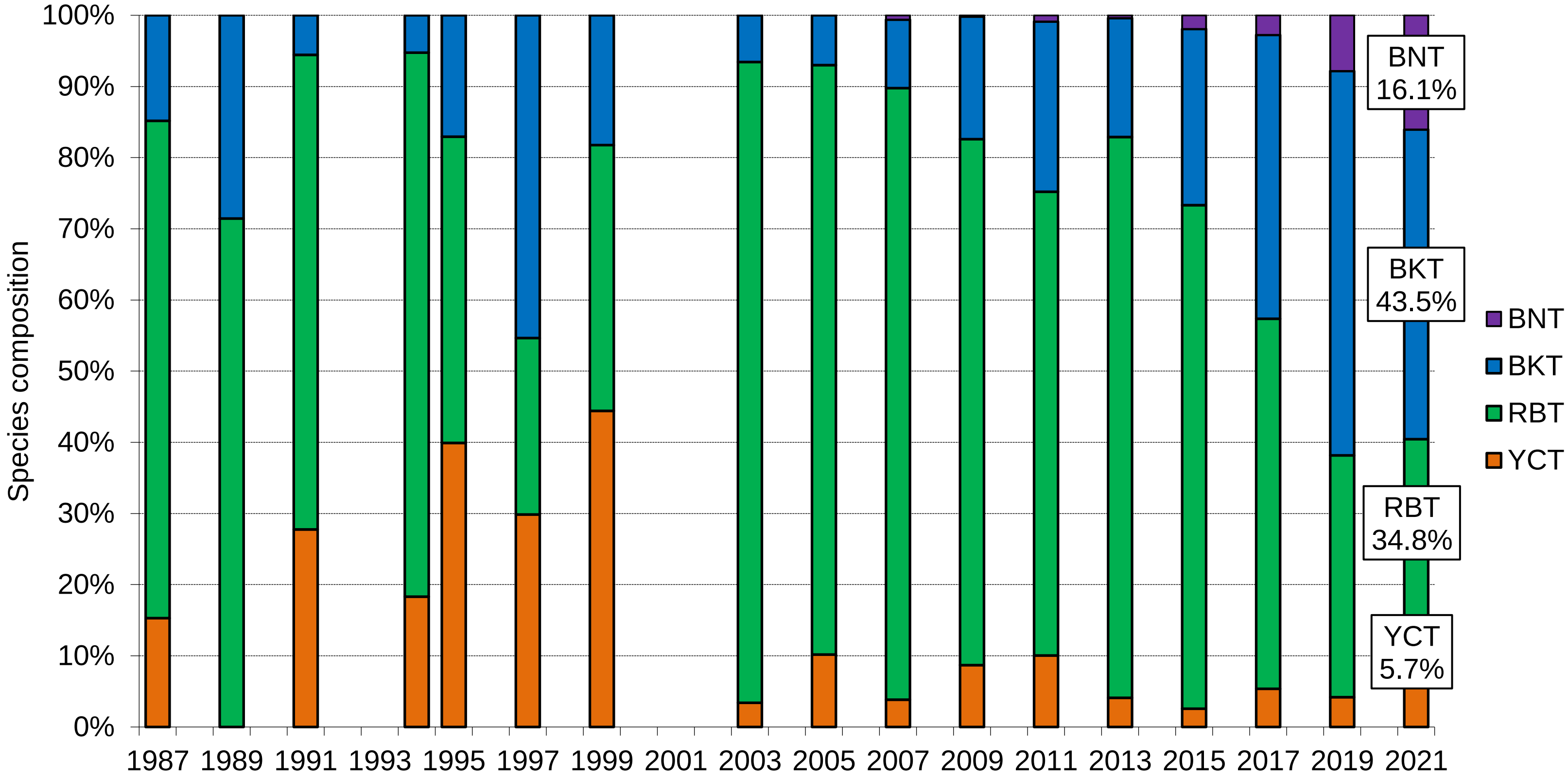

Figure 33. Species composition (%) of Brown Trout (BNT), Brook Trout (BKT), Rainbow Trout (RBT), and Yellowstone Cutthroat Trout (YCT) during each survey from 1987 to2021 in the Teton River at the Breckenridge site, 2021.

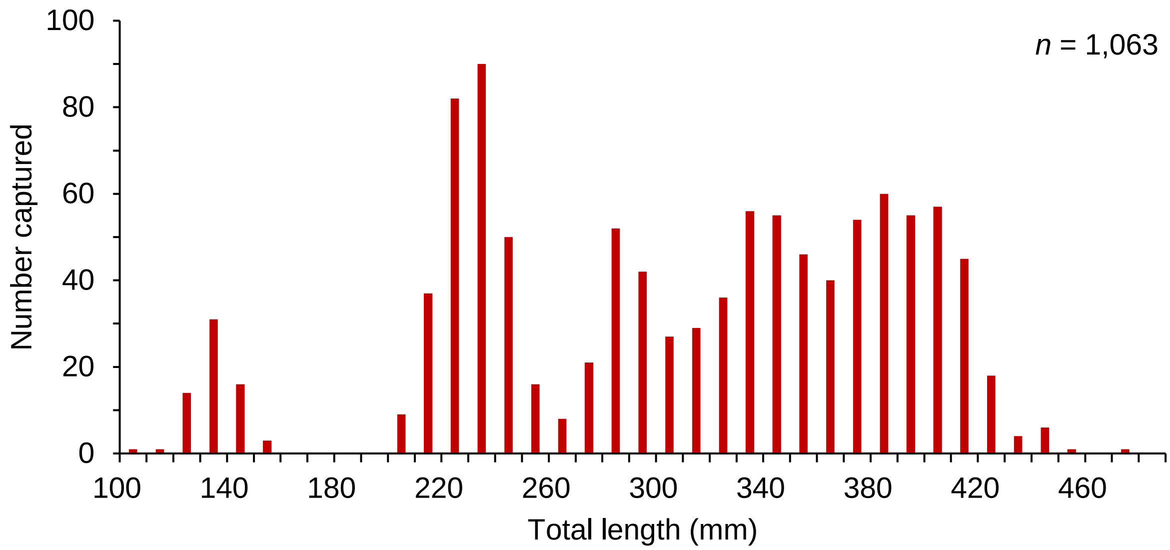

Figure 34. Length-frequency distribution of Mountain Whitefish at the Buxton site of the Teton River, 2021.

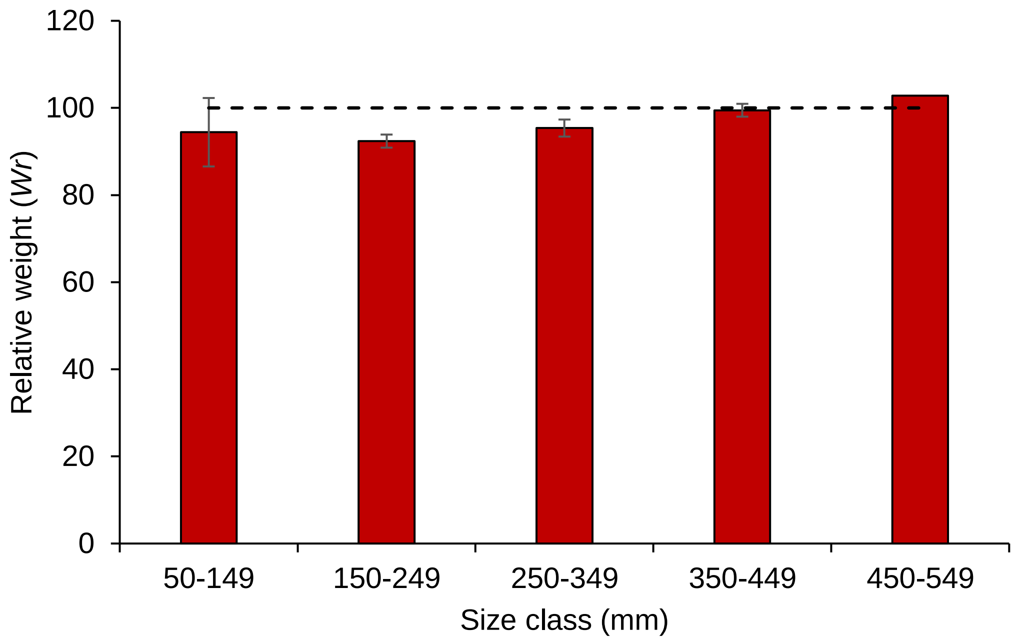

Figure 35. Mean relative weight and 95% confidence intervals of Mountain Whitefish in the Teton River at the Buxton site, 2021. Missing confidence intervals are attributed to low sample sizes. The dashed line is the standard for the species or W_r = 100.

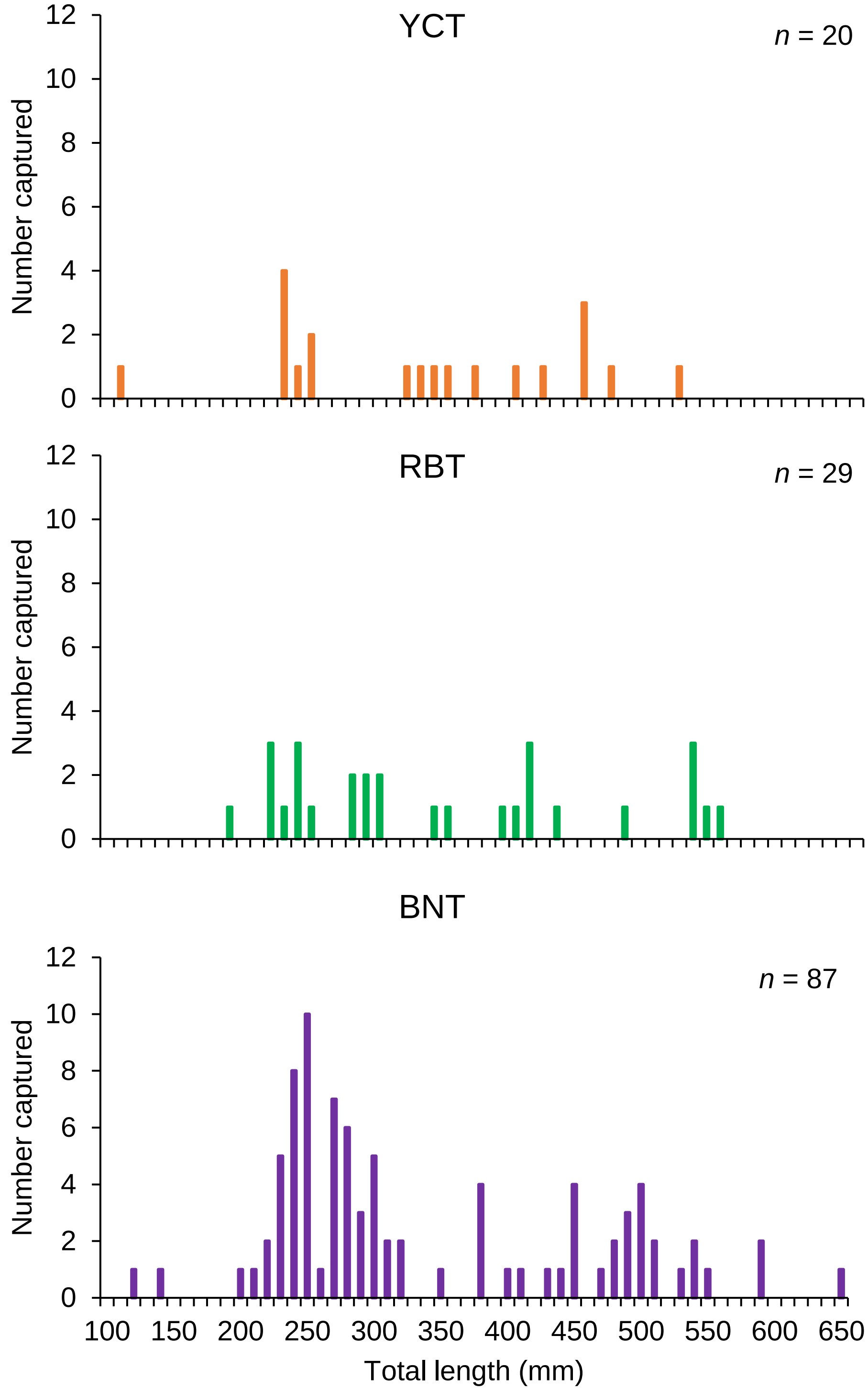

Figure 36. Length-frequency distributions for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) at the trout site of the South Fork Teton River, 2021.

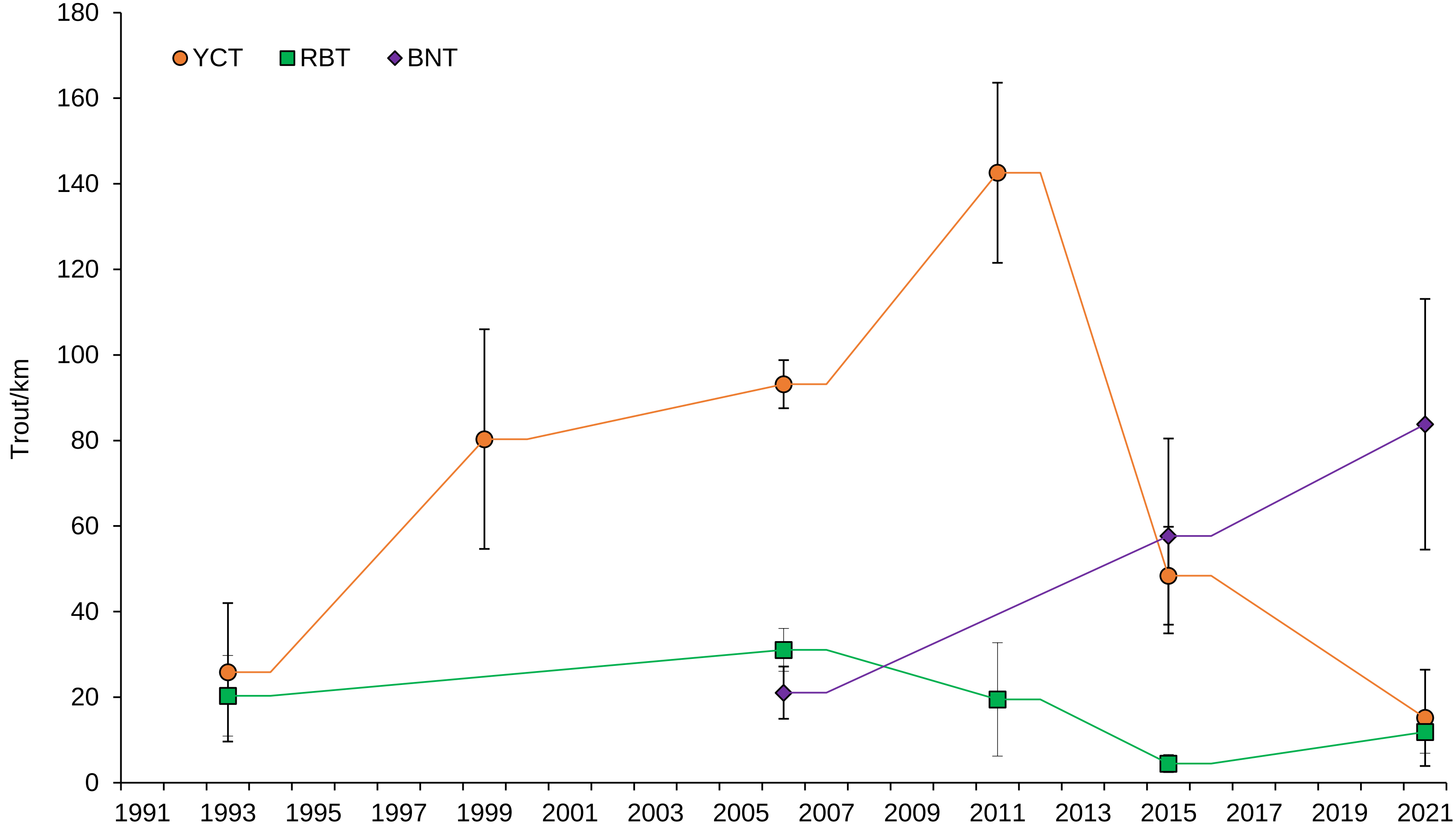

Figure 37. Estimated abundance (trout/km) with 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) in the South Fork Teton River at the trout site from 1993 to 2021.

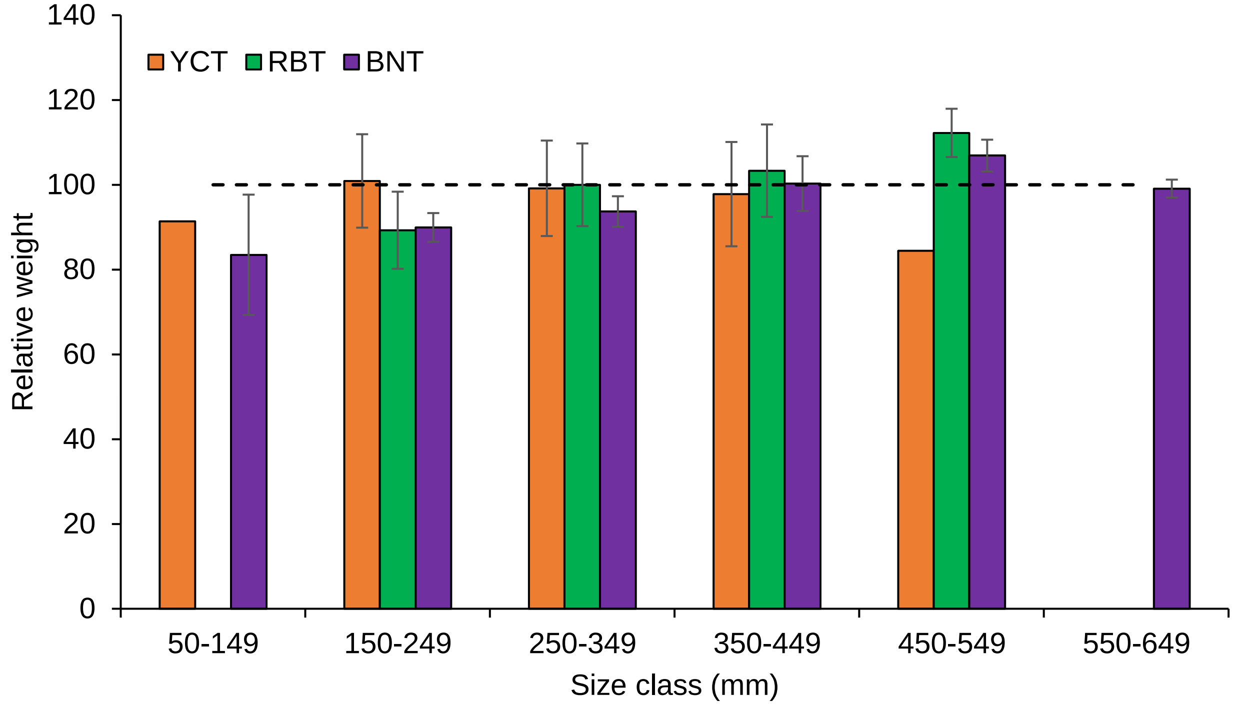

Figure 38. Mean relative weight and 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) in the South Fork Teton River at the trout site, 2021. Missing confidence intervals are attributed to low sample sizes. The dashed line is the standard for the species or W_r = 100.

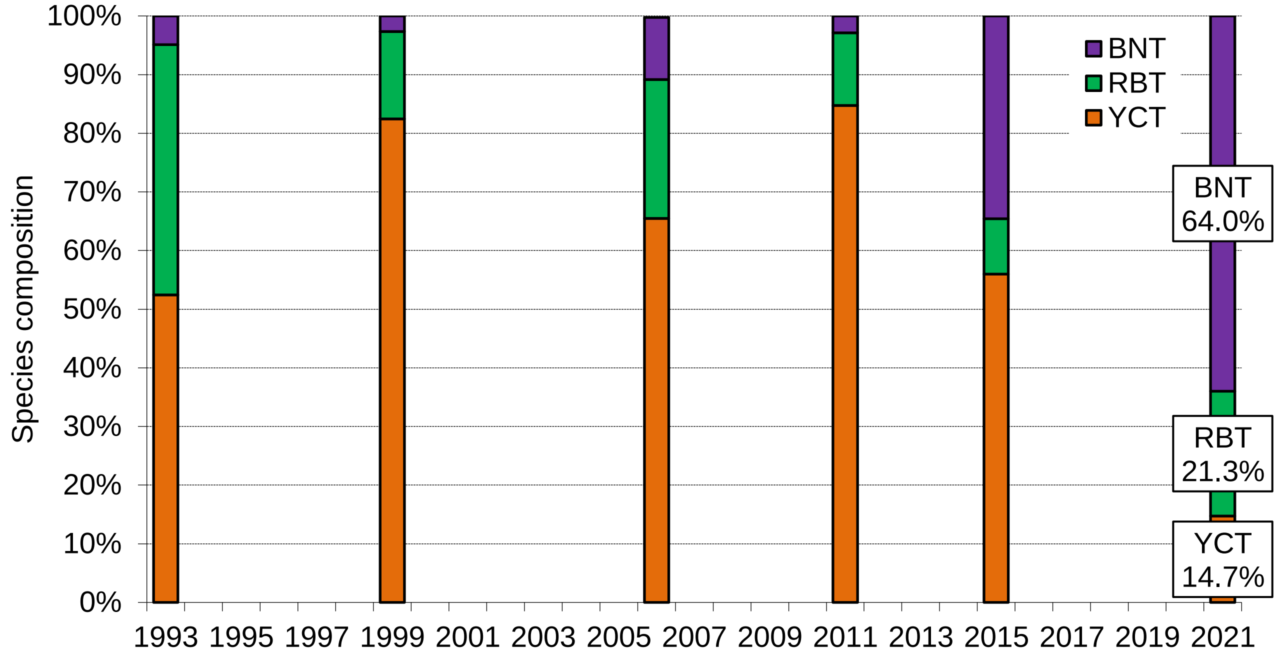

Figure 39. Species composition (%) of Brown Trout (BNT), Brook Trout (BKT), Rainbow Trout (RBT), and Yellowstone Cutthroat Trout (YCT) during each survey from 1993 to2021 in the South Fork Teton River at the trout site, 2021.

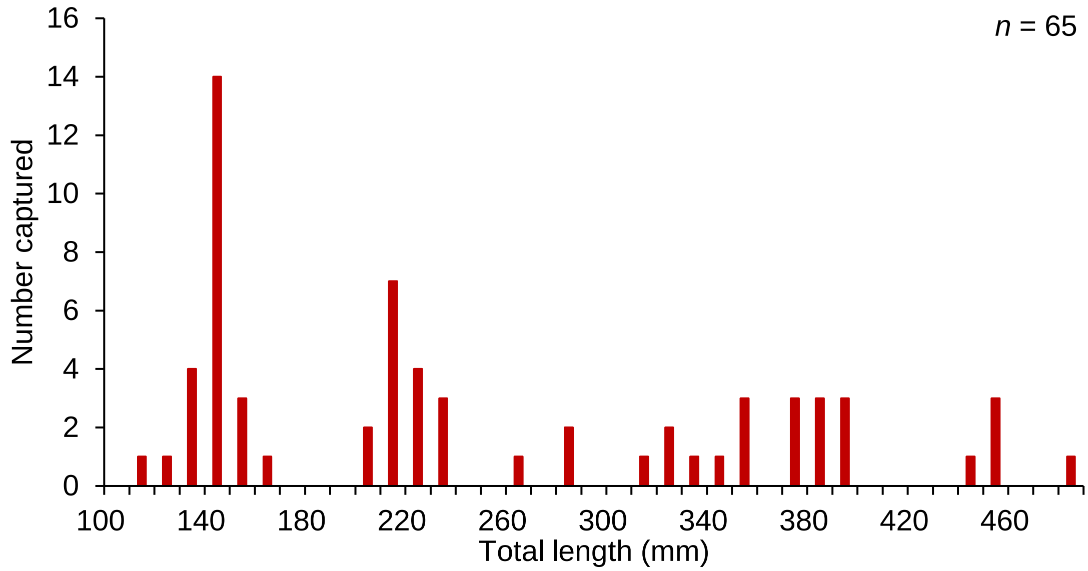

Figure 40. Length-frequency distribution for Mountain Whitefish at the Mountain Whitefish site of the South Fork Teton River, 2021.

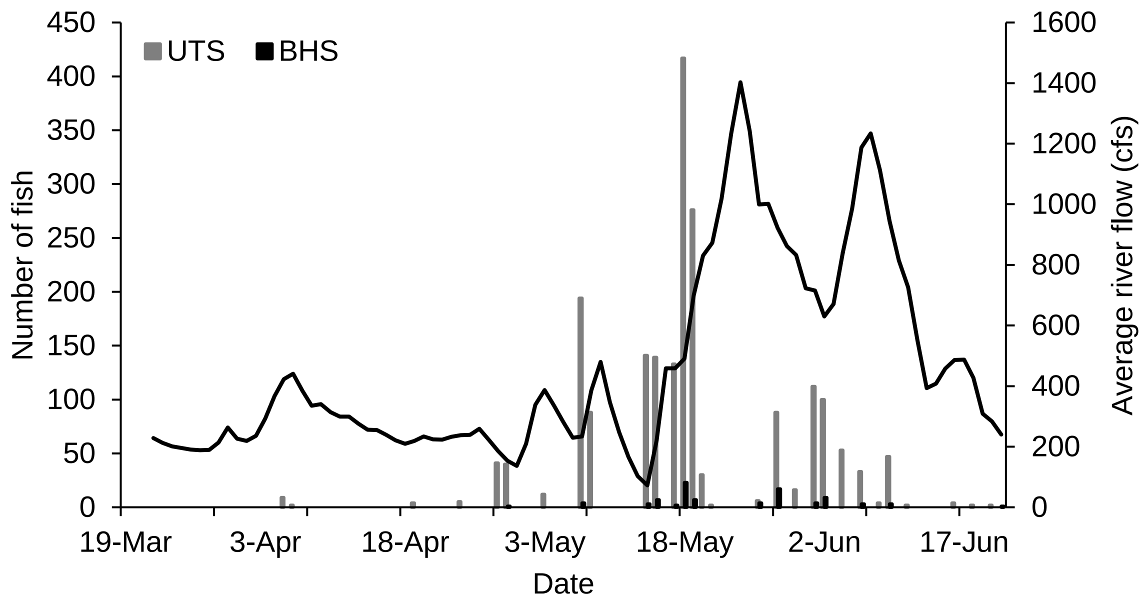

Figure 41. Number of Utah Sucker (UTS) and Bluehead Sucker (BHS) captured in the South Fork Teton Fish trap, 2021. The solid black line represents the daily average stream flow (cubic feet per second) for the South Fork Teton River at the USGS SF Teton River gauge.

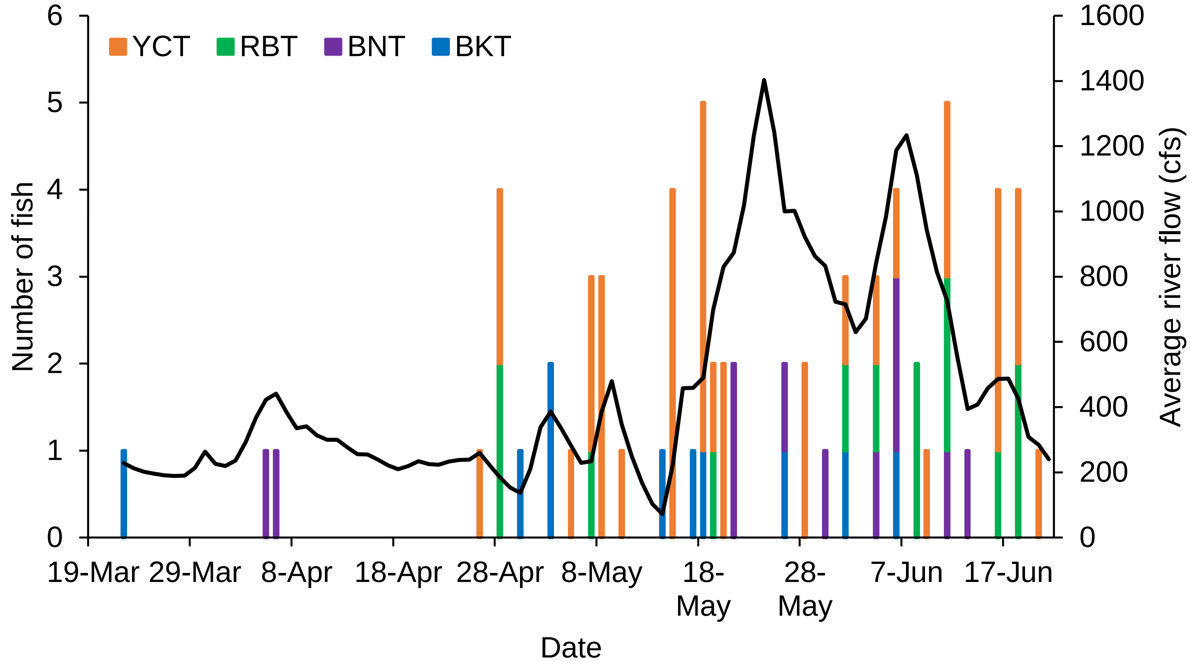

Figure 42. Number of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), Trout (BNT), and Brook Trout (BKT) captured in the South Fork Teton Fish trap, 2021. The solid black line represents the daily average stream flow (cubic feet per second) for the South Fork Teton River at the USGS SF Teton River gauge.

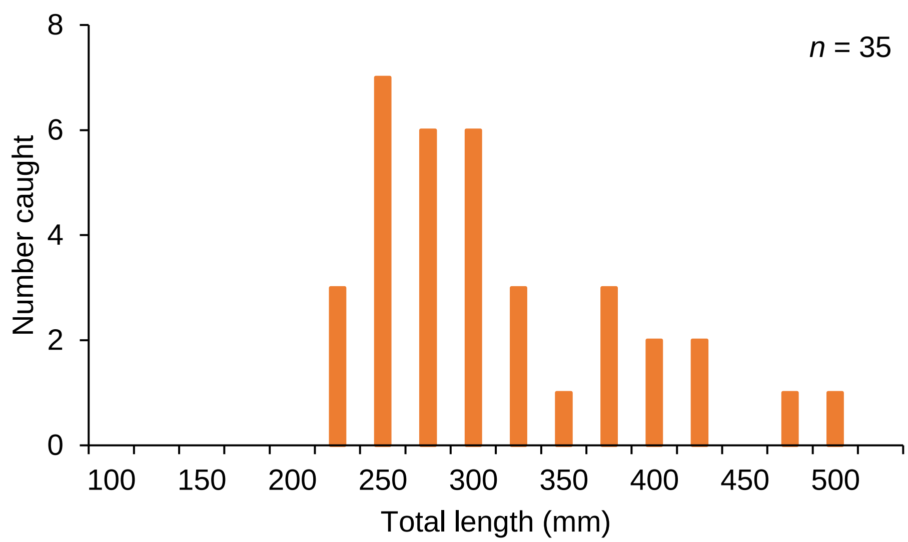

Figure 43. Length-frequency distribution of Yellowstone Cutthroat Trout at the South Fork Teton Fish trap, 2021.

BIG LOST RIVER

ABSTRACT

We electrofished four sites in the East Fork Big Lost River, one site in Wildhorse Creek, and one site in Star Hope Creek in 2020 to monitor trends in abundance and to collect fish for disease sampling. We also conducted a sentinel fish study at two sites in the East Fork Big Lost River, one site in Star Hope Creek, and one site in Wildhorse Creek. A number of fish (n = 50), which included samples from each sentinel fish site, tested positive for the parasite MC that causes whirling disease, and all samples (n = 124) that were tested for the parasite *Tetracapsuloides bryosalmonae* that causes proliferative kidney disease were negative. In April 2021, we electrofished at two long-term monitoring reaches (i.e., Campground and Leslie) in the Big Lost River downstream of Mackay Dam. We estimated the relative abundance of Rainbow Trout *Oncorhynchus mykiss* (RBT) in the Campground reach at 1,039 fish/km (95% CI ± 130.5) to serve as an index of population abundance, and we estimated Mountain Whitefish *Prosopium williamsoni* (MWF) abundance at 23 fish/km (± 5.3). In the Leslie reach, we estimated RBT abundance at 61 fish/km (± 6.5), Brook Trout *Salvelinus fontinalis* abundance at 184 fish/km (CI ± 123.5), and we did not capture enough MWF to estimate their abundance in that reach. In addition to estimating fish abundance, we tagged 238 RBT with T-bar anchor tags to evaluate exploitation and total angler use using the Idaho “Tag You’re It” angler reporting program. We estimated the reporting rate at 11% and angler use at 8%, which includes only fish that were reported as caught and released because no fish were reported as harvested. In July of 2021, we salvaged at the Moore and Blaine diversions and relocated 196 MWF, 470 RBT, 706 Brook Trout, and 3 kokanee *O. nerka* upstream into the Campground reach of the river.

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INTRODUCTION

The Big Lost River watershed is located in central Idaho, originating in the Copper Basin and eventually flowing southward to the sinks on the Idaho National Engineering Laboratory site (Figure 44). Climatic conditions in the watershed are relatively dry, with an average annual precipitation of about 25 cm. Approximately 40% of the precipitation occurs as snow. Because of the high scenic quality of the area, its numerous recreational opportunities, and its proximity to the resort area of Sun Valley, the Big Lost watershed receives a considerable amount of recreational use. Fishing is a popular recreational activity in the area with anglers spending thousands of errors fishing in the watershed each year (Corsi 1989; Heckel et al. 2020).
Numerous gamefish species are present in the watershed, including; Rainbow Trout *Oncorhynchus mykiss* (RBT), Yellowstone Cutthroat Trout *O. clarkii bouvieri* (YCT), Brook Trout *Salvelinus fontinalis* (BKT), Golden Trout *O. mykiss aquabonita*, Tiger Trout *Salmo trutta* × *S. fontinalis*, Arctic Grayling *Thymallus arcticus*, kokanee *O. nerka*, and Mountain Whitefish *Prosopium williamsoni* (MWF). Sculpin, including Piute *Cottus beldingi* and Shorthead Sculpin *C. confuses* also occupy various waterbodies in the watershed. Mountain Whitefish are believed to be the only salmonid native to the watershed and are recognized as the most genetically divergent population from other MWF present in the Pacific Northwest (Whiteley et al. 2006). Abundance of MWF has declined compared to the 1980s (IDFG 2007). Factors such as habitat alteration (e.g., channelization and impacts from grazing), irrigation (e.g., entrainment, barriers, dewatering, and changes in flow regime), non-native fish interactions (e.g., competition and predation), disease, and exploitation have all been identified as possible contributors to the decline in MWF. To address the decline in abundance and to expedite recovery efforts, the Idaho Department of Fish and Game (IDFG) developed the MWF Conservation and Management Plan for the Big Lost River Drainage, Idaho (IDFG 2007). The intent of this document is to ensure the MWF population in the Big Lost River drainage persists in response to natural and anthropogenic changes at levels capable of providing a recreational fishery. Specific population objectives are outlined, and management actions believed to be critical to the attainment of population objectives are identified.

A long history of fish stocking has occurred throughout the Big Lost watershed to provide more opportunities for anglers since MWF are the only native salmonid, but stocking downstream of Mackay Dam ceased in 2006. Upstream of Mackay Reservoir, stocking of RBT and YCT has continued, but anglers were reporting low catch rates of RBT throughout Copper Basin streams and abundance estimates conducted in 2018 indicated that RBT abundance had declined since the 1990s (Heckel et al. *in press*). In response to angler concerns of low catch rates, we conducted a creel and tagging study in 2019 (Heckel et al. 2020) to evaluate exploitation in Copper Basin. Our results indicated that exploitation was not having a population-level effect, so we then chose to investigate the extent of disease, which may be a limiting factor in the abundance of fish populations in Copper Basin. Regulations of the 2021 season prohibited MWF harvest, whereas the daily limit for trout is six from the Saturday of Memorial Day weekend through November 30, and the trout limit is zero for the rest of calendar year. Although fish stocking occurs annually in the watershed upstream of Mackay Dam, an assessment of angler use and exploitation has not occurred in the watershed downstream of Mackay Dam for wild trout since 2007 (Garren et al. 2009). Following previous studies (Corsi 1989; Garren et al. 2009; Heckel et al. 2020) conducted in the basin, we estimated angler use and harvest through a creel survey conducted downstream of Mackay Dam, and we estimated exploitation of wild trout using “Tag You’re It” (Meyer and Schill 2014).

OBJECTIVES

1. Monitor trends in abundance and collect fish for disease sampling in the Big Lost River upstream of Mackay Reservoir.

2. Replicate the sentinel fish study conducted by IDFG in 1996 (Elle 1997) to evaluate the presence of whirling disease and proliferative kidney disease in the Copper Basin area.

3. Evaluate RBT and MWF abundance in the Campground and Leslie reaches of the Big Lost River.

4. Conduct an age and growth evaluation of wild RBT in the Big Lost River downstream of Mackay Reservoir.

5. Estimate angler use and exploitation of wild trout in the Big Lost River downstream of Mackay Dam using the "Tag You're It" program.

METHODS

Sentinel Fish and Disease Sampling

Age-0 YCT and Hayspur-strain, triploid RBT were obtained from the Mackay fish hatchery and placed into traps that were located in four different streams within Copper Basin in June, July, August, and September 2020 (Figure 44). Traps were constructed of PVC and measured 15.24 cm × 60.96 cm with a cap that was glued to one end and a threaded cap on the opposite end so that it could be screwed shut and unscrewed. An eye bolt was drilled into one cap so that a rope could be tied to the trap and attached to a tree, bush, or T-post located on the bank of the trap site. Hundreds of 0.50-cm wide holes were drilled in rows along the long axis of the trap and in the two end caps to allow water flow through the trap once submerged in each stream. Two traps were set at each sentinel site; one trap contained 40 YCT and the other trap contained 40 RBT. Sentinel fish were submerged in the stream, a rope was attached to each eye bolt, and tied off to a tree, bush, or T-post on the bank nearby. Sentinel traps were placed either in pools or runs that we thought would have water flow and retain water depth from June through September so that multiple cohorts could be placed at each site. We chose four (June, July, August, and September) sentinel fish cohorts at four sites (East Fork Swamps, East Fork at Castle Rock, Wildhorse Creek, and Star Hope Creek) with each cohort consisting of 40 YCT and 40 RBT. Each cohort was collected at Mackay Fish Hatchery, wherein each site's fish (YCT were adipose fin clipped) were placed into high mountain lake stocking bags with spring water from the hatchery, filled with 100% oxygen, and then placed in coolers for transportation. At each site, fish were emptied into the traps while partially submerged in the stream water and the end cap was screwed shut. Traps were held at each site for ten days (Elle 1997). After ten days (Baldwin et al. 2000), we filled coolers with spring water from Mackay Hatchery to assure there was no contaminated water transfer from sentinel fish to the rearing facility. We emptied the sentinel fish into an aquarium dip net in which the water drained through. We then placed spring water into a pitcher that was cleaned with a 10% bleach solution and rinsed three times with spring water. Sentinel fish were then placed into the clean pitcher with spring water, poured into stocking bags, filled with 100% oxygen, and placed into a cooler. Each bag was labeled with the cohort number, site, and species. Five fish from each species by site were measured for total length (TL). In addition, Hobo Tidbit®

v2 temperature loggers were placed in one trap at each site for each cohort to monitor water temperature during the exposure.

Fish were then transferred to the Juvenile Corrections Facility in St. Anthony, ID where Benjamin Call operates a fish rearing facility within his biology classroom. We used parasite-free well water (13.25°C) that was recirculated through a flow-through system every 12 hours at about 15 L/min, and tanks were cleaned once a week. Each cohort and site was reared in separate 130-L, rectangular tanks. All YCT were adipose fin clipped to identify species and fish were reared for four months before they were processed as described for the wild fish disease sampling (Elle 1997). After the rearing period was complete, we then processed head wedges and kidney samples. Prior to processing each head wedge sample, we disinfected scalpels using a 10% bleach wipe followed by immersion in 100% propanol, and lastly a flame from an oxygenated torch. Head wedges were stored individually in sterile whirl-paks that were labeled with a unique identification number, the species, TL, location sampled, and date of sample, then frozen. Scalpels and forceps were sterilized in the same manner as described for head wedges prior to taking each kidney sample for proliferative kidney disease analysis. A section of posterior kidney about the size of a pea (e.g., < 25 mg) was removed and placed in a 1.5-ml tube filled with bio-grade ethanol (e.g., ≥ 95% ETOH concentration). Tubes were labeled with an ethanol resistant marker that included a unique identification number that corresponded to the data written on the whirl-pak. After fish processing, we sent head wedges and kidney samples to the IDFG Eagle Fish Health lab for pepsin-trypsin digest (PTD) followed by polymerase chain reaction (PCR) to confirm positive PTD samples for the presence of *Myxobolus cerebralis* (MC). Samples were pooled in groups of five for PCR. Each group consisted of YCT or RBT per site, per cohort so that we could evaluate the presence of the parasite by location, species, and temporally by cohort (Qureshi et al. 2002). The Eagle Fish Health Lab also used PCR confirmation to test for the presence of the parasite *Tetracapsuloides bryosalmonae*, which causes proliferative kidney disease (PKD).

In all stream reaches upstream of Mackay Reservoir we collected a number of wild fish from each reach near sentinel trap sites where we conducted an abundance estimate in 2020. We took fish (i.e., Brook Trout, Yellowstone Cutthroat Trout, and RBT) that we estimated to be age 1 (e.g., TL ≤ 200 mm) to extract head wedges and kidney samples. Dissecting instruments were cleaned and the same processing was used in the field as described for sentinel fish processing.

Population Monitoring

In the East Fork Big Lost River swamps and Burma reaches, Wildhorse Creek and Star Hope Creek reaches we used two backpack electrofishing units with two to four netters to sample the fish populations (Figure 44). For all backpack electrofishing we standardized output to 100 W using pulsed direct current (DC) with a 25% pulse width and a frequency of 60 Hz (Meyer et al. 2021). In the East Fork Big Lost River, Fox Creek, and Whitworth reaches we used a canoe electrofishing unit and four to six netters to sample the fish populations. We measured water temperature (°C) and conductivity (µS/cm) prior to active electrofishing using a handheld probe. For the canoe electrofishing, pulsed direct current power was provided by a 3200-W generator and standardized to 2,750-3,250 W of power output based on water conductivity (Miranda 2009). We applied electricity to the water using an Infinity model electrofisher (Midwest Lake Management, Inc., Polo, Missouri). Electrofishing began at the downstream point of the sampling reach and proceeded in an upstream direction. Reach lengths were between 100 and 300 m, and reaches began and ended at a riffle or other break in habitat types. We measured the wetted width at ten systematic transects in each reach at riffles, runs, and pools to capture the average wetted width for each reach. Three pass depletion estimates were conducted in each reach to estimate fish density and abundance.

In the Campground and Leslie reaches downstream of Mackay Dam, we used the same canoe electrofishing methods as described for the East Fork Big Lost River, but reach lengths were 1-km long each. We also measured the wetted width at ten systematic transects per reach following the same methods as described for the Copper Basin streams. Furthermore, we conducted single-pass marking and recapture runs in both reaches in which all fish were identified to species and measured for total length (TL) to the nearest mm and weighed using a digital scale to the nearest tenth of a gram. All trout and whitefish encountered that were ≥ 150 mm were marked with a hole punch in the caudal fin prior to release. The marking run was conducted on one day and the recapture run was conducted two days later. Because we were uncertain on what proportion of the population was entrained, triploid RBT from Mackay Reservoir versus wild RBT, we removed a small clip from the anal fin for genetic testing at the IDFG Eagle Fish Genetics Lab.

We kept ten RBT per 10-mm length group for age and growth analysis. We extracted sagittal otoliths for assessing age, growth, natural mortality, and survival estimates. We recorded TL and weight (g) of all RBT kept for age and growth analysis. Otoliths were sectioned across the transverse plain following standard techniques (Koch and Quist 2007; Long and Grabowski 2017), and used a compound microscope in conjunction with imaging software (LAS 4.6; Leica; Buffalo Grove, IL) to take digital images of each otolith. We used ImageJ 1.53e (National Institutes of Health, USA) software to measure annuli growth increments, and then we estimated back-calculated length-at-age using the Dahl-Lea method (Shoup and Michaletz 2017). Growth was evaluated with a von Bertalanffy growth model (Ogle et al. 2017). We built an age-length key from our subsample of RBT and applied the key to our unaged sample. Then, we used a Chapman-Robson estimator (Chapman and Robson 1960) to estimate survival rate (S), instantaneous total mortality rate (Z), and actual total mortality rate (A). All analyses were conducted using the FSA package (Ogle et al. 2021) in program R (R Core Team 2021).

Fish Tagging

We randomly tagged RBT that were sampled in the recapture run in April 2022 in the Campground and Leslie reaches with non-reward, T-bar anchor tags at the base of the dorsal fin according to standard methods (Dell 1968). Tags were printed with a unique identification number, phone number, and website address where anglers could report the tag using the “Tag You’re It” statewide tag reporting system (Meyer and Schill 2014). We used data obtained from reported tags to estimate exploitation, caught-and-released fish, and total angler use. We used the abundance estimate of RBT from the 2018 sampling (Heckel et al. *in press*) in the Campground and Leslie reaches to estimate what 20% of the abundance was to choose how many RBT to tag in the recapture runs of both reaches.

Data Analysis

We estimated abundance and 95% CI in reaches upstream of Mackay Reservoir using a method for k-pass depletion estimates (Carle and Strub 1978) using the removal() function in the FSA package in Program R (R Core Team 2022). We then calculated density as the number of fish per 100 m^2 for reaches upstream of Mackay reservoir.

For the Campground and Leslie reaches, we estimated abundance for all salmonids ≥ 150 mm using a Peterson estimator with a Chapman modification using the mrClosed() function in the

FSA package in program R (R Core Team 2022). We adjusted for size selectivity by distributing the catch into size classes to fit the recapture data, and we then calculated 95% CI for the abundance estimates. We evaluated species composition in the reach and CPUE for each species on the marking run as the number of fish caught per minute of active electrofishing. We estimated capture efficiency as the number of marked fish that were recaptured divided by the total number of fish that were available for capture multiplied by 100 to report that value as a percent (Chiaramonte et al. 2020). After calculating abundance estimates, we estimated trends in the RBT and MWF population growth for the Campground reach using an exponential model and the intrinsic rate of population change (*r*) as described by Maxell (1999) using α = 0.10. We estimated the average TL and SD of RBT, BKT, and MWF. In addition, fish condition was evaluated by calculating mean relative weight (W_r) and plotting against TL. We used the standard weight (W_s) equation

$$log_{10}(W_s) = -5.023 + 3.024 * log_{10}(L),$$

for RBT (Simpkins and Hubert 1996) where *L* is the total length, then used the equation

$$W_r = \left(\frac{W}{W_s}\right) * 100,$$

where W_r is relative weight and *W* is measured weight. We used the same equations for MWF (i.e., intercept = -5.086 and slope = 3.036; Rogers et al. 1996).

We calculated proportional size distribution (PSD) for RBT as the number of fish ≥ 300 mm divided by the number of fish ≥ 200 multiplied by 100. Similarly, relative stock densities (RSD-400 and RSD-500) used the same formula, with the numerator replaced by the number of fish ≥ 400 and 500 mm (Anderson and Neumann 1996; Neumann et al. 2012). We also investigated the relationship between winter discharge from Mackay Dam and stocking in Mackay Reservoir to the abundance of RBT in the population estimates downstream of Mackay Dam using linear regression, and we tested the linear relationship for significance using ANOVA in Program R (R Core Team 2022).

We used the angler reporting rate (λ) of 53.8% (Meyer et al 2012) for wild RBT non-reward tagged fish and a reporting rate of 88.4% for \$50 reward tagged fish to estimate exploitation. We estimated angler exploitation (u') using the equation:

$$u' = \frac{u}{\lambda\,(1-Tagl)(1-Tagm)},$$

where *u was* the number of non-reward tagged fish that were reported as harvested divided by the total number of non-reward tagged fish stocked, $Tagl$ was the first year tag loss rate (i.e., 0.088), and $Tagm$ was the tagging mortality rate (i.e., 0.01). We used the tag loss and tagging mortality as reported by Meyer and Schill (2014). We also estimated angler use by modifying *u* to also include fish reported as caught and released.

RESULTS

Sentinel Fish and Disease Sampling

All sentinel fish sites tested positive for the parasite that causes whirling disease and no sites tested positive for the parasite that causes PKD (Table 14). Rainbow Trout tested positive at greater proportions than YCT, and wild BKT tested positive that were sampled from Star Hope Creek. Three of four cohorts tested positive for (MC). Only one RBT displayed visible signs of whirling disease, which was from cohort one at the East Fork Big Lost River Swamps. No other sentinel fish or wild fish displayed physical signs of whirling disease.

Population Monitoring

We estimated the abundance and density of all species of sportfish that we encountered in the reaches upstream of Mackay Reservoir, which included RBT, BKT, YCT, and MWF (Table 15; Table 16). Brook Trout were present in all reaches, followed by YCT in 67% of reaches, RBT in 50%, and MWF in 33%, respectively. All RBT sampled in the Burma reach of the East Fork Big Lost River were hatchery-origin fish, whereas other reaches included predominantly wild-origin fish, but all fish were grouped by species.

In the Campground reach, we marked 523 RBT, 17 MWF, and four BKT on April 6. On April 8 during the recapture run, we captured 449 RBT (253 recaptures), 11 MWF (eight recaptures), and three BKT (0 recaptures). We estimated 1,039 RBT/km ± 130.5 (i.e., 5 RBT/100 m^2) and 23 MWF/km ± 5.3 (i.e., 0.1 MWF/100 m^2; Figure 45), but we did not recapture any BKT and could not estimate their abundance. Catch rate (CPUE) and capture efficiency for RBT were 6.50 fish/min and 48%, and for MWF were 0.16 fish/min and 47%, respectively. We did estimate a significantly negative change, suggesting that the population abundance is decreasing over time, in the intrinsic rate of population growth for RBT since 1987 in the Campground reach ($F_{1,5}$ = 6.252, P = 0.055), and the MWF population has been declining since the 1991 abundance estimate ($F_{1,4}$ = 15.67, P = 0.017). We estimated the average TL of all RBT in the Campground reach at 317 mm (± 150), but excluding RBT ≤ 150 mm the average TL was 403 mm (± 16; Figure 46). The proportional size distribution of RBT was 94, RSD-400 was 74, and RSD-500 was 1. We estimated the average W_r of RBT at 91 (SD ± 11; Figure 47). Using the Von Bertalanffy growth function for RBT, we estimated L_∞ = 557.5, K = 0.39, and t_0 = 0.45. Mountain Whitefish average TL was 378 mm (± 13) and BKT average TL was 233 mm (± 57).

We did not estimate a significant (α = 0.10) relationship between the number of catchable-sized RBT stocked into Mackay Reservoir the year prior to the abundance estimates ($F_{1,5}$ = 1.04, P = 0.36, r^2 = 0.10; Figure 48), but we did estimate a significant relationship in the number of fingerlings stocked into Mackay Reservoir two years prior to the abundance estimate ($F_{1,5}$ = 4.31, P = 0.07, r^2 = 0.52; Figure 49), and in the average winter flow from Dec. 1 to Feb. 28 two years prior to the abundance estimate ($F_{1,5}$ = 5.08, P = 0.07, r^2 = 0.41; Figure 50). All linear relationships included all abundance estimates in the Campground reach from 1987 to 2021 (n = 7). The IDFG Eagle Fish Genetics Lab found that all (n = 70) RBT samples that we provided tested as diploid (2N), thus we conclude these were not hatchery-origin fish.

In the Leslie reach, we marked 43 RBT, 0 MWF, and 32 BKT on April 13. On April 15 during the recapture run, we captured 41 RBT (29 recaptures), 0 MWF, and 27 BKT (4 recaptures). We estimated 61 RBT/km ± 3 (i.e., 0.6 RBT/100 m^2; Figure 51), 184 BKT/km ± 63 (i.e., 1.84 BKT/100 m^2), and we did not capture any MWF in this reach. Fish were marked on April 13 and the recapture run was conducted on April 15. Catch rate (CPUE) and capture efficiency for RBT were 0.57 fish/min and 67%, and for BKT were 2.14 fish/min and 13%, respectively. The average TL of RBT in Leslie reach was 242 mm ± 150 (Figure 52) and BKT average TL was 111 ± 28. The proportional size distribution of RBT was 84, RSD-400 was 61, and RSD-500 was 2, and for BKT was 38, 0, and 0. We estimated the average W_r of RBT at 86 (SD ± 10; Figure 53) and the average W_r of BKT was 78 ± 10 (Figure 54). Using the Von Bertalanffy growth function to describe RBT growth, we estimated L_∞ = 557, K = 0.39, and t0 = 0.45. We estimated S at 35%, Z at 1.04, and A at 65% (Figure 55).

We salvaged 470 RBT, 706 BKT, 196 MWF, and 3 kokanee from pools in the main stem Big Lost River near the Moore and Blaine diversions upstream to the Campground reach on July 28 and 29, 2021 after the Big Lost Irrigation District diverted all water into irrigation canals. We were able to measure TL for 84 MWF sampled from downstream of the Blaine diversion (Figure 56). The average TL was 256 mm ± 105 with multiple size classes present in the salvage sample.

Fish Tagging

We tagged 200 RBT with non-reward, T-bar anchor tags in the Campground reach, which was 22% of the abundance estimate in 2018. In the Leslie reach, we tagged 38 RBT, which was 14% of the abundance estimate in 2018. Only six tags have been reported (i.e., 3%), and all fish were reported as released. Therefore, there was no estimate available for exploitation of fish based on tag returns. We estimated the total angler use at 5%, which includes the estimated tagging loss and tagging mortality.

DISCUSSION

Sentinel Fish and Disease Sampling

The parasite MC that causes whirling disease was detected in all three streams (East Fork Big Lost River, Star Hope Creek, and Wildhorse Creek) where sentinel fish were located and in one stream (Star Hope Creek) where wild fish were sampled. We confirmed that MC was still present in the East Fork Big Lost, which is where the first investigations into the presence of the parasite were previously conducted (Elle 1997). Based on these findings, we assume MC is present throughout Copper Basin and the Upper Big Lost River. The presence of MC represents a limiting factor to wild RBT and MWF (Americus et al. 2021) populations in the Big Lost River upstream of Mackay Reservoir. In laboratory study, MWF mortality increased up to 66% in the first week post-hatch if infected with MC compared to uninfected MWF (Americus et al. 2021). The implications of mortality rates for MWF if infected with MC and additional losses associated with entrainment into irrigation canals (Kennedy 2009) suggests that these two factors could be major reasons for the declining abundance of MWF in the Big Lost River watershed. We recommend continuing fish salvage efforts in the river upstream of Mackay Reservoir to reduce losses associated with irrigation. However, there is little that can be done to reduce the MC infection rates. By limiting the influence of entrainment to the extent possible by salvaging MWF, we hope to limit, halt, or reverse further declines in MWF abundance.

Results from our fish health investigations suggest that a higher proportion of RBT became infected with the parasite compared to YCT, and we found no positive MC results in wild YCT. This is concordant with other studies comparing infection rates between these species (Thompson et al. 1999; Sipher and Bergersen 2005). Following our creel and tagging investigations (Heckel at al. 2020), which indicated low exploitation in Copper Basin streams, the presence of MC may be a major limiting factor contributing to lower wild RBT abundance in the basin in recent years (Heckel et al. *in press*). Continuing to stock hatchery-catchable RBT, particularly in the East Fork Big Lost River, will provide opportunities for harvest anglers while we continue stocking YCT fingerlings and catchable trout to help support wild YCT populations in the basin. In the results of our creel and tagging study, some anglers commented on the “poor fin condition” of RBT in the East Fork Big Lost River. If possible, stocking fingerling RBT in the East

Fork Big Lost River could provide anglers with hatchery-origin fish that have a better aesthetic condition as they grow than catchable-sized hatchery fish, and they could contribute to the overall RBT population during a longer timeframe than the average lifespan of a hatchery-catchable RBT (High and Meyer 2009).

Population monitoring

We estimated that there has been a significantly declining trend in RBT abundance in the Campground reach, but abundance has increased since 2018. We investigated what factors may be limiting abundance using linear regression to better understand the fishery, and we did learn that some factors may be contributing to the population dynamics of wild RBT in the Big Lost River downstream of Mackay Dam. We have ceased stocking fry and fingerling RBT in Mackay Reservoir, which was strongly correlated ($r^2 = 0.52$) to the abundance estimates, and statistically significant at $\alpha = 0.10$, in the Campground reach suggesting that a proportion of those fish could have become entrained into the tailrace fishery and contributed to the population in our abundance estimates. For example, the two lowest abundance estimates in our record (2018, 2021) were in years after no fingerling stocking was conducted in Mackay Reservoir, but we do not have genetics data from those sampling events to confirm whether this is true. It's possible that fingerling trout stocking may have subsidized the tailrace population. However, large differences in annual reservoir stocking totals prior to 2018 didn't necessarily correlate with large population changes, so the effect of stocking remains uncertain (Figure 49).

There was also a correlation ($r^2 = 0.41$) between the average winter discharge from Mackay Reservoir two years prior to the abundance estimates. Our two lowest abundance estimates occurred two years after the highest winter discharge included in the analysis. This relationship is counterintuitive, as higher winter baseflows are often regarded as beneficial to trout recruitment. In the Henrys Fork of the Snake River, we have found that higher winter flows equate to greater RBT abundance estimates (Heckel et al. 2020). However, our analysis suggests there may be more nuance between higher winter discharge and RBT abundances in the Big Lost River. These two flow events were 1.5 and 2.5 times greater than the average winter flow in the analysis. Winter streamflow at these levels may not be beneficial to RBT in the Big Lost tailwater. But, streamflow may have an effect on the population (Hurst 2007) at some level, which we have not been able to uncover at this point. Entrainment into irrigation canals and river dewatering are suggested to contribute to unstable populations and low abundance of MWF in the Big Lost River (Kennedy 2009), which could also be occurring for the Big Lost River RBT population. One possible mechanism is that higher winter flows are likely indicative of an above-average water year. Higher average flows in the Big Lost River could disperse RBT more broadly throughout the system, which become entrained in canals and in into reaches with poor fish passage that prevents upstream movement and are subsequently lost to the populations. In this respect, higher flows 2 years prior could correlate to lower trout abundance from mortality from sinks such as entrainment and downstream dispersal.

The length-frequency distribution of RBT gives us a snapshot into the size structure of the population at one point in time, but because the Big Lost River is sampled once in every 5-year period we cannot draw many conclusions about why there is low abundance in one size class (e.g., RBT 200-350 mm) or when that abundance changed. The length frequency includes a greater proportion of RBT ≤ 130 mm and ≥ 360 mm. But, we are uncertain of the mechanism causing this gap in age and size classes since we have no data on the population since 2018. Because we observed a missing age class between two prominent age classes, we can only speculate on the cause, which may be that intraspecific predation and competition could have occurred and we are observing a density dependent mechanism in the population (J. McCormick, IDFG, personal communication). The fluctuating strong year classes suggests that there may be low habitat availability for juvenile fish, particularly at low winter flows. We anticipate changing the frequency of sampling this reach by conducting abundance estimates in the Campground reach on a biennial basis to provide us with more up-to-date data and to better understand the population. Our estimate of *A* at 65% indicates that mortality is greater than in the Box Canyon reach of the Henrys Fork of the Snake River (*A* = 52%; see this report Henrys Fork of the Snake River), but we do allow general trout regulations in the Big Lost from Memorial Day weekend until the end of November, whereas Box Canyon is catch and release for trout all year. The difference in mortality estimates could be a result of density-dependent mortality.

Regardless of a missing age class, the abundance in the Campground reach is 45% greater than the RBT population in the Big Wood River Boulder reach (657 RBT/km ± 61; Thiessen et al. 2020), which is managed with similar regulations. Furthermore, the RBT abundance for all reaches in the Big Wood in 2018 was 1,736 RBT/km (± 178; 90% CI), but that estimate includes fish ≥ 100 mm, whereas our abundance estimate includes only RBT ≥ 150 mm. So, the two rivers produce similar abundances of RBT despite varying regulations. Compared to the Boise River, the Big Lost River wild RBT abundance is greater per km and the size structure averages a larger size (D'Amico et al. 2020). For example, PSD in the Big Lost River was 94 in and in the Boise River it was 55, furthermore, RSD-400 in the Big Lost River was 74 and PSD-406 in the Boise River was 10.

Farther downstream in the Leslie reach, RBT and MWF abundance is in decline from previous surveys. However, RBT abundance has been steady since 2007 and MWF fluctuates from year to year, which is likely an artifact of changes in discharge and capture efficiency during the surveys. Overall, MWF abundance has been declining throughout the system. When flows are higher during a spring survey, then fish are likely more spread out longitudinally in the system and not as concentrated in one area, but when dewatering occurs for irrigation then MWF and RBT are lost to the canal system. Based on our fish salvage efforts, as the river becomes dewatered for irrigation MWF and RBT migrate to deeper pools and not necessarily to the head gates of diversions where there are fish ladders. Therefore, instead of migrating back into river reaches with water these fish in pools end up dying from dewatering. So, it may be a combination of factors contributing to fluctuations in abundance in different reaches. Additional evidence that movement patterns vary from year to year and even with the year is evident from the number of fish that we relocated during our salvage efforts. The greatest number of fish were salvaged from immediately downstream of the Blaine diversion, which is about 1.5 km from the Leslie reach. We believe that conducting our surveys in September/October instead of April will provide us with better information on fry/fingerling abundance and fish should be more widespread in the system rather than in overwintering habitat, which is the case in April. Despite low MWF abundance in both surveys, our salvage efforts indicate that their abundance may not be as low as estimated. We sampled multiple age classes of MWF in the catch during salvage including juveniles and adults, suggesting that there is not recruitment failure and if MC is present in the system, there is still a large enough proportion of the population that survives to recruit into maturity. The salvage effort also indicates that dewatering and entrainment into canals could be having a population-level effect on MWF and RBT throughout the Big Lost River. For example, we salvaged 45% of the RBT abundance estimate in the Campground Reach, and we salvaged 852% of the MWF abundance estimate in the Campground reach. We need to continue working with the Big Lost Irrigation District to be better informed on when the river will be dewatered and to encourage them and local irrigators to screen their canals.

Fish Tagging

We estimated no exploitation of tagged, wild RBT in the Big Lost River even though RBT were tagged and available during the open harvest portion of the fishery. Additionally, we did receive few reports of tagged fish being caught, e.g., only 3% of tagged RBT were reported. These results suggest that anglers in the Big Lost River tailrace generally do not harvest fish. Our current creel study that began in April 2021 and ends in March 2022 will provide us with additional information on harvest and if it varies by location on the Big Lost River. Upstream of Mackay Reservoir we found contrasting results with hatchery-origin RBT in the East Fork Big Lost River. In a 2019 tagging investigation in the East Fork, reporting rate was estimated at 45%, exploitation at 34%, and total use at 40%. Considering these results of exploitation from tag returns and assuming that harvest practices for wild RBT are similar, exploitation does not appear to be a factor limiting abundance in the Big Lost River fishery.

MANAGEMENT RECOMMENDATIONS

1. Replicate the sentinel fish study in the tailrace portion of the Big Lost River to further our knowledge on the distribution of MC in the Big Lost River basin.

2. Continue population monitoring in the Big Lost River tailrace reaches at biennial intervals, instead of on a 5-year cycle.

3. Conduct future abundance estimates in September or October to better capture age-0 presence and to capture a more robust distribution of fishes.

Table 14. Test results for the parasites *Myxobolus cerebralis* (whirling disease) and *Tetracapsuloides bryosalmonae* (proliferative kidney disease) in sentinel fish and wild fish sampled during abundance estimates in Copper Basin streams in 2020. Species included Rainbow Trout (RBT), Yellowstone Cutthroat Trout (YCT), and Brook Trout (BKT).

Site	Species	Cohort; dates	Date processed	*M. cerebralis*	*T. bryosalmonae*
EF Swamps	RBT	1; 6/1-6/11/2020	10/14/2020	Positive	Negative
EF Swamps	YCT	1; 6/1-6/11/2020	10/14/2020	Positive	Negative
EF Castle Rock	RBT	1; 6/1-6/11/2020	10/14/2020	Positive	Negative
Star Hope Cr.	RBT	1; 6/1-6/11/2020	10/14/2020	Positive	Negative
Wildhorse Cr.	RBT	1; 6/1-6/11/2020	10/14/2020	Positive	Negative
Star Hope Cr.	RBT	2; 7/3-7/13/2020	11/10/2020	Positive	Negative
Star Hope Cr.	YCT	2; 7/3-7/13/2020	11/10/2020	Positive	Negative
EF Castle Rock	RBT	3; 7/31-8/10/2020	12/15/2020	Positive	Negative
Star Hope Cr.	BKT	Wild; 7/22/2020	7/22/2020	Positive	Negative

Table 15. Abundance (± 95% CI) and density estimates for reaches in the East Fork Big Lost River Swamps (EF Swamps), East Fork Big Lost River at Burma Rd. (EF Burma), East Fork Big Lost River at Fox Creek (EF Fox Cr), East Fork Big Lost River at Whitworth (EF Whitworth), Wildhorse Creek, and Star Hope Creek that were sampled in 2020.

Location, sampling date (mo./day)	BKT	#BKT/100m^2	RBT	#RBT/100m^2	YCT	#YCT/100m^2	MWF	#MWF/100m^2
EF Swamps (7/20)	91 (26.3)	28.9	3 (1.4)	1.0	1 (0)	0.3	0 (0)	0.0
EF Burma (8/25)	96 (1.3)	23.1	43 (0.7)	10.4	0 (0)	0.0	0 (0)	0.0
EF Fox Cr. (8/24)	12 (1.8)	0.7	0 (0)	0.0	17 (3.9)	1.0	3 (3.8)	0.2
EF Whitworth (8/24)	3 (1.4)	0.1	5 (1.5)	0.1	35 (8.9)	1.0	8 (0.6)	0.2
Wildhorse Cr. (7/21)	207 (18.9)	9.1	0 (0)	0.0	0 (0)	0.0	0 (0)	0.0
Star Hope Cr. (7/22)	195 (58.2)	7.6	0 (0)	0.0	16 (7.6)	0.6	0 (0)	0.0

Table 16. The average TL (± SD) of Brook Trout (BKT), Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Mountain Whitefish (MWF) sampled in in the East Fork Big Lost River Swamps (EF Swamps), East Fork Big Lost River at Burma Rd. (EF Burma), East Fork Big Lost River at Fox Creek (EF Fox Cr), East Fork Big Lost River at Whitworth (EF Whitworth), Wildhorse Creek, and Star Hope Creek in 2020. All RBT sampled in EF Burma were hatchery catchables. Sampling dates are the same as stated in Table 2.

Location	BKT	YCT	RBT	MWF
EF Swamps	73 (37)	63 (0)	62 (4)	-
EF Burma	138 (40)	-	260 (25)	-
EF Fox Cr	134 (59)	163 (101)	-	327 (26)
EF Whitworth	180 (56)	177 (85)	245 (99)	285 (47)
Wildhorse Cr.	93 (45)	-	-	-
Star Hope Cr.	73 (39)	124 (113)	-	-

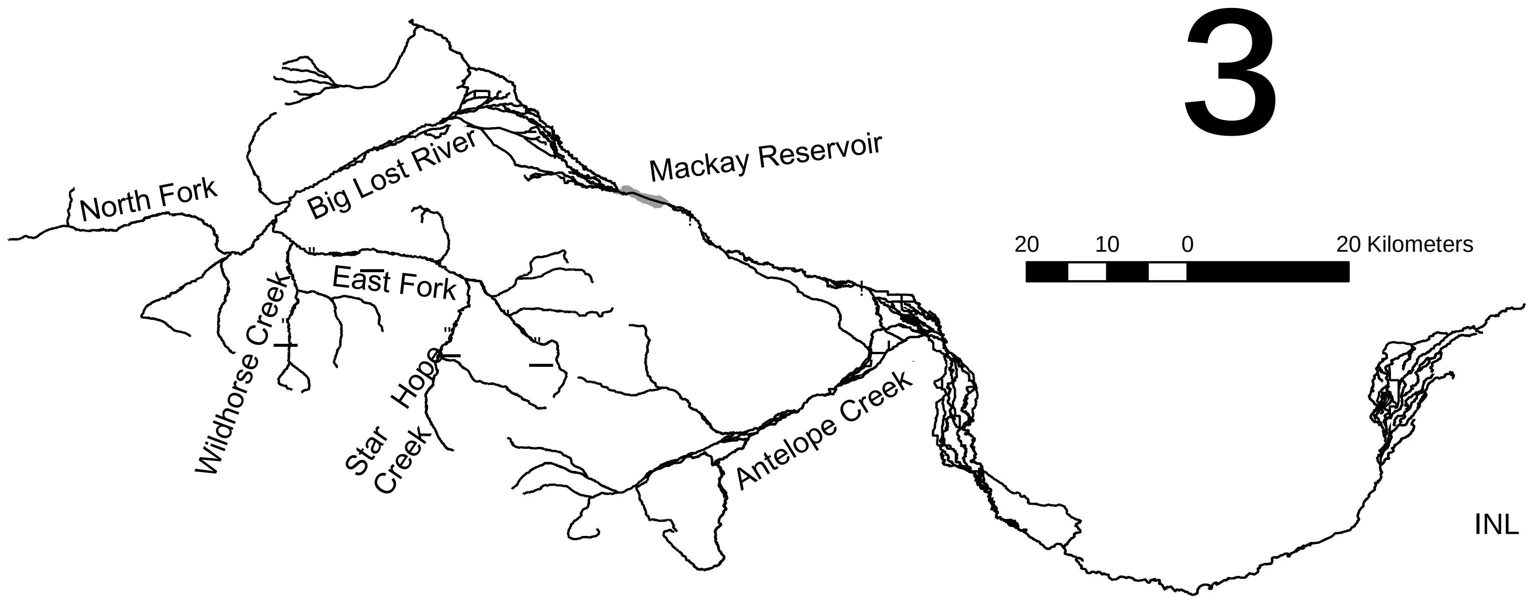

Figure 44. The Big Lost River basin from the headwaters to the sinks on the Idaho National Laboratory (INL). Abundance estimates conducted in 2020 are represented by black squares and sentinel trap sites are represented by stars. Abundance estimates conducted in 2021 are represented by black circles. There was overlap among electrofishing reaches and sentinel fish trap sites, so some stars and squares overlap in the figure.

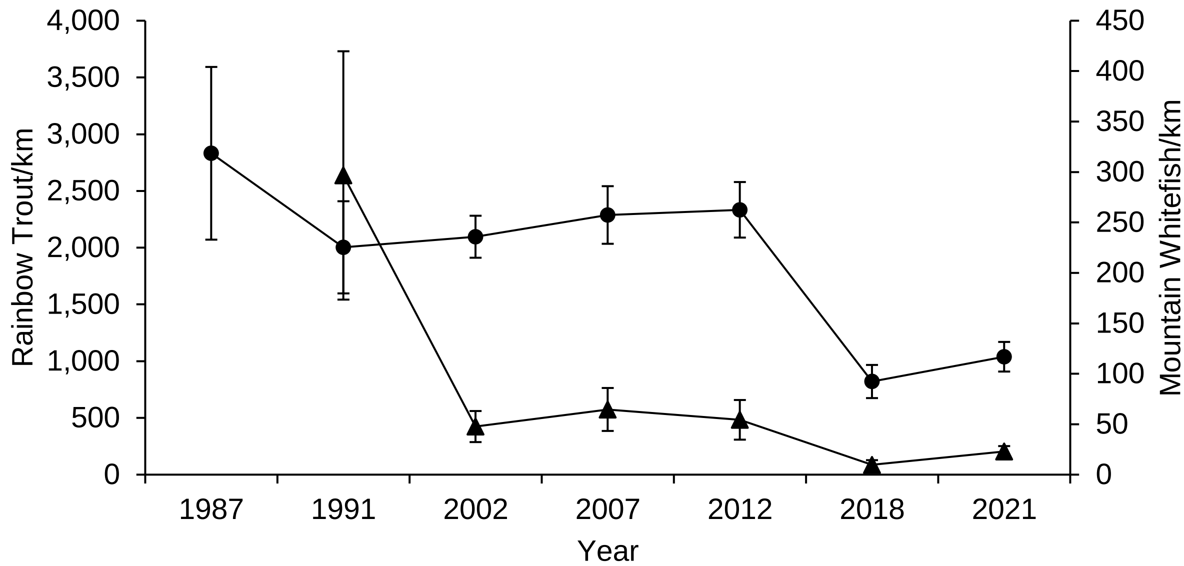

Figure 45. The estimated spring abundance of Rainbow Trout (primary y-axis; black circles) and Mountain Whitefish (secondary y-axis; black triangles) in the Campground reach of the Big Lost River from 1987–2021.

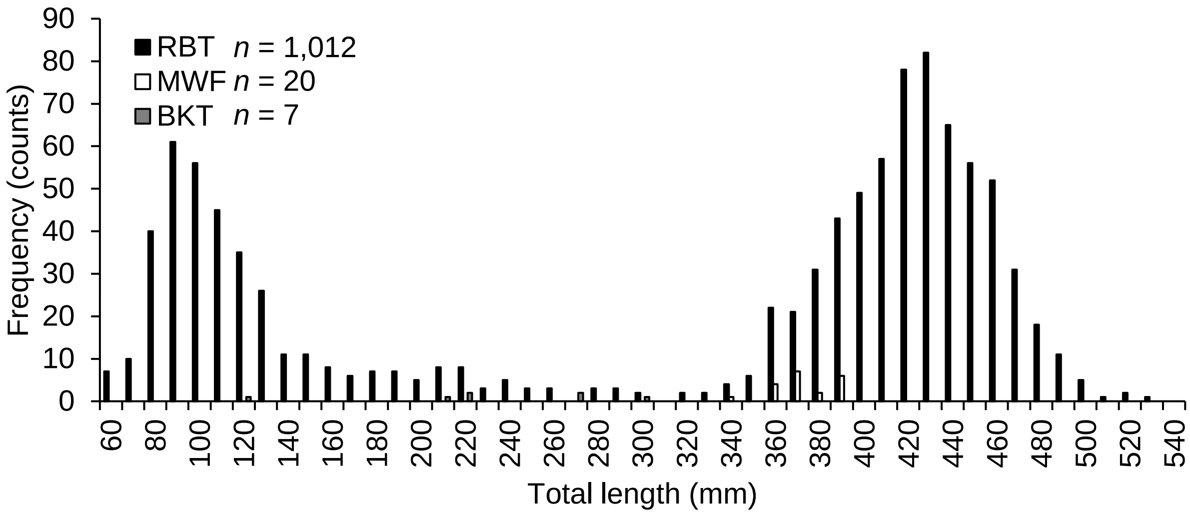

Figure 46. The length-frequency distributions of Rainbow Trout (RBT), Mountain Whitefish (MWF), and Brook Trout (BKT) in the Campground reach of the Big Lost River in April 2021.

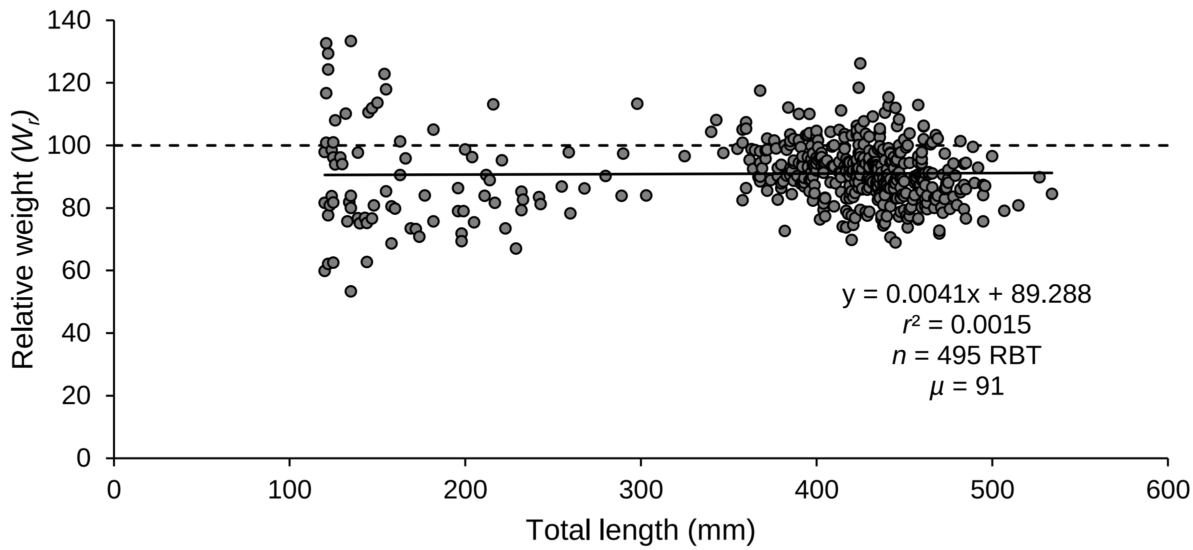

Figure 47. The estimated relative weight W_r for Rainbow Trout (RBT) sampled in the Campground reach of the Big Lost River in April 2021.

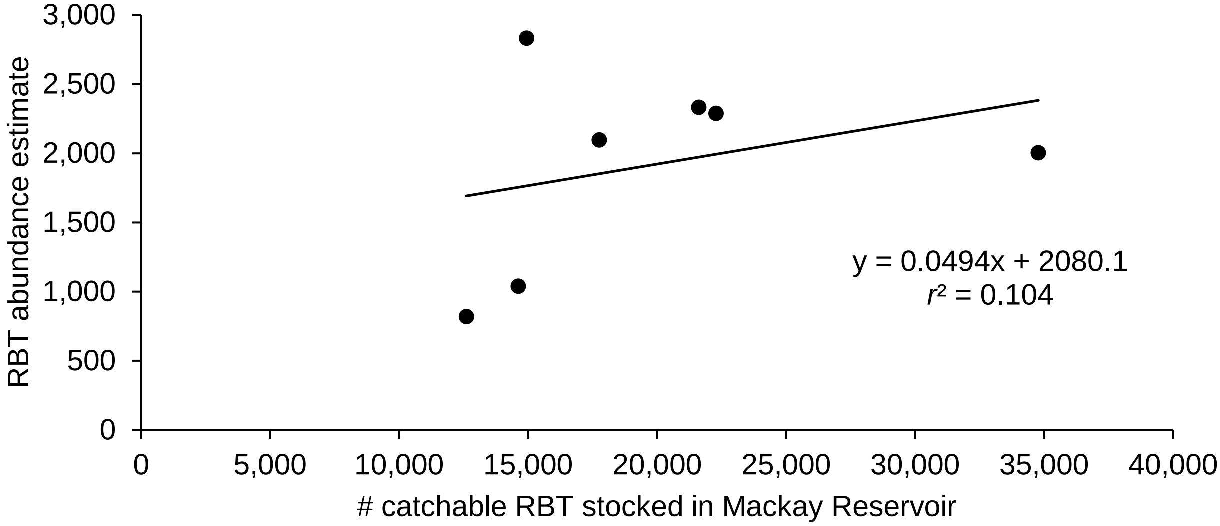

Figure 48. The linear relationship between the Rainbow Trout (RBT) spring abundance estimates from 1987–2021 to the total number of catchable RBT stocked into Mackay Reservoir the year prior to the respective abundance estimate.

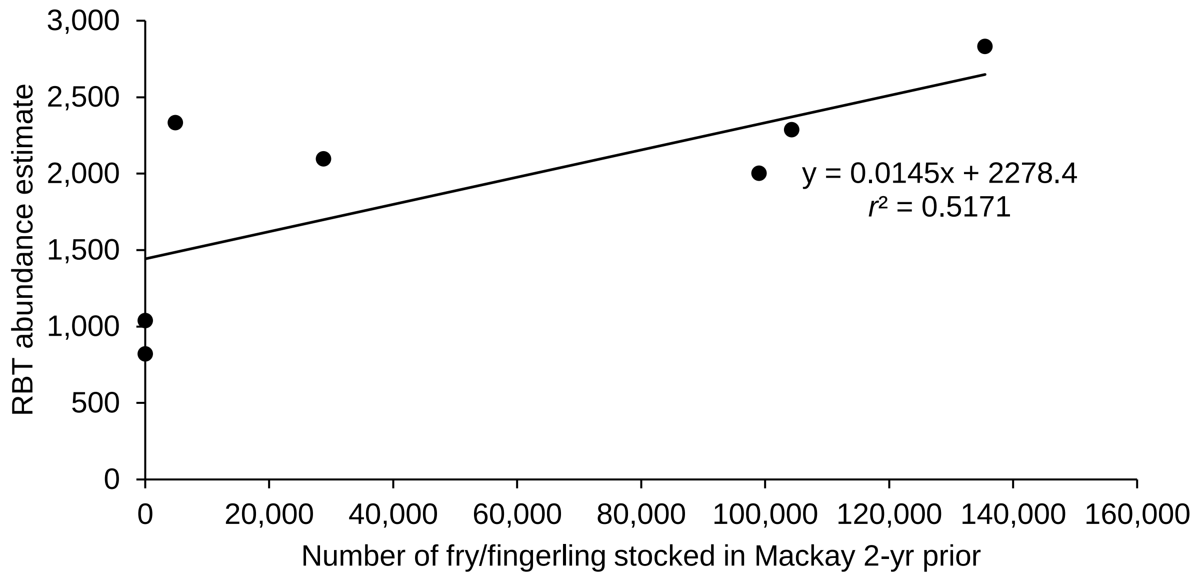

Figure 49. The linear relationship between the Rainbow Trout (RBT) abundance estimates from 1987–2021 to the total number of fry and fingerling RBT stocked into Mackay Reservoir two years prior to the respective abundance estimate.

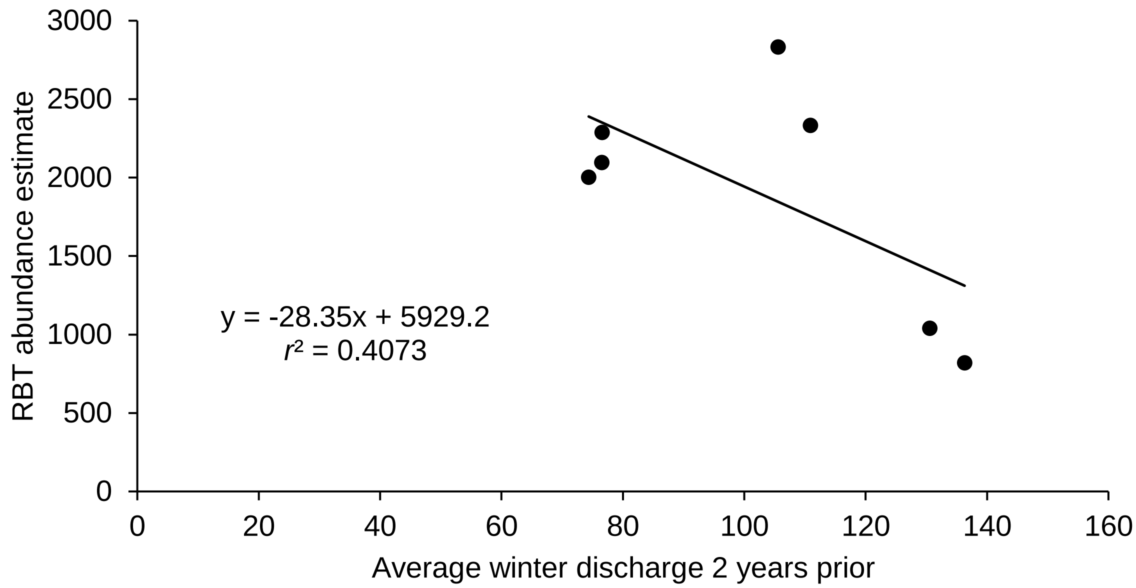

Figure 50. The linear relationship between the Rainbow Trout (RBT) abundance estimates from 1987 to 2021 to the average winter discharge from Dec. 1 to Feb. 28 two years prior to the respective abundance estimate.

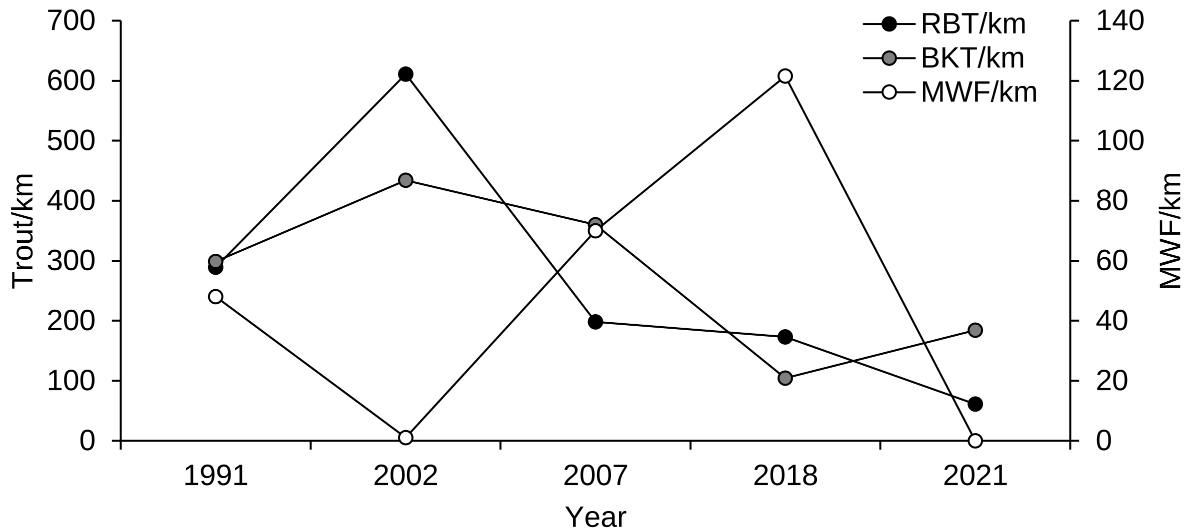

Figure 51. Rainbow Trout (RBT), Brook Trout (BKT), and Mountain Whitefish (MWF) abundance estimates in the Leslie reach of the Big Lost River 1991–2021.

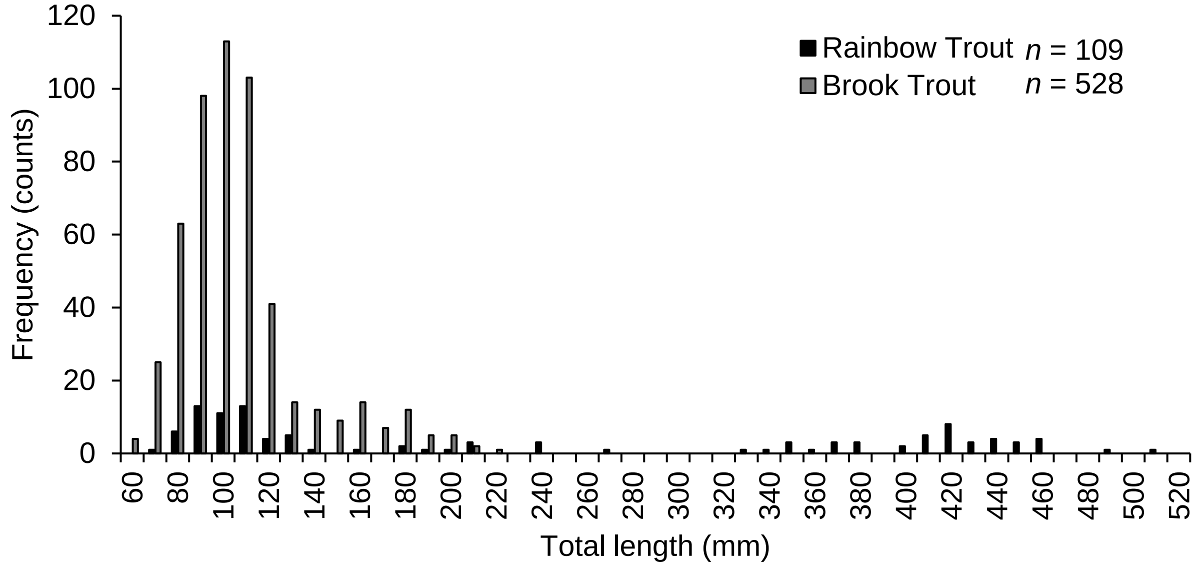

Figure 52. The length-frequency distributions of Rainbow Trout and Brook Trout sampled from the Leslie reach of the Big Lost River 2021.

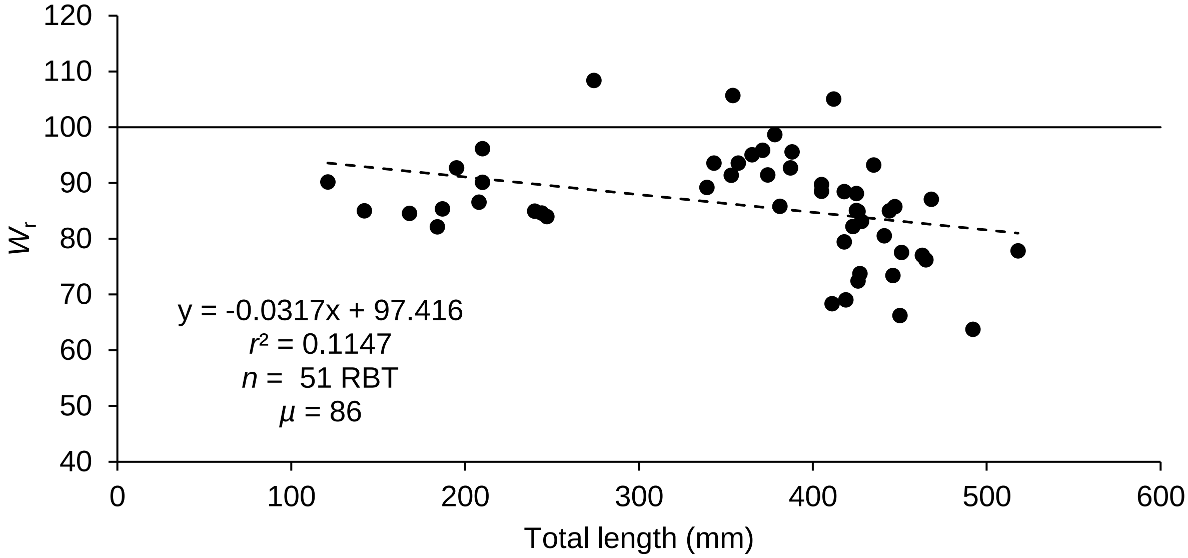

Figure 53. The distribution of relative weights W_r for Rainbow Trout (RBT) sampled from the Leslie reach of the Big Lost River 2021.

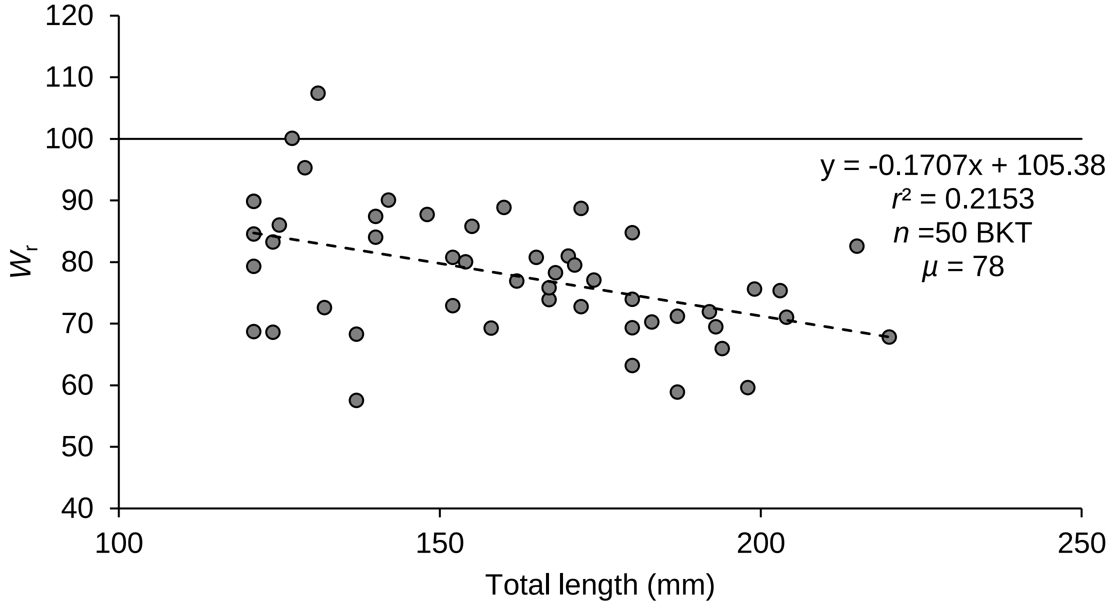

Figure 54. The distribution of relative weights W_r for Brook Trout (BKT) sampled from the Leslie reach of the Big Lost River 2021.

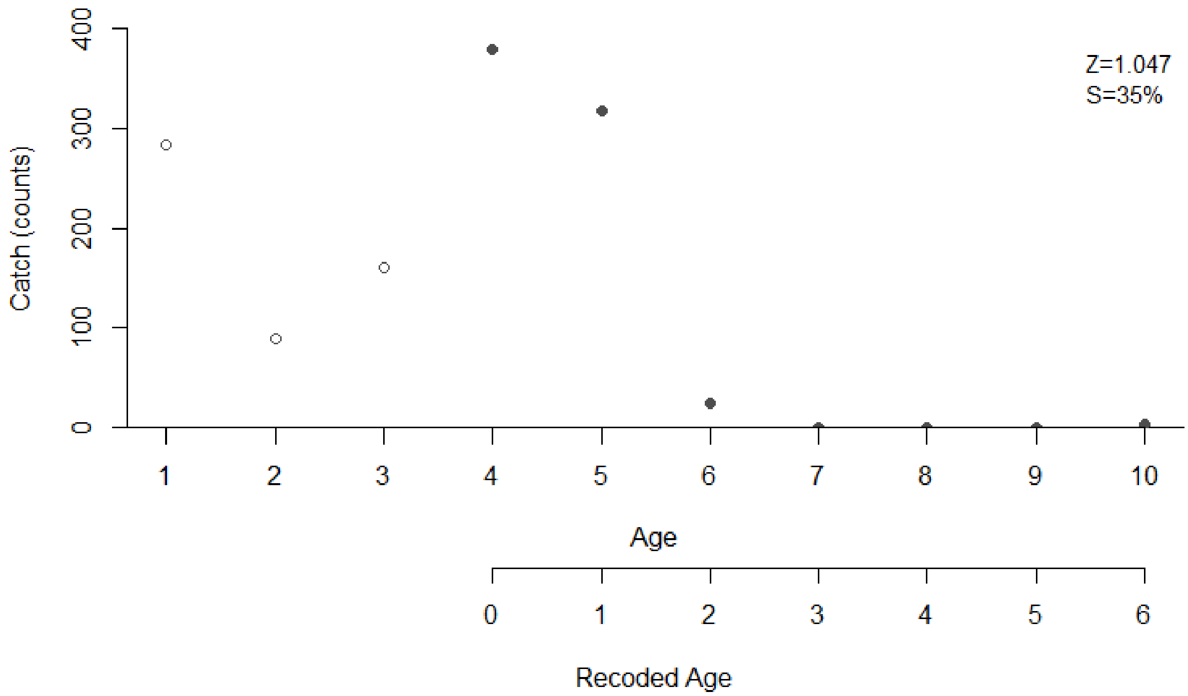

Figure 55. Catches at estimated ages for Rainbow Trout captured in the Campground reach of the Big Lost River in 2021. The solid points were used to calculate the Chapman-Robson estimates of *S* and *Z*.

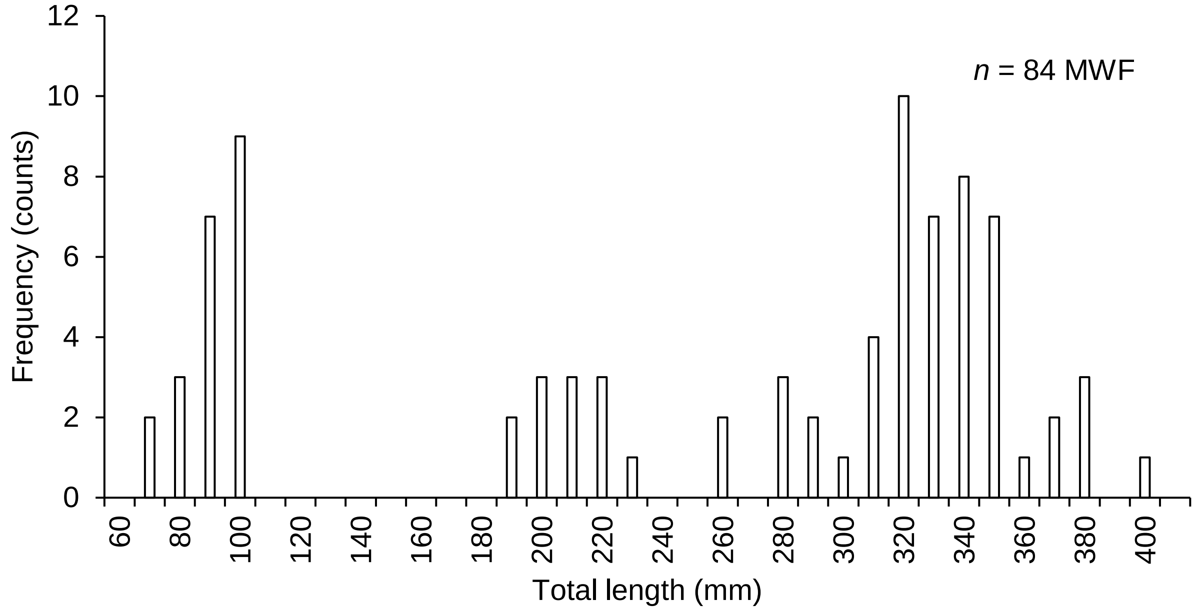

Figure 56. The length-frequency distribution of Mountain Whitefish (MWF) that were salvaged from downstream of the Blain Diversion and relocated to the Campground reach in the Big Lost River in July 2021.

ASSESSMENT OF YELLOWSTONE CUTTHROAT TROUT POPULATIONS IN THE WILLOW CREEK DRAINAGE

ABSTRACT

We conducted our second abundance estimate, as a follow-up survey to our survey in 2020, using raft-mounted electrofishing in the tailwater portion of Willow Creek downstream of Ririe Reservoir to evaluate Yellowstone Cutthroat Trout *Oncorhynchus. clarkii bouvieri* (YCT) abundance and life history. We sampled a 1.6-km long reach downstream of the dam in the tailrace portion of the creek. Species composition in the tailrace consisted of 98% YCT, 1% kokanee *O. nerka*, and 1% Lake Trout *Salvelinus namaycush*. The average total length (TL) of YCT was 196 mm (± 67; SD), kokanee was 265 mm, and Lake Trout was 386 mm. The electrofishing catch-per-unit-effort (CPUE) for the marking pass was 1.96 fish/min for YCT, 0.01 for kokanee, and 0.01 for Lake Trout. We estimated the abundance of YCT for the reach at 1,534 (± 906; 95% CI), which was 1.61 km (i.e., 1 mile) in length. The proportional size distribution of YCT was 14 and RSD-400 was 1. In 2020, we conducted the same type of survey and PIT tagged 92 YCT. In the 2021 survey, we recaptured 13 fish that were tagged in 2020. The YCT population in the Willow Creek tailrace appears robust with multiple age-classes present, and based on the number of YCT that we PIT tagged in 2020 and recaptured in 2021, this appears to be an isolated resident population. In addition to sampling the tailrace fishery, we installed two PIT-tag antennas in Willow Creek upstream of Ririe Reservoir and subsequently conducted several density estimates in conjunction with PIT tagging in Willow Creek, Tex Creek, Hell Creek, and Indian Fork to tag YCT. To investigate life history and migration dynamics of YCT and Smallmouth Bass *Micropterus dolomieu* (SMB), we sampled and PIT tagged YCT (*n* = 41) and SMB (*n* = 93) in Willow Creek, and YCT in Hell Creek (*n* = 4), Tex Creek (*n* = 3) and Ririe Reservoir (*n* = 71).

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INTRODUCTION

Willow Creek is a minor tributary of the Snake River. The creek originates from small springs in the Willow Creek lava field and drains portions of the Caribou Range and Blackfoot Mountains then flows northward into Ririe Reservoir (Figure 57). The reservoir is a U.S. Bureau of Reclamation (BOR) impoundment built primarily for flood control (completed and first stocked with fish in 1975; Ball and Jeppson 1978) and is also used for agricultural irrigation upstream and downstream of Ririe Reservoir. As such, Willow Creek is diverted multiple times downstream of Ririe Reservoir into a floodway channel east of Idaho Falls in Ucon. A portion is then diverted from the floodway channel back into Willow Creek, while another portion is diverted into Sand Creek. In 1976, the Idaho Department of Fish and Game (IDFG) suggested stream maintenance flow recommendations to the BOR based upon fish spawning and rearing requirements (Jeppson 1979). However, water releases from Ririe Dam were curtailed in 1978 to prevent the formation of ice plugs in irrigation channels that would cause the stream to overflow its banks during subsequent runoff (Jeppson 1979). As a result, the cooperative agreement between the IDFG, BOR, and U.S. Army Corps of Engineers decided that flows could be reduced to zero during periods of extreme icing conditions; otherwise, minimum releases for stream maintenance flow would be reservoir inflow or the following, whichever is less: May 16 to April 14, 30 cfs; and 15 April–15 May, 45 cfs (Jeppson 1979). Aside from minor leakage, no winter discharge (i.e., Dec.–Feb.) has occurred out of Ririe Reservoir since 1980.

Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (YCT) and Mountain Whitefish *Prosopium williamsoni* are the only native salmonids in the Willow Creek drainage. Current fishing regulations prohibit harvest of YCT in Willow Creek and its tributaries; however, anglers can harvest YCT in Ririe Reservoir. There was public access and a day use area on the west side of the Willow Creek tailwater (i.e., Creekside Park) that was accessible via the Ririe dam road and was managed by the BOR. However, that road was closed in 2005 and current access to the tailwater fishery is limited. Nearly all land surrounding the tailwater is privately owned and used for agricultural purposes aside from approximately 1.5 km directly downstream of Ririe Dam, which is property of BOR. When the tailwater access was open (via Ririe Dam road), anglers were able to walk and wade, and float fish downstream to N115E (i.e. East Shelton Rd.; Ball and Jeppson 1978). Creel results indicate that anglers predominately caught YCT, but also caught Rainbow Trout *O. mykiss*, and Brown Trout *Salmo Trutta* in the tailwater reach. Catch rates averaged 1.84 fish/h in 1976 when the tailwater fishery was first investigated using a creel survey (Jeppson 1977). In the following year, overfishing was attributed to declining catch rates of 0.44 fish/h, and 0.32 fish/h in 1978 (Jeppson 1979). Thick vegetation, lack of roads, unwadeable pools, and private property limited access to the 8.4 km of stream downstream of the dam to the first public road (N115E); therefore, angler effort was concentrated near the dam and resulted in overfishing in that reach (Ball and Jeppson 1978). The IDFG recommended developing an access site near N115E and the Eagle Rock Canal for a floating access takeout and to better distribute angling effort along the tailwater reach (Ball and Jeppson 1978), but this access was never developed. The tailwater fishery has not been investigated since 1978, but there has been an increase in fishing effort on Ririe Reservoir (e.g., 90,024 angler hours; Heckel et al. 2020). An additional fishery near Idaho Falls could provide increased angling opportunities for our constituents and become very popular if an access site, which is stipulated in the BOR Resource Management Plan (BOR 2001), is developed for anglers.

The Willow Creek drainage upstream of Ririe Reservoir supports a wild, genetically pure Yellowstone Cutthroat Trout *O. clarkii bouvieri* population. This population is managed under restrictive regulations including no harvest and tributary closures during spawning to protect this native species (IDFG 2019). Water flows vary from extremes of several thousand cubic feet per second during runoff to a few cubic feet per second in late summer and winter. Intense grazing combined with drought conditions have contributed to poor riparian habitat in the upper watershed. Although the fisheries in the Willow Creek drainage are challenged with substantial habitat and flow related issues, the persistence and recent expansions of native Yellowstone Cutthroat Trout make managing for native fish a priority for this drainage (IDFG 2019; Vincent et al. *in press*).

OBJECTIVES

The objectives for this study were threefold; (1) estimate abundance and size structure of YCT in the tailrace of Willow Creek, (2) assess the life history strategies of YCT in the Willow Creek tailrace by evaluating marked and passive integrated transponder (PIT)-tagged fish from the 2020 population survey, and (3) assess the migration and life history dynamics of PIT-tagged YCT and Smallmouth Bass *Micropterus dolomieu* (SMB) in the drainage upstream of Ririe Dam.

METHODS

Tailrace Electrofishing

We estimated fish abundances using mark/recapture techniques using one raft-mounted electrofishing unit. We measured water temperature (°C) and conductivity (µS/cm) prior to active electrofishing using a handheld probe. Pulsed direct current power was provided by a 5000-W generator and standardized to 2,750-3,250 W based on water conductivity (Miranda 2009). We applied electricity to the water using an Infinity model electrofisher (Midwest Lake Management, Inc., Polo, Missouri). Electrofishing began at the uppermost point of the sampling reach and proceeded in a downstream direction. One netter positioned at the bow of the raft used a 2.4-m long dip net with 6-mm bar knotless mesh. The netter attempted to net all fishes and place them into a live well located in the raft. We conducted single-pass marking and recapture runs in the reach. All sportfish encountered from mark-recapture surveys were collected, identified to species, measured for total length (TL) to the nearest mm, and all fish ≥ 120 mm were marked with a hole punch in the caudal fin prior to release. We took a fin clip from all YCT for genetic analysis; and because there is a possible Willow Creek connection to the South Fork Snake River via the Eagle Rock Canal, all YCT that were 120–350 mm were implanted with a PIT tag in which they could be identified in South Fork Snake River PIT arrays and in abundance estimates. We marked all PIT-tagged YCT with an adipose fin clip to further identify tag loss and tag retention. Furthermore, we evaluated life history in 2021 based on the number of YCT that were PIT tagged in 2020, or had a clipped adipose fin.

Fish Sampling For PIT Tagging

Two PIT-tag antennas were installed in September 2021 in Willow Creek upstream of Ririe Reservoir: one was installed about 80 m upstream from the Willow Creek and Tex Creek confluence and the other was installed about 30 m downstream from the confluence. In the Willow Creek drainage upstream of Ririe Dam we PIT-tagged YCT in opportunistic, single-pass electrofishing surveys in Willow Creek, Tex Creek, Hell Creek, and Indian Fork (Appendix A). Depending on stream width, either a single backpack electrofishing unit (Smith Root LR-24) standardized to 100 W of output (Meyer et al. 2021), or a towed raft electrofishing unit were used to sample fish for PIT tagging. In addition to tagging YCT in the lotic portion of the drainage upstream of Ririe Reservoir (Figure 57), we also conducted night electrofishing in the Willow

Creek arm of Ririe Reservoir using a motorized boat outfitted with the same electrofishing gear as the raft used in the tailwater. In Ririe Reservoir, netters were instructed to only net YCT. In all electrofishing upstream of Ririe Reservoir, we used pulsed direct current (DC) with a 25% pulse width and a frequency of 60 Hz. We recorded the minutes of active electrofishing to calculate electrofishing CPUE in all electrofishing locations. All YCT captured were measured for TL to the nearest mm and YCT ≥ 120 were implanted with a 12.5-mm full-duplex RFID PIT tag.

Data Analysis

In the tailrace survey, we estimated abundance using a Peterson estimator with a Chapman modification using the mrClosed() function in the FSA package in program R (R Core Team 2021). We adjusted the abundance estimate for size selectivity by distributing the catch into size classes to fit the recapture data. We estimated capture efficiency as the number of marked fish that were recaptured divided by the total number of marked fish that were available for capture multiplied by 100 and represented as a percent (Chiaramonte et al. 2020). We then calculated 95% CI for the abundance estimate. We calculated proportional size distribution (PSD) as the number of YCT ≥ 300 mm divided by the number of YCT ≥ 200 multiplied by 100. Smallmouth Bass *Micropterus dolomieu* (SMB) PSD was calculated as the number of SMB ≥ 280 mm divided by the number of SMB ≥ 180 mm multiplied by 100. Similarly, relative stock density (RSD-400) used the same formula, with the numerator replaced by the number of YCT ≥ 400 mm, and RSD-350 for SMB ≥ 350 (Anderson and Neumann 1996; Neumann et al. 2012). We also evaluated species composition and fish condition was evaluated by calculating relative weight (W_r) and plotting against TL. We used the standard weight (W_s) equation

$$log_{10}(W_s) = -5.189 + 3.099 * \ log_{10}(L),$$

for Cutthroat Trout (Kruse and Hubert 1997) where L is the total length, then used the equation

$$W_r = \left(\frac{W}{W_s}\right) * 100,$$

where W_r is relative weight and W is measured weight.

RESULTS

Tailrace Electrofishing

In the tailwater reach, we collected 251 fish (including YCT < 120 mm) and PIT tagged 106 YCT during two days of electrofishing. We marked fish on August 12, 2021 and recaptured fish on August 19. Water temperatures at time of sampling were 11°C on the marking run and 11°C on the recapture run. Species composition was dominated by YCT (98%; Table 17), followed by kokanee *O. nerka* (1%), and Lake Trout *Salvelinus namaycush* (1%). Yellowstone Cutthroat Trout abundance was estimated at 1,534 fish for the reach (± 906; i.e., 953 YCT/km). Our capture efficiency was 8%. We were unable to estimate abundance or capture efficiency of all other species because they were caught on the recapture pass. The mean TL of YCT was 196 mm (± 67; SD; Figure 58), we caught one kokanee (TL was 265 mm), and one Lake Trout (TL was 386 mm). The proportional size distribution (PSD) of YCT was 14 and RSD-400 was 1. The electrofishing CPUE for the marking pass was 1.96 fish/min for YCT. We recaptured 13 YCT from 2020, four fish shed tags and nine fish still had a PIT tag intact.

Fish Sampling for PIT Tagging

Yellowstone Cutthroat Trout abundance varied among the reaches in the Willow Creek drainage (Table 18) upstream of Ririe Reservoir. Catch rates of YCT were < 1 fish/min at all locations including Ririe Reservoir; however, we PIT tagged 119 YCT that were sampled from 5 stream reaches upstream of Ririe Reservoir and in the Willow Creek arm of Ririe Reservoir. We sampled 71 YCT from Ririe Reservoir during 2 nights (Nov. 3 and 18) of electrofishing including 1 recapture that was tagged in night one and recaptured in night two. The mean TL of YCT from Ririe Reservoir was 364 mm (± 37; Figure 59), PSD was 97, RSD-400 was 17, and the mean relative weight (W_r) was 93. The mean TL of YCT sampled in Willow Creek and its tributaries was 251 mm (± 77), PSD was 44, and RSD-400 was 3. We also sampled 93 SMB in Willow Creek upstream of Ririe Reservoir and we opportunistically PIT tagged 54 Smallmouth Bass to better understand their movement dynamics and life history. The mean TL of SMB was 209 mm (± 51; Figure 60), PSD was 52, and RSD-350 was 19.

At the PIT-tag array, we detected 13 of the 54 SMB migrating downstream in September and October. In addition, one YCT that was tagged in Tex Creek was detected migrating upstream past the upstream antenna in November.

DISCUSSION

One of the objectives of this research was to investigate the life history of YCT in the tailwater reach using YCT that were PIT tagged in 2020 and recaptured in 2021. We recaptured 13 YCT that were PIT tagged in 2020 and marked with an adipose fin clip. These fish either shed the tag or the tag was still intact, but we identified them by the clipped adipose fin from 2020 sampling. As such, we believe this is a resident YCT population because a proportion of the 2020 tagged fish were recaptured in the same reach, and no YCT that were originally tagged in the South Fork Snake River were observed during Willow Creek tailrace sampling. We also observed that the length-frequency distribution included smaller size classes and a mean TL shift to smaller size classes from 2020 to 2021. The two dominant length groups suggest that there is survival despite no base winter flow and recruitment is strong for juvenile YCT, and with no predatory fish (e.g., Walleye *Sander vitreus*, SMB) or angler exploitation in the reach it creates a unique fishery opportunity and environment. This survey provides insight into how a wild, resident YCT population can persist in severely flow-limited habitats.

Density was greater in the current survey (953 YCT/km) compared to the 2020 survey (213 YCT/km) because we recaptured fewer fish, but we also sampled more fish in total and we estimated a higher density of juvenile fish in 2021 compared to 2020. We anticipate size structure will increase as the juveniles we sampled grow and recruit to the fishery. The size structure of the Willow Creek YCT population is comparable to the YCT population in portions of the Teton River, and YCT are more abundant per km in Willow Creek compared to some portions of the Teton River. For example, in 2019 in the Nickerson reach of the Teton River YCT abundance was estimated to be 436 YCT/km and the average TL was 234 mm, compared to 953 YCT/km in Willow Creek and average TL 196 (Heckel et al 2020). However, in the Breckenridge reach of the Teton River YCT abundance was much lower (42 YCT/km) in 2019 when compared to Willow Creek. The Teton River YCT population does have the potential of hybridization and introgression with Rainbow Trout *O. mykiss* (RBT) as well as competition with that species. Therefore, YCT density could be lower in the Teton River because of the presence of RBT, which makes the Willow Creek tailrace population of YCT unique from our other YCT fisheries in the region.

Genetic analysis will describe hybridization and introgression with RBT and potentially the current and historical connections with the South Fork Snake River YCT population, or lack thereof. Furthermore, because we PIT-tagged YCT in Willow Creek downstream of Ririe Dam and there may be connectivity to the South Fork Snake River through Eagle Rock Canal, there is a possibility those fish could be encountered in South Fork Snake River population surveys. Due to the leakage out of Ririe Reservoir (e.g., no measurable discharge), a number of large, deep pools exist through the winter providing adequate habitat for this resident YCT population to persist. Anglers that responded to a statewide angler opinion survey (Koenig 2020) indicated that vehicle access ranked 8th out of 18 different factors that were important on deciding where to fish. Furthermore, shore/bank access was the number one method used to access a fishery. Providing access to this unique tailwater fishery near Idaho Falls would provide anglers an additional opportunity to fish for wild, native YCT in a small stream setting.

The PIT tag antennas were installed in September and within the first several months we collected data about the life history structure of two sportfish (YCT and SMB) species in the Ririe-Willow Creek system. This is an indication that there is great potential for learning about YCT in this system, which we could use for conservation and habitat improvement actions. Assuming that some YCT tagged in Ririe Reservoir will migrate upstream to spawn in Willow Creek in the spring, we anticipate learning more about YCT life histories as we increase tagging. Our research in the 1980s estimated that there were two migrations in the Willow Creek drainage; one migration during runoff to spawning tributaries and juveniles emigrating from tributaries, and a second emigration out of tributaries into Willow Creek or Ririe Reservoir during the late fall (Corsi 1986). One YCT emigrating is informative, but is evidence that a fluvial life history persists in the population, which is promoting population viability. We captured only larger, presumably adult, YCT in the reservoir, and presumably more juveniles in the tributaries suggesting that the tributaries are rearing areas and the reservoir is dominated by adults seeking more forage opportunities.

We observed PIT tag data that implicated downstream migration of SMB into Ririe Reservoir. We are left with a question of whether these fish hatched in tributaries then emigrate to the reservoir and follow an adfluvial life history strategy, or do they migrate from the reservoir to Willow Creek to forage then return back to the reservoir to overwinter. We plan to monitor both YCT and SMB life history strategies to gain a better understanding for these population demographics. For YCT, we intend to learn more about important spawning tributaries and linkages between the reservoir and tributaries to guide management practices and habitat restoration as outlined in the Fisheries Management Plan (IDFG 2019).

MANAGEMENT RECOMMENDATIONS

1. Work with BOR to regain public access to the tailwater fishery of Willow Creek.

2. Continue to monitor the trends in abundance and size structure of the YCT population in the Willow Creek tailwater, and describe genetic structure of this population relative to YCT in the South Fork Snake River, Teton River, and upstream of Ririe Dam.

3. Continue monitoring the species composition in the tailrace of Willow Creek to maintain our knowledge of potential threats to the YCT population.

4. Continue PIT tagging YCT and SMB in Ririe Reservoir and the Willow Creek drainage upstream of Ririe Dam to describe the life histories of YCT and SMB in this system.

Table 17. Yellowstone Cutthroat Trout (YCT), kokanee (KOK), and Lake Trout (LKT) population index summaries for the Willow Creek tailwater. Mean and median total length (TL; mm) are represented ± SD, the density estimate is represented with ± 95% CI, and capture efficiency is represented. Proportional size distribution (PSD) and relative stock density (RSD-400) are also reported in the table.

Species	Number captured	Mean TL (mm)	Median TL (mm)	PSD	RSD-400	Density (No./ km)	Species Composition (%)	Capture efficiency (%)
YCT	237	196 (± 67)	171	14	1	953 (± 563)	98	8
KOK	1	265	--	0	0	--	1	--
LKT	1	386	--	0	0	--	1	--

Table 18. Electrofishing catch rates (CPUE; fish/min) of species sampled in the Willow Creek drainage upstream of Ririe Dam in 2021. Species codes refer to Yellowstone Cutthroat Trout (YCT), Smallmouth Bass (SMB), Walleye (WLY), Mountain Sucker *Catostomus platyrhynchus* (MTS), Utah Sucker *Catostomus ardens* (UTS), and hatchery Rainbow Trout *O. mykiss* (HRBT).

Site; Date of sampling (m/d/yr)	YCT	CPUE	SMB	CPUE	WLY	CPUE	MTS	CPUE	UTS	CPUE	HRBT	CPUE
Tex-Willow confluence; 6/17/21	24	0.39	58	0.94	0	0	0	0.00	0	0.00	13	0.21
Willow Cr at Clowards; 6/15/21	17	0.19	35	0.40	0	0	1	0.01	1	0.01	2	0.02
Hell Creek 1; 4/21/21	4	0.22	0	0.00	0	0	0	0.00	3	0.17	0	0.00
Tex-Indian confluence; 4/21/21	2	0.12	0	0.00	0	0	2	0.12	0	0.00	0	0.00
Tex Creek 1; 4/21/21	0	0.00	0	0.00	0	0	5	0.42	0	0.00	0	0.00
Tex Creek 2; 4/22/21	1	-	0	-	0	-	3	-	0	-	0	-
Ririe Reservoir; 11/3, 11/18/21	71	0.29	-	-	3	0.01	-	-	-	-	-	-

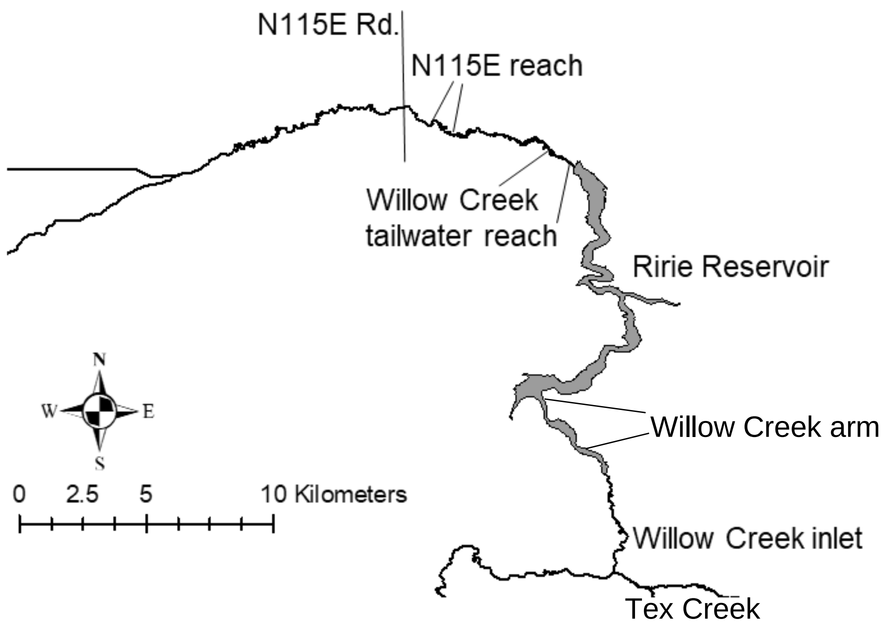

Figure 57. The Willow Creek inlet to Ririe Reservoir (south of the reservoir) and the Willow Creek tailwater reach to the northwest of Ririe Reservoir.

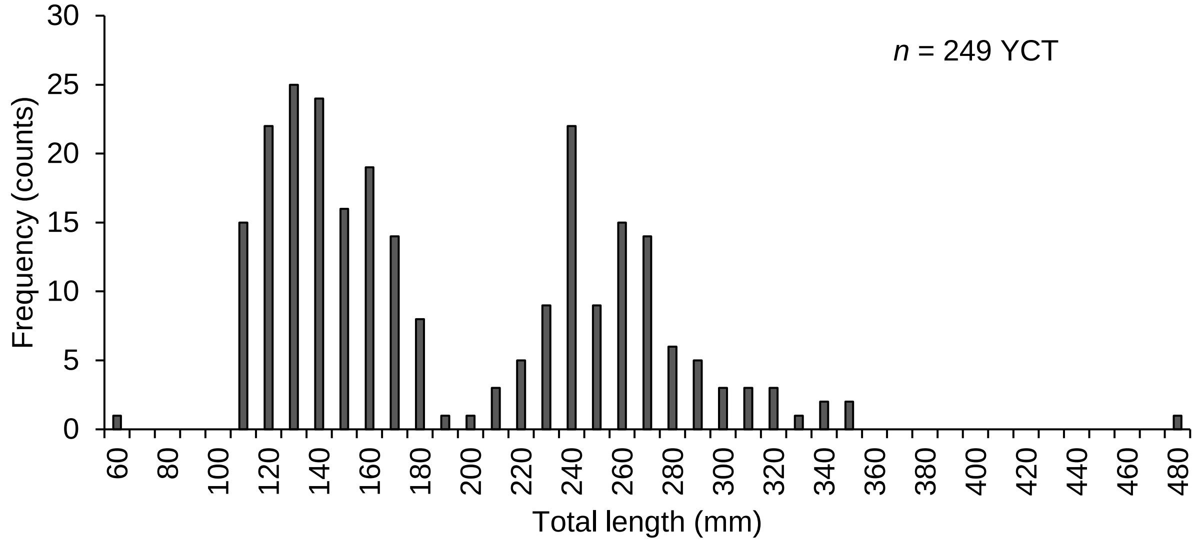

Figure 58. Length-frequency distribution of Yellowstone Cutthroat Trout (YCT) in the tailwater reach of Willow Creek

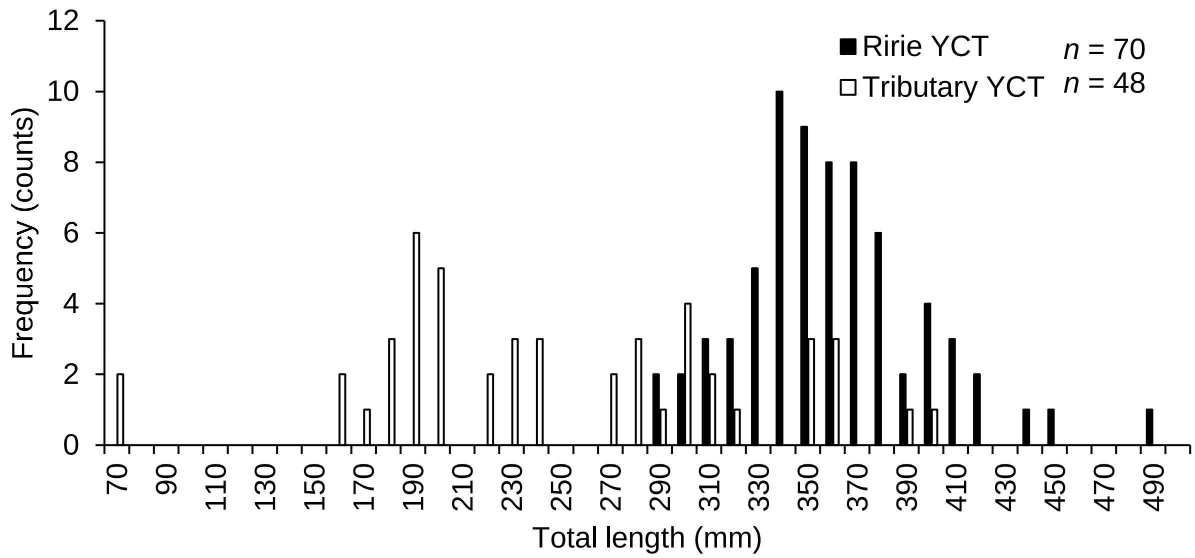

Figure 59. The length-frequency distribution of Yellowstone Cutthroat Trout (YCT) sampled in Ririe Reservoir (black bars) and upstream of Ririe Reservoir in Willow Creek and its tributaries (white bars) in 2021.

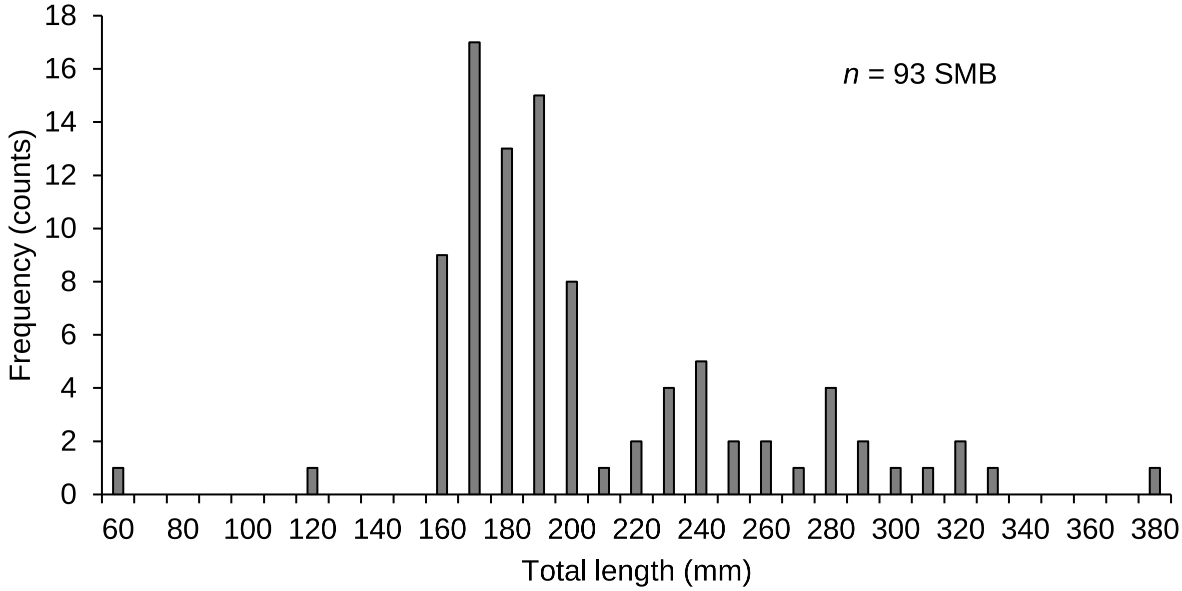

Figure 60. The length-frequency distribution of Smallmouth Bass (SMB) sampled in Willow Creek in 2021.

	Start		End	
Reach	Latitude	Longitude	Latitude	Longitude
Tailwater	43.58326	-111.74582	43.58723	-111.75385
Tex-Willow Creek confluence	43.44588	-111.72638	43.44278	-111.72878
Willow Creek at Clowards	43.44341	-111.78607	43.43662	-111.79044
Hell Creek	43.33660	-111.66169	43.33661	-111.66112
Tex Creek 1	43.43160	-111.69429	43.43131	-111.69377
Tex Creek 2	43.43328	-111.72048	43.43301	-111.72040
Tex Creek-Indian Fork confluence	43.42406	-111.66036	43.42455	-111.65976

Appendix A. Locations of population surveys in the Willow Creek drainage, Idaho 2021.

SMALL STREAMS

ABSTRACT

We conducted electrofishing surveys in Warm River, Medicine Lodge Creek, and Corral Creek to investigate trout abundances in streams that are not sampled on a regular cycle. In Warm River, we conducted abundance estimates in two reaches. In the downstream reach, we conducted a mark/recapture abundance estimate in which we estimated 1,132 trout/km (95% CI ± 187), which was split between Brown Trout *Salmo trutta* 480 (± 67) and Rainbow Trout *Oncorhynchus mykiss* (RBT) 468 (± 125). Mountain Whitefish *Prosopium williamsoni* abundance was 34 fish/km (± 15). In the upstream reach of Warm River, we only captured Brook Trout *Salvelinus fontinalis* (BKT) and estimated abundance using a depletion estimate in which we estimated 43 trout/100 m^2. We sampled two sites in Medicine Lodge Creek to investigate the RBT abundance using a mark/recapture estimate. In the downstream reach, we estimated 1,144 RBT/km (± 102) and in the upstream reach we estimated 460 (± 22). In Corral Creek, we investigated Yellowstone Cutthroat Trout *O. clarkii behnkei* (YCT) abundance in two reaches using depletion estimates. In the downstream reach, we estimated 9 YCT/100 m^2 (± 2.1) and in the upstream reach 39 (± 2.3). We also sampled Brook Trout density in the downstream reach at 32 BKT/100 m^2 (± 3.1).

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INTRODUCTION

During the winter of 2020, Upper Snake Region fisheries staff received phone calls from anglers asking about the fish population status in Warm River and Medicine Lodge Creek. Recent information about fisheries conditions and trout abundances were not available, leading staff to develop sampling plans for 2021. Warm River is approximately 59-km long from its headwaters in the Island Park Caldera to its confluence with the Henrys Fork Snake River (HFSR). At roughly 16 km upstream from the confluence, Warm Springs contributes about 6 m^3/s of water to the main stem of Warm River where it is characterized as a spring creek with aquatic vegetation and clear water. This is the primary reach targeted by anglers as a fishery. We began stocking fingerling and catchable *Oncorhynchus mykiss* (RBT) in Warm River in the 1960s and Brown Trout *Salmo trutta* (BNT) fingerling stocking was implemented in the 1980s through 1990s. The fishery is currently managed under general trout regulations from Memorial Day through November 30 with no harvest of Yellowstone Cutthroat Trout *O. clarkii bouvieri* (YCT). We currently stock catchable-sized RBT in the lower Warm River near the confluence with the HFSR (Figure 61), but we have not investigated the Warm River fishery since the 1980s (Brostrom and Spateholts 1985; Brostrom 1987). During the 1980s, we implemented a number of fishery investigations including creel surveys, fish tagging, electrofishing, snorkeling, and redd surveys in several sections of the Warm River to identify spawning and rearing areas, estimate population abundances, and estimate species composition. Based on data collection from the 1980s, the fishery consists of wild BNT, RBT, and Brook Trout *Salvelinus fontinalis* (BKT), as well as catchable RBT that are continued to be stocked annually.

In addition to Warm River, we have not investigated Medicine Lodge Creek (Figure 62) since the early 2000s (High 2006) and we ceased stocking this fishery regularly in the 1980s. Medicine Lodge Creek is classified as a Sinks Drainage (IDFG 2019) because the creek flows into the Snake River Plain aquifer and is not a tributary to a river or lake. At this time, the fishery is managed as a wild RBT fishery with a two trout limit and no harvest of YCT (IDFG 2019). We revisited Medicine Lodge Creek and Warm River to fill data gaps and collect up-to-date information on the status of these fisheries.

Corral Creek was last sampled in the early 2000s as part of the statewide status assessment of YCT (Meyer et al. 2006). Corral Creek is a small tributary in the Beaver Creek drainage, which is also part of the Sinks Drainage area. Corral Creek is often disconnected from its confluence tributary of Rattlesnake Creek due to the sinks nature of streams in the Sinks Drainage, but the headwaters consist of perennial flows that support wild, native YCT. We revisited this stream to collect up-to-date data on its status and to collect tissue samples for genetics.

METHODS

Warm River

We sampled two reaches in Warm River; the Springs reach with a downstream boundary about 1.5 km downstream from the Warm River springs origin, and the Pole Bridge reach which was upstream of the Pole Bridge campground and about 12 km upstream from the spring's origin. In the Springs reach, we used a towed barge electrofishing unit, which consisted of a 4-m long inflatable raft outfitted with an aluminum frame, and two mobile anodes connected to 15-m cables. The cathodes consisted of three octopus cable bars that totaled 2.0 m in length and consisted of 15 cable danglers. We applied pulsed direct current (DC) electricity to the water using a 5,000-W generator and an Infinity model electrofisher (Midwest Lake Management, Inc., Polo, Missouri). We measured water temperature (°C) and conductivity (µS/cm) prior to active electrofishing using a handheld probe. We standardized to 2,750-3,250 W of power output based on water conductivity (Miranda 2009). Electrofishing began at the downstream point of the sampling reach and proceeded in an upstream direction for 1 km. Electrofisher settings were 25% duty cycle and 60 Hz. Two people operated the mobile anodes, and one person guided the raft and operated the control box. Four additional people were equipped with dip nets and captured stunned fish. All trout and Mountain Whitefish *Prosopium williamsoni* (MWF) were placed in a live well stationed in the raft. Oxygen was pumped into the live well through a fine bubbler air-stone (2 L/min). When the live well was at capacity, the crew stopped at the nearest riffle and processed fish by identifying species, measuring fish to the nearest mm (TL), and marking fish with a hole punch in the caudal fin or looking for marks.

In addition to barge electrofishing, we also used two Smith-Root LR-24 backpack electrofishing units to sample stream margins and areas that the raft anodes could not reach. The backpack electrofishing was conducted by two additional people who operated the units and netted fish, then placed fish in buckets with fresh water on the bank. When the buckets were filled to capacity, those fish were transported to the aerated live well. For all backpack electrofishing we standardized output to 100 W using pulsed DC with a 25% pulse width and a frequency of 60 Hz (Meyer et al. 2021). We conducted single-pass marking and recapture runs in Warm River. All trout and Mountain Whitefish *Prosopium williamsoni* encountered that were ≥ 150 mm were marked with a hole punch in the caudal fin prior to release. The marking run was conducted on one day and the recapture run was conducted seven days later.

In the Pole Bridge reach, we used two backpack electrofishing units with four additional netters (i.e., a six-person crew) to sample the fish population. Electrofishing units were standardized as described for the Springs reach. Personnel operating the electrofishing unit also netted fish. The reach length was 200 m, and the reach began and ended at a riffle. We measured the wetted width at ten systematic transects in the reach at riffles, runs, and pools to estimate the average wetted width. A depletion estimate was conducted to estimate fish density and abundance.

Medicine Lodge Creek

Two people operated two Smith-Root LR-24 backpack electrofishers with three additional netters to sample two reaches of Medicine Lodge Creek. Personnel operating the electrofishing units also netted fish. We standardized the electrofishing units as described in the Warm River section. Both reaches in Medicine Lodge Creek were 500 m in length. We conducted single-pass marking and recapture runs in Medicine Lodge Creek in which all fish were identified to species and measured to the nearest mm (TL). All trout encountered that were ≥ 150 mm were marked with a hole punch in the caudal fin prior to release. The marking run was conducted on one day and the recapture run was conducted the following day (High 2006). We used the average wetted width and the abundance estimate from each reach to calculate density as the number of fish/100 m^2.

Corral Creek

In Corral Creek, we used a single backpack electrofishing unit and two netters to sample the fish population in two reaches of the creek. We standardized the electrofishing units as described in the Warm River section. Personnel operating the electrofisher also netted fish. We standardized the electrofisher as described in the Warm River section. Reach lengths were 100 m, and reaches began and ended at a riffle or other break in habitat types. We measured the wetted width at ten systematic transects in each reach at riffles, runs, and pools to capture the average wetted width for each reach. Three pass depletion estimates were conducted in each reach to estimate fish density and abundance.

Data Analysis

For Warm River and Medicine Lodge Creek, we estimated abundance for all salmonids ≥ 150 mm using a Peterson estimator with a Chapman modification using the mrClosed() function in the FSA package in program R (R Core Team 2022). We adjusted for size-selectivity by distributing the catch into size-classes to fit the recapture data, and we then calculated 95% confidence intervals (CI) for the abundance estimates. We evaluated species composition in the reach and catch-per-unit-effort (CPUE; number of fish/min of electrofishing) for each species on the marking run as the number of fish ≥ 150 mm caught per minute of active electrofishing. We estimated capture efficiency as the number of marked fish that were recaptured divided by the total number of fish that were available for capture multiplied by 100 (Chiaramonte et al. 2020). We estimated abundance and 95% CI in Corral Creek and upper Warm River reaches using a method (Carle and Strub 1978) for k-pass depletion estimates using the removal() function in the FSA package in Program R (R Core Team 2022). We then calculated density as the number of fish per 100 m^2 for both reaches. We estimated the average TL and standard deviation (SD) of all salmonids in all sampling locations.

RESULTS

Warm River

In the Springs reach, we marked 386 trout (sampled 1,117 trout in total) and 21 MWF on September 7, then we captured 383 trout (recaptured 149) and seven MWF (recaptured four) on September 14 (Table 19). Because we sampled over 500 fry and fingerlings on the marking run, we chose not to net fry and fingerlings on the recapture run to reduce fish processing time and mortality. Species composition of trout ≥ 150 mm was 54% BNT, 37% RBT, and 9% BKT. We estimated 1,132 trout/km (95% CI ± 187) including 480 BNT/km ± 67 and 468 RBT/km ± 125. We also estimated 34 MWF/km ± 15. We did not recapture enough BKT to estimate their abundance in the reach. The average TL of BNT was 234 mm (SD ± 102), RBT was 161 mm (± 62), BKT was 179 mm (± 36), and MWF was 259 mm (± 89), respectively (Figure 63).

We estimated 570 BKT ± 251 in the Pole Bridge reach, which equated to 43 BKT/100 m^2 and 3 BKT/min of electrofishing. The average TL of BKT in the Pole Bridge reach was 137 mm ± 38 (Figure 64).

Medicine Lodge Creek

We marked fish on October 18 and conducted the recapture run on October 19. In the downstream reach, we estimated 572 RBT (95% CI ± 51), which equated to 1,144 RBT/km ± 102, or 22 RBT/100 m^2. For RBT the average TL was 200 mm (SD ± 35; Figure 65). We estimated capture efficiency at 0.62 and CPUE was 5.3 RBT/min. In the upstream reach, we estimated 230 RBT ± 11, which equated to 460 RBT/km ± 22, or 7 RBT/100 m^2, and the average TL was 222 mm ± 39. We estimated capture efficiency at 0.64 and CPUE was 2.2 RBT/min.

Corral Creek

We sampled two reaches of Corral Creek on July 8, 2021. We estimated 38 BKT (95% CI ± 3.7) and 11 YCT ± 2.5 in the downstream reach of Corral Creek, which equated to 32 BKT/100 m^2 and 9 YCT/100 m^2. The catch rate (CPUE) was 2 BKT/min and 0.4 YCT/min. The average TL of BKT in the downstream reach was 104 mm (SD ± 48; Figure 66) and YCT was 115 mm (± 47; Figure 67). We estimated 49 YCT ± 2.9, which equated to 39 YCT/100 m^2 in the upstream reach of Corral Creek. The catch rate was 2.5 YCT/min and the average TL was 107 mm (± 37).

DISCUSSION

Warm River

The investigations into the Warm River fishery in the 1980s provided insights into the success of BNT introductions and the population structure of BNT and RBT (Brostrom 1987). Unfortunately, abundance estimates were only conducted in the lower portion of Warm River near the confluence with the HFSR, but there was an overwhelming number of age-0, or young-of-the-year, BNT and RBT. In addition, when age and growth analyses were conducted on scale samples the majority of ages were 0–2, suggesting that Warm River is a natal and juvenile rearing area for migratory trout. In the current study, we observed a high abundance of age-0 trout such that we avoided netting them in the recapture run, furthering our understanding that Warm River is an important spawning and rearing area. However, in our length frequency we observed two peaks of abundance of RBT and BNT between 60-110 mm and 150-210, but we also observed another peak of BNT abundance between 300-410 mm. The third peak in BNT abundance, but not in RBT abundance, in the length frequency suggests that these fish were likely migrating into the reach to spawn. In the 1980s, we also observed shifts in abundance of fish estimated to be >age-2 in the fall and spring suggesting immigration to spawn. The prevalence of cold, clean spring water, gravel, and aquatic vegetation makes this reach important for the success of wild BNT and RBT in the Warm River–HFSR complex.

The total trout abundance in our Springs reach estimate was 1,132 trout/km, which is very similar to what we estimated in the 1980s (1,129 trout/km) in a reach downstream several km from the Springs. But, the 1980s estimate for that reach consisted of only BKT and RBT whereas in the current study BNT composed the greatest proportion of the species at 54% in the Springs reach. With this shift in species composition in mind, it is obvious that BNT out-competed BKT and BKT continued to shift distribution farther upstream into the Pole Bridge area where they were the only salmonid that we encountered. Although Warm River is overshadowed by the popularity of the HFSR, it offers small stream fishing opportunities for wild trout with abundances over 1,000 fish/km.

Medicine Lodge Creek

Medicine Lodge Creek is not near any large town or fishing hub, but it does offer opportunities for people wanting to find high abundances of RBT in a small stream setting. Rainbow Trout abundance has increased in both reaches since it was last investigated in the early 2000s (High 2007). In the early 2000s, we estimated 4.94 RBT/100 m^2 in the downstream reach and 1.37 RBT/100 m^2 in the upstream reach. In the current study, abundance increased to 21.58 RBT/100 m^2 in the downstream reach and up to 7.30 RBT/100 m^2 in the upstream reach. Medicine Lodge Creek is another small stream resource where fishing for wild trout can provide anglers with high catch rates where RBT abundance is greater than 1,000 fish/km.

Corral Creek

The presence of BKT in the downstream reach of Corral Creek may be a cause for concern since BKT are known to displace native YCT in streams throughout the West (Varley and Gresswell 1988). However, this isolated population of YCT in a drainage that ultimately sinks into the Snake River Plain Aquifer is unique in that it is endemic and protects the genetic integrity of YCT in its native range. No BKT were sampled in the upstream reach, which may be a sign that there is a migration barrier between the downstream and upstream reaches. This conservation population of YCT likely does not receive many visits from anglers, but we should continue to monitor it and consider some kind of treatment to extirpate BKT from this stream if their distribution expands. The tissue samples we collected for genetic analysis are currently under processing at the Eagle Fish Genetics Lab and will be used to characterize native YCT genetic variability across the Upper Snake River Region and their native range.

MANAGEMENT RECOMMENDATIONS

1. Continue monitoring Warm River and Medicine Lodge Creek on an opportunistic basis to keep up-to-date records on these fisheries.

2. Investigate potential migration barriers in Corral Creek and continue monitoring BKT abundance and distribution.

3. Determine whether use of a piscicide is feasible for extirpating BKT in Corral Creek.

Table 19. Population estimate summary data for the Springs reach of Warm River sampled in September 2021.

Species	Marked	Captured	Recaptured	Abundance (± 95% CI)	Capture efficiency (recaptured/marked)	CPUE (fish/min)	Mean TL (± SD)
Brown Trout	220	224	108	480 (67)	0.49	0.87	234 (102)
Rainbow Trout	137	129	38	468 (125)	0.28	0.54	161 (62)
Brook Trout	27	29	2	-	0.07	0.11	179 (36)
Hatchery Rainbow Trout	2	1	1	-	0.50	-	-
Mountain Whitefish	21	7	4	34 (15)	0.19	0.08	259 (89)
All trout	386	383	149	1,132 (187)	0.39	-	-

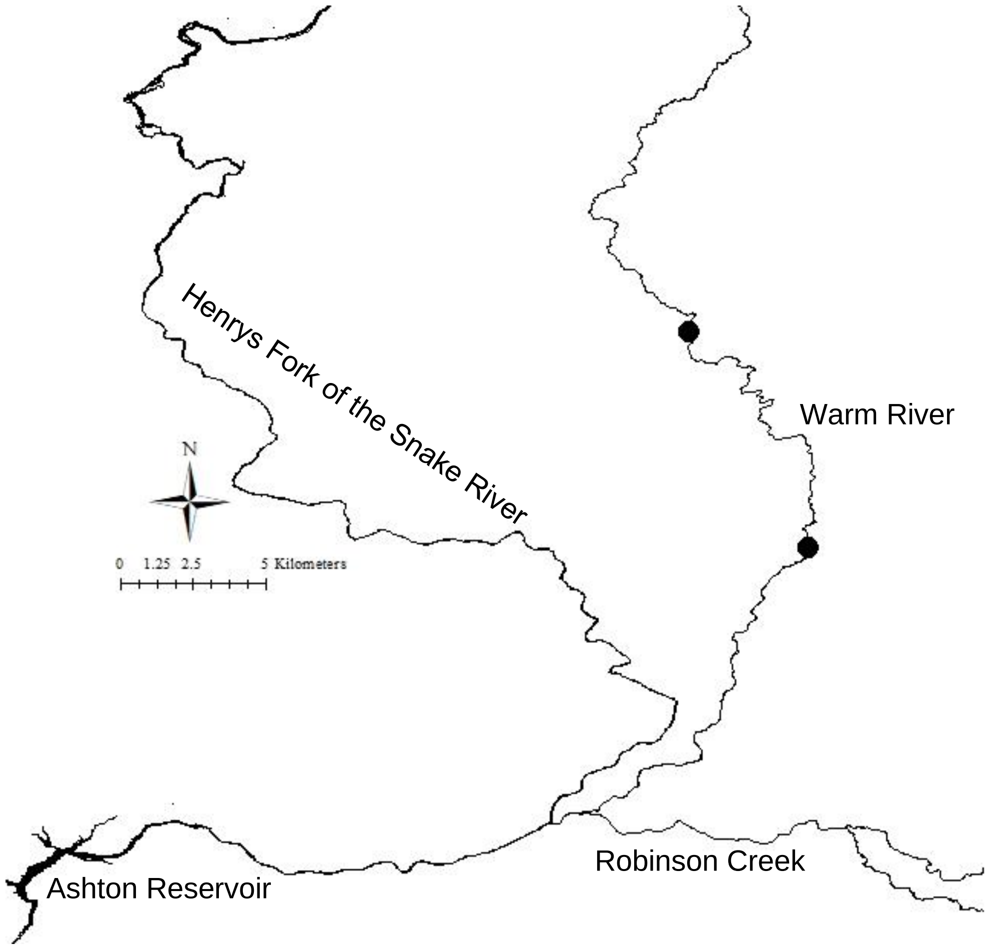

Figure 61. The Henrys Fork of the Snake River confluence with Warm River. The black circles represent the Warm River reaches sampled in 2021. The Springs reach is the downstream circle and the Pole Bridge reach is the upstream circle.

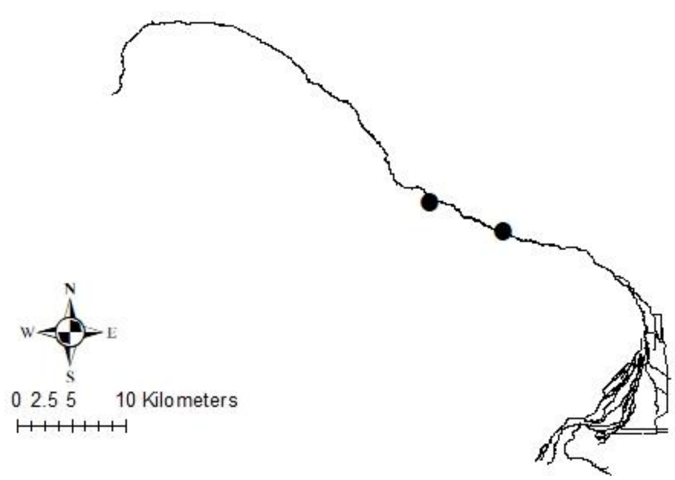

Figure 62. Medicine Lodge Creek with reaches sampled in 2021 represented by black circles.

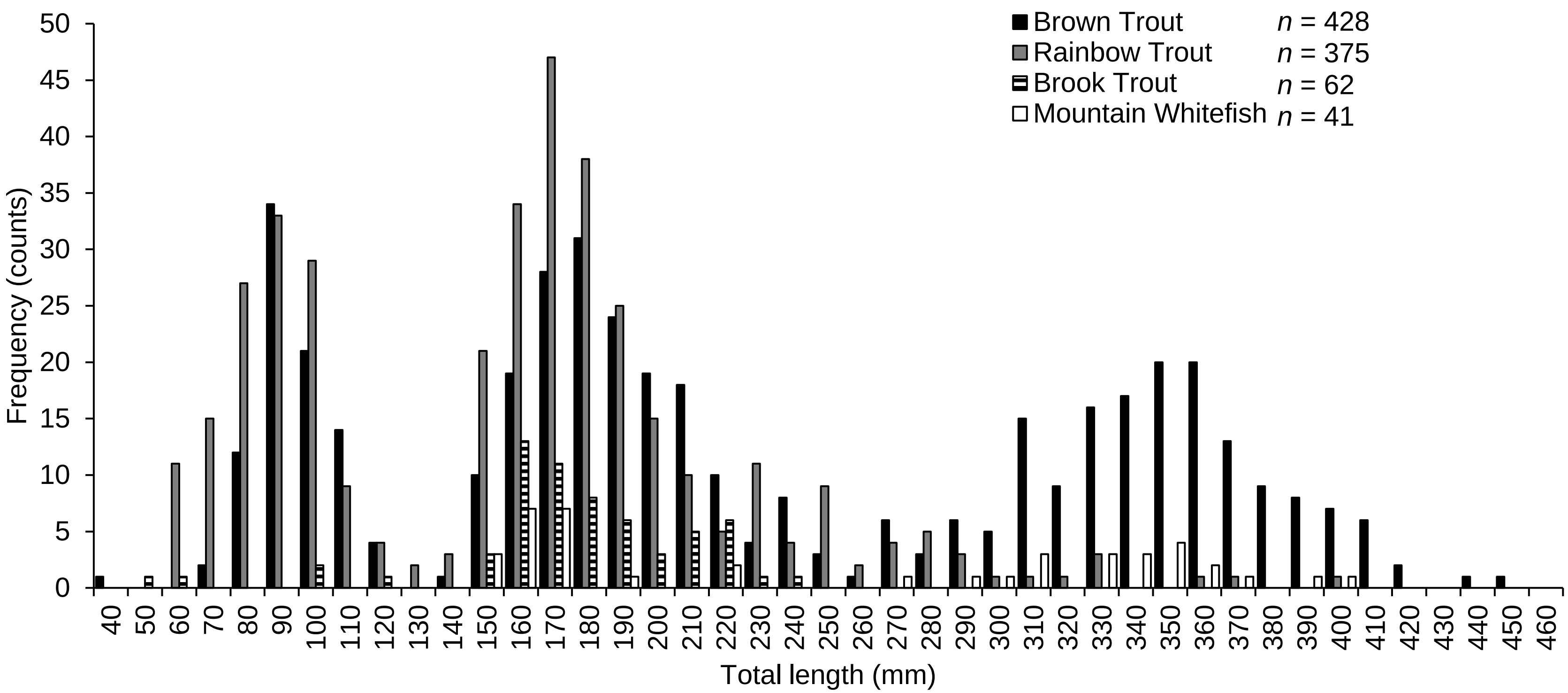

Figure 63. The length-frequency distributions of Brown Trout, Rainbow Trout, Brook Trout, and Mountain Whitefish sampled from the Spring reach of the Warm River in September 2021..

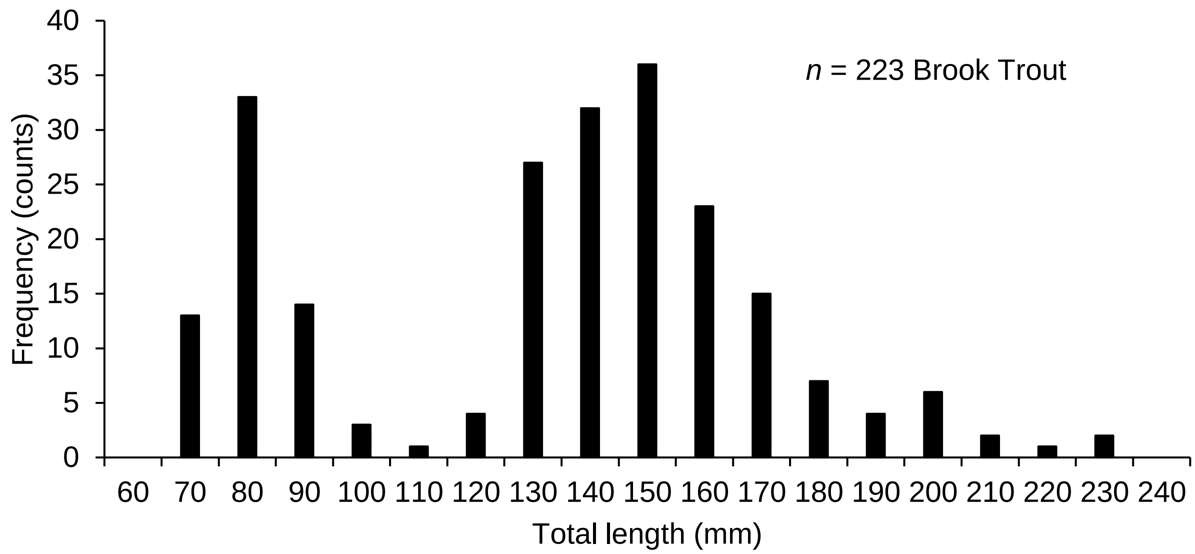

Figure 64. The length-frequency distribution of Brook Trout sampled in the Pole Bridge reach of Warm River in September 2021.

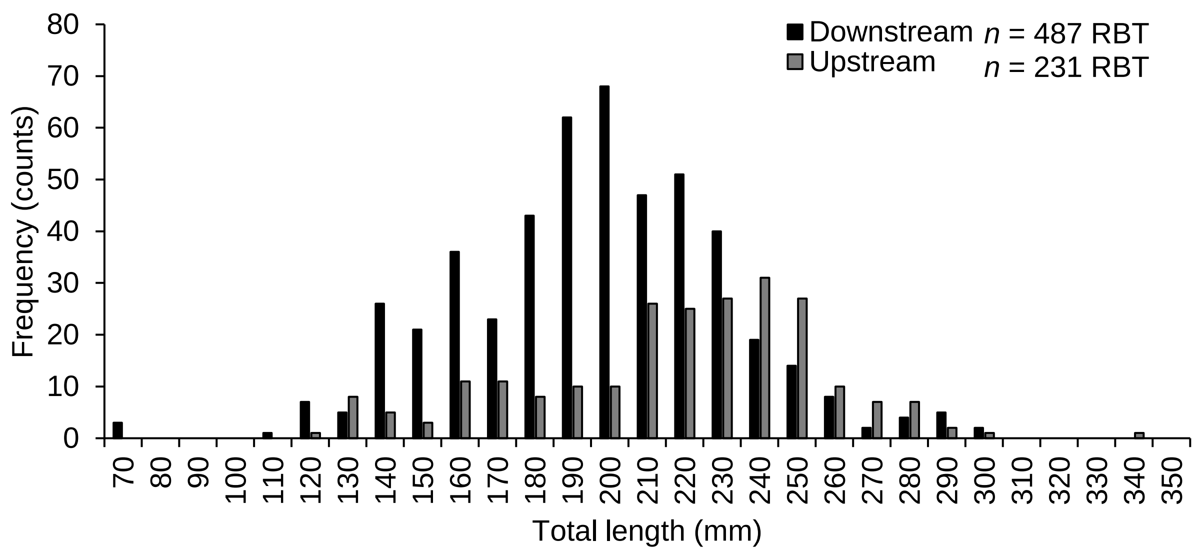

Figure 65. The length-frequency distribution of Rainbow Trout (RBT) sampled in two reaches of Medicine Lodge Creek in October 2021.

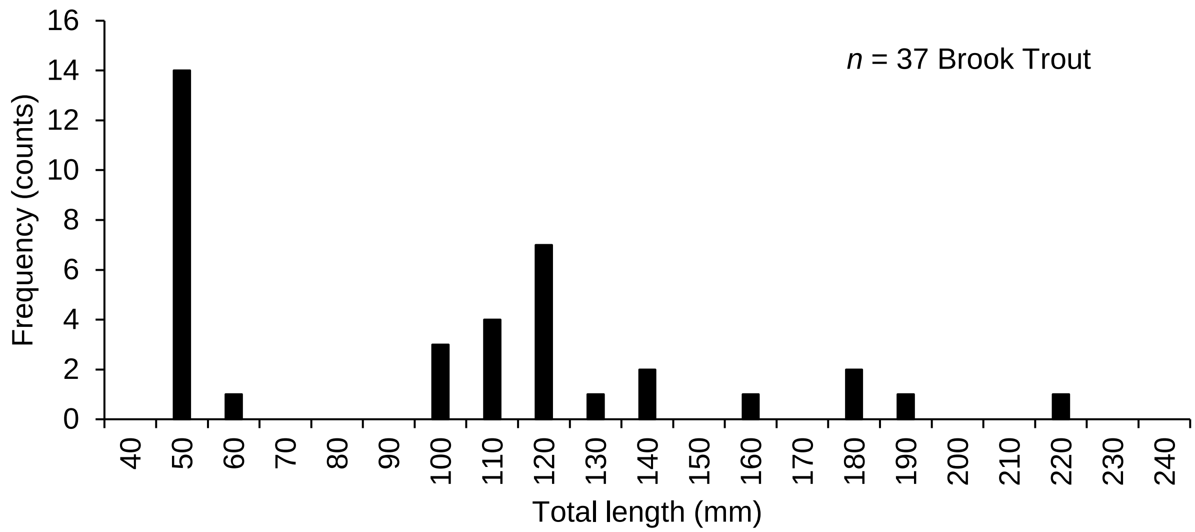

Figure 66. The length-frequency distribution of Brook Trout sampled from the downstream reach of Corral Creek in July 2021.

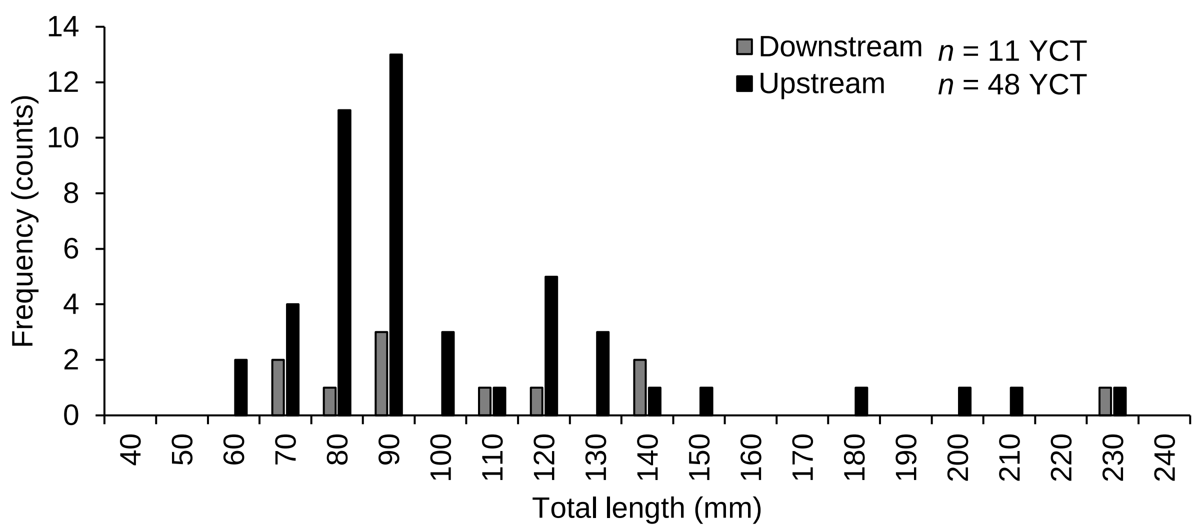

Figure 67. The length-frequency distribution of Yellowstone Cutthroat Trout (YCT) sampled in two reaches of Corral Creek in July 2021.

Appendix 1. Sampling locations, dates, and coordinates (WGS84) for small streams sampled in 2021.

Sampling location	Date (m/d/y)	Latitude	Longitude
Warm River Pole Bridge	9/7/2021	44.26410	-111.29043
Warm River Springs	9/7/21; 9/14/21	44.19684	-111.25316
Medicine Lodge downstream	10/18/21; 10/19/21	44.28711	-112.49160
Medicine Lodge upstream	10/18/21; 10/19/21	44.31246	-112.55309
Corral Creek downstream	7/8/2021	44.37324	-112.04752
Corral Creek upstream	7/8/2021	44.38664	-112.04054

HENRYS LAKE

ABSTRACT

Henrys Lake is one of the most popular recreational fisheries in Idaho and supports a robust trout fishery which includes Yellowstone Cutthroat Trout *Oncorhynchus clarkia bouvieri*, Yellowstone Cutthroat x Rainbow Trout *O. mykiss*, and Brook Trout *Salvelinus fontinalis*. We used 50 gill-net nights of effort in the spring of 2021 in historical locations to evaluate the trout populations of Henrys Lake, hereafter referred to as traditional nets or locations. Total trout catch per unit effort (CPUE) was 10.6 ± 1.7 (95% CI) trout per net night, which was below our 2020 CPUE of 14.9, the 30 year long-term mean of 11.8 and very close to the management target of 11 trout per net night. Mean relative weights for all trout species (all sizes combined) ranged from 84 to 91 and has slightly decreased compared to prior years. Utah Chub *Gila atraria* (UTC) catch per unit effort (CPUE) was 23.7 UTC per net night (± 12.6). In order to assess whether net location affects catch, we completed the second year of using an additional 50 gill-net nights at randomized locations around the lake, hereafter referred to as randomized nets. Randomized nets exhibited higher total trout CPUE and included a more inclusive range of size classes than the traditional nets. We recommend adoption of a netting design that incorporates the traditional and random net designs. Through parentage-based tagging analyses of YCT captured in our gillnet survey, we estimated that 9.5% of Henrys Lake Yellowstone Cutthroat Trout are of wild-origin. We monitored dissolved oxygen levels under the ice to assess the likelihood of a winterkill event from December 9, 2020 through February 4, 2021. Based on depletion estimates, we predicted dissolved oxygen would not reach critical levels (10 g/m^3) prior to the April 1 recharge date and did not deploy aeration pumps.

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INTRODUCTION

Henrys Lake, located in eastern Idaho in the Greater Yellowstone Ecosystem, has provided a recreational trout fishery since the late 1800s (Van Kirk and Gamblin 2000). A dam was constructed on the outflow of the natural lake in 1924 to increase storage capacity for downstream irrigation. This dam increased total surface area to 2,630 ha with a mean depth of 4 m and inundated lower portions of tributary streams. The mouths of tributary streams historically provided spawning habitat for adfluvial Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri*, prompting concerns for recruitment limitations. To mitigate for this potential loss of recruitment, the Idaho Department of Fish and Game (IDFG) acquired a private hatchery on the shores of Henrys Lake and began a fingerling trout stocking program that continues today (Garren et al. 2008). The lake supports a robust fishery for wild and hatchery native Yellowstone Cutthroat Trout (YCT), hatchery hybrid trout (Yellowstone Cutthroat Trout x Rainbow Trout *O. mykiss*; HYB) and hatchery Brook Trout *Salvelinus fontinalis* (BKT), with a mean of approximately 130,000 hours of annual angling effort. Surveys of Idaho's anglers indicate Henrys Lake has been the most popular lentic fishery in the state (IDFG 2001). Since 1923, IDFG has stocked a total of over 92 million YCT, 11.5 million HYB, and 4.3 million BKT. Stocking ratios averaged 84% YCT, 12% HYB, and 4% BKT from 1966 through 2010. Beginning in 1998, all HYB were sterilized prior to release to reduce the potential for hybridization with native YCT. Although hybridization was not a concern with BKT, only sterile fingerlings have been stocked since 1998 (with the exception of 50,000 fertile fish in 2003) to reduce the potential for naturally-reproducing BKT to compete with native cutthroat.

Anglers view Henrys Lake as a quality fishery capable of producing trophy trout, and the lake is currently managed as a trophy trout fishery (IDFG 2019). As early as the mid-1970s, 70% of interviewed anglers preferred the option of catching large fish even if it meant keeping fewer fish (Coon 1978). Since that time, management of Henrys Lake has provided restrictive harvest regulations consistent with providing a quality fishery as opposed to liberal harvest regulations that are more consistent with a yield fishery. In 1984, fisheries managers created specific, quantifiable objectives to measure angling success on Henrys Lake. Based on angler catch rate information and harvest data collected during creel surveys conducted between 1950 and 1984, managers thought it was possible to maintain angler catch rates of 0.7 trout per hour, with a size objective of 10% of harvested YCT ≥ 500 mm. These objectives remain in place today, although the size objective is now measured from gill-net sampling as opposed to fish caught by anglers and measured during creel surveys (IDFG 2019). To evaluate these objectives, annual gill-net monitoring occurs in May, immediately after ice off and full season creel surveys are conducted every three to five years.

Catch rates of trout estimated from annual gill-net surveys from 2013 through 2018 were lower than expected despite annual increases of fall stocked hatchery trout. This suggested trout may have experienced higher than normal mortality rates between 2012 and 2017. Some potential factors limiting trout survival may include abiotic factors (e.g. temperature, dissolved oxygen, nutrient concentrations) or biotic factors (e.g. food availability, intra or interspecific competition). A recent extensive water quality study was performed on the lake in 2018 and 2019. Phosphorus concentrations did not vary throughout the lake and indicated seasonal internal cycling of phosphorus instead of an external point source location. Throughout the two years, nitrogen to phosphorus ratios indicated the lake to be phosphorus limited with the highest limitations in May to June. Total algae and harmful blue-green algae peaked in the lake summer to early fall with small harmful algal blooms occurring in early October for both years (Vincent and Van Kirk, unpublished data). As expected, zooplankton biomass tracked primary production by location in the lake and through time. Zooplankton quality index (ZQI) as outlined by Teuscher 1999 indicated an abundance of zooplankton of preferred prey size for trout to support annual stockings of up to 1.9 million trout which has only been exceeded once in the past 35 years. Although zooplankton are an abundant prey resource in the lake, a 2018 dietary study indicates the dominant prey for hatchery fingerling YCT 1 to 4 weeks post stocking is amphipods (~84%), while zooplankton comprised only 3% of the diet (IDFG unpublished data). This indicated zooplankton may not be the most pertinent prey resource to monitor, rather aquatic invertebrate studies are more important to understand food-web dynamics in Henrys Lake. Stable isotope analysis (Flinders et al. 2016) indicated little overlap in dietary niches between Utah Chub and trout, but some dietary overlap most likely exists between salmonids due to niche similarities between YCT and HYB (Flinders et al. 2016; IDFG unpublished data) with BKT as more opportunistic feeders. Interspecific competition for certain prey may be an important mechanism in Henrys Lake especially at certain life stages. Abiotic factors, nutrient availability and food availability may continue to affect trout growth, survival, and fitness and should continue to be monitored along with the trout populations. Understanding these potential limiting factors is important for maintaining stable trout populations and achieving management objectives for these trout populations.

STUDY AREA

Henrys Lake is located 1,973 m above sea level, between the Henrys Lake and Centennial mountain ranges. The lake is approximately 29 km west of Yellowstone National Park. The lake is approximately 6.4-km long and 3.2-km wide, with a surface area of 2,630 ha (Figure 68). The outlet of Henrys Lake joins Big Springs Creek (~ 16 km downstream) to form the headwaters of the Henrys Fork Snake River.

OBJECTIVES

To obtain current information on the fish population, and to develop appropriate management recommendations to achieve management objectives stated in the State Fish Management Plan (IDFG 2019).

METHODS

Population Monitoring

Traditional nets

We set paired gill nets (floating and sinking) at six standardized locations for a total of 50 gill-net nights in Henrys Lake to monitor trout populations (Figure 68). All gill nets consisted of either floating or sinking types measuring 46-m long by 2-m deep, with equal-length panels ordered from smallest to largest of 2-, 2.5-, 3-, 4-, 5- and 6-cm bar mesh. All gill nets were set at dusk and retrieved the following morning.

Randomized nets

We set paired gill nets (floating and sinking) at 25 random netting locations for a total of 50 gill-net nights in Henrys Lake to quantify if our traditional nets capture a representative sample of the fish community. Paired gill nets at each random location, hereafter referred to as randomized nets were deployed at least three days apart (Figure 68). Sites were randomly selected by overlaying a grid system (100 m x 100 m) using mapping software. Gill net specifications were identical to our traditional nets as described above.

We identified captured fish in all gill-nets to species and recorded TL and weights (g). Gill-net catch per unit effort (CPUE) was calculated as the mean number of fish captured per gill-net night with 95% CI and divided by the total number of nets for traditional and randomized nets separately. Due to the high variability and schooling behavior of Utah Chub *Gila atraria* (UTC), we could not assume normal distribution of UTC throughout the lake, so we calculated CPUE as the median UTC caught per gill-net night.

We calculated relative weights (W_r) by dividing the actual weight of each fish (in grams) by a standard weight (W_s) for the same length for that species and multiplied by 100 (Anderson and Neumann 1996). Relative weights were averaged (ratio of the means) for each length class (< 200 mm, 200-299 mm, 300-399 mm and fish > 399 mm). We used the formula, $\log W_s = -5.194 + 3.098 \log TL$ (Anderson 1980) to calculate W_r of HYB, $\log W_s = -5.189 + 3.099 \log TL$ for Cutthroat Trout (Kruse and Hubert 1997) and $\log W_s = -5.186 + 3.103 \log TL$ for BKT (Hyatt and Hubert 2001a). For Utah Chub, we used the formula $\log W_s = -4.984 + 3.049 \log TL$ (IDFG, unpublished data).

We calculated proportional size distribution (PSD) and relative stock density (RSD-400 and RSD-500) to describe the size structure of trout populations in Henrys Lake. We calculated PSD for YCT, HYB and BKT using the following equation:

$$\text{PSD} = \frac{\text{number} \geq 300 \text{ mm}}{\text{number} \geq 200 \text{ mm}} \times 100$$

We calculated RSD-400 for YCT, HYB, and BKT using the following equation:

$$\text{RSD-400} = \frac{\text{number} \geq 400 \text{ mm}}{\text{number} \geq 200 \text{ mm}} \times 100$$

Criteria used for PSD and RSD-400 values for YCT, HYB, and BKT populations were based on past calculations and kept consistent for comparison purposes. We also calculated RSD-500, using the same equation as above, but used the number of fish greater than 500 mm as the numerator. This methodology (and size designation) is used on other regional waters to provide comparison between lakes and reservoirs throughout the Upper Snake Region.

Ages

We removed the sagittal otoliths of trout captured in all gill nets for age and growth analysis. After removal, all otoliths were cleaned and stored in individually-labeled vials and were analyzed as whole otoliths. Whole otoliths were immersed in water on a slide and the annuli were counted. Two trained readers independently assigned ages for each structure without reference or knowledge of fish length. A total of ten otoliths per 20 mm size class for each species (BKT, HYB and YCT) were randomly subsampled and their ages were assessed. When less than ten otoliths were present per size class, all otoliths were used to assign ages to fish. Images of otoliths were captured using a microscope interfaced with a desktop computer and digital images were taken of whole otoliths. The von Bertalanffy (1957) growth model was used to fit length:

$$l_t = L_\infty\left(1 - e^{-K(t-t_0)}\right),$$

where l_t is length at time t, L_∞ is the asymptotic length, *K* is a growth coefficient, and t_0 is the time coefficient at which length would theoretically be 0. We created an age-length key from our subsample of BKT, HYB, and YCT. We calculated mean and standard deviation length-at-age using Isermann and Knight (2005) method and the FSA package in program R (R Core Team 2021).

Parentage-Based Tagging

Parentage-based tagging (PBT) has been ongoing in Henrys Lake from 2017 to assess the composition of wild and hatchery trout in the lake. We structured our genetic sampling into three separate efforts to address this question. First, we collected genetic samples from wild broodstock during spawning sessions each winter to establish the PBT baseline. Second, we collected genetic samples from YCT and HYB caught in gill nets during spring population sampling. Finally, we used backpack electrofishing to capture YCT juveniles in four main tributaries to Henrys Lake (i.e. Howard, Targhee, Timber, and Duck Creeks). We sampled on these dates using backpack electrofishing methods outlined in Heckel et al. (2019). We removed a small portion of the caudal fin from fish captured across all methods, and we stored fin clips on pre-labeled Whatman sheets that we sent to Eagle Fish Genetics Lab in Eagle, Idaho for genotyping (Delomas et al. 2021).

Each spawn season we collected genetic samples from all YCT, and we stored samples on Whatman paper appropriately labeled by spawn date and lot number. Whatman paper was pre-labeled with seven horizontal sample locations on each plane. The first seven slot plane was identified as male with the next plane identified as female. These two horizontal planes were identified as Family 1. This was then repeated vertically down the Whatman paper with the next two male/female planes identified as Family 2 and so forth.

We collected genetic samples from all phenotypically-identified YCT and HYB trout encountered during our annual gill-net survey to assess wild vs hatchery origin of fish and provide validation to staff phenotypic identification of YCT and YCT x RBT trout hybrids in the field. Genetic samples were obtained via upper caudle fin clips from all phenotypically identified YCT and HYB and stored on Whatman Paper. All samples were sent to the Eagle Fish Genetics Lab located in Eagle, Idaho for genotyping. Genetic samples were obtained from fry in Howard, Targhee, Timber, and Duck Creeks to assess annual wild YCT fry production in the lake. We used a backpack electrofishing unit to sample the n the main tributaries to Henrys Lake including to capture out migrating YCT fingerlings. A 100-m transect was sampled at each creek on July 7 and 29, 2021, except Targhee Creek which as only sampled on the second date. We collected a genetic sample from each phenotypically identified YCT captured (>100 mm) and stored on Whatman paper. All samples were also sent to the Eagle Health Genetics Lake in Eagle, Idaho for genotyping.

Winter Dissolved Oxygen

Winter dissolved oxygen concentrations (mg/L), snow depth (m), ice thickness (m), and water temperatures (°C) were measured at five established sampling sites (Pittsburg Creek, Outlet, County Boat Dock, Wild Rose, and Hatchery; Figure 68) on Henrys Lake between December 9, 2020 and February 4, 2021. Holes were drilled in the ice with an electric ice auger prior to sampling. A YSI model Pro-20 oxygen probe was used to collect dissolved oxygen and temperature readings at the bottom of the ice and at subsequent one-meter intervals until the bottom of the lake was encountered. Dissolved oxygen mass was calculated from the dissolved oxygen probe's mg/L readings and converted to total mass in g/m^3. This was a direct conversion from mg/L to g/m^3 (i.e., 1000 L = 1 m^3). The individual dissolved oxygen readings at each site were then summed to determine the total available oxygen within that sample site. To calculate this value, we used the following formula: [Mean (bottom of ice + one m)] + [Sum (readings from two m to lake bottom)] = Total O_2 mass.

The total mass of dissolved oxygen at each sample site was then expressed in g/m^3 (Barica and Mathias 1979). Data were then transformed using the natural logarithm (ln*)* for regression analysis. We used linear regression to develop a dissolved oxygen depletion model used to predict the likelihood of the Henrys Lake environment reaching the critical threshold for fish survival. Historically, the critical threshold at Henrys Lake has been 10 g/m^3. The likelihood of reaching the critical dissolved oxygen threshold prior to April 1, the projected recharge date, is one factor which was used to decide whether to deploy aeration at the mouth of Hatchery Creek and to provide indication if low oxygen levels would be an impediment for hatchery YCT spawn operations relative to egg quality, eye-up rate, and survival rates.

RESULTS

Population Monitoring

Traditional nets

We collected 1,713 fish in 50 net nights in May 2021 (May 5 – 15), with our traditional gill net survey. Species composition of the gill-net catch was 69.1% UTC, 18.3% YCT, 10.2% HYB, and 2.5% BKT. Gill-net catch rates (CPUE) for all trout species combined was 10.6 (95% CI; ± 1.7) trout per net night (Figure 69). Mean trout CPUEs were highest for YCT at 6.3 (± 1.1), followed by HYB at 3.5 (± 1.0) and BKT at 0.9 (± 0.43; Figure 69 and Figure 70). The mean TL of HYB was 400 mm (range 212 - 570 mm; Figure 71). The mean TL of YCT was less than HYB with a mean TL of 389 mm (range 202 - 702 mm). Brook Trout TL ranged from 168 to 500 mm, with a mean of 337 mm (range 168 – 500 mm; Figure 71, Table 20). The mean CPUE of UTC was 23.7 (±12.6; Figure 72). Utah Chub gill-net CPUE has increased from our 2020 CPUE of 13.8 (± 18.6). Utah Chub TLs ranged from 131 to 371 mm with a mean of 241 mm (Figure 73).

Proportional size distribution (PSD) was highest for HYB and YCT at 76, followed by BKT (68). Relative stock density (RSD-400) was highest for YCT (59), followed by HYB (56) and BKT (49; Table 21). Overall, the mean W_r was 93 ± 0.4 (Table 21; Figure 74). For trout species, mean W_r for all size classes combined was 91 ± 1.4 for HYB, 89 ± 2.9 for BKT, and 84 ± 0.8 for YCT (Table 20; Figure 75).

Randomized nets

We collected 2,939 fish in 50 net nights with our randomized location gill nets from May 6 – 25, 2021. Catch composition was 67.5% Utah Chub, 21.7% YCT, 7.1% HYB, and 3.6% BKT. CPUE and 95% CI for all trout species combined was 19.1 ± 4.9, which was higher than CPUE in the traditional gill nets (10.6; Figure 70). Mean trout CPUE were highest for YCT at 12.7 (± 4.2), followed by HYB at 4.2 (± 1.4), and BKT at 2.1 (± 0.8; Figure 70). Hatchery hybrid trout TL ranged from 216 to 706 mm with a mean of 419 mm (Figure 71). Lengths for YCT ranged from 162 to 602 mm with a mean of 411 mm. Brook Trout TL ranged from 172 to 572 mm with a mean of 383 mm (Table 22). The mean CPUE and 95% CI for UTC was 39.7 (± 16.4). This was not significantly higher, as indicated by overlapping confidence intervals, than the traditional location nets which had a mean 23.7 (± 12.6; Figure 3). Utah Chub TL ranged from 134 to 381 mm with a mean of 229 mm (Figure 73).

Proportional stock density (PSD) was highest for YCT (90) followed by BKT (85), and HYB (81). Relative stock density (RSD-400) was highest for YCT (73), followed by BKT (62) and HYB (59; Table 21). Mean W_r and 95% confidence intervals for all size classes combined was 92 ± 1.1 for HYB, 91 ± 2.6 for BKT, and 86 ± 1.4 for YCT (Table 21). Mean W_r for UTC (all sizes combined) was 94 ± 0.8 (Table 21).

Ages

We estimated the ages of 279 YCT, 275 HYB, and 122 BKT. Ages ranged from age-1 to age-8 for YCT, age-2 to age-7 for HYB, and age-1 to age-5 for BKT (Table 23). Hatchery hybrid trout neared L_∞ the quickest relative to the other trout species in Henrys Lake with (K = 0.32, t_0 = 0.58 years, and L_∞ = 774 mm) followed by Brook Trout (K = 0.24, t_0 = -0.30 years, and L_∞ = 781 mm. Yellowstone Cutthroat Trout did not fit the model where the L_∞ and K starting values were negative. To fit the model, we manually set the values of t_0 and K to 0 and 0.3, respectively, and the L_∞ value to the largest observed total length of 702 mm, as outlined by Ogle (2016). Yellowstone Cutthroat Trout model parameters included (K = 0 and 0.3, t_0 = 0.0 and 0.3, and L_∞ = 702 mm; Figure 76).

Parentage-Based Tagging

A total of 1,764 YCT were genotyped for the 2021 Henrys Lake YCT spawn. A combined total of 1,315 YCT and HYB genetic samples from our gill-net survey were analyzed for parentage. Of the samples collected, 21% of the samples (278 fish) were genetically identified as HYB, 73% (963 fish) as YCT, and 6% (74 fish) failed to genotype. Of the 963 genotyped YCT, 867 were of hatchery origin and 96 had no PBT data on file. However, 5 YCT without parental data exhibited lengths > 470 mm, suggesting these fish are older than 4 years (older than Brood Year 2017 and prior to the institution of our PBT monitoring program) and would not assign to a PBT marked brood year (BY). This left a total of 91 YCT estimated to be born within the PBT monitoring window (2017 – 2021) without documented hatchery parents, suggesting that 9.5% of Henrys Lake YCT are of wild-origin. Of the hatchery-origin YCT, 23% (199 fish) were from BY2019, 9% (79 fish) were from BY2018, and 68% (589 fish) were from BY2017. We compared species identification from field staff during surveys to genotype results to evaluate the accuracy of determining YCT and hybrids. Field staff were 89% accurate in correctly identifying YCT during surveys using phenotype, and 82% accurate for HYB.

Winter Dissolved Oxygen

Total dissolved oxygen diminished from 55 to 30.2 g/m^3 at the Pittsburgh Creek site, from 32.2 to 19.4 g/m^3 at the Outlet site, from 38 to 20 g/m^3 at the County dock site, from 41 to 22.4 g/m^2 at the Wild Rose site, and from 42.4 to 25.7 g/m^3 at the Hatchery site from December 9, 2020 to February 4, 2021 (Table 24). The regression line using depletion estimates remained above the level of concern (10 g/m^3) for the duration of the annual Henrys Lake spawn. This year, the spawn began on February 18 and was completed on March 4, 2021. Trout did not congregate in large numbers at the mouth of Hatchery Creek following the spawn and no aeration was initiated (Figure 77).

DISCUSSION

Total trout CPUE and angler catch rates in Henrys Lake are at or above management objectives (IDFG 2019). Overall, the fishery is doing well with good growth rates and generally high relative weights. The CPUE for YCT (6.3) was slightly higher than the 10-year average of 6.2 YCT per net night. This indicates a stable population. Currently, no age classes are missing from the population suggesting a stable YCT population is likely to continue.

Total trout CPUE and angler catch rates in Henrys Lake are at or above management objectives (IDFG 2019). Overall, the fishery is doing well with good growth rates and generally high relative weights. The CPUE for YCT (6.3) was slightly higher than the 10-year average of 6.2 YCT per net night. This indicates a stable population. Currently, no age classes are missing from the population suggesting a stable YCT population is likely to continue. However, our PBT results suggest that year-class strength of hatchery YCT was variable and dominated by BY2017 fish (68% of the population). Managers should closely monitor YCT year-class strength in the coming years to better inform our knowledge of recruitment dynamics.

Utah Chub densities have been an ongoing concern at Henrys Lake due to the potential negative interactions between Utah Chub and trout. It is likely that Utah Chub were illegally introduced into Henrys Lake and were first documented in 1993. Utah Chub abundance began to increase in the late 1990s through the early 2000s and reached its highest mean CPUE of 50.5 UTC/net night in 2008. High UTC CPUE in 2008 was coupled with the highest CPUE for trout of 63.6 trout/net-night. Relative weights have varied across years since weights were first recorded to monitor body condition (2004) but are now the lowest observed across the dataset for YCT. McCarrick (2021) investigated interactions between UTC and YCT in Henry's Lake and identified that UTC abundance had a negative influence on YCT growth.

There is evidence of competition between UTC and trout for both food resources and space in many lakes and reservoirs. For example, in Schofield Reservoir, Utah, small (< 360 mm) Cutthroat and Rainbow Trout exhibited a high degree of dietary overlap with small (< 250 mm) UTC (Winters and Budy 2015). Furthermore, in Flaming Gorge Reservoir which borders Utah and Wyoming, introduced Utah Chub had a 99.7% diet overlap with Rainbow Trout (Schneidervin and Hubert 1987). As densities of Utah Chub continued to increase in Flaming Gorge Reservoir, kokanee growth declined dramatically (Teuscher and Luecke 1996). Although dietary overlap and competition for prey has been reported in these reservoirs, it is not likely occurring in Henrys Lake as stable isotope analysis has indicated minimal diet overlap exists between the two species (Flinders et al 2016; IDFG unpublished data).

Competition for resources is not limited to food only but could apply to available to habitat as well. McCarrick (2021) investigated factors influencing spatial distribution and overlap of trout and UTC in Henrys Lake. This research indicates there is limited spatial overlap between YCT and UTC in Henrys Lake (McCarrick 2022). Of the factors included in the analysis (i.e. water temperature, water depth, visibility, dissolved oxygen levels, and percent aquatic macrophyte cover) water temperature influenced species' distribution the most. Yellowstone Cutthroat Trout locations were negatively associated with warm water temperatures, while warmer water temperatures were positively associated with UTC presence (McCarrick et al. 2022). Both species occupied habitats with macrophyte cover and researchers observed limited overlap. Continued monitoring is warranted to fully understand the interaction between UTC and trout in Henrys Lake.

Annual gillnet surveys utilized to track changes in population size, structure, and composition have occurred for several years, but we are concerned that current netting locations may not be the most appropriate to sample the population. Surveys are conducted in the spring as soon as possible after ice out. During this time of year, YCT are congregated on shoreline areas (McCarrick et al. 2022). However, the designated gillnet locations for the survey are not located near shorelines. Thus, we set nets in the traditional locations in pelagic areas as well as randomized locations around the lake to be able to compare if results from traditional netting sites differ from randomized sites which include near shore habitats. Trout CPUE in the randomized nets was greater than CPUE in the traditional nets and the randomized nets captured more size classes at the upper and lower ends of the size structure compared to nets in traditional sampling locations. Thus, it appears netting location affects catch at Henrys Lake and it is important to select netting sites that incorporate all habitats available in the lake to obtain an accurate picture of the current state of the trout fishery. Based on our observations, a randomized netting design would be a more appropriate way to monitor trout populations in Henrys Lake and in the future. However, we would need to ensure that data collected with this design would be comparable with previous years when traditional netting locations were used. Thus, we recommend spending additional effort and sampling both traditional and randomized locations for a few years so that a correction factor could be created to allow us to compare data from randomized netting locations from future surveys with data collected from traditional pelagic areas during past surveys. No significant difference between and within net location with year was observed for UTC CPUE, indicating we can obtain a representative sample of the UTC population regardless of using traditional or randomized nets.

Winter dissolved oxygen profiles have been monitored annually for over three decades to inform managers of the potential for pending winter fish kills so that preventative measures could be taken. The dissolved oxygen profiles have proven to be useful in predicting hypoxic conditions in the lake which can result in a winterkill. Aeration with a Helixor system permanently installed on the shore of the lake near the hatchery has been the primary mitigation measure in the event of hypoxic conditions since it was installed in 1993. Although the aeration system has been periodically maintained, it was last utilized in 2015, and is due for a thorough evaluation of the effectiveness of the current system.

MANAGEMENT RECOMMENDATIONS

1. Continue our annual traditional gill-net monitoring survey while incorporating a randomized gill netting design to evaluate relative abundance, age, and growth of all trout species in order to quantify the current state of the fishery.

2. Continue to monitor UTC densities with annual gillnet surveys and evaluate potential effects of increased densities of UTC on trout if UTC numbers increase statistically.

3. Utilize PBT to evaluate the percentage of wild YCT production in Henrys Lake to help inform stocking densities.

4. Continue to monitor winter dissolved oxygen levels to evaluate if and/or when the aeration system should be used.

Table 20. Summary statistics of total length (TL), weight (WT), and relative weights (*W_r*) for Brook Trout (BKT), Yellowstone Cutthroat x Rainbow Hybrid Trout (HYB), Yellowstone Cutthroat Trout (YCT), and Utah Chub (UTC) collected using gill nets set at traditional netting locations in Henrys Lake, May 2021.

	BKT			HYB			YCT			UTC		
	TL (mm)	WT (g)	W_r	TL (mm)	WT (g)	W_r	TL (mm)	WT (g)	W_r	TL (mm)	WT (g)	W_r
Mean	337	553	89	400	791	91	389	646	84	241	212	93
Confidence level (95.0%)	33.8	141.8	2.9	14.2	73.9	1.38	9.0	38.5	0.83	3.4	8.6	0.4
Median	338	417	88	426	800	90	420	755	84	234	166	92
Minimum	168	48	75	212	87	70	202	67	52	131	25	68
Maximum	500	1404	112	570	2080	145	702	2852	111	371	818	143
Count	43	43	43	174	174	174	313	313	313	1183	1181	1181

Table 21. Stock density indices (Proportional Stock Density [PSD], Relative Stock Density – 400 mm [RSD-400], and RSD-500) and relative weights (W_r) for all trout species collected with traditional (T) and randomized (R) gill nets in Henrys Lake, May 2021.

	BKT		HYB		YCT		UTC	
	T	R	T	R	T	R	T	R
PSD	68	85	76	81	76	90	--	--
RSD-400	49	62	56	59	59	73	--	--
RSD-500	3	1	18	21	3	2	--	--
W_r								
<200 mm	82	82	--	--	--	92	90	90
200-299 mm	83	91	89	91	85	90	94	94
300-399 mm	93	96	94	92	87	84	93	92
>399 mm	93	91	91	93	83	83	--	--
Mean	89	92	91	92	84	85	93	93

Table 22. Summary statistics of total length (TL), weight (WT), and relative weights (*Wr*) or Brook Trout (BKT), Yellowstone Cutthroat x Rainbow Hybrid Trout (HYB), Yellowstone Cutthroat Trout (YCT), and Utah Chubs (UTC) collected using randomized gillnets at Henrys Lake, May 2021.

	BKT			HYB			YCT			UTC		
	TL (mm)	WT (g)	W_r	TL (mm)	WT (g)	W_r	TL (mm)	WT (g)	W_r	TL (mm)	WT (g)	W_r
Mean	383	732	92	419	937	92	411	732	85	229	175	93
Confidence level (95.0%)	18	79	1.8	14.3	99	1.7	4.7	20	0.7	2.2	5	0.4
Median	419	784	91	434	864	92	430	783	85	226	146	92
Minimum	172	44	66	216	97	69	162	42	60	134	31	56
Maximum	572	1,974	116	706	4,762	123	602	2,166	177	381	700	174
Count	107	107	107	192	192	192	637	636	636	2,003	2,001	2,001

Table 23. Mean total length-at-age, sample size (n), and range of lengths based on sagittal otoliths from Yellowstone Cutthroat Trout (YCT), Yellowstone Cutthroat x Rainbow Hybrid Trout (HYB), and Brook Trout (BKT) captured with combined samples from traditional gill nets and randomized gill nets set in Henrys Lake, May 2021.

		Mean Total Length-at-Age (mm)							
Species	Summary statistic	1	2	3	4	5	6	7	8
YCT	Mean TL (mm)	245	287	401	463	495	602	--	702
	Min TL	162	202	282	345	--	--	--	--
	Max TL	286	430	535	570	--	--	--	--
	No. Analyzed	12	90	87	87	1	1	--	1
HYB	Mean TL (mm)	--	280	395	491	548	700	674	--
	Min TL	--	212	272	383	421	--	629	--
	Max TL	--	385	515	554	663	--	706	--
	No. Analyzed	--	73	98	88	12	1	3	--
BKT	Mean TL (mm)	212	328	448	475	572	--	--	--
	Min TL	168	206	400	454	--	--	--	--
	Max TL	292	421	500	489	--	--	--	--
	No. Analyzed	24	33	61	3	1	--	--	--

Table 24. Dissolved oxygen (DO) (mg/l) levels recorded in Henrys Lake winter monitoring 2020 – 2021. Dissolved oxygen total mass was calculated from the dissolved oxygen probe's mg/L readings using the following equation: [Mean (bottom of ice + one meter)] + [Sum (readings from two meters to lake bottom)] = Total O_2 mass. Total mass was converted to g/m^3 using a direct conversion from mg/L to g/m^3 (i.e., 1000 L = 1 m^3).

Location	Date	DO Ice bottom	DO 1 m	DO 2 m	DO 3 m	DO 4 m	DO 5 m	Total g/m^3
Pittsburg Creek	12/29/2020	13	12.5	11.9	11.8	10	8.6	55.05
	1/6/2021	13.6	12.7	12.1	11.4	9.6	8	54.25
	1/13/2021	11.9	11.5	10.4	10.6	7.7	5.6	46.00
	2/4/2021	11.9	11.5	10.3	10	7.8	5.4	45.20
	12/9/2020	11.9	11.3	9.6	7.4	4.8	3.8	37.20
	12/22/2020	10.6	9.6	8.3	5.6	3.1	3.1	30.20
Outlet	12/29/2020	13	10.8	7.6	5.9	4	--	32.20
	1/6/2021	--	--	--	--	--	--	--
	1/13/2021	12.1	12	8	4.9	2.7	1.5	29.15
	2/4/2021	11.1	10.1	7.4	3.6	1.5	0.8	23.90
	12/9/2020	11.3	8.3	6	3	1.1	--	20.70
	12/22/2020	9.4	8.6	6.1	3.3	1	--	19.40
County Ramp	12/29/2020	13.4	12.9	11.8	8.1	5	--	38.05
	1/6/2021	--	--	--	--	--	--	--
	1/13/2021	12.2	11.5	9	6	1.5	--	28.35
	2/4/2021	12	11.3	7.9	4.1	0.9	--	24.55
	12/9/2020	--	--	--	--	--	--	--
	12/22/2020	10.2	7.6	5.6	3.3	2.2	--	25.70
Wild Rose	12/29/2020	12.9	12.5	12.2	10.3	4.6	--	41.40
	1/6/2021	12.2	11.5	9.4	3.9	2.8	--	40.20
	1/13/2021	11.8	11.7	10.4	5.7	2.4	1.3	31.55
	2/4/2021	11	10.7	9	4.9	2	1.1	27.85
	12/9/2020	13.2	10.3	8.9	6.8	5.5	--	28.45
	12/22/2020	9.9	9.8	6.4	3.6	1.5	1	22.35
Hatchery	12/29/2020	13.9	13	11.6	10.5	6.8	--	42.35
	1/6/2021	15	13.3	10.5	9.1	6.5	--	40.25
	1/13/2021	13.2	11.2	10.4	8.2	5.4	--	36.20
	2/4/2021	12.7	10.9	9.6	7.6	3.8	--	32.80
	12/29/2020	13.2	10.3	8.9	6.8	5.5	--	32.95
	1/6/2021	11.4	9.6	7.5	5	2.7	--	25.70

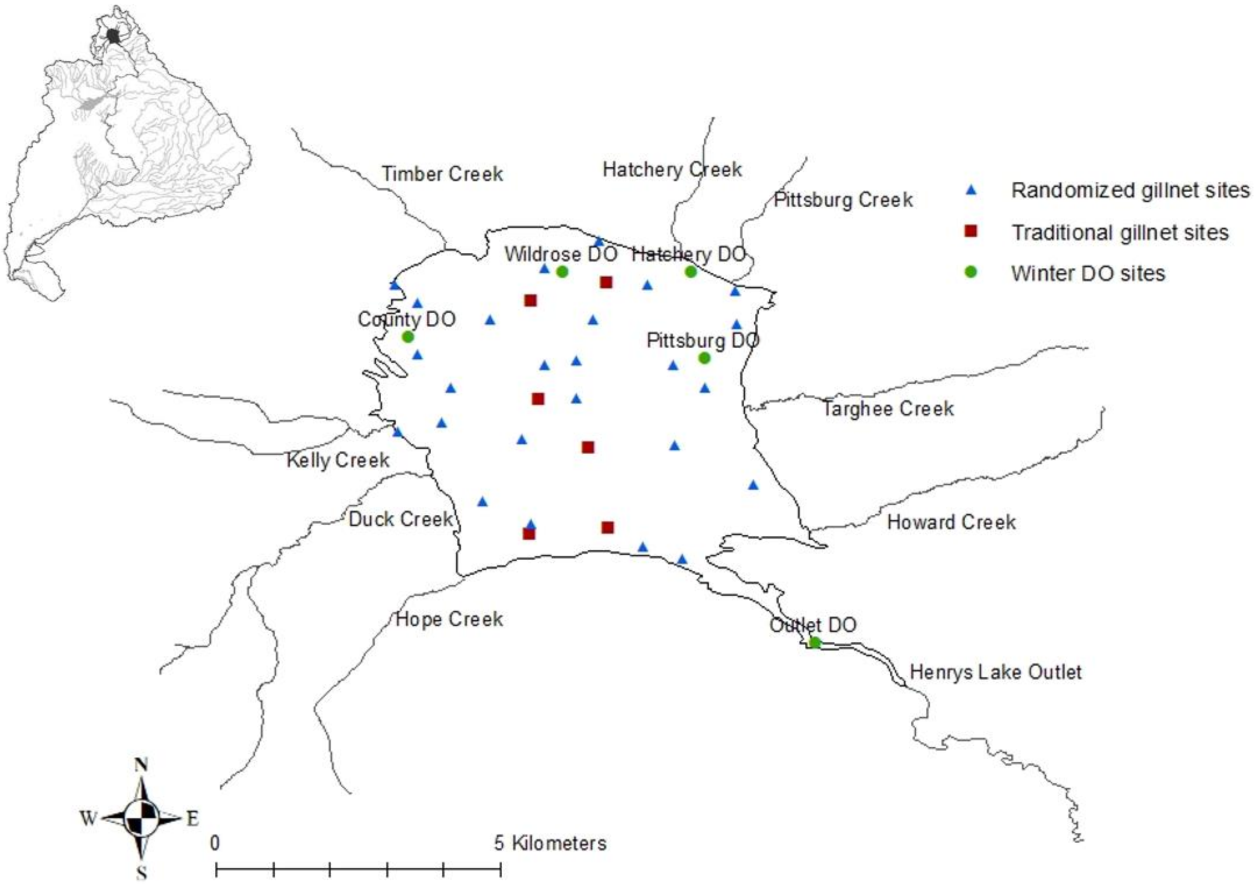

Figure 68. Location of randomized gill net sites (blue triangles), traditional gill net sites (red squares), winter dissolved oxygen sites (green circles), and major tributaries of Henrys Lake, May 2021.

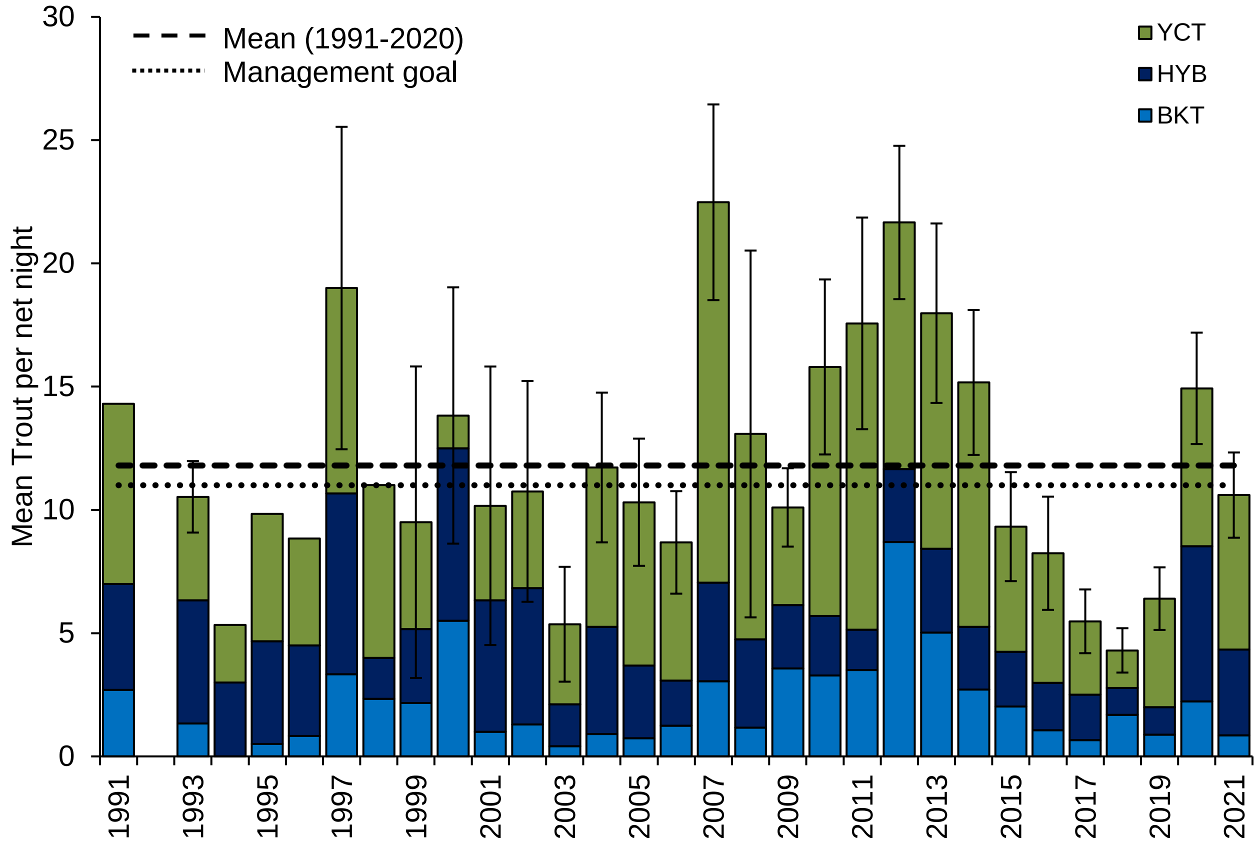

Figure 69. Catch per unit effort (CPUE) of trout per net night of traditional gillnetting sites for Yellowstone Cutthroat Trout (YCT), Yellowstone Cutthroat x Rainbow Trout (HYB), and Brook Trout (BKT) in Henrys Lake from 1991 – 2021. Error bars represent 95% confidence intervals. Horizontal lines represent the mean gillnetting CPUE from years 1991 – 2020 (dashed line) and management target of 11 trout per net night (dotted line).

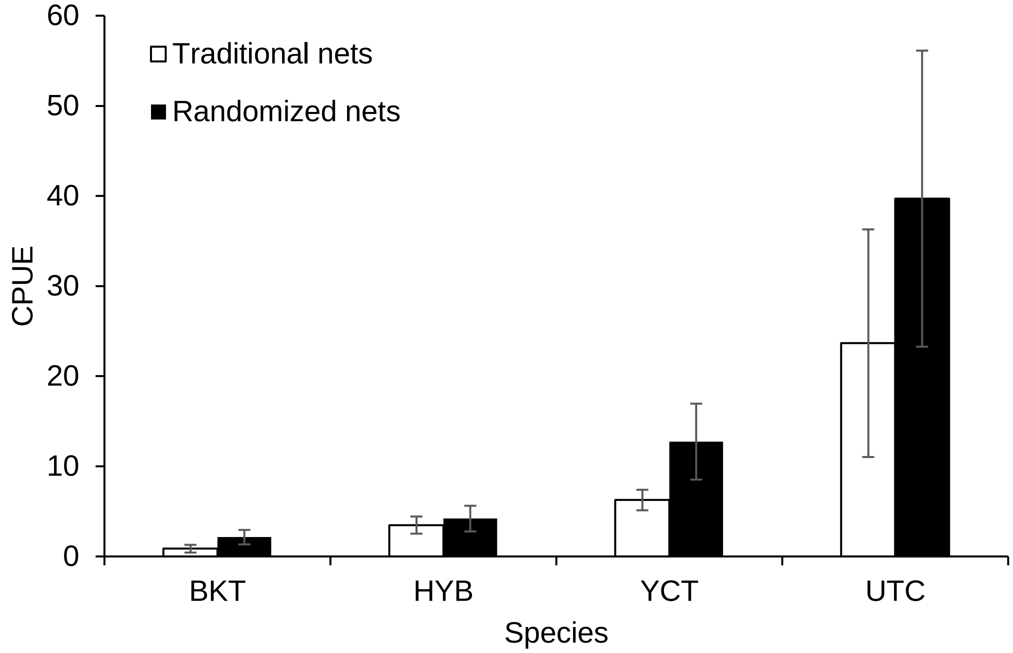

Figure 70. Catch per unit effort (CPUE) of trout per net night of traditional and randomized gill setting sites for Brook Trout (BKT), Yellowstone Cutthroat x Rainbow Trout (HYB), Yellowstone Cutthroat Trout (YCT), and Utah Chub (UTC) in Henrys Lake, May 2021. Error bars represent 95% confidence intervals.

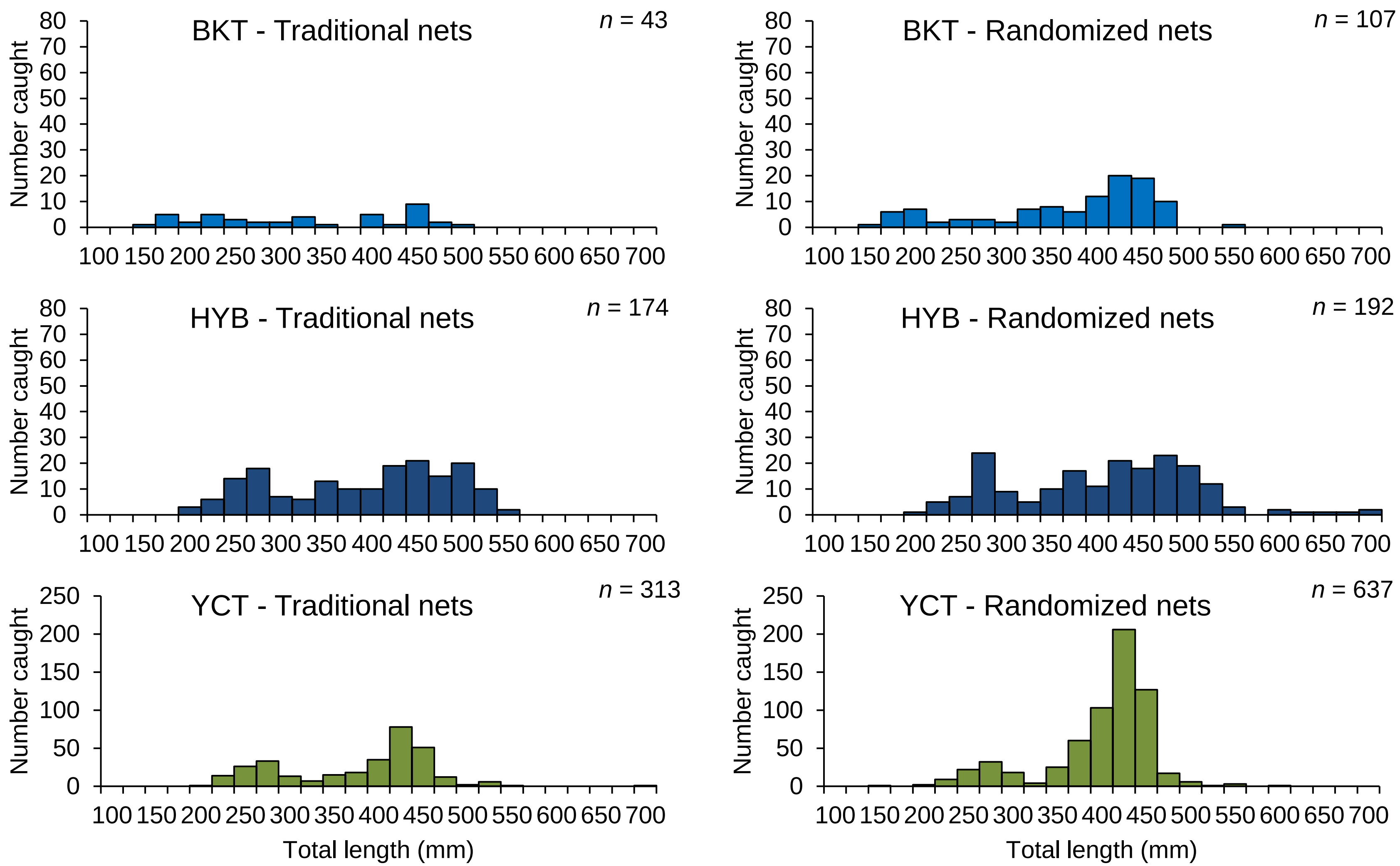

Figure 71. Brook Trout (BKT), Yellowstone Cutthroat x Rainbow Trout (HYB) and Yellowstone Cutthroat Trout (YCT) length frequency distributions from traditional gill nets (Left), and randomized gill nets (Right) set in Henrys Lake, May 2021. Note: Axis range change for YCT.

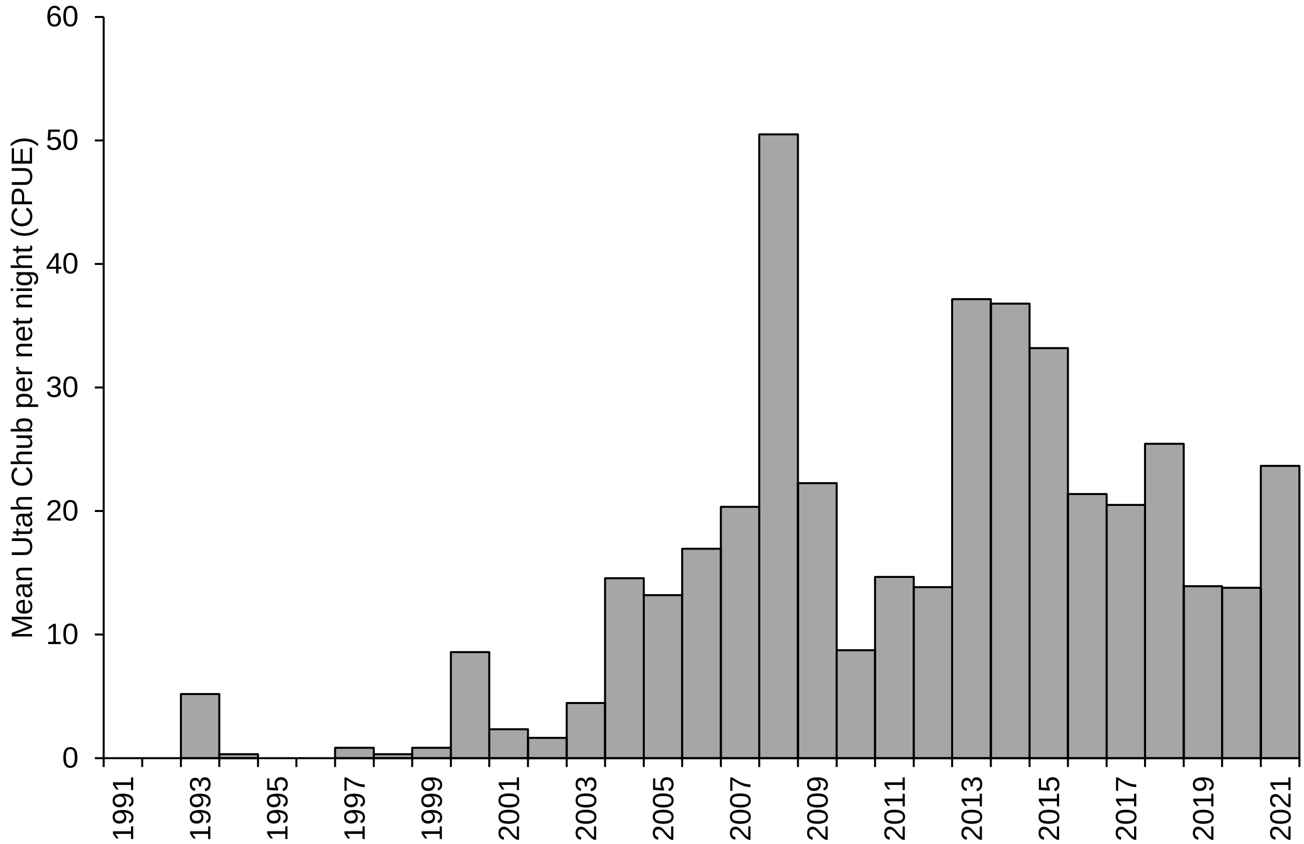

Figure 72. Mean catch per unit effort (CPUE) for Utah Chub in Henrys Lake, Idaho between 1991 and 2021 using traditional gill nets set in Henrys Lake.

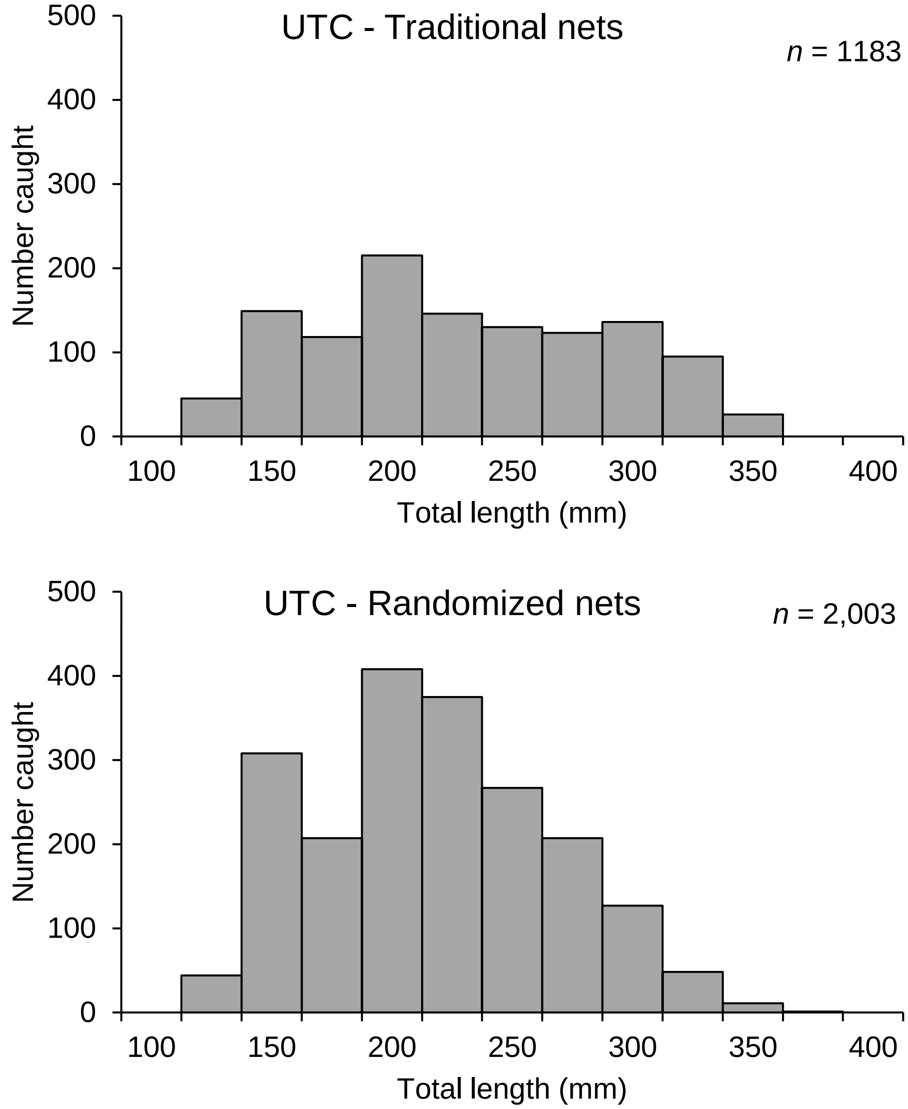

Figure 73. Utah Chub length-frequency distribution from traditional gill nets (Top), and randomized gill nets (Bottom) set in Henrys Lake, May 2021.

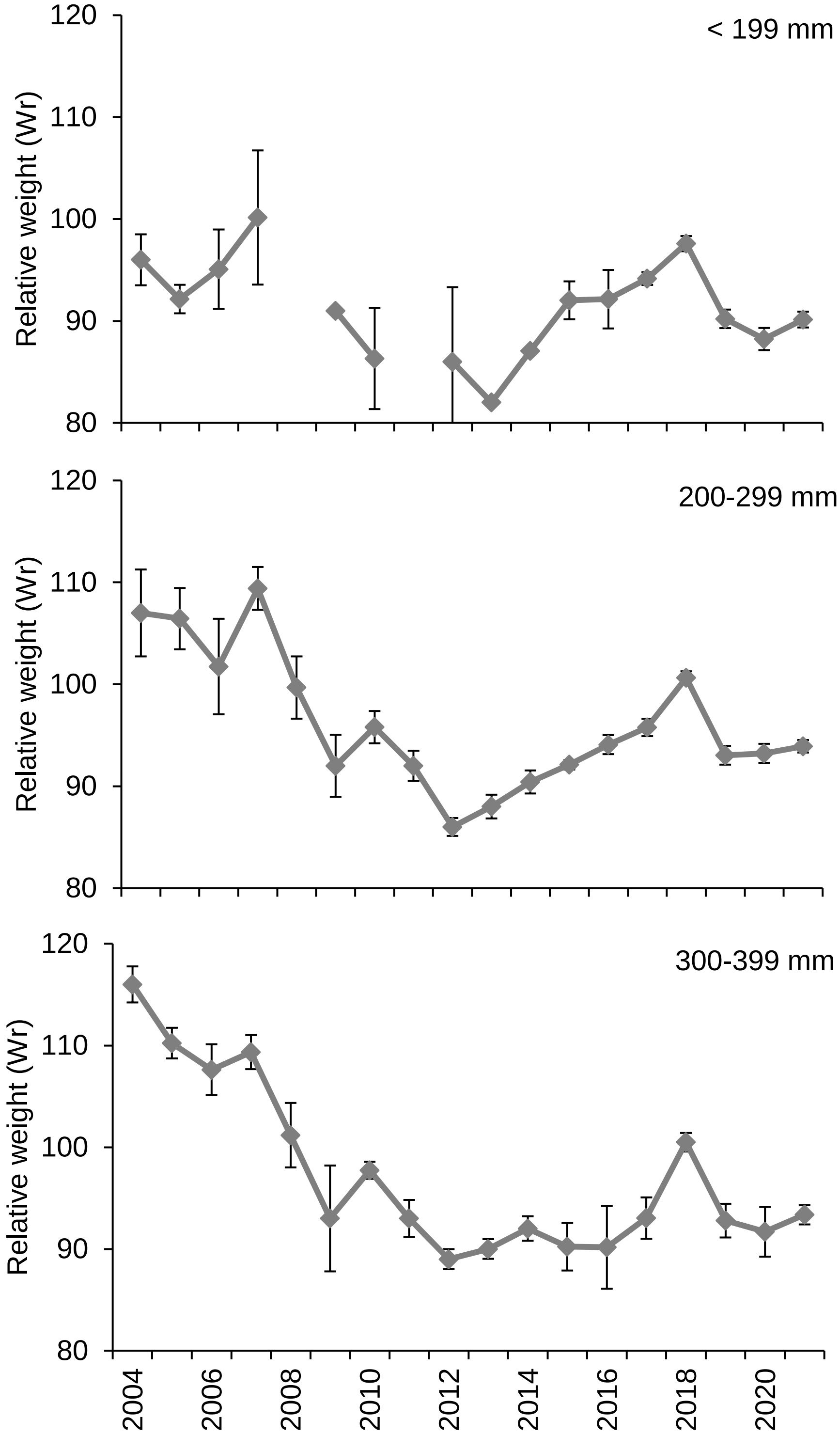

Figure 74. Relative weights (W_r) for three size classes (< 199 mm, 200 – 299 mm, 300 – 399 mm) of Utah Chub from traditional gill nets set in Henrys Lake, 2004 – 2021. Error bars represent 95% confidence intervals.

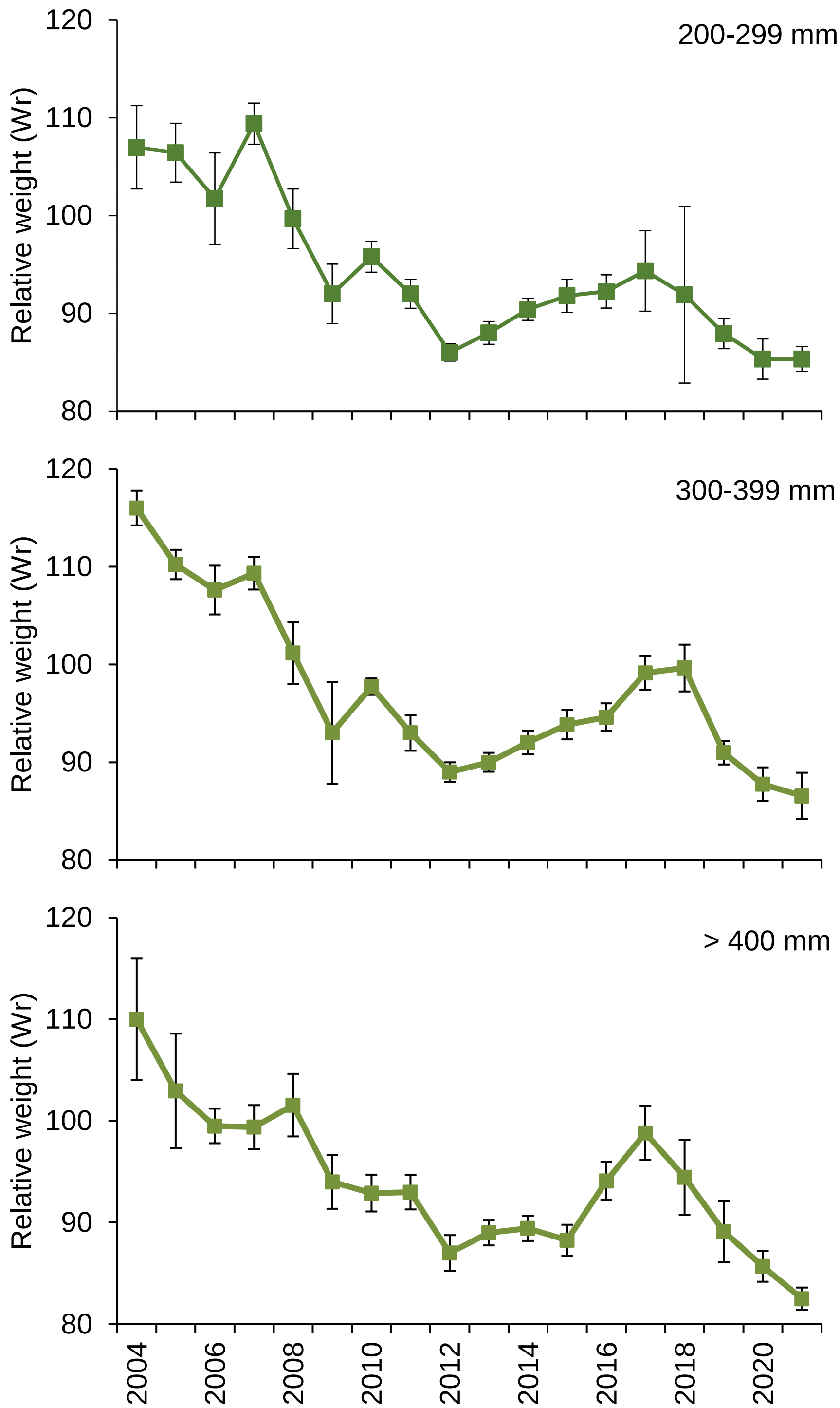

Figure 75. Relative weights (W_r) for three size classes (200 – 299 mm, 300 – 399 mm, and > 400 mm) of Yellowstone Cutthroat Trout from traditional gill nets set in Henrys Lake, 2004 – 2021. Error bars represent 95% confidence intervals.

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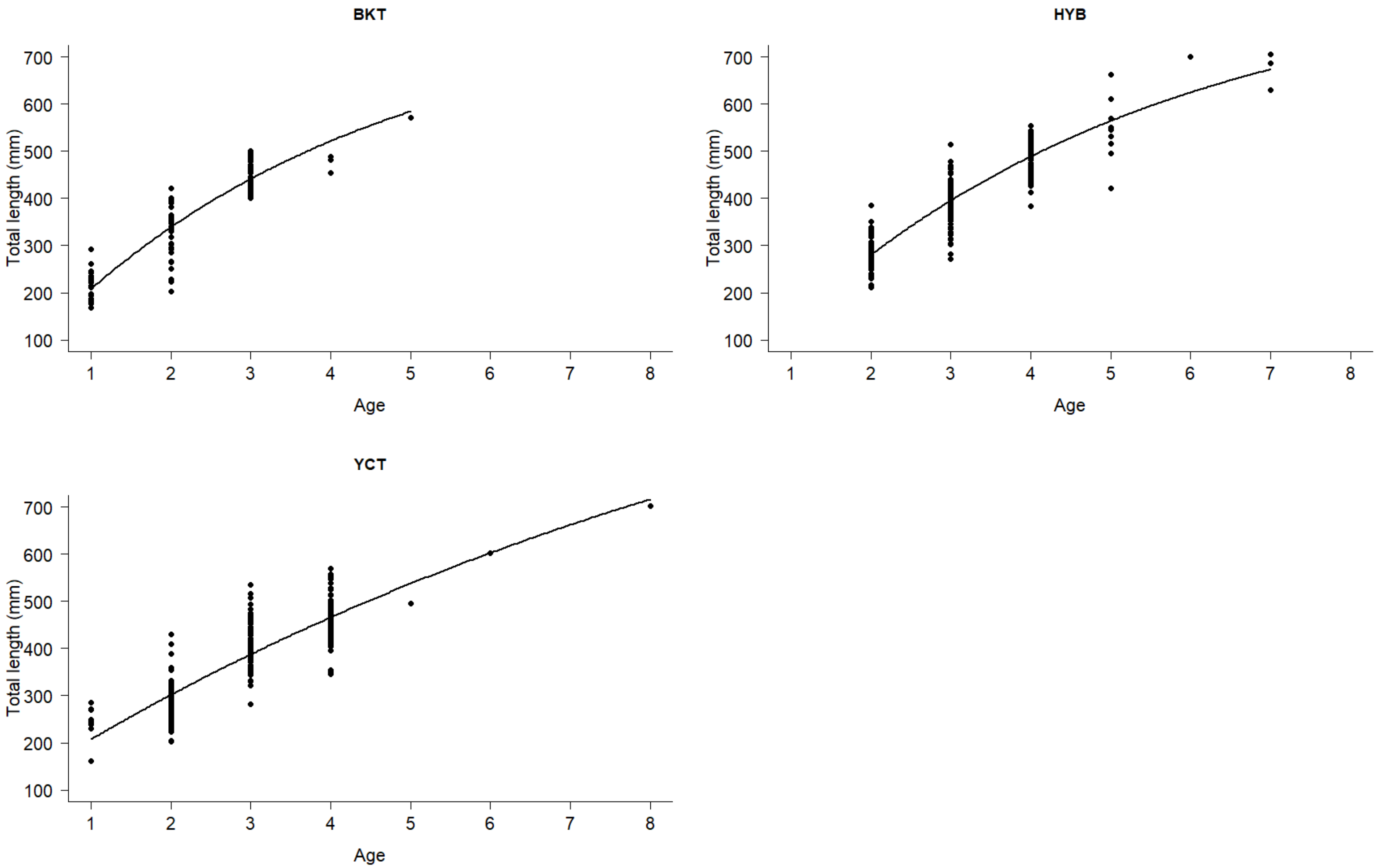

Figure 76. Von Bertalanffy growth models for Brook Trout (BKT), Yellowstone Cutthroat x Rainbow Trout (HYB), and Yellowstone Cutthroat Trout (YCT) using estimated ages from sagittal otoliths obtained from combined samples in the traditional and randomized gill nets set in Henrys Lake, May 2021.

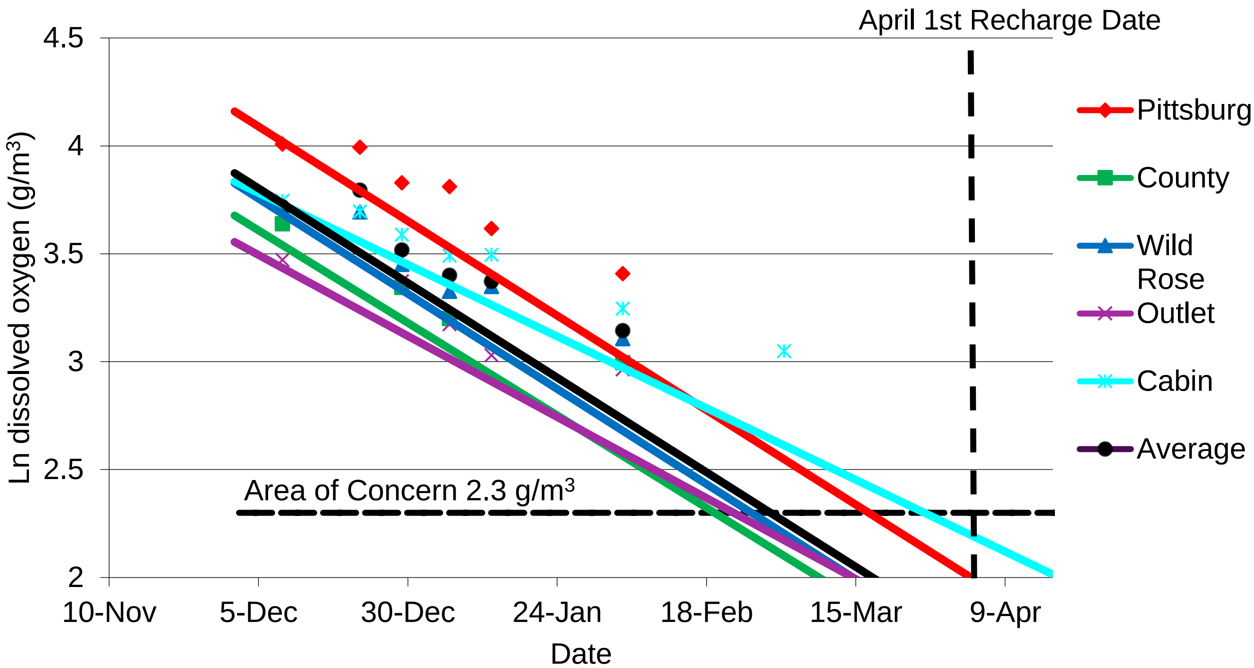

Figure 77. Dissolved oxygen depletion estimates from Henrys Lake, 2020 – 2021. Dotted lines indicate the threshold of concern (2.3 g/m^3) for trout survival, and recharge date (April 1).

ASHTON RESERVOIR

ABSTRACT

In July 2021 we conducted a standardized lowland lake survey on Ashton Reservoir. Yellow Perch *Perca flavescens* relative abundance increased since they were first documented in Ashton Reservoir in 2008, but the species assemblage was otherwise similar to prior surveys. The gill-net catch (mean fish/net night ± 95% CI) was dominated by Utah Chub *Gila atra* at 79 (± 50) and Utah Sucker *Catostomus ardens* 20 (± 10) fish/net night, while Yellow Perch at 295 (± 395) fish/h dominated the electrofishing catch. Sportfish made up a smaller part of the gill-net catch. We captured 3 (± 2) Rainbow Trout *Oncorhynchus mykiss*, 2 (± 2) Brown Trout *Salmo trutta*, and 7 (± 5) Yellow Perch per net night. Size-structure of Rainbow Trout, Brown Trout, and Yellow Perch were all lower than 2008, but small sample size limits the utility of comparisons. Although size-structure declined, Yellow Perch ≥ 200 mm increased in abundance by 54% since 2008. Average relative weight for sportfish was < 100 for both Rainbow Trout and Brown Trout, but > 100 for Yellow Perch. Further assessment of the growing Yellow Perch fishery is warranted. Despite the additional fishing opportunity, public access to Ashton Reservoir is limited to a single boat ramp above the upstream end of the reservoir, and opportunities to expand access to the resource should be explored.

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INTRODUCTION

Ashton Reservoir is located on the Henrys Fork of the Snake River approximately 4 km northwest of the town of Ashton, Idaho (Figure 78). The reservoir is 4.4 km long and 0.3 km wide, with a surface area of approximately 150 ha. Ashton Reservoir Dam is a rock and earth filled structure that reaches 60 feet tall, 226 feet long, and is operated by PacifiCorp Energy. The Ashton hydroelectric plant provides electric generation services to Rocky Mountain Power and Pacific Power. Construction of this dam began in 1914 and was completed in 1918. The plant consists of the dam, powerhouse, and three power generating units.

Ashton Reservoir is managed as a put-and-take fishery under general regulations with a management goal of at least 1 fish/h (IDFG 2019). The Idaho Department of Fish and Game (IDFG) stocks Rainbow Trout *Oncorhynchus mykiss*. A mitigation agreement between IDFG and PacifiCorp on Article 402 of the Ashton-St. Anthony Hydroelectric Project, FERC No. 2381, funds the stocking program for Ashton Reservoir, which is stocked annually with ~37,400 catchable-sized Rainbow Trout. This agreement began in 1991 and will end in 2028. Other sportfish in Ashton Reservoir includes Brown Trout *Salmo trutta*, kokanee *Oncorhynchus nerka*, and Yellow Perch *Perca flavescens*. Yellow Perch were first documented in the reservoir during a gill-net survey conducted in 2008 (Schoby et al. 2010). Brook Trout *Salvelinus fontinalis*, Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri*, and Mountain Whitefish *Prosopium williamsoni* have been documented historically (Ball and Jeppson 1980; Maiolie 1987), but have not been detected in surveys after 1986 (Schoby et al. 2010). The fish assemblage in Ashton Reservoir is dominated by Utah Chub and Utah Sucker. These two species cumulatively made up > 90% of the numerical catch in 1986 and 2008 (Maiolie 1987; Schoby et al. 2010). Ashton Reservoir is not surveyed regularly, as the most recent sampling event was conducted in 2008. Given the time elapsed since the last survey, the recent documentation of a new species (Yellow Perch), and angler reports of a growing Yellow Perch fishery, we deemed it necessary to assess the status of the Ashton Reservoir fishery in 2021.

OBJECTIVES

1. Describe the species composition, size-structure, and condition of the fish community in Ashton Reservoir.

2. Assess the level of Yellow Perch establishment and population growth within Ashton Reservoir relative to the survey conducted in 2008.

METHODS

We sampled fish populations in Ashton Reservoir using paired (floating/sinking) gill-nets and boat electrofishing. In total, we set five experimental-mesh gill-net pairs (floating/sinking) overnight and electrofished six transects (~600 s per transect, 300 – 325 v).

We identified all captured fish to species and recorded TL in mm. We recorded weights (g) of fish caught in gill nets. Gill-net catch per unit effort (CPUE) was calculated as mean fish/net night with 95% confidence intervals (CI), and electrofishing CPUE was calculated as mean fish/h with 95% CI. However, one of the floating gill-net sets drifted into the spillway buoys and we excluded these data from CPUE analyses. We assessed size-structure and condition of sportfish species by calculating proportional stock distribution (PSD), relative stock density (RSD), and relative weight (W_r). We calculated PSD and RSD according to the following equations:

$$\text{PSD} = \frac{\text{number} \geq \text{Quality Size}}{\text{number} \geq \text{Stock Size}} \times 100$$

$$\text{RSD-Preferred} = \frac{\text{number} \geq \text{Preferred Size}}{\text{number} \geq \text{Quality Size}} \times 100,$$

where preferred (P), quality (Q), and stock (S) size are specific to each species (Rainbow Trout: P = 500 mm, Q = 400, S = 250; Brown Trout: P = 400 mm, Q = 300, S = 200; Yellow Perch: P = 250 mm, Q = 200, S = 130; kokanee: P = 300 mm, Q = 250, S = 120). We calculated W_r according to the following equation:

$$W_r = \left(\frac{W}{W_S}\right) \times 100,$$

where W is the measured weight of the fish in grams and W_S is the calculated standard weight relative to TL according to species-specific W_S equations for Rainbow Trout (Simpkins and Hubert 1996), Brown Trout (Hyatt and Hubert 2001b), Yellow Perch (Willis et al. 1991), and kokanee (Hyatt and Hubert 2000). Finally, we evaluated the relationship between W_R and TL for each species using simple least-squares regression with accompanying t-tests.

RESULTS

In 9 net nights of gill-netting effort and six electrofishing transects, we captured 1,0 and 691 fish, respectively, for a total of 1,691 fish sampled from Ashton Reservoir in July, 2021. The gill-net catch (± 95% CI; Figures 79, 80) was dominated in abundance by Utah Chub (72%, 79 fish/net night ± 50 and Utah Sucker (18%, 20 ± 10), followed by Yellow Perch (6%, 7 ± 5), Rainbow Trout (3%, 3 ± 2), Brown Trout (2%, 2 ± 2), and kokanee (<1%, <1). By weight, the gill-net catch was dominated by Utah Sucker (50%, 18,833 g/net night ± 10,886) and Utah Chub (43%, 16,237 ± 11,104), followed by Brown Trout (3%, 1,024 ± 883), Yellow Perch (3%, 988 ± 729), Rainbow Trout (2%, 872 ± 583), and kokanee (<1%, 39 ± 53). In our electrofishing survey, average Yellow Perch CPUE was higher than any other taxa (44%, 295 fish/h ± 395), followed by Utah Sucker (34%, 224 ± 83), Utah Chub (8%, 56 ± 38), Redside Shiner *Richardsonius balteatus* (8%, 56 ± 48), Brown Trout (2%, 13 ± 10), Longnose Dace *Rhinichthys cataractae* (2%, 11 ± 14), *Cottus* sp. (1%, 7 ± 7), and Rainbow Trout (<1%, 3 ± 3; Figure 81).

Rainbow Trout captured in gill nets ranged in size from 155 to 453 mm (mean TL ± SD = 311 ± 50), had a PSD of 3 and an RSD-P of 0 (Figure 82). We captured three Rainbow Trout electrofishing and did not evaluate PSD and RSD-P separately due to low sample size (mean TL ± SD = 224 ± 96; Figure 83). Rainbow Trout mean W_r (± 95% CI) was 75 ± 3 (Figure 84) and was not correlated with TL (r = -0.26, F = 2.68, df = 38, P = 0.11). Brown Trout captured in gill nets ranged in size from 222 to 460 mm (mean TL ± SD = 367 ± 65), had a PSD of 79 and an RSD-P of 32 (Figure 82). Brown Trout captured electrofishing ranged in size from 82 to 454 mm (mean TL ± SD = 257 ± 125), had a PSD of 56 and an RSD-P of 22 (Figure 83). Brown Trout mean W_r was 80 ± 4 (Figure 84), and was negatively related to TL (r = -0.53, F = 6.47, df = 17, P = 0.02). Yellow Perch caught in gill nets ranged in size from 140 to 305 mm (mean TL ± SD = 207 ± 45), had a PSD of 53 and an RSD-P of 22 (Figure 82). Yellow Perch captured electrofishing ranged in size from 50 to 225 mm (mean TL± SD = 82 ± 41), had a PSD of 23, and an RSD-P of 0 (Figure 83). Yellow Perch mean W_r was 101 ± 2 (Figure 84), and was not related to increasing TL (r = -

0.10, F = 0.58, df = 57 P = 0.45). Few kokanee captured in gill nets (n = 4) and ranged in size from 170 to 280 mm (mean TL ± SD = 209 ± 49). Kokanee PSD was 25 and RSD-P was 0 (Figure 82). No additional kokanee were captured via boat electrofishing. Kokanee mean W_r was 78 ± 8 (Figure 84).

DISCUSSION

Prior to 2021, the most recent survey of Ashton Reservoir was conducted in 2008 and utilized only gill nets. This survey provides a more comprehensive assessment of the fishery, as it includes results from both gill nets and boat electrofishing gears, which is supported by our documentation of several taxa that were not detected in 2008 (e.g., Redside Shiner, Longnose Dace, and *Cottus* sp.). Species composition from 2021 gill nets was similar to 2008 and other prior surveys (Ball and Jeppson 1980; Maiolie 1987; Schoby et al., 2010), with Utah Sucker and Utah Chub continuing to dominate the gill-net catch. The most notable change in the fish assemblage was the increase in Yellow Perch relative abundance, which grew from only making up 1% of the gill-net catch in 2008 to 6% in 2021 and dominated the 2021 electrofishing catch. However, 54% of the total Yellow Perch catch was < 60 mm, potentially indicating a strong year class that could affect the fishery in coming years. Although this strong year-class could be concerning since Yellow Perch are a non-native species that is becoming established in the Henrys Fork drainage, it may be a yearly occurrence that has gone un-noticed given the infrequent nature of Ashton Reservoir surveys. Future efforts should focus on describing the Yellow Perch population in greater detail and monitoring age-structure to assess recruitment consistency.

Size-structure of Rainbow Trout and Brown Trout was marginally smaller than that observed in 2008. Sampling in 2008 yielded a small number of memorable- and trophy-size trout that were absent from 2021 surveys, but small sample sizes of individuals in those size-classes limit our ability to determine if this is an accurate depiction of the populations. Alternatively, these larger individuals could be utilizing riverine habitats of the Henrys Fork upstream of where sampling took place in Ashton Reservoir or be absent altogether. Condition of Rainbow Trout and Brown Trout were below 100, potentially indicating limited food resources or competition for ideal foraging habitat with other more abundant taxa. Particularly for Brown Trout, significantly declining Wr with increasing TL suggests a lack of intermediately sized prey which is counterintuitive given the abundance of appropriately-sized Yellow Perch observed in our gill-netting and electrofishing surveys. Alternatively, Yellow Perch can occupy a similar trophic niche to Brown Trout, and Yellow Perch may compete with smaller size classes of Brown Trout for forage (Hansen et al. 2019). Lacustrine Brown Trout ontogenetic diet shifts to piscivory take place once they reach >300 mm (Jensen et al. 2012), and given that 38% of our captured Brown Trout are <300 mm in Ashton Reservor, there is ample opportunity for dietary overlap with a mesopredator such as Yellow Perch. Only eight Yellow Perch were sampled in 2008 and although size structure declined in 2021, gill-net CPUE of quality-sized Yellow Perch increased by 54% between 2008 and 2021. Further, Yellow Perch relative abundance was higher than both Rainbow Trout and Brown Trout in 2021 regardless of capture method (gill nets or electrofishing). The increase in Yellow Perch relative abundance, presence of multiple year and size classes, and healthy condition relative to other sportfish in Ashton Reservoir suggests a healthy and growing population that will add opportunity for anglers and potentially increase use in Ashton Reservoir as the Yellow Perch fishery gains popularity. Our current management objective for Ashton Reservoir intends it to be a high catch-rate yield fishery appropriate for beginner anglers (IDFG 2019). Although this goal was intended for a trout fishery, Yellow Perch provide an additional option for beginner anglers, and are not likely to affect trout catch rates given the ample yearly stocking of catchable-sized Rainbow Trout. Regardless, we still know very little about angler utilization of the Yellow Perch fishery in Ashton Reservoir or angler preferences for this water body. In the future, we should conduct a creel survey to assess angler preferences and use, and implement a Yellow Perch Tag-You're-It study to assess exploitation of the resource.

From a multiple user-group perspective, one of the most challenging aspects of utilizing the resources of Ashton Reservoir is the limited access. Currently, only one public access point exists and is located on the more riverine portion above the upstream end of the reservoir. Not only does this preclude multiple user groups from utilizing the majority of the reservoir during the open water season, but ice fishing access is all but impossible due to faster moving water at the upper end of the reservoir and unsafe ice conditions at the only boat ramp.

MANAGEMENT RECOMMENDATIONS

1. Monitor the growing Yellow Perch population and continue to investigate potential impacts to the fishery.

2. Conduct a creel survey on the reservoir to understand fishing pressure, use, target species, and angler preferences.

3. Conduct a Tag-You're-It study on Yellow Perch in Ashton Reservoir to evaluate exploitation

4. Explore opportunities to increase access on the lower portion of Ashton Reservoir.

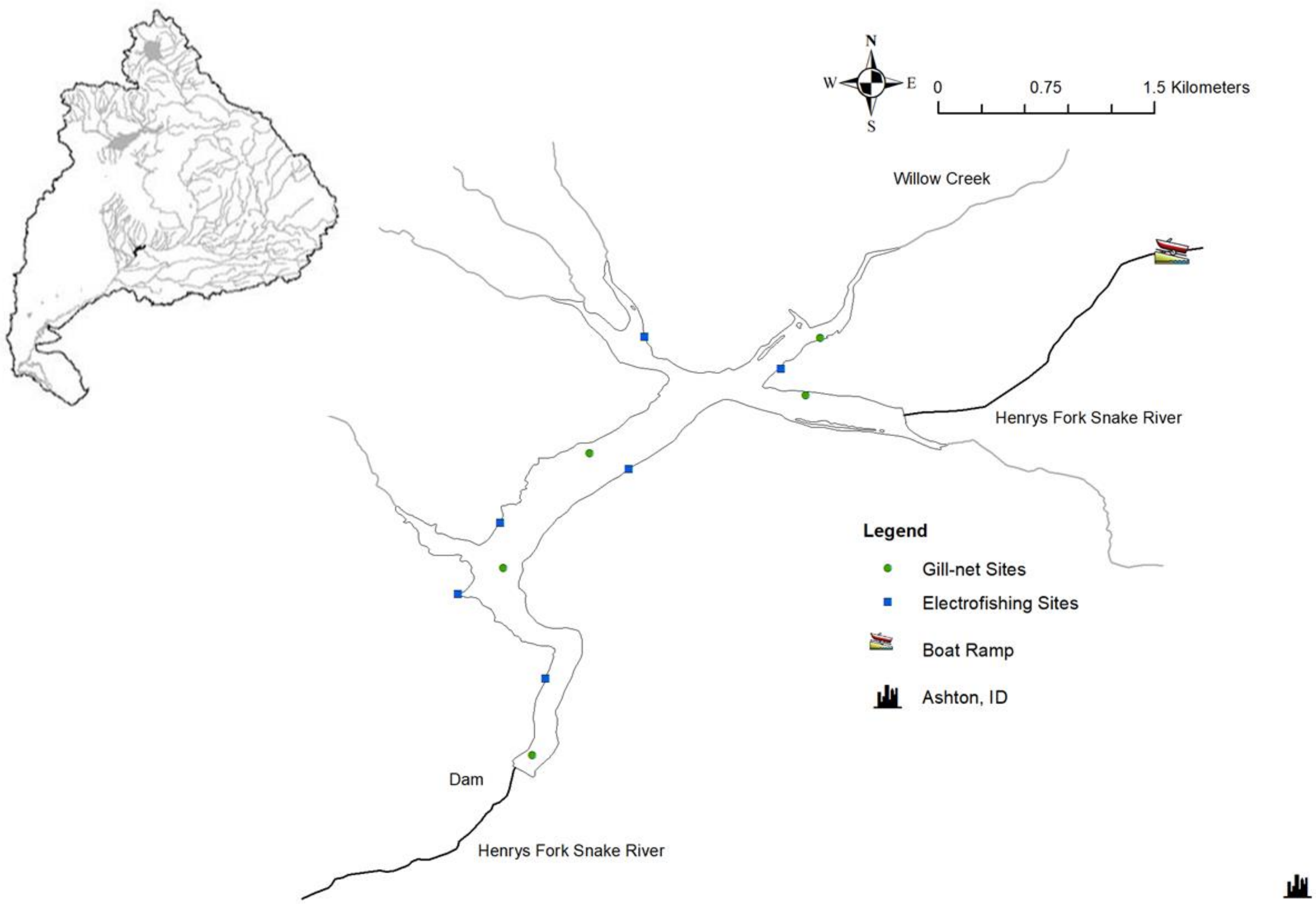

Figure 78. Location of experimental gill-net sites (Green circles), electrofishing sites (Blue squares), and tributaries of Ashton Reservoir, 2021.

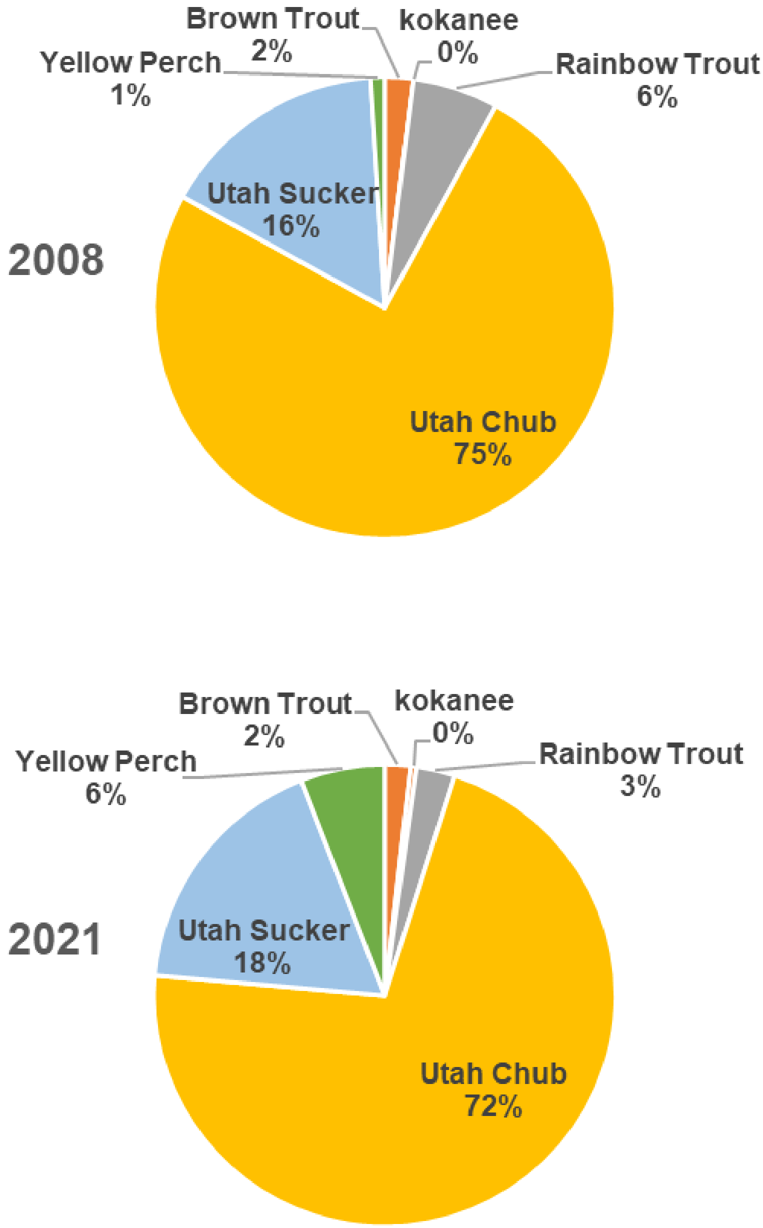

Figure 79. Species composition of fish captured using gill nets in Ashton Reservoir during sampling in 2008 (top) and 2021 (bottom).

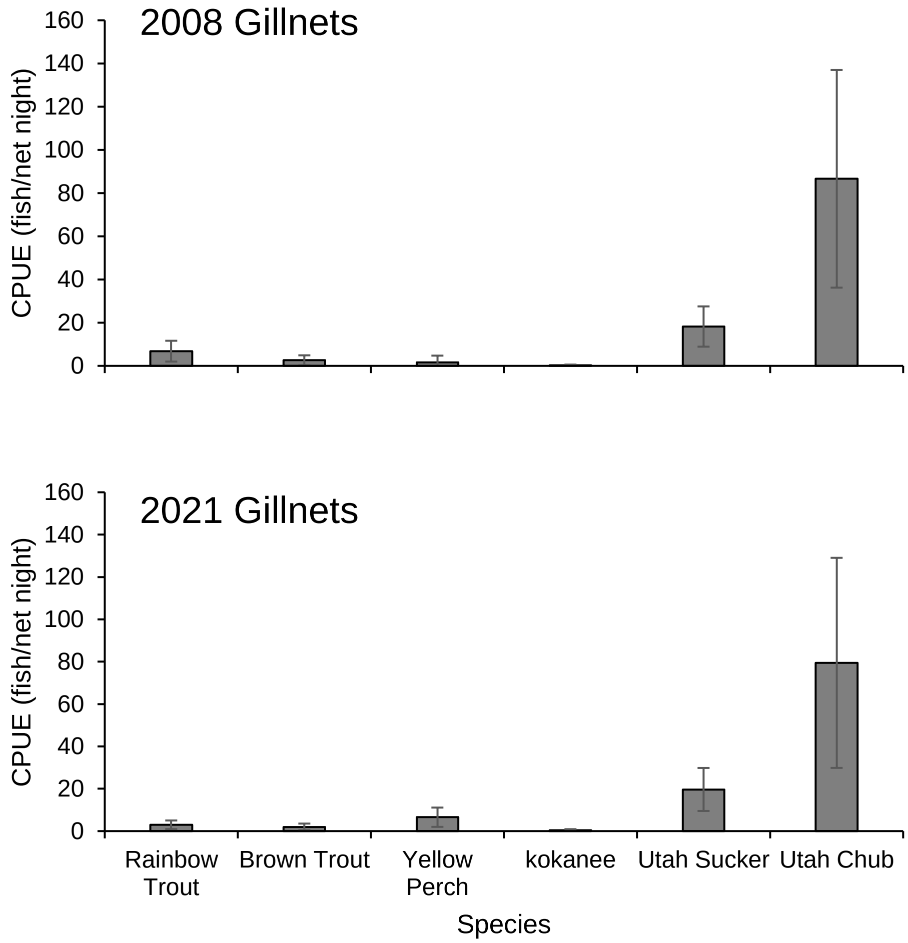

Figure 80. Gill-net catch rates (fish per net night) in Ashton Reservoir from surveys conducted in 2008 (top) and 2021 (bottom). Error bars represent 95% confidence intervals.

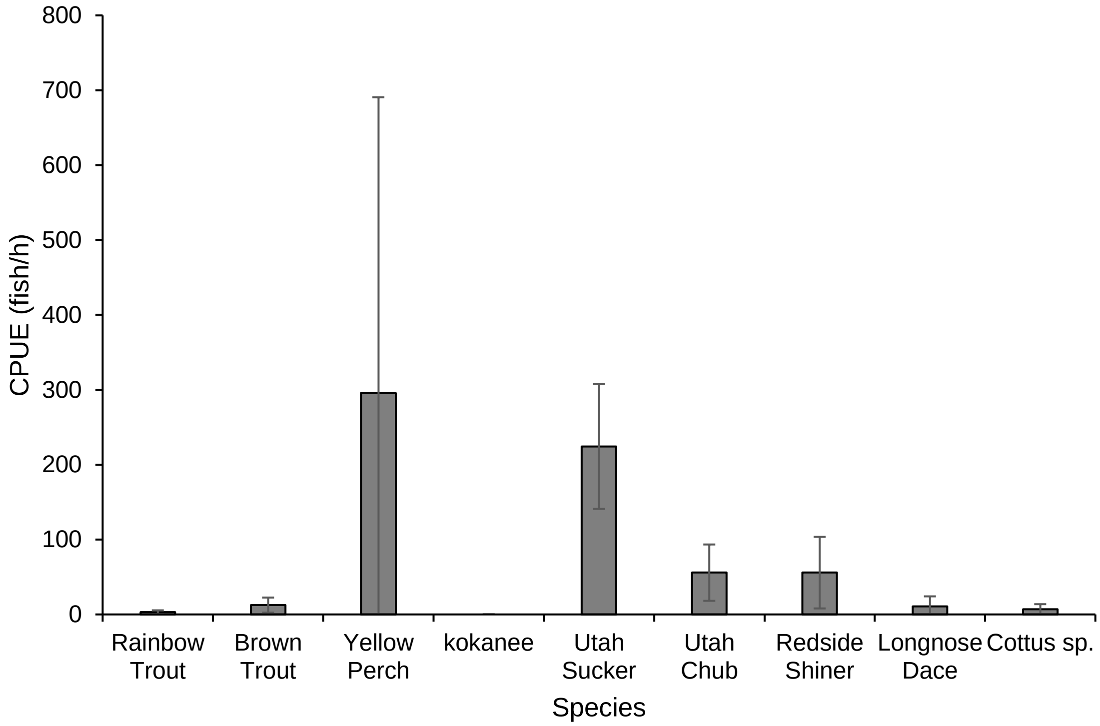

Figure 81. Boat electrofishing catch rates (fish per h) from Ashton Reservoir in 2021. Error bars represent 95% confidence intervals.

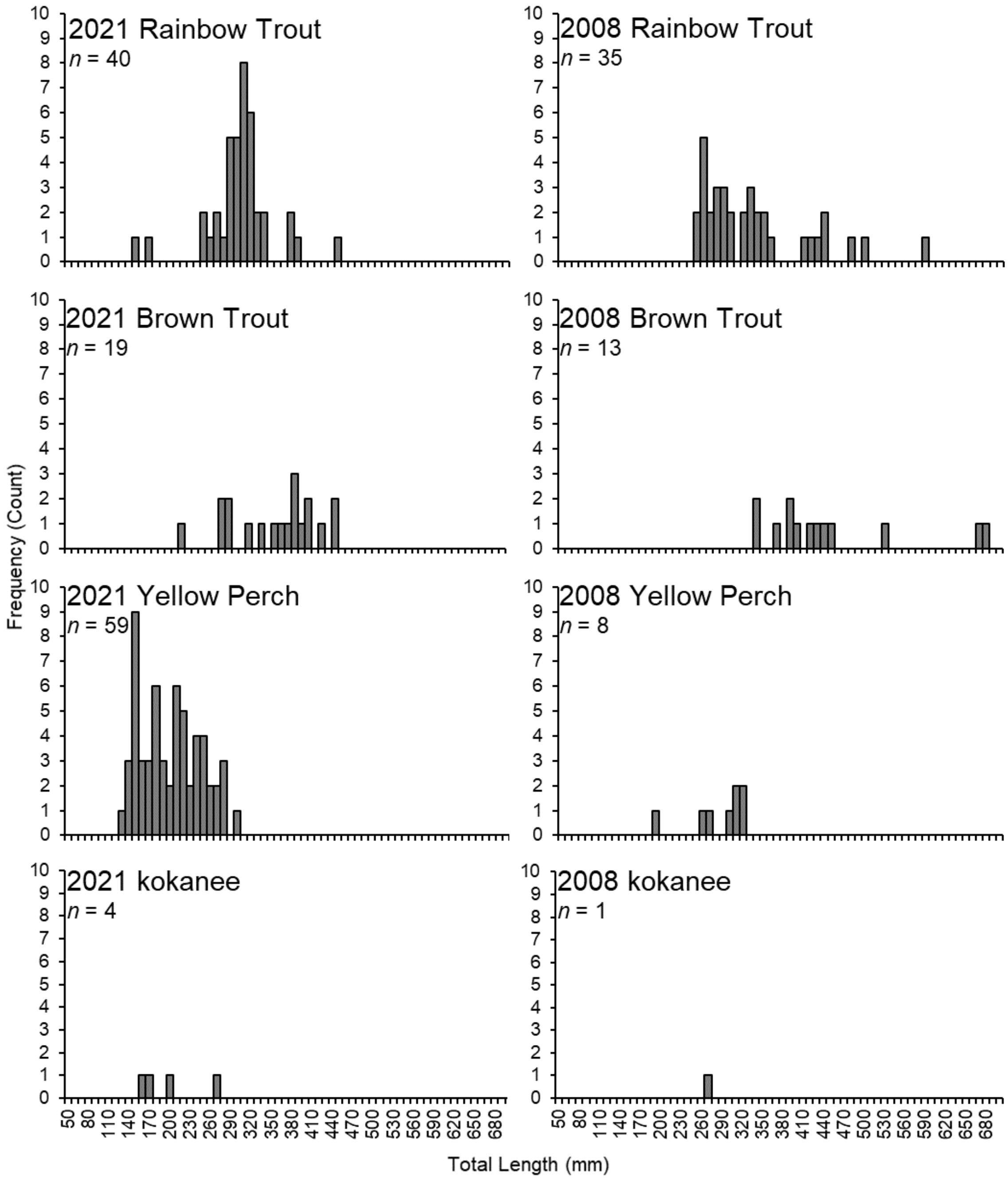

Figure 82. Length-frequency histograms of Rainbow Trout (top), Brown Trout (upper middle), Yellow Perch (lower middle), and kokanee (bottom) captured in gill nets in Ashton Reservoir during 2008 (left panel) and 2021 (right panel).

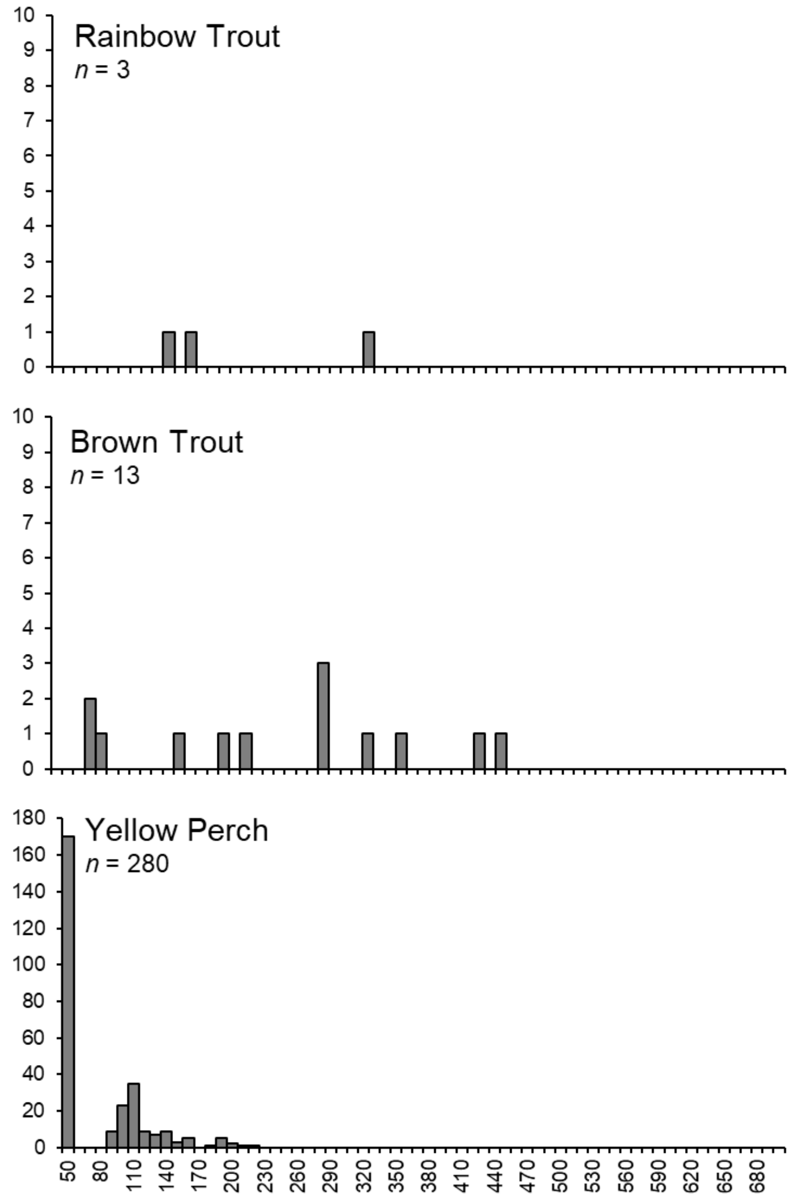

Figure 83. Length-frequency histograms of Rainbow Trout (top), Brown Trout (upper middle), Yellow Perch (lower middle), and kokanee (bottom) captured using boat electrofishing in Ashton Reservoir during 2021.

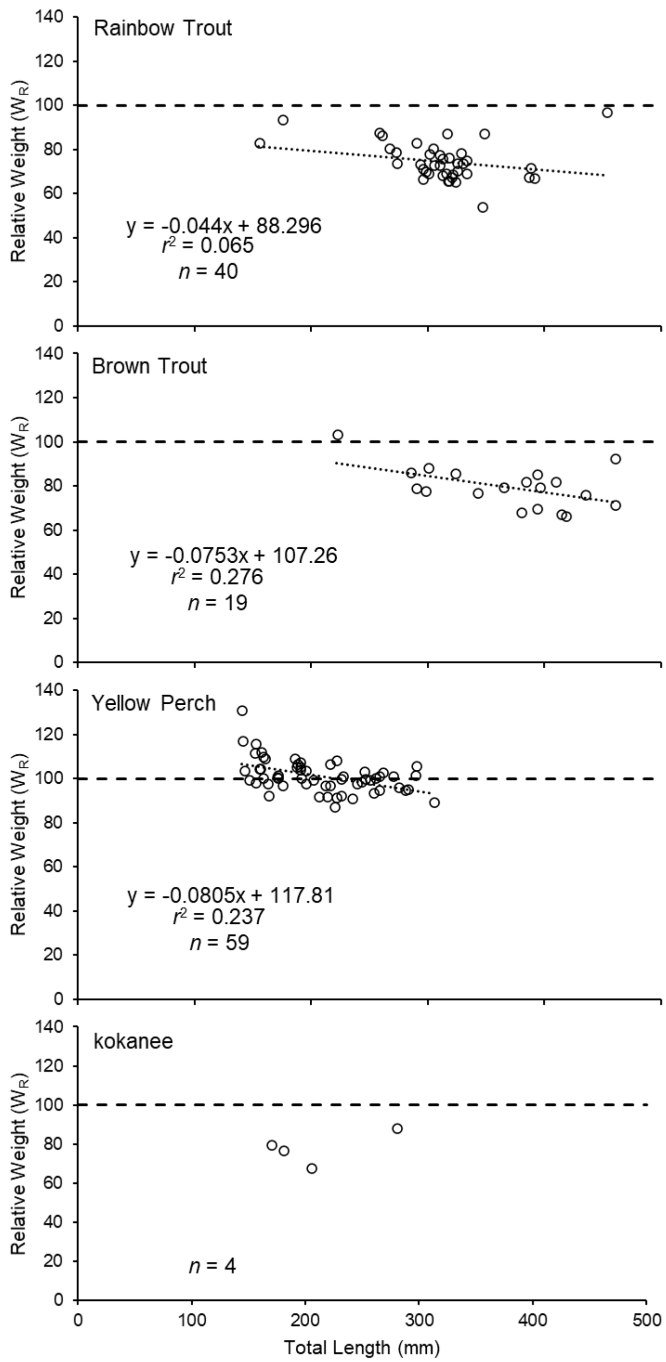

Figure 84. Relative weights of Rainbow Trout (top), Brown Trout (upper middle), Yellow Perch (lower middle), and kokanee (bottom) captured in gill nets in Ashton Reservoir during 2021. Dashed line at W_R = 100 indicates a fish in the 75th percentile of weight at a given length. Dotted line is the trend line of the model relating W_R to total length. We did not model kokanee W_R due to low sample size (n = 4).

RIRIE RESERVOIR

ABSTRACT

We conducted our seventh year of monitoring the kokanee *Oncorhynchus nerka* population in Ririe Reservoir using gill nets suspended in the thermocline. Average gill-net catch-per-unit-effort (CPUE) of kokanee was 98 fish/net night ± 4 (95% CI), which was greater than catch rates in 2020 (i.e., 24 ± 18) and greater than the six-year average (i.e., 2014-2020; 60 ± 26). Kokanee comprised the majority of the overall species composition (i.e., 82%) followed by Yellow Perch *Perca flavescens* (14%), and Utah Sucker *Catastomus ardens* (4%). Kokanee proportional size distribution (PSD) and relative stock density-preferred (RSD-P) were 74 and 61, respectively. Continued monitoring of kokanee will allow managers to adjust stocking rates when necessary, in an effort to produce a quality fishery with adequate catch rates evaluated by creel surveys. We will continue monitoring kokanee to inform anglers about the population and to follow trends in abundance in order to allow for informed fish stocking decisions in Ririe Reservoir.

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INTRODUCTION

Ririe Reservoir is located on Willow Creek, approximately 32 km east of Idaho Falls (Figure 85). A thorough description of Ririe Reservoir can be found in our previous report (Vincent et al. *in press*).

OBJECTIVES

1. Use annual summer gill netting to describe size structure, age, and growth of kokanee *O. nerka* in Ririe Reservoir to assist in developing appropriate stocking rates and to aid efforts to inform anglers of the population status of the fishery.

2. Estimate the relative abundance and size structure of Yellow Perch *Perca flavescens* in Ririe Reservoir to describe the Yellow Perch fishery.

METHODS

We sampled the kokanee population from June 7 through 10, 2021 using experimental gill nets with a neutrally buoyant design suspended in the thermocline. We used a water quality meter (YSI Inc., Yellow Springs, Ohio) to take water temperature at the surface and every subsequent meter down the water column until the thermocline was identified by a several degree water temperature difference from the previous depth. Experimental gill nets measured 49-m long by 6-m deep consisting of 16 panels that were 3-m long with two panels for each mesh size randomly positioned. The mesh sizes of the panels were 13-, 19-, 25-, 38-, 51-, 64-, 76-, and 102-mm bar mesh monofilament. We set nets at dusk and retrieved them the following morning. Sites were randomly selected by overlaying a grid system (100 × 100 m) in mapping software (IDFG 2012). For site selection, Ririe Reservoir was stratified into three strata; lower, middle, and upper. Nets were set in depths ranging from 10 to 16 m to ensure adequate coverage in the thermocline. All fish captured were identified to species, measured for total length (TL) to the nearest millimeter (mm), and weighed to the nearest gram (g). We calculated gill-net catch-per-unit-effort (CPUE) for each species as the number of fish/net night and calculated 95% confidence intervals.

We removed sagittal otoliths from a subsample (i.e., 10 fish per 10-mm length group) of kokanee collected from gillnetting for age and growth analysis. For each sample in the subsample of kokanee, one whole otolith was sanded to evaluate thermal marking to determine the true age. We created an age-length key from our subsample of kokanee, then applied the age-length key to unknown age kokanee. We calculated mean (i.e., ratio of means; and standard deviation) length-at-age using the Isermann and Knight (2005) method in the FSA package in program R (R Core Team 2022).

We calculated proportional size distribution (PSD), relative stock density-preferred (RSD-P), and relative weights to describe the size structure and condition of kokanee and Yellow Perch in Ririe Reservoir. We used the standard weight (W_s) equation

$$log_{10}(W_s) = -5.062 + 3.033 * log_{10}(L),$$

for kokanee (Hyatt and Hubert 2000) where L is the total length, then used the equation

$$W_r = \left(\frac{W}{W_s}\right) * 100,$$

where W_r is relative weight and W is measured weight. We used the same equations for Yellow Perch, but changed the intercept to -5.386 and the slope to 3.230 (Willis et al. 1991).

Kokanee PSD was calculated as the number of fish greater than or equal to 250 mm divided by the number of fish greater than or equal to 120 mm, multiplied by 100. Kokanee RSD-P was calculated as the number of fish greater than or equal to 300 mm divided by the number of fish greater than or equal to 120 mm, multiplied by 100. Yellow Perch PSD was calculated as the number of fish greater than or equal to 200 mm divided by the number of fish greater than or equal to 100 mm, multiplied by 100. Yellow Perch RSD-P was calculated as the number of fish greater than or equal to 250 mm divided by the number of fish greater than or equal to 100 mm, multiplied by 100.

RESULTS

We collected 982 kokanee, 162 Yellow Perch, 1 Tiger Trout *Salmo trutta* × *Salvelinus fontinalis*, 1Yellowstone Cutthroat Trout *O. clarkii bouvieri* (YCT), and 53 Utah Sucker *Catostomus ardens* from 10 gill nets set in the thermocline. Species composition was 82% kokanee, 13% Yellow Perch, < 1% Tiger Trout, < 1% YCT, and 4% Utah Sucker. The mean (± 95% CI) gill-net CPUE of kokanee (Figures 86 and 87), Yellow Perch, Tiger Trout, YCT, and Utah Sucker was 98.2 (± 28.0), 16.5 (± 13.3), 0.1 (± 0.2), 0.1 (± 0.2), 5.3 (± 3.9), respectively (Figure 86). We further evaluated catch rates from 2015 to 2021 excluding age-0 kokanee (Figure 88). Kokanee varied in length from 80 to 430 mm with a mean TL of 292 (± 67; Figure 89). Kokanee mean relative weight was 89 (SD ± 10) and exhibited an increase in condition with length indicated by a positive slope (y = 0.0168x + 83.99; r^2 = 0.02; Figure 90). Kokanee PSD and RSD-P were 74 and 61, respectively. We sampled more age-2 kokanee (63%) than age-1 (34%), age-0 (5%), age-3 (1%), and age-4 (1%) kokanee, and this was reflected in the gill-net CPUE (Figure 91). Total length by respective age of kokanee was similar to other years (Table 25; Figure 92), and no age class was missing in the sample. We evaluated 223 kokanee otoliths for thermal marks to obtain true age and only 3otoliths did not have thermal marks identifiable.

Gill-net CPUE for Yellow Perch (16.5 fish/net night ± 13.3; Figure 93) was lower than the average from 2015 to 2020 (58 fish/net night ± 30). Yellow Perch varied in length from 81 to 265 mm with an average total length of 235 (± 22.5; Figure 94), PSD was 94, RSD-P was 21, and Yellow Perch composed 14% of all fish caught. Yellow Perch mean relative weight was 78 (± 7.2) and did not exhibit an increase in condition with length as indicated by a negative slope (y = -0.1567x + 114.7; r^2 = 0.17; Figure 95).

DISCUSSION

The kokanee population at Ririe Reservoir has rebounded from recent lows and is currently composed of strong and multiple age classes, suggesting this fishery should perform well in the coming years. We observed strong year classes of age-1 and age-2 kokanee in our gill-netting efforts this year. Based on our catch-at-age data, the fishery is dependent on age-1 and age-2 kokanee, then we see a precipitous decrease in abundance after age-2. After poor survival of the 2018 kokanee year class caused by using late-run kokanee instead of early-run kokanee (Heckel et al. 2020), which would currently be age-3 kokanee, the fishery has rebounded to a point where there are again high abundances of catchable-sized kokanee in the fishery. These fish will continue to provide anglers with kokanee fishing opportunities during open water and ice fishing seasons for the next two seasons. The high mean relative weight (i.e., 89) with an increasing condition as TL increases suggests that current stocking rates are suited to forage availability in the reservoir. We identified thermal marks on otoliths (i.e. kokanee of hatchery origin) in 99% of kokanee sampled suggesting there is little to no natural reproduction occurring. Recruitment to the Ririe Reservoir kokanee population appears to be almost entirely dependent on hatchery supplementation, which is consistent with thermal mark evaluation in other years. Catch rates in gill nets were above the previous 6-year average and the second highest on record since we began kokanee trend monitoring.

Yellow Perch abundance was below average during our June netting, but that netting effort is focused on pelagic areas inhabited by kokanee, not deep water (aphotic) areas where we generally sample Yellow Perch. But, because we catch Yellow Perch in the suspended nets we do evaluate those trends. The low gill-net CPUE for Yellow Perch is likely an artifact of sampling design because suspended gill nets are used to target kokanee specifically. However, our fall gill netting in 2020 with sinking and floating gill nets was likely a better study design to sample and evaluate Yellow Perch abundance in Ririe, which is a similar design that the McCall sub-region uses to monitor the Yellow Perch population in Lake Cascade (Janssen et al. 2020). We recommend using the gill-net CPUE of Yellow Perch during Fall Walleye Index Netting (FWIN) as a better index of monitoring the relative abundance of Yellow Perch in Ririe Reservoir than using suspended gill netting. However, because we capture Yellow Perch in suspended gill nets when monitoring kokanee, we should continue reporting Yellow Perch size structures and relative weights. The Yellow Perch fishery in Ririe is a popular fishery which requires monitoring to evaluate population trends and predict how the fishery will perform. We recommend using our Fall Walleye Index Netting as the method to evaluate the status of the Yellow Perch population because we sample using paired floating and sinking gill nets when the reservoir storage is at its lowest level during the year.

Up until 2012, approximately 18,000 catchable-sized YCT were stocked annually to provide angler opportunity. Following relatively poor performance of those fish (e.g., low fish growth, poor recruitment to creel, and dissatisfied anglers), they were replaced by similar numbers of sterile Rainbow Trout. Based on creel results in 2013, anglers caught an estimated 14,128 of the 18,000 (78%) Rainbow Trout stocked (High et al. 2015). The high angler use of Rainbow Trout observed in 2013 suggests that hatchery Rainbow Trout are returning at a high rate and diversifying angling opportunity, as well as meeting angler expectations. In the absence of annual stocking of hatchery YCT for almost a decade, the current YCT population in Ririe Reservoir is likely of wild origin. We observed increased YCT abundance in our FWIN netting in 2020 (Vincent et al. *in press*) and anglers reported catching and harvesting YCT in our 2019 creel (Heckel et al. 2020). The IDFG Fisheries Management Plan (IDFG 2019) and the Management Plan for Conservation of Yellowstone Cutthroat Trout (IDFG 2007) include goals to ensure the long-term persistence of the subspecies within its current range and restore the subspecies to those parts of its historical range in Idaho where practical. The operation of two PIT antennas upstream of Ririe Reservoir will aid us in understanding the population dynamics and life history strategies employed by YCT in this system (see Willow Creek chapter in this report for more on YCT in Ririe Reservoir), and ultimately to manage for the proliferation of YCT in an important drainage in its native range.

MANAGEMENT RECOMMENDATIONS

1. Continue biennial early summer suspended gill net monitoring to evaluate kokanee relative abundance, age, and growth, and species composition to predict how the fishery will perform.

2. Evaluate abundance of Yellowstone Cutthroat Trout in Willow Creek and whether there is an adfluvial component to the population to help direct future stream restoration efforts.

3. Begin monitoring the Yellow Perch population on a triennial basis in years when Fall Walleye Index Netting occurs. Describe the Yellow Perch population by using a subsample to estimate age, growth, and mortality to better understand what drives Yellow Perch abundance in Ririe Reservoir and predict fishery performance.

Table 25. Mean length-at-age for kokanee sampled from suspended gill nets in Ririe Reservoir 2021.

Age	*n*	Mean TL (mm)	95% CI
0	51	100	2.8
1	330	240	2.1
2	581	337	1.8
3	13	356	18.5
4	7	324	2.0

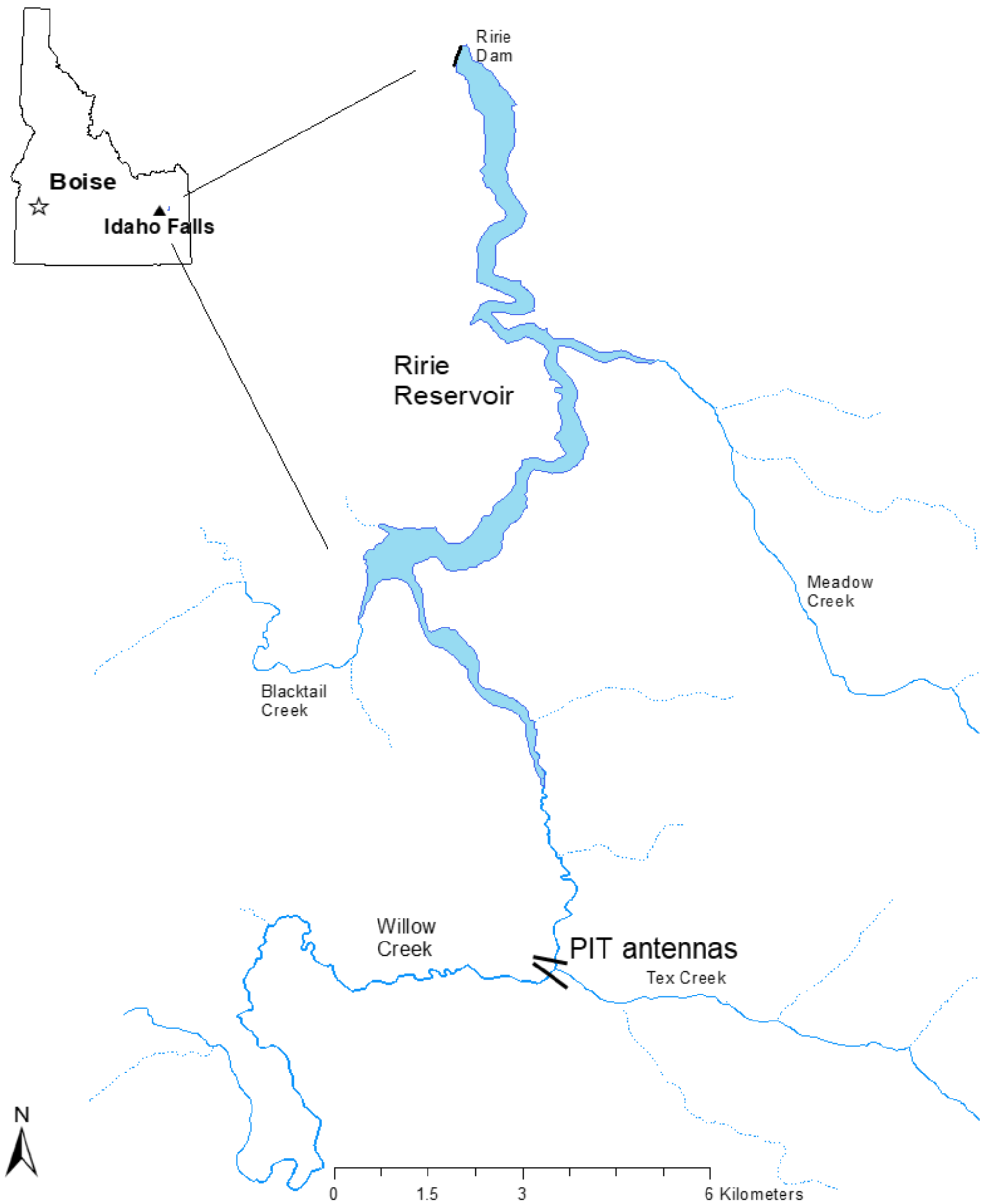

Figure 85. Location of Ririe Reservoir and major tributaries. The two PIT antennas installed in 2021 are indicated by the bold, black lines upstream and downstream of Tex Creek.

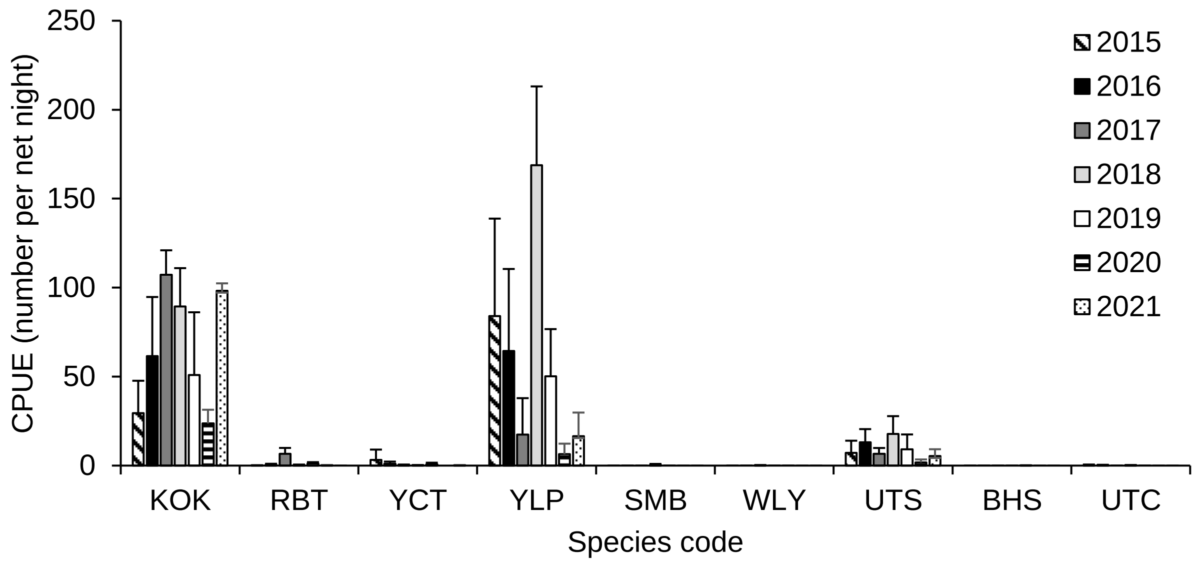

Figure 86. The number of fish/net night (CPUE) from suspended gill nets for kokanee (KOK), Rainbow Trout (RBT), Yellowstone Cutthroat Trout (YCT), Yellow Perch (YLP), Smallmouth Bass (SMB), Walleye (WLY), Utah Sucker (UTS), Bluehead Sucker *Catostomus discobolus* (BHS), and Utah Chub (UTC) in Ririe Reservoir during 2015–2021. Error bars represent 95% confidence intervals.

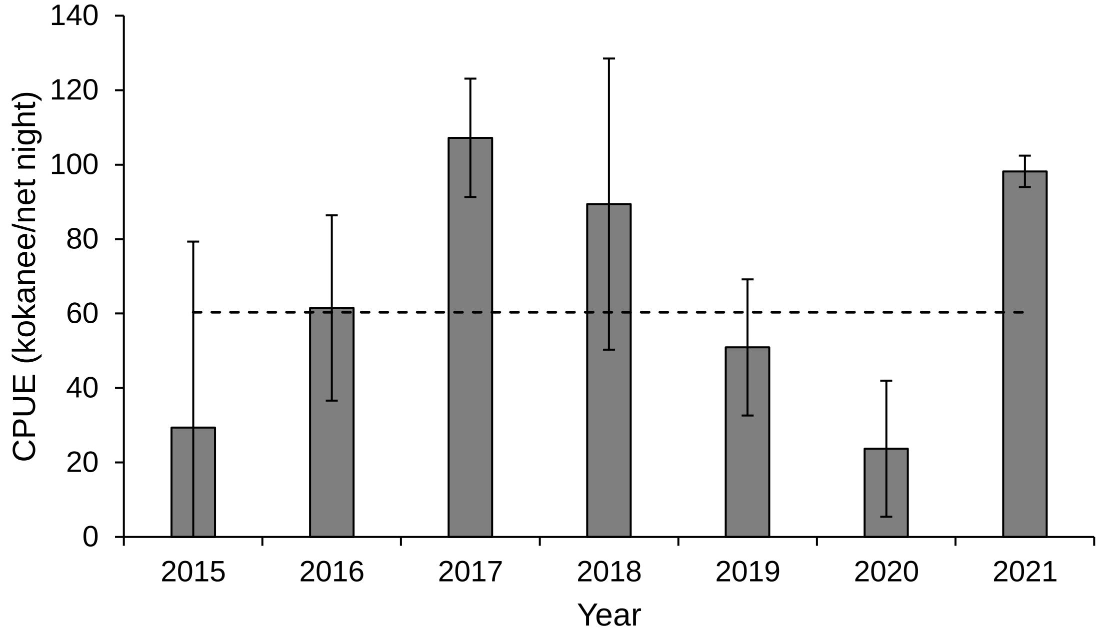

Figure 87. The mean number of kokanee caught per net night from 2015 to 2021. Error bars represent 95% confidence intervals. The dashed line represents the mean catch per net night from 2015 to 2020 (i.e. 60.4 kokanee/net night).

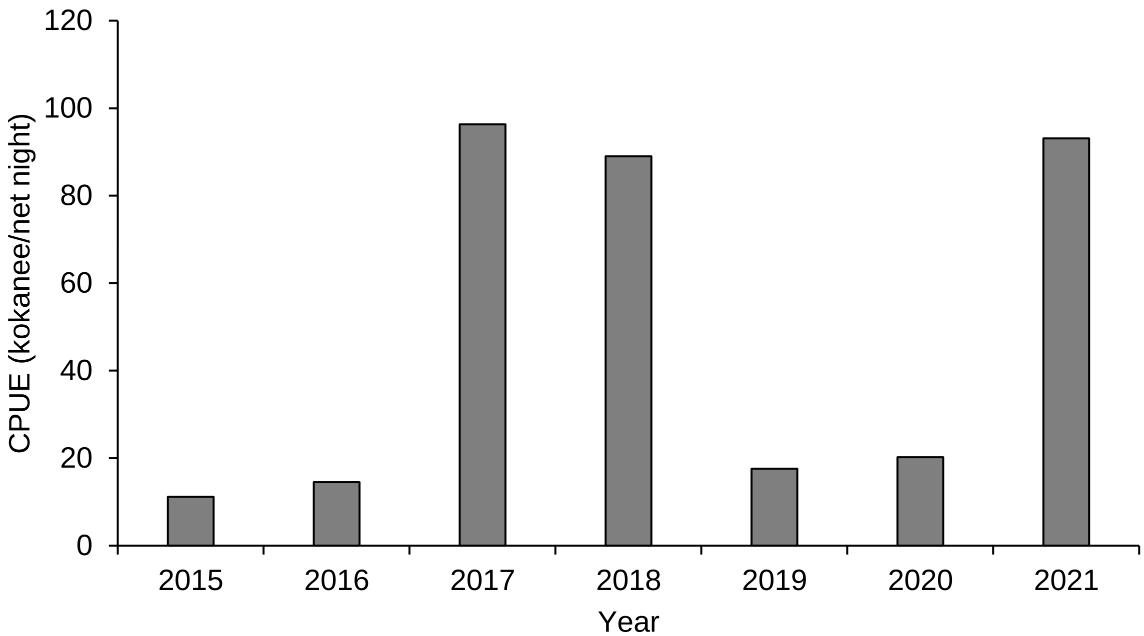

Figure 88. Catch rates of kokanee/net night in suspended gill nets from 2015-2021 excluding age-0 kokanee.

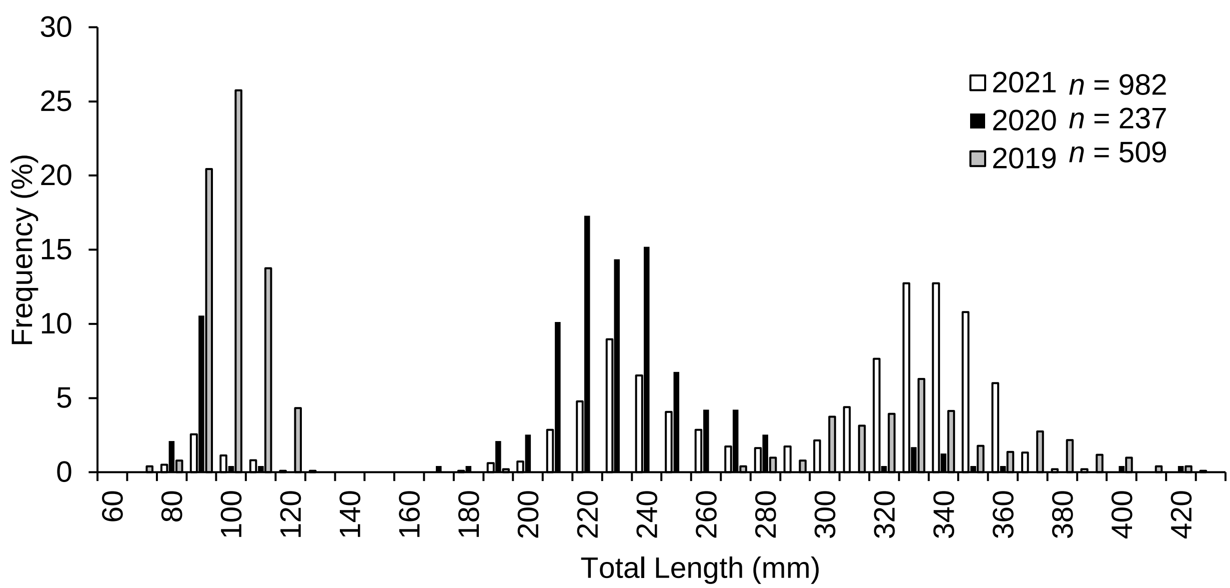

Figure 89. The length-frequency (%) distribution of kokanee caught in suspended gill nets in Ririe Reservoir in 2019-2021.

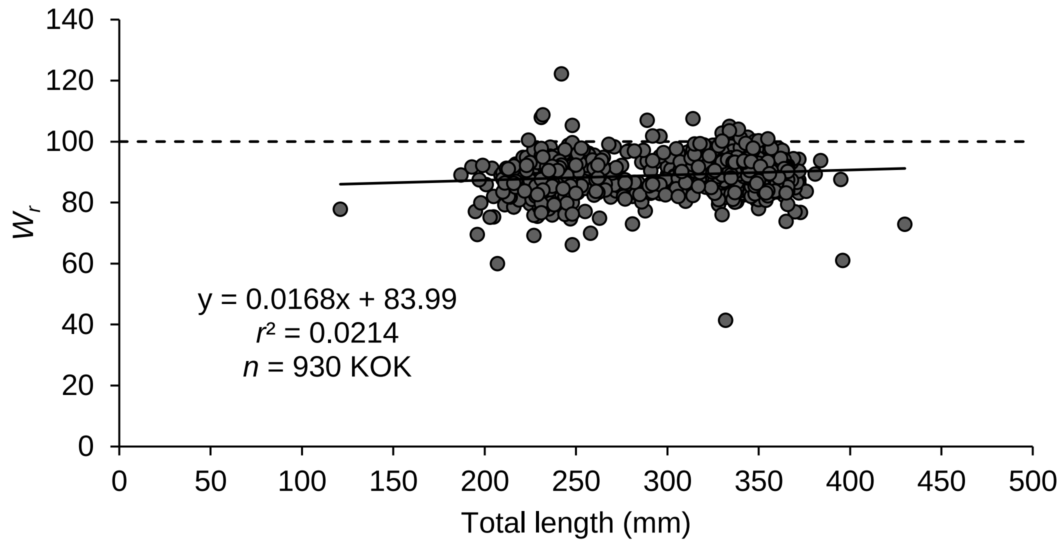

Figure 90. The relative weight (W_r) of kokanee (KOK) across total length (mm) caught in suspended gill nets in Ririe Reservoir in 2021. The dashed line represents a relative weight of 100 and the solid line is the regression line for the points.

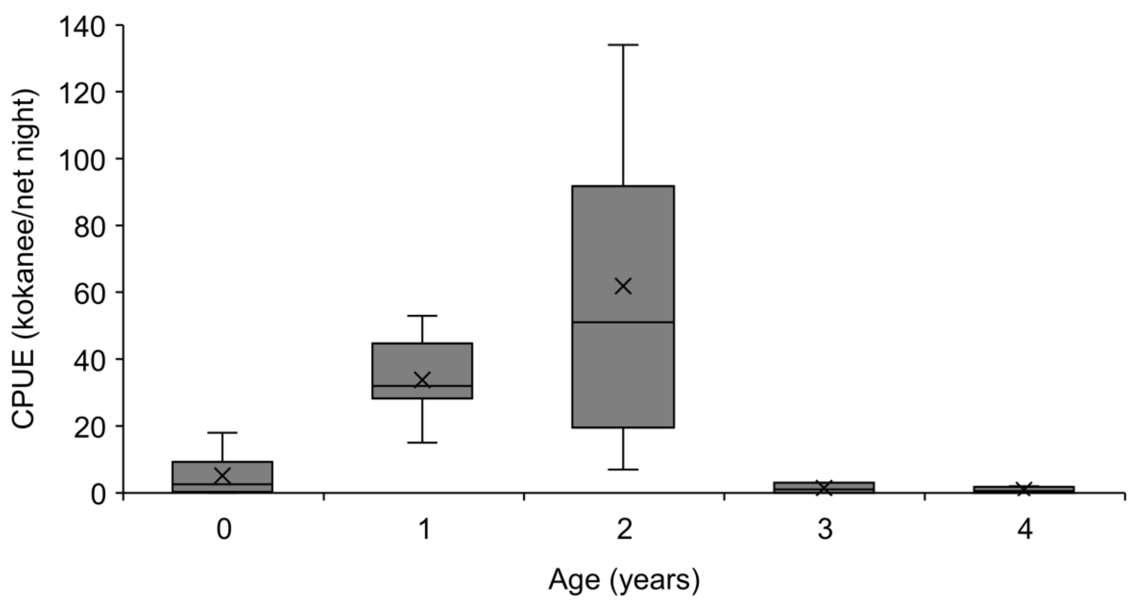

Figure 91. Catch rates (CPUE; number of kokanee/net night) by age class for kokanee in Ririe Reservoir in 2021. The line in each box represents the median, the X represents the mean, the box represents the range between the first and third quartiles, and the whiskers represent the maximum and minimum values.

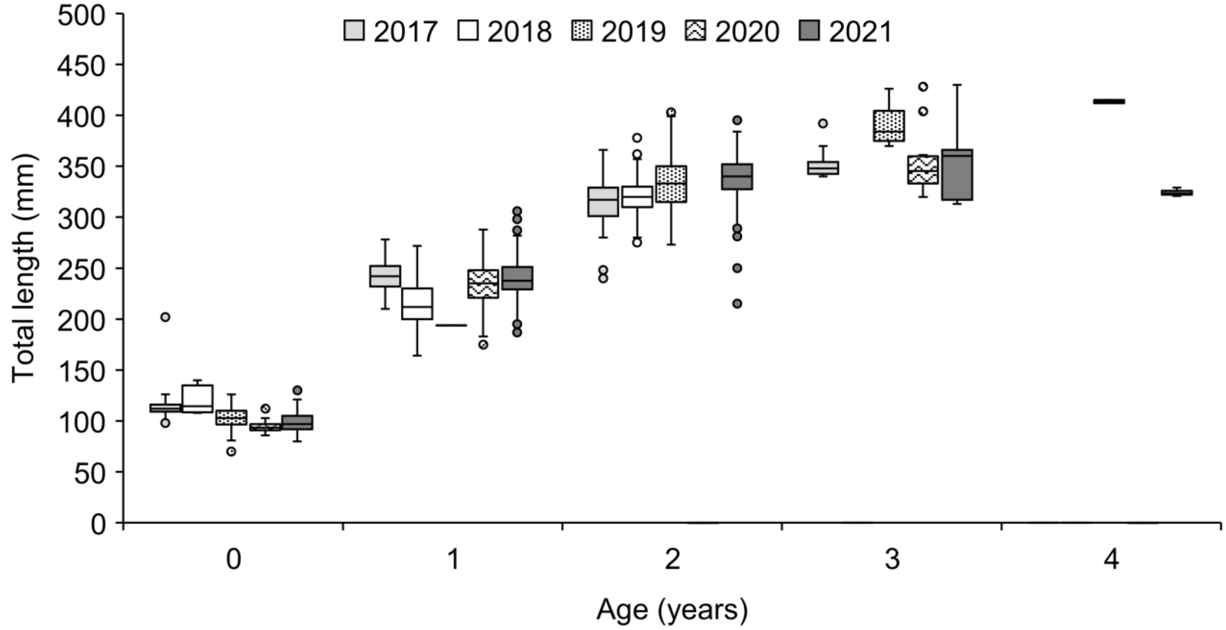

Figure 92. Total length by age of kokanee caught in suspended gill nets in Ririe Reservoir from 2017-2021. The line in each box represents the median, the box represents the range between the first and third quartiles, and the whiskers represent the maximum and minimum values.

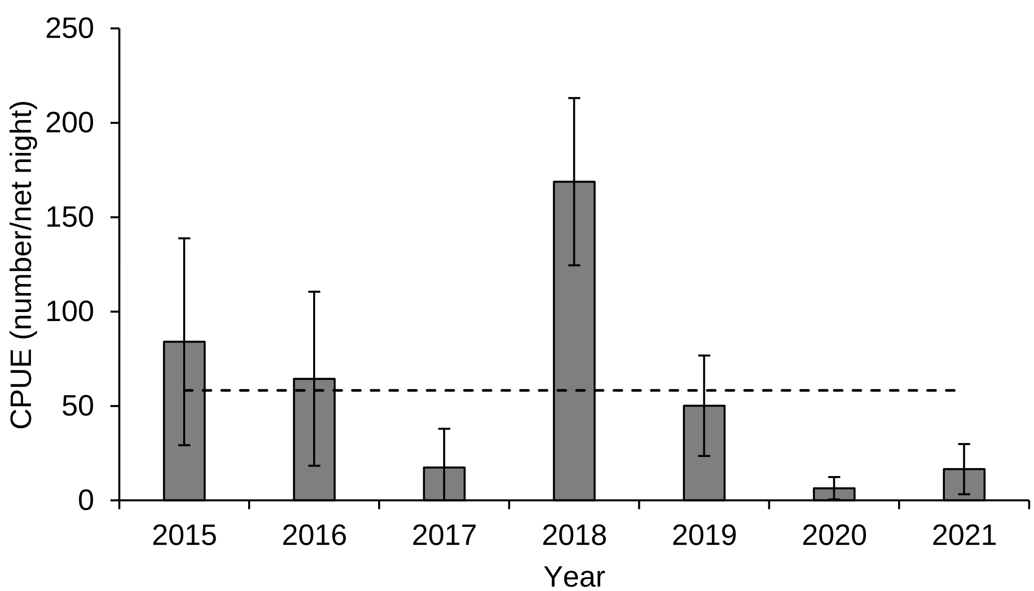

Figure 93. Catch rates (CPUE; number of fish/net night) of Yellow Perch in suspended gill nets in Ririe Reservoir in June from 2015 to 2021. The dashed line represents the mean from 2015-2020 (58 fish/net night) and error bars represent 95% CI.

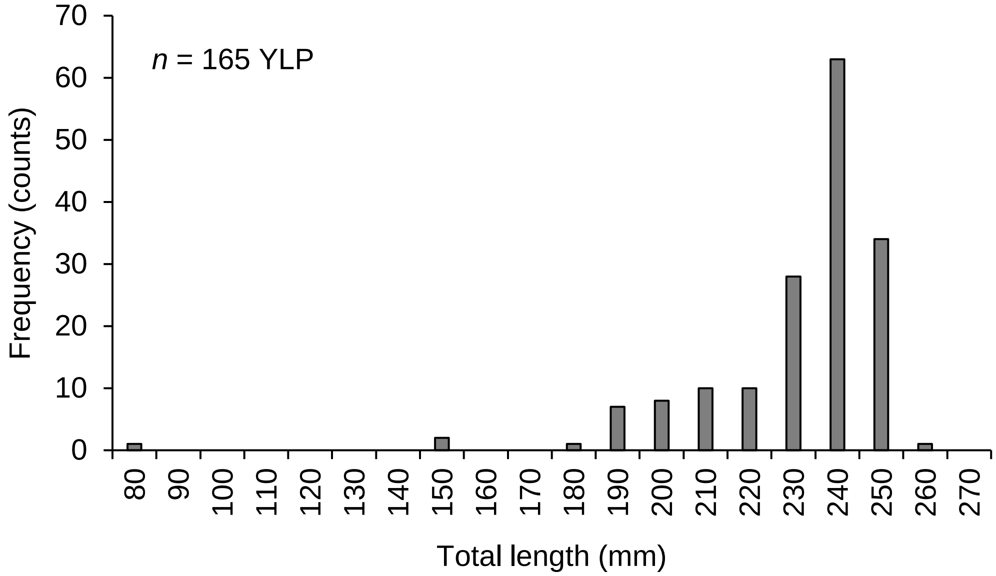

Figure 94. The length-frequency distribution of Yellow Perch (YLP) caught in suspended gill nets in Ririe Reservoir in 2021.

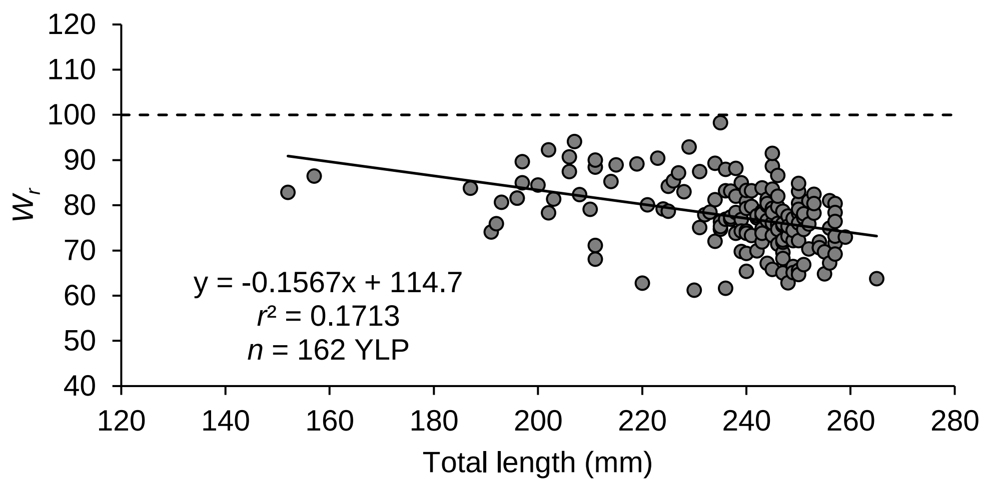

Figure 95. Relative weight (W_r) of Yellow Perch (YLP) across total length caught in suspended gill nets in Ririe Reservoir in 2021. The dashed line represents a relative weight of 100 and the solid line is the regression line for the points.

MACKAY RESERVOIR

ABSTRACT

We conducted a standard lowland lake survey in Mackay Reservoir to assess the fish population and species composition. We set two pairs of gill nets consisting of one floating gill net and one sinking gill net per location. Mean CPUE (number of fish per net night) in gill nets was 29.5 (95% CI ± 5.7) for Rainbow Trout *Oncorhynchus mykiss*, 21.8 (± 10) for kokanee *O. nerka*, and 0.5 (± 0.98) for Yellow Perch *Perca flavescens*. We also set five trap nets on the shoreline in systematic locations. Mean CPUE (number of fish per trap night) in trap nets was 1.0 (± 1.52) for Rainbow Trout. In the final component of the lowland lake survey, we electrofished the entire shoreline because the reservoir storage was at about 19% capacity. The mean CPUE (number of fish per hour of electrofishing) for Rainbow Trout was 122 (± 44), 34 (± 32) for kokanee, 5.4 (± 5.1) for Brook Trout *Salvelinus fontinalis*, 2.5 (± 1.7) for Yellowstone Cutthroat Trout *O. clarkii bouvieri*, and 0.5 fish/hr (± 1) for Yellow Perch. Due to the low abundance of Yellow Perch sampled, we will continue monitoring the fish population in Mackay to maintain up-to-date data on their distribution and abundance.

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INTRODUCTION

Mackay Reservoir is located in Custer County, Idaho (Figure 96) and has a storage capacity of 5,557 hectare meters. Mackay Dam impounds the Big Lost River, which was constructed in 1918 for irrigation storage. The dam was originally owned by the Utah Construction Company until 1936 when it was purchased by the town of Mackay, and the dam is currently owned and operated by the Lost River Irrigation District. The reservoir is stocked annually with triploid, catchable Rainbow Trout *Oncorhynchus mykiss* with detailed records dating back to the 1960s. Kokanee *O. nerka* fry are infrequently stocked (e.g., 2009 and 2019), but wild kokanee reproduction regularly occurs. We manage the reservoir as a put-and-take fishery for trout under general trout and kokanee regulations. Currently, Rainbow Trout, kokanee, Brook Trout *Salvelinus fontinalis*, Yellowstone Cutthroat Trout *O. clarkii bouvieri*, and Mountain Whitefish *Prosopium williamsoni* are the sportfish inhabiting the reservoir. Yellow Perch *Perca flavescens* were illegally introduced sometime prior to 2013 when we first sampled them in gill nets. Anecdotal and substantiated (i.e., photographs) reports of Yellow Perch harvest by anglers prompted the current study to investigate their distribution and abundance in Mackay Reservoir.

METHODS

Mackay Reservoir was drawn down to about 19% of total reservoir capacity when sampling occurred in June 2021, so we adjusted our standard lowland sampling protocol based on the current reservoir capacity (IDFG 2012). We set four experimental gill nets in total at two sites, which included one pair of nets at each site consisting of one floating gill net and one sinking gill net. These sites were selected at locations where sinking and floating gill nets were used in a previous survey (High et al 2015). Gill nets were 45-m long × 1.8 deep with six panels composed of 1.9-, 2.5-, 3.2-, 3.8-, 5.1-, and 6.4-cm bar mesh. In addition, we set five trap nets at randomized locations. Trap nets were constructed with a 23 m lead, 0.9- × 1.8-m frame with five hoops, crowfoot throats on first and third hoops, and 1.9-cm bar mesh. Catch-per-unit-effort (CPUE) for trap nets and gill nets was calculated as the number of fish caught per net night, then we calculated 95% confidence intervals around CPUE estimates.

We measured water temperature (°C) and conductivity (µS/cm) prior to active electrofishing using a handheld probe. Electrofishing began at dusk near the dam, then we proceeded to electrofish the entire shoreline in 600-s sampling increments (IDFG 2012). Pulsed direct current power was provided by a 5000-W generator and standardized to 2,750-3,250 based on water conductivity (Miranda 2009). We applied electricity to the water using an Infinity model electrofisher (Midwest Lake Management, Inc., Polo, Missouri). Two netters positioned at the bow of the boat used 2.4-m long dip nets with 6-mm bar knotless mesh. All fish captured were identified to species, measured for total length (TL) to the nearest millimeter (mm), and weighed to the nearest gram (g). We calculated electrofishing CPUE for each species as the number of fish caught per hour (IDFG 2012) and calculated 95% confidence intervals around those estimates.

We calculated mean TL (± SD), proportional size distribution (PSD), relative stock density (RSD), and relative weights (W_r) to describe the size structure and condition of kokanee and Rainbow Trout in Mackay Reservoir. Kokanee PSD was calculated as the number of fish ≥ 250 mm divided by the number greater than or equal to 120 mm, multiplied by 100. Kokanee RSD-P was calculated as the number of fish ≥ 300 mm divided by the number ≥ 120 mm multiplied by 100. Rainbow Trout PSD was calculated as the number of fish ≥ 300 mm divided by the number ≥ 200 mm, multiplied by 100. Similarly, relative stock densities (RSD-400 and RSD-500) used the same formula, with the numerator replaced by the number of fish ≥ 400 and 500 mm (Anderson and Neumann 1996; Neumann et al. 2012).

We removed sagittal otoliths from a subsample (i.e., 10 fish per 10-mm length group) of kokanee and Rainbow Trout collected from gill-netting for age and growth analysis. We sectioned, polished, and estimated age under a dissecting scope in cross-section view with transmitted light. We created an age-length key from our subsample of kokanee and Rainbow Trout, then applied the age-length key to unknown age fish. We calculated mean (and standard deviation) length-at-age using the Isermann and Knight (2005) method in the FSA package in program R (R Core Team 2019). We further estimated annual survival (*S*) and instantaneous total mortality (*Z*) using a catch curve with the Chapman-Robson (1960) method and Peak Plus criterion in the FSA package in program R (Pauly 1984; Smith et al. 2012; R Core Team 2019). We also calculated annual mortality (*A*) where:

$$A = 1 - e^{-Z}$$

for kokanee and Rainbow Trout (Ricker 1975).

RESULTS

We collected 207 fish during four nights of gill-netting effort in Mackay Reservoir. Species composition was dominated by Rainbow Trout (57%), followed by kokanee (42%), and Yellow Perch (1%). We sampled one non-game species in electrofishing and trap netting, which was Speckled Dace *Rhinichthys osculus*. Mean CPUE in gill nets was 29.5 (± 5.7) for Rainbow Trout, 21.8 (± 10) for kokanee, and 0.5 fish/net-night (± 0.98) for Yellow Perch (Table 26). We sampled five Rainbow Trout in trap nets and CPUE was 1.0 fish/net-night (± 1.52). We sampled 332 fish during night electrofishing consisting of 74% Rainbow Trout, 21% kokanee, 3% Brook Trout, 2% Yellowstone Cutthroat Trout, and <1% Yellow Perch. The mean electrofishing CPUE for Rainbow Trout was 121.8 (± 44), 34.0 (± 32) for kokanee, 5.4 (± 5.1) for Brook Trout, 2.5 (± 1.7) for Yellowstone Cutthroat Trout, and 0.5 fish/h (± 1) for Yellow Perch. Size structure varied by sampling gear type. The average TL of Rainbow Trout sampled in gill nets was 363 mm (SD ± 30), PSD was 97, RSD-400 was 15, and RSD-500 was 2 (Figure 97). During electrofishing, the average TL of Rainbow Trout was 345 mm (± 147), PSD was 89, RSD-400 was 33, and RSD-500 was 20. The average TL of kokanee sampled in gill nets was 236 mm (± 30), PSD 39, and RSD-P was 0 (Figure 98). However, the average TL of kokanee sampled from electrofishing was 141 mm (± 68), PSD was 30, and RSD-P was 0. We sampled 11 Brook Trout with electrofishing and the average TL was 270 mm (± 24), and five Yellowstone Cutthroat Trout with an average TL of 305 mm (± 45). We sampled Yellow Perch in gill nets (*n* = 2) with an average TL of 218 mm (± 78), and with electrofishing (*n* = 1; TL = 170 mm). The average relative weight (W_r) for kokanee across gear types was 79 (SD ± 8), which displayed an increasing weight with an increase in TL (Figure 99). We estimated *S* at 39%, *Z* at 0.94, and *A* at 61% for ages 1-3. The average W_r of Rainbow Trout across gear types was 80 (± 10), which displayed an increasing weight with an increase in TL (Figure 100). We estimated *S* at 39%, *Z* at 0.93, and *A* at 61% for ages 4-7.

DISCUSSION

Compared to other reservoirs in the Upper Snake Region, Mackay Reservoir has little long-term monitoring or historical data. The department sampled Mackay Reservoir with two gill nets in May 1962 (IDFG unpublished data, Stacy Gebhards, Fisheries Biologist) which yielded eight Rainbow Trout and two Mountain Whitefish. Mackay Reservoir was also sampled with two gill nets on two occasions during May 1973 (Jeppson 1975). The majority of the work conducted on Mackay Reservoir since Jeppson's gill netting has been angler surveys. However, in 2008 we conducted a survey that included six gill nets set overnight. Power analysis of this amount of effort suggested that level of sampling was capable of detecting a 25% change in Rainbow Trout abundance (IDFG unpublished data). We recommend establishing a routine sampling schedule that managers can use in future comparisons. In 2013, IDFG conducted the first thorough, comprehensive gill-netting effort on Mackay Reservoir. We used 17 floating and 13 sinking gill nets, which is excessive effort for an impoundment of this size (IDFG 2012). Therefore, in 2019 we reduced our effort to eight nets total in the mid to lower sections of the reservoir, which are the deeper portions of the reservoir that maintain pool throughout the year.

Beginning several years ago. anglers reported catching Yellow Perch in Mackay Reservoir, but we had not sampled Yellow Perch in our 2017 and 2019 surveys. Idaho Fish and Game has never stocked Yellow Perch in Mackay reservoir, so their presence suggests an illegal introduction. We are concerned because Yellow Perch may compete for prey with the wild kokanee population. During the winter of 2020 we received images of Yellow Perch that were caught through the ice by an angler. In response, we chose to conduct a thorough evaluation of the fishery using a Standard Lowland Lake Survey (IDFG 2012) to try and sample Yellow Perch and describe the current fish species assemblage. We captured three Yellow Perch in our sampling efforts, which confirmed their presence and indicated that their abundance was low at this point. Multiple size classes were present in our sample suggesting they are reproducing. The reservoir pool stage was at about 19% of capacity so it may be difficult to encounter Yellow Perch again at this low level of abundance when the reservoir is at a higher capacity, but sampling intensity would increase concurrent with reservoir increases (IDFG 2012). Therefore, we need to continue monitoring their abundance and population expansion, which should be evident if we observe higher catch rates at greater reservoir capacity. In addition, we need to continue interacting with anglers to collect catch composition data to see if they are encountering Yellow Perch through angling. We also should continue to monitor kokanee growth and survival rates as Yellow Perch presumably become more abundant, to quantify potential negative interactions between the species.

In the current survey, we conducted a more thorough evaluation of the fishery by combining gear types to sample the fish community. We sampled multiple sportfish species, but we sampled only one non-game fish species. For larger fish that may transition to a piscivorous diet, there is a higher abundance of sportfish as prey than non-gamefish. We observed different size structures of kokanee sampled via electrofishing versus gill-netting. For example, no age-0 kokanee were sampled with our gill nets, but they were sampled using the boat electrofishing gear. However, our standard lowland lake gill nets are not constructed with mesh small enough to capture age-0 kokanee, so we should combine lowland lake gill nets with kokanee gill nets for the next round of gill-net sampling in the reservoir to produce a sample representative of the populations at large, consisting of multiple species and size classes. In addition, boat electrofishing in the reservoir was more productive than gillnetting for Rainbow Trout, and we captured a wider range of size classes for kokanee and Rainbow Trout. This sample method should continue to be used to monitor the Rainbow Trout population because it was more productive, and there was no observed short-term fish mortality from electrofishing. However, the effectiveness of electrofishing could be an artifact of the low reservoir level, but conducting another lowland lake survey will provide new data for comparison to evaluate electrofishing effectiveness. Both kokanee and Rainbow Trout populations are robust with fish in multiple year classes present. The kokanee population is wild, whereas we do not know the origins of Rainbow Trout. We observed holdover hatchery-origin RBT, recently stocked Rainbow Trout, and juvenile Rainbow Trout less than 170 mm. However, the reservoir is often stocked with triploid Rainbow Trout fingerlings that are surplus from Mackay Hatchery. Therefore, in future sampling efforts tissue samples should be taken from juvenile Rainbow Trout to test ploidy level so we can better understand the population and whether stocking is conducted at an adequate level. Additionally, we need to continue monitoring this fishery after the low reservoir level this year and after confirming the presence of Yellow Perch.

MANAGEMENT RECOMMENDATIONS

1. Continue monitoring the kokanee population every three years using the increased effort with kokanee-specific gill nets to assess the population dynamics of kokanee, to evaluate 2021 stocking success, and to inform management decisions.

2. Every three years, conduct a standard lowland lake and reservoir survey to assess the Yellow Perch population, and the Rainbow Trout population to inform stocking scenarios, harvest regulations, and to assess the population dynamics of Rainbow Trout with a large sample size.

3. During the next round of sampling, take tissue samples from Rainbow Trout to test for ploidy level.

Table 26. Catch rates (± 95% CI) by species for gill nets, trap nets, and electrofishing in Mackay Reservoir in 2021.

Species	Fish/gill-net night	Fish/trap-net night	Fish/h electrofishing
Rainbow Trout	29.5 (± 5.7)	1 (± 1.5)	121.8 (± 44)
Kokanee	21.8 (± 10)		34.0 (± 32)
Brook Trout			5.4 (± 5.1)
Yellowstone Cutthroat Trout			2.5 (± 1.7)
Yellow Perch	0.5 (± 1.0)		0.5 (± 1.0)

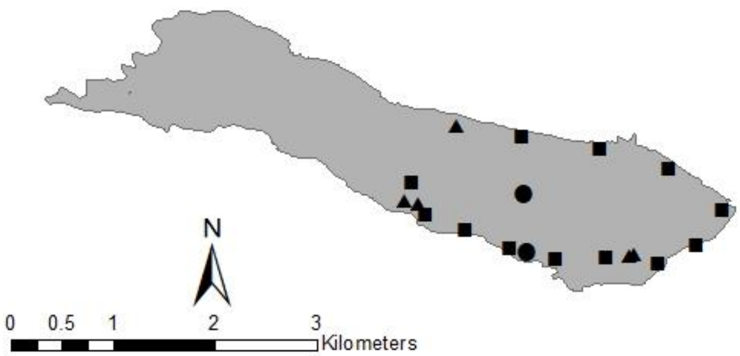

Figure 96. Gill net (black circles), trap net (black triangles), and electrofishing (black squares) sampling locations in Mackay Reservoir, 2021.

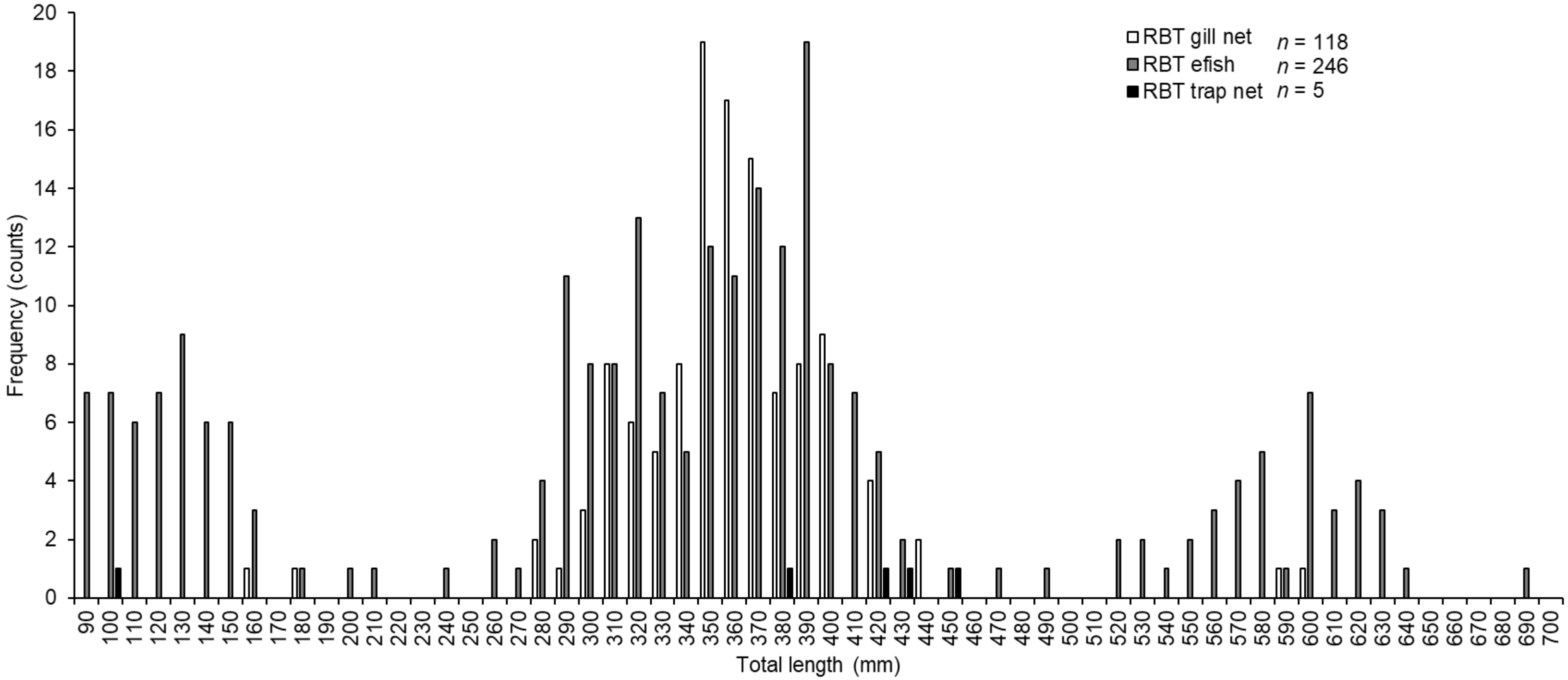

Figure 97. The length-frequency distribution of Rainbow Trout (RBT) sampled in gill nets, electrofishing (efish), and trap nets in Mackay Reservoir 2021.

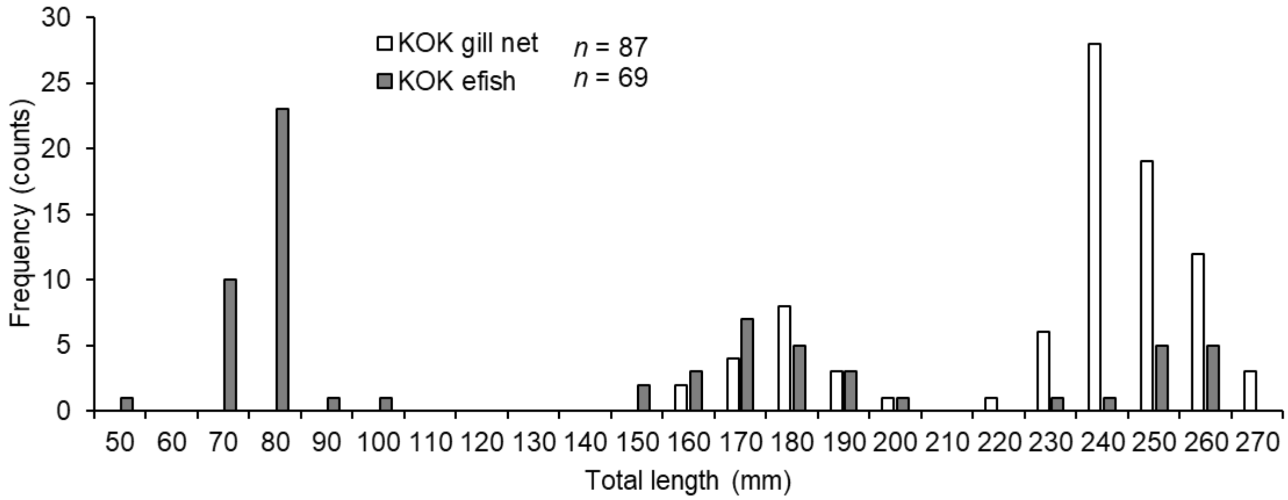

Figure 98. The length-frequency distributions of kokanee sampled in gill nets and from electrofishing (efish) in Mackay Reservoir 2021.

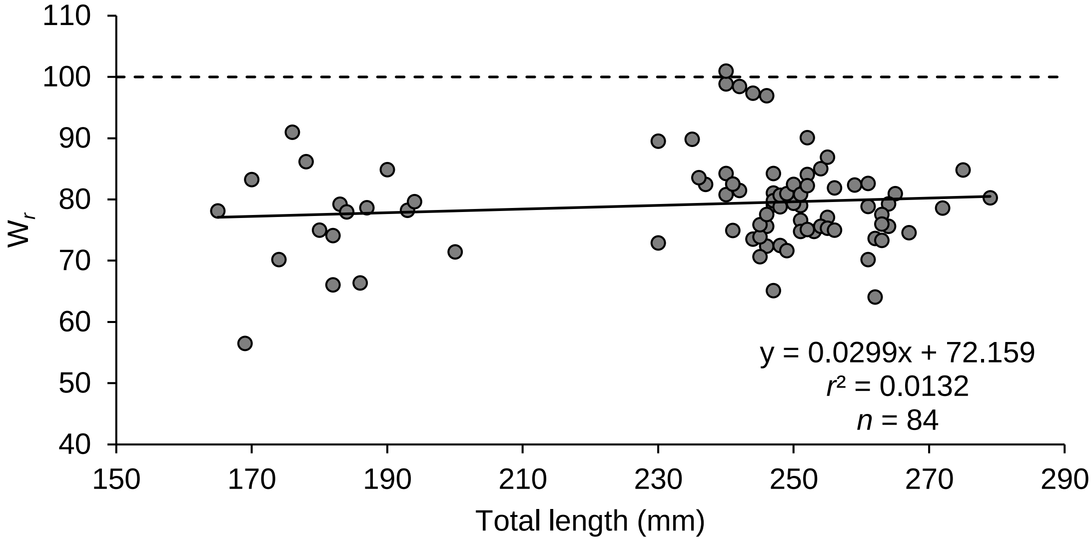

Figure 99. The relative weight (W_r) of kokanee sampled from gill nets and electrofishing in Mackay Reservoir in 2021. The dashed line represents a relative weight of 100, and the solid line represents the regression relationship between W_r and TL.

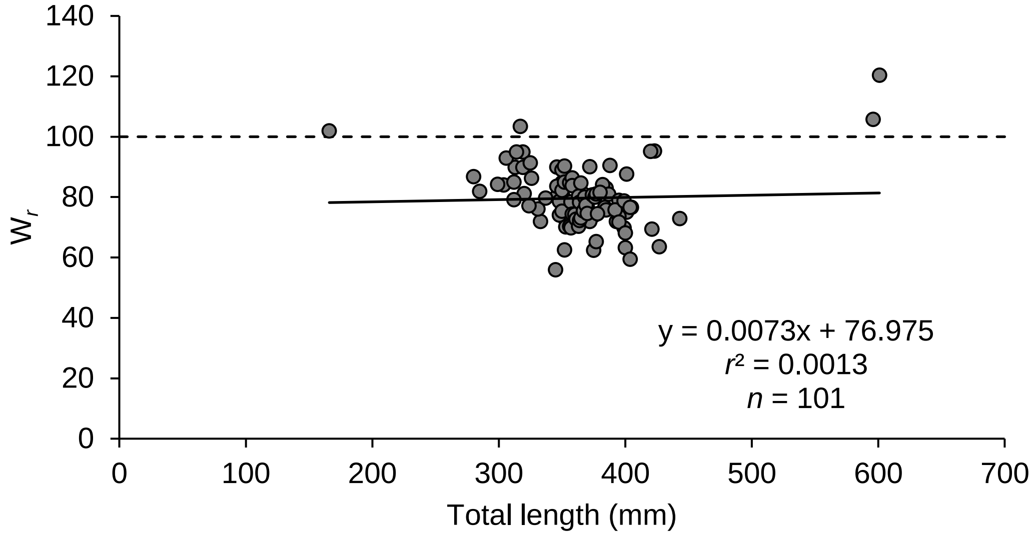

Figure 100. The relative weight (W_r) of Rainbow Trout sampled from gill nets, trap nets, and electrofishing in Mackay Reservoir in 2021. The dashed line represents a relative weight of 100, and the solid line represents the regression relationship between W_r and TL.

MUD LAKE

ABSTRACT

We conducted a lowland land and reservoir survey on Mud Lake in mid-May to collect current information on the relative abundance, species composition, body condition, and size distribution for fish species present in the lake. We also marked Largemouth Bass *Micropterus salmoides* with anchor tags in order to estimate annual angler exploitation and use. We collected dorsal spines from Largemouth Bass to assign ages and used these structures to back-calculate length-at-age to assess growth by age group. We captured a total of 850 fish during the survey with Utah Chub *Gila atraria* being the most common fish captured. Catch-per-unit-effort for species caught in the survey were 208.8 fish/unit effort for Utah Chub, 33.2 for Largemouth Bass, 25.8 for Brown Bullhead *Ameiurus nebulosus*, 12.0 for Yellow Perch *Perca flavescens*, 6.0 for Bluegill *Lepomis macrochirus*, and 3.7 for Utah Sucker *Catostomus ardens.* Relative weights for all species exceeded 100 except for Utah Chub and Yellow Perch, with Largemouth Bass averaging 122. The ages of Largemouth Bass ranged from 1 to 11, and all age classes except age-10 were represented. The annual exploitation rate (7%) was low for Largemouth Bass and the annual mortality rate was 32%. Growth rates for Largemouth Bass were fast. Plots of growth by age indicated inconsistent growth with some indication of alternate year spawning. Future efforts to describe fishery composition, performance, factors affecting recruitment, factors affecting survival, and angler preferences would benefit management efforts to improve this unique fishery in the Upper Snake Region.

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INTRODUCTION

Mud Lake is a 4,500-acre shallow lake in the Upper Snake plain located at the terminus of Camas Creek, one of the sinks drainages in the region. Historically, the Camas Creek drainage playa was more extensive than Mud Lake is now. As the area was developed, dikes were built to convert wetlands to agricultural areas, forcing water into a smaller area. Mud Lake has an average depth of 5 feet with extensive tall emergent marshes and shallow wetlands. Mud Lake is part of the 4,641 hectare (11,468 acre) Mud Lake Wildlife Management Area (WMA), which provides important stopover and nesting habitat for waterfowl and shorebirds. Management priorities for the Mud Lake WMA are focused on habitat for breeding and migratory waterfowl and for a wide variety of other game and nongame species as well as providing high quality recreational opportunities (Hendricks 2014). The fisheries management objectives for Mud Lake is to provide for a warm water fishery (IDFG 2019).

Mud Lake plays an important part in the agricultural community as a water storage and delivery system. Historically, spring flows in Camas Creek were high enough that flooding was a concern for the Mud Lake and Terreton towns adjacent to Mud Lake. Spring runoff volumes have decreased in recent years to the point where natural flow typically does not fill Mud Lake. Instead, water right holders pump ground water into the lower portions of Camas Creek to augment inflows and fill Mud Lake. This water is highly regulated and accounted for at the expense of the irrigators. Thus, water users do not typically augment inflows more than what’s needed to meet irrigation demands. This results in relatively constant lake levels from spring through summer during irrigation season, dropping lake levels with the cessation of irrigation each fall, and low lake levels each winter as water laws do not allow irrigators to carry water over from year-to-year.

In a shallow, productive lake like Mud Lake, aquatic vegetation is common across most of the lake bed. When lake levels are lowered in fall and winter, and as macrophyte biomass decomposes, especially under ice cover, low dissolved oxygen conditions occur and can cause fish kills. Winter kills have happened at Mud Lake to some extent on a near-annual basis (IDFG 2019).

Based on trends of the number of boat trailers observed at boat ramps and access areas, there appears to be an increasing amount of angler effort on Mud Lake, with anglers targeting Largemouth Bass *Micropterus salmoides* (Brett Panting, IDFG wildlife, personal communication). Mud Lake is currently managed under general fishing regulations which includes a six-bass limit and no size restrictions. Because Mud Lake has a history of frequent wintertime fish kills, a small bass population, and the apparent increase in angling effort targeting Largemouth Bass, we used a lowland lake survey (IDFG 2012) to collect basic life history information for Largemouth Bass and other fish species at Mud Lake to inform future management decisions.

OBJECTIVES

1. Describe the species composition and size structure of the fish community at Mud Lake.

2. Describe Largemouth Bass growth and survival rates.

3. Determine current levels of angler use and exploitation of Largemouth Bass in Mud Lake.

METHODS

Standardized lowland lake survey methods employed by IDFG (IDFG 2012). Fish populations in Mud Lake were sampled with standard IDFG lowland lake sampling gears during May 2021. In total, two trap nets, two gill net pairs, and one electrofishing unit (composed of seven sub-samples) were used in Mud Lake (Figure 101).

Relative weight (*Wr*) was calculated for each species captured, except Utah Sucker *Catostomus ardens*, as an index of general fish body conditions where a value of 100 is considered average (Anderson and Nuemann 1996). We used the standard weight equations for Utah Chub *Gila atraria* (IDFG unpublished data), Largemouth Bass (Wege and Anderson 1978), and Brown Bullhead *Ameiurus nebulosus* (Carlander 1969), Yellow Perch *Perca flavescens* (Willis et al. 1991), and Bluegill *Lepomis macrochirus* (Carlander 1969). Relative weight values greater than 100 describe abundant food sources and healthy fish, whereas values less than 100 indicates less than ideal foraging conditions. Catch-per-unit-effort (CPUE) and weight-per-unit-effort (WPUE) in kg were calculated by standardizing the catch of each gear type to one unit of effort and then summing across the three gear types using a ratio of means estimator. Summing across gear types yielded a combined CPUE estimate as well as a combined WPUE estimate.

We collected the 2nd dorsal fin spine of all captured Largemouth Bass, to assess fish age. We embedded the spine in epoxy and cut thin sections using a slow speed saw, and then took digital images of the spines using a compound microscope attached to a digital camera and computer. Two readers estimated the age of each fish without knowing fish length. When discrepancies were encountered, the two readers reconciled those differences for each fish while adding in fish length information. The Dahl-Lea method was used to estimate back-calculated lengths-at-ages:

$$L_i = (S_i/S_c) \times L_c ,$$

where L_i is the back-calculated length of the fish when the *i*th increment was formed, L_c is the length of the fish at the time of capture, S_c is the radius of the spine at the time of capture, and S_i is the radius of the spine at the *i*th increment (Francis 1990; Quist et al. 2012). Mean back-calculated lengths-at-ages were reported by year class. Total annual mortality (*A*) was estimated using weighted catch curves for age-3 and older fish (Ricker 1975; Smith et al. 2012).

We implanted all of the Largemouth Bass we captured during May 2021 electrofishing with non-reward, T-bar anchor tags at the base of the dorsal fin according to standard methods (Dell 1968). Anchor tags were printed with a unique identification number, phone number, and website address where anglers could report the tag using the “Tag You’re It” statewide tag reporting system (Meyer and Schill 2014). We used data obtained from reported tags to estimate exploitation, caught and released fish, and total angler use. We downloaded reported tag data from the database a year after tagging (June 2022).

We estimated the angler reporting rate (λ) using the average reporting rate of non-reward tags in the current study relative to the high-reward tags of hatchery Rainbow Trout as estimated by Meyer et al. (2012). In the current study, we used statewide averages to estimate tag loss and tagging mortality (Meyer and Schill 2014). We estimated angler exploitation (u') using the equation:

$$u' = u/(\lambda\ (1\text{-[Tag]\_l})(1\text{- [Tag]\_}m)),$$

where *u* is the number of non-reward tagged fish that were reported as harvested divided by the total number of non-reward tagged fish stocked, λ is the corrected reporting rate for Largemouth Bass (i.e., 0.407) reported by Meyer et al. (2012), [Tag]_l is the first year tag loss rate (i.e., 0.088), and [Tag]_m is the tagging mortality rate (i.e., 0.01). We used the tag loss and tagging mortality estimates reported by Meyer and Schill (2014). We also estimated angler use by modifying *u* to include fish reported as caught-and-released.

RESULTS

A total of 850 fish were captured during the standard lowland lake survey at Mud Lake in 2021. Catch was predominately Utah Chub (*n* = 671) followed by Largemouth Bass (*n* = 74) and Brown Bullhead (*n* = 59). Other species captured, included Yellow Perch, Bluegill, and Utah Sucker (Table 27). Combined CPUE and WPUE indices for all species and gear types when totaled were 289.4 fish/unit effort and 747.0 kg/unit effort (Table 27). Based on combined CPUE, Utah Chub made up 72.2% of the total catch, followed by Largemouth Bass (11.5%), and Brown Bullhead (8.9%). All of the other species collected contributed < 8.0% of total catch (Table 27). Based on combined WPUE, the fish total biomass consisted primarily of Utah Chub (94.2%). All the other species collected contributed < 6.0% to total biomass (Table 27). Electrofishing was the most effective gear type both in terms of CPUE (total CPUE = 135.9 fish/h) and species diversity as all species observed during the survey were represented in the electrofishing sample.

Utah Chub were the most abundant fish sampled by number (*n* = 671). They were captured with a combined CPUE of 208.8 fish/unit effort and a combined WPUE of 703.7 kg/unit effort (Table 27). Gill nets yielded the highest CPUE (108.3 fish/net night) of the individual gear types followed by electrofishing (CPUE = 60.1 fish/h) and trap nets (CPUE = 15.3 fish.net night). Total length of Utah Chub ranged from 99 to 400 mm (Figure 102). Relative weights for Utah Chub averaged 93 and had a stable trend across all lengths (Figure 103).

Largemouth Bass were the second most abundant fish caught by number (*n* = 74). They were captured with a combined CPUE of 33.2 fish/unit effort and a combined WPUE of 29.8 kg/unit effort (Table 27). Electrofishing was the only gear type used that captured Largemouth Bass. Total length of Largemouth Bass ranged from 166 to 480 mm (Figure 104). Largemouth Bass were also the second most sampled fish by combined WPUE (29.8 kg/unit effort) although this was only 4.0% of the biomass sampled. Relative weights for Largemouth Bass averaged 122 and had an increasing trend with total length (Figure 105). Largemouth Bass ages ranged from 1 to 11 years (Figure 106). Annual mortality rate for Largemouth Bass was estimated to be 32%. Annual growth for Largemouth Bass ranged from 159 to 293 mm per year (Table 28). Growth was cyclical, meaning that growth during odd year ages was relatively consistent and good, while growth during even year age classes was less, but also consistent. Growth to the quality size class for Largemouth Bass was fast, with fish reaching this size in their third year (Figure 106). We marked all 74 Largemouth Bass that we captured with anchor tags. Anglers reported catching 12 marked bass between June 21, 2021 and June 6, 2022 (2 to 350 d after tagging). Two of the 12 Largemouth Bass captured by anglers were harvested, and both anglers indicated they harvested the fish because it was tagged. The estimated angler exploitation rate for Largemouth Bass in Mud Lake after accounting for non-reporting bias, tag retention, and tagging mortality, was 7%. The estimated total angler use of Largemouth Bass, after accounting for these same sources of bias, was 44%.

Brown Bullhead were the third most abundant fish caught by number (*n* = 59). They were captured with a combined CPUE of 25.8 fish/unit effort and a WPUE of 8.2 kg/unit effort (Table

27). Electrofishing was the most effective capture method (CPUE = 23.8 fish/h) followed by trap nets (2.0 fish/net night). Brown Bullheads were not captured in the gill nets. Total length averaged 259 mm and ranged from 135 to 313 mm (Figure 107). Brown Bullheads comprised 1.1% of the biomass in the combined sample, with electrofishing having the highest individual WPUE (7.6 kg/unit effort) compared to trap nets (0.5). Relative weights for Brown Bullhead averaged 133 and had an increasing trend with fish length (Figure 108).

Yellow Perch (*n* = 30) were captured with a combined CPUE of 12.0 fish/unit effort and WPUE of 1.4 kg/unit effort. Electrofishing was the most effective gear type (CPUE = 11.0 fish/h) followed by gill nets (1.0 fish/net night). No Yellow Perch were captured in trap nets. Total length averaged 192 mm and ranged from 82 to 330 mm (Figure 109). Relative weights averaged 92 with a decreasing trend with fish length (Figure 110).

Bluegill (*n* = 30) were captured with a combined CPUE of 6.0 fish/unit effort and WPUE of 1.3 kg/unit effort. Electrofishing was the only method used that captured Bluegill. Total length averaged 198 mm and ranged from 177 to 275 mm (Figure 111). Relative weights averaged 141 with a decreasing trend with fish length (Figure 112).

We captured five Utah Sucker during the lowland lake survey at Mud Lake. This included a single sucker during electrofishing, three in trap nets, and one in gill nets. The combined CPUE for Utah Sucker was 3.7 fish/unit effort and WPUE was 2.6 kg/unit effort. Total length averaged 422 mm and ranged from 198 to 494 mm.

DISCUSSION

The relative abundance of species in Mud Lake has changed throughout the years. Prior to the 1960s, Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* were abundant in Mud Lake, but their densities declined as water inputs declined, and water temperatures increased (Andriano 1955; IDFG 2019). Starting in 1990, Lahontan Cutthroat Trout *O. clarkia henshawi* were experimentally stocked into Mud Lake to see if they could persist in the warmer temperatures. This experiment was unsuccessful. Other unsuccessful fish introduction efforts include Black Crappie *Pomoxis nigromaculatus* and Smallmouth Bass *Micropterus dolomieu* (IDFG 2019). Tiger muskie (Northern Pike *Esox Lucius* x Muskellunge *E. masquinongy*), were introduced into Mud Lake from 1988 through 2018. We occasionally receive reports of anglers catching or observing these fish. However, we did not capture a tiger muskie during our survey, which indicates their abundance is likely very low. In 1980, experimental gill netting indicated the fish population was 88% Utah Chub and 11% Utah Sucker (Ball and Jeppson 1980). Over a decade later, Utah Chub were no longer the dominant species. Gill-netting efforts in 1992 indicated Utah Sucker were the most prevalent species (91%) in Mud Lake followed by Utah Chub (9%). During this survey, tiger muskie, Lahontan Cutthroat Trout, Largemouth Bass, Yellow Perch, and Brown Bullhead were also observed, but made up less than 1% of the catch (Gamblin 1995). The survey conducted in 1992 was the only other time a survey has been completed using standard lowland lake survey methodology. A substantial winterkill the winter following this survey was observed and documented. Three gill nets set overnight did not capture a single fish, and plans were made to aggressively restock Mud Lake with the species observed during the 1992 lowland lake and reservoir survey (Gamblin et al. 2001). Results from the current study indicate Utah Chub are again the dominant species in Mud Lake, but the lake is currently supporting a number of game species including Largemouth Bass, Brown Bullhead, Yellow Perch, and an occasional Bluegill.

In general, the body condition for fish in Mud Lake is extremely high. This is particularly true for Largemouth Bass, and is likely a reason for the recent increasing trend of angling use at Mud Lake. Relative weights in Mud Lake averaged 122. This appears to be the highest relative weight recorded for a Largemouth Bass population in Idaho. For comparison, Largemouth Bass in Glendale Reservoir (Southeast Region) averaged 101 to 105 between 1995 and 1999 (Scully et al. 2003). In 1992 at Crane Falls, Largemouth Bass relative weights averaged 98 (Allen et al. 2000). In 2015, average relative weight of stock length fish at Brush Lake in the Panhandle Region was 82 and Avondale Lake had bass with average relative weights of 91 (Ryan et al. 2018). In Bonner Lake, relative weight averaged 92 for Largemouth Bass in 2014 and was 95 for bass in Smith Lake (Watkins et al. 2018). Relative weights for Largemouth Bass in CJ Strike Reservoir in 1984 were close to the standard for all lengths of fish (Reid 1985), and in 1988 at CJ Strike Reservoir, relative weight increased with fish size, but no individuals exceeded 130 (Mabbott and Holubetz 1990). Relative weights for the other species we captured during our survey (except Utah Chub and Yellow Perch) were also high and exceeded 100 indicating Mud Lake is very productive.

The shallow depths of Mud Lake causes this water body to be very productive, but it also leads to frequent winterkill events which have been a concern for fishery managers and anglers. It is unknown if lake productivity is the primary reason for high body conditions for fish in Mud Lake, or if there is a combination of factors affecting body condition, such as productivity and limited fish abundance due to winterkill. If limited abundance caused by frequent winterkill worked in concert with high productivity to result in high body condition, we would have expected natural and total mortality rates for Largemouth Bass to be higher than normal. Results from this study suggest lake productivity is the main factor. A recent review of Largemouth Bass total annual mortality rates reported an overall average of 57% for Largemouth Bass from 30 separate surveys spanning 51 years (Allen et al. 2008). Our estimate for annual mortality of Largemouth Bass in Mud Lake was 32%, which is lower than what is reported for other Largemouth Bass populations. With relatively low annual mortality rates, it seems winterkill events are not major factors directly influencing abundance of Largemouth Bass and indirectly enhancing body condition in Mud Lake. Furthermore, Largemouth Bass growth in Idaho have not demonstrated density dependence, likely due to variable annual recruitment rates (Dillon 1992).

While the popularity of fishing for Largemouth Bass in Mud Lake appears to be increasing, the exploitation rate is still low, suggesting angler harvest does not affect this population. We estimated the annual angler exploitation rate to be 7% for Largemouth Bass in Mud Lake. In comparison with our estimated annual mortality rate (32%) and a relative high rate of annual mortality for Largemouth Bass (57%) reported in other studies (Allen et al. 2008), it is unlikely an exploitation rate of 4% could have an impact on the population of bass in Mud Lake. Additionally, our estimate of 7% may be biased high, as the anglers who harvested the two bass in our study reported they harvested the fish because it was tagged. If these bass were not tagged, it is likely those anglers would have released them.

Harvest was low, but anglers caught an estimated 44% of the Largemouth Bass population in Mud Lake. While 44% is not the majority of the population, it is impressive given the context of the fishery at Mud Lake. The window for fishing for Largemouth Bass in Mud Lake is fairly small (i.e. late spring through early summer). Water temperatures and water levels limit opportunity in spring and by mid-summer aquatic vegetation is thick enough that it limits fishing opportunity via access and fishability. So for nearly half of the population to be captured by anglers in this short window at Mud Lake, it is apparent angler use is substantial. The estimate of angler use in 2021 will be a good starting point to gauge changing effort by, if an increasing angler effort trend continues. While bass fishing tournaments have not been held recently, tournaments were held in the 1990s (Elle and Gamblin 1993) and may become popular again, leading to additional angler use on this population.

Our back-calculated-length-at-age data suggested an interesting trend in growth rates of Largemouth Bass. Growth rates of Largemouth Bass in Mud Lake were consistent and good for bass with odd year ages, while growth rates for fish during their even age years was again consistent, but not as good. The back and forth, fast/slow, repeated trend indicates Largemouth Bass in Mud Lake may be alternate year spawners, with bass using more of their growth resources for spawning versus growing, during their even age years, starting as young as age-2 (Nieman et al. 1979). Largemouth Bass are known to mature at young ages (Laarman and Schneider 1985) and maturity may be more linked to sizes exceeding 250 mm than age (Nieman et al. 1979). Age-2 Largemouth Bass in Mud Lake are larger than 250 mm and may be mature, though we didn’t examine gonads. Interestingly, growth rates from back-calculated length-at-age data do not slow down with age for Largemouth Bass in Mud Lake (Table 28). However, there was a clear asymptote in the length-at-age data for our age groups of bass we captured and aged (Figure 106). This suggests there may some be bias in our back-calculated length-at-age data, which should be assessed during the next lowland lake survey at Mud Lake. Our quantitative findings on growth rates differed from qualitative observations from an earlier report on Mud Lake, when Largemouth Bass in Mud Lake were described as “slow-growing” after assessing size of fish caught during a bass tournament (Gamblin 1992). It is likely that sample size, analysis technique, and capture methodology caused this discrepancy.

Historically, Yellow Perch have periodically been a large component of the fishery at Mud Lake, and this species being an important component of the warmwater fishery at Mud Lake is included in the current Fisheries Management Plan (IDFG 2019). Currently, Yellow Perch are present, but appear to be in low abundance. Yellow Perch are a species that can tolerate low dissolved oxygen levels (Petrosky and Magnuson 1973). Thus, winterkill events are likely not the main factor affecting Yellow Perch recruitment or survival in Mud Lake. Future efforts to describe factors limiting Yellow Perch recruitment and survival in Mud Lake would benefit the department’s ability to work to improve this component of the Mud Lake fishery.

Brown Bullhead were the second-most common game species caught during our survey, and are likely contributing to the fishery at Mud Lake due to their abundance and size. With an average length of 259 mm and an average relative weight of 133, these fish provide an opportunity to catch a quality bullhead. Indeed, we have heard from local anglers that Brown Bullhead are being targeted by anglers more frequently, particularly by harvest-oriented anglers (Brett Panting, IDFG wildlife, personal communication). We currently have no data for angler catch or harvest rates for Brown Bullhead, or estimates of angler effort or preference for this species. Creel surveys to quantify these parameters in addition with “Tag You’re It” studies, would provide valuable information to aid future fishery management decisions.

Tiger muskie were notably absent from the 2021 Mud Lake survey. Previous surveys have resulted in observations of tiger muskie in Mud Lake (Gamblin et al. 2001), but these surveys followed events with much higher stocking numbers. Annual stocking from 1997 through 1999 ranged from 34,000 to 36,000 tiger muskie, while more recent stocking events (2016 through 2018) have included totals ranging from 336 to 2,900. This suggests that we could see some recruitment to the Mud Lake fishery with extremely high stocking densities, but the angling benefits may be minimal given the level of stocking necessary to translate to larger fish contributing to the fishery.

MANAGEMENT RECOMMENDATIONS

1. Perform a creel survey to quantify effort and catch.
2. Collect angler opinion survey data to better inform management decisions at Mud Lake.
3. Expand “Tag You’re It” marking of Largemouth Bass and Brown Bullhead to better estimate use and harvest.
4. Do not stock tiger muskie until a cost: benefit analysis and public opinion survey has been completed regarding tiger muskie fishing opportunity at Mud Lake.
5. Identify factors affecting Yellow Perch recruitment and survival in Mud Lake.

Table 27. Species, number captured (*n*), catch per unit effort (CPUE) and weight per unit effort (WPUE) of fish sampled during the Mud Lake lowland lake survey in 2021.

Species	*n*	CPUE	WPUE
Bluegill	11	6.0	1.3
Brown Bullhead	59	25.8	8.2
Largemouth Bass	74	33.2	29.8
Utah Chub	671	208.8	703.7
Utah Sucker	5	3.7	2.6
Yellow Perch	30	12.0	1.4
Total	850	289.4	747.0

Table 28. Mean annual growth intervals for each age class of Largemouth Bass estimated using back-calculated lengths for bass captured during the Mud Lake lowland lake survey in May 2021.

Age	Year Class	*n*	1	2	3	4	5	6	7	8	9	10	11
1	2020	8	207										
2	2019	9	172	114									
3	2018	30	186	106	239								
4	2017	12	172	120	231	158							
5	2016	3	165	96	228	140	260						
6	2015	3	180	88	255	133	288	158					
7	2014	3	176	87	246	127	276	152	293				
8	2013	3	154	109	225	146	264	166	284	179			
9	2012	1	160	103	223	146	241	166	251	182	259		
10	2011	0	0	0	0	0	0	0	0	0	0		
11	2010	1	145	82	202	128	221	148	240	163	255	189	272
	Weighted means		181	107	236	147	266	158	278	176	257	189	272

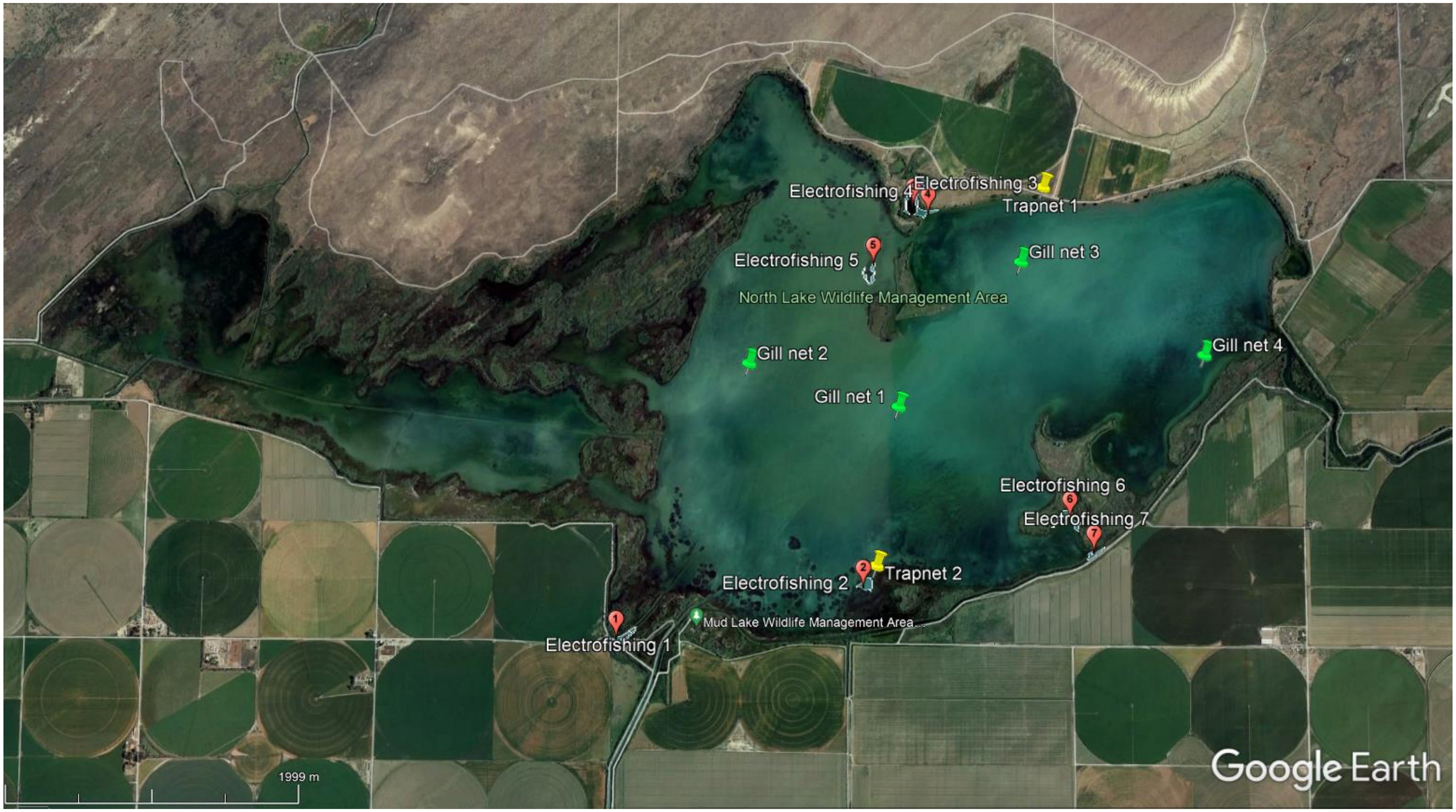

Figure 101. Lowland lake survey sampling locations for electrofishing, trap-netting, and gillnetting efforts on Mud Lake in May 2021.

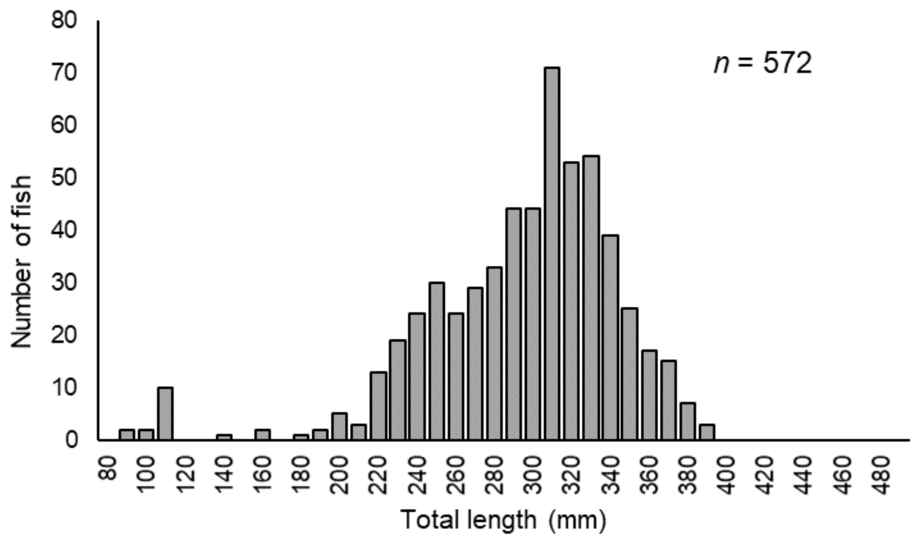

Figure 102. Length-frequency histogram for Utah Chub captured using all gear types during the Mud Lake lowland lake survey in May 2021.

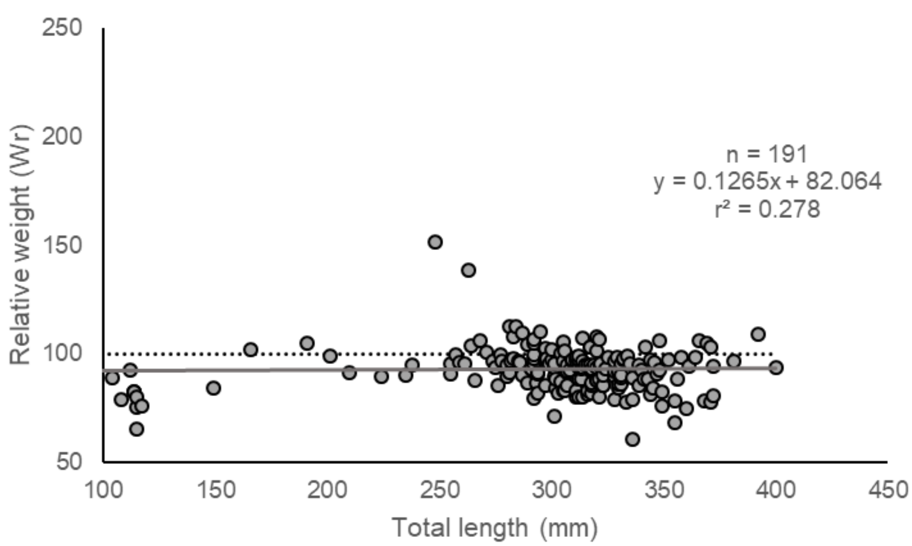

Figure 103. Relative weights of Utah Chub captured during the May 2021 Mud Lake lowland lake survey using all gear types. The dashed line is the species standard (100) and the solid line is the best fit linear regression.

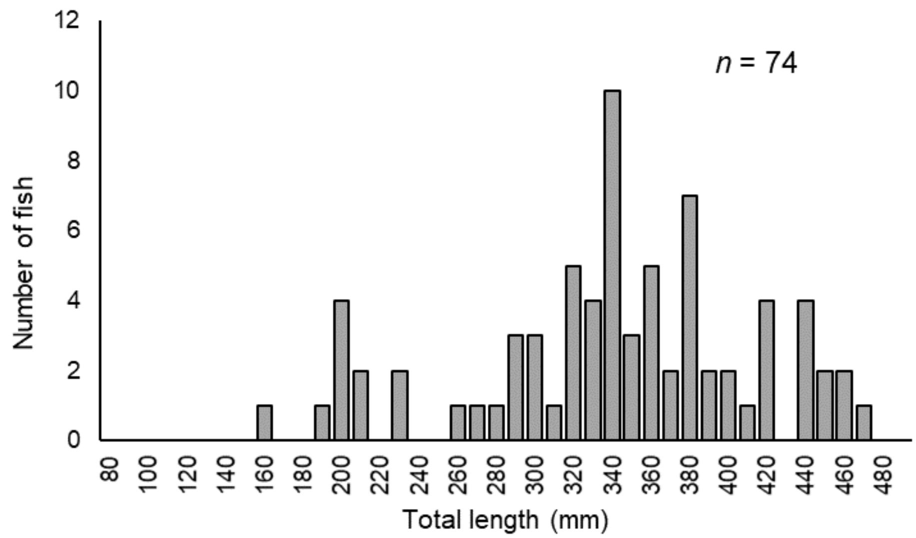

Figure 104. Length-frequency histogram of Largemouth Bass captured using electrofishing during the Mud Lake lowland lake survey in May 2021.

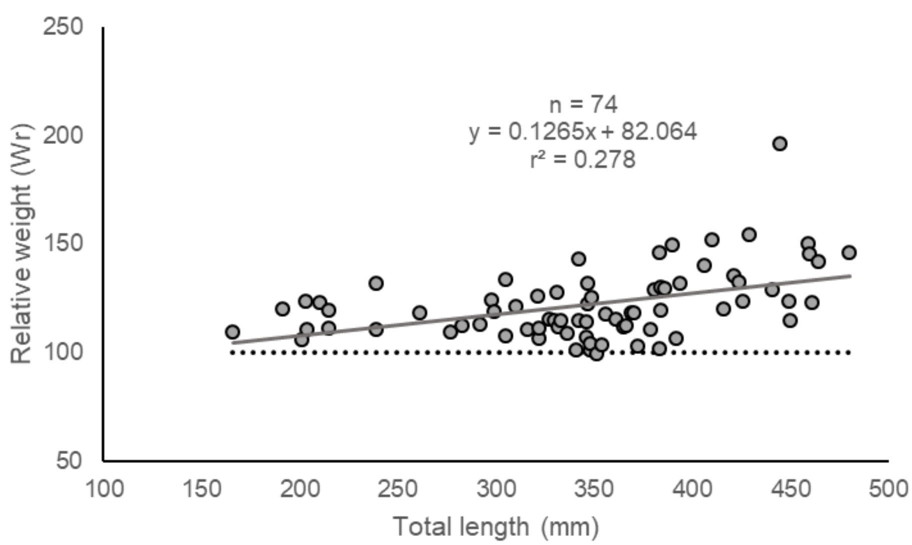

Figure 105. Relative weights of Largemouth Bass captured during the May 2021 Mud Lake lowland lake survey. The dashed line is the species standard (100) and the solid line is the best fit linear regression.

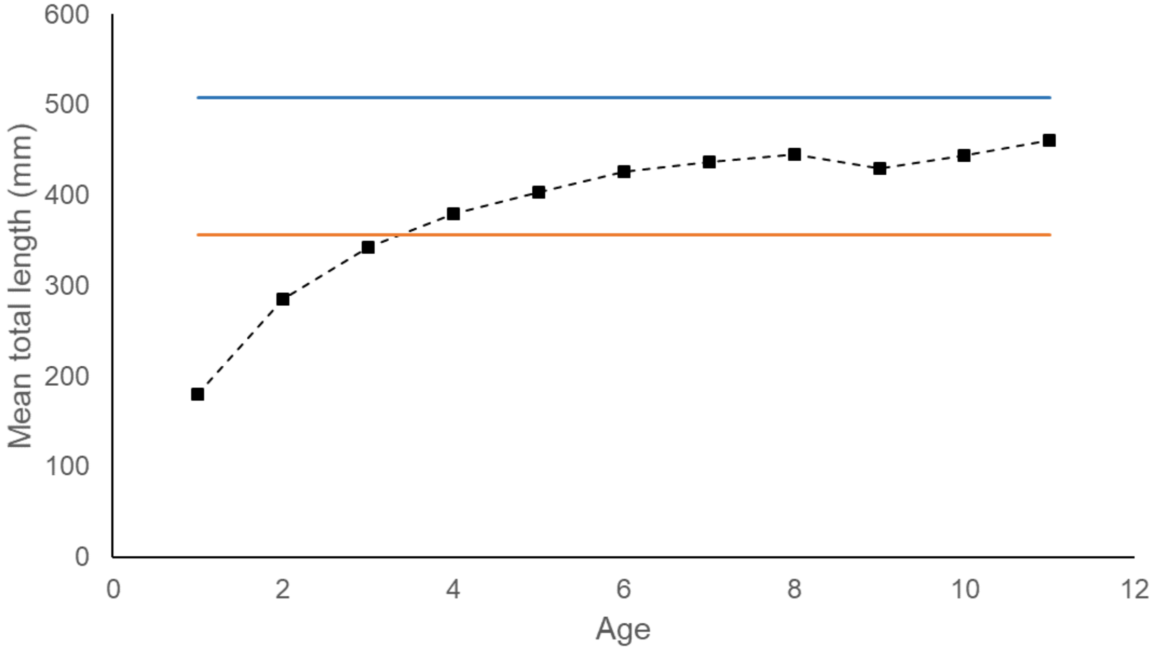

Figure 106. Weighted mean lengths at age for Largemouth Bass captured during the May 2021 Mud Lake lowland lake survey. The orange line represents the size for quality Largemouth Bass and the blue line represents the size for trophy Largemouth Bass.

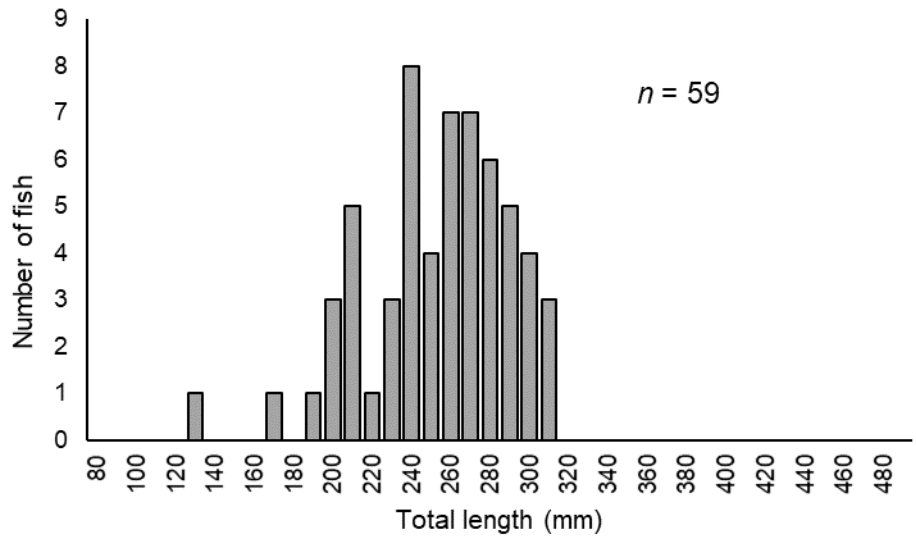

Figure 107. Length-frequency histogram of Brown Bullhead captured using all gear types during the Mud Lake lowland lake survey in May 2021.

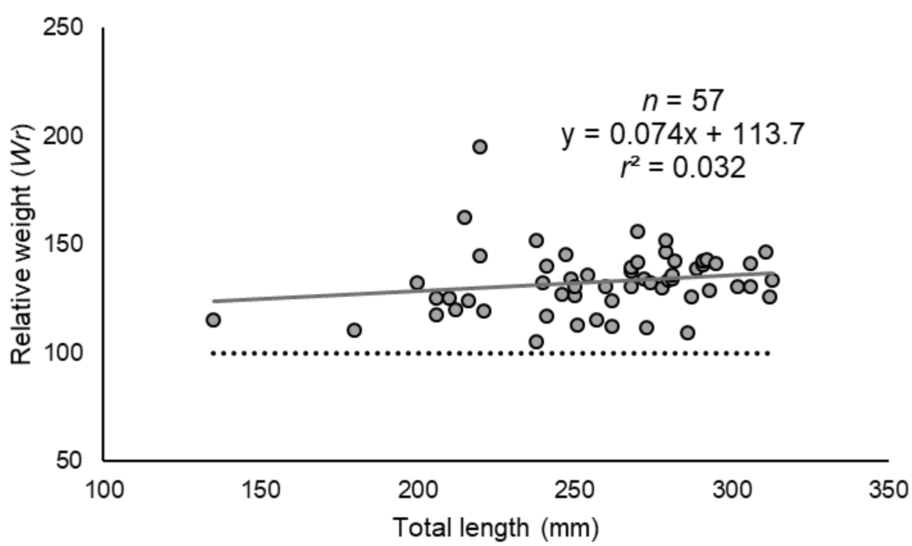

Figure 108. Relative weights of Brown Bullhead captured during the May 2021 Mud Lake lowland lake survey using all gear types. The dashed line is the species standard (100) and the solid line is the best fit linear regression.

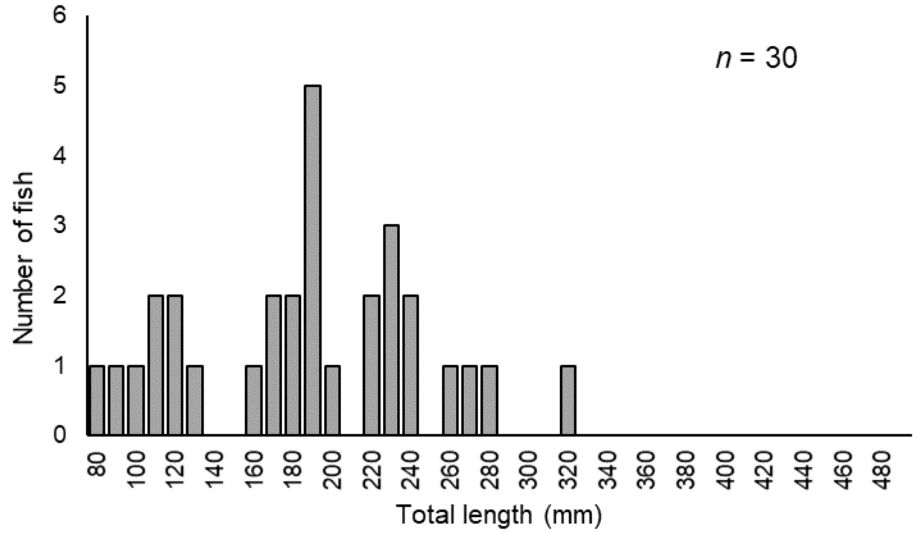

Figure 109. Length-frequency histogram of Yellow Perch captured using all gear types during the Mud Lake lowland lake survey in May 2021.

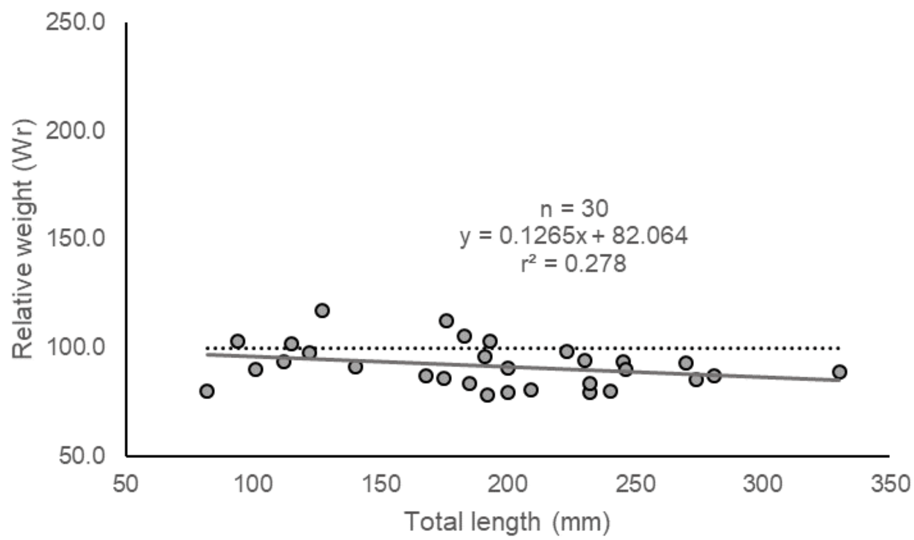

Figure 110. Relative weights of Yellow Perch captured during the May 2021 Mud Lake lowland lake survey using all gear types. The dashed line is the species standard (100) and the solid line is the best fit linear regression.

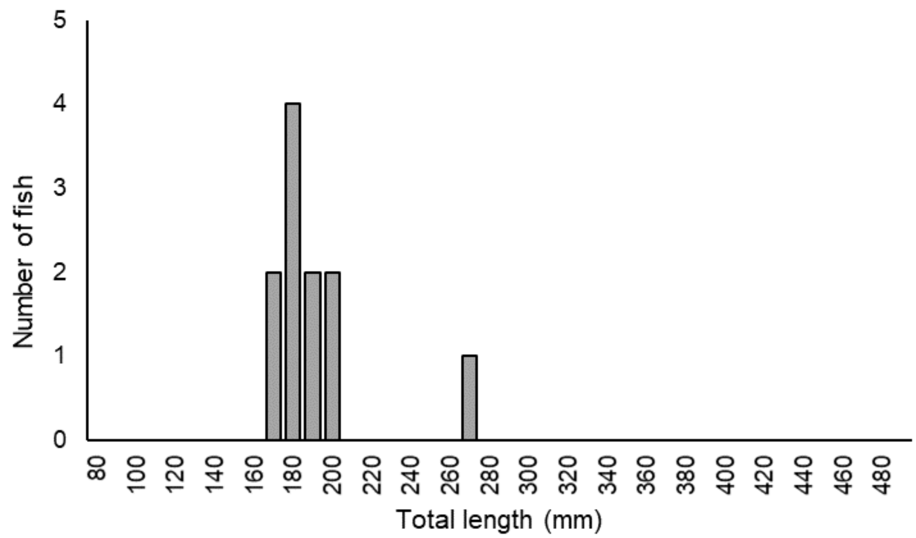

Figure 111. Length-frequency histogram of Bluegill captured using electrofishing during the Mud Lake lowland lake survey in May 2021.

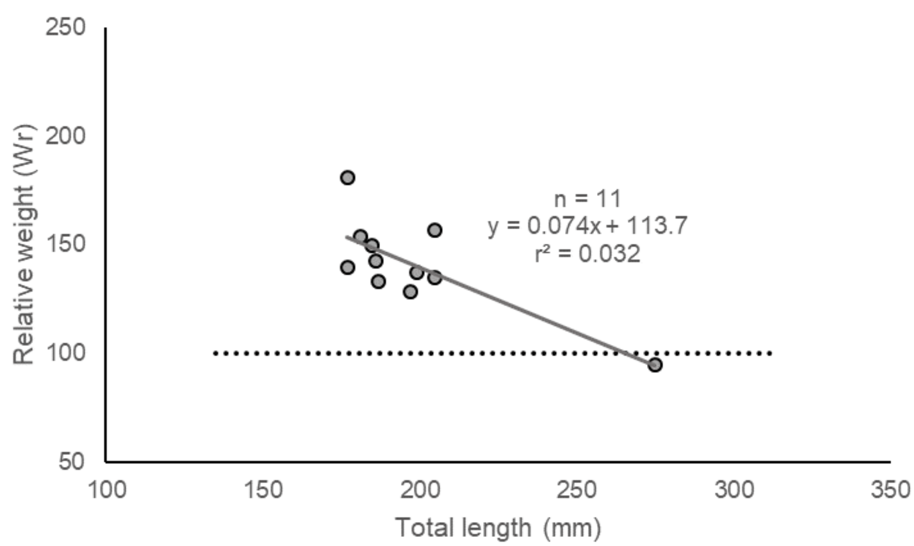

Figure 112. Relative weights of Bluegill captured during the May 2021 Mud Lake lowland lake survey using electrofishing. The dashed line is the species standard (100) and the solid line is the best fit linear regression.

ISLAND PARK RESERVOIR

ABSTRACT

Island Park Reservoir (IPR) is an irrigation storage reservoir located on the Henrys Fork Snake River which provides recreational opportunities to pleasure craft operators and anglers alike. To assess the current state of the entire IPR fishery, we conducted a lowland lake survey utilizing 12 gill-net nights of effort and 7 shoreline electrofishing transects. Non-game fish species, including Utah Chub *Gila atraria* and Utah Sucker *Catostomus ardens* dominated the catch of both gear types. Electrofishing was the most effective gear type used with a mean catch-per-unit-effort (CPUE) of 720 fish/h, followed by gill nets with a mean CPUE of 70.3 fish/net-night. Rainbow trout *Oncorhynchus mykiss* CPUE was 3.7 fish with relative weights averaging 86. To assess the kokanee *O. nerka* fishery we utilized suspended gill nets. Kokanee CPUE was low compared to other waters in the region at 16.8 fish/net-night and kokanee had a mean relative weight at 87. Of the kokanee otoliths examined for thermal marks (n = 75), only 4% (n = 3) contained thermal marks indicative of hatchery-origin fish and the remaining 96% were presumably wild (i.e. no thermal marks).

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INTRODUCTION

Island Park Reservoir (IPR) has been recognized as a quality recreational fishery since the early 1950s, supporting as much as 176,000 h of angling effort annually, with angler catch rates averaging 0.68 fish per hour. Rainbow Trout *Oncorhynchus mykiss* have provided the majority of angler catch, with kokanee *O. nerka*, Brook Trout *Salvelinus fontinalis*, Mountain Whitefish *Prosopium williamsoni,* and Yellowstone Cutthroat Trout *O. clarkii bouvieri* adding to the creel. Supplemental stocking of hatchery fish has played a large role in the management of the fishery, which is supported by hatchery releases of Rainbow Trout (RBT) and kokanee, although spawning by both species occurs in the Henrys Fork Snake River (HFSR) upstream of the reservoir. Annual RBT fingerling stockings have averaged 448,633 fish over the past 84 years with the highest of 2.5 million RBT in 1959. Beginning in 2010, the Idaho Department of Fish and Game (IDFG) increased the size of fingerling RBT (> 6 inches) stocked in IPR to reduce the potential for entrainment through the dam. The number of fingerlings released were reduced to approximately 150,000 fish which equated to the same biomass as the nearly 500,000 smaller fingerlings stocked in earlier years. Nearly 120,000 kokanee were stocked into IPR in 1944 and 1945, followed by 144,000 stocked into Moose Creek in 1957. These initial stockings resulted in a self-sustaining population of kokanee, which spawned in Moose Creek. The Idaho Department of Fish and Game established a kokanee trapping facility on Moose Creek to collect eggs to support stocking in other waters. The Moose Creek kokanee trap was operated intermittently between 1963 and 1975, with over 5 million eggs collected in 1969. Between 1976 and 1979, IPR was drawn down to near record levels on two occasions, and treated with rotenone during the 1979 draw down to remove nongame fish species. Annual kokanee fry stocking of nearly 500,000 fish in 1981, 1982, and 1984 re-established the spawning run, and trapping at Moose Creek resumed in 1987, though most fish were passed over the trap and allowed to spawn naturally. The trap was operated again in 1990 and 1991, but low numbers of fish were captured. Drought conditions and low populations prohibited trap operations from 1992 to 1994. In 1995, over 200,000 eggs were again collected at the Moose Creek trap, but future trap operations were discontinued due to low returns, combined with the identification of other, more easily obtained egg sources (e.g., Deadwood Reservoir) in the state. The trap was installed once again in 2003, but too few fish were captured to provide the necessary egg collection, so all fish were passed over the trap and allowed to spawn naturally. Since 1997, all hatchery kokanee eggs stored at Cabinet Gorge Fish Hatchery and then stocked into IPR have been marked with thermal mass-marking techniques (Volk et al. 1990). Hatchery-origin kokanee display distinct thermal marks which are used to determine brood year.

Historically, the proliferation of nongame fish, primarily Utah Chub *Gila atraria* and Utah Sucker *Catostomus ardens*, were blamed for declines in the sport fishery in IPR. Several rotenone projects had been undertaken to reduce overall nongame fish abundance and improve angler catch rates. The efficacy of these treatments was questioned as early as 1982, when Ball et al. (1982) observed that the three chemical rehabilitations of IPR over the previous 25 years had not been successful at permanent or long-term eradication of nongame species. Furthermore, improvements in the trout fishery appeared to be the result of increased stocking rates, especially noticeable with the mean annual introduction of 72,491 catchable RBT between 1980 and 1985. Ball et al. (1982) further noted that the observed declines in the RBT fishery two to four years after treatment were the result of decreased levels of hatchery inputs and were not due to increased Utah Chub (UTC) and Utah Sucker (UTS) densities. The most recent chemical treatment of the reservoir, conducted in 1992, yielded similar results, with post-treatment catch rates not improving from catch rates prior to the treatment (Gamblin et al. 2002). More recently, Garren et al. (2008) found that nongame fish exceed pre-rotenone treatment levels within five years following treatments and that angler catch rates within those five years following were not significantly different than angler catch rates prior to treatments, suggesting that rotenone treatments had no effect on improving angler catch rate in IPR.

Island Park Reservoir is operated as an irrigation storage reservoir with a total capacity of 135,205 acre-feet. This water is stored for agricultural users downstream, and is, therefore, subject to fluctuations in annual water levels. Increases in reservoir storage normally begins at the close of the irrigation season in October and lasts until demand for water increases, typically in late May or early June. Fall reservoir storage levels have fluctuated from the lowest storage level recorded of 270 acre-feet in 1992 (0.19%) to over 100% full (143,470 acre-feet), as seen in 1971 and averaged 68% from 1939 through 2021. In years with low reservoir storage, sport fish densities in gillnets decrease the following year (Garren et al. 2008). Although the relationship between carryover and gill-net catch rates has been identified, it is unclear what mechanism is affecting salmonid populations. Possible mechanisms may be increased mortality due to entrainment through the dam due to increased outflow, lost habitat and poor water quality associated with drawdowns, and/or reduction in zooplankton forage base. A study focusing on factors regulating kokanee populations in a northern Idaho reservoir found kokanee population losses as high as 90% due to entrainment as kokanee distributed throughout the reservoir (Maiolie and Elam 1998). Congregations of all age-classes of kokanee were found in the reservoir near the IPR dam, making them susceptible to entrainment due to high volumes of water being released. Consistent with the observed decline in kokanee populations, Island Park Dam was modified in 1994 with a new intake structure to facilitate power generation as part of the Island Park Hydroelectric Project (Ecosystems Research Institute 1994), thereby altering the location of water withdrawals from the reservoir. The hydroelectric facility is capable of handling up to 960 cubic feet per second (cfs). Throughout most of the year, the entire outflow is routed through the hydroelectric facility intake. To prevent entrainment, the hydroelectric intake structure features wedge wire screens with 9.5-mm openings. National Marine Fisheries Service (NMFS) screening criteria requires screen mesh with openings no larger than 2.4 mm to prevent passage of juvenile salmonids (NMFS 2011). Although this criteria is designed for anadromous fishes, it is the only reviewed criteria for juvenile salmonids and has been implemented in non-anadromous waters for screening juvenile salmonids. Surveys of the HFSR immediately below Island Park Dam have documented kokanee, indicating that some size classes are able to pass though the screened intake.

In response to low kokanee angler catch rates, and to lessen the potential impacts of entrainment and possibly establish self-sustaining spawning runs, IDFG altered its stocking practices in 2009. Historically, juvenile kokanee were stocked directly into IPR between May and June when inflow and outflow from the reservoir are increasing. This may contribute to the potential for entrainment as kokanee may actively follow river currents while migrating downstream (Fraley and Clancey 1988). Beginning in 2009, IDFG released half (approximately 125,000) of the annual kokanee stocking directly into IPR, with the remaining releases split between Big Springs Creek and Moose Creek (Figure 1). In-reservoir stockings occur throughout the reservoir, although most commonly at the Buttermilk Campground in early June.

Warm dry summers tend to increase irrigation demand downstream resulting in IPR drawdowns. These smaller volumes of remaining water in IPR are more likely to be influenced by solar input. As the dam outflow is located at the bottom of the reservoir, cold oxygenated water from the hypolimnion is removed from the reservoir downstream. This leads to a loss of the hypolimnion, reducing thermal stratification in IPR and increased mixing warmer water in the top layers of the water column (McLaran et al. 2019). Cold-water dependent species like Rainbow Trout and kokanee in IPR are forced to compete for thermal refuge space. McLaren et al. (2021) found the number of kokanee out-migrating from IPR to spawn was negatively correlated with IPR reservoir volumes in the previous two years. Kokanee spend their first two years entirely within IPR before out-migrating at age-3 to spawn and die. If excessive drawdowns occur during the first two years that a cohort spends in the reservoir, juvenile fish have limited opportunities to find cold water refugia which could reduce survival and recruitment (McLaren et al. 2021). Stable and high reservoir volumes will increase the volume of cold, well oxygenated water increasing juvenile habitat and may help to bolster the kokanee population in IPR.

STUDY AREA

Island Park Reservoir is located on the HFSR 40 km north of Ashton, Idaho and 150 km upstream from the confluence with the South Fork Snake River (Figure 1). Island Park Dam is a 23 m high earth-fill rock-faced structure operated by the United States Bureau of Reclamation to provide water for irrigation in Fremont and Madison Counties. The drainage area upstream from the dam is 774 km^2, varying in elevation from 1,920 to 3,017 m. At gross pool capacity (143,430-acre feet), the reservoir covers 3,388 hectares and has a shoreline of about 97 km. The sources of water input are a mix of snowmelt runoff streams and spring influenced streams.

OBJECTIVES

To obtain current information on the entire fish community and to develop appropriate management recommendations to achieve the management objective to produce and maintain a quality consumptive salmonid fishery in IPR as stated in the State Fish Management Plan (IDFG 2019).

METHODS

Lake Survey

A lake survey consisting of six gill-net pairs (sinking and floating) and boat electrofishing at seven near shore electrofishing transects was conducted in IPR to assess the entire fish community. We set paired gill nets (floating and sinking) at six locations for a total of 12 gill-net nights in IPR (Figure 113). All experimental gill nets measured 49-m long by 6-m deep with 16, 3-m long panels, which were randomly positioned in the net. The monofilament bar mesh panels measured 13, 19, 25, 38, 52, 64, 76, and 102 mm with each mesh representing two panels of the net. We set nets at dusk and retrieved them the following morning. Sites were randomly selected by overlaying a grid system (100 m × 100 m) using mapping software. We identified captured fish to species and recorded TL (mm), weights (g), and sex. Species-specific gill-net catch per unit effort (CPUE) was calculated as the mean number of fish captured per gill-net night divided by the total number of nets with 95% confidence intervals based on variance of the catch in separate nets. Gill-net catch in terms of weight per unit effort (WPUE) was calculated as the mean number of kg of fish captured per gill-net night with 95% confidence intervals.

We utilized ArcMap GIS software (Ersi® ArcMap™) to divide up the entire IPR shoreline into 500 m transects. A total of seven transects were then randomly selected for sampling in 2021. We measured water temperature (°C) and conductivity (µS/cm) using a handheld probe prior to electrofishing. Electrofishing began at dusk beginning at transects closest to the west end of IPR working back towards the dam. Each transect was shocked for 600 s (IDFG 2012) using one jet boat outfitted with electrofishing equipment using pulsed direct current power provided by a 5000-

W generator and standardized to 2,750-3,250 based on water conductivity (Miranda 2009). We applied electricity to the water using an Infinity model electrofisher (Midwest Lake Management Ins., Missouri). Two netters were positioned at the bow of the boat using 2.4-m long dip nets with 6-mm bar knotless mesh. All captured fish were identified to species and measured for TL. We calculated CPUE for each species as the number of fish captured per hour.

Kokanee Abundance Gillnetting

A gill-net survey was conducted to evaluate the kokanee population of IPR. A total of four experimental gill nets were suspended at the thermocline of the reservoir (Appendix A). We used a water quality meter (YSI Inc., Yellow Springs, Ohio) to take water temperature at the surface and every subsequent meter down the water column until the thermocline was identified by a several degree water temperature difference from the previous depth. Gill net specification were identical to our experimental gill nets used in the lake survey as described above. Sites were randomly selected by overlaying a grid system (100 m × 100 m) using mapping software. We identified captured fish to species and recorded TL (mm), weights (g), and sex. Gill-net catch per unit effort (CPUE) was calculated as the mean number of fish per gill-net night divided by the total number of nets with 95% confidence intervals.

We pooled relative weights from all gillnetting (sinking, floating, and suspended) combined for RBT and kokanee. Relative weights (W_r) were calculated by dividing the actual weight of each fish (g) by a standard weight (W_s) for the same length for that species and multiplied by 100 (Anderson and Neumann 1996). Relative weights were averaged for each of the following length classes: < 200 mm, 200 - 299 mm, 300 - 399 mm and fish > 399 mm. We used the formula, log W_s = -4.898 + 2.990 log TL for RBT (Simpkins and Hubert 1996) and, log W_s = -5.062 + 3.033 log TL for kokanee (Milewski and Brown 1994). Mean *Wr* for all sizes classes were compared using linear regression individually for both RBT and kokanee. Linear regression was also used to compare mean annual *Wr* from 2012 to 2021 for RBT and 2014 to 2021 for kokanee using 95% confidence intervals where non-overlapping confidences intervals indicate significant differences.

We calculated proportional size distribution (PSD) and relative stock density (RSD) to describe the size structure of kokanee and RBT in IPR. Kokanee PSD was calculated as the number of fish greater than or equal to 250 mm divided by the number of fish greater than or equal to 120 mm, multiplied by 100. Kokanee RSD was calculated as the number of fish greater than or equal to 300 mm divided by the number of fish greater than or equal to 120 mm, multiplied by 100. Rainbow Trout RSD was calculated as the number of fish greater than or equal to 300 mm divided by the number of fish greater than or equal to 200 mm, multiplied by 100.

To describe age composition, length at maturity, hatchery origin, growth, and survival, we collected sagittal otoliths from all captured kokanee. After removal, all otoliths were cleaned and stored in individually-labeled vials to be used to estimate age. Otoliths were sanded and examined with a compound microscope at 200 and 400 power magnification. Thermal marks were compared with reference samples taken from the hatchery during fry rearing to make brood year assignments.

RESULTS

Lake Survey

We captured a total of 1,632 fish during our lake survey at IPR in 2021. This included UTC (*n* = 779), UTS (*n* = 555), Redside Shiner *Richardsonius balteatus* (*n* = 178), RBT (*n* = 60), Longnose Dace *Rhinichthys cataractae* (*n* = 42), kokanee (*n* = 11), Brook Trout (*n* = 5), and Mountain Whitefish (*n* = 2; Table 29; Figure 114). Electrofishing was the most effective gear type with a mean CPUE of 720 fish/hour, followed by gillnetting which had a mean CPUE of 70.3 fish/net-night and a WPUE of 10.3 kg/net night. Based on CPUE for both gear types, catch was predominately UTC (48%) and UTS (34%). Other species in the catch included Redside Shiner (10.9%), RBT (3.7%), Longnose Dace (2.6%), kokanee (0.7%), Brook Trout (0.3%), and Mountain Whitefish (0.1%; Table 29).

Kokanee Abundance Gillnetting

We conducted our kokanee abundance gill-net survey on June 15 through 17, 2021. We collected a total of 197 fish in the suspended gill nets which included 67 kokanee. Mean CPUE was highest for UTS (18.5 ± 16.5 95% CI), followed by kokanee (16.8 ± 37.8), UTC (13 ±14.5), and RBT (1 ± 3.2; Figure 115). Kokanee CPUE slightly decreased from 2019 (21.9 kokanee per net-night) and was below the five-year average CPUE of 15.8. Similarly, mean WPUE was highest for UTS (1.5; ± 1.5 95% CI), followed by UTC (2.4 ± 1.1), KOK (2.1 ± 0.9), and RBT (0.2 ± 0.2). Catch was uneven among nets with one net contributing 78% of the kokanee catch (Figure 116). Kokanee lengths ranged from 95 to 459 mm, with a mean of 218 mm (± 15.7; Table 30). A distinct size class from 170 to 209 mm comprised 70% of the catch (Figure 117). The PSD for kokanee was 35 with an RSD of 8 (Table 30). Mean W_r of kokanee across all size classes was 87 (± 1.4; Table 30) and did not increase as total length increased (r^2 = 0.08; Figure 118).

Of the 75 kokanee otoliths examined, 4% (*n* = 3) exhibited thermal marks and were identified as hatchery-origin kokanee from spawn year 2019. Wild-origin kokanee represented the remaining 96% (i.e. no thermal marks present). We were unable to fit a logistic regression model to size (TL) at sexual maturity due the absence of larger (> 300 mm) sexually mature kokanee in the nets.

Of the RBT captured across both gear types, RBT had a mean TL of 369 mm (± 18.3; Table 30) which ranged from 79 to 550 mm (Figure 119). Mean W_r for all size classes combined was 76 ± 3.1 (Table 30). Mean W_r of RBT decreased slightly as TL increased (r^2 = 0.05; Figure 120). Rainbow Trout mean W_r was the lowest observed since 2012, although the declining trend was not significant (r = 0.37, F = 2.4, df = 5, P = 0.2). Proportional stock density was 83 and RSD was 41 for RBT.

DISCUSSION

We used a combination of fish sampling gears in Island Park Reservoir to reduce sampling bias and provide metrics for the entire fish assemblage through a multi-gear approach with broad spatial distribution. Gill nets were the most efficient gear to capture salmonids and have been utilized to monitor game fish species in IPR for over three decades. Although, nearshore community electrofishing has not been conducted previously in conjunction with gill-net surveys on IPR. Our electrofishing survey captured non-game fish Redside Shiner, and Longnose Dace which were not recruited to standardized gill nets. In addition, this survey captured smaller size classes of RBT, UTC, and UTS indicating this survey is an important tool to use in conjunction with traditional gill-net surveys to better describe the species assemblage of IPR.

The IPR fish species assemblage continues to be dominated by non-game species (UTC and UTS) with no change in catch composition since the last gill-net survey conducted in 2016 (Heckel et al. 2019; Flinders et al. 2016). Rainbow Trout total abundance has also not changed since the 2016 gill-net survey and although not significant, RBT W_r were the lowest observed over the dataset (2012-2021) indicating a potential declining trend in RBT body condition. While some evidence exists to suggest competition between RBT and UTC may play a role, for example isotopic niche overlap has been found between RBT and UTC in Schofield Reservoir in Utah (Winters and Budy 2015). Competition is likely not the mechanism affecting RBT in IPR as UTC CPUE have not changed. Rather, water quality impacts from drought conditions are likely the reason why recent W_r values for RBT are lower. Water quality would also affect prey resources in IPR. Zooplankton sampling in IPR was last conducted in 2014 and found high availability of larger Cladocera preferred by RBT and kokanee during the summer months with decreasing numbers in the fall and winter following irrigation drawdown (Flinders et al. 2016). As such there may be a potential foraging bottleneck during the fall and winter as kokanee, RBT, and UTC compete for zooplankton resources and this effect may be exacerbated with drought conditions. As prey resources become limited this may result in the decreased body conditions of RBT and kokanee observed in this study. We suggest managers to evaluate the current seasonal zooplankton abundance and species composition in IPR.

Kokanee are an important gamefish species in IPR. Our standardized sampling methods of gill nets and electrofishing did not adequately characterize the kokanee fishery because these methods captured low numbers of kokanee with similar size lengths. Kokanee are an obligate planktivore which tend to prefer the thermocline. As such, traditional floating and sinking gill nets which sample the epilimnion and hypolimnion, respectively, may not be the most appropriate gear type to sample kokanee. Suspended gill nets at the thermocline have proven to effectively capture kokanee and is the current method utilized to sample kokanee in IPR (Heckel et al. 2019; Flinders et al. 2016; IDFG unpublished data). During the 2021 survey we reduced the number of kokanee nets set from past surveys for logistical reasons from 9 nets (2019) to 4 nets. It was difficult to compare kokanee abundance between years as one net out of four captured 78% of the kokanee catch. To mitigate this disparity in catch between nets, managers may want to conduct a power analysis to identify the number of suspended kokanee nets required to detect a 20% change in the kokanee population.

The current kokanee population is dominated by wild-origin kokanee. Although the population has been largely wild-origin kokanee in previous surveys (2018: 88% and 2016: 90%) this is the highest estimated to date. This indicates wild recruitment is successful but produces low numbers of kokanee. These results also indicate how stocking has had limited impact on kokanee numbers. From 2014 through 2016, the years that stocking would have resulted in kokanee large enough to be captured during gill-net surveys in 2016 and 2018, stocking rates ranged from 248,704 to 258,500 fingerlings. Kokanee gill-net catch rates have decreased in IPR, and these gill-net catch rates are lower than those observed in other kokanee fisheries in the region (e.g., Mackay Reservoir and Ririe Reservoir; Heckel et al. 2019). Recent studies on water quality have indicated suitable habitat for kokanee in Island Park Reservoir is extremely limited from mid-June through August due to a combination of high water temperatures in the upper water column, and low dissolved oxygen levels in the hypolimnion (McLaren Dissertation – citation available early 2023).

One objective in the Fisheries Management Plan (IDFG 2019) is to identify limiting factors on kokanee in IPR. A variety of efforts have been implemented over the years with the aim of bolstering the wild kokanee population in IPR. Eyed kokanee eggs were planted in artificial redds from 2013 to 2015 in Moose and Lucky Dog creeks, two tributaries to the HFSR upstream of IPR with historical kokanee spawning grounds. This project did not bolster the kokanee population or increase spawning in these tributaries. High zooplankton densities have been found in IPR to support high densities of hatchery fish in 2014 (Flinders et al. 2016), although we are not aware of any current macroinvertebrate surveys conducted to assess additional prey abundance. An updated survey on the current zooplankton and macroinvertebrate abundance in IPR may help evaluate the current prey density available.

Drawdowns of reservoir levels in IPR may have negative impacts to kokanee survival. The level of drawdown is negatively associated with suitable habitat available to kokanee in IPR (McLaren dissertation). Thus, after a poor water year in 2016 when water demand caused higher-than-normal drawdown in IPR, it is not surprising that kokanee gill-net catch rates were significantly lower in 2017. Gill-net catch rates then rebounded in 2018 after the 2017 fall reservoir pool retained over 82,000 acre-feet more water than 2016. This high reservoir winter carryover is a likely factor to the increased overwinter survival of kokanee. This is further supported by long-term dataset analysis conducted by McLaren et al. (2021) who found the number of kokanee out-migrating from IPR to spawn was negatively correlated with IPR reservoir volumes for the two years prior. As kokanee spend the entirety of their first two years in IPR before out-migrating to spawn at age-3, increased reservoir drawdowns are likely destroying juvenile habitat and reducing thermal refuge for fish, thus causing crowding at refuge locations, increasing predation, and entrainment of juvenile fish. Retaining water in reservoirs across the region is important for fish health and survival in future years.

Current information quantifying angler harvest and angler catch rates of this fishery are not currently available. Our most recent 2013 creel survey indicated higher angler catch rates than any prior survey since 1980, but we do not know if this trend has continued. As such, a creel survey should be conducted to fully assess the current state of the fishery and help direct future management actions.

MANAGEMENT RECOMMENDATIONS

1. Continue using a multi-gear approach which includes standardized gill nets, suspended gill nets, and electrofishing to monitor the entire IPR fishery.

2. Conduct a power analysis to identify the number of suspended kokanee nets required to detect a 20% change in the kokanee population.

3. Monitor trends in kokanee and Rainbow Trout prey resources through zooplankton and macroinvertebrate surveys to assess the current hatchery stocking numbers.

4. Monitor water quality parameters and reservoir levels in IPR and how these metrics relate to kokanee survival in the IPR.

Table 29. Number of fish captured by species and gear type in Island Park Reservoir, 2021.

	Brook Trout	kokanee	Longnose Dace	Mountain Whitefish	Rainbow Trout	Redside Shiner	Utah Chub	Utah Sucker
				Lake survey				
Experimental gill nets	5	11	0	2	56	0	359	410
Electrofishing	0	0	42	0	4	178	420	145
% of catch	0.3	0.7	2.6	0.1	3.7	10.9	48.0	34.0
				Kokanee abundance gill nets				
Suspended gill nets	0	67	0	0	4	0	52	74
Total	5	78	42	2	64	178	556	904

Table 30. Stock density indices (PSD = proportional stock density; RSD = relative stock density) and relative weights (W_r) for Rainbow Trout collected using standard gill nets, suspended gill nets, and electrofishing and kokanee collected using suspended gill nets in Island Park Reservoir, June 2021. Sample size (n) for relative weight values is noted in parentheses.

	Rainbow Trout (*n*)	Kokanee (*n*)
Mean length (mm)	369	218
Min length (mm)	79	95
Max length (mm)	550	459
PSD	83	35
RSD-400	41	8
W_r		
< 250 mm	--	86 (54)
250 – 349 mm	82 (17)	94 (8)
350 – 449 mm	72 (39)	81 (2)
450 – 549 mm	80 (3)	88 (2)
550 – 649 mm	95 (1)	--
Mean W_r	76	87

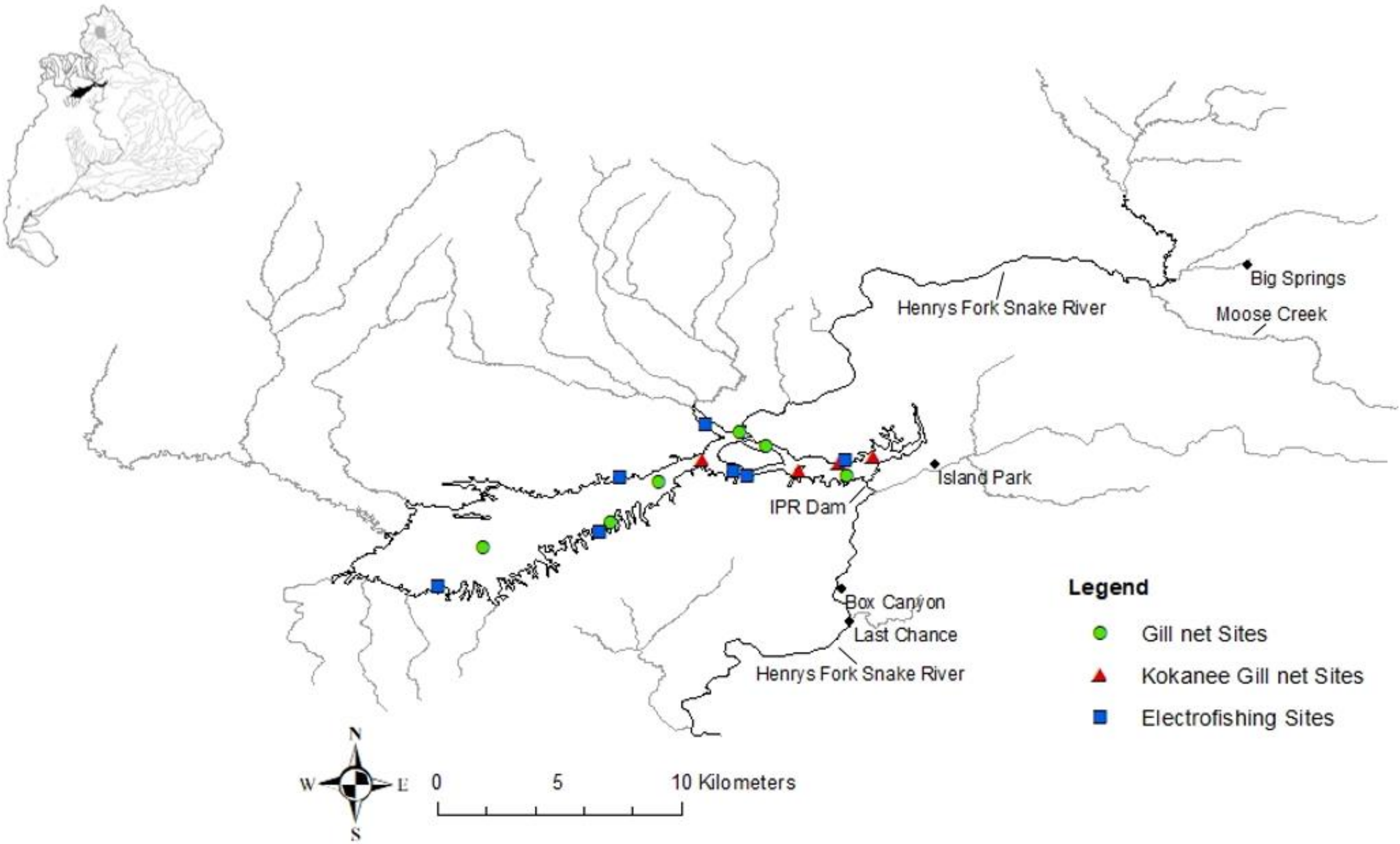

Figure 113. Location of experimental gill net sites (green circles), targeted kokanee suspended gill net sites (red triangles), electrofishing sites (blue squares), and major tributaries of Island Park Reservoir, June 2021.

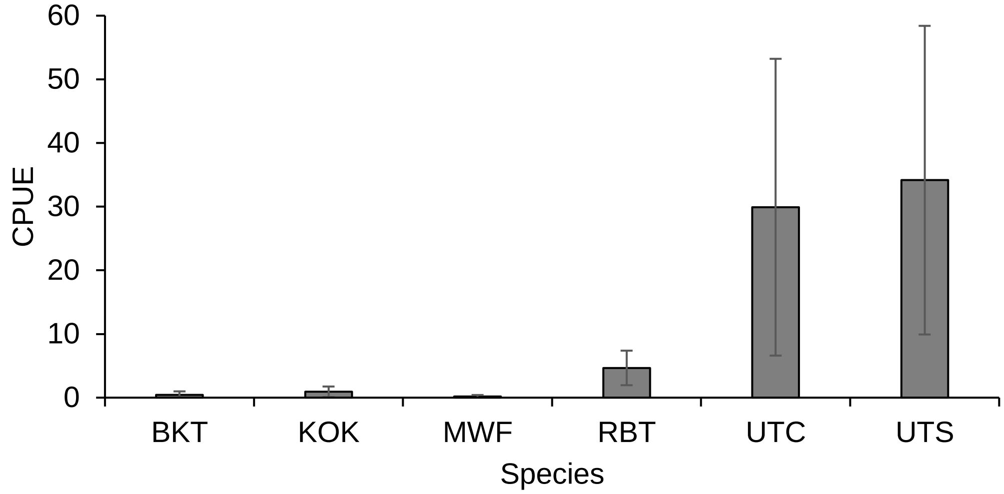

Figure 114. Catch-per-unit-effort (CPUE; number of fish per net-night) and 95% confidence intervals for Brook Trout (BKT), kokanee (KOK), Mountain Whitefish (MWF), Rainbow Trout (RBT), Utah Chub (UTC), and Utah Sucker (UTS) collected using gill nets in Island Park Reservoir, June 2021.

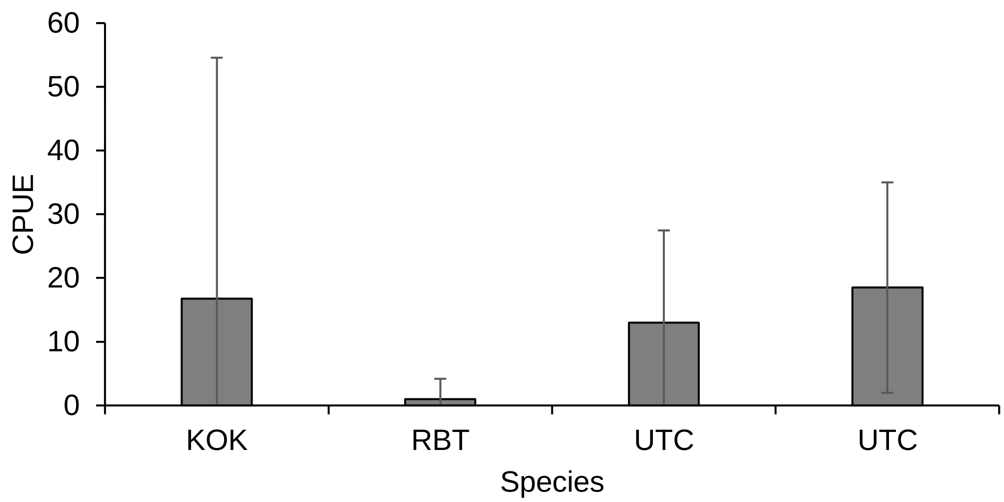

Figure 115. Catch-per-unit-effort (CPUE; number of fish per net-night) and 95% confidence intervals for Brook Trout (BKT), kokanee (KOK), Mountain Whitefish (MWF), Rainbow Trout (RBT), Utah Chub (UTC), and Utah Sucker (UTS) collected using suspended gill nets in Island Park Reservoir, June 2021.

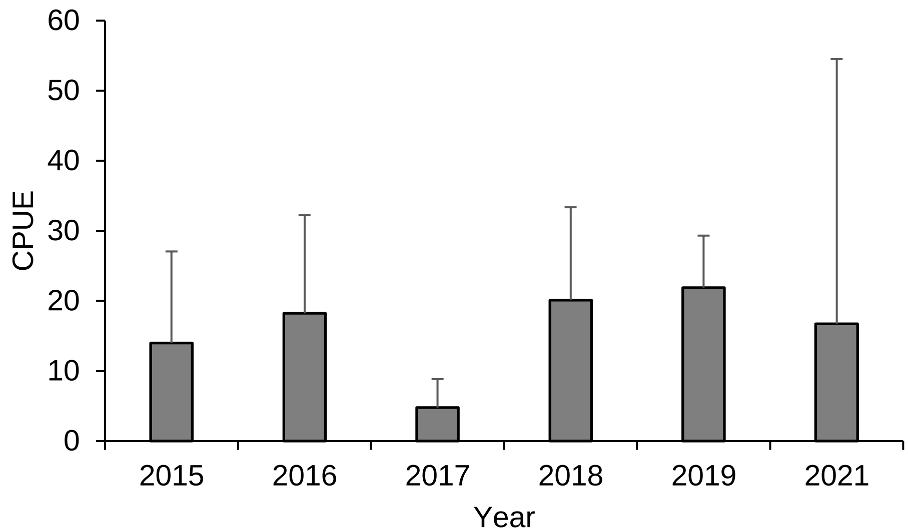

Figure 116. Catch-per-unit-effort (CPUE; number of kokanee per net-night) and 95% confidence intervals for kokanee collected using suspended gill nets in Island Park Reservoir from 2015 – 2021.

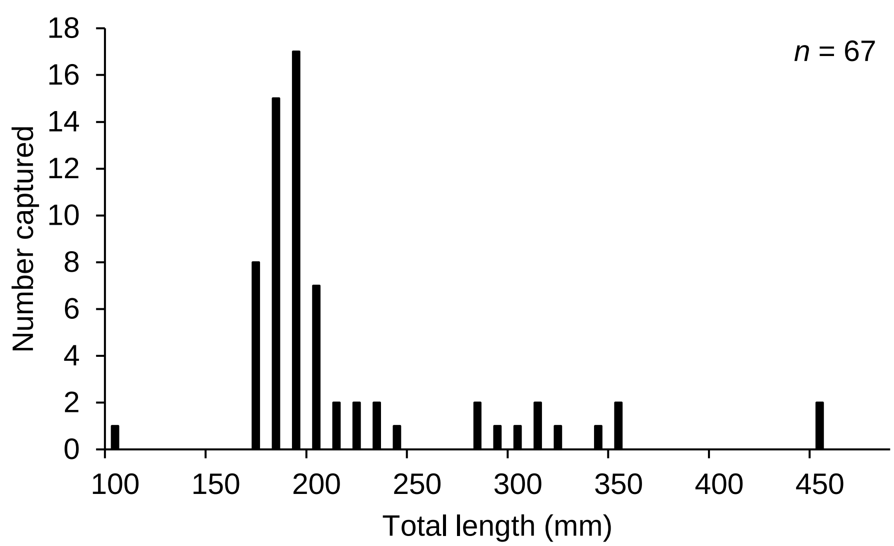

Figure 117. Length frequency distribution of kokanee captured using suspended gill nets in Island Park Reservoir, June 2021.

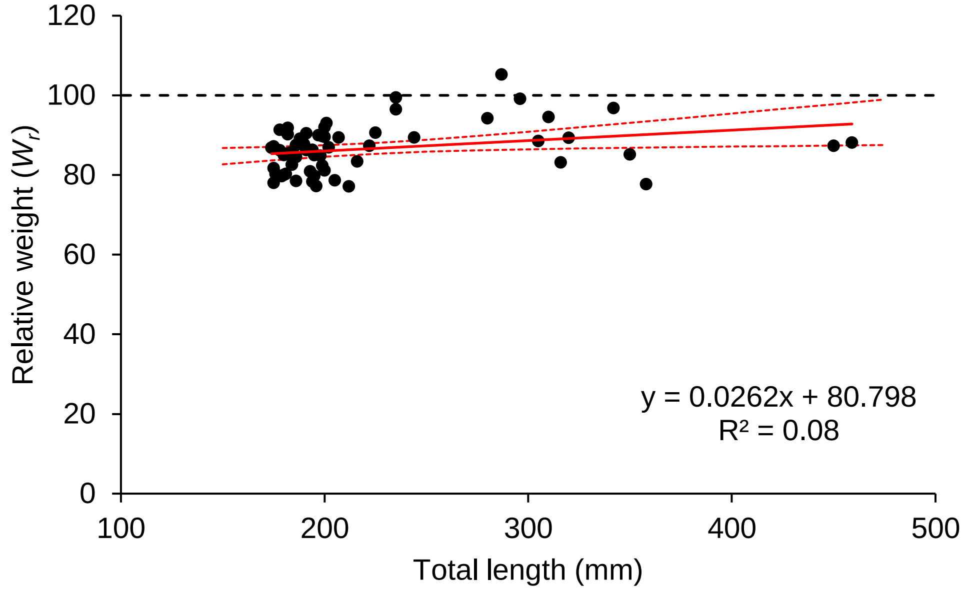

Figure 118. Relative weights of kokanee using suspended gill nets in Island Park Reservoir, June 2021. The best fit linear regression curve is represented by the solid red line, 95% confidence intervals are represented by the dotted red line, and the dashed line is the species standard (100).

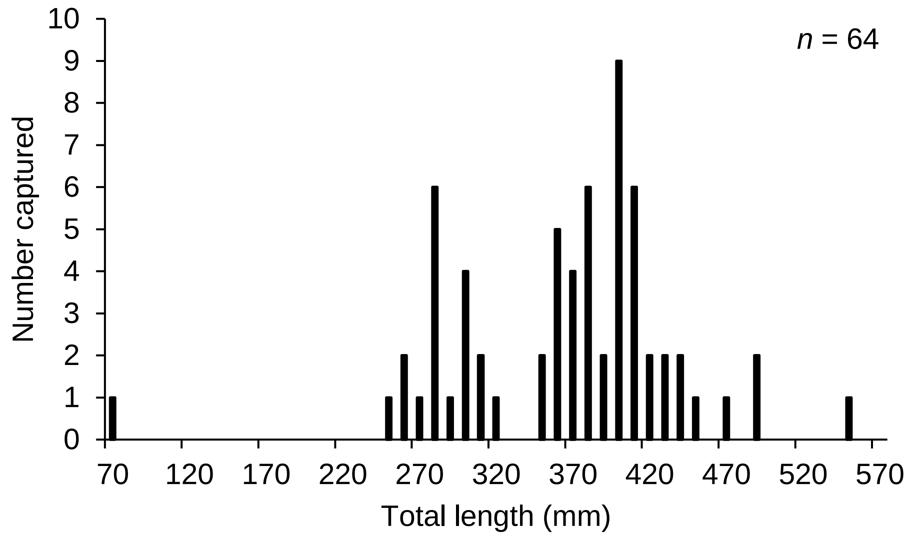

Figure 119. Length-frequency distribution of Rainbow Trout captured using floating and sinking gill nets, suspended gill nets, and electrofishing combined in Island Park Reservoir, June 2021.

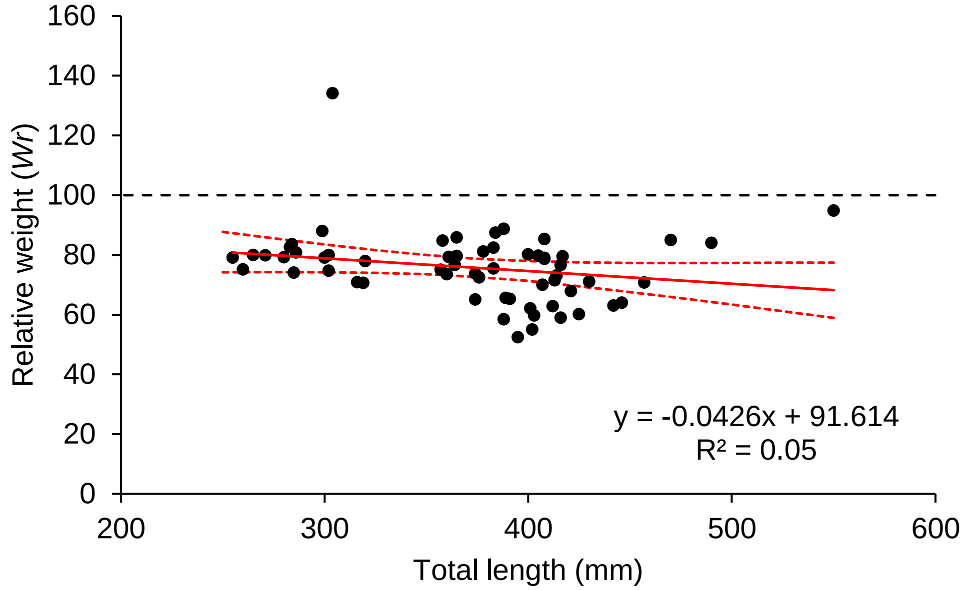

Figure 120. Relative weight of Rainbow Trout across total length (mm) captured using floating and sinking gill nets, suspended gill nets, and electrofishing in Island Park Reservoir, June 2021. The best fit linear regression curve is represented by the solid red line, 95 % confidence intervals are represented by the dotted red lines, and the dashed line is the species standard (100).

Appendix A. Gill net locations in Island Park Reservoir, June 2021. All coordinates used NAD27 and are in Zone 12.

Site Location	Latitude	Longitude
Standard gill net		
Dam	44.420538	111.403602
Bills Island	44.431547	111.433480
West Mouth	44.418247	111.472712
Lakeside	44.436354	111.442915
MP56	44.402404	111.491471
West End	44.394120	111.537647
Suspended gill net		
25	44.427720°	111.393840°
104	44.424930°	111.406520°
188	44.422200°	111.421520°
293	44.426590°	111.456700°
Electrofishing		
18	44.420394	111.440059
21	44.422337	111.445428
52	44.399712	111.445428
107	44.380037	111.554534
156	44.419923	111.487053
168	44.439215	111.455906
195	44.426356	111.404384

Appendix I. Annual kokanee stocking in Island Park Reservoir, Moose Creek, and Big Springs Creek, 1944 – 2021.

	Island Park Reservoir		Moose Creek			Big Springs Creek	
Year	Fingerling	Fry	Fingerling	Fry	Eggs	Fingerling	Fry
1944	67,770	--	--	--	--	--	--
1945	51,510	--	--	--	--	--	--
1968	360,000	--	--	107,724	--	--	--
1969	200,000	--	--	--	--	--	--
1981	--	--	--	503,198	--	--	--
1982	--	--	--	199,800	--	--	--
1984	--	--	--	760,300	--	--	--
1985	833,690	--	--	--	--	--	--
1988	--	--	--	104,720	--	--	25,200
1989	--	--	--	233,020	--	--	--
1990	189,000	--	--	--	--	--	--
1991	104,745	--	20,000	135,660	--	--	--
1992	142,142	--	115,905	--	--	--	63,000
1993	200,624	--	--	--	--	--	--
1994	596,250	--	--	--	--	--	--
1995	500,000	--	--	--	--	--	--
1996	5,000	--	419,100	--	--	--	--
1997	554,315	--	--	--	--	--	--
1998	125,304	--	--	--	--	--	--
1999	41,600	--	304,807	--	--	--	--
2000	--	--	579,128	--	--	--	--
2001	474,640	--	--	--	--	--	--
2002	402,648	--	--	--	--	--	--
2003	30,000	--	--	--	--	--	--
2004	203,695	--	--	--	--	--	--
2005	248,000	--	--	--	--	--	--
2006	418,575	--	--	--	--	--	--
2007	620,760	--	--	--	--	--	--
2008	--	223,040	--	--	--	--	--
2009	125,875	--	62,938	--	--	62,938	--
2010	108,575	--	54,287	--	--	54,287	--
2011	54,515	--	59,955	--	--	59,955	--
2012	120,391	--	65,400	--	--	65,400	--
2013	125,000	--	62,500	--	--	62,500	--
2014	129,250	--	64,625	--	53,050[a]	64,625	--
2015	248,704	--	--	--	60,000[b]	--	--
2016	252,340	--	--	--	--	--	--
2017	250,349	--	--	--	--	--	--
2018[c]	--	--	--	--	--	--	--
2019	250,177	--	--	--	--	--	--
2020[c]	--	--	--	--	--	--	--
2021	153,450	--	--	--	--	--	--

[a]Includes 9,929 eggs stocked in Lucky Dog Creek.
[b]Includes 10,000 eggs stocked in Lucky Dog Creek.
[c]Due to the shortage of kokanee eggs statewide no kokanee were stocked.

FISHING AND BOATING ACCESS PROGRAM

ABSTRACT

Staff maintains 55 fishing and boating access sites within IDFG's Region 6. Sites need continual maintenance, repair, and cleaning. These duties were completed in 2021 at all access sites. In addition, staff facilitated the completion of several improvement projects at IDFG-owned properties including the regional office, Henrys Lake cabin and water lines, wildlife management areas (WMAs) throughout the region, and the completion of the flumes at Duck Creek. These projects include light retrofit, carpet replacement, conference room remodels at the Regional Office, flow meter installation at Henrys Lake, and roof repairs to the Henrys Lake cabin. At several of the region's WMAs, access staff work on road replacement, road grading, a vaulted toilet installation at Mackay Fish Hatchery, and general maintenance projects. Access staff also worked to finish the flumes on two Duck Creek diversions to finish this two-year project. Upper Snake Region Access program staff also facilitated the development of fishing and boating infrastructure at other properties including Clowards Crossing, and dock repairs in the region. Regional Access staff spent considerable time working on partnerships to cooperatively manage sites in the Upper Snake Region, including Frome Park, Henrys Lake State Park, and South Shore sites in Island Park Reservoir, South Fork Snake River access sites, and the Antelope Creek access site in the Big Lost River drainage.

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INTRODUCTION

The goal of the Idaho Department of Fish and Game's Fishing and Boating Access program is to provide high-quality developed access sites and amenities that allow hunters, anglers, and trappers to safely use and enjoy different waters throughout Region 6. The staff maintains 55 fishing and boating access sites within the region. Also, access to properties owned by others (other state, federal, or non-governmental organizations) is provided with cooperative agreements, memorandums-of-understanding, or right-of-ways. Access facilities and properties require a high amount of maintenance. Maintenance activities and frequencies are adjusted to account for use, weather, and other reasons. Typical maintenance activities include: pumping and cleaning vault toilets, inspecting and maintaining property and water control infrastructure, grading roads and parking lots, managing contractors, installing and removing docks (to avoid ice damage), removing sediment and snow from boat ramps and parking areas, managing vegetation, maintaining brush and overgrown willows, as well as posting and replacing worn or damaged signs. In addition to normal maintenance responsibilities and activities, regional staff participates in capital improvement projects that often involve constructing new access amenities at new or existing sites or replacing dilapidated infrastructure at existing sites. Furthermore, staff encourages and facilitates the development of fishing and boating access sites and opportunities on properties owned by others such as city or county governments. Funding for this program originates from a variety of sources including the Dingell-Johnson and Pittman Robertson excise taxes administered by the U.S. Fish and Wildlife Service; license money generated from the sales of IDFG licenses, tags, depredation fees, and permits; mitigation settlements; as well as through a variety of grants, and fee programs with partner agencies. The access program in the Region 6 also has charge over maintenance and improvement projects of the Regional Office. This Includes toilet repairs, light repairs, door repairs, maintenance on the HVAC systems, and general building safety including ADA compliance, fire sprinklers, and fire extinguishers. Funding for work completed at the Regional Office is supplied by license funds.

ACCOMPLISHMENTS

The Region 6 access staff completed normal operations and maintenance activities as usual and expected with our limitations in the recent pandemic. Also, staff contributed directly to the completion of several large-scale, and small-scale renovation or repair projects on department-managed properties during 2021, including the pole barn installation, sewer repairs at the regional office, Grizzly Bear *Ursus arctos horribilis* trap repairs, and weir repairs on Pine Creek. Access staff also assisted the Idaho Department of Fish and Game's (IDFG) Nampa Region in early spring. The Nampa Region of IDFG evaluated three properties that are owned and operated by IDFG for development of backcountry airstrips to increase hunter and angler access to the main Salmon River corridor in the Frank Church River of No Return Wilderness.
Region 6 access staff also assisted the Mackay Fish Hatchery by installing a new vault toilet. This effort included preparing the site for installation, securing the necessary permit, installing the vault, setting the toilet, pouring an ADA parking pad, and painting and signing the new facility (Figure 121).

Region 6 access crew plowed the snow out of multiple access areas first thing in the spring, allowing sportsmen to have quicker access to the river for early season Rainbow Trout *Oncorhynchus mykiss* redd fishing opportunities. These sites were along the South Fork Snake River.

Ice broke a dock section at our Warm Slough access site in the winter of 2020/ 2021. The dock section drifted through three diversions before getting lodged onto a bank above our Mike Walker boat ramp on the Snake River. IDFG crews retrieved the dock and salvaged sections to be placed at Warm Slough again.

The Southeast Region needed some help building a new campground on the Portneuf River near Lava Hot Springs. With limited crews during the COVID-19 Pandemic, regional access crews worked together to accomplish projects that were already in process and funded. The Region 6 crew and machinery helped accomplish the building of this site.

Mud Lake south boat dock was replaced with a plastic dock. This new L-shaped dock system is longer and allows for more boats to be at the ramp while loading and unloading which increased efficiency and decreased ramp congestion. This style dock is non-marring which protects the boats that use the ramp and dock system. The plastic dock has more cleats than the previous dock for securing boats and is completely ADA compliant.

The Access crew worked at several sites enhancing our signage and making repairs and updates to sign kiosks all across Region 6 (Figure 122).

Louis Pond was finished last year, and it now has an ADA fishing dock, gazebo for shelter, and an ADA sidewalk from a paved parking lot. This means our newest fishing pond is fully ADA compliant.

The Antelope Creek access area in the Big Lost Drainage had a fence and cattle guard installed defining the property boundaries. This helped protect the neighbor’s cattle while the fence was being built, and it protects our site when cattle are herded down the road to and from rangeland. IDFG should finish this project in the field season of 2022 offering a new fishing access site with camping along Antelope Creek.

Cloward's Crossing had several changes in the last several years. This year we noticed that UTV users had cut down several willow clusters and were driving around the gate accessing a restricted area on our WMA. The access crew assisted WMA staff in placing concrete blocks to deter others from using the same path.

ADDITIONAL ACCOMPLISHMENTS

Region 6 access staff helped the fisheries management team with several projects throughout the season. The crew helped electrofish on the Teton River; this not only helped a short-handed crew this gave expanded knowledge to the crew by improving their rowing skills and river knowledge.

We also helped with high mountain lake stocking at Boulder Lake, Boulder Lake #2, and Washington Lake. Participating in field surveys enhances our knowledge of our region’s resources, so we can direct our users to where more recreation opportunities are.

ACKNOWLEDGEMENTS

We would like to thank Bingham County Search and Rescue (SAR). Their SAR team played a vital role in helping IDFG find a water quality sonde in Henrys Lake. The SAR team used a state-of-the-art sonar to help isolate the sonde so divers could retrieve it. This particular sonar is one of two in the state of Idaho. In my opinion, we could not have found the sonde without this technology and the volunteers that made this recovery possible.

Figure 121. Installing a vault toilet at Mackay Fish Hatchery.

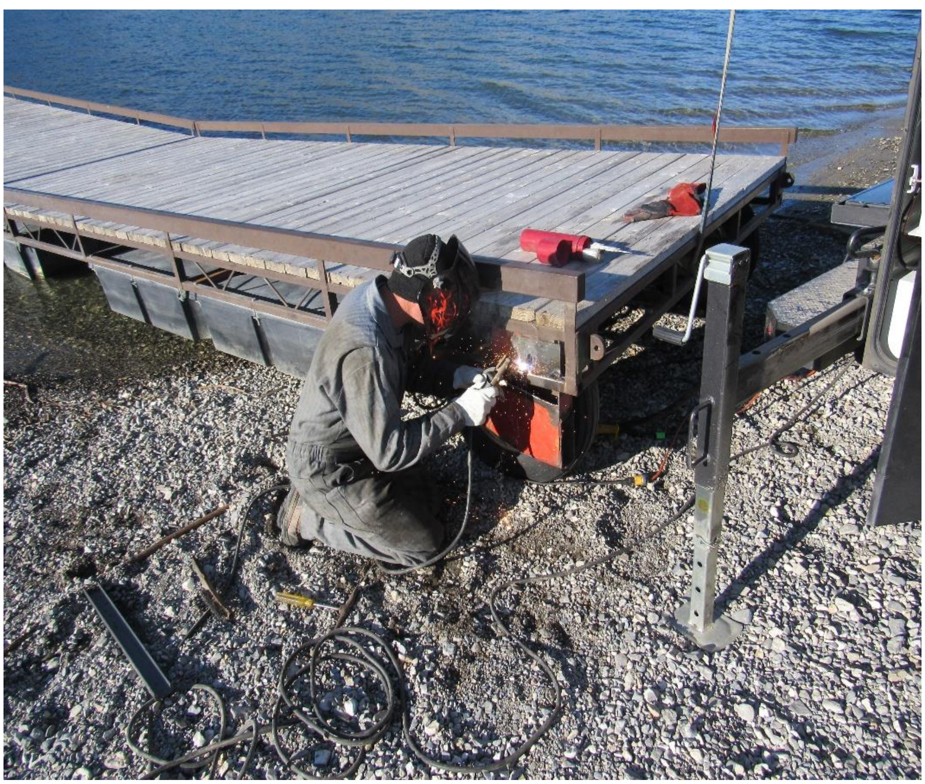

Figure 122. Repairing a section of dock at Mackay Reservoir's upper access site.

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