FISHERY MANAGEMENT INVESTIGATIONS

IDAHO DEPARTMENT OF FISH AND GAME
FISHERY MANAGEMENT ANNUAL REPORT

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UPPER SNAKE REGION
2019

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August 2020
IDFG 20-103

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SOUTH FORK SNAKE RIVER

ABSTRACT

The South Fork Snake River (SFSR) supports the largest fluvial population of native Yellowstone Cutthroat Trout *Ocorhynchus clarkii bouvieri* (YCT) in Idaho, and one of the only remaining populations throughout their range with both fluvial and resident life histories intact. The SFSR also supports important nonnative, sport fish populations of Rainbow Trout *O. mykiss* (RBT) and Brown Trout *Salmo trutta* (BNT). Rainbow Trout are the largest threat to native YCT through hybridization and competition. In autumn of 2019, we estimated 1,217 YCT/km and 1,371 RBT/km at the Conant monitoring reach. The IDFG Management Plan has an objective for the South Fork that RBT (and RBT x YCT hybrids) represent less than 10% of the trout species composition. Currently, RBT compose 44.6% of the trout species. Suppression of RBT at tributary weirs, with boat electrofishing of spawning areas, and with incentivized angler harvest combined may be effective tools for managing RBT abundance in the SFSR. The RBT harvest incentive program has been in place for 10 years. In 2019, we marked 1,360 RBT with coded wire tags. Anglers turned in 4,536 RBT heads including 114 with tags (2.5%) worth $7,950, the highest payout on record. During the course of the program, anglers turned in 29,068 RBT heads including 625 with tags (2.2%) worth $51,800. Manual suppression via boat electrofishing of RBT in the South Fork occurred on a trial basis, and we effectively removed substantial numbers of RBT. During 19 days of boat electrofishing, 5,857 RBT were removed from the South Fork and transported to local ponds. Weirs were operated on four tributaries in 2019. Where estimated, trapping efficiencies at the weirs averaged 83%, slightly below the ten-year average. Spawning run sizes for YCT indexed at four large tributaries exceeded the ten-year average, but only 70 YCT were captured at Rainey Creek. Threats to YCT populations remain in the SFSR, but consistent and adaptive management can help YCT abundance continue to increase and maintain population viability.

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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; and is one of only a handful of large rivers in the species' range which 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 listing petition was denied, 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, 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 and the Idaho Department of Fish and Game (IDFG) 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 time period, RBT increasingly pioneered tributary habitat for spawning. Rainbow Trout and RBT x Cutthroat Trout hybrids (collectively RBT hereafter) were identified as the biggest threat to the continued persistence of YCT in the SFSR (Moller and Van Kirk 2003, IDFG 2007; Van Kirk et al. 2010) because of risks through competition (Seiler and Keely 2007a) and hybridization (Henderson et. al 2000). Hybridization may result in the loss of genetically distinct YCT as gene flow transfers 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). 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). 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 or competition.

Since 2004, the IDFG and collaborators have implemented several YCT conservation management strategies in the SFSR drainage to ensure viability and genetic integrity of these populations. Associated objectives were outlined in the State Fish Management Plan (IDFG 2019) and included preserving the genetic integrity and population viability of native YCT and limiting RBT prevalence to less than 10% of the trout species composition of the catch at the Conant monitoring reach measured during annual fall electrofishing surveys. The 10% threshold would be similar to species compositions documented during the early to mid-1980s.

The primary RBT suppression strategy utilizes fish weirs and traps on four, primary spawning tributaries. At weirs, RBT and hybrids are trapped and removed from spawning runs. The IDFG started constructing fish weirs and traps on spawning tributaries in 1996 and have been manually removing RBT from spawning runs since 2001 to limit RBT invasion and hybridization with YCT. The IDFG was limited by the low effectiveness of previous weirs and traps during high flows (Schrader and Fredericks 2006). Weir modifications (from picket or floating weirs to electrical weirs or vertical/velocity barrier) have increased trap efficiency, especially during high spring flows (High et al. 2011). The trap and weir program on these tributaries has greatly reduced the possibly that RBT can access YCT 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. Weirs in the SFSR tributaries are unique because they maintain the YCT fluvial life

history while managing threats from RBT invasion and hybridization. Commonly, impassable barriers are used to manage hybridization, which unfortunately reduce life history variation and gene flow (Neville et al. 2006). If RBT were allowed to invade the major spawning tributaries, there may be little chance of securing long-term viability of YCT in the SFSR (Van Kirk et al. 2010). However, weirs alone only provide areas of refuge for YCT to spawn in the absence of RBT, and do not affect status of RBT in other portions of the watershed.

The second management strategy for maintaining status of YCT has relied upon managing discharges from Palisades Dam. Previous research indicated flows similar to a natural hydrograph (i.e. unregulated) in both timing and shape, benefit riparian zones and river function (Hauer et al. 2004) as well as YCT recruitment, while limiting recruitment of RBT (Moller and Van Kirk 2003). Pursuing the second strategy of an annual freshet was discontinued as a management goal in 2019. Modelling identified the freshet needed to exceed 25,000 cfs to result in reduced recruitment of RBT (Oldemeyer and Van Kirk 2018). Discharges of this magnitude exceed the flood-stage of 23,000 cfs set by the Army Corps of Engineers, and thus was not practical due to the potential for damage of existing infrastructure.

In 2004, harvest regulations were modified on the SFSR to year-round seasons and unlimited bag limits for RBT. This modification resulted in a brief increase in harvest (Schrader and Fredericks 2006). However, now that the regulations have been in place for several years, harvest has decreased. To counter this, we developed a program that provided a monetary incentive for RBT harvest. The third management strategy intended to increase suppression of RBT through angler harvest in the main stem SFSR. Anglers have the ability to play a key role in YCT management efforts on the SFSR should they choose to, and contribute to cutthroat conservation through RBT harvest. However, annual harvest rates were low due largely to the prevalent catch-and-release ethic embraced by many trout anglers. Despite attempts to incentivize, angler harvest rates were generally less than 20% except for one year since 2004 (High et al. 2011). Population modeling indicates suppression must exceed 20% annually in combination with other management strategies including managed high spring discharges and suppression of RBT at tributary weirs to result in a decreasing RBT population in the SFSR (Van Kirk et al. 2010; Devita 2014).

Beginning in 2018, IDFG chose to test the effectiveness of boat electrofishing to suppress RBT. In effect, this strategy could apply additional mortality to the RBT population and if sufficiently high could decrease abundance of RBT. 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 would be more feasible. This pilot study in 2018 identified that catch did not decline after initial suppression, suggesting larger suppression efforts might be effective. In 2019, IDFG tested available equipment and staffing to understand what a moderate level of suppression could achieve and what logistics it required.

All management strategies were designed to achieve the same goals, which are the preservation of the genetic integrity of YCT in the SFSR and maintaining the population's long-term viability (IDFG 2007; IDFG 2019). Results from the annual electrofishing surveys at two monitoring reaches were used to assess recruitment, population trend, and population abundance, which in turn were used to assess the effectiveness of these management strategies. This report summarizes YCT management actions in the SFSR during 2019. A fluvial population of YCT persists in the SFSR, but its long-term viability is threatened primarily by an abundant nonnative RBT population.

OBJECTIVES

1. Operate weirs on Palisades, Rainey, Pine, and Burns creeks to limit RBT access to important YCT spawning and rearing areas.

2. Increase suppression of RBT in the main stem SFSR by maintaining the RRBT harvest incentive program and by other creative solutions.

3. Determine if removing large numbers of RBT with boat electrofishing at main stem SFSR spawning areas is efficient and pragmatic considering existing resources.

4. Suppress resident RBT in Palisades Creek to reduce hybridization and competition with YCT.

5. Continue monitoring trout abundance and species composition at Conant and Lorenzo monitoring reaches.

6. Increase production of YCT by planting locally sourced eyed eggs in Rainey Creek.

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 106 km (65 mi) portion of the Snake River that flows from Palisades Dam to the confluence with the Henrys Fork is referred to as the South Fork Snake River. This section of the SFSR is regarded as a world-class trout fishery and is an important factor to local economies. Anglers and biologists divide the SFSR into three segments. The first segment, called "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. All but one of the four main YCT spawning tributaries enter the SFSR in this upper river, including Palisades, Rainey, and Pine creeks. The second segment of the SFSR flows from Pine Creek downstream to Heise, and is commonly referred to as "the canyon." Burns Creek, the fourth major YCT spawning tributary enters the SFSR in the canyon. The last segment of the SFSR flows from Heise to the confluence with the Henrys Fork, and is commonly referred to as "the lower river." There are no major YCT spawning tributaries in the lower river, and while constant 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 (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 native YCT, other salmonids in the SFSR include RBT, Brown Trout *Salmo Trutta* (BNT), Lake Trout *Salvelinus namaycush*, Brook Trout *S. fontinalis* and native Mountain Whitefish *Propsopium williamsoni* (MWF). Utah Sucker *Catostomus ardens*, Bluehead Sucker *C. discobolus*, and Mountain Sucker *C. platyrhynchus* are the native catostomids. Native Cottids are represented by the Paiute Sculpin *Cottus beldingii* and members of the Mottled Sculpin complex *C. sp.*

METHODS

Main Stem South Fork

Abundance Monitoring

Trout abundances have been estimated annually at the Lorenzo and Conant monitoring reaches of the SFSR since 2009. Surveys at Conant have been attempted annually since 1982 and at Lorenzo since 2009; though the frequency of surveys at Lorenzo was more sporadic in the late 1990s and early 2000s. Surveys are conducted during the fall when river flows decrease after the main irrigation season ends. The Conant reach is representative of the upper SFSR, and begins at the Swan Valley Bridge continuing downstream 4.9 km. The Lorenzo reach is representative of the lower SFSR and is 4.8 km long, approximately equally distributed upstream and downstream of U.S. Highway 20. We used pulsed direct current (DC) at 7–10 amps, 200–350 volts, 50% pulse width, and a frequency of 60 Hertz. Captured fish were identified to species and measured (total length; TL), and were marked with a hole punch in the caudal fin during our marking pass. This mark was used to identify previously captured fish during our recapture pass 5–7 days later.

In 2019, we sampled the Lorenzo monitoring reach September 17–18 (marking runs) and September 24–25 (recapture runs). We sampled the Conant monitoring reach October 8–9 (marking runs) and October 15–17 (recapture runs). Abundance (fish/km) was estimated separately for each species for age-1 and older trout only. The minimum total length of age-1 fish for each species (YCT ≥ 102 mm, BNT ≥ 178 mm, and RBT ≥ 152 mm) was previously estimated (Schrader and Fredericks 2006a). Abundance was also estimated for all trout species combined which included al fish ≥102 mm. The Fisheries Analysis+ program (developed by the Montana Department of Fish, Wildlife, and Parks) was used 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.

Trends for estimated abundance were calculated for the past 16 years for each trout species at each monitoring reach. This duration was selected to monitor changes with trout abundances since regulations on the SFSR were last modified. From 2000 to 2003, the trout bag limit was six fish, but only two could be YCT or BNT with none under 16 inches. In 2004, the RBT and hybrid bag limit was removed, allowing unlimited harvest, and YCT harvest was changed to zero. Brown Trout regulations were not changed at this time. Linear regression was used to estimate the intrinsic rate of change in abundance for each species where sample year was used as the independent variable and the $\log_e$ transformed abundance estimate (fish/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). 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 ($\alpha = 0.10$) around the slope of the regression line. If 90% CIs included zero, the trend was significant. We used $\alpha = 0.10$ for more power to detect significant trends in abundance (Peterman 1990; Maxell 1999).

Rainbow Trout Harvest Incentive Program

In March, RBT were individually marked with coded wire tags (CWT) in the snout. Boat electrofishing was conducted to capture RBT. Only RBT from 150–400 mm were tagged 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. The tags were etched with five different six-digit

numbers corresponding to the following monetary values: $50, $100, $200, $500, and $1,000. Currently, we attempt to tag 575 fish annually from Palisades Dam to Byington boat ramp including $50-300, $100-200, $200-50, $500-20, and $1,000-5. Anglers wishing to participate in the program were required to turn in the heads of RBT to the IDFG regional office directly or via freezers placed at the Byington and Conant boat ramp areas. When anglers turned in a head, they were asked to fill out a short questionnaire. The questionnaire asked for the angler's contact information (name, address and phone number), fishing license number, if they released any RBT, how many RBT were released, and if they used bait. On the first Friday of every month, "Fishead Friday", we scanned the heads that had been turned in for CWTs. Anglers were welcomed to observe the scanning process in the fish lab at the IDFG region office. When CWTs were found, the angler was notified to verify the address and inform them of the amount of money they would receive.

Rainbow Trout Suppression

In 2019, IDFG tested available equipment and staff to determine what a moderate level of suppression effort could achieve. Suppression was planned for mid-April through May, which corresponded with spawn timing of RBT in the SFSR. Salmonids, including RBT, congregate in shallow areas during spawning, where they are more vulnerable to electrofishing equipment. Three sections are identified to focus effort in areas of known high densities of RBT. The first section was from the Palisades Bridge downstream to the mouth of Palisades Creek. The second section was from the mouth of Indian Creek downstream to the Spring Creek boat ramp. The third section was from Dry Canyon downstream to Lufkin Bottom. One section was chosen for each day of sampling. Sampling was planned for Tuesday, Wednesday, and Thursday to reduce the potential for conflict with other boaters. We used 1–2 electrofishing boats with two netters and an operator using pulsed DC current at 7–10 amps, 200–350 volts, 50% pulse width, and a frequency of 60 Hertz. Electrofishing effort was concentrated on known RBT spawning locations (Brett High, IDFG, personal communication), or was focused on areas that typified spawning habitat including the downstream tails of pools, upstream head of riffles, or shoreline areas lateral to the channel.

Captured RBT were held in 100-gallon, partially-submerged, in-stream live-wells until sampling was complete as determined by capacity. One live-well accompanied each boat, so the number of live-wells was determined by the number of boats. When live-wells were at full capacity, RBT were scanned for CWTs, and tagged fish were released to incentivize RBT harvest. Remaining RBT were transported by boat to 200- or 300-gallon transport tanks in pickups where they were translocated to local ponds. Dissolved oxygen levels were maintained at 100% during transport and temperature was mitigated as necessary before releasing in ponds.

South Fork Tributaries

Weirs

One combination vertical and velocity barrier (Burns Creek) and three electric weirs (Pine, Rainey, Palisades creeks) were installed, maintained, and operated at the four main spawning tributaries of the SFSR during the 2019 spring spawning run. Dates for starting electrical weirs and trap box installation were selected as at least one day prior to the earliest dates RBT were captured in the respective traps in previous years. Traps were checked every three days until daily catch exceeded 20 fish, then were checked daily. Weirs were operated through mid-July, until the number of trapped fish was less than one YCT per day. Weirs were modified by adding check boards before peak discharge to increase the head of the pool upstream of the weir.

Additionally, the combination of the increased head and the check board increased efficiency of some SFSR weirs in the past (Brett High, IDFG, personal communication).The electric weir at Palisades Creek was operated through August to prevent late-spawning RBT from accessing tributary habitat. The fish trap and ladder boards were removed from Burns Creek after the YCT spawning was complete to prevent BNT and RBT from re-colonizing habitat upstream of the weir.

All fish captured at weirs were identified to species, sexed according to expression of gametes or based on head morphology, and measured to the nearest mm (TL). Cutthroat Trout were marked with a PIT tag/adipose clip or a caudal fin punch and released upstream of the weir. We removed the adipose fin from YCT that received PIT tags as a secondary mark to make future scanning for PIT tags more efficient. All YCT captured in the trap with adipose fin clips were scanned for PIT tags. Caudal fin punches or fresh adipose clip scars were used to identify if the fish had been interrogated at the weir already this season as well as marks used to evaluate weir and trapping efficiency. Rainbow Trout were removed from the tributary. Yellowstone Cutthroat Trout that fell back below the electric barrier or over the vertical/velocity barrier were accounted for to calculate accurate escapement numbers and fallback rates.

We used backpack electrofishing units on Burns and Pine creeks during the spawning season to estimate weir efficiencies. We used a downstream trap on the Palisades Canal screen bypass pipe to estimate weir efficiencies at Palisades Creek. We could not evaluate weir efficiencies at Rainey Creek because of the limited number of YCT passed upstream. To estimate weir efficiencies using backpack electrofishing, we captured fluvial YCT upstream of the fish weir, which we assessed for marks and evidence of prior interrogation at the weirs. Efficiencies for the Burns Creek, Pine Creek, and Palisades Creek weirs were calculated 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 annually from 2009 to 2012. Since the cutoffs were similar from year to year, we averaged the yearly length cutoffs for 2009 through 2012 to form a standard cutoff length (300 mm) to be used for all the SFSR tributaries (High et al. 2011). The yearly length cutoffs were identified by subtracting 1.96 standard deviations from the mean total length of YCT caught at the weirs during each respective year, and effectively eliminated skewing error resulting from erroneously including YCT with resident life history in the efficiency calculations.

We described run size, timing, and fallback rates at weirs for each of the four spawning tributaries. Total run sizes were the sum of new YCT captures at the weirs and were calculated for each sex. The run timing was described for each tributary by determining the date when 50% of the spawning run of YCT had been passed upstream of the weir. We monitored fallback rates so we could exclude those fish from run-size calculations, which produce a more accurate picture of the spawning run. Fallback rates were calculated for each tributary weir by summing the total of freshly marked (ad-clipped or caudal fin punch) observed daily at the traps, divided by the total run size, which did not include fish that fell back and re-ascended into the traps.

Palisades Creek Rainbow Trout Suppression

We used backpack electrofishing units to capture trout in Palisades Creek and then manually removed RBT using phenotypic traits for identification (Meyer et al. 2017b). Electrofishing was conducted during midsummer during base flow conditions to maximize capture efficiencies. In 2019, stream flows were high because of a large snowpack the preceding winter. High stream flows in combination with complex habitat in the upper 6.4 km of the suppression reach made electrofishing inefficient and unsafe. Because of this, and the fact previous efforts had successfully reduced RBT in this section (Meyer et al. 2017a), we did not attempt suppression

efforts in this upper reach in 2019. We performed one removal pass in the lower 3.2 km of Palisades Creek.

Electrofishing suppression started at the weir and proceeded upstream. Teams consisted of 2–4 people (depending on stream flow) with backpack electrofishers and two or more people with nets and buckets. We used a pulsed DC waveform operated at 60 Hz, 200–600 V, and a 2–5 ms pulse width. During sampling, persons with backpack electrofishers covered all available habitats. Where gradient was too steep to effectively net fish, one electrofisher was used to chase trout downstream out of the steep section and into an area with slower water velocity where fish could be more easily immobilized and netted, while the remaining electrofishers were used to block the downstream end of the slower water, where immobilization and netting occurred. Captured fish were identified to species and measured to the nearest mm (TL), and checked for marks and tags. We removed RBT after capture and released all YCT.

Rainey Creek Eyed Eggs

Adult wild YCT brood fish were collected from the SFSR near the mouth of Rainey Creek near the end of June. Ripe brood fish were transported to the Rainey Creek weir where they were spawned by staff from IDFG Grace and American Falls hatcheries. Spawned male brood fish were released above the Rainey Creek weir. We sacrificed all female brood fish and submitted samples to the Eagle Fish Health Lab for further testing for pathogens and viruses while fertilized eggs were rearing at Henrys Lake Hatchery. Eggs from each fish pairing were kept in separate heath trays, which allowed culling diseased eggs prior to stocking into Rainey Creek if warranted, depending on test results from the IDFG Eagle Fish Health lab.

Developing eggs reached the eyed stage on July 16. We estimated the number of eggs, and packed the eggs into nine Whitlock-Vibert boxes. We transported the eggs/boxes, from Henrys Lake Hatchery to Rainey Creek and buried them into the gravel of tail-outs from pools and run habitats in three different reaches. The upper site was near the U.S. Forest Service boundary at the old weir site, the middle stocking location was 10.5 river km upstream from the mouth of Rainey Creek, and the lower stocking location was in Third Creek (a tributary of Rainey Creek) near its mouth which is 2.1 river km upstream from the mouth of Rainey Creek.

We captured YCT fry from Rainey Creek using backpack electrofishing (spot shocking) in shallow, low-velocity areas lateral to the thalweg commonly inhabited by fry. Sampling was conducted between Third Creek and the U.S. Forest Service boundary in Rainey Creek with a goal of 100 tissue samples from YCT fry smaller than 80 mm (TL). Genetic samples from fry as well as the adults used for brood were analyzed at the Eagle Genetics Lab using Parental Based Tagging (PBT) techniques to identify fry produced by the eyed eggs stocking.

Rainey Creek Habitat Restoration

Following the stream restoration completed at Third Creek in 2018, post-project monitoring was conducted through the entire treatment reach in 2019. We conducted single-pass fish survey using one backpack electrofishing unit with an additional technician for netting fish. We used a pulsed DC waveform operated at 60 Hz, 200–600 V, and a 2–5 ms pulse width. During sampling, persons with backpack electrofishers covered all available habitats. Data collected from captured fish included: species, length (TL), and any marks/tags. Stream temperature (°C) and conductivity (µS) were also documented at the time of survey.

In preparation for restoration of other impaired reaches of Rainey Creek, vegetation surveys were conducted for Ute ladies'-tresses (*Spiranthes diluvialis*) during their flowering stage in August. Ute ladies'-tresses are a perennial orchid species listed as 'Threatened' under the Endangered Species Act in 1992 (USFWS 1992). Identification of Ute ladies'-tresses in areas characterized as impaired stream habitat would prevent unnecessary impacts to threatened plant species where present, if stream habitat restoration is planned. One IDFG fish Biologist, three IDFG Botanists, and one U.S. Forest Service Hydrologist conducted the surveys together. Surveys were conducted by walking all riparian habitat and additional lowland areas adjacent to the riparian corridor that typified Ute ladies'-tresses habitat, and identifying flowering plants to the genus level.

RESULTS

Main Stem South Fork

Abundance Monitoring

During 2019, 1,531 trout were captured at the Lorenzo monitoring reach, including 284 YCT, 35 RBT, 1,211 BNT, and 1 BKT. The total trout abundance at Lorenzo for BNT, YCT, RBT, and BKT combined was 1,300 (±122) trout/km. There were an estimated 300 age one and older YCT/km (±84) and 1,000 age one and older BNT/km (±103; Figure 1). Too few RBT were captured to calculate an estimate using mark-recapture techniques, but RBT did compose 2.2% of the catch. Extrapolating 2.2% with the total trout estimate (1,300 trout/km) results in a RBT abundance of about 29 RBT/km at Lorenzo. Trends in abundance from 2004 to 2019 at Lorenzo were stable with no significant changes for BNT (r = -0.011; ±0.030 90% CI), but a significant increase for YCT (r = 0.061; ±0.045 90% CI),

At the Conant monitoring reach, 3,615 trout were captured in 2019. This included 1,294 YCT, 1,586 RBT, and 735 BNT. Total trout abundance was estimated at 3,038 (±155) trout/km. There were an estimated 1,217 age-1 and older YCT/km (±127; Figure 2), 485 age one and older BNT/km (±69), and 1,371 age one and older RBT/km (±121). Currently, RBT compose 44.6% of the total trout composition, while YCT represent 39.6% of the trout composition. At Conant, the intrinsic rate of change for all three trout species increased significantly in abundance from 2004 to 2019 [YCT (r = 0.029; ±0.025 90% CI), BNT (r = 0.056; ±0.032 90% CI), RBT (r = 0.056; ±0.031 90% CI)].

Rainbow Trout Harvest Incentive Program

In 2019, we continued the RBT harvest incentive program for the tenth year. We marked a total of 1,020 RBT with coded wire tags (CWT) in March 2019, including 310 RBT with $50 tags, 192 with $100 tags, 50 with $200 tags, 16 with $500 tags, and 6 fish with $1,000 tags. Additionally, we marked 754 RBT (597 with $50 and 157 with $100) during the fall surveys at the Conant monitoring reach. A total of 182 anglers turned in 4,466 RBT in 2019 (Table 1; Figure 3). Of the 4,536 RBT turned into the program, there were 114 tagged fish. The tag values and number that were turned in were $50 (93), $100 (13), $200 (5), and $500 (2) for a total of $7,950, which was the highest year of reward payout in the program. The tags that were turned in during 2019 were originally placed in RBT during multiple years. In 2019, 56% of recovered tags were originally released in 2018–2019, 28% were released in 2017, 12% were released in 2014–2016, and 4% were released in 2010–2011.

During the ten years of the RBT harvest incentive program, 2,709 surveys were conducted when anglers turned in fish. Since 2010, 94.6% (2,564 of 2,709) of respondents identified if they were using bait. Anglers who used bait (36.7%; *n* = 941) composed a lower percentage than anglers who did not use bait (63.3%; *n* = 1,623). Since 2010, 93.9% (2,544 of 2,709) of respondents stated whether they had released RBT. Of the anglers who responded, 17.3% had released RBT. We were not able to enumerate the number of unique anglers who participated in this program due to variability in personal information (e.g., phone numbers and addresses) between years. The number of unique anglers was estimated for each year (Table 1), though some anglers participated in multiple years with different personal information, so this is likely a minimum estimate.

Since 2010, we tagged 10,204 RBT (average = 1,020) and 625 (6.1%) were returned by anglers (Table 1). From 2010 through 2015, 650 tags were released on average, but from 2016 through 2019 the average number of tags released increased 142.6% (average = 1,577). The average tag return rate (2.2%) varied from a low (0.63%) in 2010 to a high (3.18%) in 2018, but has remained relatively consistent since 2013. Anglers have turned in 28,998 RBT for a total of $51,800 rewarded to anglers.

Rainbow Trout Suppression

During 2019, we suppressed RBT on 19 separate work days and utilized approximately equal amounts of effort among the three reaches. A total of 5,857 RBT were removed from the main stem, with 2,278 from Palisades Bridge to Palisades Creek, 1,858 from Indian Creek to Spring Creek boat ramp, and 1,721 from Dry Canyon to Lufkin Bottom. An additional 284 (4.8%) were captured, and then released when a CWT was identified. Daily catch of RBT averaged 308 (range 114–545).

Fish were transferred to three ponds, where 1,480 were released in Becker Pond, 1,161 were released in Jim Moore Pond, and 2,064 were released in Trail Creek Pond. Daily mortality between time of capture and release averaged 3.1% (range 0.0–21.9%).

South Fork Tributaries

Weirs

From April 8 through July 8, we captured 1,328 migrating trout at the Burns Creek weir, including 4 male RBT, 2 female RBT, and 1,309 YCT (628 males and 681 females; (Table 2). The YCT spawning run peaked in Burns Creek on June 17, when 50% of the total run were passed above the weir. Fallback averaged 14% for male YCT and 8% for female YCT at the Burns Creek weir. We captured 52 fluvial-sized YCT upstream of the Burns Creek weir using backpack electrofishing gear, and found 49 of 52 were marked indicating they were interrogated at the fish weir. Thus, the 2019 trapping efficiency estimate for the Burns Creek weir was 94%.

We operated the Pine Creek weir from April 8 through July 5, capturing 3,199 trout, of which 8 were RBT (4 males and 4 females; Table 2). The 3,191 YCT included 1,194 males and 1,997 females. The YCT spawning run in Pine Creek on June 12. The fallback rates were 13% for female and 14% for male YCT. Upstream of the weir, we sampled fluvial-sized trout and caught 54 YCT, of which 39 had marks, so the 2019 efficiency estimate for the Pine Creek weir was 72%.

We operated the Rainey Creek weir from April 8 through June 24, capturing 70 YCT, which included 23 male YCT and 47 female YCT (Table 2). The YCT spawning run peaked in Rainey Creek on June 6, when 50% of the total run were passed above the weir. We observed one of each the male and female YCT captured had fallen back through the Rainey Cr weir and later re-entered the trap, so fallback rates were 4% for male and 2% for female YCT.

At the Palisades Creek weir, we caught 633 trout between April 8 and July 23. We caught six RBT including two males and four females, which was the lowest number of RBT captured at this weir since 2009, and the second lowest on record. The remaining 627 fish were YCT and included 261 male and 366 female YCT. The YCT spawning run in Palisades Creek peaked when 50% of the new YCT were passed above the weir on June 26. Fallback rates for male YCT were 6% and 5% for female YCT. We captured 47 YCT migrating downstream through the Palisades Canal bypass channel. Of these, 39 were marked yielding a Palisades Creek Weir trap efficiency estimate of 83%.

Palisades Creek Rainbow Trout Suppression

One pass of backpack electrofishing suppression and removal of RBT was completed in the lower 3.2 km of Palisades Creek during three days on July 16, July 23, and July 24. We caught 256 trout, including 190 YCT and 66 RBT. Rainbow Trout made up 26% of the catch, which was comparable to 2017 and 2018 (average-23.4; range 19.3-25.6%). Average size of RBT was 162 mm. Fin tissue was collected from 60 RBT and preserved on Whatman sheets to determine if these RBT were of wild or hatchery origin through future genetic analysis.

Rainey Creek Eyed Eggs

We spawned a total of 62 YCT collected from the SFSR on June 17 and June 21 using boat electrofishing to capture adult fish. This included 30 females and 32 males. Disease testing did not indicate eggs needed to be culled prior to stocking. Eye-up rates averaged 74% (range 0.0%–99.8%). We stocked an estimated 30,480 eggs in three different locations of the lower half of Rainey Creek.

We captured 82 YCT fry using backpack electrofishing from Rainey Creek on September 28, and obtained a tissue sample from each fish. Using PBT, genetic results are pending analysis.

Rainey Creek Habitat Restoration

Post-restoration monitoring at Third Creek was conducted on August 22, 2019. Stream temperatures were 16 °C and conductivity was 300 µS at 11:30 a.m., and 61 trout were captured over 0.8 km. Species composition was 67% YCT, 31% BNT, and 2% RBT.

On August 20, 2019, approximately 7.5 kilometers were surveyed that typified Ute ladies'-tresses habitat in areas that might be impacted by potential stream restoration projects. No Ute ladies'-tresses were identified in these areas during the survey.

DISCUSSION

Combined, angler harvest and manual suppression of RBT are currently the most promising tools for managing threats to YCT in the main stem SFSR. The primary threats to YCT in the SFSR are the reduced abundance of YCT through hybridization and competition (Van Kirk

et al. 2010). Hybridization may result in the loss of genetically distinct YCT, as gene flow transfers from one species to another through backcrossing of interspecific hybrids until no genetically distinct YCT remain (Young 1995; Huxel 1999; Kruse et al. 2000; Kozfkay et al. 2007; Gunnell et al. 2008). 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).

At Conant, RBT continued to be the most abundant species of trout, though a significant decrease in RBT abundance may have been the result of suppression via boat electrofishing and above average angler harvest in the main stem SFSR. Anglers harvested a similar number of RBT (77%) as were manually suppressed, which represents an important contribution from our constituency. During the ten years of this program, anglers have harvested over 29,000 RBT. This program provides important interactions between biologists and their constituency. Through many conversations with anglers on “Fishhead Friday,” it is apparent the program does successfully motivate additional angler harvest of RBT. However, the number of anglers motivated to harvest when they normally would not, is low. Increasing publicity of this program and increasing reward rates are likely to motivate additional anglers. The number of anglers that participated in the program in 2010, when the program was new, was significantly higher than in recent years. However, perhaps the waning participation is due to low reward rates, or a lack of incentive. Most anglers win $50, which may not result in much attention. More $500 to $1,000 winners could result in attention to the program through word-of-mouth or social media, which are the most effective means to publicize the program as opposed to signage or news articles. From 2010 to 2015, we tagged an average of 650 RBT/yr; however, during the last five years, we increased tagging to an average of 1,577/yr. However, the abundance of RBT at Conant increased significantly from 2015 to 2018 also, which may have countered our increased tag rate, keeping reward rates low. Suppression of RBT should help increase this reward rate as we returned tagged fish to further incentivize RBT harvest.

The intrinsic rate of change estimated for main stem trout species since 2004 was generally positive. Significant increases in abundance were estimated for all species except BNT at Lorenzo, which were stable. This trend analysis was used to evaluate changes to harvest regulations last made in 2004. This time series (16 years) may currently be too long for this purpose, where the effect of regulation changes are likely muted by stochastic changes within the environment. Still, it is important to note that increasing opportunities for harvest of RBT did not have a negative effect on RBT abundance. This is likely because most anglers practice catch and release.

It appears that given IDFG’s current resources, manual suppression may be an important tool for reducing abundance of RBT and managing threats to YCT, though several more years of data are needed before firm conclusions are drawn. One benefit of reducing competition with RBT from the main stem SFSR through suppression is that survival and recruitment to age one of juvenile YCT are likely to increase. The benefits to YCT may not be identifiable until at least 2020, because juvenile YCT were too small in 2019 to be recruited to our electrofishing equipment. If recruitment of juvenile YCT to age one is increased through suppression in 2019, the YCT abundance estimates in 2020 should continue the increasing trend currently observed, while RBT should continue to trend downward.

Removing adult RBT before or during their spawning season will reduce the likelihood of hybridization with YCT. In general, YCT in the SFSR spawn later than RBT, though spawn timing can overlap resulting in mixed species using the same spawning habitat (Henderson et al. 2010). Manual suppression also represents the most efficient method 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).

Weir operation, boat electrofishing, and angler harvest allowed combined suppression rates to approach 18% of the RBT population from Palisades Dam to Dry Canyon. Previous modeling suggested that suppression rates need to exceed 20% to significantly benefit YCT (Devita 2014). Our 2019 estimate of 18% RBT suppression is a minimum estimate considering that many harvest-oriented anglers do not turn in their RBT to IDFG (High et al. 2014). In contrast, modeling efforts were completed before RBT populations increased significantly about five years ago likely making the 20% target insufficiently low. Because of this, we think targeting RBT suppression rates of 30% are necessary. Our 2019 efforts suggest that approaching 30% rates are achievable if high-intensity boat electrofishing efforts are completed in future years

We recognize the SFSR weir program as the most important strategy for managing threats from RBT to fluvial and resident YCT in tributaries. This was only the sixth consecutive year since 2010 that we were able to operate weirs and traps on all four major spawning tributaries of the SFSR with high efficiency, and we observed relatively strong spawning runs of YCT in all tributaries except Rainey Creek. The high number of YCT captured at the weirs is the result of high trapping efficiencies and high abundance of adult spawners.

IDFG has been manually removing RBT from 10.5 km of Palisades Creek between the weir and Lower Palisades Lake since 2010. These efforts have reduced RBT abundance in this important section of Palisades Creek used for spawning by fluvial YCT and as rearing areas for juvenile YCT. In 2010, the first year of targeted removal, RBT and hybrids composed 32% of the trout community. By 2016, RBT composed 12% of the trout community in this reach. Considering the amount of work completed thus far, we expected RBT catch to decrease farther. Utilizing similar methods, staff were able to extirpate RBT from Burns Creek, though initial abundance of RBT was much lower and capture efficiencies were higher in this system (High et al. 2014). We are unsure how weir efficiency and additional escapes of RBT from private ponds have lessened our ability to extirpate RBT from this reach (Meyer et al. 2017a). Recently, weir efficiency has been improved by operating the weir through August to prevent RBT that may have been pioneering into Palisades Creek later in the summer. Still, significant catch of RBT occurred in 2019, suggesting that RBT may be escaping from a private pond adjacent to Palisades Creek and upstream from the weir, which would hamper RBT suppression efforts. The outlet of the pond flows directly into Palisades Creek. Considering the past and ongoing efforts in SFSR tributaries, it is critical to address illegal introduction of RBT into ponds within the SFSR watershed to achieve the objectives set forth by the Fisheries Management Plan (IDFG 2019) and the Yellowstone Cutthroat Management Plan (IDFG 2007).

Rainey Creek is the only major tributary that is not demonstrating the success relative to the other tributaries. Poor trapping success from 2001 to 2009 resulted in low catch and reduced effectiveness in removing RBT. In 2011, a new weir was constructed downstream closer to the mouth of Rainey Creek to protect more of the system from invading RBT. We anticipated higher catches with a trap located upstream from the mouth only 5.1 km versus 14 km. The higher catches have yet to materialize. Cutthroat Trout spawning runs at Rainey Creek continues 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 over the same period. It is possible that fluvial YCT in Rainey have gone through a bottleneck, which is defined as a severe reduction in the demographic size of a population (Campbell 1990). 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.

To increase the abundance of spawning YCT in Rainey Creek, and to mitigate the potential genetic or demographic bottleneck, the eyed egg program was initiated in 2017. This program was planned for five years, followed by evaluation to determine if these efforts increased the number of fluvial adult YCT using Rainey Creek. During the first three years, little benefit has been observed as evidenced through fry samples; however in 2020, juveniles from the 2017 out-plants will be old enough to spawn and may be encountered at the Rainey Creek weir. All YCT captured at the weir are routinely tissue sampled, so future tissue samples from spawning fluvial YCT can be compared against past parentage-based tagging (PBT) baselines from adult brood stock for the egg-boxes.

Trout habitat in several areas of Rainey Creek has been degraded. Past farming and grazing practices have resulted in over-widened channels, limited spawning habitat due to sedimentation, little riparian cover, and high stream temperatures, limiting carrying capacity for trout. In 2018, a first order tributary (Third Creek) to Rainey Creek was restored. During fish surveys conducted prior to the project, no fish were observed and maximum daily stream temperature was 22 °C. In 2019, habitat conditions for trout were significantly improved and the trout were again present in this restored reach. More work is needed to improve degraded habitat condition in this tributary and to increase the abundance of fluvial YCT.

MANAGEMENT RECOMMENDATIONS

1. Address threats associated with naturalized populations of RBT in ponds in Palisades and Rainey creeks. Replacing these RBT populations with sterile trout or hatchery YCT will reduce the threats of further introgression and competition at critical hybridization zones in these populations.

2. Alter monetary rewards for the Rainbow Trout harvest incentive program to maximize harvest. This could be accomplished by evaluating the CWT tagging rate, RBT abundance, and reward rate to estimate the number of winners and total reward value in a year. The objective will be to maximize the number of RBT turned in by offering the right reward composition to stimulate word-of-mouth advertising, yet keeping program costs less than $8,000 per year.

3. Increase outreach or signage to increase awareness of the harvest incentive program with anglers.

4. Utilize more than 20 days of boat electrofishing in known spawning areas of the main stem SFSR to suppress RBT and reduce hybridization and competition between RBT and YCT.

5. Use 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.

6. Maintain trap/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.

7. Continue planting of eyed eggs in Rainey Creek for a minimum of one more year and compare returning adult samples to PBT sampling to evaluate impact of the program on increasing YCT returns.

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Table 1. Summary of Rainbow Trout Harvest Incentive Program from 2010 through 2019.

Year	Tagged	Anglers	Rainbow Trout	Tags found	Reward rate (%)	Tag value ($)					Annual payout ($)
						50	100	200	500	1,000	
2010	575	640	2,861	18	0.63	12	3	2	0	1	2,300
2011	600	237	2,001	16	0.80	5	8	1	2	0	2,250
2012	860	207	1,854	37	2.00	23	9	3	2	0	3,650
2013	530	239	2,441	75	3.07	55	13	2	2	1	6,450
2014	705	175	3,587	75	2.09	40	29	7	0	0	6,300
2015	628	137	2,599	72	2.77	48	13	12	0	0	6,100
2016	1,338	117	2,681	45	1.68	35	7	2	1	0	3,350
2017	1,540	87	3,303	71	2.15	58	10	2	0	0	4,300
2018	2,068	104	3,205	102	3.18	75	18	8	2	1	9,150
2019	1,360	182	4,466	114	2.55	93	13	5	2	0	7,950
Total	10,204	2,125	28,998	625		444	123	44	11	3	51,800

Table 2. Weir summary statistics at the four primary spawning tributaries on the South Fork Snake from 2001 through 2019.

Year	Weir type	Operation dates	Weir efficiency (%)[a]	Catch: Cutthroat Trout	Catch: Rainbow Trout	Catch: Total
Burns Creek						
2001[b]	Floating panel	March 7 – July 20	16	3,156	3	3,159
2002[b]	Floating panel	March 23 – July 5	NE[c]	1,898	46	1,944
2003[d]	Floating panel	March 28 – June 23	17–36	1,350	1	1,351
2004	ND[e]	ND	ND	ND	ND	ND
2005	ND	ND	ND	ND	ND	ND
2006	Mitsubishi	April 14 – June 30	NE	1,539	ND	ND
2007	ND	ND	ND	ND	ND	ND
2008	ND	ND	ND	ND	ND	ND
2009	Vert./velocity	Apirl 9 – July 22	98	1,491	2	1,493
2010	Vert./velocity	March 26 – July 14	100	1,550	2	1,552
2011	Vert./velocity	March 23 – July 12	90	891	5	896
2012	Vert./velocity	March 24 – July 11	90	496	0	496
2013	Vert./velocity	April 4 – July 2	98	888	6	894
2014	Vert./velocity	April 1 – July 3	90	833	12	845
2015	Vert./velocity	April 6 – July 3	94	1,357	1	1,358
2016	Vert./velocity	April 4 – July 3	98	1,528	7	1,535
2017	Vert./velocity	April 1 – June 27	87	759	4	763
2018	Vert./velocity	April 3 – July 6	100	1,570	9	1,579
2019	Vert./velocity	April 8 – July 8	94	1,322	6	1,328
Pine Creek						
2001[b]	ND	ND	ND	ND	ND	ND
2002[b]	Floating panel	April 2 – July 5	NE	202	14	216
2003[f]	Floating panel	March 27 – June 12	40	328	7	335
2004	Hard picket	March 25 – June 28	98	2,143	27	2,170
2005	Hard picket	April 6 – June 30	NE	2,817	40	2,857
2006[g]	Mitsubishi	April 14 – April 18	NE	NE	NE	NE
2007	Mitsubishi	March 24 – June 30	20	481	2	483
2008	Hard picket	April 21 – July 8	NE	115	0	115
2009	Hard picket	April 6 – July 15	49	1,356	1	1,357
2010	Electric	April 13 – July 6	NE	2,972	3	2,975
2011	Electric	April 11 – July 9	49	1,509	1	1,510

Table 2 (continued)

Year	Weir type	Operation dates	Weir efficiency (%)[a]	Catch: Cutthroat Trout	Catch: Rainbow Trout	Catch: Total
			Pine Creek			
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	2,155	6	2,131
2019	Electric	April 8 – July 5	72	3,191	8	3,199
			Rainey Creek			
2001[b]	Floating panel	March 7 – July 6	NE[c]	0	0	0
2002[b]	Floating panel	March 26 – June 27	NE	1	0	1
2003	ND[d]	ND	ND	ND	ND	ND
2004	ND	ND	ND	ND	ND	ND
2005	Hard picket	April 7 – June 29	NE	25	0	25
2006	Hard picket	April 5 – June 30	NE	69	3	72
2007	Hard picket	March 19 – June 30	NE	14	0	14
2008	Hard picket	June 19 – July 11	NE	14	0	14
2009	Hard picket	April 7 – July 6	NE	23	0	23
2010	Hard picket	April 13 – June 29	NE	145	1	146
2011	Electric	March 28 – June 28	NE	0	0	0
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	41	0	41
2019	Electric	April 8 – June 24	NE	70	0	70
			Palisades Creek			
2001[b]	Floating panel	March 7 – July 20	10	491	160	651
2002[b]	Floating panel	March 22 – July 7	NE	967	310	1,277
2003	Floating panel	March 24 – June 24	21 – 47	529	181	710
2004	ND	ND	ND	ND	ND	ND
2005	Mitsubishi	March 18 – June 30	91	1,071	301	1,372
2006	Mitsubishi	April 4 – June 30	13	336	52	388

Table 2 (continued)

Year	Weir type	Operation dates	Weir efficiency (%)[a]	Catch: Cutthroat Trout	Catch: Rainbow Trout	Catch: Total
		Palisades Creek				
2007	Electric	May 1 – July 28	98	737	20	757
2008	ND	ND	NE	ND	ND	ND
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	474	18	492
2019	Electric	April 8 – July 23	83	627	6	633

[a]Weir efficiency was estimated using several different methods

[b]From Host (2003)

[c]NE = no estimate

[d]Weir was shut down on June 10, but the trap was operated until June 23

[e]ND = no data; weir either not built or not operated

[f]Weir was shut down early due to high cutthroat trout mortality

[g]Weir was destroyed during high runoff

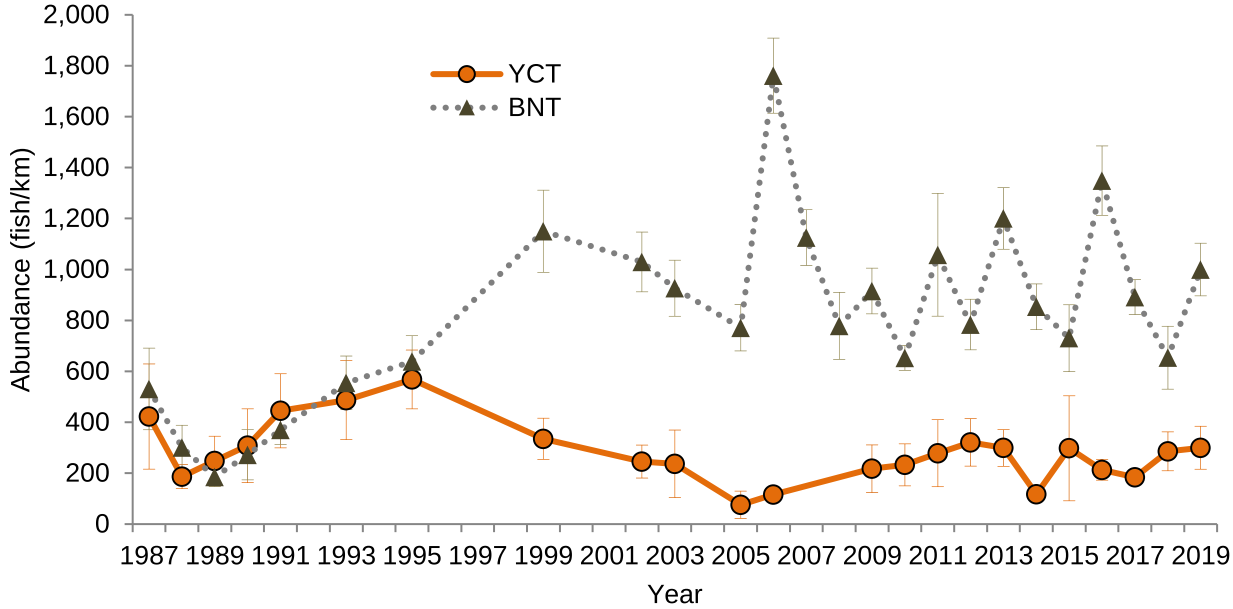

Figure 1. Abundance estimates and 95% confidence intervals for Yellowstone Cutthroat Trout (YCT) and Brown Trout (BNT) at the Lorenzo monitoring reach on the South Fork Snake River from 1987 through 2019.

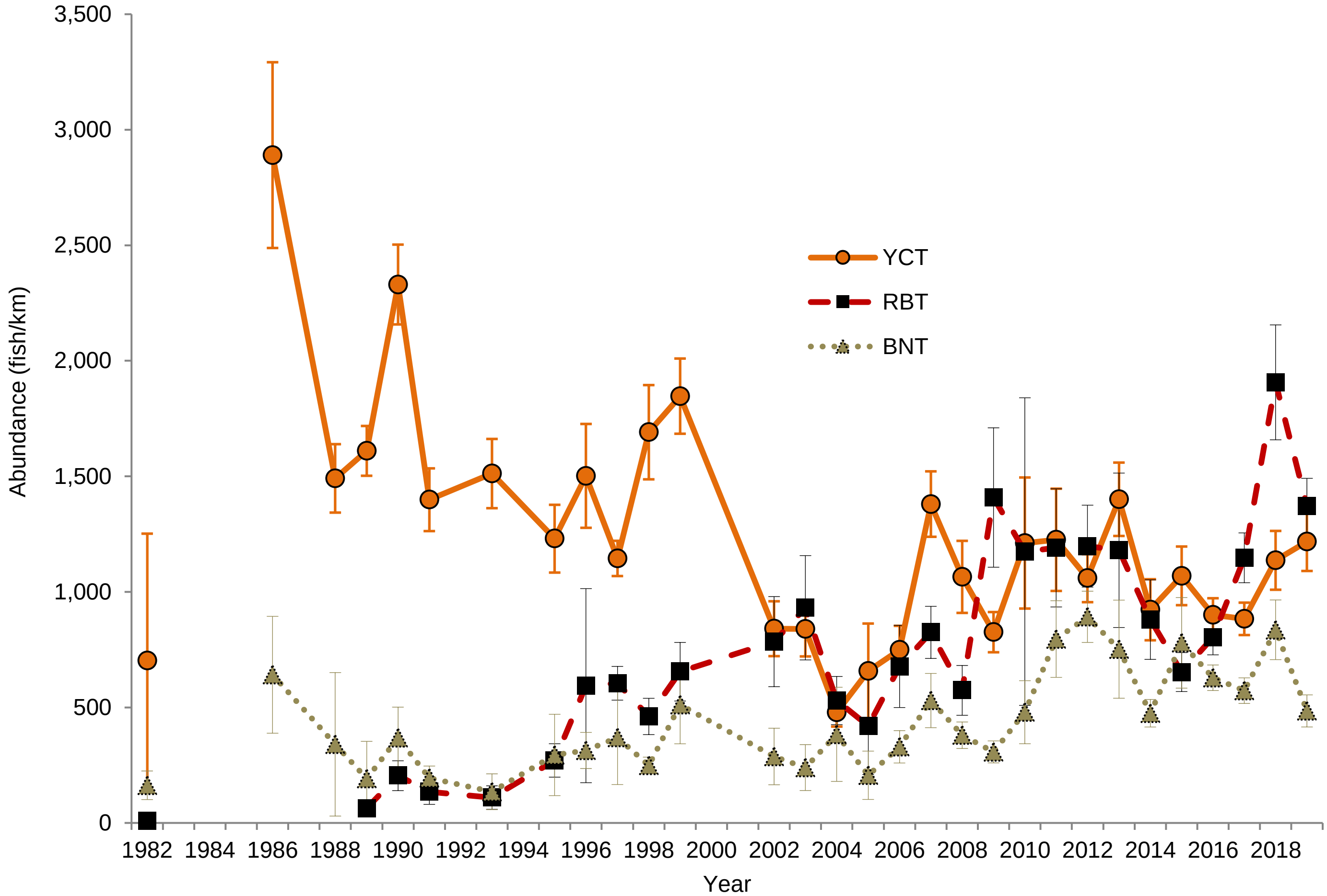

Figure 2. Abundance estimates and 95% confidence intervals for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) at the Conant monitoring reach on the South Fork Snake River from 1982 through 2019.

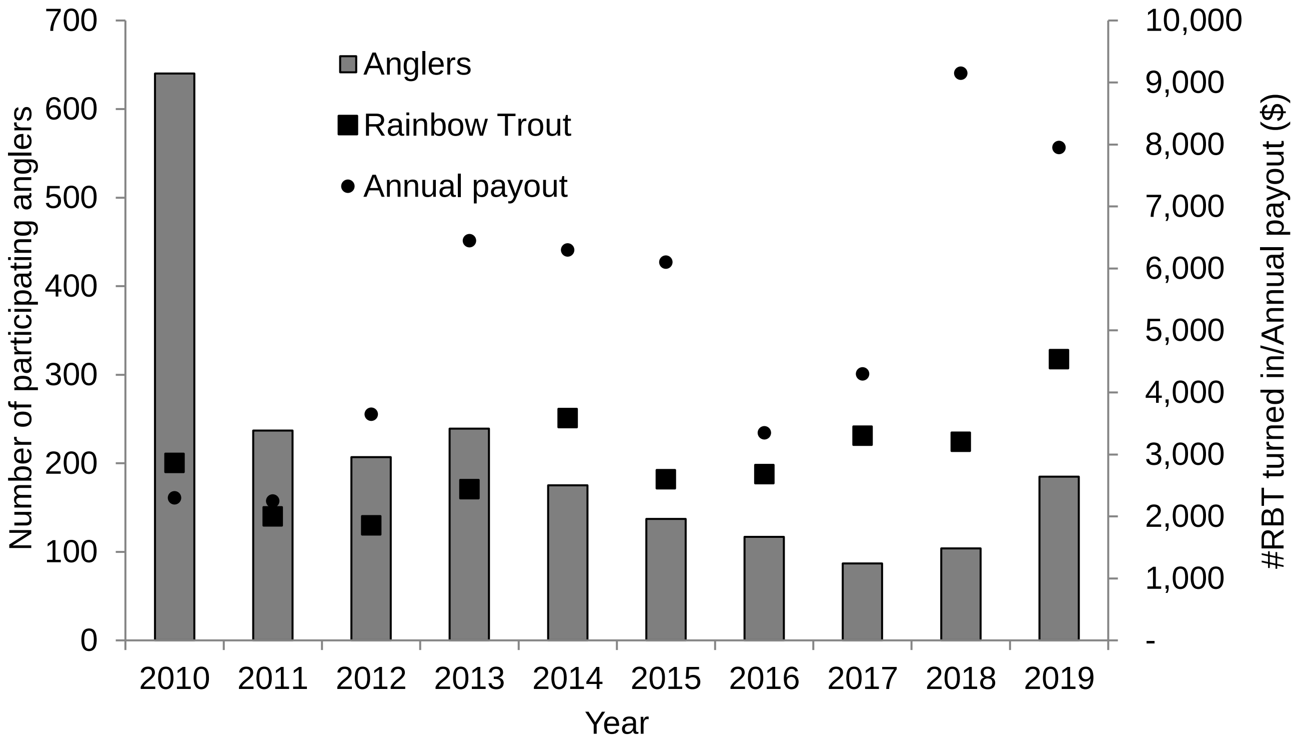

Figure 3. The annual number of anglers participating by submitting Rainbow Trout, the number of Rainbow Trout turned in, and the annual payout to anglers by the Harvest Incentive Program during ten years from 2010 through 2019.

HENRYS FORK

ABSTRACT

We used boat-mounted electrofishing equipment to assess fish populations in the Box Canyon and Stonebridge reaches of the Henrys Fork Snake River during 2019. In Box Canyon, Rainbow Trout *Oncorhynchus mykiss* (RBT) densities were greater (3,061 trout per km; 95% CI ± 237) than densities observed in 2018 (1,738 trout per km; 95% CI ± 176). Overall, RBT densities in 2019 were greater than the 25-year average (1,929 trout per km). The effect of winter discharge (December–February) from Island Park Reservoir on RBT first-winter survival continues to be significantly related to age-2 RBT abundance in our population estimates (Log_{10} age-2 RBT abundance = 0.6191 × $\log_{10}$ mean winter flow at age 0 + 1.9086; $r^2 = 0.44$). Age-2 RBT abundance for 2019 was predicted by flows to be 3,817, but age-2 RBT abundance was estimated by our population estimate to be 5,662. In the Stonebridge reach, we estimated 1,294 trout per km (95% CI ± 149) with a species composition of 77% RBT (964 RBT per km; 95% CI ± 131) and 23% Brown Trout *Salmo trutta* (BNT; 282 BNT per km; 95% CI ± 62).

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INTRODUCTION

The Henrys Fork Snake River 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 Henrys Fork 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 Henrys Fork upstream of Island Park Reservoir provides a general regulation fishery primarily supported by stocked trout. The fishery is also supported by trout that migrate out of Henrys Lake or Island Park Reservoir. Management of the Henrys Fork downstream of Island Park Dam emphasizes wild, natural populations without hatchery supplementation. The Henrys Fork 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.

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 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 April and May of 2019, we sampled the Box Canyon and Stonebridge reaches of the Henrys Fork Snake River (Figure 4). The Box Canyon reach is sampled on an annual basis as part of our long-term monitoring program for the Henrys Fork Snake River. 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. The Stonebridge reach is approximately 4.6 km in length and begins about 3.2 km downstream of the confluence with Warm River and ends at the old bridge piers. Coordinates for all mark-recapture transect boundaries are presented in Appendix A.

OBJECTIVES

To obtain current information on RBT and Mountain Whitefish *Prospoium williamsoni* (MWF) population characteristics to inform fishery management decisions on the Henrys Fork Snake River.

METHODS

All survey reaches were sampled using three electrofishing rafts. Water temperature (°C) and conductivity (µS/cm) were taken 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). Electricity was applied 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 was positioned at the bow of the raft and used a 2.4-m long dip net with 6-mm bar knotless mesh. Netters were instructed to net all trout and MWF and place fish into an aerated live well that was located in the raft. All fish were identified to species and measured for total length (TL) to the nearest mm. In the Box Canyon reach, we marked fish on May 13, 2019 and recaptured fish on May 15. Two passes per boat were made on each marking and recapture day for a total of six passes per day for both marking and recaptures. In the Stonebridge reach, we marked fish on April 29 and recaptured fish on May 1 with only one pass conducted per day. All trout encountered from mark-recapture surveys were collected, identified to species, measured for total length, and those 150 mm or greater were marked with a hole punch in the caudal fin prior to release.

In all reaches, we estimated abundance for all trout ≥ 150 mm using the log-likelihood method in Fisheries Analysis+ software (FA+; Montana Fish, Wildlife, and Parks 2004). We used the Peterson estimate with the Chapman modification to compare the current MWF estimate to historical MWF population estimates. We calculated 95% CI for all abundance estimates. Proportional size distributions (PSD) were calculated as the number of individuals (by species) ≥ 300 mm divided by the number of individuals ≥ 200 mm multiplied by 100. Similarly, relative stock densities (RSD-400) used the same formula, with the numerator replaced by the number of fish ≥ 400 mm (Anderson and Neumann 1996; Neumann et al. 2012). After population estimates were calculated, we estimated trends in population growth in the Stonebridge and Box Canyon reaches using an exponential model and the intrinsic rate of population change (r) as described by Maxell (1999) using $\alpha = 0.10$. We investigated species composition in the Stonebridge reach where RBT and BNT were both present.

We also evaluated the effectiveness of using the mean winter (Dec. 1–Feb. 28) discharge (cubic feet per second [cfs]) from Island Park Dam 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 age-2 RBT abundance and mean winter flow data from the past 25 surveys to establish the following relationship:

$$\log_{10} \text{age-2 RBT abundance} = 0.6191 \times \log_{10} \text{winter stream flow at age 0} + 1.9086$$

Using this equation, we estimated the expected abundance of age-2 RBT in our 2019 population estimate based on mean winter discharge measured during December 2017–February 2018. Winter streamflow data represented conditions experienced by age-0 trout during their first winter (i.e. the 2017 RBT year-class). To investigate this relationship, we estimated age-2 RBT abundance during the 2019 electrofishing surveys, which correlates to the TL (230–329 mm) of age-2 trout 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 flow. Data from 2019 was added to the flow vs. age-2 abundance regression

model, which will continue to be used in management of winter flow releases from Island Park Dam.

RESULTS

Box Canyon

We collected 2,467 trout during two days of electrofishing in Box Canyon. Species composition of trout collected was 88% RBT, 1% Brook Trout *Salvelinus fontinalis*, and 11% MWF. The TL of RBT ranged from 89 to 518 mm with an average of 261 mm (± 158; 95% CI; Figure 5; Appendix B). Rainbow Trout PSD and RSD-400 were 70 and 39, respectively (Table 3). We used the Log-likelihood Method (LLM) in FA+ to estimate 11,326 RBT ≥ 150 mm (± 877; 95% CI) CV = 0.04, Table 4, Appendix C) in the reach, which is approximately 3,061 RBT per km (Figure 6). Our efficiency rate (i.e., the ratio of marked fish during the recapture runs [R] to total fish captured on the recapture run [C]), unadjusted for size selectivity was 14% (Appendix C). Higher than average winter flows in 2017 (i.e., 504 cfs) resulted in a strong age-2 year-class that was evident in the frequency of RBT measuring 230-329 mm. We estimated 2,955 (± 1,324) MWF ≥ 150 mm for the Box Canyon reach (Table 4; Figure 7). We estimated 779 MWF per km with a capture efficiency of 6%.

Our linear regression model, based on 504 cfs, predicted 3,817 age-2 RBT in the 2019 (Figure 8). However, we estimated 5,662 RBT using our length-based estaimte in the Box Canyon reach during 2019 (Figure 8). The regression model continues to be a significant predictor of the relative year-class strength of RBT using mean winter stream flow (Linear regression, r^2=0.44, $F_{1,21}$ = 16.49, P<0.01) and is a useful tool to evaluate the effects of variable winter flows. The intrinsic rate of population growth (i.e., 0.005) has been stable ($F_{1,\ 22\ =}$ 0.36, P = 0.55) in the Box Canyon reach from 1994 to 2019.

Stonebridge

We collected 1,157 trout and 917 Mountain Whitefish during two days of electrofishing in the Stonebridge reach of the Henrys Fork. Species composition of all fishes collected was 44% MWF, 43% RBT, and 13% BNT. Of the trout collected, 77% were RBT and 13% were BNT. Using the log-likelihood method, we estimated 5,953 trout (i.e., RBT and BNT; ± 685; Figure 9) and 13,757 MWF (± 3,651) in the sampling reach. We estimated 4,435 RBT ≥ 150 mm for the reach (± 604), which approximates to 964 RBT per km (Table 4; Figure 10). The intrinsic rate of population growth for RBT has been declining slightly since 2002 (r = -0.04); however, r was not significant ($F_{1,\ 4}$ = 2.74, P = 0.17) indicating a stable trend. The TL of RBT ranged between 100 and 455 mm (Figure 11) with a mean and median of 285 (± 125 mm) and 285 mm, respectively (Table 3). Rainbow Trout PSD and RSD-400 values were 49 and 5, respectively. The total number of BNT ≥ 150 mm estimated for the sampling reach was 1,298 fish (± 286), which is about 282 BNT per km. Brown Trout TL ranged between 125 mm and 576 mm, with a mean and median of 345 mm (± 181 mm) 351 mm, respectively. Brown Trout PSD was 74, RSD-400 and RSD-500 values were 33 and 5, respectively. Brown Trout species composition has a relative increase about 7% since the population estimate in this reach in 2002 (linear regression, r^2 = 0.96, $F_{1,\ 4}$ = 104.6, P < 0.01; Figure 12). Although BNT species composition has been significantly increasing over time, the intrinsic rate of population growth (i.e., r = 0.03) has been stable statistically ($F_{1,\ 4}$ = 3.37, P = 0.14). The abundance of MWF ≥ 150 mm was estimated to be 2,991 fish per km. Our efficiency rate (unadjusted for size selectivity) for all trout was 17% (i.e., 14% for RBT and 25%

for BNT) and for all fishes combined was 12% (e.g., 6% for MWF). Brown Trout composition in the Stonebridge reach continues to increase (Figure 13).

DISCUSSION

Rainbow Trout abundance in Box Canyon exhibited a 76% increase in density from 2018 to 2019. The current population estimate is the third highest estimate in the history of monitoring at this site. Furthermore, the current density estimate was 39% greater than the 23-year average suggesting that the RBT population in Box Canyon appears robust following strong year-classes in 2017 and 2018. A high frequency of RBT ranging from 200 to 300 mm is an indication that age-2 RBT are the dominant age-class in the population, and that higher mean winter flows (e.g., 504 cfs in winter 2017) contributed to this high abundance of age-2 RBT in 2019. These fish will likely continue to contribute to the fishery in upcoming years and provide more opportunities for anglers. However, there appears to be some evidence of reduced growth rates comparing recent length-at-age data from 2014 (IDFG *unpublished data*) to the current length frequency of RBT. This may indicate a density dependent decline in growth and fewer RBT are reaching quality or trophy lengths in this reach, or larger RBT are moving downstream out of the reach. Conducting a population estimate on the Harriman Ranch and/or Riverside reaches may be warranted to investigate the population of RBT in those reaches.

In addition, the capture efficiency of Mountain Whitefish was less than half of that for RBT, which warrants conducting an additional population estimate focusing on Mountain Whitefish to garner data focused on the species to better monitor the population and inform management decisions.

Winter stream flows continue to be a driving factor in RBT abundances within the Box Canyon section (Garren et al. 2006a). Observed age-2 RBT abundance in 2019 was slightly greater than the upper 95% confidence interval from our regression model that incorporated flows during the winter of 2017–2018. Fausch et al. (2001) found Rainbow Trout recruitment was higher in tailwaters exhibiting high winter and/or low spring flows. Spring flows can affect year-class strength. High spring flows can reduce year-class strength due to substrate scouring that displaces eggs and fish larvae, while low spring flows can lead to dewatering redds and egg desiccation (Reiser and White 1983). Spring flows may play a limited role in reducing/increasing year-class strength in Henrys Fork and subsequently cause slight divergences in predictions of the winter flow model. In the preceding five years (2014–2018), average spring flows have ranged from 399–789 cfs. However, previous studies in the Henrys Fork have found winter flows are the primary driver regulating the survival of YOY due to the reduction of complex habitat along the river margins (Meyer and Griffith 1997; Mitro et al. 2003). Thus, the continued use and refinement of the winter flow model appears to be necessary when considering altering dam operations to improve winter flow conditions in Box Canyon.

Trout densities in the Stonebridge reach were declining after the 2003 population estimate, however, the wide margin of error suggests that the abundance of trout per kilometer may not have been as high as projected in 2003. The current survey suggests that trout densities increased about 23% since the 2016 estimate and are comparable to 2013 (e.g., -2.6% change). We observed a longer average TL in BNT than RBT in this reach, but there appear to be strong year-classes for age-2 and age-3 RBT based on length frequencies. Brown Trout species composition is increasing in the Stonebridge reach and follows a similar pattern of compared to reaches of the Henrys Fork downstream of Ashton Dam (e.g., Vernon and Chester reaches). This trend in the Stonebridge reach may continue based on connectivity from Ashton Reservoir to the

Henrys Fork and Warm River; locations where adfluvial BNT can spawn. Although the trend of increased BNT abundance in species composition is significant, there has not been a significant increase in the intrinsic rate of population growth for BNT in the Stonebridge reach. Age composition and diet studies on BNT in the Stonebridge reach would provide insight into the feeding behavior and age structure of the BNT population in this section of the Henrys Fork. Rainbow Trout abundance was declining from 2010 to 2016 while BNT abundance was stable, but both populations increased since 2016. Furthermore, there is no overlap in confidence intervals in the BNT and RBT trends from 2010 to 2016, which suggests that the RBT decline is independent of an increase in BNT abundance. The mechanisms responsible for these trends are not clear. Trout in the Stonebridge reach may use Ashton Reservoir to overwinter and may move between the reservoir and the river for foraging opportunities. We recommend conducting a creel and/or "Tag You're It" study to investigate harvest of trout that may use both lotic and lentic habitats.

RECOMMENDATIONS

1. Continue annual population surveys in the Box Canyon to quantify population response to changes in the flow regime over time and guide future management decisions.

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

3. Investigate the relationship of spring discharge periods from Island Park Dam on the abundance of age-2 RBT.

4. Investigate the age composition of Brown Trout in the Stonebridge reach of the Henrys Fork.

5. Investigate angler harvest of RBT and BNT in Ashton Reservoir and the Stonebridge reach of the Henrys Fork.

6. Conduct a population estimate dedicated to Mountain Whitefish to gather data explicit to the Mountain Whitefish population in the Box Canyon and Stonebridge reaches to better manage the population.

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Table 3. Trout population index summaries (± 95% confidence intervals) for the Henrys Fork Snake River, Idaho 2019.

River Reach	Species	Mean TL (mm)	Median TL (mm)	PSD	RSD-400	RSD-500	Density (No./km)	Species Composition (%)
Box Canyon	Rainbow Trout	261 (±158)	262	70	39	9	3,061 (±237)	100
Stonebridge	Rainbow Trout	285 (±125)	351	49	5	0	964 (±131)	77
	Brown Trout	345 (±181)	351	74	33	5	282 (±62)	23

Table 4. Log-Likelihood Method (LLM) population estimates of trout and Mountain Whitefish (≥ 150 mm) from the Henrys Fork Snake River, Idaho during 2019.

River reach	Species	No. marked	No. captured	No. recaptured	Population Estimate	Confidence Interval (± 95%)	Density (No./km)	Discharge[1] (ft^3/s)
Box Canyon[2]	Rainbow Trout	1,388	900	123	11,326	877	3,061	880
	Mountain Whitefish	194	132	8	3,201	1,743	845	
	Brook Trout	21	2	0	--	--	--	
Stonebridge[3]	Rainbow Trout	629	302	44	4,435	604	964	3,000
	Brown Trout	199	95	24	1,298	286	282	
	Mountain Whitefish	601	336	20	13,757	3,651	2,991	

[1] Represents the mean discharge value between marking and recapture events

[2] Data obtained from USGS gauge (13042500) near Island Park Dam.

[3] Data obtained from USGS gauge (13046000) below Ashton Dam.

Figure 4. Map of the Henrys Fork Snake River watershed, Idaho and location of electrofishing reaches (Box Canyon and Stonebridge) that were sampled during 2019.

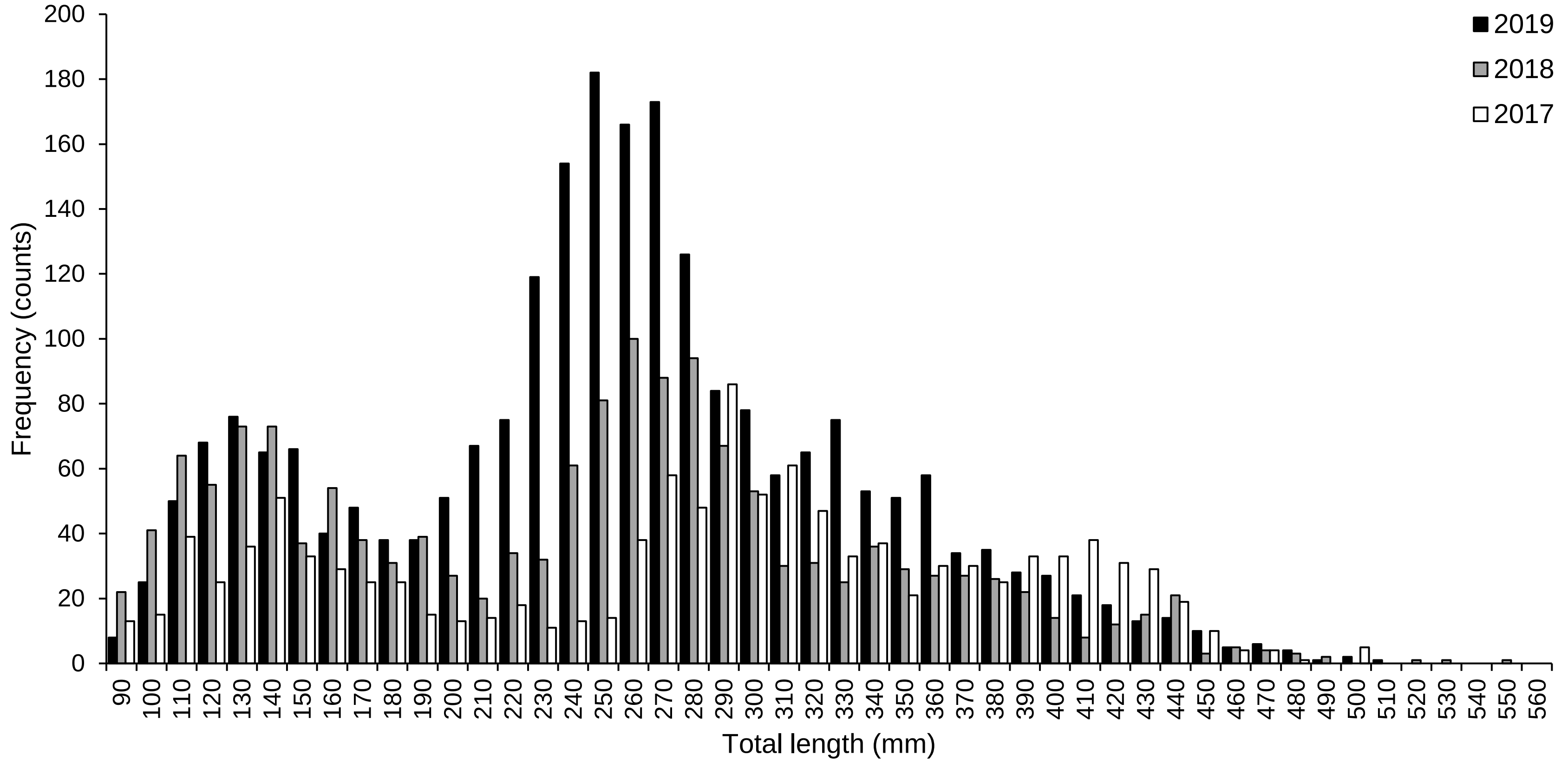

Figure 5. Length-frequency distribution of Rainbow Trout collected by electrofishing in the Box Canyon reach of the Henrys Fork Snake River, Idaho, 2017-2019.

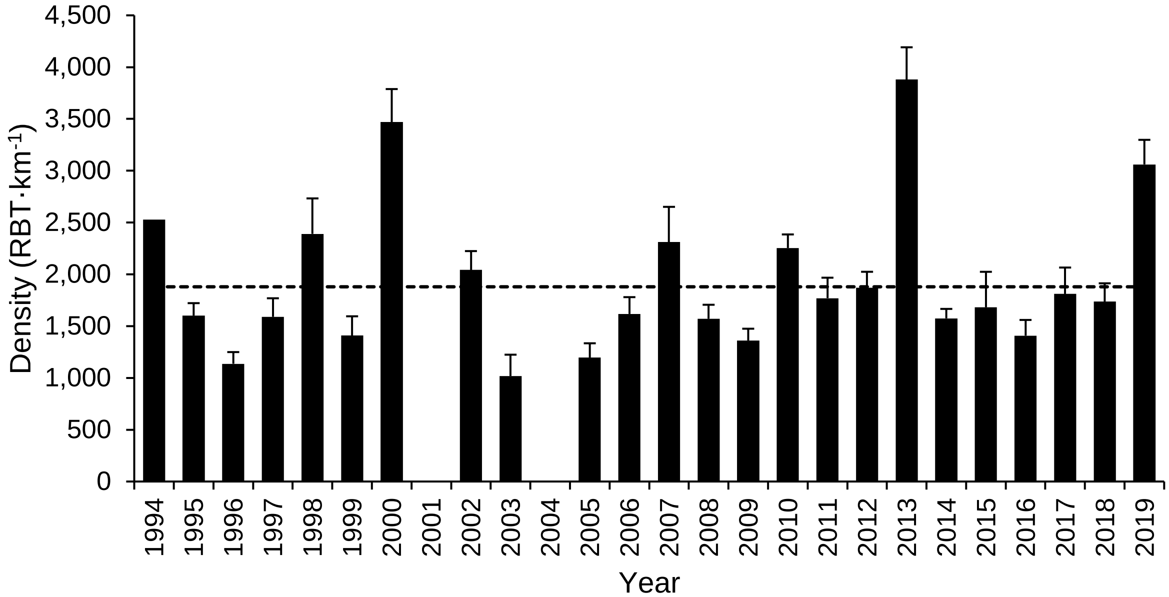

Figure 6. Rainbow Trout densities (fish per km) for the Box Canyon reach of the Henrys Fork Snake River, Idaho 1994 - 2019. Error bars represent 95% confidence intervals. The dashed line represents the long-term average (i.e., 1,880 RBT per km) Rainbow Trout density, excluding the survey from 2019.

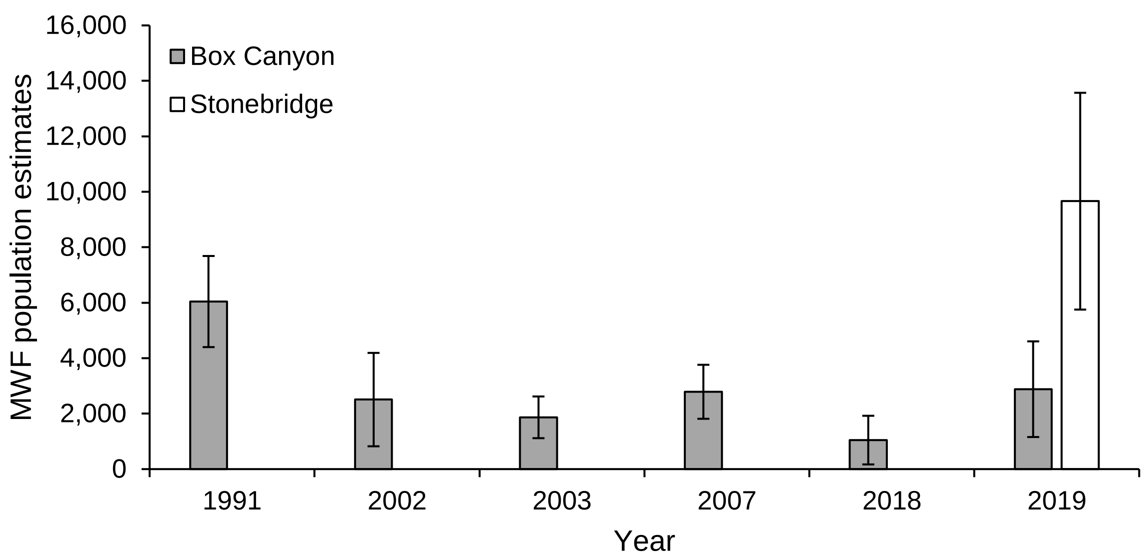

Figure 7. The Mountain Whitefish population estimate for the Box Canyon and Stonebridge reaches of the Henrys Fork Snake River from 1991 to 2019. Population size for the reach was estimated using the Petersen estimate with the Chapman modification and error bars represent 95% confidence intervals.

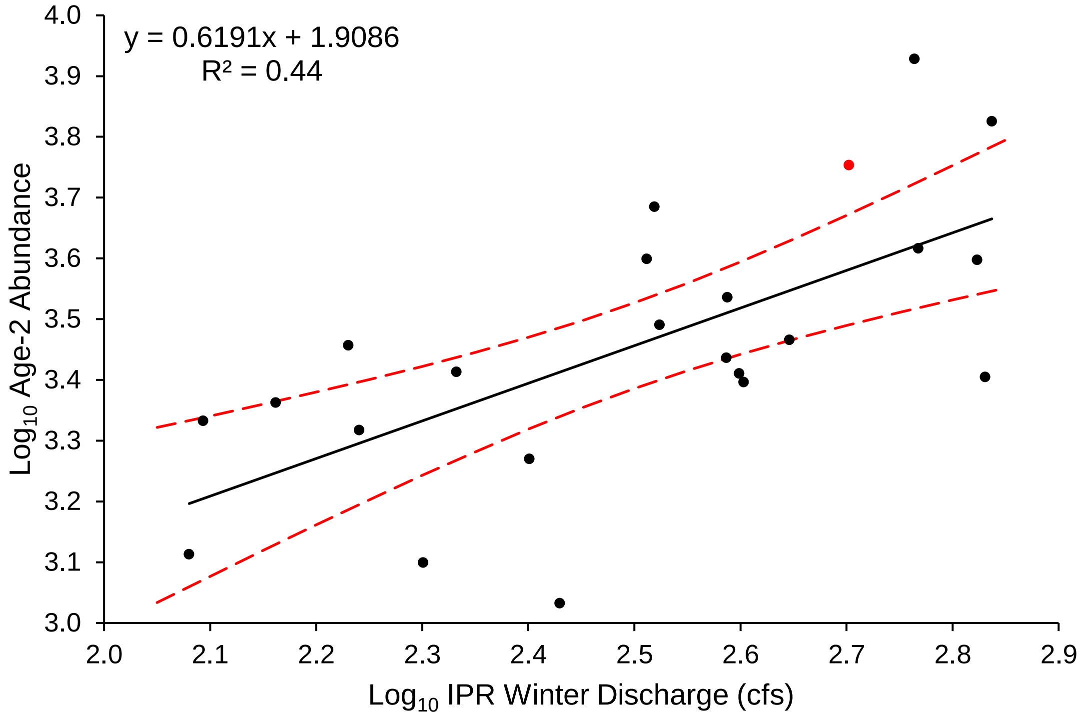

Figure 8. Linear regression (solid black line) and 95% confidence intervals (red dashed lines) of the relationship between age-2 Rainbow Trout abundance and mean winter flow (cubic feet per second; cfs) during the first winter of a fish's life from 1995–2019 (red circle). Log_{10} age-2 Rainbow Trout abundance = 0.6191*log_{10} flow (cfs) + 1.9086, (r^2=0.44, $F(1,21)$=16.5, p <0.01).

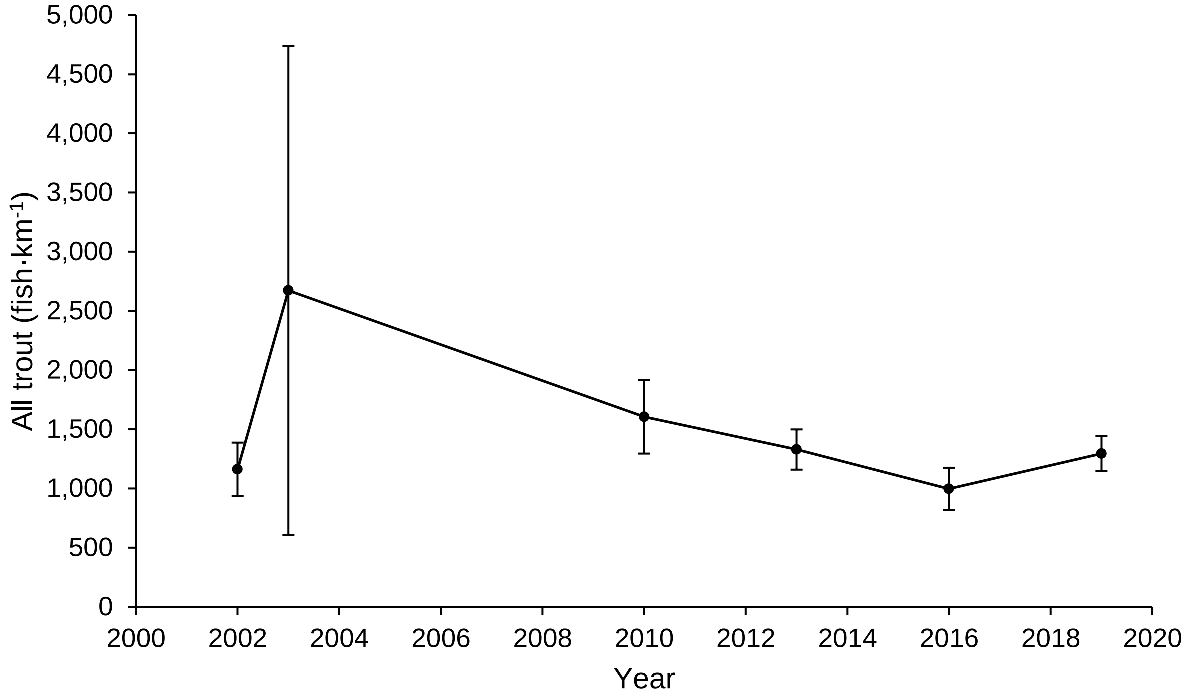

Figure 9. Abundance estimates for all trout captured in the Stonebridge reach of the Henrys Fork Snake River 2002–2019.

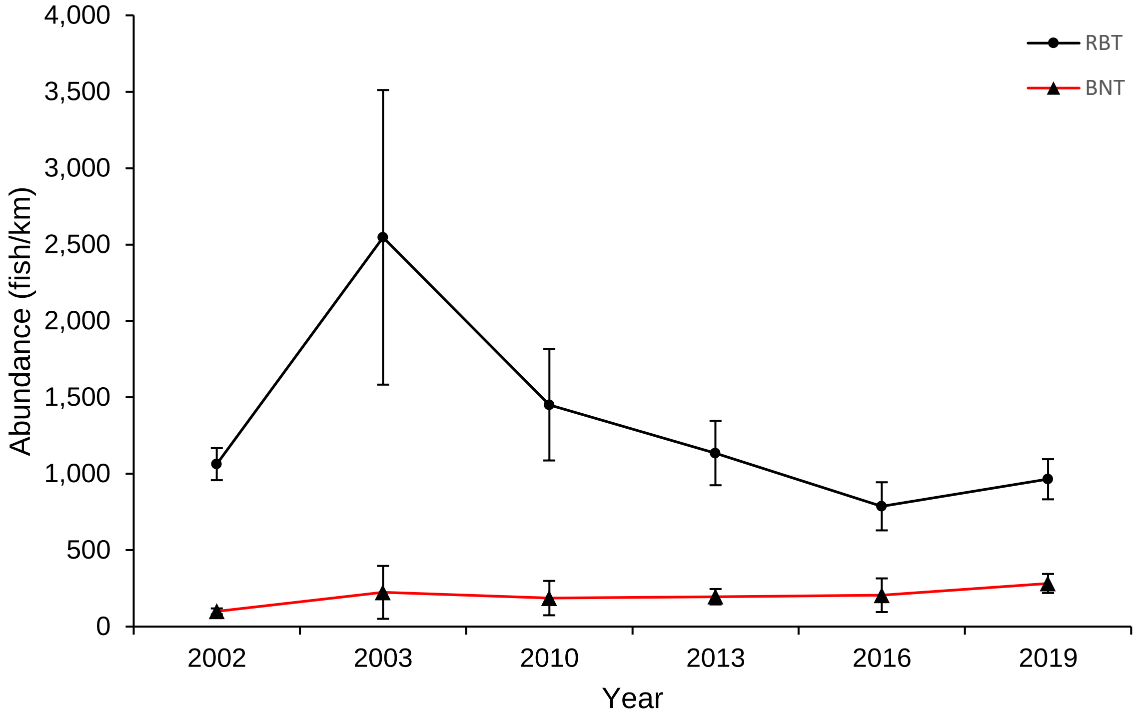

Figure 10. Abundance estimates for RBT (black line) and BNT (red line) in the Stonebridge reach of the Henrys Fork Snake River 2002–2019.

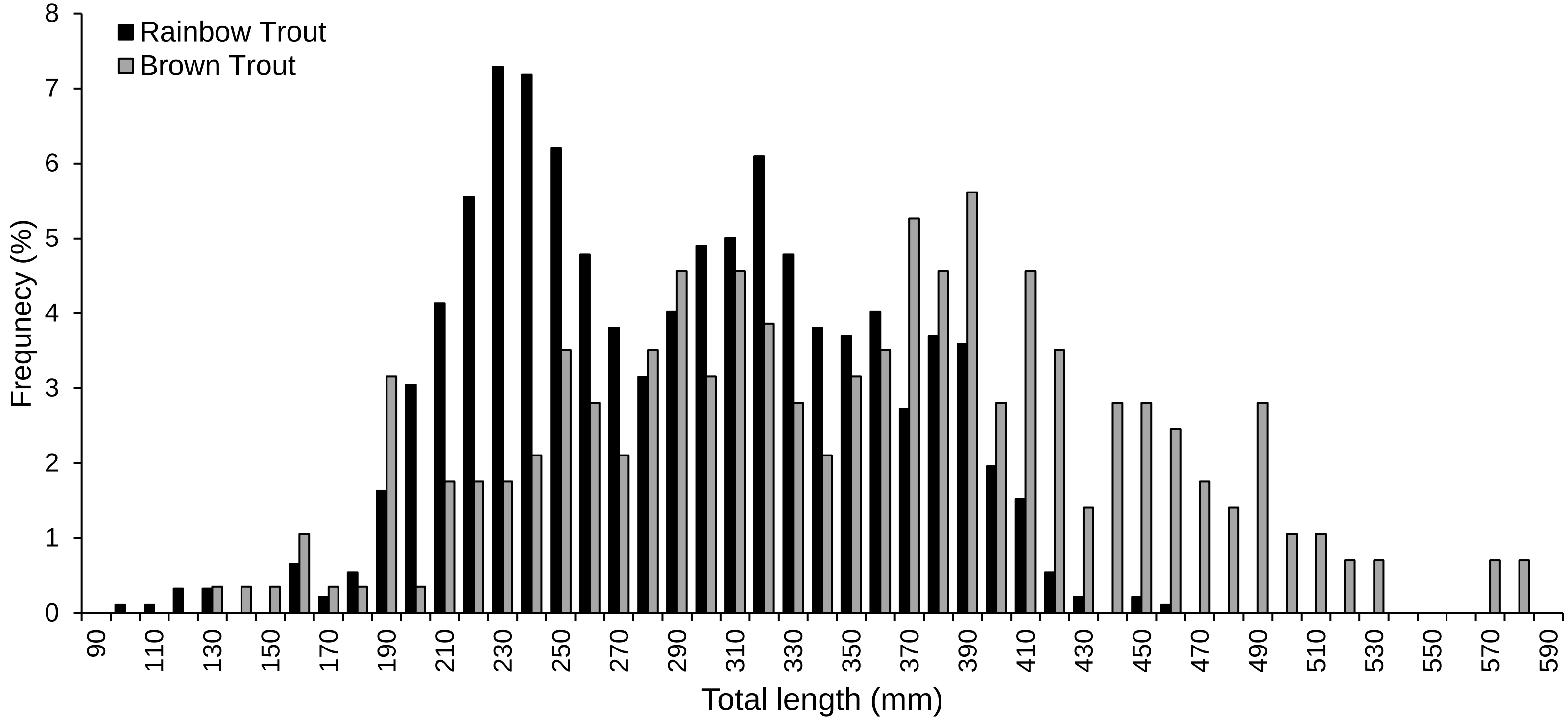

Figure 11. Length-frequency distribution of Rainbow Trout and Brown Trout captured with electrofishing in the Stonebridge reach of the Henrys Fork Snake River during the spring of 2019.

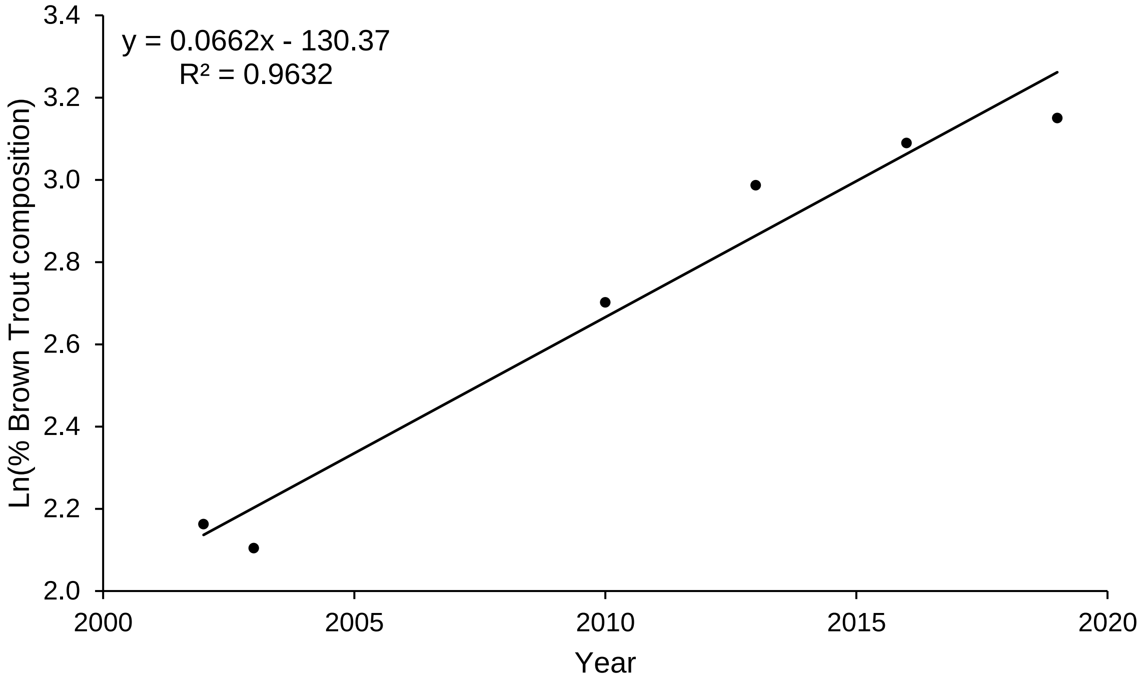

Figure 12. Brown Trout percent composition in the Stonebridge reach of the Henrys Fork Snake River (2002–2019).

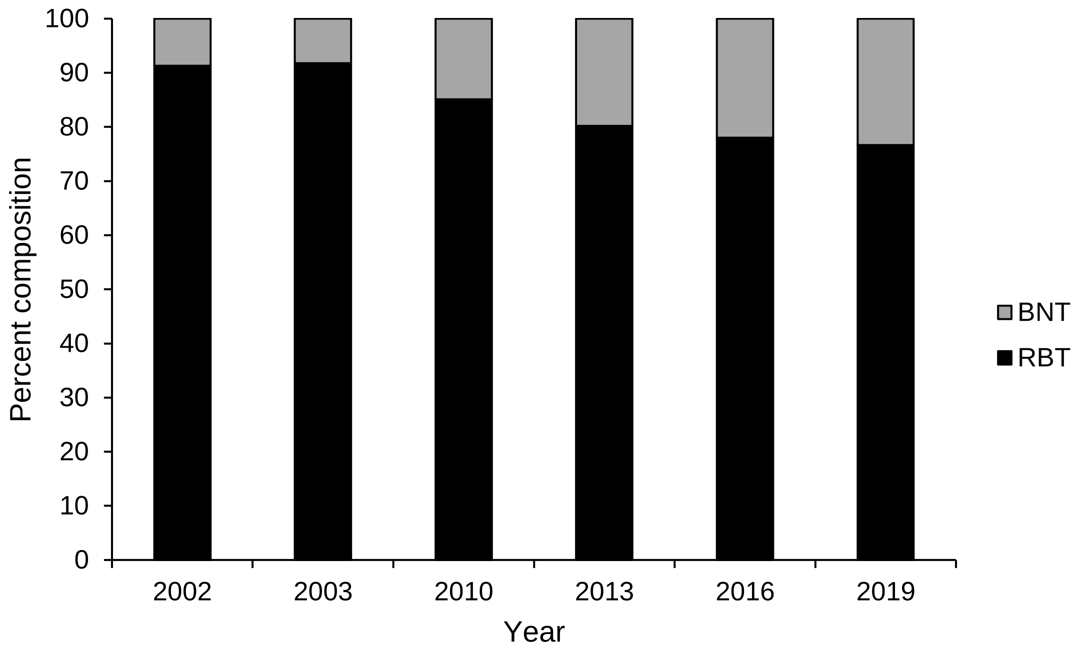

Figure 13. Trout species composition over time (2002–2019) in the Stonebridge reach of the Henrys Fork Snake River

TETON RIVER

ABSTRACT

Abundances of Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (YCT), Rainbow Trout *O. mykiss* (including Cutthroat x Rainbow hybrids; hereafter collectively referred to as RBT) and Brook Trout *Salvelinus fontinalis* (BKT) have been increasing in the Teton River since historic lows in 2003, and now are similar to, or exceed abundances observed in the 1980s when quantitative monitoring efforts were initiated. Trout density estimates (± 95% CI) at the Nickerson monitoring reach equaled 436 YCT/km (± 89), 329 RBT/km (± 55), and 1,003 BKT/km (± 223). Relative weights for each species were near nationwide averages, of 105 for YCT, 99 for RBT, and 99 for Brook Trout. Trends for all three species at Nickerson were significantly positive since 2003. Downstream in the Breckenridge monitoring reach, trout density estimates were 42 YCT/km (± 23), 501 RBT/km (± 63), 420 BKT/km (± 81), and 111 BNT/km (± 217). Relative weights were 93 for YCT, 94 for RBT, 110 for BKT, and 104 for Brown Trout *Salmo trutta* (BNT). Population trends since 2003 were significantly positive for YCT, RBT, and BKT in the Breckenridge monitoring reach. Brown Trout were observed in higher numbers than during surveys conducted prior to 2015. We captured 145 BNT in the Breckenridge monitoring reach, and 53% of these BNT were less than 150 mm in length. The abundances of total trout in the Nickerson monitoring reach and the nonnative trout abundances in the Breckenridge monitoring reach currently exceed historical highs, indicating habitat conditions have changed, angler harvest has changed, or a combination of the two. Despite new record trout abundances, fish conditions (i.e. relative weights) approximated at 100 for each species suggesting that carrying capacity has yet to be exceeded enough to negatively affect fish growth. Continued efforts to conserved native Yellowstone Cutthroat Trout will be required due to threats from increasing abundances of nonnative Rainbow Trout, Brook Trout, and Brown Trout.

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INTRODUCTION

The Teton River, a tributary of the Henrys Fork Snake River in Eastern Idaho, supports a robust population of wild trout including an important population of native Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (YCT). Other trout species present include Rainbow Trout *O. mykiss* (RBT), Brook Trout *Salvelinus fontinalis* (BKT), and Brown Trout *Salmo trutta* (BNT). Since 1987, we have routinely sampled two reaches in the upper Teton River to monitor fish population trends. This report summarizes the 2019 Teton River monitoring surveys. For a broader description of the Teton River fish assemblage and factors contributing to observed trends in abundance and species composition see Schoby et al. (2013).

OBJECTIVES

To obtain current information on fish populations of the Teton River and its tributaries to assess trends, and to develop and implement appropriate management strategies to conserve native species and benefit anglers.

METHODS

We estimated trout abundance by species using mark/recapture techniques at the Nickerson and Breckenridge monitoring reaches in Teton Valley (Figure 14). Electrofishing sampling was conducted using drift boat and raft-mounted gear in the fall when river flows reached base levels of approximately 300 cubic feet per second (cfs). Two electrofishing passes at each reach were completed with approximately one week intervals between passes. We attempted to capture all trout encountered. All fish were measured to the nearest mm (total length), and identified to species. A representative sample of fish for each species was weighed to the nearest gram. During the first pass, fish were marked using a hole punch in the caudal fin, and this mark was used to identify previously captured fish in the subsequent run. At each site, genetic samples were collected from all phenotypically identified YCT and stored on Whatman data sheets. A total of 799 YCT (730 at Nickerson and 69 at Breckenridge), 21 (15 at Nickerson and 6 at Breckenridge) and 72 BNT (all at Breckenridge) were marked with half-duplex Passive Integrated Transponder (PIT) tags as part of an ongoing general movement study in the drainage. In this chapter, all RBT include Rainbow Trout and Rainbow X Cutthroat Trout hybrids. During the second pass, captured fish were again measured, identified to species, and inspected for caudal fin marks. After calculating population estimates for each species as described in Schoby et al. (2013), we assessed population trends at the Nickerson and Breckenridge sites independently. Abundance estimates since 1995 were incorporated into an exponential model that examined the intrinsic rate of population change (r) as explained by Maxell (1999) using $\alpha = 0.10$.

Relative weights (W_r) were calculated by dividing the actual weight of each fish (in grams) 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 formula:

$$\log W_s = -5.192 + 3.086 \log TL$$

to calculate relative weights of Yellowstone Cutthroat Trout (Kruse and Hubert 1997),

$$\log W_s = -5.023 + 3.024 \log TL$$

for Rainbow Trout (Simpkins and Hubert 1996),

$$\log W_s = -5.186 + 3.103 \log TL$$

for Brook Trout (Hyatt and Hubert 2001), and

$$\log W_s = -4.867 + 2.96 \log TL$$

for Brown Trout (Milewski and Brown 1994). We compared relative weights among size groups using 95% confidence intervals.

South Fork Teton Trap

A single cement fish ladder was installed in the South Fork Teton River on a large irrigation diversion which flows into Moody Creek by the Idaho Department of Fish and Game in 1994 to facilitate upstream fish passage. A fish trap was also installed at this location to quantify passage of 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 April 1st 2019, with the first sampling occurring on April 3rd, 2019. The trap was maintained and checked every 2-3 days until the trap was pulled on June 5th, 2019. During each visit, the trap was checked and cleaned. All fish captured were identified to species, measured to the nearest mm (TL), determined by sex (if possible), and passed upstream. A subsample of YCT, RBT, and Bluehead sucker were PIT-tagged. A tissue sample was taken from every YCT encountered for genetic analysis.

The left leading pectoral fin ray was removed from a subsection of Bluehead and Utah Sucker captured by cutting as close to the pelvic girdle as possible (Koch et al. 2008). After drying, fin rays were embedded in epoxy in centrifuge tubes and a thin section (~0.3 mm thick) was cut from the base of the pectoral fin ray with a low-speed saw (Koch and Quist 2007).

RESULTS

We sampled the Nickerson monitoring reach on September 11 and 18, 2019 and captured a total of 2,406 trout. We captured 744 YCT, 462 RBT, 1,198 BKT, and 2 BNT (Figure 15). Of these, captured trout weights were taken from 301 YCT, 260 RBT, and 407 BKT. We estimated trout densities (± 95% CI) of 436 YCT/km (± 89), 329 RBT/km (± 55), and 1,003 BKT/km (± 223; Figure 16). The mean total length and relative weight of YCT were 234 mm (Figure 15) and 105 (Figure 17) respectively. For RBT, the mean total length 235 mm and average relative weight were 235 mm and 99 (Figure 17) respectively. The mean total length and relative weight for Brook Trout were193 mm and 99 respectively. Population trends for YCT at Nickerson since 1987 have been stable (r = 0.02, F = 0.45, df = 14, P = 0.51). However, since 2003 YCT abundance has experienced significant increases (r = 0.24, F = 36.45, df = 8, P < 0.001). Rainbow Trout abundance at Nickerson has significantly increased throughout the entire dataset time frame of 1991 to 2019 (r = 0.1, F = 13.84, df = 11, P = 0.004) as well as post fish stocking 2003 to 2019 (r = 0.11, F = 7.19, df = 8, P = 0.03). Brook Trout were similar to RBT at Nickerson, with significant positive abundance trends for both 1991 to 2019 (r = 0.11, F = 39.63, df = 12, P < 0.001) and 2003 to 2019 (r = 0.14, F = 30.5, df = 8, P <0.001; Figure 15).

We sampled the Breckenridge monitoring reach on September 12 and 19, 2019 and captured a total of 1,847 trout. This included 77 YCT, 628 RBT, 997 BKT, and 145 BNT (Figure 18). Weights were obtained from 53 YCT, 162 RBT, 245 BKT, and 140 BNT. We estimated trout densities (± 95% CI) of 42 YCT/km (± 23), 501 RBT/km (± 63), 420 BKT/km (± 81), and 111 BNT/km (± 217; Figure 19). The mean total length and relative weight of YCT at Breckenridge were 255 mm and 93 respectively. The mean total length and average relative weight of RBT were 255 mm and 94 respectively. Brook Trout (BKT) mean total length and average relative weight (Figure 20) were 161 mm and 110 respectively. Brown Trout were observed in multiple size groups with 145 BNT ranging in size from 72 to 576 mm in length and had an average relative weight of 104. Roughly half of the BNT captured were less than 150 mm in length (Figure 18). Since 1987 the intrinsic rate of population growth, *r*, for YCT has been slightly negative (*r* = -0.02), but *r* was not significant indicating a stable trend (F = 0.73, df = 12, P = 0.41). Since low trout abundances observed in 2003, YCT have significantly increased at Breckenridge (*r* = 0.13, F = 24.87, df = 7, P = 0.002). Rainbow Trout abundance at Breckenridge has significantly increased over the duration of the dataset (*r* = 0.06, F = 7.76, df = 13, P = 0.02) as well as since fish populations dropped in 2003 (*r* = 0.06, F = 3.02, df = 8, P = 0.13). Brook Trout trends at Breckenridge were similar to YCT in that their abundance has been stable over the range of years estimates were available, or from 1997 through 2017 with an intrinsic rate of population change (*r* = 0.13) that was not significantly different than 0 (F = 5.37, df = 7, P = 0.06), but since 2003, their abundance has increased significantly (r = 0.25, F = 30.57, df = 6, P = 0.003).

South Fork Teton Trap

A total of 1,060 fish comprised of 982 Utah Sucker, 54 YCT, 15 Bluehead Sucker, 4 BNT, 2 RBT, 1 Mountain Whitefish (MWF), 1 Utah Chub and 1 Speckled Dace were captured in the South Fork Teton fish trap. A total of 46 YCT, 1 RBT, and 9 Bluehead Sucker were PIT-tagged. Overall relative abundance of the season catch at the trap was dominated by Utah Sucker (93%). YCT, and Bluehead Sucker composed 5.1% and 1.4% of the catch respectively with <1% of the catch represented each by BNT, RBT, MWF, Utah Chub, and Speckled Dace. Utah sucker were first captured on April 20th, 2019 with the largest quantity of the season. Four subsequent capture peaks were further observed over the remaining trapping period (Figure 21). The largest, single-day catch of YCT took place on April 20th (43%; Figure 22). YCT average length at the trap was 311 mm and ranged from 215 to 585 mm (Figure 23).

DISCUSSION

Although trout species in the Nickerson monitoring reach of the Teton Valley have declined slightly from the last survey year of 2017, all trout abundances are higher than their respective 10-year averages and exceed abundances of the 1980s to early 2000s. Specifically for YCT, the current abundance (436 YCT/km) was in stark contrast to the lowest abundance estimated during the last 30 years (i.e. 9 YCT/km during the 2003 survey).

Nonnative trout (RBT and BKT) abundances in the Breckenridge monitoring reach have been steadily increasing since the all-time low in 2003, peaking in 2013 while YCT abundance has remained relatively steady. Additionally, following our survey in 2013, BNT have increased significantly throughout the Breckenridge monitoring reach. Captured BNT exhibited a wide range of size classes (72 to 576 mm), with 28% of the BNT sampled were under 150 mm. In addition, over half of the captured BKT (56.8%) exhibited a length of under 150 mm. This indicates conditions in the Teton River Valley allow for growth, reproduction, spawn, and survival of BNT.

The continued increase of these nonnative trout is a shift in the species composition, and likely indicates changing conditions in the Teton River Valley.

A variety of factors may be currently affecting fish populations in the Teton River valley including harvest (IDFG 2007), habitat alterations; including livestock and land development (Koenig 2006), water diversions, and stream flows (Van Kirk and Jenkins 2005). A no harvest of cutthroat trout rule was implemented in 2006 in response to the substantial decrease in trout abundances and is still in effect today (IDFG 2007). This indicates that harvest pressure has likely had little effect on trout populations in the Teton river over the last few years. The most recent creel survey on the Teton River also concluded that harvest rates have not increased substantially since the early 2000s (IDFG R6 Report 2016 *in progress*). Since 2003, Friends of the Teton River (FTR) have worked with land owners to complete 45 stream restoration projects (FTR unpublished data). These projects have likely had a positive affect and most likely to explain the notable increases in the abundance of YCT as well as nonnative trout.

The Teton Valley has experienced two back to back high water years (2018 and 2019). Due to irrigation diversions and the natural hydrology of the valley, many tributaries run dry during the summer and are disconnected seasonally from the main stem of the river (Van Kirk and Jenkins 2005). High water years allow tributaries to be connected to the main stem of the river later into the season, increasing the connectivity to allow access to prime habitat thus facilitating adult spawning migrations for adult fish and outmigration for YCT fry to the main stem of the river. We believe this factor strongly contributes to the increased density estimates.

Yellowstone Cutthroat Trout population densities in Teton Valley have differed by river reach since monitoring began in 1987. Currently, YCT densities are ten times higher in the Nickerson monitoring reach than the Breckenridge monitoring reach. This discrepancy exists despite telemetry data showing YCT in the entire Teton Valley section of the Teton River behave as a single population (Schrader and Jones 2004). The mechanisms causing this discrepancy between river reaches are largely unknown. The presence and subsequent increase in the population of Brown Trout in the Breckenridge reach may be contributing, although this discrepancy has been apparent before the documentation of Brown Trout in the reach. Instead, it is likely that a larger population of YCT exhibit this reach due to the close proximity to the major spawning tributaries Teton Creek and Fox Creek (Schrader and Jones 2004; Koenig 2006). Teton Creek is located at the upper boundary of the Nickerson reach and Fox Creek is located approximately 5.4 km farther upstream. In addition, many of the habitat restoration projects implemented by IDFG and FTR have been implemented in the watershed directly upstream of the Nickerson monitoring reach (FTR unpublished data). Further investigations on the factors driving YCT abundances in Teton Valley should be investigated to align with the objectives listed in the Idaho Fisheries Management Plan (IDFG 2019).

Although nonnative trout are popular sport fish to Idaho anglers, the negative effects of their introduction to native trout raise notable concerns about the future of this fishery. Low numbers of Brown Trout (< 7 fish) were captured in our surveys from 2007 to 2013. Since 2013 capture numbers have increased with 7, 28, 32 BNT in the 2013, 2015, and 2017 surveys respectively. The substantial increase to 145 BNT captured in the 2019 Breckenridge survey, coupled with the presence of multiple size classes including many fingerlings (<100 mm), and relative weights over 100 indicates that this species is successfully reproducing and growing in the river. This is also true for Rainbow and Brook Trout in the Breckenridge survey. Both species exhibited a large Age-0 size class with high relative weights.

Brown and Brook Trout have been known to displace native trout species through aggressive behavior, predation and competition for food and space throughout western North America (Buddy and Gaeta 2018; Dunham et al. 2002). Furthermore, a study on a native YCT stream in Montana showed reduced growth rates, juvenile recruitment and survival of YCT with the invasion of Brown Trout into the stream (Al-Chokhacy and Sepulved 2019). Bonneville Cutthroat Trout in the presence of Brown Tout have been found to cause changes in diet, and 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 in the Teton River.

MANAGEMENT RECOMMENDATIONS

1. Continue to monitor the abundance and trends of trout in the Teton River

2. Continue to operate the South Fork Teton fish ladder to mitigate adult fish passage and monitor fish populations

3. Monitor genetic purity of YCT to evaluate introgression with RBT.

4. Conduct YCT population surveys on the major YCT spawning tributaries of the Teton River including Fox and Teton creeks.

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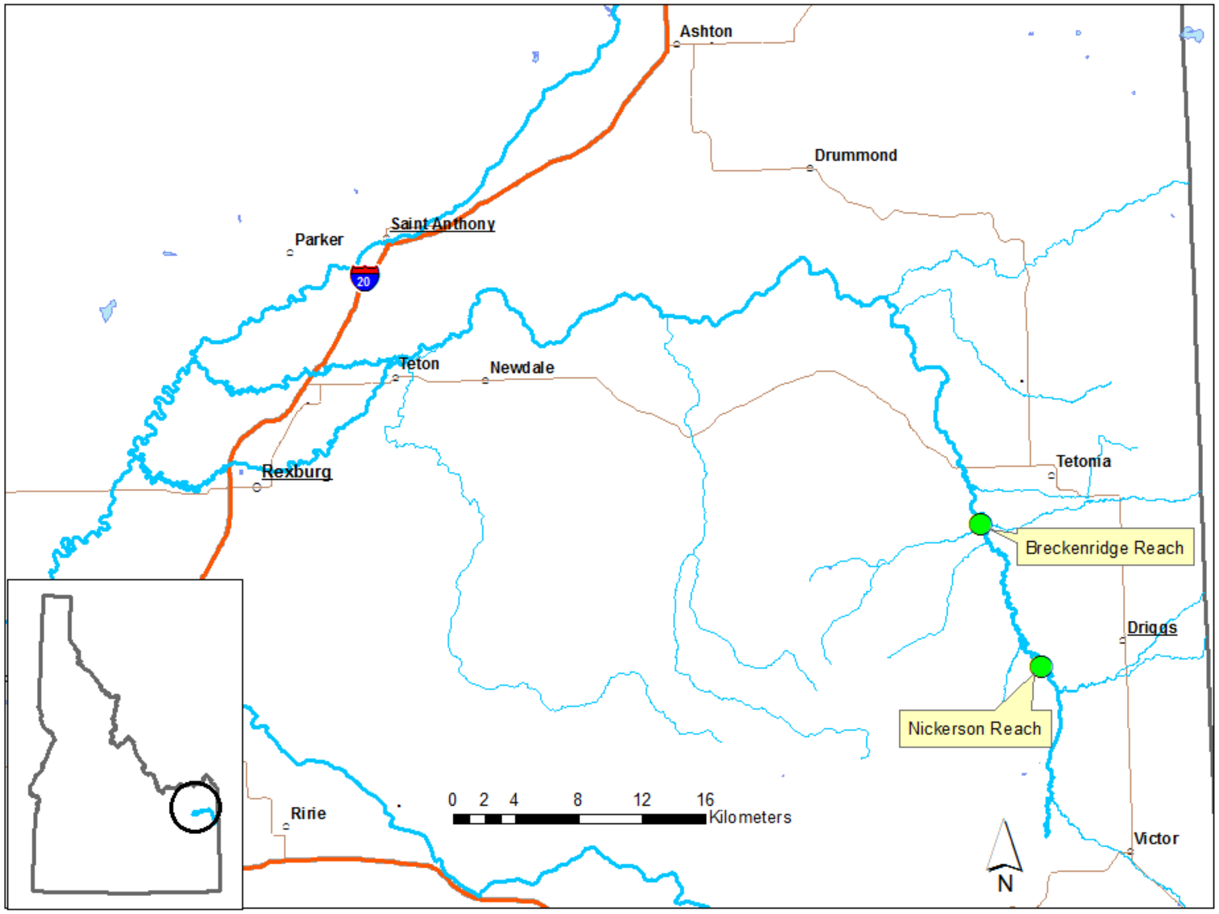

Figure 14. Electrofishing reaches sampled in the Teton River in 2019.

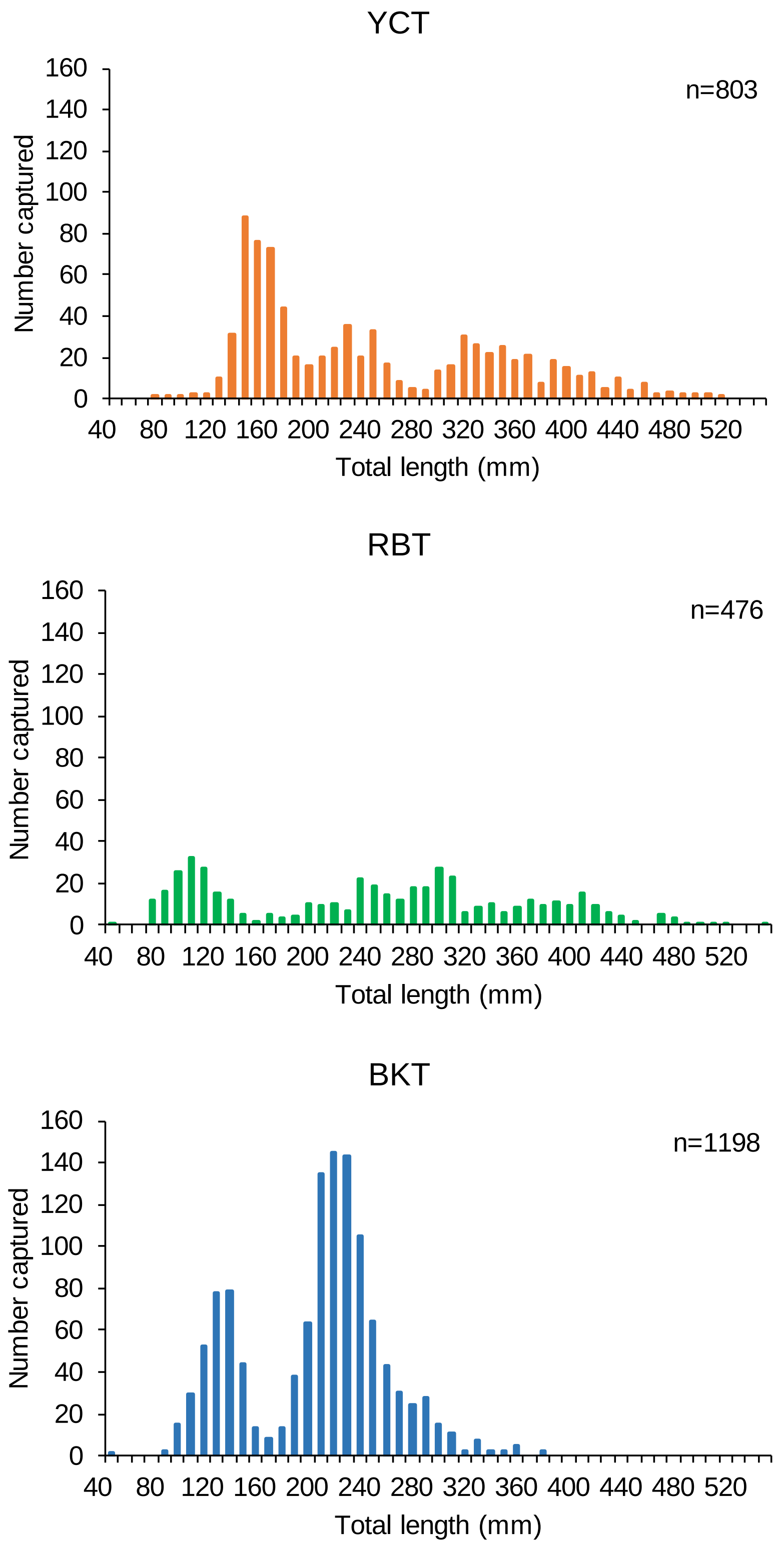

Figure 15. Length-frequency distribution for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brook Trout (BKT) at the Nickerson monitoring reach of the Teton River, 2019.

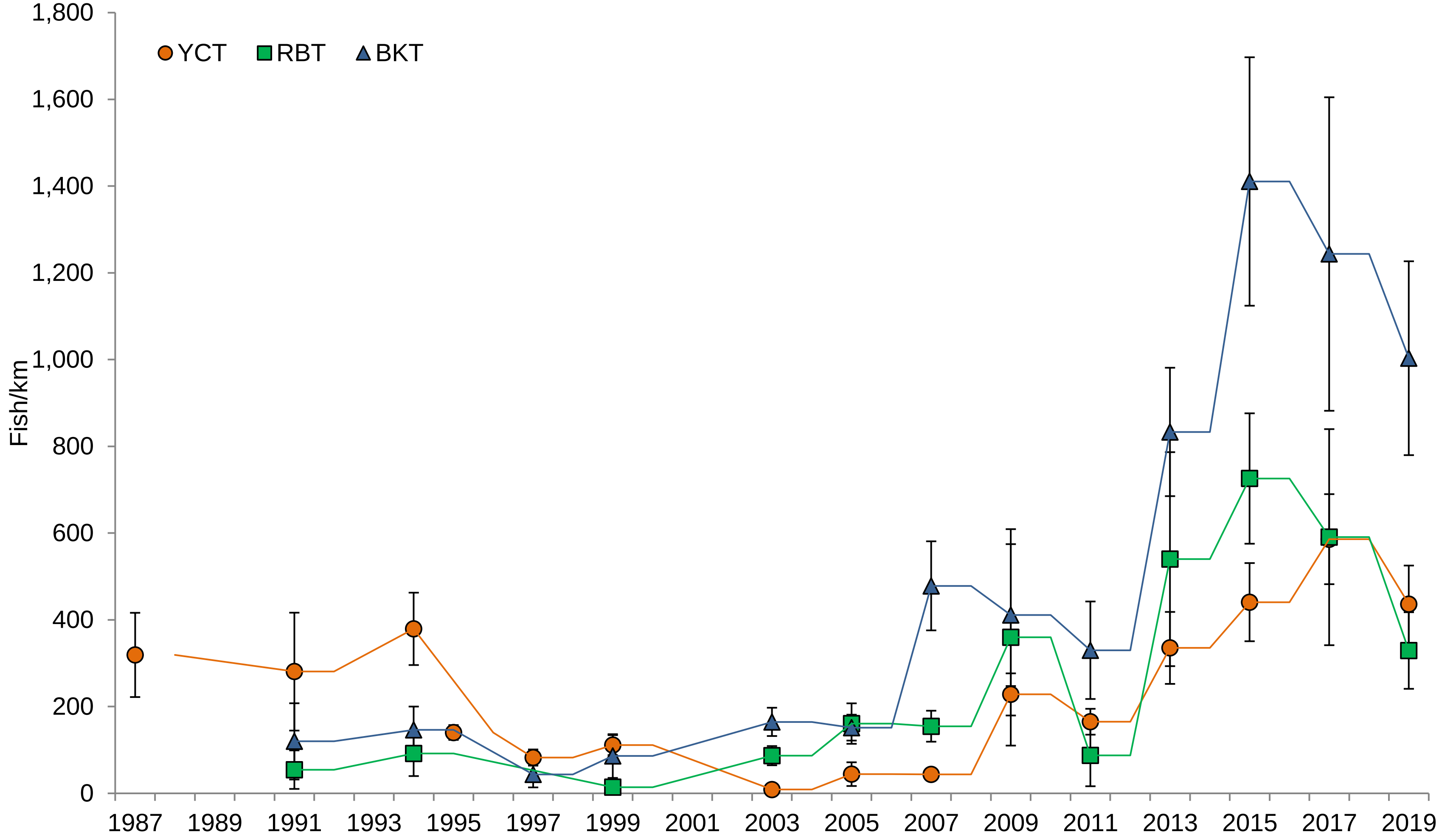

Figure 16. Estimated abundance (fish/km) with 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brook Trout (BKT) in the Teton River at the Nickerson monitoring reach from 1987 through 2019.

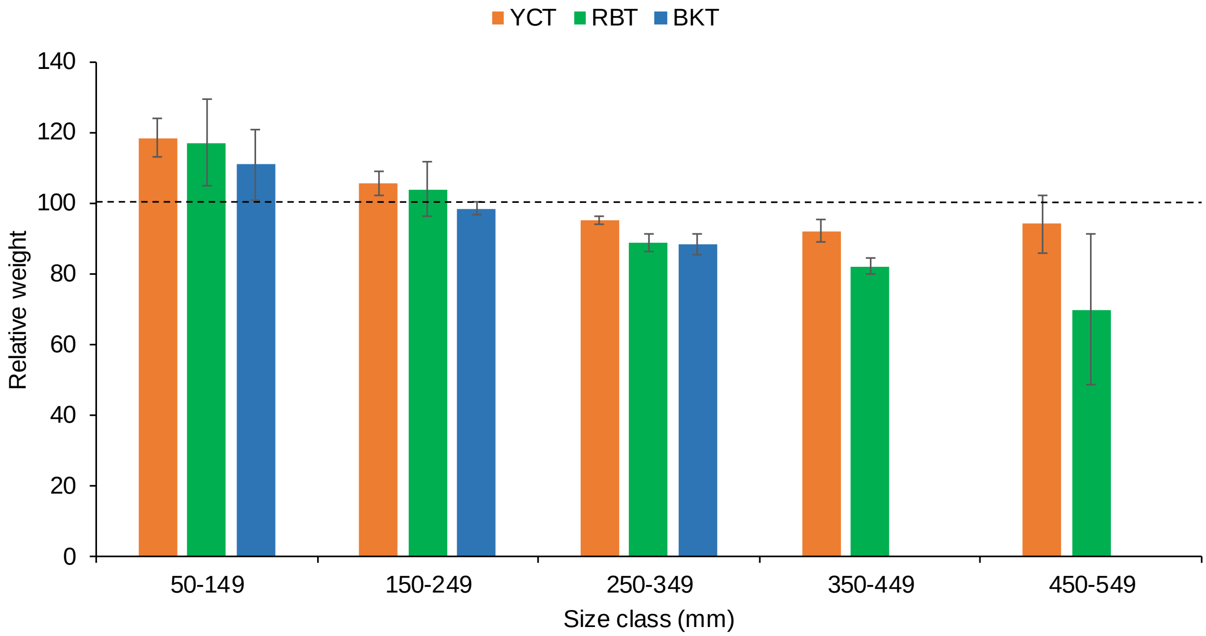

Figure 17. Mean relative weight and 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brook Trout (BKT) in the Teton River at the Nickerson monitoring reach in 2019.

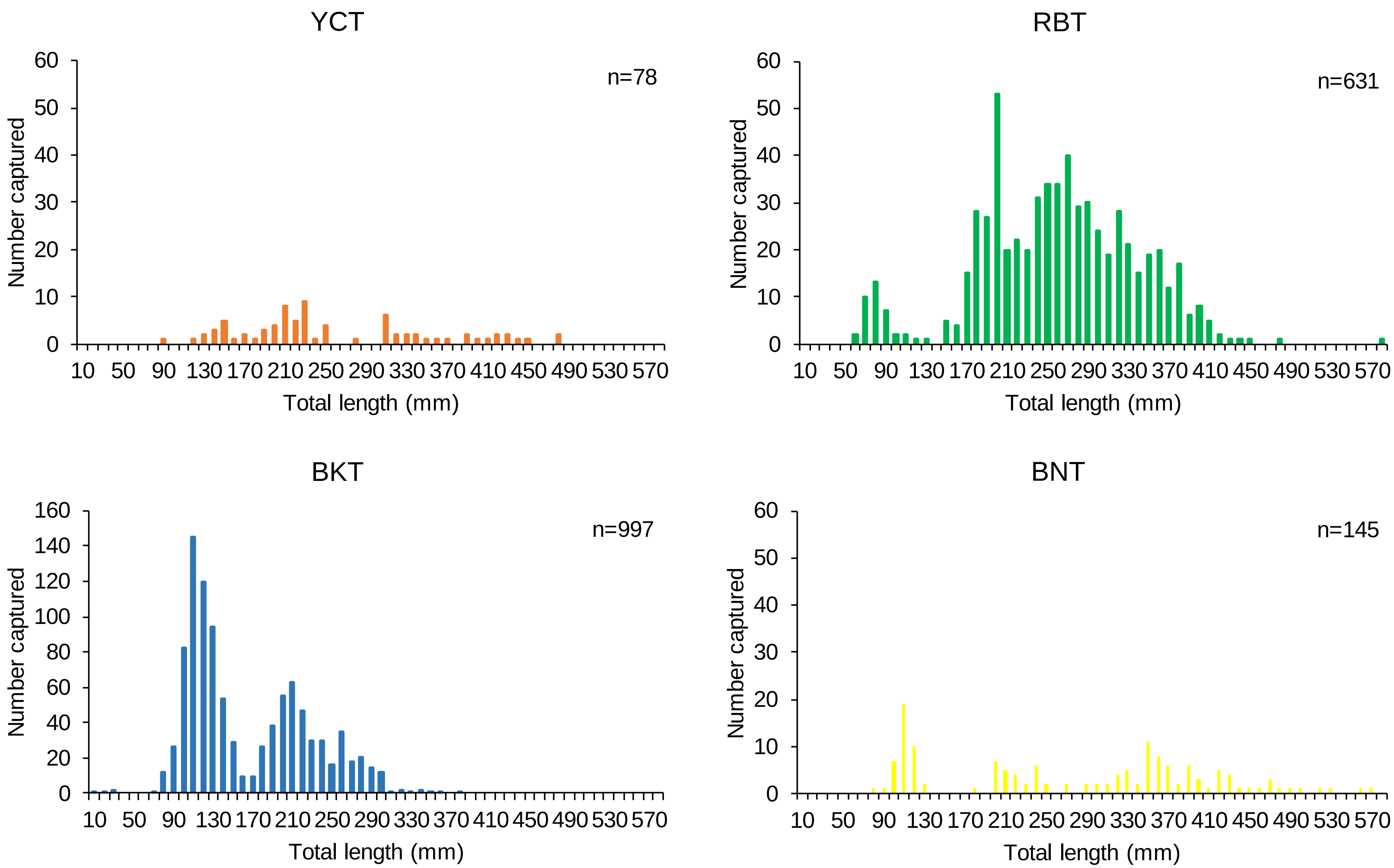

Figure 18. Length-frequency distribution for Brook Trout (BKT), Rainbow Trout (RBT), Yellowstone Cutthroat Trout (YCT), and Brown Trout (BNT), at the Breckenridge monitoring reach of the Teton River, 2019.

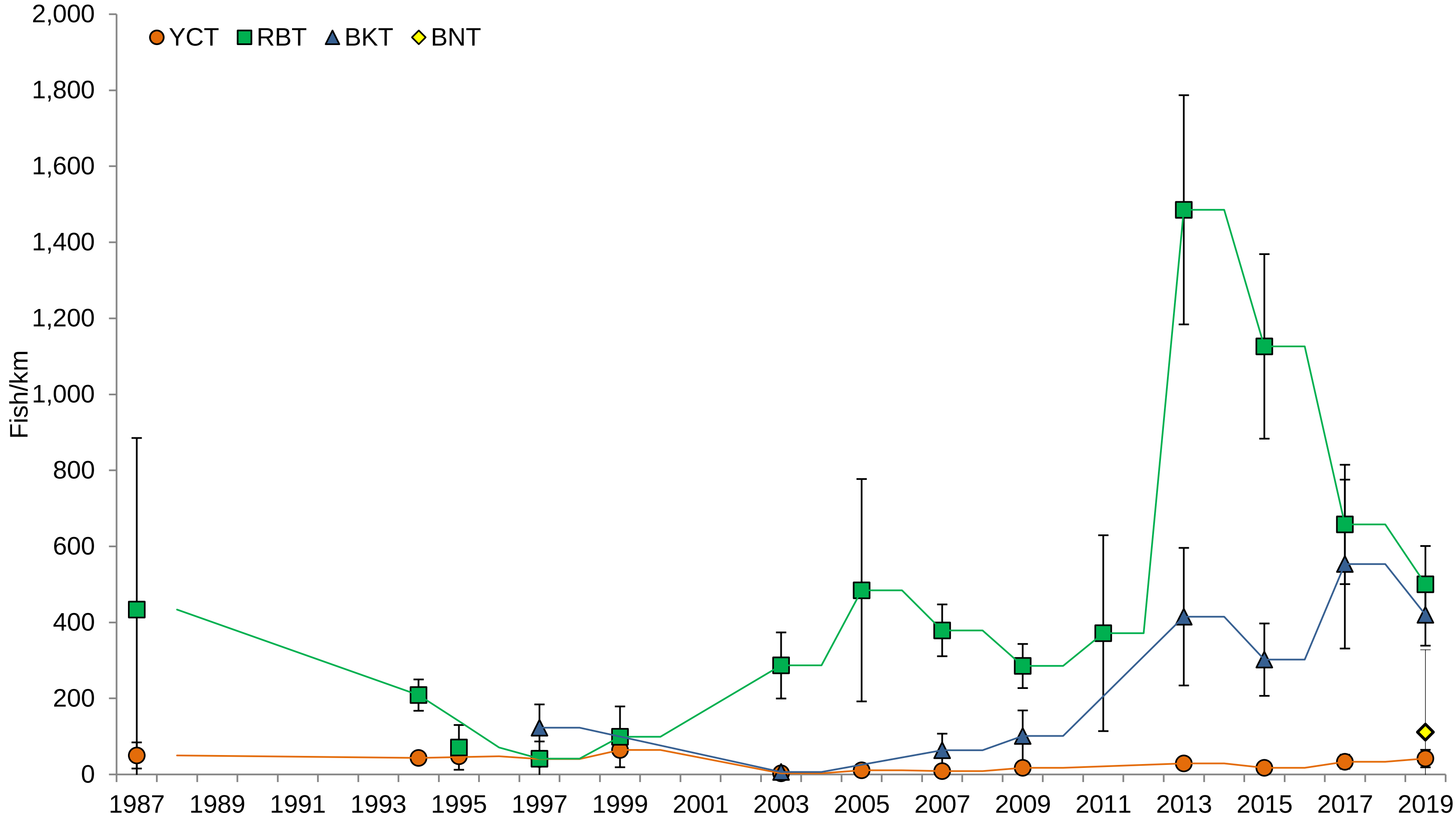

Figure 19. Estimated abundance (fish/km) with 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), Brook Trout (BKT), and Brown Trout (BNT) in the Teton River at the Breckenridge monitoring reach from 1987 through 2019.

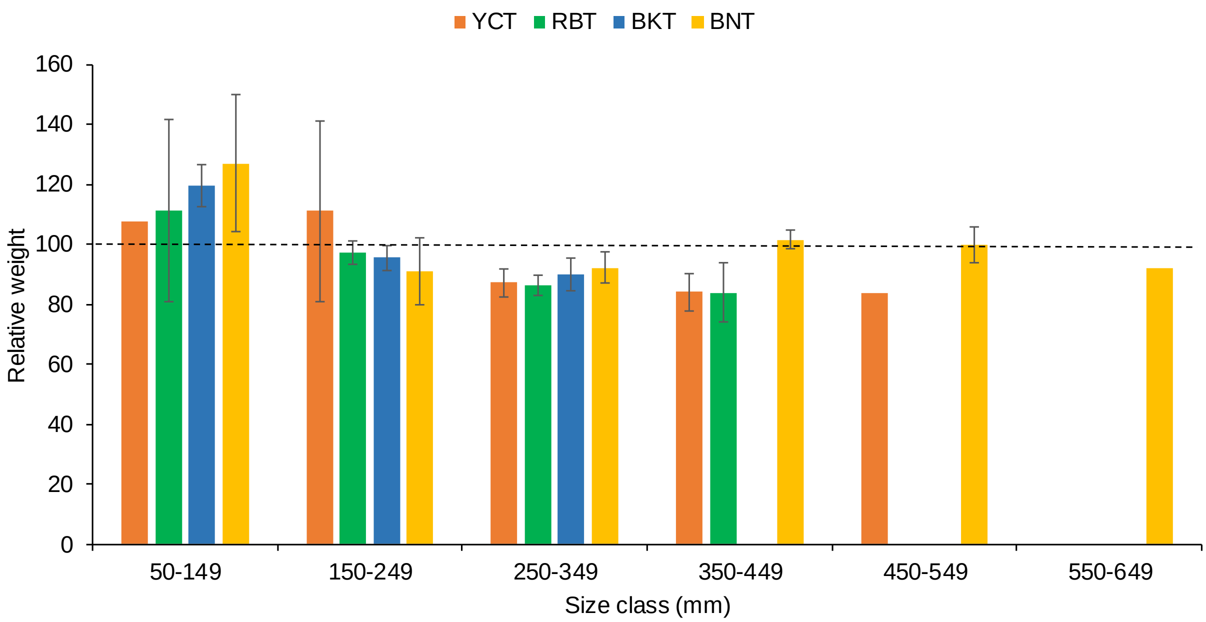

Figure 20. Mean relative weight and 95% confidence intervals of Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), Brook Trout (BKT), and Brown Trout (BNT) in the Teton River at the Breckenridge monitoring reach in 2019.

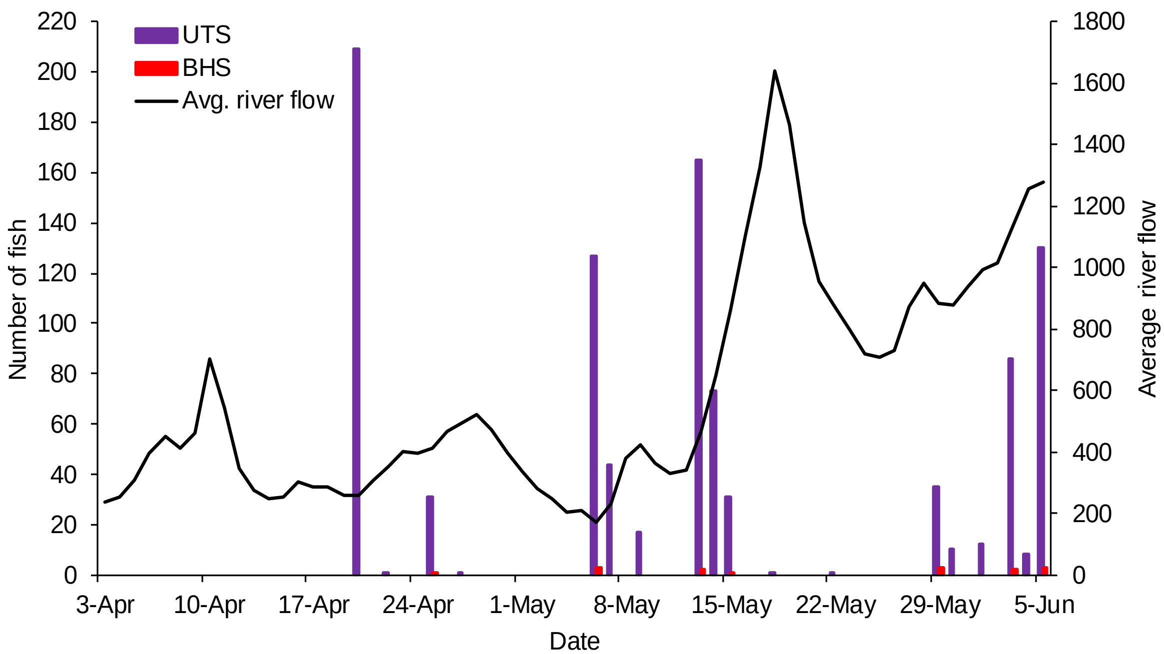

Figure 21. Number of Utah Sucker (UTS) and Bluehead Sucker (BHS) captured in the South Fork Teton Fish trap, 2019. The solid black line represents the daily average stream flow (cubic feet per second) for the South Fork of the Teton River at the USGS SF Teton River gauge.

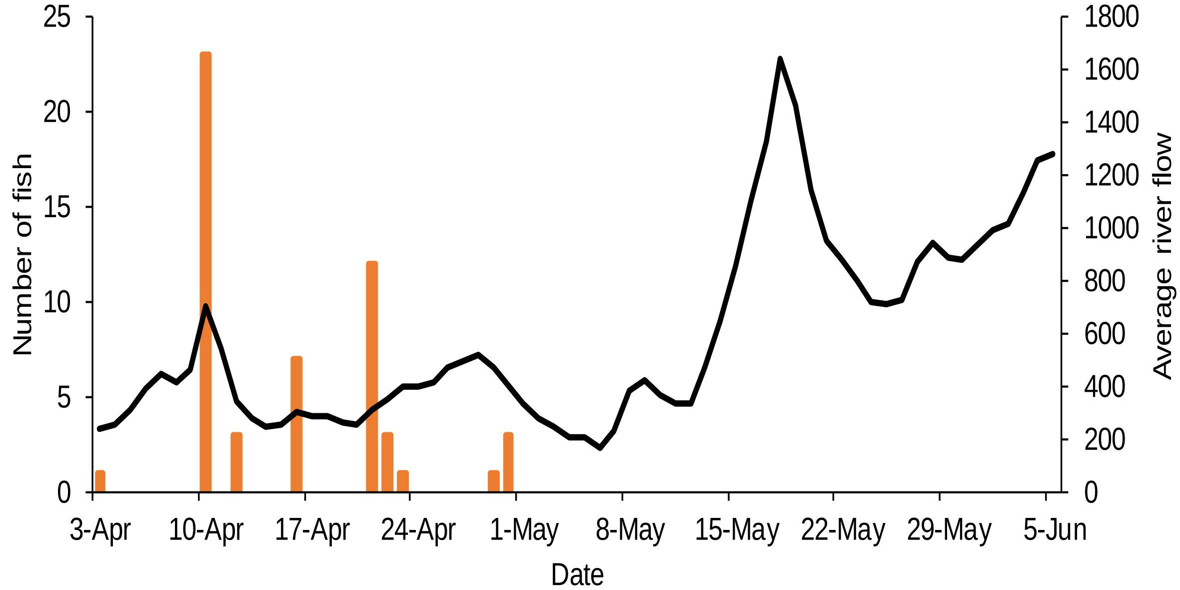

Figure 22. Number of Yellowstone Cutthroat Trout captured in the South Fork Teton Fish trap, 2019. The solid black line represents the daily average stream flow (cubic feet per second) for the South Fork of the Teton River at the USGS SF Teton River gauge.

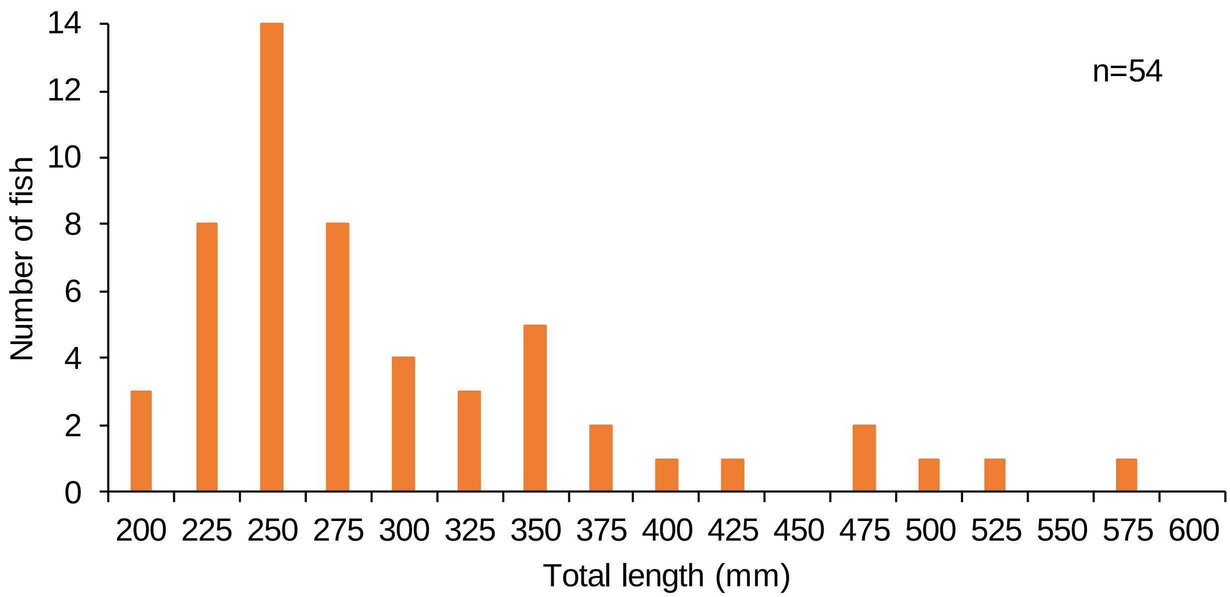

Figure 23. Length frequency distribution of Yellowstone Cutthroat Trout at the South Fork Teton Fish trap, 2019

BIG LOST RIVER

ABSTRACT

The Upper Big Lost River Fishery is supported by populations of wild trout and hatchery trout stocked primarily as catchables. Creel check stations and a “Tag You’re It” study were conducted simultaneously throughout the upper Big Lost River basin in 2019 to collect data on angler effort, angler use, catch rates, and harvest. The impetus for these studies originated from concern of declining wild trout abundances at standard sites that are sampled on a five-year cycle in the upper Big Lost River basin, and from the growing concerns of anglers that overharvest is an issue. Check station personnel interviewed 222 anglers from May through October. Angler effort was estimated at 11,775 h (± 4,975; 95% CI) and the highest amount of effort occurred in the Upper Big Lost River (i.e., 5,470 hours). Fly anglers contributed 79% of total effort, followed by bait anglers at 12%, and lure anglers at 9%. Data collected from creel suggests that catch rates were 1.4 fish/h, and we estimated that anglers caught 16,408 fish (± 8,874), all species combined. Based on creel, the estimated harvest rate (i.e., the number of fish reported as harvested out of the total number of fish reported as caught) for all anglers was 8%, with bait anglers reporting 38% harvest, lure anglers at 13%, and fly anglers at 1%. The average reporting rate of tagged hatchery trout in the upper Big Lost basin was 28% (± 17%; SD). Total angler use, which included fish harvested and caught-and-released, was 22% and ranged from 5% to 40% among four tributaries where fish were tagged and stocked, with the highest amount of use in the East Fork Big Lost River (i.e., 40%). Estimated exploitation rate (i.e., from tag returns of all catchable hatchery trout tagged in 2019) was 10% and ranged from 1% to 34% among four tributaries. Total angler use for hatchery-origin Rainbow Trout *Oncorhynchus mykiss* (RBT) was 29% and for Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (YCT) was 18%. Exploitation rate for hatchery-origin RBT was 21% and exploitation rate for hatchery-origin YCT was 4%. Although estimates of harvest from creel and fish tagging were similar, our creel estimate of harvest included both wild and hatchery trout, whereas our estimate of exploitation included only hatchery trout. Our results suggest that creel check stations can be used as an alternative to roving creel for estimating angler effort and harvest. In addition, results from check stations and fish tagging suggest that angler use and harvest varied by species and location. The most commonly-used gear type has changed from bait angling in 1986 to fly angling in 2019, and harvest has declined considerably since 1986. Creel surveys conducted in 1986 estimated harvest rates at 49% and at 19% in 2007, which are higher than 2019 (i.e. 8%).

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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 24). 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 one of the most popular recreational activities in the area (Corsi 1989).

Numerous gamefish species are present in the watershed, including; Rainbow Trout *Oncorhynchus mykiss* (RBT), Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (YCT), Brook Trout *Salvelinus fontinalis*, Golden Trout *Oncorhynchus mykiss aquabonita*, Tiger Trout *Salmo trutta* × *Salvelinus fontinalis*, Arctic Grayling *Thymallus arcticus*, kokanee *Oncorhynchus nerka*, and Mountain Whitefish *Prosopium williamsoni* (MWF). Sculpin, including Piute *Cottus beldingi* and Shorthead Sculpin *C. confusus* 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 fish from other MWF present in the Pacific Northwest (Whiteley et al. 2006). Populations of MWF have declined in abundance 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), nonnative 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 Mountain Whitefish Conservation and Management Plan for the Big Lost River Drainage, Idaho (IDFG 2007). The intent of this document is to ensure the Mountain Whitefish 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. Current regulations state that the Mountain Whitefish limit is 0, whereas the limit for trout is 6 from the Saturday of Memorial Day weekend through November 30, and the trout limit is 0 for the rest of calendar year. Although fish stocking occurs annually in the basin, an assessment of angler use and exploitation has not occurred in the watershed since 2007 (Garren et al. 2009). Following previous studies (Corsi 1989; Garren et al. 2009) conducted in the basin, we estimated angler use and harvest through creel surveys, and we estimated exploitation of hatchery trout using “Tag You’re It” (Meyer and Schill 2014).

OBJECTIVES

Our objective was to estimate angler effort and harvest using a check station creel design, and estimate angler use and exploitation of hatchery trout stocked in the upper Big Lost basin to enable comparison of current angler effort, catch, and harvest rates with those from previous surveys.

METHODS

Creel

We conducted creel check stations using a stratified, 2-stage, nonuniform, probability design (McCormick et al. 2013; McCormick and Meyer 2017). The creel design was intended to garner information for the entire Big Lost watershed, however, we focused our analyses on the Big Lost basin upstream of Mackay Reservoir to make our results comparable to previous studies. We used check stations to intercept and interview anglers that had completed their daily fishing trip to obtain method of fishing (e.g. bait, fly fishing, and spinner), time spent fishing, exact angling location, species caught, number caught, and number harvested or released. We operated check stations from Memorial Day weekend through October 31. We used the variance of effort from the previous survey (Garren et al. 2009) to estimate the number of days required to sample (McCormick and Meyer 2017). Currently, no information exists on road use, such as car counter data from the various roads that connect to the upper Big Lost basin. Therefore, we assigned percentages of use based on expert opinion (e.g., conservation officers) and applied those percentages to the amount of effort needed to sample at each check station. We stratified check stations into five locations: Fish Creek Pass Rd. (5% use), Antelope Creek Rd. (10% use), Burma Rd. (10% use), Trail Creek east (37.5% use), and Trail Creek west (37.5% use). Following previous creel studies (Corsi 1989; Garren et al. 2009), we randomly selected two weekdays and two weekend days in every two-week period throughout the creel duration (i.e., 160 days). We considered holidays that fell on a weekday as a weekend day, then we delineated the creel day to occur from one hour after sunrise until sunset; based on sunrise and sunset times for Idaho Falls. Start times for check stations occurred during one of two shifts: morning (i.e., A.M.) or afternoon (i.e., P.M.). Start time was randomly selected using the R code <runif(n, min, max)> (R Core Team 2019), where min was one hour after sunrise and max was the last minute of morning. For PM shifts, the min number was 12.00 and max number was sunset time minus the creel duration time. Creel shift durations were as follows: June = 6-h shifts, July & Aug = 8-h shifts, and Sept. & Oct. = 6-h shifts. Creel shifts in June were 6 hours long because there were road closures, while shorter days reduced the creel shift in September and October.). Once day and start time were selected, we used a weighted random number generator, and the percentages of use for each road, to select the check station location.

We estimated catch or effort ($\hat{\theta}_d$) for each day that was sampled (*d*) using the Horvitz-Thompson estimator (Cochran 1977):

$$\hat{\theta}_d = \sum_{i=1}^{n} \frac{Y_i}{\pi_i}, \quad (1)$$

where *n* was the total number of anglers interviewed on day *d*, Y_i was the number of fish caught by the *i*th angler interviewed (when estimating catch), or the hours fished by the *i*th angler interviewed (when estimate effort), and π_i was the probability of sampling the *i*th angler. The sampling probability was the product of the spatial sampling probability and the temporal probability that was the length of the shift divided by the length of the fishing day.

Catch or effort ($\hat{\theta}_k$) for the *k*th stratum was estimated as:

$$\hat{\theta}_k = N_k \frac{\sum_{d=1}^{n_k} \hat{\theta}_d}{n_k}, (2)$$

where N_k was the number of days in the stratum and n_k is the number of days surveyed in the stratum. Estimates of effort and catch were summed among strata to estimate effort and catch over the duration of the season.

Variance of catch or effort was estimated as;

$$\hat{V}(\hat{\theta}_k) = N_k{}^2 \left(\frac{s^2_{\hat{\theta}_k}}{n_k} \right), (3)$$

where $s^2_{\hat{\theta}_k}$ is the sample variance which was calculated as:

$$s^2_{\hat{\theta}_k} = \frac{\sum_{d=1}^{n_k} (\hat{\theta}_d - \bar{\theta}_k)}{n_k - 1}, (4)$$

where $\bar{\theta}_k$ was the average daily catch or effort estimate over the stratum. Similar to the point estimate, the overall season variance ($\hat{V}(\hat{\theta})$) was calculated as the sum of the estimated strata variances. A confidence interval for estimated catch or effort over the season ($CI_{\hat{\theta}}$) was estimated as:

$$CI_{\hat{\theta}} = \hat{\theta} \pm Z_{\alpha/2} \sqrt{\hat{V}(\hat{\theta})}, (5)$$

where $Z_{\alpha/2}$ was the desired critical value for the CI (e.g., 1.96 for a 95% CI).

Fish tagging

Mackay Fish Hatchery reared all YCT and RBT that were tagged and stocked in the upper Big Lost River basin. In 2019, we stocked 10,139 catchable trout (i.e., Rainbow Trout and Yellowstone Cutthroat Trout combined) in the upper Big Lost River basin. A proportion (e.g., 10-15%) of fish that were being stocked on a particular day were implanted with nonreward, T-bar anchor tags at the base of the dorsal fin according to standard methods (Dell 1968) prior to stocking. 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 estimated the angler reporting rate (λ) using the average reporting rate of nonreward tags in the current study relative to the high-reward tags of hatchery RBT as estimated by Meyer et al. (2012):

$$\lambda = \frac{\mathrm{R}r \div \mathrm{R}t}{\mathrm{N}r \div \mathrm{N}t}, (6)$$

where R*r* and R*t* are the numbers of nonreward tags released and reported, respectively; and N*r* and N*t* are the numbers of high-reward tags released and reported (Pollock et al. 2001). We assumed a $200 reward tag reporting rate of 100% (Meyer et al. 2012). In the current study, we used statewide averages to estimate tag loss and tagging mortality of hatchery RBT (Meyer and Schill 2014). We estimated angler exploitation (u') using the equation:

$$u' = \frac{u}{\lambda\,(1 - Tag_l)(1 - Tag_m)}, (7)$$

where u is the number of nonreward tagged fish that were reported as harvested divided by the total number of nonreward tagged fish stocked, 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 as

reported by Meyer and Schill (2014). We also estimated angler use by modifying u to include fish reported as caught and released.

RESULTS

Creel survey

Check station personnel interviewed 222 anglers during 48 check station shifts. Angler effort was estimated at 11,775 h (95% confidence interval ± 4,975 hours) with fly anglers contributing 79% of effort followed by bait anglers (i.e., 12%), and lure anglers at 9% of total effort (Table 5; Figure 25; Figure 26). Bait anglers reported the highest harvest rates at 38%, followed by lure anglers at 13%, and fly anglers at 1%. The estimated rate of harvest out of the total catch for all anglers was 8% (Figure 27). Total catch was estimated at 16,408 ± 8,874 fish, all species combined (Figure 28). We estimated catch rates to be 1.4 fish/h (Table 5). Most angling effort was concentrated in the Upper Big Lost River upstream from Mackay Reservoir to the confluence of the East and North forks of the Big Lost River, but the highest catch rates were in Wildhorse Creek (Figure 29). Notably, we estimated 916 MWF were caught with no whitefish being reported as harvested. Tests of significance ($\alpha = 0.05$) were conducted for total effort, total catch, catch rate, total fish stocked, total harvest, harvest rate, and gear type across years. There was not a significant difference among years between any tested independent variable. However, harvest rates have significantly declined at $\alpha = 0.10$ (Figure 30).

Fish tagging

The average reporting rate of tags in the upper Big Lost basin was 28%. Total angler use, which consisted of trout caught and released and trout harvested, was 22% (95% confidence interval ± 11%) and ranged from 5% to 40% (Table 6). Estimated angler exploitation (i.e., the total number of fish reported as harvested from tag returns) was 10% ± 6% and ranged from 1% to 34%. Total angler use for RBT was 29% ± 16% (e.g., 21% in Wildhorse Creek and 40% in East Fork Big Lost River) and for YCT was 18% ± 10% (e.g., 5% in Star Hope Creek and 32% in North Fork Big Lost River). Exploitation for RBT was 21% ± 13% (e.g., 12% Wildhorse Creek and 34% East Fork Big Lost River) and exploitation for YCT was 4% ± 3% (e.g., 1% in Star Hope Creek and 7% in North Fork Big Lost River).

DISCUSSION

We have observed downward trends in trout abundances at our standard sampling sites in the upper basin since 2007, and anglers have been voicing their concerns of historically-low catch rates in recent years. The creel and “Tag You’re It” studies were important in providing us with current data on where anglers are fishing, how much they are harvesting, and how the hatchery-origin catchables are being used in the Copper Basin area. These data provide us with information to guide our decisions on how to proceed in our management of trout in the upper Big Lost River basin. Resulting from these studies, we know that overharvest is not the issue limiting salmonid abundance, we know that catch rates are better or as good as they have been compared to past creel surveys, and we also know where our hatchery trout are being most used by anglers. Now, we can begin investigating other issues (e.g., disease, overwintering habitat) that may be limiting trout abundance and recruitment in the Big Lost watershed. In addition to trout, MWF are the species of greatest concern in the Big Lost watershed and as a result of our creel study we

know that anglers are still encountering MWF in their catch, but we are confident that anglers are not harvesting MWF because there was no reported harvest in our creel study. Based on our most recent abundance estimates conducted in the upper basin in 2017-2018, MWF abundance is increasing in the Big Lost River near Bartlett Point and in the East Fork Big Lost River. Continuing prohibitive harvest regulations on MWF is essential to promoting their abundance and distribution.

Despite some anglers reporting low catch rates, our creel survey indicated that catch rates are similar to previous surveys, but effort and harvest are lower than estimated in previous years. Although angling effort has decreased compared to previous surveys, catch rates are higher than estimated in 1986 (i.e., 1.3 fish/h Corsi 1989) and the same as estimated in 2007 (i.e., 1.4 fish/h; Garren et al. 2009). Catch rates have been consistent in response to changes in stocking rates, regulations, and the species being stocked in the watershed. Bait anglers contributed the least amount of effort, but had the highest estimated catch rates. The dominant gear type used in the basin has shifted from predominately bait angling in 1986 (i.e., 59%) to 80% use by fly anglers in 2019. Not only has there been a shift in dominant gear type, but total angler harvest has changed significantly over time. The total amount of effort estimated by our creel surveys suggests a declining trend, but there are a large proportion of anglers visiting from other states to fish the upper Big Lost basin suggesting that it remains a popular sport fishery. Data collected from creel surveys and fish tagging suggests that harvest is the lowest it has ever been estimated in the upper Big Lost basin, and catch rates are similar to what they were in past surveys suggesting that catch rates and harvest are not as bad as anglers may perceive.

Angler exploitation (i.e., 10%) of hatchery trout from the “Tag You’re It” program compared to harvest (i.e., 8%) estimates from the creel survey reveal similar estimates of harvest in the upper Big Lost basin. These low estimates of harvest are likely not having a population-level effect, and our harvest estimates in the upper Big Lost River basin are similar to other stream fisheries in Idaho. Peterson et al. (2018) estimated exploitation of hatchery trout at 9.3% and total use of hatchery trout at 13.6% in lotic fisheries of the Southwest Region. Additionally, harvest was low in the Middle Fork Boise River (2.2% and 4.5%), but more variable in the North Fork Boise River (11.2% and 20.1%; Branigan 2018). Although we estimated similar harvest using both methods, the creel estimate of harvest includes wild and hatchery fish and is based on the total number of fish harvested of the total estimated catch. Whereas fish tagging and reporting includes only the estimate of angler harvest of hatchery trout stocked in 2019. Tag returns indicated that most angler use of hatchery trout occurred in the East Fork and North Fork Big Lost rivers, while Star Hope Creek only had 5% angler use. Due to low angler use of hatchery-origin fish in Star Hope Creek we should reevaluate where those fish are stocked in Star Hope Creek by repeating the “Tag You’re It” evaluation in the future or by reallocating these fish to other waters. In regards to the creel survey, the Upper Big Lost River received the highest amount of effort followed by the North Fork and East Fork Big Lost rivers. These results suggest that our stocking regime may be put to better use if fish were stocked more heavily in tributaries with the greatest use or where there were the highest rates of harvest.

We tagged about 13% of all catchable trout that were stocked in the upper Big Lost basin, and exploitation for all stocked fish was estimated at 10%, which is about 1,000 hatchery-origin fish that were harvested. The majority of hatchery fish harvest occurred for RBT in the East Fork Big Lost River (34%). However, harvest of hatchery-origin YCT was considerably lower at 7% in the North Fork Big Lost River and only 1% in Star Hope Creek. Our creel study included a harvest estimate of hatchery-origin and wild-origin fish combined, as well as wild Brook trout. The total number of RBT reported as harvested by anglers was greater than our tagging exploitation estimate, which suggests that a portion of the wild RBT population was harvested (i.e., about 80

fish). Conversely, a greater number of hatchery-origin YCT were harvested than our estimated harvest from the creel study; suggesting that no wild fish were harvested. Although we did not tag wild fish in this study, we have estimates based on the difference of tagged fish harvested and the creel estimate of harvest. Referring to the creel estimates of harvest by species, Brook Trout had the highest rate of harvest at 31%, followed by RBT at 12%, and YCT at 3%. Anglers are permitted to harvest up to 25 Brook Trout per day, whereas anglers can harvest 6 trout of all other species combined. Even though YCT are not native to the Big Lost River, anglers generally release them and harvest them at lower rates than RBT.

The check station creel design used in the current study introduced zero bias, whereas the 2007 (Garren et al. 2009) study may have inflated total effort and catch resulting from roving surveys coupled with instantaneous car counts multiplied by a standard factor (e.g., 1.61 anglers per car). On the other hand, the current study may have underestimated total angler effort and catch because check stations occurred one at a time per day, and we assumed that all anglers would stop at a check station if encountered. Our estimates did not include a noncompliance factor. Replicating this study to specifically target the Copper Basin fishery would best be conducted using check stations on Trail Creek Rd. because that was where we had the greatest number of interviews. Additionally, working more diligently with Conservation Officers to ensure their presence at check stations could help us obtain a noncompliance factor.

MANAGEMENT RECOMMENDATIONS

1. Evaluate stocking location effect on hatchery trout use in Star Hope Creek using “Tag You’re It.”

2. Continue following the recommendation from 2007 to sample fish populations throughout the upper Big Lost River basin on a five-year rotation; the next sampling year will be 2024.

3. In the next sampling period, tag wild fish during population estimate sampling to obtain estimates of angler use and harvest of wild fish, specifically targeting those tributaries with high reported use.

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Table 5. Data collected from angler creel surveys conducted in the Big Lost River watershed in 1986, 2007, and 2019.

	2019	2007	1986
Number interviewed	222	547	645
Hours of effort	11,775	50,079	29,133
Number caught	16,408	71,899	37,873
Catch rate (fish/h)	1.4	1.4	1.3
Harvest (fish/h)	0.1	0.3	0.6
Release (fish/h)	0.9	1.2	0.7
Mountain Whitefish caught	916	748	1,505
Cutthroat Trout caught	7,110	8,487	0
Rainbow Trout caught	7,227	36,645	28,509
Brook Trout caught	2,485	25,943	10,990
Mountain Whitefish harvested	0	202	2,193
Cutthroat Trout harvested	253	883	0
Rainbow Trout harvested	833	8,069	12,440
Brook Trout harvested	758	4,557	4,075

Table 6. Total angler use and angler exploitation with 95% confidence intervals (CI) for hatchery Rainbow Trout and Cutthroat Trout stocked in the upper Big Lost basin in 2019.

Water body	Species	Angler use		Angler exploitation	
		Estimate	95% CI	Estimate	95% CI
Upper Big Lost basin total	Combined	0.22	0.11	0.10	0.06
Wildhorse Creek	Rainbow Trout	0.21	0.15	0.12	0.08
East Fork Big Lost River	Rainbow Trout	0.40	0.25	0.34	0.23
Star Hope Creek	Cutthroat Trout	0.05	0.05	0.01	0.02
North Fork Big Lost River	Cutthroat Trout	0.32	0.19	0.07	0.07

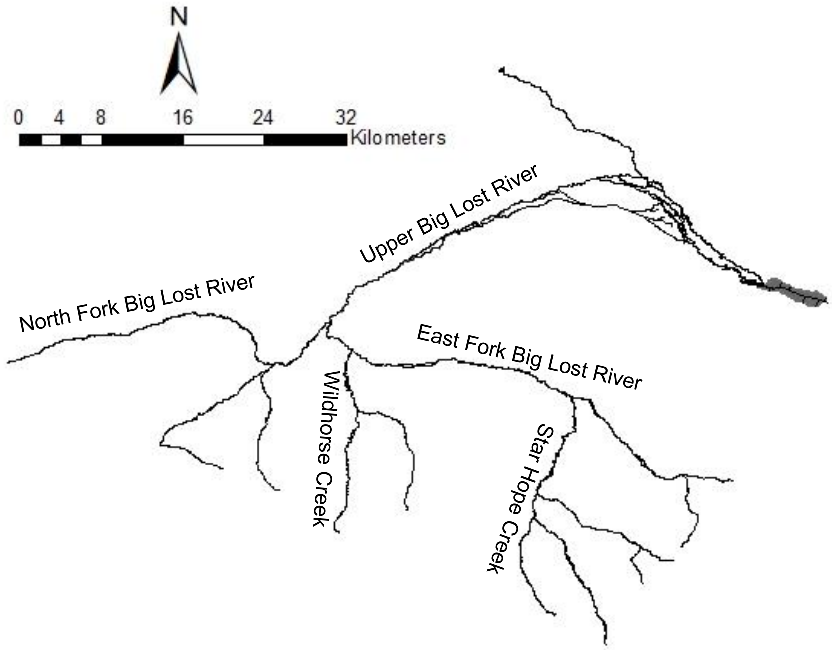

Figure 24. Map of the the upper Big Lost basin, Idaho upstream of Mackay Reservoir.

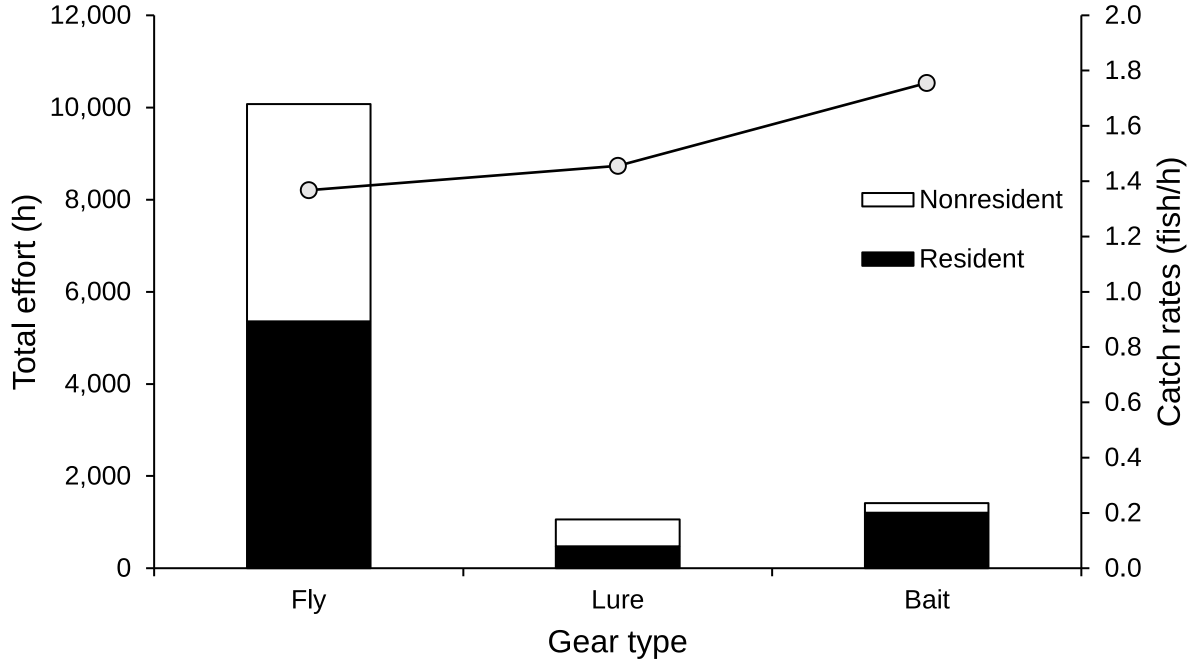

Figure 25. Total effort (hours; i.e., bars) and catch rates (number of fish per hour; i.e., line with points) by gear types. Bars are divided into nonresident (i.e., white portion) and resident (i.e., black portion) effort to equal the total amount of estimated effort by fishing gear type.

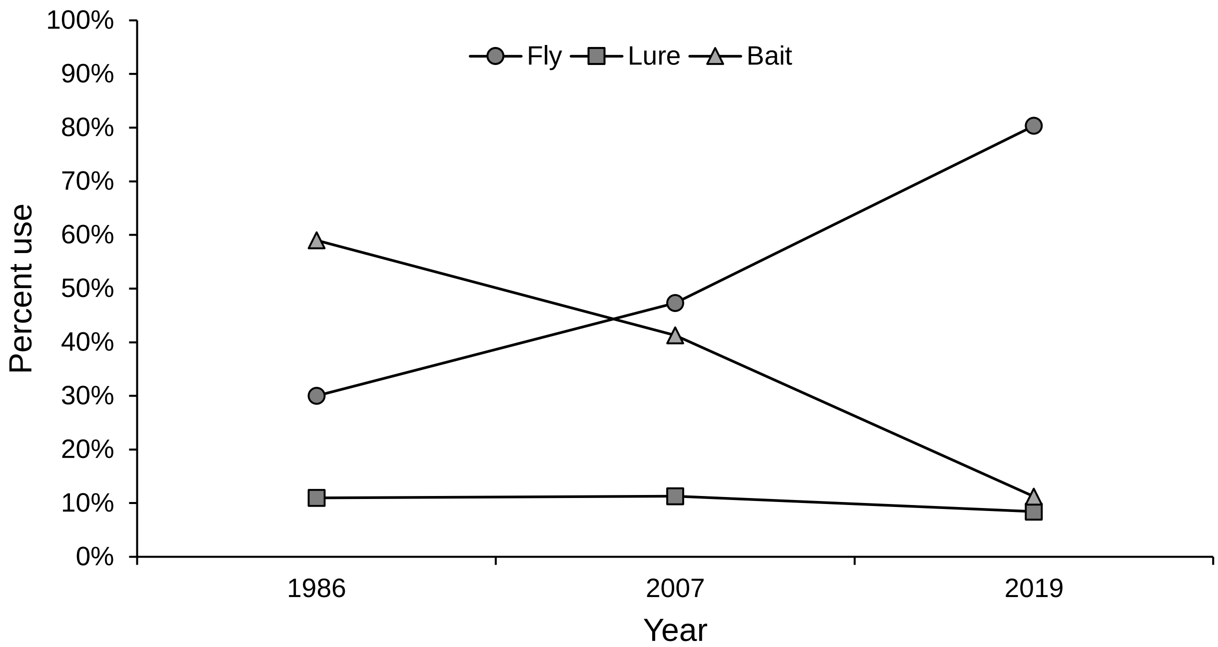

Figure 26. The percent use by each gear type over time in the Big Lost River and its tributaries upstream of Mackay Reservoir.

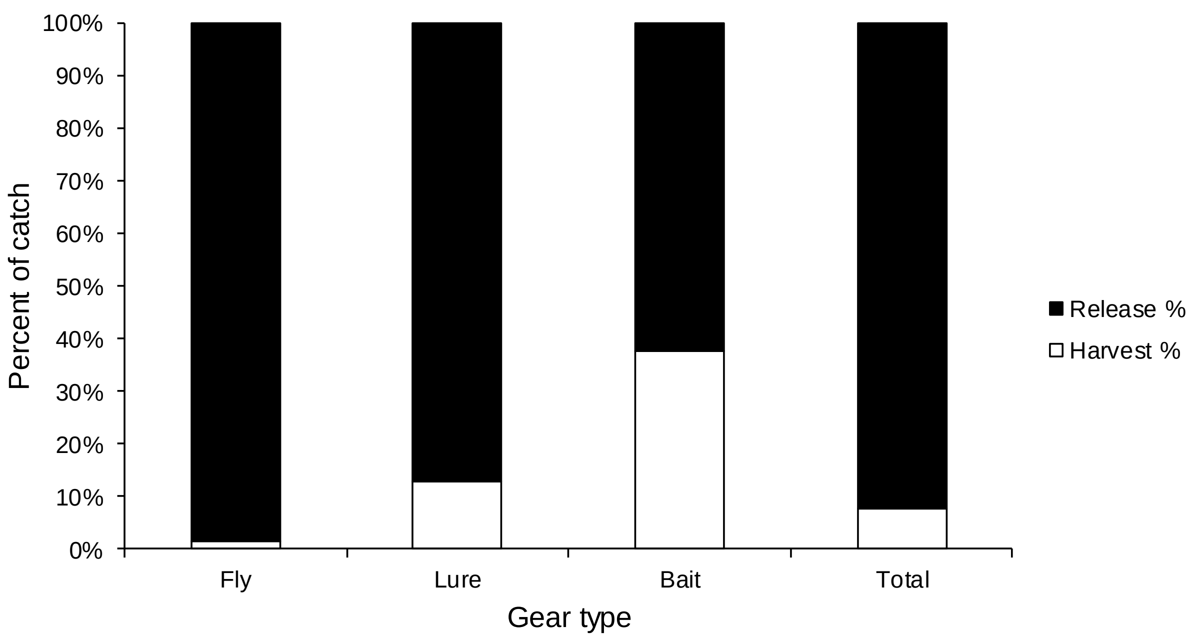

Figure 27. The estimated rates of harvested and caught and released fish out of the total catch by angler gear type.

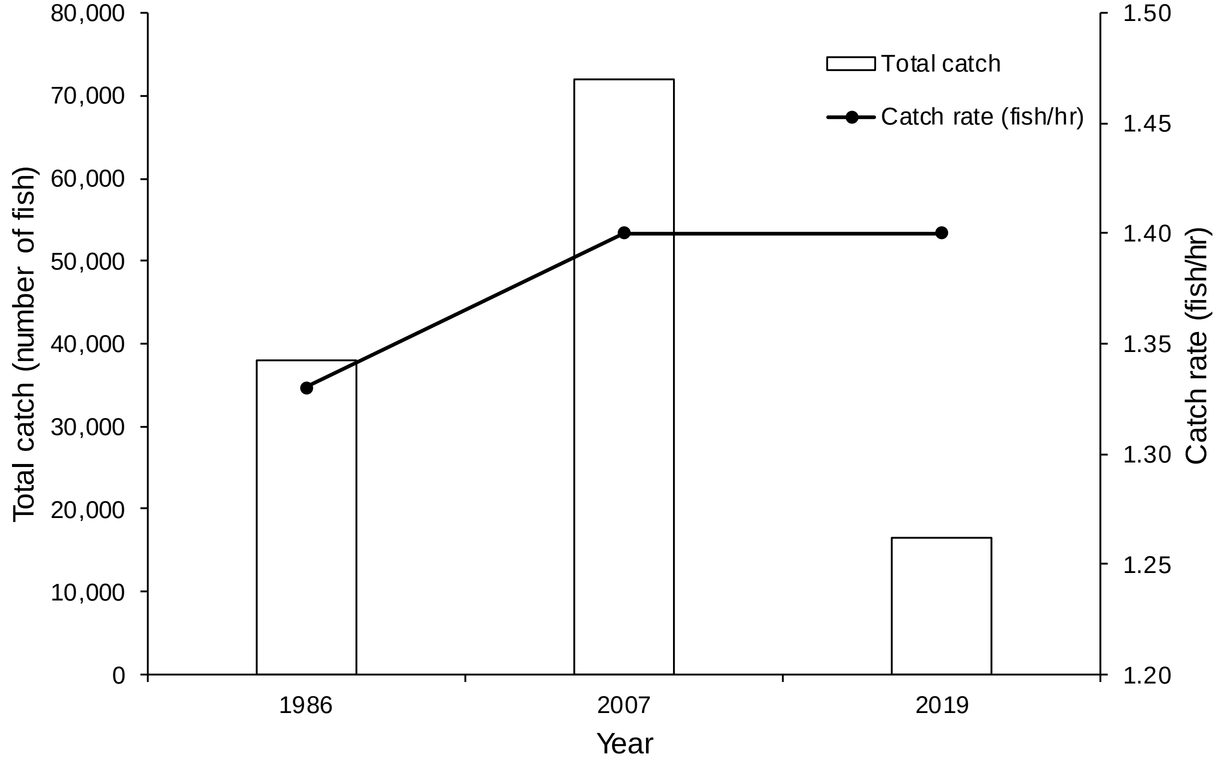

Figure 28. Total estimated catch and catch rates (fish/h from 1986, 2007, and 2019 creel surveys conducted in the upper Big Lost basin.

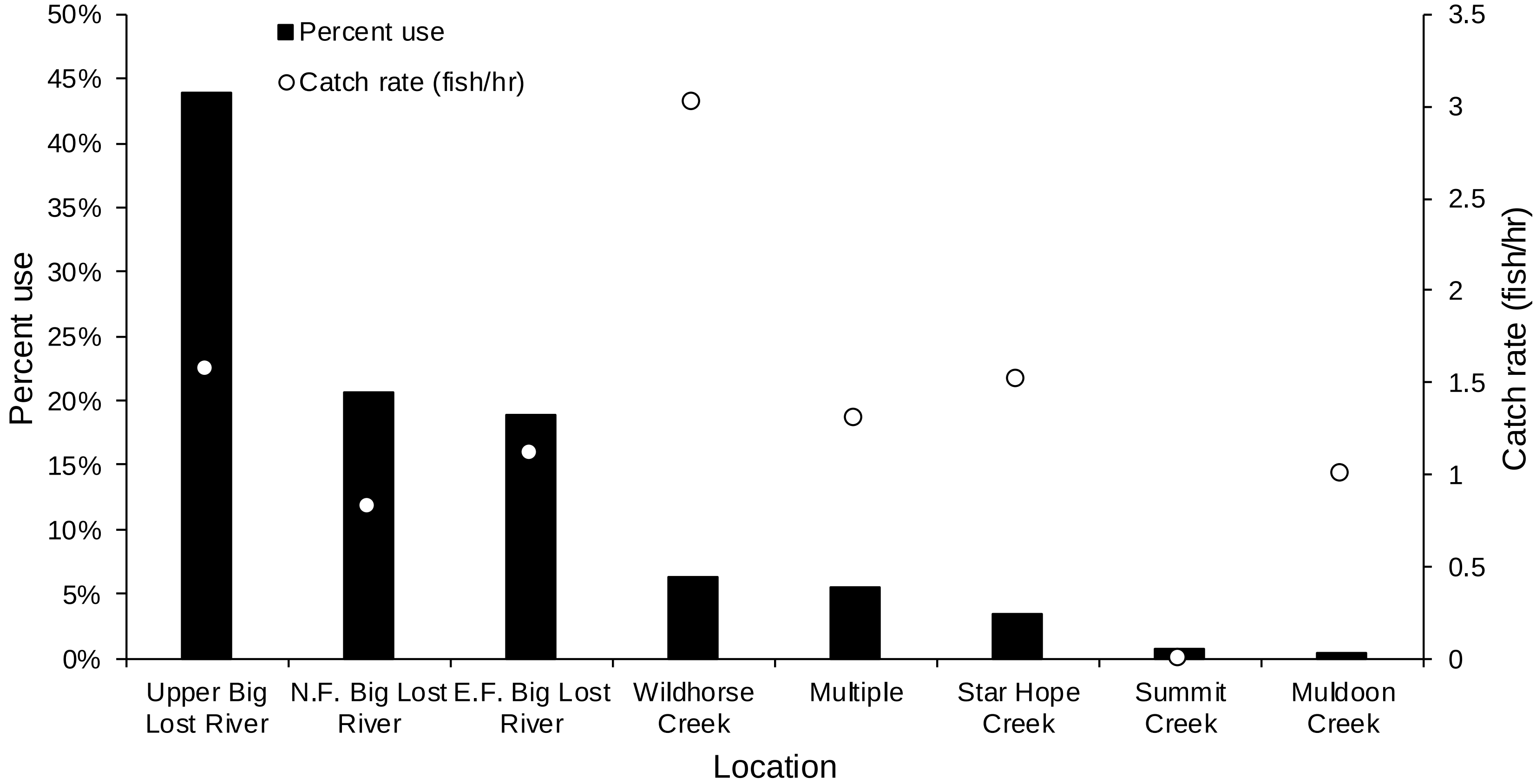

Figure 29. The percent use and catch rates per tributary in the upper Big Lost basin as estimated from creel surveys conducted in 2019.

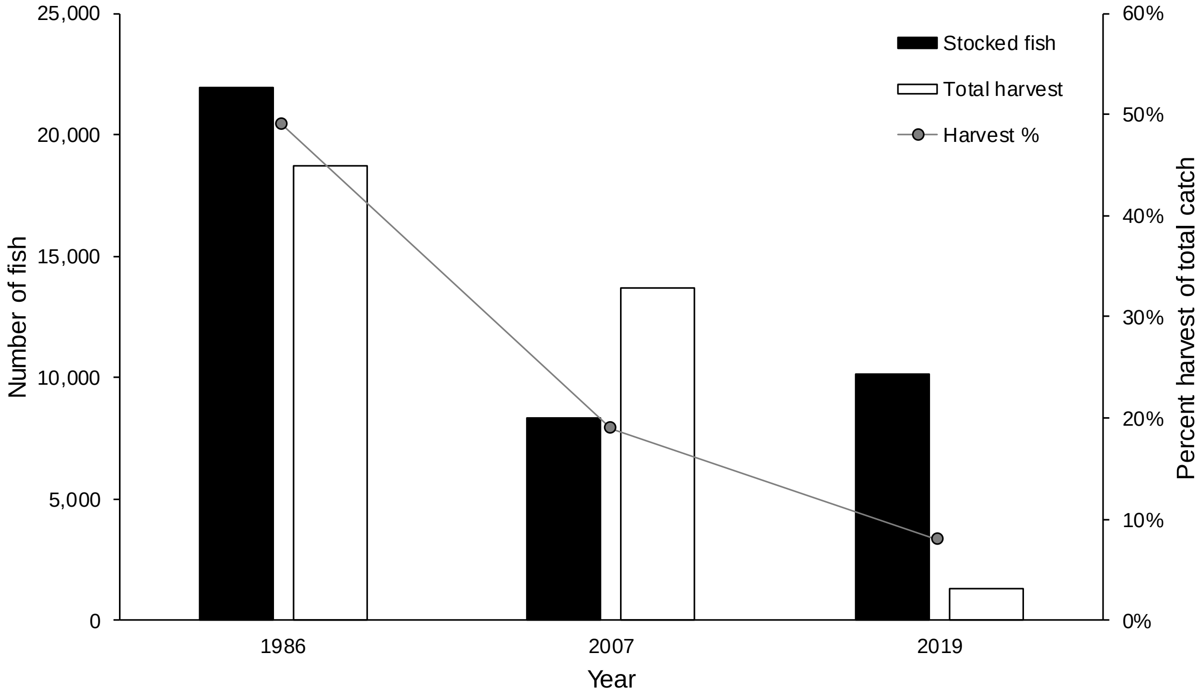

Figure 30. The total number of fish stocked from 1986, 2007, and 2019 compared to total fish harvested out of total catch as reported by anglers interviewed in creel surveys.

FALL RIVER

ABSTRACT

We used boat-mounted electrofishing equipment to conduct mark-recapture population estimates during August 2019 on two sections of the Fall River to obtain and allow monitoring of fish population densities and species composition. In the Sheep Falls reach, Rainbow Trout *Oncorhynchus mykiss* (RBT) and Mountain Whitefish *Prosopium williamsoni* (MWF) represented 77% and 22% of the catch, respectively, while Brook Trout *Salvelinus fontinalis* (BKT) constituted 1% of the catch. We estimated 853 RBT per km (± 406; 95% CI) in the Sheep Falls reach; however, we did not obtain an estimate on MWF or BKT. The mean total length of RBT and MWF in the Sheep Falls reach was 218 and 354 mm, respectively. Rainbow Trout PSD and RSD-400 were 7 and 1, whereas MWF PSD and RSD-400 were 91 and 4, respectively. In the Kirkham Bridge reach, we estimated 2,200 RBT (± 857) and 808 MWF (± 350) per km. Rainbow Trout made up 65% of all fish caught, MWF comprised 35% of all fish caught, while Brook Trout and Brown Trout *Salmo trutta* made up less than 1% of the total catch. Mean total length of RBT and MWF in the Kirkham Bridge reach was 210 and 299 mm, respectively. Rainbow Trout PSD and RSD-400 were 14 and 0, whereas MWF PSD and RSD-400 were 61 and 4, respectively. Although the size structure of RBT in the Fall River is smaller than the Henrys Fork, it does provide additional opportunities for anglers seeking a less crowded river in the region.

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INTRODUCTION

The Fall River originates in Yellowstone National Park and joins the Henrys Fork Snake River northeast of St. Anthony, Idaho (Figure 31). The Rainbow Trout *Oncorhynchus mykiss* (RBT) and Mountain Whitefish *Prosopium williamsoni* (MWF) populations have not been surveyed in the Fall River since 2006 (Garren et al. 2008). Based on fishery investigations conducted in the early 2000s (Garren et al. 2006; Garren et al. 2008), the size structure of RBT in the Fall River was dominated by smaller, younger fish. An unpublished, radio-telemetry study (Idaho Department of Fish and Game, unpublished data 2018) tracked RBT in the Fall River that were initially tagged in the Vernon reach of the Henrys Fork Snake River. These findings suggest that there may be population connectivity between the Henrys Fork and Fall rivers, and the Fall River may be important for juvenile RBT that migrate to the Henrys Fork for a portion of their life. Our objective was to establish long-term monitoring sites for conducting population estimates in upper and lower portions of the Fall River.

OBJECTIVES

To establish two, long-term monitoring sites in different reaches of the Fall River and evaluate the population characteristics of Mountain Whitefish and Rainbow Trout in both reaches.

METHODS

Previous fishery investigations of the Fall River were conducted near Kirkham Bridge using two drift boats. A single, 10-km reach was sampled in June for those surveys (Garren et al. 2006, 2008). In order to sample a wider range of habitats in the Fall River, we established two sampling reaches that were 5.63-km long (i.e., 3.50 miles) each. The upper site is upstream of all water diversions and begins near Sheep Falls. The lower site begins at Kirkham Bridge and is downstream of three diversions that lead to canals ranging in size from 0 to 251 cfs diversion rates (Appendix D). In the current study, survey reaches were sampled using two electrofishing rafts. Water temperature (°C) and conductivity (µS/cm) were taken 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). Electricity was applied 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 was positioned at the bow of the raft and used a 2.4-m long dip net with 6-mm bar knotless mesh. Netters were instructed to net all trout and MWF and place fish into a live well that was located in the raft. In the Sheep Falls reach, fish were marked on August 20 and the recapture run occurred on August 22. In the Kirkham Bridge reach, fish were marked on August 26 and the recapture run occurred on August 28. Due to logistical constraints, single-pass marking and recapture runs were conducted in each reach. All trout encountered from mark-recapture surveys were collected, identified to species, measured for total length to the nearest mm, and fish 150 mm or greater were marked with a hole punch in the caudal fin prior to release.

For both reaches, we estimated densities using a Peterson estimator with a Chapman modification in Fisheries Analysis+ software (FA+; Montana Fish, Wildlife, and Parks 2004) to compare our results to earlier studies. We then calculated 95% CI for all abundance estimates. Proportional size distributions (PSD) were calculated as the number of individuals (by species) ≥

300 mm divided by the number of individuals ≥ 200 mm multiplied by 100. Similarly, relative stock densities (RSD-400) used the same formula, with the numerator replaced by the number of fish ≥ 400 mm (Anderson and Neumann 1996; Neumann et al. 2012). We also investigated species composition in both reaches.

RESULTS

We collected 596 fish during two days of electrofishing in the Sheep Falls reach. Species composition was dominated by RBT (77%), followed by MWF (22%), and Brook Trout *Salvelinus fontinalis* (1%). Other species that we encountered included Longnose Dace *Rhinichthys cataractae*, Speckled Dace *Rhinichthys osculus*, Mottled Sculpin *Cottus bairdii*, Paiute Sculpin *Cottus beldingii*, Mountain Sucker *Catostomus platyrhynchus*, and Bluehead Sucker *Catostomus discobolus*. Rainbow Trout density was estimated at 853 fish per km (± 406; Table 7). We were unable to estimate MWF density due to lost data from the marking run. The mean TL of RBT was 219 mm (± 45; SD; Table 8; Figure 32) with PSD equal to 7 and RSD-400 was 1. Mean TL of MWF was 354 mm (± 39; SD; Figure 33) with a PSD of 91 and RSD-400 of 4.

In the Kirkham Bridge reach, we collected 1,894 fish in two days of electrofishing. Species composition was 65% RBT and 35% MWF. Brown Trout *Salmo Trutta*, Brook Trout, and Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* compose less than 1% of total catch. We encountered the same nongame species as in the Sheep Falls reach with the addition of Redside Shiner *Richardsonius balteatus*. We estimated 2,200 RBT per km (± 857) and 808 MWF per km (± 350). The mean TL of RBT was 210 mm (± 112; SD) with a PSD of 14 and RSD-400 of 1. The mean TL of MWF was 299 mm (± 126; SD), PSD was 61, and RSD-400 was 4. The Kirkham Bridge reach overlapped with earlier studies (Garren et al. 2006, 2008).

Although we sampled at a different time of year, using different watercraft, and sampled a shorter reach, we compared our density estimates of RBT and MWF in the Kirkham Bridge reach to the earlier studies (Figure 34), as well as RBT length frequencies in 2006 to 2019 (Figure 35). We estimated RBT abundance to be 2,200 fish per km (± 857) in 2019, compared to 359 fish/km (± 177) in 2006 and 474 fish per km (± 349) in 2004. In 2019, we estimated MWF abundance to be 808 fish per km (± 350), compared to 1,160 fish per km (± 259) in 2006 and 1,046 fish per km (± 510) in 2004.

DISCUSSION

Brown Trout composition in the Vernon to Chester reach (i.e., the section where the Fall River joins the Henrys Fork) of the Henrys Fork has been steadily increasing over time (e.g., 3% in 2005 to 33% in 2018). But, we did not observe the same trend in the Fall River. During 2019, we caught four Brown Trout in total; whereas during the 2006 survey, we caught three Brown Trout in total. In addition, the length-frequency histogram from the most recent population estimate in the Vernon to Chester reach of the Henrys Fork- indicated a high relative abundance of RBT at 300 mm and greater (unpublished data 2018). The size structure of RBT sampled in the Fall River in 2019 was dominated by fish less than 300 mm, which is comparable to earlier surveys and suggests that the Fall River is an important tributary for juvenile RBT (Garren et al. 2008). However, Garren et al. (2008) estimated mortality of RBT in the Fall River at 59%, which was markedly higher than mortality estimates for RBT in the Stone Bridge (i.e., 16%) and Box Canyon (i.e., 39%) reaches of the Henrys Fork during the same time period. Based on the current

survey and earlier studies, it is unknown why there are not a higher abundance of larger RBT in the Fall River, but we hypothesize that larger RBT migrate to the Henrys Fork, or conditions in the Fall River are such that it cannot support a high density of adult RBT. An unpublished, radio-telemetry study conducted by IDFG in 2017 tracked RBT into the Fall River that were tagged in the Henrys Fork, which is an indication that adult RBT use the Fall River to some degree, possibly for spawning.

Rainbow Trout densities increased when compared to the surveys conducted in the early 2000s (Garren et al. 2006, 2008), but there are discrepancies in the sampling methods. For instance, we sampled during baseflow periods instead of during higher water in June, with rafts instead of drift boats, and we sampled shorter reaches. We parted from the methods of earlier surveys because we had gear that allowed us to sample at lower flows when we believed our electrofishing equipment would be more effective. We captured considerably more RBT (n = 1,223) in our survey in the Kirkham Bridge reach when compared to both surveys conducted in 2004 (n = 337; Garren et al. 2006) and 2006 (n = 419; Garren et al. 2008). Additionally, our capture efficiency (R/C = 0.05) improved from 2004 (R/C = 0.03) in the Kirkham Bridge reach, but it was similar to the 2006 (R/C = 0.05) survey. In the current study, we sampled a variety of size classes of MWF, which suggests that MWF recruitment is occurring and that multiple age-classes are present in the Fall River. The surveys we conducted in 2004 and 2006 (Garren et al. 2006; 2008) were valuable population estimates for monitoring long term trends in species composition, abundance and size structure of RBT and MWF.

RECOMMENDATIONS

1. Continue monitoring the Mountain Whitefish and Rainbow Trout populations using the same methods in the same reaches of the Fall River every three years.

2. In the next round of surveys, take a subsample of Mountain Whitefish and Rainbow Trout in each 10-cm length group for age and growth analysis and to estimate mortality.

3. Use otoliths and water samples to investigate origins and life histories of Rainbow Trout and Mountain Whitefish in the Fall River using otolith chemistry.

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Table 7. Total counts and population estimates of Rainbow Trout and Mountain Whitefish from the Fall River, Idaho during 2019.

Reach	Species	Count	Population estimate	95% CI	Fish per km	95% CI	R/C
Sheep Falls	Rainbow Trout	522	4,805	2,283	853	406	0.06
	Mountain Whitefish	70	-	-	-	-	-
	Brook Trout	4	-	-	-	-	-
Kirkham Bridge	Rainbow Trout	1,223	12,386	4,824	2,200	857	0.05
	Mountain Whitefish	662	4,549	1,968	808	350	0.07
	Brook Trout	6	-	-	-	-	-
	Brown Trout	3	-	-	-	-	-
	Yellowstone Cutthroat Trout	2	-	-	-	-	-

Table 8. Population index summaries (± 95% confidence intervals) for the Fall River, Idaho 2019.

Reach	Species	Mean TL (mm)	Median TL (mm)	PSD	RSD-400	Species Composition (%)
Sheep Falls	Rainbow Trout	218 ± 6	216	7	1	77
	Mountain Whitefish	354 ± 9	360	91	4	22
	Brook Trout	184 ± 28	182	-	-	1
Kirkham Bridge	Rainbow Trout	210 ± 3	204	14	1	65
	Mountain Whitefish	299 ± 5	321	61	4	35
	Brook Trout	250 ± 50	270	25	0	0
	Brown Trout	354 ± 174	393	100	50	0
	Yellowstone Cutthroat Trout	286 ± 174	286	100	0	0

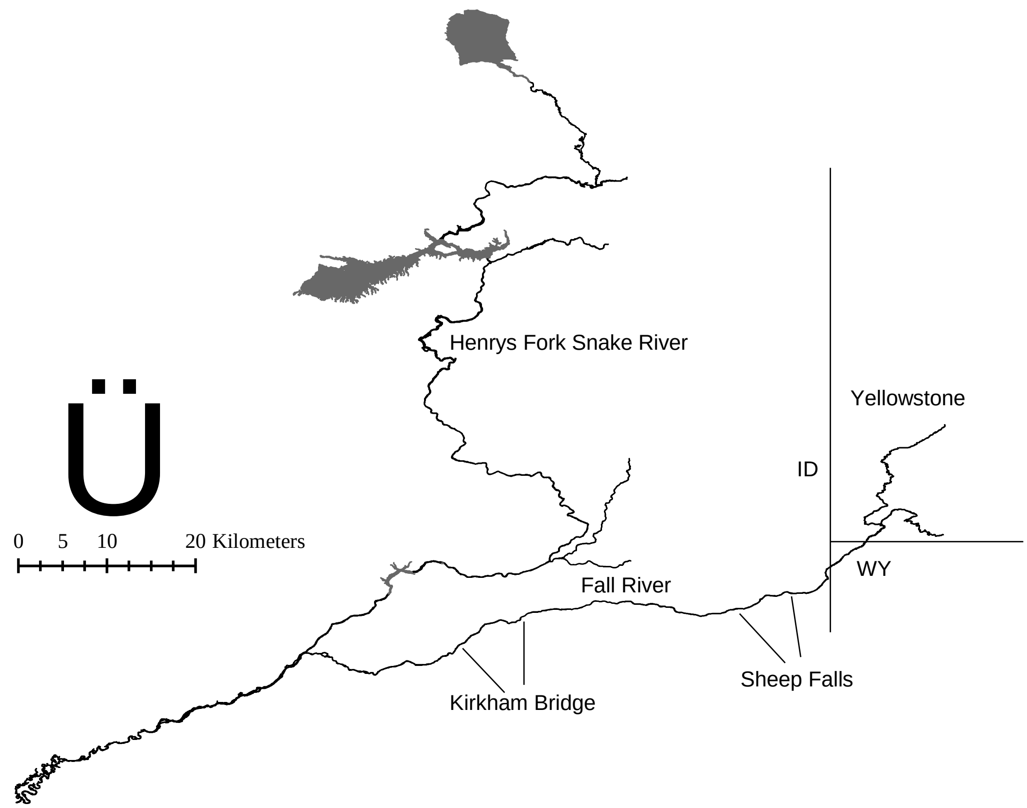

Figure 31. Map of the upper Henrys Fork and Fall River drainages, Idaho including locations of the Sheep Falls and Kirkham Bridge sampling reaches.

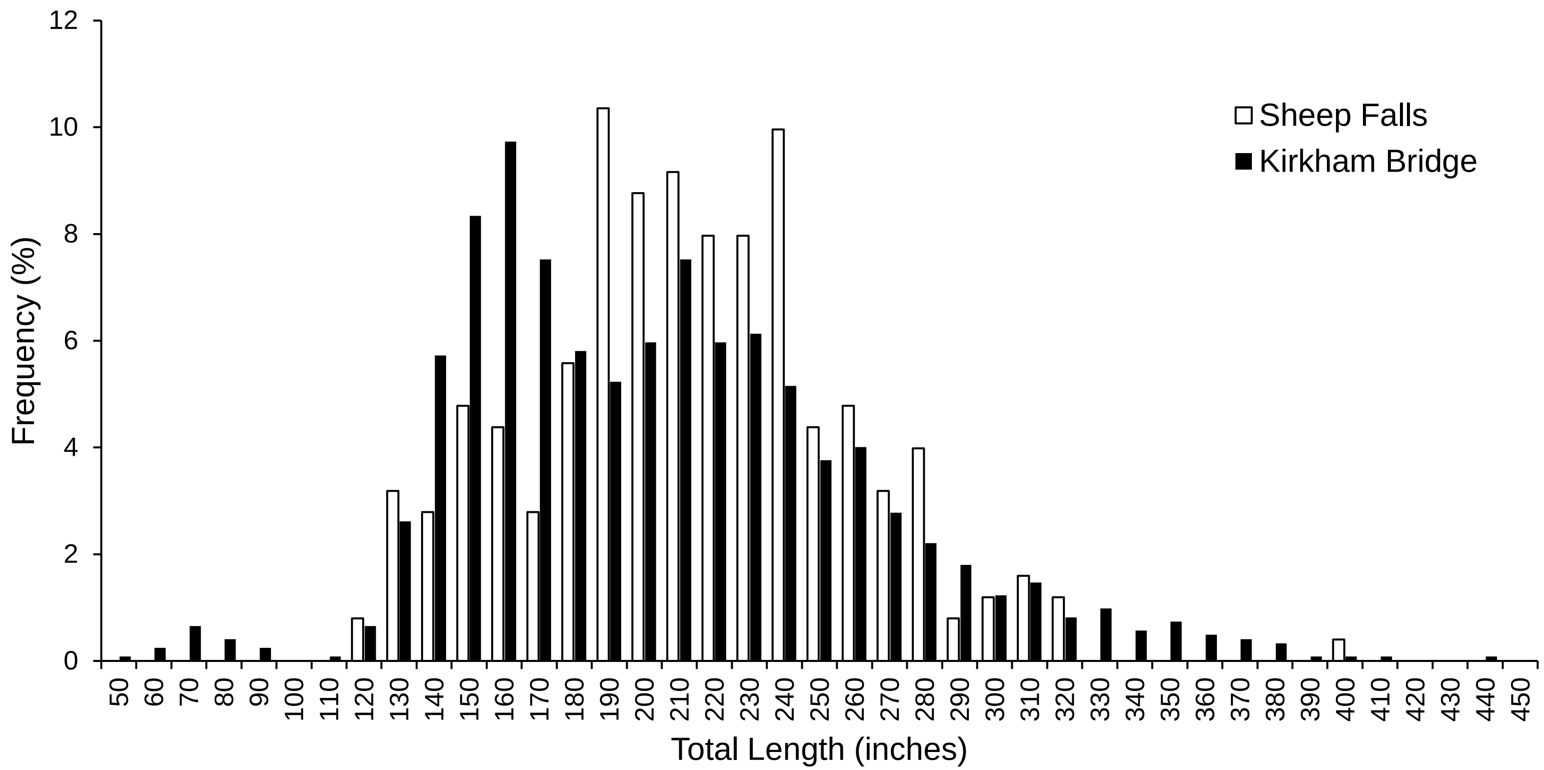

Figure 32. Length frequency distribution of Rainbow Trout caught in the Sheep Falls (n = 522 fish) and Kirkham Bridge (n = 1,223 fish) reaches of the Fall River in August 2019.

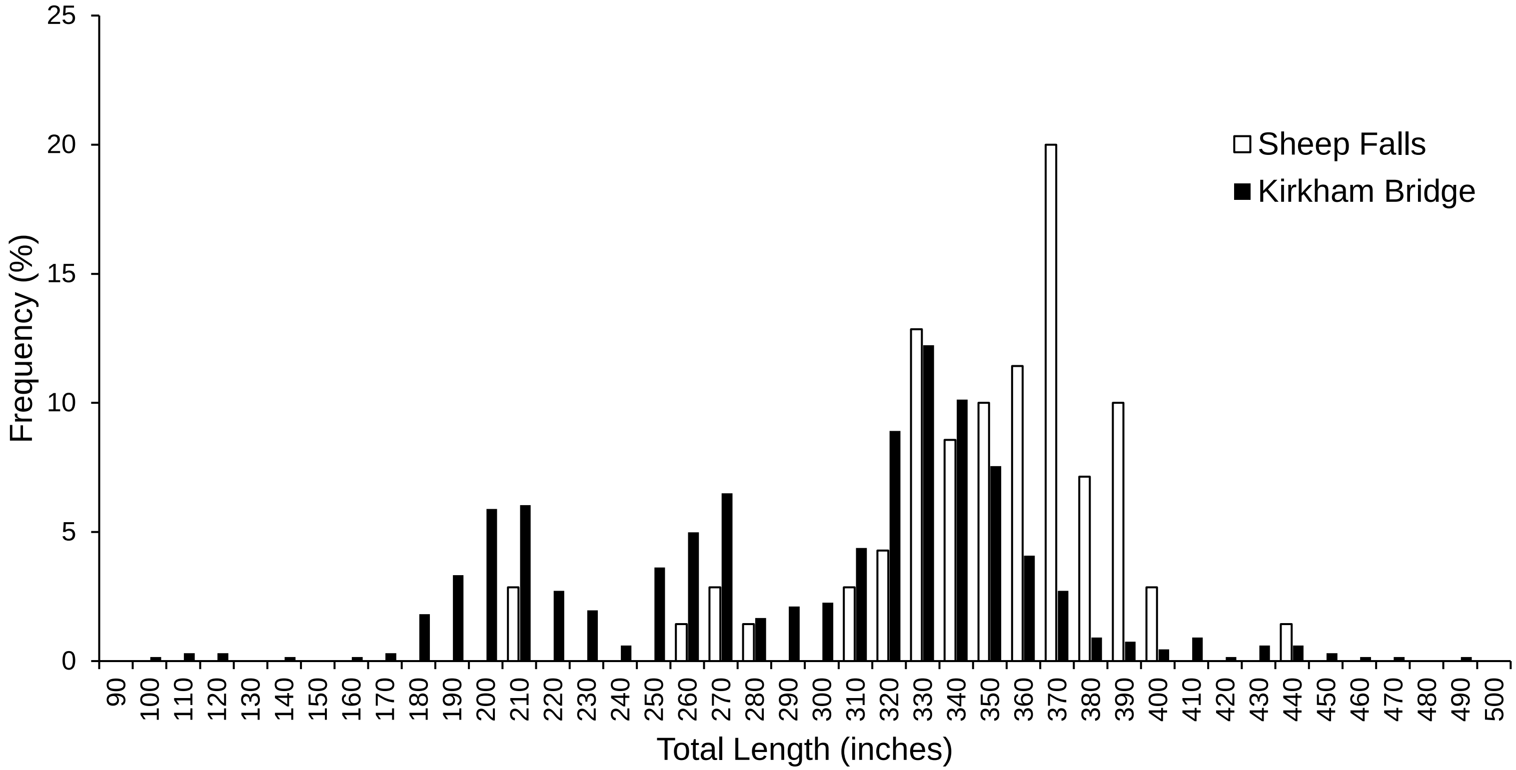

Figure 33. Length-frequency distribution of Mountain Whitefish in the Sheep Falls (n = 70) and Kirkham Bridge (n = 662) reaches of the Fall River sampled in August 2019.

Figure 34. Rainbow Trout and Mountain Whitefish population estimates for the Kirkham Bridge reach from 2004 to 2019. Population estimates were calculated using a modified Petersen estimator in Fisheries Analysis+ Software (Montana Fish, Wildlife, and Parks 2004). Bounds represent 95% confidence intervals.

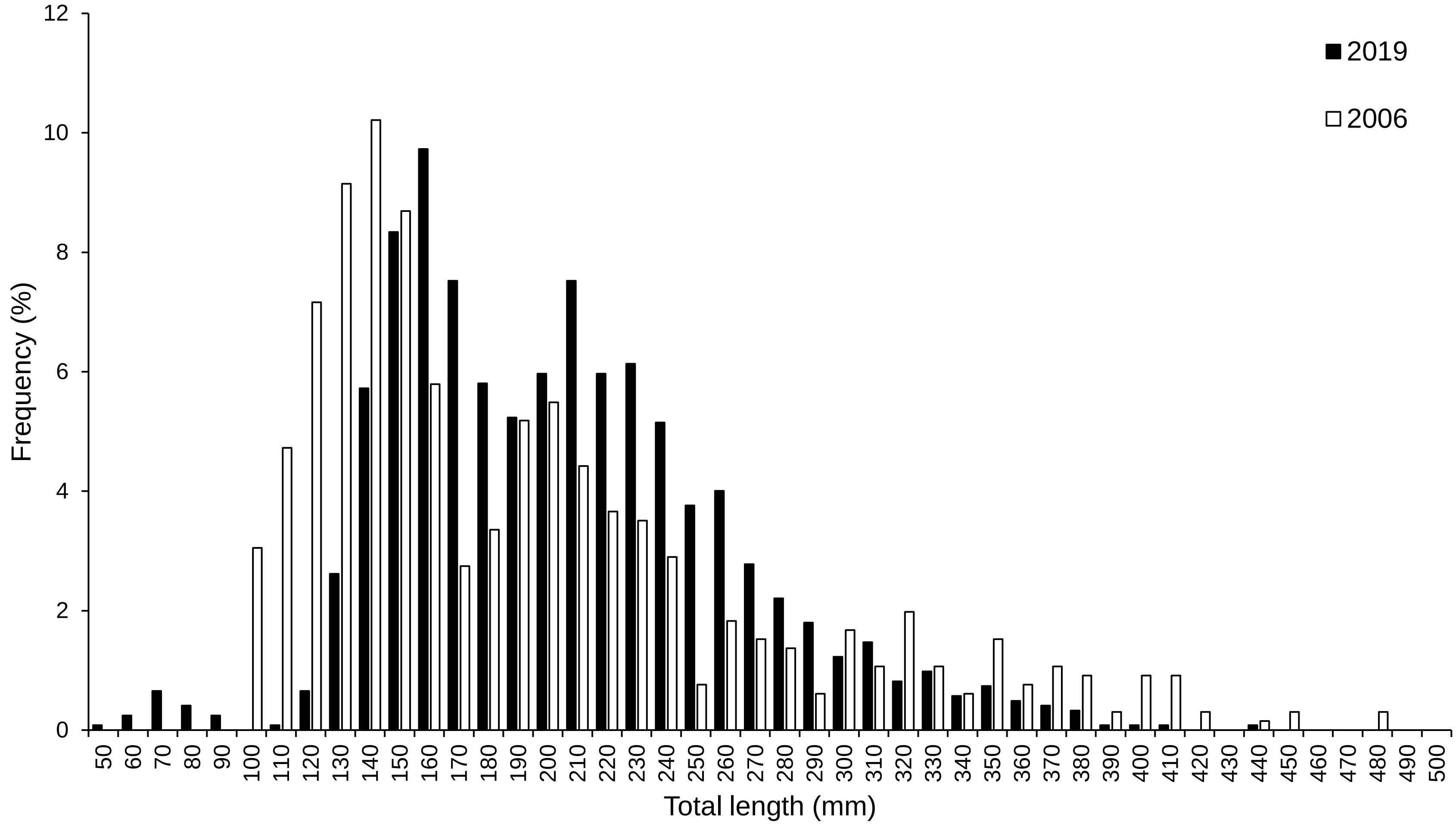

Figure 35. Length-frequency distribution of Rainbow Trout in the Kirkham Bridge reach of the Fall River in 2006 (white bars) and 2019 (black bars).

HENRYS LAKE

ABSTRACT

Henrys Lake is one of the most popular recreational fisheries in Idaho, and is known to support a robust trout fishery. We used 50 gill-net nights of effort in the spring of 2019 to evaluate the trout populations in Henrys Lake. Total trout catch per unit effort (CPUE) was 6.4 trout per net night (± 1.3; 95% CI), which was below the 25-year, long-term average of 12.3 and the management target of 11 trout per net night. Mean relative weight (W_r) for all trout species (all sizes combined) ranged from 90 to 96 and has decreased compared to prior years. Utah Chub *Gila atraria* CPUE decreased from the previous survey years to 13.9 chub per net night (± 1.3, 95% CI). In order to assess net location as a factor of net recruitment we set an additional 50 gill-net nights at randomized locations around the lake. Nets set in in random locations had higher CPUEs (21.5 ± 5.2) per net night than nets set at our traditional netting locations (6.4 ± 1.3). We monitored dissolved oxygen levels under the ice to assess the possibility of a winterkill event from December 18th, 2018 through January 24th, 2019. Based on depletion estimates, we predicted dissolved oxygen would not reach critical levels (10 g/m^2) and did not start aeration pumps. Parentage based tagging (PBT) indicated a 1.5% wild YCT contribution to the lake in 2019. We completed a full season creel on Henrys Lake in 2019. Angler catch rate was 1.09 fish/hour which exceeded our management goal of 0.7 fish/hour and was the highest catch rate observed over the last decade. Total angler effort was 207,989 hours with an estimated total harvest of 33,109 fish. Favorable water conditions, high reservoir volumes, and increased stocking rates have all lead to increased survival, increased numbers of trout and increased angling opportunities in Henrys Lake.

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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. The now-inundated lower portions of tributary streams historically provided spawning habitat for adfluvial Yellowstone Cutthroat Trout, 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 native Yellowstone Cutthroat Trout *Oncorhynchus clarkii*, Hybrid Trout (Rainbow Trout x Yellowstone Cutthroat Trout) and Brook Trout *Salvelinus fontinalis*, with an average 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 Yellowstone Cutthroat Trout, 11.5 million Hybrid Trout, and 4.3 million Brook Trout. Stocking ratios averaged 84% Yellowstone Cutthroat Trout, 12% Hybrid Trout, and 4% Brook Trout from 1966 to 2010. Beginning in 1998, all Hybrid Trout were sterilized prior to release to reduce the potential for hybridization with native Yellowstone Cutthroat Trout. Although hybridization was not a concern with Brook Trout, 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 Brook Trout to compete with native salmonids.

Anglers view Henrys Lake as a quality fishery capable of producing large trout. 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 emphasized 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 Yellowstone Cutthroat Trout exceeding 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 prior to the fishing season, while creel surveys are conducted on a three- to five-year basis.

Catch rates of trout observed in recent years during annual gill-net surveys over the past six years were lower than expected despite annual increases of fall stocked hatchery trout. This suggests trout may have experienced a higher than normal mortality rate for the last few years. 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 and interspecific competition). Understanding these potential limiting factors is key to maintaining a stable trout population to promote angling opportunities in the lake.

For the past five years, ice cover has left Henrys Lake earlier than the previous ten years, based on the first gillnet date each year. Open water present earlier in the season increases the amount of solar radiation and may have reduced the amount of thermal refuge available to trout during the warm summer months. High water temperatures and solar radiation may also have increased the frequency of cyanobacteria blooms, which were documented in the summers of

2016 and 2017. During and following an algal bloom, dissolved oxygen demands are high. As such, it is possible that the algal bloom observed in the summer of 2017 contributed to the low winter oxygen levels likely resulting in low overwinter trout survival.

Primary and secondary productivity may also be a limiting factor for trout growth and survival in Henrys Lake. Increased temperatures and the subsequent cyanobacteria blooms, may be shifting primary productivity away from beneficial phytoplankton which serve as a forage base for zooplankton and macroinvertebrates to harmful cyanobacteria. The ratio of Nitrogen to Phosphorus (TN:TP) can be used as an indicator of water conditions which contribute to this shift in phytoplankton (Levich 1996). Limited information is available on the role of water quality, phytoplankton and zooplankton effecting trout abundances in Henrys Lake. The most recent water quality assessment performed in Henrys Lake occurred more than two decades ago (Hill and Mebane 1998). As such, there is a need to conduct an in-depth water quality assessment to determine the abiotic factors, nutrient availability and food availability constraints on trout in Henrys Lake.

STUDY SITE

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

OBJECTIVES

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

METHODS

Population monitoring

As part of routine population monitoring, we set gill nets at six traditional locations in Henrys Lake in paired floating and sinking nets. Nets were set from May 7 – 15, 2019 for a total of 50 net nights (Figure 36). In addition to our traditional netting locations, we set gill nets at an additional 25 random locations in Henrys Lake in paired floating and sinking nets (50 net nights total) from May 7 – May 19, 2019 (Figure 36). The paired sets were deployed at least three days apart. All gill nets consisted of either floating or sinking types measuring 46-m long by 2-m deep, with equal length panels of 2-cm, 2.5-cm, 3-cm, 4-cm, 5-cm and 6-cm bar mesh. Nets were set at dusk and retrieved the following morning. We identified captured fish to species and recorded total lengths (TL) and weights. We calculated catch rates as fish per net night with 95% confidence intervals.

We examined all Yellowstone Cutthroat Trout sampled through the year for adipose fin clips as part of our evaluation of natural reproduction. Beginning in the 1980s and continuing through 2016, 10% of all stocked Yellowstone Cutthroat Trout (YCT) have been marked with an adipose fin clip prior to stocking (Appendix E). To estimate contributions to the YCT population from natural reproduction, we calculated the ratio of marked to unmarked fish collected in annual

gill-net surveys and trout captured ascending the fish ladder on Hatchery Creek. Since 10% of all stocked fish were marked with an adipose clip, ratios near 10% in the at-large population would be expected in the absence of additional, un-marked fish (natural reproduction). When the ratio of marked fish was less than 10%, we assumed that natural reproduction was contributing to the population. In 2017, the program shifted to using Parentage Based Tagging to gather information on hatchery vs. wild production which is described in detail below.

We removed the sagittal otoliths of all trout captured in 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 to fish length. A total of 10 otoliths were randomly subsampled and aged per 20-mm size class of each trout species. When less than 10 otoliths were present per size class, all otoliths were aged.

Ages of Utah Chub were estimated using fin rays (Griffin et al. 2017). The left leading pectoral fin ray was removed from each individual fish by cutting as close to the pelvic girdle as possible (Koch et al. 2008). After drying, fin rays were embedded in epoxy in centrifuge tubes and a thin section (~0.3-mm thick) was cut from the base of the pectoral fin ray with an isomet saw (Koch and Quist 2007). Fin rays were read by a graduate student at the University of Idaho, Moscow. Otoliths and fin rays were examined using a microscope (Lieca DM 1000 LED, Lieca Microsystems, Wetzlar, Germany) using transmitted light supplied from dual-strand fiber optics. Otoliths and fin rays were imaged using the microscope interfaced with a desktop computer and digital images were taken of whole otoliths and sectioned fin rays. A total of 10 fin rays were randomly subsampled and aged per 10-mm size class. When less than 10 fin rays were available in a size class, all rays were aged.

Relative weights (W_r) were calculated 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 then averaged 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 relative weights of Hybrid Trout, 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 Brook Trout (Hyatt and Hubert 2001). For Utah Chub, we used the formula log W_s = -4.984 + 3.049 log TL (IDFG, unpublished data).

We calculated proportional stock density (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 Yellowstone Cutthroat Trout, Hybrid Trout, and Brook Trout 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 Yellowstone Cutthroat Trout, Hybrid Trout, and Brook Trout using the following equation:

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

The criteria used for PSD and RSD-400 values for Yellowstone Cutthroat Trout, Hybrid Trout, and Brook Trout populations was 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.

Hybrid evaluation

In order to assess the effect of paternal strain on hybrid trout performance, we used fluorescent grit to mark two strains of HYB stocked into Henrys Lake. In 2015 and 2016 a total of 126,797 Gerrard and 209,088 Hayspur strain Hybrid Trout were marked with fluorescent grit for identification and stocked into Henrys Lake as fingerling trout. Trout collected in the gill nets (<500 mm TL) were visually examined using a black light for the presence of the fluorescent mark to determine strain of Hybrid Trout (chartreuse = Gerrard, and orange = Hayspur).

Parentage based tagging

Parentage Based Tagging (PBT) has been implemented since 2017 in conjunction with the Yellowstone Cutthroat Trout (YCT) spawning operations each year. All YCT from the entire season spawn take were sampled. Genetic samples were stored on Whatman paper appropriately labeled by spawn date and lot number. Whatman paper was pre-labeled with seven 7 horizontal sample locations on each plane. The first 7 slot plane was identified as male with the next plane identified as female. These 2 horizontal planes were identified as Family 1. This was repeated vertically down the Whatman paper with the next two male/female planes identified as Family 2 and so forth. Genetic samples were obtained from all phenotypically-identified YCT and HYB encountered during our annual gillnet survey.

Winter dissolved oxygen

Winter dissolved oxygen concentrations, snow depth, ice thickness and water temperatures were measured at five established sampling sites (Pittsburg Creek, Outlet, County Boat Dock, Wild Rose, and Hatchery) on Henrys Lake between December 18, 2018 and January 24, 2019. Holes were drilled in the ice with an 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 (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:

Average (bottom of ice + 1m) + Sum (readings from 2 m to lake bottom) = Total O_2 mass

The total mass of dissolved oxygen at each sample site was then expressed in g/m^2 (Barica and Mathias 1979). Data were then transformed using the natural logarithm (ln*)* for regression analysis. We used linear regression to estimate when oxygen levels would deplete to the critical threshold for fish survival (10.0 g/m^2).

Dissolved oxygen profiles were recorded each year 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^2. 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 Hatchery Creek's mouth.

Creel

We conducted a season-long creel survey, from May 25, 2019 through January 1, 2020. The fishing season was stratified into two-week intervals through October 31, when ice began to cover the lake, with the opening weekend separated out as its own strata. From October 31 through January 1, ice fishing occurred. Effort during the open water season was estimated using aerial counts on two weekend days and two week days which were randomly chosen for each strata. During the ice fishery, creel clerks counted the number of anglers and huts at each access point at two, randomly-selected times during each creel day. We used weighted time periods (morning = 60%, afternoon = 20%, evening = 20%) to select when creel clerks conducted interviews during the open water fishery. For the ice fishery, we assumed equal effort across the time periods (morning = 33%, afternoon = 33%, evening = 33%). Creel clerks interviewed anglers on the same day as the aerial counts. Creel clerks collected information on the time anglers spent fishing, the number of anglers in the party, number of vehicles per party, gear type, and number and species of fish both caught and harvested. When harvested fish were encountered, clerks measured fish for total length (mm) and identified any fin markings present.

RESULTS

Population monitoring

Traditional nets

We collected 1,015 fish in 50 net nights with our traditional gill-net survey. Gill net catch rates (CPUE) for all trout species combined was 6.4 ± 1.3 (95% CI; Figure 37). Catch composition was 68.6% Utah Chub, 54.3% Brook Trout (BKT), 35.5% Hybrid Trout (HYB), and 21.6% Yellowstone Cutthroat Trout (YCT). Mean (± 95% CI) trout CPUEs were highest for YCT at 4.4 (± 0.9) fish per net night, followed by HYB at 1.1 (± 0.4) and BKT at 0.88 (± 0.6) fish per net night (Figure 38). Hybrid Trout total lengths ranged from 288 to 299 mm with a mean of 404 mm (± 28.2). Yellowstone Cutthroat Trout total lengths ranged from 168 to 625 mm with a mean of 326 mm (± 10.1, Figure 39). Brook Trout total lengths ranged from 162 to 470 mm, with a mean of 335 mm (± 19.1; Table 8). We did not observe an adipose-clipped YCT during this gill-net survey (Appendix E).

The mean CPUE of Utah Chub was 13.9 (± 95% CI 10.4; Figure 40). This catch rate was approximately half of last year's mean of 25.5 (± 13.1) fish per net night. Utah Chub total lengths ranged from 134 to 363 mm with a mean of 230 mm (± 4.3; Figure 41).

Proportional stock density (PSD) was highest for HYB (96) followed by BKT (83), and YCT (65). Relative stock density (RSD-400) was highest for HYB (32) followed by YCT (12) and BKT (5; Table 9). Mean relative weight (W_r) for all size classes combined was 93 for both BKT and HYB while slightly lower at 90 for YCT (Table 9, Figure 42). Mean relative weight (W_r) for Utah Chub (all sizes combined) was 92 and ranged between 90 and 93 for size classes (100 – 199, 200 – 299, and 300 – 399 mm) (Table 9). Overall mean relative weights of Utah Chub decreased from 2018 (Figure 43).

Randomized nets

We collected 1,525 fish in 50 net nights with our randomized location gill nets. CPUE for all trout species combined was 21.5 ± 5.2 (95% CI), which was over three times higher than CPUE in the traditional gill nets (6.4; Figure 38). Catch composition was 44.9% YCT, 29.4% Utah Chub, 16.8% HYB, and 9.0% BKT. Mean trout CPUE (± 95% CI) were highest for YCT at 13.68 (± 3.8), followed by HYB at 5.1 (± 1.5), and BKT at 2.7 (± 1.1; Figure 38). Hybrid Trout total lengths ranged from 218 to 713 mm with a mean of 391 mm (± 12.8; Figure 39). Yellowstone Cutthroat Trout total lengths ranged from 169 to 661 mm with a mean of 347 mm (± 5.8). Brook Trout total lengths ranged from 169 to 574 mm with a mean of 349 mm (± 10.5; Table 10). We observed one YCT with an adipose clip of the 684 captured (Appendix E).

The mean CPUE for Utah Chub was 8.7 (± 95% CI 5.2). This was lower than the traditional location nets which averaged 13.9 (± 10.4; Figure 38). Utah Chub total lengths ranged from 133 to 392 mm with a mean of 223 mm (± 5.3; Figure 41).

Proportional stock density (PSD) was highest for BKT (92), followed by HYB (90), and YCT (78). Relative stock density (RSD-400) was highest for HYB (29), followed by YCT (21), and BKT (7) at similar stock densities (Table 9). Mean relative weight (W_r) was highest for BKT (98), followed by HYB (96) and YCT (91; Table 9). Mean relative weight (W_r) for Utah Chubs (all sizes combined) was 94 and ranged between 41 and 128 (Table 9).

Ages

We aged 205 YCT, 154 HYB, 92 BKT, and 229 Utah Chub. Ages ranged from age to for YCT, age 2 to 8 for HYB, age-1 to 4 for BKT, and age- 1 to 6 for Utah Chub (Table 11). Mean lengths for age-2 fish were slightly higher for phenotypically-identified HYB at 326 mm TL (range 218 to 415 mm) compared to YCT at 304 mm TL (range 225 to 500 mm; Table 12). The same trend was evident for age-3 HYB and YCT.

Hybrid evaluation

We found orange (Hayspur) fluorescent marks on one Hybrid Trout in 2019 (Table 12). This trout was captured using one of our randomized gillnets, had a total length of 608 mm and was 5 years-old.

Parentage based tagging

A combined total of 1,171 YCT and HYB genetic samples from our gill-net surveys were analyzed for parentage. Of the samples collected, 248 samples were genetically identified as HYB, 877 as YCT, and 46 (3.9%) failed to genotype. For the YCT genotyped samples, 162 fish exhibited total lengths >370 mm indicating these fish were older than 2 years and would not be able to assign to a PBT brood year. This left a total of 715 YCT which should assign to BY2017 or BY2018. A total of 704 YCT were assigned to two hatchery parents for BY2017, leaving 11 fish which failed to assign. This indicates the wild contribution of YCT for 2019 was 1.5%.

Winter dissolved oxygen

Total dissolved oxygen diminished from 41 to 18.8 g/m^2 at the Pittsburgh Creek site, from 32.9 to 17.8 g/m^2 at the Wild Rose site, from 25.2 to 11.4 g/m^2 at the County dock site, from 14.2 to 13.3 g/m^2 at the Hatchery site, and from 25.05 to 9.05 g/m^2 at the Outlet site (Table 13). The Hatchery site was sampled additionally during the spawning season once in both February and March showing a further depletion to 10.75 and 7.35 g/m^2 respectively. Depletion estimates indicated dissolved oxygen would remain below the level of concern throughout the winter and no aeration was initiated (Figure 44).

Creel

We conducted 920 interviews during which residents composed 75% of the anglers. Total angler effort over the entire season was 207,989 hours. Season angler catch rate was 1.09 fish/hour, and exceeded our management goal of 0.7 fish/hour (Figure 45). The ice fishery had a slightly higher catch rate than the open water fishery, (1.28 and 1.06 fish/hour, respectively). Anglers caught an estimated 227,490 trout with 183,484 (80.7%) and 44,006 (19.3%) caught during the open water and ice fishery, respectively. Species composition of angler catch was 45% Hybrid Trout, 39% Yellowstone Cutthroat Trout, and 15% Brook Trout. A total of 33,109 trout were harvested during the entire season (Table 14), including 24,773 open water during the open water fishing season, and 8,339 during the ice fishing season. Of the trout measured by creel clerks, 16% of the Brook Trout were larger than 450 mm TL, 25% of the Hybrid Trout were larger than 500 mm TL, and 20% of the Yellowstone Cutthroat Trout were larger than 500 mm TL (Table 14).

DISCUSSION

Gill-net monitoring surveys have indicated total trout CPUE in the 2019 traditional gill nets increased from the all-time low observed in 2018. Although, this catch rate is still less than both the 25 year average and our management goal of 11 trout per net night (IDFG 2019). This increase in trout abundance was coupled with a decline in trout relative weights indicating a decrease in trout condition. Various factors may be limiting trout growth in the lake such as changing abiotic factors (ex. temperature and dissolved oxygen, water-level fluctuation; Johnson et al 1992), decreased or community shift in the forage base (Flickinger and Bulow 1993), or intraspecific and interspecific competition (Blackwell et al 2000). It is not likely that abiotic factors have led to the decrease in relative weights as water quality monitoring on the lake has shown high water quality and quantity throughout the year. This monitoring program also estimated high densities of phytoplankton and zooplankton in the lake. In 2018, we found amphipods, instead of zooplankton to dominate the diet of recently stocked fingerling YCT in Henrys lake (IDFG 2018 in progress), suggesting a macroinvertebrate survey of the lake is warranted to determine current prey densities. Another likely factor causing reduced condition is the increase in stocking rates of hatchery trout over the past five years. In years where trout densities are low, trout condition improves in Henrys Lake. Managers should decrease stocking rates in the lake when water quality parameters are favorable, leading to average or above average condition.

Gill net catch at the randomly selected sites was higher for all trout species and captured a more inclusive range of size classes. These nets captured trout at the lower and upper thresholds of size classes missing from the traditional net catch. One major factor in this difference between net catches is the netting locations in the lake. Our traditional netting sites only encompass the pelagic zone of the lake whereas the randomized net locations were spread

throughout the lake to encompass both the pelagic and near-shore habitats. Guzzo et al (2017) found that directly following ice-off, Lake Trout were able to access the littoral zones of a lake due to the cold-water temperatures and as the water temperatures increased, trout began to seek thermal refuge in deeper portions of the lake. This movement is apparent for trout in Henrys Lake, which occupy near shore habitats in the weeks following ice-off. As such, it is not surprising that the randomized nets yielded higher catch rates and a wider range of size classes of trout.

In addition, the proportional stock density (PSD), and both relative stock density (RSD-400 and RSD-500) indices were higher for the randomized nets than the traditional nets for BKT and YCT, while the traditional nets yielded higher percentages in each index for HYB. This indicates that quality-sized BKT and YCT are more likely to occupy near-shore habitats directly following ice off while quality-sized HYB trout occupy deeper habitats in the early spring. In general, high PSD values relate to low population densities of faster growing fish and are associated with high relative weight values (Anderson 1978). This is apparent in our randomized netting where the average relative weight for each species was also higher than the traditional nets. An additional year of net location comparison is warranted to compare and allow for a shift to a randomized netting design if these net locations continued to yield notable differences in species composition, age structure, stock densities and relative weights.

A large component of the catch was age-2 Yellowstone Cutthroat Trout. This increase in trout catch may be partly due to the increased stocking rates of YCT over the last three years (Appendix F). However, age-2 YCT did not dominate the population of the lake in 2018 or 2017 to the degree observed in 2019. In addition to increased stocking, high water quality and quantity throughout 2018 may have bolstered the age-2 YCT population through increased survival throughout the year. Both air and water temperatures during the summer of 2018 and 2019 were lower than those observed in years prior. This is likely a factor in reducing the occurrence and severity of cyanobacteria blooms, as only one small fall bloom was observed in 2018 compared to the large summer blooms observed in 2015, 2016, and 2017. The Henrys Fork Watershed experienced a high water year in 2018 with high snowpack allowing for continual introduction of cold, well-oxygenated water into the lake later into the summer. During years of low snowpack and increased summer temperatures, there is a lack of thermal refuge for Trout in Henrys Lake. This is due to high wind action and the shallow nature of Henrys Lake which leads to a thermal stratification in the lake. In addition, no large draw down of water for irrigation from the lake was observed with the lowest capacity over 2018 and 2019 at 85% (USGS). Lower summer water temperatures, high concentrations of dissolved oxygen and increased quantity of water are all essential to the growth and survival of trout (Selong et al 2001), indicating high survival rates for stocked trout in 2017.

We have limited inferences on Utah Chub densities due to high variability in gill net catch (i.e. high variance). Utah Chub abundance first began to increase in Henrys Lake in the late 1990s, following an initial documentation of their presence in 1993. Even though Utah Chub densities have decreased from 2018 we continue to be concerned about potential growth of Utah Chub population in Henrys Lake. Utah Chub are currently the most numerous species caught in our gillnet surveys most likely due to their high reproductive potential. There is evidence of competition between chubs and trout for both food resources and space in many lakes and reservoirs. In Scofield Reservoir, Utah, potential competition has been noted between young Utah Chub and Cutthroat Trout (Johnson and Belk 2006). The lower Utah Chub catch observed in the randomized nets may be due to their schooling nature and preference for the pelagic zone of lakes (Winters 2014). Due to their schooling behavior, our traditional nets will commonly catch a school of chub or there will be no chub present, and these nets are generally set in the deeper locations of the lake.

Furthermore, literature suggests that the thermal tolerance for Utah Chub is between 15 – 31°C (Sigler and Sigler 1987), while temperatures over 20°C can cause lethality in Cutthroat Trout (Bear et al 2007). This higher thermal threshold for Utah Chub may allow this species to feed and grow when summer temperatures limit the activity of Cutthroat Trout. In the future, the interaction between Utah Chub and YCT in Henrys Lake should be monitored on a continual basis to ensure the lake is maintained as a trophy trout fishery.

To maintain Henrys Lake as a trophy trout fishery and to provide the best angling experience for our constituents, we conducted a study evaluating two strains of male Rainbow Trout to fertilize YCT and produce the Henrys Lake hybrid trout. Our goal was to guide future stocking efforts on which strain of hybrid trout is best suited for Henrys Lake by comparing egg eye-up rates, fry survival, and the post-stock growth and survival of these trout. The Hayspur-strain HYB were found to be significantly longer in total length at age-2 (IDFG 2017 in progress), while there was no significant difference between age-3 fish (IDFG 2018 in progress). The cost-benefit of using the Hayspur HYB coupled with the increased growth rate observed this HYB, since 2016, all HYB trout stocked into Henrys Lake have been of the Hayspur strain. This year only one hybrid was found with fluorescent grit in our gill net catch. This fish, based on age and grit color was identified as a Hayspur HYB from BY2015. The low number of marked fish captured is expected as Henrys Lake is managed as a trophy fishery. After five years, it is expected that many of these fish have been lost to either natural mortality or harvest. The capture of a marked Hayspur HYB shows the potential for our hybrid trout to persist in the lake and provide long-term angling opportunities for the public.

Angler catch rates for the 2019 season were the highest observed over the last decade and exceeded our management goal (IDFG 2019). Total angler effort, catch and harvest rates were significantly higher than the last two creel surveys conducted in 2016 and 2013. High water quantity and quality in both 2018 and 2019 coupled with increased hatchery stocking were likely the main factors influencing the survival of trout in Henrys Lake as outlined above and consequently increased angler catch rates. The release of cold, well oxygenated water from snowpack later into the season, generated a productive fishing experience later into the summer than in previous years. The reduced thermal and oxygen stress during the summer allowed trout to be more active and focus on growth and reproduction leading to increased angler opportunity.

Wild YCT contribution to the lake was estimated as 1.5% for 2019 which is similar to the estimate of 1.6% for 2018. Prior to 2017, 10% of the stocked YCT in Henrys Lake were marked by an adipose fin clip. In the last three years prior to Parentage Based Tagging (PBT; 2014-2016), the ratio of adipose clipped YCT to unclipped trout was above or equal to 10%, although an exact percentage of the contribution was not possible with this method. Although there is some error associated with mass marking techniques including miss clips (partial or no clip), improper identification of miss clips and the low rate at which fish are clipped. Only 10% of approximately one million stocked YCT each year were identified as hatchery origin trout. These sources of error have likely led to an over-estimation of the wild YCT contribution to the lake. From 2009 to 2013, the average ratio was 5% indicating an increase in wild YCT contribution. This increase in wild contribution was likely due to the habitat restoration projects which have occurred over the last two decades on the major tributaries of Henrys Lake (High et al. 2014). These projects have included fish passage improvements, irrigation canal screening and riparian fencing. Favorable stocking conditions and water quality over the last 2 seasons have likely increased the survival of hatchery trout offsetting the limited wild production of YCT in the lake. PBT is a cost effective method which allows for mass marking of all YCT stocked into Henrys Lake. This coupled with our continued gill net and juvenile trout surveys will allow us to answer a seemingly unlimited number of management questions which can help inform our stocking methods (ex. size, time of

year), habitat restoration projects, population analysis, and spawning operations.

MANAGEMENT RECOMMENDATIONS

1. Continue annual gill-net surveys at historical sampling locations at 50 net nights of effort.

2. Complete at least an additional year of 50 net nights of effort at randomized sites to determine if randomized set locations capture a more representative catch and establish a correction factor between our historical gillnet survey data to transition to a randomized site gillnet survey .

3. Collect otolith samples from all trout species caught during gill-net surveys to conduct dynamic rate analysis.

4. Continue to monitor Utah Chub densities and evaluate potential impacts of increased densities of Utah Chub on trout.

5. Collect fin rays from Utah Chub for aging and mortality estimates.

6. Utilize Parentage Based Tagging (PBT) to evaluate the percentage of wild production in Henrys Lake.

7. Conduct a macroinvertebrate survey to determine current community structure and densities of trout forage base in the lake.

8. Continue to monitor winter dissolved oxygen levels to determine when using the aeration system is required.

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Table 9. Summary statistics of total length (mm), weight (g), and relative weights (W_R) for Brook Trout (BKT), Hybrid Trout (HYB), Yellowstone Cutthroat Trout (YCT), and Utah Chub (UTC) collected using gillnets set at traditional netting locations in Henrys Lake, 2019.

	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	334	478	96	404	941	96	326	429	90	230	184	92
Confidence level (95.0%)	19	90	6	28	272	3	10	56	0.9	4.3	10	0.6
Median	350	463	91	364	501	97	311	314	90	220	131	92
Minimum	162	47	67	288	268	78	168	41	72	134	29	29
Maximum	470	1,813	188	699	4,541	123	625	2,680	109	363	638	154
Count	44	44	44	56	56	56	219	219	219	696	696	695

Table 10. Stock density indices (PSD, RSD-400, and RSD-500) and relative weights (W_r) for all trout species collected in the traditional (T) and randomized (R) gill nets locations in Henrys Lake, Idaho 2019.

	BKT		HYB		YCT		UTC	
	T	R	T	R	T	R	T	R
PSD	83	92	96	90	65	78	--	--
RSD-400	5	7	32	29	12	21	--	--
RSD-500	0	4	16	19	6	7	--	--
W_r								
<200 mm	133	99	--	97	81	80	90	91
200-299 mm	88	95	99	93	88	91	93	96
300-399 mm	94	98	96	95	91	92	93	95
>399 mm	126	108	98	100	89	88	--	--
Mean	96	98	96	97	90	91	92	94

Table 11. Summary statistics of total length (mm), weight (g), and relative weights (W_R) for Brook Trout (BKT), Hybrid Trout (HYB), Yellowstone Cutthroat Trout (YCT), and Utah Chubs (UTC) collected using randomized gillnets at Henrys Lake, 2019.

	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	349	562	98	391	849	96	347	513	91	223	174	94
Confidence level (95.0%)	10.5	64.2	2.0	12.8	106.3	1.2	5.8	33.9	0.8	5.29	12.3	1.2
Median	356	519	97	351	457	95	320	345	91	215	129	93
Minimum	169	46	69	218	103	46	1699	42	22	133	3	6
Maximum	574	2,662	148	713	4,424	134	661	3,319	268	392	627	265
Count	137	137	137	256	256	256	684	684	684	448	448	448

Table 12. Mean length-at-age data based on otoliths from Yellowstone Cutthroat Trout (YCT), Hybrid Trout (HYB), and Brook Trout (BKT); and mean length-at-age based on pectoral fin rays for Utah Chub (UTC) captured with combined samples from traditional and randomized gill nets in Henrys Lake, Idaho 2019. Mean length-at-ages were estimated using nonlinear regression.

		Age							
Species	Summary statistic	1	2	3	4	5	6	7	8
YCT	Mean TL (mm)	187	304	414	501	530	536	639	--
	Min TL	168	225	333	406	436	524	--	--
	Max TL	236	500	508	633	661	545	--	--
	No. Analyzed	6	84	30	52	29	3	1	--
HYB	Mean TL (mm)	--	326	433	490	587	645	642	617
	Min TL	--	218	328	393	445	545	592	--
	Max TL	--	415	575	625	643	713	699	--
	No. Analyzed	--	64	19	35	23	9	4	1
BKT	Mean TL (mm)	194	338	441	536	--	--	--	--
	Min TL	162	224	280	509	--	--	--	--
	Max TL	241	435	535	574	--	--	--	--
	No. Analyzed	12	69	8	3	--	--	--	--
UTC	Mean TL (mm)	167	149	179	216	239	297	--	--
	Min TL	--	135	134	150	165	172	--	--
	Max TL	--	178	316	273	315	363	--	--
	No. Analyzed	1	15	43	23	38	109	--	--

Table 13. Mean total length (mm), weight (g), and (W_R) by year, strain, and age of Hybrid Trout collected in the spring gillnetting at Henrys Lake, 2016 – 2019, that exhibited fluorescent marks.

				Mean						Range		
Year	Strain	Age	N	TL (mm)	± 95% CI	WT (g)	± 95% CI	W_R	± 95% CI	TL (mm)	WT (g)	W_R
2016	Hayspur	1	5	191	± 39.8	68	± 39.4	87	± 10.6	152-227	37-102	80-101
2017	Gerrard	1	1	274		221		97		--	--	--
		2	8	343	± 27.8	444	± 110.4	95	± 6.7	275-376	207-621	82-107
	Hayspur	1	2	200	± 501.9	87	± 622.6	90	± 18.1	160-239	38-136	88-91
		2	20	379	± 13.8	679	± 71.5	108	± 4.1	327-429	386-913	91-123
2018	Gerrard	2	2	340	± 76.2	435	± 114.4	98	± 93.7	334-336	426-444	91-105
		3	4	404	± 87.1	801	± 631.8	99	± 11.6	348-479	432-1434	90-106
		4	1	506		1493		98		--	--	--
	Hayspur	2	2	363	± 25.4	641	± 972.0	117	± 152.8	361-365	717-1281	105-129
		3	2	328	± 95.3	456	± 203.3	115	± 52.4	320-335	440-472	111-119
		4	2	537	± 19.1	1,947	± 883.1	106	± 36.6	535-538	1,877-2,016	104-109
2019	Hayspur	5	1	608		3460		128.3		--	--	--

Table 14. Dissolved oxygen (DO) (mg/l) levels recorded in Henrys Lake, Idaho winter monitoring 2018 – 2019.

Location	Date	DO Ice bottom	DO 1 meters	DO 2 meters	DO 3 meters	DO 4 meters	DO 5 meters	Total g/m^2
Pittsburgh Creek	Dec. 18, 2018	12	11.3	10.7	8.4	5.8	5.1	41.65
	Jan. 8, 2019	11.2	10.2	8.4	5.9	4.0	2.1	31.1
	Jan. 16, 2019	11.1	10.2	8.2	4.5	2.8	16	27.75
	Jan. 24, 2019	9.9	8.2	5.4	3.2	0.7	0.4	18.8
Outlet	Dec. 18, 2018	22.9	10	9.5	5.8	4.8	4.7	25.05
	Jan. 8, 2019	9.7	9.5	7.7	4.6	0.2		22.3
	Jan. 16, 2019	--	--	--	--	--		--
	Jan. 24, 2019	8.8	3.9	1.7	1.0			9.05
County Ramp	Dec. 18, 2018	11.7	10.9	7.4	3.7	2.8		25.2
	Jan. 8, 2019	8.8	7.2	3.5	2	1	0.2	14.7
	Jan. 16, 2019	--	--	--	--	--	--	--
	Jan. 24, 2019	9.4	5.4	3.3	0.7			11.4
Wild Rose	Dec. 18, 2018	11.3	10.5	9.8	7.5	3.1	1.6	32.9
	Jan. 8, 2019	10.4	8.9	5.3	2.3	1		18.25
	Jan. 16, 2019	--	--	--	--	--		--
	Jan. 24, 2019	10.3	9.1	5.0	2.8	0.3		17.8
Hatchery	Dec. 18, 2018	9.4	8.4	5.3				14.2
	Jan. 8, 2019	11.6	7.8	6.2	2.2	0.9		19
	Jan. 16, 2019	11.3	7.1	4.5	2	1.4		17.1
	Jan. 24, 2019	10.1	5.9	3.4	1.2	0.7		13.3
	Feb. 20, 2019	9.0	5.5	2.6	0.7	0.2		10.75

Table 15. Annual estimates of angler effort, catch and harvest collected from creel surveys on Henrys Lake, Idaho.

Year	Effort (*1,000)	No. Caught (*1,000)	No. Harvested (*1,000)	Total CR[a]	Harvest CR[a]	% Released	Catch composition (%)			% Exceeding goals			Mean size (mm)			Residency (%)	
							YCT	HYB	BKT	YCT (500 mm)	HYB (500 mm)	BKT (450 mm)	YCT	HYB	BKT	Res	Non Res
1950	17	--	12.3	0.82	0.72	12	77	0	23	--	--	--	--	--	--	--	--
1951	27.9	--	12.3	0.49	0.44	12	80	0	20	--	--	--	--	--	--	--	--
1971	102.2	--	36.7	0.36	0.36	0	70	14	16	--	--	--	--	--	--	--	--
1972	83.8	--	27	0.32	0.32	0	69	19	12	--	--	--	--	--	--	50	50
1975	86.3	--	29.9	0.38	0.35	10	89	0	11	--	--	--	--	--	--	49	51
1976	68.1	36.7	18.7	0.54	0.27	49	81	<1	19	2	--	2	426	--	371	50	50
1977	66.1	29.2	16.5	0.44	0.25	44	71	<1	29	4	--	4	420	339	362	50	50
1978	85.3	40.5	25.5	0.48	0.3	32	48	20	33	9	--	9	429	389	381	51	49
1979	93.9	29.8	18.7	0.32	0.2	37	35	42	24	11	8	6	452	456	378	53	47
1980	68.5	14.6	9.2	0.21	0.14	37	31	59	10	11	16	5	429	459	391	67	33
1981	65.9	14.2	7.5	0.21	0.11	47	30	54	16	13	11	19	445	450	389	--	--
1982	63.3	28.7	7.1	0.45	0.11	75	62	25	13	7	17	25	416	451	405	--	--
1983	96	122	25.4	1.23	0.23	81	84	9	7	3	14	17	388	448	392	64	36
1984	162.9	271	47	1.7	0.29	83	92	5	3	1	5	30	388	427	393	64	36
1985	125.7	159.4	37.9	1.3	0.3	76	92	4	4	0	0	0	378	416	364	60	40
1986	172.8	154.7	67.7	0.9	0.39	55	85	14	1	0	12	0	407	441	364	--	--
1987	150.2	81.1	35.7	0.54	0.24	56	60	34	6	5	26	3	436	447	371	--	--
1988	100.5	81.6	19.5	0.82	0.2	76	49	39	12	8	17	21	430	432	383	--	--
1989	340	262.5	103.7	0.77	0.31	60	50	45	5	4	11	10	404	435	387	--	--
1990	344.2	174.5	63.1	0.51	0.18	64	53	41	5	2	24	0	427	461	433	--	--
1991	124.4	50.5	16.1	0.36	0.13	68	49	49	2	21	35	20	460	473	369	--	--
1992	115.5	53	12.2	0.45	0.11	72	38	52	10	27	42	22	452	474	417	--	--
1993	144.3	92.5	26.7	0.64	0.18	71	76	21	3	7	35	23	410	485	382	--	--
1994	177.8	116.6	21	0.66	0.12	82	52	43	5	5	15	29	418	437	425	71	29
1995	172.6	99.3	20.6	0.58	0.12	79	37	60	3	9	21	27	434	442	432	65	35
1997	228.9	127.7	32.4	0.54	0.25	74	51	46	3	5	15	9	423	434	389	--	--
1999	228	148.6	27.3	0.65	0.12	72	22	65	13	8	12	16	442	447	405	--	--
2001	165.8	93.3	17.7	0.56	0.11	81	35	58	7	12	57	43	447	503	452	--	--

Table 15 (continued)

Year	Effort (*1,000)	No. Caught (*1,000)	No. Harvested (*1,000)	Total CR[a]	Harvest CR[a]	% Released	Catch composition (%)			% Exceeding goals			Mean size (mm)			Residency (%)	
							YCT	HYB	BKT	YCT (500 mm)	HYB (500 mm)	BKT (450 mm)	YCT	HYB	BKT	Res	Non Res
2002	--	--	--	0.41	--	--	42	49	9	17	71	50	454	540	462	--	--
2003	108.5	16.9	5.4	0.17	0.05	68	45	51	4	18	65	82	476	543	464	68	32
2005	95	45	8.9	0.48	0.1	80	53	42	5	4	38	0	413	497	379	66	34
2009	124.6	78.9	13.8	0.63	0.11	83	49	41	10	5	50	55	450	502	419	75	25
2010[c]	3.8	5.6	0.8	1.48	0.21	86	52	15	33	15	39	33	469	509	425	92	8
2011[b]	18.3	13.5	2.7	0.74	--	80	47	20	32	--	--	--	--	--	--	91	9
2013	19.1	17.6	2.4	0.95	0.12	86	58	28	14	4	27	27	424	465	418	74	26
2016	7.5	1.6	0.5	0.36	0.11	69	57	32	11	28	67	63	471	532	468	75	25
2019	207.9	227.5	33.1	1.09	0.16	85	39	45	15	20	25	16	442	458	413	75	24

[a] = Total catch rate and harvest rate expressed as fish per hour.

[b] = Creel survey conducted from 11/21/10 through 11/30/11.

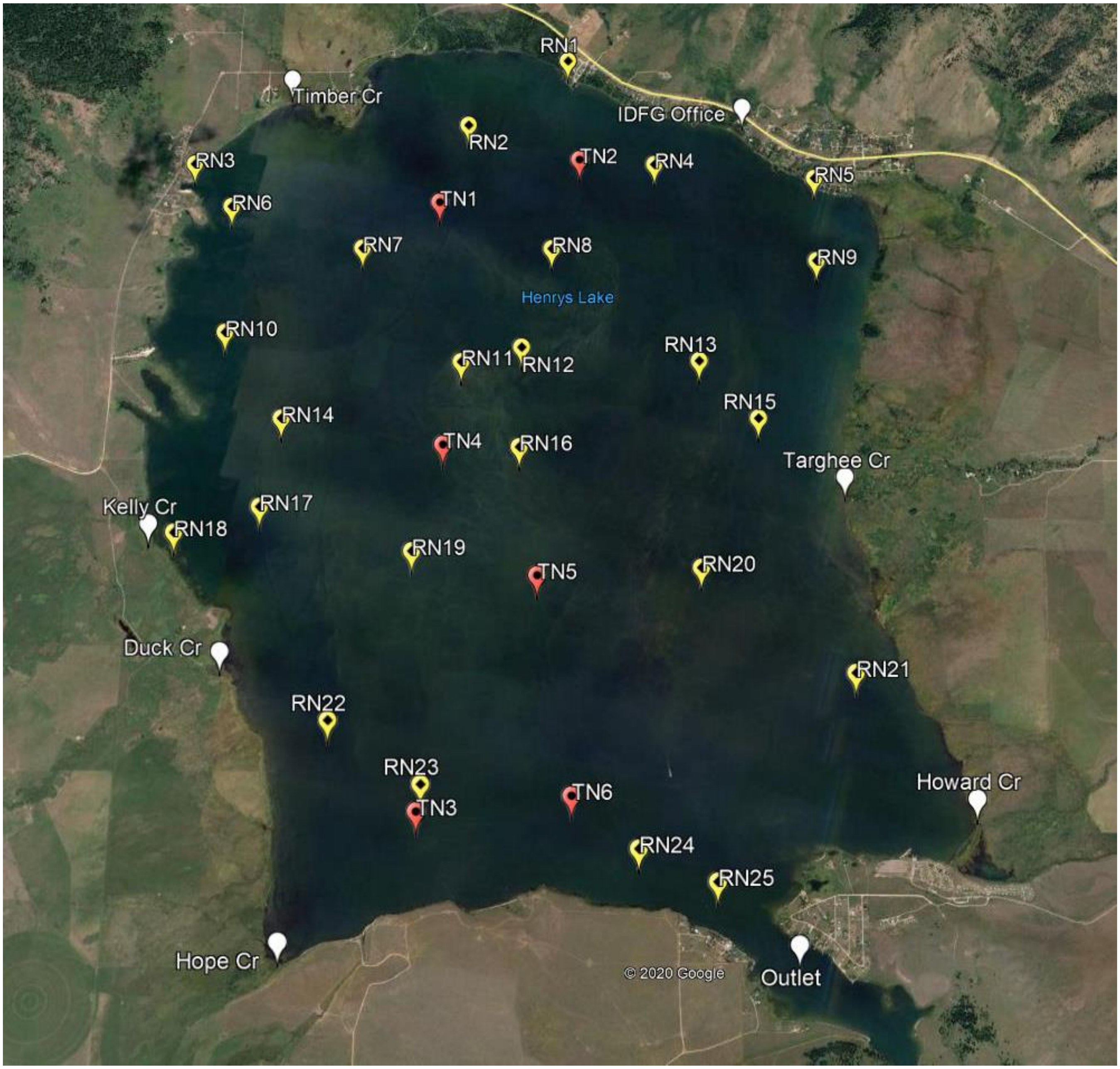

Figure 36. Spatial distribution of traditional gill net (TN 1 through 6; Red), randomized gill net (RN 1 through 25; Yellow) locations and the major tributaries in Henrys Lake, Idaho, 2019.

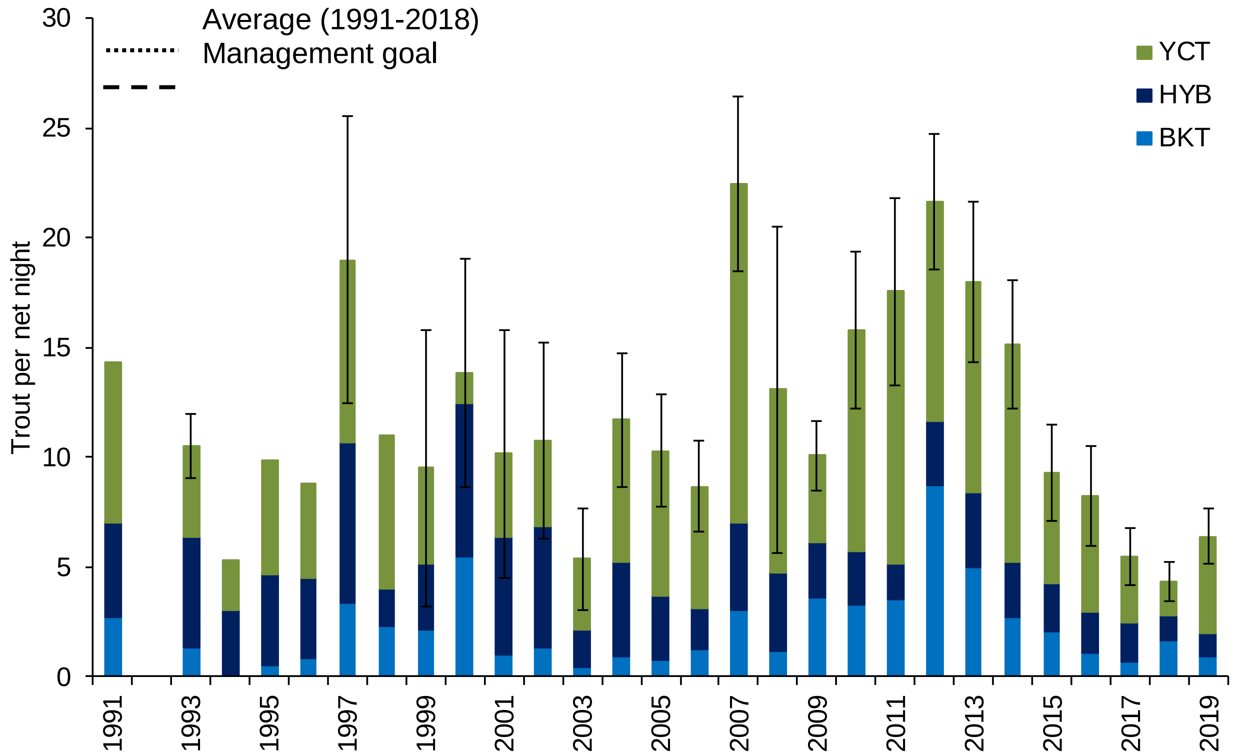

Figure 37. Catch per unit effort (CPUE) of trout per net night of traditional gillnetting sites for Yellowstone Cutthroat Trout (YCT), Hybrid Trout (HYB), and Brook Trout (BKT) in Henrys Lake, Idaho from 1991 – 2019. Error bars represent 95% confidence intervals. Lines represent the average gillnetting CPUE from years 1991 – 2018 (dashed line) and management target of 11 trout per net night (dotted line).

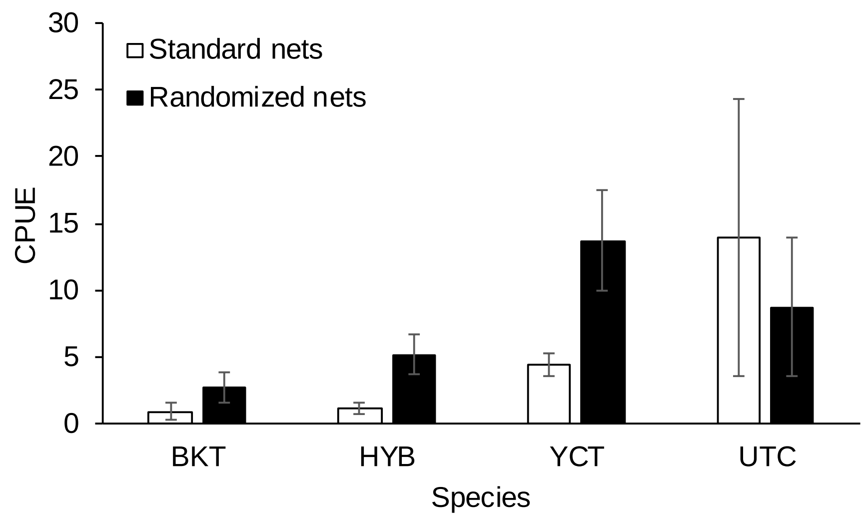

Figure 38. Catch per unit effort (CPUE) of trout per net night of traditional and randomized gill netting sites for Brook Trout (BKT), Hybrid Trout (HYB), Yellowstone Cutthroat Trout (YCT), and Utah Chub (UTC) in Henrys Lake, Idaho, 2019. Error bars represent 95% confidence intervals.

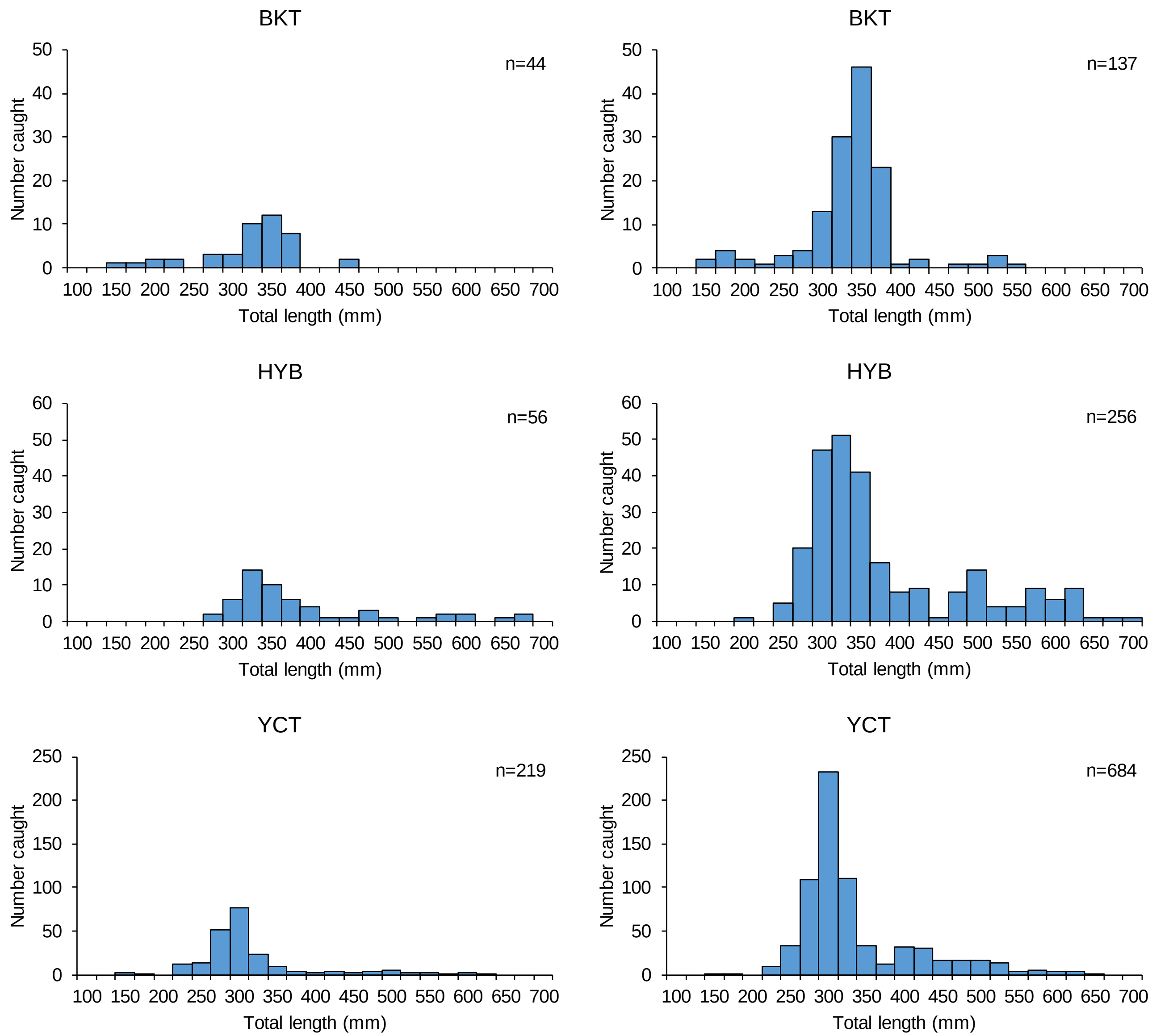

Figure 39. Brook Trout (BKT), Hybrid Trout (HYB) and Yellowstone Cutthroat Trout (YCT) length frequency distribution from Left panel: traditional gill nets, and right panel: randomized gill nets set in Henrys Lake, Idaho, 2019.

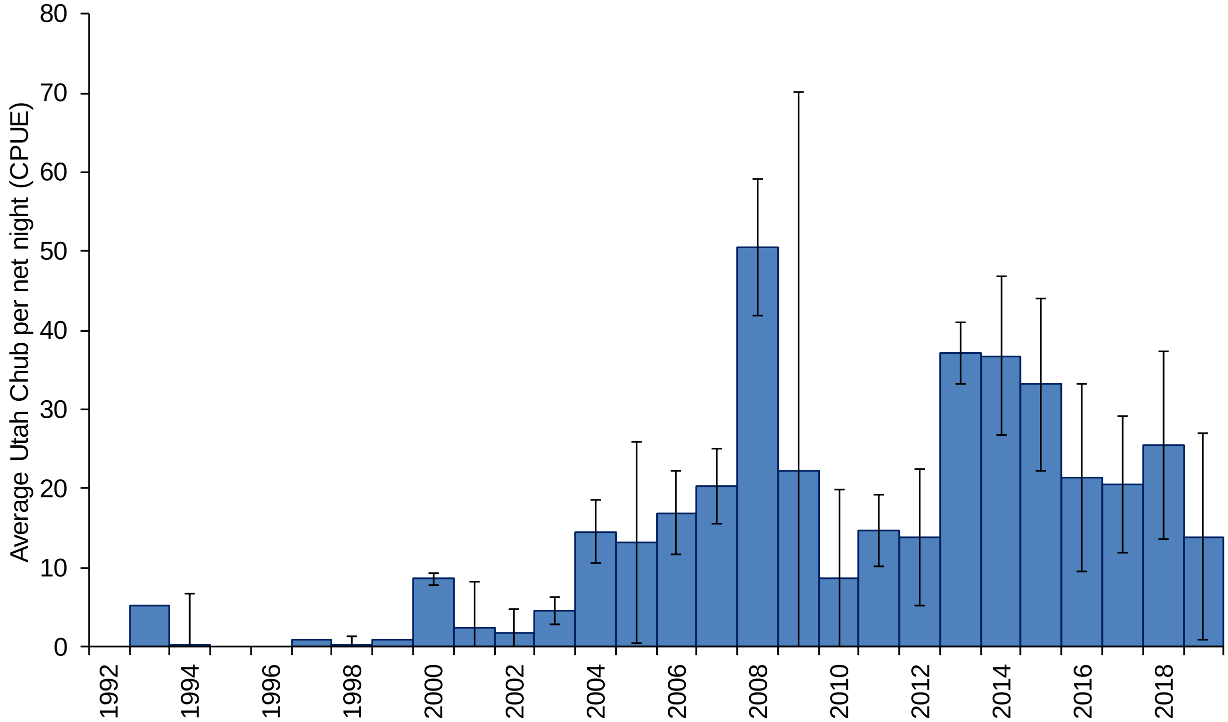

Figure 40. Catch per unit effort (CPUE) for Utah Chub in Henrys Lake, Idaho between 1991 and 2019 using traditional gill nets set in Henrys Lake. Error bars represent 95% confidence intervals and the dashed line represents the average gill netting CPUE from years 1991 to 2018.

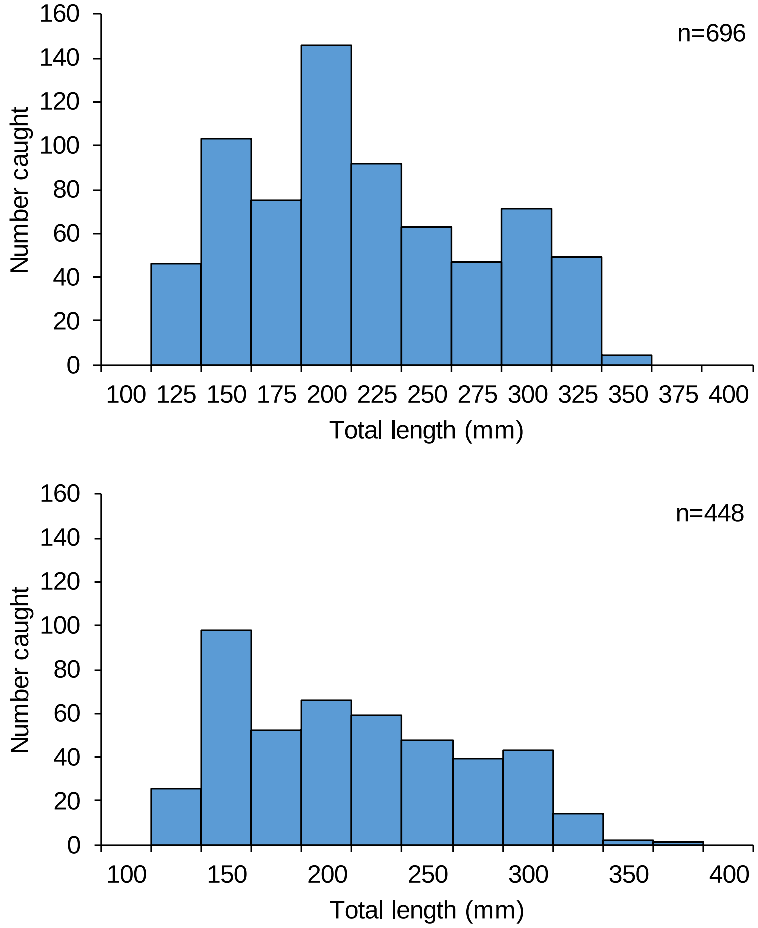

Figure 41. Utah Chub length frequency distribution from Top: traditional gill net sites, and Bottom: randomized gill net sites set in Henrys Lake, Idaho 2019.

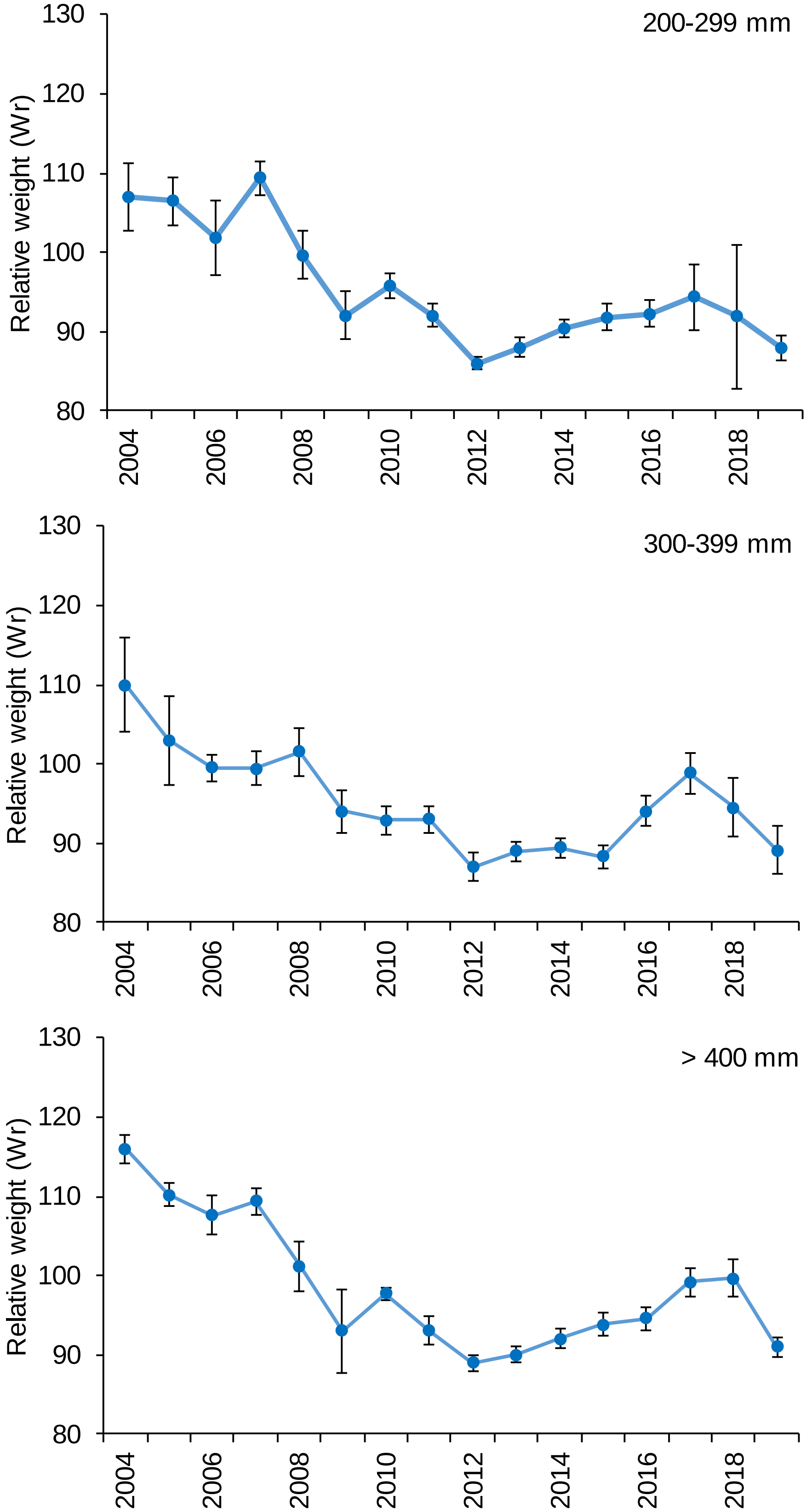

Figure 42. Relative weights (W_r) for three size classes (200 – 299 mm, 300 – 399 mm, and > 400 mm) of Yellowstone Cutthroat Trout from traditional gill netting in Henrys Lake, 2004 – 2019. Error bars represent 95% confidence intervals.

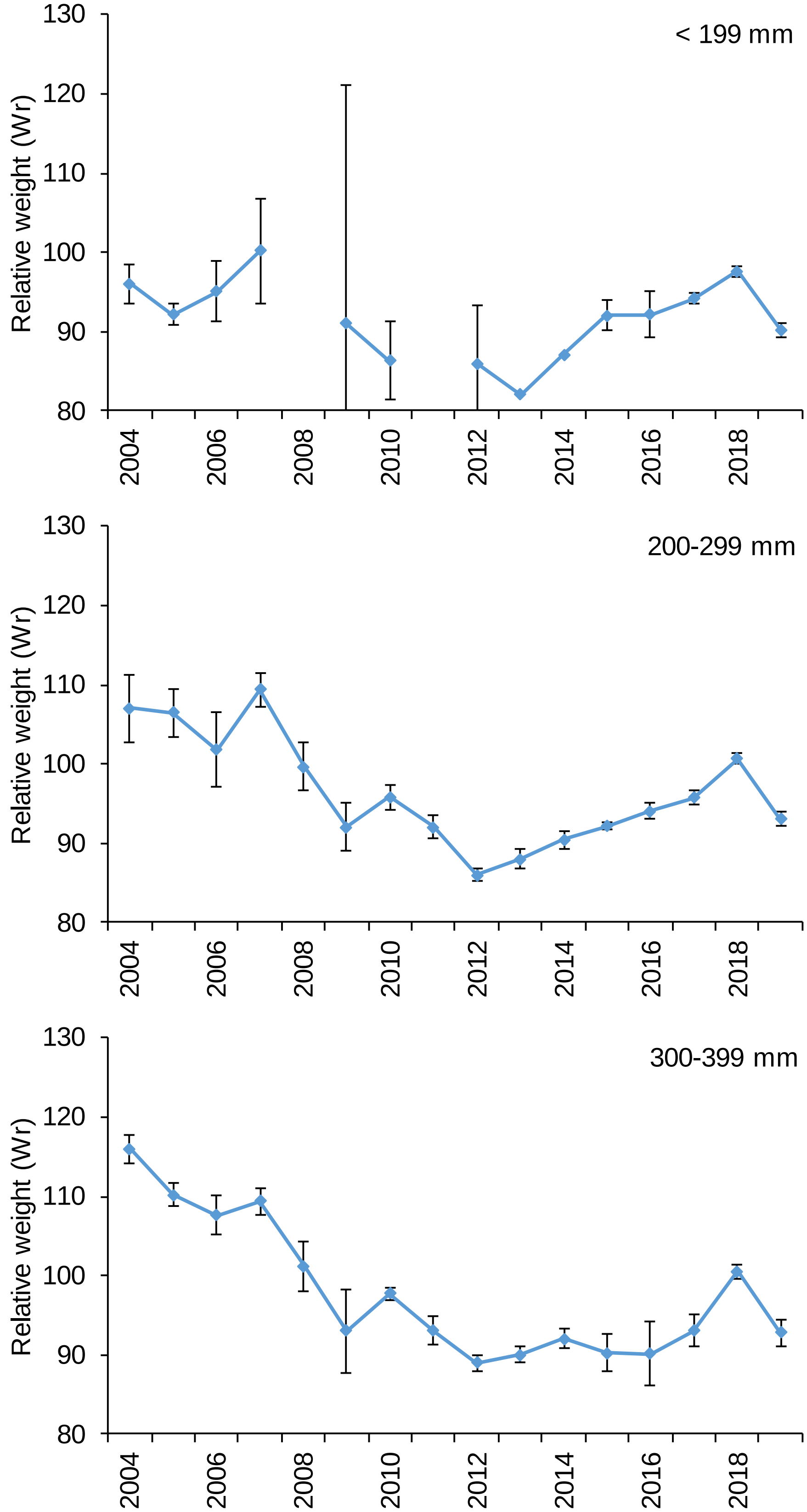

Figure 43. Relative weights (W_r) for three size classes (< 199 mm, 200 – 299 mm, 300 – 399 mm) of Utah Chub from traditional gill netting in Henrys Lake, 2004 – 2019. Error bars represent 95% confidence intervals.

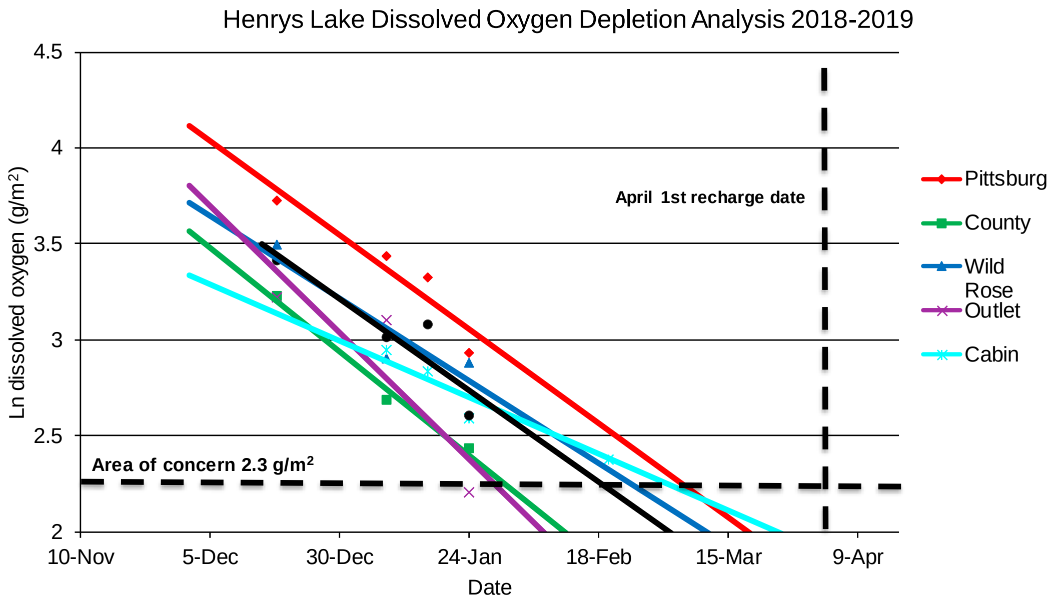

Figure 44. Dissolved oxygen depletion estimates from Henrys Lake, Idaho, 2018 – 2019. Dotted lines indicate dissolved oxygen levels indicating area of concern and recharge date (April 1).

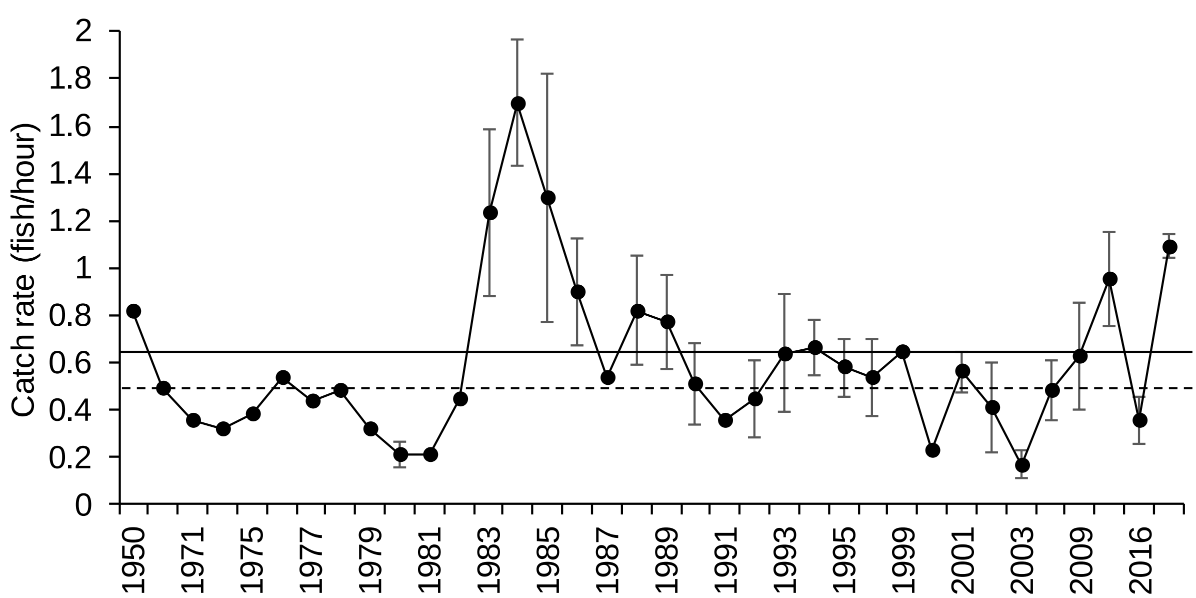

Figure 45. Angler Catch rates (fish per hour) with 95% confidence intervals from 1950 to 2019 on Henrys Lake, Idaho. Dotted and dashed lines represent the long-term average and management target (0.7 fish per hour), respectively.

RIRIE RESERVOIR

ABSTRACT

We conducted our fifth year of monitoring the kokanee *Oncorhynchus nerka* population in Ririe Reservoir using experimental gill nets suspended in the thermocline. Average catch-per-unit-effort (CPUE) of kokanee was 51 fish/net-night ± 58 (estimate ± 95% CI), which was lower than catch rates in 2018 (i.e., 89 fish/net-night ± 39) and lower than the 4-year average (i.e., 2014-2018; 72 fish/net-night ± 54). Kokanee composed the majority of the overall species composition (i.e., 46%) followed by Yellow Perch (i.e., 45%). 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. We also conducted creel surveys from May through October of 2019. Creel clerks interviewed 378 anglers in 256 parties with the majority of anglers fishing from boats (i.e., 70%). Total angling effort for the season was estimated at 90,024 hours. Total catch for all species combined was estimated to be 74,534 fish with a combined species catch rate of 0.83 fish/h. Total harvest of all species combined was 53% while 47% of all fish caught were released.

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INTRODUCTION

Ririe Reservoir is located on Willow Creek, approximately 32 km east of Idaho Falls (Figure 46). Ririe Dam was constructed in 1977, with the reservoir being filled to capacity for the first time in 1978. Ririe Reservoir is fed by approximately 153 km of streams in the Willow Creek drainage, and has a total storage capacity of 100,541 acre-feet. Ririe Reservoir is approximately 17-km long, is less than 1.5-km wide along the entire length, has a surface area of approximately 631 ha, and mean depth of 19.5 m. Ririe Reservoir is managed primarily for flood control and irrigation storage (BOR 2001).

Ririe Reservoir supports a popular fishery for kokanee *Oncorhynchus nerka*, Yellowstone Cutthroat Trout *O. clarkii bouvieri*, Rainbow Trout *O. mykiss*, Smallmouth Bass *Micropterus dolomieu*, and Yellow Perch *Perca flavescens*. Utah Chub *Gila atraria*, Utah Sucker *Catostomus ardens*, and Bluehead Sucker *Catostomus discobolus* are also found in Ririe Reservoir in relatively high numbers. In our 2013 creel survey, we estimated angler use to approximately 43,000 hours, and we estimated that the reservoir had averaged 47,000 hours of angler use during the previous 20 years (High et al. 2015). Since 1990, fingerling kokanee have been stocked annually in the spring. In 2004, we increased the total number of kokanee stocked in the reservoir from approximately 70,000 to 210,000 in an effort to improve catch rates and meet increased angler demand. Very little natural reproduction of kokanee occurs in Ririe Reservoir, therefore the kokanee fishery is supported by the number of kokanee that we stock. From 2014 to 2018, kokanee stocking numbers were increased an additional 50,000 to 110,000 to approximately 260,000 to 320,000 fingerlings per year (e.g., about 475 kokanee/ha). Up until 2012, approximately 18,000 catchable-sized Yellowstone Cutthroat Trout were stocked annually to provide angler opportunity. Following relatively poor performance of those fish (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 an effort to diversify the fishery for anglers even more, approximately 1,700 catchable-sized Tiger Trout *Salmo trutta* × *Salvelinus fontinalis* were stocked in 2019. A Yellow Perch fishery also exists in Ririe Reservoir and has become more popular over the past several years as spring reservoir levels have remained high with a resultant increase in abundance of perch (Schoby et al. 2010). A self-sustaining population of Smallmouth Bass has developed from purposeful introductions into Ririe Reservoir from 1984-1986. Although limited by the short growing season at this latitude and elevation (Dillon 1996), Smallmouth Bass provide angling diversity for anglers in the Upper Snake Region.

OBJECTIVES

1. Use annual summer gill netting to describe size structure, age, and growth of kokanee in Ririe Reservoir to assist in developing appropriate stocking rates.

2. Estimate the relative abundance, size structure, and age of Yellow Perch in Ririe Reservoir to describe the Yellow Perch fishery.

3. Evaluate angler use, catch rates, and harvest rates by conducting creel interviews from May through October.

METHODS

Population Monitoring

We targeted the kokanee population from June 12 to 14, 2019 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 with 16 panels that were 3-m long with two panels for each mesh size randomly positioned. The mesh sizes of the panel 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 (Appendix G). 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 to the nearest millimeter (mm), and weighed to the nearest gram (g). We calculated catch-per-unit-effort (CPUE) for each species as the number of fish per net-night and 95% CI for each estimate.

We removed sagittal otoliths from a subsample of kokanee and Yellow Perch 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. The von Bertalanffy (1938) growth model was used to fit length-at-age:

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

where l_t is length at time t, L_∞ is the asymptotic length, K is a growth coefficient, and t_0 is a time coefficient at which length would theoretically be 0. We created an age-length key from our subsample of kokanee and Yellow Perch, 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 calculated proportional size distribution (PSD), relative stock density (RSD), and relative weights to describe the size structure and condition of kokanee and Yellow Perch in Ririe Reservoir. Kokanee PSD was calculated as the number of fish greater than or equal to 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 greater than or equal to 300 mm divided by the number 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 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 greater than or equal to 100 mm multiplied by 100.

Creel Survey

We conducted an open water creel survey from May 4 to October 29, 2019. Ground-based interviews were conducted on two randomly selected weekdays and two randomly selected weekend days in every two-week period. Creel clerks interviewed anglers as they returned to one of two boat ramps (i.e., Blacktail or Juniper), which was randomly selected. Based on 2012 aerial counts, the creel day was divided into three shifts (i.e., AM = 0.17, MID = 0.36, and PM = 0.47) and each shift was assigned a probability based on the number of anglers observed during those counts.

Each creel shift was 5-h long and start times were randomly selected based on sunrise and sunset times in Idaho Falls, such that the creel day extended from one hour after sunrise until sunset. Creel clerks collected information on the number of hours anglers spent fishing, the number of anglers in the party, gear type, species caught, and the number of each species harvested or released. We estimated the total amount of effort for the season, total catch, and catch rates. Catch rates for each species were estimated as the total catch for the respective species (e.g., kokanee) divided by the total amount of effort for the season, because anglers could have been targeting multiple species or caught a species they were not targeting.

Total angling effort in angler-hours on day *d* ($\hat{E}_d$) is estimated as

$$\hat{E}_d = T_d \bar{I}_d,$$

where T_d is the total number of hours in the fishing day and $\bar{I}_d$ is the mean of the angler counts conducted on day d. Angling effort for the kth stratum is estimated as

$$\hat{E}_k = N_k \frac{\sum_{d=1}^{n_k} \hat{E}_d}{n_k},$$

where N_k is the number of days in the stratum and n_k is the number of days surveyed in the stratum. Estimates of effort among strata were summed to estimate effort ($\hat{E}_k$) over the duration of the fishing season.

Multi-day catch rate in stratum $k(\hat{R}_{2k})$ was estimated as

$$(\hat{R}_{2k}) = \frac{\sum_{i=1}^{j_k} c_i}{\sum_{i=1}^{j_k} h_i},$$

where j_k is the total number of anglers interviewed in the stratum. Estimates of effort and catch were summed among strata to estimate effort and catch over the duration of the season using the above formulas described in McCormick and Meyer (2017).

RESULTS

We sampled 509 kokanee, 2 Rainbow Trout, 7 Yellowstone Cutthroat Trout, 501 Yellow Perch, 91 Utah Sucker, and 1 Bluehead Sucker from 10 gill nets set at the thermocline. The mean (± 95% CI) CPUE of kokanee, Rainbow Trout, Yellowstone Cutthroat Trout, Yellow Perch, Utah Sucker, and Bluehead Sucker was 50.9 (± 18.3), 0.2 (± 0), 0.7 (± 0.9), 50.1 (± 26.6), 9.1 (± 8.4), and 0.1 fish per net-night (± 0), respectively (Figure 47). We caught 50.9 kokanee per net-night (± 18.3) in gill nets, which was less than the average from 2015-2018 (i.e., 71.9 ± 33.3; Figure 48). Kokanee ranged in length from 70 to 426 mm with a mean length of 185 mm (± 114; SD; Figure 49) and comprised 46% of all fish caught. Kokanee mean relative weight was 90.9 (± 15.9; SD) and exhibited an increase in condition with size (r^2 = 0.48; Figure 50). Kokanee PSD and RSD-P were 88 and 83, respectively. The highest catch rates occurred for age-0 (i.e., 33.3 fish/net-night ± 15.3) and age-2 (i.e., 15.9 fish/net-night ± 4.8) kokanee (Figure 51). Total length (mm) by respective age of kokanee is similar to other years (Figure 52). Although we only caught one fish estimated to be age-1, there was a greater abundance of age-0 and age-4 kokanee in 2019 when compared to previous studies.

Catch rates for Yellow Perch (i.e., 50.1 fish/net-night ± 26.2) were lower than the average from 2015-2018 (i.e., 76.9 fish/net-night ± 49.8). Yellow Perch ranged in length from 75 to 252 mm with an average total length of 207 mm (± 16.7; SD; Figure 53) and comprised 45% of all fish caught. Yellow Perch mean relative weight was 93.2 (± 10.4; SD) and did not exhibit an increase in condition with size (r^2 = 0.05; Figure 54). Yellow Perch PSD and RSD-P were 75 and 1, respectively.

Creel clerks interviewed 378 anglers in 256 parties with the majority of anglers fishing from boats (i.e., 70%; Table 15). Total angling effort for the season was estimated at 90,024 h. Total catch for all species combined was estimated to be 74,534 fish with 0.83 fish/h catch rate for all species combined (Figure 55). Catch rates were highest for Yellow Perch (i.e., 0.3 fish/h), followed by kokanee (i.e., 0.2 fish/h), Smallmouth Bass (i.e., 0.2 fish/h), and Rainbow Trout (i.e., 0.1 fish/h). Catch rates for Tiger Trout, Brown Trout, and Yellowstone Cutthroat Trout were less than 0.01 fish/h (Table 16). The percent of caught fish that were harvested for all species combined was 53% while 47% of all fish caught were released (Figure 56). All Yellowstone Cutthroat Trout that were caught were estimated to be harvested. Kokanee had the second highest harvest at 88%, followed by Yellow Perch (61%), Rainbow Trout (46%), and Smallmouth Bass 3%).

DISCUSSION

A diverse fishery is important when one species may not be as abundant in one year as compared to other years. A low abundance of age-1 kokanee was reflected in gill nets with only one fish estimated as age-1. In 2018, there was a statewide shortage of early-run kokanee fingerlings; therefore, Ririe was stocked with late-run kokanee fingerlings, which did not survive well in Ririe Reservoir. It may be challenging to avoid this in the future, but IDFG has begun researching other locations for kokanee egg-take to prevent an egg shortage in the future. The current catch rates of kokanee in gill nets were lower than the 5-year average, and the poor year-class may contribute to low catch rates by anglers over the next couple of years. Additionally, Yellow Perch abundance was highest at the southern end of the reservoir near the Blacktail boat ramp, which may be a gauntlet for age-0 kokanee that are stocked at that location and may contribute to higher predation when stocked compared to stocking near the Juniper boat ramp at the northern end of the reservoir. Stocking kokanee at a larger size and distributing stocking locations to both boat ramps, or stocking kokanee via boat and distributing them throughout the reservoir may help improve age-0 survival and prevent poor year-classes from occurring. However, reported catch rates of kokanee by anglers was the second highest next to 2005. The abundance of catchable kokanee was such that anglers were still able to maintain high catch rates. Catch rates may decline for anglers if we have another year of poor age-0 survival or when the current age-2 kokanee are no longer part of the fishery. Due to density dependence, low abundance of age-1 kokanee could have improved the overall condition of age-2 and older kokanee as condition improved with size (Rieman and Myers 1992). Compared to other years, there was a high abundance of kokanee greater than 300 mm, which is reflected in the high PSD and RSD-P values.

The salmonid fishery continues to be an important component in Ririe Reservoir with angler catch composition for salmonids comprising 43% of the total catch in creel surveys from 2019, which is slightly less than in 2013 at 49% (High et al. 2015). The Yellow Perch fishery is gaining popularity with 36% of total catch being represented by Yellow Perch, and anglers are also expending effort on catching Smallmouth Bass with 21% of the total catch composition represented by Smallmouth Bass. No Walleye *Sander vitreus* were reported as being caught by anglers, nor were Walleye sampled in our gill net sampling. Predation and consequent reduction

in salmonid abundances has the potential to impact angler catch rates and success if the Walleye population increases. Stocking larger catchable salmonids in reservoirs with top predators has been shown to be more effective in reducing predation (Flinders and Bonar 2008), but places additional financial and spatial demands on the limited resources available at IDFG hatcheries. Currently, Rainbow Trout are stocked as catchables and kokanee as fingerlings. Limited hatchery capacities and associated higher feed costs in producing catchable kokanee instead of fingerlings make this highly unrealistic as a stocking strategy. It is more likely that if Walleye populations expand drastically, and they continue to prey heavily on salmonids, stocking of these fish could become unfeasible. Fall Walleye Index Netting (FWIN) is currently conducted every three years and will continue at that rate as long as Walleye abundance remains at current levels or lower. An additional approach to the presence of Walleye in Ririe Reservoir is to stock YY-male Walleye to saturate the population with male fish and shift the sex ratio until the population crashes. This approach relies on the development of YY-male Walleye, which is currently in the works (Schill 2020). However, stocking more Walleye may have a negative impact on the salmonid fishery rather than a positive effect to crash the Walleye population. If Walleye abundance remains low, then it may not be a viable option to stock YY-male Walleye. We recommend modelling the impacts of increasing the abundance of Walleye (e.g., YY-male Walleye), which will inform our decision making moving forward.

We compared the size structure of Yellow Perch in Ririe Reservoir to Lake Cascade. The highest frequency of total lengths in Ririe were 200 to 229, whereas in Lake Cascade the highest length frequencies were from 320 to 369 (Janssen et al. 2018). Although not as large as Yellow Perch in Lake Cascade, the Yellow Perch fishery is an additional option for anglers in Ririe Reservoir and harvest remains high at 61%. We used gill netting to target the kokanee population in Ririe Reservoir, which is not the best method for estimating relative abundance of Yellow Perch because they are more abundant in littoral zones rather than the pelagic zone of a reservoir. For example, Janssen et al. (2014) used a combination of floating and sinking gill nets to target the Yellow Perch population in Lake Cascade. This sampling protocol follows the methods in the statewide sampling protocol for lowland lakes and reservoirs in Idaho (IDFG 2012).

Low numbers of Yellowstone Cutthroat Trout were caught by anglers, but they were harvested even though stocking has not occurred since 2012, suggesting that wild recruitment is occurring in the Willow Creek watershed and there is an adfluvial component to that Yellowstone Cutthroat Trout population. 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 abundance of Yellowstone Cutthroat Trout in Ririe Reservoir is currently unknown and very few were collected in our gill netting efforts. Ririe Reservoir is a component of the Willow Creek watershed, which contains headwaters that are critical for Yellowstone Cutthroat Trout and their abundance is high in some headwater streams. Increasing the distribution of Yellowstone Cutthroat Trout in the Willow Creek watershed is necessary to better manage the species and help restore its abundance in Ririe Reservoir and Willow Creek. Current fishing regulations state that there is no harvest of Yellowstone Cutthroat Trout in Willow Creek and its tributaries, but allows harvest of up to 6 trout combined species in Ririe Reservoir including Yellowstone Cutthroat Trout. To aid in Yellowstone Cutthroat Trout restoration efforts, more restrictive regulations for harvest of Yellowstone Cutthroat Trout in Ririe Reservoir may be warranted if Yellowstone Cutthroat Trout are utilizing an adfluvial life history strategy in Ririe Reservoir.

The Ririe Reservoir fishery is an important fishery for residents that live in the surrounding Idaho Falls area, which was evident in the estimated angler effort in the 2019 creel survey. Angler

effort was estimated to be the highest of any creel survey conducted at Ririe since 1993. In addition, angler effort in 2019 on Ririe Reservoir (i.e., 90,024 h) exceeded estimates on Henrys Lake in 2016 (i.e., 75,432 h; unpublished data), Palisades Reservoir in 2015 (i.e., 44,623 h; Flinders et al. 2016), and on Island Park Reservoir in 2013 (i.e., 59,636 h; High et al. 2015). The close proximity of Ririe Reservoir to Idaho Falls lends to easy access for anglers, and the diversity of species offers many opportunities for anglers. Consequently, we concluded that anglers use the fishery predominately as a harvest fishery where we estimated overall harvest at 53%, and our management strategy reflects how anglers use the fishery. Harvest rates were highest for salmonids and Yellow Perch, whereas Smallmouth Bass were only harvested at 3%. As such, continued stocking of kokanee and Rainbow Trout will be important for anglers to utilize this fishery as represented in the creel survey. Compared to other lentic waterbodies in the Upper Snake Region, catch rate at Ririe Reservoir (i.e., 0.4 fish/h) was comparable to Island Park Reservoir (i.e., 0.4 fish/h; High et al. 2015); however, harvest in Ririe was much greater than Palisades Reservoir (i.e., 0.08 fish/h; Flinders et al. 2016) and Henrys Lake (i.e., 0.07 fish/h; unpublished data). Ririe Reservoir is becoming an increasingly important reservoir fishery for anglers in the region, especially for anglers who fish to harvest their catch. Considering its importance to anglers, Ririe Reservoir needs to be a management priority for the Upper Snake Region.

We introduced Tiger Trout in 2019, but anglers reported in our creel survey catching about 2% of the 1,700 that we stocked. Therefore, if we continue stocking Tiger Trout we need to conduct a “Tag You’re It” study to evaluate angler use and harvest of these fish.

RECOMMENDATIONS

1. Continue annual early summer gill net monitoring to evaluate kokanee abundance and growth.

2. Increase the number of kokanee stocked if catch rates in gill nets remain below the 5-year average

3. Evaluate the effect of stocking kokanee fingerlings at Juniper instead of Blacktail, which is where the highest abundance of Yellow Perch are located. Furthermore, the highest catch rates of kokanee occur near the Juniper boat ramp, suggesting that environmental conditions are more suitable for kokanee at that end of the reservoir.

4. Evaluate abundance of Yellowstone Cutthroat Trout in Willow Creek and whether there is an adfluvial component to the population.

5. Evaluate appropriate methods to describe population growth rates, age structure, and exploitation of Yellow Perch to properly manage this population.

6. Monitor Smallmouth Bass abundances on a five-year cycle.

7. Model the effects of increasing Walleye (i.e., YY-male) abundance on the salmonid population in Ririe Reservoir.

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Table 16. Creel summary statistics from 1993-2019.

	1993	2003	2005	2010	2013	2015	2019
Season effort (hours)	56,612	25,981	43,825	68,364	43,643	65,607	90,024
Residents (%)	98	96	96	97	96	95	94
Nonresidents (%)	2	4	4	3	4	5	6
Number of interviews	747	271	546	384	731	167	256
Anglers per interview	2.42	2.34	2.14	2.30	2.40	2.30	1.48
Bait (%)	100	45	60	61.6	-	-	-
Lure (%)	0	55	40	37.5	-	-	-
Fly (%)	0	0.5	0.1	0.9	-	-	-
Boat (%)	-	84	-	-	-	80	70
Shore (%)	-	16	-	-	-	20	30
Number of completed trips	337	43	216	334	304	167	235
Average trip length (hours)	3.34	2.69	3.00	4.00	-	3.50	5.34

Table 17. Catch rates, percent harvest, percent caught and released by species in Ririe Reservoir in 2019 for kokanee (KOK), Rainbow Trout (RBT), Yellow Perch (YLP), Smallmouth Bass (SMB), Tiger Trout (TGT), Yellowstone Cutthroat Trout (YCT), and Brown Trout (BNT).

	KOK	RBT	YLP	SMB	TGT	YCT	BNT	Total
Total catch	19,848	12,144	26,566	15,837	26	75	40	74,536
Catch rate (number/h)	0.2205	0.1349	0.2951	0.1759	0.0003	0.0008	0.0004	0.8280
Harvested (%)	88	46	61	3	0	100	100	53
Released (%)	12	54	39	97	100	0	0	47

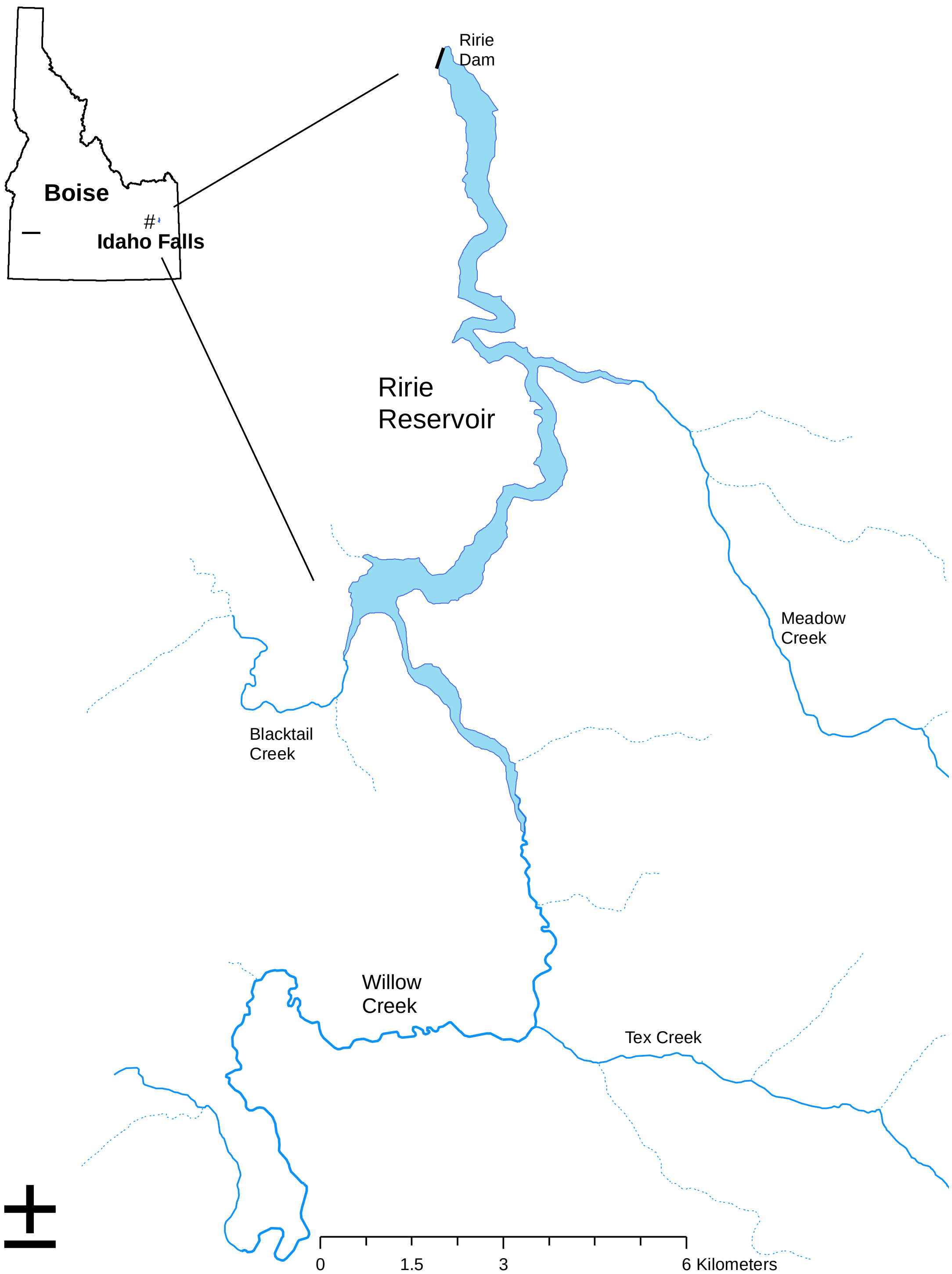

Figure 46. Location of Ririe Reservoir and major tributaries.

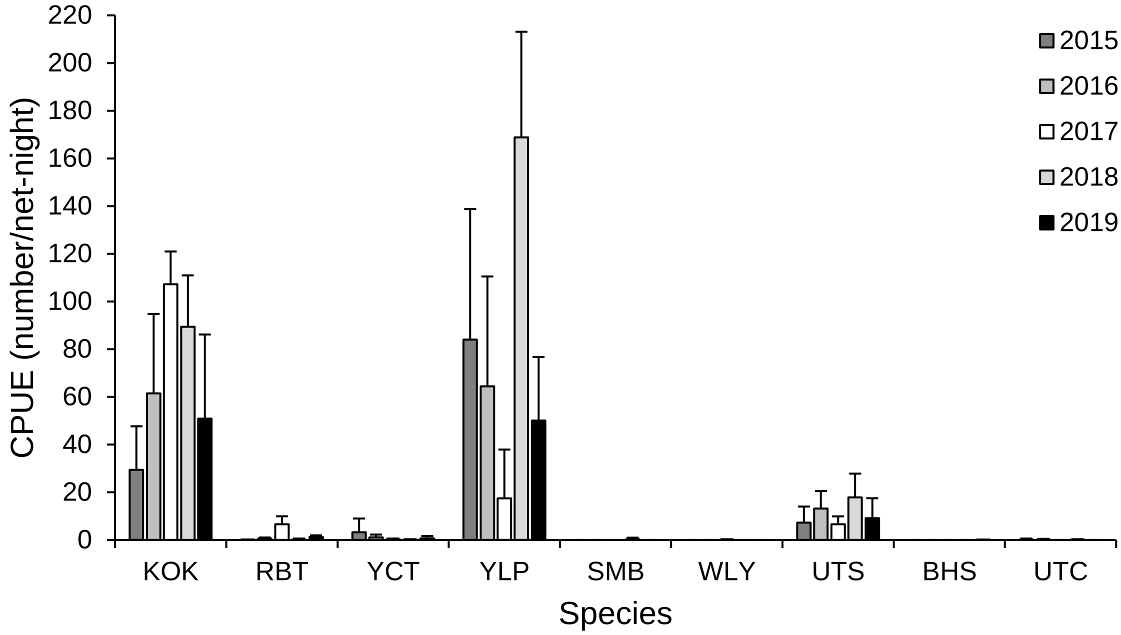

Figure 47. The number of fish per net-night ([CPUE] = catch-per-unit-effort) from 10 gill nets for kokanee (KOK), Rainbow Trout (RBT), Yellowstone Cutthroat Trout (YCT), Yellow Perch (YLP), Smallmouth Bass (SMB), Walleye (WLY), Utah Sucker (UTS), Bluehead Sucker (BHS), and Utah Chub (UTC) in Ririe Reservoir during 2015–2019. Error bars represent 95% confidence intervals.

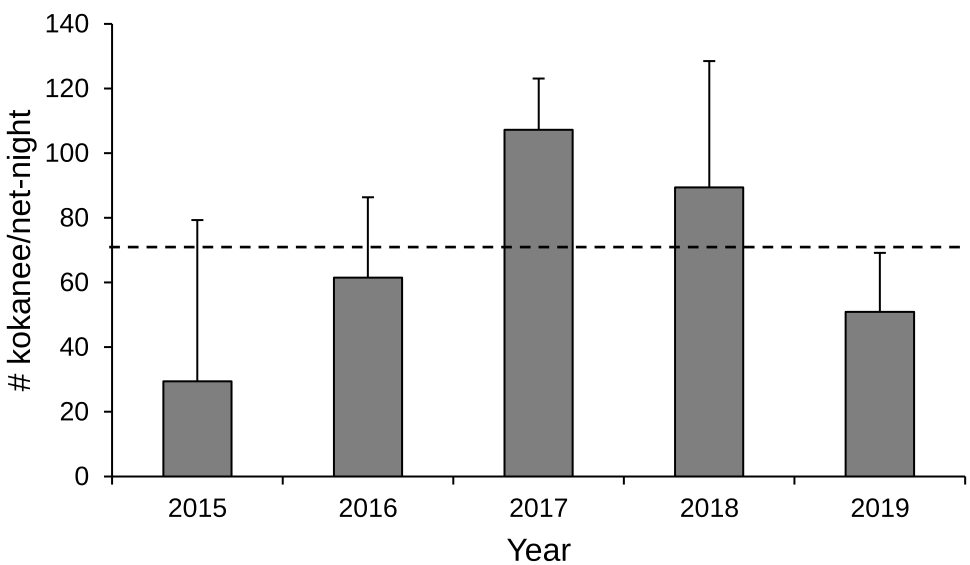

Figure 48. The mean number of kokanee caught per net-night from 2015 to 2019. Error bars represent 95% confidence intervals. The dashed line represents the mean catch per net-night from 2015 to 2018 (i.e. 71.8 kokanee/net-night).

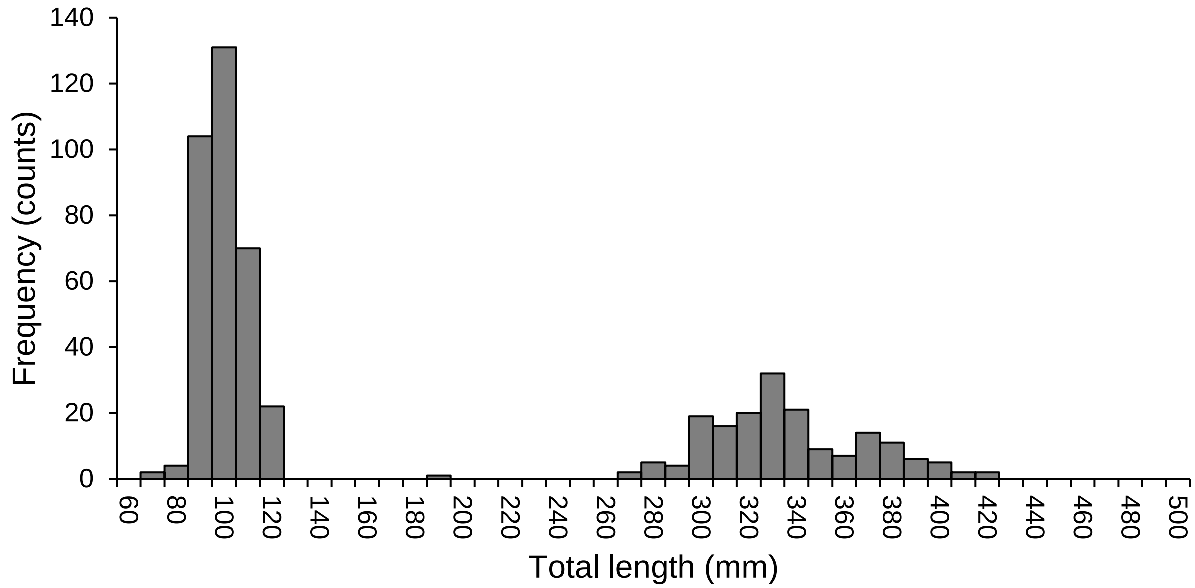

Figure 49. The length-frequency distribution of kokanee caught in gill nets in Ririe Reservoir in 2019.

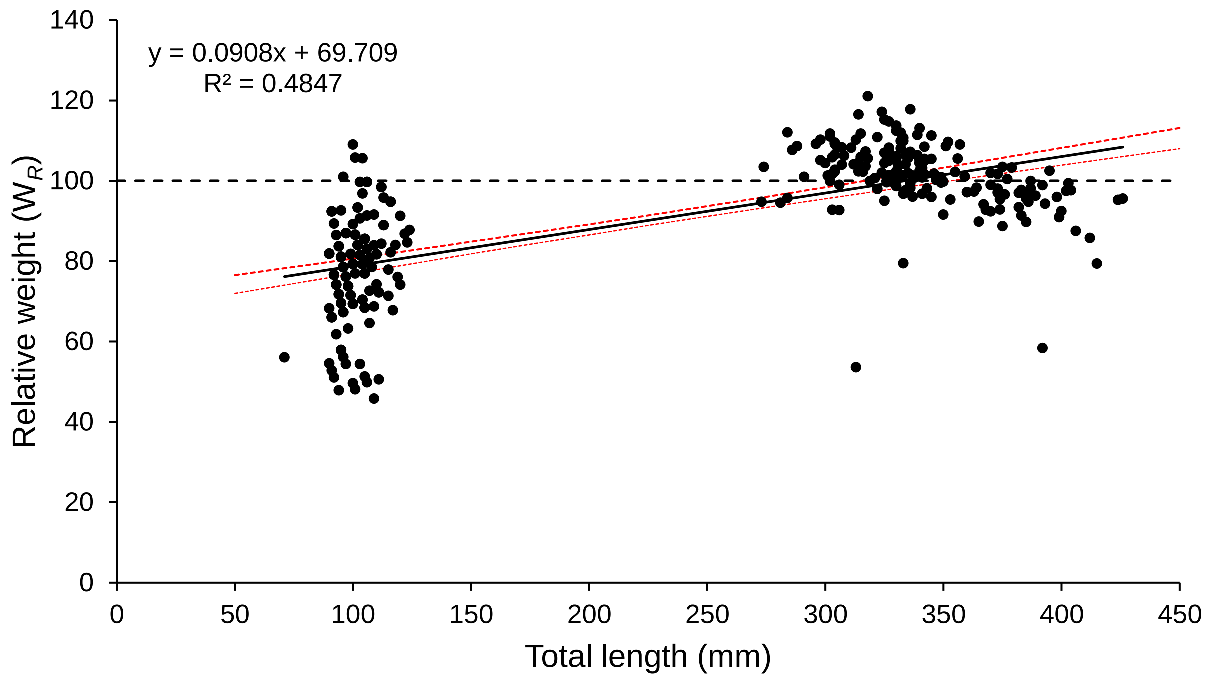

Figure 50. The relative weight (W_r) of kokanee across total length (mm) in Ririe Reservoir in 2019. The linear regression curve is represented by the solid black line and 95% confidence intervals are represented by the dotted line.

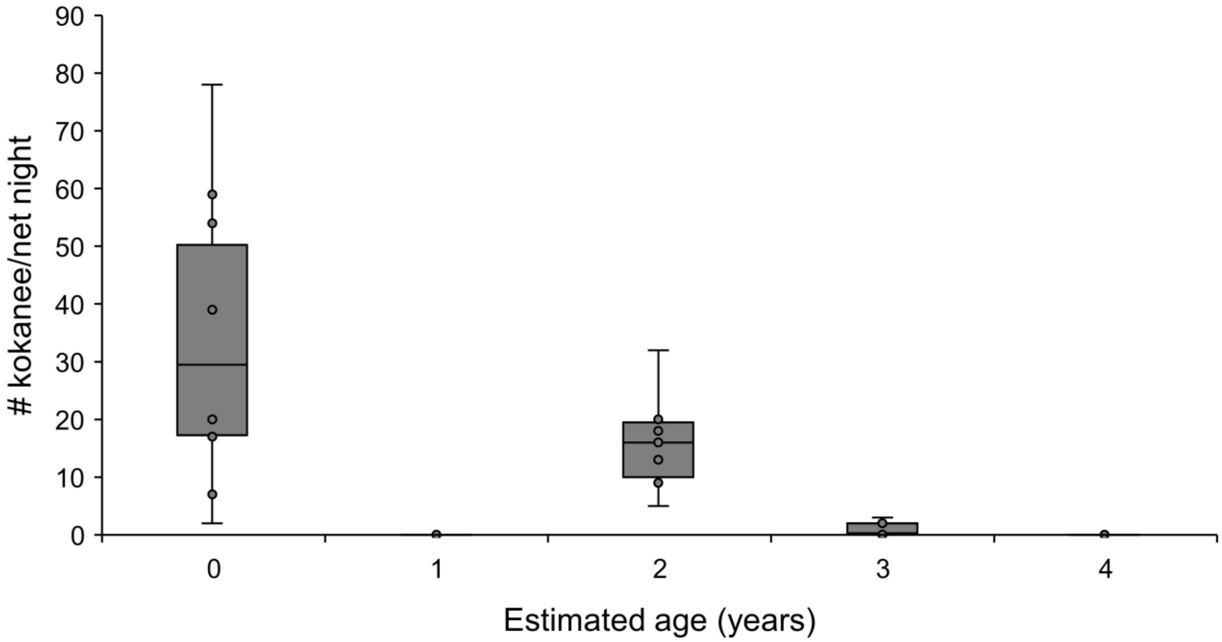

Figure 51. The catch rate (number/net-night) of kokanee per age (years) caught in gill nets in Ririe Reservoir in 2019. The mean is represented by the solid line in each box.

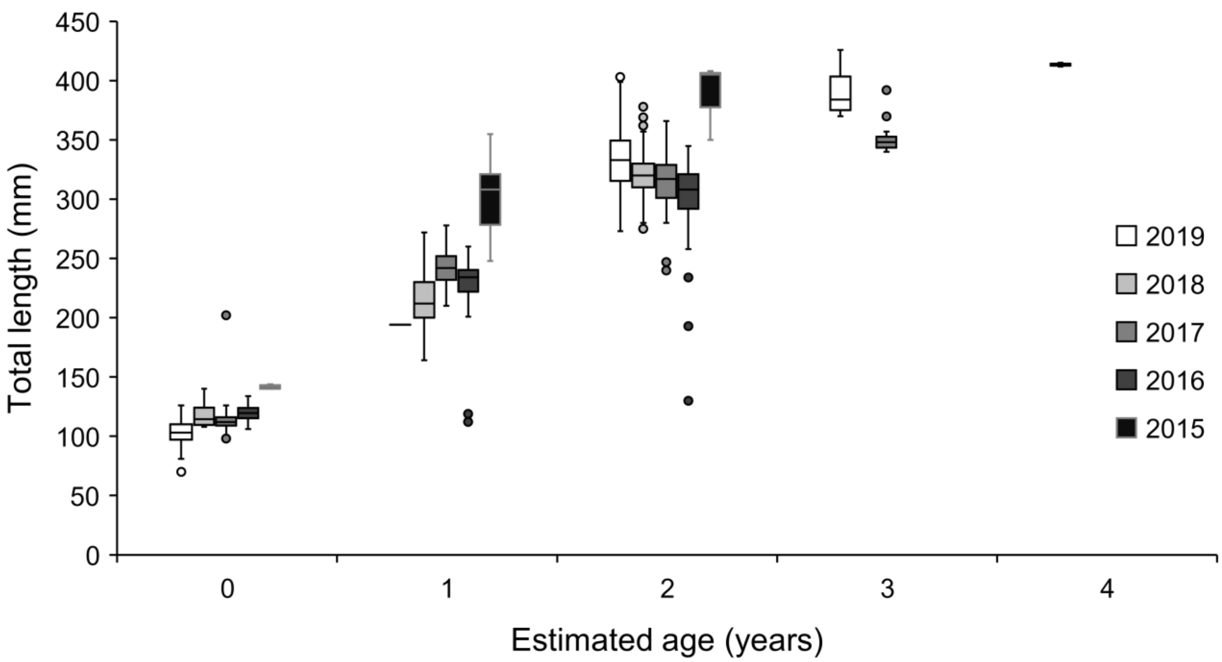

Figure 52. Total length (mm) by estimated age (years) of kokanee caught in gill nets in Ririe Reservoir from 2015 to 2019.

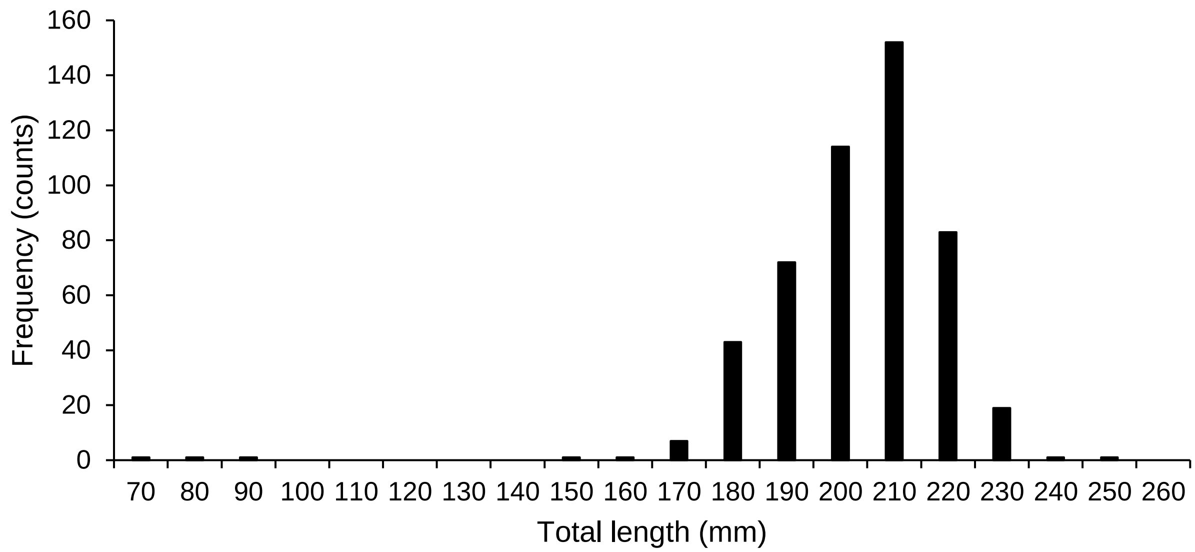

Figure 53. The length-frequency distribution of Yellow Perch caught in gill nets in Ririe Reservoir in 2019.

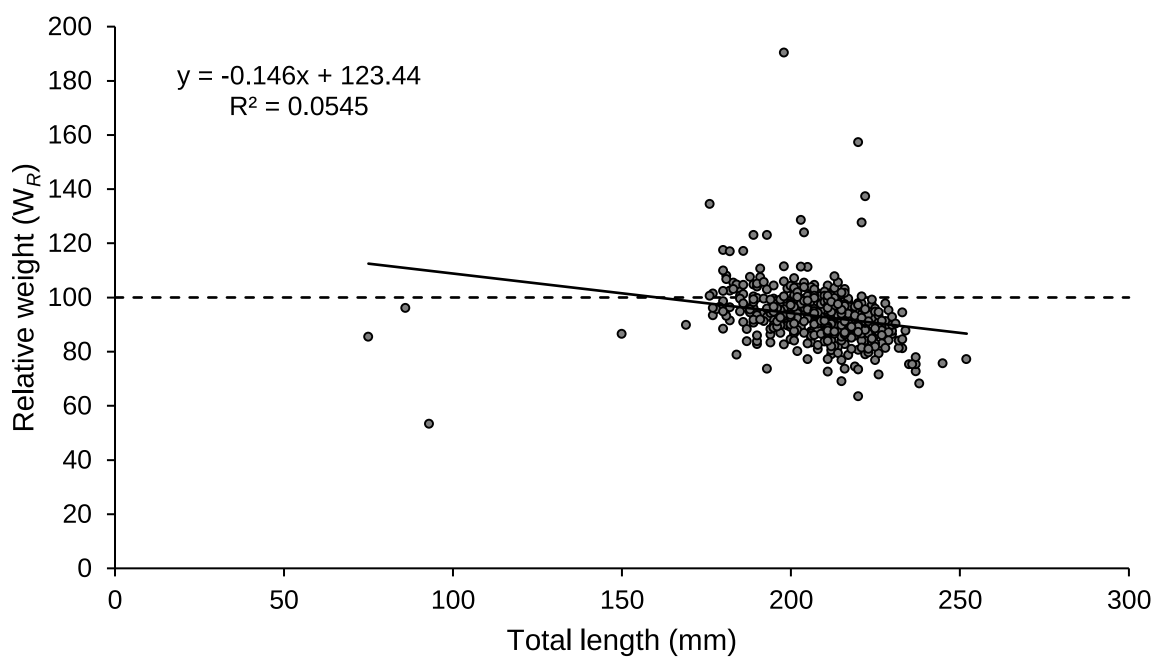

Figure 54. Relative weight (W_r) of Yellow Perch across total length (mm) caught in gill nets in Ririe Reservoir in 2019.

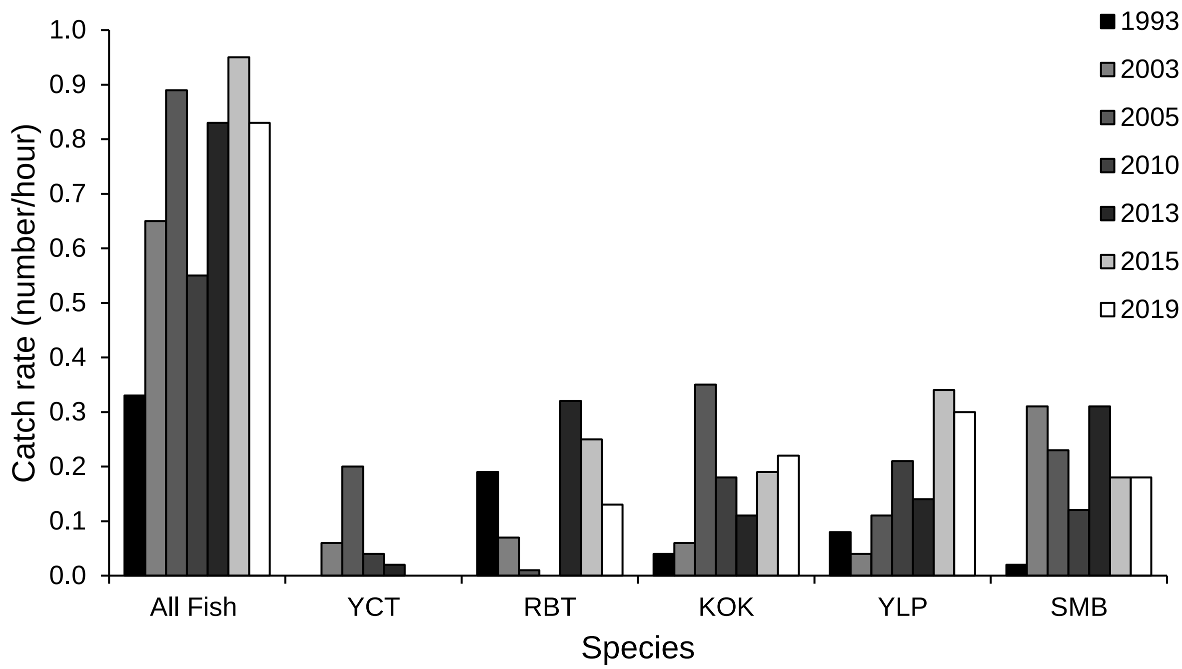

Figure 55. Angler catch rates on Ririe Reservoir by species from 1993-2019. Species codes are Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), kokanee (KOK), Yellow Perch (YLP), and Smallmouth Bass (SMB).

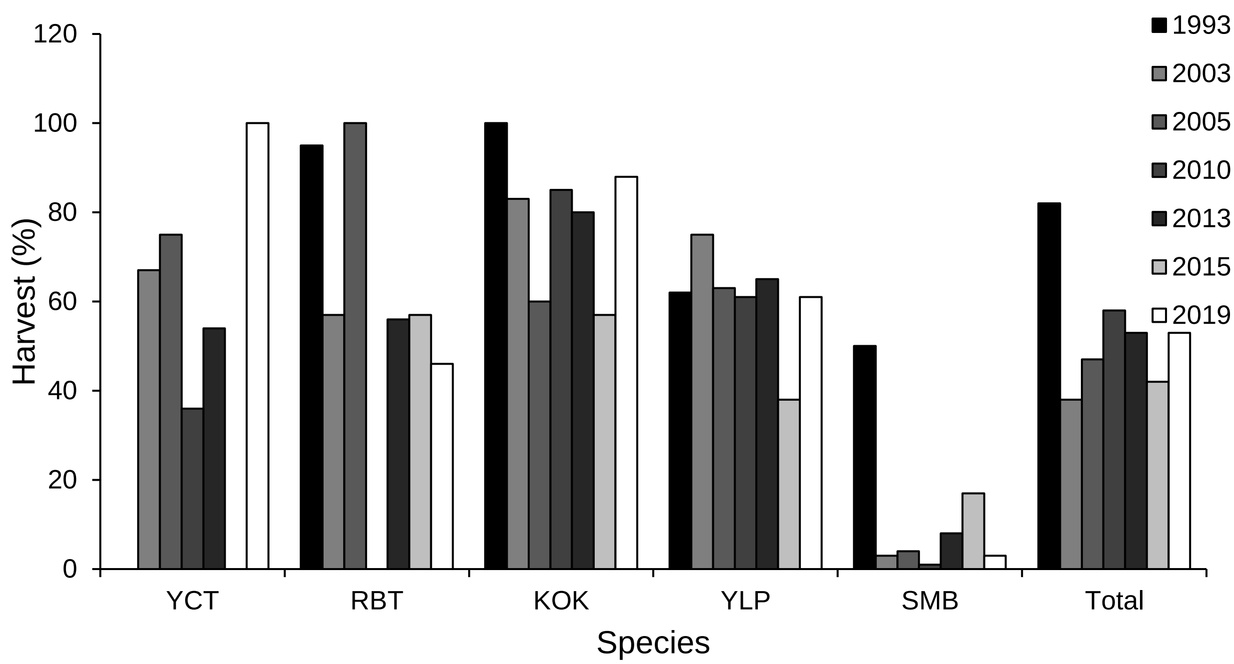

Figure 56. The proportion of caught fish that were harvested in Ririe Reservoir from 1993-2019 as estimated from creel surveys. Species codes are Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), kokanee (KOK), Yellow Perch (YLP), and Smallmouth Bass (SMB).

ISLAND PARK RESERVOIR

ABSTRACT

We used suspended gill nets to assess the kokanee population in Island Park Reservoir (IPR) during June 2019. We collected 989 fish in nine net nights of effort. Overall, relative abundance was comprised of Utah Chub *Gila atraria* (37%), Utah Sucker *Catostomus ardens* (28%), Kokanee *Oncorhynchus nerka* (20%), Redside Shiner *Richardsonius balteatus* (11%), Rainbow Trout *Oncorhynchus mykiss* (3%), Mountain Whitefish *Prosopium williamsoni* (<1%), and Brook Trout *Salvelinus fontinalis* (<1%). Average catch rates of kokanee were 21.9 per net night (± 7.4) and Rainbow Trout catch rate was 3.33 per net night (± 1.4). The catch rate for kokanee was greater (21.9) than the 2018 catch rate (20.1) and the 5 year average (15.8). While Rainbow Trout catch rates were lower than estimated for 2018 (8.6) and the 5-year average (6.2). Relative weights for both kokanee and Rainbow Trout were below 100 indicating prey resources may be limited in the reservoir promoting a need for a current evaluation on the zooplankton densities of IPR. Kokanee abundances remain relatively low in Island Park compared to other regional waters, such as Ririe and Mackay Reservoir where catch rates for surveys conducted in 2019 were 51 (± 58) and 60.1 (22.3), respectively. Additionally, to fully understand the effect that angling effort and harvest have on the IPR fishery, biologist should obtain current creel information to further evaluate the current state of the fishery.

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INTRODUCTION

Island Park Reservoir has been recognized as a quality recreational fishery since the early 1950s, supporting as much as 176,000 hours of angling effort annually, with catch rates averaging 0.68 fish per hour. Rainbow Trout *Oncorhynchus mykiss* have provided the bulk 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 stockings have played a large role in the management of the reservoir fishery, which is primarily supported by hatchery releases of Rainbow Trout and kokanee, although some spawning by both species occur in the Henrys Fork Snake River upstream of the reservoir. Annual Rainbow Trout fingerling stockings have averaged 458,000 over the past 82 years and have been as high as 2.5 million fish in 1959. Beginning in 2010, IDFG increased the size of fingerling Rainbow Trout (> 6 inches) stocked in Island Park to reduce the potential for entrainment through the dam. Fingerling numbers were reduced to approximately 150,000 fish that accounted for the same biomass as the nearly 500,000 smaller fingerlings stocked in earlier years. Nearly 120,000 kokanee were stocked into Island Park Reservoir in 1944-1945, 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. IDFG established a kokanee trapping facility on Moose Creek to collect eggs for 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, Island Park Reservoir was drawn down to near record levels on two occasions, and treated with rotenone during the 1979 draw down. The purpose of these rotenone treatments were to remove nongame fish species. Annual kokanee fry stocking of nearly 500,000 fish in 1981, 1982, and 1984 re-established the 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 1992-1994. In 1995, over 200,000 eggs were again collected at the Moose Creek trap, but future trap operations were ceased due to low returns combined with the identification of other, more easily obtained egg sources (Deadwood Reservoir). 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.

Historically, the proliferation of nongame fish, primarily Utah Chub *Gila atraria* and Utah Sucker *Catostomus ardens*, had been blamed for declines in the sport fishery in Island Park Reservoir. Several rotenone projects had been undertaken to reduce overall nongame fish abundance and improve angler catch rates. The efficacy of these treatments were questioned as early as 1982, when Ball et al. (1982) observed that the three chemical rehabilitations of Island Park Reservoir 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 levels, especially noticeable with the large introductions of catchable Rainbow Trout. Ball et al. (1982) further noted that the observed declines in the Rainbow Trout fishery two to four years after treatment were the result of decreased levels of hatchery inputs and were not due to increased chub and sucker densities. The most recent chemical treatment of the reservoir, conducted in 1992, yielded similar results, with catch rates not improving upon levels prior to the treatment (Gamblin et al. 2002). More recently, Garren et al. (2008) found that nongame fish exceed prerotenone treatment levels within five years following treatments and that angler catch rates within five years following rotenone treatments were not significantly different than catch rates prior to treatments, suggesting that rotenone treatments had no effect on improving angler catch rate in Island Park Reservoir.

Island Park Reservoir is operated as an irrigation storage reservoir for agricultural users downstream, and is therefore subject to fluctuations in annual water levels. Increases in reservoir storage normally begins at the close of irrigation season in October, and lasts until demand for water increases, typically in late May or early June. Fall reservoir storage levels can fluctuate from the lowest storage level recorded of 270 acre-feet in 1992, to nearly 90% full (121,561 acre-feet), as seen in 1997. Recent analysis of reservoir storage indicates that reservoir carryover is positively related to gill net catch rates for salmonids. Garren et al. (2008) found a significant relationship between reservoir carryover and salmonid gill net catch rate the following year by examining spring gill net catch and the previous year's reservoir level. Years following low reservoir storage typically show a reduction in sport fish densities in gill nets the following year. 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 lost habitat associated with drawdowns, entrainment through the dam due to increased outflow, 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%. The losses were due to entrainment as kokanee distributed throughout the reservoir (Maiolie and Elam 1998). Congregations of all age-classes of kokanee were found near the 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. Although both intake structures are located at the reservoir bottom, the hydroelectric intake is 206 m east of the pre1994 intake structure, and closer to the river channel. The hydroelectric facility is capable of handling up to 960 cfs. Therefore, 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 criterion is designed for anadromous fishes, it is the only reviewed criteria for juvenile salmonids, and has been implemented in nonanadromous waters for screening juvenile salmonids. Additionally, the approach velocities near the hydroelectric intake are unknown, and blockage to any area of the screen could result in areas of increased velocity that could increase the likelihood of entrainment or impingement. Based on the current screen design, entrainment or impingement of juvenile kokanee is a possible source of mortality. Surveys of the Henrys Fork Snake River immediately below Island Park Dam have documented kokanee, indicating that some size classes are able to pass though the screened intake. Additionally, recent gillnetting in Island Park Reservoir (Schoby et al. 2010) found high net catch rates of kokanee in the deep water in front of Island Park dam, in the proximity of the existing water intake structures.

Although drought, reservoir operation, and other environmental conditions may have impacted kokanee since the early 1990s, the alteration of intake facilities may be substantially inhibiting the re-establishment of the Island Park Reservoir kokanee fishery. In response to low kokanee 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 Island Park Reservoir between May and June, when inflow and outflow from the reservoir is 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 Island Park Reservoir, with the remaining releases split between Big Springs Creek and Moose Creek (Figure 57). In-reservoir stockings occur throughout the reservoir, although the west end is the preferred location when it is accessible in the spring when stocking

occurs. Tributary releases are intended to reduce downstream migration through the reservoir, to allow fingerlings a chance to grow larger before encountering the intake structures, and to allow kokanee to imprint on tributaries to establish spawning runs in these locations.

STUDY AREA

Island Park Reservoir (IPR) is located on the Henrys Fork of the Snake River 40-km north of Ashton, Idaho and 150 km upstream from the confluence with the South Fork of the Snake River (Figure 57). 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 square km, varying in elevation from 1,920 to 3,017 meters. At gross pool capacity (143,430 acre feet), the reservoir covers 3,388 hectares and has a shoreline of about 97 km. Since first filling in 1939, the minimum storage was 270 acre-feet, occurring in 1992. Runoff and numerous springs supply water to streams entering the reservoir. Maximum reservoir level or storage capacity generally occurs in May and June. Thereafter, gradual drawdown through the summer and fall lowers the reservoir to varying degrees, depending upon irrigation needs. Ice generally covers the reservoir from December to May. Approximately 25-km upstream of Island Park Dam, Moose Creek, a historically an important spawning tributary for kokanee, joins the Henrys Fork Snake River, just downstream of the confluence of the Henrys Lake outlet and Big Springs Creek. Moose Creek is approximately 13-km long, and flows from numerous spring sources, including Lucky Dog Creek.

OBJECTIVES

To obtain current information on fish populations for fishery management decisions on Island Park Reservoir and its tributaries, and to develop appropriate management recommendations.

METHODS

We targeted kokanee using experimental gill nets (Appendix H). Gill nets were set from June 18 to 20th, 2019. 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 was 13, 19.25. 38, 52, 64, 76, and 102 mm with each mesh representing two panels. We set nets at dusk and retrieved them the following morning. Gill nets were deployed in the reservoir in areas with a maximum depth of 20 m and were set at the thermocline. Sites were randomly selected by overlaying a grid system (100 X 100 m) using mapping software (IDFG staff 2012). All fish captured were identified, measured for total length to the nearest millimeter and weighed to the nearest gram. We calculated relative abundance as well as catch per unit effort (CPUE: fish per net night). Relative weights (W_r) were calculated by dividing the actual weight of each fish (in grams) by a standard weight (W_s) for the same length for that species multiplied by 100 (Anderson and Neumann 1996). We used the formula:

$$\log W_s = -4.898 + 2.990 \log \mathrm{TL}$$

for Rainbow Trout (Simpkins and Hubert 1996) and,

$$\log W_s = -5.062 + 3.033 \log \mathrm{TL}$$

for kokanee (Milewski and Brown 1994). We compared relative weights among size groups using 95% confidence intervals.

RESULTS

We sampled 989 fish in nine net nights of effort (110 fish per net night) using gill nets. Overall, relative abundance of the gill net catch was dominated by Utah Chub (37%), Utah Sucker (28%), and kokanee (20%), with Redside Shiner (11%), Rainbow Trout (3%), Mountain Whitefish (<1%), and Brook Trout (<1%) comprising a less abundant portion of the catch. Catch rate (fish per net night; CPUE) was highest for Utah Chub (40.8), followed by Utah Sucker (31.2), Redside Shiner (28), kokanee (21.9), and Rainbow Trout (3.3; Figure 58). Kokanee CPUE for 2019 was the highest observed over the last five years and exceeded the 4 year average CPUE of 15.8 (Figure 59). Kokanee ranged from 59 to 424 mm TL, with a mean length of 180 mm (± 16.2; Figure 60). Proportional stock density (PSD) was 73 and RSD-400 was 1.28 for kokanee (Table 17). Kokanee relative weights were highest at 96 for the 300 - 399 mm size class, followed by the 200 - 299 mm size class (78), and <200 mm size class (83). All size classes were below the standard 100 (Table 17). Rainbow Trout ranged from 113 to 502 mm TL, with a mean length of 341 mm (±33.8; Figure 61). Proportional stock density (PSD) was 61 and RSD-400 was 29 for Rainbow Trout. Mean relative weight of Rainbow Trout was 83 (±3.4), which was below the standard 100 (Table 17).

DISCUSSION

Kokanee catch rates have increased over the last 5 years in IPR. We collected kokanee over a wide size range (59 - 424 mm TL), with multiple age-classes present. Rainbow Trout catch rates have decreased compared to 2018 and the 5 year average. Relative weights for both kokanee and Rainbow Trout were less than 100, suggesting that food resources may be limited. In 2014, a survey of zooplankton abundance and was conducted on waters throughout the region. High zooplankton densities were found in IPR suggesting the reservoir could support high densities of stocked hatchery fish (Flinders et al. 2016). One objective in the Fisheries Management Plan (IDFG 2019) is to identify limiting factors on kokanee in IPR. By repeating a zooplankton abundance survey managers will be able to quantify the current zooplankton densities in IPR and evaluate prey abundances for the IPR kokanee fishery.

There was strong recruitment of juvenile kokanee to our gill nets with the large percentage of our catch representing the juvenile size class (<100 mm; Figure 60). The neutral buoyancy of the gill nets allow for the net to be placed directly at the desired depth, which is near the thermocline for kokanee. Kokanee are an obligate planktivore and 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. Kokanee fingerlings were stocked in Island Park Reservoir (IPR) in early June 2019 (Appendix I). This may be playing a part in our large catch < 100 mm in length fish. Due to the shortage of kokanee eggs statewide in 2018, kokanee were not stocked into IPR. The absence of stocking in 2018 is apparent in the reduced number of kokanee captured in the age-2 size class. Although we did capture some kokanee indicating that there is some natural spawning of kokanee in the tributaries of Island Park Reservoir. To fully evaluate the hatchery and wild ratios of individual age-classes in the future, biologists can use thermal marks on otoliths. All kokanee stocked in IPR were reared at Cabinet

Gorge Fish Hatchery and are marked with thermal mass marking at the eyed egg stage since 1997. All kokanee from the facility contain distinct thermal marks by age-class.

The current kokanee catch rate of Island Park Reservoir are lower than those observed in other kokanee fisheries in the region of Mackay Reservoir and Ririe Reservoir in 2019. One factor may be fluctuations in reservoir water levels from year to year during critical kokanee life stages. Fall drawdowns of reservoirs have been found to result in the loss of kokanee spawning habitat and lead to reduced egg-to-fry survival (Maiolie et al. 2006). This is apparent in our catch rates for the last five years as seen in Figure 59. Kokanee catch rates were significantly lower in 2017 following a large fall reservoir drawdown in 2016. Catch rates then rebounded in 2018 after the 2017 fall reservoir retained over 82,000 acre-feet more water than 2016. Large fluctuations in reservoir levels during the fall spawn and early winter may be playing a large factor influencing kokanee populations in IPR. The last two years have been high water years for the Upper Snake region. This has helped to keep reservoirs in the region full, but we should keep in mind the importance of keeping water in reservoirs for fish health and survival in future years.

Evaluations of kokanee spawning in the tributaries of Island Park Reservoir which have not been conducted since 2016 would also allow managers to quantify the wild contribution of kokanee in IPR. In 2016, a carcass survey was conducted on the Henrys Lake Outlet, with 95% of the carcasses determined to be wild fish due to the lack of thermal marking on otoliths (IDFG in progress). In addition, eyed egg were planted in artificial redds from 2013 to 2015 in Moose and Luck Dog creeks in an aim to re-establish a wild kokanee spawn in IPR. An updated spawn survey should be conducted to evaluate the hatchery vs. wild component of the IPR kokanee fishery.

Island Park currently supports the second largest American white pelican *Pelecanus erythrorhynchos* breeding colony of Idaho. This breeding colony was established in 2012 with the population increasing annually since the first viable fledglings were produced in 2014 (IDFG 2016). High avian predation rates on both hatchery and wild stocks of fish throughout the state have been documented (IDFG 2016. Meyer et al. 2016). In response, a pelican hazing program began in 2018 (IDFG 2019). The goal of this project is to limit the number of nesting pelicans on Island Park Reservoir to 150 nests and has been successful in both 2018 and 2019. The success of this dissuasion study may be a factor reducing avian predation rates on kokanee attempting to migrate into spawning tributaries leading to an increase in the population.

The population of kokanee in Island Park should be able to support a fishery although, we do not have current information on the harvest and catch rates of this fishery. An evaluation on the fishery has not been conducted since 2013. Our 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 roving creel survey should be conducted and will help to dictate future management actions.

MANAGEMENT RECOMMENDATIONS

1. Continue using gill nets to monitor the entire Island Park Reservoir fishery

2. Conduct a roving creel survey to obtain up to date information on angler use, catch, and harvest on the Island Park Reservoir fishery

3. Conduct kokanee visual spawner surveys in Moose Creek and Big Springs Creek to monitor trends in adult abundance and determine if past IDFG juvenile/eyed egg releases in these locations have established spawning runs

4. Monitor trends in kokanee and Rainbow Trout prey resources through zooplankton and macroinvertebrate surveys to assess current Rainbow Trout stocking practices.

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Table 18. Stock density indices (PSD: proportional stock density and RSD: relative stock density) and relative weights (W_r) for Rainbow Trout and kokanee collected using gill nets in Island Park Reservoir, Idaho 2019. Sample size (n) for relative weight values is noted in parentheses.

	Rainbow Trout (*n*)	**kokanee (*n*)**
PSD	60.7	73.1
RSD-400	28.6	1.3
RSD-500	3.6	--
W_r		
<200 mm	81 *(2)*	83 *(119)*
200 – 299 mm	86 *(11)*	87 *(21)*
300 – 399 mm	84 *(9)*	96 *(56)*
>399 mm	78 *(8)*	--
Mean	83	87

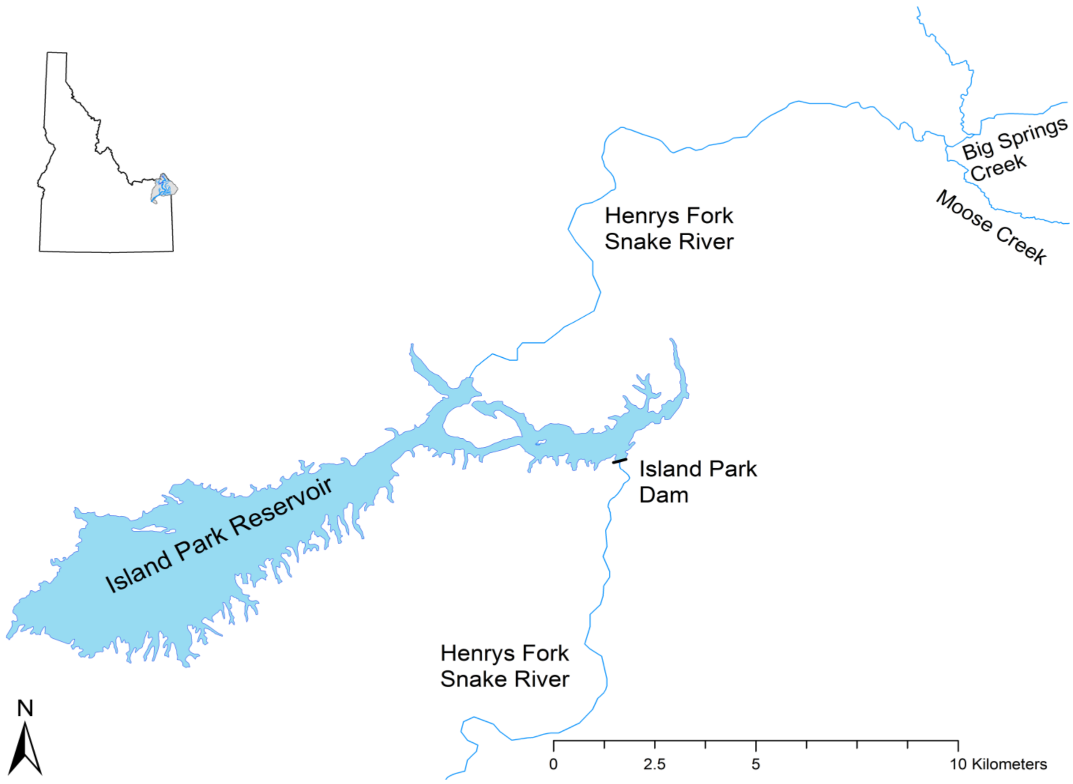

Figure 57. Map of Island Park Reservoir and the major tributaries in southeastern Idaho.

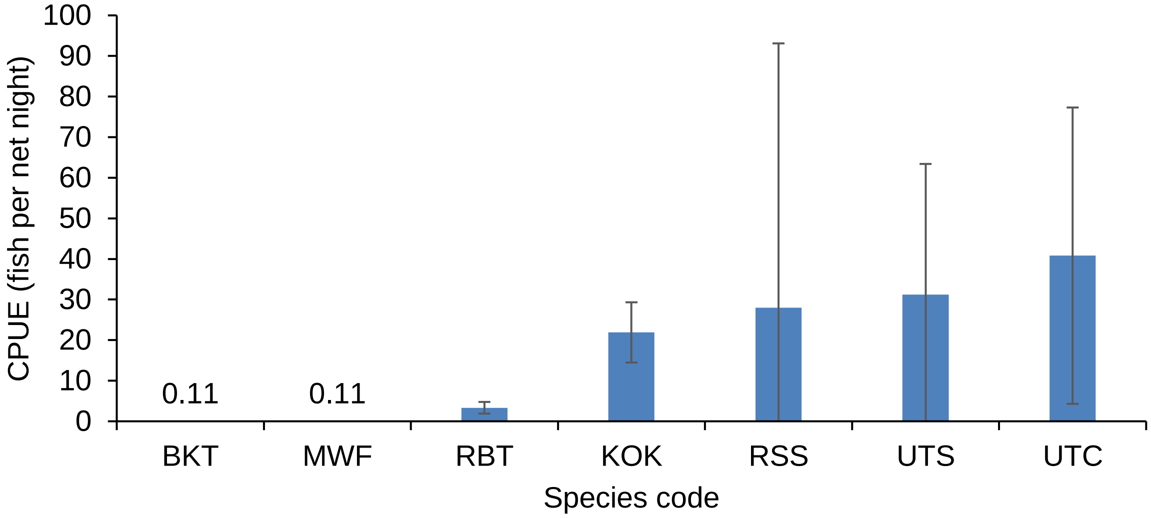

Figure 58. Catch per unit effort (CPUE, number of fish per net night) and 95% confidence intervals for Brook Trout (BKT), Mountain Whitefish (MWF), Rainbow Trout (RBT), kokanee (KOK), Redside Shiner (RSS), Utah Sucker (UTS), and Utah Chub (UTC), collected using gill nets in Island Park Reservoir, 2019.

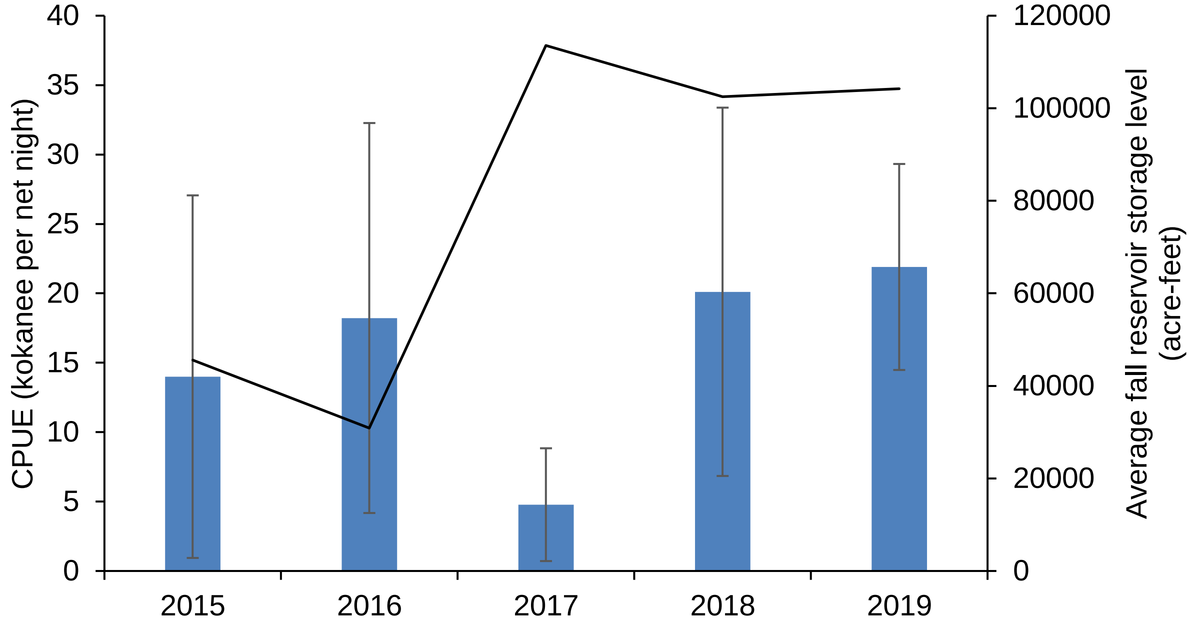

Figure 59. Kokanee catch per unit effort (CPUE) with 95% confidence intervals sampled using gillnets and the average reservoir storage level in acre-feet of Island Park Reservoir during the fall (August through October) from 2015—2019. Reservoir level data was provided by the Bureau of Reclamation Island Park Dam site.

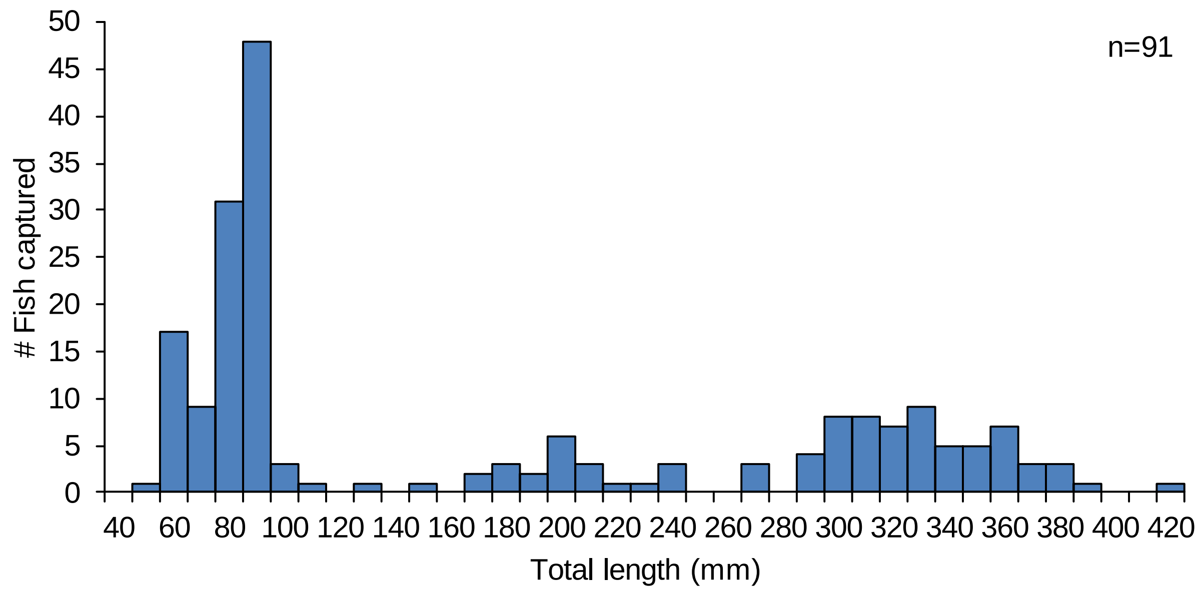

Figure 60. Length-frequency distribution of kokanee captured using suspended gill nets in Island Park Reservoir in 2019.

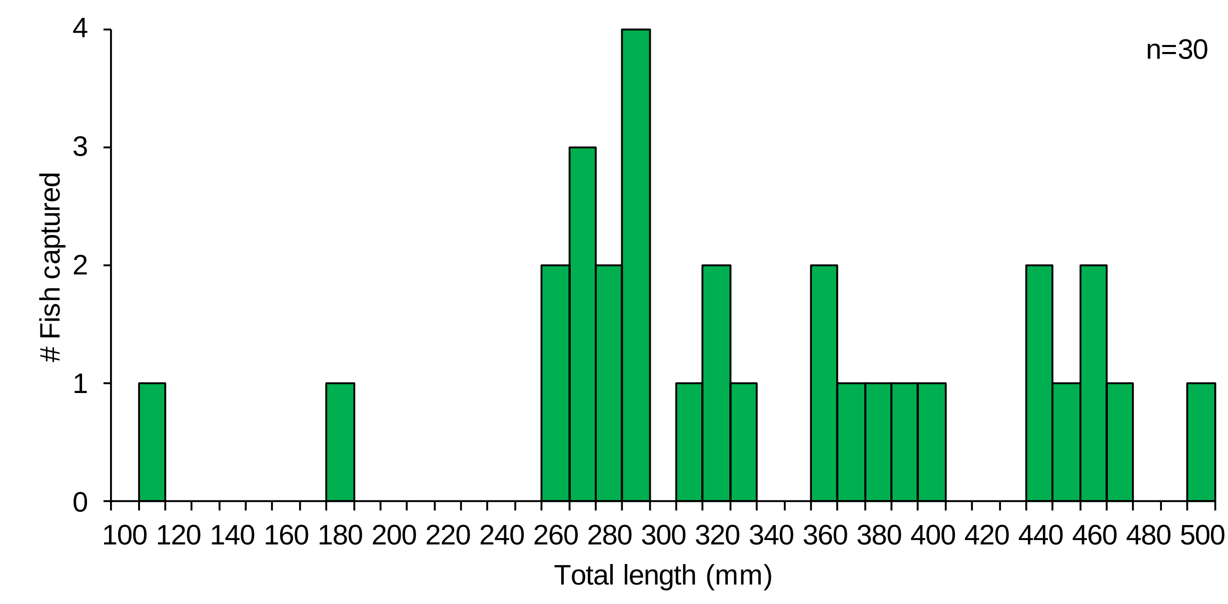

Figure 61. Length-frequency distribution of Rainbow Trout captured using gill nets in Island Park Reservoir in 2019.

MACKAY RESERVOIR

ABSTRACT

Mackay reservoir is sampled every three years to assess the kokanee and Rainbow Trout populations in the reservoir. We used eight experimental gill nets suspended in the thermocline to assess the kokanee and Rainbow Trout populations in Mackay Reservoir during July 2019. Mean catch (number of fish per net-night) was 5.7 (± 2.5; 95% CI) for Rainbow Trout and 60.1 (± 22.3; 95% CI) kokanee. The average total length (mm) for Rainbow Trout was 367.6 mm (± 16.3; 95% CI) and for kokanee was 207.7 mm (± 9.1; 95% CI). Kokanee PSD and RSD-P were 15 and 13, respectively. Rainbow Trout PSD and RSD-P were 28 and 0, respectively.

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INTRODUCTION

Mackay Reservoir is located in Custer County, Idaho (Figure 62) and has a storage capacity of 45,050 acre feet. The reservoir is on the Big Lost River and is impounded by Mackay Dam, 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-sized Rainbow Trout *Oncorhynchus mykiss* with detailed records dating back to the 1960s. The current annual Rainbow Trout stocking total is about 12,000 catchable-sized fish. Kokanee *Oncorhynchus nerka* fry are infrequently stocked (e.g., 2009 and 2019), but wild kokanee reproduction does regularly occur. The current kokanee stocking strategy is to stock surplus kokanee fingerlings when available from the IDFG Mackay Fish Hatchery. Currently, Rainbow Trout, kokanee, Brook Trout *Salvelinus fontinalis*, Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri*, and Mountain Whitefish *Prosopium williamsoni* are the sportfish inhabiting the reservoir. Yellow Perch *Perca flavescens* were sampled in gill nets in 2013, but there has been no record of their presence since that sampling effort. There are no special fishing regulations for Mackay Reservoir except for Mountain Whitefish, which is zero harvest.

METHODS

We set eight experimental gill nets at long-term monitoring locations to assess the Rainbow Trout and kokanee populations in Mackay Reservoir from July 1-2, 2019 (Appendix J; Figure 62). 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 with 16 panels that were 3-m long with two panels for each mesh size randomly positioned. The mesh sizes of the panel 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. Nets were set at dusk in the thermocline and retrieved the following morning. All fish captured were identified to species, measured for total length to the nearest millimeter (mm), and weighed to the nearest gram (g). We calculated catch-per-unit-effort (CPUE) for each species as the number of fish per net-night and calculated 95% confidence intervals around those estimates. We removed sagittal otoliths from a subsample 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).

We calculated proportional size distribution (PSD), relative stock density (RSD), and relative weights to describe the size structure and condition of kokanee and Rainbow Trout in

Mackay Reservoir. Kokanee PSD was calculated as the number of fish greater than or equal to 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 greater than or equal to 300 mm divided by the number greater than or equal to 120 mm multiplied by 100. Rainbow Trout PSD was calculated as the number of fish greater than or equal to 400 mm divided by the number greater than or equal to 250 mm, multiplied by 100. Rainbow Trout RSD-P was calculated as the number of fish greater than or equal to 500 mm divided by the number greater than or equal to 250 mm multiplied by 100.

RESULTS

We collected 523 fish over 8 nights of gillnetting effort in Mackay Reservoir in 2019. Species composition was dominated by kokanee (92%), followed by Rainbow Trout (8%), and Speckled Dace (<1%). No Yellow Perch were captured in our gill nets. Catch rates for kokanee were 60 fish/net-night (± 22) and the average total length (mm) for kokanee was 208 mm (± 73; SD; Figure 63; Figure 64). Kokanee PSD and RSD-P were 15 and 13, respectively, and the average relative weight (W_R) for kokanee was 87 (± 16; SD; Figure 65). We estimated kokanee age to vary from 0 to 3 years old with the highest frequency of fish estimated at 1 year old (Table 18). We estimated annual survival (*S*) for kokanee to be 31%, instantaneous total mortality (*Z*) to be 1.18, and annual mortality (*A*) to be 69%. Catch rates for Rainbow Trout were 6 fish/net-night (± 3) and the average total length for Rainbow Trout was 368 mm (± 53; SD; Figure 66). The average relative weight for Rainbow Trout was 76 (± 10; SD; Figure 67), PSD and RSD-P were 28 and 0, respectively. We estimated annual survival for Rainbow Trout to be 33%, instantaneous total mortality to be 1.10, and annual mortality to be 67%. Several high precipitation years recently have contributed to the highest overwinter storage levels in the past 20 years (Figure 68). This will likely help survival of all age-classes of fish in the reservoir.

DISCUSSION

Compared to other reservoirs in the Upper Snake Region, Mackay Reservoir has little long-term monitoring or historical data. Gebhards sampled Mackay Reservoir with two gill nets in May 1962 (IDFG files) which yielded eight Rainbow Trout and two Mountain Whitefish. Mackay Reservoir was sampled with two gill nets on two occasions during May 1973 (Jeppson 1975). The first survey (May 15) was a short net set (3 hours and 15 minutes) which yielded one Rainbow Trout in each net. The second survey (May 20) was an overnight set which yielded 36.5 Rainbow Trout, six Brook Trout, 7.5 Mountain Whitefish, and 0.5 kokanee per net. Jeppson (1975) also surveyed Mackay Reservoir during April 1974, and collected 22 Rainbow Trout, 11 Brook Trout, and 4 Mountain Whitefish per net-night. The majority of the work conducted on Mackay Reservoir since Jeppson's gill netting has been angler surveys. We conducted a survey in 2008 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% shift in the Rainbow Trout population. However, we believe that increased netting should be used periodically to establish baseline conditions that can be used in future comparisons. In 2013, our survey served as the first thorough, comprehensive gillnetting effort on Mackay Reservoir. We used 17 floating and 13 sinking gill nets, which is a large amount of effort for an impoundment of this size. Therefore, we reduced our effort to 8 nets total in the mid to lower sections, which are the deeper portions of the reservoir that maintain pool throughout the year.

Catch rates in gill nets for kokanee have been increasing since 2008 with 2019 having the highest catch rates for kokanee to date. Although we did sample a high proportion of age-0 kokanee, multiple year-classes were present and the high abundance of age-0 fish will provide many opportunities for anglers in the future. Catch rates for Rainbow Trout have remained stable compared to 2017, but our gill netting methods are better suited for estimating the kokanee population because nets are set in the thermocline where kokanee school rather than near the surface or benthos. Compared to netting efforts prior to 2017 Rainbow Trout abundance is down; however, if we continue our current netting protocol, we will have comparable methods from 2017 onward. Therefore, we should be able to notice shifts in Rainbow Trout abundance based on gill netting methods. On the contrary, this shift in declining Rainbow Trout abundance could warrant additional netting efforts to target Rainbow Trout using floating gill nets near the surface and sinking gill nets near the bottom rather than suspended gill nets. Relative weights for both kokanee and Rainbow Trout were less than 100, suggesting that food resources are becoming limited as fish densities increase. This warrants assessing fish condition (e.g., Fulton's condition factor) in the future to infer the condition of fishes relative to food resources and density, which can influence our management decisions regarding stocking and harvest.

Mackay Reservoir historically has been drafted to less than 5% of volume annually, which severely reduced or eliminated reservoir carryover of many fish and likely made adult migration into tributaries difficult. Reservoir drawdown reduces the amount of available habitat for fishes and can negatively impact fish abundance and the size structure of fish populations (Paller 2011). The shift in water management provides habitat that results in better carryover of fish, which is likely responsible for the increase in kokanee abundance in the current survey.

RECOMMENDATIONS

1. Continue monitoring the kokanee population every two years using the same amount of netting effort to make surveys comparable over time, to assess the population dynamics of kokanee, to evaluate stocking success, and to inform management decisions

2. Every three years, expend more netting effort using sinking and floating nets to assess the Rainbow Trout population to inform stocking scenarios, harvest regulations, and to assess the population dynamics of Rainbow Trout with a larger sample size.

3. Continue assessing the condition of kokanee using a condition factor and monitor relative weight.

4. Consider liberalizing fishing regulations if kokanee condition decreases.

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Table 19. The mean length-at-age (i.e., total length [mm]) with standard deviations (SD) for kokanee and Rainbow Trout in Mackay Reservoir in 2019. The proportion of the aged sample represented by that age is included with the standard error (SE) for that estimate.

Species	**Age**	***n***	**Mean length (mm)**	**SD**	**Proportion**	**SE**
Kokanee	0	205	71	10	0.25	0.04
	1	211	211	24	0.36	0.05
	2	156	260	58	0.37	0.05
	3	4	345	26	0.30	0.02
Rainbow Trout	2	20	306	25	0.27	0.05
	3	35	353	38	0.42	0.06
	4	18	420	34	0.25	0.05
	5	3	439	8	0.04	0.03
	6	1	473	-	0.01	0.00

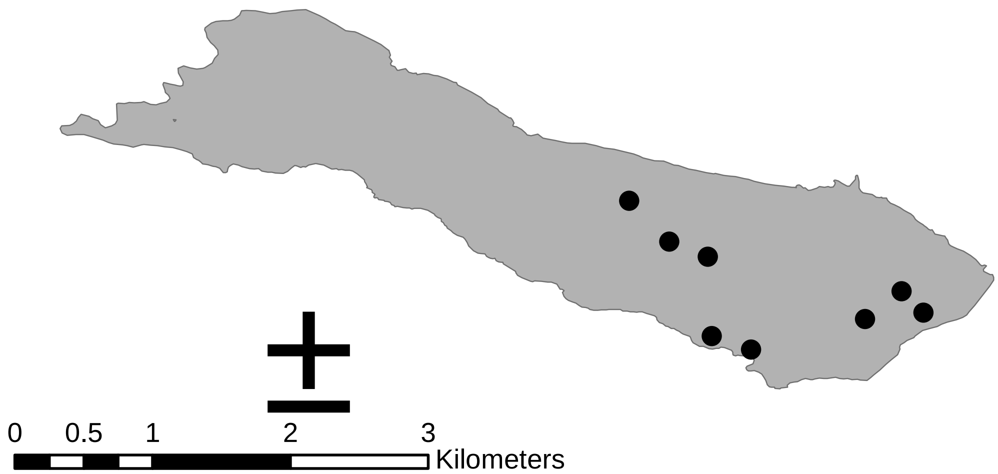

Figure 62. Gill net sample site locations in Mackay Reservoir, Idaho, 2019.

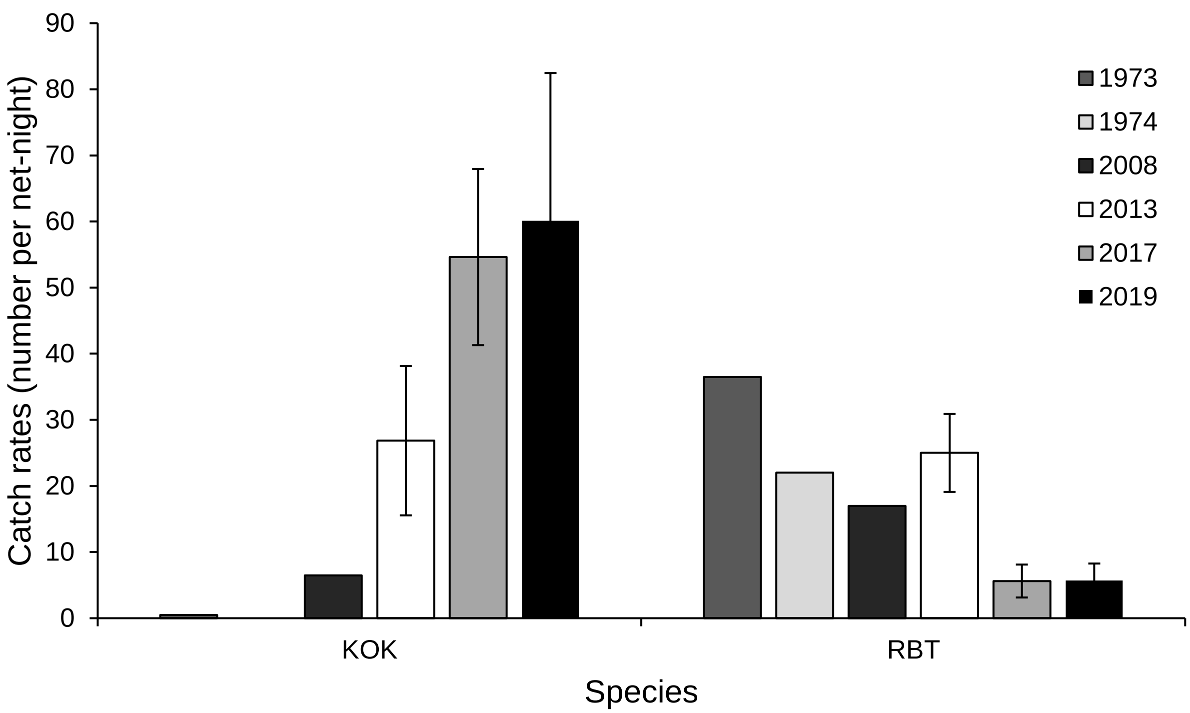

Figure 63. Gill net catch rates (number of fish per net-night) for kokanee (KOK) and Rainbow Trout (RBT) in Mackay Reservoir from 1973-2019.

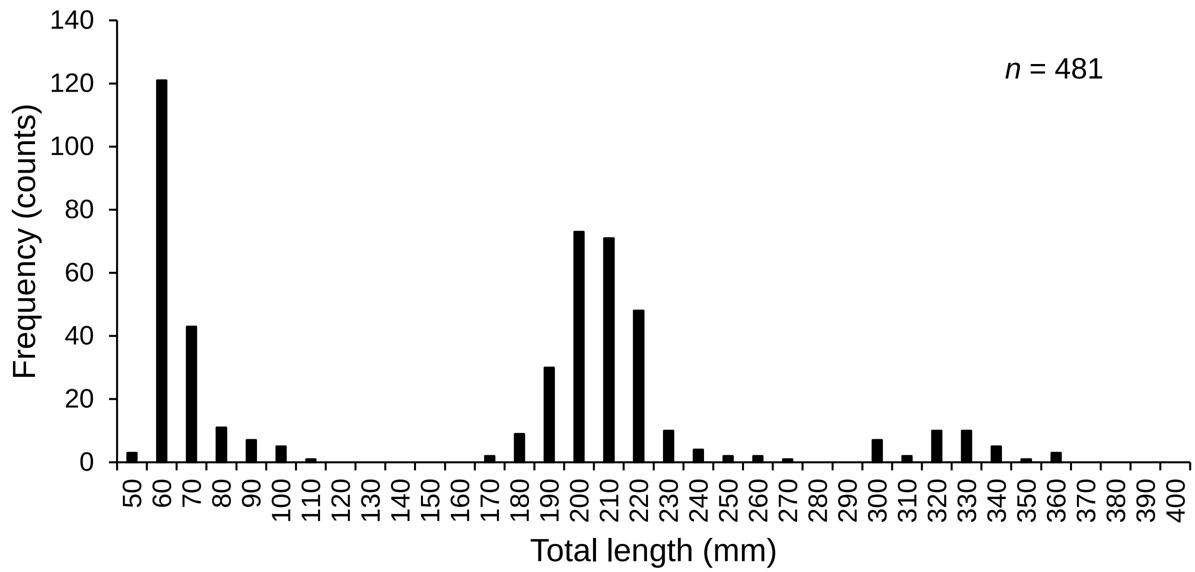

Figure 64. Length-frequency distribution of kokanee (KOK) captured with gill nets in Mackay Reservoir in 2019.

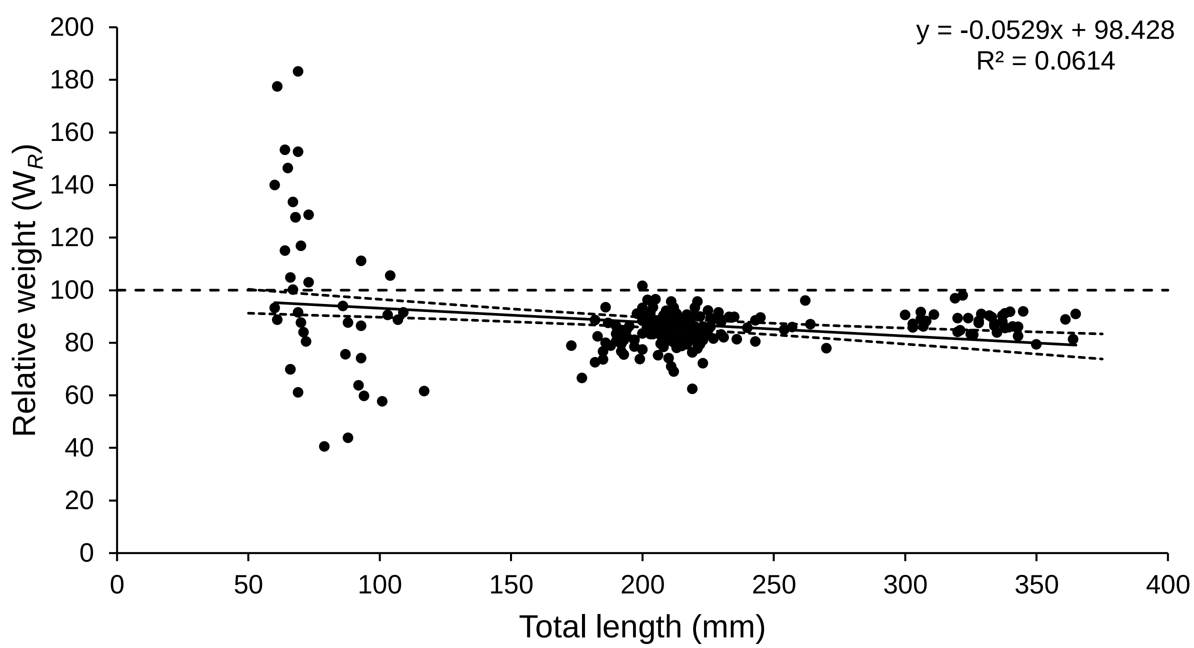

Figure 65. Kokanee (KOK) relative weights (W_R) by total length from gill netting surveys in Mackay Reservoir, 2019. Linear regression and 95% confidence intervals are represented with a solid and dotted line, respectively. Dashed line represents mean W_R of 100, which are based on 75^{th} percentile of weight at a given length.

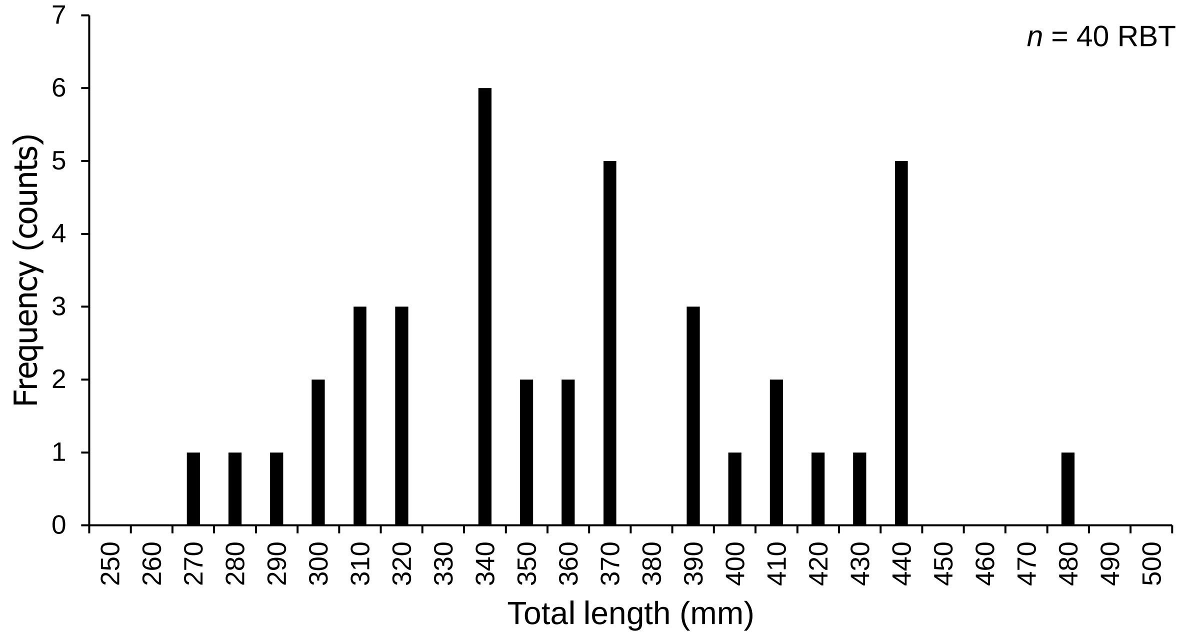

Figure 66. Length frequency distribution of Rainbow Trout (RBT) captured with gill nets in Mackay Reservoir in 2019.

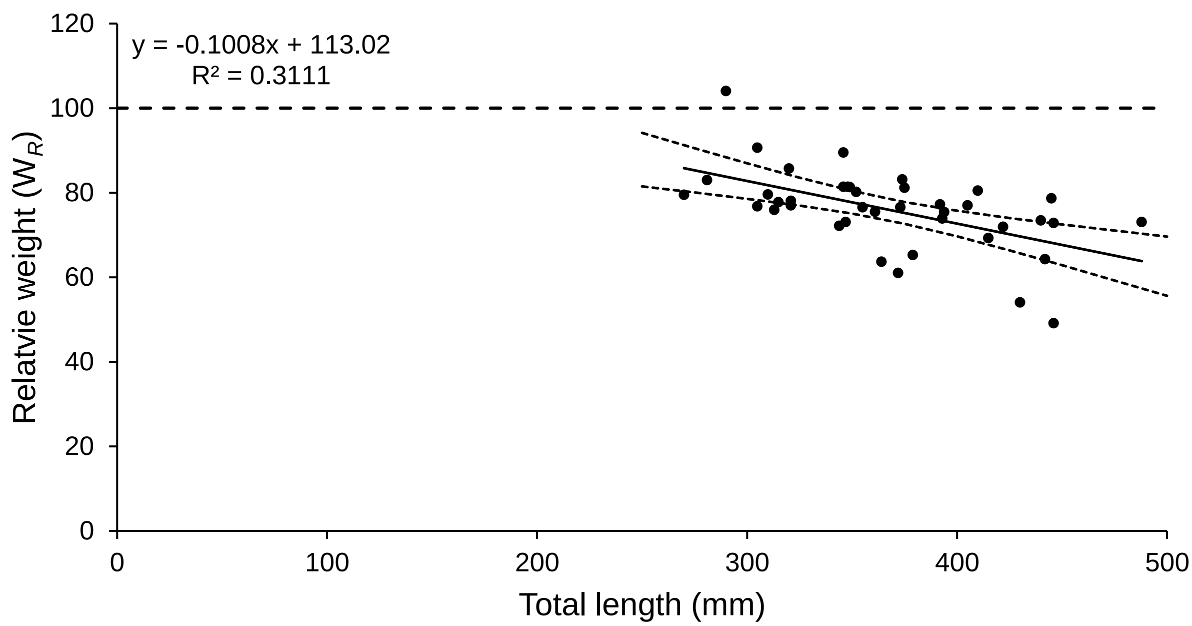

Figure 67. Rainbow Trout (RBT) relative weights (W_R) by total length from gill netting surveys in Mackay Reservoir, 2019. Linear regression and 95% confidence intervals are represented with a solid and dotted line, respectively. Dashed line represents mean W_R of 100, which are based on 75th percentile of weight at a given length.

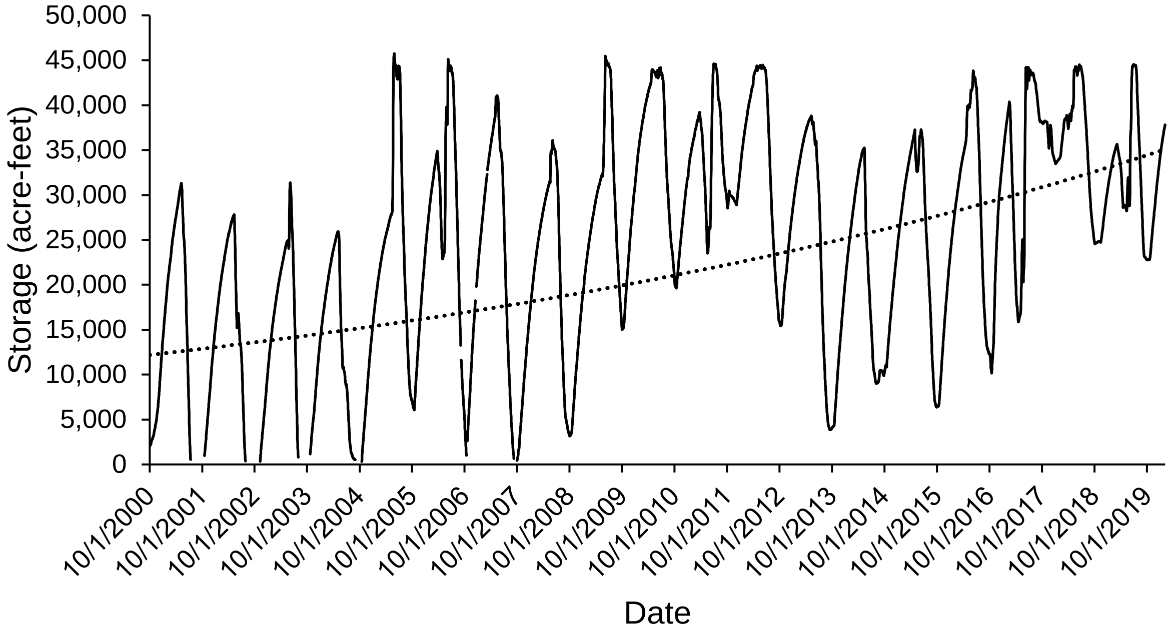

Figure 68. Storage capacity trend in Mackay Reservoir from 2000-2020. The solid line is the reservoir capacity over the time period and the dotted line is the exponential trend over time.

FISHING AND BOATING ACCESS PROGRAM

ABSTRACT

We maintain 55 fishing and boating access sites within the Upper Snake Region. Sites need continual maintenance, repair, and cleaning. These duties were completed in 2019 at all access sites. Additionally, we completed several improvement projects at IDFG-owned properties including the regional office, Henrys Lake Cabin, and the Cougar Creek backcountry airstrip in the Salmon Region. These projects included replacing office lights with more efficient LED lights controlled with motion sensors, office roof repairs to fix water leaks, regional office carpet replacement, office drinking fountain updates. Also included were updates at the Henrys Lake Cabin, including window replacement, sealing and insulating of the loft, and replacing exterior paneling. Upper Snake Region Access program staff also facilitated the development of fishing and boating infrastructure at other properties including Spring Hollow Access site (owned by the Bureau of Reclamation), Palisades Dam parking lot (U. S. Forest Service property), and a stream restoration project to improve flows on a Henrys Lake tributary, Rock Creek, located on private property. Lastly, staff spent considerable time working on partnerships to cooperatively manage sites in the Upper Snake Region, including Frome Park, State Park and South Shore sites in Island Park, and the Antelope Creek access site in the Big Lost River drainage.

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INTRODUCTION

The goal of 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 utilize and enjoy a wide variety of water types throughout the Upper Snake Region. Staff maintains 55 fishing and boating access sites within the region. Also, access to properties owned by others (other state agencies, federal, or nongovernmental 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 over grown 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. 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 grant sources. The access program in the Upper Snake Region 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

Upper Snake Region Access staff completed normal operations and maintenance activities in 2019. In addition, staff contributed directly to the completion of several larger-scale renovation or repair projects on department-managed properties.

We built a new access road into the Henrys lake Hatchery utilizing 200 tons of rejected gravel that we acquired from the Idaho Transportation Department at no cost. This work was necessary for reestablishing the right of way into our property as the previous road was located on a neighboring property that had recently sold. We also added a wildlife friendly property line fence that helps direct the traffic from the old road. We also assisted in the remodel of the cabin at Henrys Lake to address structural concerns and bat problems. Eliminating the bat access to the loft of the cabin restored sleeping accommodations and safe habitation which had been lost. At this facility, we also helped clean the ladder for Hatchery Creek at the spawn shed.

Using funding acquired from the Bureau of Reclamation, we rebuilt the Spring Hollow Access site, where we added a new boat ramp, additional parking, a vaulted toilet, and sign kiosks. The signs will help users know what hazards they will face on this adventurous section of river downstream. This was $100,000 grant in partnership with the BOR. Fremont County also assisted with this project by rebuilding the 6mile stretch of road that users travel to get to this site. The county first bulldozed the roadway into the site, hauled in hundreds of tons of pit run, and finally finished it off with road mix.

We purchased several piece of large equipment to improve our efficiency including a newer Motor grader, skid steer, and backhoe. We utilized state surplus from Idaho DOT and were able to update machinery by replacing equipment which was 30 years newer. Thus, making it easier to find parts and cut costs on both parts and repairs. We also worked with our fleet program and updated our dump truck, front end loader, and bulldozer, items that were left in Boise from the engineering crew. This helped the regional access crew be more efficient in our work efforts. For example, we were able to grade and repair almost 100 miles of road in 2019, something that hasn't been done for several years. The roads provide critical access into fishing and boating access sites and even Wildlife Management Areas. We have updated our primary truck and set it up to be mobile work shop, which will cut down on fuel costs and time of transportation, making IDFG more efficient. We spend a lot of time at our access sites and hope that the public will see our presence and understand how we are actively working on projects to make access better for them.

We assisted staff from the Salmon Region finish the Cougar Creek Air strip within the Middle Fork Salmon River Wilderness. Two of us flew in and helped finish grading this landing strip, which is the first landing strip constructed in this backcountry area in about 50 years.

We worked with the South Fork Coalition, making sure our users of the South Fork Snake River can benefit from this partnership. We developed cost proposals for paving several sites and worked together to maintain fishing and boating sites in good condition.

We provided the equipment and labor to install head gates on Rock Creek, a tributary to Duck Creek which is an important spawning tributary for Yellowstone Cutthroat Trout in Henrys Lake. The head gates replaced push up dams and tarps which had no ability to control how much water was diverted at these points of diversion. Installation of these new head gates required quite a bit of planning and preparation from a collaborative group to accomplish the project. In 2020, we will plan to finish installing flumes so the amount of water diverted can be accurately measured at these new structures. The result of this effort should be increased flows in Duck Creek and increases in fish spawning and rearing habitat.

We worked with several partners to develop relationships within local communities. We have recently reached out and made contact with Bonneville County Road and Bridge for upcoming work at the Clowards Access Site. We have strengthened our relations with Custer County and the South Fork Coalition.

Another project we completed in 2019 was a collaborative effort with Friends of the Teton River and the Teton Regional and Trust. . This collaborative worked with local artists to paint landscape and wildlife scenes on the interior walls of vault toilets in Teton Valley and at Spring Hollow. We now have six toilets painted with some remarkable art, which we believe will help deter vandalism and lower maintenance costs at these facilities while highlighting some of the regional resources and local talent. This project is growing and we are looking forward to gaining more partners and expanding work into the South Fork Snake River and the Big Lost River drainages.

In 2019, the recreation site maintenance foreman was in charge of working with contractors making changes to the Regional Office. Within this time frame, we finished retrofitting half of the building's interior and exterior lights to LED, and added many motion sensors to help cut back on wasted energy. We worked with contractors to repair the roof, and resealed the roof top and replaced all the metal roofing. This project was a priority because of the amount of water damage the facility had sustained because of a leaky roof. We replaced two thirds of the carpet

in the building, which was not an easy task considering how much moving this required. During this process, we were tasked with redesigning the cubical areas to accommodate three additional office spaces. We also added more filtered water drink stations by updating the drinking fountains, which should cut back on recycling and trash in the office. Other noteworthy accomplishments included our staff working on getting better equipped to handle the snow plowing on our access sites, ensuring that our users can get into sites year round.

ACKNOWLEDGEMENTS

We have been fortunate to work with several loyal employees that return year after year including William (Dee) Killian and Colten Hewlett. We would also like to thank our partners and specifically the employees and volunteers including Colby Serr who made an impact and will be missed. Without their help we would not be able to provide the level of service to our hunters, anglers, and trappers at our access sites.

Appendix A. Locations used in population surveys on the Henrys Fork Snake River, Idaho 2019. All locations are in Zone 12 using the NAD27 datum.

	Start		End	
Reach	Easting	Northing	Easting	Northing
Box Canyon	468677	4917703	467701	4914352
Stonebridge	470486	4882921	464168	4884320

Appendix B. Mean total length, length range, proportional size distribution (PSD), and relative stock density (RSD-400 and RSD-500) of Rainbow Trout captured in the Box Canyon electrofishing reach, Henrys Fork Snake River, Idaho, 1991-2019. RSD-400 = (number ≥400 mm/ number ≥200 mm) x 100. RSD-500 = (number ≥500 mm/ number ≥200 mm) x 100.

Year	Number	Mean TL (mm)	Length Range (mm)	PSD	RSD-400	RSD-500
1991	711	293	71–675	65	46	9
1994	1,226	313	46–555	90	46	3
1995	1,590	316	35–630	61	30	1
1996	1,049	300	31–574	66	20	1
1997	1,272	307	72–630	47	14	1
1998	1,187	269	92–532	45	13	0
1999	874	330	80–573	63	16	1
2000	1,887	293	150–593	45	11	1
2002	1,111	352	100–600	75	28	0
2003	599	365	100–520	86	42	1
2005	1,064	347	93–595	76	44	2
2006	1,200	320	95–648	64	26	2
2007	1,092	307	91–555	58	21	2
2008	1,417	341	92–536	73	20	1
2009	1,371	350	80–587	79	27	1
2010	2,700	307	75–527	51	23	1
2011	1,224	348	111–550	74	27	1
2012	1,583	302	77–560	57	22	1
2013	2,072	295	110–535	39	14	1
2014	1,916	341	106–635	80	17	1
2015	1,219	296	90–509	83	25	0
2016	1,755	267	99–520	62	31	0
2017	1,165	292	84–512	72	24	1
2018	1,532	256	90–560	45	11	0
2019	2,165	261	89–518	70	39	9

Appendix C. Electrofishing mark-recapture statistics, efficiency (R/C), coefficient of variation (CV), Modified Peterson Method (MPM) and Log-Likelihood Method (LLM) population estimates (N) of age-1 and older Rainbow Trout (≥150 mm), and mean stream discharge (ft^3/s) during the sample period for the Box Canyon reach, Henrys Fork Snake River, Idaho, 1995-2019. Confidence intervals (±95%) for population estimates are in parentheses.

Year	M[a]	C[a]	R[a]	R/C (%)	CV	N/reach MPM	N/reach LLM	N/km LLM	Discharge (ft^3/s)
1995	982	644	104	16	0.04	6,037 (5,043-7,031)	5,922 (5,473-6,371)	1,601 (1,479-1,722)	2,330
1996	626	384	69	18	0.05	3,456 (2,770-4,142)	4,206 (3,789-4,623)	1,137 (1,024-1,250)	1,930
1997	859	424	68	16	0.06	5,296 (4,202-6,390)	5,881 (5,217-6,545)	1,589 (1,410-1,769)	1,810
1998	683	425	42	10	0.07	6,775 (4,937-8,613)	8,846 (7,580-10,112)	2,391 (2,049-2,733)	1,880
1999	595	315	38	12	0.07	4,844 (3,484-6,204)	5,215 (4,529-5,901)	1,409 (1,224-1,595)	1,920
2000	1,269	692	74	11	0.05	11,734 (9,317-14,151)	12,841 (11,665-14,017)	3,471 (3,153-3,788)	915
2002	1,050	511	81	16	0.05	6,574 (5,329-7,819)	7,556 (6,882-8,230)	2,042 (1,860-2,224	820
2003	427	167	20	12	0.10	3,472 (2,147-4,797)	3,767 (3,005-4,529)	1,018 (812-1,224)	339
2005	735	401	90	22	0.06	3,250 (2,703-3,797)	4,430 (3,922-4,938)	1,197 (1,060-1,334)	507
2006	887	356	61	17	0.05	5,112 (4,005-6,219)	5,986 (5,387-6,585)	1,618 (1,456-1,779)	1,783
2007	737	332	51	15	0.08	4,725 (3,598-5,852)	8,549 (7,288-9,810)	2,311 (1,970-2,652)	542
2008	887	615	93	15	0.04	5,818 (4,842–7,089)	5,812 (5,312-6,312)	1,571 (1,436–1,706)	894
2009	673	775	112	14	0.04	4,628 (3,910-5,540)	5,034 (4,610-5,458)	1,361 (1,246-1,476)	1,377

Appendix C (continued)

Year	M[a]	C[a]	R[a]	R/C (%)	CV	N/reach MPM	N/reach LLM	N/km LLM	Discharge (ft^3/s)
2010	1,309	1,292	262	20	0.03	6,439 (5,820-7,058)	8,341 (7,857-8,825)	2,254 (2,123-2,385)	626
2011	639	652	74	11	0.06	5,571 (4,516-6,988)	6,548 (5,816-7,280)	1,770 (1,572-1,968)	1,159
2012	793	901	116	13	0.04	6,120 (5,178-7,313)	6,915 (6,339-7,491)	1,869 (1,713-2,025)	911
2013	1,115	1,301	120	9	0.04	12,008 (10,148-14,349)	14,358 (13,207-15,509)	3,881 (3,570-4,129)	648
2014	1,532	636	175	28	0.03	5,547 (4,901-6,335)	5,828 (5,491-6,165)	1,575 (1,484-1,666)	971
2015	765	351	67	19	0.10	3,964 (3,216-4,989)	6,220 (4,950-7,490)	1,681 (1,338-2,024)	709
2016	1,107	397	107	27	0.06	4,082 (3,486-4,850)	5,208 (4,645-5,771)	1,408 (1,255-1,560)	464
2017	625	425	56	13	0.07	4,679 (3,689-6,065)	6,699 (5755-7,643)	1,811 (1,556-2,066)	918
2018	715	646	93	14	0.05	4,927 (4,097-6008)	6,430 (5,780-7,080)	1,738 (1,562-1,913)	1,067
2019	1,352	903	123	14	0.04	9,896 (8,258-11,470)	11,326 (10,449-12,203)	3,061 (2,824-3,298)	880

[a]M = number of fish marked on marking run; C = total number of fish captured on recapture run; R = number of recaptured fish on recapture run.

Appendix D. Locations used in population surveys on the Fall River, Idaho 2019. All locations are in Zone 12 using the WGS84 datum.

	Start		End	
Reach	Easting	Northing	Easting	Northing
Sheep Falls	492528	4880826	487838	4878774
Kirkham Bridge	471273	4878201	466984	4876026

Appendix E. Fin clip data from Yellowstone Cutthroat Trout (YCT) stocked in Henrys Lake, Idaho. Annually, ten percent of stocked YCT receive an adipose fin clip. Fish returning to the Hatchery ladder and fish captured in annual gillnet surveys are examined for fin clips.

Year	No. Clipped	No. checked at Hatchery	No. detected	Percent clipped	No. checked in gillnets	No. detected	Percent clipped	Overall percent clipped
1996	100,290	--	--	--	--	--	--	--
1997	123,690	178	5	3%	--	--	--	3%
1998	104,740	--	--	--	--	--	--	--
1999	124,920	160	20	13%	--	--	--	13%
2000	100,000	14	1	7%	--	--	--	7%
2001	99,110	116	22	19%	--	--	--	19%
2002	110,740	38	7	18%	--	--	--	18%
2003	163,389	106	37	35%	273	47	17%	22%
2004	92,100	--	--	--	323	28	8%	9%
2005	85,124	2,138	629	29%	508[a]	55	11%	26%
2006	100,000	2,455	944	39%	269[a]	20	8%	35%
2007	139,400	--	--	--	770	70	9%	9%
2008	125,451	4,890	629	13%	100	10	10%	13%
2009	138,253	4,184	150	4%	91	9	10%	4%
2010	132,563	4,253	90	2%	505	31	6%	3%
2011	112,744	3,037	137	5%	1,097[b]	72	7%	5%
2012	75,890	2,880	215	7%	500	52	10%	8%
2013	75,600	3,360	268	8%	478	47	10%	8%
2014	72,900	6,226	651	10%	626[b]	60	10%	10%
2015	95,500	5,211	627	12%	254	24	9%	12%
2016	100,750	4,689	548	12%	238	27	11%	12%
2017[c]	--				149	12	8%	
2018[c]	--							
2019[c]	--							

[a] Includes fish from gill net samples and creel survey.
[b] Includes fish from annual spring gill net monitoring and fish collected in monthly stomach sample gill netting
[c] No fish were clipped due to the parentage based tagging implemented in 2017.

Appendix F. Historic annual stocking (x 1,000) of Henrys Lake, Idaho, 1923 – 2019 for Yellowstone Cutthroat Trout (YCT), Hybrid Trout (HYB), and Brook Trout (BKT).

YEAR	YCT	HYB	BKT	TOTAL
1923	40	0	0	40
1924	0	0	0	0
1925	1	0	1	2
1926	140	0	0	140
1927	222	0	0	222
1928	116	0	0	116
1929	0	0	0	0
1930	0	0	0	0
1931	634	0	0	634
1932	170	0	0	170
1933	50	0	0	50
1934	980	0	0	980
1935	632	0	3	635
1936	0	0	0	0
1937	719	0	0	719
1938	753	0	0	753
1939	370	0	0	370
1940	750	0	0	750
1941	0	0	0	0
1942	1589	0	0	1589
1943	1665	0	0	1665
1944	1537	0	0	1537
1945	818	0	0	818
1946	1670	0	0	1670
1947	238	0	0	238
1948	584	0	0	584
1949	684	0	2	686
1950	779	5	6	790
1951	2070	0	0	2070
1952	610	8	0	618
1953	600	0	0	600
1954	1223	0	0	1223
1955	1243	0	0	1243
1956	985	0	0	985
1957	640	0	0	640
1958	534	0	0	534
1959	454	0	0	454
1960	1024	138	0	1162
1961	1570	390	0	1960
1962	1366	385	0	1751
1963	1300	565	0	1865
1964	1455	0	0	1455
1965	1755	0	0	1755
1966	1481	563	0	2044

Appendix F (continued)

YEAR	YCT	HYB	BKT	TOTAL
1967	1159	448	0	1607
1968	847	132	0	979
1969	111	476	0	587
1970	391	133	0	524
1971	763	184	0	947
1972	834	0	0	834
1973	1145	0	0	1145
1974	1105	0	0	1105
1975	1024	0	101	1125
1976	862	200	167	1229
1977	825	200	137	1162
1978	946	179	89	1214
1979	1134	125	96	1355
1980	1040	32	91	1163
1981	2251	146	20	2417
1982	2442	242	18	2702
1983	2179	229	22	2429
1984	2041	135	0	2175
1985	995	33	111	1139
1986	989	292	0	1281
1987	663	256	0	919
1988	1011	312	0	1323
1989	1090	251	95	1436
1990	1001	200	157	1358
1991	1326	201	129	1656
1992	943	203	189	1336
1993	1060	217	112	1388
1994	1048	201	115	1363
1995	1381	144	136	1662
1996	661	200	196	1057
1997	1237	180	204	1621
1998	1047	204	207	1459
1999	1249	204	0	1453
2000	978	0	0	978
2001	991	135	0	1126
2002	1107	331	0	1438
2003	1634	264	99	1996
2004	921	38	117	1077
2005	851	201	152	1204
2006	1124	150	107	1381
2007	1394	146	104	1644
2008	1254	196	198	1648
2009	1382	220	171	1773
2010	1326	138	93	1557
2011	1127	205	100	1432
2012	768	221	101	1090
2013	756	213	110	1079

Appendix F (continued)

YEAR	YCT	HYB	BKT	TOTAL
2014	729	167	83	979
2015	955	167	71	1193
2016	1105	177	5	1288
2017	1012	212	202	1426
2018	1160	239	105	1504
2019	1209	190	106	1505

Appendix G. Location of Ririe Reservoir gill netting locations during the summer of 2019. All coordinates are Zone 12, and WGS 84 datum.

Date	Net	Strata	Latitude	Longitude
June 12	52	Lower	43.57151	-111.73504
June 12	120	Lower	43.55620	-111.73372
June 12	145	Lower	43.54991	-111.73966
June 13	215	Middle	43.53648	-111.72854
June 13	299	Middle	43.51664	-111.72681
June 13	307	Middle	43.51741	-111.73172
June 14	316	Middle	43.51208	-111.73313
June 14	382	Upper	43.50300	-111.74031
June 14	392	Upper	43.50483	-111.74905
June 14	415	Upper	43.50472	-111.75525

Appendix H. Gill net locations in Island Park Reservoir, 2019. All coordinates used NAD27 and are in Zone 12.

Net Site	**Latitude**	**Longitude**
25	44.427720°	-111.393840°
38	44.419560°	-111.396360°
104	44.424930°	-111.406520°
109	44.423220°	-111.407680°
188	44.422200°	-111.421520°
211	44.428500°	-111.425200°
241	44.423030°	-111.445450°
269	44.429230°	-111.452890°
293	44.426590°	-111.456700°

Appendix I. Annual kokanee stocking in Island Park Reservoir, Moose Creek, and Big Springs Creek, 1944 – 2019.

	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		167,850				
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						

[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 unable to stock

Appendix J. Net numbers and locations for gill netting in Mackay Reservoir.

Net number	Latitude	Longitude
26	43.95162	-113.67741
34	43.95304	-113.67883
62	43.95123	-113.68120
149	43.94923	-113.68863
175	43.95011	-113.69120
195	43.95626	-113.69397
217	43.95895	-113.69660

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