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ABSTRACT

We used 48 gill net nights of effort in the spring to evaluate trout populations in Henrys Lake, 2014. Mean trout catch rates of 15.2 per net (95% CI ± 2.9) were above the long-term trend of 12.7 trout/net (95% CI ± 2.0). Abundances of trout by species varied when compared to the long-term trend. Brook Trout *Salvelinus fontinalis*, hybrid trout, and Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* were at, below, and above historic abundances, respectively. Mean relative weight (W_r) for all trout species (all sizes combined) ranged between 91 and 97 and has increased compared to prior years. Median Utah Chub *Gila atraria* catch rate of 30.5 fish/net-night was the highest reported and suggests the chub population has increased. For the past three years, stocking rates have been reduced ~500,000 to achieve lower trout densities in an effort to improve condition and growth. Future stocking rates should continue to take into account contributions from natural reproduction and relative weights, and be adjusted accordingly until management goals are attained. We examined the stomach contents of 114 Brook Trout, 146 hybrid trout, 365 Yellowstone Cutthroat Trout, and 586 Utah Chub from May to October to assess diet composition and evaluate dietary overlap between the species. We also employed a stable isotope analysis approach to investigate the potential interactions of trout and Utah Chub. For stable isotope analysis we collected white muscle tissue across a wide size range of Brook Trout (n=9), hybrid trout (n=9), Utah Chub (n=20), and Yellowstone Cutthroat Trout (n=10). Brook Trout exhibited the widest dietary niche (SEA_c = 1.77) while Utah Chub contained the narrowest dietary niche (SEA_c = 0.87) suggesting a more specialized diet than trout. We found no biologically significant diet overlap between trout and Utah Chubs based on diets and stable isotopes analysis, which suggests trout performance (e.g., growth, condition) may be minimally impacted from competitive interactions with Utah Chub for pelagic and benthic food resources. We monitored dissolved oxygen levels to assess the possibility of a winterkill event from January 8, 2013 through February 23, 2014. Based on depletion estimates, we predicted dissolved oxygen would not reach critical levels (10 g/m^3). However, larger than average summer drawdown prompted concerns of low winter lake level and aeration was deployed to potentially mitigate for any impacts from the low lake level.

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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 bouvieri*, 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. Since 1929, IDFG has stocked a total of over 77 million Yellowstone Cutthroat Trout, 9 million hybrid trout, and nearly 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 consistent with providing a quality fishery as opposed to liberal bag limits 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 the anglers creel. 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.

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 km 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 (Figure 1).

OBJECTIVES

To obtain current information on fish populations and limnological characteristics on Henrys Lake, and to develop appropriate management recommendations to benefit anglers.

METHODS

Population Monitoring

As part of routine population monitoring, we set gill nets at six standardized locations in Henrys Lake from May 12 - 21, 2014 for a total of 48 net nights (Figure 1). Gill nets consisted of either floating or sinking types measuring 46 m by 2 m, with mesh sizes 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 measured to total length (TL) in mm. We calculated fish per net, and also calculated 95% confidence intervals.

We examined all Yellowstone Cutthroat Trout handled through the year for adipose fin clips as part of our evaluation of natural reproduction. Beginning in the 1980s, 10% of all stocked Yellowstone Cutthroat Trout have been marked with an adipose fin clip prior to stocking (Appendix A & B). To estimate contributions to the Yellowstone Cutthroat Trout 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 around 10% in the at-large population would be expected without additional natural reproduction. When the ratio of marked fish is less than 10%, we assume that natural reproduction is contributing to the population.

We removed the sagittal otoliths of all trout caught in our gill nets for age and growth analysis. After removal, all otoliths were cleaned on a paper towel and stored in individually-labeled envelopes. Ages were estimated by counting annuli under a dissecting microscope at 40x power. Otoliths were submerged in water and read in whole view when clear, distinct growth rings were present. We sectioned, polished, and read otoliths in cross-section view with transmitted light when the annuli were not distinct in whole view. Aged fish were then plotted against length using a scatter plot, and any outliers were selected, re-read, and the ages corroborated by two readers. For each species, we selected size classes (10 mm) that contained more than 5 otoliths for subsampling. Otoliths were then randomly selected for the subsampling. We aged all of the fish in size classes with less than 5 otoliths. The von Bertalanffy (1938) growth model was used to fit length:

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

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. The model was fitted to length-at-data by using the nonlinear model (NLIN) procedure in program R. We estimated mortality rate (Z) for each trout species by catch curve analysis. Age-1 trout were excluded from the analysis due to lack of recruitment to the gear type. We estimated Brook Trout, hybrid trout, and Yellowstone Cutthroat Trout mortality rates between the ages of 2 to 4.

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 (<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).

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. This methodology (and size designation) is used on other regional waters to provide comparison between lakes and reservoirs throughout the Upper Snake Region. We also calculated RSD-500, using the same equation as above, but used the number of fish greater than 500 mm as the numerator.

Diets

We collected Yellowstone Cutthroat Trout, hybrid trout, Brook Trout, and Utah Chub via gillnets from May through November to assess diet composition and evaluate dietary overlap between the species. We collected fish during standard population monitoring (May gill netting), followed by 6 net nights monthly from June to October. A combination of floating and sinking nets were set at standard monitoring locations at each monthly interval (Figure 1). All fish collected were weighed (g) and measured (TL: mm), and all Yellowstone Cutthroat Trout and Brook Trout were examined for adipose fin clips. Stomachs were removed, stored in individually labeled containers, and preserved in formalin for samples collected in May, June, and July. Stomachs collected in August, September, and October were preserved in containers with 95% EtOH. We determined dietary composition by manually removing, separating, and identifying materials contained in the foregut of each fish. Foregut was defined as the area anterior to the pyloric sphincter for trout and the anterior 50% of the stomach for Utah Chub in an effort to avoid compositional bias that may have arisen from differential digestion of diet items (Hyslop 1980). For each stomach, we identified individual food items using a dissecting scope, separated items by genus and then counted and weighed each genus to the nearest hundredth of a gram (0.01 g). Identified food items were summarized as percent weight of the total contents and percent of the total contents by number. In stomachs with highly masticated prey, we were unable to count the number of prey. However, when possible we did attempt to partition the prey into the appropriate categories for weighing. In instances where extremely high densities of *Daphnia* were encountered, we estimated numbers based on linear relationship of wet weight estimated in the lab by counting the number of *Daphnia* present in several trout stomachs (Number = 4140.6 × wet weight [g]; r^2=0.96). We then expanded the results to the total amount contained within the stomach. We removed trichoptera larvae from their casings prior to weighing and categorized casings in the vegetation section. Vegetation was excluded from the analysis given the lack of assimilated energy (Weiland and Hayward 1997). For partially digested sculpin *Cottus sp.*, we used the relationship between standard length (SL) and total length to estimate TL (TL range 58-101 mm). Conversion for Sculpin to estimate TL from SL was TL = 1.11903(SL); r^2 = 0.98. Prey taxa that were consumed

infrequently or in low proportions were combined into the other invertebrates category and included: Coleoptera, Ephemeroptera, Hemiptera, Hymenoptera, Odonata, and Simuliidae.

We estimated diets of trout and Utah Chubs using mean percent by weight:

$$MW_i = \frac{1}{P}\sum_{j=1}^{P}\left(\frac{W_{ij}}{\sum_{i=1}^{Q} W_{ij}}\right)$$

where MW_i is the mean percent by weight, P = number of fish with food in their stomachs, Q = number of food categories, and W_{ij} = weight of prey type i in fish j. In the trout diets we also estimated diets using mean percent by number:

$$MN_i = \frac{1}{P}\sum_{j=1}^{P}\left(\frac{N_{ij}}{\sum_{i=1}^{Q} N_{ij}}\right)$$

where MN_i is the mean percent by number, P = number of fish with food in their stomachs, Q = number of food categories, and N_{ij} = weight of prey type i in fish j. We excluded Utah Chub from the MN_i analysis due the low proportion of prey that where observed whole in the stomachs.

Diet overlap was assessed among species with Schoener's index (Schoener 1968):

$$D = 1 - 0.5\left(\sum_{i=1}^{s}\left|p_{ij} - p_{ik}\right|\right)$$

where D is the index value, S the number of prey items, p_{ij} and p_{ik} is the average percent weight (% MW_i) of each prey item i for species j and k, respectively. The index value of diet overlap can range from 0 (no overlap) to 1 (complete overlap) between two species being compared. Index values >0.60 indicate significant dietary overlap (Wallace 1981).

We assessed the percentage of Utah Chub susceptible to predation by each age class of trout. We used gape width estimates of 50% developed for Bear Lake Cutthroat Trout (Winters 2014) and summed the number of Utah Chub collected in gill nets that were within the gape widths of each age class of trout estimated from fitted length-at-age data.

Diet contents were summarized by species and compared to results from past studies on Henrys Lake (Garren et al. 2006; Schoby et al. 2014; Schoby et al. 2013). We tested for differences in diets among months for each species using a permutational multivariate analysis of variance (PERMANOVA), which tests the simultaneous response of one or more variables to factors in an ANOVA experimental design on the basis of a distance measure using permutation methods (Anderson 2001). The response variables were the proportion of the prey group by wet weight from the diet analysis and the predicator variable was season or species. Prey groups that represented <5% of the proportion of wet weight were excluded from the analysis. PERMANOVA was performed using the packages VEGAN (Oksanen et al. 2006) in the R-program (R Development Core Team 2007).

Stable isotope analysis

Brook Trout, Cutthroat Trout, hybrid trout, and Utah Chub used for stable isotope analysis were collected from monthly gillnetting in June and July 2014. A small portion (about 1 cm^3) of white muscle tissue without skin was dissected below the dorsal fin and above the lateral line from each fish, placed in a labeled bag, and immediately placed on ice. Potential prey sources (e.g. scuds, *daphnia*, sculpin, etc.) were sampled in order to compare the $\delta^{13}C$ and $\delta^{15}N$ signatures of trout with the isotopic signature of their prey. Benthic macroinvertebrates were collected by dredging the bottom of the lake, immediately picked from the samples alive, and placed on ice. *Daphnia* were collected using a Wisconsin zooplankton net (mesh 153 µm). Sculpin were collected electrofishing using a backpack electrofisher (Smith Root) along the shoreline. Utah Chub used for isotope analysis were collected from monthly gillnetting. All prey samples collected in the field were immediately placed on ice and transported to the laboratory and frozen (-20°C).

Because of the small size of many macroinvertebrates, multiple organisms of the same species were pooled to obtain enough sample to achieve the minimum mass required for reliable analyses (i.e. 0.25 mg). Whole bodies of at least 3 individual macroinvertebrates were pooled for isotope analysis. Trout tissue and prey samples were then freeze-dried for at least 48 h. Macroinvertebrates, prey fish, and trichoptera (removed from their casings) were analyzed whole. Trout white muscle tissue was homogenized into a fine powder using a Wig-L-Bug (DENTSPLY Rinn Digital Wig-L-Bug Mixer/Amalgamator, Model MDS). Prey fish (e.g., sculpin, Utah Chub) were homogenized whole using a Wiley Mill (40 mesh) and reground, if necessary, into a fine powder to insure homogeneity within each sample. Zooplankton were analyzed whole without homogenization.

Carbon (^{13}C) and nitrogen (^{15}N) stable isotope ratios were performed using a Finnigan Delta Plus continuous-flow isotope ratio mass spectrometer and elemental analyzer (Thermo Fisher Scientific Inc., Waltham, Massachusetts, USA) at the University of Arkansas, Stable Isotope Laboratory. Samples were weighed to 0.25-0.35 mg in individual 3.5 mm x 5 mm tin capsules. Stable isotope ratios were given using the standard delta notation ($\delta^{13}C$; $\delta^{15}N$) per mil (‰) according to the following formula:

$$\delta I = [(R_{sample}/R_{standard}) - 1] \times 10^3$$

where I is the isotope of interest (^{13}C or ^{15}N) and R is the $^{13}C/^{12}C$ or $^{15}N/^{14}N$ ratio in the sample and the standard. Standards employed were Vienna Pee Dee Belemnite for $^{13}C/^{12}C$ and atmospheric N_2 for $^{15}N/^{14}N$. Analytical precision (standard deviation) calculated from internal standards was 0.04‰ for ^{13}C and 0.06‰ for ^{15}N.

Stable isotope analysis (SIA) was completed on a total of 9 Brook Trout, 12 Yellowstone Cutthroat Trout, 9 hybrid trout, and 20 Utah Chub. We stratified fish by 10 mm size classes and randomly selected samples within the strata in an effort to evaluate SIA across a broad size range. The total number of stable isotope analyses completed on prey samples was 24. To evaluate the relative contributions of the different prey sources to the trout and chubs, we used the Bayesian stable isotope mixing model SIAR (Stable Isotope Analysis R) package (Parnell et al. 2010) in the R computing program (R Development Core Team 2012). The model uses a Bayesian approach, therefore diet contribution estimates were reported as 95% Bayesian credibility intervals (Parnell et al. 2010). Widely applied trophic isotopic fractionation factors for aquatic fish of $\delta^{13}C$: 0.05 ± 0.23‰ and $\delta^{15}N$: 3.49 ± 1.08‰ (Vander Zanden and Rasmussen 2001) were used in the model. The potential food sources were grouped into benthic

invertebrates (chironomids, scuds, leeches, caddisflies), zooplankton, and fish (Utah Chub, sculpin) for the trout simulations. For the Utah Chub simulations we only considered benthic invertebrates and zooplankton as the potential food sources in the mixing model.

To evaluate niche overlap of trout and Utah Chubs, we used population metrics originally developed by Layman et al. (2007) and recently improved by Jackson et al. (2011). We used the stable isotope Bayesian ellipses in R (SIBER) package (Parnell et al. 2008) in the R computing program (R Development Core Team 2012). We calculated standard ellipse areas (SEA_c), with a small sample size correction (subscript 'c'), for each fish species. A SEA_c is the bivariate measure of the distribution of individuals in trophic space. The estimated ellipse represents approximately 40% of the isotopic distribution of the individuals and represents the core dietary niche and typical resource use (Jackson et al. 2011).

Water Quality

The purpose of recording dissolved oxygen profiles is to develop a dissolved oxygen depletion model 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 around 10 g/m^2. Upon determining the likelihood of reaching the critical dissolved oxygen threshold prior to the projected recharge date of April 1st, a determination can be made of whether or not to use aeration.

We measured winter dissolved oxygen concentrations, snow depth, ice thickness, and water temperatures at four established sampling sites (Pittsburg Creek, County Boat Dock, Wild Rose, and Hatchery on Henrys Lake) between December 5, 2013 and February 23, 2014 (Figure 1). Holes were drilled in the ice with a gas-powered ice auger prior to sampling. We used a YSI model Pro-20 oxygen probe to collect dissolved oxygen readings at ice bottom and at subsequent one-meter intervals until the bottom of the lake was encountered. Dissolved oxygen mass is calculated from the dissolved oxygen probe's mg/L readings converted to total mass in g/m^3. This is a direct conversion from mg/L to g/m^3 (1000 L = 1 m^3). The individual dissolved oxygen readings at each site are then summed to determine the total available oxygen within that sample site. To calculate this value, we used the following formula:

Avg (ice bottom + 1 m) + Sum (readings from 2 m to lake bottom) = total O_2 mass

The total mass of dissolved oxygen at each sample site is then expressed in g/m^3 (Barica and Mathias 1979). Data are then natural logarithm (ln) transformed 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^3).

RESULTS

Population Monitoring

We collected 2,495 fish in 48 net nights in May with our standard gill net survey. Catch composition was 5% Brook Trout, 5% hybrid trout, 19% Yellowstone Cutthroat Trout, and 71% Utah Chub (Figure 2). Mean TL (with 95% confidence intervals) of Brook Trout was 395 mm (±14.7) and ranged from 180 - 515 mm (Figure 3). Mean TL of hybrid trout was 472 mm (± 12.8) and ranged from 312 - 706 mm, and was higher than the Yellowstone Cutthroat Trout which had a mean TL of 363 mm (± 29.2) and ranged in size from to 145-582 mm. Gill net catch rates for

trout were highest for Yellowstone Cutthroat Trout at 9.9 fish/net-night (±2.4), followed by Brook Trout at 2.7 fish/net-night (±0.6), and hybrid trout at 2.5 fish/net-night (±0.6) (Figure 4). Gill net catch rates for all trout species combined were 15.2 fish/per-night (±3.0). The median catch rate of Utah Chub was 30.5 per net-night (Figure 5) and was the highest observed median catch rate to date. Mean gill net catch rate for Chubs was similar to last year and among the higher observed catch rates. Yellowstone Cutthroat Trout gill net catch rate in 2014 was over 50% higher than the 22-year average catch rate (9.9 vs. 6.5 fish/net-night), whereas hybrid trout gill net catch rate was lower than the long-term average (2.5 vs. 3.7 fish/net-night). Brook Trout catch rate (2.7 vs. 2.4 fish/net-night) was similar to the 22-year catch rate average. Historical comparison of Utah Chub length frequencies (2011-2014) indicate several different age classes present in 2014 compared to past years as the strong year class present at 150 mm TL in 2011 slowly moved through the population (Figure 6).

We collected 15 Brook Trout, 60 hybrid trout, 210 Yellowstone Cutthroat Trout, 1,419 Utah Chub in 30 net nights from June to October gill netting effort. Utah Chub had the highest mean gill net catch rate (shown with 95% confidence intervals) of 47.3 (± 17.7) fish/net-night followed by Yellowstone Cutthroat Trout (7.0 fish/net-night;±1.8; Figure 7). Both Brook and hybrid trout were collected in low numbers and were 0.5 (±0.4) and 2.0 (±0.8) fish/net-night, respectively. Based on all gill net effort combined, we observed 60 of 626 (10%) captured Yellowstone Cutthroat Trout were adipose-clipped (Table 1). Similarly, we observed a 10% fin clipped cutthroat ratio in fish that returned to the hatchery in 2014 (651 marked out of 6,226 checked for marks).

We aged 120 Brook Trout, 121 hybrid trout, and 151 Yellowstone Cutthroat Trout. Brook Trout age ranged from 1 to 5 years old. Hybrid trout age ranged from 1 to 7 years old. Yellowstone Cutthroat Trout age ranged from 1 to 4 years old. Mean length at age-2 for trout was lowest for Brook Trout at 339 mm TL (±328-351) and highest for hybrid trout at 459 mm TL (±453-466) (Table 2). Mean length at age two for Yellowstone Cutthroat Trout was intermediate at 403 mm TL (±392-414). Brook Trout had the slowest growth rates with a starting age of t_0 = -1.08 years (± -2.14 to -0.01) toward their asymptotic length of L_∞ = 804 mm (±313-1296) at an instantaneous rate of K = 0.18/year (±-0.03-0.38; Figure 8). Yellowstone Cutthroat Trout grew from a starting age of t_0 = -0.09 years (±-0.42 to 0.24) toward their asymptotic length of L_∞ = 533 mm (±469-597) at an instantaneous rate of K = 0.68/year (±0.37-0.98). Hybrid trout had the fastest growth rates and grew from a starting age of t_0 = -0.57 years (±-0.99 to -0.15) toward their asymptotic length of L_∞ = 705 mm (±650-760) at an instantaneous rate of K = 0.41/year (±0.29-0.53). Annual mortality estimates from catch curve analysis of hybrid trout and Yellowstone Cutthroat Trout from age-2 to age-4 was 89% and 81%, respectively. We were unable to successfully estimate mortality in Brook Trout from age two to four using catch curve analysis. Proportional stock density (PSD) was highest for Brook Trout (95) followed by hybrid trout (93) and Cutthroat Trout (82). Relative stock density (RSD-400) was highest for Brook Trout (64) followed by hybrid trout (49) and Cutthroat Trout (28) (Table 3). Mean relative weight (W_r) for all trout species (all sizes combined) ranged between 91 and 97 (Table 4) and W_r of Yellowstone Cutthroat Trout size classes (0 - 199 mm, 200 – 299 mm, 300 – 399 mm, and >400 mm) ranged between 87 and 92 (Figure 9). We compared the gill net catch of trout per night to trends in trout mean relative weight (W_r) from 2005 to 2014 and generally found that the relative weights of trout increased or decreased inversely 1-2 years after trout abundances increased or decreased (Figure 10), suggesting density dependent growth. The highest relative weights occurred in 2005 and 2006 when trout abundances were the lowest across the 10 years.

Diets

We examined the stomach contents of 114 Brook Trout, 146 hybrid trout, 365 Yellowstone Cutthroat Trout, and 586 Utah Chub from Henrys Lake between May to October (Table 5). The percent of empty stomachs for Brook Trout, hybrid trout, Yellowstone Cutthroat Trout, and Utah Chub was 12%, 8%, 7%, and 35%, respectively. Overall, diet composition (by weight) for Brook Trout over the entire season was dominated by caddisflies (43%) followed by fish (15%), leeches (14%), scuds (11%), mollusks (6%) and *Daphnia* (4%) (Figure 11). Hybrid trout diet was dominated by caddisflies (24%), followed by scuds and fish (19% each), *Daphnia* (14%), leeches (11%), and mollusks (5%). Yellowstone cutthroat trout diet was dominated by *Daphnia* (31%), followed by scuds (21%), caddisflies (18%), chironomids (13%) and fish (9%). Utah Chub diet was dominated by scuds (37%), followed by mollusks (27%), caddisflies (15%), and chironomids (12%).

We also compared overall diet composition (by number) for Brook Trout over the entire season, which was dominated by caddisflies (48%), and followed by scuds (19%), leeches (8%), fish (7%), and *Daphnia* (5%). hybrid trout diet was dominated by *Daphnia* (26%), followed by scuds (24%), caddisflies (22%), fish (8%), chironomids (7%), and leeches (6%). Yellowstone Cutthroat Trout diet was dominated by *Daphnia* (44%), followed by scuds (24%), chironomids (13%), and caddisflies (11%). Utah Chub diet was dominated by chironomids (41%), followed by scuds (19%), mollusks (14%), caddisflies (11%), and *Daphnia* (9%).

Brook Trout diets differed monthly (PERMANOVA, P <0.01), despite the lower number of Brook Trout collected in nets after May (Table 6). In May, diets were dominated by caddisflies and shifted towards more scuds and mollusks (snails) as fall progressed (Figure 12). Fish was a fairly significant component of the diet by weight in May and June at 15% and 20%, respectively. No Brook Trout were collected in July, August, and October gill netting, so diet information during these months is lacking.

Hybrid trout diets varied monthly (PERMANOVA, P <0.01) and was comprised primarily of fish and caddisflies in May (Figure 13). Scuds increased proportionately in the diet through the fall and became the dominant food item in September and October. Caddisflies, though comprising the highest portion of the diet in May, June, and July, were almost nonexistent after the summer. Leeches were important food source in May and July. Fish was an important component of the diet in May (29% by weight), but relatively less important throughout the remaining months. Daphnia were present in the diet in May and then became more significant later in the summer as the caddisfly proportion diminished. Chironomids and mollusks (snails) were primarily in the diets of hybrids during the summer.

Yellowstone Cutthroat Trout diets differed from May to October (PERMANOVA, P <0.01). Yellowstone Cutthroat Trout diet was dominated by *Daphnia* and caddisflies in May, with chironomids comprising an important component of the diet in June (Figure 14). Leeches were present in the diets in May, June, July, and September, albeit at low amounts (<4%). *Daphnia* was an important food source early (May) and later (August – October). Caddisflies were found early in the season and then became less prevalent in the diets in the fall. Fish were observed in the diets of Yellowstone Cutthroat Trout in May and to a lesser extent June through August.

Utah Chub diets were predominantly comprised of mollusks (bivalves and snails) and scuds (Figure 15) from May to October and diets were found to differ monthly (PERMANOVA, P <0.01). Scuds became progressively more preferred food item in the fall. Chironomids were dominant by number throughout all months, yet comprised a relatively insignificant component

by weight. Caddisflies were also found in the diets throughout all the months, but were a particularly important food item in July (38% by weight). *Daphnia* never exceeded 7% in any month and was a minor component of the diet seasonally. Leeches were a scant food source for chubs in June and July.

A high diet overlap existed between Brook Trout and hybrid trout (Schoener index = 0.74) and Yellowstone Cutthroat and hybrid trout (Schoener index = 0.70) in Henrys Lake (Table 7). We found no biologically significant (Schoener index >0.60) diet overlap of Utah Chub with any trout.

Most fish (~60%) found in the trout stomachs in Henrys Lake were too decomposed to correctly identify to the species level. However, we were able to accurately identify 42 fish in trout stomach of which 21 (50%) were sculpin and 21 (50%) were Utah Chub. The average size of fish observed in the trout stomachs were 96 mm TL (95% CI 4.4; range 74-110 mm) for sculpin and 101 mm TL (95% CI 36.4; 34-235 mm) for Utah Chub. The largest Utah Chub (235 mm TL) was found in the stomach of a hybrid trout that was 681 mm TL. The largest Utah Chub collected in the gill nets was 366 mm TL. Based on gape width estimates of 50% (Mittelbach and Persson 1998), a trout would need to exceed 732 mm TL to consume a Utah Chub that was 366 mm TL. The largest trout we collected in gill nets was 706 mm TL hybrid trout. Given the faster growth rates of hybrid trout, approximately 74% of the Utah Chub were susceptible to predation by age-2 hybrids (Figure 16). By age-3, hybrids would be able to readily consume > 90% of the Utah Chub in the population. In contrast, Brook Trout growth was much slower with age-1 entirely gape limited with the exception of age-0 chubs not collected in the gillnets. Larger age-4 Brook Trout would be able to ingest 81% of the Utah Chubs available. Yellowstone Cutthroat Trout were much less gape limited than Brook Trout and by age-2, 53% of the Utah Chubs would be vulnerable to predation.

Historical comparisons of trout diets indicated a general trend away from scuds as the dominant prey in 2003, towards leeches in 2010 and 2011, then caddisflies in 2014 for Brook Trout and hybrid trout, whereas Yellowstone Cutthroat Trout shifted to Daphnia in 2010, 2011, and 2014 (Figure 17, 18, and 19). Utah Chub diets increased in scud and mollusks (e.g. bivalves, gastropods) as the major prey sources in 2014 compared to 2004 when chironomids and caddisflies were the primary prey (Figure 20).

Stable Isotope Analysis

Utah Chub was more enriched with the carbon isotopic signature ($\delta^{13}C$), indicative of more littoral primary carbon sources (e.g. invertebrates), than trout (Table 8). In contrast, Yellowstone Cutthroat Trout was the most depleted in carbon isotopes, reflecting usage of more pelagic habitat (e.g. zooplankton). Brook Trout and hybrid trout became more significantly enriched in carbon with size suggesting a shift to more littoral feeding habitats as they become larger (Table 6; Figure 21). Utah Chub and Yellowstone Cutthroat Trout utilized similar carbon sources across size with no statistical differences between carbon and total length. Utah Chub and Yellowstone Cutthroat Trout exhibited an ontogenetic shift to higher trophic positions with size based on $\delta^{15}N$ signatures. The increase in nitrogen signatures for Utah Chub suggests a shift towards more littoral habitat as they become larger. Yellowstone Cutthroat Trout increase in nitrogen with size suggests a shift towards more piscivory. The most enriched prey in carbon isotopic signature ($\delta^{13}C$) and depleted in nitrogen isotopic signature ($\delta^{15}N$) was caddisflies (Table 9). Sculpin and Utah Chub were both highly enriched in nitrogen isotopic values and were 8.64‰ and 8.51‰, respectively. In contrast, the prey most depleted in carbon isotopic

signatures ($\delta^{13}C$) were chironomids (-27.56‰), zooplankton (-27.11‰), and Utah Chub (-27.04‰).

Brook Trout exhibited the widest dietary niche (SEA_c = 1.77) while Utah Chub contained the narrowest dietary niche (SEA_c = 0.87; Table 8; Figure 22). Hybrid and Yellowstone Cutthroat both displayed moderate dietary niche widths of 1.22 and 1.40, respectively. Niche overlap between ellipses was nonexistent in the SEA_c between Utah Chub and hybrid trout (Table 7; Figure 23). Utah Chub and Yellowstone Cutthroat Trout also exhibited no overlap in SEA_c. Utah Chub only slightly overlapped niches with Brook Trout (0.26), but was not significant. Brook Trout SEA_c also overlapped to some extent with hybrid (0.43) and Yellowstone Cutthroat Trout (0.19). Only the SEA_c of hybrid and Yellowstone Cutthroat Trout showed significant overlap in niches (>0.60) of 0.76.

For the mixing model, the mean isotopic values (± SE) we used for the pooled invertebrates was 21.6‰ (±_1.6) for $\delta^{13}C$ and 6.2‰ (± 0.4) for $\delta^{15}N$ (Figure 24). Pooled fish isotopic values were -24.7‰ (±_1.2) for $\delta^{13}C$ and 8.6‰ (±_0.2) for $\delta^{15}N$ and zooplankton isotopic values were -27.1‰ (± 0.1) and 5.5‰ (±_0.1) for $\delta^{13}C$ and $\delta^{15}N$, respectively. The SIAR mixing model found zooplankton was the primary dietary item for Brook Trout (Mean = 0.58; CI 0.41-0.74), hybrid trout (Mean = 0.70; CI 0.55-0.83), Utah Chub (Mean = 0.61; CI 052-0.70), and Yellowstone Cutthroat Trout (Mean = 0.71; CI 0.58-0.84; Figure 25). Fish represented a minor component of the diet for the trout and overlapped to some extent with invertebrates in the mixing model. The mean proportion of fish in the diet was 0.10 (CI 0.00-0.24), 0.10 (CI 0.00-0.21), and 0.11 (CI 0.00-0.22) for Brook Trout, hybrid trout, and Yellowstone Cutthroat Trout, respectively. For Brook Trout, hybrid trout, Utah Chub, and Yellowstone Cutthroat Trout invertebrates comprised a mean proportion in the diet of 0.32 (CI 0.18-0.47), 0.21 (CI 0.07-0.35), 0.39 (0.30-0.48), and 0.18 (0.05-0.31), respectively.

Water Quality

Between January 8, 2013 and February 23, 2014, total dissolved oxygen diminished from 32.0 g/m^2 to 30.3 g/m^2 at the Pittsburgh Creek site and from 37.4 g/m^2 to 34.4 g/m^2 at the Wild Rose site. Between December 5, 2013 and February 23, 2014 total dissolved oxygen diminished from 35.3 g/m^2 to 24.8 g/m^2 at the County dock site, and between December 5, 2013 and January 28, 2014 levels fell from 38.84 g/m^2 to 27.5 g/m^2 at the Hatchery site (Table 10). Depletion estimates predicted dissolved oxygen would remain above the level of concern throughout the winter (Figure 26). Based on predictions of dissolved oxygen depletion rates, aeration would not have been necessary. However, Henrys Lake had a larger than average drawdown during the summer of 2014 combined with a high biomass of fish. Therefore, aeration was deployed throughout the winter to mitigate potential impacts.

DISCUSSION

Overall, the gill net catch rate of trout in 2014 decreased from that observed in 2013. In 2012, we reduced stocking rates of Yellowstone Cutthroat Trout (~500,000 per year) in an effort to stabilize the trout population near the target management goal of 11 trout/net-night and improve growth and size of the trout. For the last five years, Yellowstone Cutthroat Trout gill net catch rates have remained stable and exceeded the long-term average despite lower stocking rates. The wild component of Yellowstone Cutthroat Trout from natural reproduction in the primary tributaries such as Duck Creek, Howard Creek, and Targhee Creek influences their abundance in the lake. From 2009 to 2013, we observed a much lower percentage of clipped

Yellowstone Cutthroat (average 5%, range 3-8%), which suggests increased recruitment of wild trout in the lake. In 2014, 10% of the Yellowstone Cutthroat Trout returning to the hatchery and collected in the nets were clipped; indicating limited wild production in the current population. Stocking conditions in the lake during the fall (e.g. temperature) along with fish size at stocking often influence the subsequent fish survival. Favorable stocking conditions and/or increased size at stocking in the last few years may have contributed to higher survival of hatchery Yellowstone Cutthroat Trout offsetting any expected decreases in abundance from reduced stocking numbers and limited wild recruitment. Similar to 2013, the hybrid trout gill net catch was less than the long-term average with few age-1 fish in the samples chronicling the discrepancy of likely misidentifying between younger/smaller hybrid trout and Yellowstone Cutthroat Trout. Brook Trout densities for the past three years have precipitously decreased and are currently at the long-term average. Brook Trout were stocked at higher rates in 2008 and 2009 (198,000 and 171,000, respectively) when compared to the last five years (range 83,000-110,000, target goal of 100,000). The peak density of Brook Trout per net-night observed in 2012 and subsequent decrease is likely a result of the Brook Trout cohorts from 2008 and 2009 moving through the population. Natural production of Brook Trout from tributaries (e.g., Targhee Creek) is known to occur in Henrys Lake. However, we currently have limited information on Brook Trout natural production rates and how this component may influence overall densities. Relative weights of trout improved over weights from last year, but all size classes of trout were still below 100, with the exception of >400 mm TL Brook Trout, suggesting food availability may be limiting trout (Flickinger and Bulow 1993). Food availability constraints are likely due to intraspecific and interspecific competition among trout species as trout densities remain high (>15 per net-night as opposed to the target of 11 per net-night). In years with lower trout densities, trout condition improves in Henrys Lake. For example, in 2005 and 2006 trout densities were approximately 10 trout per night and relative weights for all trout exceeded 100. For the past three years stocking rates have been reduced by ~500,000 to achieve lower trout densities, and we expect trout conditions to continue to improve as densities decrease and food resources increase as result of this management action.

Median catch rates for Utah Chub continue to increase and were the highest reported in our gill net catch. Catch rates of Utah Chubs first began to increase in the late 1990s, following initial documentation of their presence in 1993. We have limited inferences on Utah Chub densities due to high variability in the catch (i.e. high variance) and insufficient gill net samples necessary to detect changes based on power analysis. This analysis recommends netting efforts in excess of 250 net nights of effort, which is unachievable with current funding and manpower constraints. However, we continue to be concerned that the population of Utah Chub may be increasing, which merits continued monitoring.

Growth provides one of the most important measurable life history metrics for individual fish and populations (Ricker 1975). As such, growth rates offer a reliable measure with insights into life histories, lifespans, and mortality rates of fish populations that may be influenced by exploitation. We expressed trout growth with a commonly used model, the von Bertalanffy growth function or VBGF (von Bertalanffy 1957), to describe the total length of an individual at any given age. Growth is modeled with the VBGF using a three-parameter logarithmic function of theoretical asymptotic maximum length (L_∞), growth coefficient (k; in year^{-1}), and length-at-age zero (L_0). We found the VBGF model had poor fit for brook trout data. We collected only one older brook trout greater than age-4. With few older brook trout represented, restricting the model to a non-linear relationship was difficult because the fit was more of linear relationship given the limited age ranges (ages 1-4). We found reasonable fit for the Yellowstone Cutthroat Trout and hybrid trout. However, larger fish may be underrepresented in sampling due to difficulties in capturing larger old fish (e.g. gear selectivity) and high mortality rates, which

results in underestimated L_∞. It is worth noting that in interpretation of VBGF, L_∞ is the maximum mean length and not the maximum length of an individual fish; thus an individual fish can exceed the estimated L_∞, as has been observed by biologists and anglers in Henrys Lake in the past. Further refinements of the VBGF model are still required to determine whether growth estimates of trout in Henrys Lake can provide consistent results over time. However, the VBGF appears to provide an additional index of monitoring trout growth in Henrys Lake that can provide insight into fish population dynamics.

We examined feeding niches of trout and Utah Chub utilizing a rather novel technique of stable isotope analysis. Overall Utah Chub had the narrowest trophic niche, which suggested a more specialized diet. The lower dietary plasticity and trophic position of Utah Chub when compared to trout indicates competitive interactions from a foraging perspective may be minimal. Based on the gut content analysis, we found Utah Chub utilized gastropods and bivalves much more frequently than trout and may account for their more specialized diets and lower trophic position. When we did find gastropods in the diet of trout they were often still whole. Gastropods and bivalves found in the Utah Chub stomachs were often masticated indicating they may have a morphological ability to crush the shells of mollusks more readily and extract the assimilated energy within the shell. We found no biologically-significant diet overlap between trout and Utah Chubs based on diets and stable isotopes analysis which further suggests trout performance (e.g. growth, condition) was minimally impacted from shared food resources between the species. There was limited overlap of isotopic niches between Brook Trout and Utah Chub and some interactions appear to occur. Brook Trout niches appeared to have wide breadth of diet preferences and likely rely on a diversity of prey. As expected the niche widths of two very similar species, Yellowstone Cutthroat Trout and hybrids, overlapped extensively based on the SIBER and Schoener's index. A similar study was conducted utilizing stable isotope analysis and diets to examine competition for food resources between trout (Bear Lake Cutthroat Trout, Tiger Trout, Rainbow Trout) and Utah Chubs in Scofield Reservoir, Utah (Winters 2014). Winters (2014) found significant overlap in diets of Utah Chubs (Schoener's index >60) with small Yellowstone Cutthroat Trout and small Rainbow Trout (<350 mm). Only small cutthroat trout occupied similar feeding niches with Utah Chubs based on stable isotope analysis. In Scofield Reservoir, the larger trout shifted to a more piscivorous diet comprised mainly of Utah Chubs with a resultant increase in trophic position and shift in niche space. Although competition for food resources between Utah Chub and trout does not appear to be occurring in Henrys Lake, a shift in prey assemblages both spatially and temporally could cause an alteration in predator-prey dynamics and indirectly induce competition for preferred prey. For example, if Utah Chub production is driven primarily from mollusk consumption, and a reduction in mollusk availability occurred, then Utah Chub foraging patterns may shift to higher consumption of scuds – a prey item preferred by trout. Also, if Utah Chub populations increased and the trout population increased via stocking or natural recruitment, then competition for a more limited prey base may become more pronounced among the species. Thus, the interaction of Utah Chubs and trout should be monitored periodically to ensure predator-prey dynamics are stable and the fishery continues to produce trophy-sized trout.

In an attempt to control undesirable species, highly piscivorous species are frequently introduced as a management tool (Courtney and Kohler 1986). Often the addition of these piscivores improves or creates novel angling opportunities. Given the higher numbers of Utah Chub observed in the gill net catch, stocking different predators in an effort to control the Utah Chub population may be an option if deemed effective and appropriate by fishery managers and anglers. In Scofield Reservoir, Utah, Bear Lake Cutthroat Trout were the most effective predator in consuming Utah Chub, when compared to Tiger Trout and Rainbow Trout (Winters 2014). Stocking a non-native subspecies of Cutthroat Trout like the Bear Lake subspeciesa within a

native Yellowstone Cutthroat Trout lake would allow for the possible risk of hybridization among the subspecies, resulting in a dilution of the genetics of the lake. Henrys Lake is currently managed to protect and ensure a viable population of native Yellowstone Cutthroat Trout. Thus, stocking a different cutthroat subspecies would be unacceptable despite any potential added biological control advantage unless all fish stocked were sterile. Tiger Trout are a sterile (3N) hybrid species and have been used as a biological control in other systems. However, there is still limited information regarding their effectiveness as a biological control. In Washington, Tiger Trout preferred *Daphnia*, but did exhibit some piscivory seasonally when *Daphnia* abundances decreased (Miller 2010). In Scofield reservoir, the diet of Tiger Trout was comprised of Utah Chub ~30% seasonally (Winters 2014). We currently stock Brook Trout which is the male cross in creating a Tiger Trout. Brook Trout have been known to be highly piscivorous as they become larger (>250 mm TL; Browne and Rasmussen 2009).Surprisingly in Henrys Lake, Brook Trout exhibit limited piscivory with only approximately 20% of their diet in 2014 being comprised of fish. Tiger Trout foraging behavior would be expected to be similar to Brook Trout in the lake and likely limited effectiveness as a biological control for Utah Chubs. Another consideration in Henrys Lake with biological controls would be the rate of predation on Utah Chubs compared to sculpin. Schoby et al. (2014) evaluated trout diets at Henrys Lake in 2010 and found mostly unidentifiable fish in the stomachs. Of the 15 identifiable fish, only one was a Utah Chub and the others were all sculpin. In our study, identifiable fish in the stomachs were comprised of 50% Utah Chubs and 50% sculpin. Regardless of potential biological controls or based on the current trout population, the ability to completely eradicate Utah Chubs utilizing a top down approach appears unachievable because a small proportion of larger mature adult Utah Chub would exceed the gape limitations of the largest trout. Further, interactions between Chub and trout at current levels are not impacting the trout population, so biological controls are not warranted at this time.

Trout have shifted from scuds as the primary prey resource in the early 2000s towards other invertebrates and zooplankton in the more recent diet analysis. The dietary shift away from scuds as the dominant prey is likely the result of reduced scud abundances. However, little is known regarding the historical or current invertebrate community composition and abundance other than ancillary information from anglers and biologists familiar with the lake. Scuds contain a higher caloric content (~3,300 J/g) when compared to other benthic invertebrates (~3,000 J/g) and represent a more energetically profitable prey (Flinders 2012). In 2003 when scuds were the dominant prey, trout abundances were also the lowest observed in the long-term gill net monitoring, suggesting trout could choose their most preferred food resources. As trout abundances have increased, predation on scuds may have reduced the scud population. Environmental conditions (e.g. temperature) may also play a role in regulating scud abundance. A decrease in trout abundances suggests scuds should become more prolific and prevalent in the invertebrate community, if predation pressure by trout is indeed regulating the scud population. Utah Chubs may also be a factor regulating scud abundances through direct predation. Zooplankton contributed a significant component of the diet yearly in Cutthroat Trout (>40%) and to a lesser extent in hybrids (26%). We found limited zooplankton in Brook Trout and Utah Chub diets (<10%). In contrast, the mixing model estimated zooplankton was the most important prey source in the diet for all species with a mean contribution greater than 50%. One of the major benefits with stable isotope analysis is that it provides time-integrated assimilated dietary information when compared to the traditional temporally limited “snapshot” gut content analysis method for dietary studies. Stable isotope analysis may be more effective in detecting the importance and incorporation of zooplankton into diets owing to their small size and high surface to volume ratios which facilitates a quick digestion, evacuation, and assimilation (Hyslop 1980). However, the mixing model results may be viewed cautiously for Utah Chubs. We were unable to collect gastropods and bivalves for analysis in the isotopic mixing model analysis, and

they are an important prey item for Utah Chubs. Depending on the isotopic signatures of mollusks, the output from the mixing model could have changed as it source partitions, which would likely have further separated Chub from trout. Zooplankton is known to be important prey for Utah Chubs in other systems. In Flaming Gorge, Utah-Wyoming, the decline in kokanee growth was correlated with increased Utah Chub densities with both species feeding extensively on zooplankton resulting in a reduction in zooplankton biomass (Teuscher and Luecke 1996). In Scofield Reservoir, Utah, the diets of Utah Chub were comprised of zooplankton from 20 to 100% depending on the season and size of fish. However, in Henrys Lake zooplankton comprises only a minimal component of the diet (<10%) and may be a less important prey for Utah Chubs when compared to other western reservoirs.

The ratio of marked to unmarked Yellowstone Cutthroat Trout collected in gill net surveys (10%) and in the spawning operation (10%) suggests that natural reproduction is currently contributing minimally to the Henrys Lake fishery. Past years have found significant natural reproduction/recruitment likely due to the many tributary stream habitat improvement projects that have occurred over the last decade (e.g. riparian fencing, instream passage improvements, fish irrigation screening) (High et al. 2014). We currently only mark 10% of the stocked Yellowstone Cutthroat Trout with clipped adipose fins to evaluate natural recruitment. However, there is likely an unknown error associated with this ratio estimate. Developing a cost effective method that would allow for mass marking all Yellowstone Cutthroat Trout and hybrid trout we stock would remove any error associated with the ratio estimate and provide a more accurate assessment of the annual natural tributary production. This is particularly important as we continue to improve spawning habitat along the lake and tributaries, which should result in higher contributions of naturally spawned Yellowstone Cutthroat Trout in future years.

Water quality and winter dissolved oxygen profiles have been monitored for over two decades, with aeration deployed when indicators point towards potential lethal conditions. Aeration was implemented preemptively this past winter due to concerns for low lake levels (resulting in less stored oxygen) coupled with high biomass of fish and plants. Monitoring indicated that oxygen levels remained high throughout the winter, which can in part be attributed to warm winter temperatures that caused substantial melting of low elevation snowpack. The subsequent runoff likely provided supplemental oxygen that helped keep lakewide oxygen levels high. The current oxygen depletion model has proven useful in predicting hypoxic conditions that could result in winterkill. Given the success of this model, the public and other vested interests can be given notice of the likelihood of such events and mitigation measures can be implemented at the earliest possible time. Aeration is the primary mitigation measure that we have available. The current aeration system was installed in 1993 and has been maintained since that time. Given that maintenance costs may be substantial in the coming years and more efficient equipment may now be available, a thorough evaluation of the effectiveness of the current system should be undertaken. It is recommended that a winter evaluation should be completed as soon as practical.

RECOMMENDATIONS

1. Continue annual gill net samples at 50 net nights of effort.
2. Collect otolith samples from all trout species; use for cohort analysis and estimates of mortality/year class strength and compare to previous years.
3. Continue to monitor Utah Chub densities and evaluate potential impacts to trout with increased densities of chubs.

4. Periodically conduct diet and stable isotope analysis of trout and Utah Chubs to evaluate dietary and isotopic overlap between the species
5. Collect fin rays from Utah Chubs for aging and mortality estimates.
6. Evaluate the effectiveness and necessity of the current aeration system and cost/benefit of replacing new equipment to reduced long-term costs.

Table 1. 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%

[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

Table 2. Mean length at age data based on otoliths of trout caught with gill nets in Henrys Lake, Idaho 2014. Ages were estimated using non-linear regression.

		Age						
Species		**1**	**2**	**3**	**4**	**5**	**6**	**7**
Brook Trout	Mean TL (mm)	248	339	415	478	531	--	--
	Lower 95% CI	226	328	406	465	501	--	--
	Upper 95% CI	267	351	424	490	556	--	--
	No. Analyzed	13	24	52	30	1	--	--
Cutthroat Trout	Mean TL (mm)	278	403	467	499	--	--	--
	Lower 95% CI	267	392	453	472	--	--	--
	Upper 95% CI	288	414	480	528	--	--	--
	No. Analyzed	65	58	23	5	--	--	--
Hybrid Trout	Mean TL (mm)	335	459	542	597	633	657	674
	Lower 95% CI	314	453	532	582	614	633	644
	Upper 95% CI	356	466	551	610	651	682	705
	No. Analyzed	9	87	19	1	1	1	3

Table 3. Stock density indices (PSD, RSD-400, and RSD-500) and relative weights (W_r) with 95% confidence intervals in parenthesis for all trout species collected with gill nets in Henrys Lake, Idaho 2014.

	Brook Trout	**Hybrid trout**	**Yellowstone Cutthroat Trout**
PSD	95	93	82
RSD-400	64	49	28
RSD-500	3	25	2
W_r			
<200 mm	80 (13.1)	--	87
200 – 299 mm	86 (3.2)	--	90 (1.1)
300 – 399 mm	93 (6.9)	95 (6.3)	92 (1.2)
>399 mm	101 (2.1)	97 (1.4)	89 (1.2)

Table 4. 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 in the spring gillnetting at Henrys Lake, 2014.

	BKT			**HYB**			**YCT**			**UTC**	
Summary statistic	**TL (mm)**	**WT (g)**	**W_r**	**TL (mm)**	**WT (g)**	**W_r**	**TL (mm)**	**WT (g)**	**W_r**	**TL (mm)**	**WT (g)**
Mean	395	846	97	472	1,289	97	363	569	90	210	132
Confidence level (95.0%)	14.7	80.1	2.3	12.8	113.3	1.3	7.3	29.3	0.7	3.2	4.2
Median	426	924	97	460	1,139	97	375	562	91	206	109
Minimum	180	50	50	312	301	67	145	75	49	113	28
Maximum	515	1,747	143	706	3,783	124	944	1,968	130	2,336	732
Count	130	130	128	122	122	121	476	476	468	1,767	1,761

Table 5. The number of stomach samples analyzed that contained prey (full) and no prey (empty) along with percent empty stomachs for Brook Trout (BKT), hybrid trout (HYB), Yellowstone Cutthroat Trout (YCT), and Utah Chubs (UTC) collected monthly in gill nets at Henrys Lake, 2014.

	BKT			**HYB**			**YCT**			**UTC**		
Month	**Full (*n*)**	**Empty (*n*)**	**Empty (%)**	**Full (*n*)**	**Empty (*n*)**	**Empty (%)**	**Full (*n*)**	**Empty (*n*)**	**Empty (%)**	**Full (*n*)**	**Empty (*n*)**	**Empty (%)**
May	89	10	10	86	2	2	148	19	11	16	138	90
June	8	0	0	6	0	0	47	0	0	60	13	18
July	1	0	0	11	0	0	45	1	2	76	14	16
August	0	0	0	5	9	0	26	4	0	86	5	0
September	2	4	33	10	0	0	26	2	4	87	16	16
October	0	0	0	17	0	0	46	1	0	58	17	0
TOTAL	100	14	12	135	11	8	338	27	7	383	203	35

Table 6. Linear regression results for stable isotope analysis (SIA) of $\delta^{13}C$ and $\delta^{15}N$ by total length for Brook Trout (BKT), hybrid trout (HYB), Utah Chub (UTC), and Yellowstone Cutthroat Trout (YCT) and PERMANOVA results for gut content analysis (GCA) across seasons by species in Henrys Lake, 2014.

	$\delta^{13}C$				**$\delta^{15}N$**				**GCA**		
Species	***R^2***	**df**	**F**	***P***	***R^2***	**df**	**F**	***P***	**df**	**F**	***P***
BKT	0.47	1, 7	6.26	0.041	0.28	1, 7	2.71	0.144	2, 88	3.24	<0.01
HYB	0.49	1, 7	6.77	0.035	0.05	1, 7	0.36	0.569	5, 124	6.53	<0.01
UTC	0.10	1, 18	0.19	0.669	0.25	1, 18	5.89	0.026	5, 333	19.62	<0.01
YCT	0.005	1, 10	0.05	0.824	0.60	1, 10	15.06	0.003	5, 323	13.21	<0.01

Table 7. Comparison of SIBER and Schoener overlap fraction index for Brook Trout, hybrid trout, Utah Chub, and Yellowstone Cutthroat Trout in Henrys Lake, 2014. Bolded values indicate a significant diet overlap (> 0.60) for SIBER or Schoener index.

Species		**SIBER**	**Schoener**
Brook Trout	Cutthroat Trout	0.19	0.51
Brook Trout	Hybrid trout	0.43	0.74
Brook Trout	Utah Chub	0.26	0.41
Hybrid trout	Utah Chub	0.00	0.52
Hybrid trout	Cutthroat Trout	0.76	0.70
Utah Chub	Cutthroat Trout	0.00	0.55

Table 8. Sample size (n), $\delta^{13}C$ and $\delta^{15}N$ (mean ± SD), total length (mean ± SD; minimum and maximum), and standard ellipse area corrected for small sample size (SEA_c) of Brook Trout, Cutthroat Trout, hybrid trout, and Utah Chub collected in Henrys Lake, 2014.

Species	**n**	**$\delta^{13}C$ (‰)**	**$\delta^{15}N$ (‰)**	**Total length (mm)**	**SEA_c**
Brook Trout	9	-24.26 ± 0.9	9.03 ± 0.6	379 ± 44 (313 - 451)	1.77
Hybrid trout	9	-25.50 ± 1.1	9.02 ± 0.4	454 ± 54 (375 – 527)	1.22
Utah Chub	20	-23.49 ± 0.6	8.56 ± 0.5	229 ±59 (135 – 324)	0.87
Cutthroat Trout	12	-25.88 ± 1.3	9.20 ± 0.3	345 ± 111 (172 – 513)	1.40

Table 9. Comparison of stable isotope ratios (mean ± SD) and total length (mean ± SD; minimum and maximum) of prey samples collected in Henrys Lake, 2014.

Common name	**Order/Family/Genus**	**n**	**$\delta^{13}C$ (‰)**	**$\delta^{15}N$ (‰)**	**Total length (mm)**
Midges	Chironomidae	3	-27.56 ± 1.2	6.12 ± 0.2	
Sculpin	*Cottus*	3	-22.29 ± 1.6	8.64 ± 0.4	103 ± 10.6 (95 – 115)
Zooplankton	*Daphnia*	3	-27.11 ± 0.1	5.53 ± 0.2	
Scuds	*Gammarus*	3	-21.93 ± 0.9	6.07 ± 0.7	
Utah Chub	*Gila*	3	-27.04 ± 0.8	8.51 ± 0.5	152 ± 11.1 (140 – 162)
Leeches	Hirudinea	3	-23.77 ± 1.3	7.89 ± 0.6	
Damselflies	Odonata	3	-24.03 ± 1.0	8.22 ± 0.1	
Caddisflies	Trichoptera	3	-13.05 ± 0.4	4.84 ± 1.0	

Table 10. Dissolved oxygen (DO) (mg/l) levels recorded in Henrys Lake, Idaho winter monitoring 2013-2014.

Location	Date	Snow depth (cm)	Ice thickness (cm)	DO Ice bottom	DO 1 meters	DO 2 meters	DO 3 meters	Total g/m^2
Pittsburg Creek	Jan 8, 2014	8	22	11.4	10.7	9.0	6.1	32.0
	Jan 28, 2014	6	24	11.7	11.5	9.7	8.8	38.5
	Feb 23, 2014	8	33	9	9	7.6	7.1	30.3
County Boat Ramp	Dec 5, 2013	0	5	12.7	12.6	12.5	10.1	35.3
	Jan 3, 2014	8	15	13.3	13.2	12	8.1	33.4
	Jan 28, 2014	7	22	10.5	9.8	8.8	7.2	26.2
	Feb 23, 2014	7	33	9.3	9.3	8.3	7.2	24.8
Wild Rose	Jan 8, 2014	5	22	12.1	11.8	10.7	8.9	37.4
	Jan 28, 2014	10	20	11	10.7	9.8	8.5	35.9
	Feb 23, 2014	8	36	10.5	10.5	9.1	8.1	34.4
Hatchery	Dec 5, 2013	2	5	13.9	13.4	13	12.1	38.8
	Jan 3, 2014	10	12	9.3	8.6	8.3	6.4	23.7
	Jan 28, 2014	6	20	9.2	8.8	8.9	9.6	27.5

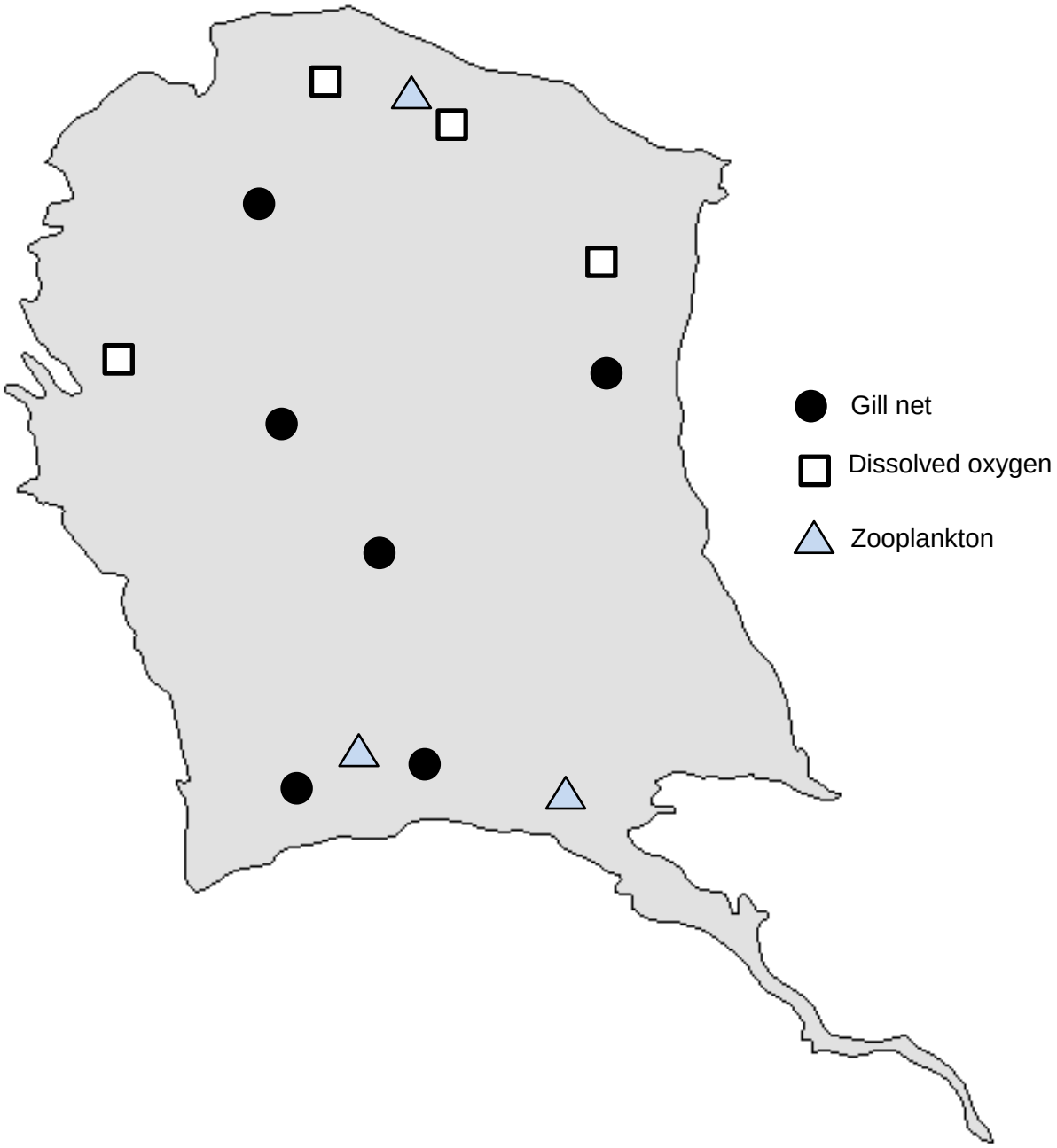

Figure 1. Spatial distribution of gill net, dissolved oxygen, and zooplankton monitoring sites in Henrys Lake, Idaho, 2014.

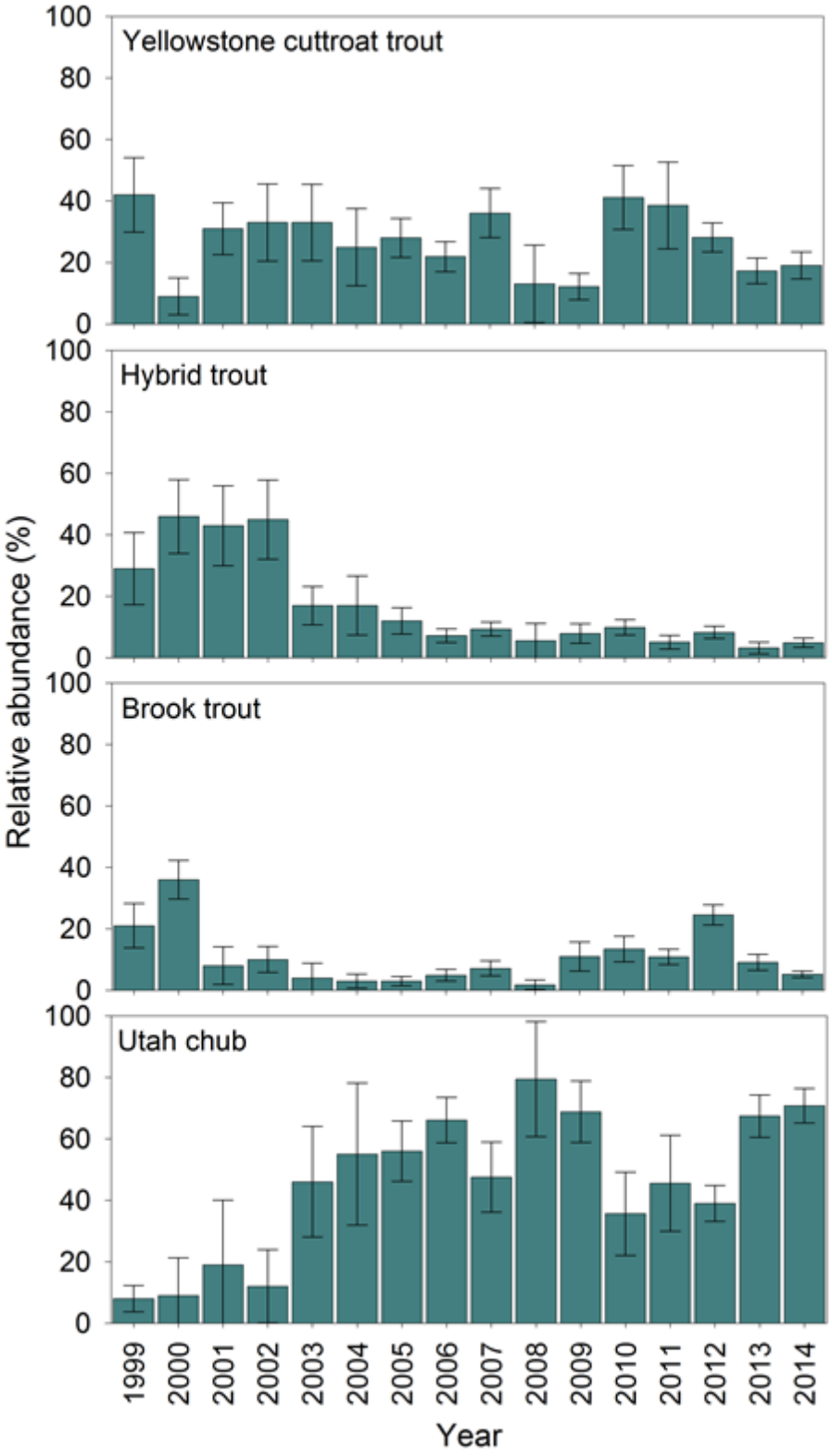
Yellowstone cuttroat trout
Hybrid trout
Brook trout
Utah chub
Relative abundance (%)
0
20
40
60
80
100
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
2010
2011
2012
2013
2014
Year

Figure 2. Relative abundance (percent composition of catch) of Yellowstone Cutthroat Trout, hybrid trout, Brook Trout, and Utah Chub caught in gill nets in Henrys Lake, Idaho between 1999 and 2014. Error bars represent 90% confidence intervals.

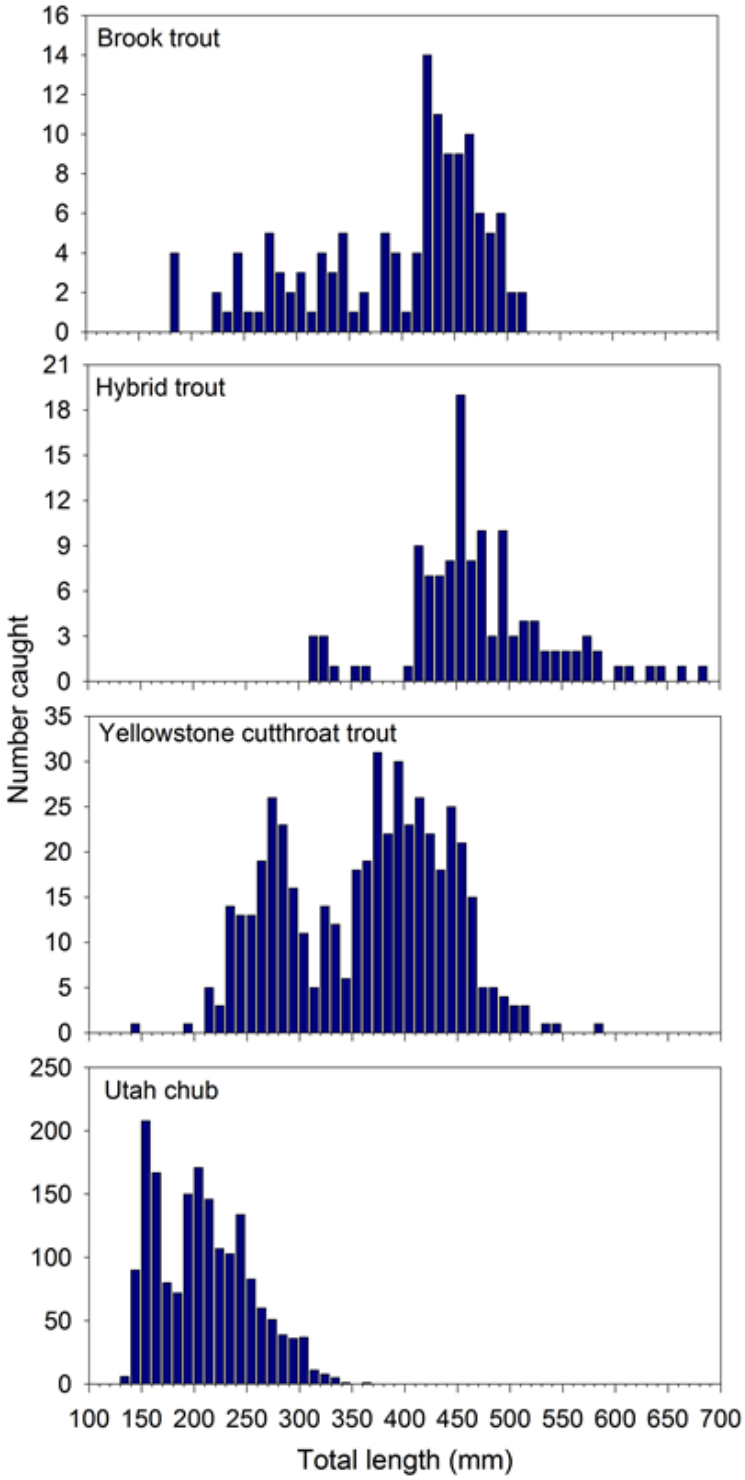

Figure 3. Brook Trout, hybrid trout, Yellowstone Cutthroat Trout, and Utah Chub length frequency distribution from gill nets set in Henrys Lake, Idaho, 2014.

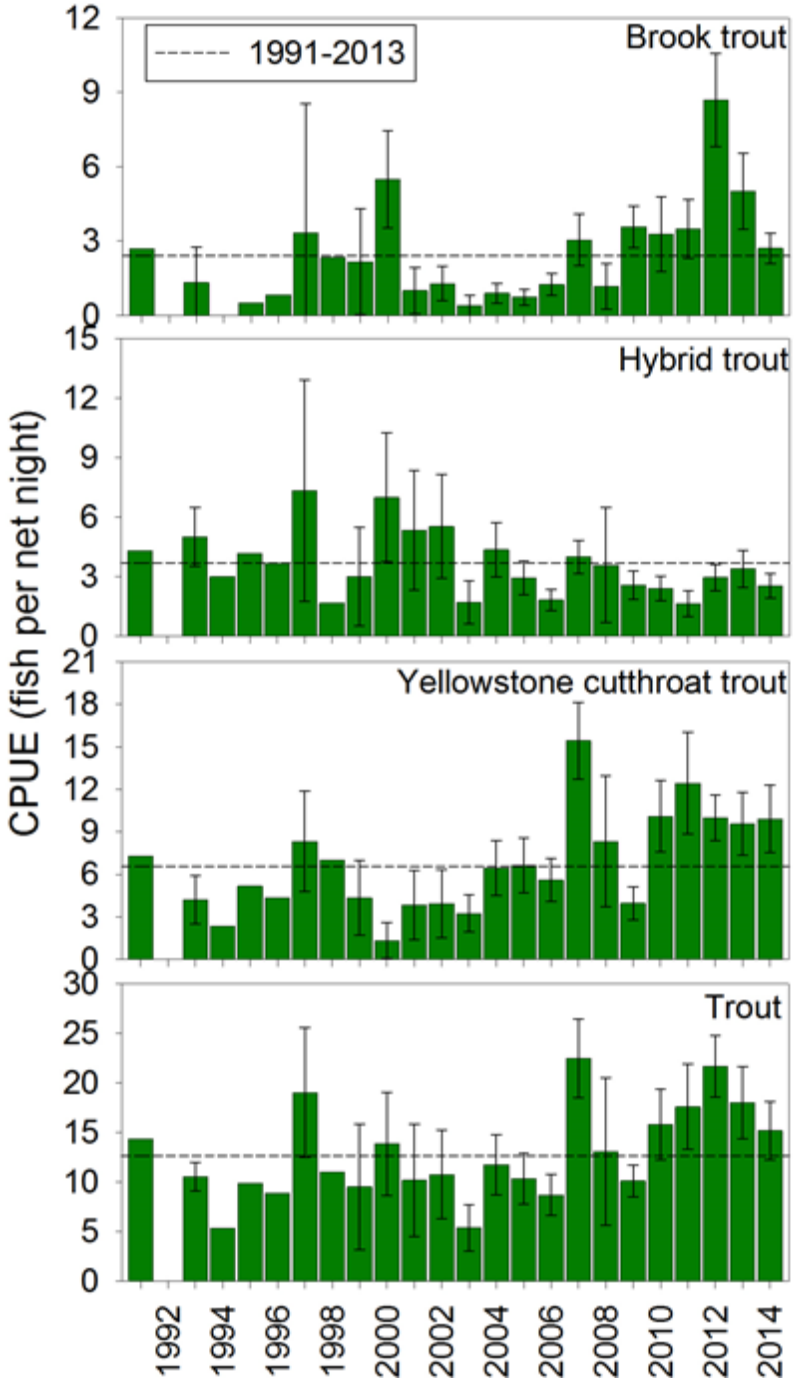

Figure 4. Catch per unit effort (CPUE) of per net-night fish/net-night for Brook Trout, hybrid trout, Yellowstone Cutthroat Trout, and all trout combined (brook, hybrid, cutthroat) in Henrys Lake, Idaho between 1991 and 2014. Error bars represent 95% confidence intervals. The dashed line represents the average gill netting CPUE from years 1991 to 2013.

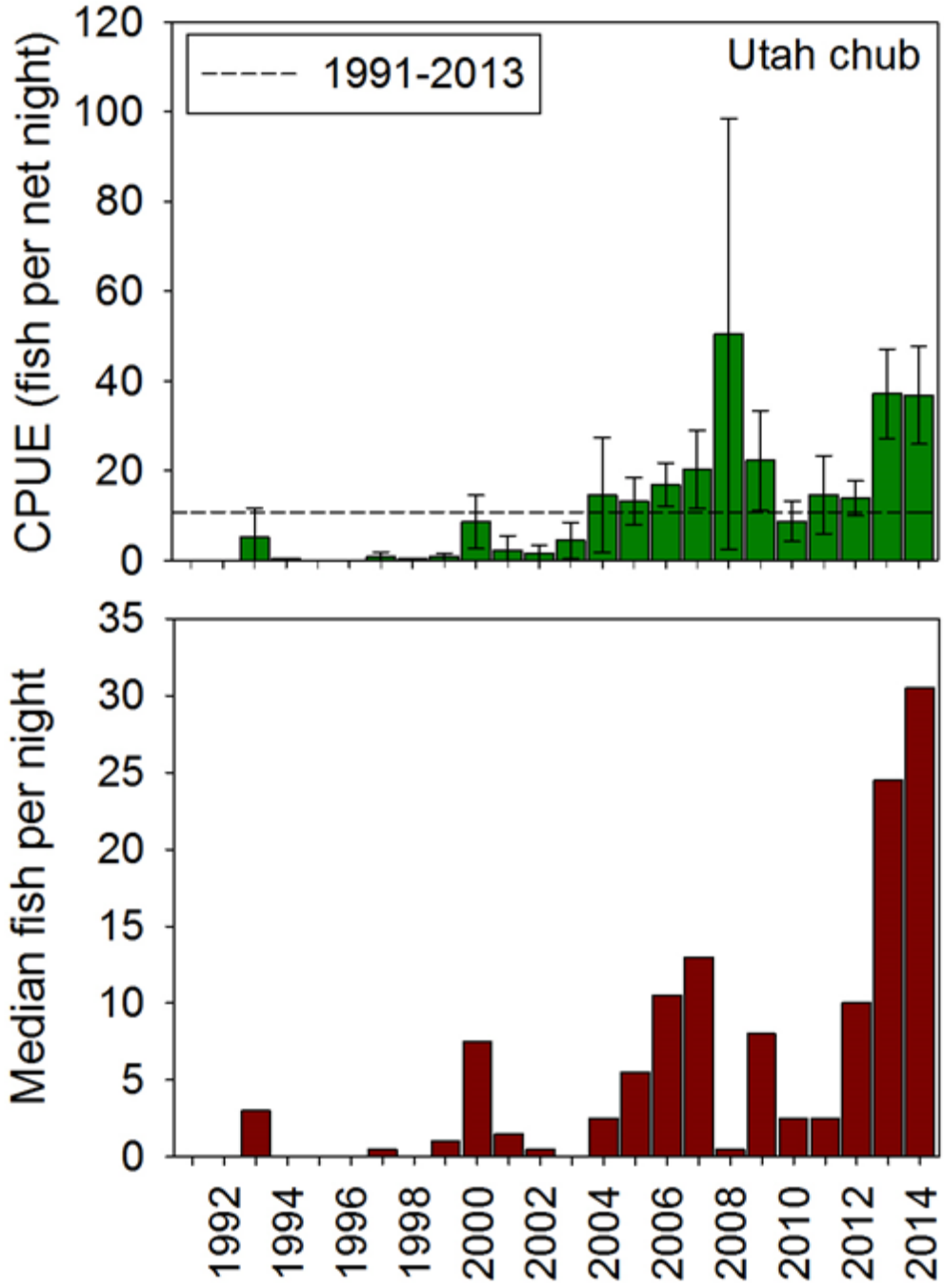

Figure 5. Catch per unit effort (CPUE) and median fish per net-night for Utah Chub in Henrys Lake, Idaho between 1991 and 2014. For the CPUE graph error bars represent 95% confidence intervals and the dashed line represents the average gill netting CPUE from years 1991 to 2014.

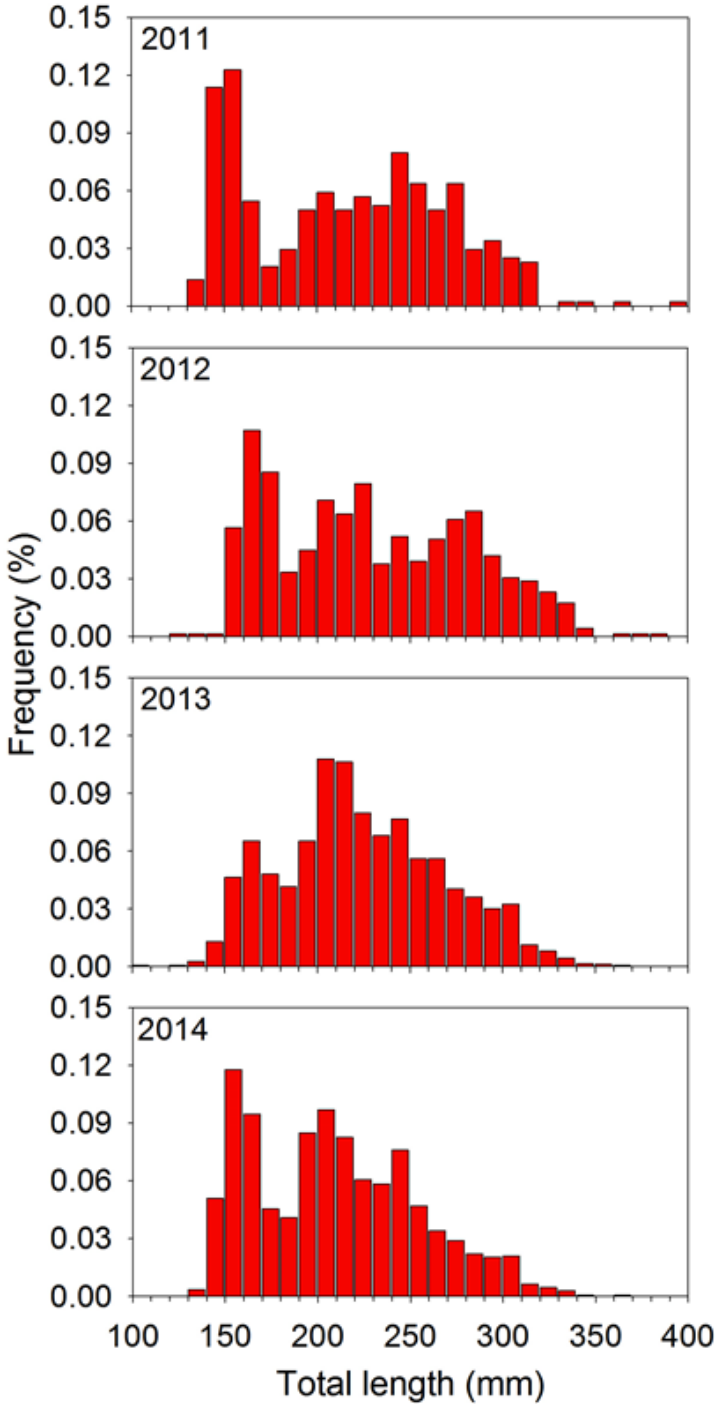

Figure 6. Utah Chub length frequency (%) from gill nets set in Henrys Lake, 2011-2014.

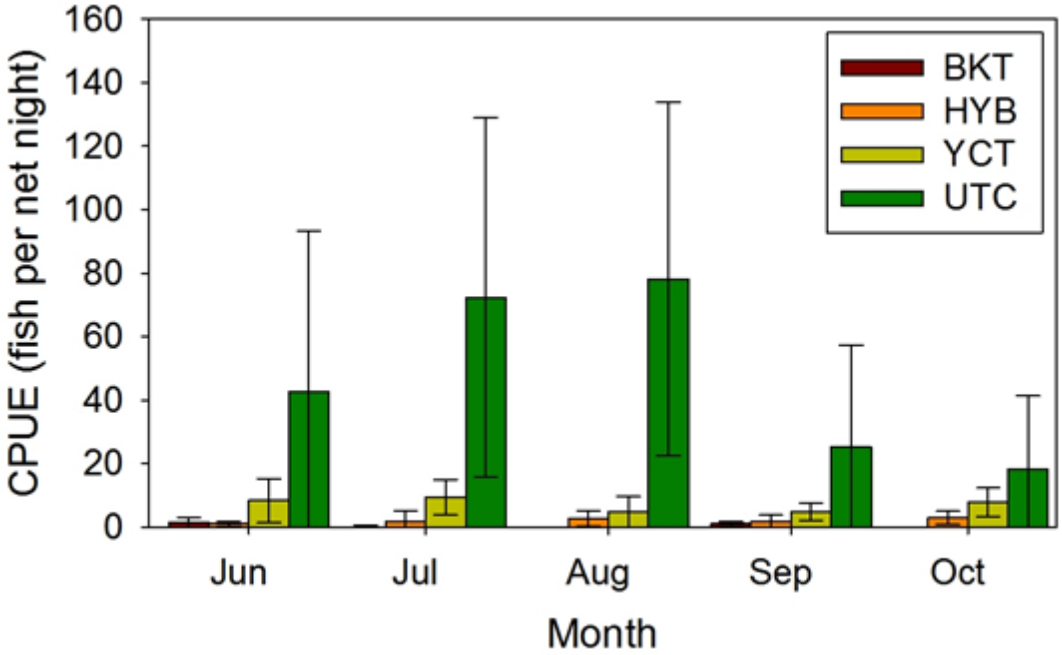

Figure 7. Catch per unit effort (CPUE) of fish per net-night for Brook Trout (BKT), hybrid trout (HYB), Yellowstone Cutthroat Trout (YCT), and Utah Chub (UTC) from gill nets in June to October Henrys Lake, Idaho 2014. Error bars represent 95% confidence intervals.

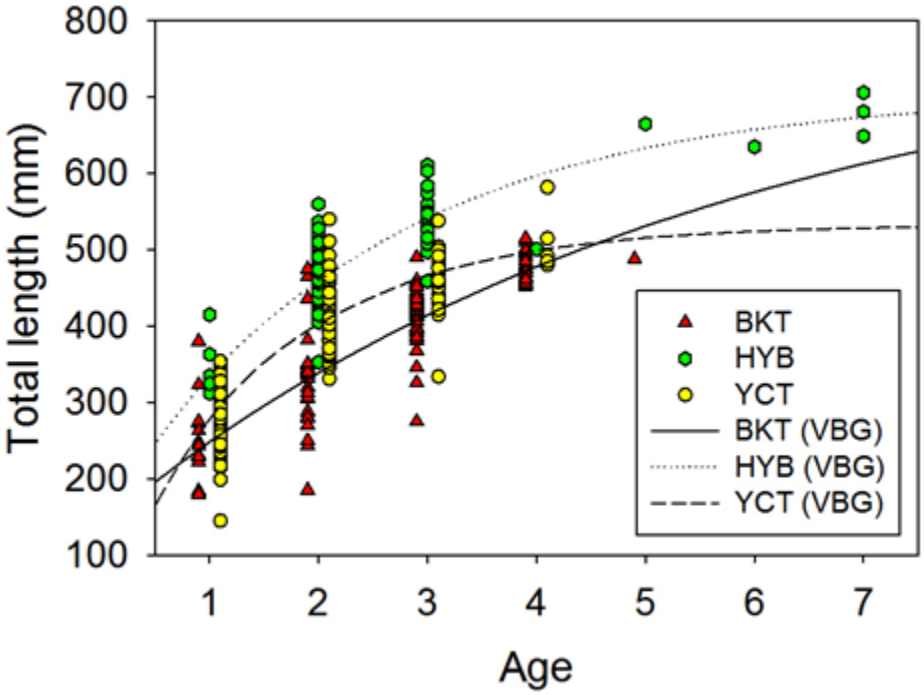

Figure 8. Length-at-age based on non-linear regression for Brook Trout (BKT), hybrid trout (HYB), and Yellowstone Cutthroat Trout from spring gill netting in Henrys Lake, 2014. Growth is described by the fitted von Bertalanffy growth (VBG) curves fitted for each species.

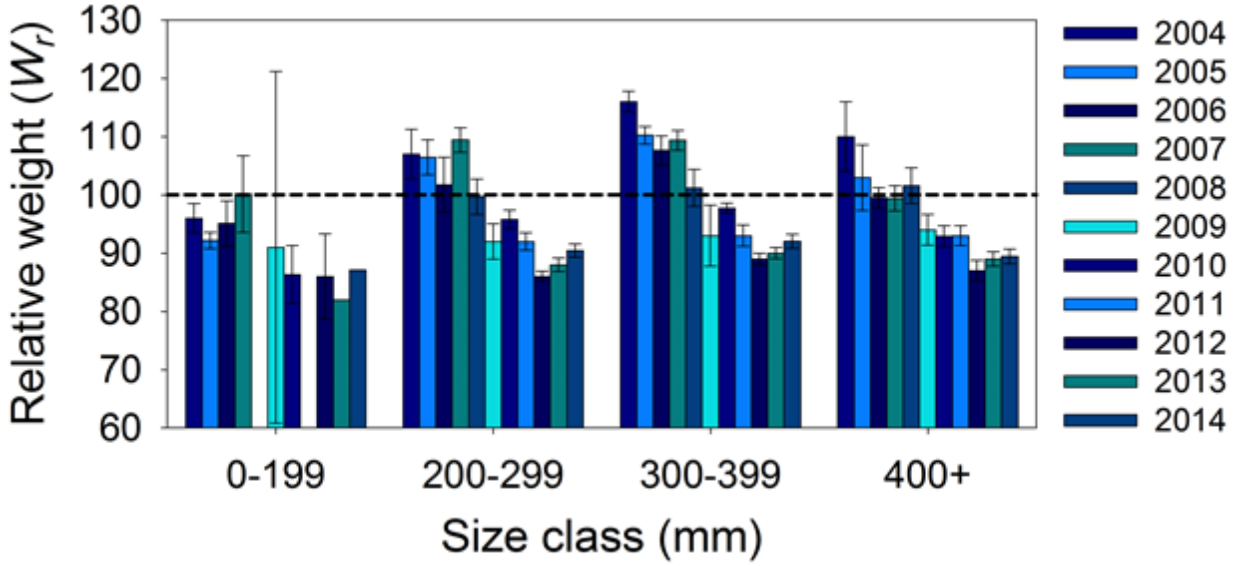

Figure 9. Relative weights (W_r) for four size classes (0 – 199 mm, 200 – 299 mm, 300 – 399 mm, and 400+ mm) of Yellowstone Cutthroat Trout from spring gill netting in Henrys Lake, Idaho 2004-2014. Error bars represent 95% confidence intervals.

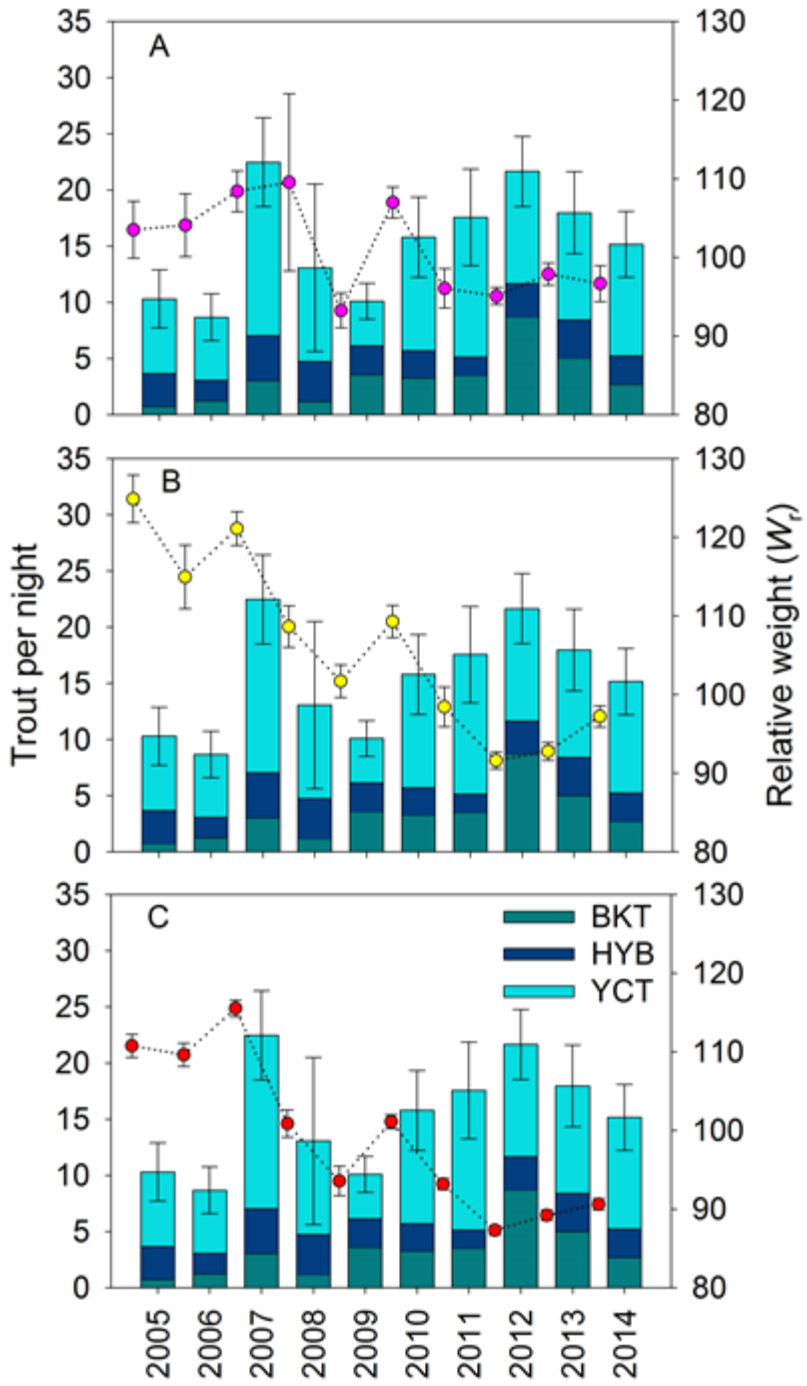

Figure 10. Bar stack plot of trout per night from spring gill netting (left-axis) with 95% confidence intervals for Brook Trout, hybrid trout, and Yellowstone Cutthroat Trout in Henrys Lake from 2005-2014. Line-scatter plot of relative weights (right-axis) with 95% confidence intervals by Brook Trout (Plot A; Pink), hybrid trout (Plot B; Yellow), and Yellowstone Cutthroat Trout (Plot C; Red).

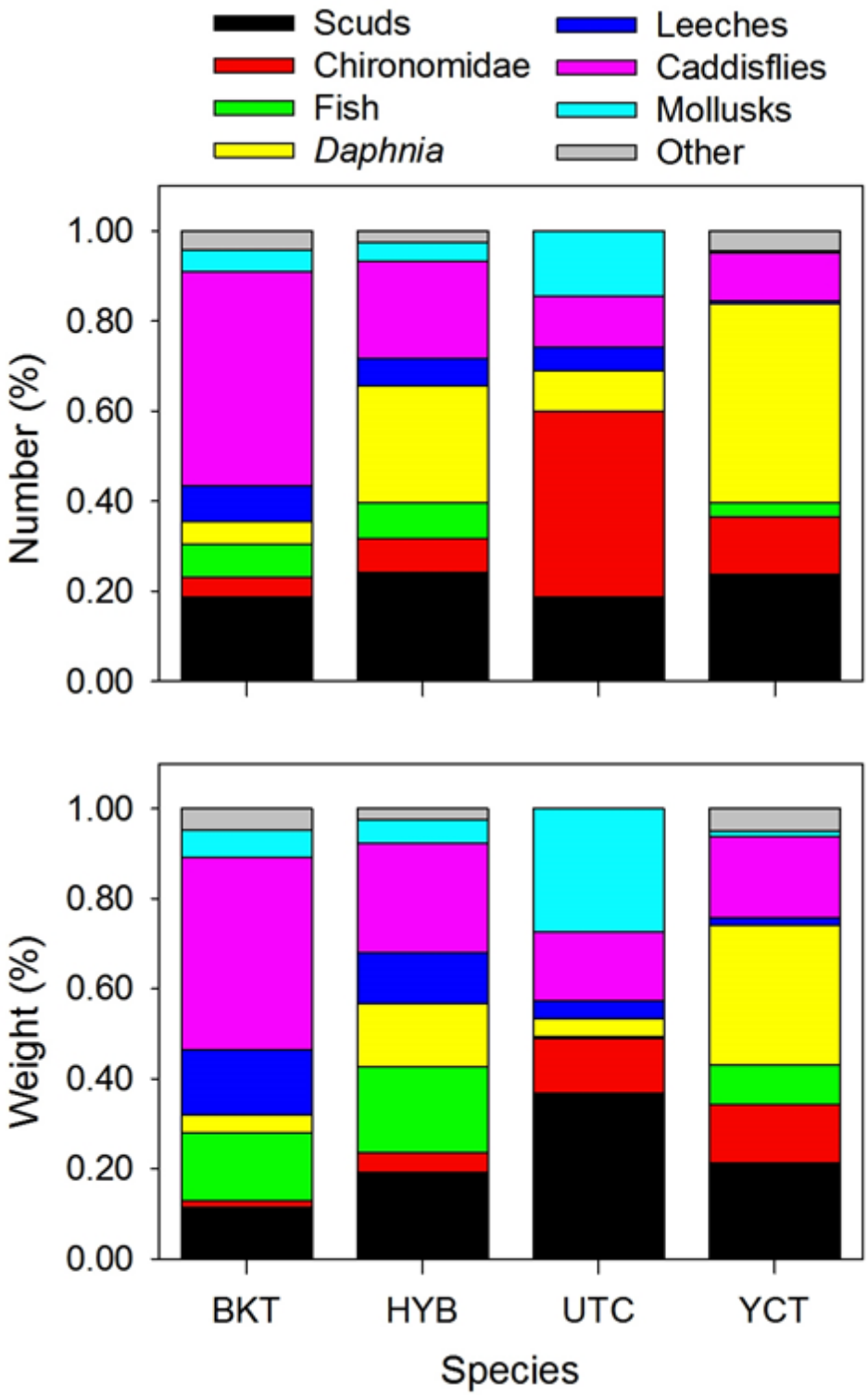

Figure 11. Diet composition, mean percent by weight and number, of Brook Trout (BKT), hybrid trout (HYB), Utah Chub (UTC), and Yellowstone Cutthroat Trout (YCT) across all months sampled (May-October) in Henrys Lake, 2014.

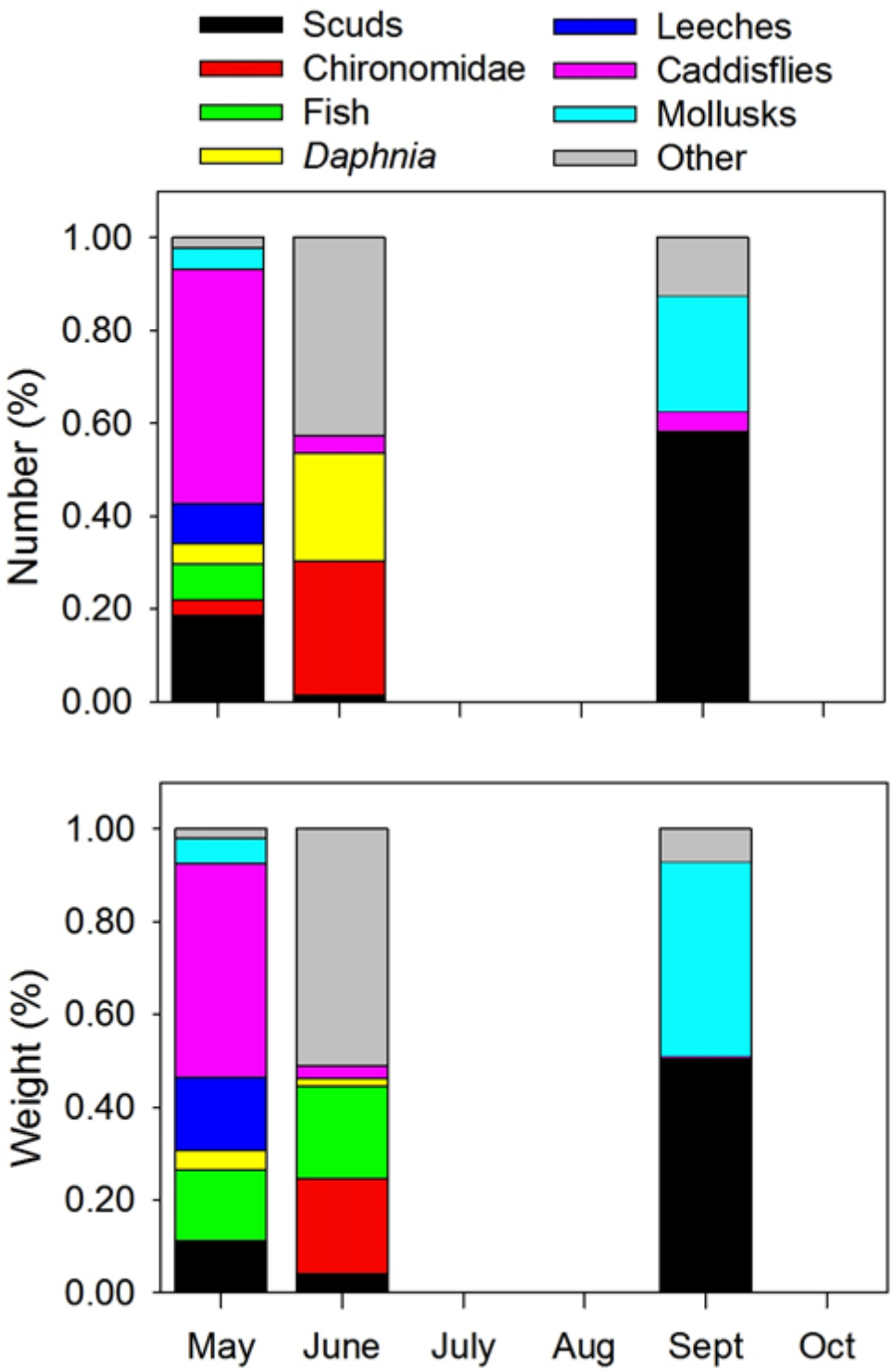

Figure 12. Brook Trout diet composition, mean percent by weight and number, by month (May-October) in Henrys Lake, 2014.

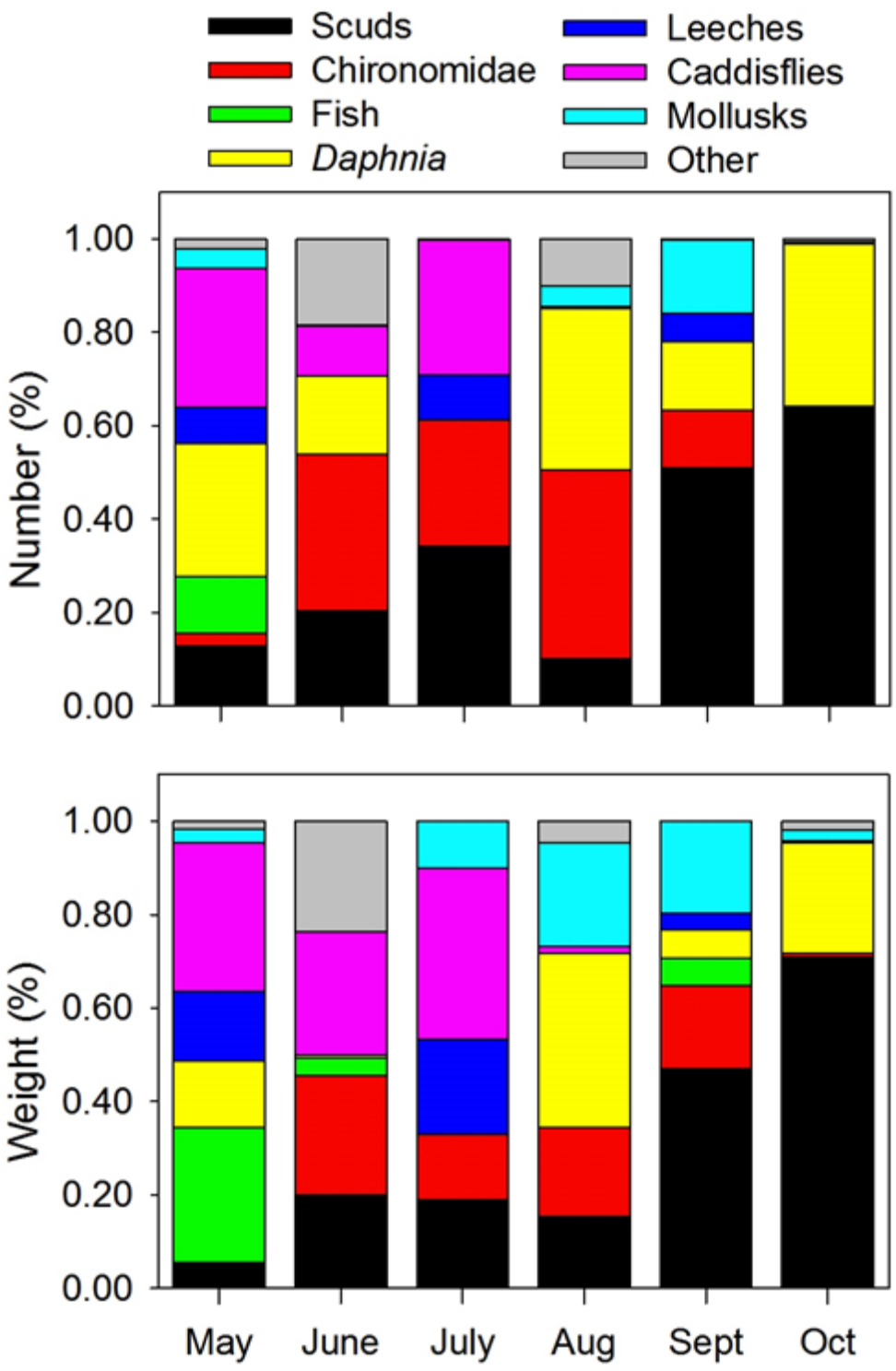

Figure 13. Hybrid trout diet composition, mean percent by weight and number, by month (May-October) in Henrys Lake, 2014.

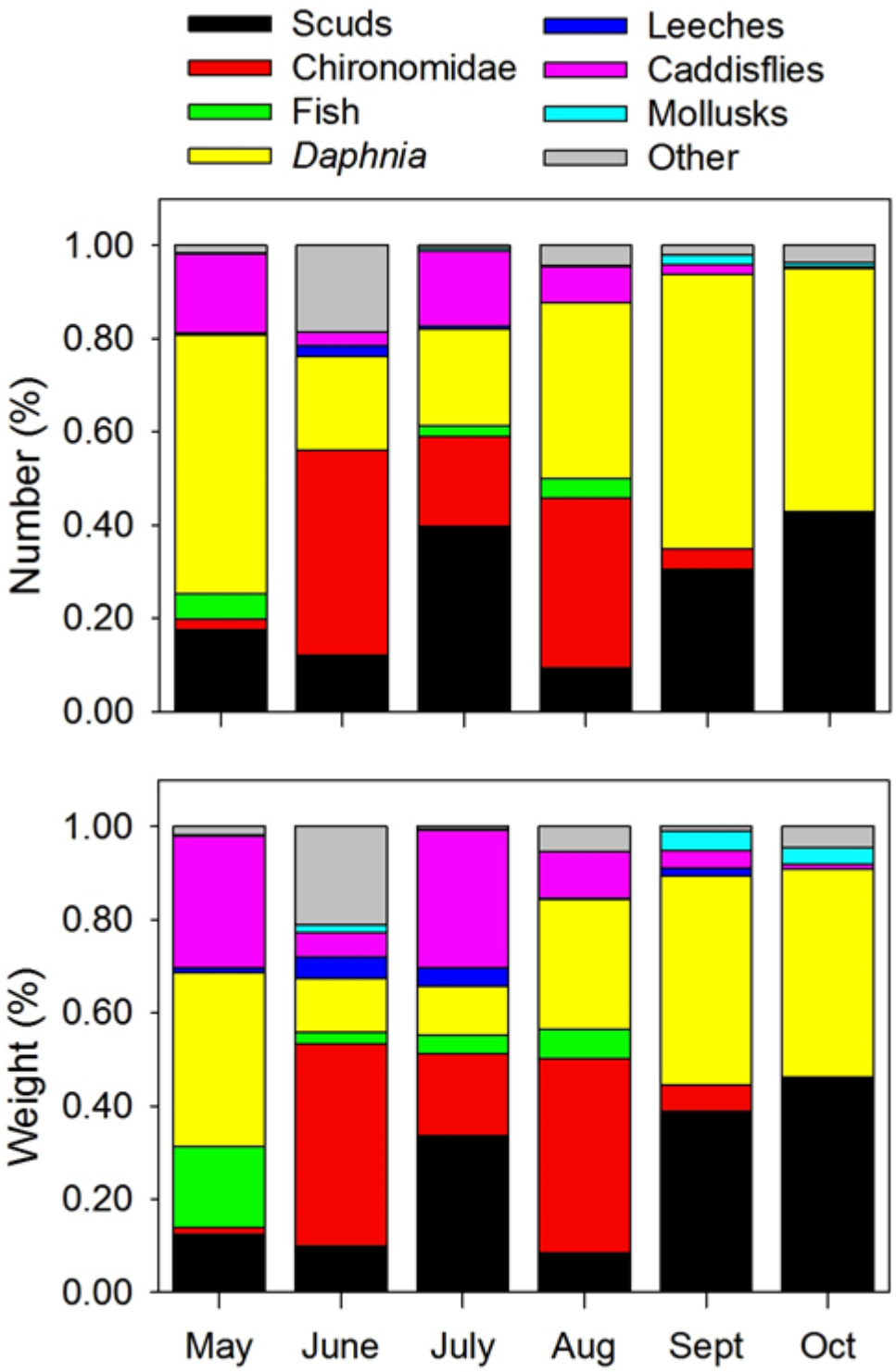

Figure 14. Yellowstone Cutthroat Trout diet composition, mean percent by weight and number, by month (May-October) in Henrys Lake, 2014.

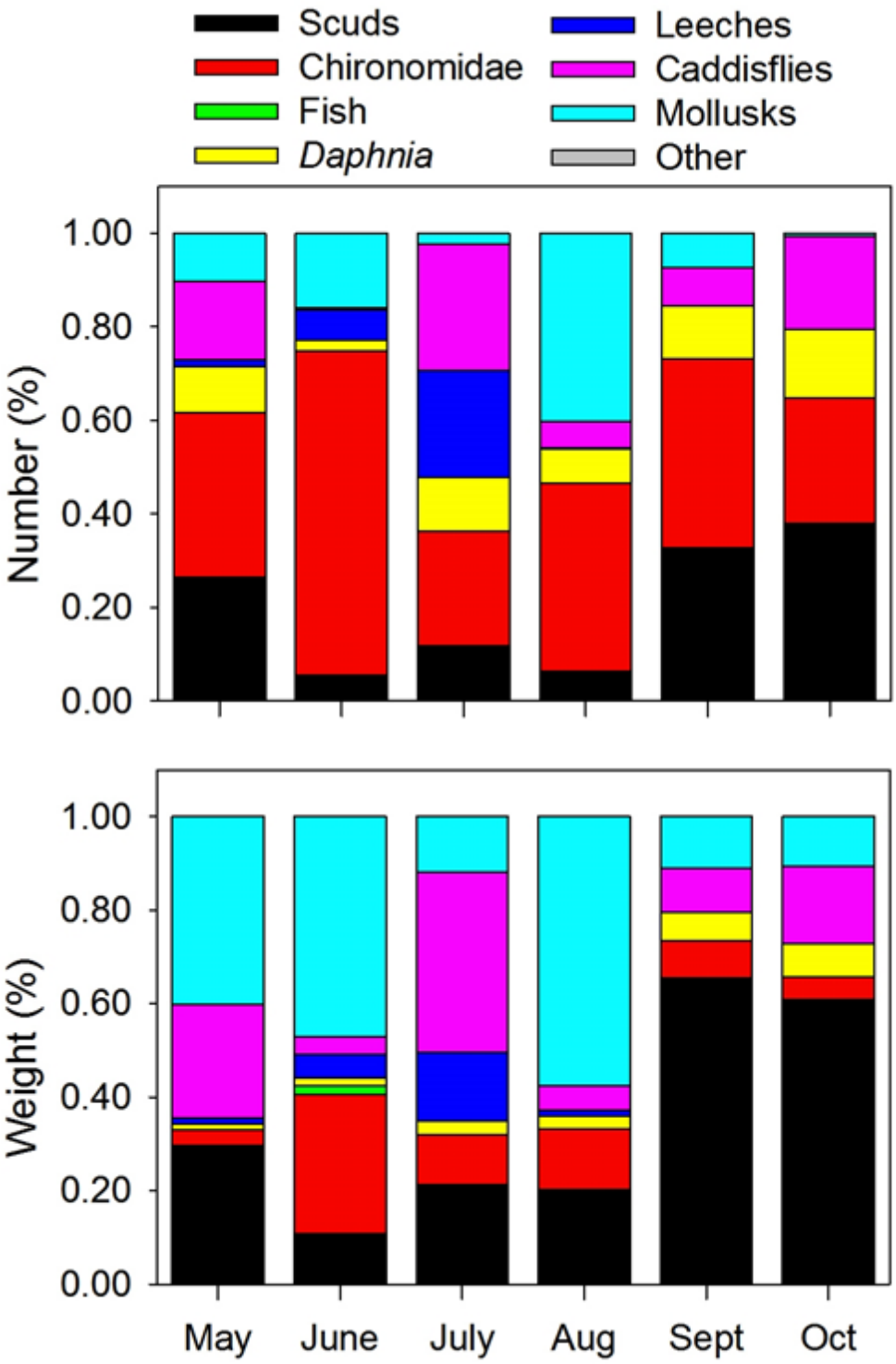

Figure 15. Utah Chub diet composition, mean percent by weight and number, by month (May-October) in Henrys Lake, 2014.

Figure 16. The percent of Utah Chub collected in gill nets that were within the estimated gape widths (i.e. susceptible to predation) of age class for Brook Trout, Yellowstone Cutthroat Trout, and hybrid trout in Henrys Lake, 2014.

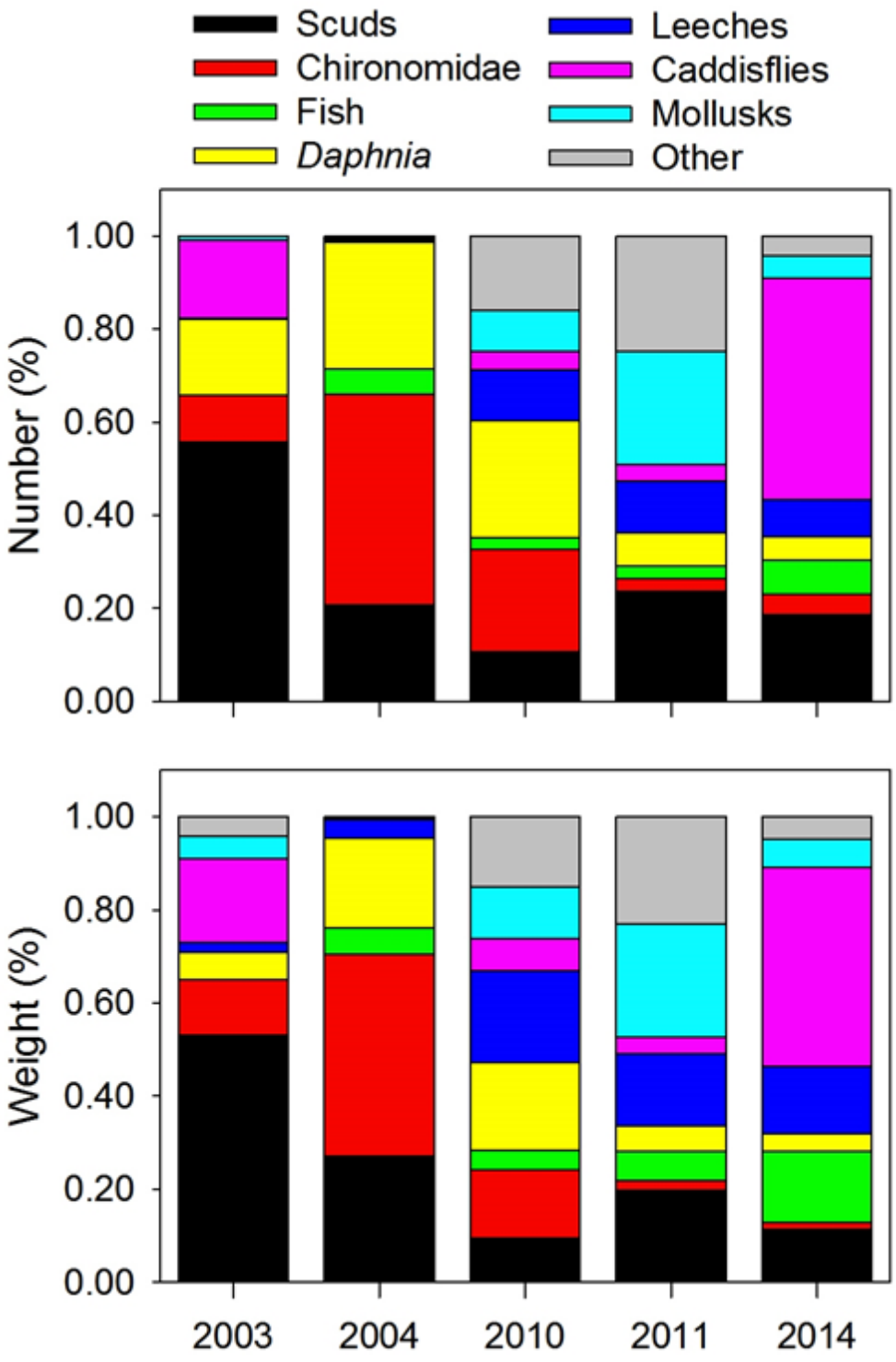

Figure 17. Diet composition, mean percent by weight and number, of Brook Trout across all months sampled (May-October) in years 2003 (*n* = 12), 2004 (*n* = 18), 2010 (*n* = 144), 2011 (*n* = 95), and 2014 (*n* = 89) in Henrys Lake.

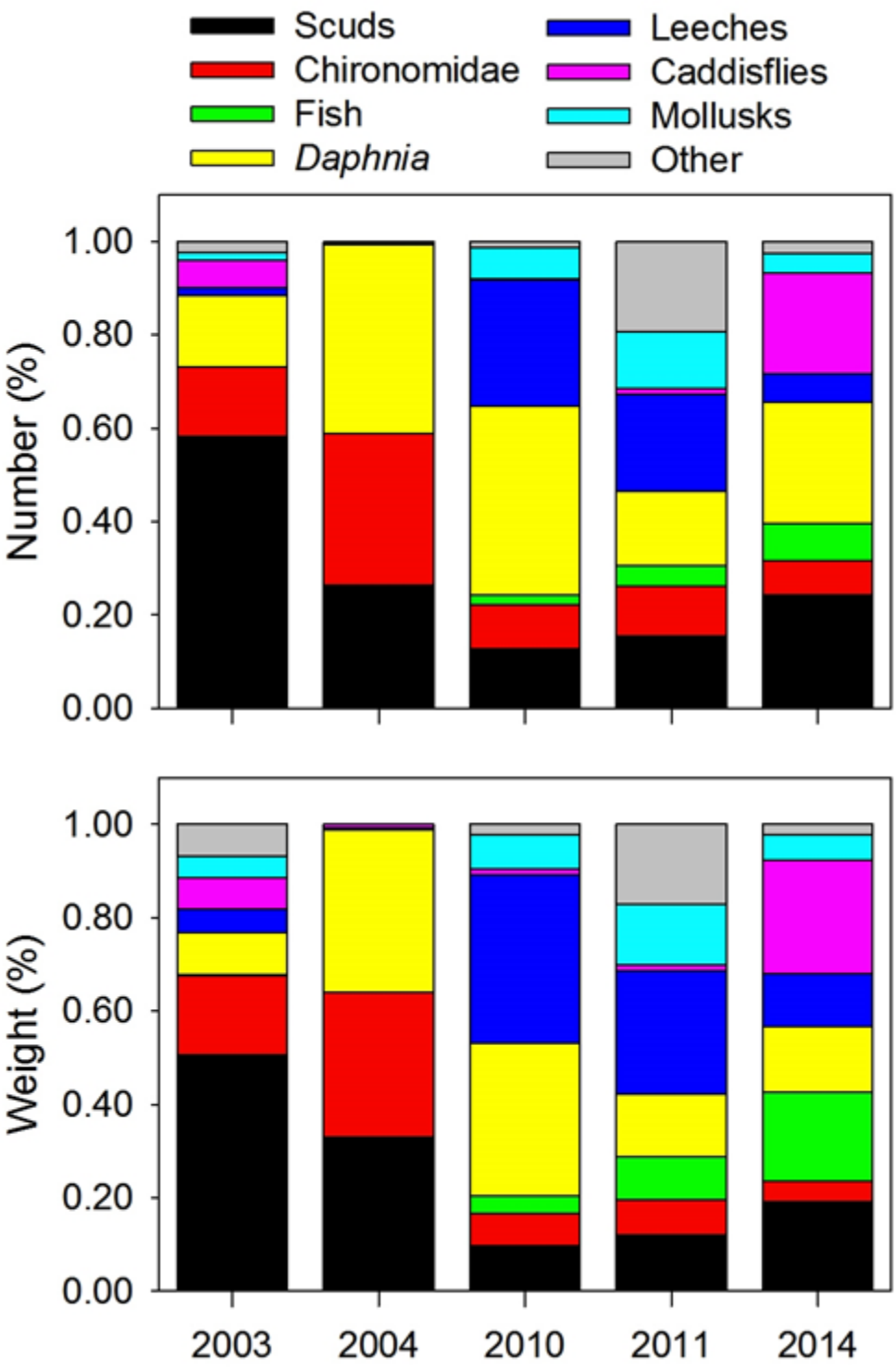

Figure 18. Diet composition, mean percent by weight and number, of hybrid trout across all months sampled (May-October) in years 2003 (*n* = 104), 2004 (*n* = 107), 2010 (*n* = 119), 2011 (*n* = 211), and 2014 (*n* = 129) in Henrys Lake.

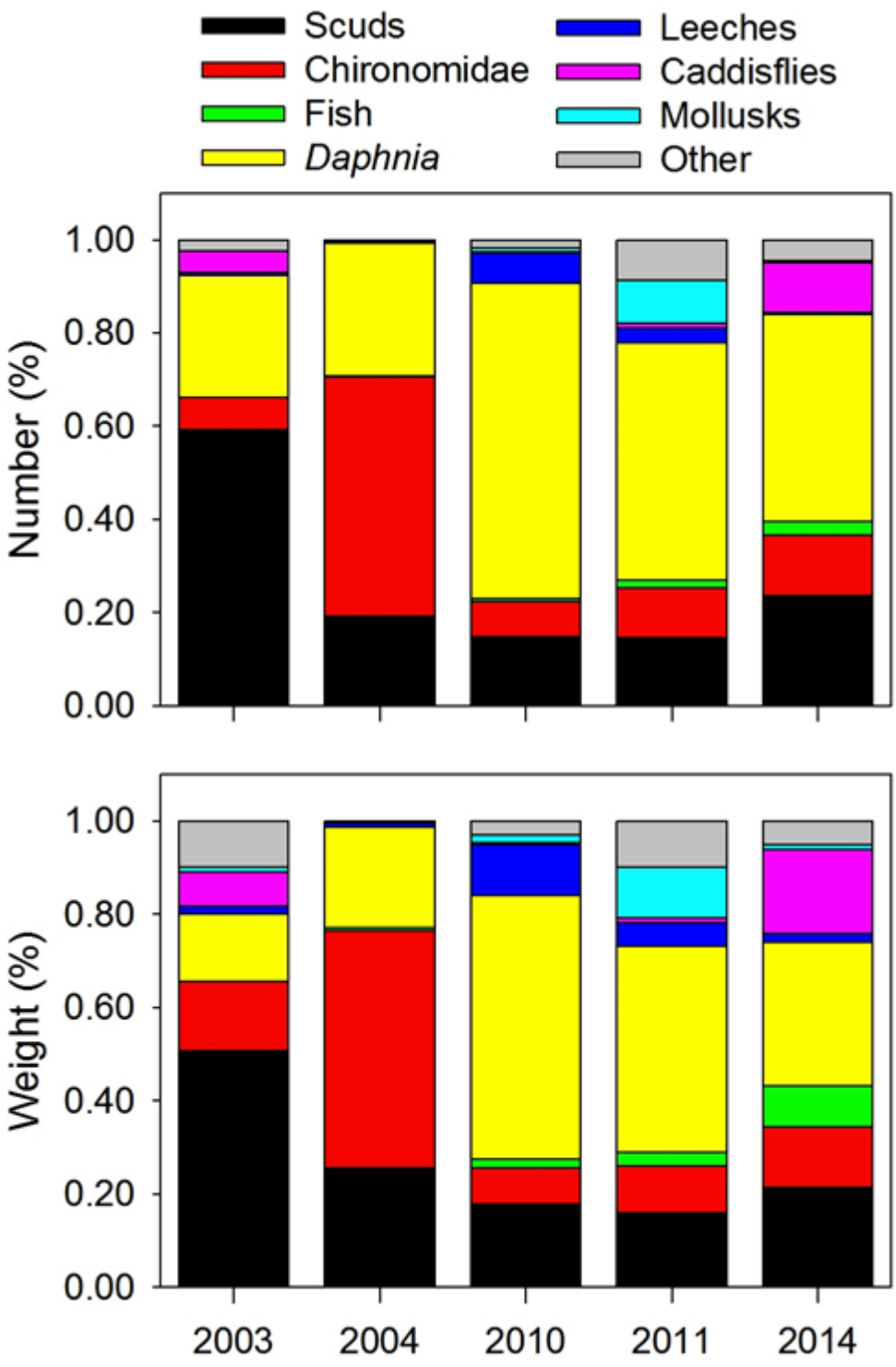

Figure 19. Diet composition, mean percent by weight and number, of Yellowstone Cutthroat Trout across all months sampled (May-October) in years 2003 (*n* = 156), 2004 (*n* = 101), 2010 (*n* = 476), 2011 (*n* = 409), and 2014 (*n* = 338) in Henrys Lake.

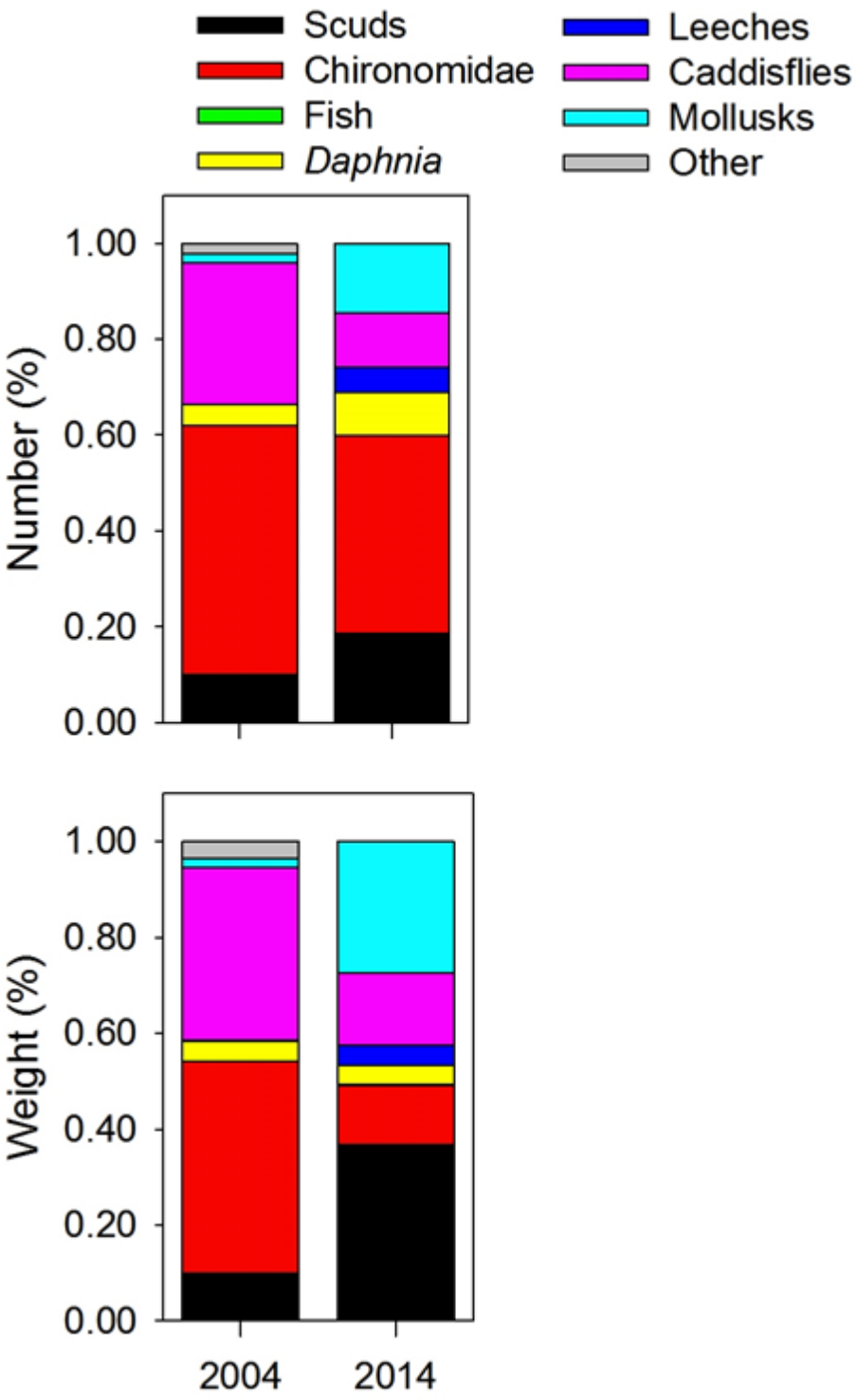

Figure 20. Diet composition, mean percent by weight and number, of Utah Chub across all months sampled (May-October) in years 2003 (*n* = 203) and 2014 (*n* = 341) in Henrys Lake.

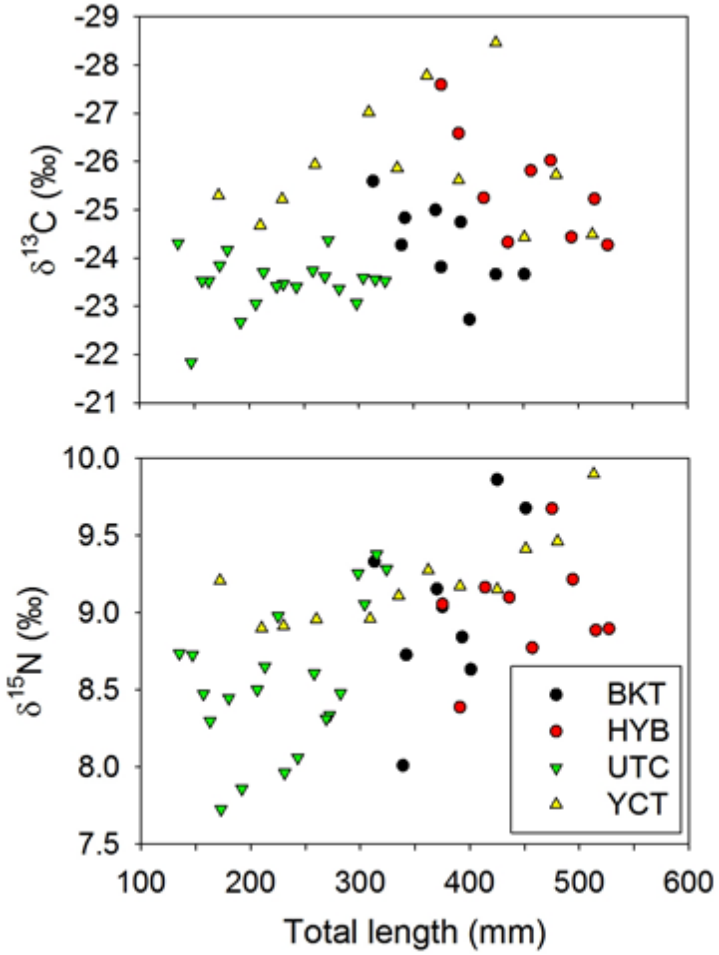

Figure 21. Stable isotopes, carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$), against total length of Brook Trout (BKT), hybrid trout (HYB), Utah Chub (UTC) and Yellowstone Cutthroat Trout (YCT) in Henrys Lake, 2014.

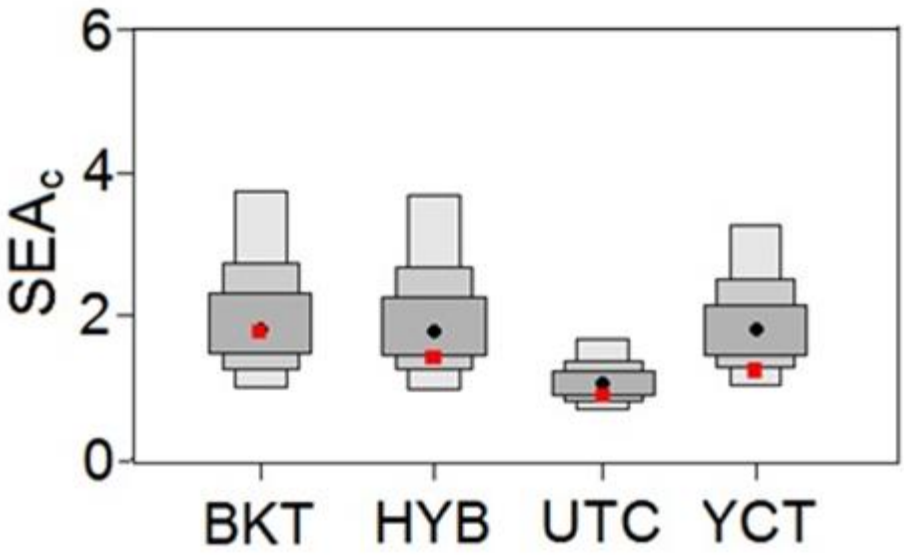

Figure 22. Bayesian estimated posterior distributions of standard ellipse areas corrected for sample size (SEA_c) for Brook Trout (BKT), hybrid trout (HYB), Utah Chub (UTC), and Yellowstone Cutthroat Trout (YCT) as determined by stable isotope analysis Henrys Lake, 2014. Black points represent mode of posterior distribution and rectangles with increasingly lighter shades of grey represent 95, 75, and 50% credible intervals, respectively.

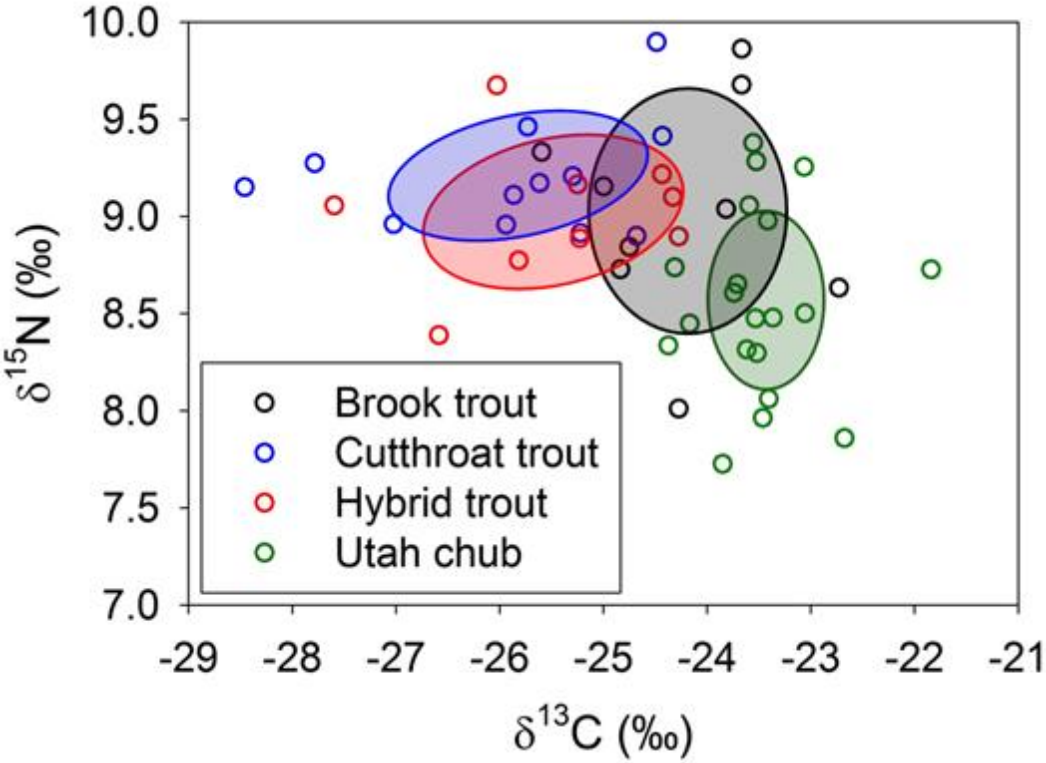

Figure 23. Stable isotope bi-plot of the isotopic niches of Brook Trout, Cutthroat Trout, hybrid trout, and Utah Chub in Henrys Lake, 2014. Lines enclose the core ellipse areas (SEA_c), which represents approximately 40% of the core dietary niche for each species.

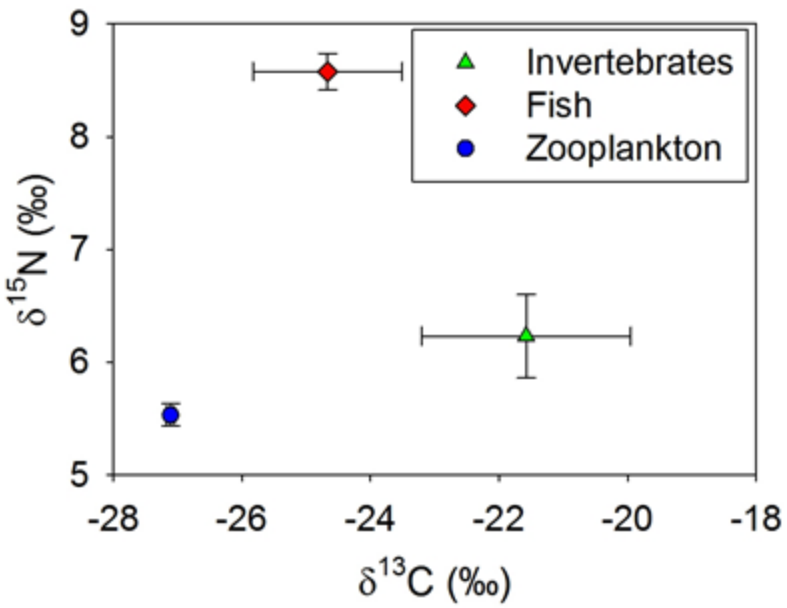

Figure 24. Carbon and nitrogen SIA with associated SE of prey used for mixing model. Fish (chubs and sculpin) and invertebrates (scuds, chironomids, leeches, caddisflies). Only invertebrates and zooplankton were used in the mixing model for Utah Chub.

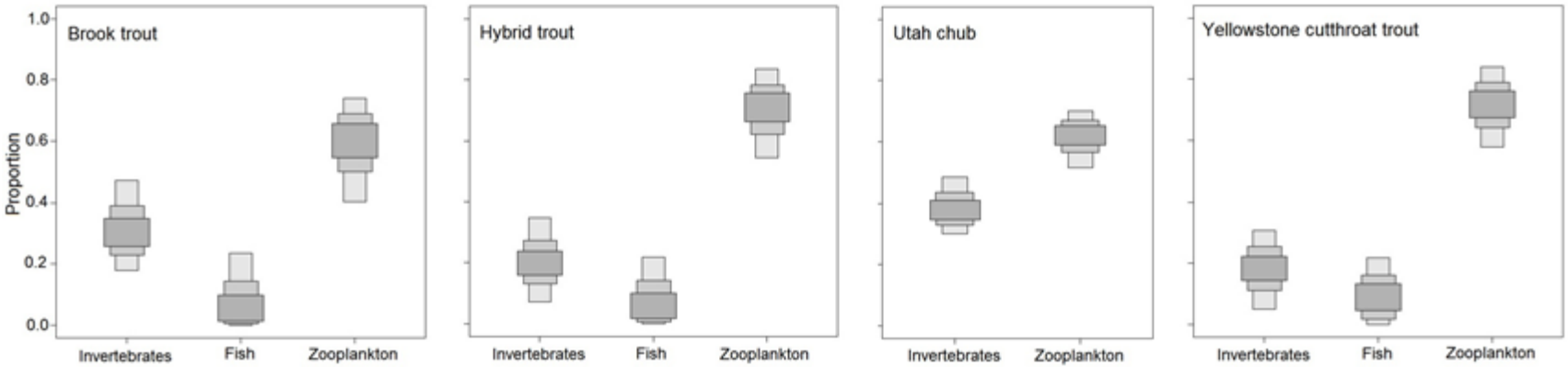

Figure 25. Diet proportions of invertebrates, fish, and zooplankton for Brook Trout, hybrid trout, Utah Chub, and Yellowstone Cutthroat Trout estimated from the Bayesian mixing model (SIAR) in Henrys Lake, 2014. Rectangles with increasingly lighter shades of grey represent 95, 75, and 50% credible intervals, respectively.

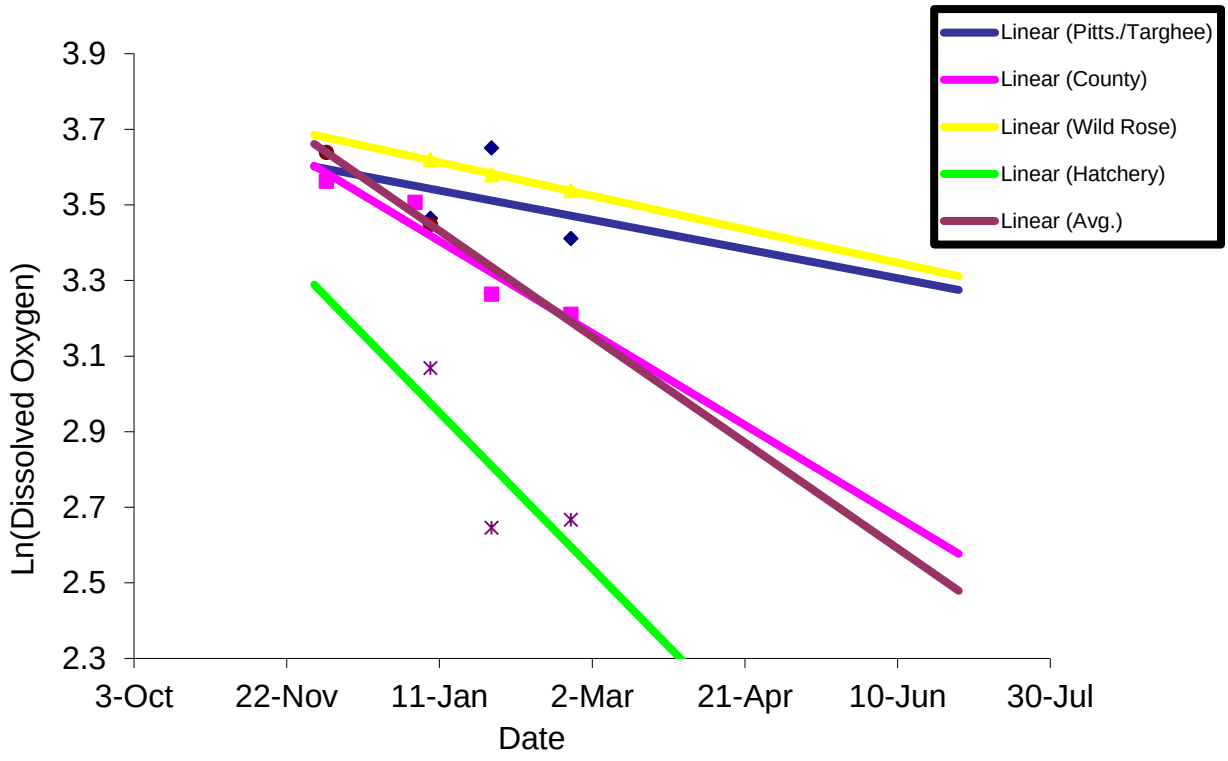

Figure 26. Dissolved oxygen depletion estimates from Henrys Lake, Idaho, 2013-2014.

ISLAND PARK RESERVOIR

ABSTRACT

We used 6 standard experimental gill nets (3 floating, 3 sinking) to assess fish populations and relative abundance in Island Park Reservoir during June 2014. Mean catch (fish per net night) was 24.5 (SE ± 12.1) for Utah Sucker *Catostomus ardens*, 11.2 (SE ± 9.0) for Utah Chub *Gila atraria*, 5.7 (SE ± 2.0) for Rainbow Trout *Oncorhynchus mykiss*, and 0.2 (SE ± 0.2) for Brook Trout *Salvelinus fontinalis*. Catch rate of Rainbow Trout in floating gill nets only in 2014 was 10 fish/net-night, which was similar to previous catch rates in floating gill nets in 2012 (12 fish/net-night) and 2013 (9 fish/net-night). Mean relative weight (W_r) for Rainbow Trout was 87 (SE ± 1.6) and we found condition decreased with size suggesting some food limitation may be occurring in larger Rainbow Trout as they switch from a hatchery environment to foraging in the reservoir. We also examined historical gill net catch rates in floating and sinking gill nets from 2002 to 2014 to determine the most appropriate method to monitor trout populations. Historically, biologists suspected sinking gill nets caught large numbers of non-target species (e.g. Utah Chub, Utah Sucker), and may not have been as effective at sampling salmonid populations as a result. Analysis suggests Utah Chubs and Utah Suckers were caught most frequently in the sinking gill nets (P <0.0001). In contrast, Rainbow Trout and Mountain Whitefish *Prosopium williamsoni* were caught most frequently in the floating gill nets (P <0.0001). Based on this analysis, floating gill nets may be the most appropriate sampling method for monitoring salmonid trends in Island Park Reservoir, particularly for Rainbow Trout. We attempted to determine the most effective way to sample kokanee *O. nerka*, and explored utilizing curtain gill nets deployed in the thermocline. We collected a total of 44 kokanee in one curtain net. In comparison we have captured a total of 76 kokanee in the last 12 years of sampling (2002-2014) using floating and sinking gillnets in 86 net nights. Thus, curtain nets appear to be more effective than standard experimental gillnets in collecting kokanee for monitoring purposes. In effort to reestablish a spawning of run of kokanee in Moose and Lucky Dog creeks, we placed eyed eggs from 60 pairs of kokanee in artificial redds created using an artificial egg depositor. Our intent is to have these eggs hatch in an area that historically supported wild kokanee production in the hopes of restoring that spawning run and improving the fishery in Island Park Reservoir.

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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/hour. Rainbow Trout *Oncorhynchus mykiss* have provided the bulk of angler catch, with kokanee Salmon *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 salmon, although some spawning by both occurs in the Henrys Fork Snake River upstream of the reservoir. Annual Rainbow Trout fingerling stockings have averaged 467,000 over the past 71 years and have been as high as 2.5 million fish in 1959. Beginning in 2010, IDFG increased the size of fingerling Rainbow Trout 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.

In addition to Rainbow Trout, Island Park Reservoir also sustains a kokanee salmon fishery. While historically abundant, the kokanee population has declined in recent 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 spawning run of kokanee in Moose Creek, upon which IDFG established a kokanee trapping facility 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 drawdown. 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 in 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 non-game 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 non-game fish abundance and improve angler catch rates. The efficacy of these treatments was questioned as early as 1982, when Ball et al. (1982) observed that the three chemical rehabilitations of Island Park Reservoir over the previous 25 years had not been successful at long-term eradication of non-game species. Improvements in the trout fishery had been the result of increased stocking levels, especially noticeable with the large introductions of catchable rainbow. Ball et al. (1982) further noted that the observed declines in the Rainbow Trout fishery two to four years after treatment are the result of decreased levels of hatchery inputs and are not due to the increase in 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 2002). More recently, Garren et al. (2008) found that non-game fish exceed pre-rotenone 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 have no effect on improving angler catch rate.

Island Park Reservoir is operated as an irrigation storage reservoir for agricultural users downstream, and is therefore subject to annual fluctuations in water levels. 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 related to gill net catch rates. 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 exactly is impacting 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% due to entrainment (Maiolie and Elam 1998). Congregations of all age-classes of kokanee were found near Dworshak Dam, making them susceptible to entrainment due to high volumes of water being released through the dam.

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 pre-1994 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 being 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 non-anadromous 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, and recent gill netting in Island Park Reservoir (Schoby et al. 2013) 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 reservoir operations, drought, 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 have been 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 or Henrys Lake Outlet and Moose Creek (Figure 27; Appendix C). 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 27). 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 storage 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. Moose Creek – 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, approximately 25 km upstream of Island Park Dam. 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 and limnological characteristics for fishery management decisions on Island Park Reservoir and its tributaries, and to develop appropriate management recommendations.

METHODS

As part of routine population monitoring, we set 6 gill nets in Island Park Reservoir overnight on June 5, 2014 (Figure 27; Appendix D). Gill nets consisted of either floating or sinking types measuring 46 m by 2 m, with mesh sizes 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: mm) and weights (g). We calculated relative abundance as well as catch per unit effort (CPUE: fish per net night). We compiled historic gill net catch data from 2002, 2005, 2012, 2013, and 2014 to determine whether the different net types (floating and sinking) targeted different fish species. We ran a randomized ANOVA block design with year as the block, number of fish per net as the response and net type as the treatment. 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, $\log W_s = -4.898 + 2.990 \log TL$

(Simpkins and Hubert 1996) to calculate relative weights of Rainbow Trout, log W_s = -5.062 + 3.033 log TL for kokanee (Hyatt and Hubert 2000). Relative weights for catchable sized Rainbow Trout (>250 mm) were compared to total length using linear regression. We removed the sagittal otoliths of all trout caught in our gill nets for age and growth analysis. After removal, all otoliths were cleaned on a paper towel and stored in individually-labeled envelopes. Ages were estimated by counting annuli under a dissecting microscope at 40x power. Otoliths were submerged in water and read in whole view when clear, distinct growth rings were present. We sectioned, polished, and read otoliths in cross-section view with transmitted light when the annuli were not distinct in whole view. The von Bertalanffy (1938) growth model was used to fit length:

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

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. The model was fitted to length-at-data by using the nonlinear model (NLIN) procedure in program R.

We targeted the kokanee population on June 17, 2014 using experimental gill net curtains suspended at the depth of the suspected thermocline (Appendix E). Experimental net curtains measured 55 m long by 6 m deep. The “small” mesh net was composed of panels with 19, 25, 32, 38, 51, and 64 mm bar mesh monofilament and the “medium” mesh net was composed of panels with 64, 76, 89, 102, 127, and 152 mm bar mesh monofilament. We set nets at dusk and retrieved them the following morning. Curtain nets were deployed in the reservoir in area with a maximum depth of 22 m. Sites were randomly selected by overlaying a grid system (100 X 100 m) in mapping software (IDFG staff 2012). The nets were set in depth range of 9 to 15 m to ensure adequate coverage in the thermocline. 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/net-night).

We electrofished 60 pairs of kokanee from Big Elk Creek, a tributary to Palisades Reservoir, on September 12, 2014 to collect gametes which were then combined to produce viable fertilized eggs to be stocked in Moose Creek and Lucky Dog Creek (Appendix F). Genetic information was collected from all male and female kokanee pairs for Parental Based Tagging to evaluate the success of different stocking locations. Genetic samples were archived at the Eagle Genetics Lab and may be used in the future. Kokanee eggs were reared in the Ashton Hatchery facility until eye-up (~October 10) after disease testing was run at Eagle Fish Health Lab. Eyed eggs were placed in artificial redds created using an artificial egg depositor in Lucky Dog Creek (one location) and Moose Creek (five locations) on October 15 and 16. The egg depositor consisted of an aluminum rod that was attached to a hydraulic hose and backpack mounted water pump. The rod was imbedded into the substrate to a depth of approximately 23-30 cm. Water was then flushed through the rod for approximately 60 seconds to remove fine silt and sediments from the artificial redd. Eyed eggs were then poured through the top of aluminum rod into the flushed gravel substrate and the rod was slowly removed from the substrate. On October 15, we measured turbidity created by the artificial egg depositor in Moose Creek using a handheld turbidity probe which measured in Nephelometric Turbidity Units (NTU). The probe was placed approximately 0.5 m downstream of the turbidity plume created by the egg depositor. We timed the approximate duration of the plume until the NTUs became less than five and estimated the length of the plumes visually to determine impacts from our activities. We measured turbidity plumes at three different artificial redds.

RESULTS

We collected 249 fish in six net nights of effort (41.5 fish/net-night) using standard experimental gill nets. Overall, relative abundance of the gill net catch was dominated by Utah Sucker (59%), Utah Chub (27%) with Rainbow Trout (14%) and Brook Trout (1%) comprising a lower portion of the catch. Relative abundance ± 90% CI of Utah Sucker in sinking gill nets was 67 ± 26% compared to 9 ± 10.7% in floating gill nets. In contrast, Rainbow Trout relative abundance was 91 ±10.7 in the floating gill net and 2 ± 2.3% in the sinking gill net. Overall catch rate (fish/net-night) was highest for Utah Sucker (24.5), followed by Utah Chub (11.2), and Rainbow Trout (5.7) (Appendix G). Rainbow Trout ranged from 214 to 481 mm TL, with a mean and median length of 337 mm and 338 mm, respectively. Proportional stock density (PSD) was 82, and RSD-400 and RSD-500 were 12 and 0, respectively (Table 11). Mean W_r of Rainbow Trout was 87.5 (SE ± 1.6) (Figure 28). Analysis of larger Rainbow Trout (>250 mm) found a decrease in relative weight with size (Linear regression, $y = -0.120x + 129.7$, $r^2=0.42$). Ages of Rainbow Trout collected ranged from 1 to 4. Rainbow Trout mean total length at age (95% CI) was 252 mm (226-267) for age-1, 337 mm TL (328-350) for age-2, 414 mm TL (396-434) for age-3, and 484 mm TL (436-517) for age-4. Rainbow Trout grew from a starting age of t_0 = -1.45 years (95% CI -3.78 to 0.87) toward their asymptotic length of L_∞ = 1118 mm (95% CI -1885 to 4121) at an instantaneous rate of K = 0.1038/year (95% CI -0.301 to 0.508) (Figure 29).

Historical analysis of the gill net catch data indicated Utah Chubs (ANOVA, F= 36.324, P <0.0001) and Utah Suckers (ANOVA, F = 66.885, P <0.0001) were collected more frequently in the sinking gill nets (Figure 30). In contrast, Rainbow Trout (ANOVA, F = 16.782, P <0.0001) and Whitefish (ANOVA, F = 10.568, P <0.0001) were collected more frequently in the floating gill nets. There were no statistical differences observed with Brook Trout (ANOVA, F = 0.009, P = 0.926), kokanee (ANOVA, F = 0.009, P = 0.924), and Yellowstone Cutthroat Trout (ANOVA, F = 0.004, P = 0.948) in the different net types. Gill net catch rates for Rainbow Trout captured in floating nets was relatively stable over the last three years (2012-2014) with an average of 10.3 Rainbow Trout per net-night (Figure 31). In 2002 and 2005 gill net catch rates of Rainbow Trout were much lower (<4 Rainbow Trout per night).

We collected 44 kokanee, 59 Utah Chub, and 14 Utah Suckers in the small mesh curtain net. In the medium mesh curtain net we only collected 7 Utah Suckers. Kokanee ranged from 90 to 458 mm TL, with a mean length of 149 mm (SE ± 12.4) (Figure 32). Proportional stock density (PSD) was 100 and RSD-400 was 67 (Table 1). Mean relative weight of kokanee was 82.0 (SE ± 1.51) (Figure 28). Smaller kokanee (<150 mm TL) were only collected in the 19 and 25 mm bar mesh size in the curtain nets, which is similar to the size of kokanee collected in these mesh sizes in other kokanee waters in the state (Figure 33). Larger kokanee (<150 mm TL) were collected in the larger bar mesh (32, 38, and 64 mm).

We eyed up approximately 53,000 kokanee eggs at Ashton Fish hatchery that were then placed in the artificial redds in the Moose and Lucky Dog creeks. We placed approximately 43,000 and 10,000 eyed eggs in Moose and Lucky Dog creeks, respectively (Appendix F). We estimated time at emergence from temperature loggers placed in the creeks in an effort to inspect redds and possibly evaluate hatch success. Based on thermal units (TU), fry emergence was likely mid-December for Lucky Dog and late-February for Moose Creek. Prior to fry emergence we placed nets over three artificial redds in the Lucky Dog Creek from December 12 to 20. However, we were unsuccessful in collecting emerging fry using this method.

We performed three trials to evaluate the turbidity plumes created using the artificial redd depositor. In all three trials we observed that NTUs returned to less than 5 within 90

seconds (Table 12). The highest reported NTU was 49.15. The turbidity plumes created by the egg depositor were approximately 2 meters in length (Figure 34).

DISCUSSION

Overall, the catch rate of Rainbow Trout in floating gill nets (10 fish/net-night) was similar to catch rates in 2012 and 2013, which were 12 and 9 fish/net-night, respectively. Annual stocking numbers for catchable Rainbow Trout has been relatively stable over the last five years with an average of approximately 42,000 trout. Although Rainbow Trout abundance remains stable, we did observe a decrease in condition with size in catchable Rainbow Trout (>250 mm), which suggests food limitation was occurring for larger fish. The higher condition observed in the 250-300 mm size range was likely a result of collecting recently stocked fish in the nets. Zooplankton sampling conducted in 2014 found a zooplankton quality index value of 1.40 (95% CI ± 0.7), indicating a high availability of larger Cladocera (>2 mm) present during the summer months in the system (Teuscher 1999). During the fall and winter the reservoir levels remain low after irrigation drawdown. Zooplankton abundances may decrease during this timeframe and become limiting as an available food resource for trout. Increased zooplankton sampling into the fall at low pool may provide insight into the temporal dynamics of zooplankton abundances in Island Park Reservoir. Other possible factors regulating growth and condition of Rainbow Trout may be environmental factors, such as oxygen levels during the summer. If oxygen levels in the hypolimnion fall below the lower threshold tolerated by Rainbow Trout (<6 mg/L) they may be excluded from foraging within that spatial portion of the reservoir. Future research should examine potential limiting biotic (e.g., competition) and abiotic (e.g., temperature, oxygen) factors and/or refining stocking strategies in an effort to improve Rainbow Trout performance.

Based on historic gill net data (2002-2014), floating gill nets may be the most appropriate sampling method for monitoring salmonid trends in Island Park Reservoir. Concerns for the efficiency of sinking gill nets have been raised, particularly due to the possibility of reduced efficiency of capturing trout when nets are filled with non-target species. This could result in inaccurate estimates of population trends over time. Island Park Reservoir has high densities of both Utah Chub and Utah Sucker, and several rotenone projects have been undertaken in prior decades to reduce these species (Gamblin et al. 2002). We recommend using floating gill nets to avoid these benthic species. Reducing bycatch and increasing efficiency of monitoring trout will reduce net processing time and allow for more nets to be deployed nightly. More effort in the form of floating gill nets will help increase the precision on indices (e.g. relative weights) and estimates (e.g., CPUE) for target salmonids.

Curtain nets appear to be an appropriate method for monitoring of the kokanee population in Island Park Reservoir. We collected 44 kokanee in one curtain net across a wide size range (90 to 458 mm TL) with two apparent age classes present. In contrast, we did not collect any kokanee in the six floating and sinking gill nets used in 2014. The neutral buoyancy of the curtain 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. The traditional floating and sinking gill nets spatially only occupy the epilimnion and hypolimnion and avoid thermocline during summer stratification, particularly when using nets that are only 2 m deep. Thus, they are ineffective at sampling kokanee in Island Park Reservoir, which adds to the management challenges associated with this species. Due to time constraints this year, we were only able to deploy one curtain net of each size (small and medium) for one night. As a result we were unable to perform power analysis (i.e. lack of variance) to determine the number of nets needed to achieve a desired level of precision. Future sampling protocols

using curtain nets will need to incorporate power analysis to help guide effective monitoring of the population. Collecting higher numbers of kokanee will also allow biologists the ability evaluate hatchery and wild ratios of individual age classes using thermal marks on otoliths (Volk et al. 1990). Kokanee stocked in Island Park Reservoir were reared at Cabinet Gorge Fish Hatchery where all kokanee eyed eggs have been marked with “thermal mass marking” since 1997. As a result, all hatchery kokanee contain distinct thermal marks by age class. Future kokanee monitoring should evaluate hatchery versus wild ratios using otolith thermal marks to assess the importance of hatchery and wild contributions to the fishery.

Hydroacoustic surveys have been incorporated into traditional curtain net sampling methods for monitoring pelagic species like kokanee (Yule 2000), and can provide estimates on the spatial distribution of pelagic species along with density estimates. In 2004, IDFG conducted preliminary hydroacoustic surveys in Island Park Reservoir to estimate pelagic fish abundance and size structure (Butts and Nelson 2006). To estimate individual species abundance from the hydroacoustic surveys, researchers use species composition obtained from gill netting efforts from 2000-2002. These regional gill net efforts consisted of floating and sinking gill nets, which are not effective in collecting kokanee in Island Park Reservoir. As a result, kokanee were not incorporated into the fish density estimates from the hydroacoustic efforts, and thereby produced inaccurate results. Curtain nets were effective at capturing kokanee which could be used to more accurately partition relative abundance of fish in the reservoir if future hydroacoustic efforts are considered. Hydroacoustic sampling is not without additional challenges, however. Curtain gill nets also collected high numbers of Utah Sucker and Utah Chub, suggesting these species were occupying the pelagic areas as well, and hydroacoustic surveys may not be able to distinguish kokanee from non-game species as a result. This concept may be worth considering with additional research in the future.

The kokanee spawning project continues to evolve as we attempt to reestablish a self-sustaining kokanee run in Moose and Lucky Dog creeks. This year, we were successful in spawning our target of 60 kokanee pairs. However, we had to collect the kokanee from Big Elk Creek, a tributary to Palisades Reservoir, due to the low run return in Henrys Lake Outlet. A visual survey of adult kokanee in Henrys Lake Outlet during September found few kokanee returned to spawn. In 2013, a strong run of adult kokanee were in the Henrys Lake Outlet and biologists were successful in obtaining 60 pairs for spawning (High et al. 2015). Additional sources for adult kokanee may need to be considered in future years if the run in Henrys Lake Outlet remains low. Similarly, the methods used to plant eyed eggs may continue to evolve. In 2013, streamside egg incubators were used, but icing affected hatch success in a portion of these incubators (High et al. 2015). As a result we used a different strategy in 2014 (egg depositor) to avoid some of the icing issues observed in 2013. Eyed eggs were deposited at depths of 23-30 cm in artificial redds created by the egg depositor. Also, a smaller number of eggs were placed in each artificial redd (200-400 eggs), and should avoid some of the disease issues (e.g., fungus) that become prevalent in egg incubators when high numbers of eggs are in close proximity. We expect adult fish from egg depositions in artificial redds to return to their spawn location as early as 2017, but more likely in 2018 as three-year-old adults. Annual escapement surveys of adult kokanee should be conducted in Moose and Lucky Dog Creeks to evaluate the success of this and past projects. If successful, these projects may help reestablish the wild component of kokanee in Island Park, and ultimately improve fishing for this highly desirable species.

RECOMMENDATIONS

1. Continue gill net monitoring using floating nets to evaluate the Island Park Reservoir Rainbow Trout fishery.

2. Assess available data on temperature and dissolved oxygen in Island Park. Consider additional sampling to evaluate if temperature and dissolved oxygen may be limiting Rainbow Trout growth in Island Park Reservoir.

3. Develop a robust kokanee monitoring program in Island Park Reservoir using curtain nets.

4. Continue visual kokanee escapement surveys in Moose Creek and Big Springs Creek to monitor trends in adult abundance and determine if juvenile/eyed egg releases in these locations have established spawning runs.

5. Continue using kokanee from appropriate sources to establish a spawning population in Moose Creek, either through adult releases or egg collection and incubation in Moose Creek.

Table 11. Stock density indices (PSD: proportional stock density and RSD: relative stock density) and relative weights (W_r) for Rainbow Trout and kokanee collected with gill nets and curtain nets, respectively, in Island Park Reservoir, Idaho 2014. Standard error (SE) for relative weight values is noted in parentheses.

	Rainbow Trout (SE)	**kokanee (SE)**
PSD	82	100
RSD-400	12	67
RSD-500	0	-
W_r		
<200 mm	--	81 (1.4)
200 – 299 mm	88 (3.6)	--
300 – 399 mm	89 (1.6)	97
>399 mm	78 (6.1)	91 (1.4)
Mean	87 (1.6)	83 (1.5)

Table 12. Maximum nephelometric turbidity units (NTU) from the turbidity plume created by the egg depositor and amount of time (seconds) before the water clarity returned to less than 5 NTU.

Trail	**Maximum NTU**	**Time (secs) to <5 NTU**
1	20.85	35
2	49.15	90
3	48.10	75

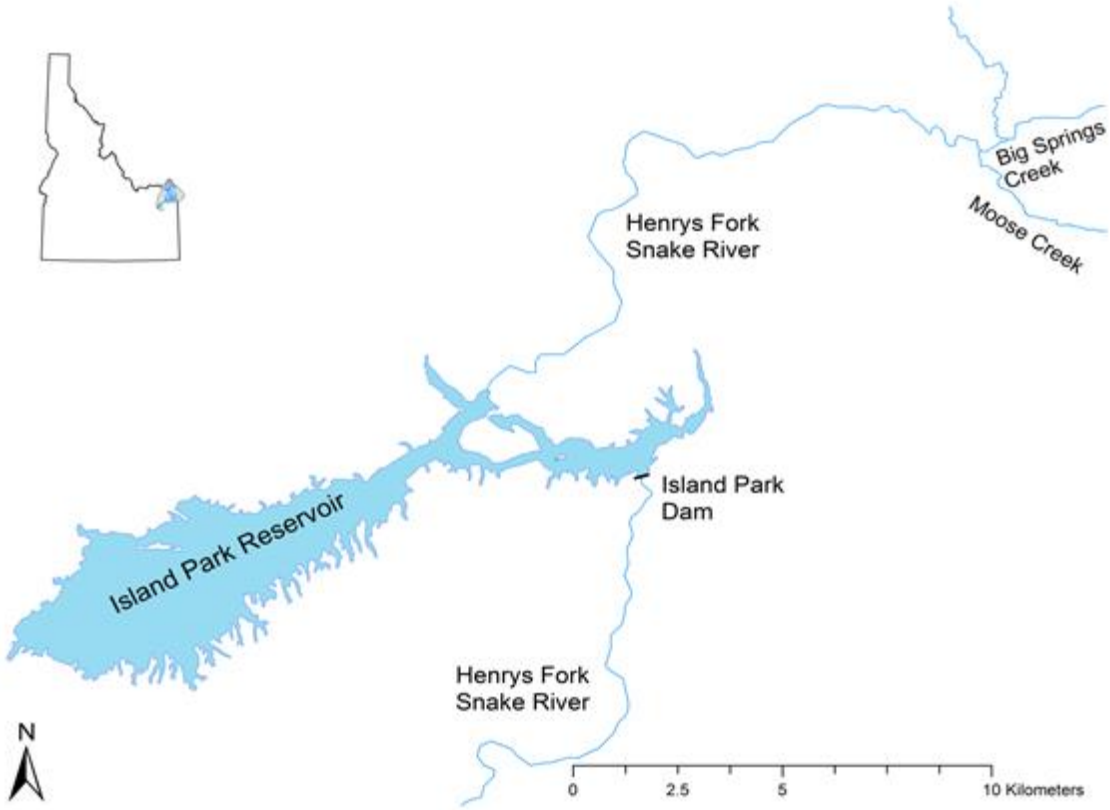

Figure 27. Map of Island Park Reservoir and the major tributaries in southeastern Idaho.

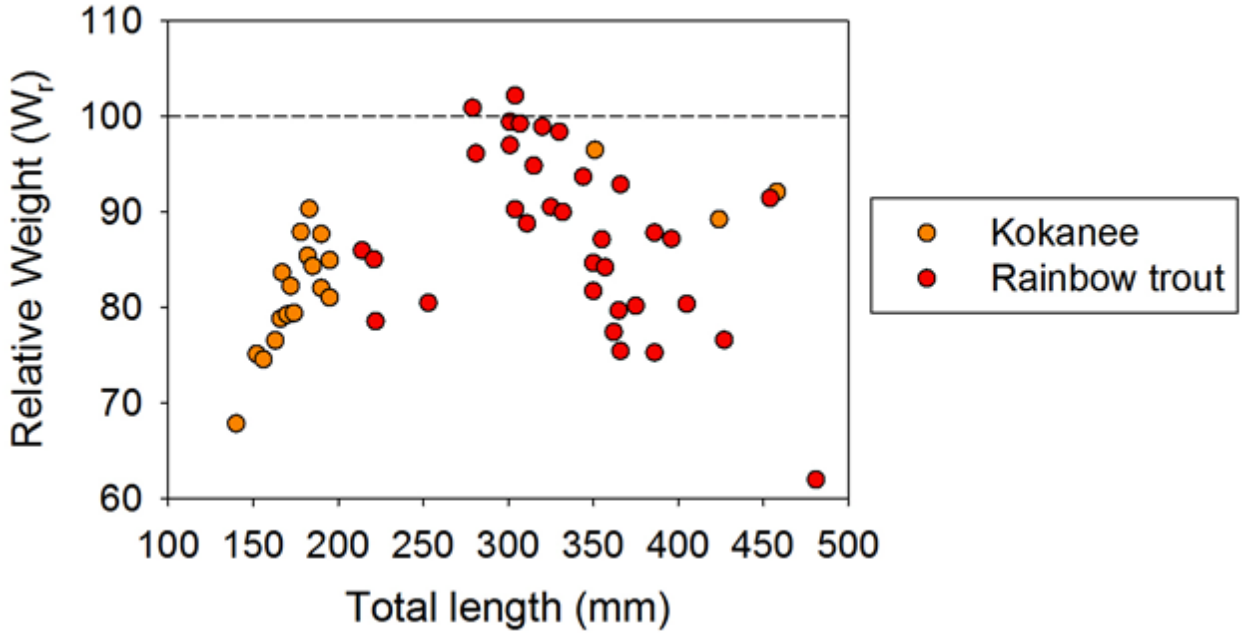

Figure 28. Relative weight (W_r) of kokanee and Rainbow Trout across total length (mm) collected in Island Park Reservoir, 2014.

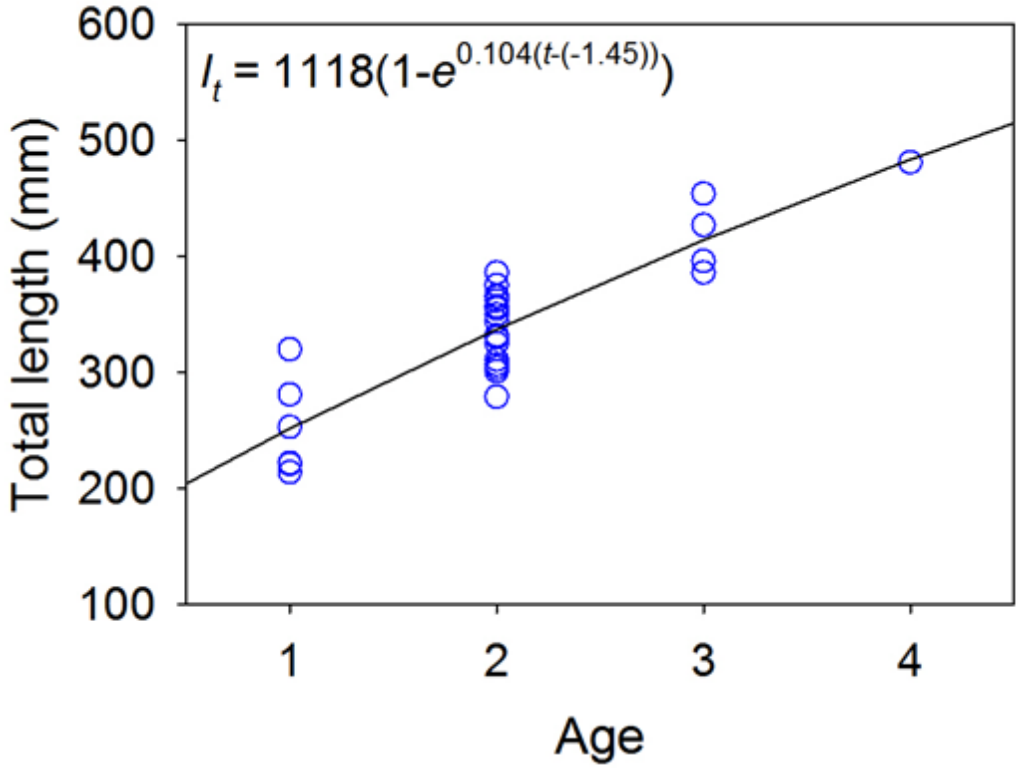

Figure 29. Length-at-age based on non-linear regression for Rainbow Trout collected from gill netting in Island Park Reservoir, 2014. Solid line represents the fitted von Bertalanffy growth model.

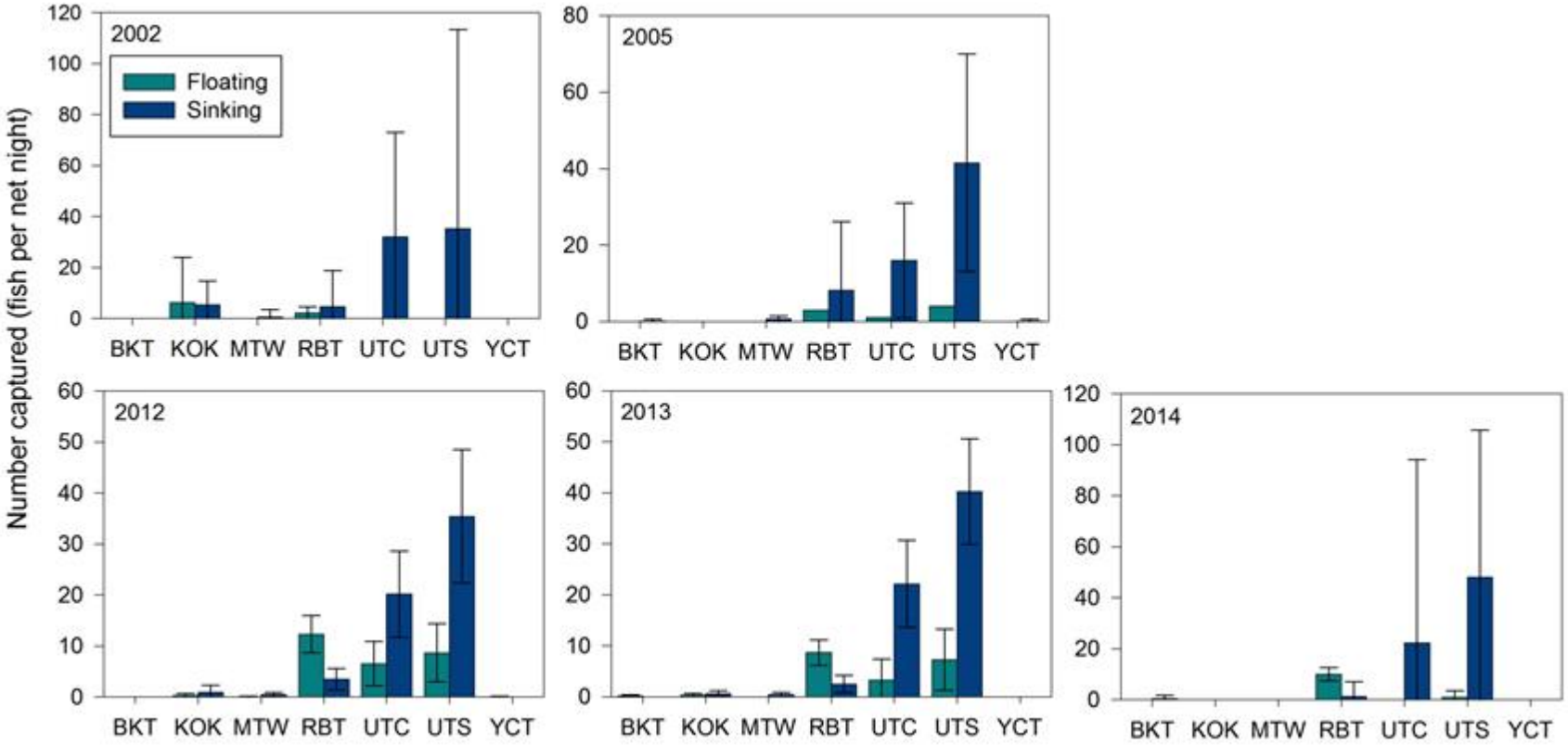

Figure 30. Comparison of floating and sinking gill net catch rates (fish/net-night) for Brook Trout (BKT), kokanee (KOK), Mountain Whitefish (MTW), Rainbow Trout (RBT), Utah Chub (UTC), Utah Sucker (UTS), and Yellowstone Cutthroat Trout (YCT) in Island Park Reservoir, from 2002, 2005, 2012, 2013, and 2014.

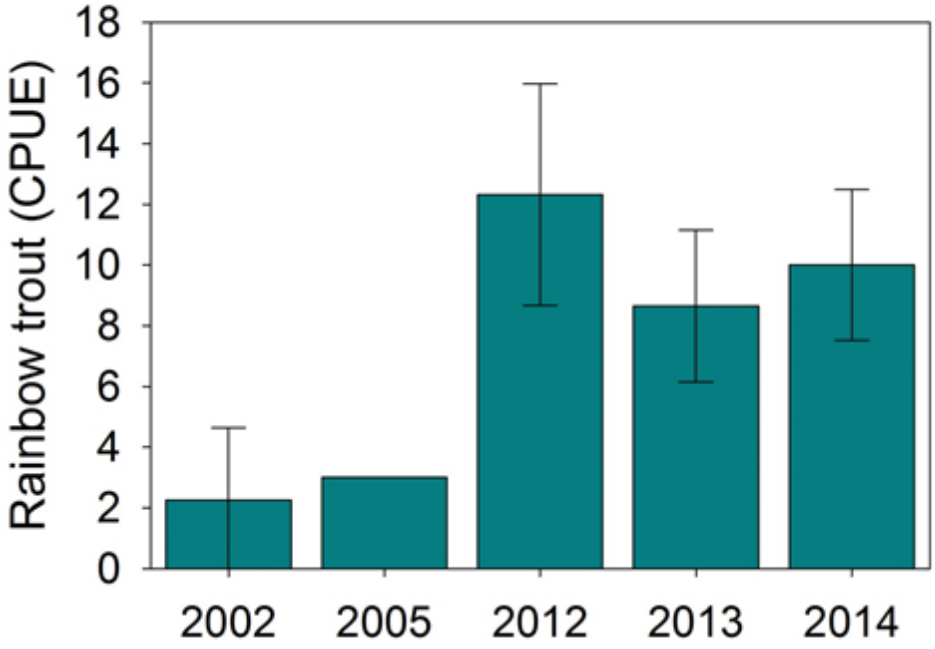

Figure 31. Floating gill net catch rates (fish/net-night) for Rainbow Trout in Island Park Reservoir, from 2002, 2005, 2012, 2013, and 2014.

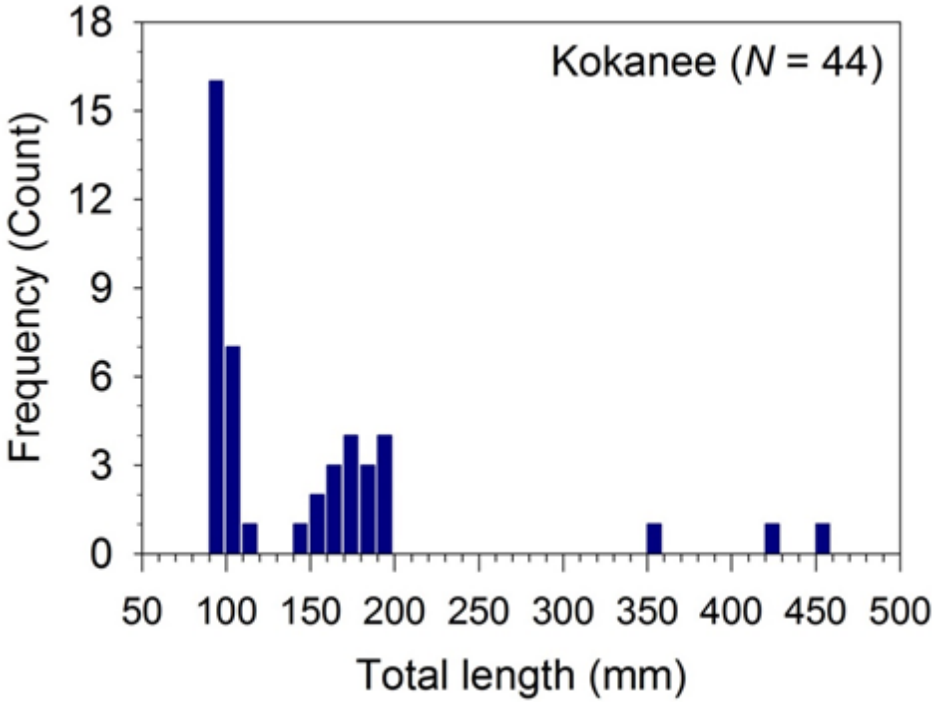

Figure 32. Length frequency of kokanee collected in curtain gill nets in Island Park Reservoir during 2014.

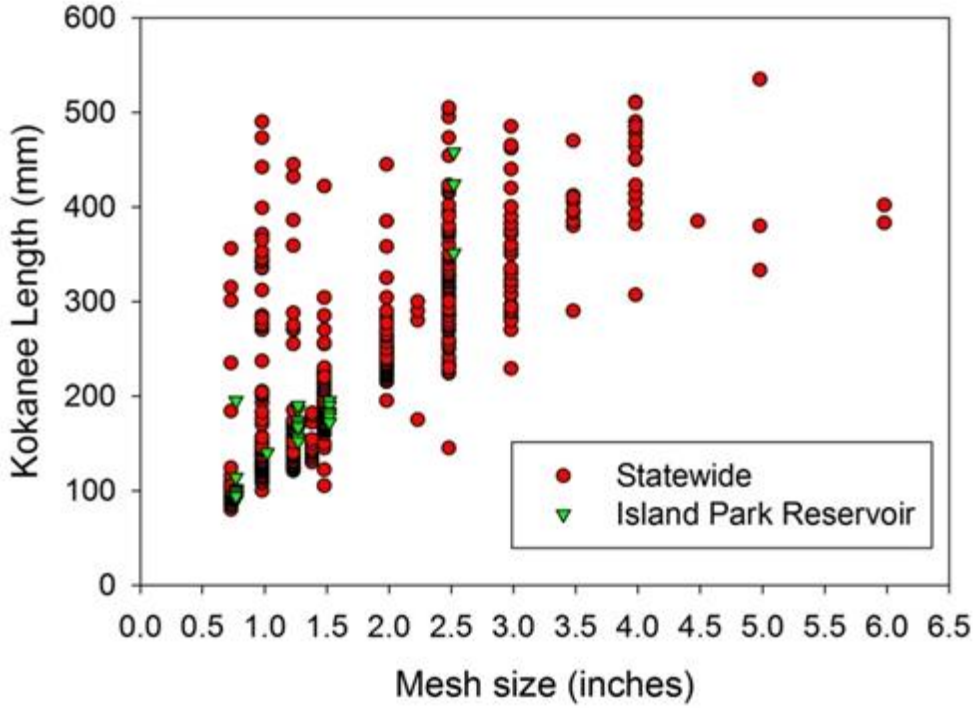

Figure 33. Kokanee total length (mm) collected in curtain nets by bar mesh size (inches) in Island Park Reservoir, 2014, and from other statewide kokanee waters.

Figure 34. A side profile of a typical turbidity plume created by the egg depositor in Moose Creek.

RIRIE RESERVOIR

ABSTRACT

We conducted our fifth consecutive year of fall Walleye *Sander vitreus* index netting (FWIN) in Ririe Reservoir, and captured eight Walleye, ranging from 303 mm to 600 mm TL. Average Walleye per net-night (± 95% CI) was 0.4 ± 0.7, which was similar to number of Walleye per net-night from 2010-2013 (0.4 ± 0.24). Estimates of Walleye ages based on otoliths range from 1 to 7, with five different age classes represented, suggesting yearly recruitment likely occurs albeit at low levels. Walleye only comprised a small proportion of the overall species composition (0.4%) with the majority of gill net catch being dominated by Yellow Perch (75%). Continued annual monitoring of the Walleye population is necessary to evaluate population expansion and to evaluate effects (i.e. trophic cascade) on the other fish populations, particularly stocked salmonids in Ririe Reservoir. We conducted boat electrofishing in June to collect information on the Smallmouth Bass population. The electrofishing effort yielded 35 ± 14.4 Smallmouth Bass per hour. Smallmouth Bass ranged in size and age from 81 to 302 mm TL and zero to eight years, respectively. We were unsuccessful in collecting any Smallmouth Bass above the minimum size limit allowed for harvest (>305 mm TL) in our electrofishing surveys. The slow growth rates of Smallmouth Bass in the Ririe Reservoir suggests the minimum size limit may not be necessary to protect juvenile bass until maturity and liberalizing the bass limit would provide additional opportunity for harvest-oriented anglers. We conducted power analysis to determine appropriate sampling effort and found future boat electrofishing surveys should sample at least 24 sites for 10 minutes (600 seconds) to acquire Smallmouth Bass CPUEs that are within 80% confidence limits and ±25% of the population mean.

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INTRODUCTION

Ririe Reservoir is located on Willow Creek, approximately 32 km east of Idaho Falls (Figure 35). 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, and is less than 1.5 km wide along the entire length, with a surface area of approximately 1,560 acres 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 salmon *Oncorhynchus nerka*, Yellowstone Cutthroat Trout *O. clarkii bouvieri*, Smallmouth Bass *Micropterus dolomieu*, and Yellow Perch *Perca flavescens*. Utah Chub *Gila atraria* and Utah Sucker *Catostomus ardens* are also found in Ririe Reservoir in relatively high numbers. In 2013, creel surveys showed angler use was approximately 43,000 hours and has averaged 47,000 hours of angler use over the last 20 years (High et al. 2015). Beginning in 1990, 70,000 juvenile kokanee were stocked annually, with an increase to 210,000 annually in 2004 to improve catch rates and meet increased angler demand. Up until 2012, approximately 18,000 catchable Yellowstone Cutthroat Trout were stocked annually to provide angler opportunity. Following relatively poor performance of those fish, 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 providing a diverse angling opportunity as well as meeting angler expectations. 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 condition and size of perch (Schoby et al. 2010). A self-sustaining population of Smallmouth Bass has developed from introductions into Ririe Reservoir from 1984-1986. Although limited by the short growing season at this latitude and altitude, Smallmouth Bass provide a diverse and popular angling opportunity for fishermen in the Upper Snake Region. Currently we have a minimum size limit restriction for bass (>12 inches) in the region to protect spawning-sized bass and allow them to spawn at least once before being legal to harvest. However, given the slow growth rates of Smallmouth Bass in Ririe Reservoir, the minimum size limit provides no biological protection and is not meeting its intended objective.

Walleye *Sander vitreus* were first documented in Ririe Reservoir in 2008 (Schoby et al. 2010), which prompted further investigations by IDFG fisheries personnel. Gill netting effort increased in 2008, followed by a walleye telemetry study in 2009 and 2010 (Schoby et al. 2014). Fall Walleye index netting (FWIN, Morgan 2002) was initiated in 2010 as an annual monitoring tool to document trends in the Walleye population in Ririe Reservoir. No Walleye were captured in 18 gill net nights of effort during 2010, and only small numbers of Walleye are encountered in annual netting to date. Low catch rates suggest a low abundance population, but the threat of increasing abundance exists. The impact Walleye may have on the existing fishery is unknown. Managing an apex predator such as Walleye can be difficult because they have the potential to alter fish communities (Knight and Vondracek 1993). In Lake Roosevelt, Washington predation by introduced Walleye accounted for a 31-39% loss of stocked kokanee (Baldwin and Polacek 2002). There are also concerns that the Walleye in Ririe Reservoir may provide a source population for future illegal introductions in the surrounding waters (McMahon and Bennett 1996) and the possibility exists for them to spread downstream of the reservoir. In Washington, personnel with the Department of Fish and Wildlife have cited irrigation canals as the mechanism for Walleye expansion from Banks Lake throughout the Columbia River basin. Additionally, in a study conducted to assess the potential for Walleye introductions in Idaho

(IDFG 1982), Ririe Reservoir was identified as having the biological suitability to sustain a healthy Walleye population, but conflicts with maintaining the existing trout fishery were cited as the main reason for not introducing Walleye into Ririe Reservoir.

OBJECTIVES

1. Use annual fall gill netting to describe population characteristics of Walleye in Ririe Reservoir as a long-term monitoring tool and to monitor changes in abundances of other species in the presence of a new apex predator.
2. Estimate the relative abundance and growth of Smallmouth Bass in Ririe Reservoir to evaluate potential changes to current harvest rules.

METHODS

The fall of 2014 marked the fifth year of FWIN to monitor trends in the Walleye population in Ririe Reservoir. From October 22-24, we set six gill nets per night, for a total of 18 gill net nights of effort. Netting effort was based on FWIN protocol recommendations for water body size (Morgan 2002). Gill nets were 61 m long x 1.8 m deep, and consist of eight panels (7.6 m long) containing 25 mm, 38 mm, 51 mm, 64 mm, 76 mm, 102 mm, 127 mm, and 152 mm stretched mesh. The reservoir was divided into three strata (Upper, Middle. Lower), with 6 nets set at previously established sites in each stratum (Figure 36). FWIN protocol recommends stratifying net sets between two depth strata (shallow: 2-5 m; deep: 5-15 m). Steep shoreline topography limits the amount of shallow water habitat in Ririe Reservoir; therefore we set a combination of floating and sinking gill nets over a variety of depths (Appendix H).

We identified all fish collected with gill nets to species and recorded total length (mm) and weight (g). Additionally, we recorded sex and maturity of all Walleye captured, and collected otoliths and stomach samples for aging and diet analysis. We calculated proportional stock density (PSD) and relative stock density of preferred sized fish (RSD-P) for all game fish (Anderson and Neumann 1996).

We used pulsed DC boat-mounted electrofishing gear to sample the littoral fish community in Ririe Reservoir from June 24 to 26, 2014. We used two netters to collect fish and electrofishing surveys were started each night at dusk. A survey consisted of 10 minutes (600 seconds) of electrofishing at each site (IDFG staff 2012). For site selection, we stratified Ririe Reservoir into three strata (Upper, Middle, Lower). Sites were developed in each strata by measuring 500 m along the shoreline, which is approximately the maximum length of shoreline that can be electrofished in 10 minutes (IDFG staff 2012). The number of sites for the Upper, Middle, and Lower strata was 31, 32, and 25, respectively. Based on previous sampling of smallmouth (CPUE, variance) in 2005 and 2007, we estimated approximately 8 sites per strata (n = 24) were necessary to estimate CPUE within 80% confidence of 25% precision of the mean (Willis 1998). We used a random number generator to select the sites used in our sampling program. Fish collected were identified to species and measured for total length before being released to the general location of capture. We calculated electrofishing CPUE of for each species as fish per hour.

We estimated the number of sites required to monitor Smallmouth Bass CPUE within certain bounds (e.g. 80% confidence limits, ±25% of the mean) based on Willis (1998):

$$n = \frac{(t^2)(s^2)}{[(a)(x)]^2}$$

Where n = sample size required, t = t value from t-table, s^2 = variance, x = mean CPUE, a = precision in describing the mean as a proportion.

We removed the sagittal otoliths from Smallmouth Bass collected from electrofishing for age and growth analysis. After removal, all otoliths were cleaned on a paper towel and stored in individually-labeled envelopes. Ages were estimated by counting annuli under a dissecting microscope at 40x power. Otoliths were submerged in water and read in whole view when clear, distinct annuli were present. We sectioned, polished, and read otoliths in cross-section view with transmitted light when the annuli were not distinct in whole view. The von Bertalanffy (1938) growth model was used to fit length:

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

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. The model was fitted to length-at-data by using the nonlinear model (NLIN) procedure in program R. We estimated mortality rate (Z) for Smallmouth Bass from ages 2 to 8 by catch curve analysis. Age-1 Smallmouth Bass were excluded from the analysis due to lack of recruitment to our sampling gear.

RESULTS

We collected 1,996 fish in 21 net nights of effort in Ririe Reservoir, including eight Walleye. The mean percent catch across gill nets (± 90% CI) in 2014 was dominated by Yellow Perch (75 ± 10.8%) and non-game fish, mainly Utah Sucker (19 ± 8.6%) and Utah Chub (3% ± 2.3). Walleye only comprised 0.4% (± 0.7) of the relative abundance of our gill net catch. We captured a total of eight Walleye in the nets. Walleye CPUE (± 95% CI) per net-night was 0.4 ± 0.7 (Figure 37), which was similar to number of Walleye per net-night from 2010-2013 (0.4 per net-night; ± 0.24). Walleye ranged in length from 303 to 600 mm with a mean TL of 428 mm (± 100.2; Figure 38, Table 13). Mean relative weights (± 95% CI) of Walleye were 98 (± 8.3) and ranged from 88 to 120 (Figure 39). Walleye PSD and RSD-P were 63 and 25. Von Bertalanffy model results show Walleye grew from a starting age of t_0 = -0.923 years (95% CI -1.39 to -0.45) toward their asymptotic length of L_∞ = 650 mm (95% CI 605-692) at an instantaneous rate of growth (K) = 0.34/year (95% CI 0.24-0.44) (Figure 40). We analyzed diet of all Walleye captured; 4 stomachs were empty, while the remaining four stomach samples contained unidentifiable salmonids, with two prey fish vertebrate length measured at 130 mm and 101 mm. Total weight of stomach contents ranged from 0 g to 12.4 g (mean: 2.8 g; 95% CI 3.7).

We captured 83 Yellow Perch per net-night (n = 1,497; Figure 37) that ranged from 72 mm to 292 mm with a mean TL (± 95% CI) of 194 mm (± 1.0; Figure 41). Yellow Perch PSD was 49, while mean relative weights were 90 (± 0.5; Figure 42). We captured 1.6 kokanee per net-night (n = 27) that ranged from 162 mm to 382 mm with a mean TL of 189 mm (± 16.2; Figure 40). Kokanee PSD and RSD-P values were 3.6 and 3.6, respectively. Kokanee mean relative weights were low at 77 (± 9.4). We captured 0.9 Yellowstone Cutthroat Trout per net-night (n = 17) that ranged from 222 mm to 485 mm with a mean TL of 347 mm (± 31.9). Yellowstone Cutthroat Trout PSD and RSD-P values were 41.2 and 5.9, respectively. Yellowstone Cutthroat Trout mean relative weights were relatively low at 81 (± 4.9). We only

captured two Smallmouth Bass, which was 0.1 fish/net-night. Smallmouth Bass were 181 mm and 293 mm in length and had relative weights that were 95 and 77, respectively. We captured 0.5 Rainbow Trout per net-night (n = 10) that ranged from 257 mm to 387 mm with a mean TL of 316 mm (± 23.8). Rainbow Trout PSD and RSD-P values were both 0. Rainbow Trout mean relative weights were low at 79 (± 4.9).

We collected 139 Smallmouth Bass, 74 Yellow Perch, and 24 Rainbow Trout from shoreline boat electrofishing. Total electrofishing effort was 4 hours across the 24 sites. Smallmouth Bass mean length (± 95% CI) was 158 mm (± 8.0; range 81-302 mm; Figure 43), with a mean CPUE of 35 bass per hour (95% CI 14.4; Figure 44). Yellow Perch mean length was 141 mm (± 5.8; range 73-182 mm), with a mean CPUE of 19 perch per hour (± 24.0). Rainbow Trout mean length was 288 mm (95% CI ± 12.8; range 221-340 mm), with a mean CPUE of 5 trout per hour (± 3.8). Mean relative weights of Rainbow Trout, Smallmouth Bass, and Yellow Perch were 94 (± 3.3), 94 (± 4.4), and 101 (± 1.6), respectively (Figure 45). Based on power analysis from 2014, we estimated the sample sizes needed to calculate CPUE for Smallmouth Bass within 80% confidence limits and ± 25% of the mean to be 24 sites (or 27% of the total sites), and 90% confidence limits and ± 25% of the mean to be 42 sites (or 48% of the total sites).

We aged 126 Smallmouth Bass otoliths, and estimated ages that ranged from 0 to 8 years between the lengths of 81 to 302 TL mm (Table 14). We compared ages to statewide averages developed by Dillon (1996) and with the exception of age-1 Smallmouth Bass, length-at-age was much lower than the historic statewide average (Figure 46). Smallmouth Bass grew from a starting age of t_0 = -1.47 years (95% CI -2.16 to -0.77) toward their asymptotic length of L_∞ = 502 mm (95% CI 285-719) at an instantaneous rate of growth (*K*) = 0.10/year (95% CI 0.03-0.17). Smallmouth Bass mortality from age two to eight was estimated at 65% annually using catch curve analysis (Figure 47).

DISCUSSION

Similar to years past (2010-2013) Walleye continue to comprise a small proportion of the overall species composition (<1%) in the gill net catch. Walleye gill net CPUE was identical to the average number of Walleye per net-night from 2010-2013. Low and stable catch rates suggest the Walleye population is persisting at low abundance levels, and that the population is not rapidly expanding. The identification of five different year classes of Walleye suggests yearly recruitment likely occurs albeit at low levels. A telemetry study conducted in 2009-2010 found that the majority of Walleye spawning activity likely occurred within the lower 2 km of Willow Creek between mid-April and mid-May (Schoby et al. 2014). Population expansion of Walleye may be limited by adequate availability of spawning habitat or perhaps other abiotic and biotic factors. The high growth rates of Walleye compared to North American average estimated by Quist et al. (2003) along with the good relative weights (Mean 98; 95% CI 8.3), indicates a readily available prey base exists for Walleye. Surprisingly, diets of the Walleye were comprised of salmonids, even though salmonids were collected in low numbers in the gill nets compared to Utah Chub, Utah Sucker, and Yellow Perch. Currently, we only stock kokanee and Rainbow Trout into Ririe Reservoir. If the Walleye population expands, predation on salmonids may result in losses of valuable stocked fish with such a top predator (Lepak et al. 2014). The salmonid fishery is an important component in Ririe Reservoir with angler catch for salmonids comprising 49% of the total angler catch in creel surveys in 2013 (High et al. 2015). Additional predation and consequent reduction in salmonid abundances has the potential to impact angler catch rates and success. 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 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 prey heavily on salmonids, stocking of these fish could become unfeasible. Continued yearly monitoring of the Walleye population is necessary to determine factors that drive Walleye populations, and to monitor impacts to valuable sportfish populations.

We compared growth of Smallmouth Bass to the statewide average developed by Dillon (1996) and found Smallmouth Bass growth was much slower in Ririe Reservoir. The lower growth rates are likely due to the differences in latitude with the shorter growing season (i.e. lower water temperatures) in southeastern Idaho. The slow growth rates may limit the effectiveness of imposing a minimum length limit. Minimum length limits prohibit harvest of fish less than a specified length and have been implemented to reduce harvest and improve catch rates (i.e. increased abundances) and size structure or as a tool to ensure fish remain in the system long enough to spawn before being available to angler harvest. Often reductions in harvest may not be popular among harvest-oriented anglers, but are very popular with catch and release anglers. A minimum length limit would likely improve the size structure of the fishery if growth was relatively fast and angling mortality was high. Conversely, slow growth as observed in this study might prevent improvements in size structure if most fish are lost to natural mortality before they reach minimum length. We estimated mortality at 65%, which suggests high natural mortality. The slow growth rate of Smallmouth Bass along with their high natural mortality indicates the current length limit is not necessary to protect juvenile bass until maturity, and it is not improving the size structure of the existing population of bass. Liberalizing the bass limit and allowing some harvest would provide additional opportunity for harvest-oriented anglers. Based on creel surveys in 2013, anglers indicated their support and opposition for the removal of the current 300 mm minimum length limit on bass at 32% and 30%, respectively (High et al. 2015). The remaining 38% of the anglers were neutral in their responses on the removal of the length limit. Given the indifference or strong objections for or against by anglers in maintaining the minimum size limit, lack of justification from a biological perspective, and the unnecessarily complexity such a regulation creates strongly suggests that removing the size limit restriction is warranted.

Although we did not collect Smallmouth Bass >305 mm TL in our electrofishing surveys, past surveys indicate Smallmouth Bass are present above the minimum size limit but in limited abundance. For example, creel surveys in 2013 estimated 8% of the Smallmouth Bass caught were harvested, but this survey did not determine if the harvested bass were of legal size. Gill netting efforts in 2013 also collected two Smallmouth Bass 341 and 421 mm TL. Electrofishing generally selects for larger sized fish due their larger surface area. However, the steep canyon walls in Ririe Reservoir may make littoral zone electrofishing less efficient than electrofishing in shallower environments. Also, larger Smallmouth Bass may be occupying deeper waters that we were ineffective in sampling with electrofishing, which could result in overestimating annual mortality rates. Power analysis found at least 24 sample sites are necessary for future monitoring trends to acquire Smallmouth Bass CPUEs that are within 80% confidence limits and ± 25% of the mean. Sampling at a higher level than this is not feasible with current manpower and funding availability, nor is it necessary to obtain adequate data to manage the bass population.

RECOMMENDATIONS

1. Continue annual gill net monitoring (FWIN) to continue monitoring abundance, growth, mortality, reproduction, and foraging behavior of Walleye.

2. Collect biological information on all fish (including non-game species) captured during FWIN monitoring to determine impacts from Walleye establishment.

3. Evaluate stocking rates of kokanee to provide maximum benefits to anglers.

4. Continue stocking sterile Rainbow Trout.

5. Develop sampling methods and protocols for monitoring kokanee population trends over time.

Table 13. Summary statistics for Walleye captured during 2014 FWIN in Ririe Reservoir.

Date	Net type	Net (#)	Mesh size (mm)	FL (mm)	TL (mm)	Weight (g)	Sex	Maturity	Age	Visceral Fat (g)	Gonad (g)
October 29	Floating	11	64	--	303	254	UNK[a]	Immature	1	6.7	N/A
October 30	Sinking	1	51	384	404	650	M	Mature	2	36.45	18.05
October 30	Sinking	1	127	568	600	2356	M	Mature	7	82.73	79.44
October 30	Sinking	1	25	565	594	2052	M	Mature	6	88.15	64.17
October 30	Sinking	1	64	460	479	1416	F	Mature	3	67.66	76.17
October 30	Sinking	1	51	381	412	727	M	Mature	2	48.24	15.34
October 30	Sinking	4	64	307	325	313	M	Mature	1	7.56	7.92
October 30	Sinking	1	51	295	310	300	UNK	Immature	1	10.35	N/A

[a] UNK: Unknown sex

Table 14. Mean length at age data based on otoliths of Smallmouth Bass collected via electrofishing in Ririe Reservoir, Idaho 2014. Ages were estimated using non-linear regression.

	Age								
Statistic	**0**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Mean TL (mm)	70	112	150	184	215	242	268	290	311
Lower 95% CI	58	106	146	179	209	236	259	279	294
Upper 95% CI	82	118	153	189	220	249	276	303	329
No. Analyzed	2	20	71	10	8	3	7	3	2

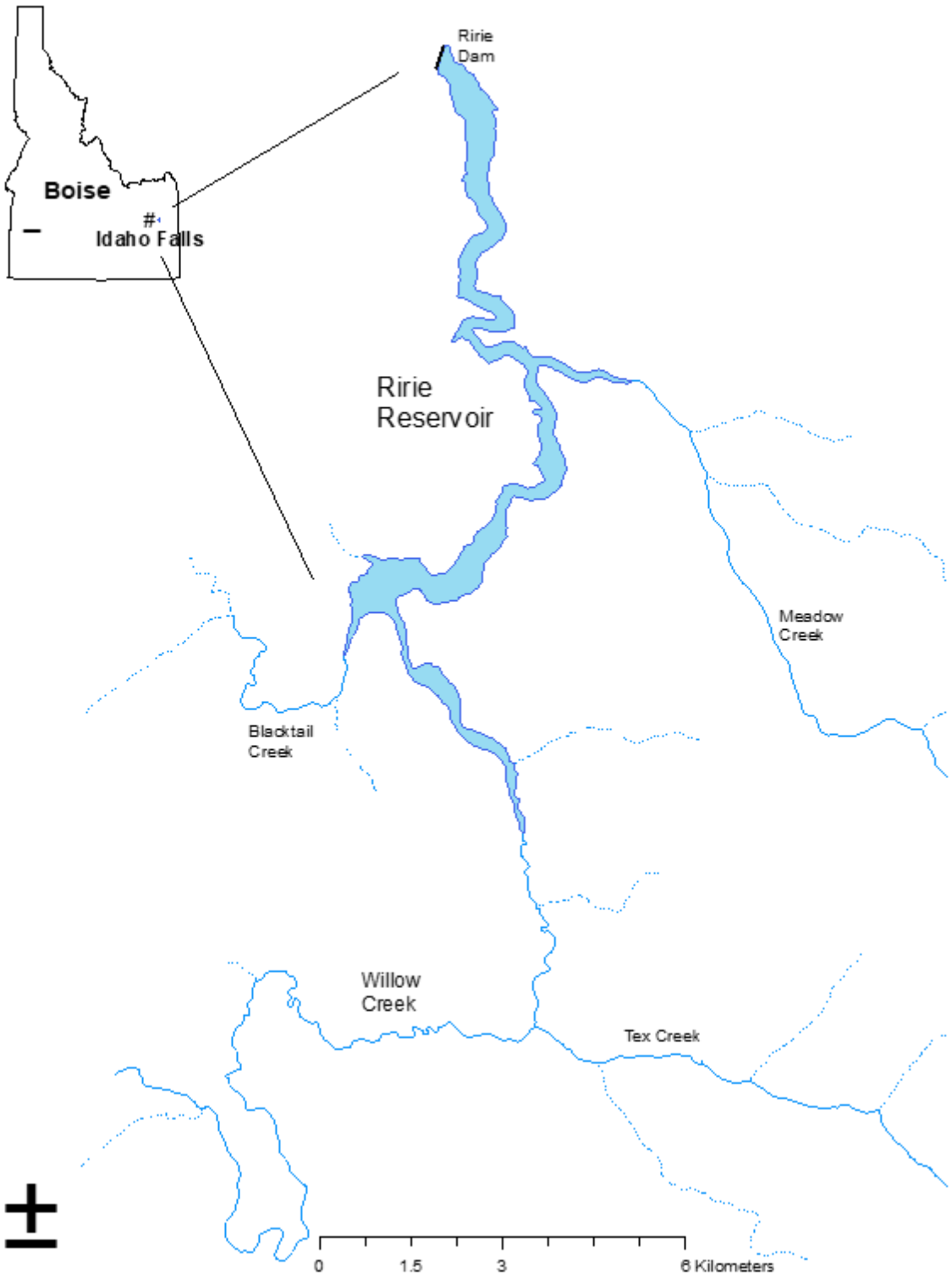

Figure 35. Location of Ririe Reservoir and major tributaries.

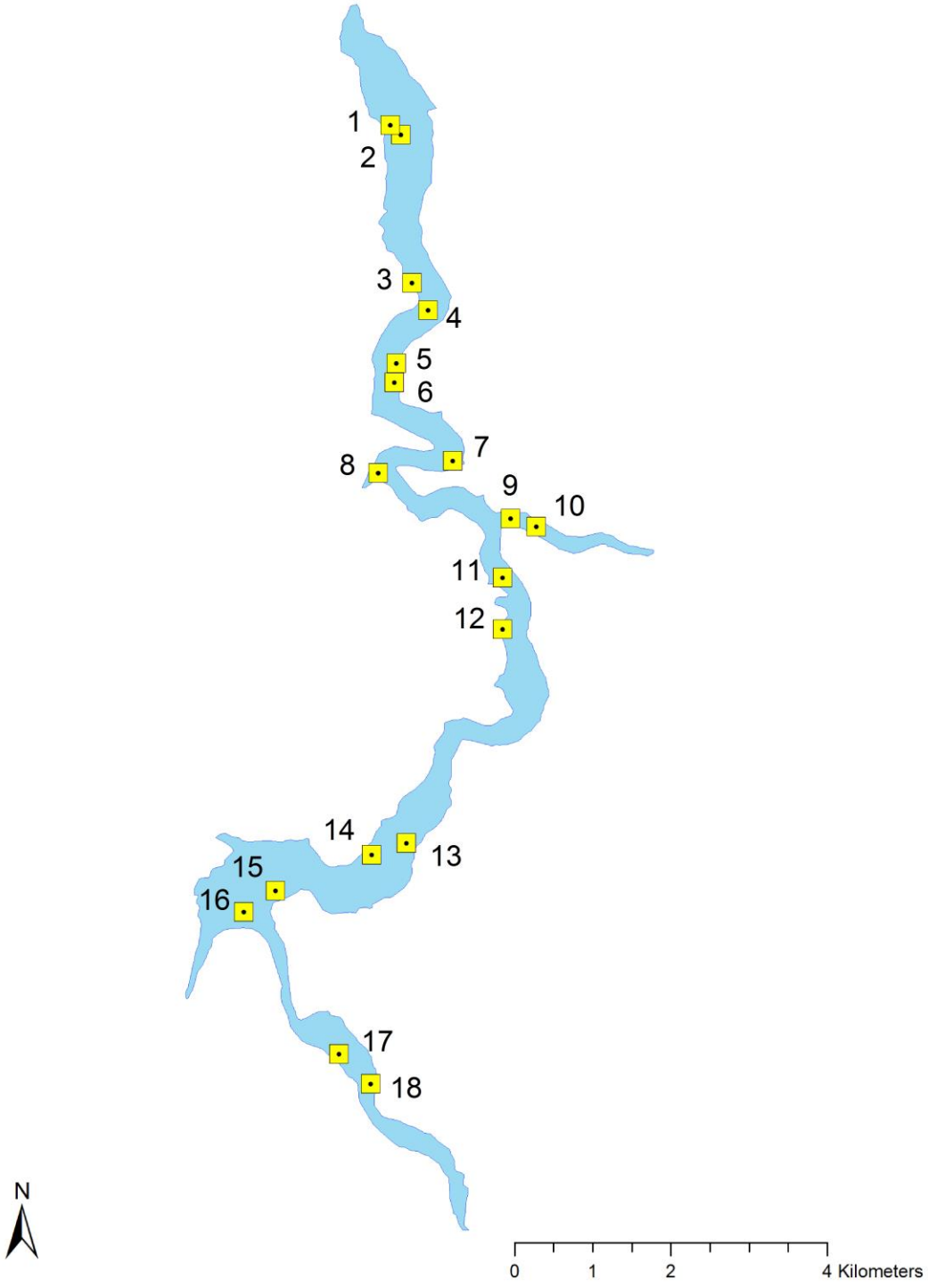

Figure 36. Location of 2014 fall Walleye index netting (FWIN) in Ririe Reservoir.

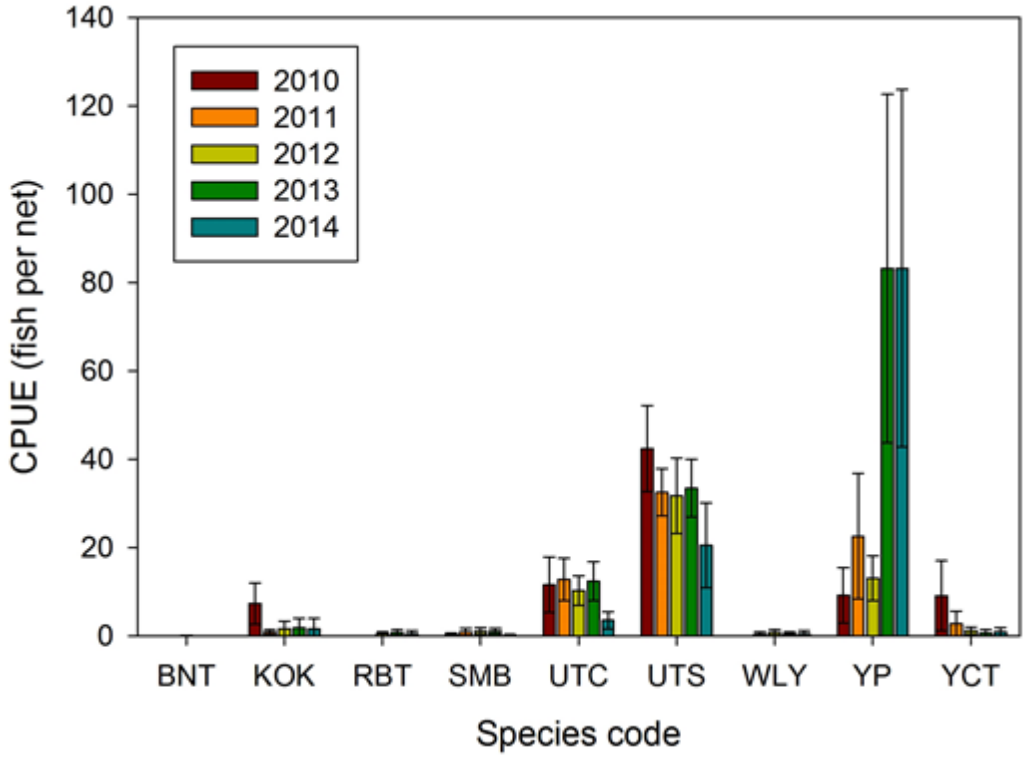

Figure 37. Mean catch per unit effort (fish/net), for 18 net nights of FWIN in Ririe Reservoir, during 2010-2014. Error bars represent 95% confidence intervals.

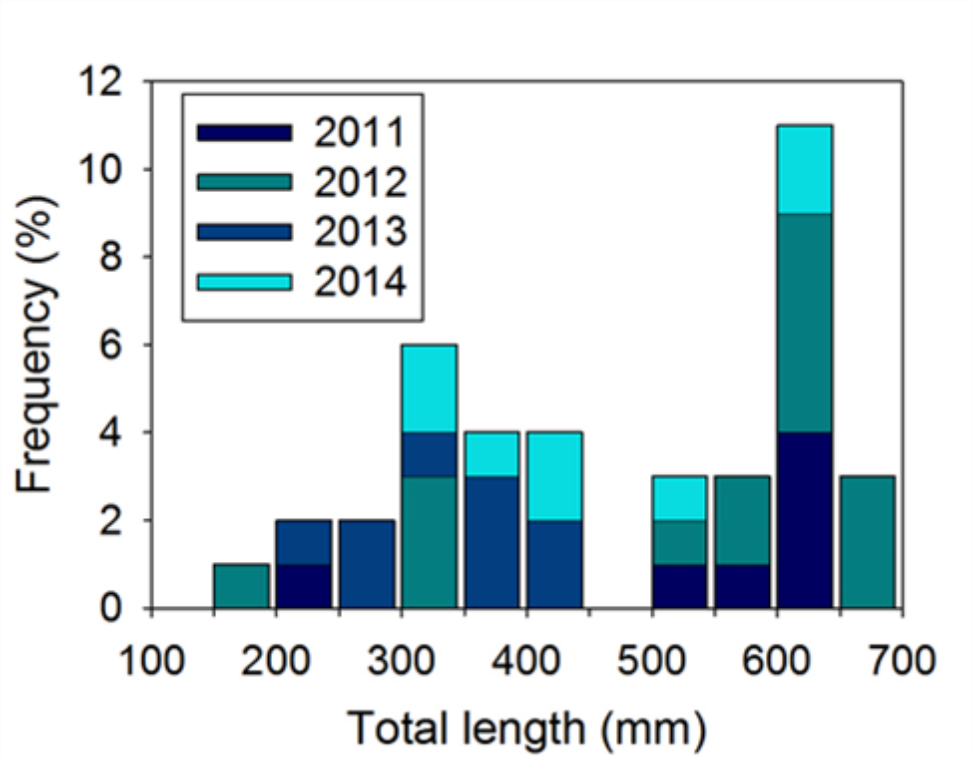

Figure 38. Length frequency of Walleye captured in gill nets during 2011-2014 FWIN in Ririe Reservoir.

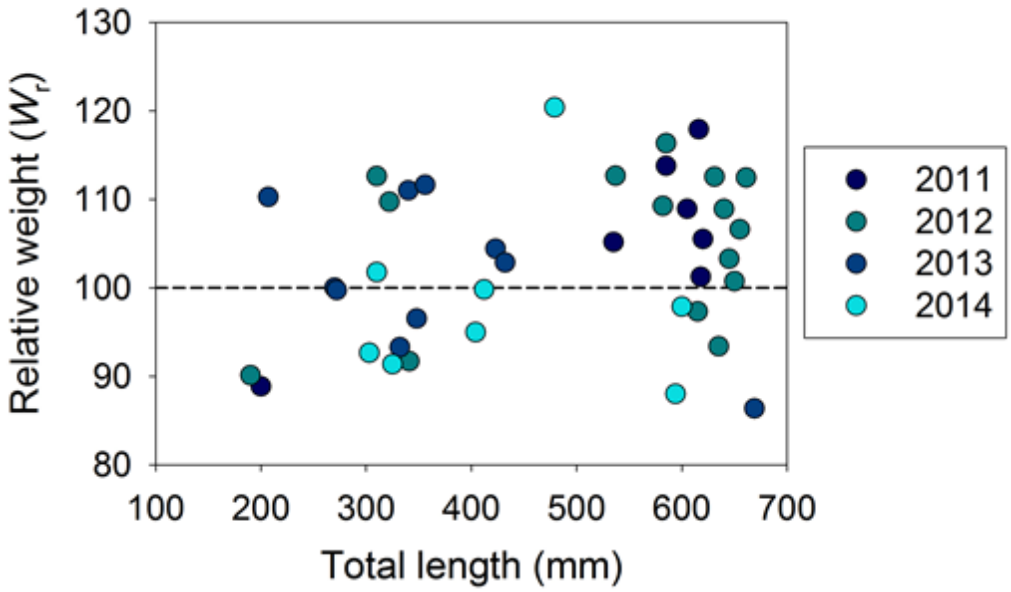

Figure 39. Walleye relative weights by total length collected in gill nets from FWIN surveys in 2011, 2012, 2013, and 2014.

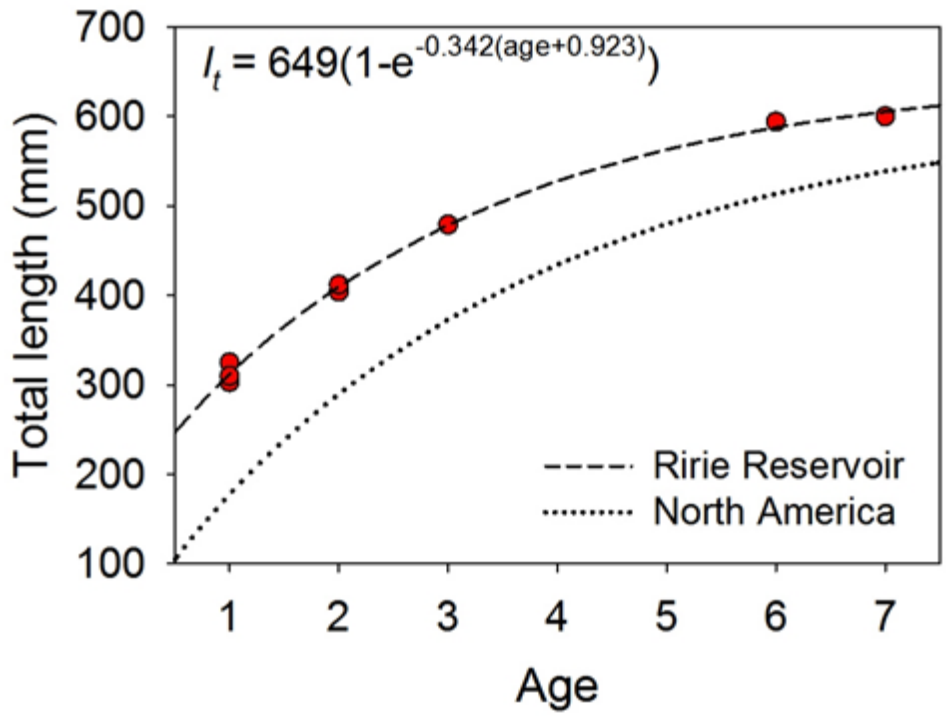

Figure 40. Walleye length-at-age based on non-linear regression in Ririe Reservoir, 2014. Walleye growth in Ririe Reservoir represents the fitted von Bertalanffy growth model (dashed line) compared to the North America average of both male and female Walleye ($l_t = 610(1-e^{-0.300(age+0.148)})$) developed in Quist et al. (2003).

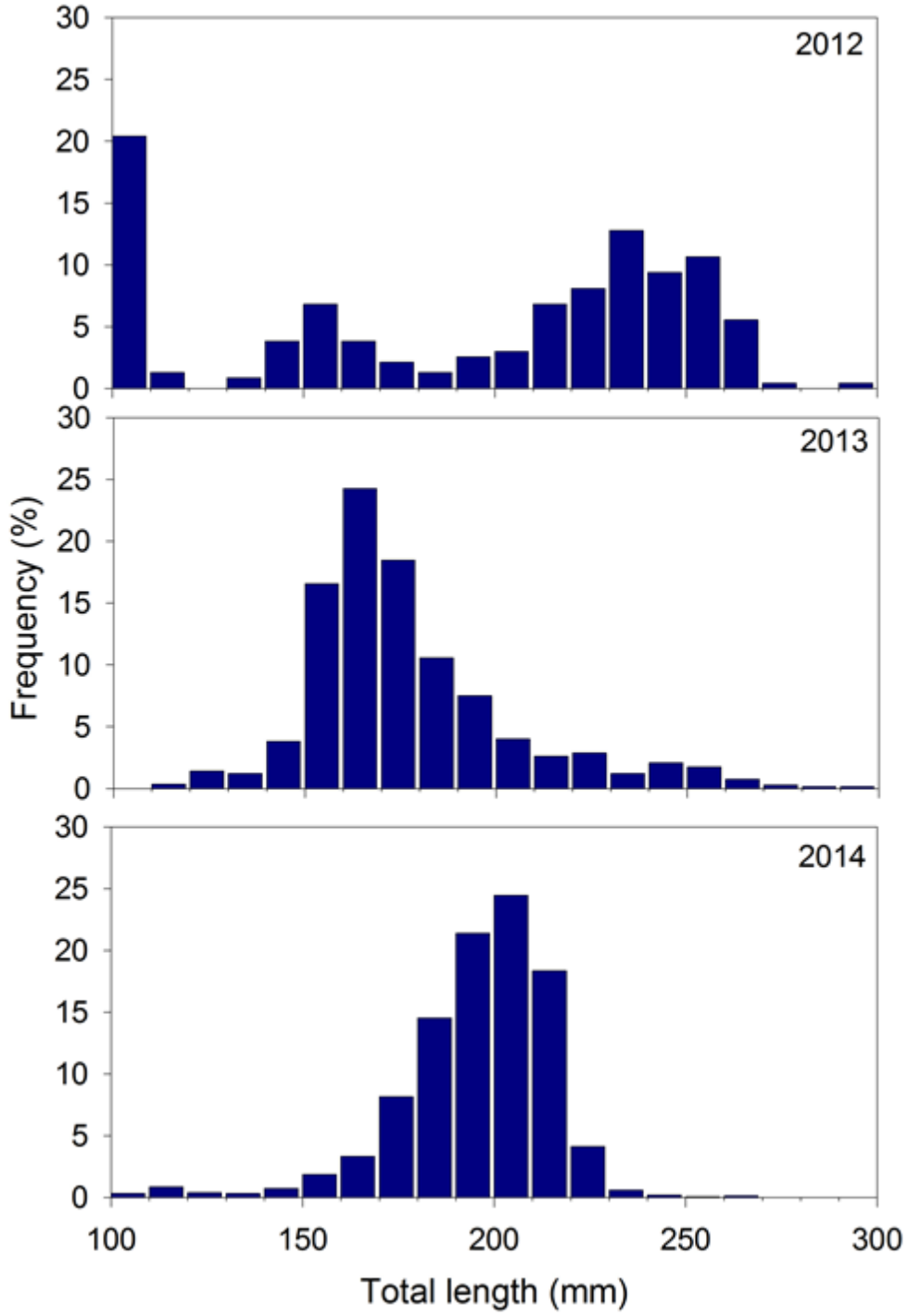

Figure 41. Length frequency of Yellow Perch captured during 2012-2014 FWIN in Ririe Reservoir.

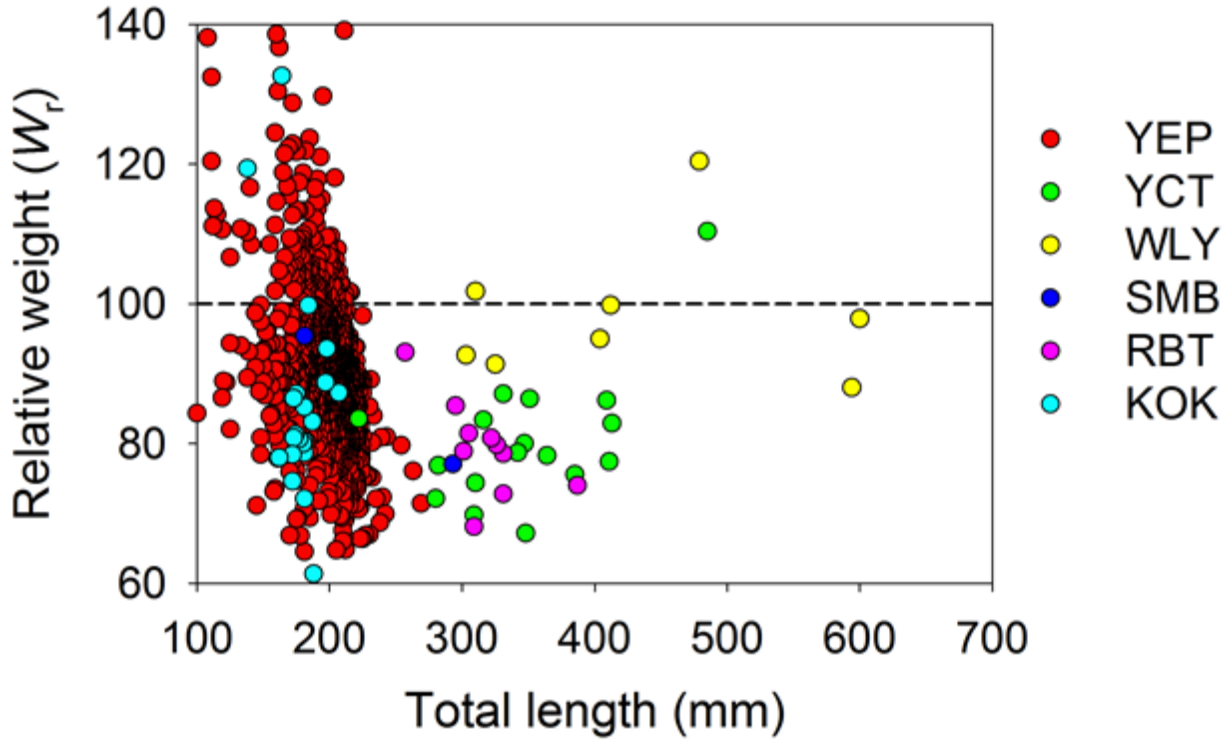

Figure 42. Relative weights by total length of Kokanee (KOK), Rainbow Trout (RBT), Smallmouth Bass (SMB), Walleye (WLY), Yellowstone Cutthroat Trout (YCT), and Yellow Perch (YEP) collected in gill nets from FWIN surveys, 2014.

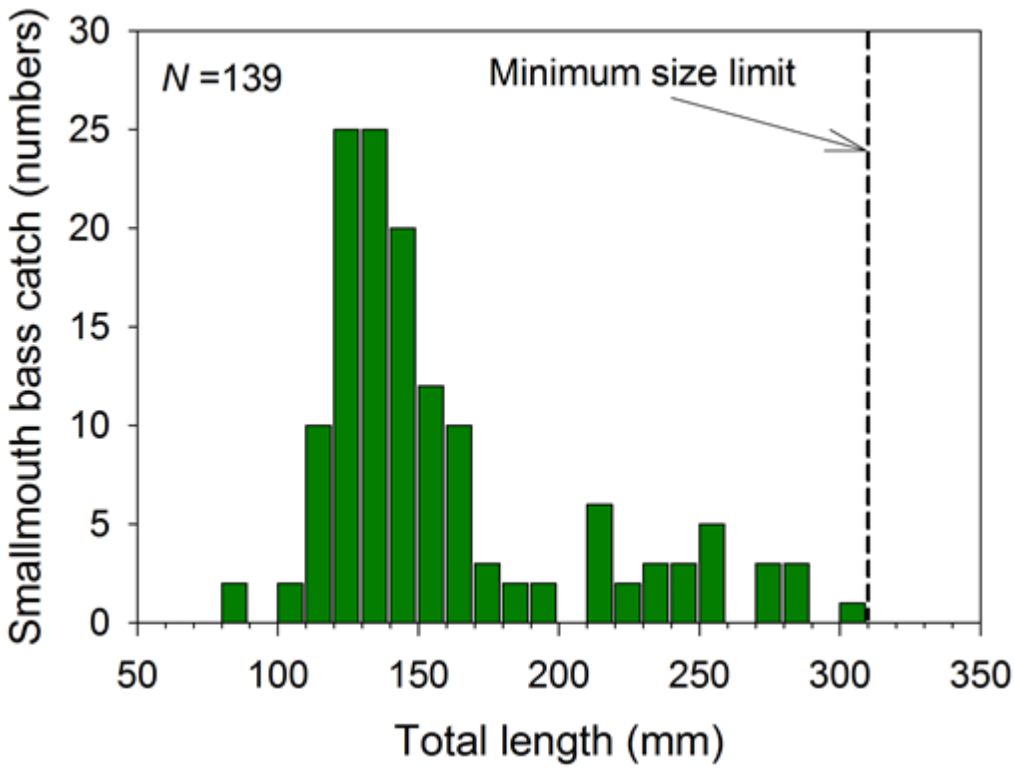

Figure 43. Length frequency histogram of Smallmouth Bass from nighttime electrofishing in Ririe Reservoir in 2014. Dashed line indicates the current minimum size limit (>305 mm) regulation for the Smallmouth Bass fishery.

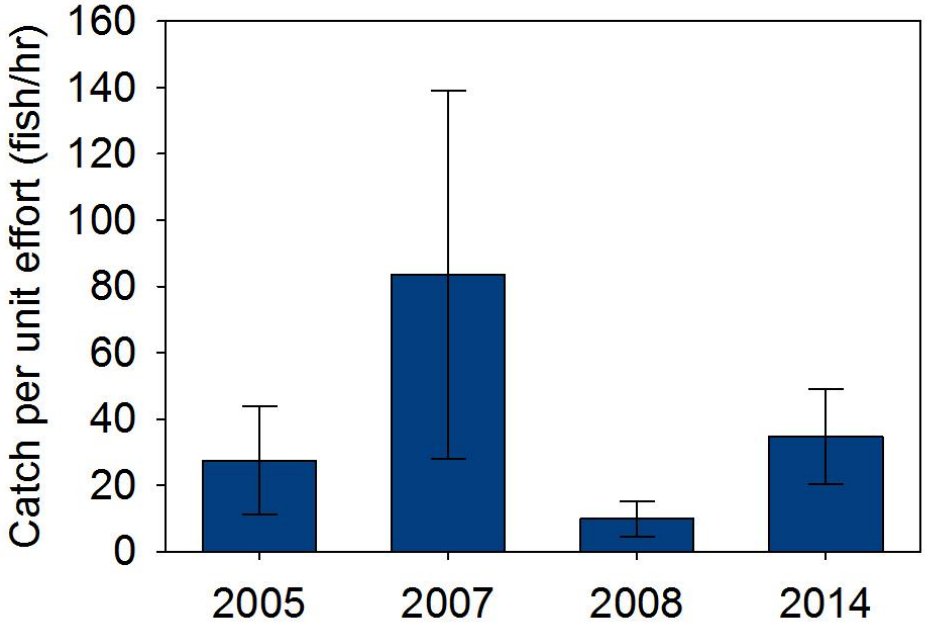

Figure 44. Catch per unit effort (fish/hr) with 95% confidence intervals for Smallmouth Bass sampled with night boat electrofishing in Ririe Reservoir in 2005, 2007, 2008, and 2014.

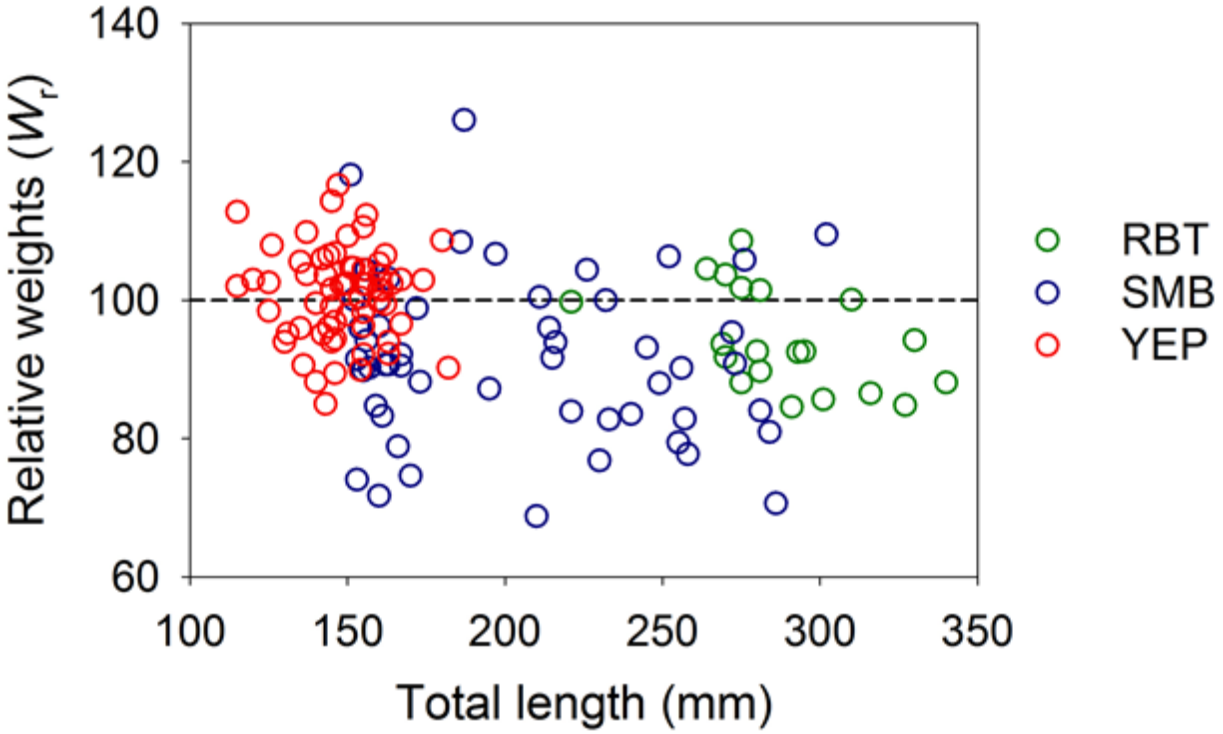

Figure 45. Relative weights by total length of Rainbow Trout (RBT), Smallmouth Bass (SMB), and Yellow Perch (YEP) collected from summer boat shoreline electrofishing surveys.

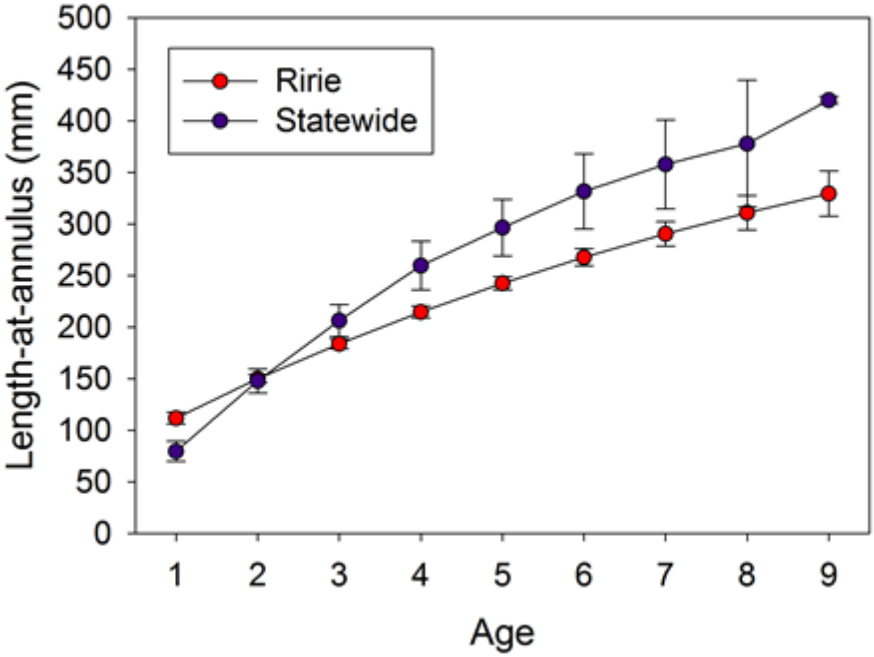

Figure 46. Comparison of Smallmouth Bass length-at-age with 95% CI estimated using non-linear regression to historic statewide age estimates developed by Dillon (1996) from 11 Idaho lakes and reservoirs.

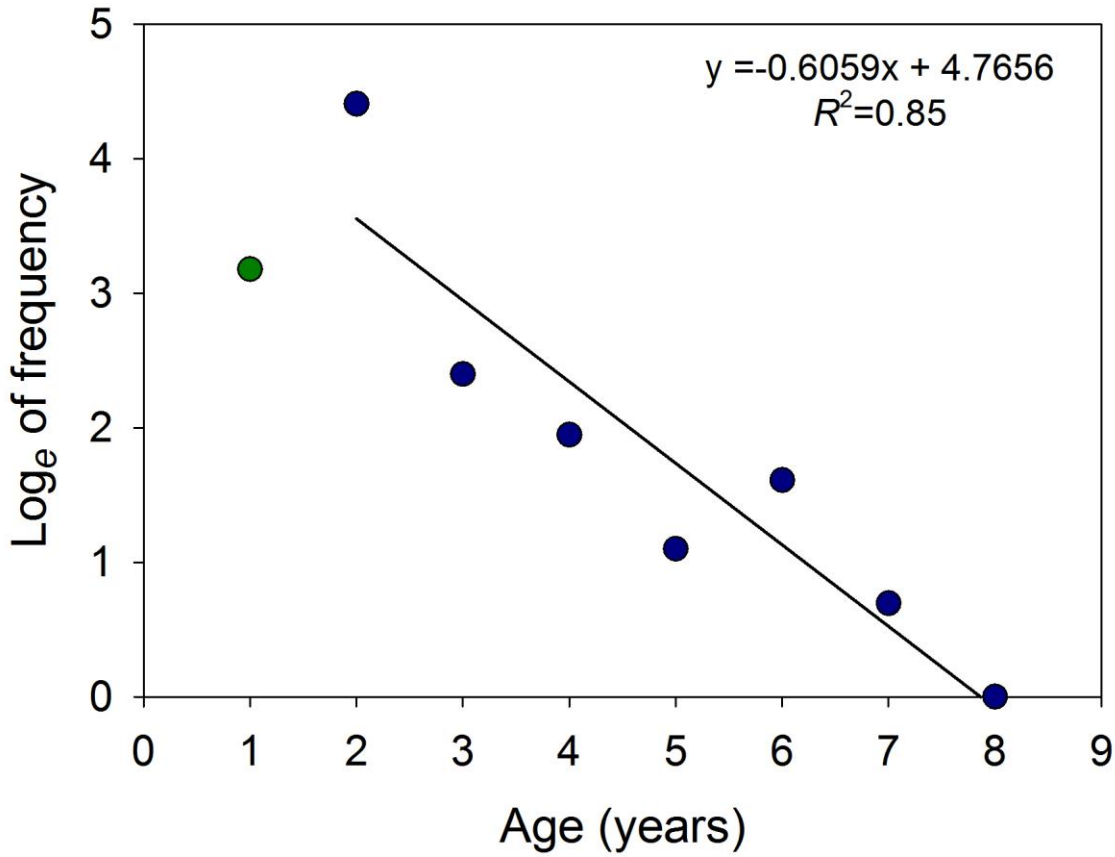

Figure 47. Catch curve for Smallmouth Bass sampled with boat electrofishing in Ririe Reservoir in 2014. Solid line represents the linear regression of the expected catch of Smallmouth Bass from ages-2 - 8 (blue circles). Age-1 (green circle) was excluded from the analysis due to lack of gear recruitment.

ZOOPLANKTON MONITORING

ABSTRACT

We monitored zooplankton abundance and biomass in five regional lakes and reservoirs to assess the forage resources and evaluate stocking rates where applicable. We assessed the cropping impacts by fish using the zooplankton ratio method (ZPR) and determined that with the exception of Mackay Reservoir, preferred zooplankton were not being cropped by fish in any of the waters sampled. Mackay Reservoir currently has a stunted kokanee population resulting in limited production of larger zooplankton. We used the zooplankton quality index (ZQI) to assess the overall abundance of preferred zooplankton, and ZQI values in 2014 across the region were generally stable when compared to previous years. Similar to previous year's findings, the high variability between zooplankton samples limits our ability to detect changes in the zooplankton abundance across years, which suggests a rigorous evaluation of sampling protocols is warranted.

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INTRODUCTION

Zooplankton provides the forage base of aquatic food webs in lakes and reservoirs and influences fish growth in lentic environments for planktivorous fish (Luecke et al. 1990). In Idaho lakes and reservoirs, Dillon and Alexander (1996) observed that the presence of large zooplankton was directly linked to the success and survival of fall hatchery trout fingerling stocking. However, fish stocking programs often fail to provide basic zooplankton monitoring data in evaluations of stocking rates. Estimates of zooplankton abundances can be used to help evaluate hatchery trout stocking programs by providing an indication of the relative production potential of a water body and potential availability of preferred zooplankton as a food source for stocked fish. In lakes and reservoirs with self-staining wild fish populations, monitoring zooplankton abundances may provide insight into forage availability and possibly survival of fish species of interest at younger ages.

METHODS

We collected zooplankton samples from five lakes and reservoirs throughout the Upper Snake Region during 2014 (Figure 48), following the protocol described by Teuscher (1999). We collected zooplankton samples between July 22 and 28 from Henrys Lake, Island Park Reservoir, Mackay Reservoir, Palisades Reservoir, and Ririe Reservoir. During each sampling event, we collected samples from three different locations spread around the lake or reservoir. We collected samples with three nets fitted with small (153 µm), medium (500 µm), and large (750 µm) mesh. We preserved zooplankton in denatured ethyl alcohol at a concentration of 1:1 (sample volume:alcohol). After ten days in alcohol, phytoplankton were removed from the samples by re-filtering through a 153 µm mesh sieve. The remaining zooplankton were blotted dry with a paper towel and weighed to the nearest 0.1 g. Biomass estimates were corrected for tow depth and reported in g/m. We estimated the relative production potential of each lake by estimating overall zooplankton biomass collected from the 153 µm net. We measured competition for food (or cropping impacts by fish) using the zooplankton productivity ratio (ZPR) which is the ratio of preferred (750 µm) to usable (500 µm) zooplankton. We also calculated the zooplankton quality index (ZQI) to account for overall abundance of zooplankton using the formula developed by Teuscher (1999) where ZQI = (500 µm + 750 µm) × ZPR.

ZQI values obtained from zooplankton monitoring are used to assess stocking rates based on the recommendations from Teuscher (1999) (Table 15). Additionally, we calculated 95% confidence intervals for zooplankton data (ZQI) from previous years to determine if the current levels of zooplankton sampling were adequate to detect changes in zooplankton abundance throughout the region.

RESULTS AND DISCUSSION

Throughout the Upper Snake Region, mean zooplankton biomass from the 153 µm net ranged from 0.62 g/m (Island Park Reservoir) to 1.56 g/m (Henrys Lake) (Table 16). Teuscher (1999) recommends conservative fingerling stocking densities in water bodies with mean ZQI estimates <0.10, as the necessary forage to support higher densities is lacking. During 2014, Mackay Reservoir zooplankton biomass estimates for ZQI values were less than 0.10, suggesting low zooplankton forage availability. Low biomass of zooplankton in Mackay Reservoir is likely limited by the high density of naturally reproducing kokanee Salmon. Based on population surveys conducted in 2013, the kokanee population appeared to be stunted with few fish observed over 300 mm TL (<10%) (High et al. 2015). Kokanee are an obligate

planktivore with small gill raker spacing and are effective zooplankton predators (Brunger-Lipsey and Stockwell 2001). Water management in the reservoir has shifted towards maintaining higher water volume in the fall drawdown period, resulting in higher survival and recruitment in the kokanee population with a resulting decrease in fish size. The current stocking strategy at Mackay Reservoir for Rainbow Trout is larger catchable trout (~10”). The lower zooplankton abundances may not be as problematic for catchable-sized trout mainly used as a put-and-take, but stocking fingerling trout in Mackay Reservoir would likely result in poor survival with the current zooplankton abundances. In 2014 the fall stocking rate of Rainbow Trout fingerlings in Island Park Reservoir was 23 fish per acre. The high ZQI values observed in Island Park Reservoir (>1.0) suggests stocking higher densities of trout fingerlings up to at least four times above the current levels is possible, if deemed appropriate and necessary by managers. However, during the fall and winter the reservoir levels remain low after irrigation drawdown and may result in reduced zooplankton abundances. ZQI values would need to be obtained throughout the fall to determine whether sufficient zooplankton abundances exist for increased densities of fingerlings. The moderate ZQI values observed in Henrys Lake suggests a stocking rate of 75-150 fingerlings per acre. In 2014 we stocked approximately 979,000 Brook, hybrid, and Yellowstone Cutthroat Trout fingerlings, which was a stocking rate of 150 trout fingerlings per acre. Based on ZQI value the current stocking rate in Henrys Lake was near the upper range recommended to maximize survival of fingerlings.

The high variability in yearly ZQI values based on the 95% confidence intervals suggests that ZQIs may provide limited inferences for trend monitoring of zooplankton abundance at current sampling levels (Figure 49). The likely spatial and temporal variability of zooplankton populations within each lake or reservoir allows for a high degree of variability between samples. A more intensive (i.e. increased samples) based on power analysis, spatially or temporally designed sampling protocol would be required for meaningful long-term monitoring. However, the logistical constraints of sampling each water body every year are currently unfeasible. A sampling approach with more intensive sampling ever 3-5 years per lake or reservoir will likely be more feasible in the future and provide a reasonably robust measure of the current zooplankton productivity and assist in making stocking decisions than the current less intense sampling strategy with a high degree of variability.

RECOMMENDATIONS

1. Develop strategy to effectively identify the errors in the current sampling strategy.

2. Design statistically appropriate guidelines for zooplankton monitoring regional waters.

Table 15. Zooplankton quality index (ZQI) ratings and the recommended stocking rates from Teuscher (1999).

ZQI	Stocking recommendation
>1.0	High density fingerlings (150 – 300 per acre)
<1.0, >0.1	Moderate density fingerlings (75 – 150 per acre)
<0.1	Low density fingerlings (< 75 per acre) or stock catchables

Table 16. Mean zooplankton biomass (g/m) by net mesh size (µm), preferred to usable (750:500) zooplankton ratio (ZPR), and zooplankton quality index (ZQI = [500 µm +750 µm] × ZPR) with 95% confidence intervals in parenthesis for lakes and reservoirs in the Upper Snake Region of Idaho, July 2014.

	Net mesh (µm)				
Waterbody	**153**	**500**	**750**	**ZPR**	**ZQI**
Henrys Lake	1.56	0.89	0.39	0.41 (±0.1)	0.58 (±0.4)
Island Park Reservoir	0.62	0.74	0.59	1.07 (±0.6)	1.40 (±0.7)
Mackay Reservoir	0.64	0.11	0.05	0.39 (±0.2)	0.07 (±0.1)
Palisades Reservoir	0.78	0.41	0.31	0.78 (±0.7)	0.56 (±0.6)
Ririe Reservoir	0.63	1.03	0.50	0.60 (±0.6)	0.75 (±0.9)

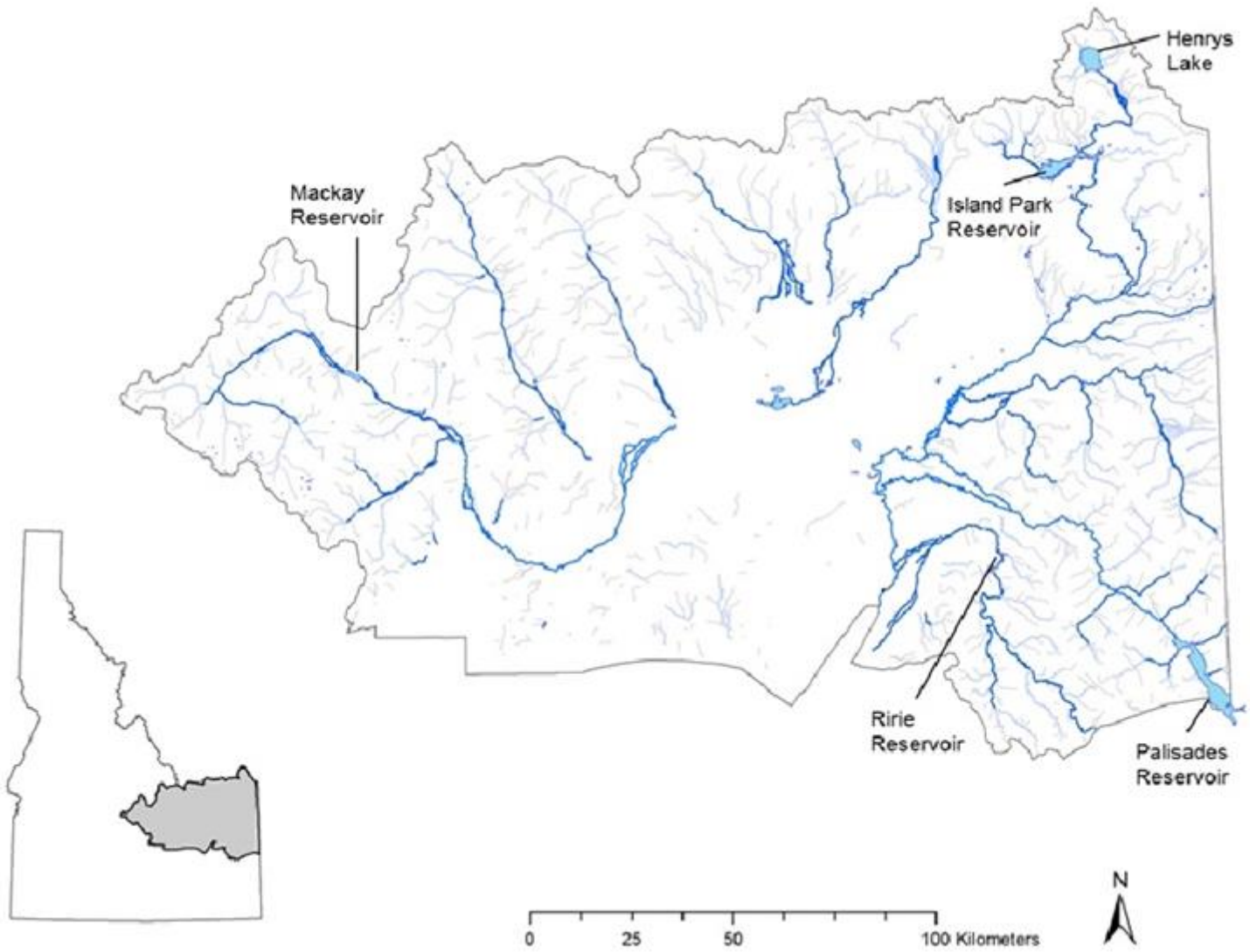

Figure 48. Zooplankton sample locations in the Upper Snake Region, 2014.

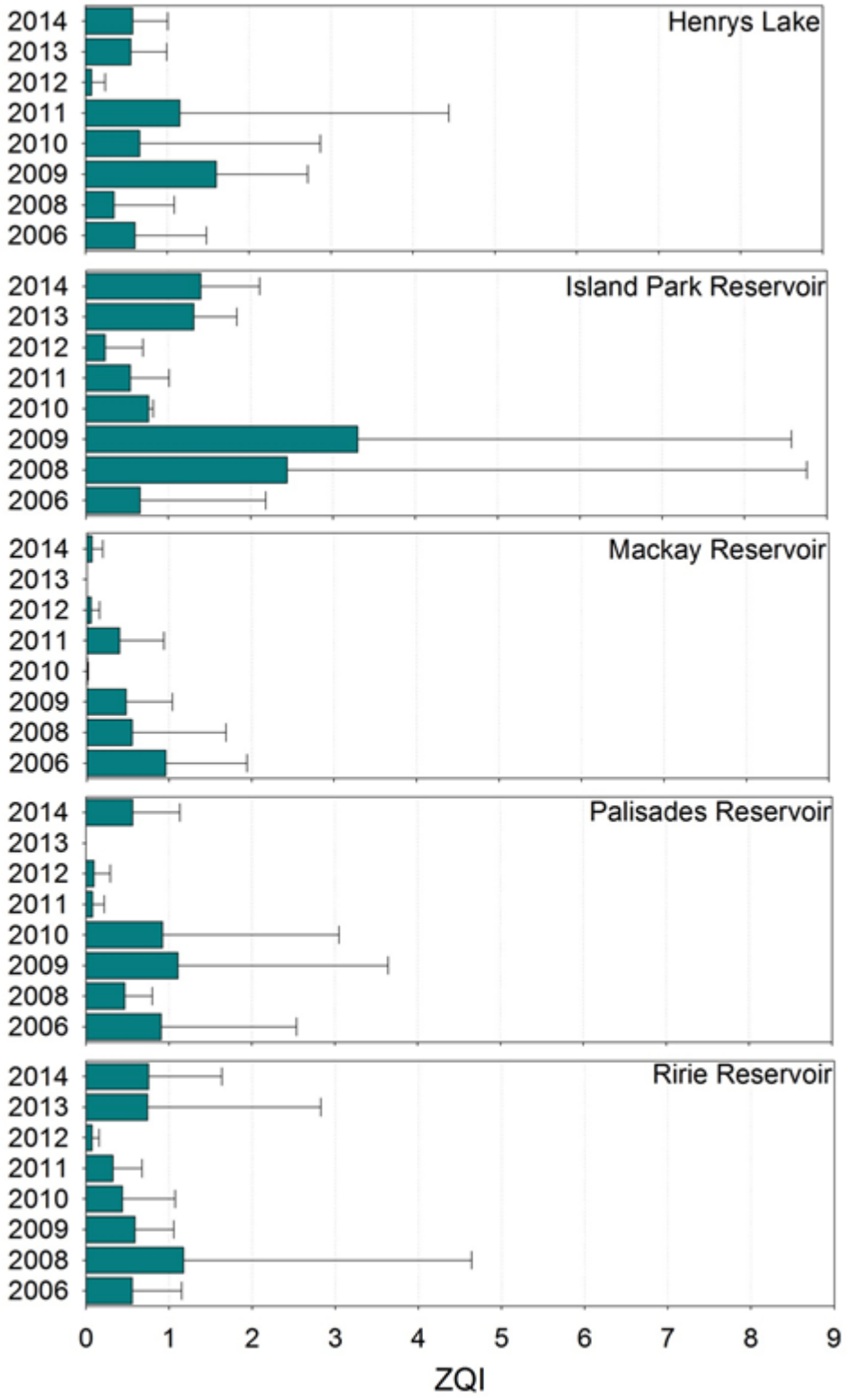

Figure 49. Zooplankton quality index (ZQI) values, with 95% confidence intervals, for lakes and reservoirs in the Upper Snake Region, from 2006-2014.

HENRYS FORK

ABSTRACT

We used boat mounted electrofishing equipment to assess fish populations in the Box Canyon and St. Anthony reaches of the Henrys Fork Snake River during 2014. In Box Canyon, Rainbow Trout *Oncorhynchus mykiss* densities were substantially lower (1,575 fish per km (95% CI ± 91) when compared to 2013 (3,881 trout/km; 95% CI ± 311), which were the highest observed density since 1994. Mean total length of Rainbow Trout increased almost 50 mm in 2014 than in 2013 due to the strong year class of age-3 currently present in the population. Overall, Rainbow Trout densities in 2014 were slightly lower than the 20-year average (1,945 trout per km). The effects of winter flows on Rainbow Trout first-winter survival continue to be significantly related, predicting age-2 abundance similar to our population estimates. Using the model of winter flows to predict age-2 trout abundances, we estimated there should have been 3,348 rainbow trout, and population estimates found 2,575. In the St. Anthony reach, we estimated 716 trout per km (95% CI ± 89) with a species composition of 83% Brown Trout *Salmo trutta* and 17% Rainbow Trout. Trout densities in the St. Anthony reach have increased more than 200% compared to spring surveys conducted in 2004 and 2010. Analysis of length frequency figures suggests two distinct year classes for Rainbow Trout and three-year classes for Brown Trout in St. Anthony reach. The trout population appears to be improving in this reach despite a liberal six trout limit in the current fishing regulations.

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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. An economic survey conducted in 2003 showed that Fremont County, which encompasses a large portion of the Henrys Fork drainage, ranked first out of the 44 counties in Idaho in terms of angler spending, and generated nearly $51 million for the local economy (Grunder et al. 2008). Similarly, an IDFG economic survey in 2011 estimated that anglers fished 165,236 days in Fremont County and spent nearly $62 million during angling trips (IDFG, unpublished data).

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. Below 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 above Island Park Reservoir provides a family fishery primarily supported by stocked trout. The fishery is also supported in part by trout that move out of Henrys Lake or Island Park Reservoir. Management of the Henrys Fork downstream of Island Park Dam emphasizes wild populations without hatchery supplementation. The Henrys Fork below Island Park Dam, particularly the Box Canyon, Harriman Ranch, and Pinehaven reaches, supports world famous wild Rainbow Trout *Oncorhynchus mykiss* fisheries. Downstream of the Harriman Ranch, the Henrys Fork flows over Mesa Falls and is joined by the Warm River, before it is impounded by Ashton Dam. Brown Trout *Salmo trutta* are present in the Henrys Fork downstream of Mesa Falls, and increase in numbers in downstream reaches, eventually dominating the species composition (>80%) in and around the town of St. Anthony and below.

Previous research has identified the importance of winter river flows to the survival of age-0 Rainbow Trout in the Box Canyon reach (Garren et al. 2006a, Mitro 1999). Higher winter flows in this reach result in significantly higher overwinter survival of juvenile trout and subsequent recruitment to the fishery below 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 to maximize wild trout survival, based on timing and magnitude of winter releases from Island Park Dam.

STUDY SITE

During 2014, we sampled the Box Canyon and St. Anthony reaches of the Henrys Fork Snake River (Figure 50). 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 started below Island Park Dam at the confluence with the Buffalo River and extended downstream 3.7 km to the bottom of a large pool. The St. Anthony reach was approximately 7 km in length and started 0.3 km downstream of the railroad tracks located below the city of St. Anthony and ended 0.2 km upstream of the Red Road Bridge. Coordinates for all mark-recapture transect boundaries are presented in Appendix I.

OBJECTIVES

To obtain current information on fish population characteristics for fishery management decisions on the Henrys Fork Snake River, and to develop appropriate management recommendations.

METHODS

During 2014, we sampled all survey reaches using three electrofishing rafts. In the Box Canyon reach, we marked fish on May 6 and 7, and recaptured fish on May 13. Two passes per boat were made on each marking and recapture day for a total of 6 passes per day for both marking and recaptures. In the St. Anthony reach, we marked fish on June 10 and 11, and recaptured fish on June 18. All trout encountered from mark-recapture surveys were collected, identified to species, and measured for total length. Those exceeding 150 mm TL were marked with a hole punch in the caudal fin prior to release.

In all reaches, we estimated densities for all trout >150 mm TL using the log-likelihood method in Fisheries Analysis+ software (FA+; Montana Fish, Wildlife, and Parks 2004). Biomass was estimated with $\hat{B} = \hat{N}\overline{w}$, where $\hat{B} =$ estimated biomass (g), $\hat{N} =$ estimated abundance, and $\overline{w} =$ mean weight of fish (g) collected sampling. Proportional stock densities (PSD) were calculated as the number of individuals (by species) ≥300 mm / by the number ≥200 mm. 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).

We removed the sagittal otoliths from Rainbow Trout collected for the length-at-maturity evaluation. After removal, all otoliths were cleaned on a paper towel and stored in individually-labeled envelopes. Ages were estimated by counting annuli under a dissecting microscope at 40x power. Otoliths were submerged in water and read in whole view when clear, distinct growth rings were present. We sectioned, polished, and read otoliths in cross-section view with transmitted light when the annuli were not distinct in whole view. The von Bertalanffy (1938) growth model was used to fit length:

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

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. The model was fitted to length-at-age data by using the nonlinear model (NLIN) procedure in Program R. We estimated mortality rate (Z) for Rainbow Trout by catch curve analysis. Age-1 trout were excluded from the analysis due to lack of recruitment to the gear type. We estimated Rainbow Trout mortality rates between the ages of 2 to 4. For catch curve analysis, we used the age estimates from trout collected for length-at-maturity and combined it with the length data from the mark-recapture estimates to build an age-length key.

We examined length-at-maturity for Rainbow Trout in Box Canyon. We collected fish on April 15 using an electrofishing raft. Fish were collected randomly throughout the section and then placed on ice and brought back to the laboratory. In the laboratory, all fish were measured for total length, weighed, and sexed. For females, each ovary was assigned a maturity stage of either immature (small, translucent) or mature (large, orange, opaque). Logistic regression was used to fit sigmoid curves to the proportion mature by length in the form,

$$p_{x1} = e^{(b_0+b_1+x_1)}/(1+e^{(b_0+b_1+x_1)})$$

where, p is the probability that a fish is mature in a given length (mm) interval x_1, and b_0 and b_1 are parameters that define the shape and location of the fitted sigmoid curve. The predicated length of 50% maturity was calculated as, $L_{50} = -b_0/b_1$.

We also evaluated the effectiveness of winter flows in Box Canyon by using linear regression to examine the relationship between age-2 Rainbow Trout abundance and mean winter (Dec 1 – Feb 28) stream flow (cubic feet per second [cfs]) as described by Garren et al. (2006a). We log-transformed age-2 Rainbow Trout abundance and mean winter flow data from the past 15 surveys to establish the following relationship:

$$\log_{10} \text{age-2 Rainbow Trout abundance} = 0.5995 \log_{10} \text{winter stream flow} + 1.9668$$

Using this equation we predicted the expected abundance of age-2 Rainbow Trout in our 2014 sampling based on mean winter stream flows observed during 2013 (December 2012 - February 2013). To validate this relationship, we determined age-2 Rainbow Trout abundance during the 2014 electrofishing surveys by estimating the number of fish between 230 and 329 mm, which correlates to the lengths of age-2 trout in past surveys. Age-2 Rainbow Trout were determined to be the first year class fully recruited to the electrofishing gear (Garren 2006b). We then compared predicted and observed age-2 Rainbow Trout abundance in Box Canyon to evaluate the equation above to predict year class strength based on winter flow. Data from 2014 was added to the flow vs. age-2 abundance regression model and this model will continue to be used in management of winter flow releases from Island Park Dam.

RESULTS

Box Canyon

We collected 2,168 trout during two days of electrofishing in the Box Canyon section. Species composition of trout collected was 99% Rainbow Trout and 1% Brook Trout *Salvelinus fontinalis*. Rainbow Trout ranged in size from 106 mm to 635 mm, with a mean (± 95% CI) and median total length of 341 (± 3) mm and 337 mm, respectively (Figure 51; Appendix J). Rainbow Trout PSD and RSD-400 were 80 and 17, respectively (Table 17). We used the Log-likelihood Method (LLM) to estimate 5,828 ± 337 Rainbow Trout >150 mm (cv = 0.06, Table 18, Appendix K) in the reach, which equates to 1,575 fish per km (Figure 52). Our efficiency rate (ratio of marked fish during the recapture runs [R] to total fish captured on the recapture run [C]), unadjusted for size selectivity was 28% (Appendix K). Rainbow Trout biomass was 100.6 kg/ha (± 5.8). As the strong year class of age-3s became more pronounced in the population, mean length of all trout increased in 2014 (mean = 341 mm TL; ± 3 mm) compared to mean length in 2013 (mean = 294 mm TL; ± 3 mm; Figure 53). Rainbow Trout mean relative weights were 93 (±1.1; Figure 54). We found no relationship between relative weight and length (linear regression, y = -0.042x + 107.2, R^2=0.06). We aged 28 male and 118 female Rainbow Trout from the Box Canyon reach of the Henrys Fork. When we combined male and female Rainbow Trout they grew from a starting age of t_0 = -0.81 years (± -2.24 to 0.001) toward their asymptotic length of L_∞ = 503 mm (95% CI 442-667 mm) at an instantaneous rate of K = 0.38/year (± 0.17-0.68) (Figure 55). Female Rainbow Trout grew from a starting age of t_0 = -0.81 years (± -2.07 to -0.015) toward their asymptotic length of L_∞ = 503 mm (95% CI 438-640 mm) at an instantaneous rate of K = 0.38/year (± 0.18-0.67). Male Rainbow Trout grew from a starting age of t_0 = -0.43 years (± -1.84 to 0.39) toward their asymptotic length of L_∞ = 595 mm (95% CI 452-1328 mm) at an instantaneous rate of K = 0.32/year (± 0.08-0.80). Length at age estimates for two-year old Rainbow Trout was 332 mm (±6). Mean length at age-2 for Rainbow Trout was 329

mm (± 6.4, n = 83) (Table 18). Catch curve analysis of Rainbow Trout estimated mortality at 74% from age two to four.

We collected 64 female Rainbow Trout for maturity analysis, which ranged in size from 229 to 453 mm. Of the 64 fish, 26 were considered mature based on inspection of the ovaries. The logistic regression curve (Maturity proportion = $1 + e^{(-(-19.31 + 0.053 \times TL))}$) estimated the proportion of female Rainbow Trout that were mature as a function of total length reasonably well despite a low sample size (*n* =64, Figure 56). Female Rainbow Trout were 1% mature (L_1) at approximately 277 mm TL and 50% mature (L_{50}) at approximately 364 mm TL or approximately age-3. We attempted to assess maturity of 30 male Rainbow Trout ranging in size 226 to 470 mm. However, we were unsuccessful in fitting a logistic regression curve (i.e. no relationship with size at maturity) to male Rainbow Trout.

The regression model between winter flow (December-February) estimated an abundance of 3,348 age-2 Rainbow Trout in the 2014 survey based on winter flows that averaged 397 cfs. However, the length-based estimates of abundance from our Log Likelihood model estimated age-2 Rainbow Trout abundance at 2,575 fish in the Box Canyon during 2014 (Figure 57). Across most years, the regression model accurately estimates the relative year class strength of Rainbow Trout using mean winter stream flow (R^2 = 0.49, $F(1,16)$ = 15.3, P = 0.001) and is a useful tool to evaluate the effects of variable winter flows.

St. Anthony

We collected 1,009 trout during three days (June 10, 11, and 18) of electrofishing in the St. Anthony reach of the Henrys Fork. Species composition of trout collected was 83% Brown Trout, 17% Rainbow Trout, <1% Brook Trout, and <1% Yellowstone Cutthroat Trout. Rainbow Trout ranged between 164 mm and 545 mm (Figure 58), with a mean (± 95% CI) and median total length of 327 mm (± 10 mm) and 331 mm, respectively (Table 17). Rainbow Trout PSD and RSD-400 values were 64 and 12, respectively. Brown Trout ranged between 130 mm and 555 mm (Figure 59), with a mean and median total length of 303 mm (± 6 mm) and 310 mm, respectively (Table 17). Brown Trout PSD and RSD-400 values were 63 and 12, respectively. We estimated 1,650 Rainbow Trout >150 mm for the reach (± 587 – 2,713), which equates to 120 Rainbow Trout per km (Table 18; Figure 60). Our efficiency rate (unadjusted for size selectivity) was 14%. We estimated 3,895 Brown Trout >150 mm for the reach (± 3,388 – 4,402; cv = 0.07), which equates to 597 Brown Trout per km (Table 18; Figure 60). Our efficiency rate (unadjusted for size selectivity) for Brown Trout was 17%. Based on regression analysis of Brown Trout species composition across time, Brown Trout have increased 0.9% percent composition per year since 2004 (linear regression, y = 0.0927x – 1782.1, R^2 = 0.71; Figure 61).

DISCUSSION

Estimates of trout abundance in Box Canyon exhibited a statistically significant decrease in density when compared to 2013 (59%). However, the current density estimate was only slightly below the 19-year average (19%) and trout biomass (kg/ha) was at the 19-year average suggesting the trout population in Box Canyon appears to have stabilized following the strong year class from 2011. While it is unlikely trout which spawned in 2011 suffered unusually high mortality, their abundance moderated between last year and the current survey. This shift in abundance may be due to movement downriver. Increased dispersal of trout downriver may occur in years with high recruitment as intraspecific competition increases from higher trout densities (Dibble et al. 2015). We are currently not as effective in conducting raft electrofishing

for population surveys in the river sections downriver of the Box Canyon (e.g. Harriman, Pinehaven) due to slower river velocities and wide shallow channels which allows trout to be more effective in eluding the raft electrical field. Since we have limited population information downriver, we were unable to determine whether trout dispersal occurred in the lower sections (i.e. increased densities) as a result of increased intraspecific competition from the strong recruitment class of age-0 from 2011. Anecdotal reports from anglers suggest dispersal may have occurred, as some anglers reported catching high numbers of smaller fish in areas where they rarely have encountered those fish in prior years. We also observed a similarly higher density of trout in 2000. Although we were unable to sample the following year in 2001, trout densities returned near the long-term average by 2002 suggesting possible dispersal downriver due to higher densities. Based on current length frequency analysis, the strong year class of age-3 trout in 2014 was the dominant age class and there appears to be limited recruitment of age-1 and age-2 trout. The weaker year classes of age-1 and age-2 may be due to the lower winter flows they experienced as age-0 trout. Winter flows were 200 and 269 cfs for age-1 and age-2 trout, respectively, in the past two years. In the previous five years (2009-2013) winter flows have exceeded 300 cfs which produced slightly stronger year classes and subsequently higher trout densities.

Estimates of fishing (F) and natural mortality (M) are important for understanding and managing recreational fisheries such as Henrys Fork. The Box Canyon section of Henrys Fork is currently managed under catch-and-release angling regulations. Due to these restrictive regulations fishing (*F*) mortality is likely a minor component to the total mortality (Z) and driven in large part by natural morality (*M*) within this section. We observed a much higher than expected mortality rate (74%) based on catch curve analysis. Total morality in catch curve analysis is based on the natural logarithm of catch plotted against ages. Otoliths were used to assign ages based on annuli patterns. If distinct annuli patterns are not observed on the otoliths then the accuracy of the estimated age may not be the true age of the fish. Trout in the Box Canyon section experience a more thermally stable environment than trout in the lower sections due to the hypolimnetic releases from the dam. As a result, otolith annuli patterns may not be distinct and affect the readability of otoliths for aging. If estimated ages tend to over or underestimate the true age (e.g. biased), total morality estimates would be influenced. Currently no studies have been undertaken to assess the accuracy of otolith age estimates of trout within the thermally stable Box Canyon section to determine if the mortality estimates derived from catch curve analysis are reasonable to expect given the restrictive angling regulations.

Winter stream flows continue to be the main factor in driving Rainbow Trout abundances within the Box Canyon section (Garren et al. 2006a). Observed age-2 abundance (2,575) in 2014 was slightly outside the 95% CIs predicted from our regression model (3,348) that incorporated flows during the winter of 2012-2013 which would have affected age-2 fish in the 2014 survey. Fausch et al. (2001) found Rainbow Trout recruitment was higher in tailwaters exhibiting high winter and/or low spring flows. High spring flows can reduce year class strength due to substrate scouring displacing eggs and fish larvae or from redd desiccation. Higher or lower 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 past five years (2010-2014) average spring flows have ranged from 468-1,057 cfs. However, most of the past studies in the Henrys Fork have found winter flow is 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, we recommend continuing to use and refine the winter flow model when considering altering dam operations to improve winter flows conditions in Box Canyon.

Trout densities in the St. Anthony reach have increased approximately 200% with statistically significant differences compared to spring surveys conducted in 2004 and 2010, respectively. We observed two strong year classes for Rainbow Trout and three year classes for Brown Trout in the trout population size structure. The limited length frequency distribution suggest variable recruitment in this section of the river. The length frequency for Brown Trout and Rainbow Trout also suggests a lightly exploited population as evident by the number of large trout, despite a 6 trout daily bag limit allowed in the current fishing regulations. The hypothesized low exploitation is likely due to angler behavior (i.e. catch-and-release) like that observed in other reaches of the Henrys Fork and other area waters (Schoby et al. 2010). IDFG has not conducted a creel survey below St Anthony, but a creel survey conducted just upstream from Chester Dam to the railroad bridge in 2008 found anglers caught 1,003 Rainbow Trout and 747 Brown Trout. Survey estimates showed angler harvest was low for Brown Trout (no harvested Brown Trout found in that survey) and only 46 Rainbow Trout were estimated as being harvested. Similar angler behaviors and harvest rates would be expected in the St. Anthony reach, which would result in low exploitation (well below 5%) being removed for harvest. In prior surveys, we have observed a shift in species composition in the HFSR downstream of Mesa Falls, from a Rainbow Trout dominated system to one with an increasing abundance of Brown Trout (High et al. 2011). In the St. Anthony reach, Brown Trout comprised 46% of the species composition in 1999, increasing to 74% in 2004, and more recently 85% and 83% of the species composition in 2010 and 2014, respectively. The shift towards a more Brown Trout dominated system may due to both abiotic (e.g. temperature) and biotic factors (e.g. competition). Brown Trout often displace Rainbow Trout from their preferred habitat in sympatric populations owing to their larger size and higher innate aggression (Gatz et al. 1987;). Rainbow Trout forced to shift towards less profitable foraging habitat would have reduced energy intake and growth. The timing of surveys (spring vs. fall) may affect species composition or overall densities in this reach if spawning fish from downstream reaches move to the upstream sample reach. If upstream spawning movement occurs, Rainbow Trout may be overrepresented in spring surveys, and Brown Trout may be a larger component during fall surveys. Past research (High et al. 2011) looked at seasonal variation in fish densities and species composition in the St. Anthony reach, and found that spawning migrations were likely not responsible for shifts in either variable. More so, it is possible that thermal stresses lower in the river cause fish to move upstream during the warmer months to take advantage of thermal refugia in areas where groundwater upwellings are common. Regardless of the mechanism responsible for these shifts, standardization of the timing of the St. Anthony reach should be prioritized, and should remain constant from this point forward. Traditional spring surveys were designed to eliminate interference with anglers, as these surveys occurred during closed fishing seasons. This is no longer a concern, as most river reaches are open to year-round fishing. Given the demands currently in place for manpower during the spring, we recommend adopting a fall survey in this reach, recognizing this may cause small issues with direct comparisons between other Henrys Fork survey locations that typically are surveyed during the spring.

RECOMMENDATIONS

1. Continue annual population surveys in Box Canyon to quantify population response to changes in the flow regime over time. Collect trout otoliths annually for aging and morality estimates.

2. Work with the irrigation community and other agencies to obtain 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 role of flows outside of winter to determine reasons why population estimates vary when compared to predicted estimates.

4. Consider effects of regulations changes on fish populations in the river. Implement consistent regulations if socially acceptable, and biologically beneficial.

Table 17. Trout population index summaries for the Henrys Fork Snake River, Idaho 2014.

River Reach	Species	Mean TL (mm)	Median TL (mm)	PSD	RSD-400	RSD-500	Density (No./km)	Species Composition (%)
Box Canyon	Rainbow Trout	341 (±2.7)	337	80	17	1	1,575 (±91)	100
St. Anthony	Brown Trout	304 (±5.7)	312	63	12	2	597 (±81)	83
	Rainbow Trout	327 (±10.6)	331	64	13	1	120 (±16)	17

Table 18. Log-Likelihood Method (LLM) population estimates of trout (≥150 mm) from the Henrys Fork Snake River, Idaho during 2014.

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	1532	636	175	5828	337	91	971
	Brook Trout	1	0	0	--	--	--	
St. Anthony[3]	Brown Trout	590	253	44	3895	507	597	1,365
	Rainbow Trout	122	44	6	1650	1063	120	
	Brook Trout	1	0	0	--	--	--	
	Cutthroat Trout	1	0	0	--	--	--	

[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.

Table 19. Mean length at age data based on otoliths of trout collected electrofishing in Box Canyon of Henrys Fork, Idaho in 2005, 2011, 2013, and 2014.

Year		Mean	95% CI	Median	Min	Max	Count
2014	1	255	64.3	265	226	275	3
	2	329	6.4	330	229	390	83
	3	391	8.3	387	322	470	56
	4	433	26.6	426	402	470	6
	6	459	69.9	459	453	464	2
2013	1	170	8.0	168	130	205	35
	2	278	10.8	268	185	424	90
	3	390	10.8	390	305	460	53
	4	449	32.5	436	375	516	9
	5	484	--	--	--	--	1
	6	475	--	--	--	--	1
2011	1	260	19.3	263	170	305	15
	2	324	15.6	323	234	404	31
	3	389	16.1	396	333	450	20
	4	421	24.8	413	382	460	8
2005	1	122	23.6	134	92	142	6
	2	296	9.4	301	239	337	28
	3	348	19.7	365	290	394	14
	4	432	16.0	435	310	497	25
	5	422	28.9	428	300	479	13
	6	458	13.0	459	418	495	15
	7	494	--	--	--	--	1

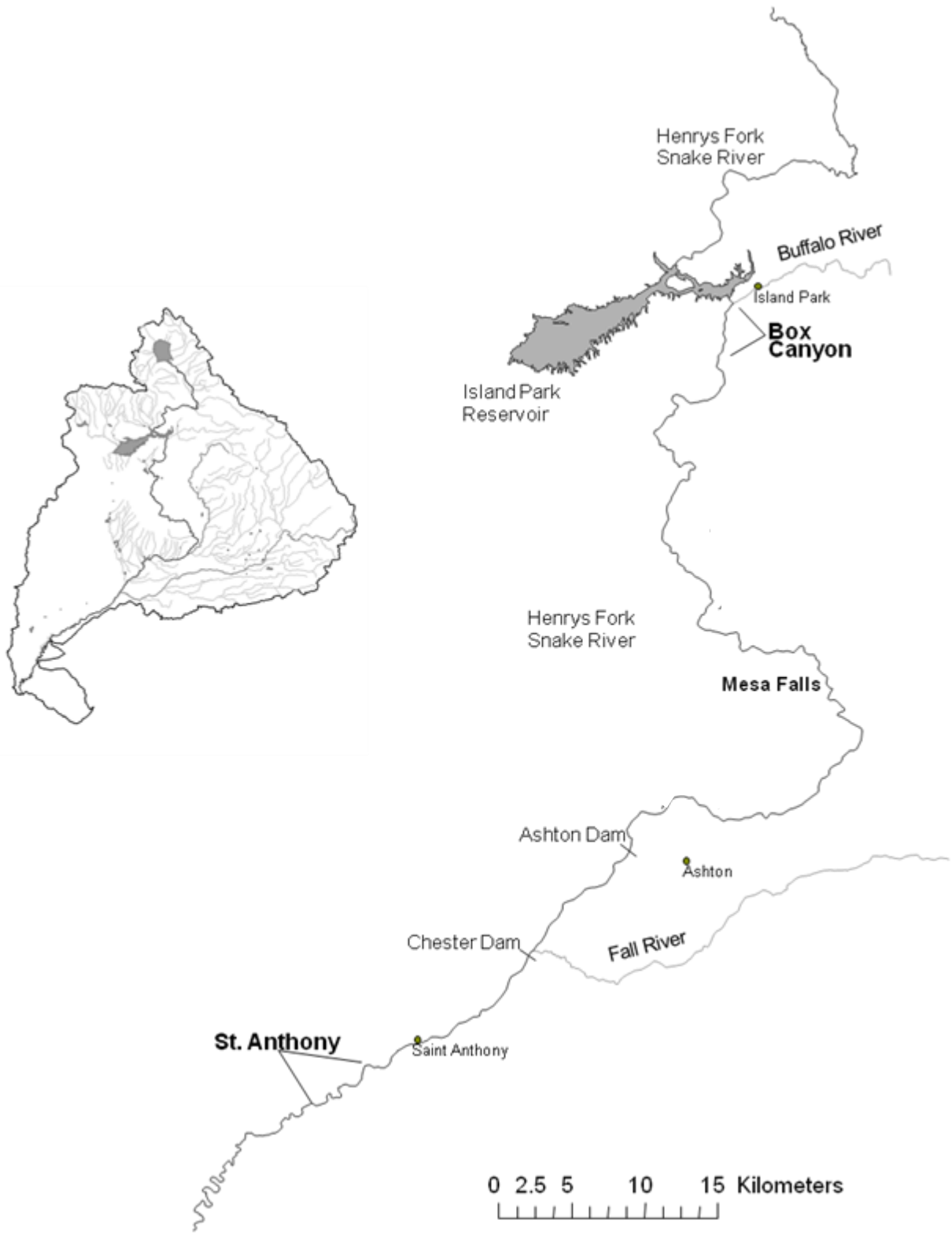

Figure 50. Map of the Henrys Fork Snake River watershed and electrofishing sample sites (Box Canyon and St. Anthony) during 2014.

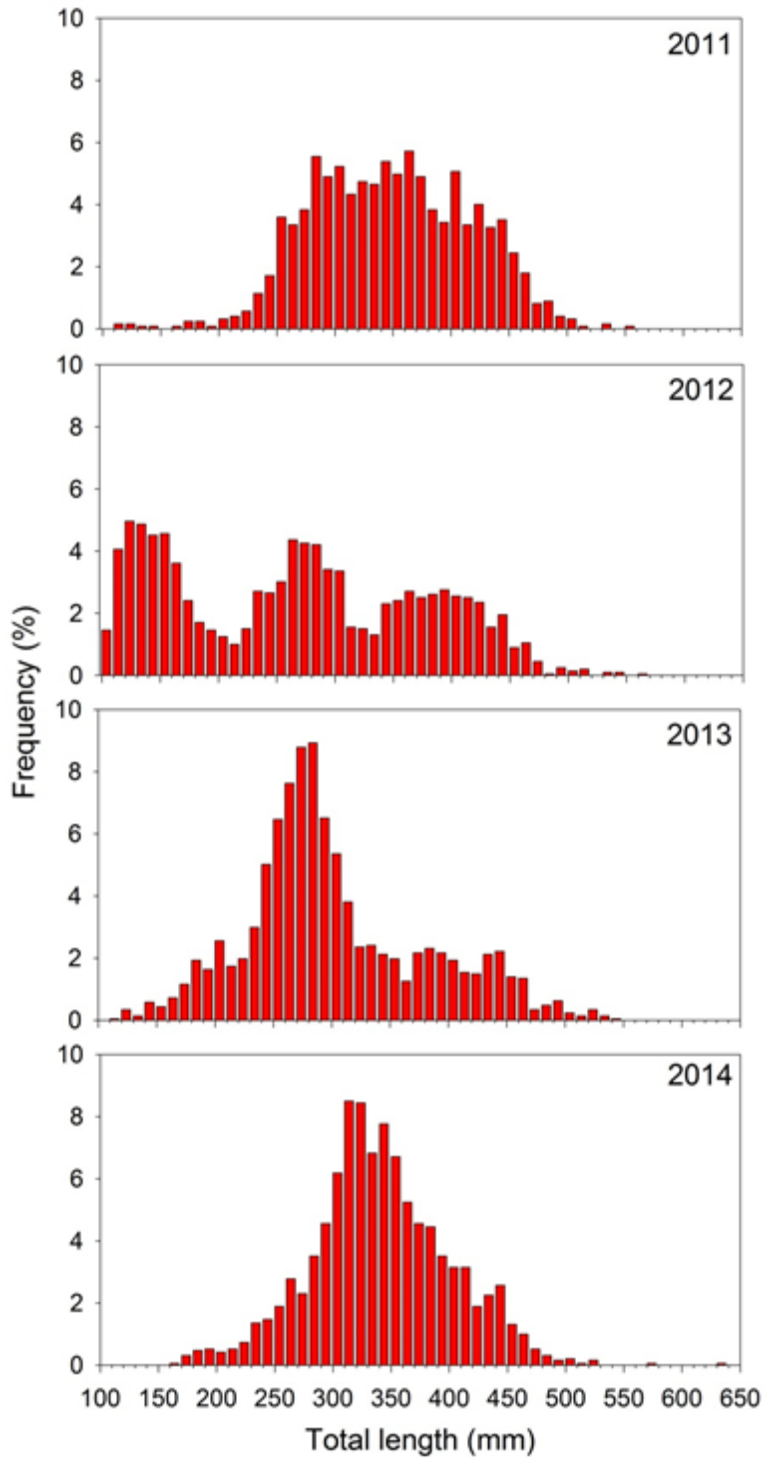

Figure 51. Length frequency distribution of Rainbow Trout collected by electrofishing in the Box Canyon reach of the Henrys Fork Snake River, Idaho, 2011-2014.

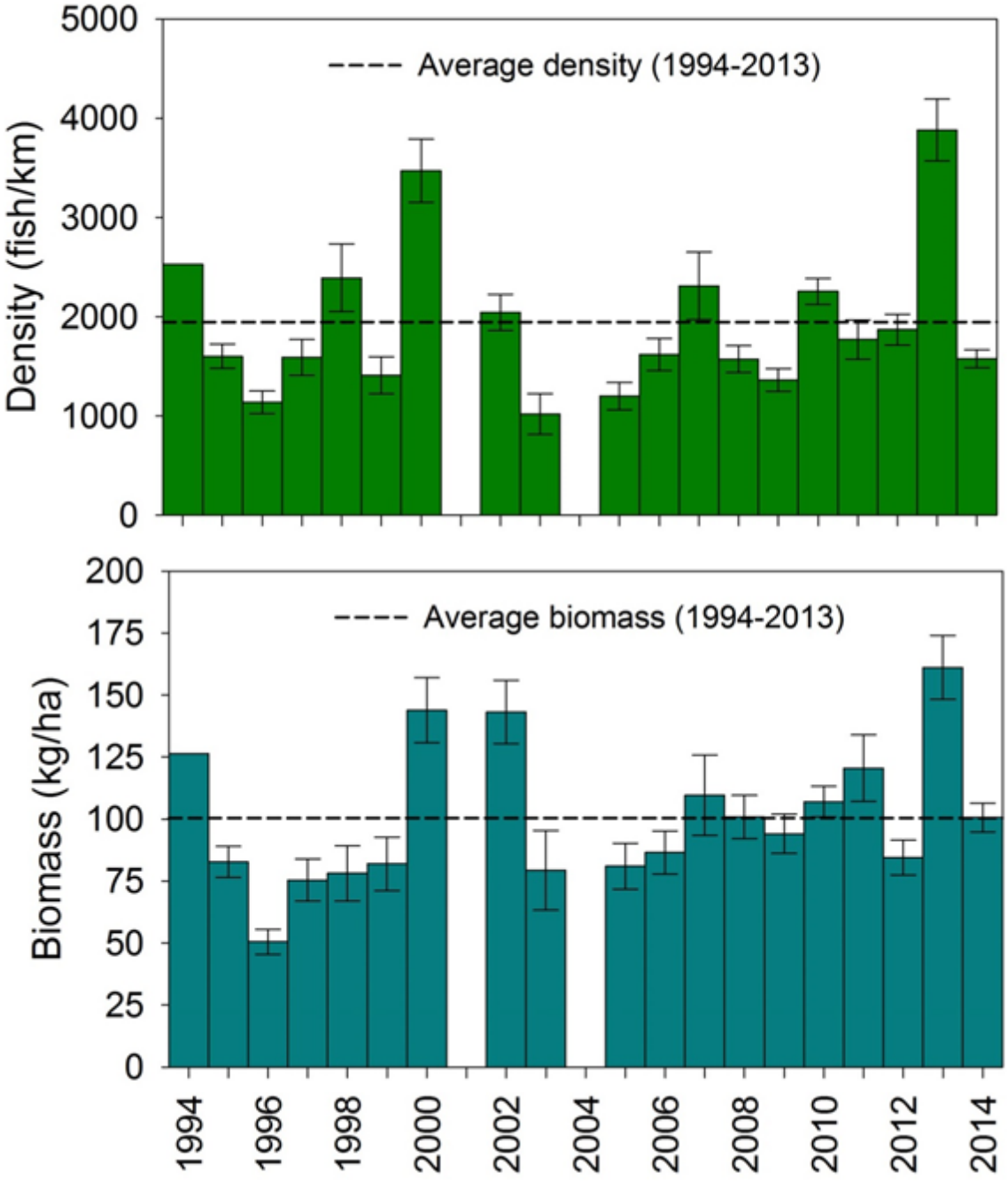

Figure 52. Rainbow Trout density (fish/km) and biomass (kg/ha) estimates for the Box Canyon reach of the Henrys Fork Snake River, Idaho 1994 - 2014. Error bars represent 95% confidence intervals. The dashed lines represent the long-term average Rainbow Trout density and biomass, excluding the current year's survey.

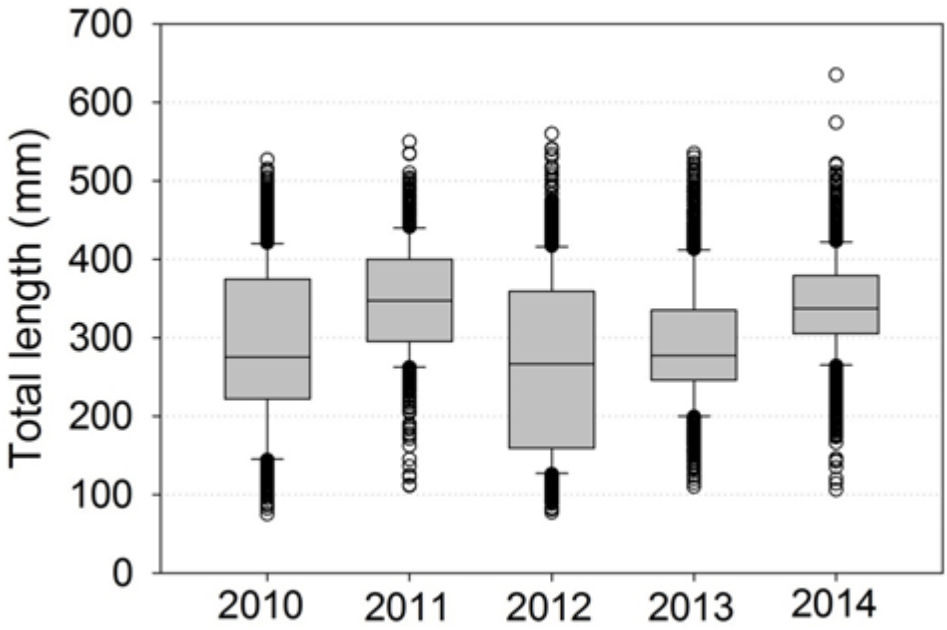

Figure 53. Boxplot of Rainbow Trout total length (mm) from 2010 to 2014 in Box Canyon reach of the Henrys Fork. The box is the interquartile range. The whiskers are the high and low values excluding outliers. The line across the box is the median.

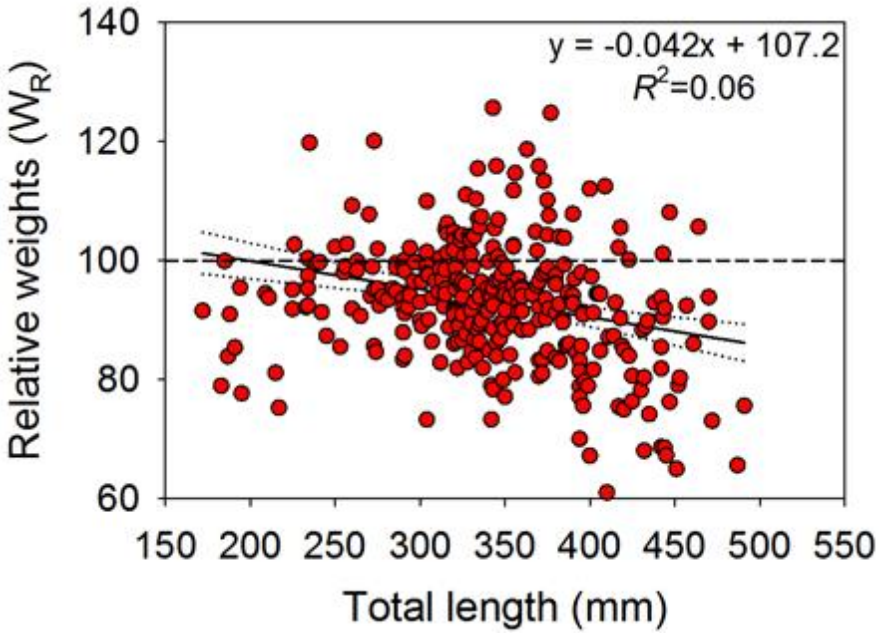

Figure 54. Rainbow Trout relative weight by total length from spring electrofishing surveys in the Box Canyon reach of Henrys Fork, 2014. Linear regression and 95% confidence intervals are represented with a solid and dotted line, respectively.

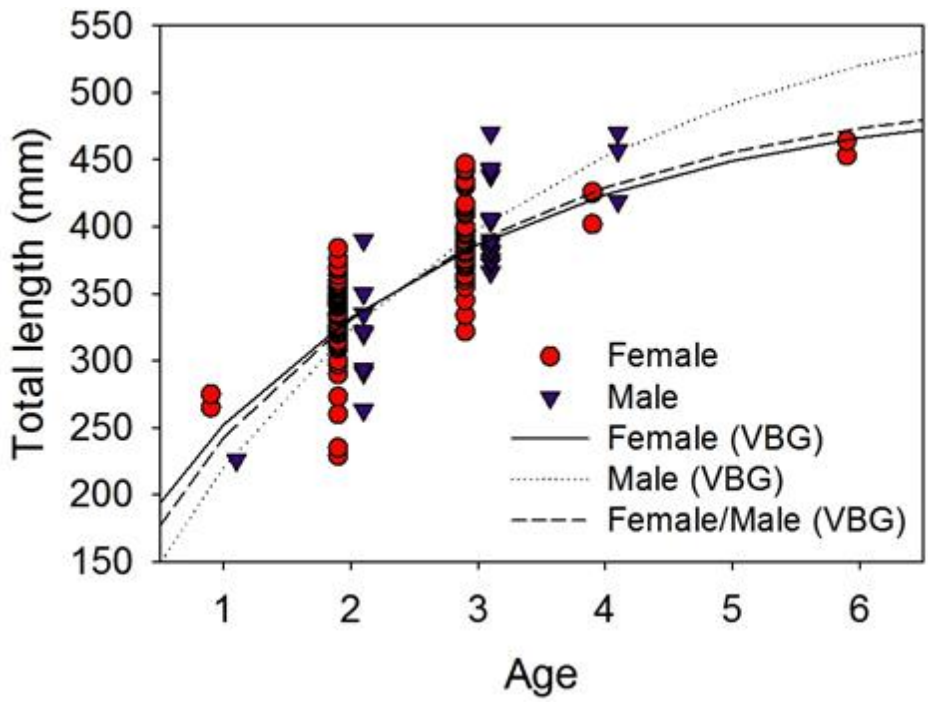

Figure 55. Length-at-age based on non-linear regression for Rainbow Trout from spring electrofishing in the Box Canyon reach of the Henrys Fork, 2014. Growth is described by the fitted von Bertalanffy growth (VBG) model for female (solid line), male (dotted line), and combined male and female (dashed line) Rainbow Trout.

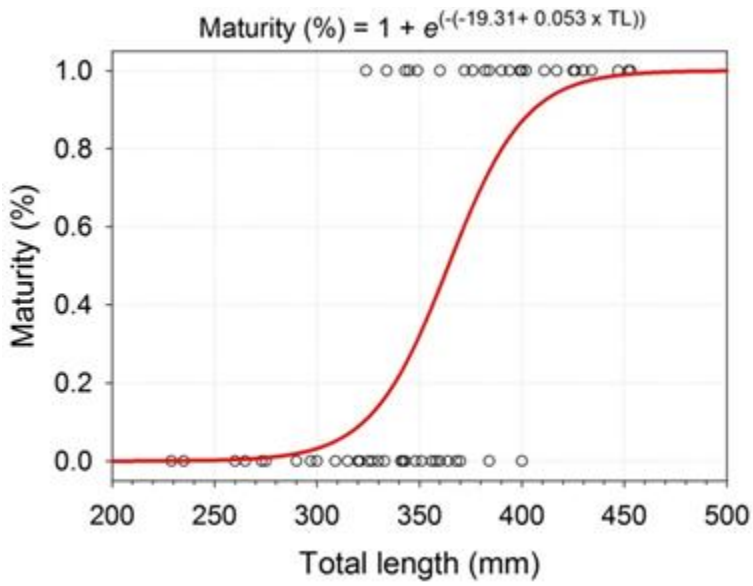

Figure 56. Proportion mature (1 = mature, 0 = immature) by total length with a fitted logistic regression curve of female Rainbow Trout collected in Box Canyon during the spring of 2014.

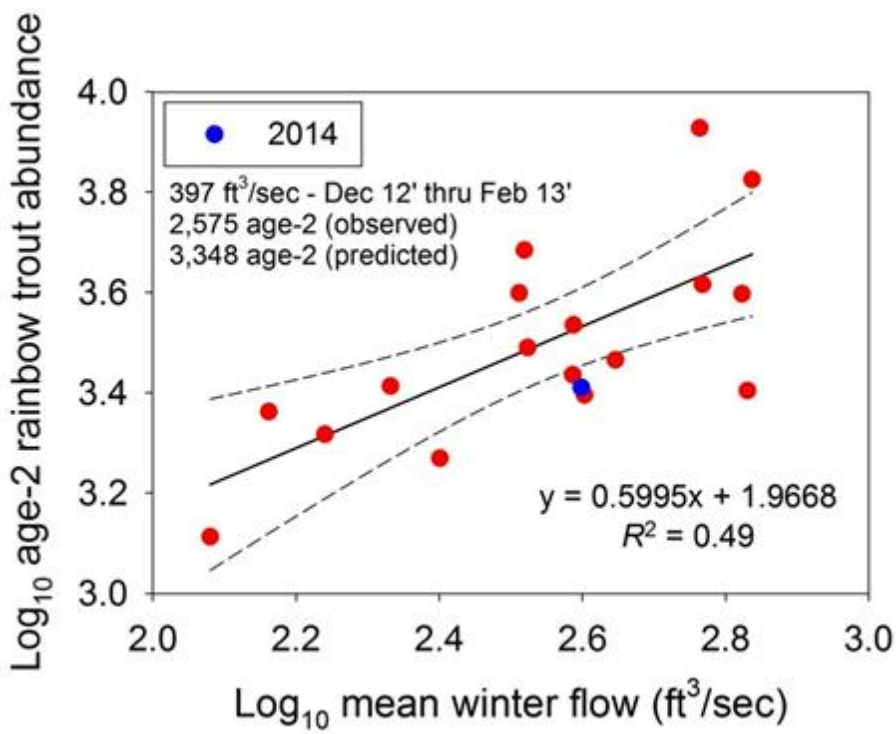

Figure 57. The relationship between age-2 Rainbow Trout abundance and mean winter flow (cfs) during the first winter of a fish's life from 1995 - 2014; $\log_{10}$ age-2 trout abundance = 0.5995 $\log_{10}$ flow (cfs) + 1.9668, (R^2= 0.49, $F(1,16)$ = 15.3, P = 0.00 1). Linear regression and 95% confidence intervals are represented with a solid and dotted line, respectively.

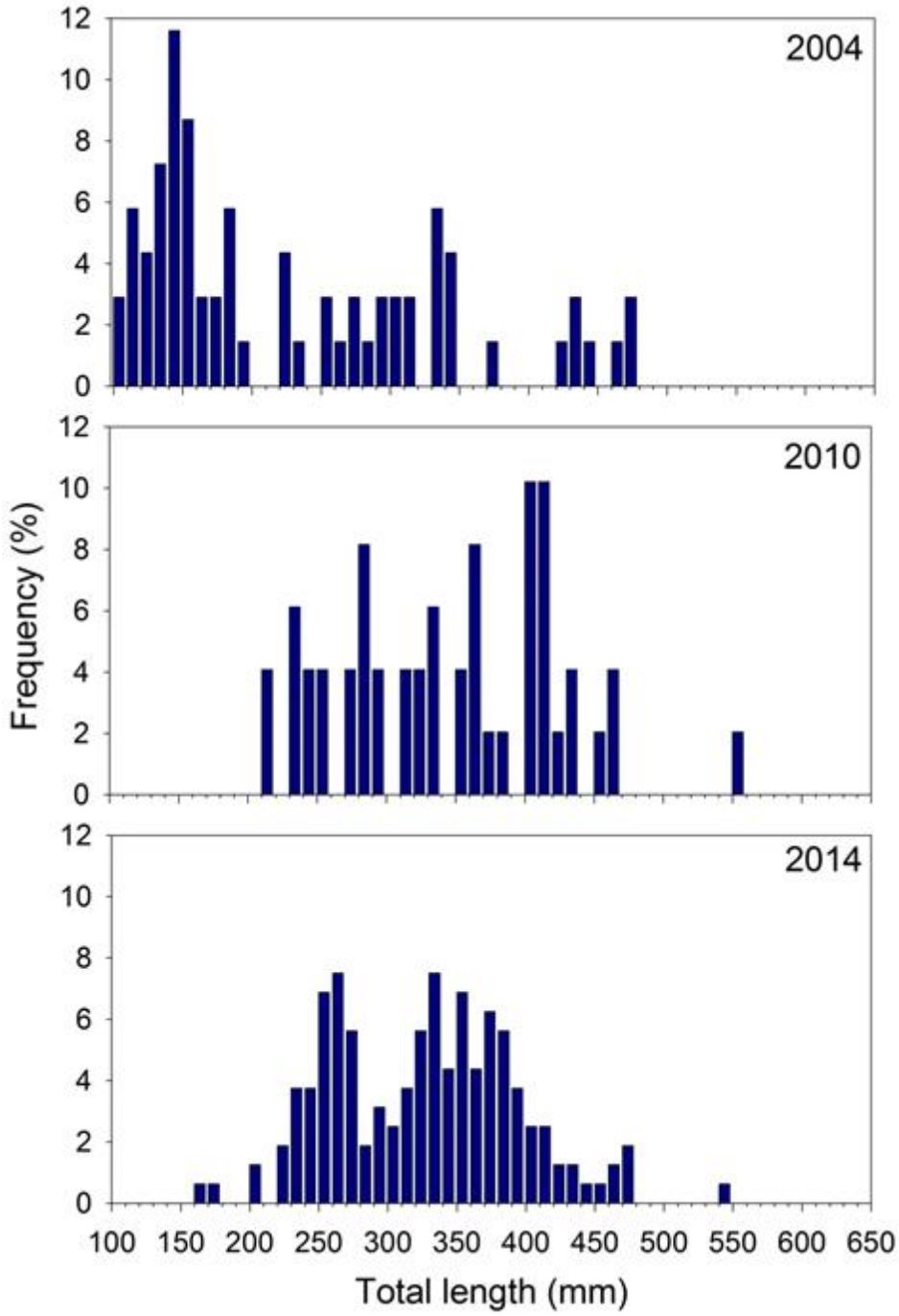

Figure 58. Length frequency of Rainbow Trout captured electrofishing in the St. Anthony reach of the Henrys Fork Snake River during the spring of 2004, 2010, and 2014.

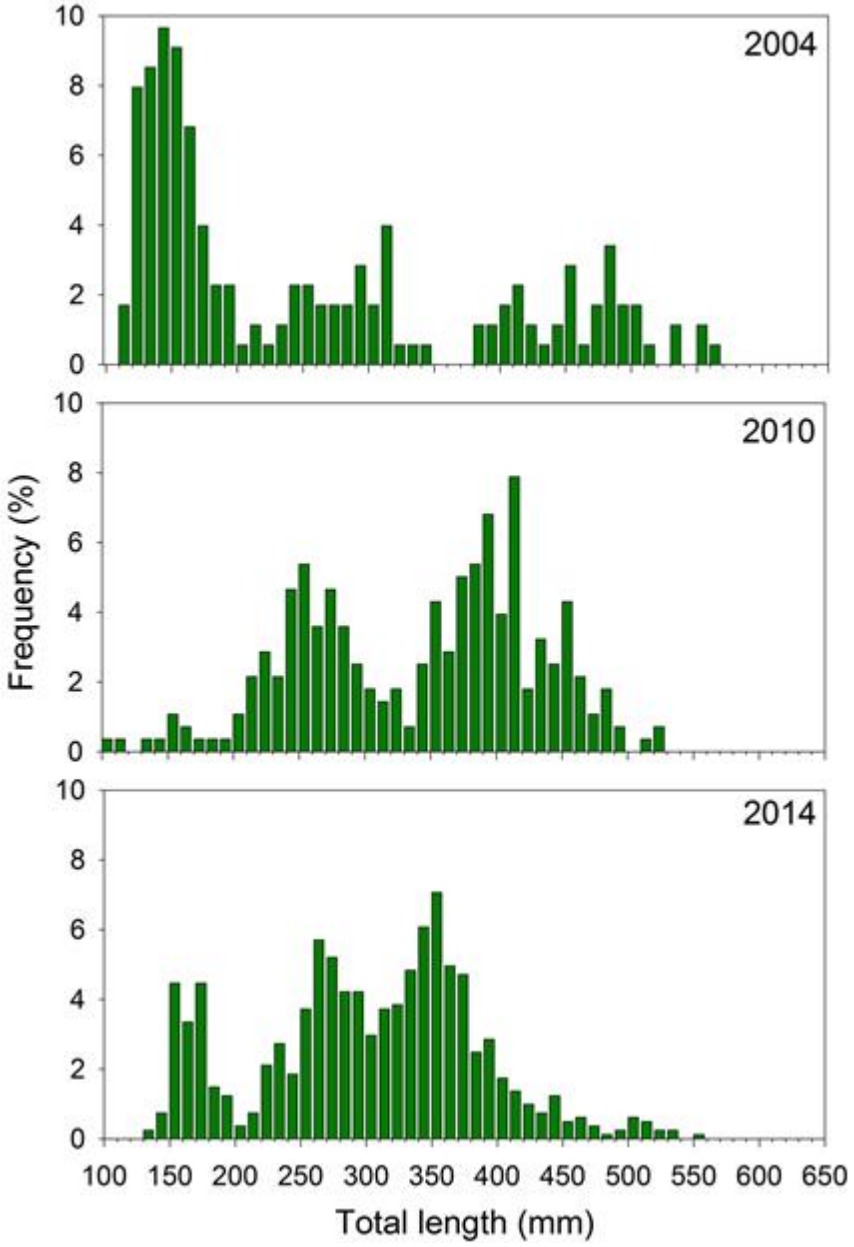

Figure 59. Length frequency of Brown Trout captured by electrofishing in the St. Anthony reach of the Henrys Fork Snake River during the spring of 2004, 2010, and 2014.

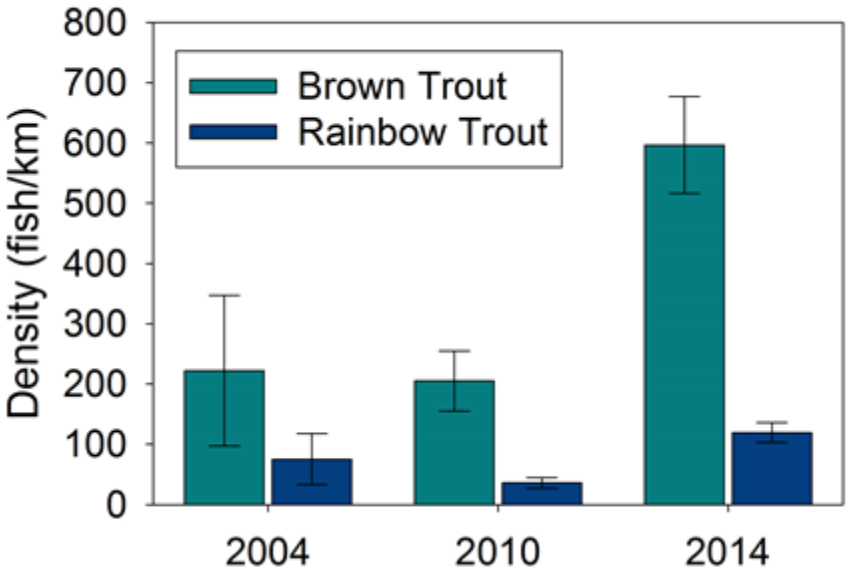

Figure 60. Rainbow Trout and Brown Trout density estimates (fish/km) from spring surveys in the St. Anthony reach of the Henrys Fork Snake River, Idaho 2004, 2010, and 2014. Error bars represent 95% confidence intervals.

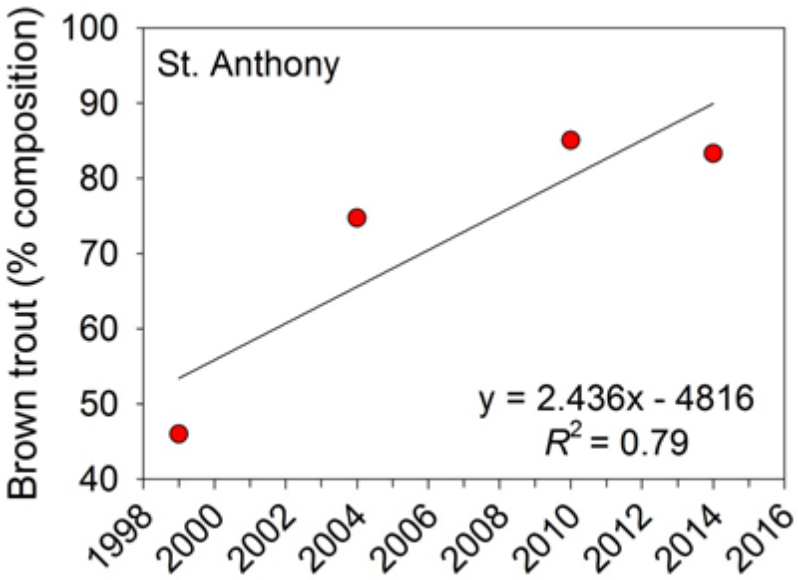

Figure 61. Percent composition of Brown Trout by year with linear regression trend line in the St. Anthony reach of the Henrys Fork Snake River, Idaho 1999, 2004, 2010, and 2010.

SOUTH FORK SNAKE RIVER

ABSTRACT

We surveyed three locations on the South Fork of the Snake River including Lorenzo and Conant, where trout abundance decreased for all trout species from recent record high numbers. Mean biomass and density of all trout are now near the 10-year average. Length frequencies suggest variable recruitment for Yellowstone Cutthroat Trout (YCT) and Brown Trout (BNT) while Rainbow Trout (RBT) recruitment has been more stable. Tributary weirs were operated on all four major tributaries with high efficiencies (>90%), except in Pine Creek which had a lower efficiency (70%). Run sizes were similar to the past few years except Pine Creek, which decreased. Telemetry data indicated 10-19% of marked trout were entrained into irrigation canals. We divided the river into three segments for the telemetry study and found marked fish from all river reaches were entrained into canals in the lower reach. All canals entrained at least one radio-tagged trout. However, entrainment was not equally distributed among canals. The Dry Bed Canal, Reid Canal, and Sunnydell Canal entrained more fish with radio tags than the other canals. Habitat use was similar for all trout species during the spawning season, with trout using transitional habitat between riffles and deep runs, and in side channels. Yellowstone Cutthroat Trout migrated further distances during spawning season than BNT or RBT, and were the only species documented to migrate to and then spawn in tributaries. Increasing spring flows were positively correlated with the abundance of age-1 YCT the following year, but no relationship was evident for RBT. The angler incentive program was continued in 2014 and continues to be an effective tool for halting RBT abundance increases by capitalizing on angler harvest. PIT tag data indicate juvenile YCT out-migrate from tributaries in spring and fall at varying ages depending on the tributary, and that spawning adults exhibit high fidelity to streams and generally spawn every year. Threats to YCT populations remain in the South Fork, but consistent and adaptive management can help YCT abundance continue to increase and maintain population viability.

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Regional Fisheries Biologist

Dan Garren
Regional Fisheries Manager

INTRODUCTION

The South Fork Snake River (South Fork) in Eastern Idaho supports robust populations of three wild trout species including an important population of native Yellowstone Cutthroat Trout (YCT). Other trout present in the South Fork include Rainbow Trout and Rainbow x Cutthroat Trout hybrids (RBT) and Brown Trout (BNT). Since 2004, a three-pronged management approach has been used to accomplish the objectives outlined in the state fish management plan including preserving the genetic integrity and population viability of native Yellowstone Cutthroat Trout and limiting RBT to less than 10% of the species composition of the catch as measured at the Conant monitoring reach during annual fall electrofishing surveys (IDFG 2013). This report summarizes management and research activities on the South Fork Snake River in 2014. For a broader description of the South Fork Snake River and additional background information see Schoby et al. (2013).

METHODS

South Fork Population Monitoring

The methodology for annually monitoring fish abundances and trends in the South Fork, operating and evaluating the tributary weirs, assessing the effects of spring flows on YCT and RBT recruitment, implementation and analysis of the South Fork Angler Incentive Program, analyses using PIT tag data, and the manual removal of RBT from Palisades Creek can be found in detail in Schoby et al. (2013). Methods used in 2014 were identical to those outlined in the referenced report.

In addition to methods used during previous years, we compared length-weight relationships for each trout species caught at the Lorenzo and Conant monitoring reaches of the South Fork. During the electrofishing surveys we weighed a subsample of each species at each of the electrofishing reaches. We then compared these observed weights with published standard weights calculated for YCT (Kruse and Hubert 1997), RBT (Simpkins and Hubert 1996), and BNT (Milewski and Brown 1994). We calculated relative weights (W_r) for sampled trout and compared these with relative weights from 2002, 2003, 2012, and 2013 at the Lorenzo Reach and with relative weights from 2002, 2012, and 2013 at the Conant Reach. Comparisons were based on 100 mm length groups among years using 95% confidence intervals where non-overlapping intervals were considered statistically significant at the $\alpha = 0.05$ level.

We investigated year class strength using length frequency plots for each species at Lorenzo and Conant for the last five years. Year classes were assigned using predetermined size ranges developed for the South Fork by Fredericks et al. (2003). Length frequency plots were based on observed catches during the fall electrofishing surveys at these sites, and ten mm intervals were used.

We compared biomass estimates by species at the Lorenzo and Conant monitoring reaches over the past 10 years using linear regression. Biomass was estimated by first calculating length:weight relationship curves for each species at the separate monitoring reaches and expanded using the population estimates. We then estimated the biomass for each species by summing the biomass for each size interval (25.4 mm groups greater than age-1) in the log-likelihood population estimates. We calculated the biomass for each size interval by using the average fish length for the size interval from the observed catch and using this length in the appropriate length:weight relationship equation specific to species and site. We then

multiplied this product by the population estimate for that size interval. We summed the size interval biomass estimates to estimate total biomass by species for each site for each year. When log-likelihood estimates (size-interval estimates) were not available, we used the overall average length of the species caught during the survey in the appropriate species and site-specific length:weight relationship equation and multiplied this result by the corresponding modified Peterson population estimate to estimate annual biomass. We calculated biomass estimates in order to investigate whether species-specific biomass or total biomass remained constant when trout abundances varied.

We assessed effects of various environmental variables on recruitment of YCT and RBT in the Conant monitoring reach. DeVita (2014) modeled the effects of river and tributary flows on recruitment of YCT and RBT. We used the model developed by DeVita (2014) and added data from 2014 to the model. Predictor variables included river winter flow (December 1 through February 28), tributary flows during rearing (July 16 through October 15), tributary flows during winter (December 1 through February 28), and tributary annual flow (October 1 through September 30) with all variables log-transformed and linearly added to a logistic function for river winter flows (see DeVita 2014).

We also tested for relationships between age-1 YCT abundance and summer temperatures, spawner abundance, and abundance of age-1 BNT to assess whether these variables could explain fluctuations in trout abundance. In lieu of summer water temperatures, which were not available for the entire data set, we used summer air temperatures recorded at Palisades Dam. We used linear regression on log-transformed data for all three variables (summer temperatures, spawner abundance, and age-1 BNT abundance) after we tested for normality. We assessed each analysis using residuals analysis plots.

Radio Telemetry

We initiated a five-year telemetry study to determine trout movements in the South Fork during the summer of 2013. In 2014, we continued this telemetry study and expanded the study area to include the entire South Fork as opposed to the 2013 efforts, which focused on the river in proximity of seven irrigation canals. These canals are ranked as follows based on their average June flows for 2014: 1-Dry Bed Canal (4,097 cfs), 2-Eagle Rock Canal (753 cfs), 3-Farmer's Friend Canal (434 cfs), 4-Anderson Canal (413 cfs), 5-Market Lake Canal (280 cfs), 6-Enterprise Canal (205 cfs), 7-Reid Canal (181 cfs), and 8-Sunnydell Canal (147 cfs: IDWR 2015). As in 2013, we stratified the study area into three sections: the lower section, canyon section, and upper section. The lower section is 38.6 river kilometers in length and is located from the confluence upstream to near the Wolf boat ramp. The canyon section is 32.2 river kilometers in length and extends from near the Wolf boat ramp upstream to near the mouth of Pine Creek. The upper section is 29.0 river kilometers in length and is located from near the mouth of Pine Creek upstream to Palisades Dam. We marked YCT, RBT, and BNT with three sizes of coded VHF radio transmitters. The larger transmitters were 12 x 53 mm, weighed 10 g in the air, and are expected to have a 528 d battery life with a 5 s burst rate for the signal. Twenty of these larger transmitters also included a motion sensor to aid in determining mortality rates. We used two different smaller sized transmitters. The non-motion sensitive tags measured 9.1 x 30.1 mm, weighed 4.5 g in the air, and are expected to have a 441 d battery life with a 5 s burst rate. The small, motion-sensitive tags we used measured 9.5 x 32 mm, weighed 4.5 g, and are expected to have a 196 d batter life. We placed 80% of the available transmitters in YCT, 15% in BNT, and 5% in RBT.

We used boat-mounted electrofishing gear to capture trout for radio tagging from March 7 to April 1 in the spring and from September 2 to October 22 in the fall. After radio tags were implanted, fish were held overnight in cages, and released the following day in the same river section they were captured. We surgically implanted radio transmitters into the body cavities of trout. Fish were measured to the nearest mm (total length) and weighed to the nearest g before being placed in an anesthetic bath. We weighed fish to ensure the transmitter used did not exceed 3% of the body weight of the fish (Brown et al. 1999). We anesthetized fish and placed them belly up in a tray mounted above the cooler once equilibrium was lost. We used a small battery-powered bilge pump attached to an adjustable hand-held sprinkler to keep the gills flooded with water during the surgery. With one worker constantly flushing the gills with water, a second would make a short 2–3 cm incision in through the body wall of the belly anterior to the pelvic fins. We then inserted a grooved director through the incision and back near the vent to shield internal organs from damage. We then inserted a catheter needle through the body wall near the vent onto the grooved director which we used to guide the point of the needle out through the incision anterior to the pelvic fins. Next we inserted the transmitter's antenna through the catheter needle and then removed the needle leaving the antenna trailing out of the fish. We used the antenna to gently pull the tag into the fish's body cavity. We also placed a PIT tag in the body cavity with the radio transmitter. Next, we used 3-0 nylon suture material to seal the incision using three to four sutures. We treated the incision and antenna wound with iodine and placed the fish in a bucket of fresh water for recovery. We recorded time to loss of equilibrium, surgery time, and recovery time (time to regain equilibrium) during each surgery. Once fish recovered equilibrium, we placed them in a PVC tube holding cage secured in calm water, and held tagged fish until the following day when they were released.

We monitored fish locations weekly through October, and then monthly in November and December. Fish locations were recorded using a hand-held GPS device during mobile tracking efforts. Proximity to a fish was estimated when power readings on the receiver exceeded 200 with the gain level at or below 50. When we were confident in the nearby location of a radio-tagged fish, we recorded the GPS location. We used jet boats, rafts, and trucks to track along the river and adjacent canals. We also used fixed receiver stations to monitor tagged fish movements throughout the system and to help fill in information gaps between mobile tracking surveys. Fixed receivers stations were placed in six locations: downstream of the Menan boat ramp, downstream of the Lorenzo boat ramp, at the Great Feeder irrigation diversion, at the Eagle Rock Canal diversion, at the Anderson Canal diversion, and near the mouth of Mud Creek. Fixed stations all had two or three antennas searching either a combination of canals and main river directions (two antenna set up) or a combination of main river upstream, main river downstream, and down canals for radio signals.

We recorded fish locations in an Access database which we used to summarize fish movements and final locations for 2014. We summarized fish movements by describing distance moved (river km) from the tagging location through December 2014 and the percentage of tagged fish entrained in the irrigation canal system.

We calculated entrainment rates into canals separately for 2013 and 2014. For 2013, entrainment rates were left uncorrected for natural mortality by simply dividing the number of radio-tagged fish entrained into canals by the total number of fish marked with radio-tags. For 2014, entrainment rates were based on the number of fish we determined were alive through the summer irrigation season (May – September). Only radio-tagged fish that were alive through the entirety of this period were included in the entrainment estimate; fish that died due to other causes during the timeframe were excluded from this analysis. These included fish that were marked in 2013 that were still alive (as evident by movement between location events) as well

as those new fish marked in March of 2014 with radio tags. Fish marked in 2013 were deemed still alive if they had moved more than one river mile between October 2013 and April 2014. Entrainment rates for 2014 were calculated separately for each species by study strata (lower river, canyon, and upper river). We used linear regression to assess the effect of canal flow on entrainment using the June flow values listed above as the predictor value. We also summarized the location of entrainment, and where sufficient data was available, the timing of entrainment as well.

We mapped the locations of radio-tagged trout during their respective spawning seasons. We inferred general spawning locations based on prior movements and "proximity" relocations assigned during the peak of spawning season and marked these on a map. The map included only one spawning location per individual fish. We also described the migratory behavior of each species by calculating the average distance in river miles from the tagging location to the spawning location as well as the minimum and maximum observed migrations.

RESULTS

South Fork Population Monitoring

We captured 1,301 trout at the Lorenzo monitoring reach, including 190 YCT, 33 RBT, and 1,078 BNT. Our abundance estimates include age-1 and older YCT (≥102 mm) and BNT (≥178 mm). We estimated YCT densities at 117 (± 27) fish/km (Figure 62). The trend for YCT abundance at Lorenzo from 1987 through 2003 was stable with an intrinsic rate of change (*r*) of -0.01 which was significantly different than zero at the α = 0.10 level (F = 0.153, df = 9, P = 0.71). In 2005, the abundance of YCT at Lorenzo did decrease below the long-term average to a low of 76 YCT/km. Since 2005, YCT abundance has increased, but these trends are not statistically significant at α = 0.10. The intrinsic rate of growth since 2005 was 0.10 (F = 3.475, df = 7, P = 0.11). The average W_r of YCT in the Lorenzo Reach was 95 (n = 143). Generally, YCT <250 mm had the highest W_r and they decreased as YCT size increased (Figure 63). Relative weights for each size class of YCT was similar among years as evidenced by overlapping 95% confidence intervals. We estimated BNT densities to be 854 (± 90) BNT per kilometer at Lorenzo (Table 20; Figure 62). The BNT population at Lorenzo has a significantly increasing trend in abundance from 1987 to 2003 with r = 0.09 (F = 17.488, df = 9, P = 0.003). Since 2005, after management actions changed on the South Fork, BNT abundance at Lorenzo have had a stable trend with r = -0.02, which was not significantly different than zero (F = 0.299, df = 9, P = 0.60). Relative weights were stable for BNT at Lorenzo except for the larger size classes in 2012, when relative weights were significantly higher for BNT in the 400 to 499 mm size group than in 2011 (Figure 64). In 2014, the average BNT relative weight was 94. We captured too few RBT to generate a population estimate using mark recapture techniques, but RBT did comprise 3% of the catch. Extrapolating 3% with the total trout estimate (971 trout/km) indicates RBT density is around 29 RBT/km at Lorenzo. Relative weights were also similar among size groups of RBT at Lorenzo between 2012 and 2014 (Figure 65), and averaged 99.

We captured 2,231 trout at the Conant monitoring reach. This included 865 YCT, 656 RBT, and 710 BNT. We estimated the total trout density at 2,473 (± 180) trout/km at Conant. We estimated there were 923 age-1 and older YCT/km (± 132) at Conant (Table 21; Figure 66). From 1982 through 2003 (prior to the current management approach), YCT have experienced a statistically significant decrease in abundance, with an intrinsic rate of growth of -0.04 (F = 11.697, df = 13, P = 0.005). Since management changed to the three-pronged management approach in 2004, YCT at Conant have experienced an increasing trend in abundance with r =

0.07 (F = 6.649, df = 10, P = 0.03). Relative weights for YCT at Conant were similar among years with YCT >300 mm having lower relative weights than smaller YCT (Figure 67). The average relative weight was 93. We estimated there to be 475 age-1 and older BNT/km (± 60) at Conant. Brown Trout abundance was stable between 1982 and 2003, with an intrinsic rate of growth of 0.01 (F = 0.542, df = 13, P = 0.48). Since 2004, BNT abundance at Conant has increased, (r = 0.09, F = 8.402, df = 10, P = 0.02). Brown Trout relative weights were stable among years as well and averaged 95 (Figure 68). We estimated there to be 880 age-1 and older RBT/km (± 172), and Rainbow Trout abundance has increased as well from 1982 through 2003 (r = 0.18, F = 85.489, df = 11, P =<0.001. From 2004 through 2014 RBT abundance continues to increase, with an intrinsic rate of growth at r = 0.09 (F = 10.574, df = 10, P = 0.010). Relative weights for RBT at Conant were more consistent across the size groups and generally higher than YCT. However, among the sample years relative weights did not differ as indicated by overlapping 95% confidence intervals (Figure 69). The average relative weight of RBT was 98.

At the new Lufkin reach, we captured 1,144 trout. This included 466 YCT, 243 RBT, and 435 BNT. For age-1 and older trout, we estimated there were 1,412 YCT/km (± 416), 773 BNT/km (± 199), and 343 RBT/km (± 115) in the Lufkin reach. The average relative weight for YCT was 95%, 94% for BNT, and 94% for RBT at Lufkin.

Visual inspection of length frequency plots and the abundance of relative size classes each year at Lorenzo suggest YCT recruitment has been fairly stable; however, sample sizes are not large, ranging from 190 to 251 fish captured each year (Figure 70). Sample sizes were even smaller for RBT at Lorenzo, such that RBT length frequency plots for 2010 through 2014 at Lorenzo were not informative and were not included in this report. We did catch numerous BNT at Lorenzo each year, and they exhibited a relatively stable recruitment trend based on age-1 abundance except for 2010 which appears to have had a relatively weak age-1 year class (Figure 71).

Yellowstone Cutthroat Trout at the Conant monitoring reach have had variable recruitment over the past five years as shown in length frequency plots based on the abundance of age-1 YCT (102 to 254 mm TL) (Figure 72). Age-1 YCT were relatively abundant in 2011, 2012, and 2013. This was not the case in 2014. Rainbow Trout recruitment over the previous five years has been stable (Figure 73) based on observations of age-1 RBT in length frequencies. Brown Trout at Conant had variable recruitment based on length frequency plots with relatively strong age-1 cohorts in 2011, 2012, and 2013 (Figure 74).

Biomass estimates for BNT and YCT at the Lorenzo monitoring reach have been variable over the past decade (Figure 75). Yellowstone Cutthroat Trout biomass at the Lorenzo monitoring reach have had an increasing trend, but this trend was not significant (F = 1.62, df = 7, P = 0.25). Conversely, BNT have exhibited and negative overall trend over the past decade at Lorenzo, and this trend was also not statistically significant (F = 2.71, df = 9, P = 0.13). Biomass of RBT at Lorenzo was too low to estimate accurately. Total trout biomass increased as trout abundance increased at Lorenzo (Figure 76).

At Conant, biomass estimates for YCT, RBT, and BNT all exhibited a positive trend over the past ten years (Figure 77). However, the positive trend in YCT biomass at Conant was not statistically significant (F = 0.40, df = 9, P = 0.56). The increasing trend of RBT biomass, however, was statistically significant (α = 0.10; (F = 4.50, df = 9, P = 0.07). Similarly, BNT had a significantly increasing trend in biomass (F = 7.14, df = 9, P = 0.03). Total biomass of trout at the Conant monitoring increased as total trout abundance increased (Figure 78).

Flows in the tributaries and main river did not explain the low level of recruitment of age-1 YCT and RBT observed at Conant. When more recent data were incorporated into the model developed by DeVita (2014), the “best” model used by DeVita (2014) based on Akaike Information Criterion corrected for small samples (AICc) scores, was still the “best” model, indicating the model was appropriate for the updated dataset. Tests for normality and variance assumptions also indicated the model was appropriate. The “best” model included main river winter flows and tributary rearing flows. While the “best” model did not change between 2012 and 2014, analysis of residuals indicated the model did not accurately predict abundance of age-1 YCT and RBT in 2014. In 2014, the abundance of age-1 YCT and RBT was 124.6% lower than predicted by the model, suggesting variables other than flow affected recruitment (Figure 79).

The abundance of age-1 YCT was not correlated with summer temperature or YCT spawner abundance, but was positively correlated with age-1 BNT abundance suggesting winter conditions that favored age-0 survival of BNT also benefitted age-0 YCT (Figure 80). Linear regression of log-transformed age-1 YCT and summer temperatures did not yield a significant result (F = 1.273, df = 25, P = 0.27). Similarly, the age-1 YCT abundance regressed with the abundance of YCT spawners was also not significant (Figure 81; F = 0.013, df = 19, P = 0.91). Age 1 YCT abundance was, however, significantly correlated with age-1 BNT abundance, but the relationship was positive (Figure 82; F = 4.789, df = 24, P = 0.04). Scatterplots of these analyses supported the statistical results.

Weirs

From April 1 through July 3 we captured 845 migrating trout at the Burns Creek weir, including 12 RBT (ten males and two females) and 833 YCT (428 males and 405 females). At Burns Creek, 18% of the male YCT captured at the trap fell back over the weir and were recaptured at the fish trap during the same spawning season. Female YCT fell back at a rate of 14%. We captured 50 fluvial-sized YCT upstream of the Burns Creek weir using backpack electrofishing gear, and found 45 of 50 were marked indicating they were handled at the fish weir. Thus, the 2014 trapping efficiency estimate for the Burns Creek weir was 90% (Table 22).

We operated the Pine Creek weir from April 7 through June 30, capturing a total of 906 fish of which seven were RBT (two males and five females). The 899 YCT included 305 males and 594 females. The fallback rates were 12% for male and 7% for female Cutthroat Trout. Upstream of the weir, we again used backpack electrofishing units to collect a sample of fluvial-sized fish and caught a total of 37 YCT, of which 26 had marks, so the 2014 efficiency estimate for the Pine Creek weir was 70%.

At the Palisades Creek weir, we caught a total of 63 RBT including 23 males and 40 females between April 2 and July 18. We also caught 309 male YCT and 425 female YCT in the trap. Fallback rates for male YCT were 17% and 9% for female Cutthroat Trout. A screened irrigation diversion near the Palisades Creek weir was used to collect out-migrant spawning fish and generate a weir efficiency estimate. Most of these fish (52 of 53) had marks indicating they were captured at the weir during their upstream migration, so the 2014 Palisades Creek electric weir efficiency estimate was 98%.

We operated the Rainey Creek weir from April 29 through June 25, capturing a total of 58 trout, two of which were female RBT. The remaining 56 fish included 12 male YCT and 44 female YCT. We did not observe any YCT that fell back through the Rainey Cr weir in 2014.

Radio Telemetry

We marked a total of 326 trout with radio transmitters in 2014. We placed 39% of these tags (126) in the lower section, 40% (130) in the canyon section, and the remaining 20% of the tags (70) in the upper section. We placed 80% of the available radio tags for each section in YCT, 15% in BNT, and 5% in RBT. In the lower section we marked 100 YCT, 20 BNT, and six RBT. In the canyon section we marked 104 YCT, 20 BNT, and six RBT. In the upper section, we marked 51 YCT, 13 BNT, and six RBT. The overall average surgery time was 3:48 from start to finish. We observed a single overnight mortality when returning the following day to release fish, which was caused by a drop in river level and not surgery-related mortality. Barring this mortality, overall surgery success was 100%.

In 2013, when this telemetry study was initiated, we tagged 331 trout and documented twenty radio-tagged fish were entrained into an irrigation canal. The uncorrected entrainment rates for 2013 were 7% for YCT (18 fish), 0% for BNT, and 11% for RBT (two fish). Of note, this first entrainment estimate did not include a full irrigation season, which may have underrepresented total entrainment.

We corrected entrainment rates for trout marked with radio tags in 2013 during the 2014 irrigation season based on the number of radio-tagged fish still alive in 2014. Between September 2013 and April 2014, 148 of the trout radio tagged in 2013 moved more than 1.6 river kilometer (our criteria for determining if a fish was alive) and were assumed to be alive going into the 2014 summer (irrigation) season. In addition to these trout that had moved a substantial distance, there were 40 radio-tagged fish with less than one river mile movement distances. We used the documented movements of our radio-tagged fish with motion sensitive tags in 2014 to determine if we should include some of the more sedentary fish in our entrainment analysis. We observed 38% of our trout marked with motion-sensitive tags were alive and had not moved more than 1 river mile during 2014. Thus, we included 163 fish originally marked in 2013 as part of our entrainment assessment for 2014, including the 148 fish we knew moved, and 15 (38%) of the sedentary group of fish we believe were likely alive, but did not meet our criteria for classification as alive. Only 13 of these radio-tagged fish were entrained in 2014. The corrected 2014 entrainment rates by species for trout marked in 2013 was 8% for YCT (eleven fish), 5% for BNT (one fish), and 17% for RBT (one fish).

There were 223 active (live) trout with radio tags during the 2014 irrigation season, and 27 (12%) were entrained into a canal between April 1 and the close of irrigation season on October 1. The entrainment rates for each study strata were 10% for radio-tagged fish in the lower river strata (15 out 153 trout), 16% for the canyon strata (8 out of 49 trout), and 19% for the upper river strata (four out of 21 trout). Entrainments rates were variable for each species (Table 23). In the lower strata, where all of canals are located, entrainment rates were 11% for YCT (13 of 119 trout), 14% for RBT (one out of seven trout), and 4% for BNT (one out of 27 trout). In the Canyon strata, entrainment rates were 17% for YCT (six out of 35 trout), 33% for RBT (one out of three trout), and 9% for BNT (one out of eleven trout). In the upper river strata, entrainment rates were 21% for YCT (three out of 14 trout), 0% for RBT (only a sample of one), and 17% for BNT (one out of six trout). Every canal entrained at least one radio-tagged fish in 2014. Of the 27 entrained trout in 2014, eight were entrained in the Dry Bed Canal, five in Reid Canal, three each in the Sunnydell, Farmers Friend, and Enterprise canals, two in Market Lake Canal, and one in both Eagle Rock Canal and Anderson Canal. Entrainment rates were not correlated with size of canal (F = 0.384, df = 6, P = 0.56). We were able to assign entrainment times for 17 of the 27 entrained fish. Of these, most entrainment occurred during the months of

June and July (41% each month or seven fish each) with the rest happening in May (18% or three fish). These were the only three months we documented entrainment in 2014.

Most of the occurrences of tagged trout encountering (passing) canal headgates took place during the irrigation withdrawal period from May through September (60% or 94 times) with 28% (44 times) in June and July. The remaining encounter dates occurred during the off-season from October through April. Fish moving upstream passed irrigation diversions primarily during early summer. Most encounters (33 of 48, or 69%) occurred between April and June (Figure 83). Encounters that involved downstream movements past irrigation diversions was more spread out through the summer months (Figure 84). We were able to document in real-time when two radio tagged trout were entrained.

We identified locations for 259 YCT during the spawning season. The average migration distance from the site of tagging was downstream an average of 3.2 river km. The range of YCT migrations from the initial capture location was -88.4 km downstream to +55.2 km upstream. Of the 259 YCT we observed during the spawning season, one YCT migrated into Burns Creek, seven YCT entered Pine Creek, and two entered Palisades Creek, suggesting as much as 96% of YCT spawn in the main river versus tributaries (assuming all of these tagged YCT spawned). The YCT in Burns Creek was tagged in the Canyon strata. In Pine Creek, two YCT were tagged in the Lower river strata, four were tagged in the Canyon strata, and one in the Upper river strata. Both YCT observed in Palisades Creek were tagged in the Upper river strata. There were fish originally tagged with radio tags in all three study strata present in each strata section during the spawning season.

We relocated 67 BNT during their spawning season, and assessed their migration distances and direction from the original point of tagging. The average migration distance was upstream 1.9 km, and ranged from -28.5 river km downstream to +66.9 river km upstream. BNT originally marked in all three river strata were observed in the Canyon and Upper river strata (Figure 85) during the spawn. We only observed BNT tagged in the Lower river and Canyon strata in the Lower river strata during the spawning season.

We identified general spawning locations for 17 RBT in 2014. All but one of these fish spawned in the same river strata they were marked (Figure 86). The average migration distance between the site of tagging and spawning for these RBT was -4.8 river km downstream, and ranged as high as -19.5 rkm downstream. The locations of these 17 RBT during spawning season was dispersed throughout the river instead of clustered in a few areas, and YCT were present in these areas during spawning season (Figure 87).

Spring Flows

Regression analysis between spring flows and trout recruitment yielded mixed results for correlations between spring maximum flows and abundance of age-1 YCT and RBT the following year. A positive relationship between maximum spring flow was correlated with age-1 YCT the following year (Figure 88; $F = 5.52$, df = 8, $P = 0.05$). Examination of the residuals from this model indicated the data were normally distributed. Maximum spring flows, however, were not statistically correlated with age-1 RBT the following year (Figure 89; $F = 0.80$, df = 9, $P = 0.40$).

South Fork Angler Incentive Program

In 2014, we marked 705 RBT with coded wire tags (CWT) between Palisades Dam and Heise for the Angler Incentive Program. We tagged 429 RBT with $50 tags, 199 with $100 tags, 51 with $200 tags, 20 with $500 tags, and 6 fish with $1,000 tags. A total of 170 anglers turned in 3,444 RBT in 2014. Overall, anglers turned in a median of three RBT and an average of five RBT. Of the 3,444 RBT brought in to IDFG there were 71 tagged fish. The tag values and number that were turned in were $50 (37), $100 (29), and $200 (five) for a total of $5,750. Based on the 17% compliance rate for turning in harvested Rainbow Trout estimated in 2012 (High et al. 2014), these Rainbow Trout likely represent 20,258 harvested rainbows in total.

PIT tags

We marked an additional 2,265 YCT with PIT tags in 2014, bringing the total number of marked YCT released in the South Fork since 2008 to 19,574. We marked YCT in the South Fork during four general field activities in 2014, including main river electrofishing during winter while marking angler incentive RBT, weir operations during spring spawning runs, tributary backpack electrofishing surveys during summer, and fall main river population surveys (Table 24). During these activities and with PIT tag array data we also recorded the recapture of 625 individual YCT.

Recapture data included information for juvenile YCT marked in one of the four main spawning tributaries and were helpful for describing life history strategies of these fluvial YCT. Since 2011 we have marked 901 juvenile YCT in Burns, Pine, and Palisades creeks which each have PIT tag arrays near their mouths. We have recorded recapture data for 96 of these juvenile fish at the various PIT array sites including 23 at the Burns Creek PIT array, one at the Pine Creek PIT array, and 72 at the Palisades Creek PIT array. The out-migration timing of these juvenile fish was variable, but generally occurred in spring or fall. In Burns Creek, most (52%) of the juvenile YCT out-migrated the fall of the year they were marked with PIT tags with a date range from September 29 to December 7. Eight juveniles (35%) out-migrated in both spring and fall of the following year with a mean departure date of July 18 (range of April 23 to June 24 for spring and August 2 to October 11 for the fall). A single juvenile YCT out-migrated from Burns Creek two years after being marked with a PIT tag on May 24. The single observed out-migrant for Pine Creek resided in the creek for a year, and left the next spring on May 21. In Palisades Creek, most (60%) of the out-migrating juvenile YCT left in the year following the year they were marked with PIT tags with an overall average date of July 21. Like Burns Creek, out-migrants from Palisades Creek left primarily in the spring (ranging from January 1 to June 18) and fall (ranging from August 2 to December 2). Many of the Palisades Creek juvenile YCT out-migrated in the fall after being tagged with a mean departure date of October 14 (dates ranged from September 12 to December 4), and a few left two years post-marking with a mean departure timing of June 6 ranging from January 6 to June 25 in the spring and a single out-migrant in the fall on November 28 (Table 25).

We observed high spawning stream fidelity for YCT. In 2014, we recaptured 87 YCT which had been recaptured or marked at a tributary weir during previous years. All 87 of these YCT returned to the same spawning tributary where they had been observed as many as three previous spawning years (Table 26). Most YCT recaptured at weirs with PIT tags (75%) were annual spawners. However, some individuals with three or more annual recapture events at a weir had missed a year. For the minority that were not annual spawners, alternate year spawning was the next most common interval for spawning YCT observed (21% or 18 fish).

One of these YCT was observed during three different spawning years. This fish was a male YCT observed in Palisades Creek during the 2010, 2012, and 2014 spawning runs.

DISCUSSION

South Fork Population Monitoring

Trout abundances in the South Fork at the Lorenzo site were lower in 2014 than in 2013. Trout populations vary annually because of a variety of factors including flows (Moller and Van Kirk 2003), temperature (Isaak and Hubert 2004), disease (Hedrick et al. 1998), genetics (Guinand et al. 2003), variable recruitment, angler harvest (but not in our case), and various other natural and anthropomorphic factors. While both YCT and BNT abundances decreased from 2013 to 2014 at Lorenzo, the ten-year trend since management actions changed on the river to the three-pronged management approach in 2004 indicate stable populations. Population estimates from recent years have been near all-time high levels. The decrease from 2013 to 2014 has brought abundances down to just below their respective ten-year averages, and should be viewed as a return to more normal abundances, as opposed to a decline in abundance that is cause for concern. These decreases were reflected in all trout species at Lorenzo at similar magnitude. The average species compositions from 2005 through 2013 were 13%, 1.4%, and 85% for YCT, RBT, and BNT, respectively. In 2014 the species composition were very similar to these averages at 15%, 2.5%, and 83% YCT, RBT, and BNT, respectively.

Decreases in trout abundances at the Conant monitoring reach from 2013 to 2014 were similar to those at the Lorenzo monitoring reach, and were reflected in all trout species. Like Lorenzo, this suggests that the variable(s) responsible for this decrease in abundance was an environmental variable that all fish were subjected to. We hypothesize that flows may have played a role in this population fluctuation, as this metric has historically had the biggest impact on trout abundance in the South Fork. When we tested the model developed by DeVita (2014), we found that adding 2014 data did not change which of the available flow models fit the data best. This indicated that winter river flows as well as tributary flows during the early rearing period for YCT are most influential, and that this relationship has not changed. However, 2014 was somewhat of an outlier and did not fit the model well. As such, we concluded something other than flows affected recruitment in 2014. As such, we tested relationships of recruitment with other available data, including stream temperature, adult spawner abundance, and abundance of age-1 BNT. However, we did not find significant correlations which would explain lower abundance estimates of trout in 2014. We did observe a positive relationship between age-1 YCT and age-1 BNT, but abundances of both species declined significantly in 2014. Thus, the relationship observed likely resulted from environmental variables, such as winter flows, that benefits both species. In summary, we were unable to determine the cause of the decline in all species of trout in 2014. Trout abundances are now back at historical averages, and are not cause for concern at this time. Future monitoring is warranted to identify and better predict other issues that are affecting trout abundances.

Although trout abundances decreased between the two most recent surveys at Conant, trends since management actions switched to the three-pronged management approach have been increasing overall. Thus, the overall result of the three-pronged management approach has been significantly more trout in the river. While RBT population trends have been increasing since 2004, the rate of population growth has not been as high as would be expected if the three-pronged management approach was not occurring (DeVita et al. 2014). Management efforts have likely limited the RBT population growth rate, but efforts to cause a decrease in

RBT abundance to mid-1990 levels (no more than 10% species composition) as stated in the state fisheries management plan (IDFG 2013) have not yet been successful. Currently, RBT are 30% of the species composition at the Conant monitoring site. Across their native range, YCT have not persisted as strong populations when RBT are abundant (Allendorf and Leary 1988; Hitt et al. 2003; Gunnell et al. 2008; Muhlfeld et al. 2009; Seiler and Keeley 2007a; Seiler and Keeley 2007b). Yellowstone Cutthroat Trout are still abundant in the South Fork at the Conant monitoring site, but RBT continue to pose a threat to their persistence. We added a new electrofishing reach to the South Fork approximate midway between Lorenzo and Conant. This new reach (Lufkin) will provide valuable data for future comparisons in this stretch of the South Fork where fish species compositions start to change towards a BNT dominated versus a YCT/RBT dominated stretch of river.

Relative weights for all trout species in the Lorenzo and Conant monitoring reaches were similar among years and were high compared to published standards. With average relative weights ranging from 93% to 99% and the fact that they do not differ among the years suggest that the South Fork is conducive for growing healthy trout on a consistent basis. Relative weights were also very similar for the same species of fish between the two monitoring reaches, indicating growth rates are similar for fish throughout the river. Even though populations decreased in 2014, relative weights did not change, indicating the change in abundance was not due to reduced diet. Further, reduced abundance of trout did not increase relative weights in 2014 either, indicating growth rates are not only stable, but are near the maximum level for trout in the South Fork as presumed increase food availability, because of a reduced fish population, did not increase relative weights.

Biomass estimates were variable between monitoring reaches and among years. Total trout biomass at the Lorenzo monitoring reach closely mimicked changes in estimated total trout abundances over the same time period. This indicates biomass at Lorenzo is density dependent. When trout are abundant, biomass is high, and vice versa. It also suggests that carrying capacity is not causing the fluctuations in total trout abundance. In other words, we did not observe constant biomass estimates among years with varying trout abundances. Thus, the lower biomass estimate for 2014 was a function of lower trout abundance.

Similar to the Lorenzo reach, biomass estimates for all three trout species dropped in 2014 at the Conant monitoring reach. However, the decreases brought the 2014 biomass estimates down to the ten-year average instead of below it like at Lorenzo. Cutthroat comprise the largest portion of biomass at Conant, but both RBT and BNT have experienced significantly positive increases in biomass over the past decade. The average size for RBT at Conant is increasing. RBT abundance has remained constant from 2010 through 2013, but biomass was increasing over that three-year period. This is likely because the large year class produced in 2009 has matured, resulting in relatively larger individuals in the population. Total trout biomass mimicked changes in total trout abundance estimates at Conant, indicating biomass estimates are density dependent and carrying capacity is not the primary driver of fluctuations in abundance.

Weirs

This was the first year since 2010 that we were able to effectively operate weirs and traps on all four major spawning tributaries. Rainey Creek was not effectively trapping fish prior to 2014 due to insufficient flows through the trap (High et al. 2015) and was not run at all in 2012 or 2013. In partnership with the Caribou-Targhee National Forest, we modified stream flows in the vicinity of the weir just prior to the beginning of the spawning runs for RBT and YCT.

Stream flows were directed toward the fish trap and water velocities were increased. These efforts allowed us to operate the Rainey Creek weir throughout the spawning run. The 56 YCT captured at the Rainey Creek weir in 2014 was the third highest number of YCT caught in Rainey Creek out of ten years of trapping efforts.

Trapping efficiencies and catch were both high at the Burns and Palisades weirs. The YCT spawning runs were average or above in both streams in 2014 and trapping efficiencies exceeded 90%. Palisades Creek was once again where we caught the largest number of RBT out of the four tributaries. To stop RBT from migrating upstream into Palisades Creek outside of the spawning season and operation of the weir, we maintained the screen boards in the fish trap, but removed the funnel trap entrance. We also did not keep the barrier electrified once the spawning run concluded as by this time the Palisades Canal operators had installed enough dam boards at the site to essentially create a physical barrier. With the barrier across the channel and the screens still in place at the upstream edge of the trap box, we are hoping to see a bigger reduction in RBT composition in the two miles of Palisades Creek immediately upstream of the weir. IDFG research staff are currently manually removing Rainbow Trout from Palisades Creek and further upstream to Lower Palisades Lake. Thus, keeping a barrier to RBT immigration into this study area is essential to the research project and to preventing upstream migration of RBT, and may prove key to maintaining the YCT genetic diversity and population viability in Palisades Creek.

Unlike Burns and Palisades creeks, the run size at Pine Creek was much lower than expected. Pine Creek is usually home to the largest YCT spawning runs with numbers usually more than double that of Burns or Palisades creeks. However, in 2014, the total runs size was comparable to both of these streams. Trapping efficiencies were also lower than last year. It is possible that we had a number of factors adversely affecting the spawning run at Pine Creek. First, throughout the spawning run there was a bridge construction project underway a couple miles downstream of the weir on Highway 31. While the construction crew did have sediment fences in place, it is possible that the construction activities affect YCT migratory behavior. Second, YCT were somehow getting past the weir without entering the trap. While trapping efficiency was not low enough to make up the difference in expected versus observed run size, it was much lower than the anticipated 90%. We did not observe YCT escaping past the electric weir. While this may have been possible, it seems unlikely that it was a big factor in our low run estimates, given the amount of time we were present during the active part of the run. Alternatively, a side channel that connects Pine Creek from downstream of the weir to upstream bypassing the weir, seems the more likely explanation. This side channel is only connected in high water, and we annually attempt to keep it blocked to fluvial YCT using a picket weir. The picket weir appeared to be in place and functioning throughout the spawning run, but this small weir should be evaluated in the future as it may have allowed fish to pass during high flows.

Radio Telemetry

We observed radio-tagged trout of all species and from all sections of the South Fork entrained into canals diverted off the main river in 2014. This was the first year we had radio-tagged fish in the river system throughout the irrigation withdrawal period (May through October). We expected entrainment rates to be highest for the lower river that includes the diversion structures for the eight large unscreened canals off the South Fork. Interestingly, entrainment rates for radio-tagged fish marked in this section were actually the lowest entrainment rates of all three river reaches, at 10%. Entrainment rates increased progressively upstream with the Upper River strata having the highest overall entrainment rate of 19% with the Canyon strata intermediate at 16% entrainment. The entrainment rate for fish in the Upper

River strata may be biased due to a low sample size (21 tagged fish), but this seems unlikely because entrainments rates between the Upper River and Canyon strata were similar, and the Canyon strata had a much larger sample size (41 tagged fish).

Entrainment occurred primarily in early summer during the months of June and July. These also happen to be the months when irrigation withdrawals were at their peak. All three species of trout marked with radio tags were entrained into canals in 2014, but the rate differed among species. Brown Trout had the lowest overall entrainment rate as a species while YCT, and RBT entrainment rates were higher. It is possible that trout movement patterns related to spawning are a substantial factor affecting entrainment. Both YCT and RBT spawn in the spring and are moving through the river drainage from mid-May through mid-July (Schoby et al. 2013). This active movement time coincided with the months (June and July) when we observed the highest entrainment rates in 2014. Although nearly half of the entrainment occurred during June and July, encounter events when radio-tagged fish swam past canal headgates was proportionally lower in June and July, at 28%. During these two months, most fish encountering headgates were traveling downstream. Given the timing (post spawning) and the direction of travel for the majority of the encounters (downstream) when the bulk of entrainment occurred, it is possible post-spawn fish were more susceptible to entrainment. In addition to potentially increased risks of entrainment due to more frequent diversion encounters associated with spawning migrations, spring spawning YCT may be more at risk to entrainment during the post-spawning migrations when energy resources are depleted and avoidance of canals may be more difficult. Conversely, BNT did not exhibit lengthy movements during the early summer when irrigation withdrawals were high. As fall spawners, their increased movement patters generally did not start until late summer/early fall when canals were diverting less water or shutting down for the season. This is likely the reason why entrainment rates for radio-tagged BNT were less than those for RBT or YCT.

All seven canals departing from the South Fork Snake River and one off the mainstem Snake River downstream of the mouth of the South Fork (the Market Lake Canal) entrained radio-tagged trout. The relative size of the canal was not a good predictor for entrainment rates based on our regression analysis. A few canals entrained proportionally more radio-tagged trout than others. The Reid Canal entrained the second highest number or radio-tagged trout (5 or 19%) despite the fact that it is the second smallest canal. Similarly, despite being the smallest canal, the Sunnydell Canal entrained 3 radio-tagged trout (11% of the fish entrained) which was more than the Anderson, Enterprise, and Market Lake canals. Thus, other canal characteristics likely drive entrainment or entrainment is infrequent enough that these are statistical anomalies. Headgate angle and location relative to the river thalweg are two variables that are known to affect entrainment (Bahn 2007). The Dry Bed Canal entrained the most radio-tagged fish (8 or 30% of the fish entrained), and is the largest canal in the system. It also has a headgate angle perpendicular to the river flow and adjacent to the thalweg. The Enterprise Canal is immediately upstream of the Dry Bed, and the Farmers Friend is nearby as well, both with headgate angles similar to the Dry Bed, but a little farther away from the thalweg. When the Dry Bed, Farmers Friend, and Enterprise canals are combined they were responsible for 52% of the entrainment of radio-tagged fish in 2014.

Most often, when radio-tagged fish encountered a canal diversion they were not entrained. With fixed receiver data, we only observed two encounters out of 156 during the irrigation season that resulted in entrainment. Most of these encounters occurred during the shoulder seasons of irrigation withdrawals. Only 28% of the encounters happened during June and July. The direction of trout movement, also likely affected the risk of entrainment during encounters with irrigation diversions. During June and July, trout passing canals were moving

upstream 39% of the time. It is possible that fish moving upstream are at a lower risk of entrainment than trout moving downstream. The timing of when fish passed irrigation diversions moving downstream was more evenly distributed from May through October.

In general, entrainment effects on adult trout in the South Fork appear to be minimal. Trout populations are very dynamic and often have relatively high (40-50%) levels of natural mortality. In the South Fork, DeVita (2014) estimated annual mortality to be 32% for YCT and 56% for RBT. These estimated annual mortality rates are much higher than the entrainment rates we observed (10–19%). It is logical that entrainment rates are a density dependent function, and that this density dependent process leads to compensatory responses in annual mortality. We believe that at current entrainment rates, compensatory responses to annual mortality of trout in the South Fork offset the losses caused by entrainment and minimizes the effects of adult entrainment into canals.

Yellowstone Cutthroat Trout moved farther than either BNT or RBT did during the spawning season in both upstream and downstream directions, and were the only species observed in tributaries during the spawning season. Although sample sizes are smaller, the single fish that entered Burns Creek and the two that entered Palisades Creek were all marked in the river strata that the respective spawning tributary flows in to. Roughly half (51%), of YCT were relocated during the spawning season within 3 river miles of where they were tagged. It is possible that some of these YCT did not spawn in 2014, and thus would negatively bias the average spawning migration distance estimates, as many YCT migrated much further. However, based on PIT tag data for returning YCT at tributary weirs, YCT spawn on average every 1.3 years in tributaries of the South Fork (IDFG unpublished data). Based on observations of radio-tagged YCT during the peak of spawning season and spawning frequency observed for PIT-tagged YCT returning to tributary weirs, the vast majority (up to 96%) of YCT spawn in the main river. This is true even for YCT in the lower portions of the South Fork. During the spawning season, YCT were often observed near shelves where riffles would drop off into deeper runs and pools, especially when these shelves were off the main river thalweg such as side channels or inside bends and coves. When we mapped the locations of individual YCT in the South Fork during the spawning season, we observed fish scattered throughout the length of the river instead of localized into spawning congregations. This suggests that YCT find suitable spawning habitat throughout the length of South Fork.

Brown Trout, like YCT, also migrated to spawning areas in both the downstream and upstream directions, and they moved less distance than YCT but greater distance than RBT. Brown Trout that moved less than three river miles from marking location to spawning location comprised 54% of the sample. As with YCT, we did not observe large groups of radio-tagged BNT in the South Fork during the spawning season suggesting BNT also find suitable spawning habitat throughout the length of the river.

Rainbow Trout were the most sedentary of the three species during the spawning seasons with 65% of the observed trout moving less than three river miles. Similar to both BNT, and YCT, RBT were also spread out throughout the river drainage during spawning season and specific locations where hybridization risks with river-spawning YCT could not be identified due to the dispersed nature of both RBT and YCT locations during the spawning season. As with YCT, RBT were also associated with shelves off riffles into deeper runs and pools.

Spring Flows

Increases in spring flows benefit YCT recruitment, but are not necessarily correlated with reduced RBT recruitment. Since 2004, increases in maximum spring flows are correlated with increasing abundance of age-1 YCT the following year. Flows during the past decade ranged from 396 to 668 m^3/s. The relationship between higher maximum spring flows and higher age-1 YCT recruitment are likely related to the fact that YCT use increasing spring flows as a spawning cue (Thurow and King 1994; Henderson et al. 2000). Tributary flows are also likely related to higher snowpack and increased tributary flows which benefit YCT recruitment in spawning tributaries (Varley and Gresswell 1988). The abundance of age-1 RBT was not significantly correlated with flows, suggesting maximum flows did not reach levels sufficient to disturb developing embryos or displace newly emerging fry. This finding corroborates previous studies on the South Fork that indicated spring flows in 2005 peaking at 422 m^3/s were not sufficient to move small radio transmitters placed in RBT redds (Schrader and Fredericks 2006) and that South Fork riverbed material is not mobilized until flow reach 736 m^3/s (Hauer et al. 2004). Previous studies performed on the South Fork indicate flows in excess of 708 cms are required for geomorphic processes to start altering stream channels (Hauer et al. 2004) or providing the most benefit to YCT (Moller and Van Kirk 2003). While we could not detect a statistically significant correlation between maximum spring river flows and age-1 RBT abundance the following year, our dataset does not include maximum flows that are near the range suggested by Hauer et al. (2004), which clouds this relationship.

South Fork Angler Incentive Study

The South Fork Angler Incentive Program plays an important role in IDFG's management of YCT in the South Fork. This Program provides a tool for outreach and education about the importance of Yellowstone Cutthroat Trout conservation in the South Fork. This of itself may be enough justification for how much benefit is derived given the program's low operational costs. However, recent population modeling efforts for how YCT populations respond to different levels of harvest and different scenarios of spring flows, indicate the Angler Incentive Program as part of the three-pronged management efforts on the South Fork is one of the key factors that is limiting the rate of RBT population growth, and has the potential to cause a population decline, particularly if harvest levels are increased (DeVita et al. 2015).

PIT tags

Information collected from PIT-tagged YCT indicates juvenile Yellowstone Cutthroat Trout out-migrate from natal streams primarily in the spring and fall, but at varying years post-hatch. In Burns Creek, the majority of the juvenile YCT that were marked with PIT tags out-migrated from the stream in the fall of the same year they were marked with tags. We were marking YCT >100 mm TL, so these YCT were likely age-2 at the time of tagging and out-migration. This corroborates data for Burns Creek from 1981 where Moore and Schill captured out-migrating YCT ranging in size from 95 to 230 mm (Moore and Schill 1984). In Palisades Creek, we observed most of the age-2+ YCT marked with PIT tags remain in the tributary to out-migrate the following year as age-3 YCT. Again, we observed that out-migration occurred in the spring and fall. For Palisades Creek, Moore and Schill (1984) reported the majority of the out-migrating fry occurred in fall with a peak in late September, but they were unable to operate a weir from mid-April through the end of June due to high flows. In 1981, Moore and Schill (1984) postulated that juveniles that out-migrated in late fall and winter were the driving factor behind the functional fluvial YCT run in Palisades Creek, as they believed most of the out-migrants in the summer were entrained and lost in Palisades Canal. Earlier observations for

Palisades Creek indicated entrainment of trout was a significant source of loss (Jeppson 1970). We observed 30% of the out-migrants in our study leaving Palisades Creek from November through March, including six individuals (9%) that out-migrated in January and February. The discrepancy in out-migration timing for age-2 YCT in Burns and Palisades creeks may be related to available habitat. Moore and Schill (1984) documented that most YCT out-migrate from South Fork tributaries as age-0 fry. As average August stream flow for Burns Creek is 15% that of Palisades Creek, it is likely that more age-2 YCT can find suitable habitat in Palisades Creek and thus would not need to out-migrate to the main river. Future work should continue to investigate out-migration timing and magnitude in all of the spawning tributaries to determine which life strategies are most successful to adding to the main river YCT population. We marked numerous juvenile YCT in Pine Creek in 2014, and expect additional data to become available in the coming years. Furthermore, as we continue to monitor these PIT-tagged YCT we can evaluate how many of these stream-spawned YCT return to their natal streams (High et al. 2015).

We again documented high spawning stream fidelity for adult fluvial YCT in the South Fork. In 2014, all of the YCT recaptured at a tributary weir which had previously been observed in a spawning tributary during spawning season, returned to the same tributary. High spawning fidelity is not uncommon among salmonids, and has been used as a means for monitoring trends in fish populations. High fidelity in South Fork spawning tributaries suggests that spawning run sizes in Burns, Pine, and Palisades creeks should continue at current levels provided the mainstem population of YCT remains constant as well. This high site fidelity may be a limiting factor for the YCT spawning run in Rainey Creek, however. Estimated run sizes in Rainey Creek have varied from zero to 145 between 2001 and 2014, suggesting the fluvial YCT spawning population here is likely depressed. As fluvial YCT in the South Fork do not often stray, it is possible the Rainey Creek fluvial YCT spawning run may need some type of management intervention. We documented 75% of the fluvial YCT that spawn in tributaries, spawn on an annual basis. However, this estimate should be viewed as conservative. Although much more effective than previous weir designs, our current weirs do not capture every upstream migrant every year. Thus, it is possible that some YCT passed by the weirs without detection, and spawned, which when captured the following year, would mistakenly be described as an alternate year spawner. Additionally, YCT may short stop the weirs and spawn in the tributaries below the weirs without being detected, which would negatively bias spawning frequency estimates as well. Thus, while we observed 75% of the YCT spawn annually in YCT tributaries, it is possibly higher than that.

In summary, the current status of the South Fork fishery is good but opportunities to improve the fishery exist in terms of continued evaluation of entrainment in large canals, increasing angler harvest of RBT, and enhancing tributary spawning runs. Although populations have returned to average densities following several years of exceptionally high abundance, angler reports are generally positive. Current research will help provide critical answers to move management forward in coming years. Yellowstone Cutthroat Trout will continue to face risks to their dominance from a sympatric, robust RBT population, but consistent and adaptive management can maintain long-term population viability and a continued upward trend in abundance.

RECOMMENDATIONS

1. Continue to monitor effects of spring freshets, the operation of tributary weirs, and angler harvest of RBT on RBT, YCT, and BNT populations and adjust management actions accordingly.
2. Continue to use tributary weirs to protect spawning YCT in South Fork tributaries from risks of hybridization and competition.
3. Increase efforts that encourage anglers to participate in the Angler Incentive Program.
4. Remove resident RBT from Palisades Creek for at least another year to determine if manual removal efforts reduce introgression rates. Work with private landowners along lower Palisades Creek to replace RBT in private ponds with YCT or sterile RBT.
5. Remove resident RBT and BNT from Burns Creek upstream of the fish trap using backpack electrofishing similar to Palisades Creek.
6. Work with the US Bureau of Reclamation to obtain flows close to 708 m^3/s (25,000 cfs) during RBT fry emergence.
7. Continue marking YCT with PIT tags in the South Fork drainage to assess spawning stream fidelity, spawning periodicity, tributary use and duration, general movement patterns, and population size and growth rates using an open population model.
8. Continue to assess entrainment rates into large irrigation canals using radio telemetry.

Table 20. Summary statistics from the Lorenzo monitoring site between 1987 and 2014 on the South Fork Snake River.

	Yellowstone cutthroat trout							Rainbow trout							Brown trout							Total trout							
Year	M	C	R	R/C	YCT/Km	SD	CV	M	C	R	R/C	RBT/Km	SD	CV	M	C	R	R/C	BNT/Km	SD	CV	M	C	R	R/C	trout/Km	SD	CV	Mean Q (cms)
1987	146	63	6	9.5	422	207	0.25	2	0	0	0.0				225	102	12	11.8	531	160	0.15	380	168	18	10.7	970	99	0.10	64
1988	133	88	13	14.8	187	47	0.13	3	2	0	0.0				241	130	23	17.7	300	88	0.15	386	225	36	16.0	529	50	0.09	33
1989	119	74	13	17.6	248	98	0.20	1	2	0	0.0				199	97	22	22.7	185	38	0.10	377	204	35	17.2	677	60	0.09	25
1990	208	91	12	13.2	308	145	0.24	2	0	0	0.0				260	93	23	24.7	272	99	0.18	549	240	35	14.6	949	75	0.08	68
1991	199	175	17	9.7	445	146	0.17	0	6	0	0.0				319	234	47	20.1	369	56	0.08	560	474	64	13.5	953	67	0.07	71
1992																													
1993	144	201	18	9.0	487	155	0.16	6	8	0	0.0				238	270	27	10.0	555	105	0.10	420	531	45	8.5	1,213	74	0.06	57
1994																													
1995	264	196	22	11.2	568	116	0.10	4	5	0	0.0				325	341	41	12.0	639	101	0.08	677	731	66	9.0	1,587	73	0.05	36
1996																													
1997																													
1998																													
1999	194	163	26	16.0	335	81	0.12	3	4	0	0.0				500	588	55	9.4	1,150	161	0.07	711	798	82	10.3	1,485	74	0.05	67
2000																													
2001																													
2002	108	138	14	10.1	246	65	0.13	4	3	1	33.3				457	579	61	10.5	1,030	117	0.06	582	750	76	10.1	1,385	66	0.05	98
2003	90	81	11	13.6	237	133	0.29	2	2	0	0.0				557	432	61	14.1	926	110	0.06	668	593	72	12.1	1,184	61	0.05	81
2004																													
2005	37	47	4	8.5	76	54	0.36	5	2	0	0.0				440	486	67	13.8	771	91	0.06	641	569	71	12.5	2,030	96	0.05	78
2006	112	71	14	19.7	116	25	0.11	10	12	1	8.3				1154	933	140	15.0	1,761	148	0.04	1,326	1,064	155	14.6	2,116	77	0.04	
2007	90	41	2	4.9				17	6	0	0.0				764	446	67	15.0	1,125	110	0.05	888	525	69	13.1	1,504	70	0.05	131
2008	30	34	0	0.0				2	2	0	0.0				373	365	40	11.0	778	132	0.09	415	418	40	9.6	988	77	0.08	157
2009	77	110	10	9.1	218	93	0.22	13	10	1	10.0				603	739	104	14.1	915	90	0.05	718	916	117	12.8	1,236	53	0.04	92
2010	110	91	10	11.0	233	83	0.18	8	11	1	9.1				600	545	110	20.2	653	49	0.04	735	790	121	15.3	956	34	0.04	91
2011	134	126	12	9.5	279	132	0.24	12	17	0	0.0				323	365	27	7.4	1,058	241	0.12	495	544	39	7.2	1,770	153	0.09	107
2012	134	106	10	9.43	321	93.3	0.15	5	11	0	0.0				437	435	51	11.7	784	99	0.06	607	642	61	9.5	1,329	66	0.05	93
2013	150	167	25	15	299	72.1	0.12	17	27	0	0.0				838	714	108	15.1	1,200	121	0.05	1,005	908	133	14.6	1,826	136	0.04	66
2014	97	98	21	21.4	117.292	26.9	0.12	20	14	1	7.1				589	481	72	15	854	90	0.05	706	593	94	15.9	1,203	95	0.04	92

Table 21. Summary statistics from the Conant monitoring site between 1982 and 2014 on the South Fork Snake River.

	Yellowstone cutthroat trout							Rainbow trout							Brown trout							Total trout							
Year	M	C	R	R/C	YCT/Km	SD	CV	M	C	R	R/C	RBT/Km	SD	CV	M	C	R	R/C	BRN/Km	x SD	CV	M	C	R	R/C	trout/Km	SD	CV	Mean Q (cms)
1982					1,899							26							412										
1983																													
1984																													
1985																													
1986	1,170	546	70	12.8	2,890	402	0.07	32	16	2	12.5				183	105	8	7.6	641	253	0.20	1,385	667	80	0.12	2,351	236	0.10	102
1987	281							5							26							312							26
1988	1,100	561	98	17.5	1,491	148	0.05	41	18	1	5.6				113	46	4	8.7	340	310	0.47	1,254	625	103	0.16	1,836	88	0.05	103
1989	1,416	1,050	200	19.0	1,610	108	0.03	57	55	10	18.2	63	26	0.21	92	76	11	14.5	191	162	0.43	1,565	1,181	221	0.19	1,791	54	0.03	86
1990	1,733	1,522	317	20.8	2,330	173	0.04	113	109	14	12.8	204	64	0.16	173	117	12	10.3	369	133	0.18	2,019	1,748	343	0.20	2,984	89	0.03	101
1991	1,145	625	140	22.4	1,399	136	0.05	98	54	9	16.7	134	54	0.20	150	119	19	16.0	195	52	0.14	1,393	798	168	0.21	1,616	58	0.04	132
1992	595							34							76							705							60
1993	972	623	100	16.1	1,512	150	0.05	74	41	6	14.6	110	51	0.24	101	64	10	15.6	135	78	0.29	1,147	728	116	0.16	1,643	66	0.04	91
1994	853							87							110							1,050							52
1995	631	542	77	14.2	1,230	147	0.06	130	140	17	12.1	270	72	0.14	150	108	13	12.0	294	176	0.31	911	790	107	0.14	1,696	79	0.05	93
1996	707	548	72	13.1	1,502	225	0.08	155	111	5	4.5	594	420	0.36	212	124	18	14.5	314	78	0.13	1,074	783	95	0.12	2,292	131	0.06	107
1997	910	895	164	18.3	1,145	76	0.03	429	467	72	15.4	604	73	0.06	344	281	82	29.2	369	203	0.28	1,683	1,643	318	0.19	1,969	48	0.02	85
1998	674	682	61	8.9	1,691	204	0.06	216	247	26	10.5	461	79	0.09	257	216	49	22.7	249	36	0.07	1,147	1,145	136	0.12	2,191	79	0.04	110
1999	1,019	883	117	13.3	1,847	163	0.04	345	241	29	12.0	654	127	0.10	293	241	31	12.9	512	169	0.17	1,657	1,365	177	0.13	2,827	90	0.03	110
2000	797							260							133							1,190							91
2001	776							321							208							1,305							117
2002	495	394	50	12.7	841	119	0.07	295	257	24	9.3	785	195	0.13	111	104	9	8.7	288	122	0.22	901	755	83	0.11	1,803	81	0.05	72
2003	422	571	72	12.6	840	119	0.07	272	360	29	8.1	931	226	0.12	143	165	27	16.4	240	99	0.21	837	1,096	128	0.12	1,821	67	0.04	108
2004	315	379	51	13.5	478	61	0.07	227	304	29	9.5	530	104	0.10	169	202	22	10.9	383	204	0.27	711	885	102	0.12	1,441	62	0.04	114
2005	391	254	30	11.8	658	205	0.16	172	142	11	7.7	421	211	0.26	115	95	10	10.5	206	105	0.26	678	491	51	0.10	1,588	200	0.13	106
2006	423	365	54	14.8	749	104	0.07	289	251	23	9.2	677	178	0.13	215	223	31	13.9	329	70	0.11	927	839	108	0.13	1,938	80	0.04	
2007	784	568	72	12.7	1,380	142	0.05	565	361	52	14.4	825	113	0.07	404	289	50	17.3	530	117	0.11	1,753	1,218	174	0.14	2,713	87	0.03	116
2008	377	554	51	9.2	1,065	156	0.07	187	318	25	7.9	574	108	0.10	205	253	29	11.5	380	57	0.08	769	1,125	105	0.09	1,882	74	0.04	170
2009	623	489	90	18.4	826	87	0.05	475	425	34	8.0	1,408	302	0.11	261	219	42	19.2	307	48	0.08	1,359	1,133	166	0.15	2,276	80	0.04	98
2010	389	307	27	8.8	1,211	284	0.12	286	139	7	5.0	1,174	666	0.29	178	154	14	9.1	479	136	0.15	853	600	48	0.08	2,295	297	0.13	127
2011	609	429	70	16.3	1,225	221	0.09	448	311	28	9.0	1,190	256	0.11	357	300	29	9.7	796	166	0.11	1,414	1,040	127	0.12	3,002	142	0.05	99
2012	721	601	102	17	1,059	104	0.05	445	518	44	8.49	1,198	177	0.08	561	573	75	13.1	892	111	0.06	1,727	1,692	221	13.06	18,306	4225	0.12	105
2013	784	536	73	13.6	1,401	159	0.06	578	393	52	13.2	1,180	334	0.14	538	314	52	16.6	752	212	0.14	1900	1243	177	14.2	3,136	241	0.4	62
2014	488	415	50	12	923	132	0.07	350	265	28	10.6	880	172	0.1	382	273	46	16.8	475	60	0.06	1220	953	124	13	2,473	180	0.4	77

Table 22. Summary tributary fish trap operation dates, efficiencies, and catches from 2001 through 2014.

Location and year	Weir type	Operation dates	Estimated weir efficiency (%)[a]	Catch		
				Cutthroat trout	Rainbow trout	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		
2007	ND	ND	ND	ND	ND	ND
2008	ND	ND	ND	ND	ND	ND
2009	Fall/velocity	Apirl 9 - July 22	98	1,491	2	1,493
2010	Fall/velocity	March 26 - July 14	100	1,550	2	1,552
2011	Fall/velocity	March 23 - July 12	90	891	5	896
2012	Fall/velocity	March 24 - July 11	90	496	0	496
2013	Fall/velocity	April 4 - July 2	98	888	6	894
2014	Fall/velocity	April 1 - July 3	90	833	12	845
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	Apirl 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
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

[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

Table 22 continued. Summary tributary fish trap operation dates, efficiencies, and catches from 2001 through 2014.

Location and year	Weir type	Operation dates	Estimated weir efficiency (%)[a]	Catch		
				Cutthroat trout	Rainbow trout	Total
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
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
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

[a]Weir efficiency was estimated using several different methods

[b]From Host (2003)

[c]NE = no estimate

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

Table 23. Entrainment rates of Yellowstone Cutthroat Trout (YCT), Brown Trout (BNT), and Rainbow Trout (RBT) from the South Fork Snake River into canals diverted from the river in 2014 relative to where fish were captured and marked with radio transmitters. Sample sizes including the number of entrained fish over the number of fish alive in the river through the 2014 irrigation season are provided in parentheses.

	Entrainment Rates					
Species	Lower river		Canyon		Upper river	
YCT	10.9%	(13/119)	17.1%	(6/35)	21.4%	(3/14)
BNT	3.7%	(1/27)	33.3%	(1/3)	16.7%	(1/6)
RBT	14.3%	(1/7)	9.1%	(1/11)	0%	(0/1)

Table 24. Locations and the number of YCT marked with PIT tags and the number of YCT recaptured that were previously marked with PIT tags.

Location	Marked	Recaptured
Main river	678	57
Burns Cr weir	194	108
Pine Cr weir	200	49
Rainey Cr weir	55	1
Palisades Cr weir	200	75
Burns Creek	193	36
Pine Creek	49	5
Palisades Creek	213	28
Lorenzo	143	28
Lufkin	145	13
Conant	195	42
Burns PIT array	NA	63
Pine PIT array	NA	87
Palisades PIT array	NA	28
Dry Bed PIT array	NA	5[a]

[a]Dry Bed PIT array has low observation efficiency due to extreme background interference.

Table 25. The summary of PIT tagged juvenile Yellowstone Cutthroat Trout (YCT) out-migrating from South Fork Snake River tributaries including the total number of YCT >100 mm (assumed to be age-1 or greater) marked with PIT tags, the number recaptured at downstream PIT tag arrays in the tributary and the number of those recaptured YCT that out-migrated in the same year, the following year, or two years post-marking. The date in parentheses represents the average out-migration date.

			Outmigrated			
Stream	**Marked**	**Recaptured**	**Same year**	**Next year**	**Two years later**	**unknown**
Burns Creek	276	23	12 (Oct 27)	8 (Jul 18)	1 (May 24)	2
Pine Creek	68	1	--	1 (May 21)	--	--
Palisades Creek	557	72	22 (Oct 14)	43 (Jul 21)	4 (Jun 6)	3

Table 26. A summary of spawning stream fidelity for Yellowstone Cutthroat Trout (YCT) observed in South Fork Snake River tributaries in 2014.

			Spawning interval			
Weir	Recaptured*	Fidelity**	Every year	Every other year	unknown	# observed spawning events
Burns Creek	66	100% (49/49)	40	7	2	118 (range 2 - 4 years per fish)
Pine Creek	87	100 % (18/18)	13	5	NA	46 (range 2 - 4 years per fish)
Rainey Creek	1	NA	NA	NA	NA	NA
Palisades Creek	28	100% (20/20)	12	6	2	42 (range 2 - 4 years per fish)

*Recaptures only include YCT that were originally marked with PIT tags someplace or sometime outside of the 2014 spring spawning run

**These records only include YCT that were observed at one of the spawning tributary weirs

Figure 62. 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 2014.

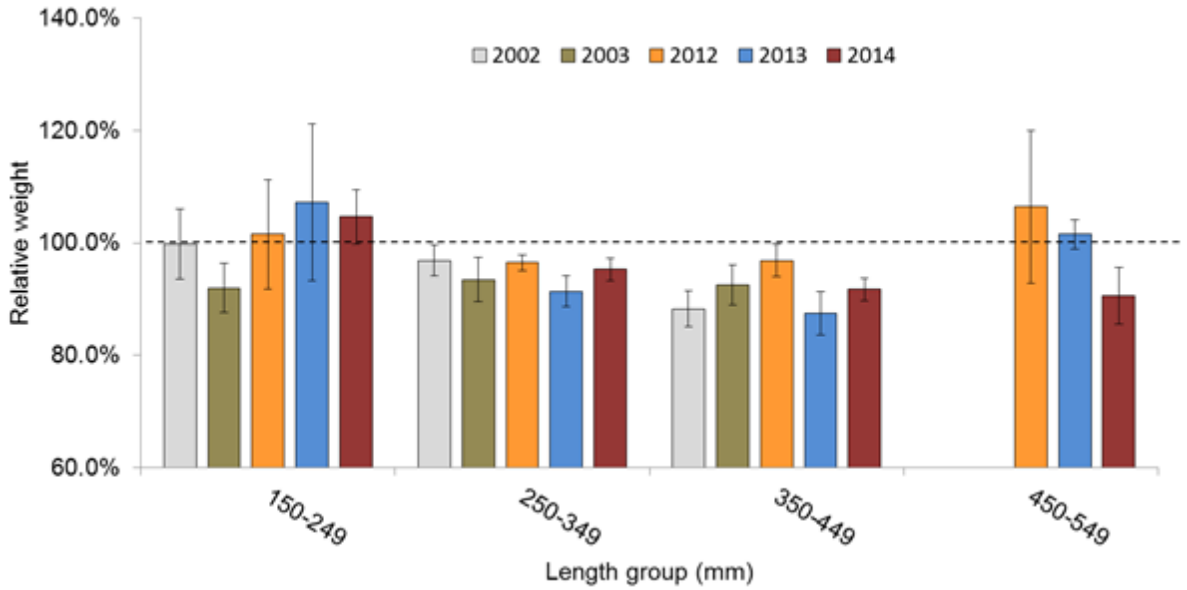

Figure 63. Relative weights for Yellowstone Cutthroat Trout at the Lorenzo monitoring reach by 50 mm size groups from 2002 to 2014.

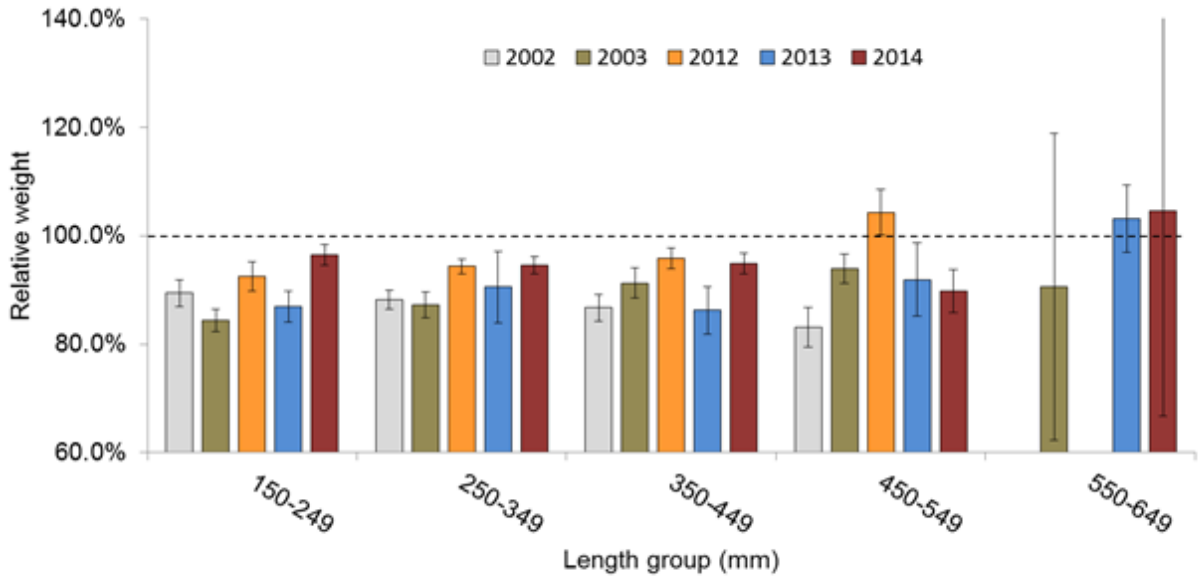

Figure 64. Relative weights for Brown Trout at the Lorenzo monitoring reach by 50 mm size groups from 2002 through 2014.

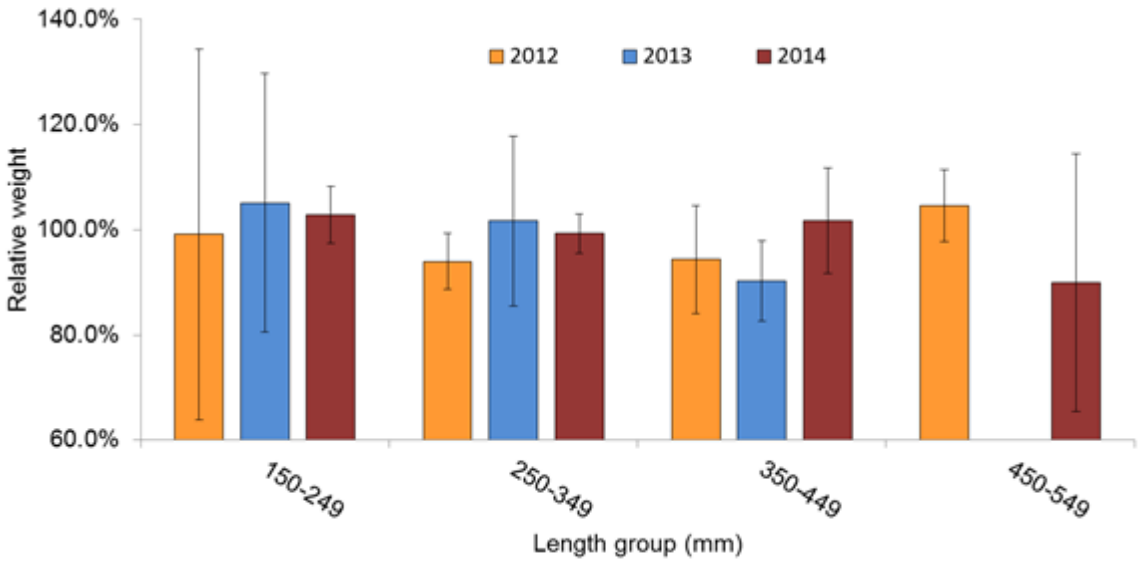

Figure 65. Relative weights for Rainbow Trout at the Lorenzo monitoring reach by 50 mm size groups from 2012 through 2014.

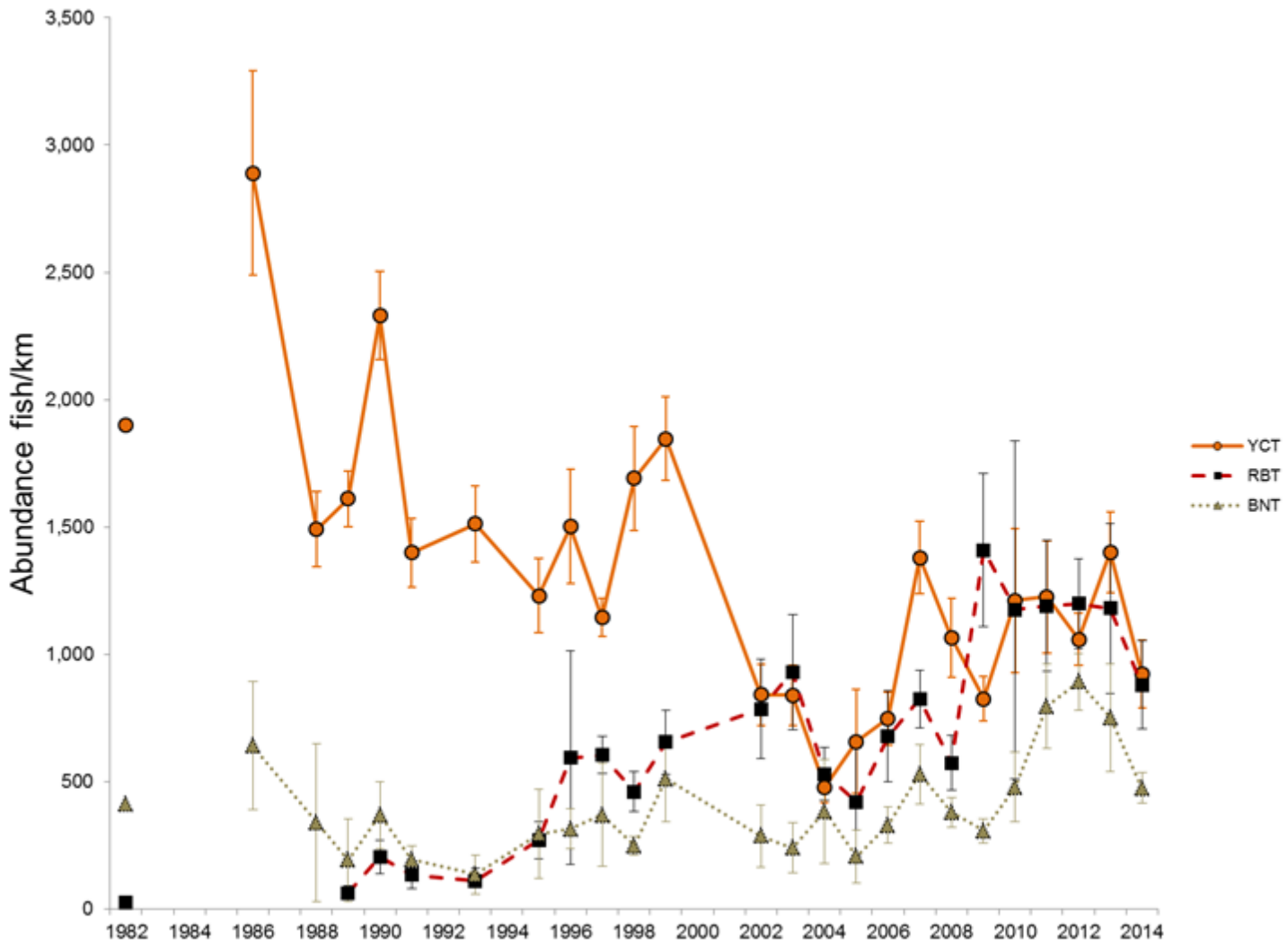

Figure 66. Abundance estimates and 95% confidence intervals for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) at the Conant monitoring site on the South Fork Snake River from 1986 through 2014.

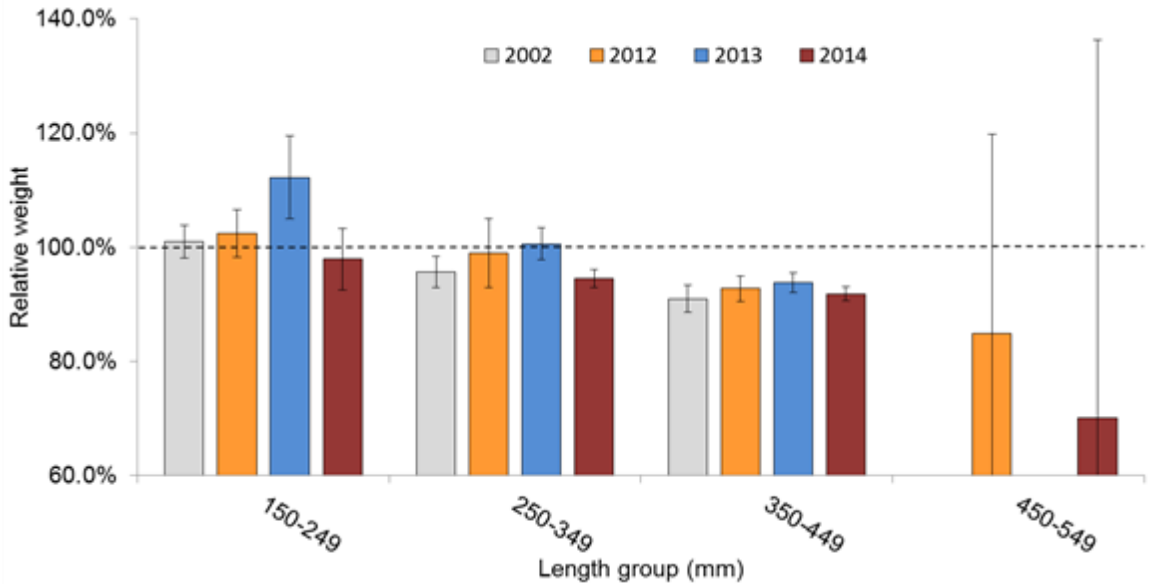

Figure 67. Relative weights for Yellowstone Cutthroat Trout at the Conant monitoring reach by 50 mm size groups from 2002 through 2014.

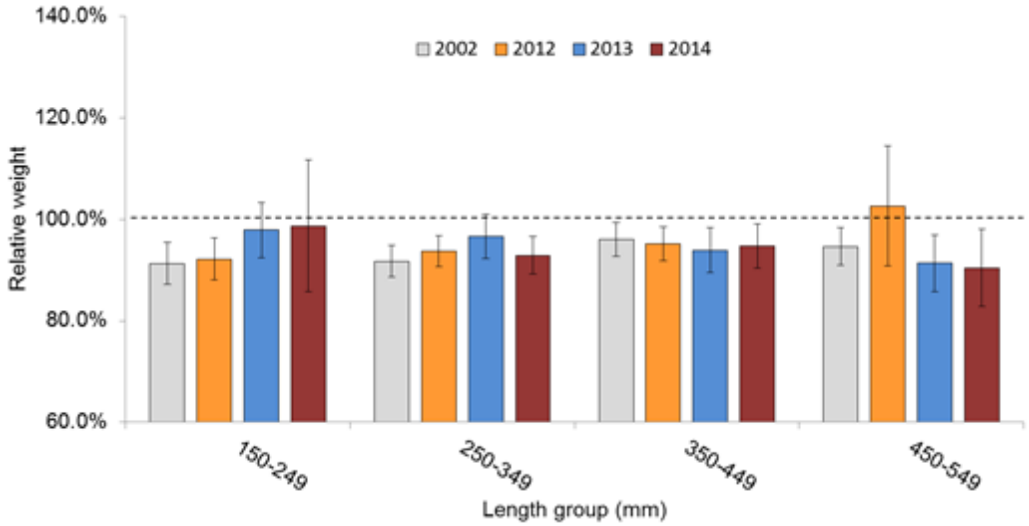

Figure 68. Relative weights for Brown Trout at the Conant monitoring reach by 50 mm size groups from 2002 through 2014.

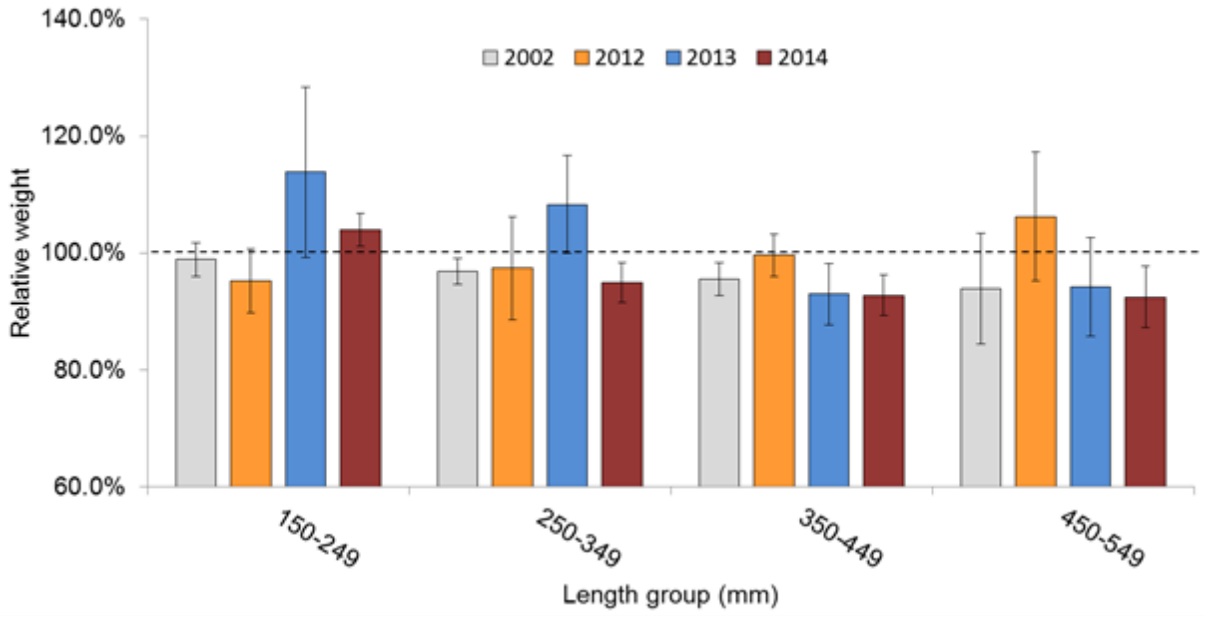

Figure 69. Relative weights for Rainbow Trout at the Conant monitoring reach by 50 mm size groups from 2002 through 2014.

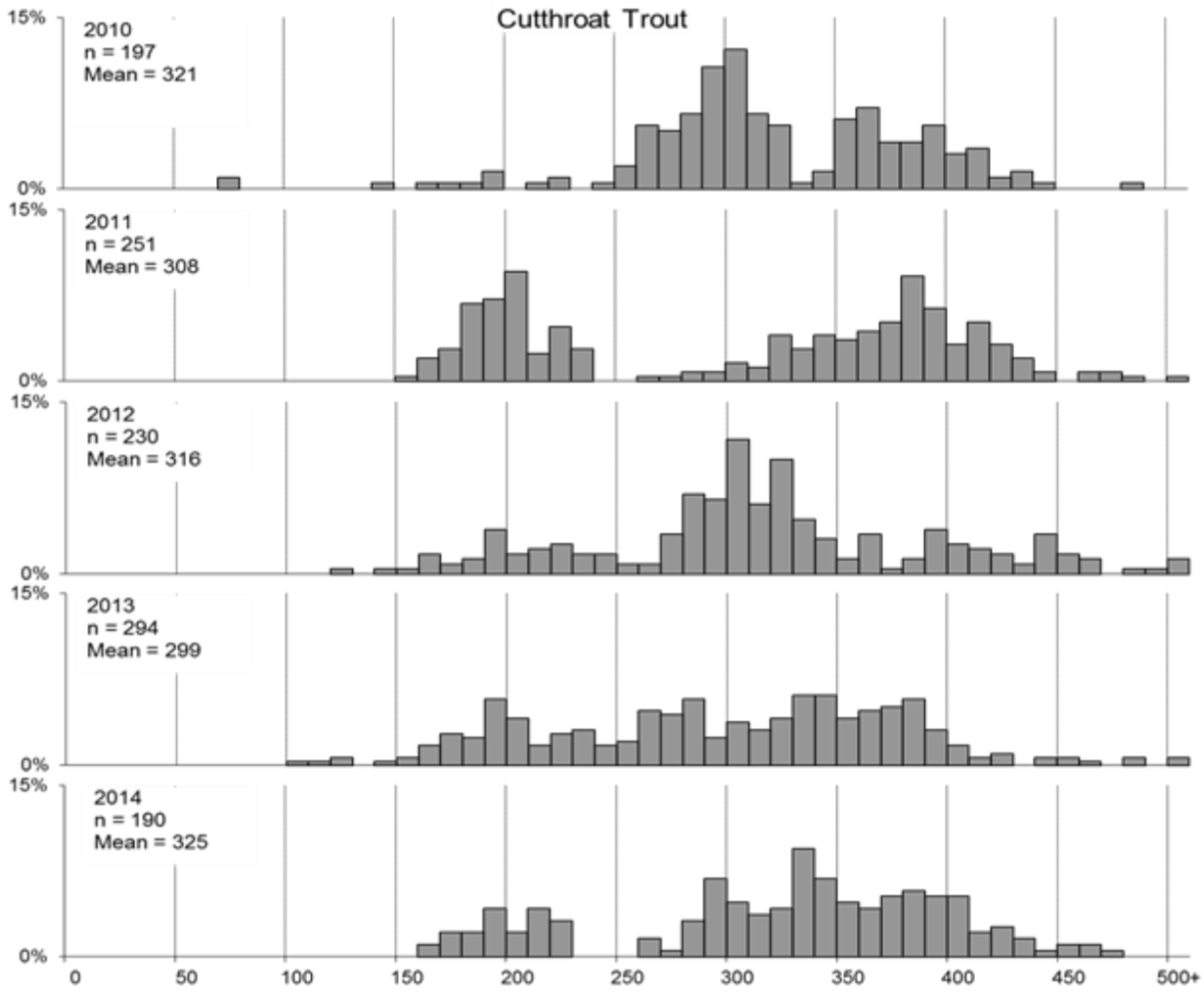

Figure 70. Length frequency plots for Yellowstone Cutthroat Trout at the Lorenzo monitoring reach of the South Fork Snake River from 2010 through 2014.

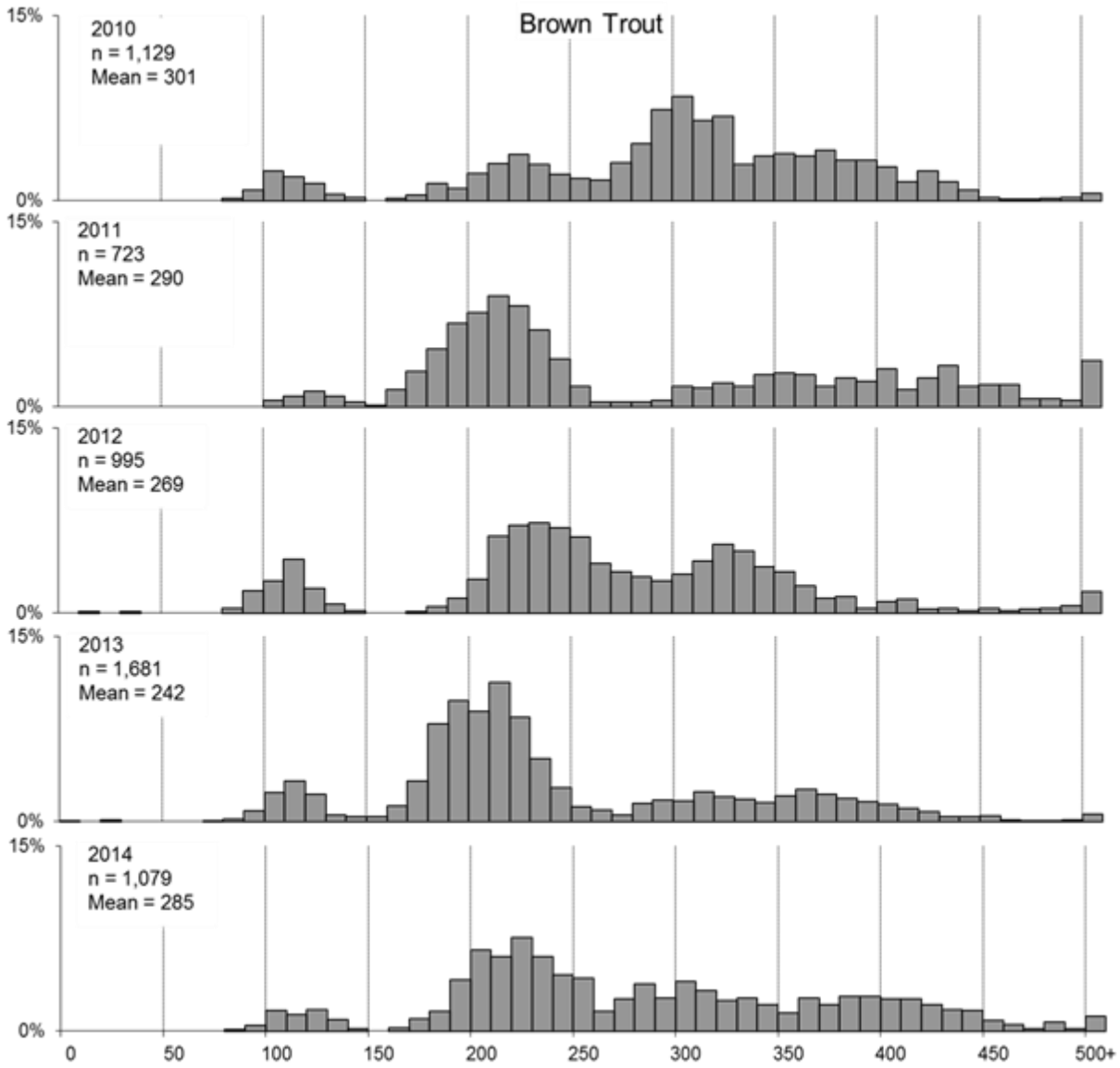

Figure 71. Length frequency plots for Brown Trout at the Lorenzo monitoring reach of the South Fork Snake River from 2010 through 2014.

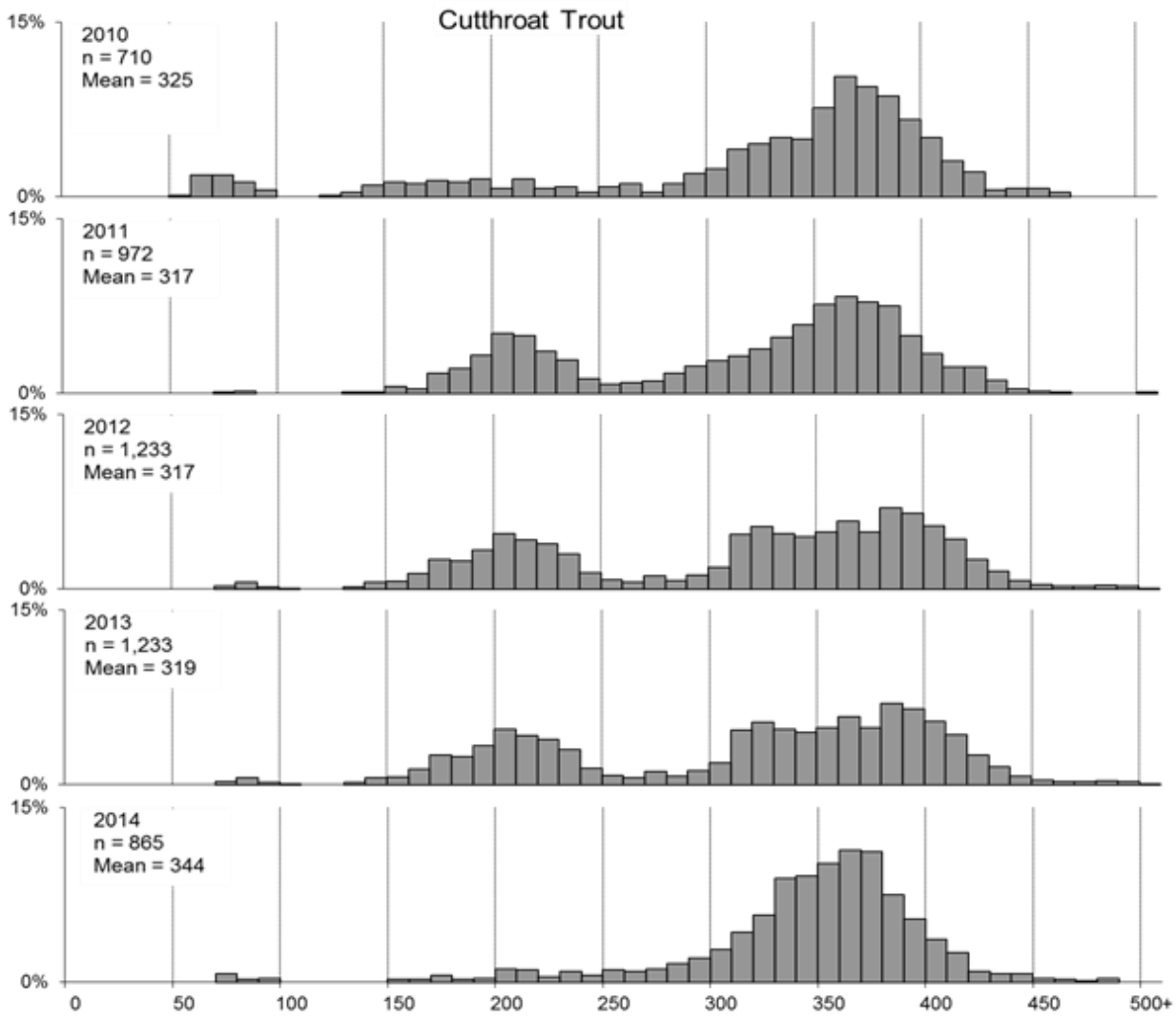

Figure 72. Length frequency plots for Yellowstone Cutthroat Trout at the Conant monitoring reach of the South Fork Snake River from 2010 through 2014.

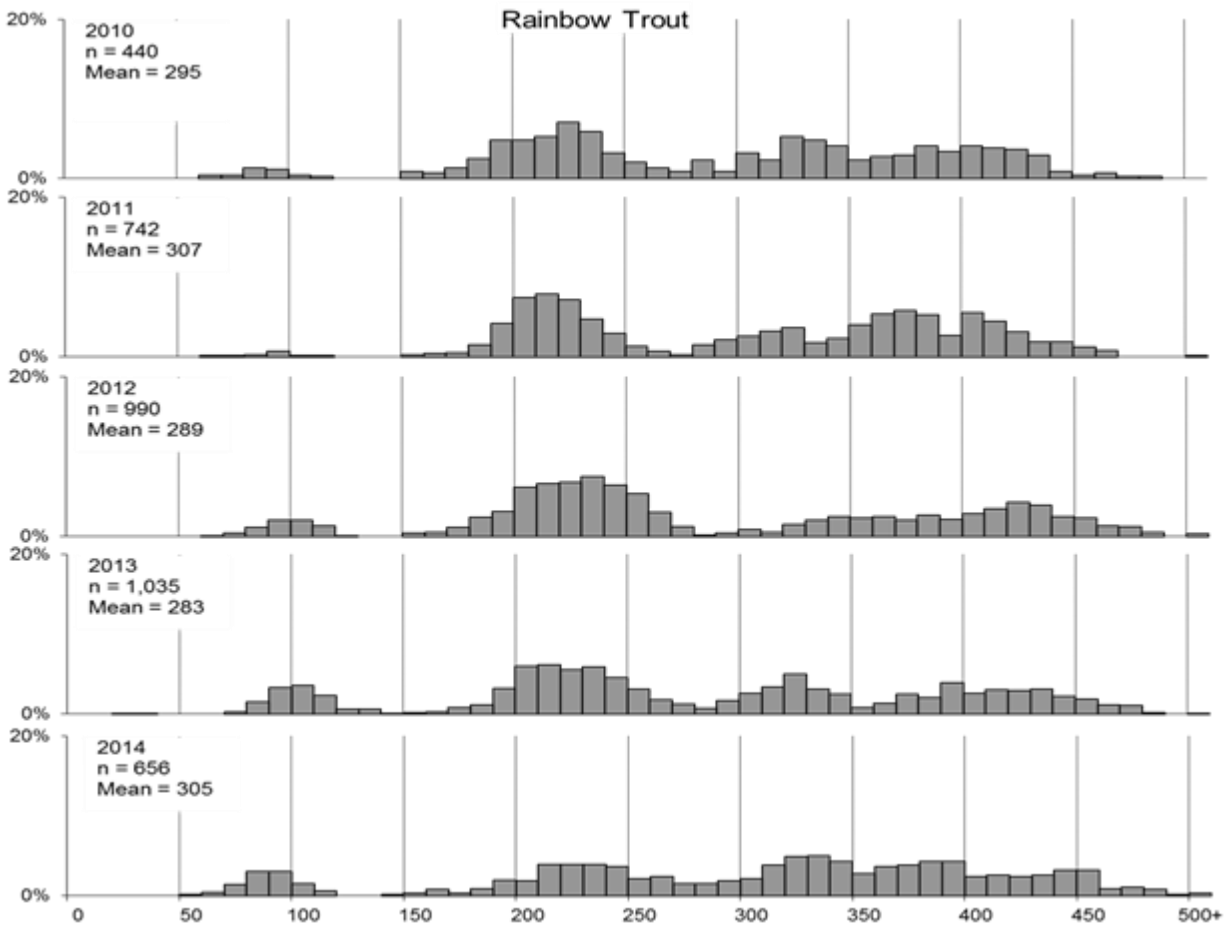

Figure 73. Length frequency plots for Rainbow Trout at the Conant monitoring reach of the South Fork Snake River from 2010 through 2014.

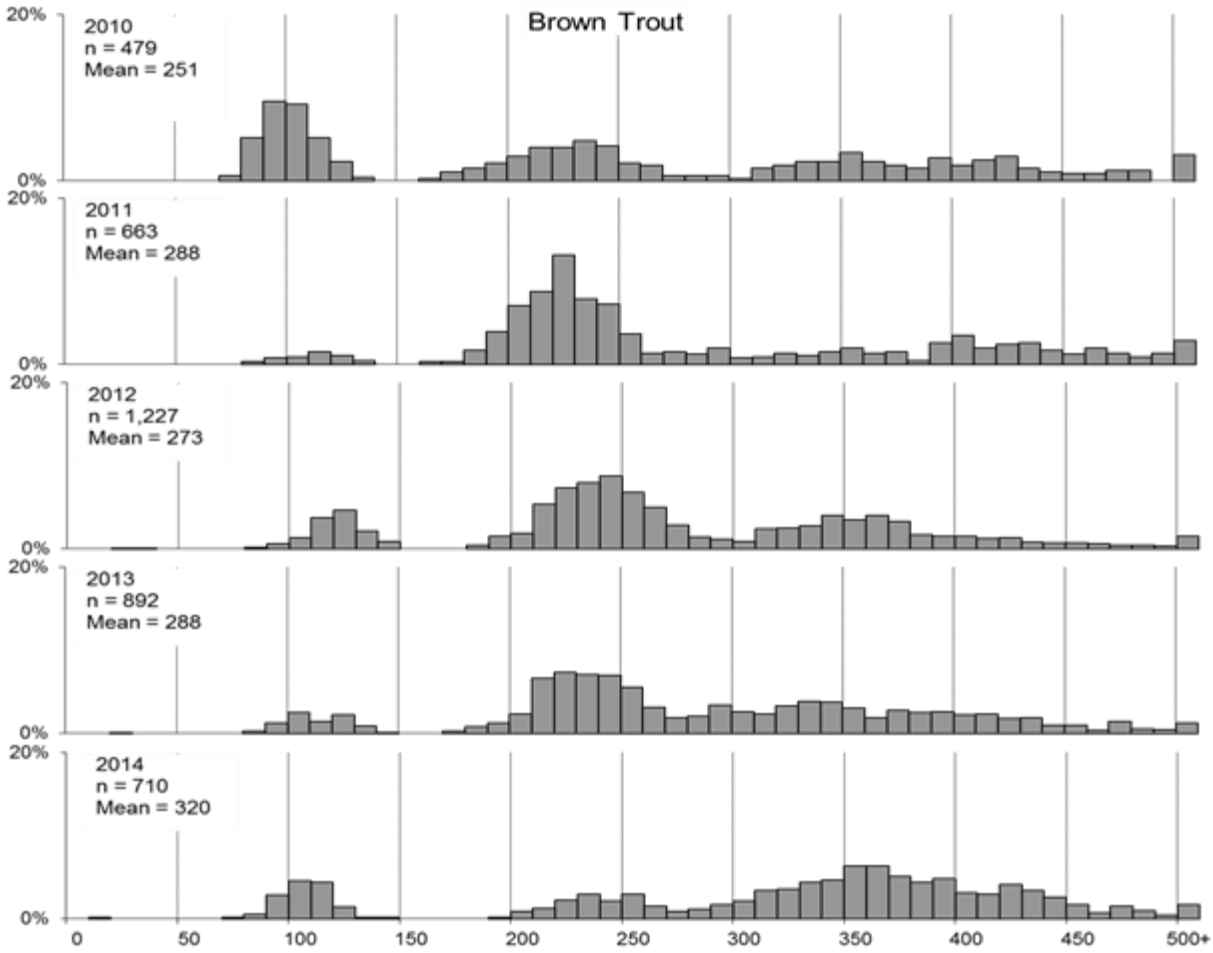

Figure 74. Length frequency plots for Brown Trout at the Conant monitoring reach of the South Fork Snake River from 2010 through 2014.

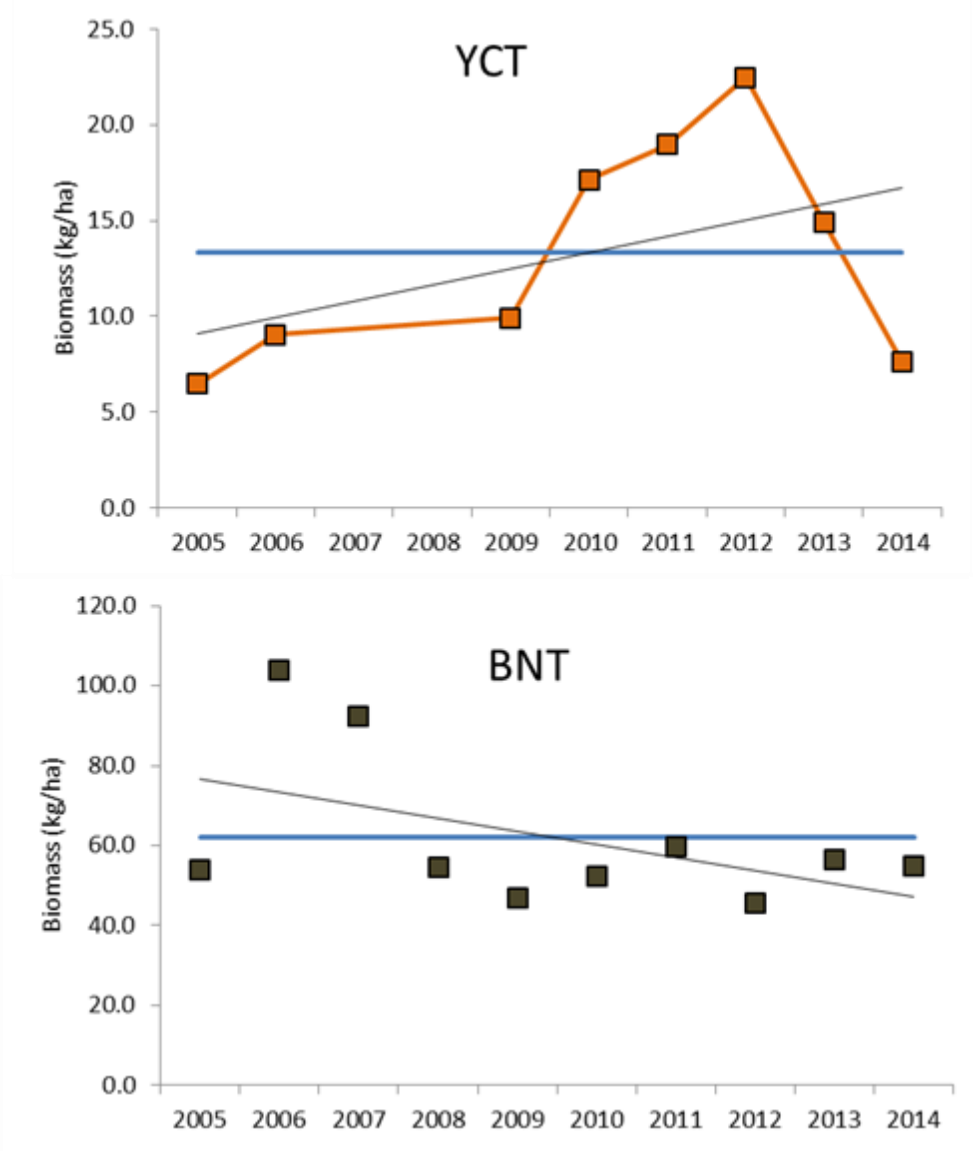

Figure 75. Biomass estimates for Yellowstone Cutthroat Trout (YCT) and Brown Trout (BNT) at the Lorenzo monitoring reach from 2005 through 2014. The solid blue lines represent the 10-year average and the dashed lines represent the best-fit linear regression trends.

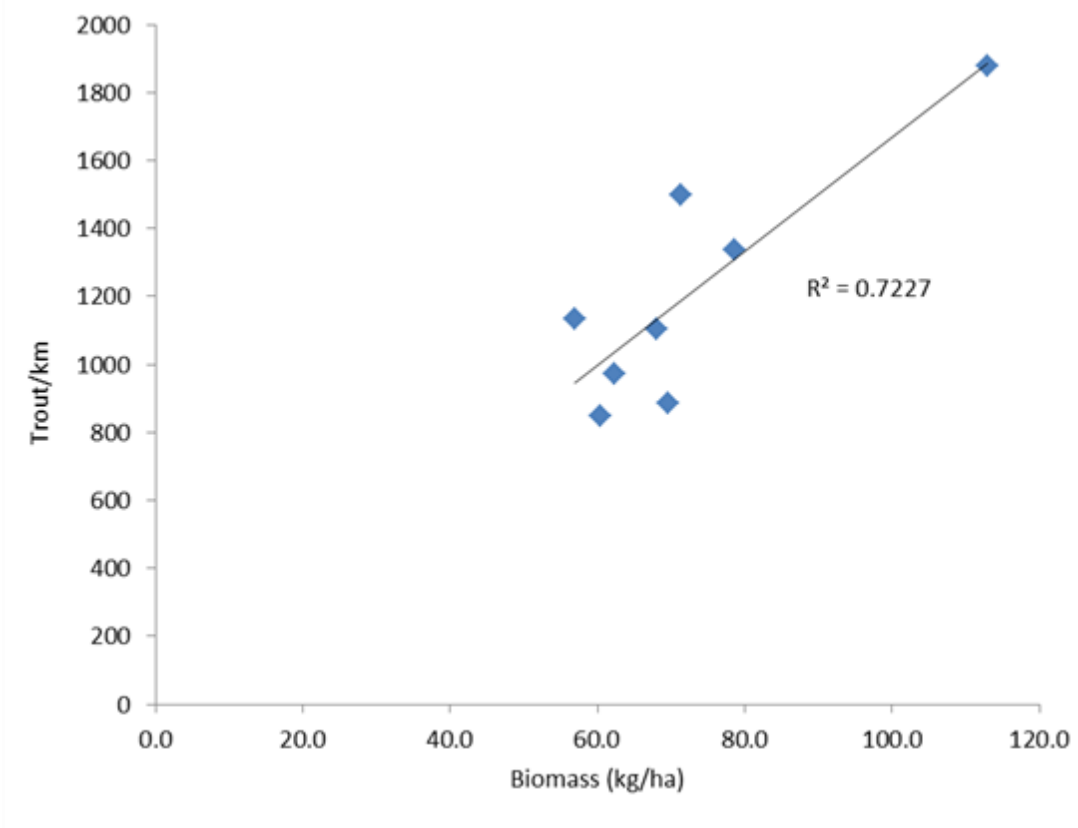

Figure 76. Scatter plot of biomass of all trout combined plotted against total trout abundance (all species combined) for Lorenzo 2005 through 2014.

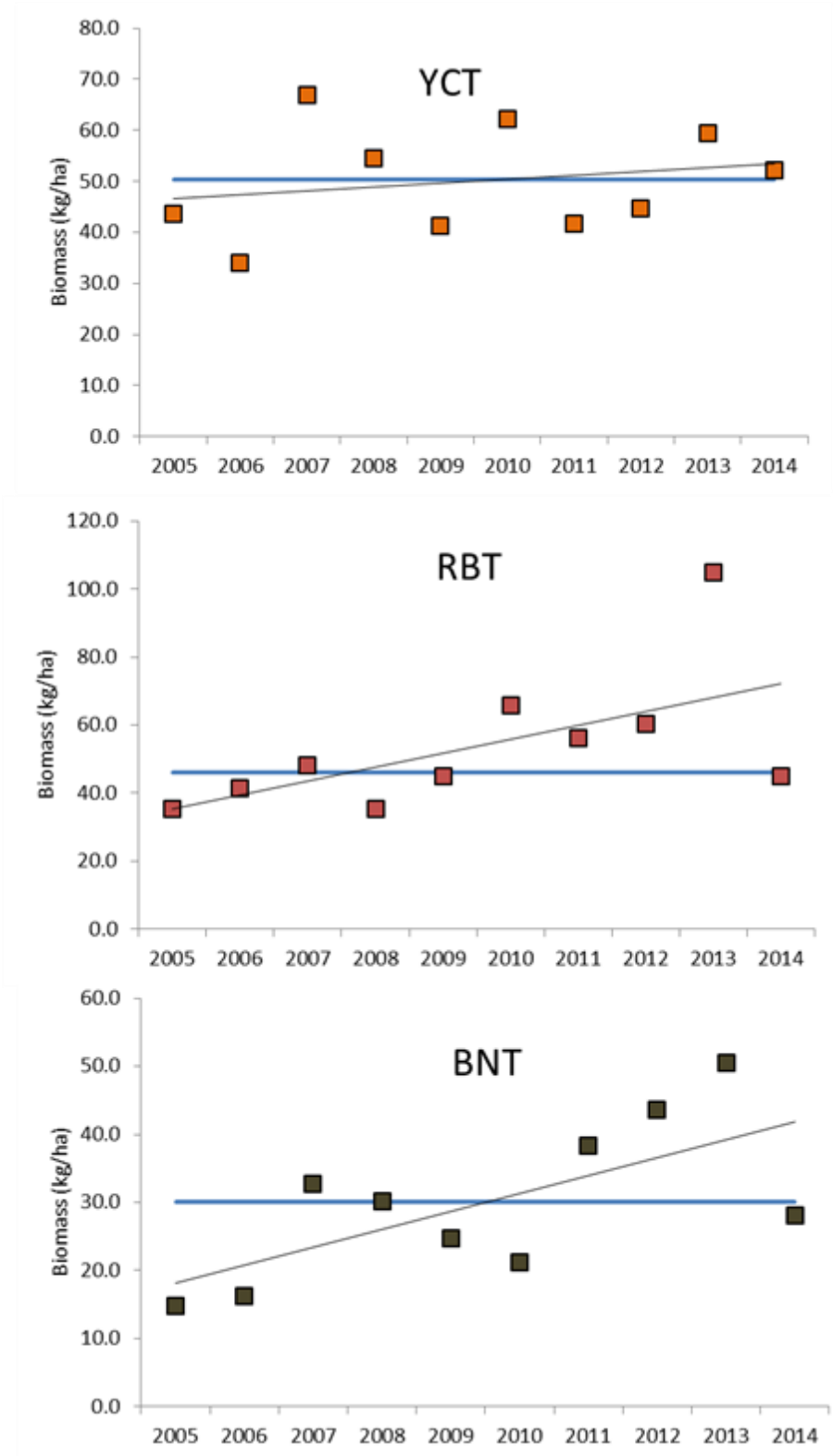

Figure 77. Biomass estimates for Yellowstone Cutthroat Trout (YCT), Rainbow Trout (RBT), and Brown Trout (BNT) at the Conant monitoring reach from 2005 through 2014. The solid blue lines represent the 10-year average and the dashed lines represent the best-fit linear regression trends.

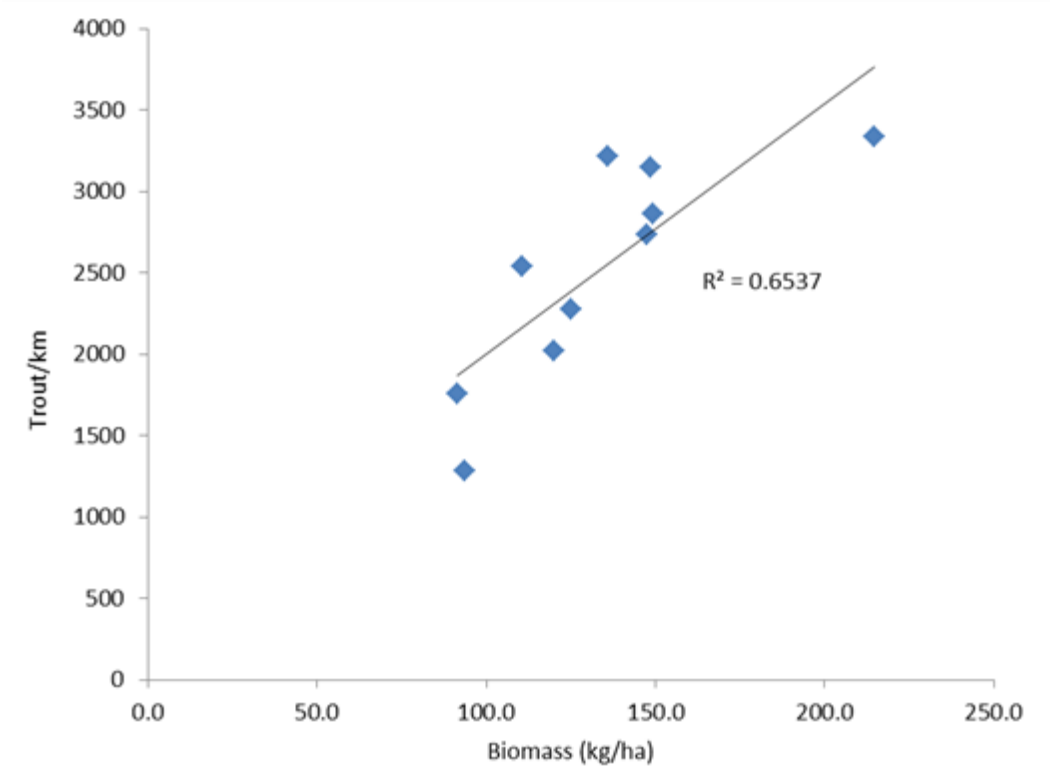

Figure 78. Biomass estimates all trout species combined at the Conant monitoring reach from 2005 through 2014 plotted against total trout numbers (all species combined) for the same time period.

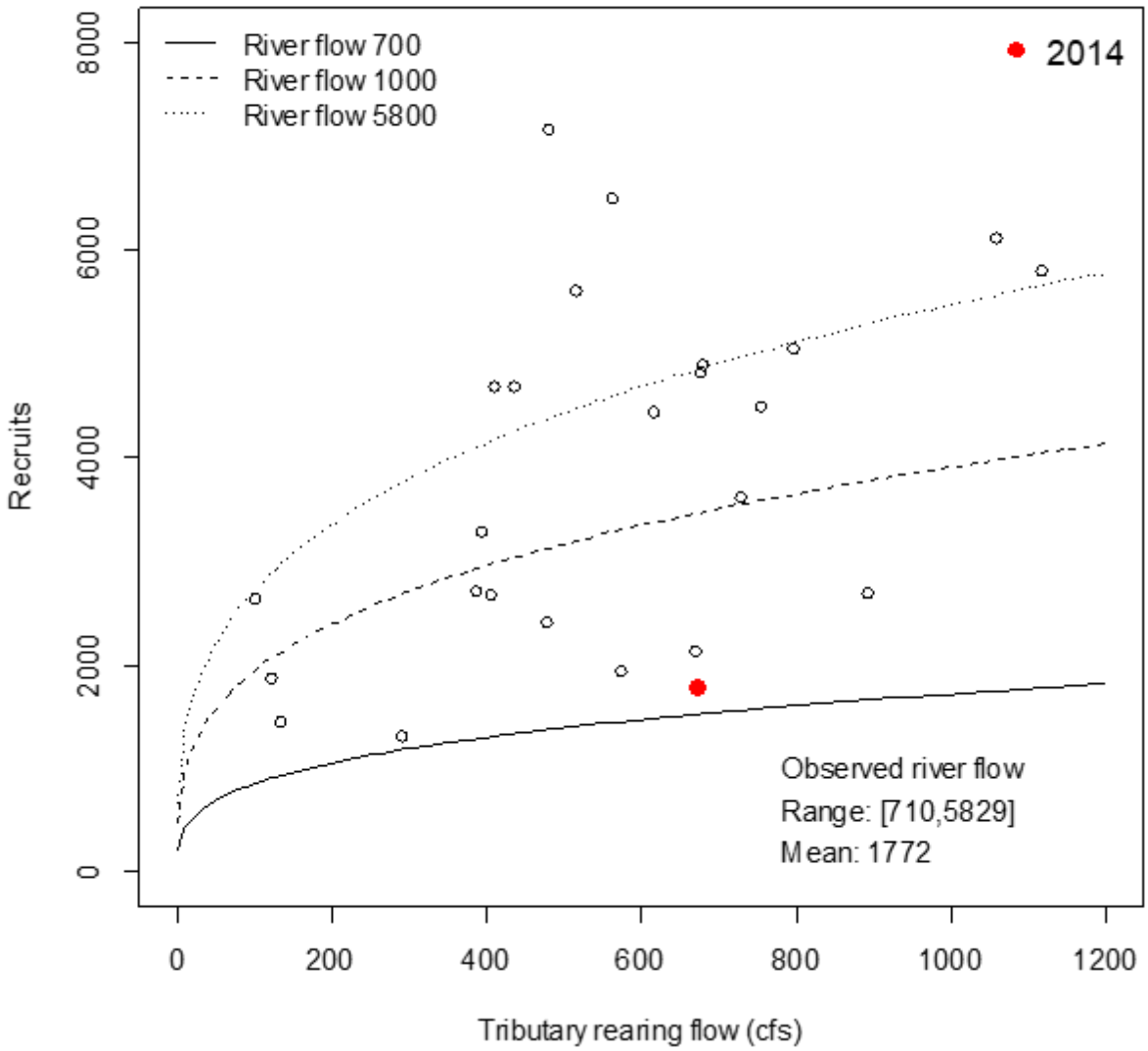

Figure 79. Observed abundance of recruits, including age-1 Yellowstone Cutthroat Trout and Rainbow Trout and hybrids at Conant relative to tributary rearing flows versus predicted abundances at three different winter flow scenarios in the main river.

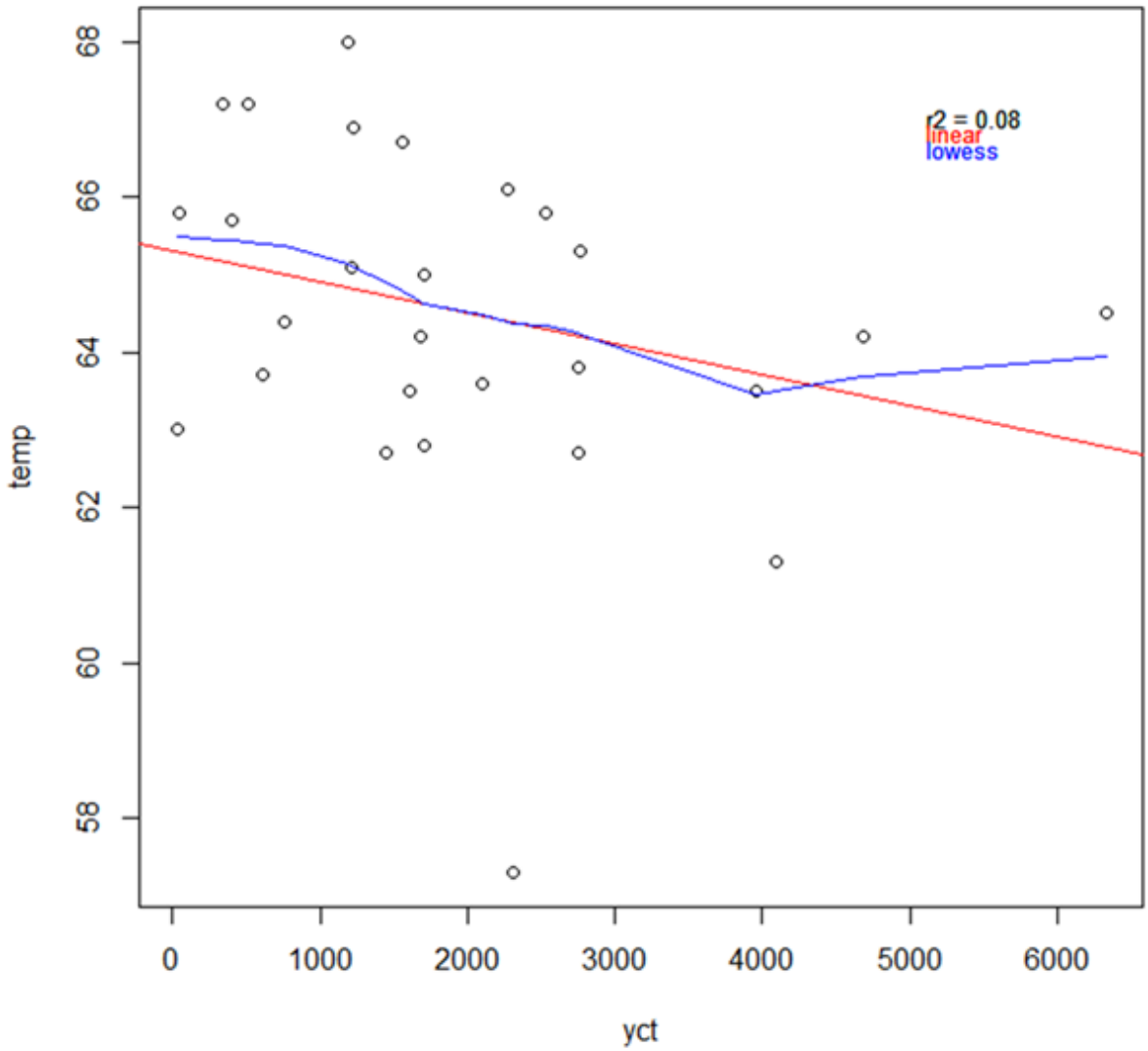

Figure 80. Linear regression (red), Pearson correlation coefficient, and Lowess regression (blue) for age-1 Yellowstone Cutthroat Trout (YCT) and summer temperatures (temp).

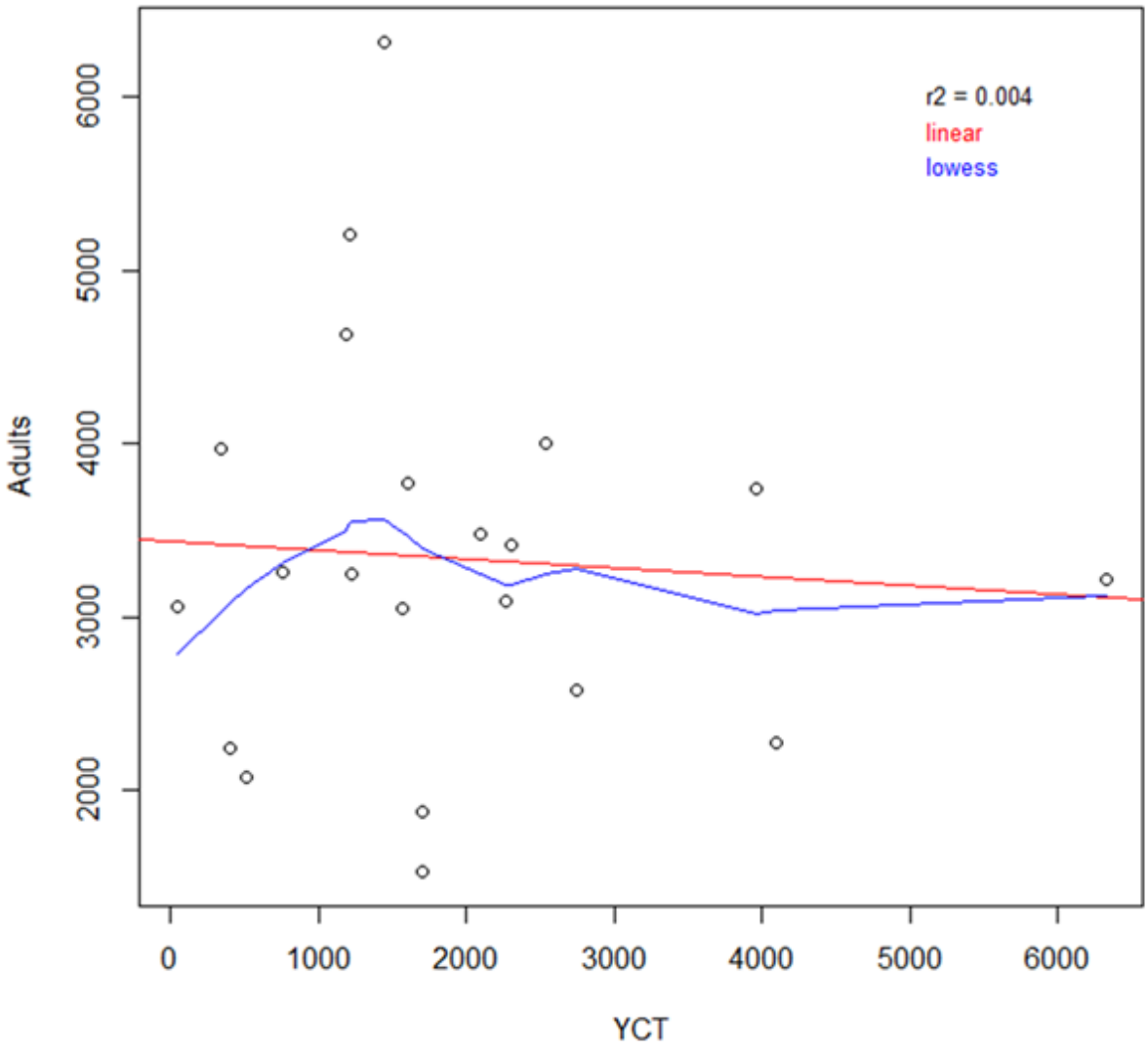

Figure 81. Linear regression (red), Pearson correlation coefficient, and Lowess regression (blue) for age-1 Yellowstone Cutthroat Trout (YCT) and abundance of adult Yellowstone Cutthroat Trout (Adults).

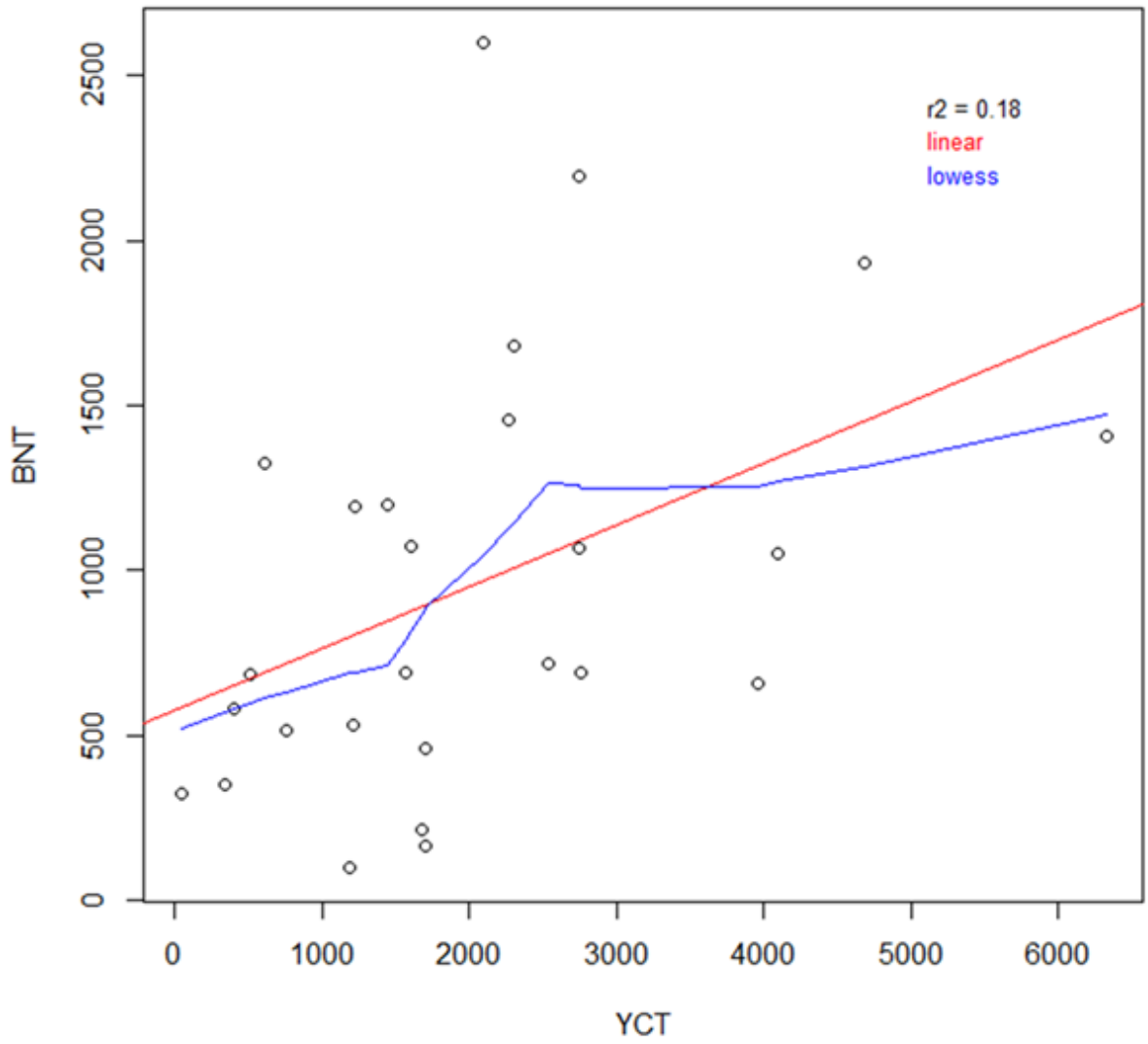

Figure 82. Linear regression (red), Pearson correlation coefficient, and Lowess regression (blue) for age-1 Yellowstone Cutthroat Trout (YCT) and age-1 Brown Trout (BNT).

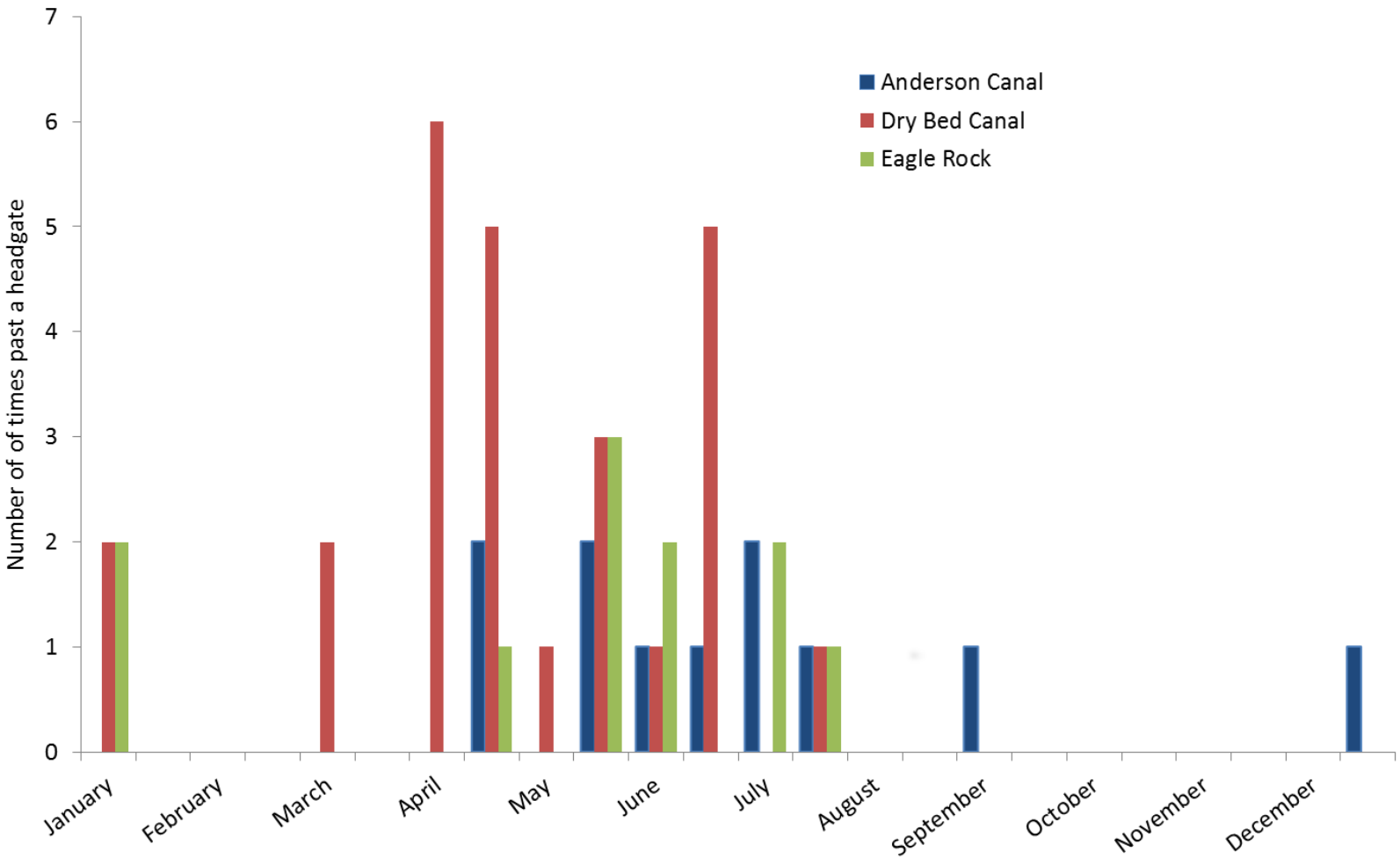

Figure 83. Timing of occurrences with radio-tagged trout passed major irrigation diversions from the South Fork Snake River while moving upstream in 2014.

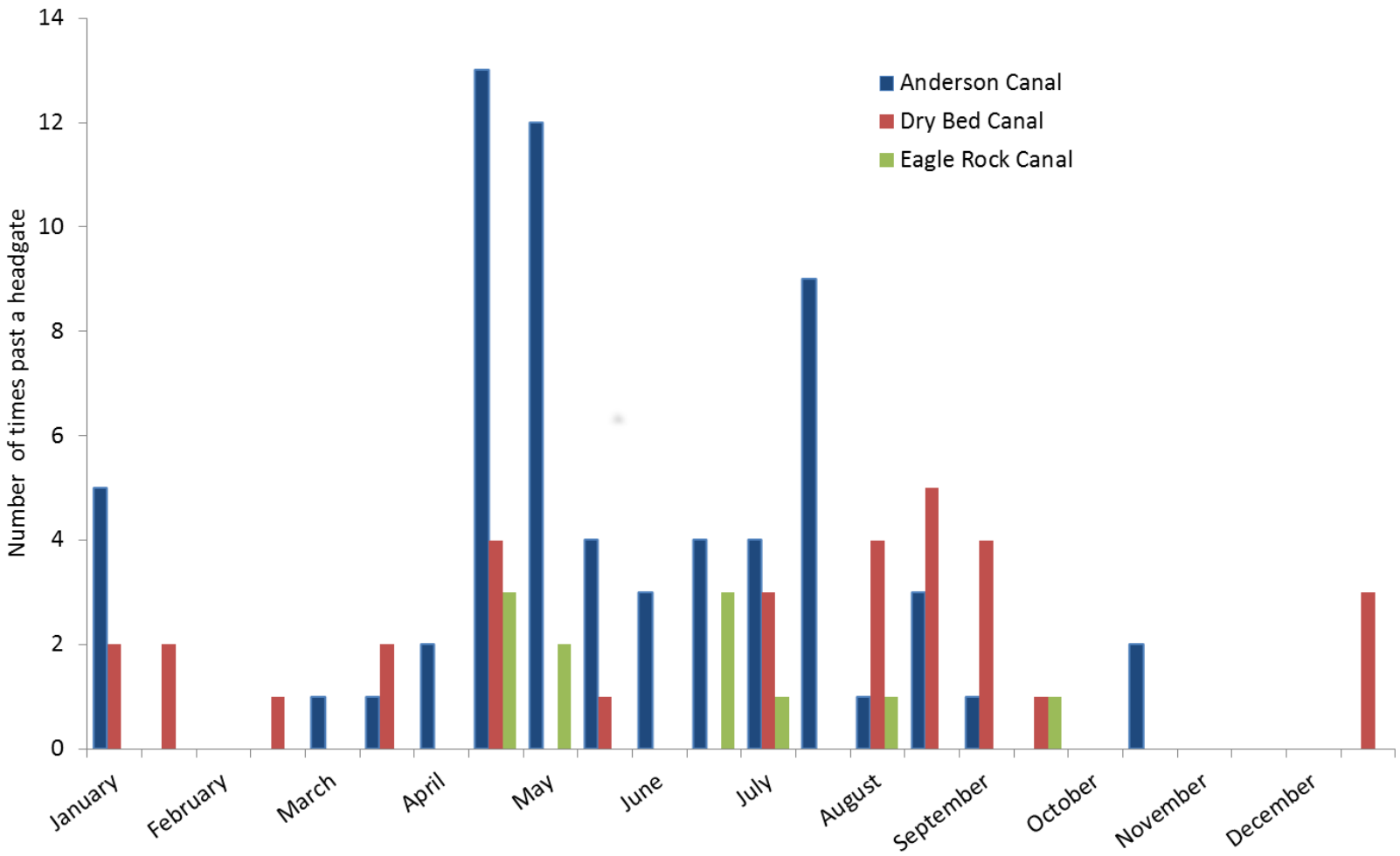

Figure 84. Timing of occurrences with radio-tagged trout passed major irrigation diversions from the South Fork Snake River while moving downstream in 2014.

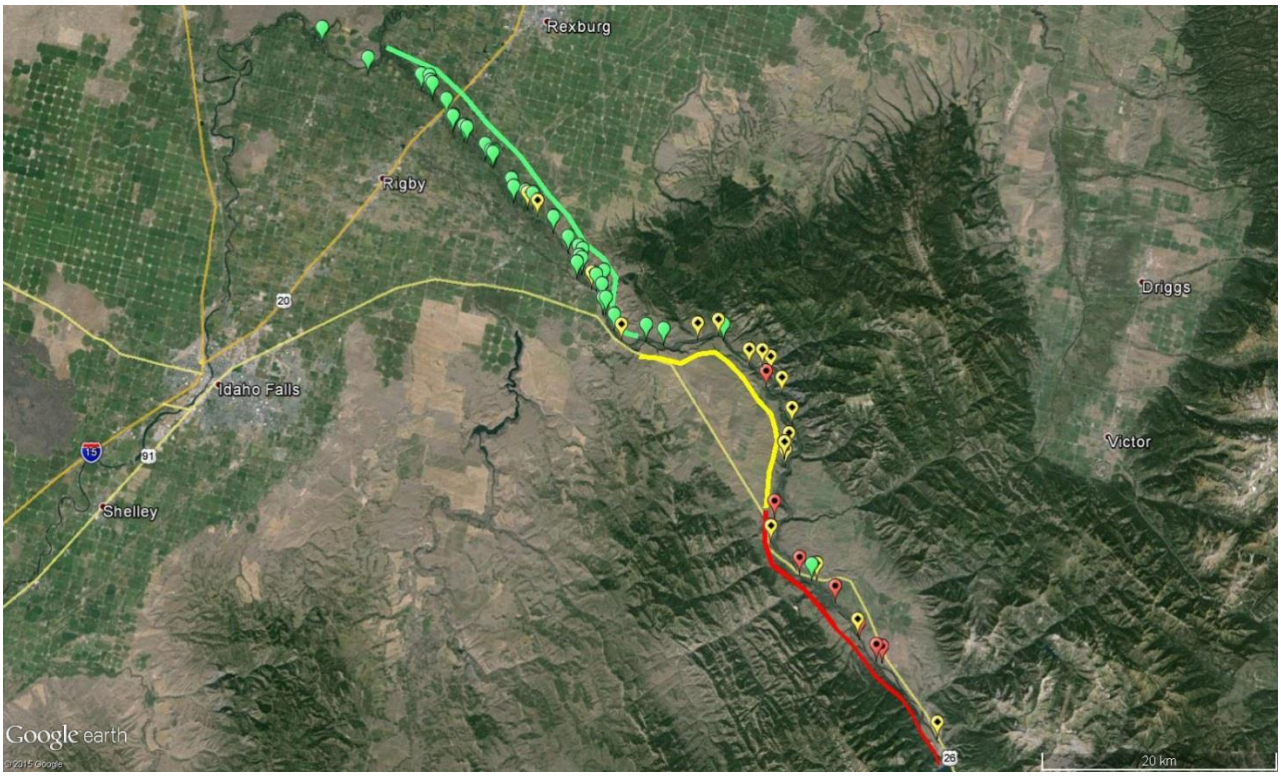

Figure 85. Map of inferred spawning locations of 67 Brown Trout marked with radio transmitters in the South Fork Snake River in 2013 and 2014. The markers indicate the spawning locations and the color of the markers indicates the section of the South Fork the fish were originally marked in, with green representing the lower river, yellow representing the canyon, and red representing the upper river. The river sections are identified with the solid lines adjacent to the river with the same coloration already described.

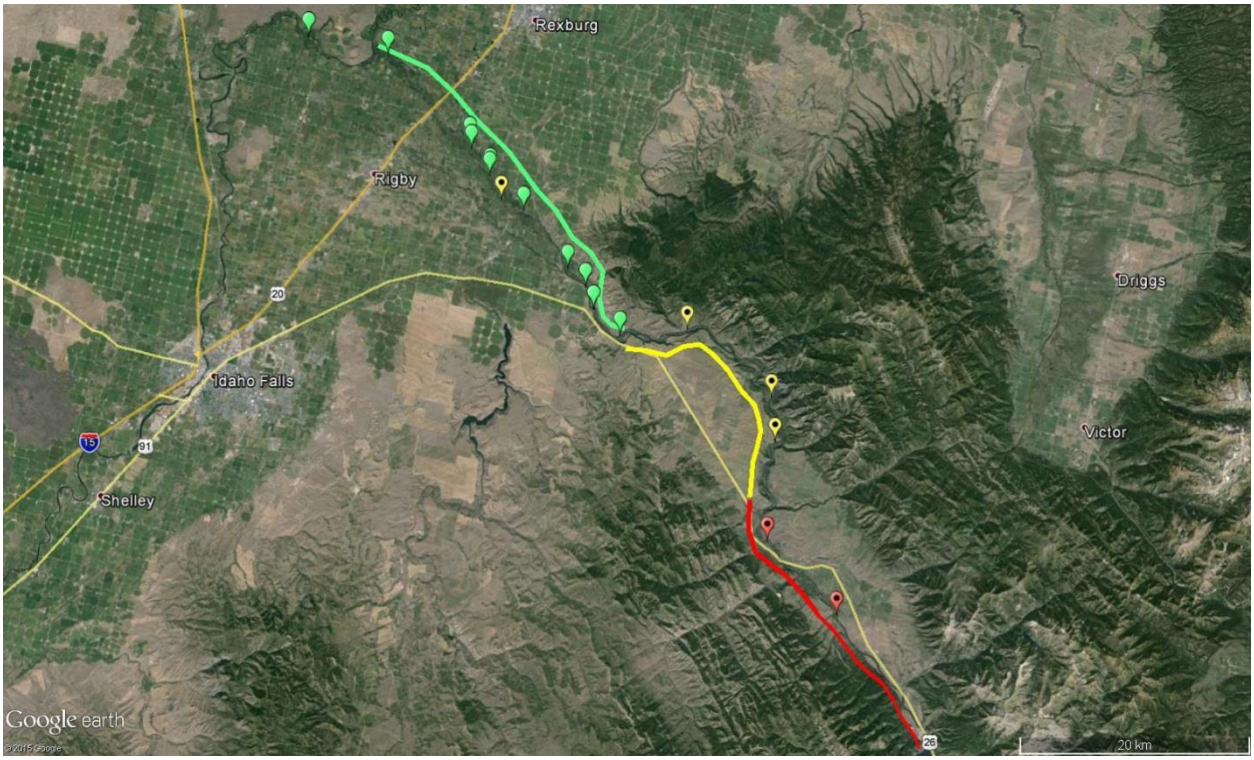

Figure 86. Map of inferred spawning locations of 17 Rainbow Trout marked with radio transmitters in the South Fork Snake River in 2013 and 2014. The markers indicate the spawning locations and the color of the markers indicates the section of the South Fork the fish were originally marked in, with green representing the lower river, yellow representing the canyon, and red representing the upper river. The river sections are identified with the solid lines adjacent to the river with the same coloration already described.

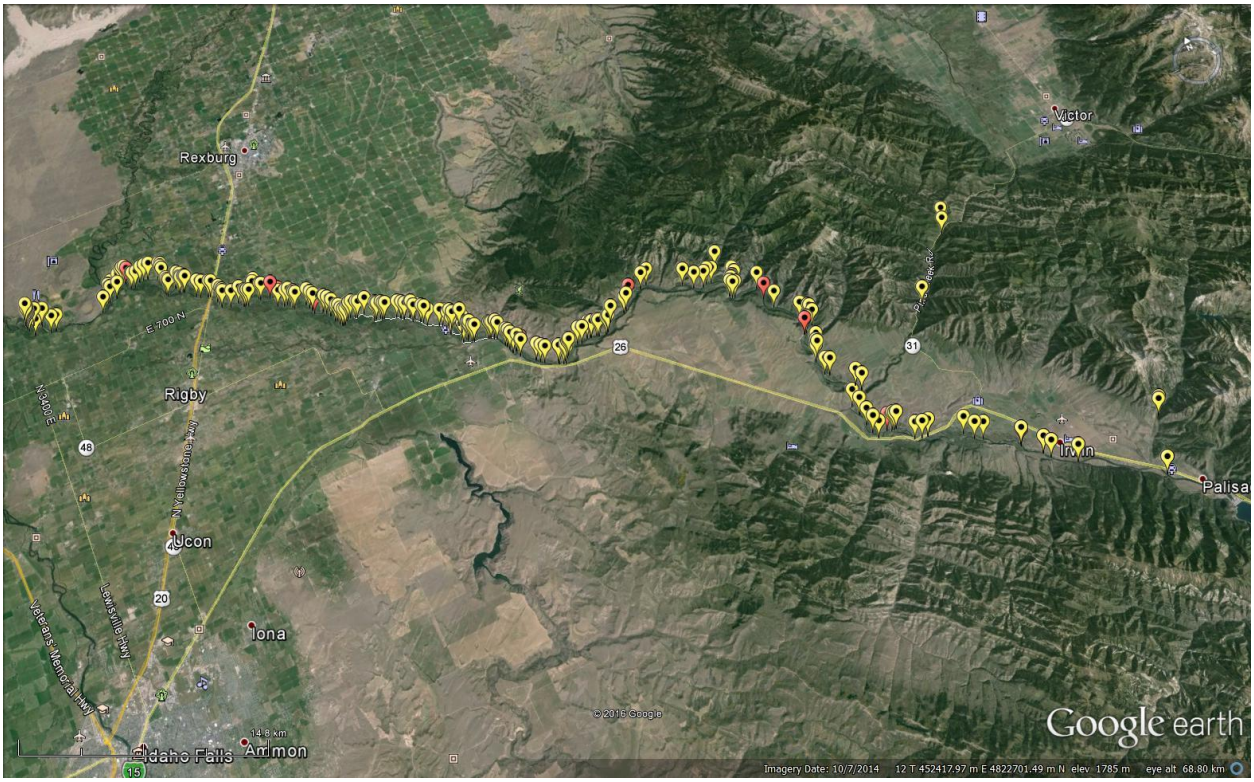

Figure 87. Map of inferred spawning locations of 17 Rainbow Trout (red markers) and 259 Yellowstone Cutthroat Trout (yellow markers) marked with radio transmitters in the South Fork Snake River in 2013 and 2014. The markers indicate the spawning locations.

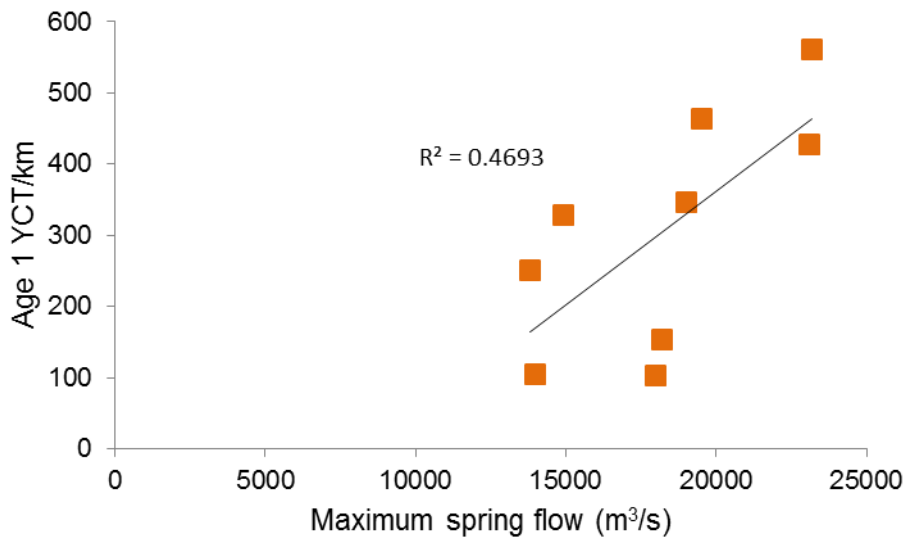

Figure 88. Correlation between age-1 Yellowstone Cutthroat Trout (YCT) abundance at the Conant monitoring site and maximum spring flows the prior year on the South Fork Snake River.

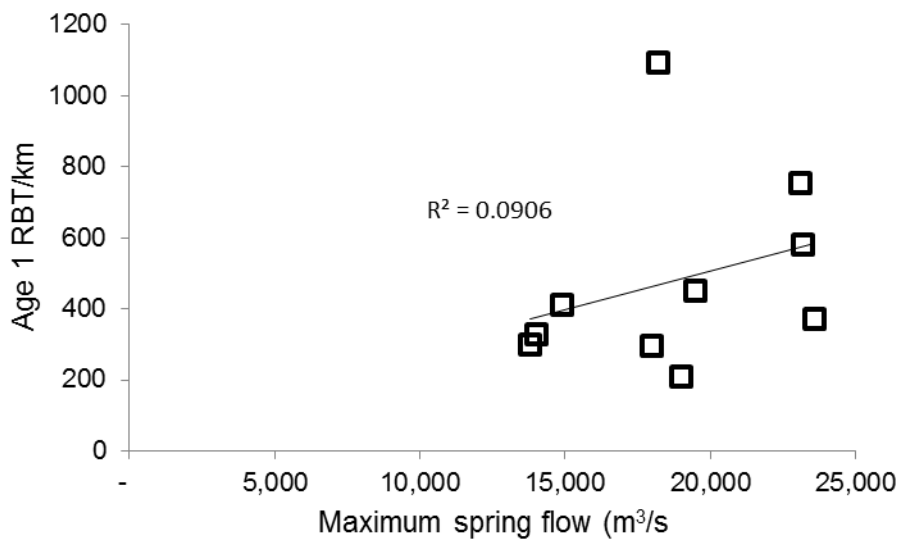

Figure 89. Correlation between age-1 Rainbow Trout (RBT) abundance at the Conant monitoring site and maximum spring flows the prior year on the South Fork Snake River.

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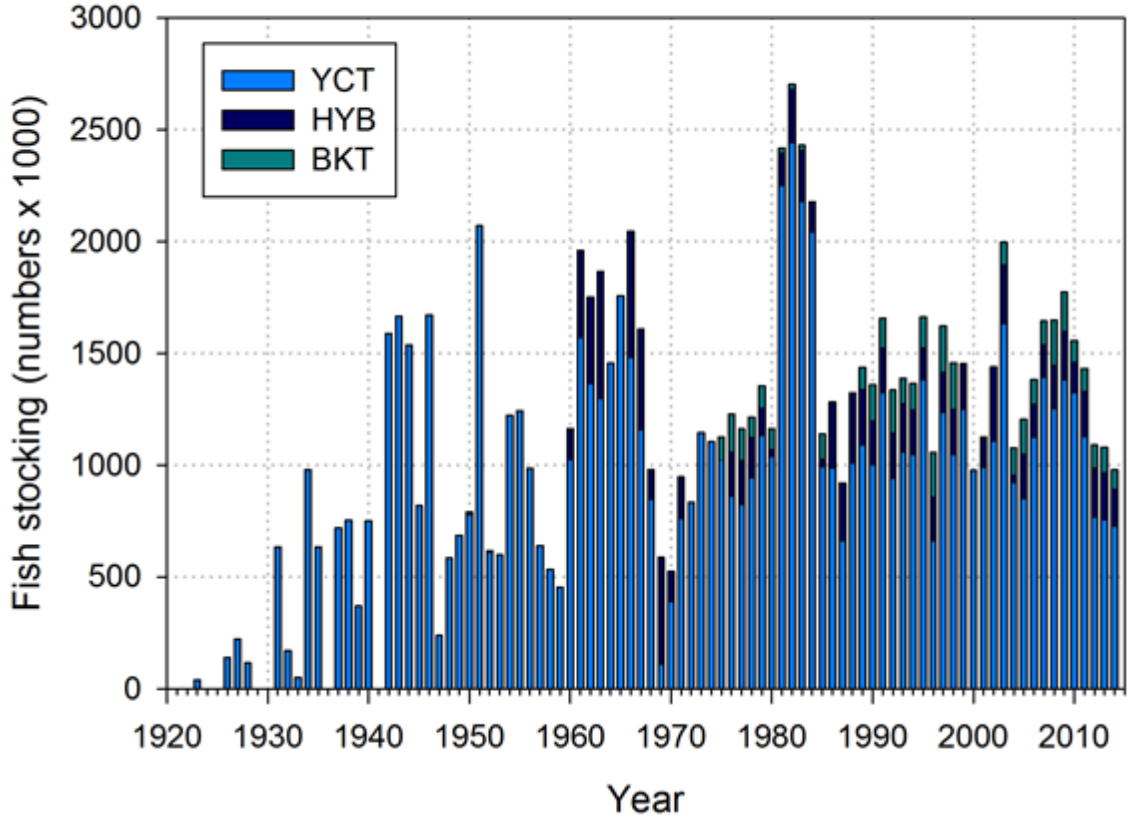
APPENDICES

Appendix A. Historic annual stocking (*1,000) of Henrys Lake, Idaho, 1925 -2014.

Year	Yellowstone Cutthroat Trout	Hybrid Trout	Brook Trout	Total Trout
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
1967	1159	448	0	1607

Year	Yellowstone Cutthroat Trout	Hybrid Trout	Brook Trout	Total Trout
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
2014	729	167	83	979



Appendix B. Historic annual stockings for Yellowstone Cutthroat Trout (YCT), hybrid trout (HYB), and Brook Trout (BKT) in Henrys Lake 1923-2014.

Appendix C. Annual kokanee stocking in Island Park Reservoir, Moose Creek, Big Springs Creek and Henrys Lake (HL) Outlet, 1944–2014.

	Island Park Reservoir		Moose Creek			Big Springs Creek		HL Outlet
Year	Fingerling	Fry	Fingerling	Fry	Eggs	Fingerling	Fry	Fingerling
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

[a] Includes 9,929 eggs stocked in Lucky Dog Creek.

Appendix D. Gill net locations in Island Park Reservoir, 2014. All coordinates used NAD27 and are in Zone 12.

Location	UTM E	UTM N
West Mouth	462368	4918437
Bills Island West	463725	4919296
Lakeside	464751	4920435
Mill Cr	466325	4921491
Bills Island	465499	4919897
Dam	467871	4918662

Appendix E. Curtain net locations in Island Park Reservoir, 2014. All coordinates used NAD27 and are in Zone 12.

Location	Mesh size	UTM E	UTM N
Dam	Small	468205	4918623
	Medium	468132	4918708

Appendix F. Locations and number of kokanee eyed eggs deposited in artificial redds in Moose and Lucky Dog Creeks, 2014. All coordinates used NAD27 and are in Zone 12.

Location	Section	Kokanee eggs (#)	UTM E	UTM N
Moose Creek	Upper	18,000	481577	4922807
	Middle	10,000	481630	4924087
	Lower	15,000	477310	4925395
Lucky Dog	Lower	10,000	477674	4925761

Appendix G. Gill net catch totals by species from Island Park Reservoir, 2014.

Date pulled	Site	Net type	BKT	RBT	UTC	UTS	TOTAL
6/5/2014	Dam	Floating	0	9	0	1	10
6/5/2014	Lakeside	Floating	0	10	0	2	12
6/5/2014	Westmouth	Floating	0	11	0	0	11
6/5/2014	Bills island	Sinking	1	4	55	73	133
6/5/2014	Bills west	Sinking	0	0	12	27	39
6/5/2014	Mill creek	Sinking	0	0	0	44	44

Appendix H. Location of Ririe Reservoir fall Walleye index netting (FWIN) net locations during the fall of 2014. All coordinates are Zone 12, and WGS 84 datum.

Date	Net	Lake Strata	Latitude	Longitude	Net Type
October 29	1	Lower	43.57389	-111.73896	Sinking
October 29	2	Lower	43.57190	-111.73730	Floating
October 29	3	Lower	43.55865	-111.73516	Sinking
October 29	4	Lower	43.55458	-111.73408	Floating
October 29	5	Lower	43.55152	-111.73736	Floating
October 29	6	Lower	43.54914	-111.73777	Sinking
October 28	7	Middle	43.54118	-111.73067	Sinking
October 28	8	Middle	43.54018	-111.73941	Floating
October 28	9	Middle	43.53589	-111.72381	Sinking
October 28	10	Middle	43.53430	-111.71858	Floating
October 28	11	Middle	43.53075	-111.72564	Floating
October 28	12	Middle	43.52612	-111.72368	Sinking
October 27	13	Upper	43.50336	-111.73889	Sinking
October 27	14	Upper	43.50400	-111.74468	Floating
October 27	15	Upper	43.50239	-111.75437	Sinking
October 27	16	Upper	43.50046	-111.76001	Floating
October 27	17	Upper	43.48119	-111.74784	Sinking
October 27	18	Upper	43.48669	-111.74451	Floating

Appendix I. Locations used in population surveys on the Henrys Fork Snake River, Idaho 2014. All locations used NAD27 and are in Zone 12.

	Start		End	
Reach	Easting	Northing	Easting	Northing
Box Canyon	468677	4917703	467701	4914352
St. Anthony	442187	4866559	437660	4864150

Appendix J. Mean total length, length range, proportional stock density (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, 1991-2014. 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

Appendix K. 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, 1995-2014. 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

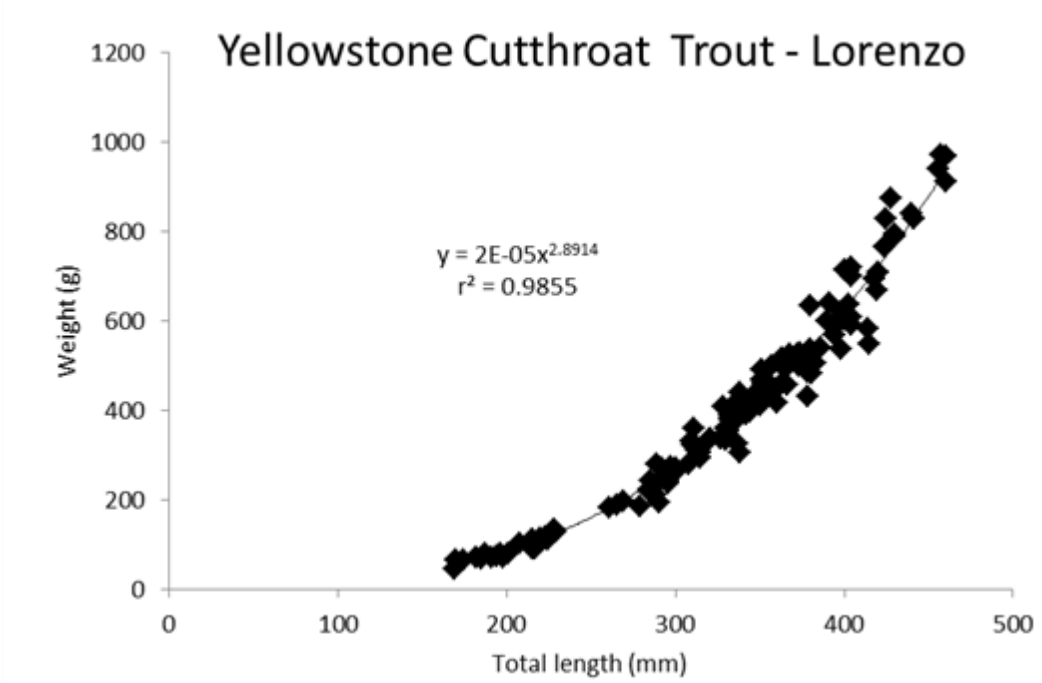
Year	M[a]	C[a]	R[a]	R/C (%)	CV	N/reach MPM	N/reach LLM	N/km LLM	Discharge (ft^3/s)
2010	1,309	1,292	262	20	0.03	6,439 (5,820-7,058)	8,341 (7,857-8,825)	2,254 (2,123-2,385)	626
2011	639	652	74	11	0.06	5,571 (4,516-6,988)	6,548 (5,816-7,280)	1,770 (1,572-1,968)	1,159
2012	793	901	116	13	0.04	6,120 (5,178-7,313)	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	0.06	5,547 (4,901-6,335)	5,828 (5,491-6,165)	1,575 (1,484-1,666)	971

[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.

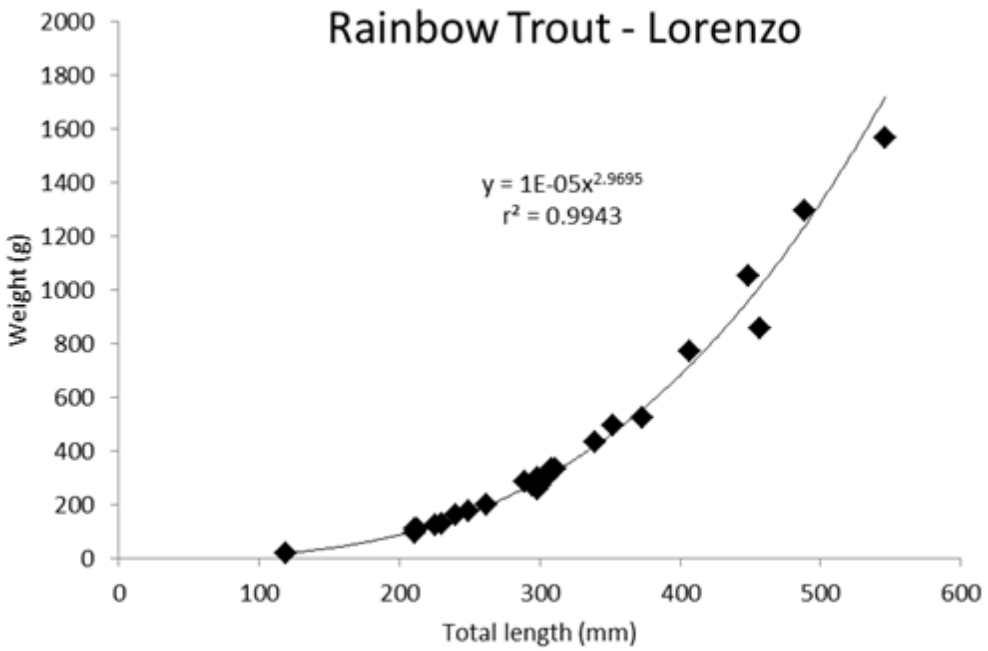
Appendix L. Locations of South Fork Snake River fish population monitoring sites, tributary weirs, and PIT tag arrays (WGS 84).

Site	Upstream boundary	Downstream boundary
Conant monitoring site	12T 467846 E 4810899 N	12T 465305 E 4814032 N
Lorenzo monitoring site	12T 430743 E 4841275 N	12T 428214 E 4844051 N
Burns Cr Weir	12T 462063 E 4827984 N	NA
Pine Cr Weir	12T 473373 E 4819000 N	NA
Rainey Cr Weir	12T 473284 E 4810412 N	NA
Palisades Cr Weir	12T 480668 E 4803039 N	NA
Dry Bed PIT array	12T 443999 E 4831136 N	NA
Burns Cr PIT array	12T 461795 E 4827725 N	NA
Pine Cr PIT array	12T 466503 E 4815440 N	NA
Palisades Cr PIT array	12T 480250 E 4802331 N	NA

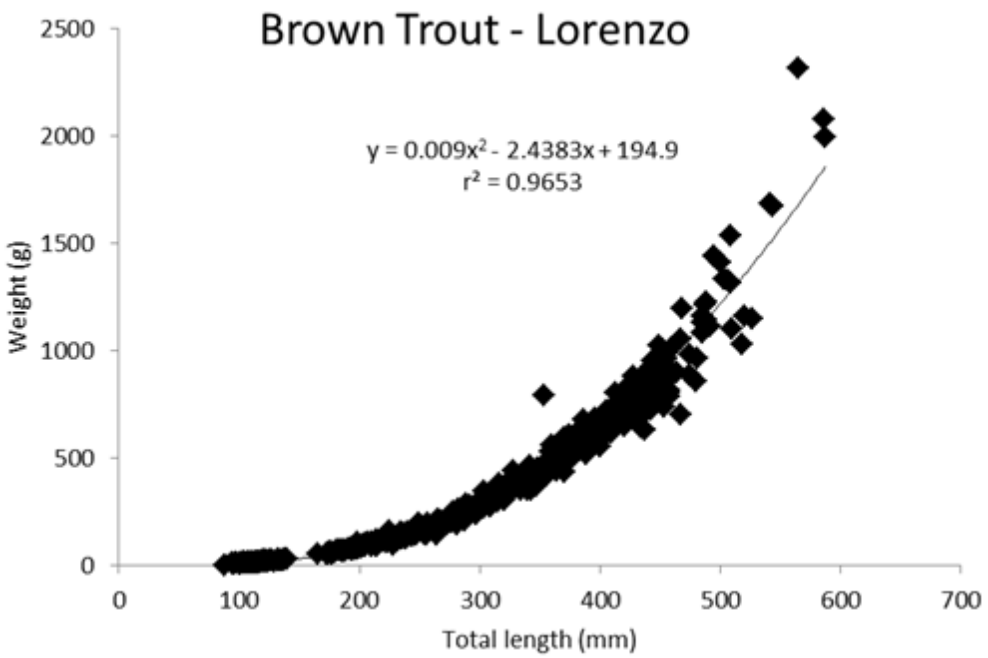
Appendix M. Length-weight regression for Yellowstone Cutthroat Trout captured at the Lorenzo monitoring reach of the South Fork Snake River in 2014.



Appendix N. Length-weight regression for Rainbow Trout captured at the Lorenzo monitoring reach of the South Fork Snake River in 2014.



Appendix O. Length-weight regression for Brown Trout captured at the Lorenzo monitoring reach of the South Fork Snake River in 2014.



Appendix P. Before and after pictures of channel width and flows at the Rainey Creek Weir. Picture A is taken before channel work (looking upstream) and picture B is looking downstream near the completion of channel work.

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