FISHERY RESEARCH

PROJECT 5: LAKE AND RESERVOIR RESEARCH

ANNUAL PROGRESS REPORT
July 1, 2017 — June 30, 2018

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PROJECT 5: LAKE AND RESERVOIR RESEARCH

Annual Performance Report

July 1, 2017 to June 30, 2018

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Project 5 – Lake and Reservoir Research

Subproject 1: Predation by double-crested cormorants and other avian piscivores on hatchery stocked rainbow trout in reservoir fisheries

Subproject 2: Air exposure and fight times for various catch-and-release fisheries in Idaho

Subproject 3: Effect of pulse frequency (Hz) on capture efficiency and injury of trout sampled with backpack electrofishing

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SUBPROJECT #1: PREDATION BY DOUBLE-CRESTED CORMORANTS AND OTHER AVIAN PISCIVORES ON HATCHERY STOCKED RAINBOW TROUT IN RESERVOIR FISHERIES

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ABSTRACT

The abundance of piscivorous birds such as double-crested cormorants *Phalacrocorax auritus* and American white pelicans *Pelecanus erythrorhynchos* has increased in recent decades in North America, resulting in increased conflict between birds and fisheries resources. Predation by these birds on fisheries in Idaho has become a concern for fisheries managers, prompting the need to quantify these specific impacts. We stocked two sizes of hatchery trout (250mm and 300mm) tagged with PIT, radio, and T-bar anchor tags into 15 Idaho waters and estimated angler catch and predation by avian piscivores. Estimated angler use, which includes fish harvested as well as caught-and-released, averaged 17% while avian predation averaged 25%. Predation specifically attributed to double-crested cormorants and American white pelicans was estimated at 16% and 6%, respectively. Avian predation rates were similar for standard (250mm) and magnum (300mm) sized catchables, suggesting that larger-sized fish are not better able to escape predation than smaller catchables. Our results suggest that in some southern Idaho fisheries, piscivorous birds, rather than anglers, are the dominant consumers of hatchery trout.

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INTRODUCTION

The abundance of many piscivorous birds such as double-crested cormorants *Phalacrocorax auritus* (hereafter cormorants) and American white pelicans *Pelecanus erythrorhynchos* (hereafter pelicans) has increased in recent decades in North America (Dorr and Fielder 2014). Generally, population increases are attributed to the discontinued use of organochlorine pesticides such as DDT and to reduced human persecution due to protections gained in 1972 under the Migratory Bird Treaty Act. Concurrent with bird population increases have been more conflicts between bird populations and fisheries resources. Avian predators such as cormorants and pelicans have exerted measurable predation pressure on several freshwater fisheries and aquaculture facilities (Dorr and Fielder 2017). The Idaho Department of Fish and Game (IDFG) stocks approximately 1.5 million catchable-sized (≥250mm) Rainbow Trout *Oncorhynchus mykiss* annually into lakes and rivers around the state. Recently, substantial levels of predation of wild and hatchery-reared trout by cormorants and pelicans in Idaho have been documented (Teuscher et al. 2015; Meyer et al. 2016).

Methods for estimating such predation typically involve tagging a proportion of stocked fish and recovering those tags at bird nesting, roosting, or loafing areas. Such estimates are considered conservative if they do not account for consumed tags that are unrecovered, (i.e., tag recovery efficiency). One approach to estimating tag recovery efficiency is by directly feeding birds with fish implanted with passive integrated transponder (PIT) tags and recovering tags deposited at nesting and roosting sites (Osterback 2013; Scoppettone 2014; Teuscher et al. 2015). Using this approach, an estimated 6.4-60.6% of adult wild Yellowstone Cutthroat Trout were consumed by pelicans in the Blackfoot River drainage of Idaho. Using these same methods, Meyer et al. (2016) estimated that pelicans consumed an average of 22% (range: 0-65%) of hatchery Rainbow Trout stocked in 12 Idaho Reservoirs. Estimated total predation of hatchery fish by cormorants averaged 21% (range: 5-69%) when using the tag recovery rate calculated for pelicans. However, estimating tag recovery efficiency for cormorants was not possible because direct feeding was unsuccessful.

In our study, we released fish tagged with both PIT and radio tags and used the recovery ratio of each tag type to calculate tag recovery efficiency and estimate total rather than minimum predation. We also estimated angler exploitation to compare to avian predation. Previous research comparing angler exploitation of different sized fish suggests that larger fish return to creel at higher rates than smaller fish (Cassinelli and Meyer 2018), and IDFG subsequently switched from stocking trout averaging ~250mm (hereafter “standards”) to trout averaging ~300mm (hereafter “magnums”) in lentic waters. We estimated avian predation on both sizes of trout to evaluate whether larger fish have an advantage in avoiding predation due to their size.

OBJECTIVES

1. Estimate total predation rates of hatchery-stocked Rainbow Trout by cormorants, pelicans, and other avian predators in southern Idaho reservoir fisheries where avian predators are known to reside.

2. Estimate angler exploitation and total use of stocked hatchery trout at these same waters.

3. Compare predation rates of “standard” and “magnum” sized fish by cormorants and pelicans to evaluate whether larger catchables are better able to avoid avian predators.

METHODS

Study areas

Study sites were selected as waters with known or suspected avian predation, and were primarily in southeastern Idaho (Figure 1). All of these waters regularly receive stockings of hatchery Rainbow Trout to sustain and provide sportfishing opportunities, typically in late spring when angler activity increases but also when these predatory bird species are nesting and foraging for their young. Potential tag recovery locations were identified based on known bird colonies and loafing/roosting areas, while others were discovered based on radio tag signals.

Fish tagging and stocking

Hatchery Rainbow Trout were stocked in all study waters in the springs of 2015 to 2017. Nearly all hatchery Rainbow Trout were stocked with T-bar anchor tags and PIT tags, but some fish were also tagged with radio tags (Table 1). T-bar anchor tags (Floy Tag & Mfg., Inc.; Seattle, WA) were placed on fish by inserting the T-bar into the base of the dorsal fin according to standard methods (Dell 1968). These anchor tags were printed with the tag number and the website/phone number where anglers can report tags to IDFG's "Tag You're It" tag reporting system, designed to track exploitation and what is herein termed total use of fish by anglers statewide (see below).

Half-duplex PIT tags (23 x 3mm) were inserted into the body cavity using a 6 gauge hypodermic needle (Prentice et al. 1990). In 2015, only standard-sized trout were tagged and stocked. In 2016 and 2017, equal numbers of standard and magnum-sized trout were tagged with anchor and PIT tags.

Radio tags (MST-093 Lotek) were surgically implanted into the body cavity of the fish by making a small incision into the ventral wall anterior to the pelvic girdle (Hart and Summerfelt 1975). A grooved needle shield was inserted posteriorly past the pelvic girdle and a 6 gauge needle was inserted between the pelvic girdle and the anal vent using the shielded needle technique to protect internal organs and direct the needle under the pelvic girdle and through the incision on the body wall (Ross and Kleiner 1982). The radio antenna was threaded through the needle so the antenna exited the hole made by the needle. While threading the antenna, the tag was inserted into the body cavity along with a PIT tag. The incision was closed using two sutures. Fish were placed in recovery water and monitored for at least 24 h prior to release. Only "magnum" sized fish were implanted with radio tags in 2016 and 2017. These radio tags were equipped with internal motion sensors to emit a mortality signal if the tag had not moved for 12 h, allowing for identification of fish mortalities due to predation or other causes, depending on location and detection history. Tagged fish were stocked in each water along with a larger group of non-tagged fish as a part of regular stocking schedules.

Radio telemetry

To monitor fish presence and potential removal by predators of radio-tagged fish, fixed radio receivers (Lotek SRX-400) were installed at each water where fish with radio tags were stocked, with the exception of Emerald and Rupert Gravel ponds due to vandalism concerns. At those two waters, mobile radio tracking was conducted in conjunction with bird surveys (see below). Fixed radio receivers were also installed at cormorant and pelican colonies at Minidoka NWR, Blackfoot Reservoir, and Island Park Reservoir to detect the arrival of depredated tagged fish. Receivers were programmed to scan tag-specific frequencies (150.380, 150.360, and

150.320 Mhz) every six seconds. Fixed telemetry stations consisted of a radio receiver powered by a 12V battery and housed in a lockable steel box. Two Yagi antennae with 3-4 elements mounted on a t-post and connected to the receiver with coaxial cable were oriented with the elements perpendicular to the ground and aimed in directions that maximized the scanning area.

Bird surveys

To gather data on relative bird abundance at each stocked water, non-random bird counts (*n* = 10) were conducted at Foster and Treasureton reservoirs in 2015 when technicians were present for radio telemetry tracking or radio receiver maintenance. In 2016 and 2017, random bird counts were conducted at each water once per week from 31 May to 3 October. Survey days and times (between 800-1800hr) were randomly assigned to each group of waters. For the sake of efficiency, waters near each other were surveyed for birds in the same day. Using binoculars either from shore or from a boat, we counted cormorants, pelicans, Great Blue Herons *Ardea herodias*, and osprey *Pandion haliaetus*.

Tag recovery

We recovered PIT tags deposited by birds at a variety of nesting, roosting, and loafing locations. Primary tag recovery locations were colony sites comprised of three small islands at Minidoka NWR, Gull Island at Blackfoot Reservoir, the island at Foster Reservoir, Trude Island at Island Park Reservoir, and Schoff's and Banbury islands along the Snake River. We also searched for and recovered several tags at bird loafing areas at Treasureton and Chesterfield reservoirs, as well as other unnamed locations along the Snake River. When scanning these areas, we detected PIT tags using Oregon RFID HDX backpack PIT tag readers attached to a 2m long pole with a 0.5m diameter hoop antenna on the end with a detection range of approximately 0.5m from the edge of the hoop. The recovery area was searched systematically by walking 2m transects while sweeping the antenna side to side until all the ground at recovery locations was scanned. When tags were detected, locations were marked with a survey flag. We used Biomark 601 handheld PIT tag readers (Biomark, Boise, Idaho) to precisely locate tags, and recovered tags by digging and scanning small amounts of material and using trowels and sieves when necessary (Teuscher et al. 2015; Meyer et al. 2016). Additionally, anchor tags were visually identified and recovered during this process.

We recovered deposited radio tags using a mobile Lotek SRX-800 telemetry receiver and handheld Yagi antenna. Upon the completion of the 2015 and 2016 bird surveys and manual tag recoveries at colonies, we completed a single pass sweep across all study waters via fixed wing aircraft outfitted with three directional telemetry antennae. In 2017, four flights were conducted along the Snake River. Tags detected aerially were marked with GPS waypoints subsequently located on foot or by boat.

Data analysis

Minimum predation estimates from recovered PIT ($Pred_{PIT}$) and radio tags ($Pred_{RADIO}$) were calculated by dividing the number of tags recovered by the number of tags stocked at each location. Variances for these proportions (Thompson 2012) were calculated using the formula:

$$Var(proportion) = \frac{P(1-P)}{n-1} \qquad \text{(Equation 1)}$$

where P is the proportion of recovered tags and n is the number of stocked tags. Ninety-five percent confidence intervals were calculated accordingly for each water. A paired *t*-test (α = 0.05) was used to compare numbers of PIT-tagged fish preyed upon that were either in the magnum or standard category for waters with paired releases. Because some overlap in fish lengths occurred between the two groups, we also tested size selectivity by comparing cumulative length frequency distributions of stocked versus eaten fish using a Kolmogorov-Smirnov test (α = 0.05).

Because an unknown proportion of consumed PIT tags go unrecovered, PIT tag recovery efficiency was not 100%. While also true for radio tags, the proportion of radio tags with known fate was much higher than for PIT tags. Therefore, PIT tag recovery efficiency (Tag_{rec}) was calculated for each water as the proportion of PIT-tagged fish where the PIT tag was recovered along with its respective radio tag. Total predation ($Pred_{Total}$) was calculated as

$$Pred_{Total} = \frac{Pred_{PIT}}{Tag_{rec}} \qquad \text{(Equation 2)}$$

For example, if 10 radio tags from double-tagged fish were recovered, but only 5 PIT tags were recovered, then the PIT tag recovery efficiency (Tag_{rec}) would be estimated to be 50%. Therefore, minimum predation estimates based on PIT tag recoveries would be doubled to estimate total predation from PIT tag recoveries.

We assigned predator species according to locations where tags were found. For example, tags were often found in nests, and were thus assumed to be eaten by that bird species. For tags found in ambiguous locations, we used relative proportions of cormorants, pelicans, herons, and osprey counted at those waters from which tags were stocked to assign predation events to a predator species (Meyer et al. 2016). Species-specific predation was assigned to predation estimates after adjusting for tag recovery efficiency.

Angler exploitation estimates

We estimated total angler use (i.e., harvested fish + caught-and-released fish) and exploitation (i.e., harvested fish only) of fish stocked into study waters using T-bar anchor tag returns reported to the IDFG "Tag You're It!" tagged fish reporting program (Meyer and Schill 2014). Estimates were adjusted to account for reporting rate using the equation:

$$\lambda = \frac{Rr/Rt}{Nr/Nt} \qquad \text{(Equation 3)}$$

where Rt and Rr are the number of non-reward tags stocked and reported and Nr and Nt are the number of \$50 reward tags stocked and reported (Pollock et al. 2001). We used a \$50 reward tag reporting rate of 88% (Meyer et al. 2012). For our study, we used statewide averages of non-reward tag reporting rates and tag loss rates. We estimated angler exploitation (u') using the equation:

$$u' = \frac{u}{\lambda(1-Tag_l)(1-Tag_m)} \qquad \text{(Equation 4)}$$

where u is the number of non-reward tagged fish harvested divided by the number of non-reward tags stocked, Tag_l is the first year tag loss rate based on returns data for double tagged fish, and Tag_m is the tagging mortality rate. To estimate total angler use, u was modified to include fish caught and released as well as harvested.

Cost analysis

The direct cost of hatchery Rainbow Trout consumed by cormorants in this study was estimated for each water by multiplying the predation rates by total number of catchable sized trout (10-12 inches) stocked into that water from March through August, a time period of stocking that roughly corresponded to our bird surveys. Costs per stocked fish were based on hatchery specific rearing expenses, which included feed, labor, and transportation. A linear regression with a zero intercept was fit for predation cost as a function of stocking numbers for these waters. Using this relationship, annual predation costs were estimated to other similar waters in southern Idaho where cormorants are known to have a sustained feeding presence.

RESULTS

Avian predation

A total of 5,294 fish were anchor tagged, 5,967 were PIT tagged, and 428 were radio tagged and released into 15 waters over 21 separate stocking events to estimate avian predation as well as angler exploitation and use of stocked hatchery Rainbow Trout. Of these, a total of 95 anchor tags, 780 PIT tags, and 91 radio tags were recovered in bird nesting, roosting, or loafing areas.

Minimum predation estimates based on recovered PIT tags averaged 16% (range, 1-58%; Table 2, Figure 1). Minimum predation estimates based solely on radio tags averaged 23.5%, (range, 0-56.3%). The estimated recovery rate of PIT tags associated with radio tags known to have been consumed by avian predators and deposited at recovery locations averaged 56% (range, 0-100%). Adjusted predation estimates from recovered PIT tags averaged 24.6% (range, 0-100%). The water with the highest predation using all three estimation methods was Rupert Gravel Pond.

Estimates of predation based on radio tags were on average about 5% higher than those based on PIT tags (Figure 2a.) However, when PIT tag estimates were adjusted to account for imperfect tag recovery rates, these corrections were not consistent across the range of observed predation estimates (Figure 2b.)

Minimum predation from recovered PIT tags on magnum and standard-sized catchable trout was 15.2% and 15.8%, respectively (Figure 3). Predation rates between magnum and standard-sized fish did not differ significantly ($t = 2.18$, $P = 0.78$;Figure 2). Cumulative length frequency distributions of stocked versus eaten fish also did not differ ($D = 0.032$, $P = 0.85$; Figure 4).

Species-specific estimates of total predation ranged widely between waters, averaging 16% for cormorants, ranging from <1% in the Snake River to 80% at Rupert Gravel Pond. We estimated that pelican predation averaged 6% (range, 0-29%; Table 3, Figure 5). Predation by herons was highest at 30% at Frank Oster pond, but generally was below 15% elsewhere. Species-specific predation estimates from Rupert Gravel Pond exceed 100% when summed because of a likely overestimate of tag recovery rate.

Of the total recovered tags, 343 (64%) were found in unambiguous locations (i.e., nests) and thus were attributed to specific avian predators. The distances of these tags from the water where they were stocked varied among species (Figure 6.). Heron foraging distances ranged from

9.5-19 km. Cormorant foraging distances averaged 28.6 km and ranged from 0-133 km. Pelican foraging distances averaged 52.2 km and ranged from 13.5-150 km.

Catch and harvest

Total angler use, which consisted of trout caught and released as well as trout harvested, averaged 16% (range, 0-26%) for magnum sized catchables and 14% (range, 0-35%) for standard sized trout (Table 4). Angler exploitation (i.e., the number of fish that anglers harvested within one year of stocking) averaged 12% (range, 0-25%) for magnums and 10% (range, 0-35%) for standards. Tag reporting rates were 45%, 45%, and 51% in 2015, 2016, and 2017. Corresponding tag loss rates were 2.5%, 0%, and 0%. No relationship was detected between avian predation and angler catch among water bodies and stocking groups (Figure 7).

Bird surveys

We conducted 94 randomized bird counts at 10 waters from 31 May 2016 to 3 October 2016 and 16 May 2017 to 05 September 2017. Cormorants were the most abundant bird species observed at four reservoirs. Chesterfield Reservoir had the highest daily average number of cormorants (12) and pelicans (9) observed. At Glendale Reservoir, herons were most numerous, while pelicans were most abundant at Treasureton Reservoir (Table 5).

Cost of predation by cormorants

The total cost of Rainbow Trout consumed by cormorants at each water in this study was estimated at $18,025 (Table 6). A significant relationship between numbers of fish stocked (x) and predation cost (y) revealed a relationship of $y = 0.1338x$ ($R^2 = 0.74$, $P = 0.001$). Using this relationship, the estimated total annual cost of fish eaten by cormorants in southern Idaho lakes and reservoirs was estimated at $46,428 (Table 7.)

DISCUSSION

Avian predation on salmonid populations has been the subject of much research across the western United States (Modde et al. 1996; Derby and Lovvorn 1997; Evans et al. 2012; Teuscher et al. 2015; Meyer et al. 2016). These studies use either diet samples or tag recoveries from tagged fish to quantify predation. In studies using tagged fish, tag detection probabilities are estimated by either sowing tags into recovery areas (Evans et al. 2012), an approach that does not account for off-site tag deposition, or by feeding tagged fish directly to birds (Osterback et al. 2013; Scoppettone et al. 2014; Hostetter et al. 2015; Teuscher et al. 2015; Meyer et al. 2016), an approach that is not feasible with some species, including cormorants. We instead used double-tagged fish (PIT and radio), allowing us to account for both consumed PIT tags that went undetected at nesting colonies and consumed PIT tags that were deposited off-colony.

Although our approach to adjusting minimum estimates differed from previous studies, our estimates of predation by cormorants and pelicans are similar to those reported for inland resident fisheries (Modde et al. 1996; Derby and Lovvorn 1997; Teuscher et al. 2015; Meyer et al. 2016). In the Blackfoot River of Idaho, total predation rates on adfluvial Yellowstone Cutthroat Trout by pelicans averaged 26% over four years and ranged from 6-70% (Teuscher et al. 2015). In a more widespread study of avian predation on Idaho fisheries, hatchery-stocked rainbow trout were preyed upon at an average rate of 18% by pelicans (0-65%), with additional predation attributed to cormorants (2-38%), and total bird predation often exceeding angler use (Meyer et al. 2016).

Our results add to the growing body of evidence demonstrating predation impacts by cormorants and pelicans on Rainbow Trout among southern Idaho fisheries. We documented avian predation on hatchery stocked Rainbow Trout in 95% of the stocking events in 15 waters from 2015-2017 using PIT tags and radio tags recovered from colony sites. We also demonstrated that the use of double-tagged fish can correct for undetected predation events and adjust minimum predation estimates. Our results suggest that in some southern Idaho fisheries, piscivorous bird predation on hatchery trout exceeds angler exploitation.

During tag recovery efforts, tags recovered directly from bird nests were assumed to be eaten by those specific birds. Many tags were found in ambiguous areas used by cormorants and pelicans. In these instances, proportional bird abundances based on counts at waters from where these tags were released were used to assign predation to a bird species. Limitations to using this method exist, as noted by Meyer et al. (2016). Using proportional abundance to assign predation events to respective bird species assumes that fish consumption occurs at equal rates. Even though pelicans require four to eight times the daily energy intake (Hall 1925; Ferguson et al. 2011) compared to cormorants (Seefelt and Gillingham 2008), numbers of PIT tags found specifically in cormorants nests at the Minidoka colony exceeded numbers found in pelican nests. This finding suggests that at the waters we included in our study, cormorants simply consumed more Rainbow Trout than pelicans, regardless of lower daily energy requirements. Perhaps cormorants are more effective at preying upon these fish and therefore the trout make up a larger proportion of their diet.

The foraging distance range inferred for pelicans (13.5-149 km; Johnson and Sloan 1978; O'Malley and Evans 1982) and herons (9.5-10.5 km; Dowd and Flake 1985) concurs with other studies, but foraging distances measured for cormorants (0-133 km) are greater than any reported in the literature (Custer and Bunck 1992; Bugajski et al. 2013). A possible explanation for this may be that cormorants sometimes incorporate shiny or flashy debris into their nests (Podolsky and Kress 1989), and therefore may have intentionally picked up the already deposited PIT, Floy, or radio tags and placed them in their nests. If such is the case, predation attributed to cormorants may be overestimated.

Total use by anglers was similar to predation by pelicans, but less than predation by cormorants. We did not see a consistent relationship between predation and angler use, though a few exceptions were evident. Our highest estimated predation (100%) occurred at Rupert Gravel pond, where 0% angler catch was reported. Overall, total avian predation was more than double the amount of estimated angler use. Because significant resources go into the rearing of hatchery Rainbow Trout, this presents not only an economic issue but an angler satisfaction issue. If a third of all Rainbow Trout that get stocked into these waters are eaten by birds, then fewer fish are available for angler harvest.

The estimated costs of predation by cormorants in this report likely underestimate the total economic costs for several reasons. First, our estimates only include lakes that were stocked between March and August with catchable sized trout. Thus any fingerling sized fish or fish that were stocked outside this time period were not accounted for in the stocking numbers or potential predation costs. Second, we did not include rivers due to the uncertainty of bird use along their gradient with respect to multiple fish stocking locations. Third, there was no way to estimate reduced angler effort and related angler spending that likely occurs when trout are consumed by predators.

RECOMMENDATIONS

1. Because magnum-sized catchables were not less vulnerable to avian predation than standard-sized catchables, switching hatchery stocking to larger catchables to reduce avian predation is not justified.

2. Management alternatives for addressing pelican predation identified in the Management plan for the conservation of American white pelicans in Idaho (IDFG 2016) are also applicable to cormorants. These management actions include, but are not limited to, modification of hatchery trout stocking strategies; hazing or lethal take of birds at foraging, loafing, and nesting locations; oiling eggs to reduce recruitment, nesting exclusion or nest destruction; and establishing population number objectives.

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LITERATURE CITED

Bugajski, A., M. W. Reudink, J. L. Doucette, S. E. Franks, B. Wissel, and C. M. Somers. 2013. The complexity of cormorants: stable isotopes reveal multiple prey sources and feeding site switching. Canadian Journal of Fisheries & Aquatic Sciences 70:271–279.

Cassinelli, J. D., and K. A. Meyer. 2018. Factors influencing return-to-creel of hatchery catchable Rainbow Trout stocked in Idaho lentic waters. Fisheries Research 204:316-323.

Custer, T. W., and C. Bunck. 1992. Feeding Flights of Breeding Double-Crested Cormorants at Two Wisconsin Colonies. Journal of Field Ornithology 63:203–211.

Dell, M. B. 1968. A New Fish Tag and Rapid, Cartridge-Fed Applicator. Transactions of the American Fisheries Society 97:57–59.

Derby, C. E., and J. R. Lovvorn. 1997. Predation on fish by cormorants and pelicans in a cold-water river: a field and modeling study. Canadian Journal of Fisheries and Aquatic Sciences 54:1480–1493.

Dorr, B., and D. Fielder. 2017. Double-Crested Cormorants: Too Much of a Good Thing? USDA National Wildlife Research Center - Staff Publications.

Dowd, E. M., and L. D. Flake. 1985. Foraging Habitats and Movements of Nesting Great Blue Herons in a Prairie River Ecosystem, South Dakota. Journal of Field Ornithology 56:379–387.

Evans, A. F., N. J. Hostetter, D. D. Roby, K. Collis, D. E. Lyons, B. P. Sandford, R. D. Ledgerwood, and S. Sebring. 2012. Systemwide Evaluation of Avian Predation on Juvenile Salmonids from the Columbia River Based on Recoveries of Passive Integrated Transponder Tags. Transactions of the American Fisheries Society 141:975–989.

Ferguson, T. L., B. J. Rude, and D. Tommy King. 2011. Nutrient Utilization and Diet Preference of American White Pelicans Consuming Either a Mono- or Multi-Species Diet. Waterbirds 34:218–224.

Hall, E. R. 1925. Pelicans versus Fishes in Pyramid Lake. The Condor 27(4):147–160.

Hart, L. G., and R. C. Summerfelt. 1975. Surgical Procedures for Implanting Ultrasonic Transmitters into Flathead Catfish (*Pylodictis olivaris*). Transactions of the American Fisheries Society 104:56–59.

Hostetter, N. J., A. F. Evans, B. M. Cramer, K. Collis, D. E. Lyons, and D. D. Roby. 2015. Quantifying Avian Predation on Fish Populations: Integrating Predator-Specific Deposition Probabilities in Tag Recovery Studies. Transactions of the American Fisheries Society 144:410–422.

IDFG. 2016. Management Plan for the Conservation of American White Pelicans in Idaho - A ten-year plan (2016-2025) to conserve American White Pelican populations and manage impacts to fisheries resources in Idaho. Idaho Department of Fish and Game. Boise.

Johnson, R. F., and N. F. Sloan. 1978. White Pelican Production and Survival of Young at Chase Lake National Wildlife Refuge, North Dakota. The Wilson Bulletin 90:346–352.

Meyer, K. A., F. S. Elle, J. A. Lamansky. Jr., E. R. J. M. Mamer, and A. E. Butts. 2012. A Reward-Recovery Study to Estimate Tagged-Fish Reporting Rates by Idaho Anglers. North American Journal of Fisheries Management 32:696–703.

Meyer, K. A., and D. J. Schill. 2014. Use of a Statewide Angler Tag Reporting System to Estimate Rates of Exploitation and Total Mortality for Idaho Sport Fisheries. North American Journal of Fisheries Management 34:1145–1158.

Meyer, K. A., C. L. Sullivan, P. Kennedy, D. J. Schill, D. M. Teuscher, A. F. Brimmer, and D. T. King. 2016. Predation by American White Pelicans and Double-Crested Cormorants on Catchable-Sized Hatchery Rainbow Trout in Select Idaho Lentic Waters. North American Journal of Fisheries Management 36:294–308.

Modde, T., A. F. Wasowicz, and D. K. Hepworth. 1996. Cormorant and Grebe Predation on Rainbow Trout Stocked in a Southern Utah Reservoir. North American Journal of Fisheries Management 16:388–394.

O'Malley, J. B. E., and R. M. Evans. 1982. Flock formation in white pelicans. Canadian Journal of Zoology 60:1024–1031.

Osterback, A.-M. K., D. M. Frechette, A. O. Shelton, S. A. Hayes, M. H. Bond, S. A. Shaffer, and J. W. Moore. 2013. High predation on small populations: avian predation on imperiled salmonids. Ecosphere 4(9):1–21.

Podolsky, R. H. and S. W. Kress. 1989. Plastic debris incorporated into double-crested cormorant nests in the Gulf of Maine. Journal of Field Ornithology 60: 248-250.

Pollock, K. H., J. M. Hoenig, W. S. Hearn, and B. Calingaert. 2001. Tag Reporting Rate Estimation: 1. An Evaluation of the High-Reward Tagging Method. North American Journal of Fisheries Management 21:521–532.

Prentice, E. F., T. A. Flagg, and C. S. McCutcheon. 1990. Feasibility of using implantable passive integrated transponder (PIT) tags in salmonids. Pages 317–322 Fish-marking techniques Symposium 7. American Fisheries Society, Bethesda, Maryland.

Ross, M. J., and C. F. Kleiner. 1982. Shielded needle technique for surgically implanting radio frequency transmitters in fish. Progressive Fish-Culturist (44):41–43.

Scoppettone, G. G., P. H. Rissler, M. C. Fabes, and D. Withers. 2014. American White Pelican Predation on Cui-ui in Pyramid Lake, Nevada. North American Journal of Fisheries Management 34:57–67.

Seefelt, N. E., and J. C. Gillingham. 2008. Bioenergetics and Prey Consumption of Breeding Double-crested Cormorants in the Beaver Archipelago, Northern Lake Michigan. Journal of Great Lakes Research 34:122–133.

Teuscher, D. M., M. T. Green, D. J. Schill, A. F. Brimmer, and R. W. Hillyard. 2015. Predation by American White Pelicans on Yellowstone Cutthroat Trout in the Blackfoot River Drainage, Idaho. North American Journal of Fisheries Management 35:454–463.

Thompson, S. K. 2012. Sampling Third. John Wiley & Sons, Inc., Hoboken, New Jersey.

TABLES

Table 1. Tag types and fish numbers stocked into study waters.

Water	Surface area (hectares)	Year	Floy	PIT	Radio	Total
American Falls Reservoir	22363.3	2016	400	430	30	430
		2017	398	428	30	428
Chesterfield Reservoir	504.1	2016	400	430	30	430
Deep Creek Reservoir	65.8	2015	300	300	0	400
East Harriman Pond	12.4	2017	150	170	20	170
Emerald Lake	13.6	2016	100	229	30	229
Foster Reservoir	52.3	2015	300	330	29	430
		2016	200	230	30	230
Frank Oster #4 Pond	2.3	2017	200	217	17	217
Glendale Reservoir	82.2	2015	300	300	-	400
		2016	400	430	30	430
Jensen Grove Pond	26.1	2017	397	427	30	427
Johnson Reservoir	17.4	2015	300	329	30	430
Lamont Reservoir	34.2	2015	300	298	-	400
Riley Creek Pond	11.7	2017	200	216	16	216
Rupert Gravel Pond	4.3	2016	50	116	16	116
Snake River		2017	399	429	30	429
Treasureton Reservoir	61.4	2015	300	328	30	430
		2016	200	230	30	230

Table 2. Study waters with numbers of tagged fish stocked and predation estimates for (a) PIT tags and for (b) radio tags.

(a)

Release Location	Year	Number of PIT-tagged fish stocked	Number of PIT tags recovered	Minimum Predation	Tag recovery efficiency	Total Predation
American Falls Reservoir	2016	400	25	0.06	0.50	0.13
	2017	398	10	0.03	0.00	0.03
Chesterfield Reservoir	2016	400	64	0.16	0.78	0.21
Deep Creek Reservoir	2015	400	0	0.00		0.00
East Harriman Pond	2017	150	27	0.18	0.57	0.32
Emerald Lake	2016	199	74	0.37	1.00	0.37
Foster Reservoir	2015	300	110	0.37	0.67	0.55
	2016	200	49	0.25	0.91	0.27
Frank Oster Pond	2017	200	24	0.12	0.20	0.60
Glendale Reservoir	2015	300	55	0.18	NA	0.18
	2016	400	36	0.09	0.40	0.23
Jensens Grove Pond	2017	397	1	0.00	0.00	0.00
Johnson Reservoir	2015	300	39	0.13	0.50	0.26
Lamont Reservoir	2015	298	63	0.21	NA	0.20
Riley Creek Pond	2017	200	10	0.05	0.50	0.10
Rupert Gravel Pond	2016	100	58	0.58	0.44	1.00
Snake River	2017	399	3	0.01	1.00	0.01
Treasureton Reservoir	2015	298	41	0.14	NA	0.13
	2016	200	25	0.13	1.00	0.13

(b)

Release Location	Year	Number of radio-tagged fish stocked	Number of radio tags recovered	Predation estimate	95% C.I.
American Falls Reservoir	2016	30	2	0.07	0.09
	2017	30	0	0.00	-
Chesterfield Reservoir	2016	30	9	0.30	0.17

East Harriman Pond	2017	20	7	0.35	0.18
Emerald Lake	2016	30	11	0.37	0.18
Foster Reservoir	2015	29	14	0.48	0.19
	2016	30	11	0.37	0.18
Frank Oster Pond	2017	17	5	0.29	0.23
Glendale Reservoir	2016	30	5	0.17	0.14
Jensens Grove Pond	2017	30	0	0.00	-
Johnson Reservoir	2015	30	6	0.20	0.15
Riley Creek Pond	2017	16	2	0.13	0.17
Rupert Gravel Pond	2016	16	9	0.56	0.25
Snake River	2017	30	2	0.07	0.09
Treasureton Reservoir	2015	30	1	0.03	0.07
	2016	30	6	0.20	0.15

Table 3. Bird-specific total predation by double-crested cormorants, American White Pelicans, Great Blue Herons, and Ospreys.*Recovery efficiency estimates adjusted total predation to >100%.

Water	Year	Cormorant	Pelican	Heron	Osprey
Foster	2015	0.42	0.01	0.12	0
Glendale		0.14	0	0.04	0
Johnson		0.20	0	0.06	0
Lamont		0.15	0.02	0.04	0
Treasureton		0.07	0.06	0.01	0
American Falls	2016	0.04	0.09	0	0
Chesterfield		0.09	0.10	0.01	0
Emerald		0.27	0.09	0.01	0.01
Foster		0.21	0	0.06	0.01
Glendale		0.10	0	0.11	0.02
Rupert Gravel Pond*		0.80	0.29	0.14	0.08
Treasureton		0.05	0.07	0.01	0
American Falls	2017	0	0.02	0	0
East Harriman		0.30	0.01	0	0
Frank Oster		0.05	0.25	0.30	0
Jensens Grove		0	0	0	0
Riley Creek Pond		0.07	0.02	0.01	0
Snake River		0	0.01	0	0

Table 4. Angler exploitation and total use with 90% confidence intervals (CI) for hatchery Rainbow Trout stocked in southern Idaho waters. Estimates of angler exploitation and use were adjusted to account for year specific tag loss and tag reporting rates. NA refers to estimates for which CIs could not be estimated.

		Standard-sized catchables				Magnum-sized catchables			
		Angler exploitation		Angler total use		Angler exploitation		Angler total use	
Water Body	Year	Estimate	90% CI	Estimate	90% CI	Estimate	90% CI	Estimate	90% CI
American Falls Reservoir	2016	0.03	0.03	0.06	0.04	0.13	0.06	0.18	0.08
American Falls Reservoir	2017	0.05	0.04	0.06	0.04	0.06	0.04	0.08	0.05
Chesterfield Reservoir	2016	0.19	0.08	0.28	0.10	0.09	0.05	0.14	0.07
Deep Creek Reservoir	2015	0.11	0.05	0.21	0.07				
East Harriman Pond	2017	0.05	0.06	0.11	0.09	0.16	0.11	0.21	0.13
Emerald Lake	2016	0.27	0.17	0.31	0.19	0.09	0.10	0.18	0.14
Foster's Reservoir	2015	0.11	0.05	0.11	0.05				
Foster's Reservoir	2016	0.14	0.09	0.16	0.10	0.25	0.12	0.24	0.12
Frank Oster Pond #4	2017	0.06	0.07	0.06	0.07	0.06	0.07	0.06	0.07
Frank Oster Pond #4	2017	0.18	0.16	0.29	0.21	0.06	0.10	0.12	0.14
Glendale Reservoir	2015	0.07	0.04	0.19	0.07				
Glendale Reservoir	2016	0.09	0.05	0.09	0.05	0.18	0.08	0.22	0.09
Jensen Grove Pond	2017	0.19	0.08	0.22	0.09	0.26	0.10	0.26	0.10
Johnson Reservoir	2015	0.07	0.04	0.16	0.06				
Lamont Reservoir	2015	0.01	0.01	0.02	0.02				
Riley Creek Pond	2017	0.35	0.23	0.35	0.23	0.18	0.17	0.24	0.19
Riley Creek Pond	2017	0.12	0.10	0.18	0.12	0.18	0.12	0.24	0.14
Rupert gravel Pond Club	2016	0.00	NA	0.00	NA	0.00	NA	0.00	NA
Snake River	2017	0.05	0.04	0.06	0.04	0.07	0.05	0.12	0.06
Treasureton Reservoir	2015	0.00	NA	0.02	0.02				
Treasureton Reservoir	2016	0.00	NA	0.07	0.06	0.00	NA	0.07	0.06

Table 5. The daily mean and total number of Double-crested Cormorants (DCC), American White Pelicans (AWP), Great Blue Herons (GBH), and Ospreys (OSP) counted at each water from 31 May to 3 October in 2016 and 2017.

Water		DCC		AWP		GBH		OSP	
	N	mean	range	mean	range	mean	range	mean	range
2015									
Foster	10	5.5	0-19	0.0	-	1.6	0-5	0.0	-
Treasureton	10	5.8	0-16	2.8	0-10	1.6	0-4	0.0	-
2016									
Chesterfield	26	12.4	0-46	9.2	0-24	1.6	0-7	0.2	0-4
Emerald Lake	17	1.6	0-9	0.3	0-3	0.1	0-1	0.1	0-2
Foster	26	7.5	2-31	0.5	0-8	2.0	0-9	0.2	0-2
Glendale	24	1.3	0-7	0.0	-	1.5	0-6	0.3	0-2
Rupert pond	18	0.6	0-4	0.2	0-4	0.3	0-2	0.2	0-1
Treasureton	30	4.7	0-14	6.8	0-31	1.1	0-2	0.1	0-1
2017									
East Harriman	15	1.6	0-4	0.0	-	0.1	-	0.6	0-2
Frank Oster #4	13	0.5	0-4	2.2	0-14	0.1	-	0.0	-
Jensen Grove	12	0.0	0-0	0.0	-	0.0	-	0.3	0-2
Riley Creek	13	1.5	0-5	2.8	0-10	0.0	-	0.1	0-1

Table 6. Estimated costs of Rainbow Trout consumed by double-crested cormorants based on estimates of known tagged trout predation, hatchery costs, and total fish stocking records.

Water	Rearing Hatchery	Surface Hectares	Year	Number stocked*	Cormorant predation	Trout consumed	Cost per fish**	Cost
Snake River	American Falls	NA	2017	6636	0.003	17	$1.28	$21
American Falls	American Falls	22363.6	2016	18426	0.04	263	$1.28	$337
American Falls	American Falls	22363.6	2017	18431	0.00	11	$1.28	$14
Chesterfield	Grace	504.1	2016	23710	0.09	2158	$1.79	$3,863
Glendale	Grace	82.2	2016	4300	0.10	416	$1.79	$745
Glendale	Grace	82.2	2015	6104	0.14	862	$1.79	$1,542
Deep Creek	Grace	65.8	2017	4048	0		$1.79	$0
Treasureton	Grace	61.4	2016	230	0.05	20	$1.79	$36
Treasureton	Grace	61.4	2015	3985	0.07	1075	$1.79	$1,925
Foster	Grace	52.3	2015	3906	0.42	1624	$1.79	$2,907
Foster	Grace	52.3	2016	2340	0.21	486	$1.79	$869
Lamont	Grace	34.2	2015	4603	0.15	697	$1.79	$1,248
Jensens Grove	American Falls	26.1	2017	5077	0.00	0	$1.79	$0
Johnson	Grace	17.0	2015	1750	0.20	350	$1.79	$627
Emerald	Hagerman	13.6	2016	1010	0.27	272	$1.90	$516
East Harriman	American Falls	12.4	2017	1500	0.30	455	$1.79	$814
Riley Creek Pond	Hagerman	11.7	2017	14688	0.07	1028	$1.90	$1,954
Rupert Gravel Pond	Nampa	4.3	2016	392	0.80	313	$1.15	$359
Frank Oster #4	Hagerman	2.3	2017	2404	0.05	131	$1.90	$248
							Total	**$18,025**

* Stocking numbers are numbers catchable sized Rainbow Trout (10-12 inches) stocked from March-August during 2015-2017 at each water.
**Cost per fish includes hatchery feed, labor, stocking, and distribution specific to each facility.

Table 7. Estimated predation costs in Idaho ponds, lakes, and reservoirs where double-crested cormorants are known to frequent.

Water	**Surface Area (ha)**	**Number stocked***	**Estimated cost****
Alexander Reservoir	409.5	14089	$1,885.11
American Falls Reservoir	22363.6	27726	$3,709.74
Blair Trail Reservoir	4.1	3111	$416.25
Camas Pond #2	1.3	3170	$424.15
Carmella Vineyards Pond	0.2	286	$38.27
Castle Rock State Park Pond	0.6	1668	$223.18
Chesterfield Reservoir	504.1	21777	$2,913.76
Crowthers Reservoir	5.1	4032	$539.48
Crystal (Springs) Lake	0.6	4494	$601.30
Crystal Springs Pond	2.1	4642	$621.10
Deep Creek Reservoir	65.8	6022	$805.74
Devils Creek Reservoir	34.6	11305	$1,512.61
Dierkes Lake	10.4	4389	$587.25
Dog Creek Reservoir	4.0	5314	$711.01
East Harriman Pond	12.4	1500	$200.70
Filer Pond	1.8	4019	$537.74
Foster Reservoir	52.3	3667	$490.64
Frank Oster Pond #1	1.2	13875	$1,856.48
Frank Oster Pond #2	1.1	1875	$250.88
Frank Oster Pond #3	2.6	1479	$197.89
Frank Oster Pond #4	2.3	1901	$254.35
Freedom Park Pond	0.4	220	$29.44
Glendale Reservoir	86.6	5589	$747.81
Hagerman Bass Pond	2.2	570	$76.27
Island Park Reservoir	2946.7	65851	$8,810.86
Jensens Grove Pond	26.1	4985	$666.99
Johnson Reservoir	17.0	2252	$301.32
Lamont Reservoir	34.2	3779	$505.63
Little Camas Reservoir	390.6	3321	$444.35
Magic Reservoir	1553.9	6184	$827.42
McTucker Pond	2.6	5747	$768.95
Mormon Reservoir	634.6	4159	$556.47
Oakley Reservoir	407.1	12814	$1,714.51
Riley Creek Pond	11.7	14334	$1,917.89
Rose pond	7.9	2150	$287.67
Roseworth Reservoir	351.6	15123	$2,023.46
Rupert Gravel Pond	4.3	392	$52.45

Salmon Falls Creek Reservoir	1072.0	40084	$5,363.24
Stoddard Mill Pond	0.2	999	$133.67
Stone Reservoir	50.2	5261	$703.92
Treasureton Reservoir	61.4	6268	$838.66
Twin Lakes Reservoir	176.8	4034	$539.75
Winder Reservoir	30.6	2053	$274.69
Wood Duck Pond	0.2	487	$65.16
		Total	**$46,428**

* Stocking numbers are averages of catchable sized Rainbow Trout (10-12 inches) stocked from March-August during 2015-2017 at each water.
** Costs (y) are based on stocking numbers (x) using the linear equation y = 0.1338x ($R^2 = 0.74$, P = 0.001), a relationship derived from the predation estimates and stocking numbers of waters used in this study.

FIGURES

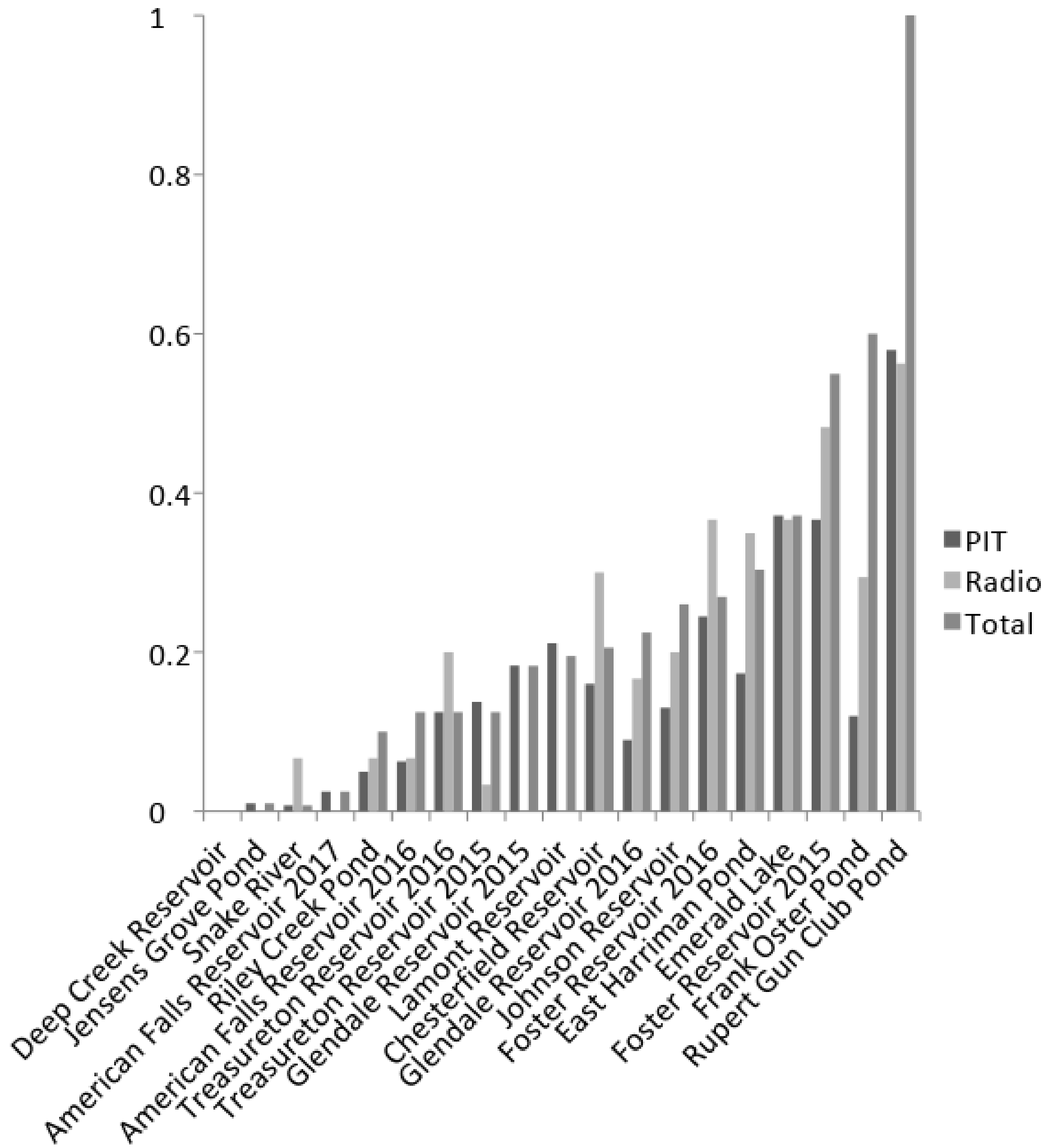
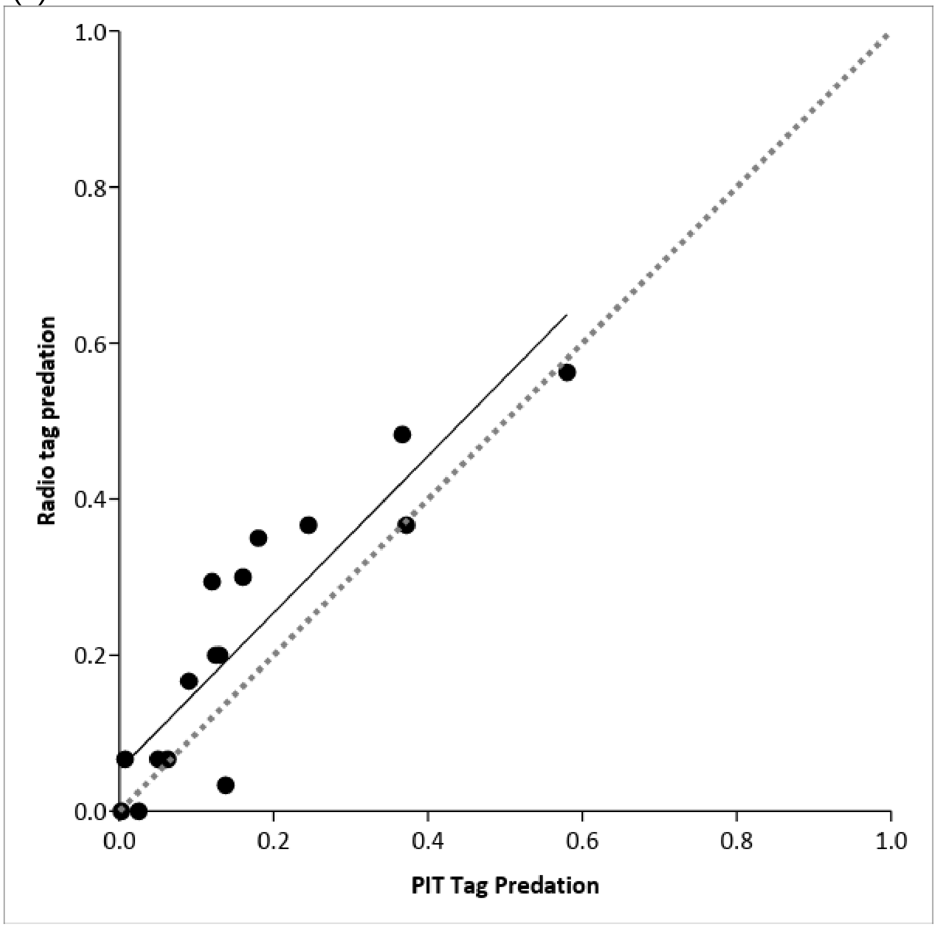


Figure 1. Avian predation estimates based on PIT tags, radio tags, and PIT tags corrected for tag recovery efficiency based on fish tagged with both PIT and radio tags (total).

(a)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Radio tag predation
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PIT Tag Predation

(b)

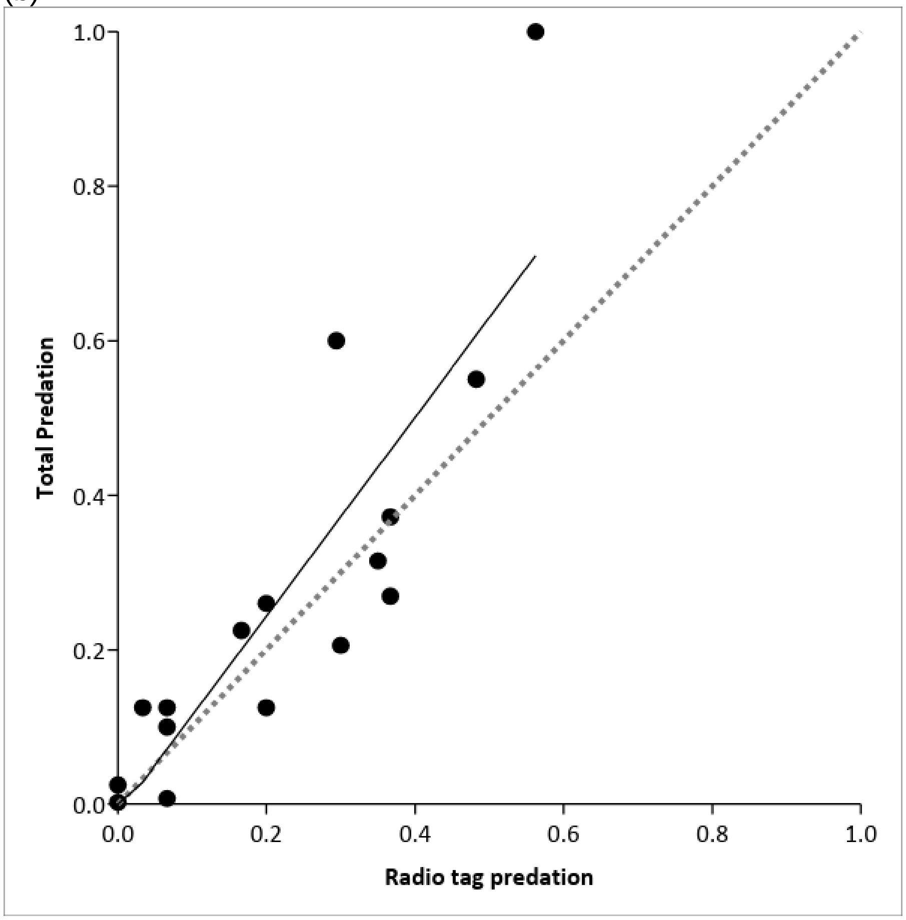

Figure 2. (a) Minimum predation estimates based on recovery of PIT tags and radio tags for each water. The solid line shows the best fit for a linear regression (y = 1.005x + 0.05, R^2 = 0.802); the dotted line shows a hypothetical 1:1 relationship between radio and PIT tag predation estimates. (b) Radio tag predation estimates and corresponding total predation estimates adjusted for tag recovery rates. The solid line shows the best fit for the linear regression (y = 1.288x - 0.014, R^2 = 0.746)

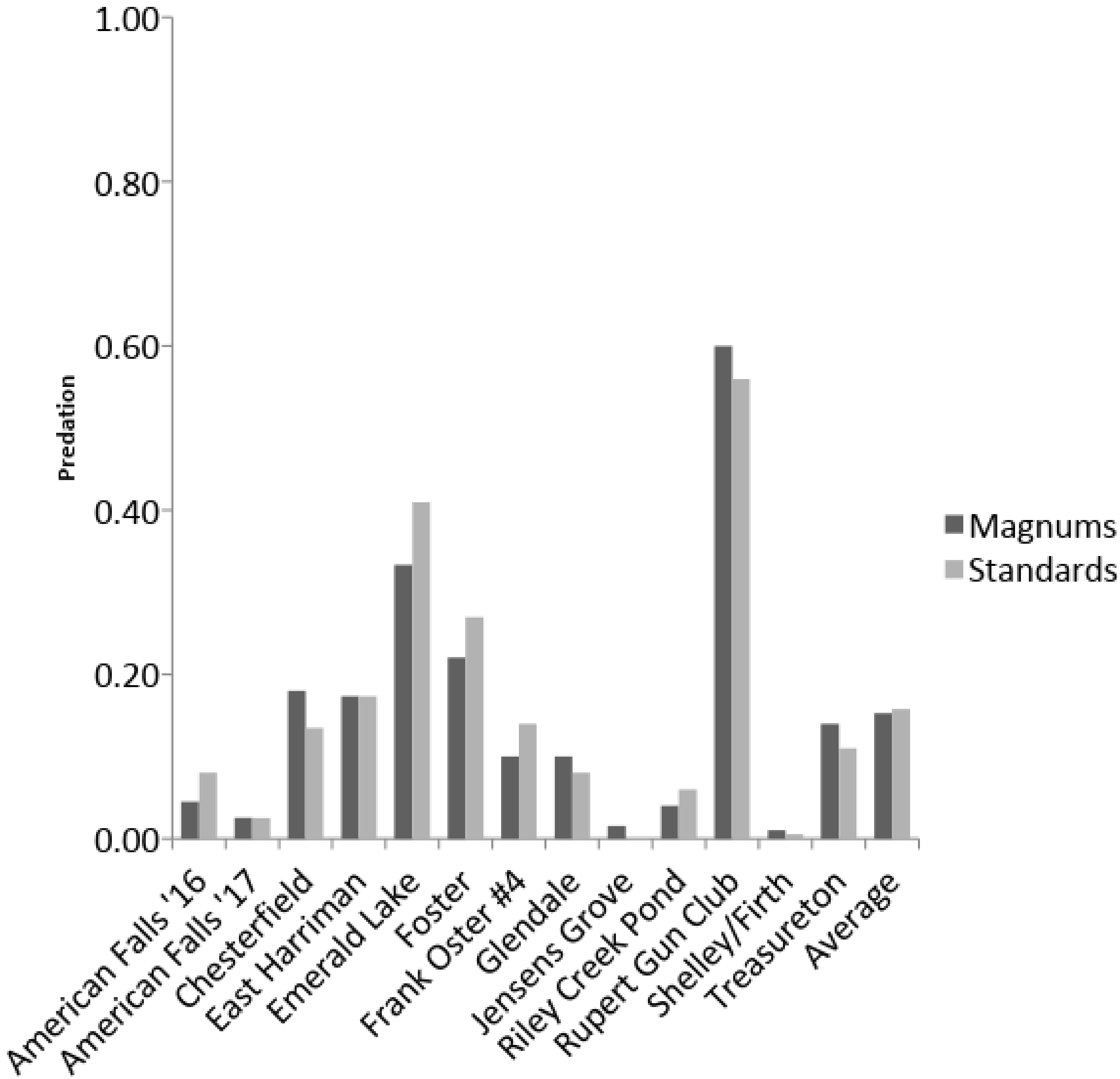

Figure 3. Minimum avian predation estimates of magnum (~300mm) and standard (~250mm) sized hatchery Rainbow Trout tagged with passive integrated transponders (PIT) and stocked into reservoir fisheries. Estimates are based on PIT tags recovered from bird nesting, loafing, or roosting locations. Only sites where paired releases of magnums and standards occurred are shown.

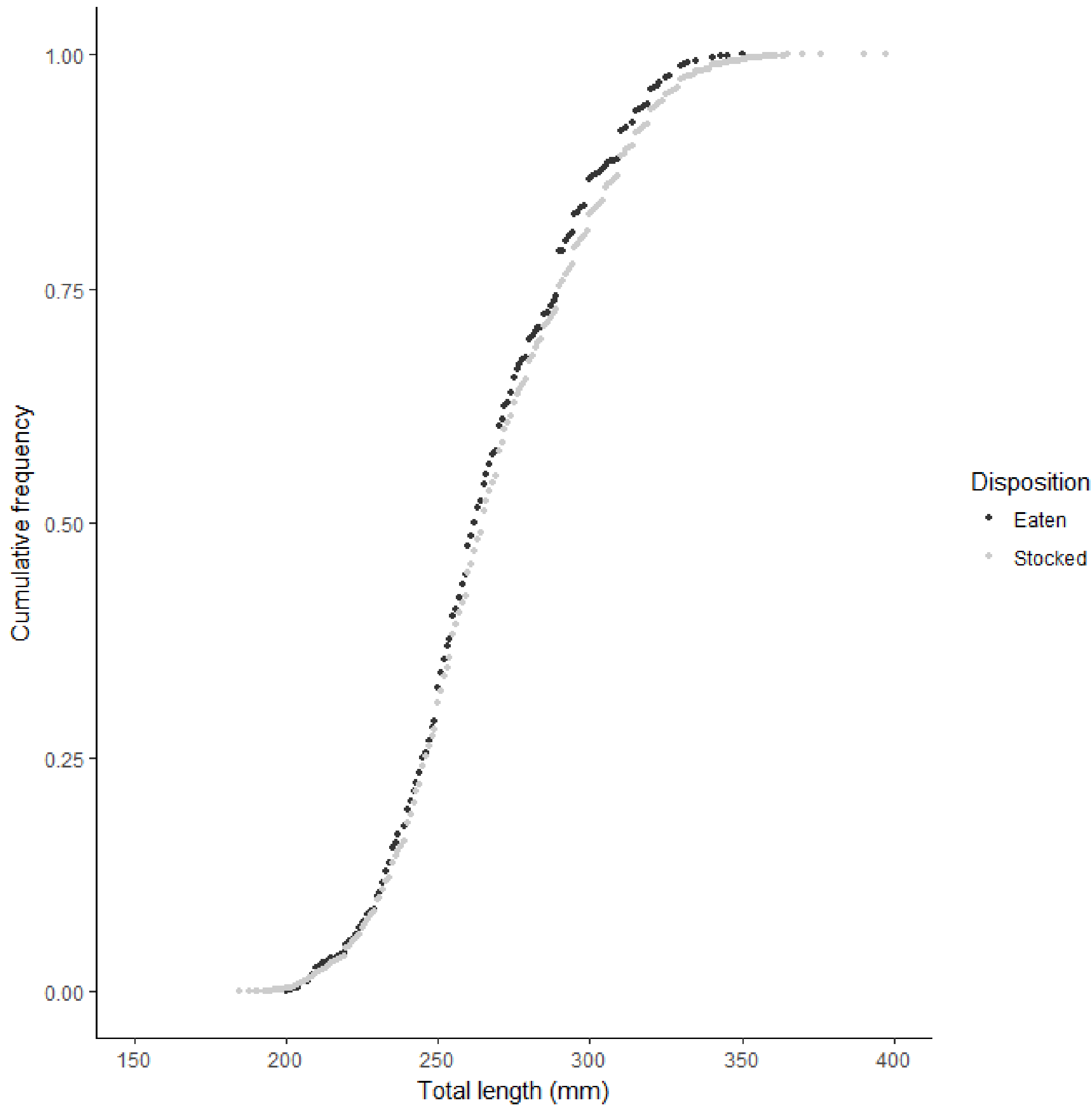

Figure 4. Cumulative length distributions (mm) for Rainbow Trout (*Oncorhynchus mykiss*) tagged and stocked into various Idaho waters between 2015-2017 and for those fish consumed by birds, indicated by tag recovery in bird use areas.

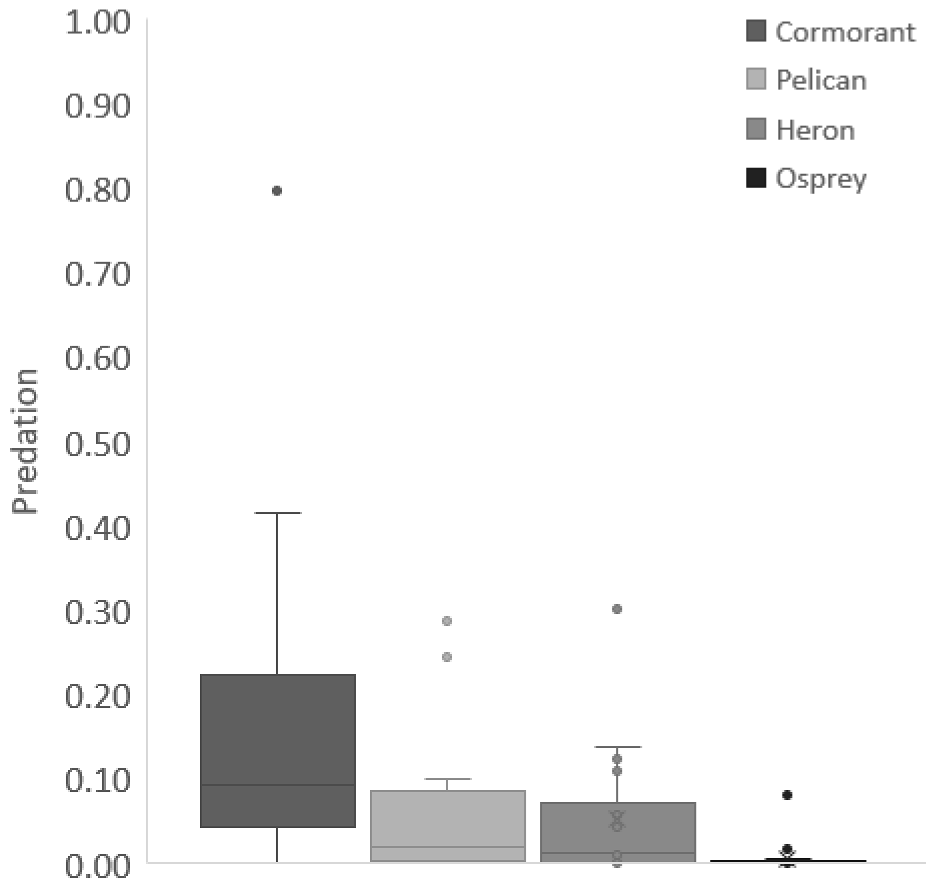

Figure 5. Species-specific predation rates on Rainbow Trout based on combination of PIT tag recovery locations (nest type) proportional bird abundance at each water to assign tags not found in specific nests.

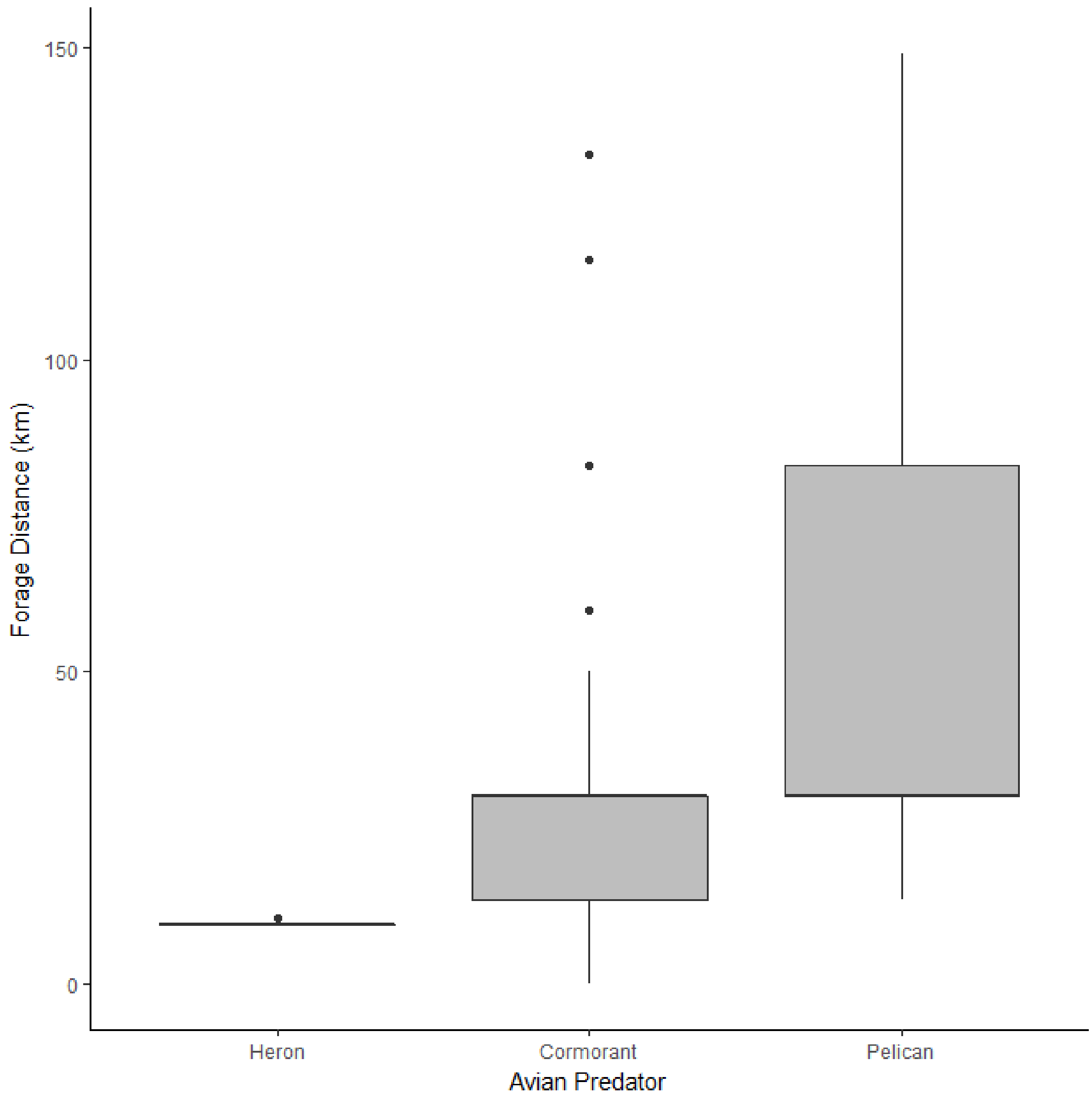

Figure 6. Foraging distances of the avian predators double-crested cormorants, Great Blue Herons, and American White Pelicans. Distances were based on locations between where PIT tagged fish were stocked and recovered from.

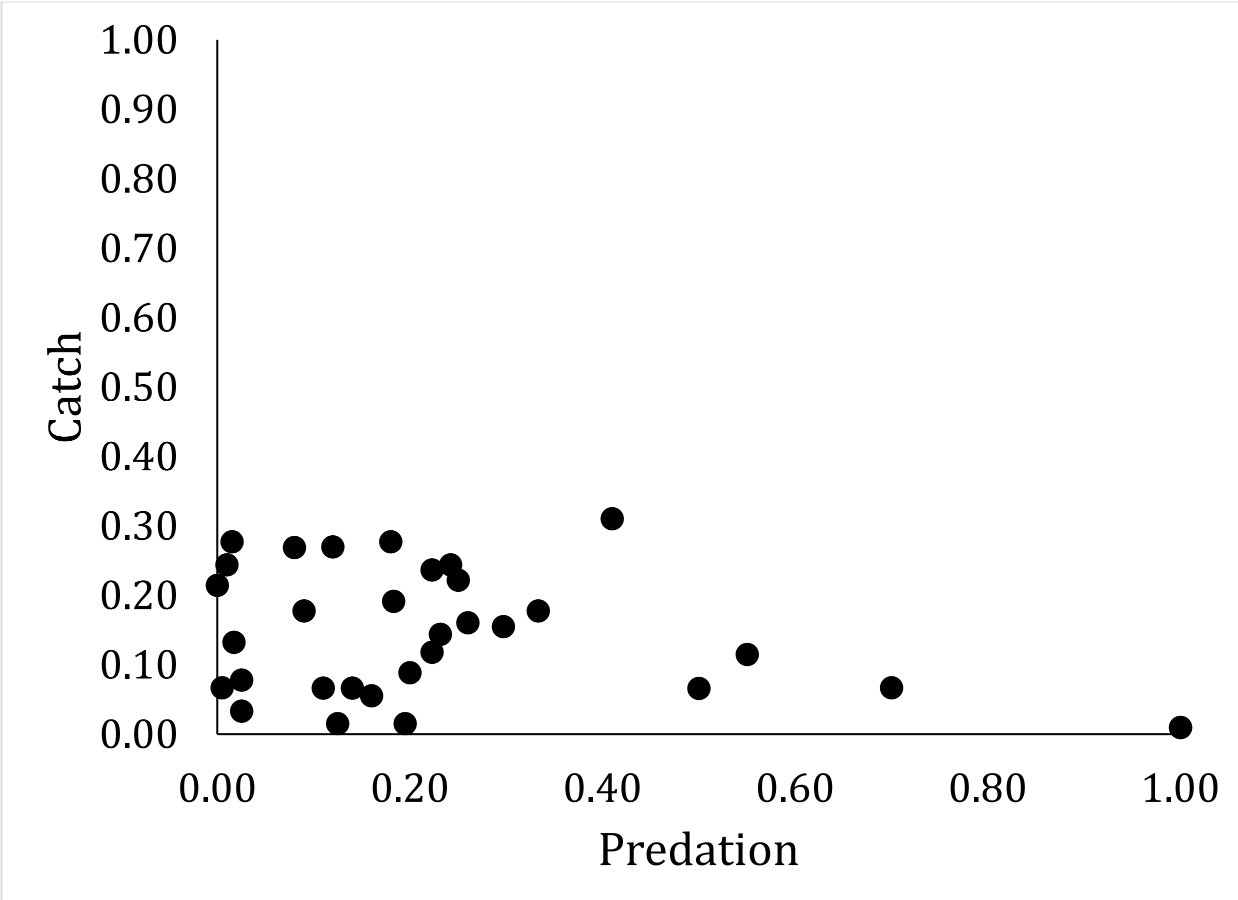

Figure 7. Scatterplot of the total angler use and total avian predation across all stocking events for both magnum (300mm) and standard (250mm) sized catchable Rainbow Trout.

ANNUAL PERFORMANCE REPORT
SUBPROJECT #2: AIR EXPOSURE AND FIGHT TIMES OF CATCH-AND-RELEASE FISHERIES IN IDAHO

State of: Idaho

Grant No.: F-73-R-39 (F17AF00851)

Project No.: 5

Title: Lake and Reservoir Research

Subproject #2: Air exposure and fight times of catch-and-release fisheries in Idaho

Contract Period: July 1, 2017 to June 30, 2018

ABSTRACT

Potential effects of air exposure and fight times on caught-and-released fish have been increasingly studied in recent years, yet little is known about how long anglers actually fight fish and expose them to air before releasing them. In the present study, air exposure and fight times were measured for anglers catching and releasing fish in popular fisheries in Idaho, and other relevant factors were also recorded such as fishing gear (fly or non-fly), occurrence of anglers photographing their catch, landing method (net or hand), and whether the fish was hooked deeply. A total of 432 steelhead were observed being landed, from which 395 fight times and 251 air exposure times were recorded. The longest interval of air exposure for all steelhead caught and released averaged 28.1 s (95% confidence interval [CI], 25.9-31.3 s), and the vast majority of anglers (88%) held steelhead out of water less than 60 s. Air exposure was not significantly different by gear type but was estimated to be 1.69 times longer if the steelhead angler took a photo of their catch; fly anglers were 58% more likely to photograph their catch than non-fly anglers. Fight time for steelhead averaged 130 s (95% CI, 119.3-140.7 s) and differed significantly by gear type, with fly anglers taking an estimated 1.54 times longer to land fish than non-fly anglers. Steelhead deep hooking rates were 0% for fly and bait/jig terminal tackle and 1% for lures. For Yellow Perch, Smallmouth Bass, and other warmwater species, air exposure averaged 27.7 s and fight time averaged 13.1 s. In the context of previous studies that have measured post-release mortality of caught and released salmonids, the effects of these fight and air exposure times and deep hooking rates in Idaho fisheries are likely negligible.

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INTRODUCTION

Potential effects of catch-and-release angling on fish mortality has been the subject of extensive research for decades (see reviews by Wydoski 1977 and Muoneke and Childress 1994), and concerns over sublethal physiological effects and general fish welfare are growing areas of research (e.g., Davie and Kopf 2006; Huntingford et al. 2006; Arlinghaus et al. 2007; Cooke and Sneddon 2007). Aspects of catch-and-release angling shown to affect post-release performance and survival include (among others) terminal tackle type (bait, lure, fly; Hunsaker 1970), fish handling (fight time and air exposure; Schisler and Bergerson 1996), and environmental conditions (e.g., water temperature; Klein 1966, capture depth; Rogers et al. 1986).

Of these factors affecting post-release performance of caught and released fish, the effect of air exposure and fight times has received the most attention in recent fisheries literature (reviewed in Cook et al. 2015) and popular culture (e.g., see www.keepemwet.org). Early studies conducted on hatchery Rainbow Trout *Oncorhynchus mykiss* suggested that air exposure and fight times greatly elevated mortality rates for fish (Ferguson and Tufts 1992; Schisler and Bergersen 1996). However, more recent research suggests that, for most salmonid species, unless released fish are air exposed for a prolonged period, and such air exposure is coupled with other stressors (i.e., high temperatures), long-term impacts are rarely life-threatening (Schreer et al. 2005; Donaldson et al. 2010, 2014; Raby et al. 2013; Gale et al. 2014). Nevertheless, some state fisheries management agencies have enacted regulations prohibiting air exposure of caught-and-released fish for some species. For example, in Washington state, it is unlawful to completely remove salmon, steelhead (anadromous *O. mykiss*), or Bull Trout *Salvelinus confluentus* from the water if the angler intends to release the fish. With regard to fight time, exhaustive exercise has been implicated as having negative consequences for caught-and-released fish (Ferguson and Tufts 1992; Schreer et al. 2001), but as with air exposure, such impacts typically do not materialize unless fight times or simulated exercise are extreme.

Considering the breadth of research on effects of various levels of air exposure and fight time on caught-and-released fish, surprisingly little information exists about how anglers actually play and release fish. In Idaho, covert angler observations at several lake and river fisheries revealed that trout anglers held fish out of water for an average for 26 s before releasing them, with only 4% of anglers holding fish out of water for >60 s (Lamansky and Meyer 2016). In the same study, fight time averaged 53 s. Another recent study showed that trout caught-and-released were exposed to air for an average of 19 s, while fight times averaged 40 s (Roth et al. 2018a). Although such fight and air exposure times are unlikely to impact mortality of caught-and-released trout, the extent to which these findings relate to other sport fisheries in Idaho is unknown.

In addition to air exposure and fight time, terminal gear type and anatomical hooking location also affect survival of caught-and-released fish (Bendock and Alexandersdottir 1993; Vincent-Lang et al. 1993; Lindsay et al. 2004; Nelson et al. 2005; Cowen et al. 2007), though other factors such as hook size and type, species, fish size, and capture conditions may also play a role because they directly influence hooking location. Bait fishing generally results in higher rates of deep hooking, which more often injures vital internal organs or gills, and frequently leads to higher rates of catch-and-release mortality (Wydoski 1977; Muoneke and Childress 1994). However, very little data exists to describe the deep hooking rates of migration salmon and steelhead that have essentially ceased feeding.

OBJECTIVES

1. Evaluate fight and air exposure times in popular steelhead and warmwater fisheries in Idaho.

2. Evaluate deep hooking rates by steelhead anglers using bait or other terminal tackle.

METHODS

The Clearwater River, its north and south tributary forks, the Little Salmon River, and the Salmon River are all popular steelhead fisheries in Idaho. In these waters, anglers may only harvest steelhead with a clipped adipose fin, indicating they are of hatchery origin; otherwise, the fish must be released. The only gear restriction on these waters is that only single-pointed barbless hooks are allowed when fishing for steelhead on the South Fork Clearwater River.

We observed anglers fishing for steelhead in all of these fisheries. At most sites, observations were made covertly because we assumed that angler behavior might be affected by the close presence of state agency staff. However, some observations on the South Fork Clearwater River were overt, i.e., they were collected opportunistically during an unrelated program involving volunteer steelhead broodstock collection by anglers. During these instances, anglers were fishing for their own personal enjoyment and potential harvest, but also assisting with occasional broodfish collection. The overt observations used in our analysis did not include any of the fish collected for broodstock. Covert observations of anglers were conducted with binoculars from inconspicuous locations, or directly by observers posing as anglers.

When a fish was hooked, we used a stopwatch or smartphone timer to measure the duration (s) it took from initial hookup to landing of the fish. At times, the initial hookup was not observed so fight times were not recorded for those fish. Once landed, we timed how long the fish was exposed to air before being released. Occasionally, fish were put back in the water and then re-exposed to air one or two times. These occurrences accounted for 14% of observations and 7% of all air exposure and did not affect model variables included in our results or our conclusions. Thus, for the purposes of analysis, we used only the longest air exposure interval. During each fish landing event, we also noted various associated factors that might influence fight times and air exposure, including the type of fishing gear used (fly, lure, bait), the method used to land the fish (net, hand), and whether a photo of the fish was taken.

Non-fly fishing tackle such as beads, yarn, or bait drifted with or without a bobber, was all fished very similarly and was not always distinguishable at a distance by anything other than the rod type and technique used, so they were combined into a non-fly gear category. Even though lures are fished distinctively, they were also included in the non-fly gear category because rods used to fish lures are very similar to those used with other non-fly gear and would be expected to have a similar effect on fight times. Thus, for fishing gear type used, we report either fly fishing or non-fly fishing when testing effects on fight time and air exposure.

Water bodies were considered separately in this analysis due to differences that could contribute to variation in the data. For example, on the North Fork Clearwater River, many anglers fish either from the Ahsahka Bridge or from the wall below Dworshak Dam, targeting fish returning to Dworshak Hatchery. At both locations, anglers are fishing 10-20 m above the surface of the water, which (1) precludes fly anglers from fishing those locations, and (2) greatly extends the fight time because the angler is required to walk across the bridge or wall and climb down a series of stairs to reach the water and land a hooked fish. Thus, observations collected at the bridge and dam were treated as a site separate from the remaining observations on the North Fork Clearwater River and other sites. In the interest of collecting independent observations, we did not knowingly collect more than one observation per angler each day.

Hooking location, collected only on the South Fork Clearwater River, was recorded for overt observations but could not be determined for covert observations; we assumed that anglers could not influence their hooking location based on their awareness of a nearby state agency staff member. Hooking location was recorded as deep (i.e., either in the gills or more deeply hooked), mouth (i.e., in the corner of the mouth or anything inside the mouth but not deep hooked), or foul hooked (i.e., on the outside of the body). For hooking location, gear was categorized into either bait/jigs, lures, or flies because of suspected differences among terminal tackle types (Wydoski 1977; Muoneke and Childress 1994).

The air exposure and fight time data represented time-to-event data that conformed to an exponential distribution, so we used accelerated failure time models to evaluate the factors affecting each response variable (Therneau and Grambsch 2000; Therneau 2015); air exposure and fight time were modeled separately. Accelerated failure time models designate a family of models that can be generalized to include covariates on the air exposure or fight time function (Kalbfleisch and Prentice 2002). Candidate models included water, gear type, photo taken (yes/no), observer status (covert/overt), landing method (net/hand), and whether or not the fish was harvested (only fight time models) as factors potentially affecting air exposure or fight time. We considered photo taking and harvest separately in candidate fight time models as surrogates for fish size, under the rationale that the larger a fish is, the more likely it is to be photographed or harvested. Candidate models were evaluated using Akaike's Information Criterion corrected for small sample size (AIC_c; Burnham and Anderson 2002). Once exponentiated, coefficients in the accelerated failure time models are multiplicative. For instance, if the coefficient for when a photo was taken was 1.5 for a given air exposure model, this means air exposure was 1.5 times longer for photographed fish than for those not photographed. We used leave-one-out cross-validation to evaluate the predictive performance of each model and reported mean error for each candidate model. Program R was used for all data analyses (R Development Core Team 2011).

Data on caught-and-released warmwater species was covertly collected at CJ Strike, Brownlee, Cascade reservoirs. Yellow Perch *Perca flavescens* and Smallmouth Bass *Micropterus dolomieu* fight times and air exposures were considered separately when the observer was able to distinguish the species. Otherwise, these species, as well as Largemouth Bass *M. salmoides*, Black Crappie *Pomoxis nigromaculatus* and White Crappie *P. annularis*, and Bluegill *Lepomis macrochirus* were combined into a warmwater category for data reporting. Summary statistics of fight and air exposure times for warmwater species are reported but not modeled as steelhead data were.

RESULTS

From September 2016 to April 2017, we observed a total of 432 steelhead caught, of which 293 were released. We recorded 395 fight times and 251 air exposure times. The longest interval of air exposure of caught-and-released steelhead averaged 28.1 s (95% confidence interval [CI], 25.9-31.3 s), and the vast majority of anglers (88%) held fish out of water less than

60 s. Only 14% (mean 13.3 s) and 3% (7.6 s) of anglers held fish out of water for two and three separate intervals. The average fight time was 130 s (95% CI, 119.3-140.7 s; Table 1).

The top three air exposure models included either photo taking, gear type and photo taking, or both terms including an interaction (Table 2). Of these factors, photo taking had the strongest effect, with air exposure time estimated at 1.69 times (95% CI 1.26-2.27) longer if the fish was photographed (Table 3). This effect appeared stronger for anglers using non-fly gear, as inferred by the inclusion of a gear × photo interaction term in one of the top three models. However, the gear and gear × photo interaction terms both had coefficients with 95% CIs that overlapped 1, indicating their effects were not significant. Nevertheless, fly anglers photographed their catch significantly more often (38%) than non-fly anglers (24%) ($\chi^2 = 5.19$, df = 1, $P = 0.02$). Anglers on the South Fork Clearwater River, the only water with covert and overt observations, held fish out of the water an estimated 1.57 times (95% CI 1.09-2.28) longer when they did not know they were being observed by agency staff.

Candidate fight time models that included gear type, water body, landing method, and either harvest or photo best supported our data (Table 4). However, the best model included just gear type and waterbody as predictors of fight time (Table 5). When fly-gear was used, anglers fought fish for an estimated 1.54 times longer than when non-fly gear was used, after accounting for differences in waterbody (Table 5). Although waterbody was included in the top model, coefficients for each waterbody had CIs that overlapped 1, suggesting no significant difference from the reference waterbody. Fight times measured on the South Fork Clearwater River were also affected by observer status (covert/overt), but in the opposite direction as air exposure. Fight times measured covertly were 0.61 (95% CI 0.43-0.85) times as long as those collected overtly.

Hooking location was determined for 188 fish, comprised of 49 caught by bait/jig, 99 caught by lure, and 40 caught by fly fishing. Deep hooking rates were 0% for bait/jigs, 1% for lures, and 0% for flies. Foul hooking rates were much higher for lures (40%) than for flies (7.5%) or bait/jigs (4%).

Observations of Yellow Perch, Smallmouth bass, and other warmwater species consisted of 110 air exposures averaging 27.7 s and 92 fight times averaging 13.1 s (Table 6).

DISCUSSION

Our finding that most Idaho anglers exposed caught-and-released steelhead to < 30 s of air concurs with previous studies of trout anglers, which also showed that fish on average were exposed to < 30 s of air before being released (Lamansky and Meyer 2016; Roth et al. 2018). Our observations of air exposures for warmwater species also concur with those studies. In contrast, our average steelhead fight time of 120 s was 2-3 fold longer than previous trout studies reporting mean fight times of 53 s (Lamansky and Meyer 2016) and 40 s (Roth et al. 2018). This disparity was most likely related to the size of the fish being caught (i.e., adult steelhead trout are an order of magnitude heavier than most resident adult trout), though fight times for Cutthroat Trout *O. clarkii* in Yellowstone National Park (102 s; Schill et al. 1986) were similar to steelhead fight times in the present study. Such short air exposure and fight time intervals suggest that most anglers are inherently conscientious of the negative impact that prolonged air exposure or exhaustive exercise can have on caught and released fish, although some anglers may also operate under the assumption that prolonging fight time increases the likelihood that a hooked fish will escape landing.

Though survival of caught-and-released steelhead was not estimated in the present study, results of nearly all prior salmonid studies suggest that air exposure and fight times reported herein would result in little to no mortality for trout and salmon in freshwater (Table 7). One notable exception includes a lab study in which hatchery Rainbow Trout were chased in a hatchery raceway for 10 minutes and exposed to air for 30 or 60 seconds to simulate a catch-and-release event (Ferguson and Tufts 1992). Though significant mortality (38-72%) of air-exposed fish was observed, even the fish exposed to no air (but still exercised) experienced 12% mortality, and the 10 minutes of exhaustive exercise was not representative of fight times observed in real-world angling scenarios (Schill et al. 1986; Lamansky and Meyer 2016; Roth et al. 2018). Furthermore, the test fish were cannulated and sampled for blood up to five times throughout the experiment, which may have exacerbated treatment effects. In a contrasting example of hardiness to handling stress, 75% of Pink Salmon *O. gorbuscha* exposed to 16 minutes of air and 100% of them exposed to eight minutes of air survived and spawned successfully (Raby et al. 2013). In the context of these and many other studies, the fight and air exposure times observed in the present study likely have negligible population-level effects in Idaho steelhead fisheries.

While hooking mortality was not estimated in the present study, deep hooking rates were lower (≤ 1%) than those reported in previous anadromous salmonid studies. For example, Chinook Salmon *O. tshawytscha* in the Willamette River experienced a 13% deep hooking rate (esophagus-stomach and gills), including rates of 15% for bait anglers and 3% for anglers using spinners (Lindsay et al. 2004). Chinook Salmon in the Yakima River experienced an 8% deep hooking rate, with 99% of anglers fishing with bait (Fritts et al. 2016). The discrepancy between these previous studies and our results may be caused by disparate foraging behavior between species during upstream spawning migrations. Indeed, adult steelhead are generally not believed to feed in freshwater before they spawn (Penney and Moffitt 2013) whereas Chinook Salmon have been shown to forage occasionally during upstream migration (Garner et al. 2009). Because deep hooking by anglers is most strongly associated with fish attempting to swallow bait attached to a hook (Wydoski 1977; Muoneke and Childress 1994), fish that actively attempt to swallow food are inherently more likely to be deep hooked by bait anglers.

Fly fishing is usually regarded as resulting in higher catch-and-release survival than bait or lure fishing due to lower deep hooking rates (Hunsaker et al. 1970) and perceived reduced handling stress. In the present study, fly anglers took considerably longer to land steelhead, exposed fish to similar air durations, were more likely to hold a fish out of water for picture taking, and did not deep hook fish less often compared to other terminal tackle. These results suggest that fish caught and released by fly anglers in Idaho steelhead fisheries may experience more stressful handling conditions (primarily in the form of extended fight time) than fish caught by non-fly anglers. Extended fight time for fly anglers should not be surprising given that fishing rods used with bait, bobbers, and lures typically have much greater resistance and strength than fly rods used in similar fisheries. However, it should be noted that these differences, though statistically significant, are likely not biologically meaningful in the context of post-release mortality rates or population-level impacts.

Our study confirms the importance of covertly collecting angler observational data so as not to bias their behavior. As suspected, anglers held fish out of the water for less time when the data were overtly collected, presumably because they were aware that their behavior was observable to agency staff. Until the study by Lamansky and Meyer (2016), previous studies that reported angler fight and air exposure times had only included anglers participating in a particular study (e.g., Landsman et al. 2011). Surprisingly, covertly collected fight times were shorter than overt observations, perhaps because most overt observations were of anglers were cooperating with biologists for hatchery broodstock collection. This situation may have placed greater importance on carefully landing the fish, making the anglers more conscious not to hurry the capture, thereby slightly prolonging the process. A limitation of this interpretation is that we assume bias due to the presence of an observer affects a typical recreational angler similar to an angler who is voluntarily assisting biologists collecting broodstock.

The results of this study add to prior work (Lamansky and Meyer 2016; Roth et al. 2018) suggesting that anglers in many Idaho fisheries already minimize stress on caught-and-released fish by fighting fish quickly and minimizing air exposure times. Even when anglers photographed their steelhead catch, air exposure times observed were not consistent with values expected to result in reduced survival. We therefore see no benefit from imposing air exposure fishing regulations in these types of fisheries. Angling interest is already declining nationwide (Maillet et al. 2017), and formally banning the practice of photographing or admiring a fish out of water before releasing it may negatively impact angler satisfaction, and ultimately angler recruitment, especially considering the rise in smartphone camera and social media use. No science has demonstrated that air exposure times and handling methods typical in catch-and-release fisheries have negative impacts on fish populations, and efforts to regulate air exposure distract attention from more important and legitimate negative impacts to managed fisheries, such as overharvest, habitat alterations, invasive species, and climate change.

RECOMMENDATIONS

1. Because Idaho anglers already minimize fight and air exposure times, implementing angling regulations for catch-and-release fishing is not warranted.

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LITERATURE CITED

Arlinghaus, R., S. J. Cooke, A. Schwab, and I. G. Cowx. 2007. Fish welfare: a challenge to the feelings-based approach, with implications for recreational fishing. Fish and Fisheries 8:57–71.

Behnke, R. J. 1987. Catch-and-release: the last word. Pages 291-299 *in* R. A. Barnhart and T. D. Roelofs, editors. Proceedings of catch-and-release fishing, a decade of experience. Humboldt State University, Arcata, California.

Bendock, T., and M. Alexandersdottir. 1993. Hooking mortality of Chinook Salmon released in the Kenai River, Alaska. North American Journal of Fisheries Management 13:540–549.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer, Verlag, New York.

Cook, K. V., R. J. Lennox, S. G. Hinch, and S. J. Cooke. 2015. Fish Out of Water: How much air is too much? Fisheries 40:452–461.

Cooke, S. J., and L. U. Sneddon. 2007. Animal welfare perspectives on recreational angling. Applied Animal Behaviour Science 104:176–198.

Cowen, L., N. Trouton, and R. E. Bailey. 2007. Effects of angling on Chinook Salmon for the Nicola River, British Columbia, 1996–2002. North American Journal of Fisheries Management 27:256–267.

Davie, P. S., and R. K. Kopf. 2006. Physiology, behavior and welfare of fish during recreational fishing and after release. New Zealand Veterinary Journal 54:161–172.

Donaldson, M. R., T. D. Clark, S. G. Hinch, S. J. Cooke, D. A. Patterson, M. K. Gale, P. B. Frappell, and A. P. Farrell. 2010. Physiological responses of free-swimming adult Coho Salmon to simulated predator and fisheries encounters. Physiological and Biochemical Zoology 83:973–983.

Donaldson, M. R., S. G. Hinch, K. M. Jeffries, D. A. Patterson, S. J. Cooke, A. P. Farrell, and K. M. Miller. 2014. Species- and sex-specific responses and recovery of wild, mature pacific salmon to an exhaustive exercise and air exposure stressor. Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology 173:7–16.

Ferguson, R. A., and B. L. Tufts. 1992. Physiological effects of brief air exposure in exhaustively exercised Rainbow Trout (*Oncorhynchus mykiss*): implications for “catch and release” fisheries. Canadian Journal of Fisheries and Aquatic Sciences 49:1157–1162.

Fritts, A. L., G. M. Temple, C. Lillquist, and D. Rawding. 2016. Post-release survival of Yakima River spring Chinook Salmon associated with a mark-selective fishery. Washington Department of Fish and Wildlife, Olympia, WA.

Gale, M. K., S. G. Hinch, S. J. Cooke, M. R. Donaldson, E. J. Eliason, K. M. Jeffries, E. G. Martins, and D. A. Patterson. 2014. Observable impairments predict mortality of captured and released Sockeye Salmon at various temperatures. Conservation Physiology 2:1-15.

Garner, S. R., J. W. Heath, and B. D. Neff. 2009. Egg consumption in mature Pacific salmon (*Oncorhynchus spp*.). Canadian Journal of Fisheries and Aquatic Sciences 66:1546-1553.

Hunsaker, D., L. F. Marnell, and F. P. Sharpe. 1970. Hooking mortality of Yellowstone Cutthroat Trout. The Progressive Fish Culturist 32: 231-235.

Huntingford, F. A., C. Adams, V. A. Braithwaite, S. Kadri, T. G. Pottinger, P. Sandøe, and J. F. Turnbull. 2006. Current issues in fish welfare. Journal of Fish Biology 68:332–372.

Kalbfleisch, J. D., and R. L. Prentice. 2002. The statistical analysis of failure time data, Second Edition. John Wiley & Sons, Inc., Hoboken, New Jersey.

Klein, W. D. 1966. Mortality of trout caught on artificial lures and released by fishermen. Transactions of the American Fisheries Society 95: 326–328.

Lamansky, J. A., and K. A. Meyer. 2016. Air exposure time of trout released by anglers during catch and release. North American Journal of Fisheries Management 36:1018–1023.

Landsman, S. J., H. J. Wachelka, C. D. Suski, S. J. Cooke. 2011. Evaluation of the physiology, behavior, and survival of adult Muskellunge (*Esox masquinongy*) captured and released by specialized anglers. Fisheries Research 110:377-386.

Lindsay, R. B., R. K. Schroeder, K. R. Kenaston, R. N. Toman, and M. A. Buckman. 2004. Hooking mortality by anatomical location and its use in estimating mortality of spring Chinook Salmon caught and released in a river sport fishery. North American Journal of Fisheries Management 24:367–378.

Maillet, E., R. Aiken, and M. Fuller. 2017. The demographic and economic trends of trout anglers: implications for recruitment, retention, and restoration and habitat management support. Pages 15-22 *in* R.F Carline and C. LoSapio, editors. Wild Trout II: Science, politics, and wild trout management: who's driving and where are we going? Wild Trout Symposium, West Yellowstone, Montana.

Muoneke, M. I., and W. M. Childress. 1994. Hooking mortality: A review for recreational fisheries. Reviews in Fisheries Science 2:123–156.

Nelson, T. C., M. L. Rosenau, and N. T. Johnston. 2005. Behavior and survival of wild and hatchery-origin winter steelhead spawners caught and released in a recreational fishery. North American Journal of Fisheries Management 25:931–943.

Penney, Z. L., and C. M. Moffitt. 2014. Histological assessment of organs in sexually mature and post-spawning Steelhead Trout and insights into iteroparity. Reviews in fish biology and fisheries 24:781-801.

R Development Core Team. 2011. R: A language and environment for statistical computing. The R Foundation for Statistical Computing, Vienna, Austria.

Raby, G. D., S. J. Cooke, K. V. Cook, S. H. McConnachie, M. R. Donaldson, S. G. Hinch, C. K. Whitney, S. M. Drenner, D. A. Patterson, T. D. Clark, and A. P. Farrell. 2013. Resilience of Pink Salmon and Chum Salmon to simulated fisheries capture stress incurred upon arrival at spawning grounds. Transactions of the American Fisheries Society 142:524–539

Rogers, S. G., Langston, H. T., and T. E. Targett. 1986. Anatomical trauma to sponge-coral reef fishes captured by trawling and angling. United States National Marine Fisheries Service Bulletin 84:697–704.

Roth, C. J., D. J. Schill, and M. C. Quist. 2018a. Fight and air exposure times of caught and released salmonids from the South Fork Snake River. Fisheries Research 201:38-43.

Roth, C. J., D. J. Schill, and M. C. Quist. 2018b. Effects of air exposure in summer on the survival of caught-and-released salmonids. North American Journal of Fisheries Management 38:886-895.

Schill, D. J., and R. L. Scarpella. 1997. Barbed hook restrictions in catch-and-release trout fisheries: a social issue. North American Journal of Fisheries Management 17:873-881.

Schill, D. J., J. S. Griffith, and R. E. Gresswell. 1986. Hooking mortality of Cutthroat Trout in a catch-and-release segment of the Yellowstone River, Yellowstone National Park. North American Journal of Fisheries Management 6:226-232.

Schisler, G. J., and E. P. Bergersen. 1996. Postrelease hooking mortality of Rainbow Trout caught on scented artificial baits. North American Journal of Fisheries Management 16:570–578.

Schreer, J. F., S. J. Cooke, and R. S. McKinley. 2001. Cardiac response to variable forced exercise at different temperatures: an angling simulation for Smallmouth Bass. Transactions of the American Fisheries Society 130:783–795.

Schreer, J. F., D. M. Resch, M. L. Gately, and S. J. Cooke. 2005. Swimming performance of Brook Trout after simulated catch-and-release angling: looking for air exposure thresholds. North American Journal of Fisheries Management 25:1513–1517.

Therneau, T. M., and P. M. Grambsch. 2000. Modeling survival data: extending the cox model. Springer, New York.

Therneau, T. M. 2015. A package for survival analysis in S. version 2.38, https://CRAN.R-project.org/package=survival.

Vincent-Lang, D. 1993. Relative survival of unmarked and fin-clipped Coho Salmon from Bear Lake, Alaska. The Progressive Fish-Culturist 55:141.

Wydoski, R. S. 1977. Relation of hooking mortality and sublethal hooking stress to quality fishery management. Pages 43–87 *in* R. A. Barnhart and T. D. Roelofs, editors. Catch-and-release fishing as a management tool. California Cooperative Fishery Research Unit, Humboldt State University, Arcata, CA.

TABLES

Table 1. Summary statistics of steelhead air exposure and fight times by anglers in five Idaho rivers, including sample size (N), mean, range, and 95% confidence intervals (CI) for gear types (fly and non-fly), observer status (covert and overt), and whether the angler photographed their catch.

	N	Mean	Range	95% CI
Air Exposure (s)				
Total	251	28.1	0-185	3.2
[1]Fly	47	22.4	0-89	5.9
[1]Non-fly	204	29.4	0-185	3.7
[2]Covert	50	36.4	0-109	8.0
[2]Overt	102	23.2	2-60	2.8
Photo	75	37.9	0-129	6.4
No photo	176	24.1	0-185	3.5
Fight time (s)				
Total	395	130.4	5-900	10.7
[1]Fly	70	169.8	13-765	31.1
[1]Non-fly	234	93.2	5-575	9.9
[2]Covert	47	74.16	5-494	26.2
[2]Overt	171	113.2	6-765	15.3
Photo	70	140	17-519	24.5
No photo	234	101.9	5-765	12.1

[1]Air exposure and fight times for non-fly gear include fish caught from the South Fork Clearwater and Little Salmon rivers only.
[2]Air exposure and fight times for covert and overt observations include the South Fork Clearwater River only.

Table 2. Comparison of accelerated failure time models that estimate air exposure times of steelhead caught and released in five Idaho rivers. Degrees of freedom (df), Akaike's Information Criteria (AIC_c), change in AIC_c value (ΔAIC_c), AIC_c weights, and mean error from leave-one-out cross-validation were used to select top models from a set of candidate models. Variables considered in the models included fishing gear type, whether the angler photographed their catch, waterbody, and whether the observer was covert.

Model	df	AIC_c	ΔAIC_c	weight	Mean error
Photo	2	1867.2	0.00	0.291	16.74
Photo + Gear	3	1867.2	0.03	0.286	16.48
Photo + Gear + Photo × Gear	4	1867.5	0.37	0.242	16.56
Water	6	1868.3	1.15	0.164	16.27
Photo + Gear + Photo × Gear + Water + Water × Gear	14	1873.1	5.98	0.015	15.98
Gear	2	1878.3	11.11	0.001	17.07
Intercept	1	1878.4	13.59	0.001	17.21

Table 3. Coefficients and 95% confidence intervals (CI) for the most highly supported accelerated failure time models used to estimate air exposure times for steelhead caught and released in five Idaho rivers. Fly gear and no photo taken are the reference categories for gear and photo taking. Coefficient estimates are interpreted as multiplicative of each parameter relative to the reference category.

Coefficient	Estimate	95% CI
Air Exposure ~ Photo		
(Intercept)	22.66	19.36-26.53
Photo Yes	1.69	1.26-2.27
Air Exposure ~ Photo + Gear		
(Intercept)	18.61	13.74-25.19
Gear (non-fly)	1.28	0.92-1.77
Photo (yes)	1.69	1.26-2.26
Air Exposure ~ Photo + Gear + Photo × Gear		
(Intercept)	20.92	14.54-30.10
Gear (non-fly)	1.10	0.74-1.65
Photo (yes)	1.16	0.63-2.14
Gear (non-fly) × Photo (yes)	1.61	0.80-3.23

Table 4. Comparison of accelerated failure time models used to estimate fight times of steelhead caught and released in five Idaho rivers. Degrees of freedom (df), Akaike's Information Criteria (AIC_c), change in AIC_c value (ΔAIC_c), AIC_c weights, and mean error from leave-one-out cross-validation were used to select top models from a set of candidate models. Variables considered in the models included fishing gear type, whether the angler photographed their catch, waterbody, whether the observer was covert, landing method, and whether the fish was harvested.

Model	df	AIC_c	ΔAIC_c	weight	Mean error
Gear + Water	6	4432.6	0.00	0.511	65.16
Gear + Water + Landing method	7	4434.5	1.95	0.193	65.26
Gear + Water + Landing method + Harvest	8	4434.7	2.16	0.174	65.01
Gear + Water + Landing method + Photo	8	4435.4	2.87	0.122	65.24
Water	5	4464.2	31.63	0	70.13
Gear	2	4485.9	53.38	0	77.89
Photo	2	4487.4	54.79	0	77.28
Intercept	1	4490.5	57.92	0	78.24
Landing method	2	4492.2	59.67	0	78.36

Table 5. Coefficients and 95% confidence intervals (CI) for the most highly supported accelerated failure time model used to estimate fight times for steelhead caught and released in five Idaho rivers. Coefficient estimates are interpreted as multiplicative of each parameter relative to the reference category. Fly gear and the Clearwater River were the reference categories for gear and water. (LS=Little Salmon River, NFBD=North Fork Clearwater River dam wall and bridge fisheries, NFCLW= North Fork Clearwater River, SFCLW=South Fork Clearwater River). No fight times were measured for fight caught in the Salmon River, so this water is not included in the model selection.

	Coefficient	95% CI
Fight ~ Gear + Water		
Intercept	251.59	119.23-530.91
Gear (Non-fly)	0.46	0.35-0.60
Water (LS)	0.55	0.26-1.18
Water (NFBD)	1.69	0.82-3.49
Water (NFCLW)	1.55	0.71-3.40
Water (SFCLW)	0.70	0.35-1.43

Table 6. Summary statistics of warmwater species air exposure and fight times by anglers at three Idaho reservoirs, including sample size (N), mean, range, and 95% confidence intervals (CI).

	N	Mean	Range	95% CI
Air exposure (s)				
Yellow Perch	36	25.5	2-78	6.2
Smallmouth Bass	55	27.9	6-141	5.7
Warmwater spp.	19	31.1	4-116	10.7
Fight time (s)				
Yellow Perch	58	12.8	1-39	1.9
Smallmouth Bass	18	12.4	2-40	4.7
Warmwater spp.	16	14.7	3-63	7.3

Table 7. Summary of studies testing exercise (fight times), air exposure, water temperatures, and sample sizes (N) for various salmonids, with resulting mortality rates.

Species	Temp-erature (°C)	Fight time (s)	Air exposure (s)	Mortality (%)	N	Reference
Rainbow Trout	8-10	600	0	12	6	Ferguson and Tufts 1992
	8-10	600	30	38	8	
	8-10	600	60	72	7	
Brook Trout	10	30	0	0	12	Schreer et al. 2005
	10	30	30	0	12	
	10	30	60	0	12	
	10	30	120	0	12	
Coho Salmon	8	180	0	0	12	Donaldson et al. 2010
		180	60	0	13	
Pink Salmon	13.2	180	0	0	29	Raby et al. 2013
	13.2	180	60	0	29	
	13.2	10	0	0	29	
	11.9	0	60	0	20	
	11.9	0	120	0	20	
	11.9	0	240	0	20	
	11.9	0	480	0	20	
	11.9	0	960	25	20	
Chum Salmon *O. keta*	11.8	180	0	0	29	
	11.8	180	60	0	29	
	11.8	10	0	0	29	
Cutthroat Trout *O. clarkii*	10-13	17	0	32	110	Roth et al. 2018[1]
		17	30	43	110	
		17	60	39	108	
Bull Trout *Salvelinus confluentus*	9-14	16	0	48	92	
		14	30	41	94	
		14	60	36	92	
Rainbow Trout		16	0	37	103	
		13	30	42	106	

		15	60	49	113	
Pink Salmon	11-12	180	60	0	44	Donaldson et al. 2014
Sockeye Salmon	11-12	180	60	0	66	

[1]Mortality estimates from Roth et al. 2018 were calculated from relative survival estimates reported for captured-marked-and recaptured fish.

ANNUAL PERFORMANCE REPORT
SUBPROJECT #3: PULSE FREQUENCY (HZ) EFFECTS ON CAPTURE EFFICIENCY AND INJURY OF TROUT SAMPLED WITH BACKPACK ELECTROFISHING

State of: Idaho

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Title: Lake and Reservoir Research

Subproject #2: Pulse frequency (Hz) effects on capture efficiency and injury of trout sampled with backpack electrofishing

Contract Period: June 30, 2017 to July 1, 2018

ABSTRACT

Recommendations for the use of low pulse frequencies when backpack electrofishing to reduce injury to fish have not evaluated how doing so affects fish capture efficiency (CE). We compared capture efficiencies (measured by the recapture of marked fish) and spinal injury rates of trout captured in reaches sampled with either 30 or 60 Hz for a given power output. For sites sampled with 30 Hz, CE averaged 0.63 on pass 1, 0.36 on pass 2, 0.13 on pass 3, 0.12 on pass 4, and 0.83 overall. For sites sampled with 60 Hz, CE averaged 0.77 on pass 1, 0.46 on pass 2, 0.25 on pass 3, 0.05 on pass 4, and 0.93 overall. Bias averaged 0.16 for the reaches sampled with 30 Hz and 0.07 for reaches sampled with 60 Hz. X-ray images revealed vertebral compressions and/or misalignments for 9% of fish captured with 30Hz (n = 149) compared to 11% of fish captured with 60 Hz (n = 140); no such injuries were observed for fish captured via angling (n = 75). No fractured vertebrae were observed in any of the x-ray fish from any of the treatments. Results suggest that in small trout streams where fish size is generally <300 mm total length, using 60 Hz for sampling purposes appears to have the benefit of greater capture efficiency resulting in better trout abundance estimates, with little to no increase in spinal injury.

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INTRODUCTION

Backpack electrofishing is one of the most commonly used methods of assessing fish composition and abundance in wadeable streams. Biologists often make multiple electrofishing passes through a particular study reach and use the catch data to obtain a maximum likelihood estimate of abundance and capture efficiency (Moran 1951; Zippin 1956, 1958), known as the removal depletion method. It has long been recognized that the removal depletion method tends to (1) produce declining fish capture efficiency (CE) with successive passes, and (2) overestimates true CE within each pass, both of which lead to underestimating true fish abundance (e.g., Mahon et al. 1979; Riley and Fausch 1992). Because the magnitude of the bias in estimates of abundance is directly related to the magnitude of the bias in estimates of CE, maximizing CE inherently improves estimates of fish abundance.

Numerous factors can affect CE, such as fish size (larger fish are easier to catch), channel complexity (more complexity reduces CE), type of electricity used (AC, DC, or pulsed DC), electrical intensity of shocker settings, and experience of crew conducting the survey. One factor that has been little studied but that theoretically should be directly linked to CE is pulse frequency when using pulsed DC electrical output. At very low pulse frequencies, fish can more easily escape the electrical current, whereas at higher pulse frequencies, fish tend to be more intensely immobilized and thus more easily captured. However, pulse frequency is also the most important factor influencing fish injury during electrofishing (Reynolds and Kolz 2012), and some agencies have established maximum pulse frequency thresholds for electrofishing to protect fish. For example, the state of Montana requires the use of 30 Hz or less "in any waters containing self-sustaining salmonid populations" (Montana Fish, Wildlife and Parks 2002, http://fwp.mt.gov/fwpDoc.html?id=9001). Similarly, NOAA recommends 30 Hz and sets a maximum level of 70 Hz for sampling streams "containing salmonids listed under the Endangered Species Act" (NOAA 2000, http://www.fwspubs.org/doi/suppl/10.3996/112016-JFWM-083/suppl_file/fwma-08-01-30_reference+s02.pdf); NOAA also sets thresholds on voltage (based on ambient conductivity levels) and restricts all electrofishing above certain water temperatures and conductivities. The fact that restrictions on pulse frequency may affect CE – and therefore the accuracy of the abundance estimate – is apparently overlooked by these guidelines or considered immaterial compared to the importance of protecting fish from injury during electrofishing surveys.

OBJECTIVES

1. Evaluate the effect of low (30 Hz) and high (60 Hz) pulse frequencies on capture efficiency of trout sampled with backpack electrofishing.

2. Evaluate the effect of low (30 Hz) and high (60 Hz) pulse frequencies on spinal injury of trout sampled with backpack electrofishing.

METHODS

Study sites

We chose study sites based on their size and our pre-existing knowledge of the presence of trout. Study sites were selected across the state of Idaho to encompass a range of water conductivities, which are known to affect power transfer to the fish and thus CE and injury. Sample sites were streams 2-4 m wide that could be effectively sampled with only one backpack electrofisher and that could be sectioned off effectively with block nets. At each stream, two sample reaches approximately 50 m long were chosen and separated on the upper and lower ends with double block nets set 1-2 m apart. Block nets consisted of 1.22 m X 4.57 m of 0.95 cm mesh with a floating line along the top and a lead line along the bottom. Ropes attached to the floating line on either end were secured to streamside vegetation and the bottom was secured to the stream bottom by lining it with cobble and boulders. Electrofishing surveys were conducted in the months of July-September, after peak streamflows had subsided for the year and before deciduous leaf inputs prevented block nets from functioning effectively overnight.

Marking pass

We used a Smith Root LR-24 backpack electrofisher for all of the study streams. Pulsed direct current (DC) was used with 24% duty cycle. Voltage was adjusted to achieve specific average power output as described below, which differed between marking and depletion passes. A pulse frequency of 30 Hz was used for capture fish during the marking pass to minimize power output experienced by marked fish. The person wearing the backpack electrofisher was the primary netter with one backup netter. We collected water temperatures and conductivities before and after each of six electrofishing passes (marking, removal [4], and x-ray sample collection).

Prior to electrofishing inside the reach, power settings were determined in a representative pool/riffle location outside either reach. To capture fish for marking, we set the voltage to achieve a power output that averaged 75 W between a riffle and a pool, given that water depth affects power output. For the marking run, 30Hz pulse frequency and 24% duty cycle were used. After tuning the power settings, we electrofished each reach and captured 10-21 fish for marking, attempting to mark roughly equal numbers for the 30Hz and 60Hz treatment reaches. Fish were measured to the nearest millimeter and given an upper caudal clip for the upstream reach and a lower caudal clip for the downstream reach. After marking, the fish were returned to their respective reaches.

Depletion passes

Prior to the 4-pass depletion, we tested power output in a riffle and pool and adjusted voltage so that it averaged 95-100 W between the two water depths because this output typically results in sufficient immobilization (taxis but not tetany) of trout in small stream settings. We used 30Hz and 60 Hz randomly assigned to each of a pair of block netted reaches for testing CE during recapture runs. The person wearing the electrofisher was not informed of the pulse frequency being used for that reach, so as not to bias their effort. During each pass fish were captured with a net and transferred to a bucket for data collection. After being measured to the nearest mm and checked for fin clips, fish were allowed to recover and placed in a net pen outside of the sample reach until the four passes were complete. We also electrofished between each set of block nets for all four passes to assess fish escapement prior to and during the removal process.

Fish collection and x-ray analysis

Power output was tested in a riffle and pool as described above and adjusted to achieve an average of 95-100 W. Ten fish were collected at 30 Hz and 60 Hz and euthanized for subsequent x-ray examination. These fish were collected outside of the block-netted study reaches so that we collected fish that had not been previously electrofished. We also collected and euthanized 10 fish per sampling basin with rod and reel as a non-shocked control group.

Each trout collected for x-ray analysis was kept frozen from the time of field collection until x-radiography (x-ray) was carried out. Dorsal and lateral views were taken from each fish. Digital files of x-rays from each fish were examined by two readers, neither of whom knew to which treatment the fish belonged. Scores indicating the injury severity, if any, were assigned to each fish following Reynolds and Kolz (2012). Disagreements in scores among the two readers were refereed by a third person. Locations and number of affected vertebra were also noted.

Habitat measurements

We collected data on several physicochemical variables that could affect CE. Temperature (C°) and specific conductivity (µS/cm) were measured with an Oakton Cond 6+ conductivity meter (Oakton Instruments, Vernon Hills, IL) at the beginning and end of electrofishing. Five to six equally spaced transects were designated for each reach. Wetted stream width (m), depth, and overhanging bank vegetation were measured with a stadia rod. We also estimated relative substrate composition, shading, undercut banks, unstable banks, and amount of large wood. Stream gradient was calculated from differences in elevation of upstream and downstream ends of study reaches obtained by GPS waypoints and topographical maps.

Data analysis

Mark-recapture estimates were calculated for each reach using the Peterson mark-recapture method with the Chapman modification for small sample size.

Capture efficiency was calculated as the number of marked fish recaptured during the depletion run (all four passes combined) divided by the total number of marked fish. Multiple linear regression was used to test the effect of pulse frequency as well as several physicochemical variables on CE. Candidate models were ranked using Akaike's Information Criterion (AIC), with only models having a AIC value of ≤2 considered plausible. Variables included in model building that could affect CE include mean wetted width, mean depth, percent boulder substrate, percent large wood, percent shading, percent undercut bank, overhanging vegetation, gradient, and specific conductivity. Model coefficients and 95% confidence intervals (CI) are reported for the top models.

Bias was calculated as $1-(N_R/N_M)$, where N_R is the number of marked fish estimated using the Carle-Strub removal depletion estimator and N_M is the known number of marked fish in each reach.

RESULTS

For sites sampled with 30 Hz, CE averaged 0.63 on pass 1, 0.36 on pass 2, 0.13 on pass 3, 0.12 on pass 4, and 0.83 overall. For sites sampled with 60 Hz, CE averaged 0.77 on pass 1, 0.46 on pass 2, 0.25 on pass 3, 0.05 on pass 4, and 0.93 overall. Bias in the estimated number of marked fish averaged 0.16 for the reaches sampled with 30 Hz and 0.07 for reaches sampled with 60 Hz (Figure 1).

The most plausible model for explaining CE included the variables pulse frequency, ambient water conductivity, and water temperature. The model estimated a 9% increase in CE using 60 Hz compared to 30 Hz, even after accounting for ambient conductivity and temperature. The coefficients for pulse frequency, ambient conductivity and temperature all had confidence intervals that did not include zero, suggesting their effects were significantly different from the intercept. The next two best models included either conductivity alone or pulse frequency and conductivity, including an interaction between the two.

Of the 149 fish sampled with 30 Hz for x-ray analysis, 7% (10 fish) had vertebral compressions and 2% (3 fish) had vertebral compressions and misalignments. Of the 140 fish sampled with 60 Hz for x-ray analysis, 3% (4 fish) had vertebral compressions and 8% (11 fish) had vertebral compressions and misalignments. No significant change in injury rate was observed between 30 Hz (9%) and 60 Hz (11%; $\chi^2=0.0289$, df=1, p=0.865). Of the 75 angled fish, no injuries were observed. No fractured vertebrae were observed in any of the x-ray fish from any of the treatments.

DISCUSSION

This study adds to the body of evidence demonstrating the underestimation of removal abundance estimates compared to mark-recapture abundance estimates (Heggberget and Hesthagen 1979; Schnute 1983; Riley and Fausch 1992). Our study is unique in that it demonstrates how pulse frequency of backpack electrofishers contributes to an underestimation by affecting capture efficiency. Furthermore, our low injury rates associated with the pulse frequencies tested indicate a viable tradeoff between maximizing capture efficiencies and minimizing electrofishing injuries.

Underestimation of abundance using removal methods is generally thought to be related to reductions in capture efficiency among removal passes. Indeed, reduced CE among passes were detected in this study as well. Declining CE is likely caused by avoidance behavior of any remaining fish that were previously exposed to the electric field but not captured.

Model selection indicated that, in addition to pulse frequency, ambient water conductivity and water temperature were important variables affecting CE. This finding is likely directly related to the power transfer theory as it relates to electrofishing, which states that electrical power transferred from the water to the fish is highest when the conductivities of the water and the fish are equal (Kolz 1989). While the effective conductivities of fish in our study were not known, they can range from 70-200 µS/cm and are affected by the cross-sectional area of the water (i.e., water depth; Kolz 2006). Additionally, physical stream attributes such as depth, width, gradient, substrate, and instream cover did not explain differences in CE like in other studies (Peterson et al. 2004; Hense et al. 2010; Meyer and High 2011).

While the CE was higher for 60 Hz than for 30 Hz, no appreciable corresponding increase in injury frequency was observed. Both pulse frequency treatments had a similar overall frequency of injury; the 60 Hz treatment had a higher proportion of vertebral misalignments than compressions. Although vertebral misalignments have a higher trauma score (2) compared to vertebral compressions alone (1) on a scale of 0-3 (Reynolds 1996), no evidence exists to suggest that misalignments are a more severe injury than compressions. Thus, using 60 Hz for sampling purposes appears to have the benefit of greater capture efficiency and better trout abundance estimates with no increase in spinal injury.

RECOMMENDATIONS

1. To maximize capture efficiency during backpack electrofishing, use 60 Hz and adjust the voltage to achieve a minimum of 100 W average power output, while always monitoring and facilitating adequate recovery of captured fish.

2. Collect another year of data to boost sample size.

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LITERATURE CITED

Heggberget, T. G., and T. Hesthagen. 1979. Population estimates of young Atlantic salmon, Salmo salar L., and brown trout, Salmo trutta L., by electrofishing in two small streams in North Norway. Report - Institute of Freshwater Research (Sweden).

Hense, Z., R. W. Martin, and J. T. Petty. 2010. Electrofishing Capture Efficiencies for Common Stream Fish Species to Support Watershed-Scale Studies in the Central Appalachians. North American Journal of Fisheries Management 30(4):1041–1050.

Kolz, A. L. 1989. A power transfer theory for electrofishing. US Fish and Wildlife Service, Fish and Wildlife Technical Report 22 (1-11).

Kolz, A. L. 2006. Electrical Conductivity as Applied to Electrofishing. Transactions of the American Fisheries Society 135:509–518.

Mahon, R., E. K. Balon, and D. L. G. Noakes. 1979. Distribution, community structure and production of fishes in the upper Speed River, Ontario: a preimpoundment study. Environmental Biology of Fishes 4:219–244.

Meyer, K. A., and B. High. 2011. Accuracy of Removal Electrofishing Estimates of Trout Abundance in Rocky Mountain Streams. North American Journal of Fisheries Management 31:923–933.

Moran, P. A. P. 1951. A Mathematical Theory of Animal Trapping. Biometrika 38:307–311.

Peterson, J. T., R. F. Thurow, and J. W. Guzevich. 2004. An Evaluation of Multipass Electrofishing for Estimating the Abundance of Stream-Dwelling Salmonids. Transactions of the American Fisheries Society 133:462–475.

Reynolds, J. B., and L. Kolz. 2012. Electrofishing. Pages 305–361 A. V. Zale, D. L. Parrish, and T. M. Sutton, editors. Fisheries Techniques, 3rd edition. American Fisheries Society, Bethesda, Maryland.

Reynolds, J. B. 1996. Electrofishing. Pages 221-253 in B. R. Murphy and D. W. Willis, editors. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

Riley, S. C., and K. D. Fausch. 1992. Underestimation of Trout Population Size by Maximum-Likelihood Removal Estimates in Small Streams. North American Journal of Fisheries Management 12:768–776.

Schnute, J. 1983. A New Approach to Estimating Populations by the Removal Method. Canadian Journal of Fisheries and Aquatic Sciences 40:2153–2169.

Zippin, C. 1956. An Evaluation of the Removal Method of Estimating Animal Populations. Biometrics 12:163–189.

Zippin, C. 1958. The Removal Method of Population Estimation. The Journal of Wildlife Management 22:82–90.

TABLES

Table 1. Physicochemical characteristics of 26 reaches of 11 streams electrofished for trout in Idaho.

Variable	**Mean**	**SD**	**Range**
Reach length (m)	56.6	25.36	29-142
Temperature (°C)	14.7	2.02	9-17.38
Specific conductivity (µS/cm)	159.67	115.32	54.37-408.34
Ambient conductivity (µS/cm)	113.7	78.18	44.86-280.89
Width(m)	2.57	0.56	1.70-3.43
Depth (m)	0.14	0.11	0.06-0.45
Gradient (%)	4.80	2.39	1.92-8.77
Substrate (%)			
Fines	8	8	1-27
Gravel	34	9	18-49
Boulder	18	13	2-40
Sand	8	5	0-19
Cobble	31	8	21-45
Bedrock	0	0	NA
LWD%	6	8	0-25
Undercut Bank (%)	18	10	6-43
Shading(%)	42	20	6-81
Overhanging vegetation width (m)	0.24	0.20	0-0.83

Table 2. Comparison of linear regression models relating stream and electrofisher variables to capture efficiency of previously marked trout. Adjusted R^2 values, degrees of freedom (df), Akaike's Information Criterion (AIC_c), and change in AIC_c values (ΔAIC_c), and AIC_c weights were used to select the best models (i.e., models with ΔAIC_c values ≈ ≤2.0) from a set of candidate models. Variables

Model	**R^2**	**df**	**AICc**	**ΔAICc**	**weight**
Capture efficiency ~ pulse frequency + ambient conductivity + temperature	0.52	5	-36.0	0	0.696
Capture efficiency ~ pulse frequency + ambient conductivity	0.41	4	-33.2	2.80	0.171
Capture efficiency ~ pulse frequency × ambient conductivity	0.40	5	-31.1	4.95	0.058

Table 3. Model coefficients, their estimates, and 95% confidence intervals (CI) for the best model relating stream and electrofisher variables to capture efficiency of electrofished trout.

Coefficient	**Estimate**	**95% CI**
Capture efficiency ~ pulse frequency + ambient conductivity + temperature		
Intercept	0.3247	0.0205-0.6289
Pulse frequency (60Hz)	0.0031	0.0005-0.0057
Ambient conductivity	0.0009	0.0003-0.0014
Temperature	0.0217	0.0029-0.0406

FIGURES

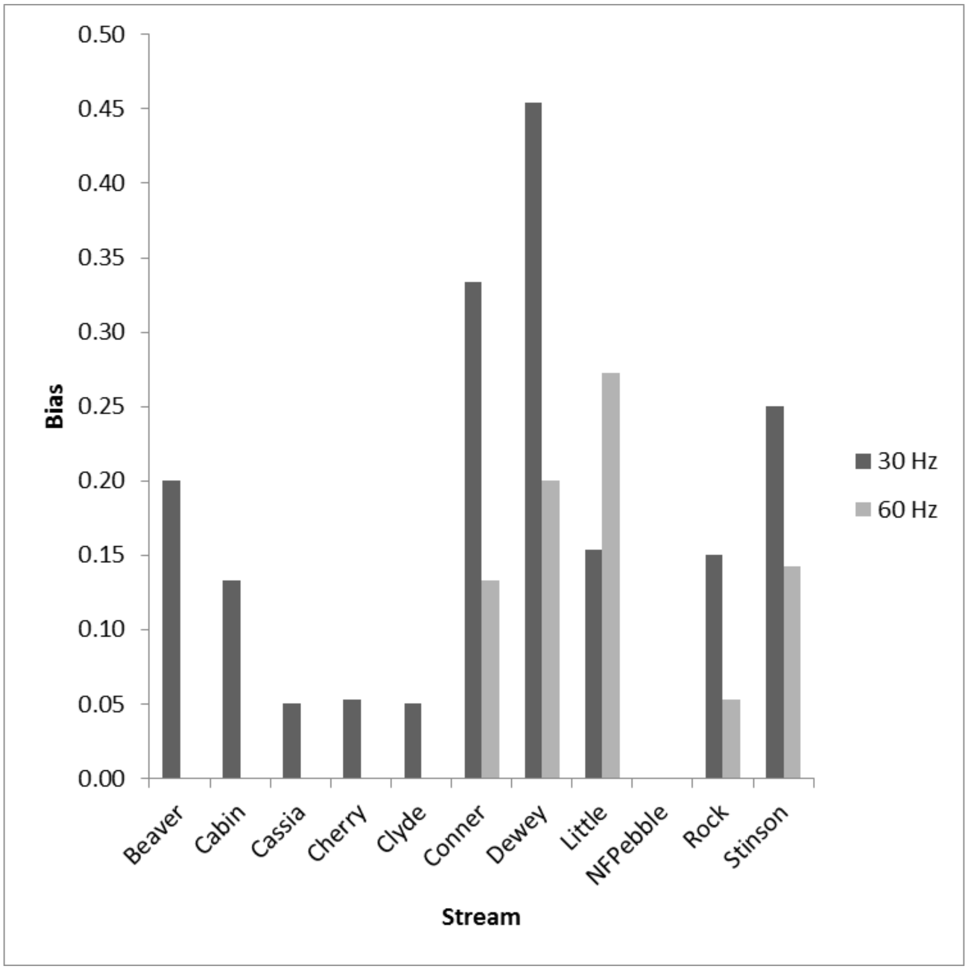

Figure 1. Bias in abundance estimates of marked fish in streams electrofished with either 30 or 60 Hz. Bias was calculated as $1-(N_R/N_M)$, where N_R is the number of marked fish estimated using a Carle-Strub removal depletion estimator and N_M is the known number of marked fish.

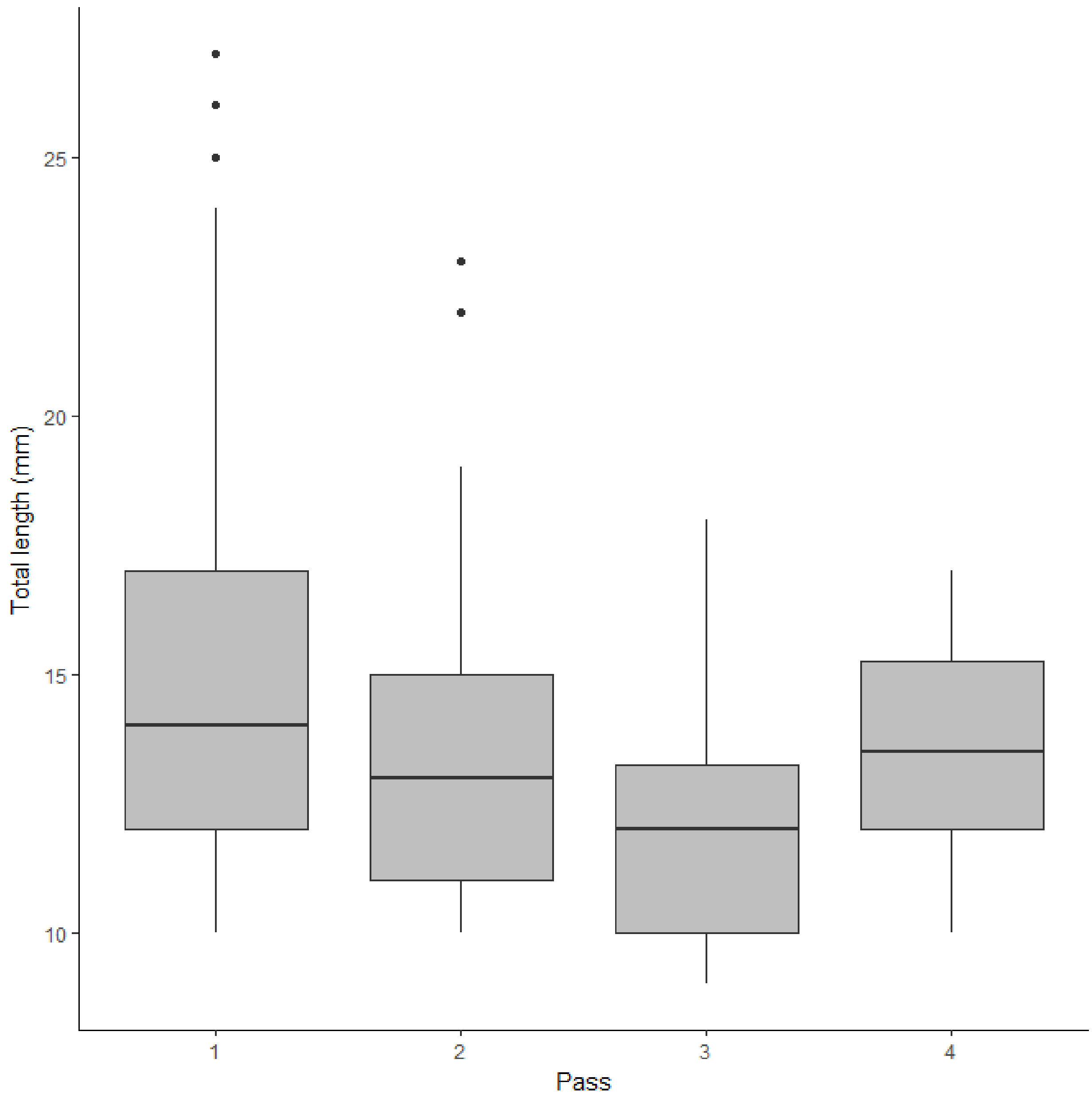

Figure 2. Boxplots of trout total length (mm) captured in each pass among 22 reaches on 11 streams.

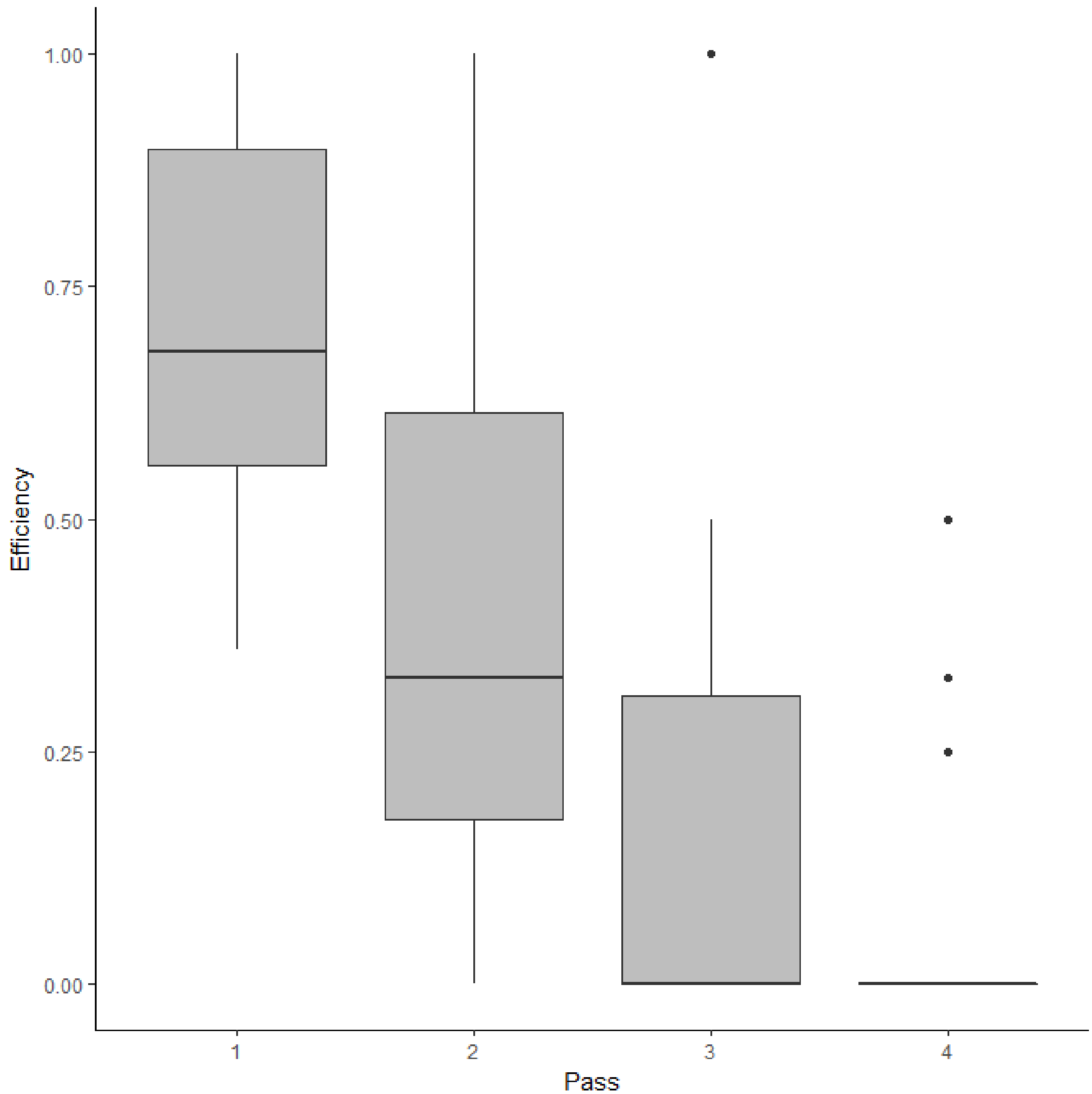

Figure 3. Boxplots of capture efficiency of marked trout during a four pass depletion removal.

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