FISHERY RESEARCH

WILD TROUT EVALUATIONS: YELLOWSTONE CUTTHROAT TROUT STREAM PURIFICATION, PELICAN AND CORMORANT PREDATION, AND PIT TAG RETENTION

Report Period July 1, 2013 to June 30, 2014

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By

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CHAPTER 1: ATTEMPTING TO PURIFY A YELLOWSTONE CUTTHROAT TROUT STREAM BY REMOVING RAINBOW TROUT AND HYBRIDS VIA ELECTROFISHING

ABSTRACT

We completed six backpack electrofishing removals between 2010 and 2014 to remove Rainbow Trout *Oncorhynchus mykiss* and Rainbow Trout X Cutthroat Trout *O. clarkii* hybrids from Palisades Creek (a tributary of the South Fork Snake River) in an attempt to purify an introgressed population of Yellowstone Cutthroat Trout *O. clarkii bouvieri*. Removals were conducted from an electric weir (0.7 km upstream from the mouth) — installed to prevent upstream migration of non-native salmonids — upstream approximately 10 km to a high velocity cascading section of stream that appears to be a complete fish passage barrier. A total of 10,925 fish were captured in Palisades Creek across all removals, of which 2,963 were Rainbow Trout or hybrids and were consequently removed from the stream. The proportion of the total catch that Yellowstone Cutthroat Trout comprised (across all size classes combined) increased slowly over time, from 67% in 2010 to 81% for the second removal in 2014. Removals were especially effective at targeting large fish (i.e., ≥300 mm in total length), with Cutthroat Trout comprising 58% of those fish in 2010 but 91% by the second 2014 removal. In general, there were fewer Rainbow Trout and hybrids in the upper reaches of the treatment section, especially in later years; by 2012, Yellowstone Cutthroat Trout comprised ~90% of the trout population in the highest 2 km of stream. Capture efficiency across all years and size classes averaged 0.36, ranged from 0.23 to 0.51, and was much higher for larger fish than smaller fish. Despite the slow pace at which the stream is being purified, our results suggest that, with one more year of removal in 2015 (especially if three removals are attempted), it may be possible to achieve the overall goal of having Yellowstone Cutthroat Trout comprise >90% of the trout composition for all size classes.

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INTRODUCTION

In eastern Idaho, the South Fork of the Snake River supports one of the few remaining fluvial populations of Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* (Thurow et al. 1988; Gresswell 2011). However, the long-term persistence of Cutthroat Trout in the South Fork of the Snake River drainage is threatened by an increasing abundance of Rainbow Trout *O. mykiss* (High 2010). Interactions between native Cutthroat Trout *O. clarkii* and non-native Rainbow Trout often reduce or eliminate pure Cutthroat Trout populations via introgressive hybridization. If introgressive hybridization occurs for many generations throughout a population, the result is sometimes a hybrid swarm, where none of the remaining individuals in the population are pure (Allendorf et al. 2001). Once a hybrid swarm is established, without population replacement, it is essentially impossible to recover the population to a genetically pure state even with new introductions of Cutthroat Trout from a genetically pure population.

Fortunately, Rainbow Trout are not ubiquitous throughout the South Fork of the Snake River drainage (Meyer et al. 2006a), which is not surprising since hybridization is often not uniform within individual populations (Woodruff 1973). In areas where hybrid swarms have not yet formed, management actions that focus on removing Rainbow Trout and hybrids will almost certainly reduce both the rate and spread of hybridization and introgression in Cutthroat Trout populations, and may actually be able to restore populations to a genetically pure or nearly pure condition (Leary et al. 1995). Debates on the value of hybridized populations of Cutthroat Trout have been fervent; opinions vary from discounting any population that is not entirely pure (Allendorf et al. 2004, 2005) to advocating preservation of populations that are up to 25% hybridized (USFWS 2003; Campton and Kaeding 2005). The Idaho Department of Fish and Game (IDFG) categorizes Cutthroat Trout into core, conservation, and sport fish populations as those with <1%, 1−10%, or >10% introgressive hybridization, respectively (Lentsch et al. 2000) and prioritizes fisheries management of each type of population differently. Converting conservation and sport fish populations to a more pure category by reducing Rainbow Trout and hybrid abundance is one of IDFG's highest management priorities for Yellowstone Cutthroat Trout populations in Idaho (IDFG 2007).

Prior attempts to eradicate non-native salmonids from streams using electrofishing removals have produced mixed results. Electrofishing removals typically reduce the non-native species but do not lead to complete eradication (e.g., Thompson and Rahel 1996; Meyer et al. 2006b; Peterson et al. 2009; Carmona-Catot et al. 2010; Buktenica et al. 2013). In the few instances where electrofishing removals have apparently resulted in complete eradication, the treatment reaches have been very short (<3 km), the streams have been very small (<3 m wide), and multiple electrofishing removals per year for many consecutive years have been needed to achieve success (e.g., Kulp and Moore 2000; Shepard et al. 2014). Although this body of work collectively demonstrates the difficulty of using electrofishing to completely eradicate unwanted fish from streams, most of these attempts have targeted non-native Brook Trout *Salvelinus fontinalis* for removal. Relative to most other salmonids, Brook Trout mature at an early age and are able to withstand intense removal pressure with little impact on their abundance or survival (Meyer et al. 2006b). Our target species was Rainbow Trout, which have hybridized with Yellowstone Cutthroat Trout in several spawning tributaries of the South Fork of the Snake River, including Palisades Creek. Rainbow Trout may not be as resilient as Brook Trout to sustained intensive removal pressure because they mature later in life. Moreover, complete eradication was not considered necessary for project success, since our goal was to reduce hybridization in Palisades Creek to <10%, thereby transforming the population from an IDFG-designation as a sport fish Cutthroat Trout population to a conservation population. Additionally, an electric weir was in place near the mouth of Palisades Creek to prevent

upstream passage of Rainbow Trout (Larson et al. 2014), thereby reducing the concern of recolonization. Removing the residualized Rainbow Trout and hybrids in Palisades Creek would therefore provide a conservation benefit for pure Yellowstone Cutthroat Trout in the South Fork of the Snake River system.

OBJECTIVE

1.   Evaluate whether a hybridized population of Yellowstone Cutthroat Trout in Palisades Creek could be reduced to <10% introgression with four years of electrofishing removals.

METHODS

Palisades Creek is a 4th order stream (1:24,000 hydrologic scale) with a mean width of approximately 10 m and an average gradient of approximately 2.0%. The fish community is predominantly Yellowstone Cutthroat Trout, Rainbow Trout, and hybrids. Although a few wild Brown Trout *Salmo trutta* and Mountain Whitefish *Prosopium williamsoni* are also present, they comprise <1% of the salmonid population and were not included in any aspects of our study. Paiute Sculpin *Cottus beldingi* are the only known non-game fish present.

The main stem of Palisades Creek is about 30 kilometers in length. However, about 11 km upstream from the confluence with the South Fork of the Snake River, there is a high-gradient, steep cascading section that fish surveys have identified as a natural velocity barrier (Meyer et al. 2006a; K. Meyer, unpublished data). Indeed, Rainbow Trout and hybrids have not been identified upstream from this barrier, although Yellowstone Cutthroat Trout and Paiute Sculpin are present above the barrier. Consequently, removal of Rainbow Trout and hybrids occurred only in this lower portion of the stream.

In an attempt to preclude upstream migration of Rainbow Trout and hybrids, several weirs have been operated on Palisades Creek since the year 2000. Most of these weirs were not efficient or could not be operated during high flows when Rainbow Trout are migrating into spawning tributaries. In 2009, a permanent electric weir was installed on Palisades Creek 0.7 km upstream from the confluence with the South Fork of the Snake River (Larson et al. 2014). The weir is generally about 90-95% efficient at capturing upstream migrating trout (B. High, unpublished data).

Rainbow Trout and hybrid removals were conducted annually in 2010 and 2012, but in 2013 and 2014 the electrofishing effort consisted of two complete removals separated by one month. Removal efforts were not possible during 2011 due to high flow conditions that rendered electrofishing inefficient and dangerous all summer and fall. Also, due to damaging high flows, the electric weir failed in 2011 early in the spawning run and had to be shut down for the remainder of the spawning period. During removals in other years, teams with backpack electrofishers and nets electrofished sequential sections of about 800 m. Removals started at the electric weir and proceeded approximately 10 km upstream to the natural barrier.

Electrofishing teams consisted of 2-4 people (depending on stream flow) with backpack electrofishers and two or more people with nets and buckets. We used a pulsed-DC waveform operated at 60 Hz, 200–500 V, and a 1–3-ms pulse width. During sampling, persons with backpack electrofishers covered all available habitats. Where gradient was too steep to effectively net fish, one electrofisher was used to chase trout downstream out of the steep

section and into an area with slower water velocity where fish could be more easily immobilized and netted, while the remaining electrofishers were used to block the downstream end of the slower water, where immobilization and netting occurred. All fish were anesthetized with MS-222, identified, and measured for total length (TL). Trout <100 mm TL comprised about 1% of the total catch; fish of this size were too small to effectively capture and thus were not considered in any of our analyses, although Rainbow Trout and hybrids of this size were removed (but not enumerated).

In the lower 4 km of stream, a road paralleled the stream, and Rainbow Trout and hybrids in this section were removed from the stream and transported with hatchery trucks to nearby community fishing ponds; in the roadless section of stream, Rainbow Trout and hybrids were euthanized after capture. All Yellowstone Cutthroat Trout were released after recovering from anesthesia.

One week prior to each electrofishing removal (except in 2010), Yellowstone Cutthroat Trout were collected, measured, and marked with a fin clip that varied for each year. These marks were used to estimate trout abundance, capture efficiency, and Rainbow Trout removal efficiency. Only Yellowstone Cutthroat Trout ≥100 mm total length were marked and released, whereas all Rainbow Trout and hybrids that were encountered during the marking runs (including those <100 mm TL) were removed from the stream.

The Fisheries Analysis + software package (Montana Fish, Wildlife and Parks 2004) was used to estimate trout abundance, using the modified Peterson estimator. Separate abundance estimates were made for the smallest size groups possible (usually 25-50 mm), having at least three marked fish per group in order to satisfy model assumptions. We assumed that there were: 1) no mortality of marked fish; 2) no movement of marked or unmarked fish out of Palisades Creek between the marking and recapture run; and 3) no difference in capture efficiency between Cutthroat Trout, Rainbow Trout, and hybrids. All trout were pooled for an overall estimate of abundance (e.g., Mullner et al. 1998; Isaak and Hubert 2004), and estimates for each species (hybrids were grouped with Rainbow Trout) were calculated based on the proportion of catch comprised by each species in each size class during the recapture run (Meyer and High 2011).

RESULTS

A total of 10,925 trout were captured in Palisades Creek across all removals, of which 2,963 were Rainbow Trout or hybrids and were consequently removed from the stream (Table 1). Fish <200 mm TL made up the bulk of the abundance estimates for both Yellowstone Cutthroat Trout (63% on average) and Rainbow Trout and hybrids (64%), whereas large fish (≥300 mm) made up <10% of the abundance of both species .

Total population abundance for Rainbow Trout and hybrids increased from a total of 959 in 2012 to 2,832 for the first removal of 2013, but then declined steadily to 1,310 for the second removal of 2014 (Table 1). On average only 3% of these fish were ≥300 mm TL. At the time of the second removal in 2014, there were only an estimated 24 Rainbow Trout ≥300 mm remaining in the stream, of which 17 were removed during that survey. At the same time, Yellowstone Cutthroat Trout abundance increased steadily, from 2,598 fish in 2012 to 6,245 fish by the second removal of 2014.

The proportion of the total catch that Yellowstone Cutthroat Trout comprised (across all size classes combined) increased slowly over time, from 67% in 2010 to 81% for the second removal in 2014. However, within each size class, the proportion of the total catch that Yellowstone Cutthroat Trout comprised showed different trends (Figure 1). For the smallest fish (<200 mm), Cutthroat Trout comprised 74% of the total catch at the beginning of the study but declined to only 53% of the catch by the first 2013 removal; by the second 2014 removal, Cutthroat Trout comprised 83% of the catch for the smallest fish. For intermediate-sized fish (200-299 mm), Cutthroat Trout comprised only 59% of the catch in 2010, but by the second 2014 removal they comprised 75% of the catch. The largest impact of the removals was on spawning-sized fish (≥300 mm), with Cutthroat Trout comprising 58% of the catch in 2010 but 91% of the catch for the second 2014 removal.

In general, there were fewer Rainbow Trout and hybrids in the upper reaches of the treatment section for all size classes of fish (Figure 1). In fact, the uppermost reach was the only reach which met the goal of having Cutthroat Trout comprise 90% of the fish population for all size classes. In contrast, the lower 1.6 km of stream was the only section where trout composition did not shift toward a higher percentage being comprised of Yellowstone Cutthroat Trout (Figure 2).

Capture efficiency for each removal ranged from a low of 23% for the first removal of 2014 (when stream flow was the highest of any removals) to a high of 51% for the 2012 removal (when flow was lowest of any removals). Capture efficiency generally increased as fish size increased for all removal events for which capture efficiency was estimated (Table 2).

Size structure for Rainbow Trout and hybrids shifted to smaller fish with each subsequent removal through 2013, but in 2014 the size of Rainbow Trout and hybrids shifted back toward a larger average size (Figure 3). Alternatively, size structure for Yellowstone Cutthroat Trout changed very little over time (Figure 3).

DISCUSSION

Palisades Creek is a relatively wide, steep, swift, and deep stream that is difficult to sample effectively with electrofishing equipment. Thus, it is not surprising that capture efficiency of marked fish was only between 23 and 51%. Even in much smaller streams that are less complex and with significantly less stream flow, achieving capture efficiency above 50% with backpack electrofishers is difficult (Meyer and High 2011). Capture efficiency was highest for spawning-sized fish, which is promising since removing the largest fish should reduce recruitment of Rainbow Trout and hybrids in the stream over time. Unfortunately, the abundance of small and intermediate-sized Rainbow Trout and hybrids is diminishing at a slow rate (Table 1). This disconnect suggests one or more of the following issues may have diminished the success of the removals. First, the weir may not be completely blocking Rainbow Trout and hybrids when in operation, which may be allowing adult Rainbow Trout to successfully enter and spawn in Palisades Creek. This explanation seems implausible because the abundance of spawning-sized Rainbow Trout and hybrids has declined since the removals began (Table 1), and the number of Rainbow Trout and hybrid spawners captured at the weir is already very low in most years. Second, the lack of operation of the weir in 2011, combined with our inability to remove Rainbow Trout and hybrids that year, may have allowed Rainbow Trout and hybrids to enter the stream, spawn, and produced a strong age-0 year class in 2011. This explanation also seems unlikely because, based on length frequency analyses, a strong age-0 year class of Rainbow Trout and hybrids in 2011 would have been about 150-200 mm in length in 2012, and

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no strong year class was evident during the 2012 removal. Third, perhaps new Rainbow Trout and hybrids (especially subadults) are entering the stream when the weir is not being operated (i.e., sometime other than the spawning migration). This explanation seems more plausible since the only reach that is not showing a positive trend toward a higher composition of Cutthroat Trout is in the lowest reach. Also, there are several private fishing ponds adjacent to the creek that are apparently stocked with rainbow trout (B. High, personal observation); while such ponds are required to be screened, they may not be, and if they are being stocked with fertile rainbow trout that at least occasionally escape the pond, this may be diminishing our removal success. Finally, the remaining Rainbow Trout and hybrids that are missed during the removals may be undergoing a compensatory response to the removals via increased survival or reproductive success, thereby diminishing the success of the removal efforts (see Meyer et al. 2006b).

Regardless of the reason that the removals are not achieving the desired result, after six removals in four years, it now appears unlikely that continued removal efforts in Palisades Creek will result in a nearly pure Yellowstone Cutthroat Trout population in the foreseeable future. Nevertheless, with one more year of removal in 2015, especially if three removals are attempted, it may be possible to achieve the overall goal of having Yellowstone Cutthroat Trout comprise >90% of the trout composition for all size classes. Moreover, although returning the Yellowstone Cutthroat Trout population in Palisades Creek to a nearly pure condition may not be feasible, removals of Rainbow Trout and hybrids at the electric weir during the spawning run may still be successful at maintaining a healthy population of Yellowstone Cutthroat Trout in Palisades Creek. We did not analyze the genotypes of Rainbow Trout and hybrids in 2014, and although the total number of Rainbow Trout and hybrids may not be greatly diminishing, the number of Rainbow Trout alleles in the population may be diminishing at a faster rate.

The original goal of this project was to purify a Yellowstone Cutthroat Trout population with three consecutive years of removal effort. While this goal was not achieved, there are now many fewer Rainbow Trout alleles in Palisades Creek than before the removal efforts were initiated. Next year will be the final year of removal efforts in Palisades Creek, and genetic samples will be collected during the last removal effort to estimate the percentage of Rainbow Trout and hybrids and the percentage of Rainbow Trout alleles left in the Palisades Creek trout population.

RECOMMENDATIONS

1.   Conduct one more year of Rainbow Trout and hybrid removal in Palisades Creek, with three removals being conducted to maximize removal efforts.

2.   Collect genetic samples during the last removal effort to determine the level of purity in the Yellowstone Cutthroat Trout population.

3.   Consider operating the electric weir for a longer period of time, since the amount of Rainbow Trout and hybrid removal does not match the amount of reduction observed in our catch data (especially in the lowermost reaches), suggesting that Rainbow Trout and hybrids are perhaps still colonizing the stream.

4.   Contact the landowners along the creek with private fishing ponds to determine if the ponds are stocked with fertile rainbow trout and if the inlets/outlets are screened. Work with the landowners to minimize any escape of fertile rainbow trout from these ponds.

ACKNOWLEDGEMENTS

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TABLES

Table 1. Population abundance estimates and the number of Rainbow Trout and hybrids removed during each removal effort at Palisades Creek, Idaho.

| | Rainbow trout and hybrid abundance | | | | | | Yellowstone Cutthroat Trout abundance | | |
| | <20 cm | | 20-29 cm | | ≥30 cm | | | | |
Year	Estimate	Removed	Estimate	Removed	Estimate	Removed	<20 cm	20-29 cm	≥30 cm
2010	-	260	-	426	-	148	-	-	-
2012	398	125	512	236	49	56	1,400	981	217
2013-1	2,409	288	397	62	27	15	2,776	1,801	298
2013-2	954	381	539	206	16	8	2,840	1,517	437
2014-1	1,197	148	404	169	92	23	3,574	715	513
2014-2	766	194	521	201	24	17	4,481	1,528	235

Table 2. Number of marked fish (m) and capture efficiency (CE; i.e., the proportion of marked fish caught in the recapture run) for various size groups of Yellowstone Cutthroat Trout during one removal in 2012 and two removals in 2013 at Palisades Creek, Idaho.

| Size group (mm) | 2012 | | 2013-1 | | 2013-2 | | 2014-1 | | 2014-2 | |
	m	CE	m	CE	m	CE	m	CE	m	CE
100-149	13	0.00	13	0.00	24	0.13	35	0.14	24	0.13
150-199	32	0.38	16	0.25	20	0.50	38	0.21	12	0.33
200-249	33	0.45	15	0.20	25	0.24	19	0.26	21	0.43
250-299	44	0.64	13	0.08	30	0.57	6	0.33	10	0.60
300-349	20	0.85	13	0.54	22	0.32	8	0.63	10	0.70
350-499	8	0.75	11	1.00	8	0.25	12	0.17	7	0.71
total	153	0.51	81	0.32	129	0.35	118	0.23	84	0.39

FIGURES
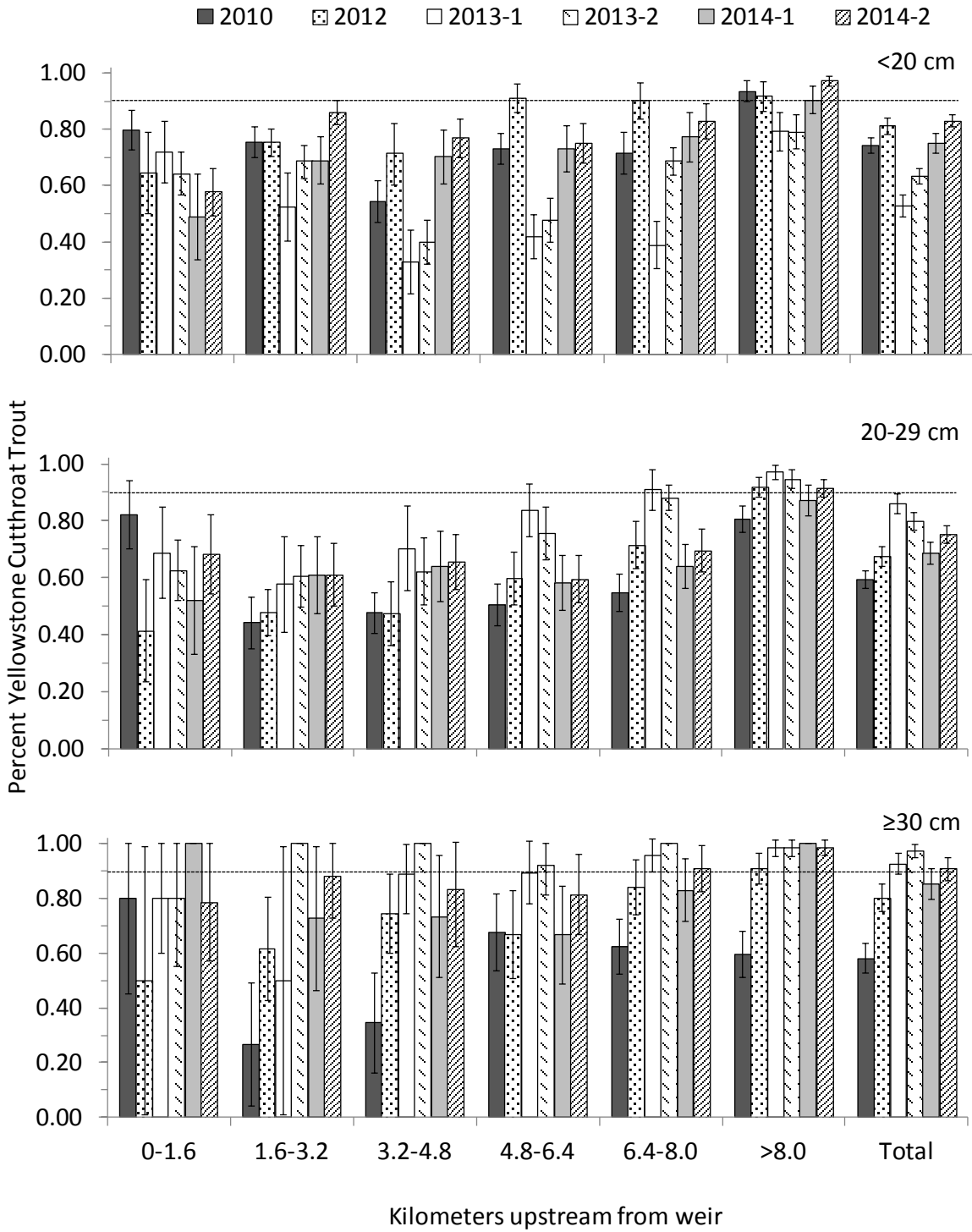
Figure 1. Proportion of fish that were Yellowstone Cutthroat Trout during electrofishing surveys from 2010 to 2014 in Palisades Creek, Idaho. Dashed bar represents the goal of at least 90% of the trout in each reach being comprised of Cutthroat Trout.

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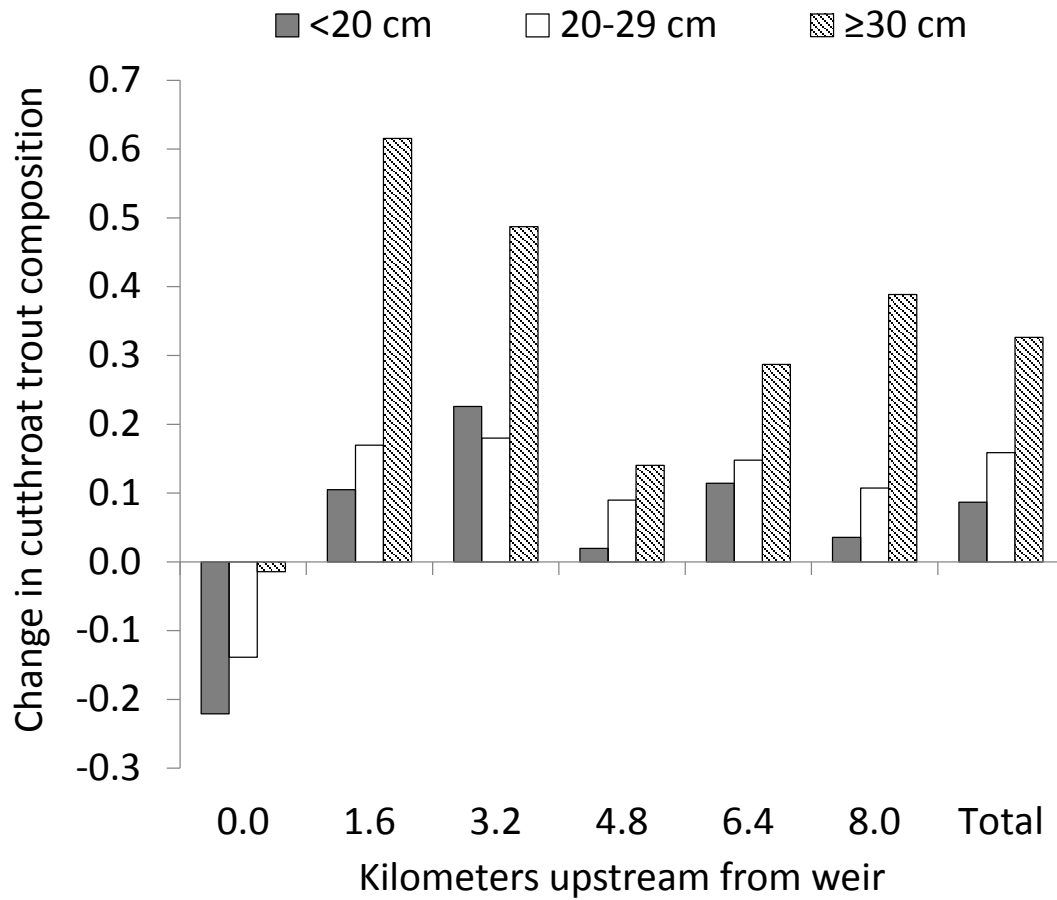
Figure 2.   Change (from the first to the last removal effort) in the proportion of the trout population being comprised of Yellowstone Cutthroat Trout in Palisades Creek, Idaho.

Figure 3.   Cumulative length frequencies for fish caught during electrofishing removals from 2010 to 2014 at Palisades Creek, Idaho.

CHAPTER 2: PREDATION BY AMERICAN WHITE PELICANS AND DOUBLE-CRESTED CORMORANTS ON CATCHABLE-SIZED HATCHERY RAINBOW TROUT IN SELECT IDAHO LENTIC WATERS

ABSTRACT

In southern Idaho, population growth of American white pelicans *Pelecanus erythrorhynchos* at the Blackfoot and Lake Walcott colonies since the early 1990s has generated concerns about whether peli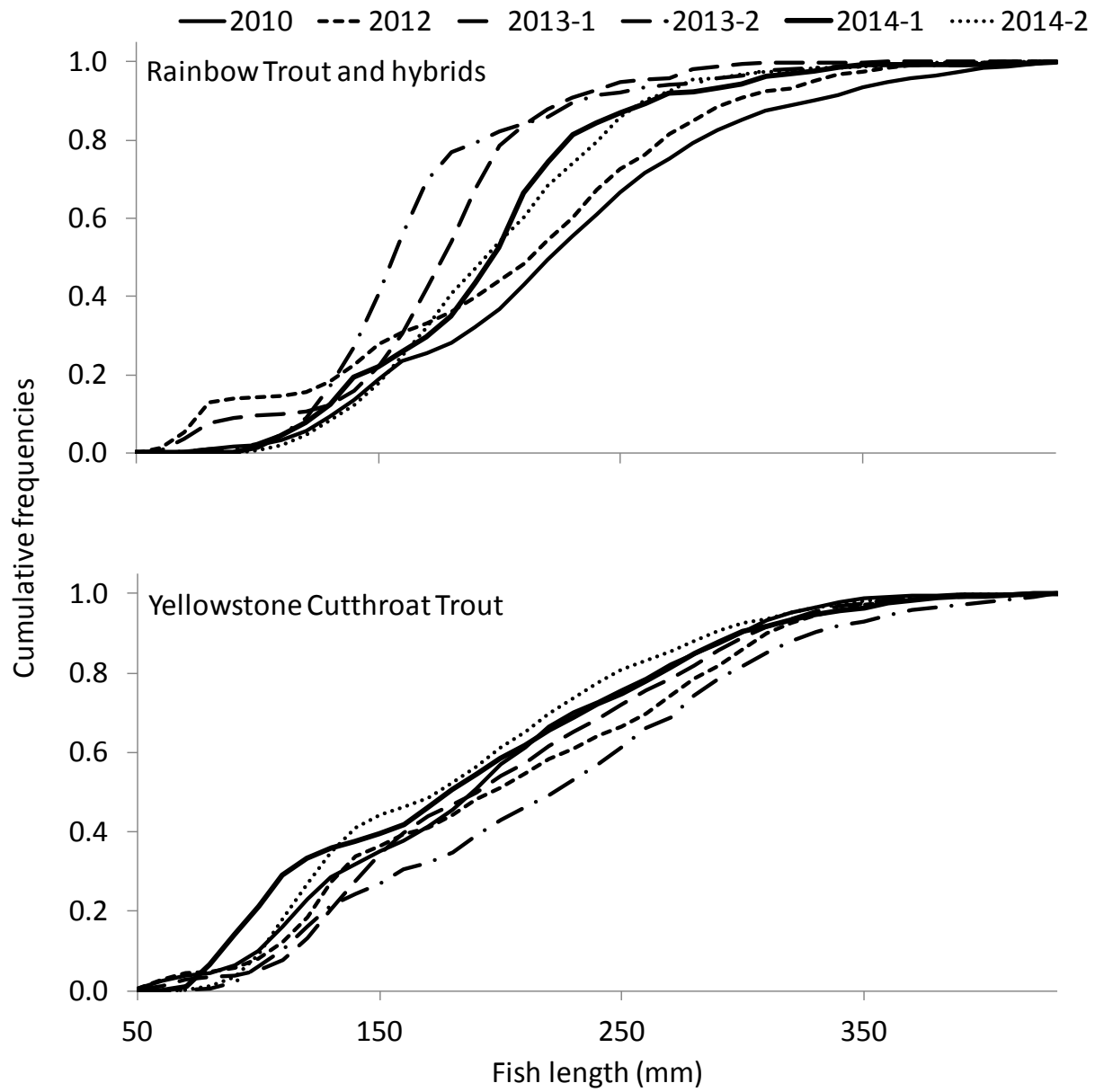can predation is impacting angler catch of hatchery trout stocked in Idaho waters. To evaluate this concern, we estimated rates of pelican predation (i.e., the proportion of fish consumed by pelicans) and angler catch (i.e., the proportion of fish caught by anglers) for 19 unique springtime fish stocking events over three years across 12 study waters; where feasible we also estimated double-crested cormorant *Phalacrocorax auritus* predation. Stocked trout averaged 247 mm in length, and were internally PIT tagged (to monitor bird predation) and externally anchor tagged (to monitor angler catch) before stocking. Additional hatchery fish were PIT tagged, euthanized, and fed directly to pelicans to estimate PIT tag deposition rates at the colonies; feeding was unsuccessful for cormorants. After the juvenile pelicans and cormorants fledged in the fall, we recovered PIT tags from stocked and fed fish that were deposited at the two colonies. Deposition rates for pelican-consumed tags averaged 21% and declined exponentially at greater distance from the colonies. Pelican predation on hatchery trout averaged 18% and ranged from 0-48%, whereas angler catch averaged 21% and ranged from 0% to 82%. Mean angler catch was nearly four times higher when pelican predation was low (i.e., <25%) than when pelican predation was high (≥25%). Cormorant predation estimates (available for 7 stocking events) were minimum estimates only (i.e., they assumed 100% of tags consumed by cormorants were recovered) and averaged 14% (range = 2-38%). Our results suggest that predation by American white pelicans and double-crested cormorants on catchable-sized hatchery trout stocked in southern Idaho waters often exceeds total catch of those fish by anglers who compete directly with avian predators for use of stocked trout.

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INTRODUCTION

American white pelicans *Pelecanus erythrorhynchos* (hereafter pelicans) experienced long-term declines in abundance across North America until the 1960s (Knopf and Evans 2004). The cause of the decline is not clear but was likely related to a lack of federal and state protection and the extensive use of pesticides prior to the 1960s (Keith 2005). Regardless of what caused the decline, since the early 1990s pelicans have experienced an almost exponential rebound in abundance (King and Anderson 2005), including in Idaho (IDFG [Idaho Department of Fish and Game) 2009.)

While recent increases in abundance are positive signs for the conservation of American white pelicans across North America, the increasing population size has also resulted in documented cases of pelican predation impacts on native fish populations and important recreational fisheries. For example, pelicans have been shown to capitalize on fish spawning migrations (Findholt and Anderson 1995; Murphy and Tracy 2005; Scoppettone et al. 2014), including in Idaho, where pelicans frequently consume substantial portions of the spawning migration of native Yellowstone Cutthroat Trout *Oncorhynchus clarkii bouvieri* in the Blackfoot River system (Teuscher and Schill 2010; Teuscher et al. 2015). Substantial levels of pelican predation have also been documented on hatchery trout within days of individual stocking events (Derby and Lovvorn 1997). Such impacts are not surprising for a generalist predator such as the American white pelican that exhibits plasticity in its opportunistic feeding habits (Hall 1925; Knopf and Kennedy 1980). With the noticeable increase in the presence of pelicans at local fisheries, anglers and fisheries management agencies are increasingly interested in quantifying the impact that pelicans may be having on fisheries.

Recent innovative research investigating avian predation on salmonids in the Pacific Northwest has focused on recovery of Passive Integrated Transponder (PIT) tags at bird colonies that were implanted in salmonids and subsequently consumed by nesting birds and deposited at the colonies (Evans et al. 2012; Sebring et al. 2013). Although PIT tag recovery efficiency at the colonies has been estimated by intentionally sowing "control tags" onto the bird colony before PIT tag recovery efforts are undertaken, a shortcoming to this approach is that off-colony deposition rate is unknown. Consequently, PIT tag recoveries using this methodology only provide minimum predation estimates since not all tags that are consumed by birds are deposited at and recovered from the colony. We used an updated modification to this approach that incorporates off-colony deposition into the predation estimates (Osterback et al. 2013; Scoppettone et al. 2014; Hostetter et al. 2015; Teuscher et al. 2015), thereby producing estimates of total predation (rather than minimum predation) by pelicans. The primary objective of this study was to estimate predation rates by American white pelicans on catchable-sized (i.e., ~250 mm in total length) hatchery Rainbow Trout *O. mykiss* stocked in several southern Idaho reservoirs and community ponds to gauge their general impact on hatchery-supported trout fisheries in southern Idaho.

In instances where pelican predation of stocked hatchery fish is relatively high, it follows that angler catch (i.e., the proportion of the stocked fish caught and therefore utilized by anglers) of those same fish would likely be minimal since a large portion of the stocked fish would have been consumed by pelicans before anglers could successfully catch them. However, angler catch of hatchery trout stocked in lentic environments is affected by numerous factors other than pelican predation, such as rearing conditions in the hatchery (e.g., Davison 1997; Barnes et al. 2009), season of stocking and size at release (Yule et al. 2000), water quality (Koenig and Meyer 2011), and the presence of piscine and other avian predators (Derby and Lovvorn 1997). Thus a low rate of pelican predation would not necessarily translate directly into high rates of

angler catch. Likewise, we expected that pelican predation, at least by breeding adults, would always be low at great distances from a colony because breeding birds would choose to forage at waters closer to their nest. However, at waters in close proximity to colonies, pelican predation would not necessarily be high since it is affected by more than just travel distance from the nest to the foraging water, such as water depth (Kaeding 2002; Ivey and Herziger 2006) and water clarity (Anderson 1991) where the birds are foraging, the vulnerability of specific prey (Findholt and Anderson 1995), and forage abundance (Kaeding 2002). Secondary objectives were to evaluate relationships between rates of pelican predation and angler catch, and rates of pelican predation and distance from colonies.

Similar to American white pelicans, double-crested cormorants *Phalacrocorax auritus* (hereafter cormorants) have also increased in abundance in recent decades throughout North America (Wires and Cuthbert 2006; Adkins et al. 2014), including in Idaho. The increased abundance of this fish predator has led to numerous conflicts with important economic fish industries in North America, especially Great Lakes sport fisheries (Burnett et al. 2002; Lantry et al. 2002; Fielder 2008) and the aquaculture industry in the Southeastern U.S. (Glahn et al. 2000; Dorr et al. 2012). As with pelicans, cormorants can also be very effective predators of hatchery trout (Modde et al. 1996; Derby and Lovvorn 1997; Skiles 2008). Accordingly, a final study objective was to estimate cormorant predation on catchable trout stocked in some of these same Idaho waters.

STUDY AREA

American white pelican nesting in Idaho occurs primarily at two adjacent islands in Blackfoot Reservoir and at three adjacent islands in Lake Walcott (also a reservoir), the latter of which is part of the Minidoka National Wildlife Refuge (NWR). Pelicans in recent years have also been attempting to nest at an island in Island Park Reservoir in eastern Idaho but success has been limited. Neighboring pelican colonies include Molly Island at Yellowstone National Park in northwest Wyoming, Gunnison Island at the Great Salt Lake in northern Utah, Badger Island on the Columbia River in southwest Washington, and the Malheur NWR in eastern Oregon (Figure 1). Double-crested cormorants also nest at the Blackfoot and Lake Walcott pelican colonies, as well as several other locations throughout Idaho.

The IDFG annually stocks about 1.8 million hatchery Rainbow Trout of catchable size (i.e., about 250 mm total length) - hereafter referred to as hatchery catchable trout - in numerous lakes and rivers of Idaho to provide put-and-take trout fisheries for Idaho anglers. For this particular study, we monitored pelican and cormorant predation and angler catch of hatchery catchable trout stocked in 12 study waters across southern Idaho (Figure 4, Table 3). Study waters were not selected at random, but instead were selected primarily to (1) investigate pelican predation in several southern Idaho fisheries known or suspected to be receiving substantial pelican use, and (2) gain perspective on possible geographical gradients in pelican predation rates across southern Idaho in relation to Idaho's primary pelican nesting locations. Cormorants were known to forage on all study waters as well. In some waters, hatchery catchable trout were the only fish present, but in most waters pelicans and cormorants could forage on a variety of other fish taxa (Table 3), including several species of centrarchids, cottids, cyprinids, catostomids, and other salmonids.

Distances from the study waters to the nearest of the two primary Idaho pelican colonies ranged from 0 to 304 km (Table 3). The soaring ability of pelicans enables them to forage at distances of up to 300 km from their nests (Johnson and Sloan 1978; Trottier et al. 1980;

O'Malley and Evans 1982). In contrast, the maximum foraging distance for cormorants is only about 50 km from their nests (Custer and Bunck 1992; Bugajski et al. 2012). Thus, for study waters within 50 km of the colonies, PIT-tagged fish consumed by avian predators and deposited at the Blackfoot or Lake Walcott colonies could have been the result of pelican or cormorant predation, whereas for waters more than 50 km from a colony, tag deposition at the colony likely could only have been the result of pelican predation. This distinction was important for our approach to estimating pelican and cormorant predation.

METHODS

Estimating the rate of pelican and cormorant predation on hatchery catchable trout involved four steps outlined in detail below but summarized here. The first step was to stock PIT-tagged hatchery catchable trout into our study waters that were then vulnerable to pelican and cormorant predation. A second step was to PIT tag other hatchery fish, euthanize them, and feed them directly to pelicans at many (but not all) of the study waters, which allowed us to estimate tag deposition rates on the colonies; direct feeding of cormorants was attempted but was unsuccessful. The third step occurred after pelicans and cormorants on the Blackfoot and Lake Walcott colonies had fledged their young; at that time we searched the two colonies (as well as a few other cormorant roosting and loafing areas) for regurgitated and/or defecated PIT tags. The final step was to apportion the recovered tags into those known or assumed to have been consumed by either pelicans or cormorants.

By recovering PIT tags at the colonies from fish stocked in our study waters, we were able to estimate a minimum rate of pelican predation at each study water, which was simply the number of tags recovered from a particular study water (and assigned to pelicans) divided by the number stocked at that water. For the four study waters within 50 km of the nearest colony, we could similarly estimate a minimum rate of cormorant predation. These minimum rates of predation did not account for stocked fish that were consumed by pelicans or cormorants but were either not deposited on or not recovered at the two colonies.

By recovering tags from fish fed directly to pelicans at various study waters, we could directly estimate water-specific tag recovery efficiency for pelicans. This was important because (1) not all tags consumed by birds nesting at one of the two colonies would necessarily be deposited on the island where they were nesting, and (2) birds foraging in Idaho that were not nesting at these two colonies (e.g., non-breeding birds, and birds nesting at other colonies) had little to no chance of depositing a tag at these colonies. Estimating tag recovery efficiency for each water allowed us to transform (for pelicans only) minimum predation estimates into estimates of total predation on hatchery catchable trout that included predation by all pelicans, not just those nesting on the colonies we were studying (cf. Teuscher et al. 2015). Because cormorant feeding was unsuccessful, tag recovery efficiency was unknown for cormorants. Thus, all tag recoveries ascribed to cormorants resulted only in minimum estimates of cormorant predation (cf. Evans et al. 2012; Sebring et al. 2013).

Fish stocking

To accomplish the first step in the methodology outlined above, we stocked PIT tagged catchable trout into each study water in conjunction with regularly scheduled hatchery trout stocking events. The PIT-tagged fish comprised on average about 2% of the total number of hatchery catchable trout stocked in any given water in any given year. Mean size of stocked fish averaged 247 mm TL (SD = 24.9). Prior to tagging, hatchery fish were sedated with peppermint

oil (in a 1:10 stock solution ratio with ethanol, using 0.3–0.5 mL of stock solution per L of water). Once sedated, PIT tags (23 mm half-duplex tags) were injected with a 7 gauge hypodermic needle into the abdominal cavity; the insertion point was posterior to the pectoral fin, offset slightly to the right or left side depending on the handedness of the individual tagger. Fish were then transferred to net pens in the raceways and held for 1-2 days prior to stocking. To reduce the rate of tagging mortality, individual fish were judged up to the point of release as to whether they were unfit for this study due to visible signs of stress from capture and handling procedures (Nielsen 1992). This monitoring protocol applied to the implantation of anchor tags as well (see below). Mortality rate for fish tagging before stocking was <1%, but individual mortalities were noted and subtracted from the number of fish actually stocked. Post-release mortality from tagging was assumed to be zero (Acolas et al. 2007).

Pelican feeding

During this same timeframe, we fed hatchery fish (also abdominally tagged with PIT tags) directly to pelicans at many (but not all) of the study waters. Feeding occurred between late May and mid-July, which encompassed much of the time when breeding pelicans were foraging and traveling between the breeding colonies and foraging sites to feed their chicks. For each pelican feeding event, hatchery fish were obtained from a state fish hatchery and were euthanized with an overdose of peppermint oil while travelling to the study water. These fish were injected with a PIT tag into the abdominal cavity and a small amount of air under the skin before being thrown individually in the direction of loafing or foraging pelicans. Although loafing and foraging pelicans were initially wary of our approaching boat, after a few days they became more comfortable with our close proximity and reticently consumed fish thrown in their direction. The purpose of the injected air in the euthanized fish was to help ensure the fish did not sink after being thrown, increasing the likelihood that a pelican would consume the PIT-tagged fish. Each fish thrown in the direction of pelicans was monitored with binoculars until a pelican captured and swallowed the fish.

Attempts were made to minimize the occurrence of individual birds consuming more than one tagged fish in any given day in order to achieve independence in tag recoveries. Although at times 100 or more pelicans were attempting to consume fish being fed to them, no more than 40 tagged fish were fed on any given day. This also allowed us to temporally disperse colony deposition of fed tags throughout more of the pelican breeding season. How many fish were successfully fed to pelicans on any given day was variable depending on pelican flock size and wariness; thus at most waters, several feeding events were employed each year throughout the feeding period.

Loafing and foraging cormorants never allowed us to be close enough in proximity to engage in direct feeding, so feeding events targeted at cormorants were abandoned. Consequently, they also did not interfere with pelican feeding.

PIT tag recoveries

We searched for regurgitated and/or defecated PIT tags from fed and stocked fish at the Blackfoot and Lake Walcott colonies after the juvenile pelicans and cormorants had fledged in the fall. We used a PIT tag reader (Oregon RFID HDX Backpack Reader) with a 0.5 m diameter hoop-antenna attached to the end of a 2-m long pole. Read range for PIT tags was generally about 0.5 m regardless of whether or not the tag was on the surface or buried slightly in a nest or below ground-level. Searchers scanned the entire colonies by "sweeping" the antenna back and forth just above the ground while slowly walking in 2-m wide transects that overlapped one

another to ensure that all of the ground was covered once. We also scanned shallow water (<0.3 m deep) surrounding the islands (submersing the antenna while sweeping these areas), and cormorant nests in bushes. When a tag was detected, surveyors noted the location as being in or very close to a cormorant nest, in or very close to a pelican nest, or not close to a nest. Surveyors then used a shovel and sieve to recover and remove the tag if it was not visible on the surface, in order to avoid interference with other PIT tags in the same area or in subsequent years. In the few instances where we were unable to recover and remove the tag, attempts were made to ensure no other PIT tags were in the same location, and individual PIT tag numbers were recorded. We assumed that any tag we recovered from stocked fish was from a live fish that a bird consumed, not from stocked fish that had died of natural causes and were later eaten by a pelican or cormorant.

Apportioning colony tag recoveries to pelicans or cormorants

For 8 of the 12 study waters (which produced 14 of the 19 individual pelican predation estimates), the nearest colony was presumably outside the foraging range of all avian predators except pelicans, so all PIT tags recovered at the colonies from those waters were assigned to pelican predation. For one other study water, tag deposition occurred at the Blackfoot colony, and this water was within the foraging range of that colony for both pelicans and cormorants. However, pelican and cormorant nesting did not overlap at Blackfoot during our study (D. Teuscher, unpublished data), so tags recovered from this water were assigned to pelican or cormorant predation based on tag recovery location at the colony. Thus assigning PIT tags recovered at the colonies to pelican or cormorant predation was unambiguous for 9 of the 12 study waters (or 15 of the 19 pelican predation estimates).

For the remaining three study waters (which produced the remaining four pelican predation estimates), tag deposition occurred at the Lake Walcott colony, and these waters were all within the foraging range of that colony for both pelicans and cormorants. At this colony, pelicans and cormorants were generally segregated in their nesting locations, but there was not complete separation. For example, cormorants often nested in bushes elevated a meter or more off the ground, and they sometimes roosted in willows, while some pelicans nested underneath these cormorant nesting or roosting areas. Moreover, some fed tags (known to have been consumed by pelicans) were recovered closer to a cormorant nest than a pelican nest (K. Meyer, unpublished data). Also, trail cameras showed that cormorants were occasionally present amidst numerous nesting pelicans, and both birds were seen loafing near one another near the island shores.

It was clear that both birds were foraging at these waters because PIT tags from stocked fish were recovered in both cormorant and pelican nests. However, because pelican and cormorant nesting and loafing was not entirely segregated at Lake Walcott, correct tag assignment for these waters at this colony was questionable. Consequently, we compared the assignment of predation to pelicans or cormorants using three approaches in order to assess variability in tag assignment (Table 4). First, recovered tags were assigned to pelicans or cormorants based on the proportional abundance of these birds at the colony. This was determined by mounting several cameras on fence posts placed strategically around the Lake Walcott islands to best capture images of birds present on the islands. The cameras captured images at hourly intervals each day from May through September each year, resulting in tens of thousands of images. We subsampled the images by randomly selecting six photographs (from daylight times only) from each camera for each month (from May to September), for a total of 180 images being used each year. We counted the number of pelicans and cormorants visible in each picture (mean = 46 pelicans and cormorants per picture; range = 0-253), and estimated

the mean number of pelicans and cormorants present across the entire period from May to September at each island. We used these estimates of bird abundance to proportionally assign tags recovered from stocked fish to either pelican or cormorant predation (Table 4). This approach assumed that pelicans and cormorants were equally successful at foraging on hatchery catchable trout, and that their energetic demands were equivalent.

A modification of this approach accounted for differences in energetic demand between these birds, which are reasonably well defined. Adult double-crested cormorants require approximately 320 g of fish/day (Hatch and Weseloh 1999) compared to 1,500 g for American white pelicans (Ferguson et al. 2011), and cormorant chicks require an estimated 8-9 kg of food from hatching to fledging (Seefelt and Gillingham 2008) compared to 68 kg for American white pelican chicks (Hall 1925). Tag assignment based solely on bird abundance was thus modified to account for these energetic differences (Table 4).

A final approach for assigning recovered tags from these three waters to either pelican or cormorant predation was based on tag recovery location relative to the nearest pelican or cormorant nest (Table 4). Although as noted above, there was not complete separation in pelican and cormorant nesting and loafing areas at the Lake Walcott colony, we nevertheless recorded the location of each recovered tag relative to the nearest pelican or cormorant nest. Under this approach, any tags recorded in or very near a pelican or cormorant nest was assigned according to the nest that the tag was in or closest to; any remaining tags recovered near shore or nowhere near a nest were assigned to pelican or cormorant predation based on estimates of bird abundance, as outlined above. This approach assumed that all tags found in or near pelican nests were consumed by pelicans and vice versa for cormorants.

All three approaches generally resulted in similar numbers of tags being assigned to either pelican or cormorant predation (Table 4). Considering this similarity, we felt that for the three study waters in question, assigning pelican or cormorant predation to recovered tags based solely on bird abundance was the best approach because it appeared to balance the various assumptions of these approaches and it resulted in relative tag assignments that were intermediate to the other two approaches.

Because only 4 of the 12 study waters were within the range of cormorant foraging from the Blackfoot or Lake Walcott colonies, basing cormorant predation only on tag recoveries at colonies would have limited our ability to characterize cormorant predation. Therefore, to augment colony tag recoveries, at a few waters we scanned for additional tags at cormorant roosting and loafing areas. We only scanned cormorant roosting and loafing areas that were (1) well defined spatially, (2) rarely if ever were visited by other avian predators (namely pelicans and herons), and (3) logistically feasible to scan. We assigned all PIT tags recovered at cormorant roosting/loafing areas to cormorant predation. Recovered tags from this step were combined with colony-recovered tags assigned to cormorants before final estimates of cormorant predation were made.

Calculating pelican predation rates

For each stocking event that was coupled with pelican feeding, proportions of recovered tags were calculated independently for both the fed tags (*FT*) and stocked tags (*RT*), where:

FT = tag recovery efficiency, i.e., the number of fed PIT tags found on the colony divided by the total number of tags fed to pelicans

RT = number of stocked PIT tags found on the colony (that were assigned to pelicans) divided by the total number of tags stocked

Variance for these proportions was calculated according to the formula in Fleiss (1981) as:

$$Var(proportion) = \sqrt{\frac{PQ}{n}},$$

where P is the numerator divided by the denominator for either FT or RT, Q is 1-P, and n is the denominator in either FT or RT. We calculated the pelican predation rate ($Pred_{pel}$) for each water when both fed and stocked tags were recovered at a colony according to the following formula:

$$Pred_{pel} = \frac{RT}{FT}.$$

Because the numerator and denominator were both individual estimates, with their own estimates of variance, we used the formula for the variance of a ratio (McFadden 1961; Yates 1980) to calculate the variance for $Pred_{pel}$, using the following formula:

$$Var\left(\frac{RT}{FT}\right) = \left(\frac{RT}{FT}\right)^2 \times \left(\frac{Var(RT)}{RT^2} + \frac{Var(FT)}{FT^2}\right).$$

For each water-specific estimate of the rate of pelican predation, we then calculated 90% confidence intervals (CIs).

For stocking events that were not coupled with pelican feeding events, we could not directly estimate total pelican predation because tag recovery efficiency was not estimated. Instead, we predicted tag recovery efficiency for these stocking events based on a scatterplot of distance-to-colony (x-axis) and tag-recovery-efficiency (y-axis) for the stocking events that were coupled with pelican feeding. The relationship was curvilinear in nature, so we fitted an exponential regression to the data to evaluate the statistical significance of the relationship. Estimates of RT for waters where no feeding occurred were then adjusted by the predicted tag recovery efficiency in order to estimate total pelican predation for these waters.

Calculating cormorant predation rates

As mentioned above, because cormorants were not fed directly, tag recovery efficiency could not be estimated for cormorants for any stocking events. Therefore, all cormorant predation estimates were minimum estimates only, based simply on the number of stocked PIT tags found at cormorant loafing/roosting areas or on the colonies and assigned to cormorants, divided by the total number of tags stocked.

Estimating angler catch

To estimate angler catch, we attached T-bar anchor tags to the same hatchery catchable trout that were released with PIT tags in the study waters. Tags were inserted just below the dorsal fin following the recommendations of Guy et al. (1996). Anchor tagging occurred at the same time as PIT tagging.

For more details on anchor tagging methods and estimating angler catch, see Meyer et al. (2012) and Meyer and Schill (2014). In short, anchor tags were fluorescent orange (so

anglers could more easily notice them on fish), 70 mm in total length (51 mm of tubing), and labeled with the agency and phone number (i.e., "IDFG 1-866-258-0338") where tags could be reported. A toll-free automated telephone hotline and website were established through which anglers could report tags, although some tags were mailed to or dropped off at IDFG offices. Tag reporting by anglers in this program was voluntary, not mandatory.

We tested whether implanting hatchery catchable trout with fluorescent orange tags made them more visible to pelicans and cormorants and therefore more vulnerable to bird predation by implanting one-half of the stocked fish with dull green anchor tags at six waters in 2013 to evaluate tag recovery by tag color. We recovered a total of 108 and 99 PIT tags from fish stocked in these waters with dull green and fluorescent orange anchor tags, respectively. A Wilcoxon signed rank test indicated that tag recoveries did not differ by color ($P = 0.50$).

Unadjusted angler catch (c) for each stocking event was calculated as the number of tagged fish reported as caught by anglers (within one year of the stocking event) divided by the number of fish released with tags; variance for this proportion was again calculated according to the same formulas in Fleiss (1981) as noted above. Adjusted angler catch (c') incorporated estimates of angler tag reporting rate (λ), anchor tag loss rate (tag_l), and tag mortality rate (Tag_m) (estimated to be 49.4%, 8.2%, and 1%, respectively; see Meyer and Schill 2014), and used the following formula:

$$c' = \frac{c}{\lambda(1 - Tag_l)(1 - Tag_m)}$$

Variance estimates for λ, tag_l, and tag_m came from data reported in Meyer and Schill (2014). Variance for the entire denominator in the above equation was estimated using the approximate formula for the variance of a product in Yates (1980):

$$s_{x_1 x_2}^2 = x_1^2 \cdot s_{x_2}^2 + x_2^2 \cdot s_{x1}^2$$

where $s_{x_1 x_2}^2$ is the variance of the product, x_1 and x_2 are independent estimates being multiplied together, and $s_{x_1}^2$ and $s_{x_2}^2$ are their respective variances. Variance for c' was calculated using the approximate formula for the variance of a ratio as previously noted, from which 90% CIs were derived.

Scatterplots were constructed to evaluate relationships between rates of pelican predation and angler catch, and rates of pelican predation and distance from colonies. The relationships were more curvilinear than linear in nature (with stronger effect sizes), so we fitted exponential regressions to the data to evaluate the statistical significance of the relationships. We also used a t-test to assess whether angler catch was reduced when pelican predation was high (i.e., ≥25%) compared to when pelican predation was low (<25%); a one-tailed test was used since we assumed that higher pelican predation would not positively affect angler catch rates.

We used $\alpha = 0.10$ for all statistical significance tests and for calculating CIs. This less-stringent significance level (compared to the more standard use of α = 0.05) was adopted to balance type I and type II errors in our statistical tests (Cohen 1990; Stephens et al. 2005) and because resource managers in our agency were content with the tradeoff of having tighter bounds around the estimates of predation and angler catch at the expense of less confidence in the estimates.

RESULTS

We directly fed to pelicans a total of 1,073 PIT-tagged hatchery catchable trout over three years, of which 189 (18%) tags were subsequently recovered at the Blackfoot or Lake Walcott pelican colonies (Table 5). For the 13 water × study year combinations of pooled feeding events, tag recovery efficiency at the colonies averaged 21% and ranged from 0-65% (Table 5). There was a strong negative exponential relationship between the distance from a particular study water to the nearest pelican colony and tag recovery efficiency for the feeding events in that study water ($r^2 = 0.80$; $F = 43.99$; $P < 0.001$; Figure 5).

We stocked a total of 5,565 PIT-tagged hatchery catchable trout in 19 separate stocking events in our study waters, of which 194 (4%) tags were recovered at the Blackfoot or Lake Walcott colonies and were known or assumed to have been consumed by pelicans (Table 6). Resulting estimates of total pelican predation on stocked hatchery catchable trout averaged 18% and ranged from 0 to 48%.

In comparison, a total of 311 PIT tags implanted in stocked fish were recovered at the colonies or at cormorant loafing/roosting areas and were known or assumed to have been consumed by cormorants (Table 7). These tag recoveries came from 7 of the 19 stocking events; for the remaining 12 stocking events, cormorant tag recoveries were not attempted. Resulting estimates of minimum cormorant predation - assuming that 100% of tags consumed by cormorants were recovered - averaged 14% and ranged from 2-38% (Table 7). If we assumed that cormorant tag recovery efficiency was equivalent to pelican tag recovery efficiency, total cormorant predation was estimated to average 21% and range from 5-69%.

The maximum pelican foraging distance we documented was 248 km (Table 3). Pelican predation rates at individual waters declined exponentially at greater distances from the nearest colony ($r^2 = 0.26$; $F = 5.93$; $P = 0.03$; Figure 6).

Angler catch of anchor tagged hatchery catchable trout stocked in study waters averaged 21% and ranged from 0% to 82% (Table 6). There was some evidence of a negative exponential relationship between pelican predation and angler catch for individual stocking events ($r^2 = 0.15$; $F = 3.11$; $P = 0.10$; Figure 7), although the relationship was weak and quite variable. Nevertheless, for stocking events where pelican predation was ≥25%, angler catch averaged only 8%, whereas when pelican predation was <25%, angler catch averaged 31%; this nearly four-fold difference in mean angler catch was statistically significant ($t = 1.33$; $df = 17$; $P = 0.03$; Figure 8).

DISCUSSION

Our results suggest that predation by American white pelicans and double-crested cormorants on catchable trout stocked in southern Idaho waters can be relatively high (i.e., >25%) and often exceeds total catch of those fish by anglers who compete directly with avian predators for use of stocked trout. Although our study includes results from only a small sample of locations, our findings support the supposition that in southern Idaho, pelican predation of hatchery catchable trout will negatively affect angler catch rates for these fish in some waters. In the neighboring state of Wyoming, pelicans quickly increased their focus on trout species (relative to other prey species available) as soon as hatchery trout were stocked (Derby and

Lovvorn 1997). Rainbow Trout more often display pelagic (i.e., suspended in the water column) rather than benthic (near the substrate) behavior in lentic waters, making them particularly vulnerable to avian predation relative to other salmonids (Matkowski 1989). Moreover, fish reared in production raceways are naïve with regard to predators and once they are stocked they do not initiate avoidance behaviors similar to wild fish (Berejikian 1995).

Estimated predation rates by pelicans on stocked Rainbow Trout in the study waters we evaluated were quite variable, but were nonetheless inversely related to the distance from the study water to the nearest colony. Declines in avian predation rates related to distance from colonies have been previously demonstrated (e.g., Fasola and Bogliani 1990; Osterback et al. 2013) and would be expected for birds such as adult pelicans that rear chicks with high energy needs and that have high energy demands of their own. The highest observed pelican predation rates in this study were usually at waters within 100 km of the nearest colony except at CJ Strike Reservoir, which was over 200 km from the nearest colony yet still received relatively heavy predation pressure by pelicans in some years. The maximum recorded distance of which we are aware that American white pelicans have been shown to travel one way from colonies to foraging areas is 305 km (Johnson and Sloan 1978), suggesting that nearly all of the reservoirs and ponds in southern Idaho could be subjected to pelican predation. The maximum distance of travel we observed was 248 km, but in a concurrent related study we also recovered at the Lake Walcott colony a PIT tag implanted into a Yellowstone Cutthroat Trout at Henrys Lake, 278 km away (K. Meyer, unpublished data).

Most hatchery catchable trout fisheries in southern Idaho are within the foraging range of pelicans nesting at colonies other than Lake Walcott and Blackfoot. Pelicans from Gunnison Island at the Great Salt Lake are particularly concerning from a fisheries management perspective due to the large number of pelicans nesting there (8,000 nesting pairs in 2000; King and Anderson 2005) and their close proximity to southern Idaho fisheries. However, we searched Gunnison Island in October 2014 and found only 11 PIT tags from fish stocked in three of our study waters (up to 231 km away; Table 3). We also searched Molly Island and found 20 PIT tags but none were from hatchery catchable trout; rather, they were all from Yellowstone Cutthroat Trout implanted at Henrys Lake (97 km away). Finally, we searched for PIT tags at the Island Park colony and found one PIT tag from a hatchery catchable trout that was stocked in Lake Walcott in 2014. Taken collectively, the number of pelican-consumed PIT tags recovered at the Lake Walcott and Blackfoot colonies (n = 383), compared to the Gunnison, Molly Island, and Island Park colonies (n = 12) suggests that little of the pelican predation occurring in southern Idaho hatchery trout fisheries stems from pelicans breeding at colonies other than Lake Walcott and Blackfoot. This appears so even after factoring in the decline in tag recovery efficiency at greater distances from pelican colonies. Predation from pelicans nesting outside of southern Idaho reduces tag deposition rates at Lake Walcott and Blackfoot, but because our study design accounted for off-colony deposition, our pelican predation estimates incorporated all pelican predation that was occurring, regardless of the origin of any particular bird.

Several of our estimates of total pelican predation may have been biased low. For example, for 4 of our 19 pelican predation estimates, we assumed that pelicans and cormorants were equally successful at foraging on hatchery catchable trout, and that their energetic demands were equivalent. While the relative foraging success of pelicans and cormorants on hatchery catchable trout is unknown, energetic demands are 4-8 times higher for pelicans (Hall 1925; Ferguson et al. 2011) than for cormorants (Hatch and Weseloh 1999; Seefelt and Gillingham 2008). By apportioning tags based solely on bird abundance without adjusting for differing energy requirements, we likely underestimated pelican predation (and consequently

overestimated cormorant predation) unless cormorants were 4-8 times more successful foragers on our stocked fish.

A limitation to our approach was that, if a study water exceeded the foraging range of breeding pelicans, then there would be no chance of recovering a tag at the colony, and pelican predation would consequently be estimated to be 0% regardless of whether or not any hatchery catchable trout were actually eaten by a pelican. For example, at Cascade Reservoir, we estimated that pelican predation was 0% in 2012 and 2013 because no PIT tags were recovered at the Lake Walcott colony. However, at 304 km from Lake Walcott, Cascade Reservoir may indeed have been outside the foraging range of pelicans nesting at Lake Walcott (Johnson and Sloan 1978; Trottier et al. 1980; O'Malley and Evans 1982). Nevertheless, pelicans are generally quite abundant at Cascade Reservoir, with an average of 327 and a maximum of 989 pelicans counted by numerous ground surveys conducted between May and August 2013; similar numbers were present in 2012. Although pelican predation on catchable trout stocked in Cascade Reservoir in 2012 and 2013 may indeed have been 0%, the large number of pelicans inhabiting this water suggests otherwise. We included Cascade Reservoir in our study precisely because it was at or beyond the foraging limit of pelicans from the Lake Walcott colony; if it exceeded that limit, then our study design incorrectly resulted in an estimate of 0% predation by pelicans, unless non-breeding pelicans completely avoided consumption of hatchery catchable trout.

Compared to our estimates of total pelican predation generated for 19 fish stocking events, the estimates of minimum cormorant predation we produced for 7 of the 19 stocking events were less rigorous. For some of the cormorant predation estimates, we assumed that cormorants and pelicans deposited PIT tags at colonies at a rate commensurate with their abundance, but this approach has other assumptions associated with it. One was that energetic requirements were equivalent between birds, though as mentioned above it has been well established that pelican diets greatly exceed cormorant diets in volume, and this may have led to overestimating cormorant predation (and underestimating pelican predation) at waters <50 km from the Lake Walcott colony. Another assumption was that cormorants and pelicans exerted equal predation effort on and had equal capture efficiency of hatchery catchable trout. The fact that catchable trout are quite surface oriented after stocking makes them vulnerable to both birds, but if the diving ability of cormorants allowed them to target hatchery catchable trout more effectively in the months that followed the stocking events, our tag assignment approach may have led to underestimating cormorant predation (and overestimating pelican predation) for some waters. Expanding our estimates of minimum cormorant predation to total cormorant predation required a final assumption that tag recovery efficiencies were equivalent for pelicans and cormorants, when in reality tag recovery efficiencies for cormorants were unknown, and the likelihood of equal tag recovery efficiency curves between pelicans and cormorants is probably low (Hostetter et al. 2015). Despite these weaknesses, the similarity between tag assignments under a variety of approaches (Table 4) suggests that these assumptions likely did not lead to substantial biases in pelican or cormorant predation estimates for waters near the Lake Walcott colony. It was surprising that estimates of minimum cormorant predation exceeded total pelican predation in 3 of 7 instances, suggesting that where cormorants are abundant, their impact on catchable trout stocked in southern Idaho waters may often exceed that of pelicans. In the North Platte River of Wyoming, cormorants and pelicans ate an estimated 80% of the subcatchable-sized trout (10-16 cm long) stocked during the summer, nearly all of which was attributed to cormorants (Derby and Lovvorn 1997).

We recovered only a fraction of the tags we fed directly to pelicans, which highlights the importance of correcting predation estimates for fish consumed by nesting avian predators but

deposited off-colony. Our average tag recovery efficiency of 21% is slightly higher than several recent avian predation studies with similar direct-feeding study designs, all of which found that recovery of tags fed directly to birds was less than 10% (Osterback et al. 2013; Scoppettone et al. 2014; Teuscher et al. 2015). Hostetter et al. (2015) directly fed PIT-tagged hatchery trout to Caspian terns *Hydroprogne caspia*, double-crested cormorants, and California gulls *Larus californicus* and found tag deposition rates on nearby colonies of 71%, 51%, and 15%, respectively, but most of their feeding trials were conducted on birds while they were on or immediately adjacent to the colonies, which likely elevated their tag deposition rates greatly.

A simple explanation for the exponential decline in tag recovery efficiency at greater distances from the pelican colonies is that the increased energy demand of foraging at greater distance from the colony requires a proportional increase in food consumption to meet adult metabolism needs rather than for chick feeding, which would likely reduce tag deposition rates at the colonies. Also likely is that pelicans that forage at waters further from colonies may be more likely to be non-breeders, or as mentioned above, they may be breeding at other colonies, both of which would reduce tag deposition rates at the colonies we studied. Regardless of the causative mechanism, the strength of the relationship between tag recovery efficiency and distance from colonies allowed us to estimate total pelican predation at waters where direct feeding of pelicans was not conducted due to time constraints during our study or because pelican abundance was too low or too variable to create effective direct-feeding conditions. Future efforts to effectively feed cormorants would not only allow minimum cormorant predation estimates to be converted to total predation estimates, but might also allow predictions of total cormorant predation at waters where direct feeding was not conducted.

For several reasons we deem it unlikely that predatory birds other than pelicans and cormorants were responsible for tags that were recovered at these colonies. First, as we have already pointed out, for most of our predation estimates, pelicans were the only avian predator capable of foraging at the distance needed to consume stocked fish and subsequently transport PIT tags to the colonies. Second, although great blue herons *Ardea herodias* were present at the Lake Walcott and Blackfoot colonies, their abundance was a fraction of the abundance of pelicans and cormorants at both colonies, and their maximum foraging distance from colonies has been estimated to be only about 15 km (Parris and Grau 1979; Thompson 1979; Dowd and Flake 1985), precluding them as a meaningful source of predation that was unaccounted for. Third, although ring-billed gulls *Larus delawarensis* and California gulls are also common on both colonies, the foraging range for most gulls is generally less than 25 km (Fasola and Bogliani 1990; Belant et al. 1998), they generally have a non-fish diet (York et al. 2000), and the size of catchable trout we stocked (247 mm on average) is likely too large for these gulls to effectively consume at a meaningful level, all of which precludes them from being an appreciable source of predation as well.

The amount of pelican and cormorant predation the present study demonstrates is occurring on catchable trout stocked in some southern Idaho waters, and a low level of angler catch associated with many of those stocking events, begs the question of whether something can or should be done to either reduce predation or increase angler catch. Considering that IDFG annually stocks about 1.8 million hatchery catchable trout statewide at a cost of about $2.5 million US, maximizing angler catch of these fish by any means possible (including reducing avian predation) is of much import. In terms of stocking strategies, Derby and Lovvorn (1997) suggest that altering the timing of stocking or the size of fish at release may reduce avian predation. Indeed, most catchable trout stocking in southern Idaho occurs from April to June, which closely coincides with peak food requirements for colonial nesting avian predators. However, this also closely coincides with peak angler effort in southern Idaho fisheries, some of

which are largely or entirely supported by catchable stocking. Thus, while stocking at a later date may reduce avian predation, it may also reduce angler catch even further. Moreover, while stocking larger fish (e.g., >350 mm in length) would increase the fish's swimming speed, thereby reducing their vulnerability to pelicans and cormorants, the added costs associated with raising catchables to such a large size may economically preclude such a strategy. In the Blackfoot River drainage, an extensive hazing program to reduce pelican nesting success has been undertaken by IDFG in recent years to help preserve a wild, native population of Yellowstone Cutthroat Trout diminished by pelican predation (Teuscher and Schill 2010; Teuscher et al. 2015). However, hazing strategizes are not logistically feasible at the scale that would be required to protect catchable trout from avian predation in southern Idaho hatchery trout fisheries. A more controversial strategy would be to measurably reduce the numbers of pelicans and cormorants in an area using habitat alteration and/or lethal control, including lethal take as well as egg oiling (to reduce hatching survival). Such strategies have been considered and sometimes implemented for pelicans (Mwema et al. 2010; Teuscher et al. 2015) and cormorants (Belant et al. 2000; Glahn et al. 2000). An alternative strategy is the massive efforts currently underway on the Columbia River to reduce predation by cormorants and Caspian terns on juvenile anadromous salmonids by relocating entire colonies to areas outside of the Columbia River basin (USFWS 2005; NMFS 2010; Lyons et al. 2011). Advantages and disadvantages of each management action must be considered in light of the current status of cormorants and pelicans in North America and their cumulative impacts on economically important fisheries that anglers and policymakers value.

ACKNOWLEDGEMENTS

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TABLES

Table 3. Study water characteristics and distance (km) from nearest colonies to these waters which were stocked with PIT tagged catchable-sized trout that were then exposed to American white pelican and double-crested cormorant predation. Underlined numbers indicate study water × colony combinations where cormorants may also have contributed to consumption and deposition of PIT tags (based on maximum foraging range). Bold numbers indicate study water × colony combinations where PIT tag recoveries actually occurred at colonies. Study water numbers are used for geographical orientation in Figure 1.

Study waters	Water size (ha)	Number of fish species present	Number of catchable trout annually stocked	Nearest pelican colonies (km)				
				Yellowstone National Park	Island Park Reservoir[a]	Blackfoot Reservoir	Lake Walcott	Great Salt Lake
1 Cascade Reservoir	10,994	11	62,000	459	363	412	304	448
2 CJ Strike Reservoir	3,035	23	102,000	483	385	354	*201*	313
3 Riley Creek Pond	7	0	17,000	415	323	274	*118*	*231*
4 Filer Pond	1	0	7,600	403	314	252	*95*	202
5 Magic Reservoir	1,569	8	6,000	366	268	231	*111*	230
6 Freedom Park Pond	1	0	1,000	346	272	181	*32*	154
7 Rupert Gun Club Pond	4	0	900	347	271	181	*32*	156
8 Lake Walcott	3,335	11	40,000	315	*248*	148	*0*	152
9 American Falls Reservoir	22,369	11	51,000	259	199	95	*56*	*170*
10 Chesterfield Reservoir	504	7	57,000	213	174	*27*	*119*	*187*
11 Foster Reservoir	52	5	5,900	275	252	84	140	111
12 Glendale Reservoir	82	6	9,200	275	253	83	141	113

[a]Pelican nesting is annually attempted here but successful offspring are rarely produced.

Table 4. Summary of PIT tags recovered from stocked catchable trout and assigned to either American white pelican or double-crested cormorant predation based on three possible tag assignment approaches. See text for more details regarding each approach.

Hatchery trout stocking water	Year	Number of PIT-tagged fish: Stocked	Number of PIT-tagged fish: Recovered at nearest colony	Recovered tags assigned to pelicans or cormorants assuming tags were deposited: Proportional to pelican and cormorant abundance at colonies		Proportional to abundance and further adjusted for energetics of pelicans and cormorants		According to tag recovery location in proximity to pelican and cormorant nesting and loafing areas	
				Pelicans	Cormorants	Pelicans	Cormorants	Pelicans	Cormorants
Freedom Park Pond	2013	100	19	16	3	18	1	12	7
Rupert Gun Club Pond	2013	99	18	16	2	18	0	14	4
Lake Walcott	2013	397	82	65	17	79	3	54	28
Lake Walcott	2014	208	63	41	22	58	5	39	24

Table 5.   Summary of American white pelican feeding events at various southern Idaho waters, and subsequent estimates of tag recovery efficiency.

Water	Year	Distance to nearest colony (km)	PIT-tagged fish fed to pelicans:		Fed tag recovery efficiency
			Number fed	Number recovered at nearest colony	
Lake Walcott	2013	0	91	44	0.48
Lake Walcott	2014	0	81	53	0.65
Chesterfield Reservoir	2013	27	80	19	0.24
American Falls Reservoir	2013	56	101	9	0.09
American Falls Reservoir	2014	56	83	12	0.14
Riley Creek Pond	2012	118	64	16	0.25
Riley Creek Pond	2013	118	39	24	0.62
Riley Creek Pond	2014	118	10	2	0.20
CJ Strike Reservoir	2012	201	100	6	0.06
CJ Strike Reservoir	2013	201	100	2	0.02
CJ Strike Reservoir	2014	201	95	2	0.02
Cascade Reservoir	2012	304	104	0	0.00
Cascade Reservoir	2013	304	125	0	0.00
Total			1,073	189	

Table 6.   Number of PIT tags implanted in catchable trout stocked in study waters that were recovered at colonies and were known or assumed to have been consumed by pelicans; resulting estimates of pelican predation are also shown, as are estimates of angler catch of these same hatchery fish.

Water	Year	Distance to nearest colony (km)	PIT-tagged trout: Initially stocked	PIT-tagged trout: Recovered at nearest colony	Pelican predation Estimate	Pelican predation 90% CI	Angler catch Estimate	Angler catch 90% CI
Waters outside foraging range of cormorants								
American Falls Reservoir	2013	56	396	11	0.31	0.22	0.00	-
American Falls Reservoir	2014	56	398	17	0.30	0.17	0.07	0.03
Glendale Reservoir	2013	83	399	0	0.00	-	0.25	0.07
Foster Reservoir	2013	84	293	0	0.00	-	0.30	0.07
Filer Pond	2012	95	100	3	0.23[a]	NA	0.68	0.18
Magic Reservoir	2014	111	449	4	0.09[a]	NA	0.06	0.04
Riley Creek Pond	2012	118	100	2	0.08	0.09	0.82	0.20
Riley Creek Pond	2013	118	100	4	0.07	0.05	0.06	0.07
Riley Creek Pond	2014	118	99	3	0.15	0.16	0.77	0.20
CJ Strike Reservoir	2012	201	399	1	0.04	0.07	0.32	0.07
CJ Strike Reservoir	2013	201	400	2	0.25	0.32	0.09	0.05
CJ Strike Reservoir	2014	201	400	4	0.48	0.67	0.14	0.05
Cascade Reservoir	2012	304	393	0	0.00	-	0.02	0.02
Cascade Reservoir	2013	304	450	0	0.00	-	0.09	0.03
Waters within foraging range of cormorants								
Lake Walcott	2013	0	397	65[b]	0.34	0.09	0.00	-
Lake Walcott	2014	0	208	41[b]	0.30	0.09	0.04	0.03
Chesterfield Reservoir	2013	27	385	5[b]	0.05	0.04	0.02	0.02
Rupert Gun Club Pond	2013	32	99	16[b]	0.37[a]	NA	0.00	-
Freedom Park Pond	2013	32	100	16[b]	0.37[a]	NA	0.31	0.16
Total			5,565	194				

[a] Pelican predation estimate not based on pelican fed tags but rather on equation from Figure 2.
[b] PIT tags assigned based on results presented in Table 2.

Table 7. Number of PIT tags implanted in catchable trout stocked in study waters that were recovered at colonies or cormorant loafing and roosting areas, and were known or assumed to have been consumed by cormorants; resulting estimates of cormorant predation are also shown.

| Hatchery trout stocking water	PIT tag recovery location(s)	Year	Distance to nearest colony (km)	Number of PIT-tagged fish stocked	Number of PIT tags recovered: At nearest colony and assigned to cormorants	Number of PIT tags recovered: At cormorant loafing/roosting areas where fish were stocked	Cormorant predation estimates Assuming 100% tag recovery rate	Cormorant predation estimates Assuming equivalent tag-recovery-by-distance rates for pelicans and cormorants
Lake Walcott	Lake Walcott colony	2013	0	397	17	-	0.04	0.05
Lake Walcott	Lake Walcott colony	2014	0	208	22	-	0.11	0.13
Chesterfield Reservoir	Blackfoot colony and Chesterfield Reservoir	2013	27	385	96	52	0.38	0.69
Rupert Gun Club Pond	Lake Walcott colony	2013	32	99	2	-	0.02	0.05
Freedom Park Pond	Lake Walcott colony	2013	32	100	3	-	0.03	0.07
Glendale Reservoir	Glendale Reservoir	2013	83	399	0	20	0.05	0.06
Foster Reservoir	Foster Reservoir	2013	84	293	0	99	0.34	0.42

FIGURES
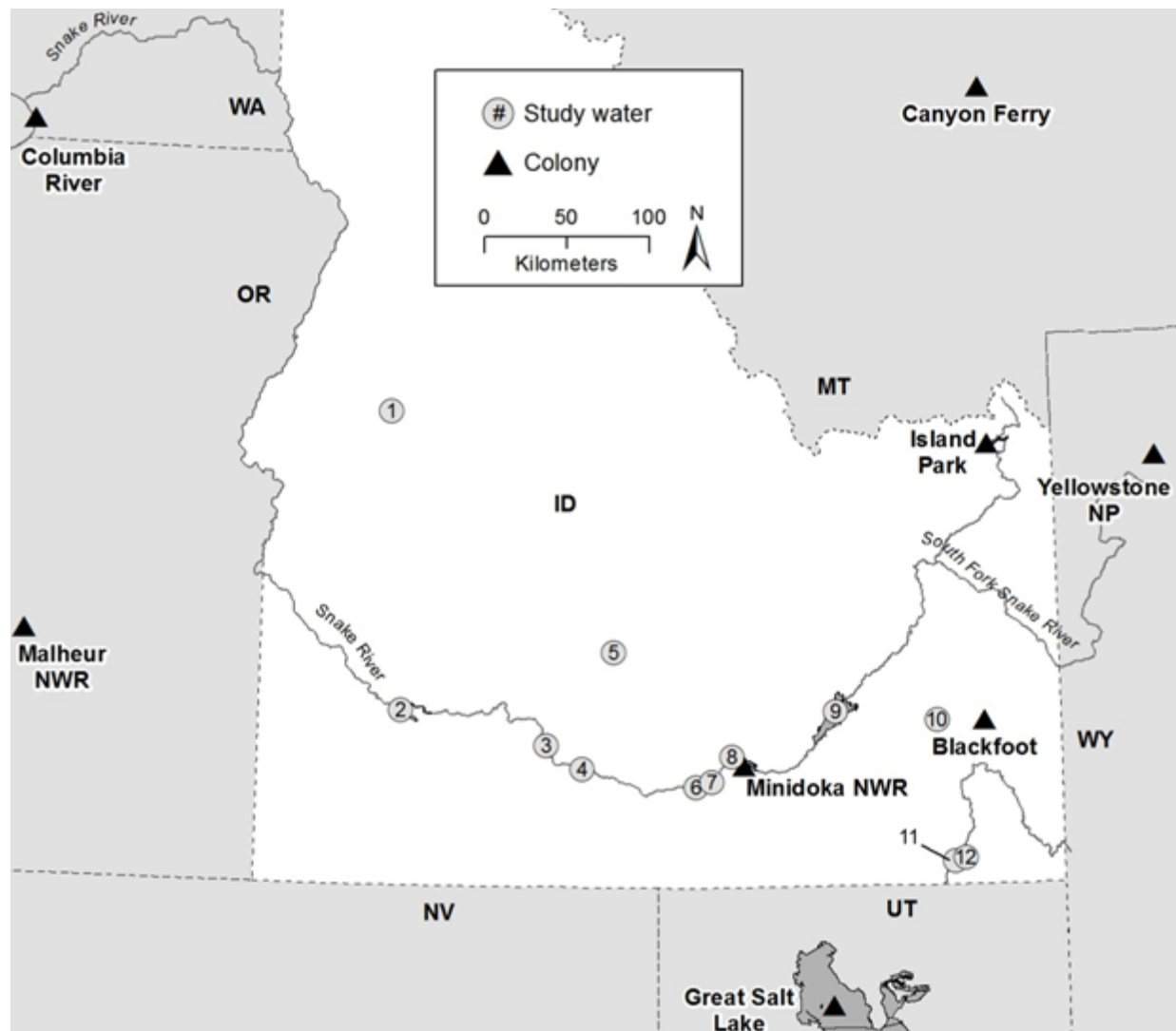
Figure 4.   American White Pelican colonies nearest to southern Idaho study waters where pelican and cormorant predation of hatchery catchable trout was evaluated. Study water numbers correspond to Table 3.
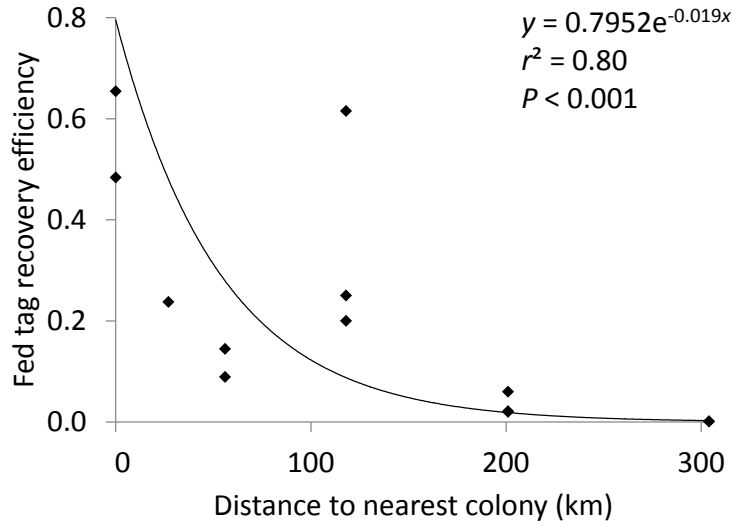
Figure 5.   Relationship between a study waters' distance from the nearest American white pelican colony and the recovery efficiency (at the nearest colony) of PIT tags implanted in hatchery catchable trout and fed directly to pelicans at that study water. The line and equation depict an exponential relationship fitted to the data.
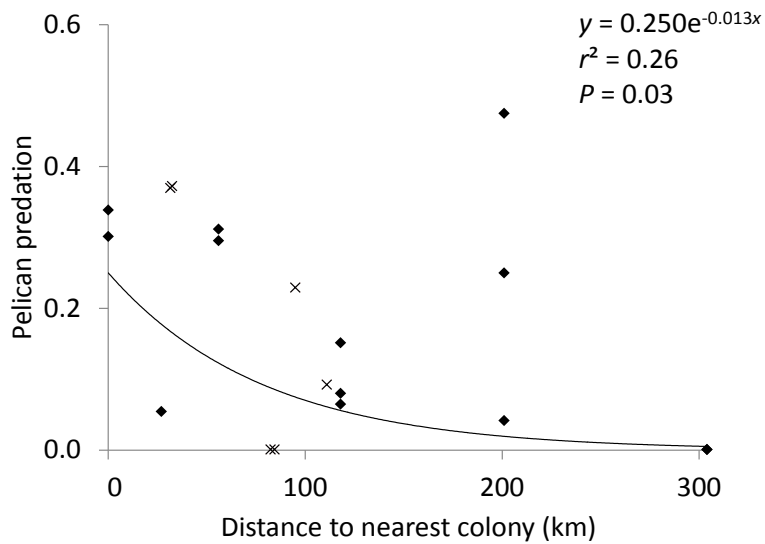

Figure 6.   Relationship between a study waters' distance to the nearest American white pelican colony and the pelican predation rate on catchable trout stocked at that water. Predation rates for the waters labeled with an "x" were predicted (rather than estimated directly) based on the relationship in Figure 5. The line and equation depict an exponential relationship fitted to the data.
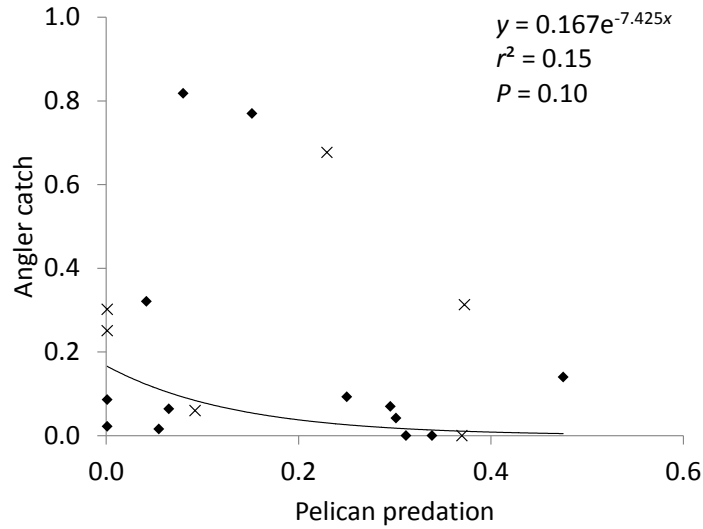
Figure 7.   Relationship between estimates of American white pelican predation and angler catch of catchable trout stocked in southern Idaho study waters. Predation rates for the waters labeled with an "x" were predicted (rather than estimated directly) based on the relationship in Figure 5. The line and equation depict an exponential relationship fitted to the data.
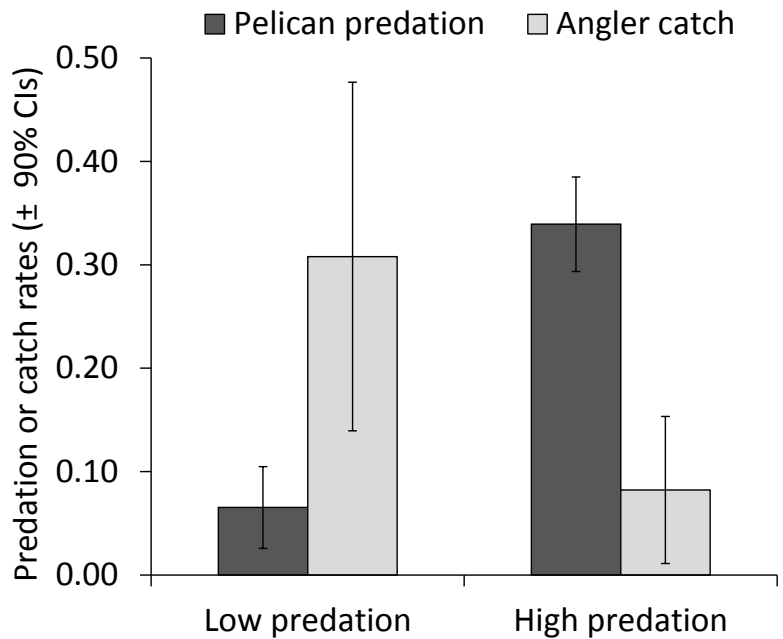

Figure 8.   Mean rates of American white pelican predation and angler catch [± 90% confidence intervals (CIs)] of catchable trout stocked in southern Idaho waters, grouped into stocking events where pelican predation was either high (i.e., ≥25%) or low (<25%).

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CHAPTER 3: RETENTION RATES OF PIT AND VIE TAGS AND MAXILLARY CLIPS IN WILD TROUT OF SPAWNING SIZE

ABSTRACT

Tagging fish is a common method to identify individuals or groups of fish, but the utility of tags can be compromised if tags are shed or deteriorate over time. We evaluated retention rates for three injection locations of passive integrated transponder (PIT) tags and for visual implant elastomer (VIE) tags in stream-dwelling Cutthroat Trout *Oncorhynchus clarkii* and Rainbow Trout *O. mykiss* of spawning size. In three streams in southeastern Idaho, 2,893 fish ≥150 mm (total length) were marked with VIE in the lower jaw, and PIT tagged in the (1) body cavity, (2) muscle tissue immediately posterior to the cleithrum, or (3) muscle tissue immediately ventral to the dorsal fin. Maxillary clips provided additional marks to identify study fish and year tagged. Retention of PIT tags was highest in the dorsal musculature location for one year at large (93%), followed by the cleithrum (84%), and the body cavity (72%). Length and sex impacted retention of PIT tags placed in the body cavity. Fish under 200 mm had 100% retention, whereas in fish ≥200 mm, females retained tags at a much lower rate compared to males (59% and 89%, respectively). The retention rates of cleithrum and dorsal musculature locations were unaffected by sex or larger size. One-year retention rates of VIE tags and maxillary clips were high (96% and 93%, respectively). The likelihood of a PIT tag remaining in the fillet tissue was highest for the dorsal musculature location (67%), followed by cleithrum (60%) and body cavity (4%) locations. Using PIT tags in muscle should be approached with caution in exploitable fish populations; however, the dorsal musculature location does increase retention rates in populations where consumption is not a concern. Batch tags of VIE and physical clip combinations provide adequate identifying ability to the subpopulation level for the at least 2 years.

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INTRODUCTION

Tags placed in fish have broad application as a tool for fisheries managers to attempt to quantify population dynamics of fish in the wild. A foremost challenge of any tag type is the ability to non-lethally and yet accurately identify individuals over time, at least to the subpopulation level. Many options exist to meet this need; however, not all withstand the test of extended retention time in the wild, requiring the validation of such rates prior to data analysis. Particularly of interest are those tagging techniques retained throughout the life cycle of the fish and capable of withstanding biological changes or behaviors exhibited by sexually mature fish without tag shedding or degradation.

Tag retention varies with fish species, tag type, and tagging location. As elaborated by Nielson (1992), some characteristics of the 'perfect mark' are: permanence throughout the fish's life cycle, ease of application, high likelihood of being observed by the untrained eye upon recapture, and low cost. Knowing that tag retention rates are affected by many factors is intrinsic to the ability to accurately describe the population under study when using tags. How effective tag types are as long-term identification techniques is greatly influenced by where the tag is applied and what effect application (location) might have on retention through developmental stages of the fish.

Three tagging systems - passive integrated transponder (PIT) tags, visible implant elastomer (VIE) tags, and maxillary clips - are frequently used to identify fish (Bruyndoncx et al. 2002; Hopko et al. 2010; Younk et al. 2010; Soula et al. 2012). Multiple studies have evaluated retention rates for PIT and VIE tags in trout (Prentice et al 1990; Bonneau et al. 1995; Hale and Gray 1998; Close 2000; Josephson and Robinson 2008; Knudsen et al. 2009); however, little information is available for long-term observations of tag retention in mature wild fish in streams.

Passive integrated transponder tags, while providing a unique identifier within a population, have demonstrated moderate retention rates in stream-dwelling trout, possibly due to natural spawning activity when the tag is forcefully ejected with eggs by females (Bateman et al. 2009; Meyer et al. 2011). These tags, when inserted in the peritoneal (body) cavity of trout, have been shown to be shed at a higher rate as fish size increases. Of the few studies addressing the effect of stream-dwelling trout size and implied sexual maturity upon PIT tag retention rates, Bateman et al. (2009) found that smaller (<140 mm) Westslope Cutthroat Trout *Oncorhynchus clarkii lewisi* residing in headwaters were 1.4 times as likely to retain PIT tags than larger (>174 mm) fish, though they were unable to identify to sex the population of fish marked. For stream-dwelling Redband Trout *O. Mykiss* in southern Idaho, Meyer et al. (2011) demonstrated a difference in long-term PIT tag retention rate of the body cavity tagging location between spawning size females (67%) and males (90%), suggesting sexual maturation and ensuing spawning activity may increase the expulsion rate. When assessing two-month retention rates of PIT tags in Brook Trout *Salvelinus fontinalis* and Brown Trout *Salmo trutta*, Dieterman and Hoxmeier (2009) found that retention in the body cavity was 70% and 56% respectively, after the fall spawning period. This same study found dorsal musculature PIT tag retention, over a two-month period not including a spawning event, was 100% for Brook Trout and 95% for Brown Trout.

The retention of VIE tags has been studied in many species and environments (Dewey and Zigler 1996; Haines et al. 1998; Goldsmith et al. 2003, Leblanc and Noakes 2012) yet the period of time involved has generally been too short to fully assess whether the tags were retained or degraded over time. Two studies found that VIE tags became fragmented in as little as 30-45 days after application (Astorga et al. 2005; Soula et al. 2012), however for long-term

observations, Fitzgerald et al. (2004) found 92% retention in net pen Atlantic Salmon *Salmo salar* after 16 months, while Willis et al. (2001) were able to identify ocean run adult Snapper *Pagrus auratus* three years post-tagging.

Historically, the first recorded marking of fish for science was noted by Izaak Walton in *The Compleat Angler* (1653), who described Sir Francis Bacon tying colored yarn onto individual fish, allowing the identification and study of fish in their natural environments. This technique was a precursor to methods later employed, specifically minor alterations to appendages or bony structures of the study subject. Physical marks became much more commonly used, evidenced by observations that the removal of a fin or bone from a salmon or trout had been evaluated repeatedly in the 20th century (Krumholz 1944). Clipping a portion of the maxillary bone is currently a routinely utilized method to provide an easily read, external mark on fish. Stauffer and Hansen (1969) found a 91% retention rate in hatchery reared Rainbow Trout *Oncorhynchus spp.* after two years, and Weber and Wahle (1969) had similar retention (94%) in wild run Sockeye Salmon *O. nerka* at large four years; however, no study has yet to evaluate the effectiveness of this mark in wild trout in streams as a tool for inland fisheries management.

This study was initiated due to the concern over the lack of information regarding internal and external tag retention in wild trout of spawning size as well as the impact different tagging technologies might have on the likelihood an angler may encounter a tag when preparing a fish for consumption. The objective of this study was to evaluate long-term (one- and two-year) PIT tag retention rates, differentiated by injection location, of mature-sized salmonids dwelling in a stream environment that would likely have gone through at least one spawning event. We attempted to assess whether alternate PIT tagging locations within the body can be used to maintain high retention rates through the growth and likely impending sexual maturity of wild trout, as well as determine the terminal location of PIT tags when injected into fish tissues. When study design allows, less-expensive batch marks created by VIE tags and maxillary clips may be useful. In this study, both types of secondary marks were applied concurrently with a PIT tag to determine retention rates of these tag types in spawning-sized fish. This study was implemented over two years, allowing us to examine retention rates over an extended period of time.

METHODS

In July and September of 2012, Yellowstone Cutthroat Trout (*O. clarkii bouvieri;* YCT), Rainbow Trout (RBT), and Rainbow x Cutthroat hybrids (RbtHYB) were collected from three streams (Fall, Rainey, and Badger creeks) in southeastern Idaho (Table 8). These streams were selected due to effectively no chance of angler exploitation as these populations are protected by regulations and limited access. This limited harvest/access was necessary to avoid putting anglers at risk of potentially consuming a tag. Additionally, the presence of large numbers of mature, resident (i.e., non-migratory) trout in all three streams would help maximize recapture potential.

Two backpack electrofishing units were used to conduct a single upstream pass in all efforts on Fall and Rainey creeks as well as the initial effort on Badger Creek. A raft electrofishing configuration was used for final recapture effort on Badger Creek. Fish ≥150 mm total length (TL) were held in 19-liter buckets while electrofishing. At periodic intervals, fish were anesthetized in an immersion bath of 15-20 ppm isoeugenol (AQUI-S®, New Zealand). Once

measured to the nearest mm (TL), fish were tagged, held in freshwater until recovered, and released near the area from which they were collected.

Three PIT tag injection site locations were used for evaluation of retention rates. A full-duplex PIT tag (Biomark, 12 mm long, 2 mm diameter, uncoated glass) was injected using a 12-gauge stainless steel veterinary hypodermic needle and modified syringe (Prentice et al. 1990) in one of three locations. PIT tag locations included (1) body cavity (injected into the peritoneal cavity, anterior to the pelvic girdle, offset from the dorsoventral axis; BC); (2) cleithrum (inserted subcutaneously, dorsoventrally, directly posterior and parallel to cleithrum; CL); and (3) dorsal musculature (injected post-anteriorly, shallow intramuscular depth, parallel to and directly ventral of dorsal fin ray process; DM) (Figure 9, Figure 10). PIT tags were read by a portable PIT tag reader once injected into the fish, prior to release. Tagging wounds were not sealed by any surgical glue or closure.

One injection site location was evaluated for VIE (Northwest Marine Technologies, Shaw Island, Washington). A biologically non-reactive elastomer, VIE was injected subcutaneously into the minimally pigmented tissue against and parallel to the bony structure of the lower mandible (Figure 11). Using a 28-gauge needle on a .33 cc syringe in a manual VIE elastomer injector, the needle tip was inserted and a volume of elastomer injected as the needle was withdrawn, stopping just prior to the needle bezel exiting the tissue. Excess elastomer was removed by gently wiping the injection site with a fingertip to avoid leaving strands of pigment to harden outside of the dermis in an attempt to decrease the likelihood of shedding. To differentiate between tagging event years, a corresponding year-specific color of VIE was used.

An additional secondary physical mark in the form of maxillary clip combinations (severed posterior of the dermal membrane connecting upper to lower jaw, either right, left, or both sides) was used to act as an indication of a PIT tag location should the tag be shed (Siepker et al. 2012). We assumed removal of either one or both tips of the maxillary bone would not result in substantial mortality, similar to observations made by Stauffer and Hansen (1969).

In September and October of 2013, using the same backpack electrofishing method mentioned above (single pass), fish were recollected and interrogated for all marking types and locations. Fish were scanned for PIT tags using a portable PIT tag reader, and VIE marks and maxillary clips were examined using incidental light. For this first collection effort, any presence of elastomer material was considered a mark, but partial or fragmented conditions were not noted. Fish identified as recaptures had marks recorded and were released. New specimens were PIT tagged, maxillary clipped, and marked with the second year VIE color. Recaptured fish that had shed a PIT tag were not retagged with a new PIT tag or VIE color.

Study sites were revisited in August and October of 2014 and all fish found to bear a mark of any kind were sacrificed, brought into the lab, and frozen until examined. In the laboratory, fish were partially thawed, scanned for a PIT tag using a PIT tag reader (Destron Fearing PTS FS2001F ISO ring-type), and x-rayed along the lateral plane using a portable digital x-ray machine (Sound Eklin). Presence and identity of a PIT tag was noted where possible. Samples were then rescreened for PIT tags on a second reader (Biomark IS0-601 handheld) and lengths, weights, and sex determined using the approach of Downs et al. (1997). The presence of all external marks and mark integrity (fragmentation) of VIE was noted at this time, using VIE handheld UV light if no mark was seen by eye. Different from the 2013 recollections, fragmentation of VIE marks was noted in fish of this second collection effort.

Study fish were filleted parallel to the spine (severing rib cage bones) to produce two fillets and a carcass (consisting of head, spine, internal organs, tail, and residual meat). The individual performing the filleting had no awareness of marks that might indicate a possible PIT tag location as mark combinations were complex when considered across both years, requiring a key to discern. Each fillet and carcass was then scanned for a PIT tag on a third reader (handheld). Determination of the terminal location (fillet or carcass) of the PIT tag was noted.

Long-term retention rates by injection location were calculated as the proportion of recaptured fish that had retained a tag relative to the number of all recaptured fish that presented a secondary visual mark (VIE and max clip) indicating that it was part of our study. Ninety-five percent confidence intervals (CIs) were calculated for the retention rate of each tag location and mark type, and differences in proportions were evaluated for statistical significance based on non-overlapping confidence intervals (Fleiss 1981). For the sake of analysis, all species and streams were considered together when calculating retention rates.

RESULTS

Collectively 891 YCT and 579 RBT were marked in 2012 and 811 YCT and 605 RBT in 2013, for a total of 2,886 fish tagged over the two years of the study. The total length of fish tagged ranged from 150-415 mm (mean = 210 mm, SD = 45; Table 9). Of those fish at-large for one year, 83 ± 2% retained PIT tags, 96 ± 1% retained a VIE tag, and 93 ± 3% retained an easily identifiable maxillary clip (Table 10). PIT tag retention rate was highest for the DM (93 ± 6%) location when compared to CL or BC (84 ± 5% and 72 ± 3%, respectively, Figure 12).

PIT tag retention rates varied widely depending on injection location, fish length, and sex for fish collected in the second recapture effort. Tags injected into the body cavity were retained well (100%) for both sexes when recapture length was <200 mm; however, retention rates dropped to 59% for females ≥200 m when compared to 89% in males of the same size class (Figure 12). Both CL and DM retention rates appeared unaffected by sex; however, both locations had lower retention rates in trout <200 mm (Table 10). Due to insufficient sample sizes for fish at large two years, we were unable to calculate a two-year PIT tag retention rate.

Using x-rays to determine terminal locations of PIT tags in fish tissue was very effective, with PIT tags being accurately identified in 100% of the digital images created. Of the 293 digital images of fish that indicated a PIT tag present, PIT tag detectors were able to identify 288 of them, suggesting that in five fish there were tags that no longer communicated electronically (1.7% tag failure) with any of the three tag reading devices employed.

The filleting process allowed for consideration of the potential for a PIT tag to be encountered by an angler when a trout is being prepared for consumption. Of those tags injected into the BC location, only 4% (± 1, $n = 72$) were found in the fillet, whereas both CL and DM were found predominantly in fillet tissue (60% ± 9, $n = 82$ and 67% ± 9, $n = 84$, respectively) (Figure 13).

Secondary marks provided an extra ability to evaluate these PIT tag retention rates and were retained at equal if not better rates, independent of size or sex. We were able to calculate both year zero-to-one and one-to-two VIE retention rates (96% ± 2, $n = 524$; 100%, $n = 36$). Fragmentation occurred in 10% of those VIE tags at large one year and 19% of those at large two years. Maxillary clip retention was also very high for year zero-to-one (93% 93% ± 3, $n = 229$) and year one-to-two (100% $n = 105$).

DISCUSSION

Ease of application, tag retention, and duration of the study are some elements to be considered when choosing an appropriate tag type in fisheries studies. In this study on stream dwelling trout of spawning size, fish length had a measurable impact on PIT tag retention. For fish <200mm, PIT tag retention was highest in the body cavity when compared to both cleithrum and dorsal musculature locations. This may be explained by the fact that these fish were physically smaller when being tagged in muscle tissue, which may suggest difficulty in assimilating the foreign body (i.e., the tag), or that a developmental aspect (e.g., sexual maturity) had yet to occur. However, as body size increased, this trend reversed, with cleithrum and dorsal musculature locations performing more effectively in larger fish than the body cavity location.

For PIT tags to remain effective throughout the life cycle of trout, a location other than the body cavity would be ideal. While both the cleithrum and dorsal musculature locations appear to be unaffected by sexual maturation factors causing higher PIT tag shedding, these two locations come with an additional consideration of angler interaction upon harvest (e.g., Dieterman and Hoxmeier 2009). Siepker et al. (2012) investigated using Plastic Infusion Process (PIP) PIT tags in response to food safety hazards created by consumption of a glass PIT tag. Working with largemouth bass, they found that tag retention, when injected in either the peritoneal cavity or the dorsal musculature, was 100% over one year but given the larger tag size, the utility of this type of tag might be limited to fish of larger size. McKenzie et al. (2006) noted that in snapper *Pagrus auratus*, PIP tags could be considered reliable for as long as two years.

A valid concern when placing a tag within a fish that could be exploited by anglers is the likelihood that the tag might be ingested by the angler. By filleting the fish as an angler might, we attempted to mimic this scenario and measure the incidence of a tag remaining in a fillet that is being prepared for consumption. We found the majority of cleithrum and dorsal musculature PIT tags were retained in the fillet tissue, compared to very few of the body cavity tags. This clearly demonstrates that cleithrum and dorsal musculature PIT tag locations should only be used on fish that are not likely to be consumed by humans. Evaluation of x-ray images suggests that the cleithrum location is more vulnerable to possible consumption than we had anticipated because, upon visual inspection of the x-rays, cleithrum tags appeared to have migrated from the initial injection location (close to the cleithrum bone) further into the fillet tissue greater than 50% of the time. Tag migration in tissue is a phenomenon noted in many species of animals (Gibbons and Andrew 2004). It was our hope in picking these alternative injection locations (cleithrum and dorsal musculature) that the tag might remain in the residual tissue left behind on the carcass after filleting. However, the fact that, for these injection locations, the tags were found in fillet tissue the majority of the time suggests these locations are not suitable for exploitable populations. Both of these sites (cleithrum and dorsal musculature) would be usable on fish protected by regulation or inaccessibility of access by anglers due to locale, though caution should be taken to minimize study designs that might allow study fish to move into areas accessible to consumers. It is interesting to note that the body cavity location, considered safe from consumption, may still carry a slight risk.

Unexpectedly, we realized a small PIT tag failure rate (tags that no longer communicated with any PIT tag reader). The presence of failed PIT tags has a minor but measurable impact on retention rates by artificially deflating these values, as these tags were

not actually shed. Although a PIT tag is expected to last the lifetime of the animal in which it is injected (Sam Breidenbach, Biomark, personal communication), there are currently few studies of fish that mention PIT tag failure rates (Parker and Rankin 2003; Daugherty and Buckmeier 2009). In this study, the tags that failed had been implanted in all three evaluated injection locations, suggesting that no specific injection location was more prone to tag failure. Additionally, these fish, as part of the laboratory examination process, were interrogated by three different PIT tag readers, physically filleted, and visually inspected for a tag. These failed tags were not found in any of these opportunities for discovery, suggesting that without prior knowledge of the presence of a tag, these tags could easily be missed in the tissue of a trout.

A VIE tag is inexpensive, easy to apply, easily observed by eye, and requires little special equipment. However, in some studies, injection location, size at tagging, and length of time at large outlasted mark longevity. In Muskellunge *Esox masquinongy* marked as fingerlings, Younk et al. (2010) found VIE to be a viable mark (100% retention) after 176 days, but rates dropped drastically 2-6 years later. Fitzgerald et al. 2004 found that adipose eye tissue VIE tags were retained at a higher rate than jaw tags (88% compared to 72%) in Atlantic Salmon smolts held in net pens for 28 months. Being able to easily see a mark is vital to its usefulness. Curtis (2006) noted that levels of tag visibility, color determination, and injection location with respect to pigmentation can all affect the utility of VIE tags. Close and Jones (2002) described some possible limitations to VIE retention in fingerling Rainbow Trout as the development of pigmentation and tissue overgrowth affect visibility, and the occurrence of mark fragmentation. While studying Brook Trout in both hatchery and lake environments, Josephson and Robinson (2008) reported 50-72% VIE tag retention rate when observed under outdoor sunlight conditions approximately 400 d post-tagging, declining to 0% at 959 d. The VIE colors used in this study (red in 2012, blue in 2013) are considered contrasting colors and of highest visibility (Astorga et al. 2005) in non-colorblind individuals. We found that for one year, VIE detection was 96% and in this study we were able to estimate a one- to two-year retention rate for those fish recaptured in both years. Of the 37 fish that could be identified as being recaptured twice, VIE was retained at 100%, suggesting most loss occurred in the first year, and 98% of those two-year-old tags were able to be seen without aid of an additional UV light. A concern for studies using VIE tags would be to quantify fragmentation of the elastomer in the tissue, which, should shedding continue to occur over time, could result in fewer VIE marks retained. In this study, rates of fragmentation increased with time at large, which could have a potential impact on studies longer than two years. Bangs et al (2013), working on Oregon chub *Oregonichthys crameri*, did not assume perfect detection (i.e., an intact mark) past 150 days due to fragmentation.

The maxillary clip is a simple form of physical mark and its impact on growth and survival has been studied by many (Stauffer and Hansen 1969; Weber and Wahle 1969; Gjerde and Refstie 1988). Generally considered a benign mark, removal of bone can occasionally result in deformities that make later mark identification difficult. Bonham (1968) noted that along with deformations, exposure to fishing can complicate maxillary clip identification, though he reported 87% retention in Chinook Salmon *O. tshawytscha* marked as fingerlings having been at large for two years. Our results suggest that maxillary clip retention in our study streams may be high, regardless of fishing pressure, due to the fish having been marked at a larger size.

A common assumption when using tags is that there is no tag loss (Pollack et al 2001); however, in reality it does occur and is typically present in two forms; immediate tag loss following soon after the tag is implanted, or continuous or chronic tag loss that occurs throughout the length of time the tag is utilized (Beverton and Holt 1993, Gaertner et al 2004). Maxillary clip regrowth, should it occur, appears to happen within the first year, which, while not

instantaneous, suggests that chronic loss does not occur after initial healing. VIE fragmentation may be a chronic process, occurring throughout the lifetime of the tag. PIT tags are affected by compounding factors of tag placement, body size, and sexual maturation. When used in fish, PIT tags suffer initial, or immediate, loss rates complicated by the ratio of fish size to tag size. The PIT tag loss type progresses to chronic loss, specifically in the case of body cavity placed tags, due to the delayed effects of sexual maturation and spawning behavior.

RECOMMENDATIONS

PIT tags, when injected into the body cavity of stream dwelling trout, are an effective method to identify individuals, up to spawning size. After that life stage has been reached, however, retention rates are compromised, especially for sexually mature females. Some alternative injection site retention rates are equivalent to prespawning body cavity retention rates; however, the application of these locations must take angler exploitation opportunities into consideration. Currently the only responsible PIT tag location to use on exploitable stream dwelling trout populations is the body cavity, which itself still carries a modicum of risk for anglers possibly encountering a tag when preparing a fish for consumption.

Injectable elastomer products are useful to identify, to the group level, fish populations of all size ranges. This tool is inexpensive, easily discerned in the field, available in a multitude of colors, and can be applied in many locations on a fish to provide an opportunity for a wide range of distinctive marks and mark combinations. Retention was excellent for the length of this study so could therefore be recommended for use in evaluations within a two-year time frame.

Maxillary clips provide a very effective, easily identified, and durable mark. While limited in scope (three clip options available), when used in conjunction with VIE technology, subgroup delineations may be made with confidence, at least for the length of this study.

Alternative technologies exist with respect to PIT tags, providing a perceived level of safety not possible with glass PIT tags. Plastic encased PIT tags (PIP) are available both locally and internationally and are used in muscle tissue in exploitable fish populations (Mountain whitefish *Prosopium williamsoni*, Rainbow Trout, Walleye *Sander vitreus*) in Canada, and are considered an angler friendly alternative and as 'food safe' in that country (Dustin Ford, Golder Associates, B.C., Canada, personal communication). These biologists found that PIP tags were less prone to breakage, easier to apply, have a wider read range and were easier for anglers to identify in the muscle tissue. Regardless of perceived safety, appropriate agency/bureau approval should be sought prior to using any tool that comes with potential consumption risk. Another tag type that would potentially avoid these risks are PIT tags encased in T-bar anchor tag material (Hallprint, Oregon RFID); however, more work to evaluate tag loss in these two technologies would be recommended prior to use.

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TABLES

Table 8.   Numbers of wild trout sampled and recaptured with days at large (including at least one spawning event) at three different streams in Southeast Idaho.

Stream	Run Mark	Recap	2nd mark	Final recap	Days at large by spawn event 1st	2nd
Badger Creek	07/23/12	10/02/13		08/26/14	436	764
			10/02/13	08/26/14	328	
Fall Creek	09/25/12	09/23/13		08/25/14	363	699
			09/23/13	08/25/14	336	
Rainey Creek	07/17/12	09/24/13		08/25/14	434	769
			09/24/13	08/25/14	335	

Table 9.   Numbers and average length (mm, Standard Deviation [SD]) at tagging and recapture of wild trout in three southeast Idaho streams that had the opportunity to experience at least one spawning event.

Sample year	PIT location	Tag		Recapture	
		n	Ave. length (mm)	n	Ave. length (mm)
2012	BC	493	208 (±43)	-	-
	CL	487	212 (±47)	-	-
	DM	490	211 (±44)	-	-
2013	BC	470	206 (±37)	110	247 (± 32)
	CL	467	207 (±37)	91	247 (± 41)
	DM	479	206 (±39)	93	243 (± 31)
2014	BC	-	-	95	231 (± 30)
	CL	-	-	101	229 (± 35)
	DM	-	-	93	228 (± 30)

Table 10.   Numbers of wild trout by sex, recaptured post tagging and one spawning event, having received a passive integrated transponder tag in one of three injection locations, and secondary marks, with corresponding tag retention rates (percent retained, 90% confidence interval [CI]) in three southeast Idaho streams.

| Type/location	All fish			Females			Males		
	Recap	With tag	Retention (%, CI)	Recap	With tag	Retention (%, CI)	Recap	With tag	Retention (%, CI)
PIT/body cavity	180	129	72 ± 6	32	19	59 ± 14	39	36	90 ± 8
PIT/cleithrum	171	145	84 ± 5	37	30	76 ± 12	39	31	77 ± 11
PIT/dorsal musculature	171	160	93 ± 3	38	34	89 ± 8	37	34	89 ± 8
Visible injectable elastomer/jaw	524	503	96 ± 2	139	133	96 ± 3	142	136	97 ± 3
Maxillary clip	229	213	93 ± 3	105	98	93 ± 4	116	108	93 ± 4

FIGURES
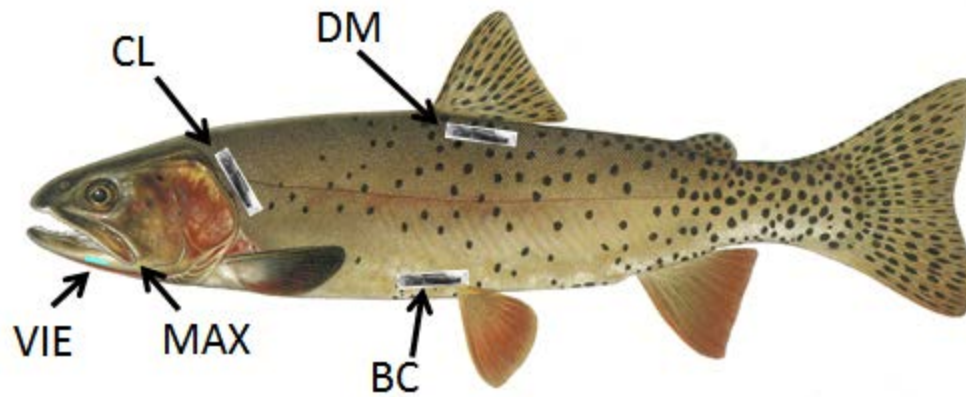
Figure 9.   Locations of PIT tag injection sites (BC, body cavity; CL, cleithrum; DM, dorsal musculature), maxillary clip (MAX) and visible injectable elastomer (VIE) injection site.

Figure 10.   Demonstration of PIT tagging technique in the body cavity (BC), cleithrum (CL), and dorsal musculature (DM) injection locations.

Figure 11.   Examples of VIE mark on lower mandible (indicated by arrow) and a maxillary clip on Rainbow Trout.
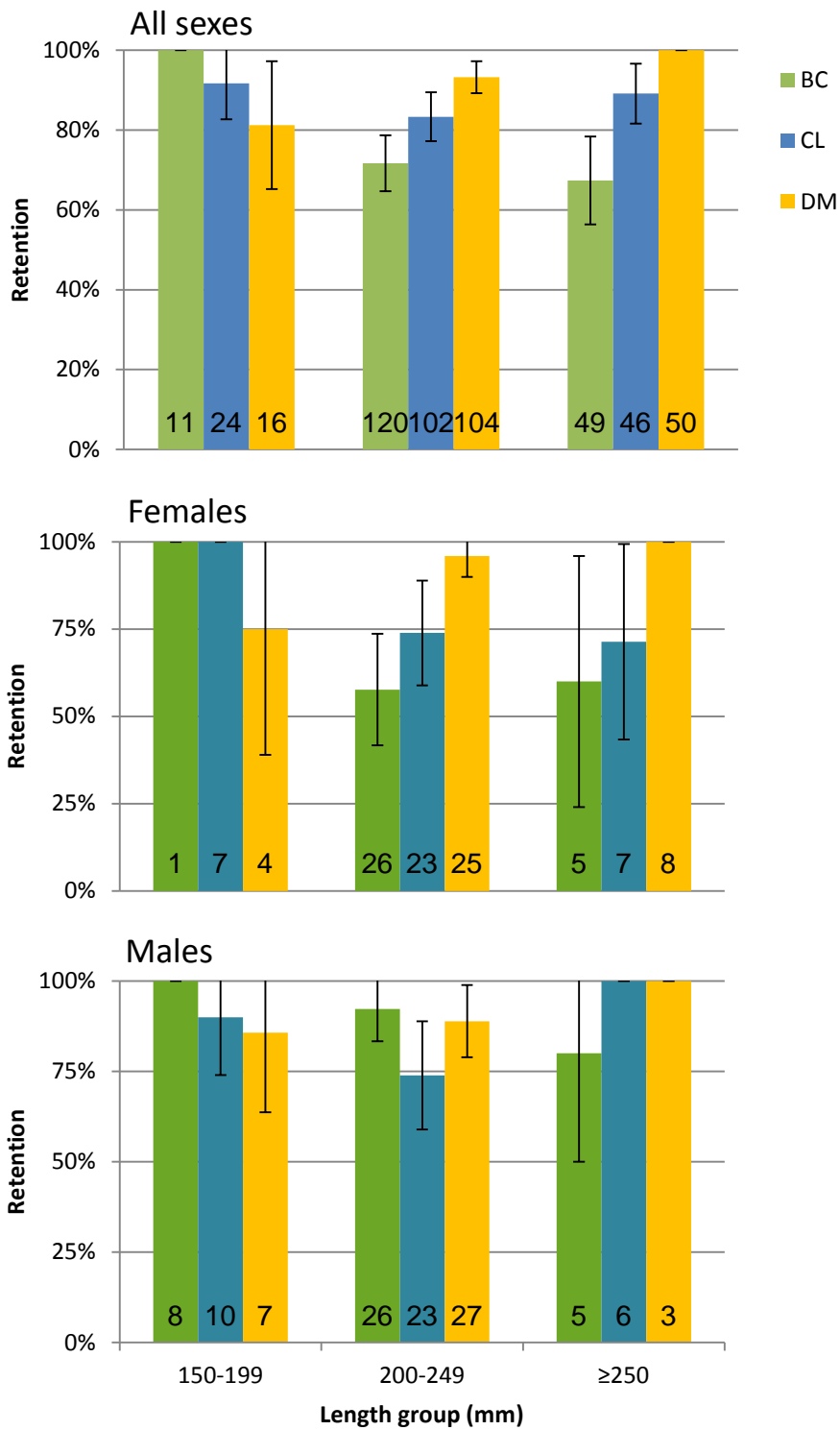
Figure 12.   Effect of size and sex of trout (at large through one possible spawning event) on tag retention rate while evaluating three PIT tag locations (body cavity - BC; cleithrum - CL; dorsal musculature - DM); 90% CI and sample size at the bottom of each column.
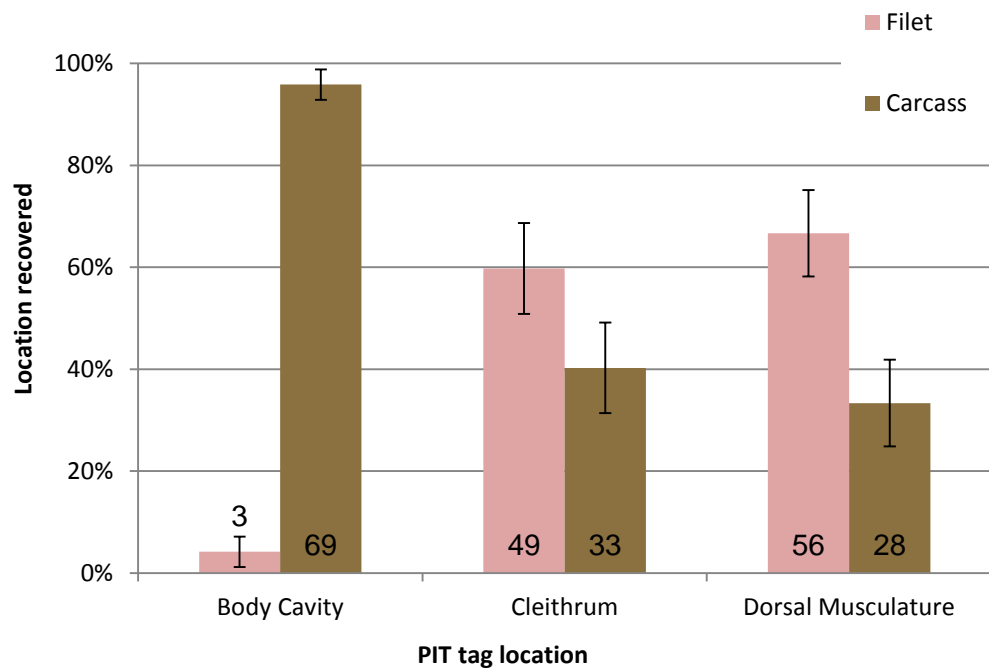
Figure 13.   Percent location recovered by injection location (body cavity - BC; cleithrum - CL; dorsal musculature - DM) with respect to the terminal location of a tag in the tissue of a trout, once having been filleted. Fish were at large effectively one year post both tagging and a spawning event; 90% CI, n's at the base of each graph.

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