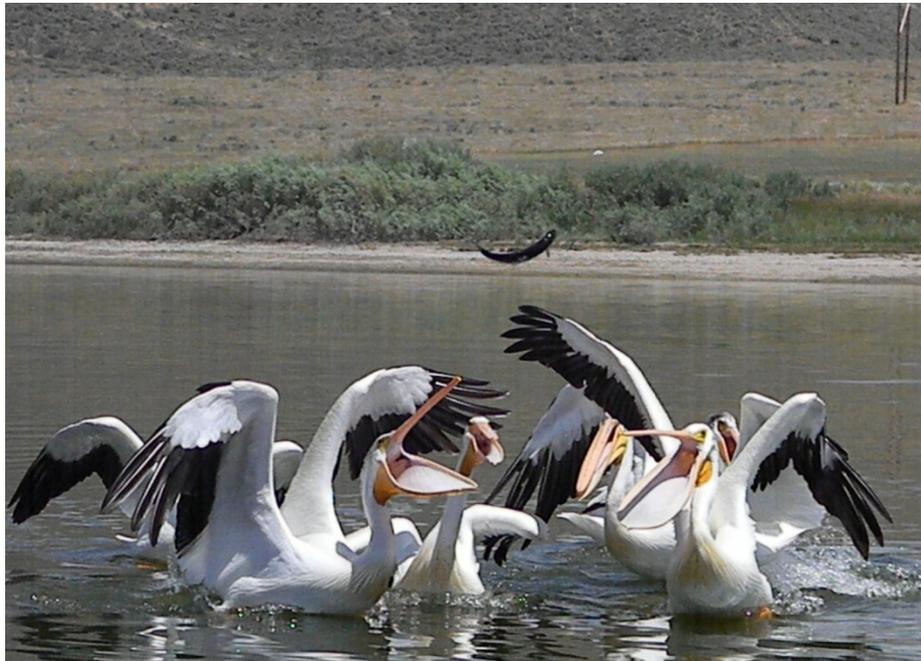




WILD TROUT EVALUATIONS: ELECTRIC WEIR FISH INJURY, CUTTHROAT TROUT STREAM PURIFICATION, AND PELICAN PREDATION



Report Period July 1, 2012 to June 30, 2013

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**IDFG Report Number 13-03
February 2013**

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**Annual Performance Report
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CHAPTER 1: EFFECTS OF ELECTRIC WEIRS ON THE INCIDENCE OF SPINAL INJURIES IN MIGRATORY YELLOWSTONE CUTTHROAT TROUT

ABSTRACT

Although the South Fork Snake River supports an abundant population of native Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* (YCT), the population is threatened by non-native rainbow trout *O. mykiss* that compete and hybridize with the native species. We sought to determine if the electric weirs that prevent rainbow trout passage upstream into YCT spawning habitat in tributaries of the South Fork Snake River cause spinal injuries in the YCT population. We sampled YCT from two electric weirs, on Palisades and Pine Creeks, and one waterfall/velocity barrier weir on Burns Creek (which served as a control) and x-rayed them to detect spinal injuries. The spinal injury rates at the Palisades Creek electric weir (11.3%) and the control weir at Burns Creek (4.5%) differed significantly (at $\alpha = 0.10$), while the injury rate at the Pine Creek electric weir (6.4%) was intermediate to the above two weirs. The average number of vertebrae involved in injuries at all weirs was 16.6. All spinal injuries involved compressions, though fractures of the spine and spinal misalignments also occurred. We conclude that there is evidence that the electric weirs may be causing a low level of spinal injuries. However, injury rates are not high enough to be of concern at the YCT population level, especially when considering the long-term benefit to the YCT population by excluding rainbow trout from the spawning tributaries.

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INTRODUCTION

The South Fork Snake River in eastern Idaho supports an abundant population of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* (Meyer et al. 2006). This population is considered important because it is one of the few robust fluvial populations of Yellowstone cutthroat trout (YCT) remaining in Idaho (Thurrow et al. 1988; Meyer et al. 2006; Gresswell 2011). However, the long-term persistence of YCT in the South Fork Snake River drainage is threatened by the increasing abundance of non-native rainbow trout *O. mykiss* (High 2010). Rainbow trout and YCT have similar life histories in the South Fork Snake River that overlap in many ways, including the fluvial nature of their spawning behavior (Henderson et al. 2000). While rainbow trout tend to spawn slightly earlier than YCT and are more likely to spawn in the main stem river (Henderson et al. 2000), both species also ascend the four main tributaries of the South Fork Snake River below Palisades Dam (Burns, Pine, Rainey, and Palisades creeks) to spawn. Rainbow trout and cutthroat trout have no isolating mechanism and readily hybridize throughout the native range of cutthroat trout (Young 1995; Behnke 2002). While rainbow trout will likely never be eliminated from the entire South Fork Snake River drainage, protection of pure YCT within the main stem and the four main spawning tributaries in Idaho has become a high priority for the Idaho Department of Fish and Game (IDFG 2007; High 2010).

Since 2001, IDFG has operated migration traps on these tributaries to prevent rainbow trout access upstream during the spawning period. Various weirs have been used over time, but most were inefficient or could not be operated in high flows during the critical period of the spring spawning migration. This led to the installation of a permanent combination waterfall and velocity weir on Burns Creek in 2009, which has been measured to be >98% efficient at capturing all upstream-migrating salmonids (High 2010). The remaining tributaries lacked sufficient channel gradient to install velocity barriers, so permanent electric weirs were installed in Palisades Creek in 2009, Pine Creek in 2010, and in Rainey Creek in 2011. Efficiencies for the electric weirs in these tributaries ranged from 49 to 86% during the first few years while trying to match the right electrical settings to various flow levels (B. High, IDFG, unpublished data).

While waterfall/velocity weirs likely do not cause fish injury, electric weirs have the potential to be more injurious to upstream or downstream migrating fish. However, little empirical data exists on the incidence of injuries at electric weirs. Electrical current in the water such as is used during electrofishing has repeatedly been shown to cause spinal and hemorrhage injuries (reviewed in Reynolds 1996). Trout species are especially vulnerable to injury from electrofishing (Edwards and Higgins 1973, Roach 1999), with injury rates ranging from 4-35% at 30 Hz or less (McMichael 1993, Hollender and Carline 1994). Larger fish are more vulnerable to injury because their increased surface area results in a greater electric potential (Reynolds et al. 1988). While both spinal injuries and hemorrhages are considered important when evaluating fish injuries from electricity (Reynolds 1996), spinal injuries are much more harmful because hemorrhages only exist for a relatively short time and therefore do not represent a long-term mortality or health risk to the fish (Schill and Elle 2000). Consequently, the primary objective of this study was to use a portable x-ray machine to evaluate spinal injuries to YCT presumably exposed to electricity at electric weirs.

METHODS

Fish were collected from traps at migration weirs on three tributaries of the South Fork Snake River on June 12-14 and June 25-26, 2012. Palisades Creek and Pine Creek have

electric weirs that prevent passage of rainbow trout upstream, whereas Burns Creek has a combination waterfall/velocity barrier weir which was therefore considered to be a control site for assessing spinal injury rates at the electric weirs. Rainey Creek also has an electric weir, but the spawning migration run was too small to include fish from this stream in our analyses. Ambient conductivity was 185, 298, and 359 $\mu\text{S}/\text{cm}$ at Burns, Palisades, and Pine creeks, respectively.

Both electric weirs in our study have six parallel electrodes made of metal railing embedded in an Insulcrete apron along the stream bottom, with the upper surfaces of the railings exposed to the water. The railings span the entire stream channel and continue up the Insulcrete walls that enclose the entire stream except for the fish trap. Fish traps at both weirs are located on the right bank (looking upstream), outside the electric field. The lowermost and uppermost electrodes are “parasitic,” meaning that electrical current does not “bleed” upstream or downstream of these electrodes. Consequently, fish that approach the electric field from a downstream location can enter the fish trap without experiencing any electrical current. The Palisades Creek weir output was set at 11.5 Hz, 2.5 ms pulse width, and 265 volts. The Pine Creek weir output was set at 13 Hz, 2 ms pulse width, and 270 volts. These settings produce similar horizontal voltage gradients at each weir, ranging from -11 to +12 V/cm but with most values in the -5 to +5 V/cm range. These values are likely sufficient to repel most fish without stunning them, given the low frequency and duty cycle (J. Reynolds, personal communication). The electric weirs are operated beginning on March 15 and ending after the spawning runs of rainbow trout and YCT have subsided, usually about mid-July.

Fish were netted from the trap box at each weir, anesthetized using MS-222, measured for total length, PIT-tagged, and given an adipose fin clip. X-rays were then taken with a MinXRay HF 100+ portable digital x-ray generator and a TruDR Ix system plate and computer program. X-rays were taken with a peak kilovoltage of 100 and an exposure of ~1.3 milliampere seconds, but settings were adjusted slightly as needed to obtain clearer images for each fish. Both vertical and horizontal x-rays were taken for nearly all injured fish and a subsample of uninjured fish to confirm that all injuries could be detected using horizontal x-rays only. After recovering from anesthesia, YCT were released upstream of the weir and fish trap.

X-ray images were analyzed for presence of spinal injuries, and injuries were rated using injury criteria in Reynolds (1996). Data were analyzed in SAS using a generalized linear model (with $\alpha = 0.10$, due to relatively small sample size), using fish total length, stream, and presence of an adipose fin clip prior to sampling (yes or no) as the explanatory variables, and fish injury as the response variable. For the response variable, we used dummy values of 0 for uninjured fish and 1 for fish with a spinal injury. Total length was included in the model because longer fish have a greater electrical potential and are therefore more likely to be injured by electricity (Reynolds et al. 1988). Presence of an adipose fin clip indicated that the fish had been previously captured and PIT-tagged by biologists. While the majority of these adipose-clipped fish in 2012 were simply trapped and tagged at the same weir in prior years, some (13% on average) were captured and tagged during main-stem South Fork Snake River electrofishing surveys a few months prior to our study (B. High, IDFG, personal communication). Any spinal injuries caused by main-stem electrofishing may not have healed by the time fish were captured at the weirs in 2012, causing us to falsely attribute these injuries to the weirs. Thus the presence of an adipose fin clip was used as a blocking factor in our model, allowing us to assume that any statistical differences in injury rates between the stream with velocity weirs and the streams with electric weirs was an indication of spinal injuries caused by the electric weirs.

RESULTS

A total of 349 YCT were x-rayed, including 134 fish at Burns Creek, 106 at Palisades Creek, and 109 at Pine Creek. A total of 25 spinal injuries were detected, for an overall spinal injury rate of 7.2%. All of the spinal injuries were categorized as being caused by electricity. Two additional fish with spinal malformations in the caudal peduncle were determined to have congenital defects and were not categorized as injured for our analyses.

The full model that included fish length, stream, and the absence of an adipose fin explained a statistically significant amount of variation in YCT spinal injury rate ($F = 2.05$, $P = 0.087$). However, only the stream main factor had a significant effect on injury rate ($P = 0.07$), with Duncan's multiple range comparisons indicating that spinal injury rate was significantly lower for Burns Creek at 4.5% (90% confidence intervals = 2.1-8.9%) than for Palisades at 11.3% (6.2-19.5%); the injury rate at Pine Creek was intermediate at 6.4% (3.2-12.1%) and did not differ significantly from the other streams (Figure 1). Fish length ($F = 1.89$, $P = 0.17$) and absence of an adipose fin ($F = 2.38$, $P = 0.12$) did not significantly influence YCT spinal injury rate.

The number of vertebrae involved in YCT spinal injuries ranged from 2 to 34, with an average of 16.6 vertebrae affected in each injured fish across all three streams. In all three streams, injuries of varying severity occurred, with compressions and fractures of the spine present in fish from all streams and spinal misalignments present at Palisades Creek and Burns Creek. However, 100% of all spinal injuries involved vertebrae compressions. Cutthroat trout at all streams had a mean (\pm SE) injury score of 1.56 (\pm 0.15), with mean scores of 1.83 (\pm 0.40) at Burns Creek, 1.58 (\pm 0.19) at Palisades Creek, and 1.29 (\pm 0.29) at Pine Creek.

DISCUSSION

Our results suggest that the electrical weirs constructed on spawning tributaries of the South Fork Snake River in Idaho may be causing a low level of spinal injury in the YCT population as fish migrate upstream. However, not all injuries that we detected were the result of electric weirs. For several reasons, we believe all of the spinal injuries at Burns Creek were caused by main-stem electrofishing, not the waterfall/velocity weir. First, we can find no evidence in the literature that challenging a waterfall/velocity weir would cause any spinal compressions in fish (especially involving on average 17 vertebrae). Second, the main stem of the South Fork Snake River was electroshocked sporadically throughout a 75 km stretch of river from February 4-28, 2012 as part of routine population monitoring. During the entire Burns Creek spawning run, 2.4% of the YCT captured at the velocity weir had also been captured and tagged during the February 2012 main stem electrofishing efforts. Since every fish that is exposed to electricity during such sampling is not netted and captured, presumably more fish near the vicinity of the Burns Creek mouth were also exposed to electricity a few months prior to being x-rayed. Because spinal compression injuries usually heal visibly within a year (Dalbey et al 1996; J. Reynolds, personal communication), the injuries at Burns Creek were likely attributable only to the February 2012 electrofishing surveys.

If this is true, then some of the injuries we observed at Palisades and Pine creeks were also likely caused by main-stem electrofishing a few months prior to our study. Indeed, the percentage of YCT captured at the electric weirs that had also been captured and tagged during the February 2012 electrofishing survey for Palisades Creek (3.2%) and Pine Creek (4.0%) was similar to those for fish caught in Burns Creek. Thus, the impact of main-stem electrofishing on

spinal injury rates at the electric weirs was probably similar across our study sites, suggesting that all of our spinal injury rates were overestimated to a similar degree. Nevertheless, the statistically significant difference between the velocity weir and one of the electric weirs suggests that the electric weirs cause spinal injuries in YCT, albeit at a very low level.

Mean spinal injury rate at the electric weirs was 8.6% in our study, which is relatively high considering the low pulse width (0.0-2.5 ms) and pulse frequency (11.5-13.0 Hz) settings for these weirs. Sharber et al. (1994) reported a spinal injury rate of 3% for wild rainbow trout using 15 Hz, whereas Hollender and Carline (1994) reported a rate of 13% for wild brook trout using 60 Hz. However, if our estimates were a few percentages high due to exposure to main-stem electrofishing prior to YCT spawning runs, then the estimates in our study would be in general agreement with the aforementioned studies. We are not aware of any estimates of fish injury rates at electric weirs to which we could compare our results directly. Additional studies of spinal injuries at electric weirs would help substantiate our results.

Not all detrimental effects to the YCT population at the electric weirs are the result of spinal injuries. For example, while sampling fish at Pine Creek, we found two dead YCT caught in the electric field of the weir, circling in an eddy. The two dead fish were x-rayed but did not have any spinal injuries, suggesting they likely died from asphyxiation or some other severe physiological stress response to the electricity (Snyder 2004). Although rare, fish are sometimes observed challenging the electric fields at the Pine and Palisades weirs, occasionally reaching the headboards before becoming immobilized and eventually washing downstream. While fish are recovering their equilibrium, they may asphyxiate or get caught in in-stream structures downstream. Mortalities observed at the electric weirs have generally been low, averaging only 0.8% of the entire spawning run and often due to handling stress rather than exposure to electricity (B. High, IDFG, unpublished data). However, unobserved mortality is likely to occur in fish over-exposed to electricity that float dead downstream without being observed by the weir operators. Annual exposure to electricity for the migratory component of the YCT population may also lead to a long-term reduction in fish growth rates (Gatz 1986), or may reduce egg survival for fish that are passed over the electric weirs (Marriott 1973; Dwyer 1993; Roach 1999). It would be worth considering a study to examine the effects of electric weirs on egg survival to alleviate concerns that spawning success is being diminished by the operation of the electric weirs on South Fork Snake River tributaries.

Given that cutthroat trout have on average 60-63 vertebrae, an average of 16.6 vertebrae involved in the injured YCT in this study constitutes a significant level of injury. Other studies have found an average of 6 to 8 vertebrae involved in spinal injuries due to electrofishing (Hollender and Carline 1994; Sharber and Carothers 1988), although these studies involved fish with lower mean lengths, thus the fish were likely not as affected by electricity as were the larger fish in our study.

Despite the fact that the electric weirs may be causing a low level of spinal injury to YCT in the South Fork Snake River drainage, any population-level effects are unlikely since injury rates were low and, consequently, mortality rates of injured fish are also probably low (Dalbey et al. 1996, Elle and Schill 2000). Dalbey et al. (1996) found that long-term survival of rainbow trout was not significantly affected by severity of electrofishing induced injuries. Also, since the weirs are only in operation from mid-March to mid-July, and outmigration of YCT usually occurs after mid-July, the majority of YCT only encounter the weirs while they are electrically active once a year, which helps minimize exposure. The injury rates were significantly different between Palisades Creek and Burns Creek, so the electric weirs are likely causing a number of YCT spinal injuries. However, we do not consider the injury rates at Palisades Creek to be high

enough to be of concern to YCT populations, considering that the weirs will block roughly 90% or more of rainbow trout and hybrids attempting passage upstream in future years. We consider the benefit the weirs provide by preventing upstream passage of rainbow trout and hybrids to outweigh the harm caused to YCT by the low injury rates found in this study.

RECOMMENDATIONS

1. Work with Region 6 fisheries staff to x-ray main-stem South Fork Snake River fish captured during electrofishing surveys to compare injuries in these fish with electric weir captures.
2. Since current electric weir settings appear to have a minimal effect on Yellowstone cutthroat trout migrating to these weirs, increase electrical field settings to increase capture efficiency and diminish the likelihood of rainbow trout or hybrids escaping upstream to spawn with Yellowstone cutthroat trout.
3. Work with Region 6 staff to capture and lethally retain any Yellowstone cutthroat trout x-rayed in 2012 in order to x-ray them again to assess injury healing. Since this will likely only involved 10-20 fish at the most, this will have little impact on the cutthroat trout spawning runs but would provide valuable additional data.
4. Consider an evaluation of the effects of exposure to electric current at these weirs on egg survival.

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FIGURES

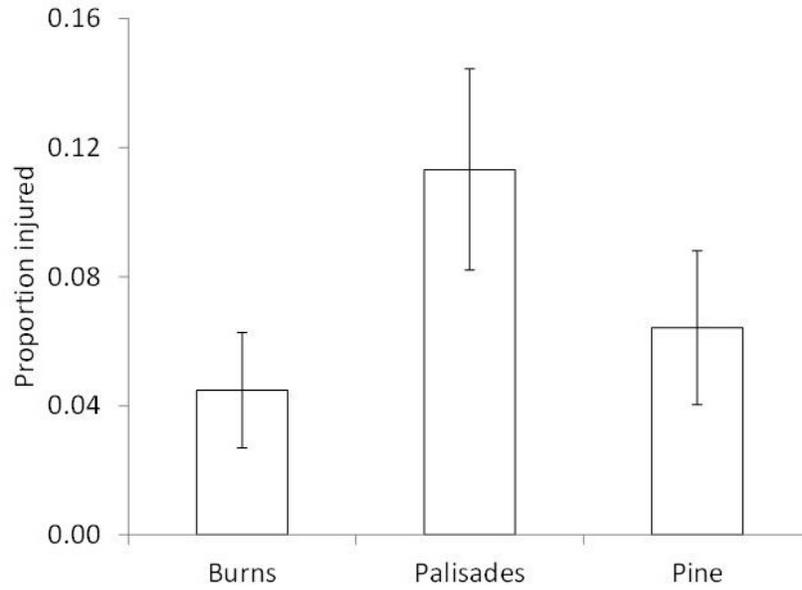


Figure 1. Proportion (\pm 1SE) of spinal injuries in Yellowstone cutthroat trout sampled during the spawning migration run in tributaries of the South Fork Snake River in Idaho.

CHAPTER 2: ATTEMPTING TO PURIFY A YELLOWSTONE CUTTHROAT TROUT STREAM BY REMOVING RAINBOW TROUT AND HYBRIDS VIA ELECTROFISHING

ABSTRACT

We used two years of backpack electrofishing removals of rainbow trout *Oncorhynchus mykiss* and hybrids to evaluate whether an introgressed population of Yellowstone cutthroat trout *O. clarkii bouvieri* in Palisades Creek (a tributary of the South Fork Snake River) could be reverted to pure or nearly pure conditions. Removals were conducted from an electric weir (0.7 km from the mouth) that blocks upstream migrating non-native salmonids from spawning in Palisades Creek, upstream approximately 10 km to a high velocity cascading section of stream that appears to be a complete fish barrier. In 2012, genetic samples were collected and the phenotypic characteristics of the sampled fish were recorded for comparison against genetic results. Despite Palisades Creek being a relatively wide (about 10 m), steep, swift, and deep stream for effectively sampling with backpack electrofishing gear, capture efficiency was surprisingly high at 51%, and was especially high for spawning-sized fish (i.e., those \geq about 250 mm TL) at 73%. Early indications are that the removal effort is working, as Yellowstone cutthroat trout comprised 68% of all fish captured in 2010 and 74% in 2012. Total phenotypic accuracy for all fish was 89%, and nearly half the mistakes were misidentifying rainbow trout as hybrids (which were removed anyway). Thus $<1\%$ of the Yellowstone cutthroat trout that were captured were mistakenly killed, and only 6% of the rainbow trout alleles that were captured were accidentally released. Our results suggest that, combined with an efficiently operated electric weir that precludes rainbow trout and hybrids from accessing Palisades Creek, and the addition of a second removal effort each year, reducing rainbow trout alleles to below 10% may only take 1-2 more years.

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INTRODUCTION

In eastern Idaho, the South Fork Snake River supports one of the few remaining fluvial populations of Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* (Thurow et al. 1988; Meyer et al. 2006a; Gresswell 2011). However, the long-term persistence of cutthroat trout in the South Fork Snake River drainage is threatened by an increasing abundance of rainbow trout (High 2010). Interactions between native cutthroat trout *O. clarkii* and non-native rainbow trout *O. mykiss* usually 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 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 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). Recent 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 higher category by eradicating rainbow trout and hybrids is one of the highest management priorities for cutthroat trout populations in Idaho by the IDFG (IDFG 2007).

While prior attempts to eradicate non-native fish species with electrofishing removals have typically not been highly successful (e.g., Thompson and Rahel 1996; Meyer et al. 2006b; Peterson et al. 2009; but see Kulp and Moore 2000 and Shepard et al. 2002), the life history traits of rainbow trout may result in higher success with removal efforts. In western North America, brook trout are the most commonly targeted non-native salmonid for removals (Dunham et al. 2002), but brook trout mature at an earlier age and are more fecund than rainbow trout (Buss and McCreary 1960), providing brook trout populations better compensatory traits to offset removal efforts (Meyer et al. 2006a) than rainbow trout populations might have. This should make removing rainbow trout by electrofishing more feasible, and in fact, it has already been successfully done in small streams in eastern North America (Kulp and Moore 2000). Additionally, electric and velocity weirs are in place on several tributaries to the South Fork Snake River, preventing current and future rainbow trout passage upstream to Yellowstone cutthroat trout spawning habitat within these tributaries (High 2010). Removing the remaining resident rainbow trout and hybrids in such tributaries could preserve these streams for pure Yellowstone cutthroat trout spawning and rearing habitat.

Although genotype remains the most accurate measure of hybridization, for practical purposes, phenotype is a much easier and cheaper characterization to make when sampling cutthroat trout populations in the field. While prior studies have genetically confirmed that Yellowstone cutthroat trout, rainbow trout, and hybrids can be differentiated phenotypically

(Campbell et al 2002; Meyer et al. 2006a), no research exists on the proportional incidence of specific phenotypic traits in genetically identified Yellowstone cutthroat trout, rainbow trout, and hybrids. Because hybrids can be difficult to identify accurately and can exhibit a range of parental phenotypes (Neff and Smith 1979), combining genetic analyses with recorded phenotypic traits would help create a more accurate metric for phenotypically differentiating Yellowstone cutthroat trout from rainbow trout and hybrids. Refining these identifications would also make efforts to remove hybrids and rainbow trout more effective.

OBJECTIVE

1. Evaluate whether a hybridized population of Yellowstone cutthroat trout in Palisades Creek (a tributary to the South Fork Snake River) could be reverted back to pure or nearly pure conditions within three years.

METHODS

Palisades Creek is a 4th order stream (1:24,000 hydrologic scale) with a mean width of about 10 m and an average gradient of about 2.0%. Although the stream is about 30 kilometers in length, near river kilometer 11 there is a high-gradient, steep cascading section of river that may form a natural velocity barrier, because rainbow trout and hybrids are absent above this section of river (Meyer et al. 2006a; K. Meyer, unpublished data). Consequently, removal of rainbow trout and hybrids occurred only in the lower 11 kilometers of stream. Various weirs have been operated on Palisades Creek since about 2000 in an attempt to halt upstream spawning migrations of rainbow trout and hybrids, but most were not highly efficient or could not be operated in high flows during the critical period of the spring spawning migration run. In 2009, this led to the installation of a permanent electric weir on Palisades Creek 0.7 km upstream from the confluence with the South Fork Snake River.

Rainbow trout and hybrid removal

Rainbow trout and hybrid removal to date has taken place in 2010 and 2012. Removal efforts were not possible during 2011 due to flows far above average that made it impossible to effectively electrofish the stream. Moreover, the electric weir failed in 2011 early in the spawning run (due to damage from extremely high flow events) and had to be shut down for the remainder of the spawning period. During removals in 2010 and 2012, teams with backpack electrofishers and nets electrofished sequential sections of about 800 m, starting at the electric weir and shocking roughly 10 km upstream to the point where habitat was considered unsuitable to maintain trout populations due to high stream gradient and elevation; this also coincided with a greatly diminished abundance of trout within this section of stream.

In 2010, three teams each consisted of two people with backpack electrofishers and two or more people with nets and buckets. In 2012, two teams each consisted of three people with backpack electrofishers and three or more people with nets and buckets. Teams moved upstream within each 800-m section, marking the beginning and end of their section with flagging tape to ensure there was no gap in shocking area covered during upstream leapfrogging of sections. During sampling, persons with backpack electrofishers covered all available habitats. When gradient was especially steep, 1-2 electrofishers were used to block downstream movement of trout, and the remaining electrofishers were used to chase trout

downstream through the steep gradient section into an area with slower water velocity where the fish were immobilized and more easily netted.

All fish were anesthetized with MS-222 and measured for total length (TL). Rainbow trout and hybrids were removed and killed, and Yellowstone cutthroat trout were released downstream after recovering from anesthesia. Genetic samples were collected from 30 randomly selected fish from each section, and the phenotypic characteristics of the fish with genetic samples were recorded for comparison against genetic results (see phenotypic characteristics in Table 1).

Recapture efficiency

Four weeks prior to electrofishing in 2012, 153 Yellowstone cutthroat trout were collected from throughout the stream, measured, marked with a maxillary clip, and released to conduct a mark-recapture estimate of trout abundance and capture efficiency and rainbow trout removal efficiency. During the marking run, 45 rainbow trout and hybrids were removed and killed. Only Yellowstone cutthroat trout ≥ 100 mm total length (TL) were marked and released.

A mark-recapture population estimate of the abundance of trout ≥ 100 TL in Palisades Creek was produced for 2012 using the Fisheries Analysis + software package (Montana Fish, Wildlife and Parks 2004). Because capture efficiency was size selective (Table 2), estimates were made for each 50-mm size group and summed for a total estimate. Since no recaptures occurred in the 100-150 mm size group, and capture efficiency declined with size, we used the partial log-likelihood model (rather than the modified Petersen model) which allowed us to estimate capture efficiency for this size group (rather than pooling it with the next largest size group, which would have resulted in overestimating abundance of this size group). We assumed there was 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, and estimates for each species were calculated based on the proportion of catch comprised by each species during the recapture run (Meyer et al. 2006a).

Genetic analyses

Genetic analyses were completed to test phenotype-based identification of Yellowstone cutthroat trout, rainbow trout, and hybrids. All samples were screened for rainbow trout hybridization/introgression with seven diagnostic nuclear DNA (nDNA) markers (Occ34, Occ35, Occ36, Occ37, Occ38, Occ42 and OM55; Ostberg and Rodriquez 2002). Individual genetic classification was based on composite nDNA genotypes following similar procedures as those outlined by Ostberg and Rodriquez (2006) and Kozfkay et al. (2007). Samples were classified as Yellowstone cutthroat trout if they were homozygous for *O. c. bouvieri* alleles at all loci, rainbow trout if they were homozygous for *O. mykiss* alleles at all loci, and hybrids if they possessed a mixture of alleles from the two parental species. Hybrids were further classified into three categories: first-generation hybrids (F1) if they were heterozygous at all loci; $>F1$ Yellowstone cutthroat trout backcross hybrid if they possessed a mix of heterozygous loci and loci homozygous for cutthroat alleles; and $>F1$ rainbow trout backcross hybrids if they possessed a mix of heterozygous loci and loci homozygous for rainbow trout alleles. With seven codominant nDNA loci, our probability of mistaking a backcross hybrid ($>F1$) as an F1 hybrid was less than 1% (Boecklen and Howard 1997). Hardy Weinberg exact tests in the software program Genepop on the Web (<http://genepop.curtin.edu.au/>) were performed to test the null

hypothesis that these samples could have been drawn from a single population (Raymond and Rousset 1995).

Introgression is the actual incorporation of genes from one taxa into the population of another through hybridization and backcrossing. Therefore, usually in an admixed sample, alleles from fish identified as pure rainbow trout and F1 hybrids are not included in estimates of introgression. However, for summary and comparison purposes we reported introgression in two ways. First, we calculated introgression as the total number of rainbow trout alleles observed out of the total examined, including all samples regardless of individual genetic classification. We also calculated introgression for the sample group as the total number of rainbow trout alleles observed out of the total examined, excluding all samples that had been phenotypically identified as rainbow trout or hybrids (and thus removed during electroshocking).

RESULTS

Rainbow trout and hybrid removal

A total of 2,625 fish were captured in 2010, of which 68% were identified phenotypically as Yellowstone cutthroat trout, 8% as rainbow trout, and 24% as hybrids. In comparison, a total of 1,671 fish were captured in 2012, of which 74% were identified phenotypically as Yellowstone cutthroat trout, 4% as rainbow trout, and 22% as hybrids. Fish ranged from 41-503 mm TL in 2010 and from 34-475 mm TL in 2012.

The proportion of the trout population that Yellowstone cutthroat trout comprised increased from 2010 to 2012 across most size categories, remaining at 0.77 for fish <15 cm TL, increasing from 0.63 to 0.64 for fish 15-25 cm TL, and increasing from 0.60 to 0.75 for fish >25 cm.

In 2012, the proportion of the catch comprised of Yellowstone cutthroat trout generally increased in an upstream manner, whereas in 2010, proportions of Yellowstone cutthroat trout were lowest in the intermediate stream reaches (Figure 2). The highest proportion of Yellowstone cutthroat trout in 2012 (0.93) occurred from river kilometer (rkm) 8.1-9.7 and the lowest proportion (0.64) occurred from rkm 1.6-3.2. In contrast, the highest proportion of Yellowstone cutthroat trout in 2010 (0.83) occurred from rkm 0-1.6 and the lowest proportion (0.50) occurred from rkm 3.2-4.8.

The partial log-likelihood population estimate in 2012 was used to estimate the efficiency of removing rainbow trout and hybrids for that year. For the largest size group (>25 cm TL), 137 (55%) of an estimated total of 249 rainbow trout and hybrids were removed in 2012. In comparison, 220 (33%) of an estimated 676 rainbow trout and hybrids 15-25 cm TL were removed, and 127 (81%) of an estimated 157 rainbow trout and hybrids <15 cm TL were removed.

Recapture efficiency

Mean capture efficiency for marked Yellowstone cutthroat trout during 2012 removals was 0.51. However, capture efficiency increased as fish size increased (Table 2). For spawning-sized fish (i.e., those generally > 250 mm TL), capture efficiency was 0.73.

Genetic analyses

A total of 326 fish were randomly sampled for genetic analysis, of which 219 were Yellowstone cutthroat trout, 31 were rainbow trout, 34 were F1 hybrids, and 42 were >F1 backcross hybrids. Of the backcross hybrids, 27 were Yellowstone cutthroat trout backcrosses and 15 were rainbow trout backcrosses. Introgression in the fish phenotypically identified as Yellowstone cutthroat trout was 1.7%, or 55 alleles out of 3,248. Of all the fish genetically sampled, 925 alleles (20.3%) out of 4,564 were rainbow trout.

Our phenotypic accuracy was 94% for identifying Yellowstone cutthroat trout, 71% for identifying rainbow trout, and 79% for identifying hybrids. Fish mistakenly identified phenotypically as rainbow trout ($n = 5$) were all actually hybrids, with each fish having at least 11 rainbow alleles (out of 14 that were screened). Fish that were mistakenly identified as hybrids ($n = 15$) were all actually rainbow trout, except for one Yellowstone cutthroat trout. Fish that were mistakenly identified as Yellowstone cutthroat trout ($n = 15$) were all actually hybrids except for one rainbow trout (which had all the phenotypic traits of a Yellowstone cutthroat trout and may have been a recording error). Of the hybrids that were mistakenly released as Yellowstone cutthroat trout, 29% had only one rainbow trout allele and the average number of rainbow trout alleles in mistakenly released hybrids was 2.9 (out of 14).

Most of the phenotypic characteristics were useful for differentiating Yellowstone cutthroat trout from rainbow trout and hybrids, but head spots and white edges on fins (especially the pelvic or anal fins) were particularly useful traits for separation (Table 1). For example, 227 fish had fewer than five spots on their head, and all but seven of those were Yellowstone cutthroat trout, whereas a total of 97 fish had five or more spots on their head, and only 12 of those were cutthroat trout. Also, only 2 Yellowstone cutthroat trout had a white leading edge on the pelvic or anal fins, whereas 16 of 17 rainbow trout had white on one of those fins. White on the pelvic or anal fins were not as definitive for hybrids, with 3 of 34 F1 hybrids and 18 of 44 >F1 hybrids lacking white on any of the three fins. The presence of a slash was not a definitive trait for separation. In fact, only 16 fish (4.9%) lacked a slash altogether (all of them were hybrids or rainbow trout), and even pure rainbow trout had some level of slash 61% of the time.

DISCUSSION

Although Palisades Creek is a relatively wide, steep, swift, and deep stream that is difficult to sample with backpack electrofishing gear, capture efficiency was surprisingly high at 51%. Capture efficiency was highest for larger fish (i.e., ≥ 250 mm TL), which is promising since rainbow trout in the South Fork Snake River basin reach sexual maturity at about this size (B. High, IDFG, personal communication), and removing the largest fish should quickly reduce new recruitment of rainbow trout and hybrids to the population. The proportion of Yellowstone cutthroat trout among all trout increased most from 2010 to 2012 in the ≥ 250 mm size group, reaffirming that our rainbow trout and hybrid removal efficiencies are probably highest for that group. The removal efforts would likely have been even more successful if high water had not prevented electrofishing removals and the operation of the electric weir in 2011. In the future, multiple removals within the same year, over the period of base flow conditions, may be needed to hasten the reduction of rainbow trout alleles in the population (Kulp and Moore 2000; Shepard et al. 2002).

Overall, genetic analyses suggest that phenotypic identification and removal of rainbow trout and hybrids from Yellowstone cutthroat trout populations can be effective in some streams. Only 0.5% of the Yellowstone cutthroat trout that were captured were mistakenly killed, so the loss of Yellowstone cutthroat trout was negligible. Moreover, only 6% of the rainbow trout alleles that were captured were released, and of the 14 hybrids released, all but one were cutthroat trout backcrosses, with most being phenotypically indistinguishable from pure Yellowstone cutthroat trout. Our identification error rate was higher for rainbow trout and hybrids than for Yellowstone cutthroat trout, but most of the errors made were misidentifying rainbows as hybrids or the reverse. From a removal action perspective, both were treated the same. Therefore, the number of hybrids and rainbows that were released as a result of misidentification was low.

Certain phenotypic traits appear to be more indicative of rainbow trout alleles and genotype than others. Fewer than five head spots, lack of white on the fins, lack of rainbow colored sides and spots clustered dorsally and posteriorly were traits that were far more prevalent in Yellowstone cutthroat trout than in rainbow trout and hybrids. Belly coloration and presence of slashes were not good indicators of pure Yellowstone cutthroat trout alleles because approximately 60% of pure rainbow trout had weak to moderate slashes. While no single phenotypic trait conclusively determined the presence of rainbow trout alleles in a fish, 5 or more head spots and a white tip on the pelvic or anal fins appeared to be the most effective traits for differentiating rainbow trout and hybrids from Yellowstone cutthroat trout. A further direction for management of the Yellowstone cutthroat trout population in the South Fork of the Snake may be to make a classification tree using the proportion of these phenotypic traits, as classification trees have been shown to be more accurate in identifying hybrids than relying on phenotypic traits alone (Weigel et al 2002).

Although returning the Yellowstone cutthroat trout population in Palisades Creek to less than 1% hybridization may not be feasible, removals of rainbow trout and hybrids may still be successful at maintaining a healthy population of Yellowstone cutthroat trout in this stream, especially if the proportion of rainbow trout alleles in the entire population is reduced below 10%. During the 2012 electrofishing efforts, (1) we captured about 50% of the trout occupying Palisades creek, (2) Yellowstone cutthroat trout comprised 74% of the trout in the stream (based on phenotype), and (3) we removed 100% of the rainbow trout that were captured and only released 6% of the hybrids that were captured (mistakenly calling them cutthroat trout). Correcting trout composition for phenotypic misidentifications, we estimate that cutthroat trout actually comprised 70% of the trout population in 2012. Of the 30% that were either rainbow trout or hybrids, we probably removed about 50%. Thus, after the removal, Yellowstone cutthroat trout should comprise about 85% of the trout population. If the electric weirs preclude rainbow trout and hybrids from accessing the stream, and removal efficiency remains the same in subsequent years, Yellowstone cutthroat trout should comprise >90% of the trout population after one more year of removal effort.

RECOMMENDATIONS

1. Due to the high gradient of some sections of this stream, teams with three electrofishers should be used for future removal of rainbow trout alleles, using 1-2 shockers as “blockers” while chasing fish downstream out of the steepest habitat and into areas where netting fish is more feasible.

2. Because each removal effort can achieve about a 50% reduction in rainbow trout alleles in the stream, we suggest that two removals should be conducted in each base flow period to speed the “purification” process.
3. Yellowstone cutthroat trout should be marked and released before each removal effort (any rainbow trout or hybrids captured during any marking run should be removed) so that capture efficiency (and therefore rainbow trout removal efficiency) can be estimated for each removal effort.

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TABLES

Table 1. Description of phenotypic characteristics used to distinguish Yellowstone cutthroat trout from rainbow trout and hybrids, and the proportional occurrence by genotype.

Phenotype	Description	Composition (%)			
		YCT	>F1	F1	RBT
Head spots	Five or more	4.6	68.2	91.2	92.9
	Fewer than five	93.4	31.8	8.8	6.1
Belly	Orange hue	26.5	9.1	5.9	0
	White	73.5	90.9	94.1	100
Pelvic Fins	White tips present on fins	0.5	52.3	58.8	96.4
	White tips absent from fins	99.5	47.7	41.2	3.4
Anal Fins	White tips present on fins	0.9	56.8	79.4	96.4
	White tips absent from fins	99.1	43.2	20.6	3.6
Dorsal Fins	White tips present on fins	0	38.6	70.6	85.7
	White tips absent from fins	100	61.4	29.4	14.3
Throat Slash	Bright orange and prominent	90.0	34.1	17.7	3.6
	Dull but continuous	4.1	11.4	17.7	0
	Faint and barely visible	5.9	43.2	64.7	57.1
	Absent	0	11.4	0	39.3
Body Spots	More evenly distributed along sides	3.7	70.5	76.5	96.4
	Clustered dorsally and posteriorly	96.3	29.5	23.5	3.6
Side Coloration	Presence of rainbow coloration	5.5	61.4	85.3	92.9
	Absence of rainbow coloration	94.5	38.6	14.7	7.1

Table 2. Recapture efficiency by 50-mm size groups of marked Yellowstone cutthroat trout in Palisades Creek during 2012 sampling.

Size group (mm)	<i>n</i>	Capture efficiency
100-149	13	0.00
150-199	32	0.38
200-249	33	0.45
250-299	47	0.60
300-349	20	0.85
350-499	8	0.75
Combined	153	0.51

FIGURES

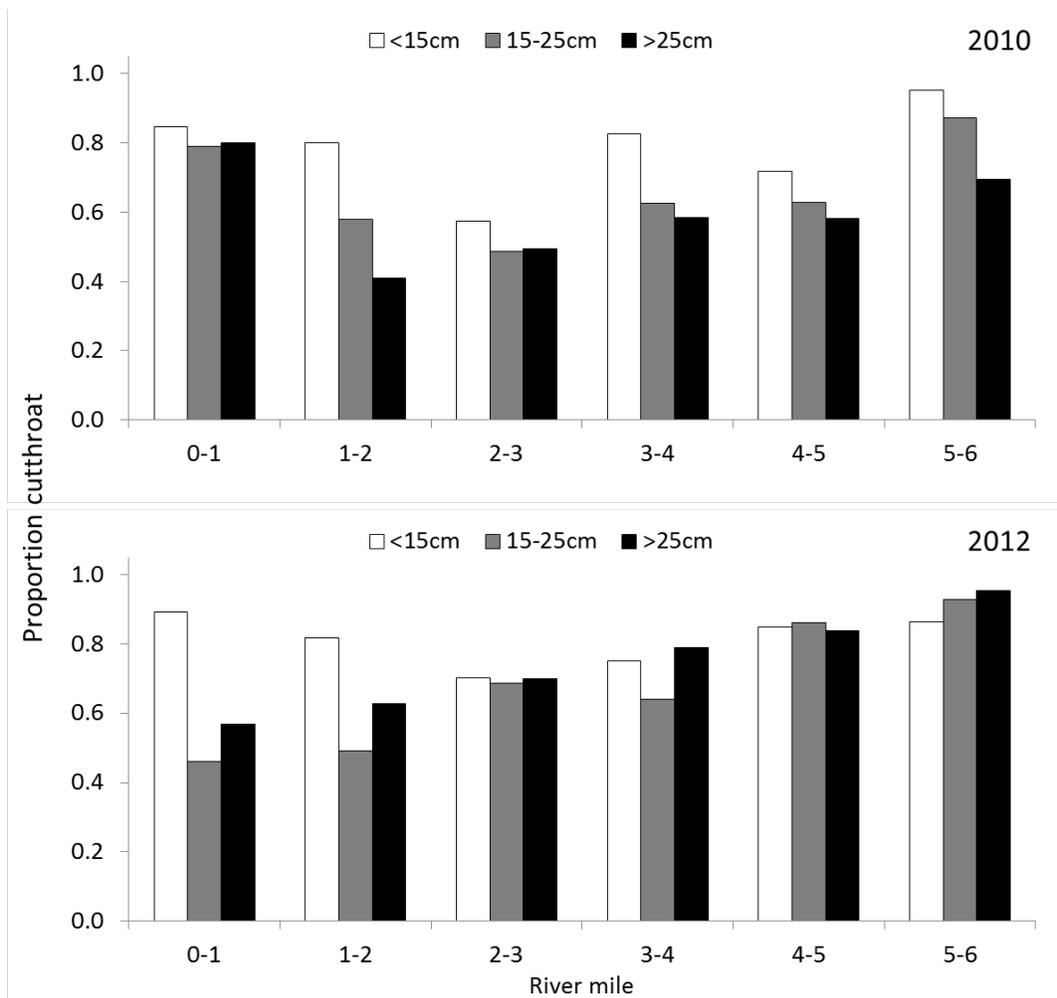


Figure 2. Proportional abundance of Yellowstone cutthroat trout by river mile and size class in 2010 and 2012.

CHAPTER 3: PREDATION RATE OF AMERICAN WHITE PELICANS ON STOCKED CATCHABLE RAINBOW TROUT IN IDAHO RESERVOIRS

ABSTRACT

In southern Idaho, growth of two American White Pelican *Pelicanus erythorhynchus* nesting colonies since the early 1990s has generated fishery concerns about the effect of pelican predation on salmonid stocks, including the Idaho Department of Fish and Game's stocking program for catchable-sized hatchery trout. To assess the level of impact that American White Pelican predation may be having on catchable rainbow trout *Oncorhynchus mykiss*, predation rates were estimated at several Idaho study waters, and comparisons were made to angler harvest at the same locations. To estimate predation, we directly fed 368 PIT-tagged fish to pelicans across all study waters, and stocked a total of 1,379 PIT-tagged catchable rainbow trout into the same waters; once the juveniles were fledged, we searched the two nesting colonies for regurgitated or defecated PIT tags from both of these PIT tag groups of fish. Recoveries of fed tags allowed us to estimate tag recovery efficiency from fish that were known to be consumed by pelicans, which was used to correct the unadjusted predation rate (estimated by recovery of tags from stocked fish) by the tag recovery efficiency, thereby enabling us to estimate the total pelican predation rate. At the nesting colonies we recovered 220 PIT tags, which constituted 10.4% of all the stocked tags and 26.9% of all the fed tags. We recovered 23 additional tags at loafing areas from two study waters. Predation by American White Pelicans on stocked rainbow trout averaged 35% but not all estimates were considered reliable. In contrast, angler harvest averaged 23%. In general, there was an inverse relationship between pelican predation and angler harvest, except at Cascade Reservoir, where both pelican predation and angler harvest were estimated to be near zero. The results of this pilot study indicate that at select waters in Idaho, pelican predation on catchable trout is high and greatly exceeds harvest by anglers. Further research in new waters in 2013 will broaden this perspective.

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INTRODUCTION

Piscivorous birds including the American White Pelican *Pelicanus erythorhynchus* can pose a substantial threat to fish stocks. Although abundance of American White Pelicans has increased dramatically in North America in the last few decades (King and Anderson 2005), their effects on fish populations remain largely undocumented. Several studies have shown that American White Pelicans exhibit plasticity in their feeding habits (Hall 1925; Knopf and Kennedy 1980) and become opportunistic under the right conditions, including fish spawning migrations (Findholt and Anderson 1995; Scopettone and Rissler 2002; Murphy and Tracy 2005). In Idaho, pelicans have been shown to consume significant numbers of adfluvial Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri* (YCT) in the Blackfoot Reservoir system, potentially reducing both spawning success and recruitment (Teuscher and Schill 2010).

Catchable hatchery rainbow trout *O. mykiss* are another important fish stock that American White Pelicans may be impacting in Idaho. Because the Idaho Department of Fish and Game (IDFG) releases catchable rainbow trout in numerous waters throughout the state to increase angler catch rates, determining the level of predation by pelicans on these fish is needed. Substantial predation by American White Pelicans on trout has been documented on the North Platte River, Wyoming, within days of the release events (Derby and Lovvorn 1997), and there is evidence that such swift consumption is occurring in Idaho as well. Indeed, pelicans at Blackfoot Reservoir have been shown to target stocking areas shortly after trout are released (Teuscher et al. 2005). In addition, anglers in southern Idaho have reported incidents of ponds being devoid of stocked trout within days after planting, and anglers and IDFG biologists have reported seeing increased pelican activity during the same timeframe (D. Megargle, IDFG, personal communication). Moreover, large numbers of T-bar anchor tags and PIT tags from hatchery trout stocked throughout southern Idaho were found during 2011 surveys of nesting colonies in Idaho (D. Teuscher, IDFG, unpublished data), suggesting pelicans may be consuming large numbers of stocked fish which were intended for angler harvest.

American White Pelicans inhabit at least two nesting colonies in Idaho, one located on islands at Blackfoot Reservoir and the other at an island complex on Lake Walcott at the Minidoka National Wildlife Refuge (NWR). There are numerous ponds and reservoirs in southern Idaho that are regularly stocked with hatchery rainbow trout and are within the foraging range of pelicans nesting at the two southern Idaho colonies, as well as other colonies outside the state boundary. American White Pelicans from Idaho colonies have been shown to travel to other reservoirs to forage, but the extent to which they influence these fisheries is currently unknown (IDFG 2009). Their soaring ability enables them to forage at distances of up to 600 km (Johnson and Sloan 1978; Trottier et al. 1980; O'Malley and Evans 1982).

The goal of this study was to estimate annual American White Pelican predation rates for catchable rainbow trout stocked in several Idaho reservoirs. We used 2012 as a pilot study year to develop appropriate methodologies and address the following objectives: 1) estimate pelican predation rate on catchable rainbow trout in six southern Idaho ponds and reservoirs, 2) estimate the mean number of anglers and American White Pelicans (birds of breeding age and non-breeding age) using the study waters from May through September, and 3) compare pelican predation rates to angler harvest rates of stocked catchable trout on the same waters and during the same time period.

METHODS

Study waters evaluated during 2012 were scattered across southern Idaho (Figure 3). Distances between the two American White Pelican colonies and the study waters ranged from 30 to 410 km at Chesterfield Reservoir and Cascade Reservoir, respectively (Table 3). Study waters were selected to investigate predation losses in several southern Idaho fisheries known to be receiving substantial pelican use, and to gain perspective on possible geographical gradients in pelican predation rates across the state for planning future studies at a larger scale.

Pelican and angler counts

Pelican and angler counts were conducted on randomly assigned days and times (daylight hours) from March through October depending on the water body. The mean daylight hours for each survey interval were divided into two time periods (AM and PM), and pelican and angler surveys were initiated at random start times within these periods. For the larger water bodies where surveys could not be completed from one location, the direction of surveys around the study water (i.e., clockwise or counter-clockwise) was also randomized for each sample. During each survey at the larger study waters, a motorized boat was driven around the reservoir, and all pelicans, boats, and anglers were counted. High-powered binoculars and digital cameras were used to enumerate birds and anglers. At CJ Strike Reservoir, in instances where pelicans were clustered in groups of at least 10 birds, were moving, or were too difficult to count or identify, digital photographs were taken and later examined with the zoom capabilities of photo software on a desktop computer. We used these photos (and those available from other waters) to classify pelicans as breeding or non-breeding aged birds based on the presence or absence of breeding horns, dark plumage on the nape of the neck, or coloration of other plumage, the bill, or legs.

Estimating pelican predation

Estimating the rate of pelican predation in 2012 involved three steps. First, we PIT tagged live hatchery rainbow trout and fed them directly to pelicans (D. Teuscher, personal communication). Second, during the same timeframe, we stocked PIT-tagged catchable rainbow trout into the study waters. Third, after pelicans on the nesting colonies fledged their young, we searched the two colonies for regurgitated and defecated PIT tags from both of these PIT tag groups of fish (fed and stocked). By recovering fed tags at pelican nesting colonies, we were able to estimate tag recovery efficiency from fish that were known to be consumed by pelicans. This allowed us to adjust the unadjusted predation rate (estimated by recovery of tags from stocked fish) by the tag recovery efficiency (D. Teuscher, personal communication), thereby enabling us to estimate total pelican predation on catchable rainbow trout.

We fed pelicans live or dead fish that were abdominally tagged with PIT tags (23 mm ½ duplex tags). Feeding began in late May and concluded by late July, which encompassed the time when breeding pelicans were foraging and traveling between the breeding colonies and foraging sites to feed their chicks. Hatchery rainbow trout >200 mm (TL) were transported from a nearby hatchery and injected with a PIT tag in the abdominal cavity and a small amount of air under the skin before being launched in the direction of loafing or foraging pelicans. Although pelicans were usually difficult to approach and feed initially, after a few days they became more comfortable with our close proximity and reticently consumed fish launched in their direction.

The purpose of the injected air was to ensure the tagged fish stayed on the surface after deployment, increasing the likelihood that pelicans would consume the fish (D. Teuscher,

personal communication). Pelicans were observed after they caught or scooped up a tagged fish until a swallowing motion was noted. Attempts were made to minimize the occurrence of individual birds consuming more than one tagged fish, but this was not always avoidable. In addition, no more than 40 tagged fish were deployed on a given day to maximize tag dispersion and subsequent tag recovery independence. Fed fish that were consumed by other birds (gulls and herons) were omitted as lost tags. While feeding efforts were ongoing, we also stocked 100-400 PIT-tagged catchable rainbow trout in each study water in conjunction with regularly scheduled stocking events to estimate pelican predation rates on stocked catchables (Table 4).

We searched for regurgitated and defecated PIT tags from fed and stocked fish at the breeding colonies located on Blackfoot Reservoir and Minidoka NWR, and at pelican loafing areas on some of the study waters. Areas targeted for scanning on the colonies were marked into grids using surveyor flagging, and searchers scanned the grid systematically to ensure that all of the ground was covered (D. Teuscher, personal communication). We used a backpack PIT tag reader (Oregon RFID HDX Backpack Reader), and when a tag was detected, surveyors used shovels and sieves to recover and remove the tags if they were not visible on the surface, in order to avoid interference with other PIT tags in the same area. If we were unable to recover and remove the tag, attempts were made to ensure no other PIT tags were in the area, and individual PIT tag numbers were recorded. Tags that were found in or near nests of Double-crested Cormorants *Phalacrocorax auritus* were not included as recovered tags so that our estimated predation rate would apply only to pelicans.

At loafing areas on some of the study waters, we searched areas that were dry or wadeable with the backpack PIT tag reader. In addition, we also searched near shore areas with a pontoon-mounted RFID antenna attached to the same reader (Fischer et al. 2012) to detect PIT tags that were deposited off shore.

Proportions of recovered tags were calculated independently for both the fed tags (p_x) and stocked tags (p_y), where:

p_x = number of fed tags found on the colony divided by the total number of tags directly fed to pelicans on the individual study waters

p_y = number of stocked PIT tags found on the colony divided by the total number of stocked PIT tags released in the study waters.

Variances and 90% confidence intervals (CIs) for each of these proportions were calculated according to Fleiss (1981). We calculated the pelican predation rate (PR) for each water body when both fed and stocked tags were recovered at a nesting colony according to the following formula:

$$PR = p_y/p_x$$

To calculate the variance for PR , we used the formula for the variance of a ratio (McFadden 1961; Yates 1980):

$$s^2 \left(\frac{p_y}{p_x} \right) = \left(\frac{p_y}{p_x} \right)^2 \times \left(\frac{S_{p_y^2}}{p_y^2} + \frac{S_{p_x^2}}{p_x^2} \right)$$

For each water-specific estimate of pelican predation rate, we then calculated 90% CIs.

For loafing areas, at locations where both fed and stocked tags were recovered, we were able to estimate pelican predation rate as well. Since all tags were potentially recoverable from either location, and the proportion of recoveries from fed or stocked fish allowed us to estimate predation, we assumed that either estimate provided the total pelican predation rate, regardless of whether the recoveries occurred at the colonies or the loafing areas.

Estimating angler harvest

To compare pelican predation rate to angler harvest rate, we used T-bar anchor tags to tag the same hatchery catchable rainbow trout that were stocked with PIT tags in the study waters (Table 4). For more details on anchor tagging methods and angler tag reporting rate calculations, see Meyer et al. (2010, 2012). In short, anchor tags were 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. In addition, informational posters and stickers were distributed to IDFG license vendors, regional offices, and sporting goods stores to publicize the tagging efforts, explain how the information was being used, and provide tag return instructions. Tag reporting for anglers in this program is voluntary, not mandatory.

Unadjusted angler harvest rate (u) at each stocking location was calculated as the number of tagged fish reported as harvested divided by the number of fish released with tags; variance was calculated according to Fleiss (1981). Typically, angler harvest is calculated as an annual rate, but at this time, a full year has not elapsed since tagged fish were released, therefore the estimates presented herein will be adjusted once a full year has elapsed. Adjusted angler harvest rate (u') incorporated estimates of angler tag reporting rate (λ), anchor tag loss (tag_l), and tag mortality (Tag_m) (estimated to be 49.4%, 8.2%, and 1%, respectively; Meyer et al. 2010), and used the following formula:

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

Variance estimates for λ , tag_l , and tag_m came from Meyer et al. (2010). Variance for the entire denominator in the above equation was estimated using the approximate formula for the variance of a product in Yates (1980):

$$s_{x_1x_2}^2 = x_1^2 \cdot s_{x_2}^2 + x_2^2 \cdot s_{x_1}^2$$

where $s_{x_1x_2}^2$ 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 u' was calculated using the approximate formula for the variance of a ratio as previously noted, from which 90% CIs were derived.

RESULTS

Pelican and angler counts

Total American White Pelican numbers in CJ Strike Reservoir ranged from 75 to 661 birds per survey. The highest mean monthly numbers of pelicans occurred in June and July,

which were also the months when angler use was lowest during our surveys (Figure 4). In contrast, we observed the highest mean numbers of pelicans at Cascade and Chesterfield reservoirs from July to September (Figure 4), which at Cascade Reservoir also coincided with the fewest number of anglers. Mean pelican abundance was 259, 261, and 35 for CJ Strike, Cascade, and Chesterfield reservoirs, respectively, compared to mean angler counts were 61, 21, and 8, respectively (Figure 4).

For the three ponds in south-central Idaho, mean pelican abundance was 8 at Riley Creek Pond and <1 at Connor Pond; counts were not made at Filer Pond. Mean angler count at Riley Creek Pond was 9 anglers, and counts were not made at the other ponds. Pelican presence appeared to coincide with the stocking of catchable-sized rainbow trout at Connor Pond. For example, from early April to the end of October, out of 74 days of surveying, pelicans were only present on two days in early June, before catchable rainbow trout were stocked, and for the six survey dates in mid-June immediately following catchable stocking (Figure 5).

Most of the pelicans we observed were breeding-aged birds until juveniles fledged in early August. For example, at CJ Strike Reservoir, the proportion of pelicans that were breeding-age averaged 94% prior to August 1 and 66% after August 1 (Figure 6). This was the only water body where extensive data was available. The only other data available was for two dates in June at Cascade Reservoir, and one date in July at Chesterfield Reservoir, for which the proportion of pelicans that were breeding-age was 93%, 95%, and 100%, respectively.

Pelican predation rate

We stocked a total of 1,379 PIT tagged rainbow trout into the six study waters during 2012, and directly fed 368 PIT tagged rainbow trout to pelicans in four of the six study waters (Table 4). We were unable to locate any pelicans to feed during our trips to Connor and Filer ponds.

At the nesting colonies we recovered 220 PIT tags; 33.2% were recovered at Minidoka NWR, and 66.8% were recovered at the Blackfoot nesting colony (Table 4). This constituted 10.4% of all the stocked tags and 26.9% of all the fed tags from 2012. Five tags were recovered in or directly adjacent to cormorant nests at the Minidoka colony and were not included in pelican predation calculations.

There was a negative exponential relationship between the distance from the nearest nesting colony and fed tag recovery efficiency at the study water (Figure 7), although this relationship was based on only four data points. We recovered 23 additional tags at loafing areas from two of the study waters (CJ Strike and Cascade reservoirs; Table 2), but only at CJ Strike Reservoir were both stocked and fed tags recovered, which allowed us to calculate a predation rate at that location based only on loafing area tag recoveries.

We were unable to calculate predation rates at all tag release locations because we did not recover both stocked and fed tags from all sites at the breeding colonies (Table 4). At Connor and Filer ponds, we used the aforementioned relationship between tag recovery efficiency and colony distance from study water (Figure 7) to predict tag recovery efficiency at these study waters, and then estimated predation rates based on stocked tag recoveries at Minidoka NWR from these study waters.

Estimated predation rates by American White Pelicans on stocked rainbow trout in the study waters were low at Cascade Reservoir, Filer Pond, and Riley Creek Pond and high at

Chesterfield Reservoir and Connor Pond (Table 5). The estimate of pelican predation at CJ Strike Reservoir was high based on loafing area tag recoveries and low based on nesting colony tag recoveries, but both estimates were based on only a few recovered tags and we consider both of these estimates to be suspect. When considering only those estimates derived from tags recovered at the nesting colonies, pelican predation rate was strongly and inversely related to the study waters' distance from the nearest Idaho breeding colony (Figure 8).

From PIT tag recoveries at the nesting colonies at Minidoka NWR, we determined that some pelicans traveled at least 203 km one way to forage. We recovered tags at the Blackfoot Reservoir nesting colony that originated 29 km away (Chesterfield Reservoir), but recovered no tags from other study waters.

Angler harvest rate

In general, at study waters where pelican predation was estimated to be high, angler harvest rate was estimated to be low, and vice versa where pelican predation was low (Table 5). One exception to this was at Cascade Reservoir where both pelican predation and angler harvest were low; previous estimates of angler harvest at Cascade Reservoir have repeatedly demonstrated that angler harvest of catchable trout is consistently low at this location. Another somewhat anomalous study water was CJ Strike Reservoir, where angler harvest was moderately high and pelican predation (based on tag recoveries at loafing areas) was also high. This incongruity at CJ Strike Reservoir further undermines the pelican predation estimate there that was based on tag recoveries from loafing areas. Despite the tenuous nature of both estimates of pelican predation at CJ Strike Reservoir, angler harvest was negatively correlated with pelican predation with the inclusion of 1) the CJ Strike Reservoir loafing estimate ($r = -0.86$), 2) the CJ Strike Reservoir colony estimate ($r = -0.76$), or 3) the average of the two estimates ($r = -0.95$). As noted earlier, the angler harvest estimates are not yet annual estimates because a year has not elapsed from the time the catchables were stocked.

DISCUSSION

Our results suggest that predation by American White Pelicans on hatchery trout stocked at catchable size in some southern Idaho waters may be high, and except for Cascade Reservoir, where predation rates are high, angler harvest rates are low. In Wyoming, pelicans (and cormorants) quickly increased their focus on trout species as soon as hatchery trout were stocked, and consumed an estimated 80% of the trout stocked in one year (Derby and Lovvorn 1997). Although our study includes results from only a handful of study waters and should therefore be considered preliminary, our findings support the supposition that in southern Idaho, pelican predation of trout stocked at catchable size will negatively affect angler harvest of these fish. Additional years of data may help clarify the strength and the geographical extent of this relationship.

Estimated predation rates by American White Pelicans on stocked rainbow trout in the study waters were variable and apparently were related to the distance of the study water to a breeding colony in Idaho. Results from CJ Strike Reservoir suggest that pelican predation in Idaho is almost exclusively associated with breeding-aged birds (Figure 6). This concurs with findings from patagial tagged pelicans in Idaho which suggest that pelicans do not return to Idaho after fledging until about age-4 (C. Moulton, IDFG, unpublished data).

American White Pelican abundance in the three reservoirs we surveyed varied by month, but in general showed a pattern of lower abundance in early spring and late fall, with peak abundance during the summer. This trend coincides with the activities associated with the breeding season and associated foraging behavior of pelicans. During May and June, breeding pelicans in southern Idaho are selecting nesting sites, incubating eggs, and are occupied with early chick rearing, all of which require the presence of one adult on the nest at all times (Madden and Restani 2005). When pelican chicks reach four weeks of age, both parents begin leaving the breeding colony for foraging trips simultaneously (O'Malley and Evans 1982); this period coincides with peak abundances observed at our reservoirs, and therefore likely represents the time of greatest predation potential on fish stocks within those reservoirs.

It was interesting that at CJ Strike Reservoir, which is 201 km from the nearest pelican nesting colony, the proportion of breeding-aged birds plummeted in early August at the same time that total numbers of pelicans dropped precipitously (Figure 6). Surprisingly, the proportion of birds that were of breeding age quickly rebounded, whereas total numbers remained low for the duration of the season. A pattern of decline in both pelican numbers and the proportion of pelicans of breeding age might be expected if adults accompanied juveniles as they fledged, and juveniles immediately after fledging had a short range of travel. However, the immediate return in total numbers of pelicans was surprising and does not support this explanation. An alternative explanation may be that adults using CJ Strike Reservoir during the summer headed south once their chicks fledged, leaving fewer adults present and thus a higher percentage of non-breeding aged birds, and then a new group of adults moved in that had finished breeding elsewhere (presumably north of CJ Strike Reservoir).

The maximum recorded distance of which we are aware that American White Pelicans have been shown to travel one way from nesting colonies to foraging areas is 305 km (Johnson and Sloan 1978), suggesting that all of the reservoirs and ponds in southern Idaho are vulnerable to pelican predation (Table 3). One of the goals of the first year of this study was to determine how far pelicans traveled from the breeding colonies to reach foraging areas. PIT tag recoveries from the breeding colony at Minidoka NWR showed that pelicans traveled a maximum distance of 203 km to forage at CJ Strike reservoir. We recovered 7 PIT tags (1 stocked fish, 6 fed fish) from CJ Strike, suggesting some regular foraging at the reservoir by pelicans from the Minidoka NWR colony. Although we did not recover PIT tags at the Blackfoot Reservoir colony from study waters >30 km away, it is likely that pelicans from that colony foraged at other locations. In 2013, several additional study waters will be added to the study design throughout southern Idaho, which will help clarify distances travelled for birds at each Idaho colony.

There are other American White Pelican breeding colonies outside of Idaho that are within the foraging range of many southern Idaho reservoirs, the largest being colonies at Gunnison Island in the Great Salt Lake in northern Utah, Malheur National Wildlife Refuge (NWR) in eastern Oregon, Badger Island on the Columbia River in southeastern Washington, and Molly Island on Yellowstone Lake in northwestern Wyoming (Figure 3). It is likely that breeding adults from these colonies forage in Idaho reservoirs, and their impact is currently unknown. For instance, we did not recover any stocked or efficiency tags from Cascade Reservoir at Idaho breeding colonies, but we regularly observed 200-500 pelicans (including birds with the phenotypic characteristics of breeding adults) at Cascade Reservoir during June-September. The most likely sources of breeding adults that may forage at Cascade Reservoir would appear to be the Badger Island and Malheur NWR colonies; however, scanning for tags in fall 2012 by staff at Oregon State University resulted in no recoveries of fed or stocked fish from Cascade Reservoir (A. Evans, Real Time Research, personal communication). The

pelicans using the breeding colony on Gunnison Island in the Great Salt Lake, Utah, also pose a potential threat to southern Idaho fisheries due to the large number of pelicans (8,000 nesting pairs in 2005) nesting there (King and Anderson 2005) and its close proximity to numerous reservoirs in southeastern Idaho. Scanning these additional colonies in future years would help gain a better understanding of the full impact of American White Pelican predation at a wider geographical scale.

We assumed that all tags we recovered that were not in or directly adjacent to Double-crested Cormorant nests were deposited by American White Pelicans. However, some tags at the breeding colonies were recovered that could not be directly assigned to pelican nests or cormorant nests. We assumed these tags were deposited by pelicans, but if some were actually deposited by cormorants, then we may have overestimated pelican predation in some study waters. Using the Minidoka NWR nesting colony as an example, only one study water (Conner Pond) was within 40 km of the Minidoka nesting colony, which is the maximum recorded foraging distance for Double-crested Cormorants from nests (Custer and Bunck 1992). At the Minidoka colony, all five tags ascribed to cormorant deposition came from Conner Pond, compared to 39 tags from Conner Pond ascribed to pelican predation. The additional bioenergetic demands for nesting pelicans compared to nesting cormorants (Derby and Lovvorn 1997) and the greater foraging distance for pelicans further bolsters the argument that pelicans likely consumed more of the PIT tags whose origin was questionable than did cormorants. Taken collectively, we believe that it is unlikely that a substantial portion of the predation we attribute to pelican predation was actually the result of cormorant predation.

RECOMMENDATIONS

1. Search additional American White Pelican nesting colonies outside Idaho for our PIT tags since they are within the distance that nesting pelicans are capable of traveling to forage.
2. During the 2013 field season, expand the number of study waters where catchable trout are stocked and pelicans are fed in order to increase sample size and strengthen final conclusions regarding pelican predation of fish in southern Idaho.
3. Attempt decoying to feed American White Pelicans at small ponds in 2013 to allow better estimation of exploitation in those locations where pelicans are repeatedly absent although anecdotal evidence suggests that pelican predation rate is high.
4. Focus more effort on feeding only one fish to each American White Pelican.
5. Improve the quality of cormorant counts, pelican pictures (for aging), and angler data to put pelican counts and predation into better context.
6. Install cameras as the Minidoka nesting colony to help parse future tag recoveries into cormorant- or pelican-consumed fish.

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TABLES

Table 3. Distance (km) from Idaho American White Pelican colonies to southern Idaho ponds and reservoirs that were evaluated for pelican predation of catchable rainbow trout stocked in 2012. Bold data indicates which breeding colonies were evaluated for pelican predation.

Nearest colony		Study waters						
		Cascade Reservoir	CJ Strike Reservoir	Riley Creek Pond	Filer Pond	Connor Pond	Chesterfield Reservoir	Blackfoot Reservoir
Name	State							
Canyon Ferry	Montana	412	524	490	492	470	450	397
Yellowstone NP	Wyoming	459	483	415	403	353	260	205
Island Park Reservoir	Idaho	363	385	323	314	271	217	163
Blackfoot Reservoir	Idaho	412	354	274	252	189	27	0
Minidoka Reservoir	Idaho	304	201	118	95	31	132	153
Great Salt Lake	Utah	417	283	206	178	124	116	169
Malheur NWR	Oregon	260	242	325	351	417	575	587
Columbia River	Washington	274	425	487	513	563	692	679

Table 4. Initial numbers of stocked (PIT tagged and anchor tagged) and fed hatchery rainbow trout and PIT tags recovered from American White Pelican nesting colonies or loafing areas in select Idaho waters in 2012.

Study waters	PIT Tags		Tags recovered																	
	implanted		CJ Strike ^a		Cascade ^a		Chesterfield ^b		Filer ^b		Riley Creek ^b		Connor ^b		Minidoka NWR		Blackfoot		Totals	
	Stocked	Fed	Stocked	Fed	Stocked	Fed	Stocked	Fed	Stocked	Fed	Stocked	Fed	Stocked	Fed	Stocked	Fed	Stocked	Fed	Stocked	Fed
CJ Strike Reservoir	399	100	6	2	0	0									1	6	0	0	7	8
Cascade Reservoir	393	104	0	0	0	14									0	0	0	0	0	14
Chesterfield Reservoir	287	100	0	0	0	0									0	0	86	61	86	61
Filer Pond	100	0	0	0	1	0									3	-	0	0	4	-
Riley Creek Pond	100	64	0	0	0	0									2	16	0	0	2	16
Connor Pond	100	0	0	0	0	0									39	-	0	0	39	-
Total or average	1379	368	6	2	1	14									45	22	86	61	138	99

^aTag recovery efforts involved loafing areas only.

^bNo tag recovery efforts were made.

Table 5. Estimates of American White Pelican predation and angler harvest in select Idaho waters in 2012.

Study waters	Hatchery release date	Pelican predation (%)				Angler harvest (%)	
		Nesting colony		Loafing areas		Estimate	90% CI
		Estimate	90% CI	Estimate	90% CI		
CJ Strike Reservoir	5/29/2012	4.2 ^a	7.2	75.2	5.1	20.4	5.5
Cascade Reservoir	5/29/2012	0.0		0		1.1	1.3
Chesterfield Reservoir	5/23/2012	49.1 ^b	9.7	0		4.7 ^e	3.1
	5/23/2012	"	"	"		4.5 ^e	2.1
Filer Pond	6/13/2012	10.5 ^c		- ^d		53.7	16.1
Riley Creek Pond	6/14/2012	8.0 ^a	9.6	- ^d		51.4	15.9
Connor Pond	6/14/2012	67.7 ^c		- ^d		6.7	6.3

^aTag recovery from Minidoka nesting colony.

^bTag recovery from Blackfoot nesting colony.

^cPredation estimate extrapolated from relationship between tag recovery rate and distance from colony (Figure 4).

^dNo attempts to recover tags at loafing areas were made.

^ePart of two estimates with (first estimate) and without (second) PIT tags; similarity of estimates demonstrates lack of mortality attributable to PIT tagging.

FIGURES

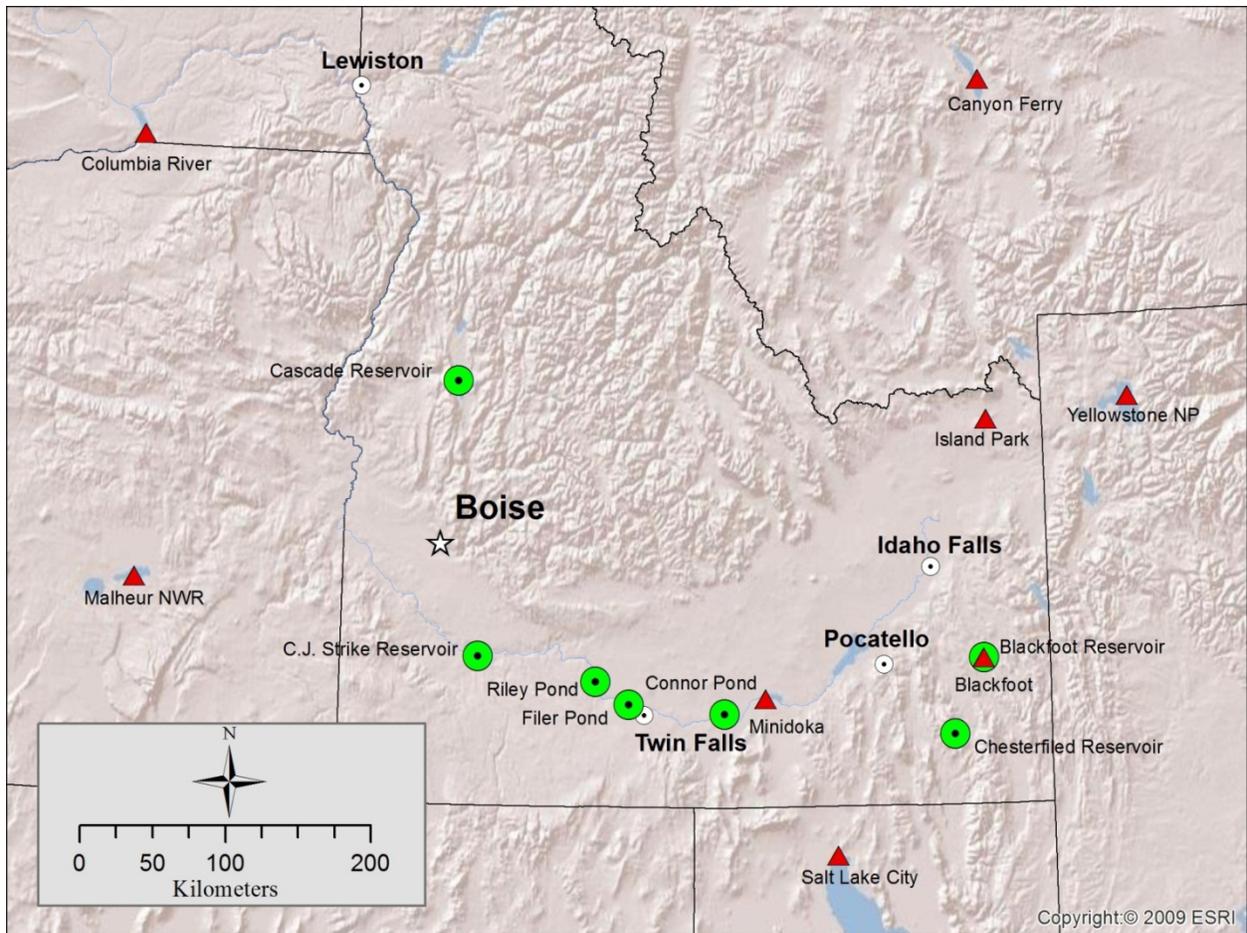


Figure 3. Map depicting study waters (circles) in southern Idaho and surrounding American White Pelican nesting colonies (triangles).

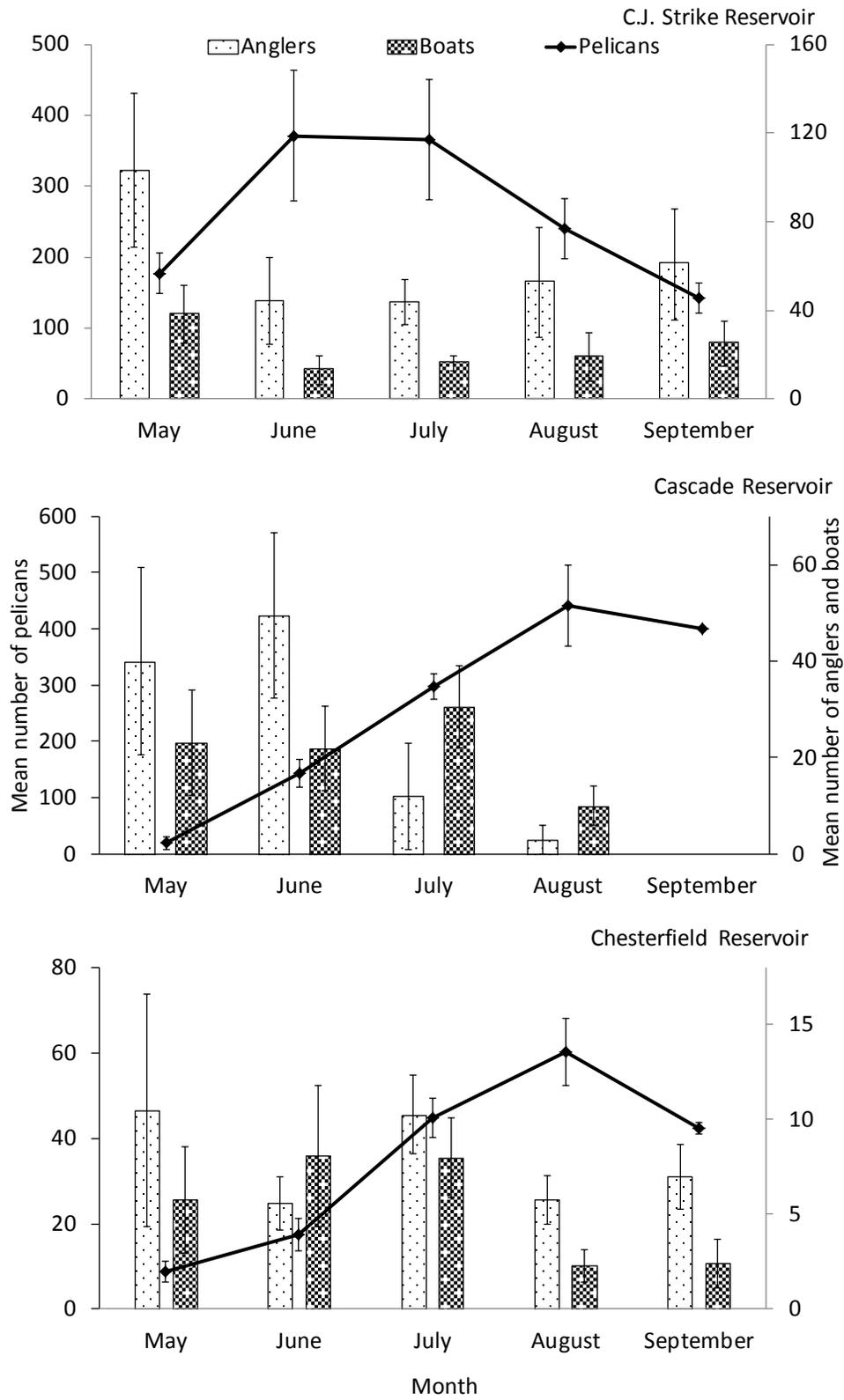


Figure 4. Mean monthly numbers (\pm SE) of American White Pelicans, boats, and anglers surveyed at C.J. Strike, Cascade, and Chesterfield reservoirs during 2012.

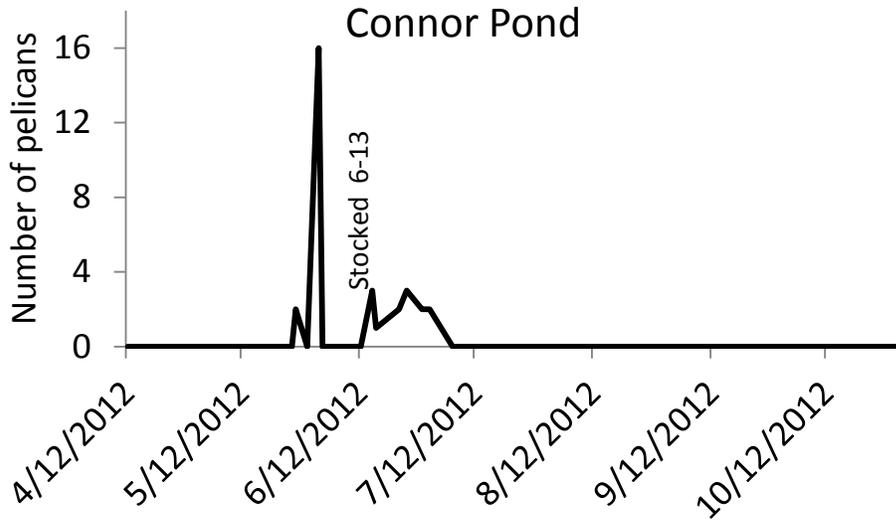


Figure 5. Number of American White Pelicans counted at Connor Pond in 2012. Figure also indicates when catchable stocking occurred.

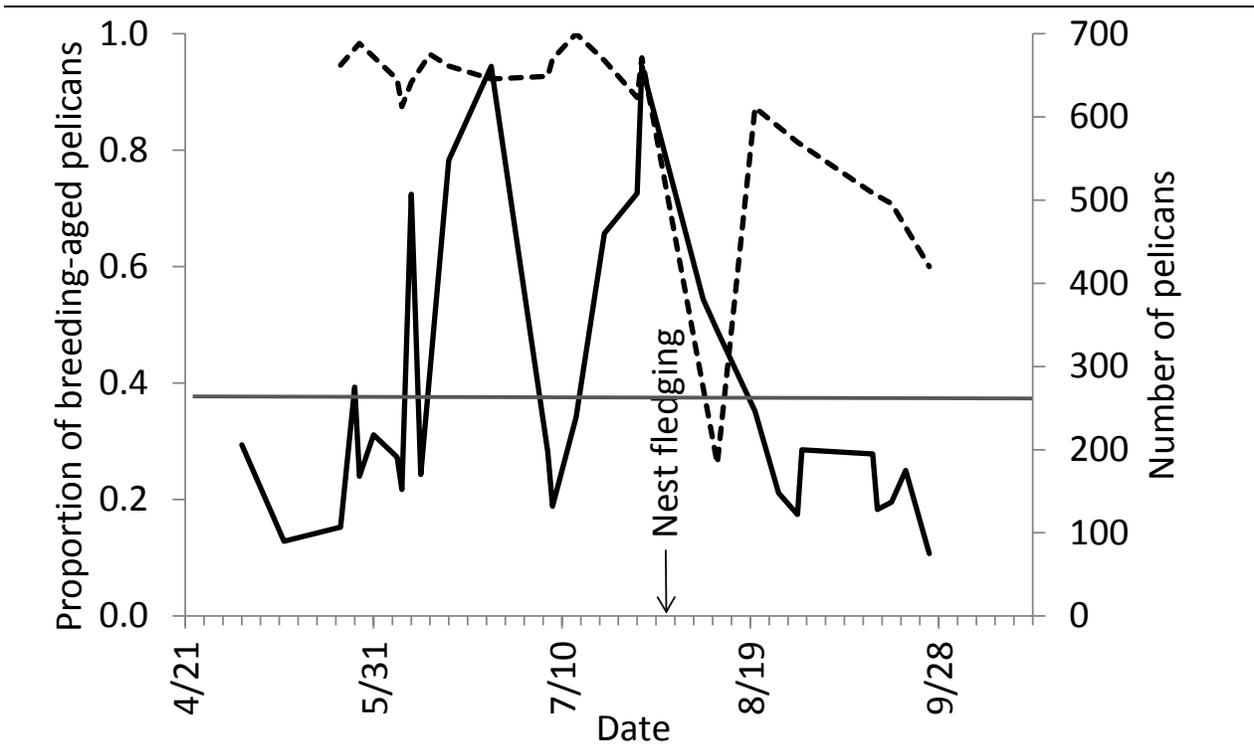


Figure 6. The proportion of breeding-aged pelicans (dashed black line) and the total number of pelicans (solid black line) at CJ Strike Reservoir in 2012, before and after juveniles fledged from Idaho nesting colonies. Breeding-aged birds were identified by digital photos. The solid gray horizontal line depicts the mean number of pelicans at CJ Strike Reservoir during the study period.

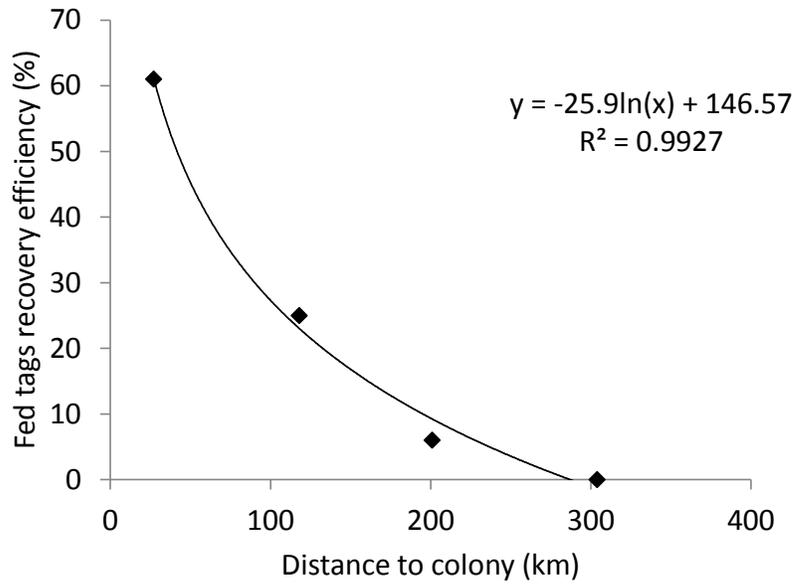


Figure 7. Relationship between study waters' distance from nearest colony and American White Pelican nesting colony recovery efficiency from PIT tags implanted in hatchery rainbow trout and fed directly to pelicans.

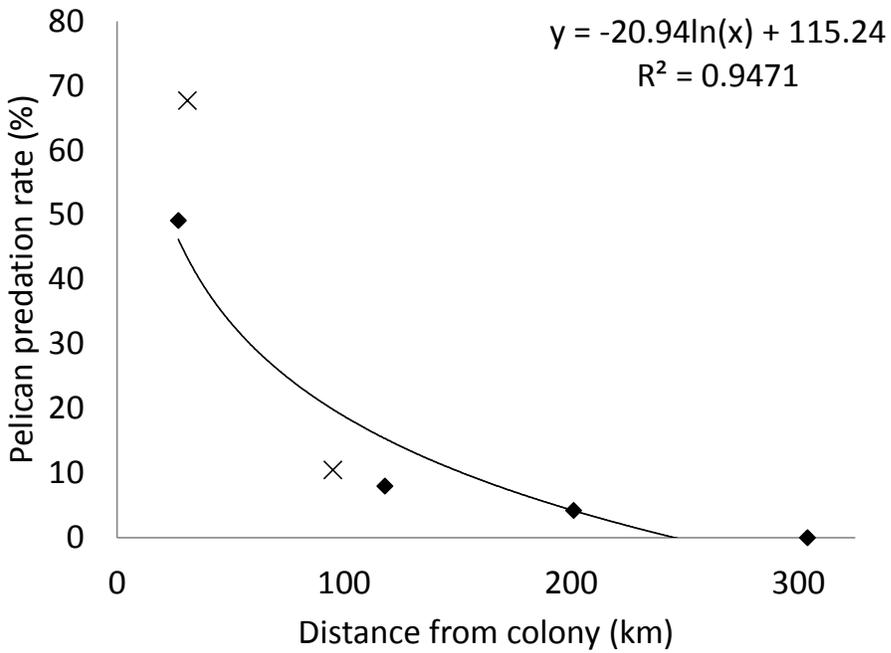


Figure 8. Relationship between study waters' distance from colony and American White Pelican predation rate on stocked rainbow trout. The "x" data points were not used to build the equation but rather were generated from the relationship in Figure 6 for the two waters where fed tags were not achieved.

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