FISHERY RESEARCH

WILD TROUT COMPETITION STUDIES

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Report Period July 1, 2009 to June 30, 2010

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Wild Trout Competition Studies

Annual Performance Report

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Grant # F-73-R-32

Project 2: Wild Trout Investigations

Subproject #1: Competition between Wild and Hatchery Rainbow Trout

Subproject #2: Hooking Mortality Comparisons with Circle Hooks

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SUBPROJECT #1: COMPETITION BETWEEN WILD AND HATCHERY RAINBOW TROUT

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ABSTRACT

Idaho Department of Fish and Game has proactively dealt with potential adverse genetic effects of stocking hatchery trout in waters that support wild salmonids by adopting in 2001 a policy whereby only rainbow trout *Oncorhynchus mykiss* pressure treated to induce sterility are stocked in flowing waters; however, concerns remain regarding competitive effects of introducing hatchery trout into streams and rivers supporting wild trout. We stocked fish in the middle three years of a five-year study to assess if stocking hatchery rainbow trout of catchable size (i.e., catchables) reduced wild rainbow trout abundance, survival, growth, or recruitment in streams. Catchables (averaging 249 mm total length; TL) were stocked from 2006-2008 at a density of 4.2 fish/100m^2 into 12 treatment reaches of stream, which were paired with control reaches in the same stream (3 km apart) where no stocking occurred. Wild rainbow trout abundance (including all fish ≥75 mm TL), recruitment, survival, and growth were determined with population estimates and PIT-tagged recaptures during mark-recapture electrofishing sampling. Wild trout abundance averaged 13.2 fish/100m^2, but ranged substantially across all sites in all years, from a low of 0.5 to a high of 131.3 fish/100m^2; similar variability was observed in recruitment to age-1. Total annual survival averaged 0.53 for estimates based on population abundance (which allowed for emigration and immigration) and 0.26 for estimates based on PIT-tagged recaptures (which allowed for emigration but not immigration). Our paired study design demonstrated that wild rainbow trout abundance, survival, growth, and recruitment to age-1 were all unaffected by stocking catchables. The lack of population-level effects from stocking catchables on wild fish was not surprising considering the high short-term mortality and socially and physiologically naive behavior typically exhibited by hatchery catchables stocked in lotic systems.

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INTRODUCTION

Maintaining put-and-take fisheries in streams, ponds, lakes, and reservoirs that cannot withstand the harvest demands of anglers is a widely accepted use of hatchery-reared fish (Utter 1994; Epifanio and Nickum 1997). However, supplementing wild trout stream fisheries with hatchery trout raises concern over potential adverse genetic and ecological effects (Krueger and May 1991; Allendorf 1991; Weber and Fausch 2003). Since 2001, the Idaho Department of Fish and Game (IDFG) has proactively dealt with potential adverse genetic effects of hatchery rainbow trout *Oncorhynchus mykiss* on existing native trout populations by only stocking hatchery rainbow trout that have been treated to induce sterility (Kozfkay et al. 2006), however, possible adverse competitive interactions in streams and rivers between wild and stocked salmonids have not been addressed.

Competition, by definition, causes a reduction in fitness of an organism due to the limited supply of a resource held in common with other organisms, or the limited ability to exploit a resource because of interference by other organisms (Birch 1957). Reduced fitness levels in wild trout populations could translate to decreased survival, growth, or reproduction (Moyle and Cech 1982). Most competition studies have indirectly assessed changes in fitness levels, or found evidence of competition by inferring causal relationships between fitness and characteristics such as ability to maintain favorable positions (Griffith 1972; Fausch and White 1986; Peery and Bjornn 1996), win agonistic bouts (Griffith 1972; Mesa 1991; McMichael et al. 1999), gain weight (Dewald and Wilzbach 1992; Harvey and Nakamoto 1996), or survive (Kocik and Taylor 1994). These studies at the microhabitat scale are much easier to replicate with different manipulations of fish compositions and densities to test interspecific and intraspecific competition. However, they do not directly address concerns at the population level (Fausch 1998), a scale at which competition investigations are rarely performed (Schoener 1983). Of the population-level studies conducted on competition between hatchery and wild trout, the two foremost of which we are aware have contradicting conclusions. Vincent (1987) concluded hatchery trout decreased the abundance and biomass of wild rainbow trout and brown trout *Salmo trutta* in the Madison River and O'Dell Creek, Montana, while Petrosky and Bjornn (1988) concluded that catchables had little effect on wild cutthroat trout *O. clarkii* in the St. Joe River and wild rainbow trout in Big Springs Creek, Idaho.

In 2005, a total of 2,200,000 catchable-sized (i.e., 200-300 mm) sterile rainbow trout (hereafter catchables) were stocked in 323 waters in Idaho by IDFG. Most (64%) of these waters were lentic systems, but IDFG stocks more than 500,000 catchables into streams annually (IDFG unpublished data). Much of this stocking occurs in stream reaches where pure native salmonids no longer exist, or where environmental conditions can no longer support healthy wild salmonid populations. In those stocked streams in Idaho that do support wild salmonids, we questioned whether stocking hatchery rainbow trout would affect wild populations at a measurable level and in a meaningful way. Specifically, we tested for population-level competition effects of stocked catchables on wild rainbow trout populations by quantifying wild rainbow trout population abundance, survival, growth, and recruitment over several years.

OBJECTIVES

1. Determine whether stocking of hatchery rainbow trout catchables appreciably reduces the abundance, growth, survival, or recruitment in wild rainbow trout populations.

METHODS

The study area for this investigation was the upper Snake River basin in southern Idaho. Eleven study streams were used to establish 12 paired treatment and control reaches that ranged from 1,094 to 2,104 m in elevation, from 0.5 to 6.2% in gradient, and from 30 to 360 µS/cm in specific conductivity (Table 1). Study reaches were on average 719 m in length, which we felt would be long enough to estimate the trout population metrics in question and apply them to the population level. Streams were grouped by three angling regulation categories: general (six fish bag limit), wild trout (two fish limit), and catch-and-release (Table 1). The three streams in the latter category were not explicitly managed with catch-and-release regulations, but they functioned as such because of slow fish growth and a 356 mm minimum length limit for two streams, and very limited public access and fishing pressure for the third stream. During our analyses, differences in the effects of competition among the three fish regulation categories were not apparent; hence, results from all study streams were eventually combined. The fishing season for all streams was Memorial Day through November 30.

In addition to the angling regulation categories, selection criteria for study streams were: 1) the stream was not already stocked nearby (i.e., within 5 km of any study reach) and had not been in the last 10 years; 2) 3 stream km could be established between two study reaches (one treatment and one control) on each stream; and 3) rainbow trout dominated the salmonid composition of the stream. In our study streams, rainbow trout on average comprised 94% of the trout population, but brook trout *Salvelinus fontinalis*, bull trout *Salvelinus confluentus*, brown trout, and cutthroat trout were also sometimes present. Other species we encountered included mountain whitefish *Prosopium williamsoni* and several species of Cottidae, Catostomidae, and Cyprinidae.

We set the distance between sites (≥3 km) to be far enough that few stocked catchables would be expected to move from a treatment to a control site (High and Meyer 2010). The treatment (i.e., stocking) reach was randomly assigned to one of the paired reaches except at Badger Creek, where logistical constraints of planting catchables required that the upper site serve as the treatment.

From 2006-2008, catchable-sized rainbow trout were stocked into the middle of treatment reaches three times during the growing season at monthly intervals with a mean monthly stocking density of 1.4 fish/100m^2 (range 0.9-2.4). Stocking density and intervals were based on typical stream stocking by IDFG, with density selected to be at the high end of ongoing stocking rates, thus serving as a worst-case scenario for competitive impacts to wild trout populations. Moreover, an equal number of catchables were also planted at the upstream and downstream reach boundaries to account for dispersal in order to maintain an elevated catchable density. In 2006, mean size of a subsample of stocked catchables was 249 mm total length (SD = 31; range 89-377 mm; n = 853), and based on hatchery records, year-to-year variation in fish size was minimal. To avoid impacting wild trout populations solely by increasing harvest after catchables were stocked (Butler and Borgeson 1965; Carline et al. 1991; Baer et al. 2007), we did not advertise any stocking events or locations.

From 2005-2009, incorporating one year before and after stocking, salmonid populations were sampled using backpack and canoe-mounted electrofishing gear for conducting mark-recapture (M-R) population sampling. To minimize the effect that seasonal changes can have on fish abundance (Decker and Erman 1992), paired sites were sampled in the same week, and sampling was repeated each year within a few weeks of the same calendar date for each paired location. All captured salmonids were identified to species, measured to the nearest mm (TL), and weighed to the nearest 0.1 g using a top-loading digital scale. Fish scales were collected from a subsample of fish for aging purposes. Passive integrated transponder (PIT) tags were implanted intraperitoneally in most captured wild rainbow trout ≥ 75 mm from 2006-2008 to estimate growth and survival in subsequent years. For each reach, one recapture run was made 1-2 days after the marking run. We used the Fisheries Analysis Plus program (Montana Fish, Wildlife, and Parks 2004) to calculate population estimates and 95% confidence intervals (CIs) using the Lincoln-Petersen M-R model as modified by Chapman (1951). In nearly every instance, we were able to create size groups (generally 25-50 mm) meeting the criteria that (1) the number of fish marked in the marking run multiplied by the catch in the recapture run was ≥four times the estimated population size, and (2) at least three recaptures occurred per size group; meeting these criteria creates modified Petersen estimates that are <2% biased (Robson and Regier 1964). Estimates were run separately for each species when possible. Size of age-0 rainbow trout varied considerably between streams over the course of each year and between sample years, but generally they were <75 mm TL. For consistency, 75 mm was subsequently used as the cutoff for population estimates to minimize bias from the variability in size of age-0 fish.

As an additional index of population size, we calculated percent habitat saturation (PHS) as developed by Grant and Kramer (1990):

$$PHS = \Sigma\ D \times T \times 100$$

where D is the density of fish (no./m²) and T is the territory size, with territory size estimated from Grant and Kramer (1990) as:

$$\log_{10}(\text{territory size}) = 2.61 \times \log_{10}(\text{fork length (cm)}) - 2.83 \times 1.19$$

We included PHS because it incorporates several parameters (i.e., fish size, fish density, and territoriality) that likely reflect carrying capacity within study reaches better than density alone (Grant and Kramer 1990). Because we recorded data as total length, fork length (FL) for rainbow trout was calculated using the equation FL = TL/1.049 (Carlander 1969).

Scale aging was used to separate age-1 fish from other cohorts. Otoliths were not used because killing fish each year to obtain otoliths could have altered population densities and other population metrics in subsequent years. Although scales usually underestimate age for older rainbow trout (e.g., Hining et al. 2000; Cooper 2003), they tend to be a reasonably accurate aging structure in Idaho for rainbow trout age-2 or less (Schill et al. 2010). Moreover, we assumed that any directional bias in scale aging would not have differed between treatment and control reaches. Scales were independently aged by two readers with no knowledge of fish length. Disparities were reconciled by a third reader using all available information (including fish length of the scale being aged, fish length frequency at the site, and sampling date) to agree on a final age. We compared aging agreement between the two initial readers by calculating the between-reader coefficient of variation (Chang 1982). Because length usually overlapped slightly from age-0 to age-1 and from age-1 to age-2, age-length keys with 10 mm length groups were used to allocate ages (i.e., age-0, age-1, or age-2 and older) for all fish in the overlapping length groups to develop size cutoffs before population estimates were made.

Using the above aging information, we computed separate population estimates for age-1+ and age-2+ fish in order to estimate total annual survival (*S*), which was calculated for each year using Heincke's estimator $S = (N - N_1)/N$ (Ricker 1975), where N and N_1 are the abundance of age 1+ and age 2+ fish, respectively. Survival for fish age-1 and older was assumed to be constant across all age classes. Following Carline (2006), we used the variance from the population estimates to estimate the variance of *S* by employing a Taylor series approximation (Som 1996). Recruitment to age-1 was estimated by computing population estimates for fish only within the age-1 size group.

For a second estimate of survival, we used capture histories of PIT-tagged fish to estimate apparent survival (ϕ) and recapture (p) probabilities using open population Cormack-Jolly-Seber (CJS) models (Lebreton et al. 1992) in Program MARK (White and Burnham 1999). We used an information–theoretic approach (Burnham and Anderson 2002) to rank candidate models (Appendix A). We constructed single stage models (i.e., fish of all stages grouped together) that had stream reach and treatment effects. We determined goodness–of–fit of the global model, $\phi(a \times g \times t)$, $p(a \times g \times t)$, by estimating the overdispersion parameter (median ĉ) using a simulation procedure in Program MARK, where a = area (stream reach), g = treatment or control group, and t = time (year). We also constructed a more parsimonious additive model $\phi(a + g + t)$, $p(a + g + t)$ plus 10 additional models that were more parsimonious than the global model. Since we did not PIT tag fish in 2005, and in program MARK ϕ is not estimable between the last two encounters because it is confounded with p in the CJS model (Amstrup et al. 2005), only two estimates of ϕ could be made for each study reach (2006-2007 and 2007-2008). There was no evidence of overdispersion in the PIT tag recapture data, as evidenced by the global model median $\hat{c}$ value of 1.07; we therefore did not adjust estimated standard errors and model-selection-parameter estimates. Only results from the global model were used to estimate ϕ since it had 94% of the model weight (second best model was 5.62 ΔAIC_c units lower).

Our methodology assumed that within-stream rates of recruitment, survival, emigration, and immigration were consistent for each of the control and treatment reach pairs among years, and that any differences between control and treatment pairs in the magnitude of fluctuations in abundance, recruitment, and survival were due to stocking. For estimates of ϕ, we assumed no PIT tags were lost. Although we estimated annual PIT tag loss from 2006 to 2007, we could not estimate annual loss in later years because after 2007 fish with missing tags could not be tied to a specific year. Thus, we did not correct ϕ for PIT tag loss.

Data Analysis

To assess the effects of stocking on abundance, PHS, recruitment, and survival, we used repeated measures analysis of variance (ANOVA). The experimental unit was each survey we conducted within each study reach, using year and stream as blocking factors to control for the variation in the response variable due to these effects. A statistically significant interaction between stocking treatment (stocked or unstocked) and year was used to indicate a stocking effect on the response variable. Because MARK produced only two estimates of ϕ (2006-2007 and 2007-2008), we used a paired *t*-test rather than repeated measures to assess differences in ϕ between treatment and control groups.

To assess possible effects of stocking on growth, we used Analysis of Covariance (ANCOVA) on PIT-tag recapture data. The experimental unit was each recaptured PIT-tagged fish (n = 1,612) and the response variable was the growth (in mm) from year x to year x+1, using year and stream as blocking factors and initial fish length (at tagging) as a covariate, and testing for differences in growth between treatment and control reaches. We also calculated relative weight (W_r) according to Anderson and Neumann (1996) and assessed differences in fish condition by comparing mean W_r between treatments and controls with a paired *t*-test.

As an additional assessment of fish growth, we compared mean length at age between control and treatment reaches. All PIT-tagged fish were retained in the recapture run in 2009 (n = 385); they were overdosed with MS-222 and transported to the laboratory where otoliths were removed. Additional fish at some locations were also retained for aging because sample size was inadequate at a number of reaches using only PIT-tagged fish. Age estimates were derived by examining digital images of whole otoliths at 40X magnification using a Leica DC 500 digital camera mounted on a Leica DM4000B compound microscope. Otoliths were immersed in water and illuminated under oblique reflected fiber optic light. The outer edge of each translucent zone was counted as an annulus and fish were assumed to reach age-1 on 1 January (Devries and Frie 1996). As with scales, otoliths were independently aged by two readers with no knowledge of fish length, and disparities were reconciled by a third reader using all available information (including fish length, and fish length frequency at the site) to agree on a final age. Mean length at age-2 and age-3 were calculated for all reaches where at least three fish from each age were available. Mean length at age was compared between control and treatment study reaches with a paired *t*-test. We only used age-2 and age-3 for these analyses because (1) these fish had been alive for 2-3 summers of stocking, and (2) there tended to be adequate sample sizes for these ages compared to older fish.

RESULTS

Sampling conditions, capture efficiency, and scale aging results varied little among years. Based on length frequency and scale aging, the transition from age-0 to age-1 and age-1 to age-2 were on average 90 mm (range 63-145 mm) and 153 mm (range 106-245 mm), respectively. The transitions were consistent between years, with coefficients of variation (CV) averaging 7.0% for age-0 to age-1 transitions and 8.2% for age-1 to age-2 transitions. Between-reader CV for scale age estimates (used only to calculate population estimates of separate age classes) ranged from 0.6% to 45.1% and averaged 10.7%.

Density of all wild rainbow trout among study streams averaged 13.2/100m^2 and ranged from 0.5 to 131.3 throughout the study (Figure 1). Density was generally highest in Rock Creek (mean = 41.1/100m^2) and Willow Creek (mean = 24.4/100m^2) and lowest in the Middle Fork Boise River (mean = 1.6/100m^2). Over the study period, mean density increased on average from 11.4/100m^2 in 2005 to 19.5/100m^2 by 2007, then decreased to 11.3/100m^2 in 2009. During the three stocking years, treatment reaches were on average 22% higher in abundance than the unstocked years, compared to an average of 24% higher for control reaches. Accordingly, the density of wild rainbow trout was unaffected by the stocking of hatchery catchables (repeated measures ANOVA; F = 1.23; df = 23; P = 0.37). Mean PHS among all sites was 19.8, ranged from 1.4 to 80.9 (Figure 2), and was also unaffected by stocking catchables (repeated measures ANOVA; F = 1.60; df = 23; P = 0.27).

Recruitment to age-1 increased initially, from an average of 6.4/100m^2 in 2005 to 12.4/100m^2 by 2007, then declined to 5.4 by 2009 (Figure 3). Most of this increase in 2007 was due to the large increase at the control reach in Willow Creek, likely related to substantial new beaver activity in the reach; without this site, mean recruitment to age-1 in 2007 was 7.2/100m^2. Recruitment to age-1 was relatively consistent from year to year, with the CV for recruitment at each site averaging 48.8, but was not as consistent as total abundance (average CV = 36.8). Recruitment of wild rainbow trout to age-1 was not affected by the stocking of hatchery catchables (repeated measures ANOVA; F = 0.29; df = 23; P = 0.88).

Estimates of survival based on scale aging and population abundance estimates ranged widely, from 0.0 to over 0.99 and with a mean across all study reaches and years of 0.55 (Figure 4). Mean S from 2005 to 2009 was 0.63, 0.49, 0.53, 0.57, and 0.48 for the control group (overall mean = 0.54), compared to 0.61, 0.54, 0.51, 0.62, and 0.51 for the treatment group (overall mean = 0.56). Stocking hatchery catchables did not have any effect on S (repeated measures ANOVA; F = 0.41; df = 23; P = 0.80). Similarly, modeling results from Program MARK indicated that stocking hatchery catchables also did not influence apparent survival (Figure 5), with mean ϕ of 0.25 (SE = 0.22) and 0.27 (SE = 0.21) for control and treatment reaches, respectively (paired t-test; t = 2.07; df = 23; P = 0.67). Much of the difference between S (mean = 0.53) and ϕ (mean = 0.26) was likely attributable to annual PIT tag loss, which from 2006 to 2007 averaged 19% and ranged from 8-33%.

A total of 1,612 PIT-tagged wild rainbow trout were recaptured one year after initial tagging. On average, fish grew 44 mm from one year to the next, and this growth appeared to be linear (Figure 6). Although growth was different from stream to stream (ANCOVA; F = 972.8; P < 0.001), there was no difference in growth between control and treatment reaches (ANCOVA; F = 0.13; P = 0.72). Mean length at age-2 and age-3 (and range) at control reaches averaged 190 mm (147–263 mm) and 208 mm (170–253 mm), respectively, compared to 187 mm (150–245 mm) and 214 mm (167–264 mm) at treatment reaches (Table 2). Mean length at age did not differ significantly between control and treatment reaches (paired t-test; t = 2.14; df = 14; P = 0.37). Relative weight also was not statistically different between control (mean W_r = 86.3; range 79.4-96.7) and treatment (mean W_r = 85.8; range 78.4-98.0) reaches (paired t-test; t = 2.20; df = 11; P = 0.45; Figure 7).

DISCUSSION

Our study provides tangible evidence that the stocking of catchable rainbow trout in southern Idaho streams had no impact on the abundance, recruitment, survival, or growth of existing wild trout populations. Indeed, despite artificially increasing the abundance of rainbow trout in the treatment reaches by an average of 78% (range 13-444%) via stocking, wild rainbow trout showed no ill effects at the population level. The lack of effect we observed on wild trout after stocking is most likely due to the poor competitive abilities of hatchery fish, and numerous studies support this conclusion. R. B. Miller was a pioneer in this work, with a series of investigations on wild and hatchery trout interactions (see Miller 1951, 1953, 1958). He reported that hatchery fish moved downstream after stocking but did not survive long, especially overwinter. Hatchery catchables reared in streams fared better than pond reared fish but not as well as wild fish transplanted from another location, and even the newly transplanted wild fish were outcompeted by resident trout. Lactic acid levels were higher in hatchery fish than wild fish and he concluded that hatchery fish died of exhaustion, probably from harassment but also naivety in holding favorable stream feeding positions.

Since Miller's pioneering work, more recent studies have shown similar differences between hatchery and wild trout, including differences in aggressive behavior, foraging behavior, movement, holding position, growth, and survival (e.g., Needham and Slater 1945; Moyle 1969; Mesa 1991; Petrosky and Bjornn 1988; Peery and Bjornn 1996; Berejikian et al. 1999; see review in Weber and Fausch 2003). These and other studies have generally found that hatchery fish are more aggressive, use less energetically profitable holding and feeding positions, consume less food, are less wary of predators, and do not persist long under most stream settings. Vehanen et al. (2009) went so far as to suggest that, rather than hatchery fish distressing wild fish, the presence of wild trout may actually benefit hatchery fish via learned feeding and holding behavior, and they and Huusko and Vehanen (2010) provide some empirical evidence to support this.

Rarely has any study shown negative impacts to wild trout at the population level due to catchable stocking. One of the most cited examples is Vincent (1987), who reported that numbers and biomass of large wild trout (250-425 mm in length) increased substantially in years with no stocking in the Madison River and O'Dell Creek in western Montana, but smaller fish were not unaffected by stocking. However, a complicating factor in this study was that uncontrolled factors might have confounded the results. For example, winter flows were higher in the Madison River in the non-stocking years, potentially increasing wild trout abundance in the non-stocking years by increasing overwinter survival. In addition, exploitation was 25% greater during stocking years (presumably from attracting anglers by stocking fish), especially for larger-sized fish (Vincent 1980), suggesting increased harvest may also have explained some of the reduced wild trout numbers during stocked years. Other studies have also reported higher fishing mortality of wild trout populations associated with hatchery stocking (Butler and Borgeson 1965; Carline et al. 1991; Baer et al. 2007), and we believe that increased angler effort is the most common cause of decline in wild trout abundance associated with hatchery stocking, when such a decline occurs. This may explain why we saw no change in any of the vital statistics of wild trout in our study, since we did not advertise our stocking to anglers and thus they were unaware of the increased abundance of fish. Moreover, in one-third of our reaches harvest was impractical because nearly all fish (wild and hatchery) were of sublegal size, and in one instance because access for anglers was virtually nonexistent due to private land and a rugged, roadless canyon.

In our study, estimates of apparent survival (ϕ) were consistently much lower (mean 0.26 compared to 0.55) than total annual survival (*S*). This was expected in part because a lost PIT tag was equated as a death in the CJS model, and annual tag loss in the first year averaged 19% (K. Meyer, unpublished data). In addition, for estimates of *S* (based on population estimates), emigrants may have been balanced out immigrants, whereas for estimate of ϕ (based on PIT tag recaptures), emigrants could not balance out immigrants since emigrants were not PIT tagged. Thus, any remaining difference between *S* and ϕ not attributable to PIT tag loss may represent the level of immigration and emigration occurring in stream reaches of southern Idaho. Regardless, the lack of impact on *S* and ϕ for wild trout suggests that not only was survival of wild trout unaffected by catchables, but stocking also did not cause any additional emigration when hatchery trout were stocked. Although displacement of wild fish by hatchery fish has been demonstrated in both laboratory and stream settings (see review in Weber and Fausch 2003), including at the reach scale (Symons 1969), our study reaches in general were nowhere near carrying capacity for wild trout (mean PHS = 20), so it is not surprising that the addition of hatchery fish caused no additional emigration of wild trout over that which is normal for stream-dwelling salmonids.

We could not measure recruitment to age-0 because sampling occurred too early in the year to effectively capture these small fish. Instead, we inferred that any effect of stocking on wild rainbow trout spawning success or age-0 survival would have translated to lower abundance of wild age-1 fish the following year. We could have missed an effect if recruitment was reduced by stocking but this effect was obscured before we sampled age-1 fish the following year; density independent survival constraints for age-0 fish during winter is one possible example. However, we would argue that if such a 'recruitment' effect were masked in all of our treatment reaches by the time fish reached age-1, then the effect to the population, though real, would still likely be meaningless in most instances.

Certainly, there are instances where stocking catchable trout can be detrimental to wild trout populations. First, and perhaps most obvious, is potential genetic impacts, where stocking hatchery fish can result in widespread hybridization of native trout populations (see review in Allendorf 1991). Although hybridization with hatchery rainbow trout has impacted several species of native salmonids in Idaho (Meyer et al. 2006; Kozfkay 2011), IDFG has since 2001 only stocked catchables that have been treated with heat or pressure to induce sterility (Kozfkay et al. 2006) in an attempt to halt the further spread of hybridization.

Second, although we saw no negative effect of stocking catchables on the vital statistics of wild trout populations, had we stocked fish at higher densities, it is more likely that the catchables would have affected wild trout populations. We stocked streams at the upper end of densities used by IDFG hatchery staff, at an annual stocking density of $4.2/100m^2$. We also stocked additional fish at the upper and lower reach boundaries, some of which likely drifted almost immediately into the treatment reaches (especially at the upstream boundary; High and Meyer 2010). Counting only the fish stocked in the middle of the treatment reaches, our stocking density was at the lower end of other comparable studies investigating wild vs. hatchery competition in natural settings, such as Carline et al. (1991; $8.1/100\ m^2$) and Vincent (1987; $16.3/100\ m^2$). Petrosky and Bjornn (1988) saw little effect of stocking catchables on wild rainbow trout until their highest stocking density, which on average was almost two orders of magnitude higher ($253/100m^2$) than our stocking rate. Although the above-mentioned negative effect of stocking catchables on wild trout shown by Vincent (1987) in Montana may instead have been due largely to effects of stream flow and angling pressure, stocking rates were also four times higher than in our study. Including all catchables that we annually stocked, mean PHS in the treatment reaches was 48% (range 26-94%), which corresponds to a probability of observing a density dependent response (in either growth, survival, or emigration) of 0.73 (range 0.48–0.90) according to the logistic model of Grant and Kramer (1990). Although inclusion of all catchables certainly over-represents true PHS in treatment reaches because most catchables likely did not survive from one stocking event in one month to the next (High and Meyer 2010), densities of fish were certainly high enough to expect some density dependent response to the stocking if competition for territorial space was occurring. Evidently our study reaches were not limited by rearing or holding habitat, but more likely by food, overwinter survival, spawning substrate, or some combination of these and other factors. Considering that hatchery trout are not efficient at foraging (Petrosky and Bjornn 1988; Mesa 1991) and would generally not be expected to survive long enough to compete for holding habitat in the coming winter or spawning habitat in the following spring (Petrosky and Bjornn 1988; Dillon et al. 2000), it is not surprising that no stocking effect was detected in the wild trout populations.

Besides stocking at higher densities, we could also have stocked for a longer period of time, and there is evidence that prolonged stocking may be more likely to impact wild fish due to cumulative effects. For example, Pearsons and Temple (2007) found little change in rainbow trout population abundance, size, or distribution after five years of supplemental stocking of juvenile Chinook salmon *O. tshawytscha*, but after nine years, statistically significant decreases in abundance and biomass were more common (Pearsons and Temple 2010). However, as Pearsons and Temple (2010) noted, it seems likely that changes in abundance and growth would be more likely among small fish, which should have become manifest within a year or two in measurements of recruitment, and we detected no such impact. Moreover, Pearsons and Temple (2010) suggested that much of the increased detectability of changes in fish populations with additional study years in their study was probably related to increased statistical power. We believe our study design had ample statistical power, with 12 paired control vs. treatment comparisons over five years of monitoring, compared to the 'increased statistical power' of

Pearsons and Temple (2010) who had only two treatments and one control that were monitored for nine years.

In summary, despite the long-held notion that stocking catchable-sized hatchery trout negatively impacts wild trout via competition (e.g., Bachman 1984; Vincent 1987), relatively few studies have been published that have investigated impacts at the population scale. Our results suggest that stocking hatchery catchables at densities at the upper end of existing IDFG stocking protocols had no measurable effect on wild trout abundance, survival, growth, or recruitment in southern Idaho streams. Certainly resource use by wild and hatchery trout in stream environments do overlap, and our results do not lead us to advocate stocking hatchery fish haphazardly in streams across the landscape. However, where genetic concerns for native fish are nonexistent, we believe that stocking catchables at normal stocking densities will have negligible effects on existing populations of wild stream-dwelling trout.

RECOMMENDATIONS

1. Continue to stock catchables in select Idaho streams at current densities, frequencies, and locations with the realization that the stocking program has little to no impact on wild trout population metrics.

ACKNOWLEDGEMENTS

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TABLES

Table 1. Study reach locations, abiotic descriptions, and treatment stocking rates (per month, stocked in the middle of the treatment reaches) for study streams in southern Idaho.

Stream	Reach	UTMs Easting	UTMs Northing	UTMs Zone	Reach length (m)	Mean width (m)	Elevation (m)	Drainage area (km^2)	Conductivity (µS/cm)	Mean summer water temperature (°C)	Stream order (1:100K)	Gradient (%)	Percent substrate composition Fines	Percent substrate composition Gravel	Percent substrate composition Cobble/ boulder	Catchables/ plant
General regulation streams																
Fourth Fork Rock Creek	Treatment	726775	4681619	11	811.6	3.6	1577	25	65	13.6	3rd	4.8	4	34	53	51
	Control	725117	4678773	11	443.0	3.0	1800	14	58	12.9	2nd	3.5	5	39	44	-
East Fork Weiser River	Control	550180	4961897	11	489.2	5.5	1260	70	86	13.1	2nd	3.3	6	20	53	-
	Treatment	553858	4962111	11	495.4	6.5	1481	50	62	9.8	2nd	5.9	5	23	63	39
Little Weiser River	Treatment	555728	4927393	11	641.4	10.6	1163	116	81	15.7	3rd	1.8	5	27	60	70
	Control	560119	4929915	11	581.0	8.7	1306	85	70	12.4	3rd	2.2	2	9	73	-
Wild trout regulation streams																
Second Fork Squaw Creek	Control	555742	4913591	11	872.8	8.7	1149	96	39	15.8	3rd	1.4	4	37	49	-
	Treatment	556947	4920060	11	684.4	9.9	1254	66	38	14.9	3rd	1.8	2	29	57	88
Clear Creek	Treatment	612447	4884548	11	545.0	11.2	1267	136	44	12.2	3rd	3.1	6	18	63	124
	Control	615483	4892526	11	370.6	10.5	1562	93	33	10.4	3rd	2.5	4	12	62	-
Sawmill Creek	Treatment	313446	4909346	12	707.2	8.2	2036	192	69	12.4	4th	1.3	7	35	48	57
	Control	310744	4914244	12	520.8	6.4	2104	116	53	11.1	4th	1.4	8	35	46	-
Willow Creek	Treatment	691836	4817640	11	461.4	4.7	1717	51	244	12	3rd	2.5	28	41	15	23
	Control	690451	4819520	11	349.2	3.6	1790	31	255	11.1	3rd	2.9	7	42	39	-
Medicine Lodge Creek	Treatment	380935	4904871	12	567.4	5.2	1737	409	392	13	4th	1.6	26	30	7	44
	Control	376226	4907857	12	527.0	5.9	1806	394	380	12.5	4th	1.7	18	29	19	-
Catch-and-release regulation streams																
South Fork Boise River	Control	660998	4827702	11	1438.8	23.3	1343	917	100	13.4	5th	1.6	3	25	63	-
	Treatment	664707	4828430	11	1083.8	23.2	1604	899	106	15.3	5th	1.7	7	35	43	227
Middle Fork Boise River	Treatment	613554	4843073	11	827.2	26.9	1094	984	68	18.1	5th	0.5	2	27	47	303
	Control	618340	4848149	11	915.4	30.8	1171	905	75	15.6	5th	0.5	8	19	57	-
Middle Fork Boise River	Treatment	626494	4849524	11	964.2	23.0	1269	757	64	15.1	5th	0.4	3	16	59	270
	Control	631618	4852102	11	925.2	24.7	1305	647	60	14.5	5th	1	4	16	61	-
Badger Creek	Control	477674	4862215	12	216.8	11.9	1646	150	248		3rd	4.4	3	19	46	-
	Treatment	480530	4863772	12	470.0	16.0	1698	145	224	10.8	3rd	0.9	6	39	43	181

Table 2. Mean length at age for wild rainbow trout collected in control and treatment reaches in 2010.

	Mean length at age 2				Mean length at age 3			
	Control		Treatment		Control		Treatment	
site	Est.	n	Est.	n	Est.	n	Est.	n
Fourth Fork Rock Creek	147.4	10	156.7	3	-		184.6	5
East Fork Weiser River	159.4	10	149.7	17	180.1	8	174.6	10
Little Weiser River	176.8	5	-		202.8	4	-	
Second Fork Squaw Creek	-		157.4	5	-		180.3	6
Clear Creek	158.1	7	153.9	7	169.7	3	166.8	4
Sawmill Creek	186.8	11	191.0	7	196.9	8	223.5	6
Willow Creek	-		181.2	5	-		-	
Medicine Lodge Creek	262.8	12	244.9	21	-		263.7	12
South Fork Boise River	201.2	21	205.0	4	252.8	4	258.3	4
Middle Fork Boise River	196.8	11	208.7	7	-		-	
Middle Fork Boise River	193.2	22	192.7	3	204.3	3	227.2	6
Badger Creek	221.6	24	218.1	25	248.1	7	249.5	6

FIGURES

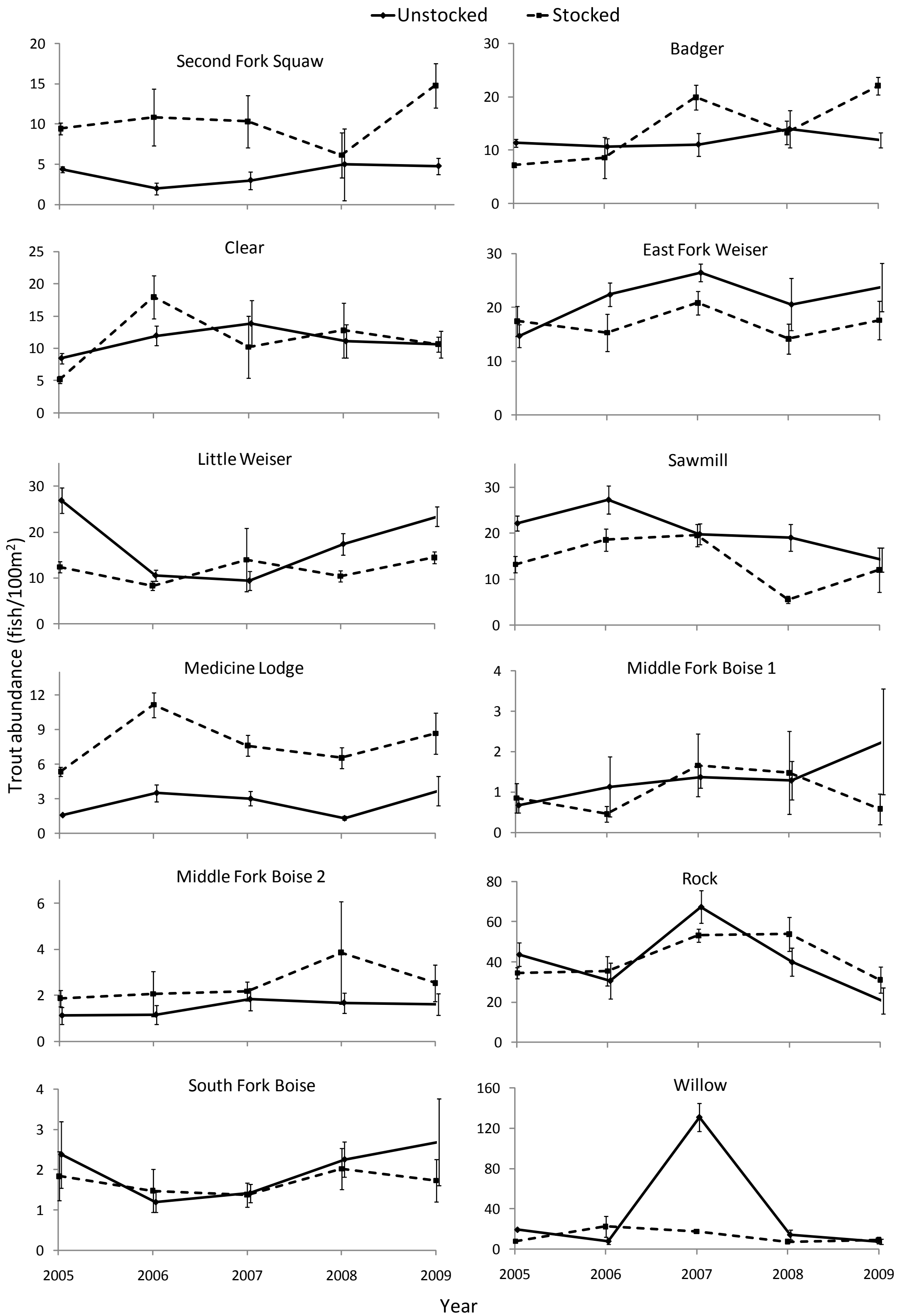

Figure 1. Estimated abundance of wild rainbow trout (fish/100m^2) and 95% confidence intervals of all trout in unstocked and stocked reaches of streams from 2005 to 2009 in southern Idaho. Catchable stocking occurred from 2006 to 2008.

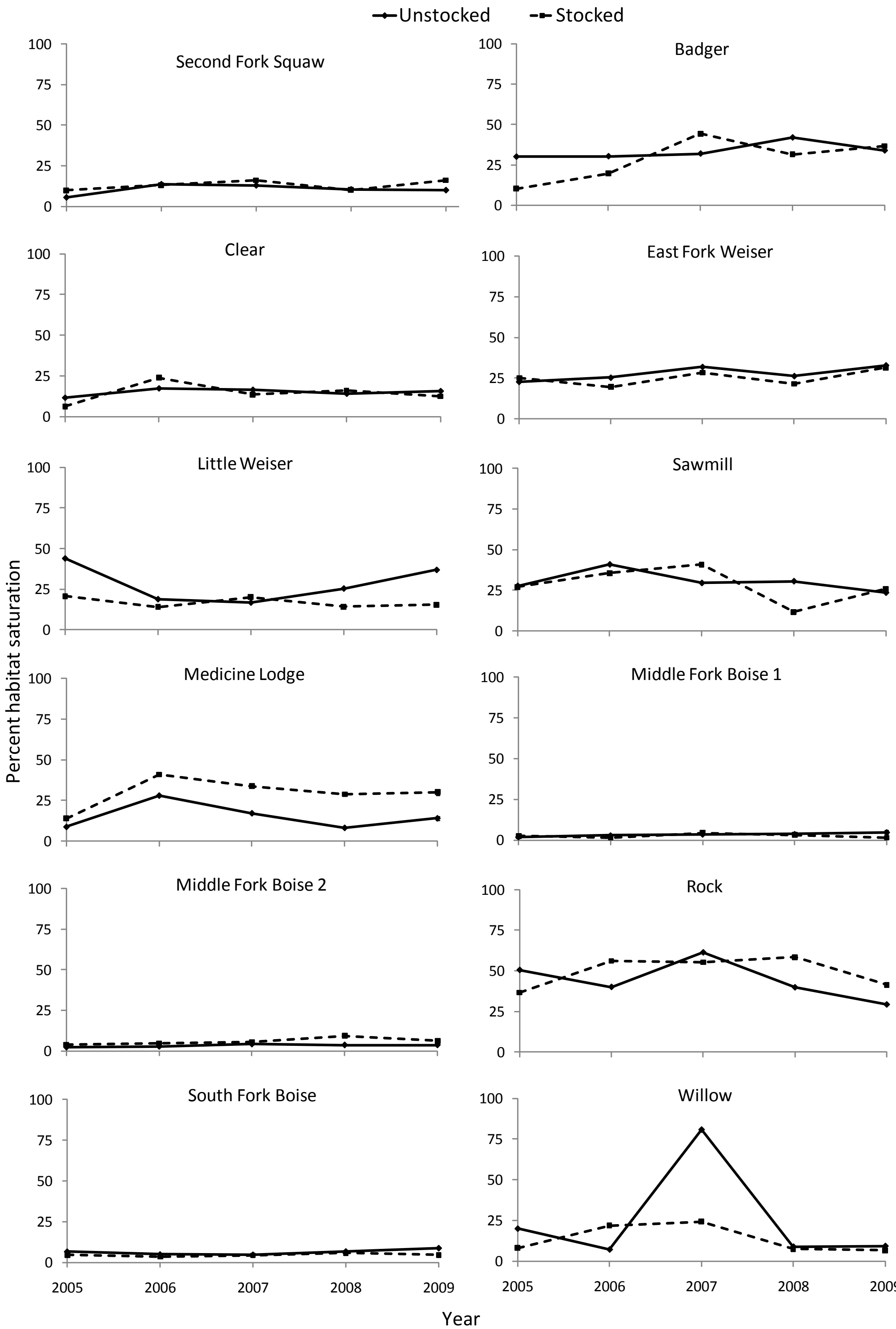

Figure 2. Estimated percent habitat saturation of wild rainbow trout in unstocked and stocked reaches of streams from 2005 to 2009 in southern Idaho. Catchable stocking occurred from 2006 to 2008.

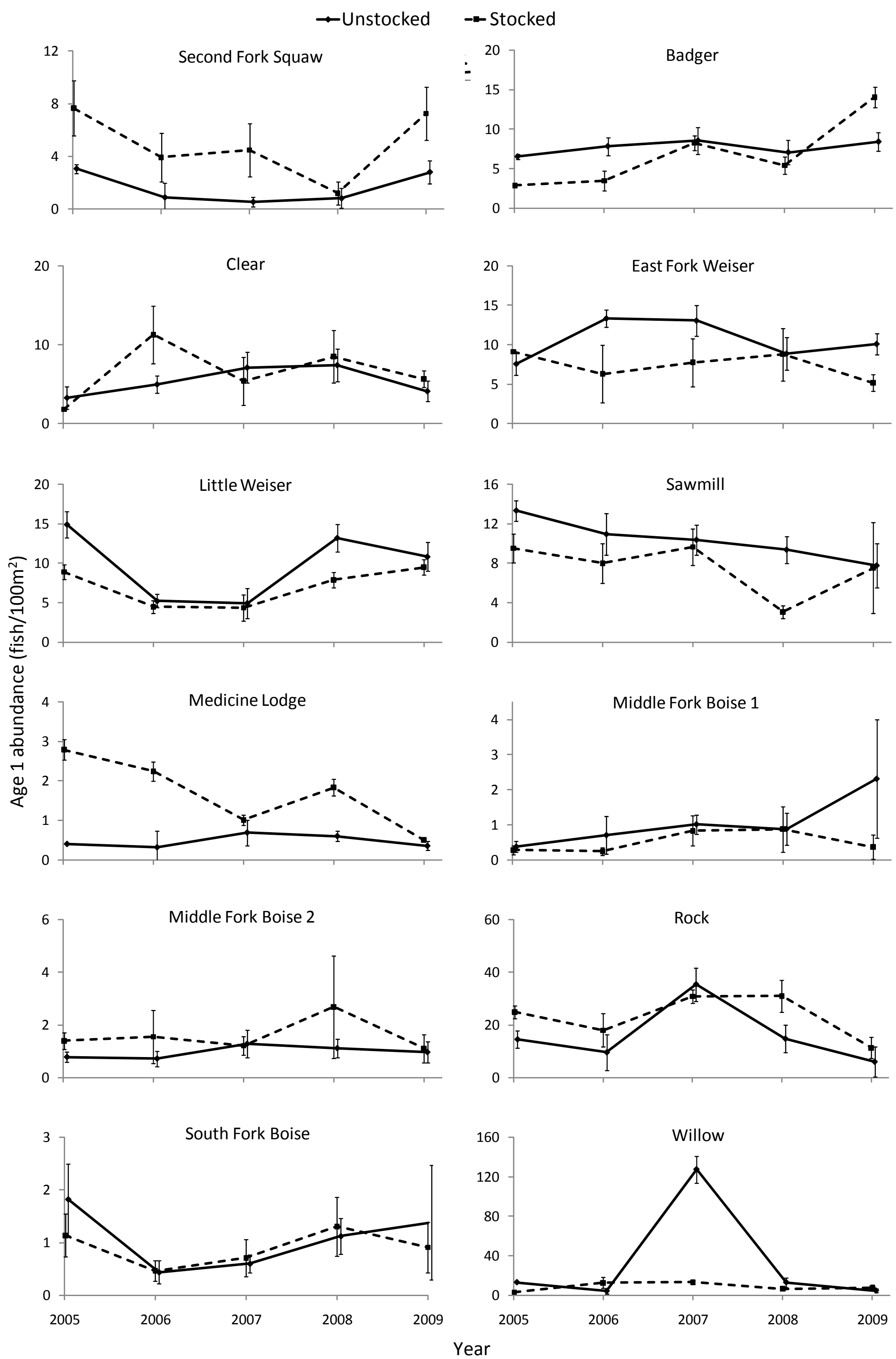

Figure 3. Estimated recruitment of wild rainbow trout (age-1 fish/100m^2) and 95% confidence intervals in unstocked and stocked reaches of streams from 2005 to 2009 in southern Idaho. Catchable stocking occurred from 2006 to 2008.

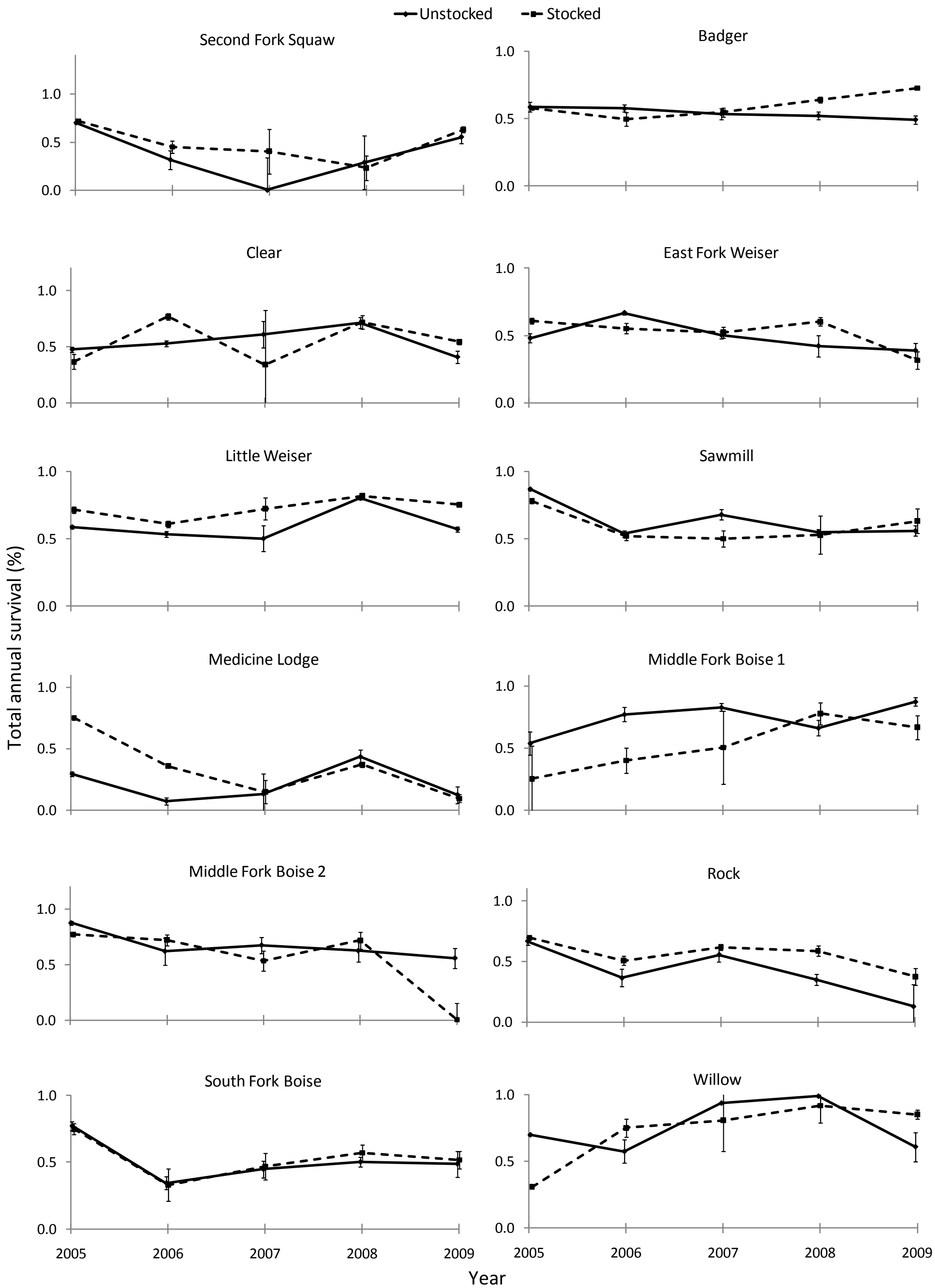

Figure 4. Total annual survival (S) and 95% confidence intervals for age-1 and older wild rainbow trout in unstocked and stocked reaches of streams from 2005 to 2009 in southern Idaho. Catchable stocking occurred from 2006 to 2008.

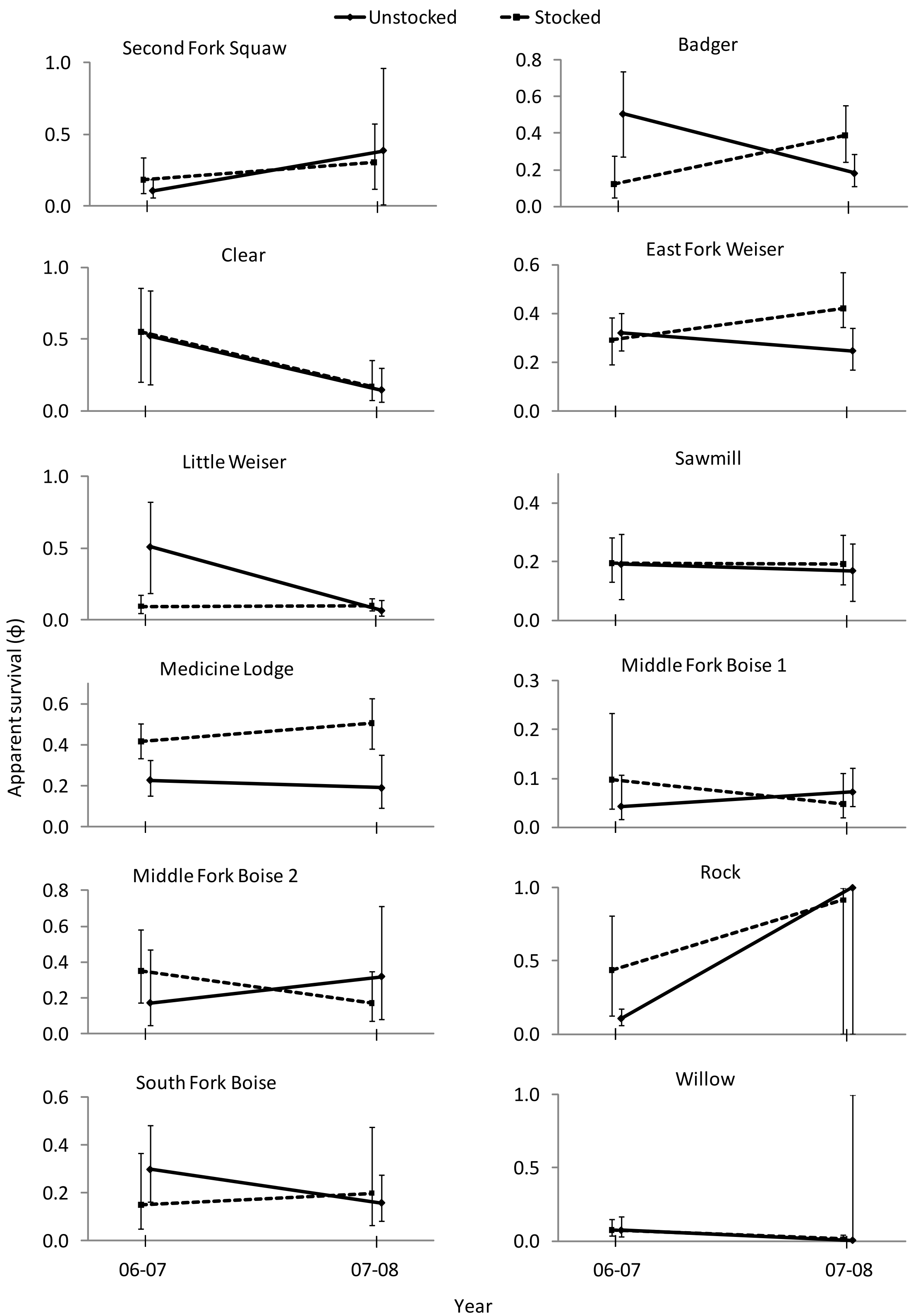

Figure 5. Apparent survival (ϕ) and 95% confidence intervals from Cormack Jolly Seber modeling based on PIT-tagged recaptures of rainbow trout in unstocked and stocked reaches of streams in southern Idaho.

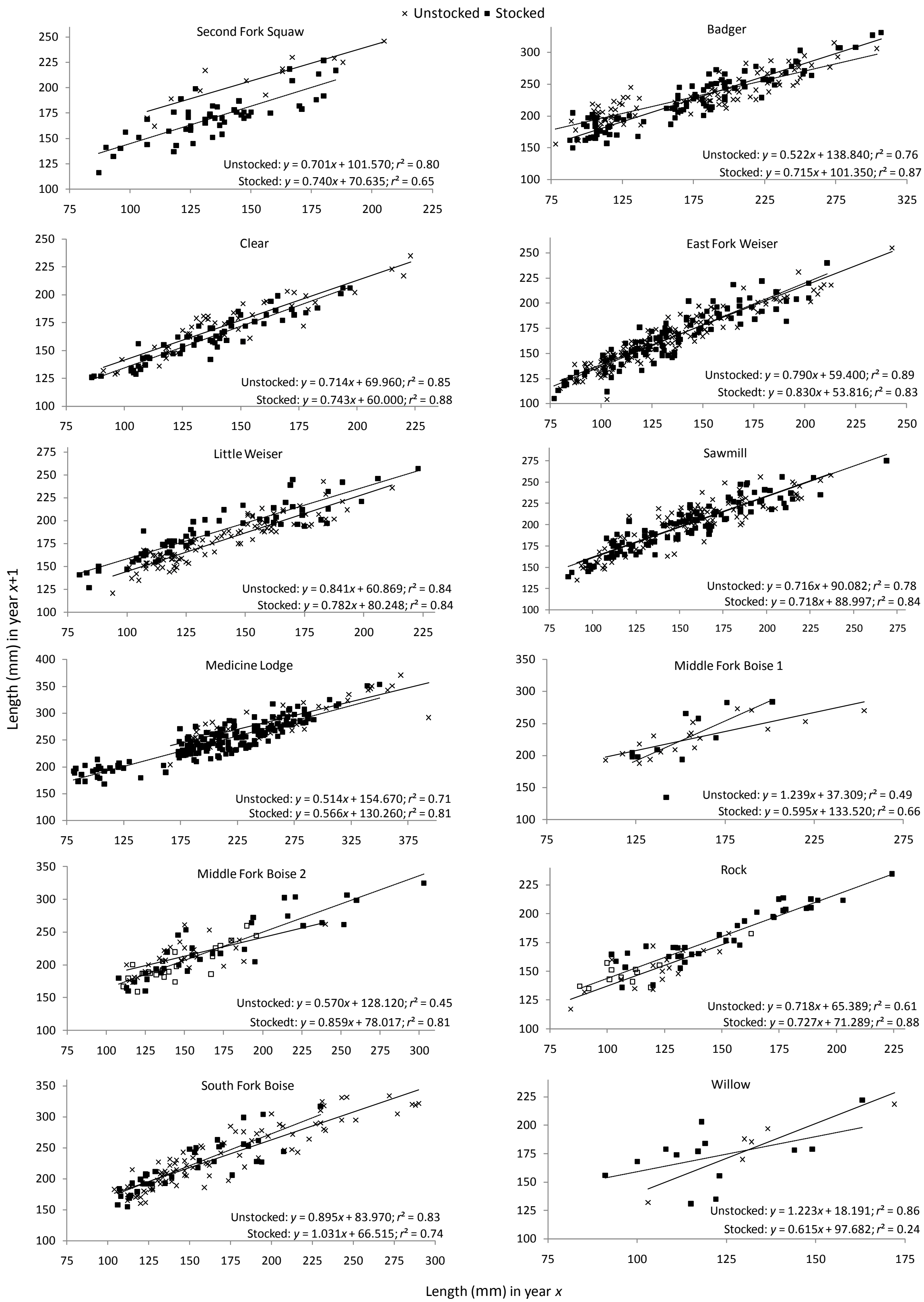

Figure 6. Growth (mm) of wild rainbow trout from year *x* to year *x*+1 for PIT-tagged fish in unstocked and stocked reaches of streams in southern Idaho.

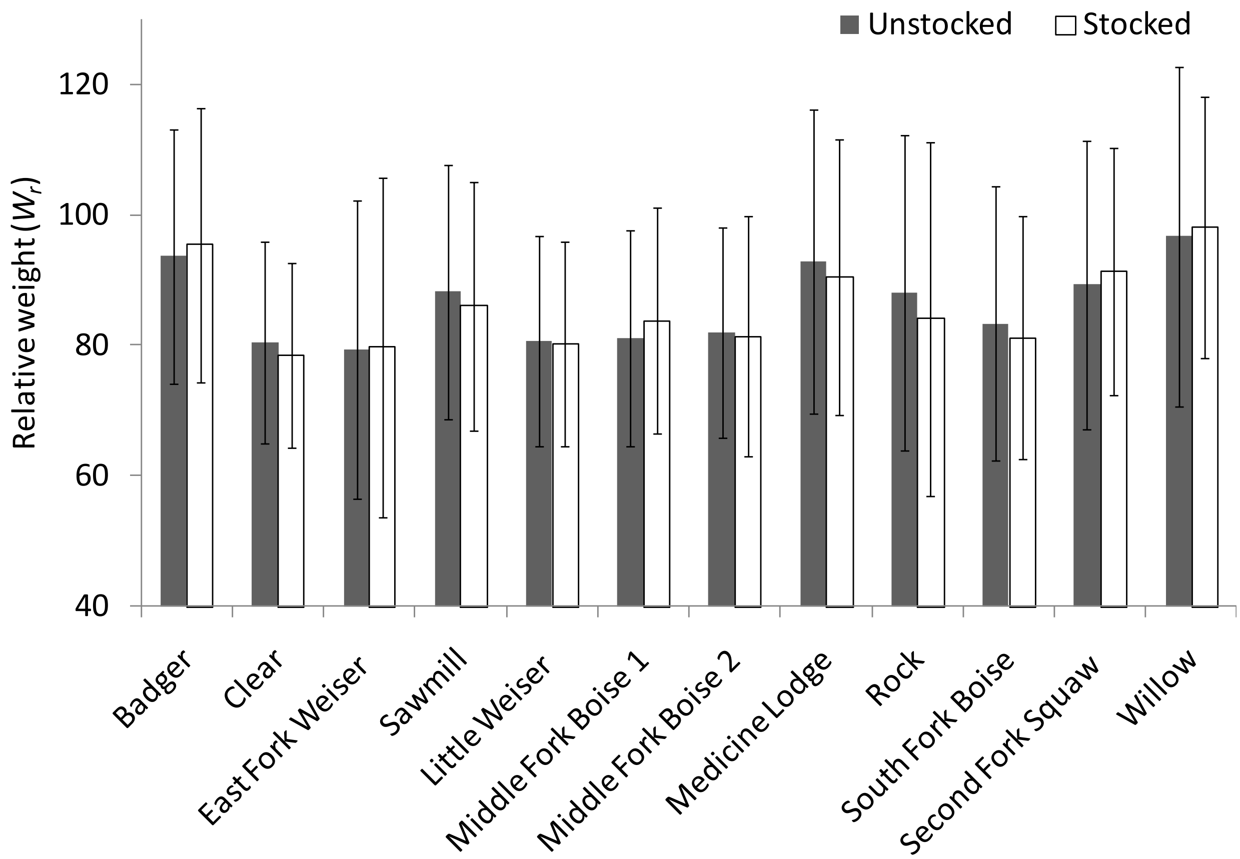

Figure 7. Mean relative weight for wild rainbow trout in unstocked and stocked reaches of streams in southern Idaho.

ANNUAL PERFORMANCE REPORT
SUBPROJECT #2: HOOKING MORTALITY FROM BAITED CIRCLE HOOKS

State of: Idaho

Grant No.: F-73-R-32, Fishery Research

Project No.: 2

Title: Wild Trout Studies

Subproject #2: Hooking mortality comparisons using circle hooks

Contract Period: July 1, 2009 to June 30, 2010

ABSTRACT

We compared hooking and landing success along with hooking mortality for trout caught with barbed baited circle hooks to barbed single-hook dry flies, barbed treble hook spinners, and barbed baited J hooks. In one experiment, 300 wild trout were caught in equal numbers in an unexploited 1 km reach of Badger Creek in eastern Idaho, marked using passive integrated transponder tags, and released for 69 d. Deep hooking rate was higher for baited J hooks (21%) than for spinners (11%), baited circle hooks (4%), and dry flies (1%). Mark-recapture population estimates indicated that mortality rates for trout captured with baited J hooks (25%) and treble hook spinners (29%) were significantly higher than for trout captured with baited circle hooks (7%) and dry flies (4%). For J-hooked fish, mortality was 3.9 times higher for those that were deep-hooked. In a second experiment (n = 604 wild trout), hooking success (i.e., number of successful hook-ups ÷ number of strikes) was about one-third lower for circle hooks (fished passively, 37%; fished actively, 40%) than all other hook types except passively-fished J hooks. Once hooked, landing success (i.e., number of fish to hand ÷ number of hook-ups) was high and relatively constant for all hook types (range 68-87%), but was lowest for both passively-fished baited hooks. Deep hooking was two and six times higher for J hooks than circle hooks for fish caught actively and passively, respectively. For circle hooks, deep hooking was over two times greater when the angler did not actively set the hook compared to actively setting the hook, which conflicts with manufacturer's recommendations. Our combined results suggest that when bait fishing for trout in streams, circle hook use may reduce deep hooking and hooking mortality regardless of whether they are fished in the recommended manner.

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INTRODUCTION

Increasing angler effort on popular wild trout fisheries has often led to implementation of so-called special regulations such as creel limits, slot limits, size limits, and gear restrictions, which are designed to reduce fishing mortality rates. Such management strategies assume negligible post-release or hooking mortality (Wydoski 1977). In trout species, nearly all past studies comparing bait hooking mortality versus artificial flies and lures have concluded that the use of bait results in mortality rates 3-6 times higher than other gear types (e.g., Shetter and Allison 1955; Hunsaker et al. 1970; Mongillo 1984).

Although several studies have demonstrated that bait fishing can be compatible with special regulations for salmonids (e.g., Turner 1986; Thurow 1990; Orciari and Leonard 1990; Carline et al. 1991), it is generally assumed that bait fishing conflicts with such regulations. This is because the elevated hooking mortality rates noted above for bait are presumed to prevent sufficient increases in fish size or abundance that could result from creel, slot, or size limits. Based on this assumption, fishery managers typically restrict the use of bait in an effort to obtain maximum trout density. In doing so, they must weigh the social risk of alienating bait fishermen against the potential for higher hooking mortality rates for fish caught and released with bait (Thurow and Schill 1994).

Hooking mortality using conventional bait fishing gear is significantly higher than for other gear types because mortality of caught-and-released fish is strongly dependent on the anatomical site of hooking and resultant injury to vital organs due to deep hooking (Mason and Hunt 1967; Schill 1996). While artificial flies and lures are not immune to hooking fish in critical areas such as the esophagus, stomach, or gills, they generally penetrate these critical areas less than 10% of the time, compared to a much higher rate (up to 50%) when bait is used with conventional J hooks (Mongillo 1984).

Circle hooks were first suggested for evaluation in salmonid special regulation fisheries by Schill (1996), and subsequently have gained notoriety as a more benign bait hook for reducing hooking mortality in a number of species relative to conventional J hooks (Cooke and Suski 2004). On a circle hook, the point of the hook is oriented perpendicular to the shank, rather than parallel as on a J hook. Because of this, once it is swallowed, the baited hook ostensibly can pull free from a fish's esophagus or stomach, until the hook's path of travel is changed at the edge of the mouth, allowing the hook to rotate and the point to become embedded. Manufacturers recommend that for circle hooks to perform properly in recreational bait fishing, anglers should not set the hook, but rather should lift lightly on the rod and slowly retrieve the fish (e.g., Montrey 1999; also see ASMFC 2003). Cooke and Suski (2004) assume this recommendation is sound but point out that no studies have tested whether angling technique (i.e., actively setting the hook or not) affects circle hook performance. To gain acceptance among bait anglers and fish managers, circle hooks performance would ideally be similar to that of conventional J hooks in terms of hooking and landing success. Past studies have produced equivocal results, but a literature summary by Cooke and Suski (2004) suggests that capture efficiency is generally lower for circle hooks than J hooks, although none of the studies in their review included freshwater salmonids in lotic systems.

Expected reductions in hooking mortality has made circle hooks widely accepted for use in commercial and recreational marine fisheries (Kaimmer and Trumble 1997; Trumble et al. 2000), but use in freshwater sport fisheries is also growing in popularity (Meka 2004; Cooke and Suski 2004). Because hooking mortality rates for circle hooks relative to J hooks have been inconsistent across species and individual settings, it has been suggested that, unless compelling comparative species-specific data exist, management agencies should restrict routine encouragement of circle hooks (Cooke and Suski 2004).

In salmonid fisheries, studies have been limited to hatchery settings, and methods and results have also been inconsistent. Parmenter (2000) found that hooking mortality was twice as high for conventional J hooks (19.4%) compared to circle hooks (10%) for rainbow trout *Oncorhynchus mykiss* caught in hatchery raceways, but, unexpectedly, deep hooking rates were similar. The author reported volunteer anglers fishing circle hooks incorrectly (i.e., actively setting the hook), which, he believed, may have confounded the results. In aquaculture pens, deep hooking of hatchery rainbow trout was 2.4 times higher for J hooks than circle hooks (Jenkins 2003); however, mortality rates (9%) for circle hooks were higher than for J hooks (0%), presumably because all circle hooks were removed from the fish regardless of hooking location, whereas for J hooks, lines were cut for deep-hooked fish. The applicability of these studies to wild trout fisheries remains questionable since wild trout may experience higher hooking mortality rates than their hatchery counterparts (Warner 1979; Mongillo 1984), presumably due to stressors associated with confining wild test fish (Schill 1996).

The purpose of this study was to assess hooking mortality rates for baited circle hooks and more conventional hook types in streams supporting wild trout fisheries. Specifically, hooking mortality was quantified for wild cutthroat trout *O. clarkii* and rainbow trout as a function of hook type (circle hook, J hook, dry fly, and treble hook spinner) and hooking location in a natural, unconfined stream setting. We also assessed whether hooking and landing success were lower when bait fishing with circle hooks, and whether deep hooking rates for circle hooks varied according to whether or not the hook was actively set by the angler.

OBJECTIVE

1. Quantify hooking mortality of wild trout caused by baited circle hooks relative to baited J-hooks, dry flies, and treble hook spinners.

METHODS

Badger Creek is a productive tributary of the Teton River in the upper Snake River basin in eastern Idaho (43º55’35” N, 111º14’53” E). The study site was in the lower section of the stream (mean width 13.6 m, mean depth 0.20 m) in a deep, narrow canyon. Water temperatures during the months of July and August in 2006 averaged 10.9°C and fluctuated between 8.8 and 14.5°C. Lower Badger Creek is surrounded by private land and access is quite limited, thus fishing pressure and harvest is extremely low despite the fact that general fishing regulations (excluding the harvest of cutthroat trout) are in force (6 fish limit, no size restrictions). A 1.0 km section of Badger Creek was isolated with hardware cloth wire mesh (1.3 cm) weirs to prevent fish from entering or leaving the study area during the holding period. The weirs were checked and cleaned frequently (at least every two days) to ensure proper function. Fish composition in Badger Creek was comprised mainly of trout (approximately 97% rainbow trout, 3% cutthroat trout and hybrids), but mottled sculpin *Cottus bairdi* were also present.

Hooking Mortality

After the weirs were built, experienced anglers fished from July 5 to 8, 2007, using dry flies (size 4 to 14), treble hook spinners (Panther Martin® 3.5 g), in-line style circle hooks baited with nightcrawlers (Eagle Claw® size 8, model L2050-12), and conventional off-set J hooks baited with nightcrawlers (Renegade snelled size 8), all barbed. While fishing with bait, anglers limited effort to pools and slackwater to maximize potential for deep hooking. Circle hooks were fished according to manufacturer's recommendations (Montrey 1999; also see reviews in ASMFC 2003, Cooke and Suski 2004) that when a strike is detected, the angler slowly reels in to retrieve the fish without setting the hook. For J hooks, the hook was actively set once a strike was detected. Angling continued until 75 fish were caught with each gear type. Captured trout were anesthetized, categorized into 25 mm size groups, and marked with an adipose fin clip. Passive integrated transponder (PIT) tags were placed intraperitoneally using a rinsed, 12 gauge hypodermic needle; the insertion point was ventral and posterior to the pectoral fin, offset slightly to the right or left side depending on the individual tagger. The anatomical location of hooking was noted as well as other observations including relative amount of bleeding, whether the hook was removed or the line was cut, and the presence of disease or existing health problems. Anglers cut the line with all hook types when fish were hooked in the esophagus or deeper, leaving the hook in the test fish. Adipose fin clips were used to quantify rates of PIT tag loss, and to indicate the presence of a PIT tag in subsequent sampling. Upon recovery, fish were released where they were captured.

The study area was maintained for a 69 d holding period after the 4 d angling event. Due to debris building up against the hardware cloth, the lower weir partially failed three times for no more than one full day before repairs could be made. A small tear at the bottom of the upper screen was also repaired once. At the conclusion of the observation period, electrofishing passes were made approximately 100 m above and below the weirs to assess the level of fish escape that may have occurred during weir failures; no study fish were captured below the lower weir, but three were captured above the upper weir. An additional 500 m was surveyed above the upper weir, but no additional test fish were captured. Escaped fish resulted in an unknown (but probably slight) overestimate of hooking mortality, but it is reasonable to assume there was no difference between hook types in fish escape, thus mortality comparisons between hook types were assumed to be unbiased.

A mark-recapture electrofishing survey was conducted within the study reach at the end of the study using backpack electrofishing units. Captured fish were measured to the nearest mm. During the marking run, caudal fin clips were used to mark each fish, and adipose-clipped fish were scanned for PIT tags. We recaptured fish on the following day. Abundance estimates and 95% confidence intervals (CIs) were made for all trout in the study reach as well as the remaining abundance of test fish for each hook type using the modified Petersen method within the software package Fisheries Analysis Plus (Montana Fish, Wildlife, and Parks 2004). To control for size selectivity bias, estimates were separated into the smallest size-groups possible (usually 100 mm) which met the criteria that (1) the number of fish marked in the marking run multiplied by the catch in the recapture run was at least four times the estimated population size and (2) at least three recaptures occurred per size group; meeting these criteria creates modified Petersen estimates that are less than 2% biased (Robson and Regier 1964).

Some test fish shed PIT tags during the holding period and thus could not be traced back to hook type. We estimated how many fish shed PIT tags by calculating a modified Petersen population estimate based on the number of fish in the mark and recapture runs with adipose clips but without PIT tags. We assumed no differences in PIT tag shedding rates between hook types, and distributed the estimate of test fish that lost PIT tags and the corresponding variance back into the four hook types. We weighted this adjustment based on the proportion of the total sample size estimated to remain after the holding period for each hook type. We calculated mortality rates over the test period for each hook type as follows:

$$Mn = (An – Bn)/An$$

where Mn is the mortality rate for fish of hook type n, An is the number of fish of hook type n initially tagged while angling, and Bn is the estimate of the abundance of fish of hook type n at the end of the study. Statistically significant differences in mortality rates were noted by non-overlapping 95% CIs around the estimates. Confidence intervals were determined by using the lower and upper bound values of the Bn estimate in the above formula for mortality rates for each hook type, respectively.

We tested whether hook type influenced the size of fish captured by comparing the mean total length of fish caught in each group; we used non-overlapping 95% CIs around the means to indicate a statistically significant difference in fish size captured between hook types.

We tested whether the anatomical site of hooking location affected survival by comparing the proportion of deep-hooked test fish caught during the angling phase of the study to that observed at the end of the holding period. Deep hooking was defined as having the hook embedded in the gill arches, esophagus, or stomach during capture.

Hooking and Landing Success Evaluation

Rates of hooking success and landing success were compared with an additional experiment in July 2009 (within the same study reach) by counting the number of strikes and the number of hook-ups it took to land 100 fish for each hook type. Hooking success rate was calculated as the number of successful hook-ups (i.e., the fish was “on” the line for at least a few seconds) divided by the number of strikes. Landing success rate was calculated as the number of fish successfully landed (i.e., reducing the fish to hand) divided by the number of successful hook-ups. In order to estimate hooking success, landing success, and deep hooking rates for baited hooks fished passively or actively, we fished circle and J hooks both ways. We calculated 95% CIs around these percentages following (Fleiss 1981), and used non-overlapping CIs to assess statistical differences between comparisons.

RESULTS

Hooking Mortality

During a 4 d period, anglers caught and PIT tagged 300 fish using four different hook types. The majority (72%) of the trout caught were hooked in the upper or lower jaw (Table 3), followed by the roof and floor of the mouth (13%). Eight percent were deep hooked, most (67%) of which occurred with J hooks. The deep hooking rate was higher for baited J hooks (21%) than for spinners (5%), baited circle hooks (4%), and dry flies (1%). Only one immediate mortality was observed, occurring after the release of a fish caught in the esophagus on a J hook. The mean length of test fish caught was 252 mm (range 126 to 370 mm), and overlapping 95% confidence intervals around the mean total lengths of fish caught with each hook type indicated fish length was not influenced by hook type.

After a 69 d experimental period, a total of 1,738 trout were handled during the mark and recapture electrofishing survey, including 240 test fish. Electrofishing capture efficiency was estimated to be 69%. We estimated 2,255 (± 66) trout ≥100 mm were present within the 1 km study area. We captured 44 test fish that had lost their PIT tags, and estimated that a total of 47 lost their tag. After adjusting for PIT tag loss, population estimates at the end of the experiment (and 95% CIs) for each hook type ranged from 53 ± 4 for lures, 58 ± 4 for J hooks, 70 ± 4 for circle hooks, and 71 ± 5 for dry flies (Table 4). Combining the population estimates with the different gear sample sizes yielded mortality rates that were statistically greater for fish caught with spinners (29%) and J hooks (25%) than for fish caught with circle hooks (7%), and dry flies 4% (Table 4). For J hooked fish, mortality was 54% (95% CI = 39-69) for deep hooked fish compared to only 14% (8-21%) for those that were not deep hooked.

Hooking and Landing Success Evaluation

Hooking success was highest for spinners (64 ± 7%) and actively-fished J hooks (63 ± 5%), and lowest for circle hooks (37 ± 5% for passive fishing, 40 ± 7% for active fishing) and passively fished J hooks (38 ± 5%; Table 5). Once fish were hooked, landing success was high and relatively constant for all hook types (range 68-87%), although landing success was statistically higher for dry flies than for spinners and both passively-fished baited hooks (Table 5). Deep hooking was more common for J hooks than circle hooks, whether they were actively fished (six times as likely) or passively fished (twice as likely; Table 5). Deep hooking for circle hooks was 3% when actively fished and 10% when passively fished, but these percentages did not differ statistically.

DISCUSSION

With a 7% mortality rate over the 69 d study, passively-fished barbed circle hooks baited with nightcrawlers performed nearly as well as dry flies at limiting hooking-related mortality of caught-and-released wild trout in Badger Creek, and outperformed spinners and actively-fished baited J hooks. Low mortality rates for trout caught with circle hooks in our study corroborated results of previous studies using circle hooks on hatchery rainbow trout, which reported 9% mortality after 26 d in a net pen (Jenkins 2003) and 10% mortality after 28 d in a hatchery setting (Parmenter 2000).

We observed more deep hooking of wild trout while fishing with J hooks than circle hooks, regardless of whether the hooks were actively or passively fished, indicating that circle hooks reduced deep hooking no matter how they were fished. This contrasts with the findings of Parmenter (2000), who found a higher deep hooking rate using circle hooks (identical to our study) in rainbow trout caught in a hatchery (55%) and attributed this to his observation that some anglers actively set the hook (against instructions). The premise for using circle hooks when bait fishing is that if the angler allows the fish to ingest the bait and hook, and then sets the hook with vigor that is typical of the common bait fisher, the hook will either not capture the fish at all (because it will be pulled loose) or will be more likely to hook deeply than if gentle, steady pressure is applied (Cooke and Suski 2004). Our results suggest that for stream-dwelling trout, neither may be the case. We suspect that actively fishing the circle hook resulted in less deep hooking than passive fishing because by setting the hook when a strike occurs, the hook was less likely to have already been deeply ingested by the fish, and subsequently was less likely to lodge there. Surprisingly, we did not see a similar reduction in deep hooking by actively fishing J hooks, although others have (Schisler and Bergersen 1996). We suggest that more research is needed to replicate these findings and test comparative deep hooking rates under a variety of lentic and lotic conditions before strong conclusions regarding the use of circle hooks for freshwater salmonid fishing can be made.

The higher mortality rate we observed for passively-fished baited J hooks relative to other hook types was likely caused by the higher rate of deep hooking with J hooks, as evidenced by the fact that a significantly lower proportion of test fish deep-hooked with J hooks remained at the conclusion of the study compared to the beginning. The anatomical site of hooking is strongly related to hooking mortality because deep-hooked trout often die due to hooking damage to organs including the heart and liver (Mason and Hunt 1967; Schill 1996). In our study, deep hooking most commonly occurred when fishing with actively-fished baited J hooks, but was low relative to other studies. For example, Jenkins (2003) reported over 60% of the hatchery rainbow trout he caught were hooked in the esophagus using J hooks with Powerbait® while fishing net pens in a pond. The stream setting may have influenced our deep hooking rates, since pool and backwater habitat in our study reach was not extensive. Thus, stream flow within or adjacent to the pools may have affected our ability to allow trout to consistently swallow the bait, as observed by Jenkins (2003) for J hooks.

Our results suggest that bait fishers angling for stream-dwelling rainbow trout may experience a 1/3 reduction in hooking success and a slight change in landing success if they switch from actively fishing J hooks to using circle hooks, regardless of whether they fish actively or passively. In total, anglers switching to circle hooks may experience a 47% reduction in catch compared to fishing actively with a baited J hook. However, for those who already fish J hooks passively with bait, such as those using rod holders, our results suggest that hooking and landing success will not change from a switch to circle hooks. The fact that landing success was lower for both bait hook types when they were fished passively suggests that the barb may not have been adequately penetrated the fish for some hook-ups.

One common drawback in the literature to the use of circle hooks is the increased eye damage that has been noted in several studies for specific species (see review in Cook and Suski 2004). However, some of the literature cited in their review pertains to fish that were caught while vertical jigging. We found low rates of eye damage using circle hooks to fish for stream trout, most likely because anglers that drift bait in streams will generally not feel a bite until the bait (and hook) are already behind the eye.

Hooking and landing success rates did not differ for circle hooks when fished actively or passively. This was contrary to our expectations because manufacturers of circle hooks recommend passive fishing techniques to minimize deep hooking. Interestingly, deep hooking rates of passively fished circle hooks (10%) were more than three times that of actively fished circle hooks (3%), although this difference was not statistically significant at $\alpha = 0.05$ and $n = 100$. Further review of differences in hooking locations when circle hooks are fished passively or actively would help determine if circle hooks could be used as a regulation tool because not all anglers will fish according to manufacturers' recommendations. Passive fishing may allow the hook to get set in the deeper, critical hooking areas for a longer period of time, and our results suggest this may increase the likelihood of deep hooking for circle hooks. Zimmerman and Bochenek (2002) reported that circle hooks appeared to be more prone to deep hooking flounder when drift speed was lowest. To our knowledge, the present study is the first to compare circle hook performance with different types of hook set as recommended by Cooke and Suski (2004). Further research with similar designs would help determine whether passively fishing circle hooks increases deep hooking for stream trout or other species.

Although deep hooking was uncommon (5%), the high mortality rates for test fish caught using Panther Martin spinners (29%) was high and not significantly different from that for test fish caught with J hooks (25%). Most previous studies have indicated that lures do not cause high hooking mortality rates within resident trout populations (Wydoski 1977; Dubois and

Dubielzig 2004). The higher mortality rates we observed for lures may have been related to eye hooking, which may directly increase mortality rate (Siewert and Cave 1990) and was 2.6 times higher for spinners than any other hook treatment in our study. Alternatively, the small size of spinner used relative to the size of most study fish may have been important, as the size of fish relative to hook size is considered to be important in hooking related damage (Cooke and Suski 2004) and has long been suspected as the cause of elevated lure hooking mortality rates in salmonids (D. J. Schill, IDFG, personal communication). We noticed a few of the test fish landed with spinners were hooked in the jaw, but had sustained damage to the gill arches. In these situations, mortality may have been caused by initial deep hooking in the gill arches with a small lure that damaged that area prior to lodging in the mouth or jaw.

PIT tag loss likely had little effect on our findings because loss was minimal, we accounted for the loss in our mortality estimates, and the loss presumably did not differ between hook types. Similarly, our electrofishing above and below the weirs suggests that fish escape from the study reach was minimal (probably ~1%) and, with no expected difference between hook types, likely resulted in only a slight overestimation of actual mortality rates. Angling harvest and natural mortality may also have accounted for some of the mortality in our study and caused further overestimation of actual hooking mortality rates, although with such limited access and a reasonably short summer study period, this overestimation bias was probably minimal and presumably not different between our test groups. No anglers were ever observed during routine (almost daily) weir maintenance.

Fishery managers often must balance social preferences for fishing regulations with the biological constraints of individual fish populations. Special regulations are typically put in place to limit annual mortality rates of fish populations by reducing angling mortality. Unfortunately, special regulations restricting bait have a tendency to alienate those constituents, sometimes with legal consequences (Gigliotti and Peyton 1993; Thurow and Schill 1994). Conventional bait-fishing gear has been shown to cause high rates of hooking mortality (Shetter and Allison 1955; Stringer 1967; Mongillo 1984), and thus is often considered incompatible with regulation schemes aimed at keeping hooking mortality as low as absolutely possible. However, the current study demonstrates that circle hooks may be fished with bait for wild rainbow trout in a natural stream setting with resultant hooking mortality rates not unlike dry flies. Thus allowing bait fishing in the development of future restrictive special regulation waters may be possible if additional studies confirm the present findings and subsequent use of properly designed circle hooks is mandated.

Our results suggest that circle hooks reduce deep hooking compared to J hooks regardless of whether they are fished in a traditional bait-fishing manner (i.e., setting the hook), or fished passively, following the manufacturer’s recommendations. However, we tested only one circle hook design and size, which is now one of many commercially available hooks on the market in sizes applicable to stream trout. Not only do the shapes of circle hooks vary by manufacturer, but the profile differs as well. We used an in-line style of hook, which potentially could translate into lower rates of deep hooking relative to offset circle hooks, where the point is off to the side of the shank when viewed from the top (Vecchio and Wenner 2007).

In conclusion, our results suggest that circle hooks may have the potential to significantly decrease bait-hooking mortality for stream-dwelling wild trout compared to conventional bait hooks such as J hooks without drastically altering hooking or landing success. In addition, deep hooking rates in trout streams such as in our study may be lower for circle hooks fished actively (i.e., setting the hook) rather than passively. However, considering the scarcity of studies on wild stream-dwelling trout hooking mortality with circle hooks, our results should be viewed as preliminary, and additional studies with different species, stream conditions, and circle hook designs and sizes would be useful. Further, the potential of other bait hook designs to reduce the incidence of deep hooking should be investigated. Any hook design that results in the hook point riding laterally or dorsally (as opposed to ventrally) would likely reduce hooking mortality from bait fishing (Schill 1996).

RECOMMENDATIONS

1. In 2010, evaluate whether in-line or offset hooks affect the effectiveness of circle hooks in reducing deep hooking compared to J-hooks, as well as the hooking and landing success of both hook types.

2. Determine how widely available circle hooks are at tackle shops throughout Idaho.

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TABLES

Table 3. Sample size (percentage) of anatomical hooking locations for trout caught in Badger Creek using four different barbed hook types.

Hook location	J hook (%)	Circle hook (%)	Spinner (%)	Dry fly (%)	Total (%)
Upper jaw	35 (46)	58 (77)	22 (29)	44 (59)	159 (53)
Lower jaw	9 (12)	4 (5)	23 (31)	20 (27)	56 (19)
Mouth roof	7 (9)	4 (5)	5 (7)	2 (3)	18 (6)
Mouth floor	5 (7)	3 (4)	6 (8)	6 (8)	20 (7)
Tongue	1 (1)		5 (7)	1 (1)	7 (2)
Gill	6 (8)		4 (5)	1 (1)	11 (4)
Esophagus	10 (13)	3 (4)	3 (4)		13 (4)
Belly (foul)			1 (1)		1 (0.3)
Eye	3 (4)	3 (4)	8 (11)		14 (5)
Unknown			1 (1)		1 (0.3)
Total	76	75	75	74	300

Table 4. Initial population, ending population estimate, and hooking mortality rate for each of the four hook types.

	Initial	Ending population		Hooking mortality rate	
Hook type	population	Estimate	95% CI	Estimate	95% CI
Spinner	75	53	48-58	29	23-36
J hook	76	57	54-62	25	19-29
Circle hook	75	70	66-75	7	1-13
Dry fly	74	71	64-74	4	1-14

Table 5. Hooking success, landing success, and deep hooking rates by hook type and fishing method during the second angling experiment.

	Hooking success		Landing success		Deep hooking	
Hook type	Estimate	± 95% CI	Estimate	± 95% CI	Estimate	± 95% CI
Fly	56.1	6.9	87.0	6.2	0.0	0
Spinner	64.5	6.5	72.5	7.6	1.0	5.2
Baited J (active)	63.2	4.8	82.0	7.7	19.0	7.8
Baited J (passive)	37.9	5.2	68.5	7.3	20.0	8
Baited circle (active)	39.9	6.9	74.1	7	2.8	5.7
Baited circle (passive)	37.1	4.9	67.6	7.7	10.0	6

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